Master thesis

Anaerobic dry fermentation of dried chicken manure and kitchen waste

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TIIVISTELMÄ

Kananlannan ja ruokajätteen määrä, jätteiden aiheuttamat ympäristöongelmat sekä resurssipula fossiilisten lannoitevalmisteiden osalta kasvavat sekä Suomessa että globaalisti. Anaerobisella käsittelyllä orgaanisten jätteiden sisältämä energia on hyödynnettävissä biokaasun muodossa. Biokaasun ohella syntyvää mädätejäännöstä voidaan hyödyntää lannoitteena, sillä syötteen ravinteet eivät katoa prosessin aikana. Tässä työssä tutkittiin kuivatetun kananlannan ja tuoreen keittiöjätteen yhteismädätystä kyseisten jakeiden sisältämien energian ja ravinteiden hyödyntämiseksi. Aluksi jätejakeiden sekä niiden erilaisten yhdistelmien metaanintuottopotentiaalia tutkittiin panoskokein 35 °C lämpötilassa. Jätejakeista muodostetun sekoituksen biokaasun tuottoa tutkittiin puolijatkuvassa reaktorissa 35 °C lämpötilassa. Syntyvän mädätteen lannoitepotentiaalia tutkittiin typen ja fosforin pitoisuuksien perusteella.

Panoskokeissa metaanin (CH_4) tuoton kannalta tuottoisin jäteyhdistelmä oli 10 % kananlantaa ja 90 % keittiöjätettä orgaanisena kiintoaineena (VS) ilmoitettuna, tuottaen 398,5 mlCH_4/gVS. Kananlannan suurempi osuus seoksessa heikensi kaasuntuottoa (340.8 – 367.7 mlCH_4/gVS) johtuen lannan heikosta biohajoavuudesta. Kananlannan ja keittiöjätteen metaanintuottopotentiaalit olivat 301.1 ja 411.1 mlCH_4/gVS. Puolijatkuvassa prosessissa jäteseos tuotti 388.2 mlCH_4/gVS 3 gVS/l/d orgaanisella syötöllä, eikä inhibitiota havaittu. Vakaita olosuhteita ei kuitenkaan saavutettu, joten pitkäaikaisia, jatkuvatoimisia kokeita tarvitaan tulosten vahvistamiseksi. Tulosten perusteella kuivatun kananlannan ja keittiöjätteen yhteismädätys kuivaprosessilla on teknisesti mahdollista ja metaanin muodostus tuottoisaa, mutta typen syötön tulee olla matala inhibition ehkäisemiseksi.

Mädätteen ravinnepitoisuudet olivat typen osalta 4 g/l ja fosforin osalta 0,3 g/l. Ravinnepitoisuus ja typen ja fosforin suhde (13:1) ylsi kaupallisten biolannoitteiden tasolle (N:P -suhde 11 – 14:1). Mädätteen kokonaistypen ja -fosforin suhde oli hieman korkea, jolloin typpipitoisuus voi rajoittaa lannoitekäyttöä. Lannoitekäyttö on mahdollista mailla, joiden viljavuus on hyvä tai joiden fosforilannoitusta on vähennettävä.

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ABSTRACT

Generation of chicken manure and food waste, their resulting environmental problems and resource scarcity on fossil fertilizer production are increasing in Finland and globally. Anaerobic digestion is a sustainable environmental technology that can be used for recovering the energy in organic waste in the form of biogas. In addition, digestate generated alongside biogas can be used as a source of fertilizer since the process preserves fed nutrients. Co-dry fermentation of dried chicken manure and fresh kitchen waste was studied in an anaerobic digestion process to recover the energy and nutrients in the studied substrates. At first, methane potential of the two waste fractions and their mixtures was determined in biological methane potential (BMP) –tests at 35 °C. The biogas production of the waste mixture was studied in a semi-continuous plug-flow reactor at 35 °C. Later, the fertilizer potential of generated digestate was evaluated based on the nitrogen and phosphorus concentrations.

BMP results show that the highest methane production of 398.5 mlCH_4/g volatile solids (VS) added was obtained when the waste mixture consisted of 10 % chicken manure + 90 % kitchen waste on VS basis. Increase in chicken manure in the waste mixture decreased the methane production (340.8 – 367.7 mlCH_4/gVS) due to low biodegradability of chicken manure. BMP's of chicken manure and kitchen waste produced 301.1 and 411.1 ml CH_4/gVS, respectively. In the semi-continuous dry fermentation process, methane production was 388.2 mlCH_4/gVS at 3 gVS/l/d organic loading rate and no process inhibition was observed. However, steady-state conditions were not achieved and further long-term semi-continuous reactor studies are needed to confirm these results. Based on the results, co-digestion of dried chicken manure and kitchen waste by dry fermentation is feasible and productive but nitrogen feed should be low in order to prevent inhibition.

Chemical analyses showed that nitrogen (N) and phosphorus (P) concentrations in the digestate were 4 g/l and 0.3 g/l respectively and were comparable to that of commercial bio-fertilizers (11 – 14:1 N:P). The total N:P ratio in the digestate was high (13:1) and thus may limit its use on soils with high N content. However, fertilizer application is possible in soils with high P level or in soils where P fertilization is to be decreased.

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ABBREVIATION AND NOMENCLATURE

AD	anaerobic digestion
CH_4	methane
CM	chicken manure
CO_2	carbon dioxide
COD	chemical oxygen demand
BMP	biochemical methane potential
GHG	greenhouse gas
H_2	hydrogen
H_2S	hydrogen sulphide
HRT	hydraulic retention time
KW	kitchen waste
N	nitrogen
NH_3	ammonia
NH_4^+	ammonium
TKN	total Kjeldahl nitrogen
TS	total solids
OLR	organic loading rate
P	phosphorus
VFA	volatile fatty acid
VS	volatile solids

1 INTRODUCTION

Global food production and energy demand are increasing rapidly as the human population and living standards and capital in developing countries grow. As living standards and incomes are rising, food consumption per person increases and diets tend to change into containing more animal protein. Change in diet increases meat demand and has led to meat sector being one of the fastest growing food production sector, currently growing at 2.4 % per year. Global annual production growth rates in 1991 – 2007 and current production volumes for bovine, ovine, pig and poultry meat are 0.9 %, 1.6 %, 2.3 % and 4.4 % and approximately 62.6, 12.9, 99.9 and 82 million tons, respectively. (Alexandratos & Bruinsma 2012)

1.1 Generation and treatment of poultry manure and kitchen waste

In Finland, poultry meat production is estimated to increase by 20 % during 2010 – 2020 (Pyykkönen et al. 2010). Global poultry meat and egg production has increased in the last decade faster than other meat production sectors, nowadays covering about 32 % of total meat production and it is estimated to be the fastest growing meat sector in the future (Alexandratos & Bruinsma 2012, AVEC 2013). Over 12 million tons of broiler meat was produced in 2012 in the European Union and about 99 300 tons in Finland (AVEC 2013, Suomen Siipikarjaliitto 2010). In addition to poultry meat production, there were about 305 million egg layers in EU-15 area in 1999 (IRPP BREF 2003). Food production facilities are growing in size and concentrating locally producing locally larger amounts of manure and waste, which creates a potential for environmental problems such as greenhouse gas emissions, nutrient run-off, eutrophication and odour problems (Luostarinen et al. 2011b).

The amounts of produced food wastes in Europe are significant (European Comission 2010). The Finnish society produces an estimate of 335 000 – 460 000 tons of food waste and EU-28 area about 92 million tons of vegetal, animal and mixed food wastes annually (Silvennoinen et al. 2012, Eurostat 2014). These waste fractions produce at least 170 million tons of CO_2 eq. emissions annually (European Comission 2010). Silvennoinen et al. (2012) estimated that Finnish catering business and households produce 75 000 – 85 000 tons and 120 000 – 160 000 tons of food waste annually, respectively. The most food waste from catering business originated in nurseries, retirement homes and hospitals (Silvennoinen et al. 2012).

1.1.1 Treatment of poultry manure and biowastes in the EU

According to regulation of European parliament and of the council (EC 1069/2009), manures can be spread to field without hygienization, can be composted or treated anaerobically. Currently, there are no official statistics about manure treatment in the EU, but composting, biodrying by aerobic microbes, thermal drying and incineration are used (Lyngsø et al. 2011). All these methods create a potential emission risk (NH_3, CH_4, VOC - compounds), produce biofertilizers in forms of compost, pellets or ash, but apart from incineration, do not utilize the residual energy within manure (Lyngsø et al. 2011). In Finland, spreading of poultry manure into fields and selling as refined fertilizer are common applications for poultry manure (Suomen siipikarjaliitto, 2010). While spreading of manure improves nutrient balance in the soil, excess spreading causes environmental problems such as nutrient leaching to water environments, emissions of greenhouse gases, spread of pathogens and phytotoxins (Kelleher et al. 2002).

Waste treatment methods for bio-wastes are composting, anaerobic digestion, incineration and landfilling (European Comission 2008). Non-separated kitchen waste in municipal solid waste is often incinerated but if the waste is treated biologically, composting is the most used treatment method (European Comission 2008, Tilastokeskus 2010). Still, most food waste is landfilled along municipal solid waste (European Comission 2008) even though it is the worst option according to EU waste hierarchy (Directive 2008/98/EC). In Finland, 98 % of source-separated bio-wastes are composted (Huhtinen et al. 2007). Composting recycles the material by producing compost but is an energy-consuming treatment method if active aeration and mixing is needed (Lampinen & Laakkonen 2010). Low demand for compost decreases profits and promotes dumping of treated compost into landfills (Huhtinen et al. 2007). For incineration, bio-waste is an impure fuel, which causes high maintenance if combusted and the high moisture content in kitchen waste makes combustion inefficient (Zhang et al. 2007, Lampinen & Laakkonen 2010). This makes incineration a less worthy treatment option.

The commission of European Union encourages member states to move towards renewable energy by proposing new energy and climate goals for the EU: a 27 % share of renewable energy of total energy consumption and a 40 % GHG emission reduction from 1995 emission level (COM/2014/015 Final). Utilization of residual carbon as energy and nutrients present in wastes would result in environmental advantages due to improved

carbon balance, better condition of aquatic environments and would increase the economic viability of waste treatment (Lampinen & Laakkonen 2010) and is approved by EU (Directive 2008/98/EC). Therefore renewable energy production from previous organic wastes would be reasonable and benefits concerning economics and the environment could be achieved by treating organic wastes by anaerobic digestion (Lampinen & Laakkonen 2010).

1.1.2 Anaerobic digestion as a waste treatment method

Anaerobic digestion (AD) is one of the most effective and flexible biological waste treatment process (Jha et al. 2011) capable of treating various types of organic wastes (Sung & Santha 2001). An AD process has several benefits compared to previously mentioned waste treatments: AD preserves nutrients better, removes organic matter efficiently and improves local energy self-sufficiency and economy (Lampinen & Laakkonen 2010). AD reduces waste volume more than composting and produces energy whereas composting is an energy consuming process due to aeration (Deublein & Steinhauser 2011). Aerated composting can consume about 20 times more energy than AD (Deublein & Steinhauser 2011). AD also removes zoonotic pathogens and parasites and prevents natural CH_4 -emissions by capturing biogas (Rasi 2009, Massé et al. 2011). Anaerobic digestion has different roles concerning topical environmental problems: it reduces negative environmental impacts by waste treatment and it mitigates climate change and resource depletion via bioenergy and bio-fertilizer. According to Callaghan et al (2002), AD process is the most likely option for waste-to-energy solution for organic wastes if economic conditions are profitable.

Currently, anaerobic treatment of food wastes, agricultural wastes and municipal solid wastes is widely used in Europe (Chen et al. 2008, Li et al. 2011). Poultry manure is not currently treated anaerobically in the EU even though AD as manure treatment in general is increasing in several EU member states, such as Germany, Austria and Italy (Lyngsø et al. 2011).

The increase of AD has decreased the amount of landfilled organic fraction of municipal waste even though the increase of AD plants has been moderate (Häkkinen et al. 2014). The national waste plan of Finland to the year 2016 promotes the production and utilization of biogas from municipal wastes (Huhtinen et al. 2007) and anaerobic treatment

is estimated to replace composting as a treatment method for source-separated organic wastes (Häkkinen et al. 2014).

Biogas is a clean and energy-rich secondary energy source, which can be used as a source of district heating and electricity and as vehicle fuel when upgraded to natural gas quality. Thermal, electrical and mechanical energy production from biogas in Finland was 256.2 GWh in 2012, a 20 % increase from 2011. (Huttunen & Kuittinen 2013). Most of the biogas was used for heat production but combustion in combined heat and power (CHP) plants is also common (Huttunen & Kuittinen 2013). In addition to previous, the use of Finnish biogas as vehicle fuel has increased from 2000 MWh in 2011 to 32 000 MWh in 2013 (Huttunen & Kuittinen 2013). Biogas production from organic wastes is more economical than cultivation of energy crops due to gate fees (Rasi 2009).

The residual material from AD can be used as a fertilizer due to preserved nutrient content in the digestate (Lampinen & Laakkonen 2010). As global mineral fertilizer resources deplete and fertilizer prices rise, the demand and value of recycled fertilizers in agriculture will grow (Albuquerque et al. 2012). Utilization of phosphorus and nitrogen present in wastes has increased due to waste politics and has been widely applied in Finland (Marttinen et al. 2013).

Thus, anaerobic digestion can be seen as a profitable method to treat waste problems. In fact, organic wastes like animal manures and kitchen wastes can be seen as sources of bio-fertilizers and bio-energy rather than just environmental problems.

1.2 Anaerobic digestion

Anaerobic digestion is a biological process where anaerobic microbial community degrades organic matter under anaerobic condition to produce nutrient rich digestate and biogas (55 – 65 % CH_4, 35 – 45 % CO_2, H_2 and traces of H_2S and NH_3) (Abbasi et al. 2012). AD is a complex fermentation process in which different microbes break up and use organic matter for their metabolism. AD consists of four anaerobic phases working in sequence, hydrolysis, acidogenesis, acetogenesis and methanogenesis (Figure 1).

1.2.1 Biology of anaerobic digestion

In the first step of AD called hydrolysis, polymeric carbohydrates, proteins and lipids are broken down into monomers: sugars, amino acids and long-chain fatty acids by hydrolytic bacteria and their extracellular enzymes (Abbasi-Guendouz, 2012, Li et al. 2011) e.g. hydrolases, proteases and lipases (Deublein & Steinhauser 2011). Efficient hydrolysis requires pH less than 5 because VFA's are more toxic to hydrolytic bacteria at pH range of 5 – 7 (Abbassi-Guendouz, 2012). Hydrolysis can be speeded up by pretreating the substrate, e.g. by shredding (Abbassi-Guendouz, 2012).

In the second phase, acidogenesis, sugars, amino acids and fatty acids are fermented to short-chain organic volatile fatty acids (VFA's): mainly lactic, propionic and butyric and valeric acids by fermentative bacteria (Abassi et al. 2012). Also alcohols, nitrogen oxide, hydrogen sulfide, hydrogen and carbon dioxide are formed (Deublein & Steinhauser 2011). Amino acids are transformed to ammonia nitrogen (Jagadabhi 2011). High partial pressure of hydrogen reduces the amount of reduced compounds such as acetate, so high formation of hydrogen reduces the formation of VFA's (Deublein & Steinhauser 2011).

In acetogenesis, acetogenic bacteria oxidize lactate, alcohols and VFAs into acetate, carbon dioxide and hydrogen (Li et a. 2011, Jagadabhi 2011). Homoacetogenic microbes convert formed hydrogen and carbon dioxide into acetate and sustain low hydrogen concentration needed for acetogenic microbial performance since oxidation reactions will happen only under low hydrogen partial pressure (Deublein & Steinhauser 2011).

Acetate and carbon dioxide are consumed by methanogenic archae during methanogenesis, the last phase of AD, to produce methane (Li et al. 2011), water and carbon dioxide (Deublein & Steinhauser 2011). Methane can be formed either from acetate or from CO_2 and H_2 (Jagadabhi 2011). About 70 % of total methane comes from acetate by acetoclastic methanogens and about 30 % from CO_2 and H_2 by hydrogenotrophic methanogens (Deublein & Steinhauser 2011, Jagadabhi 2011). Methanogens have a symbiotic relationship with acetogens as they use up hydrogen produced by acetogenic microbes (Massé et al. 2011) providing low H_2 partial pressure. Methanogenesis is the most vulnerable phase for inhibition and process stability and is a rate-limiting phase whereas hydrolysis and acidogenesis are faster phases (Karthikeyan & Visvanathan 2013). Hydrolysis can limit the process rate if substrate structure is difficult to break down, e.g.

wood (Deublein & Steinhauser 2011), if substrate is easily degradable, acetogenesis is a rate-limiting step (Jagadabhi 2011).

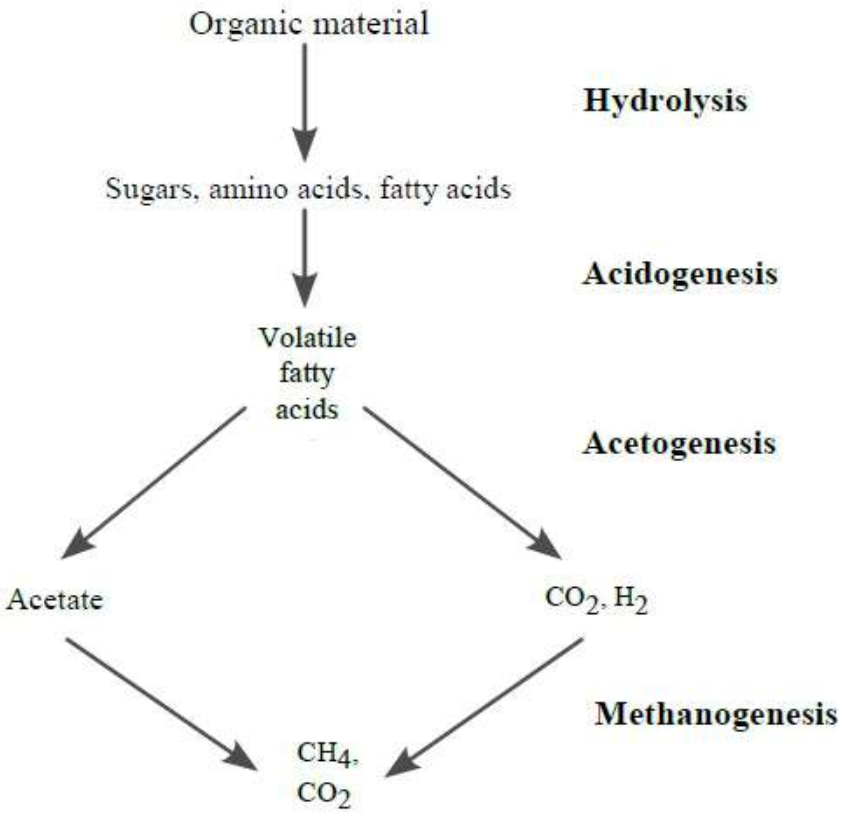

Figure 1. Anaerobic digestion process steps and their metabolites. Modified from Abbasi et al. (2012).

All organic wastes or biomass containing carbohydrates, proteins or fat as the main substrate are suitable as feedstock for anaerobic fermentation (Deublein & Steinhauser 2011). The use of lignin and lignocellulose based substrates is not profitable since their degradation is slow and incomplete (Deublein & Steinhauser 2011). The biodegradability of substrates from easily degradable to slowly degradable goes in the following order: sugars, proteins, fat, hemicellulose, lignin (Deublein & Steinhauser 2011). The degradation degree of digestate varies a lot (27 – 76 %) and is usually about 43.5 % (Deublein & Steinhauser 2011).

1.3 Factors affecting anaerobic digestion

Anaerobic digestion processes can be classified based on process temperature (mesophilic: 20 – 40 °C, thermophilic: 50 – 65 °C or psychrophilic: < 10 °C) (Abbasi et al. 2012), solids content (dry or wet), reactor type (vertical or horizontal), continuity of substrate feeding (continuous or batch) or by dividing AD process into separate biological stages that follow each other (single stage, two or multi stage reactors).

Each phase of AD process needs optimal environmental conditions as anaerobic microbes are sensitive to pH, temperature, intermediate products e.g. ammonia and volatile fatty acids and toxic compounds (Sakar et al 2009). These optimal environmental conditions, especially pH, differ between hydrolysis – acidogenesis phase and acetogenesis – methanogenesis phase. The requirements of methanogens are more important due to their low growth rate and sensitivity to disturbances and acidity in the reactor (Deublein & Steinhauser 2011). The overall process performance depends on the balance between different microbe populations and their conversion activity (Karthikeyan & Visvanathan 2013).

Results from different biogas plants and experiments vary often a lot due to the complexity of AD process and multiple factors affecting the process. Equipment and the quality of inoculum affect the methane yields and decreases reliability of comparison of separate studies (Browne & Murphy 2013). Reactor conditions such as pH, temperature, buffering capacity and volatile fatty acid (VFA) concentration as well as feedstock characteristics such as volatile solids (VS) content, nutrient content, carbon-nitrogen ratio, particle size and biodegradability affect reactor's overall performance and gas yields (Jha et al. 2011, Zhang et al. 2007). Mixing maintains process stability by preventing stratification of digester material and temperature inside the reactor and allowing even contact between microbial community and substrate (Abassi et al. 2012). The main effective operational parameters in AD are hydraulic retention time (HRT), organic loading rate (OLR), temperature, inoculation and pretreatment (Abbasi-Guendouz, 2012). Since it is unknown what is going on inside the reactor, chemical properties such as pH, VFA and nitrogen concentrations must be analyzed and the quality of the end products studied to overview reactor conditions during the process.

1.3.1 pH and alkalinity

Too high or low pH affects the abundance and chemical forms and properties of toxic compounds and the viability of micro-organisms (Jha et al. 2011). Methanogens require pH of 6.5 – 7.8 to function efficiently (Sakar et al. 2009). Optimal pH for a biogas reactor is considered to be 6.5 – 7.5 (Liu, C. et al. 2008), but usually pH stabilizes between 7.2 and 8.2 (Abassi et al. 2012).

Alkalinity in AD means the reaction equilibrium of CO_2 and bicarbonate ions in the solution that creates a buffering effect and resists significant and fast changes in pH (Ward et al. 2008). Buffering capacity of alkalinity means the ability of reactor mixture to neutralize acids and resist pH changes (Sakar et al. 2009). Hence buffering capacity increases the process stability and resistance against changes on environmental conditions.

1.3.2 Temperature

There are different anaerobic microbe communities functioning in different temperature ranges (Abbasi et al. 2012). Microbes functioning in lower, mesophilic (usually 35 °C) and psychrophilic (> 10 °C) temperatures have slower digestion rates requiring longer hydraulic retention times for AD (Karthikeyan & Visvanathan 2013) whereas thermophilic (usually 55 °C) processes have faster digestion rates and produce higher gas yields (Liu et al. 2007). This makes thermophilic processes more efficient and productive but they are also more difficult to maintain and they use more heating energy. On the other hand, Parawira et al (2007) noticed that mesophilic digestion resulted in better biogas yields compared to thermophilic digestion. Thermophilic microbes are more sensitive with process condition fluctuations and need longer time to recover from process failure (Karthikeyan & Visvanathan 2013) More stable and resistance performance causes mesophilic biogas plants to be more common (Jha et al. 2011) even though they need larger digesters and produce less biogas (Yaldiz et al. 2011). Dilution of substrates (e.g. chicken manure) lowers the energy input – output balance decreasing the energy production efficiency, especially in thermophilic temperatures (Massé et al. 2011).

1.3.3 Nutrients

Anaerobic microbes as well as other organisms require nutrients for growth, metabolism and production of enzymes (Jagadabhi 2011). Essential nutrients can be divided into

macronutrients (C, N, P and S) and nutrients needed in smaller amounts, micronutrients or trace nutrients (e.g. Fe, Ni, Co, Mo, W and Se) (Jagadabhi 2011). Addition of trace nutrients promotes microbial growth and results in improved digestion and higher methane yields (Sakar et al. 2009, Deublein & Steinhauser 2011). They also catalyze microbial metabolism as enzymes (Deublein & Steinhauser 2011). There should be an optimal availability of nutrients in the reactor since nutrient deficiency reduces degradation efficiency (Jagadabhi 2011) and can even lead to process failure (Banks et al. 2012). Nutrients can be provided by a heterogeneous feedstock or by identifying missing nutrients and supplementing them directly into the reactor (Banks et al. 2012).

1.3.4 Inhibition

Various inhibiting compounds may disturb the microbial activity by changing environmental conditions in the reactor and cause lower biogas yields and a threat of a process failure (Chen et al. 2008). Inhibition has been found out to be the most important factor that causes system failure and prevents biogas processes to be widely used (Chen et al. 2008). The level of inhibition depends on the concentration of inhibitors, microbial community and its resistance, environmental conditions and substrate composition (Deublein & Steinhauser 2011). Since harmful compounds may cause additive inhibition when present together, the complexity of overall inhibition causes variation in reported inhibiting levels of toxicants (Chen et al. 2008). Microbial community usually revives within 3 weeks from inhibition-based failure (Deublein & Steinhauser 2011) during which economic losses and energy shortages may result. Inhibition can be prevented by designing the process parameters, choosing feedstocks carefully and following process conditions before and during a steady-state.

Typical inhibiting compounds are e.g. ammonia, VFA's, hydrogen, minerals and heavy metals in high concentrations. They have a stimulating effect on microbial growth in small concentrations (Abassi et al. 2013), as many inhibitors are trace nutrients for microbes (Deublein & Steinhauser 2011) but harmful in higher amounts. Even though AD is an anaerobic process, oxygen is not usually inhibiting since acidifying bacteria consume oxygen fast and maintain anaerobic conditions (Deublein & Steinhauser 2011).

pH has an influence on the equilibrium of toxic to non-toxic forms of compounds (Jha et al. 2011), e.g. on ammonia. High concentrations of VFA's will drop pH and inhibit

methanogens (Liu et al. 2007) unless there is sufficient buffering capacity in the reactor (Kaparaju & Rintala 2005). Also other disturbances to methanogenic activity may cause over-acidification (Deublein & Steinhauser 2011). VFA's are important intermediates, which are eventually consumed but can cause high peaks and inhibition if organic loading is too intense for the microbial community (Liu et al. 2007). If VFA concentration stays high for long time, methanogens are replaced by acetogens (Sakar et al. 2009). The accumulation of VFA's has been stated to be a serious problem in dry biogas reactors (Jha et al. 2011, Abassi et al. 2012). Therefore it is important to optimize loading rates based on feedstock characteristics and to perform a careful start-up phase to achieve an efficient and stable biological process.

A portion of total nitrogen in the feed will be converted to ammonium (NH_4^+) and ammonia (NH_3) depending on pH and temperature (Karthikeyan & Visvanathan 2013). Rising pH and temperature shifts the reaction equilibrium towards ammonia, ammonium starts to transform into ammonia at pH over 6.5 and cause toxicity at over 7.0 (Sakar et al. 2009). Ammonia is the primary inhibiting nitrogen form (Nielsen et al. 2013) whereas ammonium is usually harmless (Deublein & Steinhauser 2011). Ammonia inhibits directly methanogenic enzymes and microbes by entering the cell, changing the intracellular pH and disturbing cell homeostasis (Calli et al. 2005, Nielsen et al 2013). Ammonia tends to accumulate in the digester and cause inhibition to methanogens at concentrations of 1500 – 3000 mg/l total ammonia ($NH_4^+ + NH_3$) and at 600 – 800 mg/l of free ammonia (NH_3) (Sakar et al. 2009, Karthikeyan & Visvanathan 2013) at pH higher than 8.5 (Abassi et al. 2012).

The adaptation of microbial community can increase resilience against inhibiting compounds (Chen et al. 2008) and inhibition can also be reduced by mesophilic degradation and pH adjustment (Deublein & Steinhauser 2011). Calli et al. (2005) produced biogas in total ammonia nitrogen concentration of 5000 mg/l and 800 mg/l free ammonia without any inhibition. Also Abouelenien et al. (2009b) found out that methanogens can adapt to high levels of ammonia and high solids (25 % TS).

1.3.5 Organic loading rate

Organic loading rate (OLR) means the feeding intensity and tells how much organic matter is fed to the reactor, usually given as amount of volatile solids (VS) or chemical oxygen

demand (COD) for reactor volume per day, e.g. kgVS/m^3/d. It tells the capacity of the biological conversion of the process (Karthikeyan & Visvanathan 2013). High loading rates are more economical, but not necessarily efficient (Deublein & Steinhauser 2011) because slowly degrading materials may not have enough time to degrade. High OLR enables treatment of larger amounts of waste making the process more productive in that sense but do not necessarily achieve maximum biogas yields if material leaves the reactor only partially degraded. Too high loading rate may also drop the pH due to fast accumulation of VFA's and eventually inhibit biogas production (Abassi et al. 2012). Most biogas plants work with 3 kgVS/m^3/d OLR or less (Deublein & Steinhauser 2011).

1.3.6 Hydraulic retention time

Hydraulic retention time (HRT) is the time that organic matter spends in the reactor and it corresponds on how well organics degrade before they get removed as digestate (Abassi et al. 2012). Methanogenic species grow slowly and HRT should be at least 10 – 15 days to prevent methanogens being washed away from the reactor (Deublein & Steinhauser 2011). Substrates with low degradability require at least 20 days HRTs for methanisation (Deublein & Steinhauser 2011). Longer HRTs could improve biosecurity of the digestion residue since some pathogens do not get destroyed during short retention times, especially at mesophilic temperatures (Alfa et al. 2014).

1.3.7 Pretreatment

Most substrates are usually pretreated before anaerobic digestion to increase solubility of the substrate and to speed up hydrolysis (Jha et al. 2011). Pretreatment can be done by mechanical, physico-chemical, thermal or biological methods and a technique should be economical, easy and environmentally friendly (Karthikeyan & Visvanathan 2013). Size reduction by mechanical treatment is a common procedure for bio-wastes and agricultural residues (Deublein & Steinhauser 2011). Screw mills, tearing devices and choppers are usually used since they are relatively cheap to operate and invest (Deublein & Steinhauser 2011). Particle size has been noticed to affect digestion efficiency and biogas yields due to increased active surface area and easy access for degradative enzymes (Karthikeyan & Visvanathan 2013, Deublein & Steinhauser 2011).

1.3.8 Reactor design

Reactor design between batch or continuous and single-stage or multi-stage reactors gives advantages and disadvantages to AD process. Single-stage reactors in which the whole AD process happens in a single reactor are more common in agricultural and large-scale biogas plants due to their simplicity (Liu et al. 2007, Deublein & Steinhauser 2011). In two- or multi-stage reactors, the hydrolysis – acidogenesis and acetogenesis – methanogenesis phases are separated into different reactors (Ward et al. 2008). The main challenge with single-stage reactors is to prevent pH drops due to fast hydrolysis, especially with food wastes (Liu et al. 2007). Two-stage reactors are noticed to be more efficient by the ability to degrade higher and more variable OLRs and having more stable biological process and higher gas production rates (Liu et al. 2007, Ward et al. 2008). On the other hand, multi-stage reactors are more expensive to build and operate (Ward et al. 2008).

A biogas reactor can be fed with substrate continuously or only once in the beginning. In the most simple biogas systems, batch reactors, the digester is filled once, sealed and opened when HRT is completed and organic matter supposedly degraded (Ward et al. 2008). Batch reactors are not mixed and there may be uneven distribution of nutrients and metabolites for efficient degradation (Li et al. 2011). Batch reactors often have fluctuating gas production and they lack the possibility to control the process (Li et al. 2011, Liu et al. 2007). Most of the batch reactors are used for research purposes to estimate methane potentials but are becoming more common in commercial use (Li et al. 2011, Liu et al. 2007). In continuous reactor, substrate is fed and the degraded matter is removed with regularity (Liu et al. 2007). Gas productions in continuous reactors are more stable (Li et al. 2011). Reactors are usually mixed to improve even distribution of required nutrients and metabolites and to combine substrate with microbes and to release produced gas (Ward et al. 2008). Most plants operate as continuous mixed biogas reactors (Luostarinen et al. 2011a).

1.4 Co-digestion

In anaerobic co-digestion two or more different substrates are digested simultaneously in the same reactor. If substrates are chosen right, co-digestion brings advantages to the process due to better carbon – nitrogen ratio, better buffering capacity, more diverse nutrient content or dilution of inhibiting compounds (resulting higher biogas yields) and

more biologically stable process (Karthikeyan & Visvanathan 2013, Chen et al. 2008). Adjustment of moisture and pH in the reactor can also be done by co-digestion (Esposito et al. 2012).

The right amount of total nutrients and water can be achieved by combining different substrates, e.g. nutrient-rich dry matter with wet nutrient-poor substrates. Substrate characteristics must be known for efficient co-digestion (Karthikeyan & Visvanathan 2013), e.g. substrates high in nitrogen (e.g. manures) and other inorganic nutrients should be digested with low-nitrogen and high-carbon content substrates (e.g. energy crops) to minimize ammonia nitrogen based inhibition. This kind of substrate mixture is suggested to prevent radical pH drops (Esposito et al. 2012). Karthikeyan & Visvanathan (2013) states that manures and highly biodegradable food or vegetable wastes form good mixture for co-digestion. Esposito et al. (2012) presented a faster and more resistant digestion process when highly biodegradable substrates were digested with ammonia-rich substrates. Kuglarz et al. (2011) noticed 10 – 60 % increase in methane production rates when pig manure was co-digested with kitchen waste.

1.5 Dry fermentation for biogas production

Dry fermentation is a process where high solids organic matter is converted to biogas via AD. Dry fermentation or dry anaerobic digestion is also called as solid-state or high-solid AD, dry anaerobic bio-conversion or dry digestion (Li et al. 2011, Karthikeyan & Visvanathan 2013, Abbassi-Guendouz, 2012). Classifications between wet and dry anaerobic digestion processes are based on solids content and vary between 10 – 20 % total solids (Jha et al. 2011, Karthikeyan & Visvanathan 2013, Abbassi-Guendouz 2012) but usually substrates containing more than 15 % of solids are considered dry and substrates containing less solids wet (Li et al. 2011, Liu et al. 2007).

Dry AD has advantages and disadvantages in comparison to wet AD processes. Dry processes require less water and provide smaller reactor sizes, they require less mixing and less heating energy due to smaller amounts of water but they require longer hydraulic retention times (HRT's), in some cases even three times longer (Li et al. 2011). When there is higher organic matter content and less water in the substrate, the corresponding amount of organic matter requires less space. Dry AD requires more inoculum (Li et al. 2011) but allows higher organic loading rates (OLR) and technical simplicity (Karthikeyan &

Visvanathan 2013). Dry reactor digestate is also easier to handle than wet sludges (Li et al. 2011) and there is less reject water to treat. Liu et al. (2007) stated that due to high viscosity of dry feedstocks they are more difficult to handle and feed to the reactor but easier to pretreat. Dry processes have been reported to have higher net energy gain than wet processes (Karthikeyan & Visvanathan 2013) with increased biogas yields and better gas quality (Deublein & Steinhauser 2011). Batch processes and plug-flow reactors are often used when solid substrates are anaerobically degraded (Luostarinen et al. 2011a). At least Dranco, Kompogas and Valorga processes have been noticed to be effecient at an industrial scale (Liu et al. 2007). Due to previous advantages, dry fermentation processes have been more popular and economical in the last decade (Karthikeyan & Visvanathan 2013) but most industrial sized biogas units used to have wet processes.

There are problems concerning dry AD since it is not yet a mature technology unlike wet AD (Abbasi-Guendouz, 2012). An important problem according to Karthikeyan & Visvanathan (2013) is that better biogas yields with dry fermentation processes require more inoculum, especially for batch reactors. Difficulties in mixing the substrate and microbes and removing of the produced gas from solids create problems in dry AD processes (Luostarinen et al. 2011a). Also too dry conditions lower process efficiency. Abbassi-Guendouz (2012) noticed that methane production decreased as solids content rose. There seems to be inhibition at over 35 % TS which could be due to higher concentrations of inhibiting agents (Abbasi-Guendouz 2012) but too low water content also slows down cell growth (Deublein & Steinhauser 2011). Abbassi-Guendouz (2012) also noticed that methanogens were less abundant in dry reactors compared to wet reactors. There seems to be a threshold of 30 % TS for efficient and stable methane production (Abbassi-Guendouz (2012). Also mixing may be difficult and degassing and even supply of nutrients is worse in a badly mixed reactor whereas local accumulation of inhibitors is possible (Deublein & Steinhauser 2011).

1.6 Kitchen waste and chicken manure for dry fermentation

Only few studies on dry fermentation of kitchen waste and chicken manure have been made and not a single study had studied their dry co-digestion (Table 1). Since dry fermentation studies of previously mentioned substrates are not available, studies on solid kitchen, food and vegetable wastes, poultry manures and their experimental set-ups and results are put together in table 1. Study results vary a lot due to heterogeneity of food

waste components and regional differences, which makes it harder to draw conclusions (Browne & Murphy 2013). Also different reactor types and operations differ causing variation. Two-stage systems consisted mostly of solid state leaching beds as acidogenic reactors and liquid methanogenic reactors feeding on leachate (Table 1).

In general, kitchen biowastes are considered as good substrates for AD, especially for co-digestion (Kuglarz et al. 2011). Chen et al. (2008) stated that high biodegradability and potential as a renewable energy source makes biowaste interesting for anaerobic digestion.

The maximum biogas yield according to Deublein & Steinhauser (2011) from biowaste varied between 0.3 – 1.0 m^3 biogas per 1 kgVS. Kitchen waste usually has a highly variable and heterogeneous composition, which makes AD challenging (Bodkhe & Vaidya 2012) and it is hard to predict biogas yields. Intense feeding of heterogeneous feedstocks may cause accumulation of organic matter due to differences in degradation rates (Abbasi-Guendouz 2012). The large portion of easily digested organic matter in kitchen waste (Li et al. 2011, Kuglarz et al. 2011) can cause limitations to OLR due to fast VFA production (Liu et al. 2007) if system is not adapted enough. Li et al. (2013) noticed VFA -based inhibition when OLR was increased over 5.6 gVS/l/d. Lane (1984) suggested that poultry manure would stabilize AD process by offering buffering capacity and nutrient supplement when co-digested with vegetable or fruit wastes. According to various searches from scientific databases, the combination has not been studied since.

Kitchen waste contains usually low concentrations of trace nutrients and nutrient addition may promote digestion process (Facchin et al. 2013) but it is not always needed. Zhang et al. (2007) noticed that Californian source-separated food waste contained sufficient nutrients for anaerobic digestion and there were significant variation in TS and VS contents during collection period. When reactor was fed with mixed vegetable waste in the study of Jiang et al. (2012), reactor crashed because of insufficient nutrient supply and high accumulation of VFA's. Chen et al. (2010) noticed similar results with VFA accumulation and they proposed continuous NaOH supplementation or co-digestion with animal manures to sustain a steady digestion. Even though food waste contained required nutrients, the variation in feedstock may stress the bioreactor and cause instability which is why co-digestion with homogeneous substrate could help to keep the process more stable.

Anaerobic digestion of chicken manure has been studied and put into practice but problems concerning low methane content in biogas and process stability have been found

(Callaghan et al. 1999). Chicken manure contains excreta, feathers, fodder, bedding material, mortality and water (Sakar et al. 2009). Chicken manure is usually rich in nitrogen (Abouelenien et al. 2009b) and has less organic matter, which decreases substrate's C:N -ratio and effective methane production. On the other hand, chicken manure contains sufficient nutrients for microbial growth and nutrient addition is not necessary (Güngör-Demirci & Demirer 2004). Nutrient contents in dried chicken manure are more concentrated compared to wet manure (Sakar et al. 2009). High ammonia content was concluded to be the most important problem in anaerobic digestion of chicken manure (Abouelenien et al. 2009a). Due to these properties, AD of chicken manure is prone to inhibition and system instability and therefore chicken manure is not often treated anaerobically, even though it is possible to produce biogas with methane content of 60 % (Güngör-Demirci & Demirer 2004). Sakar et al. (2009) found out that most AD processes treating poultry manure operated under mesophilic conditions. Abouelenien et al. (2009b) successfully digested chicken manure alone in mesophilic conditions even though ammonia inhibition was observed. Chicken manure is often diluted to 0.5 – 3.0 % TS content to dilute nitrogen concentration for inhibition-free AD (Bujoczek et al. 2000). Bujoczek et al. (2000) noticed an increase in ammonia inhibition as TS and VS loadings increased. Bujoczek et al. (2000) gained best biogas yields with 5 % TS dilution. Diluting creates large volumes of treatable waste and requires lots of water (Bujoczek et al. 2000) which could be avoided by causing the dilution effect by co-digestion. Even though environmental benefits are gained as biogas, a large scale dilution consumes enormous amounts of water if 95 % of the feed is water. Dilution with water also makes the process less profitable (Bujoczek et al. 2000). Luostarinen & Pyykkönen (2013) stated that nitrogen problems of chicken manure can be avoided or mitigated by diluting manure with substrate poor in nitrogen and rich in organic matter such as kitchen waste.

Table 1. Previous studies on AD of solid food wastes and poultry manures. *= acidogenic reactor, **=methanogenic reactor, UBF = upflow blanket filter, UASB = upflow anaerobic sludge blanket, HASL = Hybrid anaerobic solid-liquid system, CSTR = completely stirred tank reactor

Feedstock	TS (%)	HRT (d)	OLR (gVS/l/d)	Temp (°C)	Reactor design	CH_4 yield ($mlCH_4/gVS$)	Reference
Food waste	15 – 30	-	2 gVS/l	-	BMP	472	Cho et al. 1995
Food waste	20	-	2, 4, 10	37	Two-stage batch: solid-bed*, UBF**	405 – 415	Cho et al. 1995
Food waste	68.2 – 73.9	-	-	40	BMP	233.5	Wang et al. 1997
Food waste	22.4	15	5.7 – 7.9	35 – 38	Two-stage continuous	440	Lee et al. 1999
Food waste	12	-	-	35 ± 1	Two-stage batch	250	Wang et al. 2002
Food waste	23	20	5.72	36.5	Single-stage continuous	390	Banks et al. 2008
Food waste	16.3, 18.6	-	-	35 ± 1	Two-stage HASL	-	Liu, X.Y. et al. 2008
Food waste	23.5	20	1	35	Single-stage continuous	180	Chen et al. 2010
Food waste	18 ± 3	8* 1.2**	10.8 ± 0.6* 6.5 ± 0.5**	37 ± 1	Two-stage batch: leaching bed*, UASB**	270 ± 10	Shin et al. 2001
Food waste	27.7 – 27.8	80	2.5	42	900 m^3 single-stage continuous	402	Banks et al. 2011a
Food waste	29.4	-	-	37	BMP	467 – 529	Browne & Murphy 2013
Food waste	22.6	30	3	37	CSTR	480	Facchin et al. 2013
Food waste	24.2	20 – 150	1.1 – 8.8	27 ± 2	Single-stage continuous	353 – 488	Li et al. 2013

Table 1, continuation. Previous studies on AD of solid food wastes and poultry manures. FVW = fruit and vegetable waste, FW = food waste, CSTR = completely stirred tank reactor

Feedstock	TS (%)	HRT (d)	OLR (gVS/l/d)	Temp (°C)	Reactor design	CH_4 yield ($mlCH_4/gVS$)	Reference
Food waste	23.1 – 25.7	10.5 – 31.2	6 - 16	35 ± 1	Two-stage CSTR	390 – 405	Zhang et al. 2013
Food waste	24.8	39 – 117	2 – 6	37	Semi-continuous	405 – 483	Tampio et al. 2014
Food waste, FVW	FW: 22.6, FVW: 9.5	30	1 – 3.5	35	Single-stage CSTR	328 – 478	Shen et al. 2013
Food waste, FVW	FW: 22.6, FVW: 9.5	10	2 – 10* 1 – 5**	35	Two-stage CSTR	198 – 458	Shen et al. 2013
FVW	8 – 18	20 – 30	1.6 – 3.6	-	CSTR	370 – 470	Bouallagui et al. 2005
Chicken manure	21.7	-	-	35	BMP	13	Bujoczek et al. 2000
Chicken manure	10	-	-	35	BMP	283	Bujoczek et al. 2000
Chicken manure	25	-	-	37	Batch	31	Abouelenien et al. 2009
Poultry manure	86.7	-	-	35	BMP	282.16	Esposito et al. 2012
FVW	8 – 18	20 – 30	1.6 – 3.6		CSTR	370 – 470	Bouallagui et al. 2005
Chicken manure	21.7	-	-	35	BMP	13	Bujoczek et al. 2000
Chicken manure	10	-	-	35	BMP	283	Bujoczek et al. 2000
Chicken manure	25	-	-	37	Batch	31	Abouelenien et al. 2009
Poultry manure	86.7	-	-	35	BMP	282.16	Esposito et al. 2012

1.7 Use of digestate as a fertilizer

An AD digestate is homogeneous, stable, has low pollutant content and all the nutrients fed along feed remain in digestate in accessible forms for plants (Güngör-Demirci & Demirer 2004, Yaldiz et al. 2011). Digestate improves soil quality by improving carbon and nutrient balance and preventing erosion, has better nutrient balance than raw manures and causes less eutrophication due to decreased biological oxygen demand (Lampinen & Laakkonen 2010, Massé et al. 2011). Because of previous advantages, digestate can be used as biofertilizer to replace mineral fertilizers.

The digestate from AD process can be gasified, pyrolyzed or carbonized for energy purposes or used as a fertilizer but the most common utilization is composting and fertilizer use (Möller & Müller 2012, Deublein & Steinhauser 2011). In Finland, most AD digestates are spread on the fields (Marttinen et al. 2013). The digestate can be used as a fertilizer as received but the most is often refined by separating solids and liquids, concentrating nutrients by evaporating excess moisture and improving stability by composting (Marttinen et al. 2013). Digestate must be upgraded to meet the requirements of its planned utilization purposes (Marttinen et al. 2013).

The suitability for plants depends on biological stability, homogeneity and availability of nutrients in the digestate (Marttinen et al. 2013). The purity, quality and nutrient content of the digestate depends on used feedstocks (Abassi et al. 2012) and process operational design (Möller & Müller 2012) (Table 2), which makes especially the choice of feedstocks important if fertilizer use is planned. For example, total nitrogen (N) contents varied between 3.1 and 14 % in different studies (Möller & Müller 2012). The digestate of AD preserves and concentrates nutrients as they were, but nitrogen compounds within the digestate are mineralized to soluble ammonium, which reduces N_2O -emissions compared to artificial fertilizers because of faster penetration to the soil (Lampinen & Laakkonen 2010, Luostarinen et al. 2011a, Weiland 2010). The uptake of ammonium nitrogen is also easier and fast due to better availability (Marttinen et al. 2013, Alburquerque et al. 2012). Ammonium present in the digestate could provide better yields on crops with high nitrogen demand and during short growth seasons (Möller & Müller 2012). On the other hand, easy solubility enables nutrient runoff (Luostarinen et al. 2011b) and eutrophication risk. Haraldsen et al. (2011) found out that AD digestate caused more

leaching of ammonium than commercial fertilizer, compost or raw manure. Insoluble nitrogen in digestate mineralizes so slowly that it is not usable on short Finnish growth season (Marttinen et al. 2013) but could be profitable in warmer countries. Diluted digestates could also be effectively applied to soilless cultivation conditions (Möller & Müller 2012).

Unlike ammonium nitrogen, phosphorus (P) tends to bind into soil particles and is less soluble (Luostarinen et al. 2011b) and therefore less prone to leaching (Haraldsen et al. 2011). Digestate from AD of poultry droppings contained useful fungi and bacteria that form soluble nutrients available for plants (Alfa et al. 2014).

Table 2. Digestate characteristics based on literature and of commercial bio-fertilizers. Contents are reported as wet weight. * = Commercial fertilizer products.

Substrate	TS %	Total N g/l	Total P g/l	N:P -ratio	Study or fertilizer
Pig slurry, slaughterhouse and biodiesel wastewater,	1.9	3.8	-	-	Alburquerque et al. 2012
Food waste	4.5	5.6	0.4	14:1	Banks et al. 2011a
Food waste	3.9	2.1	0.3	7:1	Facchin et al. 2013
Household waste	1.5	2.2	0.2	11:1	Haraldsen et al. 2011
Pig slurry, industrial by-products	8.9	7.6	2	3.8:1	Marttinen et al. 2013
Biowaste, vegetable wastes, cattle manure	2.7	4	0.4	10:1	BioKymppi Oy: PeltoKymppi A*
Manure, industrial by-products	1.6	5.2	0.4	13:1	Biovakka Oy: Biovakka Moniravinne*
Manure, biowastes, industrial by-products	1.8	3.5	0.5	7:1	Jepuan Biokaasu Oy: Jepuan Kasvuvoima*
Manure, industrial by-products, wastewater sludge	6.1	4.6	0.9	5.1	VamBio Oy: VamBion Perus fertilizer*

AD typically removes 40 – 70 % of the organic matter in form of CH_4 and CO_2 (Luostarinen et al. 2011a). The remaining carbon is quite stable, less than 10 % of residual carbon was mineralized in studies of Marttinen et al. (2013).

Phosphorus and nitrogen contents in the digestate limit the utilization possibilities (Luostarinen et al. 2011b). Finnish Government Decree 366/2007 limits phosphorus fertilizing to maximum 80 kg/ha/a for field crops and 120 kg/ha/a for garden plants. The corresponding nitrogen limit for fields is 170 kg/ha/a (Government Decree 931/2000).

Phosphorus content in the fertilizer is usually more limiting than nitrogen (Luostarinen et al. 2011b). Phosphorus and nitrogen can be separated by partitioning solid and liquid fractions of digestate since most phosphorus is bound to solids while most nitrogen is in liquid fraction (Liedl et al. 2006).

There are contradictory results on digestate effect on crop yields so further field experiments are needed (Möller & Müller 2012). Liedl et al. (2006) noticed that digestate of poultry litter AD can be more effective as a fertilizer than chemical fertilizers but there are varying effects of crop biomass and yield on different crop plants. Haraldsen et al. (2011) got as good barley (*Hordeum vulgare*) yield and nutrient uptake with digestate of household waste AD as commercial NPK-fertilizer. Alburquerque et al. (2012) noticed that digestate raised concentrations of available phosphorus in the soil more effectively than cattle manure and inorganic fertilizers but digestate did not improve yields in all plant species. To achieve a maximum fertilizing effect, digestate nutrient contents and utilization timing must meet the specific nutrient requirements and growth characteristics of each crop species (Alburquerque et al. 2012). Chicken manure contains following nutrients: N, P, K, Ca, Mg, S, Mn, Cu, Zn, Cl, B, Fe, and Mo which are all essential for plant growth (Sakar et al. 2009) and therefore needed in fertilizers.

AD digestate must meet hygiene requirements based on pathogen concentrations set by EC 1069/2009 and MMMa 24/11. Food waste digestate needs to be hygienized for biosecurity as mesophilic process does not destroy all pathogens completely (Banks et al. 2011b, Marttinen et al. 2013). Poultry manure does not require hygienization or treatment before spreading to fields but source separated biowaste must be hygienized at 70 °C for at least 60 minutes (EC 1069/2009, MMMa 24/11). Digestate hygienization adds more investment costs and increases energy use extending the payback time of the treatment process (Banks et al. 2011b). Marttinen et al. (2013) did not observe significant change in microbial activity or plant toxicity of the soil after digestate adding whereas Alfa et al. (2014) found high levels of potentially harmful bacteria such as coliforms and *Salmonella* spp. in the digestate. Therefore, digestates seem to be suitable for plant fertilizing but the need for hygienization must be determined before use.

AD digestate has less odorous emissions, pathogens and harmful organic compounds such as phthalates and polycyclic aromatic hydrocarbons (PAH -compounds) than substrate before AD (Luostarinen et al. 2011a). The digestate becomes odourless in 12 – 24 hours

after being removed from the reactor (Deublein & Steinhauser 2011). Currently, there are no toxicity or stability requirements based on the law (in Finland) for AD digestates, but they are expected by EU legislation in the future (Marttinen et al. 2013).

1.8 Objective

The main objective of the study was to evaluate the technical feasibility of anaerobic co-digestion of dried chicken manure and kitchen waste under dry fermentation. Methane potential of dried chicken manure, kitchen waste and their mixtures was determined in batch experiments. Continuous process performance and methane yields of co-digestion of dried chicken manure and kitchen waste was investigated in a plug-flow reactor at 35 °C for 93 days. The fertilizing potential of the digestate was evaluated based on total nitrogen and phosphorus concentration.

2 MATERIAL AND METHODS

2.1 Substrates and inoculum

Chicken manure, dried for 6 months, was collected from GFA de Pierport poultry farm situated in Castelnau de Montmiral, France. The farm rears annually 1.2 million chickens and other poultry depending on the season. Chickens were fed on commercial ready-to-use feed. Chicken manure was shredded to reduce particle size by Blik BB 230 grinder and weighed by Kern 572 scale (type: DS100K0.5). Shredded chicken manure was stored in plastic containers at -20 °C for further use.

Fresh kitchen waste was collected during four weekdays from a nearby restaurant (N.S Restauration's kitchen, Narbonne, France). Collected kitchen waste contained only vegetables: cucumber and carrot peelings, radish tops and leaves (Figure 2). Boiled potatoes and rice along with bread were added to kitchen waste in order to simulate the ingredient diversity and to represent more Finnish kitchen waste (Table 3). Kitchen waste was grinded by Moulinex grinder (type: ME415) to break the particle structure and to reduce particle size. Grater mode was the most efficient grinding method because high moisture content caused frequent blockade formation in the grinder. After grinding, kitchen waste was stored at -20 °C. After four days of waste collecting, grinding and freezing, when enough kitchen waste was collected, the frozen waste was melted at room temperature and mixed thoroughly to gain a homogenized feed. Melted kitchen waste was dosed in 0.5 and 1.0 litre plastic containers and stored at -20 °C to prevent biodegradation. During the experiment, each week the needed amount of kitchen waste and chicken manure for one week were melted and stored at 4 °C separately.

Granular sludge from a UASB reactor treating sugar refinery waste (Marseille, France) was used as inoculum. The inoculum was mixed for 4 days prior to the experiment to break down the granular structure and to enhance the contact between the microbes and substrates. The VS content of the pretreated sludge was analyzed and adjusted to 5 gVS/l by diluting with distilled water.

Table 3. Composition of kitchen waste by weight fractions.

Waste Fraction	% (wet weight)
Vegetables (total)	75
Cucumber peelings	35
Carrot peelings	33
Radish leaves	7
Potatoes	7
Rice	13
Bread	6

Total solids (TS), volatile solids (VS), total Kjeldahl nitrogen (TKN) and ammonium (NH_4^+) concentrations were analyzed from kitchen waste (see section 2.4). Previously mentioned analyses from chicken manure were already done by INRA (The French National Institute For Agricultural Research).

Figure 2. Collected kitchen waste (left) and shredded kitchen waste (middle and right).

2.2 Batch assays

Biochemical methane potentials (BMP) of dried chicken manure, kitchen waste and their combinations were determined in batch experiment. Batch experiment was conducted in 400 ml glass bottles at 35 °C. Based on the VS content, six different substrate mixtures of chicken manure and kitchen waste were prepared as follows: 10 % VS chicken manure + 90 % VS kitchen waste, 20 % VS chicken manure + 80 % VS kitchen waste and 30 % VS chicken manure + 70 % VS kitchen waste. To each assay, 1g VS of substrate and 78 ml of

inoculum were added to attain a substrate VS to inoculum VS ratio of 0.5. Similarly, control assays were also prepared with 1g COD ethanol.

To each assay, 4.3 ml of macronutrients and 4 ml of micronutrients, 20.8 ml bicarbonate buffer, and distilled water were added to achieve a final working volume of 400 ml. The composition of fed nutrient and buffer solutions is shown in Table 4. Prepared assays were flushed with nitrogen for creating anaerobic conditions and closed with rubber stoppers. Closed bottles were incubated in a room at 35 °C. The experiment was conducted in duplicates. Bottles were manually mixed twice a day during weekdays.

Table 4. Chemical compositions of nutrient solutions for BMP's (Monlau et al. 2013).

	Concentration (g/l)
Macro nutrients	
NH_4CL	26.6
KH_2PO_4	10
$MgCl_2$	6
$CaCl_2$	2
Micronutrients	
$FeCl_2$	2
$CoCl_2$	0.5
$MnCl_2$	0.1
$NiCl_2$	0.1
$ZnCl_2$	0.05
H_3BO_3	0.05
Na_2SeO_3	0.05
$CuCl_2$	0.04
Na_2MoO_4	0.01
Bicarbonate buffer	
$NaHCO_3$	50

2.3 Semi-continuous experiment

The semi-continuous experiment was carried out in 15 litre horizontal plug-flow reactor with 10 litre working volume (Figure 3) at 35 °C. Reactor was mixed intermittently in 20 minutes interval by Bonfiglioli Riduttori mixer run by Schneider Electric timer (Type CCT15722) to create homogeneous conditions and to minimize stratification in the reactor.

Mixing was set to periods because of electricity savings. Reactor's gas production was monitored by Ritter gas meter (type: MGC-1V3.0, measuring chamber: 3.27 ml), which was connected to computer. Gas meter readings were saved by computer in every 2 minutes.

Figure 3. Plug-flow reactor for semi-continuous experiment.

Before the experiment reactor was fed daily with 1 gCOD/l/d ethanol for one week to estimate endogenous respiration of the sludge. Gas production rate was calculated from observed gas production curve after fed ethanol had been used by the microbial community. Other estimates of endogenous respiration were done by measuring gas production rates from the last 12 hours of each week. After the experiment, endogenous respiration was estimated by following gas production rates during 12 hours after 24, 48 and 72 hours after the experiment had ended. The endogenous respiration was excluded from total gas production volume to calculate gas yield derived from fed substrate. After feeding with ethanol, used feed was introduced to microbial community by feeding 0.5 gVS/l/d organic loading rate (OLR) for one day (day 0), before the actual experiment.

Overall, the experiment lasted for 13 weeks. During the experiment, OLR was gradually increased from 1 gVS/l/d to 3 gVS/l/d. After the initial start-up, reactor was fed with an OLR of 1 gVS/l/d and substrate ratio of 47 % VS of chicken manure and 53 % VS of kitchen waste (day 1). Reactor was fed every weekday by opening the top hatch, which allowed air to flow into reactor's top space. Daily inputs of kitchen waste and chicken manure were weighed into a beaker by Precisa XT1220M scale and then manually fed to the reactor. After three weeks (day 23), OLR was increased from 1 to 2 gVS/l/d. After 5 weeks from the beginning of experiment (day 36), the substrate mixture was changed to 10 % VS of chicken manure and 90 % kitchen waste to prevent possible nitrogen

accumulation in the future. Day 36 was the first day of third week (out of 6 weeks) during OLR 2 gVS/l/d. After 6 weeks of operation at an OLR of 2 gVS/l/d and 4 weeks of lower chicken manure proportion, OLR was increased further to 3 gVS/l/d (day 65) and continued at this rate for another 4 weeks (until day 93). Theoretical HRTs (hydraulic retention times) during OLR phases 1 to 3 were 222, 80 and 53 days, respectively. The experiment for OLR phases 1 to 3 lasted for 22, 41 and 31 days, respectively.

Reactor's working mass was balanced in the beginning of each week. The sludge inside the reactor was mixed well and sludge was removed to balance reactor's mass to correspond a working mass of 10 kg. Samples were taken from removed sludge. Reactor samples (figure 4) were stored at 4 °C. The following analyses were done from samples to follow the stability of the process and possible environmental changes: TS, VS, total Kjeldahl nitrogen (TKN), ammonium nitrogen, VFA, alkalinity and pH. Gas composition was measured 2 – 3 times a week, usually on every second day, every time before feeding.

Figure 4. Digestate sample.

2.4 Analytical methods

Gas production of BMP test bottles were calculated based on pressure changes in bottles. Pressure inside BMP bottles was measured with a manometer (Keller Mano 2000). Gas production and composition were analyzed in triplicates three times a week during the first two weeks, twice a week during the third week and once a week during weeks 4 – 7. After each measure, gases were let out from bottle until pressure in bottle was the same as atmospheric pressure. The experiment lasted for 42 days during which 11 measurements were done.

2.4.1 Total solids and volatile solids

TS and VS concentrations were analyzed in triplicates according to Monlau (2012). Precisa XT1220M scale was used for weighing, plastic desiccators for cooling and ovens WTB Binder (type: 0105325000100) and Vulcan (type: A550) were used for heating.

2.4.2 pH and alkalinity

Total alkalinity was analyzed in duplicates according to Björnsson et al. (2000) and standard method (ISO 9963 – 1) with titration endpoints at pH 4.3 and 4.0. Inolab pH 720 meter was used for pH measuring. Finnpipette Labsystems pipettes (4500, 1 – 5 ml) were used for pipetting.

2.4.3 Total Kjeldahl nitrogen and ammonium nitrogen concentrations

TKN and ammonium nitrogen concentrations were analyzed in duplicates by Büchi autokjeldahl unit K-370 distiller according to Jimenez et al. (2013), respectively. Samples were mineralized by Büchi Digest Automat K-438.

2.4.4 Volatile fatty acids concentrations

VFA concentrations were analyzed in triplicates by PerkinElmer Clarus 580 gas chromatograph according to Affes et al. (2013). Eppendorf minispin 5452 centrifuge was used for centrifuging samples.

2.4.5 Gas composition

Gas composition was measured from gas sample by Clarus 580 PerkinElmer gas chromatograph according to Affes et al. (2013). Samples were analyzed in triplicates and average compositions were used in further calculations. Teruma Neolus 0.4 * 20 mm needles (NN-2719R) and GSE 250 µl syringe were used for taking gas samples and PerkinElmer Clarus 580 gas chromatograph was used for gas composition analysis. Gas samples (2 ml) were taken with a syringe from reactor’s gas outlet and from BMP test bottle through permeable plastic membrane and injected into gas chromatograph.

2.4.6 Fertilizing potential

Fertilizing potential of the digestate was estimated by measuring total phosphorus concentration (mg/l) from the last reactor sample by Hach Lange LCK348 test. Four

parallel analyses were done. Diluted samples were heated at 100 °C for 60 minutes for hydrolysis, cooled and analyzed by DR LANGE Lasa 100 (v.1.20, type LPG 357).

2.5 Calculations

2.5.1 Batch experiments

Methane production in BMP's was calculated based on number of produced CH_4 moles. First, number of moles of CH_4 (N) was calculated from equation,

$$\Delta N(j) = \left[y(j)P(j)\frac{V_h}{RT}\right] - \left[y(j-1)P_{atm}(j-1)\frac{V_h}{RT}\right] \quad (1)$$

, where

y(j) = Content of CH_4 in biogas on day *j* (%)

P(j = Pressure in the bottle on day *j* (bar)

V_h = Working volume (l)

R = Gas constant = 8.314 J/mol^{-1}/K^{-1}

T = Temperature (K)

y(j-1) = Content of CH_4 in biogas on day *j-1* (%)

P_{atm}*(j-1)*= Atmospheric pressure on day *j* (bar).

Volume of CH_4 (ml) in standard temperature and pressure (V) was calculated from equation,

$$\Delta V(j) = \Delta N(j)\frac{RT_0}{P_0} * 10^6 \quad (2)$$

, where

T_0 = 273.15 K

P_0 = 1.0 Bar

Accumulated methane yields were calculated by summing up each gas production measurements and average methane yields were calculated from duplicates. Endogenous respiration was excluded to obtain methane yield for 1 gVS substrate without microbial activity. Endogenous respiration was calculated from control experiments in which the used substrate was ethanol. The corresponding CH_4 amount for 1 gCOD (394 ml CH_4) was subtracted from total methane production to gain the value of endogenous respiration.

2.5.2 Semi-continuous experiment

Daily methane yields were calculated for 1 gVS fed substrate according to average methane content of reactor samples in current days and biogas yields. Endogenous respiration for each week was extrapolated from gas production curve of the last 12 hours of each week. Endogenous respiration was excluded from methane and biogas production volumes to achieve yields. Weekly biogas and methane yields were calculated by summing daily methane and biogas yields. Data collection and processing was done by Microsoft Office Excel 2010 for all collected data.

The proportion of free ammonia nitrogen (NH_3) (g/l) was calculated from equation (El-Mashad et al. 2004),

$$[NH_3] = [NH_4^+] * (1 + \frac{10^{-pH}}{10^{-\left(0{,}1075 + \frac{2725}{T}\right)}})^{-1} \quad (3)$$

, where

$[NH_4^+]$ = concentration of ammonium (g/l)

T = temperature (K).

3 RESULTS

3.1 Substrate characteristics

The characteristics of the studied substrates are presented in Table 5. Kitchen waste (KW) had low solids content (161.9 mg/g) due to great portion of vegetables with high moisture whereas dried chicken manure (CM) contained high concentration of solids (826 mg/g) (Table 5). The proportion of volatile solids in total solids was much higher in kitchen waste (91.5 %) than in chicken manure (62.7 %). Chicken manure was observed to contain about 15 times more total nitrogen (30.4 mgTKN/g) than kitchen waste, which contained 2.4 mgTKN/g. Ammonium concentration in chicken manure was 1.5 mg/g whereas kitchen waste had only 0.009 mg/g. On the other hand, TS and VS contents of the inoculum were 31.9 mg/g and 20.9 mg/g, respectively. The characteristics of substrate mixtures are calculated from results of chicken manure and kitchen waste.

Table 5. Characteristics of feedstock used in the study.

	KW	CM	10 % VS CM + 90 % VS KW	20 % VS CM + 80 % VS KW	30 % VS CM + 70 % VS KW
Total solids (mg/g)	161.9 ± 0.5	826	182.4	206.2	234.4
Volatile solids (mg/g)	148.2 ± 0.7	518	160	172.9	188.6
TS/VS -ratio (%)	91.5 ± 1.4	62.7	87.7	83.9	80.5
Total Kjeldahl nitrogen (mg/g)	2.4 ± 0.1	30.4	3.3	4.3	5.5
Ammonium (mg/g)	0.009	1.5	0.06	0.1	0.2

3.2 Batch assays

The effect of different substrate mixtures on methane potential are presented in Figure 5 and Table 6. Methane production started immediately in all assays and lasted for 42 days. Kitchen waste had the highest methane potential of 411 ml/gVS while chicken manure the lowest (301 ml/gVS) (Table 6). The more chicken manure was present in substrate mixture, the less methane reactor yielded (Figure 6). Average methane contents between substrate mixtures did not vary significantly except for chicken manure which had about 5

% less methane in biogas (Table 6). Methane production from endogenous respiration was estimated to be 29.8 ml for 78 ml inoculum during the 42 days and is excluded from Figure 5 and Table 6.

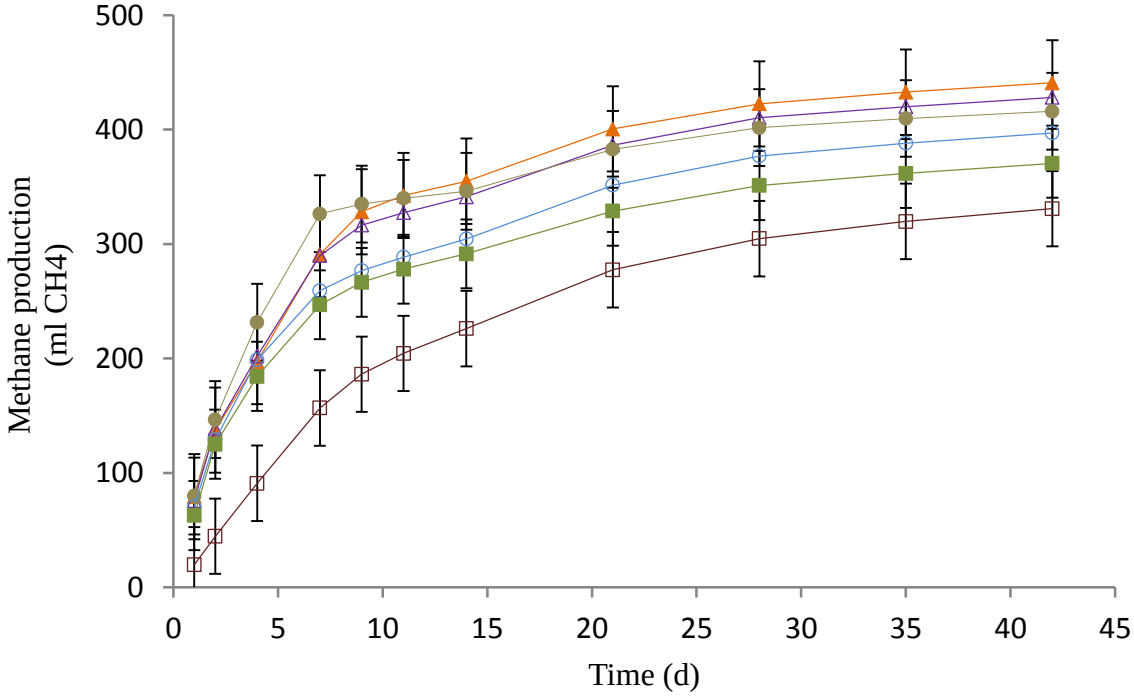

Figure 5. Cumulative methane production in BMP tests. Each BMP reactor was fed with 1 gVS substrate containing kitchen waste (KW) and chicken manure (CM) in different proportions. Endogenous respiration is excluded. ▲ = KW, ▲ = KW 90 % VS + CM 10% VS, ● = Ethanol, ○ = KW 70 % VS + CM 30 % VS, ■ = KW 80 % VS+ CM 20 % VS, □ = CM.

Table 6. Biological methane potentials of kitchen waste (KW), chicken manure (CM) and their mixtures.

Substrate	KW	CM	CM 10 % VS + KW 90 % VS	CM 20 % VS + KW 80 % VS	CM 30 % VS + KW 70 % VS
Methane yield (ml CH_4 / gVS)	411.1 ± 74	301.1 ± 14.1	398.5 ± 10.8	340.8 ± 24.7	367.3 ± 6.6
Average CH_4 content (%)	56.5 ± 1.1	51.7 ± 1.1	56.2 ± 0.7	55.8 ± 0.2	56.5 ± 0.4

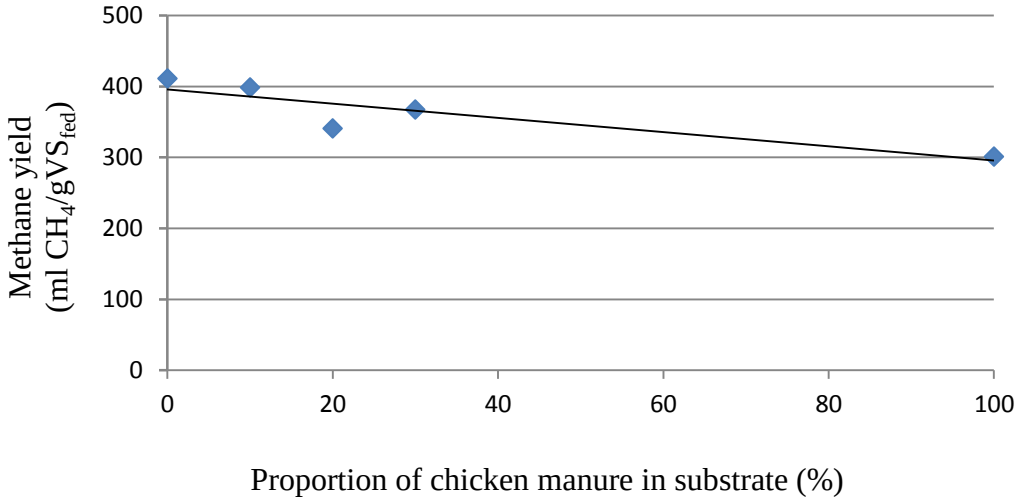

Figure 6. Effect of dried chicken manure on biological methane potential at 35 °C.

3.3 Semi-continuous experiment

In the semi-continuous experiment, a feed mixture of chicken manure and kitchen waste was fed into a mesophilic 15 l plug-flow reactor. The results of the semi-continuous experiment are presented in Figure 8 and Table 7. Biogas and methane yields were lower during the first 22 days when OLR of 1 gVS/l/d and the highest proportion of chicken manure were used. Average biogas and methane yields for this period were 570.4 and 230.4 ml/gVS_{fed}, respectively. Also the methane content in biogas increased from average of 42 % to average of 51 and 54 % as loading rates increased to 2 and 3 gVS/l/d (Table 7). There was a 24.7 % increase in methane yields after day 36 when proportion of kitchen waste in the feedstock mixture was increased from 53 % VS to 90 % VS and proportion of chicken manure was decreased from 47 % VS to 10 % VS. This change in feed increased the total average biogas and methane yields during 2 gVS/l/d OLR. Weekly biogas and methane yields increased as OLR increased from 1 to 2 gVS/l/d but both yields remained quite similar during loading rates 2 (675.1 and 350.5 ml/gVS) and 3 gVS/l/d (676.2 and 388.2 ml/gVS) (Figure 8a). The increase of OLR from 2 to 3 gVS/l/d yielded a 2.6 % increase in weekly methane yields. The average methane yield for 1 g of fresh substrate mixture during loading rates 1 to 3 were 43.8, 59 and 62.1 mlCH_4/g substrate, respectively.

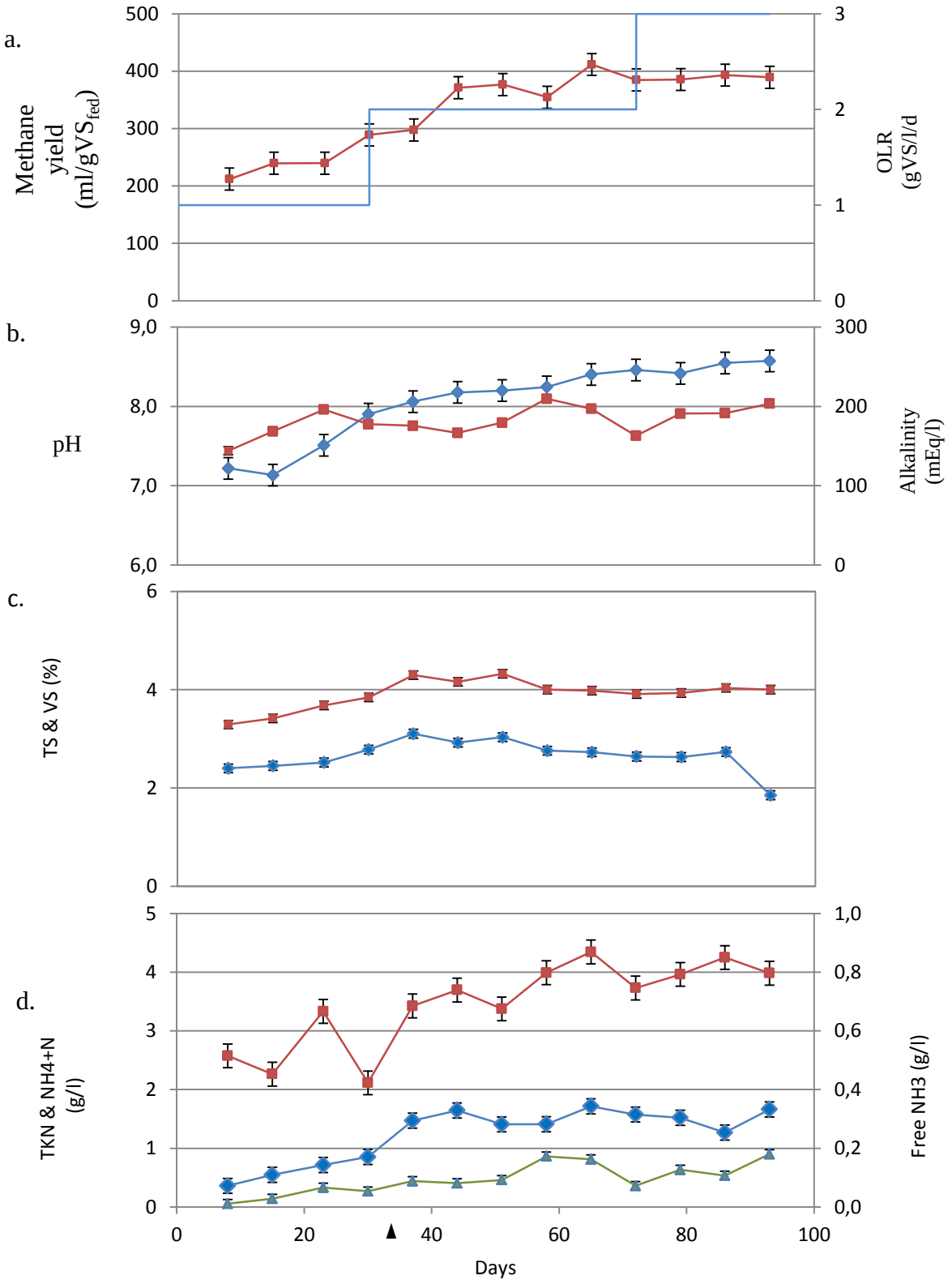

Figure 7. Process conditions in semi-continuous reactor. a: ■ = methane production. b: ■ = pH, ♦ = alkalinity. c: ■ = total solids, ♦ = volatile solids. d: ■ = total Kjeldahl nitrogen, ♦ = ammonium nitrogen, ▲= free ammonia nitrogen. ▲= day 36 when feedstock mixture was altered.

Process conditions, which were analyzed to estimate process stability during the semi-continuous experiment, are shown in figure 7. pH fluctuated between 7.4 – 8.1 but there was only slight increase during the experiment and no sign of microbial inhibition was observed during the experiment. Ammonium levels rose until week 6 and remained stable since whereas there was increase in TKN levels during the whole experiment. TS and VS contents in the digestate increased less than 1 percent during experiment, reaching 4.0 % and 2.5 % in the end, respectively.

Table 7. Digestate characteristics and biogas and methane yields for each OLR.

	OLR (gVS/l/d)		
	1	2	3
TS (%)	3.5 ± 0.2	4.1 ± 0.2	4 ± 0.1
VS (%)	2.5 ± 0.06	2.9 ± 0.2	2.5 ± 0.4
VS reduction (%)	12.6	24.9	41.03
pH	7.7 ± 0.3	7.8 ± 0.2	7.9 ± 0.2
Biogas yield (ml/gVS_{fed})	570.4 ± 13.5	675.1 ± 70.4	676.2 ± 81.6
CH_4 yield (ml/gVS_{fed})	230.4 ± 15.9	350.5 ± 47.8	388.2 ± 3.9
CH_4 content in biogas (%)	42 ± 9.2	51.4 ± 4	54.4 ± 3.9
Calculated CH_4 yield (l/kg fresh substrate)	43.8	59	62.1
TKN (g/l)	2.7 ± 0.6	3.5 ± 0.8	4 ± 0.2
NH_4-N (g/l)	0.5 ± 0.2	1.4 ± 0.3	1.5 ± 0.2
Free NH_3 (g/l)	0.03 ± 0.03	0.1 ± 0.05	0.1 ± 0.05
VFA (g/l)	0.03 ± 0.002	0.03 ± 0.01	0.03 ± 0.004

3.4 Fertilizing potential

Digestate sample taken after the experiment contained on average 0.3 ± 0.006 g/l total phosphorus. The nitrogen concentrations had a rising trend for almost throughout the experiment and reached 4 gTKN/l and 1.5 gNH_4-N/l at OLR 3 gVS/l/d (Table 7). The nitrogen to phosphorus ratio of digestate was 13:1.

4 DISCUSSION

4.1 Substrate characteristics

The characteristics of used substrates showed they were suitable for co-digestion. Kitchen waste acted as a carbon source whereas chicken manure provided required nutrients for the process. Efficient biogas production from concentrated chicken manure cannot be done without inhibition (Bujoczek et al. 2000). The high moisture of kitchen waste diluted high nitrogen contents of chicken manure and decreased the risk for ammonia inhibition during AD (Table 5). Wet kitchen waste also moistened chicken manure which was initially too dry for AD (Table 5). This way the use of excess water was avoided and thus lower heating demand was achieved. As proportion of chicken manure increased in feed mixture, TS/VS -ratio decreased and TKN rose (Table 5). This shift moves towards ammonia based inhibition and lower biodegradability.

Used kitchen waste contained less TS and VS than most studies (Table 1) due to high proportion of vegetables and lack of meat and dairy products. Cho et al. (1997) and Bouallagui et al. (2005) had similar substrate characteristics and reached similar methane yields (Table 1). Esposito et al. (2012) had similar chicken manure characteristics and achieved similar BMP results (Table 1).

4.2 Batch assays

Kitchen waste was clearly more biodegradable as was predicted based on substrate characteristics. Therefore, the best feedstock mixture methane yield-wise was 10 % VS CM + 90 % VS KW. Lower methane yields in mixtures containing more CM resulted from low biodegradability of chicken manure.

The methane yields obtained for chicken manure in the present study (301 ml/gVS_{fed}) were slightly higher than those reported in the literature (13 – 283 ml/gVS) (Bujoczek et al. 2000, Abouelenien et al. 2009, Esposito et al. 2012), which could be due to more efficient inoculum. On the other hand, this kind of effect was not seen with kitchen waste BMPs. BMPs of kitchen waste produced less methane (411 ml/gVS_{fed}) than mentioned in former studies, e.g. some studies reached methane yields up to 472 and 529 ml/gVS (Cho et al. 1995, Brown & Murphy 2013). However, some studies had relatively low BMP yields, between 94.8 – 270 mlCH_4/gVS) and high standard deviation especially with kitchen waste

weakens the comparability of the results (Wang et al. 1997, Shin et al. 2001, Wang et al. 2002). High variation in results can result from various factors, such as characteristics of substrates and inoculums, differences in the BMP procedures and in careful laboratory working methods (Browne & Murphy, 2013). Relatively low results from this experiment could be due to kitchen waste characteristics: high proportion of vegetables and lack of proteins and fats.

BMP of 10 % VS CM + 90 % VS KW mixture and the semi-continuous experiment with the same substrate mixture reached similar methane yields, 399 and 388 ml CH_4 /gVS_{fed}, respectively. Also, control BMP produced 98 % of the theoretical yield. These indicate that BMP experiment was successful, results are comparable and they represent the degradation of the substrate.

4.3. Semi-continuous experiment

The dry fermentation for biogas production was found to be technically feasible and productive. Methane yields were rising during OLR 1 gVS/l/d which lasted for three weeks (Figure 7). This time was estimated to be the start-up phase when microbial populations grew, began utilizing substrates more efficiently and shifted to a more stable operation phase. The increase of kitchen waste and decrease of chicken manure in the substrate mixture at day 36 did obviously enhance biogas production and increased the amount of methane in biogas (Figure 7, Table 7). This effect was observed also with BMP assays (Figure 6). Methane yield with 47 % VS CM + 53 % VS KW should have been 360 ml/gVS, (theoretical yield) when calculated from BMP results of CM and KW. The poor actual methane yield with 47 % VS CM + 53 % VS KW (230 ml/gVS_{fed}) feed at 1 gVS/l/d OLR could be because of a joint effect of start-up phase and high proportion of chicken manure. The actual and calculated methane yield of 10 % VS CM + 90 % VS KW mixture were quite close (388 ml/gVS_{fed} and 400 ml/gVS, respectively) indicating that the semi-continuous reactor functioned properly.

As seen in BMPs, lower methane yields (due to substrate characteristics) are also seen in semi-continuous AD of 10 % VS CM + 90 % VS KW mixture. There were differences in reactor designs and loading rates in literature (Table 1) which create variation in results. Food waste in literature had often more VS than kitchen waste (Table 1). On the other

hand, Facchin et al. (2013), Shen et al. (2013) and Bouallagui et al. (2005) reached higher methane yields by using similar feedstocks (Table 1).

No VFA accumulation was observed during the experiment which indicates a complete degradation of produced VFA's (Table 7). Fast consumption of VFAs gives evidence on stable and properly functioning AD process (Liu et al. 2007) without danger of inhibition. Also VS content of the digestate remained stable (Figure 7.c), the last lower measuring point could be caused possibly by unrepresentative sample or divergent laboratory work. Stable biogas yields with OLR 2 and 3 gVS/l/d (Figure 7.a) prove that microbial activity was stable and adapted for current loading rates. These results indicate that feeding with used substrates and OLR was within a safe range (Liu et al. 2007) and the process would have most likely allowed an increase in feeding intensity. On the other hand, rising nitrogen levels (Figure 7.d) would have probably restricted the increase of OLR and caused ammonia-based inhibition in the future.

Throughout the experiment nitrogen levels (TKN, NH_4-N) stayed below inhibiting levels (Karthikeyan & Visvanathan 2013) and showed no inhibition even though they had a rising trend (Figure 7.d). Due to the rising trend, the proportion of chicken manure in the feed mixture was decreased to lower feed's nitrogen supply to prevent ammonia based inhibition. Nitrogen supply was reduced from 8 gTKN/kg to 3.3 gTKN/kg and the chicken manure's share in the mixture reduced from 47 %VS to 10 %VS. Total ammonia levels were calculated to reach a concentration of 1.3 g/l within 2 HRT with 3.3 gTKN/kg. This provides safety in which accidental overfeeding does not crash the reactor as easily and the situation can be corrected before problems occur. A time scale of two full HRTs was used as a time frame in N-level model because AD processes may take up two HRTs to achieve a fully steady state (Ganesh et al. 2013). Even though nitrogen did not seem to be a problem within 13 weeks of AD, the experiment should have been continued further, at least for two whole HRTs (106 days with OLR of 3 gVS/l/d), for verifying a stable and inhibition-free operation.

The objective of AD, whether it aims for maximum biogas yield or optimal waste treatment, should be well defined before planning the actual process. For effective biogas production, the optimal proportion of kitchen waste for maximum biogas yield should be set. If the role of AD is to biologically treat chicken manure, its proportion cannot be lowered too much at the expense of higher methane yields. On the contrary, for optimal

chicken manure treatment purposes, the maximum proportion in feed for stable, inhibition-free AD should be studied.

1 ton of fresh feed would produce approximately 62 m^3 CH_4 which corresponds for about 620 kWh (Lehtomäki et al. 2007). The feed ratio (CM:KW) by fresh matter is about 1:32, so 1 t of treatable dried chicken manure would need about 32 t of kitchen waste to form a 10 % VS CM + 90 % VS KW mixture. With this in mind, possibilities for reasonable kitchen waste production and collection near poultry farms should be studied. Since so significant amounts of kitchen waste are needed for optimal mixture, a large scale continuous treatment would require wide kitchen waste collecting activity from various sources to provide enough feedstock. As generation and production of kitchen waste varies more than chicken manure, kitchen waste should be seen as the main feedstock and chicken manure as supplementary feedstock. This is recommended especially in smaller cities and municipalities. For enabling a full scale dry fermentation process with highly variable feedstocks, characteristics of substrates should be studied in each case to check their suitability and to form an optimal mixture.

4.4. Fertilizing potential

As is seen, characteristics of feedstocks vary a lot and the diversity affects digestate properties. Phosphorus and total nitrogen contents of 0.3 g/l and 4 g/l were only slightly lower than in general (Table 2). Most digestates in table 2 were derived from manure AD. The low manure content in the feed in this study obviously lowers nutrient contents. Still, total nitrogen and phosphorus contents were quite close to commercial bio-fertilizers (3.5 – 5.2 g/l TKN, 0.4 – 0.9 g P/l) made from AD digestates (Table 2). The N:P -ratio (13:1) was similar to BioVakka Moniravinne -fertilizer which is suitable for grain crops, sugar beet, oil plants and grass (BioVakka Oy 2013). Since digestate is quite similar to commercial products, it should be usable as a bio-fertilizer.

High nitrogen content in the digestate limits the spreading on field crops lowering the possibilities of phosphorus fertilization. With a maximum 170 kgN/ha/a spreading, the spread digestate would supply phosphorus approximately 13 kgP/ha/a. This kind of digestate would be useful in soils with high P-levels but most likely inefficient if more phosphorus is needed. Digestate could also be applied in soils where current phosphorus supply is excess. According to Valkama et al. (2009), phosphorus application in Finnish

agricultural soils should be reduced for sustainable nutrient utilization. Since nutrient requirements of crop species and soil characteristics creates the nutrient requirements (composition and volume), an optimal fertilizer for each species and field must be individually composed (Mavi 2009).

5 CONCLUSIONS

The co-dry fermentation of kitchen waste and dried chicken manure is technically feasible and a potential method for treating these waste fractions and providing bioenergy and biofertilizer. Co-digestion of chicken manure with food wastes or other wastes would increase the level of chicken manure AD. Based on this study, dried chicken manure can be efficiently digested well as a side substrate but not alone due to high nitrogen content. Chicken manure can be seen as a source of alkalinity and nutrients for nutrient-poor and easily degradable vegetable-rich kitchen waste rather than a single substrate for AD. Co-substrates are needed for diluting inhibiting agents to avoid excess water use and lower process efficiency. The nitrogen concentration in the feed should be set to reach nitrogen contents inside the reactor below inhibiting levels and stay within safe limits to provide a buffer effect in case of accidents. For a stable process, it is crucial to keep the nitrogen concentration low. The digestate could be utilized as a fertilizer in soils that require moderate or no phosphorus supplementation due to digestate’s high nitrogen to phosphorus ratio.

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