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Anaerobic digestion : its potential to improve the economic and environmental performance of organic farming systems

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Anaerobic Digestion: its potential to improve the economic and environmental performance of organic farming systems

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A thesis submitted in fulfilment of the requirements of a

Doctor of Philosophy

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Summary

The safe and effective treatment of biodegradable resources is increasingly being seen as a prerequisite for environmental protection in the context of preserving water quality, reducing greenhouse gas emissions and preventing the dispersal of human pathogens. One potential mechanism to meet this need is the treatment of organic resources by anaerobic digestion (AD), especially on-farm AD. The aim of this thesis was to place an economic value on the non-market environmental benefits of AD, above just being a source of renewable energy, through a combination of scientific and economic research. In chapter 4, digestate from manure subject to AD, undigested manure and synthetic fertilizer was applied to pasture (grass ± clover) grown in pots. It was found that there is potentially less leaching from digestate compared to synthetic fertilizer, with no negative impacts on yield. Chapter 5 was based on the same pot trial, and reports that the application of digestate affects the soil decomposer community in a similar way to that of synthetic fertilizer. Chapter 6 is the accumulation of a three year field trial comparing crop yield between synthetic fertilizers and digestate. Again, it was found that digestate application may replace synthetic fertilizer and maintain crop yields. Chapter 7 is an economic valuation of all the non-market benefits of on-farm AD. The valuation highlights the economic benefits of implementing on-farm AD as a management tool for organic residues. This thesis is multidisciplinary in nature, encompassing microbiology, soil and environmental science, agronomy, and economics. An understanding of all these disciplines is imperative to properly value the benefits of AD both at a private level to the farmer, and at a public level to the wider community. The research indicates that AD is currently economically undervalued under the current renewable energy incentive (Feed-In-Tariffs; FIT) scheme run by the UK government and it is proposed that the FIT should be increased by £0.03 - 0.15 per kWh of electricity produced via AD. This would substantially increase the FIT rate to between £0.12 - 0.30 per kWh. Increasing the FIT to reflect all the non-market benefits that on-farm AD delivers would incentivise uptake of the technology and would facilitate the long-term viability of the industry. It would also rightfully reflect the fact that AD offers an effective pollution abatement technology as well as a source of renewable energy.

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This PhD is dedicated to Mam and Dad, for their continuous support and encouragement, throughout life.

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There are a large number of people who helped greatly in the completion of this thesis. I would like to start the long list of thanks with my funders, KESS and especially Penny, Fre-Energy and Calon Wen, Chris, Richard, Jonathon and Denise, for all their assistance throughout the duration of the PhD. Thank you to the co-authors of some chapters, Johannes, Neal and Dan for excellent assistance.

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ABBREVIATIONS

AD	Anaerobic Digestion
AHR	Anaerobic hybrid reactor
ANOVA	Analysis of variance
BOD	Biological oxygen demand
C	Carbon or control treatment (depending on circumstance)
°C	Degree Celsius
Ca	Calcium
CHP	Combined heat and power
CH_4	Methane
C:N	Carbon to nitrogen ratio
CO_2	Carbon dioxide
CO_2e	Carbon dioxide equivalent
COD	Chemical oxygen demand
CS	Conventional soil
C:S	Carbon to sulphur ratio
CSTR	Continually-stirred tank reactor
DFD	Dry fibre digestate
DM	Dry matter
DOC	Dissolved organic carbon
DON	Dissolved organic nitrogen
EC	Electrical conductivity
EU	European Union

ETS	Emission trading scheme
F:B	Fungal to bacterial ratio
FIT	Feed in tariff
g	Gram
GHG	Greenhouse gas
GWP	Global warming potential
H_2S	Hydrogen sulphide
h	Hour
ha	Hectare
K	Potassium
KESS	Knowledge Economy and Skills Scholarship
kg	Kilogram
kWh	Kilowatt hour
LD	Liquid digestate
Leu	Lecuine
m	Million
m^2	Meter squared
m^3	Meter cubed

CHAPTER 1

Introduction

1.1 Funding body and aim of Funding

The Knowledge Economy and Skills Scholarship (KESS), is a joint EU and Welsh Government scholarship, set up in 2009. The aim of the scholarship is to bridge the gap between academia and industry in regards to research. To this end, during this PhD there was collaboration between Bangor University, Calon Wen (Organic milk producers) and Fre-Energy (Anaerobic Digestion (AD) plant manufacturers). The PhD specifically focussed on on-farm AD since this is the main portion of Fre-Energy's commercial activity. As part of the KESS program, the PhD participant was obliged to work for the industrial co-sponsor company for approximately 4 weeks every year. As part of this work placement, an operations manual was written for Fre-Energy's AD units. (Although the operations manual was present in the submitted copy of the thesis, it will not be submitted in the final version at the request of Fre-Energy due to commercial sensitivities.)

Chapter 2 of the thesis comprehensively reviews the literature in regards to the current technical and scientific knowledge of AD. Chapter 3 details the methodology used in the experimental trials designed to answer important research questions on AD. Chapter 4 details a short pot trial experiment, designed to better understand the agronomic (grass yield) and environmental benefits (loss of nutrients through leaching) from the application of digestate relative to other organic and synthetic fertilizers. Chapter 5 reports on the effects of application of digestate on the soil decomposer community (bacteria and fungi) in a grass system. Chapter 6 reports on a three-year long field trial which aimed to determine the long-term crop yield and feed value of a mixed ley pasture after the application of digestate. Chapter 7 is an economic valuation of all the non-market benefits brought about from the introduction of on-farm AD. This chapter extrapolated the findings from earlier chapters, as well as those from existing literature. The thesis ends with a short discussion, detailing how this body of work has furthered the knowledge of AD, and recommends areas for future research that both time and funding constraints failed to allow for during the course of the PhD.

1.2 A brief history of AD

Anaerobic digestion is a very old renewable energy technology, with reports that AD was used to heat bath water in Assyria during 10th century BC. However, the first recorded report of the production of biogas directly related to the decomposition of organic material is in 1776, by the Italian physicist, Volta. He observed how methane was generated from organic matter in the bottom sediments of ponds and streams (Kothari et al., 2010). The first evidence of the construction of an AD digester was at an Indian leper colony in 1859, but it was not until 1895 that AD first reached the UK, when recovered biogas was used to light street lamps in Exeter (Harris, 2013). However, despite AD probably being the oldest source of renewable energy in the world, the general public's understanding of the technology appears to be somewhat lower than that of some other renewable energy technologies.

1.3 What is Anaerobic Digestion?

Anaerobic Digestion (AD) is the decomposition of organic matter by a microbial consortium in an oxygen-free environment (Pain & Hepard, 1985). This produces biogas (a methane-rich gas which can subsequently be burnt to produce heat and electricity) and digestate (organic matter which can be used as a biofertilizer). Figure 1 is a simple flow diagram of how the AD process converts organic material to energy and digestate. The feedstock can go through a pre-treatment process or be digested in its current state. Pre-treatment can involve pasteurisation (though this can also occur post-digestion), maceration of large feedstock, or mixing of different feedstock. Post-treatment usually involves the separation of digestate into a dry fibre and liquid fraction, and can also include pasteurisation (if not conducted pre-digestion and the feedstock contains animal by-products). The use of pre- and post-treatment ultimately depends upon the design specification of the digester unit and feedstock.

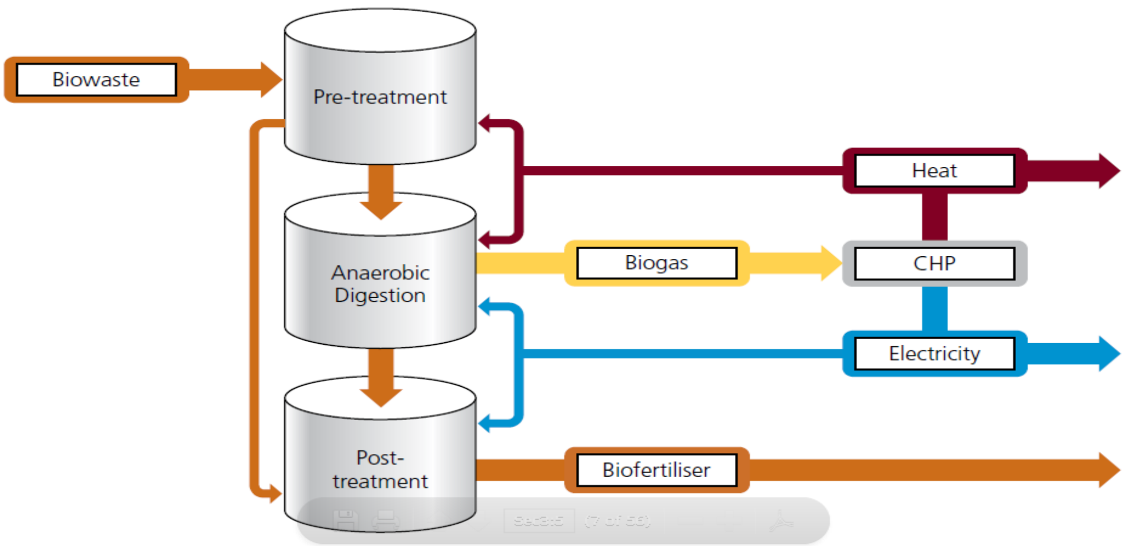

Figure 1.1: Flow diagram of Anaerobic Digestion

AD can convert almost all sources of biomass to biogas (Holm-Nielsen et al., 2009), with only strongly lignified organic products not being suitable (Weiland, 2010). However, the feedstocks (organic matter) that are most often used for AD include food waste, livestock manure) and purpose-grown crops such as maize. From here on manure incorporates both solid manures (stackable) and slurries (liquid). If manure, solid manure or slurry is referred to this is to refer to that specific type of livestock excretion. As a result, AD systems are particularly suitable on farms. Biogas is extracted from the volatile solids of the initial feedstock during digestion. The greater the proportion of volatile organic matter in AD feedstock, the greater the biogas production per unit volume of material (Nelson & Lamb, 2002); and pre-treating feedstock (by maceration) to reduce the fibrous particle sizes can improve methane production by up to 20% (Angelidaki & Ahring, 2000). During digestion of cattle manure, approximately 25-40% of the organic dry mater is converted to methane and carbon dioxide (Klinger, 1998). The biogas produced typically comprises of 50-70% methane (CH_4), 30-45% carbon dioxide (CO_2), ~ 500 ppm hydrogen sulphide (H_2S), and ~100 ppm Ammonia (NH_3) (ADAS, 2012; Mohseni et al., 2012). The digestate produced can then be passed through a separator which separates the dry fibre fraction from the liquid fraction. Livestock manures are high in nitrogen and are thus a valuable crop nutrient, with cattle/pig, chicken and turkey manure having 6, 19 and 30 kg N per tonne (ADAS, 2001), and food waste approximately 7.5 kg N per tonne (Taylor et al., 2010). Processing of such on-farm resources via AD therefore generates digestate with a potentially high nutrient value. Studies suggest that AD does not alter the total quantity of nutrients in the feedstock, but alters it into a form that is more readily available for plant uptake just the form of that particular nutrient (Field et al., 1984; Masse et al., 2011). This potentially increases the appeal of AD systems on farms due to the increasing costs associated with buying synthetic fertilizers. The liquid fraction contains the majority of nitrogen and potassium (Moeller et al., 2010); while the dry fibre can be used as a soil conditioner, with low amounts of nitrogen and the majority of phosphorus (Bauer et al., 2009).

1.4 The AD process

There are four stages in the AD process: (1) Hydrolysis, (2) Acidogenesis, (3) Acetogenesis, and (4) Methanogenesis. During hydrolysis, carbohydrates, fats and proteins are broken down into sugars, fatty acids and amino acids, respectively. After hydrolysis, the process enters the second stage, acidogenesis, which is similar to fermentation. During acidogenesis, sugars, fatty acids and amino acids are broken down to form carbonic acids, organic acids, alcohols, hydrogen, carbon dioxide and ammonia. The components from the acidogenesis stage are further broken down in the acetogenesis stage to produce hydrogen, carbon dioxide and acetic acid by acetogenic bacteria; which have a symbiotic relationship with methane forming bacteria. In the final stage of the digestion process, methanogens convert products from the intermediate process (acidogenesis and acetogenesis) into water, carbon dioxide and methane. Methane-forming bacteria are classified in the Archaebacteria domain and can be classified into three distinct groups: hydrogenotrophic methanogens, acetotrophic methanogens and methylotrophic methanogens. Hydrogenotrophic methanogens access hydrogen to convert CO_2 to methane, while acetotrophic methanogens split acateate into CO_2 and methane, and methylotrophic methanogens produce methane directly from methyl and not CO_2 (Gerardi, 2003).

A simple example of the microbial process can be seen in equation 1.1. Hydrolysis is the solubilisation of particulate organic compounds such as cellulose (1.1), and colloidal organic compounds such as proteins (1.2). These compounds are absorbed by bacterial cells, that lead to bacterial degradation which results in the production of volatile acids and alcohols (such as ethanol). The volatile acids are then converted to acetate and hydrogen gas. The degradation of acetate will lead to methane production (1.3) plus the reduction of CO_2 by hydrogen gas (1.4) (Gerardi, 2003).

Equation 1.1: Simple microbial action within a digester

Source: (Gerardi, 2003)

$$\text{Cellulose} + H_2O \text{ (hydrolysis)} \longrightarrow \text{Soluble sugars (1.1)}$$

$$\text{Proteins} + H_2O \text{ (hydrolysis)} \longrightarrow \text{Soluble amino acids (1.2)}$$

$$\text{Ethanol} \longrightarrow CH_4 + CO_2 \text{ (1.3)}$$

$$CH_4 + 4H_2 \longrightarrow CH_4 + 2\ H_2O \text{ (1.4)}$$

Due to AD being a biological process driven by microbes, there is a need to control a number of factors to provide successful and efficient digestion. The most important of these are heat, pH, and the carbon to nitrogen ratio (C:N). Microorganisms generally utilise carbon and nitrogen in the ratio of 25–30:1, and operators should aim to keep feedstock at this C:N level (Esposito et al., 2012). However, C:N ratios can be considerably lower and still provide successful digestion, if careful monitoring of the digestion process is upheld. AD can be performed under one of three temperature ranges: psychrophilic (10–20 °C), mesophilic (25–40 °C), or thermophilic (45–60 °C). Psychrophilic digestion is generally performed in large open lagoons, by letting natural process take over without any technical intervention other than filling the lagoon. This type of digestion is not commercially viable due to low biogas yield, therefore commercial on-farm AD is normally carried out under either mesophilic or thermophilic conditions. Mesophilic digestion of cattle manure requires approximately twice the retention time (the time needed for all the biogas to be extracted from the feedstock) of thermophilic digestion (typical retention time of mesophilic digestion is 20-25 days); however mesophilic digestion is still the preferred digestion temperature for on-farm AD. This is mainly due to the greater robustness of the bacterial community within mesophilic systems to temperature change (Gungor-Demirci & Demirer, 2004). Increasing the temperature of the digester to the thermophilic range will increase the rate of anaerobic conversion and consequently the overall system efficiency (Manariotis & Grigoropoulos, 2006; Zakkour et al., 2001) and potentially allows the operator to increase feedstock throughput (i.e. avoid the need to build a bigger digester). However, changing from mesophilic to thermophilic digestion should be done slowly (increasing by no more than 2 °C per day) so as to allow time for the microbial community to adapt; with a pseudo steady-state condition reached after a month and a final steady-state condition after 2 months (Cecchi et al., 1993).

Feedstock must be heated from ambient temperature to the digester temperature. The heat needed depends on the feedstock and the digester operating temperature. The wetter the feedstock, the more heat that is required per m^3 of biogas produced. The biogas is produced from the dry matter part of the feedstock, thus the operator needs to avoid heating water where possible. The digester can be heated either externally or internally. With external heating, the digestate is circulated through tubes in a heat exchanger. With internal heating, the digestate is heated by pipes or hot water heat exchangers within the digester. Somewhere between 10-33% of the biogas energy will be needed to heat a typical digester; representing the greatest running cost of the AD process (Warburton, 1997). The other main operating cost of a digester is the actual loading and unloading of the digester. The time and cost of this will heavily depend on the degree of system automation.

On an on-farm AD unit, the digestate will be handled the same way as a farmer handled their manure before the implementation of a digester. The liquid digestate can be stored in a lagoon (ideally with a cover so as to minimise rain ingress and loss of nitrogen through gaseous emissions) and the dry fraction can be piled and applied to land as required. Inclusion of an AD system on a livestock farm therefore poses comparatively little additional work to farmers and their manure management system.

1.5 AD across the globe

The UK has approximately 100 on-farm, working digesters (NNFCC, 2012) and the potential for up to 900 more, based on a combination of 200 on-farm and 700 municipal food waste fed digesters (ADAS, 2012). Germany in particular is at the forefront of AD internationally, boasting over 4000 plants (Wilkinson, 2011). There are approximately 15 in France and another 90 under construction (Peu et al., 2012). Despite the US being comparable to Europe in terms of population and landmass, the US has only 135 on-farm digesters (Parameswaran & Rittmann, 2012). China possesses over one million digesters (Wilkinson, 2011), however, most are very small household units, and would not be economically feasible in western countries. Meanwhile, a vast and developed country such as Australia has only one operational AD unit (Wilkinson, 2011).

The reason for the large difference in the uptake of on-farm AD is twofold. Firstly, in environmentally aware countries such as Germany, the fact that AD is a clean renewable energy provides a catalyst for continued and future development. Secondly, to make AD economically feasible on-farm, there is a need for large amounts of livestock manure that can be easily collected or supplementation with other feedstocks such as maize or grass silage as well as locally sourced food waste. Due to the abundance of winter housing in the EU from generous construction grants over the last number of decades, vast amounts of stored animal manure are easily accessible.

1.6 Conclusion

The thesis brings together information that is of relevance to academia, industry, and policy-makers. This is achieved by bringing together a number of different disciplines relevant to on-farm AD. This should allow both experts and non-experts to gain a deeper understanding of the practical, scientific and economic issues pertaining on-farm AD. The thesis starts in chapter 2 with an overview of the AD industry, a review of literature relevant to successfully running a digester, and the challenges to expanding the AD industry in the UK. Chapter 3 is a materials and methods chapter covering all aspects of the experimental work that was undertaken during the course of the PhD and as reported individually in chapters 4, 5 and 6. Chapter 7 brings together the findings from those experiments as well as those from other studies, to try and place an economic value on the non-market benefits of on-farm AD; specifically the agronomic and environmental benefits.

During the initial literature review, it became apparent that assigning such economic values was not possible for all the non-market benefits. For instance, although a number of studies stated that digestate is of agronomic value similar to synthetic fertilizer, there was a lack of fully replicated, scientific trials to verify such claims and this was the justification of the first experimental chapter (chapter 4). This pot trial also helped establish the potential for environmental damage through leaching of nitrate following the application of digestate and other organic and synthetic nutrient sources. This enabled an estimation of the potential of digestate to displace synthetic fertilizer and the pollution abatement this would bring. Chapter 5 studied the response of the soil decomposer community to the same amendments, so as to aid the interpretation of results from the previous chapter (e.g. in terms of nutrient dynamics). Chapter 6 expanded upon the findings of the previous two chapters through conducting a field-scale assessment of the agronomic value of digestate over a three-year period in a mixed grassland ley. Chapter 7 bases economic values on the scientific findings. By combining the results from the experimental chapters detailed previously and those from other studies, it was possible to place a range of values on the potential pollution abatement values form the digestion of 1% of livestock manure in the UK. Calculating on a 1% value would facilitate the up-scaling of results to estimate the potential value from the digestion of greater volumes of livestock manure (5%, 10%, etc.). The findings were then used to question whether the UK FIT rate currently paid for electricity generated through AD adequately reflects the non-market benefits that on-farm AD brings.

It is believed that this is the first attempt to place such non-market values on the pollution abatement benefits offered by on-farm AD. The large range in value per kWh of the proposed new FIT rate shows that a great deal more work is required to better qualify the true value of such non-market benefits. This work will help determine the added value that on-farm AD brings relative to other sources of renewable energy; values that are currently not appreciated in economic terms (Yiridoe et al., 2009). This involves multidisciplinary work and this thesis alone is a fusion of findings from microbiology, soil science, agronomy and economics. The findings of the thesis should also help bridge the divide that may be evident between academic and industry in relation to anaerobic digestion systems, and inform relevant policy-makers as to potential steps that may be taken to increase the uptake of on-farm AD in the UK

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CHAPTER 2

Understanding the process, benefits, and uptake of on-farm anaerobic digestion: a review

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Abstract

Although numerous reviews exist on specific aspects of anaerobic digestion (AD), this article brings together information from published literature on the operation, optimisation, and the agronomic and environmental benefits of AD. The review revealed notable gaps in current literature including the role of micronutrients for successful AD, and the relationship between microbial populations and digestion performance. In addition, it was apparent that procedures to deal with the problem of sodium toxicity within a digester are quite limited, thus potentially rejecting a great deal of useful biodegradable feedstock from the AD process, namely material from the seafood industry.

Keywords: Agriculture, Biogas, Greenhouse gases, Renewable energy, Waste management.

2.1 Introduction

2.1.1 Alternatives to anaerobic digestion

AD is a renewable energy technology with little public resistance (not to be confused with local resistance) due to its ability to turn a "waste" material into a source of clean energy (Anderson et al., 2013; Poeschl et al., 2010). There is also growing interest in AD due to its potential benefit for recycling nutrients back to agricultural land (Fricke et al., 2007; Salminen et al., 2001). AD also represents an alternative to landfill (Frigon & Guiot, 2010; Karagiannidis & Perkoulidis, 2009; Mata-Alvarez et al., 2000), thereby reducing greenhouse gas (GHG) emissions from the decomposition of biodegradable material (Braber, 1995).

The main alternatives to AD for dealing with organic waste material (e.g. food waste) include landfill, incineration and composting. Of these, landfill is the most undesirable option as typically there is no (or limited) energy recovery and the nutrients in the biodegradable material are not recovered for application back to land. Older landfill sites are potent sources of GHG emissions and were estimated to produce 36% of England's methane emissions in 2011 (DECC, 2012); and whilst the construction of new sites with gas collection systems reduce GHG emissions, these are costly to build. Similarly, although incineration can be used as a source of energy, and is used in some countries as a method to deal with poultry litter, the valuable crop nutrients N and S within the biomass are lost to volatilization at the high temperatures required. Although P and K are retained in the ash and can be applied back to land, the ash may include prohibitive levels of heavy metals. Hence, incineration results in the loss of valuable nutrients that farmers have to purchase in the form of synthetic fertilizer. During composting, a considerable proportion of the nutrients are lost from the initial feedstock as gaseous NH_3 and in leachate, and the process is a net user of energy. Subsequently, the total potential energy in the biomass is never recovered (Walker et al., 2009) and a commodity-rich product is not being extracted (i.e. biogas). Therefore, as a waste management strategy, AD could be considered a better option for its cleaner operation and better product range than composting (Kothari et al., 2010).

2.1.2 Types of digester

The different types of digesters typically in use today include: fixed dome plant, up-flow anaerobic filter (UAF), fixed-film reactor, anaerobic rotating biological reactor, continuously stirred tank reactor (CSTR), attached-film bioreactor, batch reactors, up-flow anaerobic sludge blanket (UASB), temperature-phased anaerobic digestion (TPAD), and an anaerobic hybrid reactor (AHR). Furthermore, there are numerous mixing systems; ranging from gas, jet, propeller or paddle. The most suitable type of digester depends upon the feedstock. The dry matter (DM) content of the feedstock determines whether the digestion is classified as dry or wet digestion. Any feedstock below 15% DM is considered wet digestion and above 15% as dry digestion. Considerable variations in dry matter can also occur within what may firstly be seen as similar feedstocks, e.g., swine, cattle and poultry manure (Moller et al., 2004).

The AD process can be carried out in either one or two stages. In single-stage AD, all four microbial metabolisms (hydrolysis, acidogenesis, acetogensis, and methanogenesis) occur concurrently within the one tank (Coats et al., 2012). Although this system offers the benefits of simplicity in design, the inefficient synchronization of the AD metabolisms within a single stage tank often leaves high-value organic matter undigested (Coats et al., 2012). Two-stage AD was developed, in part, to remedy single-stage metabolism inefficiencies (Ghosh, 1987), where hydrolysis, acidogenesis and acetogensis occur in one tank, and methanogenesis ensues in a second tank. In two-stage digestion, volatile fatty acid rich supernatant is transferred from the acidogenesis/acetogensis stage to the second stage, methanogenesis digester. Two-stage digestion allows for semi-optimization of each metabolic process thereby potentially enhancing methane production (Coats et al., 2012); although this does come at greater capital cost than a single-stage system. Using cattle manure as the only feedstock in the thermophilic range, Gannoun et al. (2007) reported increasing methane yields of 6-8% when two-stage digestion was introduced compared to one-stage digestion. A more important aspect for operators is to ensure that maximum biogas can be obtained from the feedstock after digestion.

2.1.3 Co-digestion as part of on-farm AD

Co-digestion of feedstocks is often carried out to control for factors such as the required pH level, C:N ratio, dilution of potential toxic compounds and increasing the biodegradable material content of the feedstock (Esposito et al., 2012). It became apparent in the late 1970s and early 1980s that many carbohydrate-rich feedstocks were found to require co-digestion with other feedstocks (Hills & Roberts, 1982; Knol et al., 1978). Co-digestion can increase biogas yield and improve the payback on investment in on-farm AD (Cavinato et al., 2010). For example, the digestion of cattle manure and sludge from the waste water industry has been shown to increase biogas yield by approximately 2.19 times from that of manure alone (Corton et al., 2013). Despite comparatively low yields of methane when digested alone, cattle manure is an excellent substrate to enable digestion of mixed feedstocks due to its balanced nutrient content and buffering capacity (Castrillon et al., 2013; Corton et al., 2013). The high fibrous fraction (solid manure) and high water content (slurry) of cattle manure is the reason for the low biogas yield, plus the fact that much of the methane production from the original feedstock has occurred within the cow rumen during digestion (Angelidaki & Ellegaard, 2003). Co-digestion improves the methane yield for a number of reasons including providing bacterial diversity and may supply missing nutrients by the co-substrates, leading to a more balanced nutritional composition (Jha et al., 2011; Kaparaju et al., 2008), plus the environmental and economic benefit of recycling valuable crop nutrients back to land.

A large number of substrates have been used alongside animal manure for co-digestion; including waste fruits and vegetables (Knol et al., 1978; Romano & Zhang, 2011); seafood resources (Ferreira et al., 2012); municipal solid waste (Hartmann & Ahring, 2005); brewery resources (Zupancic et al., 2012); fats, oils and grease (Long et al., 2012); and crop residues (Kavacik & Topaloglu, 2010). Food waste is potentially high in energy because of the rich organic material present in the feedstock (Digman & Kim, 2008). According to Bailey (2007), the co-digestion of fats oils and grease at a rate of 10–30% of feedstock caused a 30–80% increase in biogas. Callaghan et al. (2002) recommended that the best co-substrates for cattle manure were fruit and vegetable resources, chicken manure, and fish offal. Slaughterhouse and meat resources have been shown to prove very successful in co-digestion, resulting in high biogas yields (Alvarez & Liden, 2008; Buendia et al., 2009; Cuetos et al., 2008), although land-application of the resulting digestate within the EU would be subject to the Animal By-Products Regulations (i.e. potentially preventing land spreading). In Germany, maize silage is the dominant feedstock for on-farm AD (McEniry & O'Kiely, 2013). In the UK, approximately 67% of the utilised agricultural area is under permanent grassland (Eurostat, 2011), with some of the highest grass yields per hectare in Europe produced in the UK and Ireland (McEniry & O'Kiely, 2013). Grass silage is known to be a useful feedstock for AD, however, a late harvest will decrease digestibility and increase fibre component which will have a negative effect on biogas production (McEniry & O'Kiely, 2013).

New research into crude glycerine from the biodiesel industry (Astals et al., 2011; Castrillon et al., 2013) has provided some very promising results as a co-substrate for AD. Each year, millions of tonnes of crude glycerine biomass resources are produced by the biodiesel industry, exceeding the present commercial demand for such a commodity (Siles Lopez et al., 2009). Crude glycerine is easily storable over long periods of time, is readily digestible (Robra et al., 2010), and easily transportable. There would be a number of benefits to the co-digestion of crude glycerine, above increasing the biogas yield. The biodiesel industry is often associated with negative aspects; in particular that biodiesel production requires productive agriculture land (thus raises the issue of whether land is used for production of food or fuel). Co-digestion of crude glycerine would improve the environmental benefits of biodiesel by further reducing GHG emissions from fossil fuels, with the additional benefit of returning nutrients back to land.

2.2 Controlling factors in successful digestion

2.2.1 pH

Along with temperature and C:N ratio of the feedstock, pH is the most important variable for successful AD, with a pH of < 6.1 or > 8.3 resulting in inefficiency, and even failure of a digester (Esposito et al., 2012; Lay et al., 1997). Changes in pH affect the digestion process because the hydrogen ion concentration has a direct influence on microbial growth (Jha et al., 2011) therefore the optimal pH range for AD varies for each metabolic stages. For instance, the optimum pH for hydrolysis and acidogenesis ranges from 5.5 to 6.5 (Arshad et al., 2011); while the ideal pH for the methanogenesis ranges from 6.8 to 7.6 as methanogen growth rate is greatly reduced below pH 6.6 (Mosey & Fernandes, 1989). Rapid hydrolysis of high volatile solids (VS) feedstocks may lead to acidification of a digester and the consequent inhibition of methanogenesis (Ward et al., 2008). Naturally acidic or basic feedstock can affect pH, and high concentrations of volatile fatty acids (VFAs) will lower pH (Jha et al., 2011). Co-digestion and sourcing homogeneous material on a regular basis may help to control pH and avoid sudden changes. The addition of an alkaline buffer will also help ensure stable performance (Hills & Roberts, 1982; Knol et al., 1978).

2.2.2 Ammonia

Ammonia is essential for the microorganisms involved in the AD process, as well as contributing to the stabilization of pH within a digester (De la Rubia et al., 2010). Ammonical nitrogen exists in two forms, ammonia ions (NH_4^+) and free ammonia nitrogen (NH_3). The formation of NH_3 is a result of the degradation of proteins and amino acids present in the initial feedstock (Bayr et al., 2012; Rao et al., 2008). Higher temperatures produce higher concentration of free ammonia (Angelidaki et al., 1993; Bayr et al., 2012). A major factor in preventing digester failure is avoiding ammonia inhibition, which endangers the process stability (Desloover et al., 2012). Ammonia inhibition leads to reduced methane yields (Borja et al., 1996; Nielsen & Angelidaki, 2008; Sossa et al., 2004) by as much as 50% with concentrations of ammonia above 5 g N per litre (El Hadj et al., 2009). Conversely, excess ammonia has a negative effect on the hydrolysis stage of digestion (PoggiVaraldo et al., 1997) and free ammonia is highly toxic to methanogens (De la Rubia et al., 2010).

If the concentration of free ammonia is above a critical level, the operator can induce a drop in temperature which will have a positive net result; likewise if ammonia levels are low, then an increase in temperature can be induced (Angelidaki & Ahring, 1994). Ammonia inhibition is not really an issue with digesters fed cattle manure only. Nevertheless, it can be of major concern if co-digesting manure with feedstocks containing high levels of nitrogen, such as chicken manure. If possible, the best option for dealing with feedstock with ammonia levels above 5 g N per litre is co-digestion with feedstock of high carbon and low nitrogen, e.g. chicken manure with paper waste. If co-digestion is not possible there are a number of options available for stripping nitrogen to reduce overall levels (Mousavi et al., 2012) including: air stripping (Rao et al., 2008), ultrasound (Wang et al., 2008), electrochemical conversion (Lei & Maekawa, 2007), biological denitrification (Wett & Rauch, 2003), and microwave radiation (Lin et al., 2009).

2.2.3 Sodium

Although sodium is essential for bacterial growth (Dimroth & Thomer, 1989), high sodium concentrations increase osmotic stress that can result in decreased cell activity and cell plasmolysis (Uygur, 2006), leading to inhibition of AD. Research is abundant on the AD of high saline feedstocks ranging from tannery industries (Lefebvre et al., 2006), seafood processing (Omil et al., 1995) and oil and gas production (Ji et al., 2009). Still, solutions to the problem of inhibitory high levels of sodium are limited, with the addition of grass residues shown to be the most cost-effective (and environmentally-friendly) material to decrease sodium toxicity (Suwannoppadol et al., 2012). The lack of methods to deal with high levels of sodium in certain feedstocks mean that large volumes of methane-rich biodegradable resource from the seafood industry is largely unavailable for AD at present due to the high saline content, meaning it is disposed of via less efficient ways.

2.2.4 Micronutrients

The control of the macronutrients carbon, nitrogen, phosphorus and sulphur is widely considered essential for efficient digestion. However, there is far less of an appreciation of the significance of micronutrients, such as iron, nickel, cobalt, selenium and tungsten. These micronutrients are critical for the microorganisms involved in digestion (Takashima & Speece, 1990). As of yet, no specific optimal concentrations of micronutrients have been determined for efficient digestion and figures quoted in the literature are highly variable (Table 2.1).

Table 2.1: Micronutrient requirements for anaerobic digestion. Figures with the units mg/l are expressed in terms of wet weight, while quantities with mg/l* are on a dry weight basis.

Micronutrient	**Recommended quantity**	**Reference**
Cobalt	0.15 to 0.58 mg/l	Lo et al. (2012)
	3 mg/l*	Kayhanian & Rich (1995)
Nickel	0.801 to 5.4 mg/l	Lo et al. (2012)
	5 to 25 mg/l*	Kayhanian & Rich (1995)
Selenium	> 0.1 mg/l*	Kayhanian & Rich (1995)
Tungsten	> 0.1 mg/l*	Kayhanian & Rich (1995)
Iron	1000 to 5000 mg/l*	Takashima & Speece (1990)

Although microbial populations within AD systems have been well-characterized (Nettmann et al., 2008), the relationship between microbial populations and digestion performance have not yet been well established (Yue et al., 2013). Whilst manure-fed digesters are often very successful, there are a number of problems in relation to the digestion of the biodegradable fraction of municipal solid waste due to high protein and fat content, as well as possible nutrient deficiencies. One possible explanation may be that although all the macronutrients are present; the micronutrients that exist in manure could well be absent (De la Rubia et al., 2010; Banks et al., 2008).

2.3 Benefits of on-farm anaerobic digestion

2.3.1 Destruction of weed seeds

Certain weed seeds are consumed by animals and are then excreted by the animal or are spread when raw manure is applied to land. Weeds are a problem for farmers because they take up available nutrients that are intended for crops, restrict the growth of the desired crop, and have reduced (or no) feed value. Most farmers control weeds through the application of chemicals, however, organic farmers are unable to deal with the problem of weeds by herbicide application. It has been demonstrated that a number of weed seeds are killed during AD (Jeyanayagam & Collins, 1984); especially at thermophilic temperatures (Westerman et al., 2012b).

Digester type is also significant in regards to weed seed destruction, with greater survival probability of *Abutilon theophrasti* (velvet weed) and *Malva neglecta* (common mallow) in batch digesters, while *Persicaria lapathifolium (Polygonum)* (curlytop knotweed) survives better in CSTR digesters (Westerman et al., 2012a). Greater survival of weed species (above 50%) are seen with hard seeds during mesophilic digestion compared to those that lack a water-impermeable layer and freshly harvested seeds (Westerman et al., 2012b). Studies have shown *Rumex obtusifolius* (broad leaf dock) and *Lycopersicon lycoperscium* (tomato) seeds to be completely destroyed after 14 days o thermophilic digestion (Engeli et al., 1993). Westerman et al. (2012a) reported seed destruction after 2 days for *Abutilon theophrasti* (velvetleaf), 5.8 days for *Malva neglecta* (common mallow), 19.7 days for *Chenopodium album* (fat hen) and 1.2–9.1 days for *Fallopia convolvulus* (wild buckweed) during AD.

2.3.2 Valuable by-products

Land-application of digestate helps close the nutrient cycle and decrease dependency on synthetic fertilizer. However, it is not always possible to apply digestate to land within the vicinity of the AD unit. A number of factors may hinder the application of digestate to land, ranging from the feedstock (e.g. slaughterhouse waste) to environmental regulations (e.g. nitrate vulnerable zones).

If digestate cannot be applied to local land, the best economic and environmental use of digestate should be encouraged. Separation of the digestate is paramount, as it is easier to develop a market for the separated products, liquid digestate and dry fibre digestate. Liquid digestate is the most nutrient-rich component of the digestate and can be successfully used as a nutrient growth medium for algae, which may then be used for biofuel production in specialised set-ups (Chen et al., 2012; Wilkie & Mulbry, 2002). However, finding a market for the dry fibre fraction, with its lower nutrient value, can be difficult. Some current examples are for animal bedding to replace straw or as a peat replacement; however, the economic returns on these can be very low. New research has forecast both an interesting and potentially profitable market in the conversion of dry fibre for use in ethanol production (Teater et al., 2011). Digested AD fibre has more cellulose (32%) and less hemicelluloses (11%) than undigested cattle manure (Yue et al., 2010), making it very useful product for ethanol production. Yue et al. (2010) illustrated that the 109 million dry tonnes of solid cattle manure available annually in the US could generate 57 million tonnes of AD dry fibre and produce more than 6.32 billion litres of ethanol, which would equate to approximately 111 litres of ethanol per tonne of dry fibre. If these figures are applied to the 35 million dry tonnes of solid cattle manure produced yearly in the UK, there is the potential for 1,844 million litres of ethanol.

2.3.3 Greenhouse gas reduction

The AD of animal manure has been shown to reduce GHG emissions as it replaces fossil fuels for energy conversion (De Vries et al., 2012) and the production of synthetic fertilizer. If off-farm biodegradable material is to be imported onto farm AD systems, such economic and environmental benefits must be weighed against costs (e.g. emissions from transport of feedstock) so that there is a total net reduction in GHG emissions. For a 1 MW digester, it was found that with high energy efficiency and resource recovery, an operating distance of 192 km could yield a 35% carbon dioxide (CO_2) saving, and a 50% CO_2 saving was possible at a radius of 70 km (Capponi et al., 2012). In addition, there is a possible 90% resource saving when comparing bio-based energy to conventionally produced electricity (De Meester et al., 2012), again best case scenario.

2.4 Issues that hinder the uptake of Anaerobic Digestion

2.4.1 Lignin

If lignocelluloses-abundant materials (such as straw from cereal production) could be digested, this would dramatically increase the potential feedstock to AD units (Zeng et al., 2007). However, a high lignin concentration reduces the biodegradability of certain types of biomass, slowing the hydrolysis step of AD and limiting the production of methane (Frigon & Guiot, 2010). The complex lignocelluloses structure limits the accessibility of the sugars in cellulose and hemicelluloses, impacting on the methanogenesis stage (Nkemka & Murto, 2013). Hence, pre-treatment of lignin material is needed to gain access to the sugars bound in lignocelluloses-abundant feedstocks (Alvira et al., 2010). One method used to extract these sugars is steam pre-treatment in the presence of dilute acid, which results in efficient lignocelluloses hydrolysis and sterilisation; however, the building and operation of a steam pre-treatment unit is expensive (Nkemka & Murto, 2013). A vast majority of the excess heat is dumped from on-farm AD units, due to no use for the heat. Therefore if this dumped heat was used for steam pre-treatment this would substantially reduce the energy input requirement (Ljunggren & Zacchi, 2010).

2.4.2 Hydrogen sulphide (H_2S) within biogas

Sulphur is an important nutrient for successful AD, and the correct carbon to sulphur ratio (C:S) should be around 40:1, to help limit the concentration of H_2S in raw biogas (Peu et al., 2012). Hydrogen sulphide is pungent as well as toxic, and can damage equipment such as combined heat and power engines by causing corrosion. To prolong the working life of such machinery, the H_2S concentration of biogas is recommended to be lower than 500 ppm (Bayr et al., 2012; Peu et al., 2012; Ryckebosch et al., 2011). There are numerous measures to lower H_2S in biogas. One such method is the addition of chemical compounds such as metal ions, while another is the introduction of inhibitor producing microorganisms, or sulphide scavengers (Bayr et al., 2012; Isa & Anderson, 2005). Alternative approaches to chemicals include the creation of micro-aerobic conditions in the gas storage facility by adding 2–6% air to the biogas (Peu et al., 2012). Oxygen will encourage the growth of chemoautotroph microorganisms, which oxidises H_2S into elemental S and SO_4^{2-} (Diaz et al., 2010). Another option is to allow small amounts of oxygen under controlled conditions into the roof of the digester (i.e. still keeping the digester anaerobic), thus as the H_2S converts to S and SO_4^{2-}, it will fall back into the tank and be extracted with the digestate, allowing the sulphate to be applied to land. The amount of oxygen added to the biogas should be carefully controlled and limited to prevent explosive gas mixtures and biogas dilution to ensure satisfactory biogas combustion (Rahmouni et al., 2003).

2.4.3 Problems with the uptake of on-farm Anaerobic Digestion

In a recent study of UK farmers, it was stated that the most common barrier to uptake and operation of on-farm AD besides capital funding and planning permission, were fears about technical problems such as generators and feedstock pumps not working properly (Bywater, 2011). Planning permission can be difficult to obtain due to public perception, legal barriers about biomass resources allowable for digestion etc. Further consultation between farmers groups, AD industry and planning authorities is required to make this part of the AD process less problematic. Unlike other renewable energies, AD is a live biological process and thus not simply a “plug in and wait” technology. Other issues that are of concern to farmers is an increase in vermin due to storage of organic material, but this can be reduced dramatically with proper management, and is only of real concern if biodegradable material (particularly food waste) is brought onto the farm for digestion. From discussion with farmers who have applied and/or installed an AD unit, they report that the two main concerns they faced from the local community were in relation to gas explosion and excess road traffic.

2.5 Private and public investment in AD

2.5.1 Farmer’s decision to invest in AD

Farmers considering investing in AD must be aware of the returns from the digestion of different feedstock. Sourcing adequate amounts of the right feedstock/s is paramount. Table 2.2 reports the potential biogas yield from the manure of different farm animals and the amount of animals required to produce a tonne of manure for digestion. Anaerobic digestion is an expensive technology and therefore farmers must know what the rate of return on the investment will be before proceeding with a project.

Table 2.2: Potential biogas output from biomass resources generated by livestock agriculture. Source: (EPA, 2002)

Feedstock	No. of animals to produce 1 tonne/day	Biogas yield (m^3/tonne)	Energy value (MJ/m^3 biogas)
Cattle manure	20-40	25	23-25
Pig manure	250-300	26	21-25
Laying hen litter	8,000-9,000	90-150	23-27
Broiler manure	10,000-15,000	50-100	21-23

Note: Figures should be regarded as indicative values only

As well as revenue from energy sales, AD offers additional private savings that are often overlooked by farmers in their investment decision and are neither well represented in current literature. Figure 2.1 provides a visual representation of the private benefits that a farmer may receive from the introduction of AD to their farming system. Digestate has the potential to reduce the quantity of synthetic fertilizer required due to higher crop yields following application of digestate compared to undigested manure (Walsh et al., 2012); and the AD process kills weed seeds (Westerman et al., 2012a; Westerman et al., 2012b) hence may reduce the costs of herbicide use. Labour cost is the only factor that has a negative economic effect for a farmer who adopts AD technology in that it will require frequent (or if large, continuous) labour for running and upkeep of the digesters; however many modern systems work to a high degree of automation.

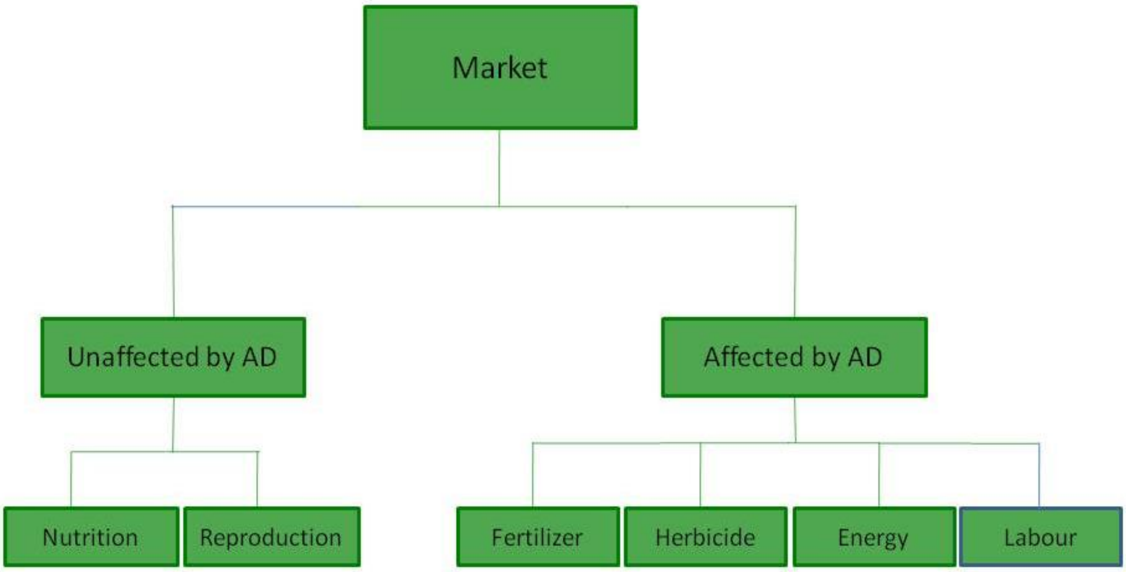

Figure 2.1: Private benefits of on-farm AD to the farmer.

2.5.2 Government investment in anaerobic digestion

Like all renewable energy technologies, AD is supported by the UK government via a Feed-In-Tariff (FIT) or Renewable Obligation Certificates (ROC) payment for the production of renewable electricity on top of what the suppler may receive from a utility company for the sale of electricity to the grid. A similar model is applied in over 75 jurisdictions around the world (Kim & Lee, 2012). However, from an environmental economics standpoint the government incentive should theoretically cover all the non-market or social benefits brought about by the introduction of the technology that is being subsidised. Non-market simply means that the benefits are not traded in a conventional market place, with buyers and sellers. Like all industries, livestock agriculture suffers from negative externalities associated with milk and meat production. If AD can reduce these negatives it should be financially supported for the additional benefits above replacement of fossil fuel.

Figure 2.2 separates the reported environmental benefits of AD into water and GHG benefits, with the baseline being no pollution. Anaerobic digestion effectively destroys pathogens (Sahlstrom, 2003; Saunders et al., 2012) and hence if greater volumes of livestock biomass resources were treated via AD, this may lead to reduced numbers of water-borne infections and the associated costs to the economy. Recent research suggests that there may be less potential for NO_3^- leaching from the land-application of liquid digestate compared to synthetic fertilizer (Walsh et al., 2012). BOD and chemical oxygen demand (COD) are dramatically reduced during digestion (Anon, 2003; Clemens et al., 2006), thus even if digestate were to contaminate waterways, the effects of BOD and COD would be reduced.

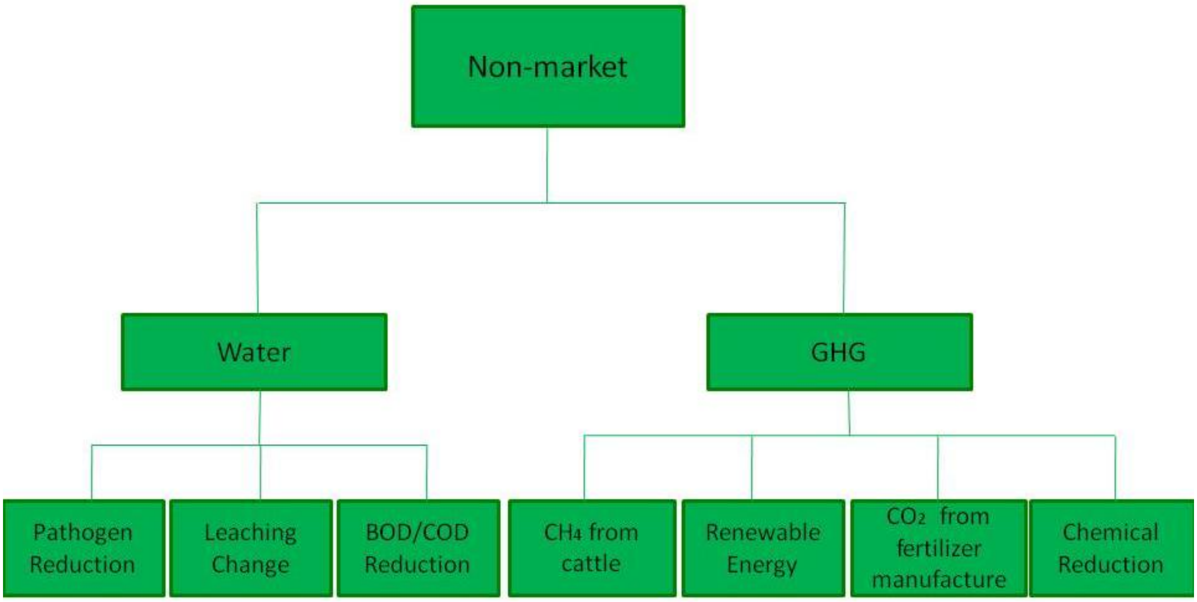

Figure 2.2: Social benefits of AD to farmers and the wider community

The other main environmental positive externality associated with the introduction of AD is the reduction in GHG emissions from livestock sector of agriculture. As can be seen in Fig 2.2, there are four variables affected in relation to the reduction of GHG: methane (CH_4) from cattle, renewable energy, CO_2 from fertilizer manufacture, and reduced use of chemicals. CH_4 is produced in the rumen of ruminants as they digest their food. Dairy cattle are the largest emitters of methane, emitting approximately 100 kg CH_4 via enteric processes and 15.9 kg CH_4 through manure management per annum, with non-dairy cattle producing approximately half that (Hynes et al., 2009). As methane has 21 times the global warming potential of CO_2 (IPCC, 2007) any reduction can have significant positive impacts. During AD, the methane in the manure is captured and thus is prevented from escaping to the atmosphere, under normal storage conditions. The extra bonus of this is the second box of the nest, "Renewable Energy". The methane that is captured is used as a source of electricity and heat production, displacing fossil fuels. Further, if AD is implemented on-farm and crop yields are higher than undigested manure, the decreased need for synthetic fertilizer to meet crop requirements and herbicide application discussed previously would reduce the GHG emissions that occur during the production of such products.

2.6 Conclusions

AD of any biodegradable material is considered an effective and environmentally friendly way to obtain the maximum use value from biodegradable commodities that may often otherwise be deemed as waste products with limited value. However, AD is an inherently complex biological process. As such, there is a need to increase our understanding of the factors that govern its effectiveness; be they biological, chemical, or technical parameters. This review has highlighted some of the knowledge gaps that need to be addressed. It has also highlighted the need for farmers and policy-makers to consider the wider non-market environmental benefits of AD both to the farmer at a private level, and the general public at a social level. This suggests that AD may at present be an undervalued technology that should be prioritised for development.

2.7 References

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Chapter 3

Materials and methods

3.1 Experimental design of the agronomic trials

3.1.1 Chapters 4 and 6

There are two agronomy-based chapters in this thesis, chapters 4 and 6. Chapter 4 was a controlled greenhouse pot experiment, while chapter 6 was a three year field trial experiment. In both cases, liquid digestate (LD), undigested slurry (US), and two types of synthetic fertilizer (N and NPK) were applied to pots and field plots, respectively, at a rate equivalent to 100 kg N ha^{-1} of each fertilizer type. To represent farmer practice, the first harvest was undertaken six weeks after the initial application. A second fertilizer addition of 50 kg N ha^{-1} was then applied one week post the first harvest of that year. The application rate for each was normalised for nitrogen, based on mineral N (ammonium N) values and total nitrogen content for the synthetic fertilizers. There were three harvests in total over one growing season in the work detailed in chapter 4; while chapter 6 had two harvests every year for three years.

For the pot trial experiment (chapter 4 and 5) soil (Eutric Cambisol) was collected from a pasture-based system at a conventional (non-organic) farm (CS) (Bangor University's Henfaes Research Station; 53°14'05''N, 4°00'50''W) and an organic farm (OS) (Wrexham; 53°08'16"N, 2°90'48"W). The soil was collected to a depth of 10 cm, passed through an 8 mm sieve and analysed in the laboratory within 24 h of collection. OS was collected from an organic dairy farm (for the past 12 years) under permanent pasture (*Lolium perenne* L., *Trifolium repens* L.). CS was collected from a conventionally managed sheep farm under permanent pasture. The experimental field site was located on the same farm as the conventional soil used in the pot trial (Bangor University's Henfaes Research Station; 53°14'05''N, 4°00'50''W). The sward contained a mixture of perennial rye grass (*Lolium perenne* L.) and white clover (*Trifolium repens*, L.) and was previously subject to sheep grazing (ca. 15 ewes ha^{-1}).

For both experimental chapters, undigested slurry (US) was collected from an organic dairy farm from cows fed 50% grass silage and 50% whole crop silage (barley and peas). Added to the silage in concentrated form was rolled wheat, rolled beans, maize flour and soya expeller (concentrate to silage ratio was 20:80 on a dry matter basis). Digestate, in liquid form (LD) and dry fibre form (DFD) was collected from the AD unit on the same farm. The AD unit is a 1000 m^3 mesophilic (38 °C) system, continually stirred digester with a retention time of 25 days and fed with mainly cow slurry, and the US was bedded on a mixture of paper and sawdust. The digestate was separated mechanically after leaving the digester and only the liquid fraction was used in the pot trial experiment, will both LD and DFD were used in the field trial. All samples were analysed within 24 h of collection.

US, and LD generated from AD of the slurry were collected from a dairy farm located in Wrexham, NE Wales. The US was collected directly from a cattle housing shed while the LD was collected from a 1000 m^3 mesophilic (38 °C), continually-stirred tank reactor (CSTR) with a retention time of 25 d. Mainly on-farm cattle slurry had been fed as feedstock into the CSTR for the previous six months. After digestion, the digestate was mechanically separated into liquid and dry fibre fractions, of which only the liquid fraction was collected for use in the experiment.

3.1.2 Pot Trial (chapter 4 and 5)

A fully randomized pot trial experiment (*n* = 100) was set up in a greenhouse under controlled conditions with mean weekly temperature of 23 ± 3 °C. Pots of average size for pot experiments were used (150 mm diameter; 1.2 l volume), and were filled with 1.3 kg of either OS or CS (dry bulk density, OS: 0.87 g cm^{-1} and CS: 0.85 g cm^{-1}) and fitted with Rhizon® suction samplers (Rhizosphere Research Products, Wageningen, The Netherlands) at approximately 100 mm depth for collection of soil solution. Soil moisture was maintained at 70% field capacity throughout the experiment by watering up to a known weight for each pot. Half the pots for each soil type were seeded with perennial ryegrass only (*Lolium perenne L.*), and the other half with a mixture of perennial ryegrass and white clover (*Trifolium repens L.*) at a rate equivalent to 40 kg of grass ha^{-1} and 12 kg of clover ha^{-1} (Emorsgate Seeds, Norfolk, UK). The final setup therefore consisted of four sub-groups (a) organic soil *Lolium perenne*, (b) organic soil *Lolium perenne-Trifolium repens* mix (c) conventional soil *Lolium perenne*, (d) conventional soil *Lolium perenne-Trifolium repens* mix, with five fertilizer treatments and five replicates for each treatment (totalling 4 × 5 × 5 = 100 pots). The five fertilizer treatments consisted of (i) an unamended control (C), (ii) undigested slurry (US), (iii) liquid digestate (LD), (iv) mineral N (NH_4NO_3) fertilizer (GrowHow Ltd., Cheshire, UK), and (v) a mineral NPK fertilizer (N:P:K = 21:8:11) (Yara Ltd., Lincolnshire, UK). Pots were regularly re-randomized during the trial.

After the grass and grass/clover mix had established (4 weeks after planting), 100 kg N ha^{-1} was applied to all treatments except the control, and this event defined the initiation of the experiment (time = 0). After 5 weeks (W5) the first herbage harvest was taken as well as soil samples for analysis of the decomposer community. One week later, a further 50 kg N ha^{-1} was applied, and on week 11 (W11) the second herbage harvest was taken and soil collected for analyses. The final herbage harvest and soil samples were collected on week 16 (W16). At the time of each harvest, soil samples were collected from the top 5 cm of three randomly selected pots. These samples were homogenised and combined to a composite sample that was used to estimate bacterial and fungal growth. For logistical reasons, four randomly selected replicates out of the total five were used for the microbial analyses at week 16.

3.1.3 Field Trial (chapter 6)

Five different fertilizer treatments were applied to 2 × 2 m plots (n = 4), organised in a randomized design. These included: a no fertilizer control (C); undigested cow slurry (US); the liquid fraction of anaerobically digested cow slurry (liquid digestate, LD); the dry fibre fraction of anaerobically digested cow slurry (dry fibre digestate, DFD); synthetic N 34.5% fertilizer (N; ammonium nitrate) and a synthetic NPK 21.8.11 (NPK) compound fertilizer. Over a three year period, six above-ground vegetation harvests were performed on the plots with two harvests taken per year (May-June and August-September). Weather patterns were recorded over the trial period and total monthly rainfall and mean monthly temperature.

With the exception of the first harvest, the harvested material was manually separated to determine the proportion of grass and clover in the sward. Only the plant biomass within the central 1 m^2 of the plots was quantitatively evaluated to avoid potential edge effects. Soil samples were taken from each plot at the very beginning of the experiment, after the third harvest and again at the end of the final harvest in year 3. All harvested plant material was weighed wet, and then a 300 g subsample was removed, dried at 85 °C for 48 h, and reweighed. Crop nutrient analysis was undertaken in both harvests in year three of the trial to determine shoot total nitrogen and carbon content. Protein content was calculated by multiplying the nitrogen reading by 6.25, which is the industry standard, however, this tends to overestimate the true protein of feedstocks (Sriperm et al., 2011). Digestibility was calculated using the MAD fibre content of each sample (Yara, 2013).

3.1.3.1 Soil and fertilizer nutrient analysis

The same methods were practiced throughout the duration of the experiments for all nutrient analyses of soil and organic fertilizer. These methods are proven over many years and are the ones employed by the whole lab group. Moisture content was determined after drying samples at 105 °C for 24 h. Nutrients from soil and fertilizer were extracted in deionised water at a ratio of 1:5 (w/v). Although KCl extraction is often used for extraction of soil samples, deionised water was used at the very start of the research work for all extraction and this was done throughout for consistency. Samples were shaken (250 rev min^{-1}, 1 h, 20 °C), centrifuged for 15 min (4,000 *g*), filtered (Whatman no. 42), and the supernatant recovered for analysis. NO_3^- NH_4^+ and P were determined colorimetrically (BioTek®, Vermont, NE) using the methods of Mulvaney (1996), Miranda et al. (2001), and Murphy & Riley (1962), respectively. Major cations (K^+, Na^+ and Ca^{2+}) were analysed using a model 410 flame photometer (Sherwood Scientific, Cambridge, UK). Total organic carbon and nitrogen were measured using a CHN2000 elemental analyzer (Leco Corp., St Joseph, MI) and dissolved organic carbon and dissolved nitrogen were measured using a TOC-V CHS analyzer (Shimadzu Corp., Kyoto, Japan). Undiluted slurry and digestate were used for the determination of electrical conductivity (EC; Jenway 4010 EC meter) and pH (Hanna Instruments pH 209 pH meter) whereas a 1:5 (soil: water, w/v) extract was used for soil.

3.1.3.2 Nutrient levels in soil solution

Sterile vacuum tubes were attached for 24 h to the Rhizon® samplers at weekly intervals throughout the experiment, one hour after a watering event. Volumes of soil solution collected were subsequently measured and concentrations of NO_3^-, NH_4^+ and P determined. Nutrient sampling was stopped ten weeks post the first fertilizer application as after this time quantities of NO_3^-, NH_4^+ and P in soil solution were all below detection levels (< 0.1 mg l^{-1}). These data were pooled so that mean concentrations of nutrients in soil solution could be determined.

3.2 Microbial analyses (chapter 5)

3.2.1 Bacterial growth

Bacterial growth was estimated using leucine (Leu; Kirchman et al., 1985) incorporation in bacteria extracted from soil using the homogenization / centrifugation technique (Bååth, 1994) with modifications (Rousk & Baath, 2011) (Bååth et al., 2001). Briefly, 2 µl of radiolabelled Leu ([^{3}H]Leu, 37 MBq ml^{-1}, 5.74 TBq mmol^{-1}, Perkin Elmer, UK) combined with non-labelled Leu was added to each tube, resulting in 275 nM Leu in the bacterial suspensions. The amount of Leu incorporated into extracted bacteria per h and g soil was used as a measure of bacterial growth. Although this is a relatively new way of determining bacterial growth in terrestrial systems, Dr. Johannes Rousk has co-authored numerous published papers on the method. We had the chance to work with Dr. Rousk in this area and capitalize on the novel opportunity to apply this method into this subject area.

3.2.2 Fungal growth and biomass

Fungal growth was assessed using the acetate incorporation into ergosterol method (Newell and Fallon 1991) adapted for soil (Bååth, 2001) with modifications (Rousk & Baath, 2011; Rousk et al., 2009). Briefly, 1-[^{14}C]acetic acid (sodium salt, 7.4 MBq ml^{-1}, 2.04 GBq mmol^{-1}, Perkin Elmer, UK) combined with unlabelled sodium acetate resulting in a final acetate concentration of 220 µM was added to a soil solution and incubated for 4 h at 22 °C without light. Ergosterol was then extracted, separated and quantified using HPLC equipped with a UV detector (Rousk & Baath, 2007). The fungal biomass was estimated assuming 5 mg ergosterol g^{-1} fungal biomass (Joergensen, 2000) (Ruzicka et al., 2000). The eluent containing the ergosterol peak was collected and the amount of incorporated radioactivity determined. The amount of acetate (Ac) incorporated into fungal ergosterol (pmol h^{-1} g^{-1} soil) was used as a measure of fungal growth. Again, Dr. Rousk is well renowned in the literature for his research using this technique to look at fungal growth and has proven its validity through the publication of over 30 research papers.

3.3 Plant yield and GHG analysis

3.3.1 Plant yield analysis (chapter 3 and 6)

In the pot trial, three herbage harvests were taken at weeks 5, 11 and 16 after the first application of fertilizer. The shoots were harvested to 2 cm above the soil surface. After harvesting, all samples were weighed fresh, dried for 48 hours at 85 °C, and then reweighed. The first harvest was ground to determine shoot total nitrogen and carbon content; protein content was calculated by multiplying the nitrogen reading by 6.25 and digestibility was tested as described in Omed *et al.* (1989). In the field experiment, two harvests were taken every year; samples were over dried as in the pot trial. For analysis, all harvests from harvest 2 onwards were manually separated into grass and clover contingent, and a 300 g subsample was oven dried for further analysis. C and N were analysed in the same manner as soil and fertilizer, while in the final year digestibility was determined using the MAD approach (Yara, 2013).

3.3.2 Predicted ammonia emissions and nitrate leaching (chapter 6)

Fertilizer application rate in tonnes, dry matter content, total nitrogen and total NH_4^+ of the undigested slurry, liquid digestate and dry fibre digestate were inputted into the computer programme MANNER v4.0 (Chambers et al., 1999). MANNER is a software application that allows the user to determine the potential N volatilisation and leaching of organic fertilizer for different regions of the UK. A 3-year average of the fertilizer value (nutrient content) was used (rather than three individual years) to determine what the potential ammonia emission reduction and leaching may have been from all three organic fertilizers over the experimental period.

3.4 Statistical analyses

3.4.1 Chapter 4, 5 and 6

There were two statistical analyses programs used in this thesis. Treatment differences in the microbial variables and the plant yield data were compared by 3-way ANOVAs (JMP 7.0 for Mac, SAS Institute Inc., Cary, NC, USA), using soil (organic or conventional), crop (grass or grass/clover) and fertilizer (control, US, LD, N, and NPK) as fixed factors. Tukey's HSD pair-wise comparisons ($p < 0.05$) were used to determine differences between fertilizer responses, in chapter 5, as Dr. Rousk was not familiar with SPSS v. 18 and thus a statistical package was chosen that all authors were comfortable with. In all other chapters, SPSS v. 18 (IBM UK Ltd., Hampshire, UK) was used. A homoscedastic two-tailed T-test was used to determine differences between soils and between the two organic fertilizers for nutrient content. For analysis of crop yield data, total yield from all three harvests was used and subject firstly to a one-way ANOVA to determine differences within each sub-group, with treatment as the factor. Then a 3-way ANOVA was performed, using soil (organic or conventional), crop (grass or grass/clover) and fertilizer (control, US, LD, N, and NPK) as fixed factors, to determine if results were continuous for all and individual subgroups. The same analysis was used for nutrients in soil solution with data for mean weekly concentrations of nutrients, for a 10 week period. Nutrients in soil solution were also subject to repeated measures ANOVA to determine if difference existed in potential nutrient loss on a weekly basis. Post-hoc tests were carried out on all ANOVAs using Tukey HSD test at the level $p < 0.05$. For the crop yield data in chapter 6, total yield from all harvests were used and subject firstly to a one-way ANOVA to determine differences within each sub-group, with treatment as the factor. The same analysis was used for carbon, nitrogen and digestibility tests. Post-hoc tests were carried out on all ANOVAs using Tukey HSD test at the level ($p < 0.05$).

3.5 Economic valuation (chapter 7)

There are a number of economic tools that may be used for valuing environmental non-market benefits. These range from hedonic valuation, travel cost method, willingness to pay and contingent valuation. Hedonic valuation is mainly used for odour valuation, and is where homes in a certain region with the same attributes (i.e. number of bathrooms etc.) are compared in price, with the only difference being the presence of foul odour at one of the locations. Travel cost method, is best explained by an example. If a person travels 1 hour to go fishing, then the cost of fuel and potentially lost wages etc. can be added together to work out the value of a clean lake/river with fish. Contingent valuation or as often referred to, stated preference, is probably the most controversial of all techniques as the researcher asks the participant how much something is worth to them in monetary value; and these valuations may be skewed by people's perceptions and ideologies.

Economic valuations are increasingly used as a way of elucidating the relative weight of different ecosystem services. It allows the relative benefits (economic generation or savings) and disbenefits (costs) to be weighed so that the net benefits of a system can be valued economically. In the context of AD, although the technology has often been proposed to offer a number of benefits, these have not seemingly been quantified economically. As a result, AD may thus be undervalued and the full potential of the technology unrecognised. The work carried out in chapter 7 attempted for the first time to collate data from different studies that have used the aforementioned valuation tools to estimate the value that on-farm AD offers. In difference to conventional approaches, the chapter took a wide angle by attempting to value the wider non-market benefits of the technology and extrapolate country-wide. It was felt that this approach would help determine whether government incentives for AD systems are proportionate to the benefits that AD could deliver.

3.5.1 Valuation tools used

There were seven variables valued in this thesis: GHG reduction; synthetic fertilizer replacement; nutrient leaching; biological and chemical oxygen demand reduction; pathogen reduction and odour. The feedstock was limited to livestock manure so as to focus on on-farm AD. A number of different valuation tools were used to arrive at the final non-market value. In each section, the non-market benefit of AD was reviewed and the available data used to estimate its value. Data were predominantly obtained from peer-reviewed sources, with additional UK-specific data from government organisations. The economic value was estimated for each per m^3 of livestock manure digested (as is standard in agriculture, 1 m^3 of manure/digestate was treated as equivalent to 1 tonne), relative to land-spreading that manure in undigested form on fields, which is the current practice. Where possible, we break down the non-market benefits from the introduction of AD to an increase in the current FIT rate per kWh of electricity produced. GHG reduction can be valued using carbon market prices, such as the European Union Emissions Trading Scheme (EU ETS), abatement costs, or the estimated social damage cost of emitting CO_2. The average EU ETS C prices for 2011 was £13 per tonne, with marginal abatement cost for the UK estimated to be £52 per tonne of C abated (DECC, 2011). This is below the mean of approximately £60 per tonne reported by Tol (2005) from peer-reviewed journals (February 2013 exchange rate). As there are three figures for per tonne of CO_2e, the highest (£60; Tol, 2005) and lowest (£13 EU ETS) figures were both applied to the available scientific data to give a range of values. All figures for GHG emissions were arrived at using either the Tol (2005) or EU ETS values.

To determine the montary beneftis from potential reduction in leaching, Pretty et al. (2003) estimates were used. They determined that the annual cost of N leaching to waterways in the UK is between \$105m – \$160m. Included in this valuation is \$7.17m – \$11.19m for GHG associated with eutrophication, though in order to prevent double counting, the GHG associated figure was subtracted, thus giving a new value of \$98 – \$148m. At the exchange rate that Pretty (2003) used, and converting to today's values, £90 – £134m (2012 £) was the cost attributed to N leaching. For damage to waterways, cost analysis are the most favourable tool for valuation, with the most up to date available source for the UK being O'Neill (2007). Biological oxygen demand reductions were determined from O'Neill (2007), who estimated that between £4m – £5m (in 2012 £) in damage costs can be attributed to BOD in UK waterways. The rationale behind these values are not clear and for the analysis in chapter 7, ⅓ of the damage cost associated with "informal recreation from poor water quality" from O'Neill (2007) were used for assumed damage caused by BOD.

To place values on potential pathogen reduction, contingent valuation modelling was used. Eftec (2002) reported a total UK value of £79m (2012 £) for a 1% chance of each person avoiding stomach upset due to poor bathing water quality from faecal contamination. Due to uncertainty of what percentage of pathogens to waterways are caused by animal manure, a low value of 50% of pathogens in bathing water emanating from animal manure and a high value of 90% were chosen.

Finally, odour reduction required the development of an equation to place values on odour reduction. The equation used for this can be seen below.

Equation 3.1 The increased value per household by the introduction of AD to reduce the odour from animal manure stores.

$$\text{Eq 1} \qquad \text{increased value} = (D-1) \times H \times X \times Y$$

Where

D = average number of households in area equal to ½ mile radius from farm minus the farmer's property as an increase in the farmer's property is a private benefit, and not a public benefit.

H = average house prices within the locality of the manure storage facility.

X = the percentage drop in house prices associated with odour

Y = the percentage drop in odour

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Chapter 4

Replacing inorganic fertilizer with anaerobic digestate can maintain agricultural productivity at less environmental cost

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†G. Edwards-Jones was involved in the original grant proposal and conceptual design of this PhD and therefore is credited posthumously.

Abstract

On-farm Anaerobic Digestion (AD) is considered by both industry and policy-makers to be an effective way of converting biodegradable resource material into two commodities: methane that can be burnt to generate renewable energy, and digestate which can be applied onto land as an alternative for synthetic fertilizer, similar to undigested animal manure. However, few studies have assessed both the agronomic and environmental benefits of replacing synthetic fertilizer with digestate. Here, we compare the yield of grass grown in a greenhouse under controlled conditions following applications of either digestate generated from slurry, undigested slurry, synthetic N (ammonium nitrate) or synthetic NPK compound fertilizer. Soil solution was also collected to compare the potential loss of key nutrients through leaching. Soils from an organic and conventional farm were sown with commercial grassland mixtures comprising either a grass or a grass-clover mix, and the different types of amendments were applied. Application rates were normalised in terms of nitrogen and were equivalent to 150 kg dissolved N ha^{-1}. Crop yield from swards applied digestate was found to be equal to those applied synthetic NPK, and were significantly better than those applied straight synthetic N or undigested slurry. Protein levels were significantly greater in grass applied synthetic N, however there were no differences in digestibility between treatments. The potential for leaching of nitrate and ammonium was significantly greater in soils applied synthetic fertilizer than from soils applied undigested slurry or digestate; however, there were no significant differences for phosphate. The results indicate that the application of digestate rather than synthetic fertilizer can maintain grassland productivity but with less impact on the environment.

Keywords: Biogas, Digestibility, Legume, Water pollution

4.1 Introduction

Agricultural demand for synthetic fertilizers is likely to increase due to the need to feed a growing global population. However, synthetic fertilizers are increasingly expensive due to the energy-intensive nature of their production, and their use is also responsible for a significant proportion of the greenhouse gas (GHG) emissions and water pollution incidences from agriculture. Spreading organic resources (e.g. animal slurries) can reduce dependency on synthetic fertilizers; however, this too can lead to nitrate and phosphate pollution of groundwater (Fraters et al., 1998; Strebel et al., 1989) and the storage and application of slurry also emits GHGs (Banks et al., 2007). There is therefore both a need to reduce the amount of synthetic fertilizers utilized and to improve the management of organic resources to reduce the environmental impact of agriculture.

Anaerobic digestion (AD) is increasingly utilized worldwide as a management strategy for organic resources. AD generates two products: methane which can subsequently be burnt to generate renewable energy (Wilkinson, 2011), and digestate, which can be separated into a dry and liquid fraction suitable for land application. AD can also bring other benefits such as reducing both the odour (Smet et al., 1999) and pathogen load (Lund et al., 1996; Sahlstrom, 2003) of wastes and improve weed seed kill (Engeli et al., 1993). AD of cattle slurries has also been shown to reduce the biological oxygen demand by 55% (Anon, 2003) and chemical oxygen demand by up to 45% (Clemens et al., 2006); reducing the risk to aquatic ecology following spreading. AD is particularly appealing to livestock systems as the resources generated (manure) are suitable for digestion and hence can provide an additional source of income and reduce costs (Demirer & Chen, 2005).

In agricultural systems, nitrogen is the most frequent limiting factor for crop growth, especially on organic farms where synthetic fertilizer cannot be applied (Berry et al., 2002). During the AD process, the ammonium (NH_4^+) content of manure increases; digestate therefore has a higher content of directly available N than undigested manure (Field et al., 1984; Gutser et al., 2005). This ultimately increases in potential plant nitrogen uptake following spreading (Vanotti et al., 2009), meaning that AD could help farmers maximise the nutrient returns from manure and therefore reduce reliance on synthetic fertilizer. There is, however, some paucity of information on the agronomic effects of applying digestate as a replacement for undigested manure and synthetic fertilizer. Although Pain and Hepherd, (1985) and Tafdrup, (1995) are extensively cited, the evidence they provide on the positive benefits of digestate application on crop yield is highly subjective and lack scientific rigour. Some ambiguity also exists as to whether the application of digestate, relative to undigested manure, leads to greater or less potential for leaching of nutrients. For instance, Möller et al. (2008) found that there was less leaching of nitrate when slurry were digested; whereas Sänger et al. (2010; 2011) reported higher levels of leaching from digestate compared to undigested slurry, in fallow soil. Furthermore, there are no available data comparing leaching from pasture systems to which digestate has been applied relative to synthetic fertilizer.

The objectives of this study were to compare how the application of liquid digestate generated from digested cattle slurry, undigested (raw) cattle slurry, and two types of synthetic fertilizer (N and NPK) to different leys affected (i) crop yield; (ii) ley protein and digestibility values; and (iii) potential for leaching of nutrients in soils from both an organic and conventional farm.

4.2 Materials and methods

4.2.1 Soil and fertilizer collection and characterization

Soil (Eutric Cambisol) was collected from a pasture-based system at a conventional (non-organic) farm (CS) (Bangor University's Henfaes Research Station; 53°14'05''N, 4°00'50''W) and an organic farm (OS) (Wrexham; 53°08'16"N, 2°90'48"W). The soil was collected to a depth of 10 cm, passed through an 8 mm sieve and analysed in the laboratory within 24 h of collection.

Undigested slurry (US) was collected from an organic dairy farm from cows fed 50% grass silage and 50% whole crop silage (barley and peas). Added to the silage in concentrated form was rolled wheat, rolled beans, maize flour and soya expeller (concentrate to silage ratio was 20:80 on a dry matter basis). Liquid digestate (LD) was collected from the AD unit on the same farm. The AD unit is a 1000 m^3 mesophilic (38 °C) system, continually stirred digester with a retention time of 25 days and fed with cow manure only. The digestate was separated mechanically after leaving the digester and only the liquid fraction was collected. Samples were analysed within 24 h of collection.

Moisture content was determined after drying samples at 105 °C for 24 h. Nutrients from soil, US and LD were extracted in deionised water at a ratio of 1:5 (w/v). Samples were shaken (250 rev min^{-1}, 1 h, 20 °C), centrifuged for 15 min (4,000 *g*), filtered (Whatman no. 42), and the supernatant recovered for analysis. NO_3^-, NH_4^+ and P were determined colorimetrically (BioTek®, Vermont, NE) using the methods of Mulvaney (1996), Miranda et al., (2001) and Murphy et al., (1962), respectively; and K, Na and Ca were measured using a Sherwood Scientific 410 flame Photometer (Sherwood Scientific, Cambridge, UK). Total organic carbon and nitrogen were measured using a CHN2000 elemental analyzer (Leco Corp., St Joseph, MI) and dissolved organic carbon and dissolved nitrogen were measured using a TOC-V CHS analyzer (Shimadzu Corp., Kyoto, Japan). Undiluted slurry and digestate were used for the determination of electrical conductivity (EC; Jenway 4010 EC meter) and pH (Hanna Instruments pH 209 pH meter) whereas a 1:5 (soil: water, w/v) extract was used for soil.

4.2.2 Experimental design

A fully randomized pot trial experiment (n = 100) was set up in a greenhouse under controlled conditions. Pots (150 mm diameter; 1.2 l volume) were filled with 1.3 kg of either OS or CS (dry bulk density, OS: 0.87 g cm^{-1} and CS: 0.85 g cm^{-1}) and fitted with Rhizon® suction samplers (Rhizosphere Research Products, Wageningen, The Netherlands), one per pot at approximately 100 mm depth for collection of soil solution. Soil moisture was maintained at 70% field capacity throughout the experiment by watering up to a known weight for each pot. Half the pots for each soil type were seeded with perennial ryegrass only (*Lolium perenne L.*), and the other half with a mixture of perennial ryegrass and white clover (*Trifolium repens L.*) at a rate equivalent to 40 kg of grass ha^{-1} and 12 kg of clover ha^{-1} (Emorsgate Seeds, Norfolk, UK). The final experimental design therefore consisted of four sub-groups, namely (1) OS, grass only; (2) OS, grass-clover mix; (3) CS, grass only; and (4) CS, grass-clover mix. These were subsequently split into sub-groups of five treatments (n = 5 of each), to which the following were applied: US, LD, commercial straight ammonium nitrate (34.5% N) fertilizer (N) (GrowHow, Cheshire, UK), a commercial compound (21.8.11 NPK) fertilizer blend (NPK) (Yara, Lincolnshire, UK), or no amendment controls (C); for both soil types. The application rate for each was normalised for nitrogen, based on mineral N (ammonium N) values and total nitrogen content for the synthetic fertilizers. Nutrients were surface applied in two stages, with an equivalent of 100 kg N ha^{-1} applied four weeks following seeding, and the equivalent of 50 kg N ha^{-1} applied thereafter, one week after the first harvest (see below). Pots were regularly re-randomized during the trial. Pictures 4.1 to

4.4 were taken to show how soil was collected from the field and how the pots were set up in the greenhouse.

Picture 4.2: Soil taken from field, used in pots

Picture 4.3: Pots in the greenhouse, with grass growing

Picture 4.3: Pots with Rhizon® samples and vacuum tubes attached from nutrient analysis

Picture 4.4: Pots after harvesting

4.2.3 Yield analysis

In total, three herbage harvests were taken at weeks 5, 11 and 16 after the first application of fertilizer. The shoots were harvested to 2 cm above the soil surface. After harvesting, all samples were weighed fresh, dried for 48 hours at 85 °C, and then reweighed. The first harvest was ground to determine shoot total nitrogen and carbon content; protein content was calculated by multiplying the nitrogen reading by 6.25 and digestibility was tested as described in Omed et al. (1989).

4.2.4 Nutrient levels in soil solution

Sterile vacuum tubes were attached for 24 h to the Rhizon® samplers at weekly intervals throughout the experiment, one hour after a watering event. Volumes of soil solution collected were subsequently measured and concentrations of NO_3^-, NH_4^+ and P determined. Nutrient sampling was stopped ten weeks post the first fertilizer application as after this time quantities of NO_3^-, NH_4^+ and P in soil solution were all below detection levels (< 0.1 mg l^{-1}). These data were pooled so that mean concentrations of nutrients in soil solution could be determined.

4.2.5 Statistical analysis

Statistical analysis was performed using SPSS v.18. A homoscedastic two-tailed T-test was used to determine differences between soils and between the two organic fertilizers for nutrient content. For analysis of crop yield data, total yield from all three harvests was used and subject firstly to a one-way ANOVA to determine differences within each sub-group, with treatment as the factor. Then a 3-way ANOVA was performed, using soil (organically farmed soil or conventionally farmed), crop (grass or grass/clover) and fertilizer (control, US, LD, N, and NPK) as fixed factors, to determine if results were continuous for all and individual subgroups. The same analysis was used for nutrients in soil solution with data for mean weekly concentrations of nutrients, for a 10 week period. Nutrients in soil solution were also subject to repeated measures ANOVA to determine if differences existed in potential nutrient loss on a weekly basis. Post-hoc tests were carried out on all ANOVAs using Tukey HSD test at the level $p < 0.05$.

4.3 Results

4.3.1 Soil analysis

Soil pH values differed significantly ($p < 0.05$) between the two soils (Table 3.1). In addition, DOC levels were approximately three times greater in the organic farmed soil. In contrast however, the conventionally farmed soil had higher levels of total N and C ($p < 0.05$). There was no difference ($p > 0.05$) in the concentrations of NO_3^- or NH_4^+ between the two soils; however, levels of P and all base cations (K, Ca and Na) were significantly greater in the conventionally farmed soil ($p < 0.05$).

Table 4.1: Physico-chemical properties of both soils used in the study. Values represent means ± SEM ($n = 3$) and are expressed in terms of dry weight.

	Organic farmed soil	Conventionally farmed soil
pH	5.30 ± 0.04	5.45 ± 0.02
EC (µS cm^{-1})	51.2 ± 3.6	44.4 ± 5.4
Dry matter (%)	80.6 ± 0.1	78.3 ± 0.2
Total C (mg g^{-1})	20.3 ± 0.4	29.1 ± 0.5
Total N (mg g^{-1})	2.02 ± 0.02	3.11 ± 0.07
C:N	10 ± 0.02	9 ± 0.13
DOC (mg g^{-1})	0.35 ± 0.02	0.11 ± 0.01
NO_3^- (µg g^{-1})	12 ± 2.1	20 ± 1.2
NH_4^+ (µg g^{-1})	10 ± 3.4	9 ±1.3
P (µg g^{-1})	16 ± 2	90 ± 5
K (µg g^{-1})	13 ± 4	30 ± 4
Ca (µg g^{-1})	23 ± 2	36 ± 3
Na (µg g^{-1})	40 ± 3	65 ± 6

4.3.2 Fertilizer analysis

Liquid digestate had a higher pH than undigested slurry ($p < 0.05$) (Table 4.2). Total C levels were greater in undigested slurry ($p < 0.05$); and although there were no differences between total N, NH_4^+ levels were over three times greater in the digestate ($p < 0.05$). NO_3^- was also slightly higher in the digestate although this did not prove statistically significant ($p > 0.05$). Conversely, the undigested slurry had approximately ten times greater levels of P in comparison to the digestate ($p < 0.05$). There was no statistical difference in the concentrations of base cations between either organic fertilizer types.

Table 4.2: Physico-chemical properties of the undigested slurry (US) and liquid digestate (LD) used in the study. Values represent means ± SEM ($n = 3$) and are expressed in terms of dry weight.

	US	LD
pH	7.55 ± 0.12	8.59 ± 0.01
EC (mS cm^{-1})	9.01 ± 0.14	12.2 ± 0.1
Dry matter (%)	14.3 ± 0.26	5.2 ± 0.3
Total C (mg g^{-1})	393 ± 8	274 ± 6
Total N (mg g^{-1})	21 ± 0.4	21 ± 1
C:N	18 ± 0.3	13 ± 0.1
DOC (mg g^{-1})	35.3 ± 0.2	30.0 ± 0.9
DON (mg g^{-1})	11.6 ± 0.1	27.4 ± 1.3
NO_3^- (mg g^{-1})	0.31 ± 0.15	0.51 ± 0.04
NH_4^+ (mg g^{-1})	6.54 ± 0.25	20.35 ± 0.53
P (mg g^{-1})	10.6 ± 0.8	1.0 ± 0.2
K (mg g^{-1})	9.1 ± 0.1	16.5 ± 0.0
Ca (mg g^{-1})	13.9 ± 0.1	19.5 ± 0.1
Na (mg g^{-1})	3.6 ± 0.1	7.2 ± 0.2

4.3.3 Crop yield

Using the one-way ANOVA, all treatments gave significantly greater yields ($p < 0.05$) than the unamended control for both grass grown in the organic farmed and conventionally farmed soil and grass-clover grown in the organic farmed soil (Fig. 4.1A-4.1C). In grass grown in the organic farmed soil (Fig. 4.1A), greatest yields were recovered in pots to which LD or NPK had been applied. A similar trend was also evident in the organic farmed grass-clover sub-group (Fig.4.1B) and the grass grown in conventionally farmed soil (Fig. 4.1C); though there was no significant difference between N and NPK in the latter. However, in conventionally farmed soil, the grass-clover yield (Fig. 4.1D) showed very different results with no significant difference ($p > 0.05$) between control and the synthetic fertilizers; but significantly greater yields ($p < 0.05$) from pots to which LD or US had been applied. When the 3-way ANOVA was applied, LD had the highest crop yield of all treatments and was significantly different from all other treatments ($p < 0.05$). There were highly significant differences ($p < 0.001$) between soil and treatments for all three harvests. All treatments were significantly greater than control ($p < 0.05$) throughout the experiment.

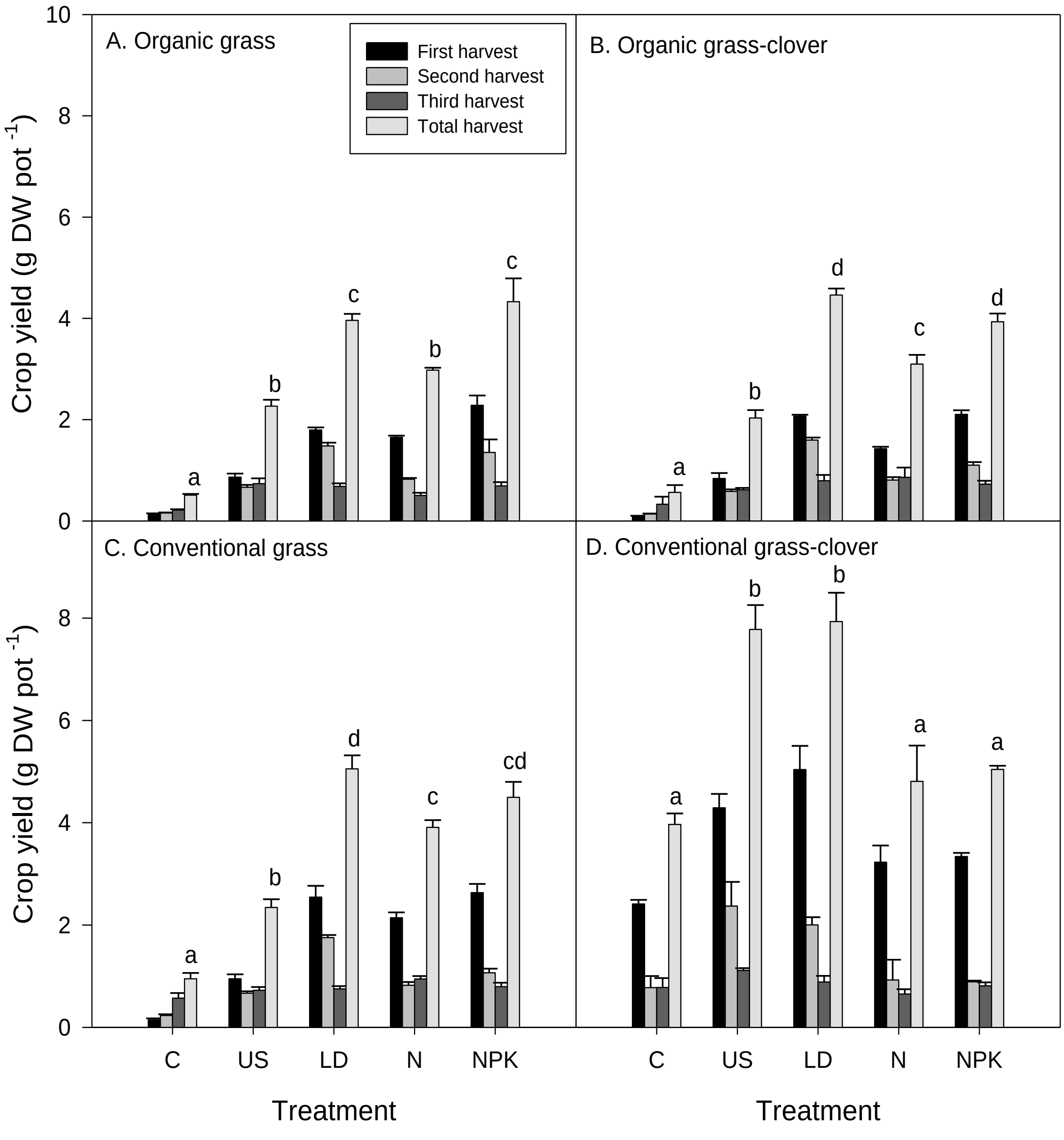

Figure 4.1: Comparison of crop yield in each of the four sub-groups: organic grass (A), organic grass-clover (B), conventionally farmed grass (C), and conventionally farmed grass-clover (D) after the application of different fertilizer types: control (C), undigested slurry (US), liquid digestate (LD), mineral nitrogen (N) and mineral NPK (NPK). Values represent the mean ± SEM ($n = 5$). Letters within graphs denote differences ($p < 0.05$) between treatments within that sub-group for the total yield.

4.3.4 Protein and digestibility

There were significant differences in protein levels between ($p < 0.001$) and within ($p < 0.05$) treatments (Fig.4.2). Protein levels were not significantly different to controls in any of the treatments with organic fertilizer ($p > 0.05$); however, levels were significantly higher for grass applied straight N for both grass and grass-clover ($p > 0.05$). In grass grown in conventionally farmed soil, again there was no difference ($p > 0.05$) between control and the organic fertilizers, with grass applied straight N having the highest level of protein ($p < 0.05$). In contrast though, conventionally farmed grass-clover there was no difference ($p > 0.05$) between control and the synthetic fertilizers, whereas there were greater levels of protein when organic fertilizer was applied ($p < 0.05$). There were no significant differences in digestibility between treatments ($p > 0.05$) and within treatments ($p > 0.05$) (data not shown).

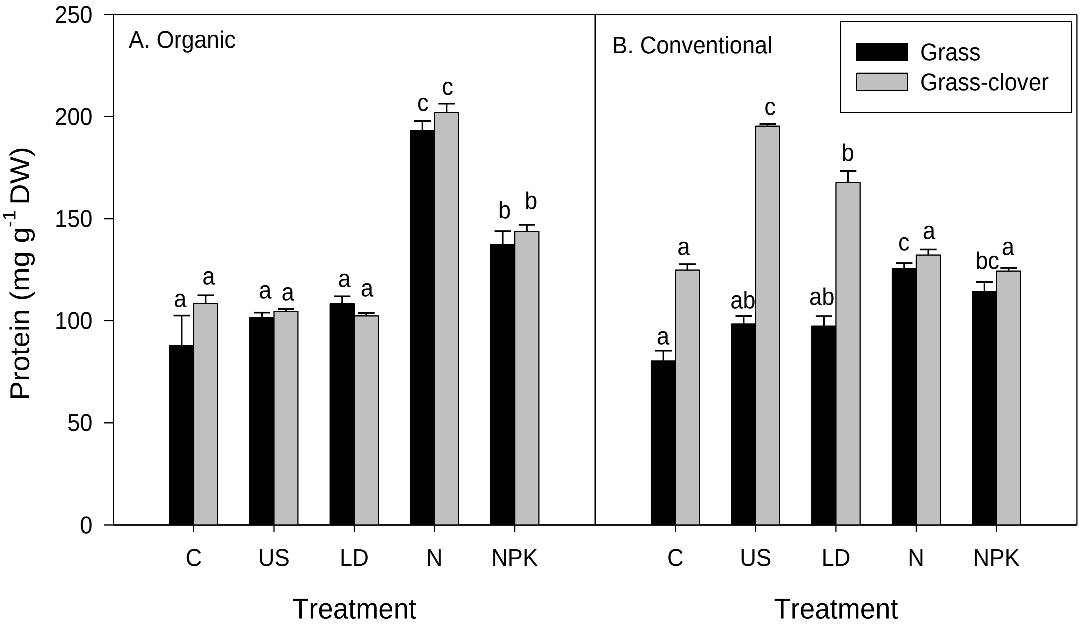

Figure 4.2: Mean protein content in leys grown in either an organically (A) or conventionally (B) managed soil after the application of different fertilizer types: control (C), undigested slurry (US), liquid digestate (LD), mineral nitrogen (N) and mineral NPK (NPK). Results are for the first harvest only (five weeks post-application). Values represent the mean ± SEM (n = 5). Letters within each graph denote significant differences ($p < 0.05$) between treatments of the same soil and vegetation cover type.

4.3.5 Nutrient levels in soil solution

4.3.5.1 Nitrate

Within the soil solution extracted from the organic farmed soil in which grass was grown, there were no differences ($p > 0.05$) in nitrate levels between the control, organic fertilizers and NPK treatments; but concentrations were significantly greater when straight N was applied ($p < 0.05$). From soil solution in the organic farmed grass-clover treatments (Fig. 4.3A), significantly greater mean concentrations of nitrate were found in soil solution when both synthetic fertilizers had been applied ($p < 0.05$). The same pattern was followed for soil solution in both conventionally farmed grass and grass-clover pots (Fig. 4.3B) with both synthetic fertilizers being different ($p < 0.05$) from control and the organic fertilizers ($p < 0.05$). When the data was analysed in a 3-way ANOVA, no differences in soil types ($p > 0.05$) emerged for either form of nitrogen, however, there was a difference in seed type ($p < 0.05$), with nitrate levels greatest where clover was also present. The interaction between seed type and treatment was also significant ($p < 0.05$). Throughout the experiment, no differences in nutrient levels in soil solution emerged between C pots and those applied US and LD; all of which were significantly lower than synthetic fertilizer ($p < 0.05$). There were no significant differences between the synthetic fertilizers ($p > 0.05$), but levels were significantly greater in comparison to the control and organic treatments ($p < 0.05$).

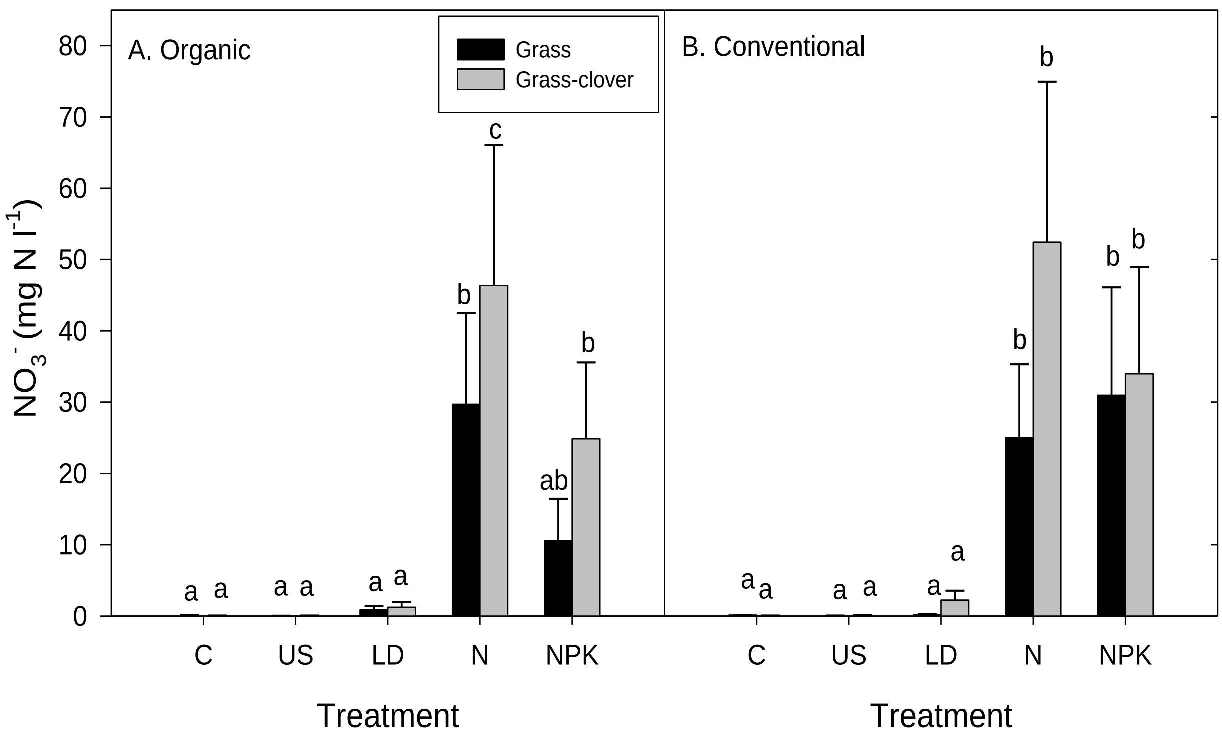

Figure 4.3: Mean soil solution NO_3^- concentrations after the application of organic and synthetic fertilizers to either an organically (A) or conventionally (B) managed soil. Capital letters represent the unamended control (C) and those applied the following fertilizers: undigested slurry (US), liquid digestate (LD), mineral nitrogen (N) and mineral NPK (NPK). Values represent the mean ± SEM (n = 5). Letters within each graph denote significant differences ($p < 0.05$) between treatments of the same soil and ley type.

4.3.5.2 Ammonium

Within the organic soil, mean concentrations of ammonium found in soil solution were only significantly greater than controls ($p < 0.05$) when pots had N applied, for both ley types (Fig. 3.4A). A similar pattern was seen with solutions extracted from the conventionally farmed soils, although application of either N or NPK to pots led to significantly raised levels of ammonium in soil solution ($p < 0.05$), particularly when a grass-clover ley was grown (Fig. 4.4B). When data was analysed in a 3-way ANOVA, differences emerged within and between seed and treatment ($p < 0.001$ for both). The 3-way ANOVA for ammonium was the same as nitrate, i.e. with no difference between C, US and LD, and there was no difference between the synthetic fertilizers ($p < 0.05$). However, levels within controls and those applied organic fertilizers were significantly lower than those applied synthetic fertilizers ($p < 0.05$).

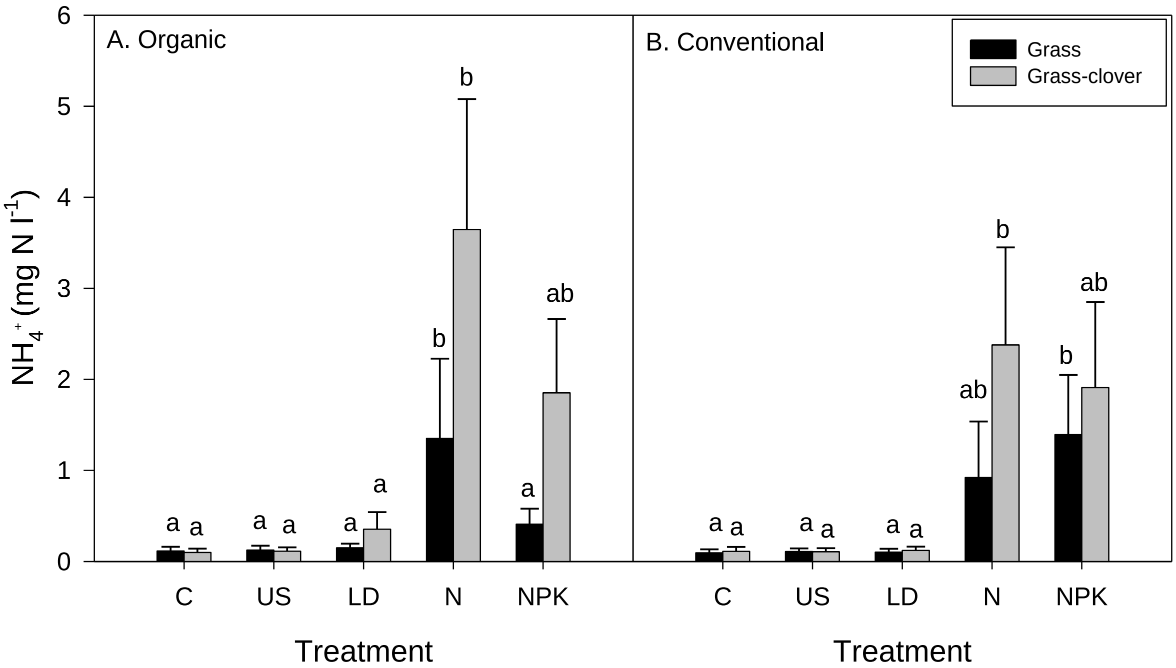

Figure 4.4: Mean soil solution NH_4^+ concentrations after the application of organic and synthetic fertilizers to either an organically (A) or conventionally (B) managed soil. Capital letters represent the unamended control (C) and those applied the following fertilizers: undigested slurry (US), liquid digestate (LD), mineral nitrogen (N) and mineral NPK (NPK). Values represent the mean ± SEM (n = 5). Letters within each graph denote significant differences ($p < 0.05$) between treatments of the same soil and ley type.

4.3.5.3 Phosphate

There was much less variability in the levels of phosphate recovered in soil solution between treatments (Fig. 4.5). In organic farmed soil, levels of P in solution were only significantly higher than controls ($p < 0.05$) when NPK was applied to a grass-clover ley (Fig. 4.5A). For solutions in conventionally farmed soil sown with grass, P concentrations were actually significantly greater in control samples ($p < 0.05$) than in all treatments; however no statistical difference existed when a grass-clover ley was grown (Fig 4.5B). When the data was analysed in a 3-way ANOVA, differences emerged with soil and seed ($p < 0.05$); as well as between soil and seed type, and soil and treatment ($p < 0.05$). Controls had higher levels of P in soil solution than any treatment ($p < 0.05$). Levels in pots applied N, US, and LD were not significantly different ($p > 0.05$) from each other. Overall, phosphate levels in those applied NPK were higher than other treatments, though still lower than the controls ($p < 0.05$).

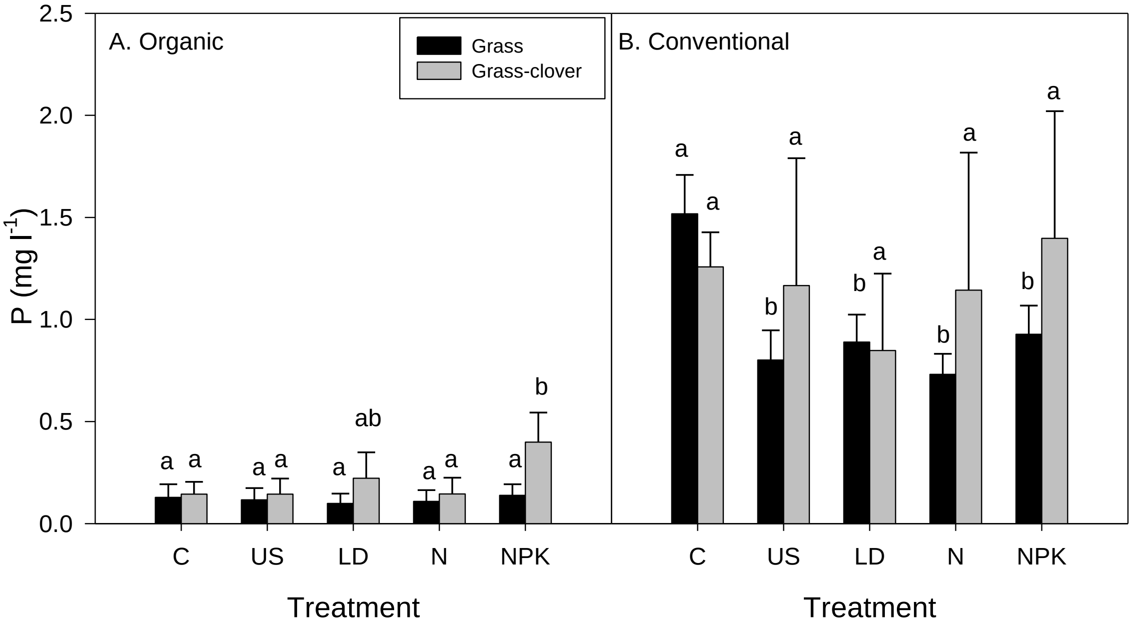

Figure 4.5: Mean soil solution P concentrations after the application of organic and synthetic fertilizers to either an organically (A) or conventionally (B) managed soil. Capital letters represent the unamended control (C) and those applied the following fertilizers: undigested slurry (US), liquid digestate (LD), mineral nitrogen (N) and mineral NPK (NPK). Values represent the mean ± SEM ($n = 5$). Letters within each graph denote significant differences ($p < 0.05$) between treatments of the same soil and ley type.

4.4 Discussion

4.4.1 Crop yield

Grasslands dominate livestock systems and the agricultural industry is increasingly seeking to maximise returns achievable through better utilization of grass with lesser inputs. Relevant to this point, this study found that grasses applied LD gave similar or better yield than those receiving either N or NPK synthetic fertilizers. This pattern was also evident when comparing LD and US in all but one case. The results concur with previous studies that implied that anaerobic digestion of organic fertilizers enhances plant uptake of nutrients and hence crop yield thereafter (de Boer, 2008; Holm-Nielsen et al., 2009; Rubaek et al., 1996).

The lower overall yield from swards applied US may in part be due to the fact that all fertilizers were surface applied. Approximately 95% of manure in the UK is surface applied (DEFRA, 2010) and the application method used in this study is therefore representative of typical agricultural practices. However, the thicker texture of US meant that a notable proportion remained on the surface of the soil and hence may be subject to loss of ammonia (and therefore N) through volatilization (Sommer & Hutchings, 2001); whereas the LD was readily absorbed into the soil. It may be deduced that the higher yield from leys applied LD compared to N in all sub-groups is at least in part due to LD incorporating other nutrients key for plant growth (e.g. P and K), as LD performed as well as NPK in three of the four treatments. Our results are in accordance with others such as Dahlberg et al., (1988) who found that digestate was as effective as synthetic fertilizer when comparing dry matter content from grain yields and Liedl et al., (2006) who found digested poultry litter performed as well as synthetic N fertilizers.

This study was conducted over a period equating to approximately one growing season under a simulated intensive grazing or cropping system. Mineralisation of N is relatively slow in organic manures, thus with repeated application of organic fertilizers, residual levels of N increases in soils, as shown in long-term field trials (Schroder et al., 2007; Sorensen, 2004). However, the results from the current study show that application of digestate can be effective in increasing yield of pasture in the short-term. It is acknowledged that yield response may differ with other crop types; however, it has been noted that crops with a short and intensive uptake of nitrogen may benefit most from the application of digestate (Svensson et al., 2004). Application of inorganic, rather than organic fertilizer is also known to suppress clover growth and hence reduce the amount of nitrogen fixed by legume roots (Nesheim et al., 1990). Although the yield of clover wasn't directly measured in this trial, visual observation suggests that there was more clover in pots applied LD than in those receiving synthetic fertilizers. This may also explain why yields were greater when LD was applied; although further work at field-scale is needed to validate this.

4.4.2 Protein and digestibility

The addition of synthetic fertilizers had a greater effect on grass protein levels in the organic farmed, rather than the conventionally farmed soil. This may partially be due to the former lacking nutrients over the years due to restrictions on fertilizer application (Table 4.1). With the conventionally farmed grass-clover ley, application of either organic fertilizer led to significantly greater levels of protein, which is of note to agricultural systems. However, this increase in protein may be due to the higher percentage of clover in the ley, as discussed previously.

No differences emerged in digestibility between different leys or on the type of fertilizer applied. This is plausibly due to the relatively short-term nature of the trial before digestibility decreased in any treatment. Conducting a trial over a longer growing period may therefore be needed to establish if any differences become apparent over time.

4.4.3 Nutrients in soil solution

The loss of nitrogen from soil is a major agricultural and environmental problem due to the cost of nitrogen and its impact on water quality and the atmosphere (Fangmeier et al., 1994; Schulze et al., 1989; Stark & Richards, 2008). Nutrient loss due to overland flow following application of manure and synthetic fertilizer has been the focus of numerous studies (Turtola & Yli-Halla, 1999; Uusi-Kamppa & Mattila, 2010). This current study focused on the potential for leaching of nutrients through the soil. Our findings in relation to potential nutrient loss concur with field trials that found there were no significant differences in levels of N leaching from digestate and undigested manure (Lukehurst et al., 2010). Schroder et al., (2010) implied that the degree of nitrogen leaching from grassland was unaffected by whether the source of nitrogen was synthetic fertilizer or cattle manure; but rather the dominant factor was the balance of supply and crop demand. Nevertheless, our results indicate that application of synthetic fertilizer, rather than organic fertilizer, could lead to far greater potential leaching of nitrogen. In addition to the environmental cost (impact) of nitrate leaching, this has important economic significance due to the cost associated with fertilizers and clean-up. Phosphate is also a major cause of eutrophication in waterways and minimising leaching of P is of considerable interest. This trial indicates that the potential for loss of P through leaching is low when LD is applied. Whilst this may be expected due to much lower levels of P relative to synthetic NPK fertilizer, the fact that LD application led to enhanced grass yields would also have facilitated efficient uptake of phosphate and hence reduce the possibility of loss due to leaching.

4.5. Conclusions

This glasshouse-based study was performed with one common agricultural soil type and with two ley compositions that are frequently used for livestock grazing. Although the findings should not be extrapolated to all soil types and management systems, the results do add further evidence as to the potential value of AD over conventional agronomic practices.

The agricultural industry is under pressure to reduce its greenhouse gas emissions and the loss of nutrients into waters, whilst at the same time improving the efficiency of production. This study indicates that replacing synthetic fertilizers with liquid digestate can maintain or improve yields from grassland systems and concurrently reduce the potential for losses of nutrients to the environment. This may ultimately reduce agricultural dependence on synthetic fertilizer and the energy and economic costs associated with their use. AD should therefore not only be considered a source of renewable energy and waste management system, but also a pollution abatement technology.

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Chapter 5

Fungal and bacterial growth following the application of slurry and anaerobic digestate of livestock manure to temperate pasture soils

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†G. Edwards-Jones was involved in the original grant proposal and conceptual design of this PhD and therefore is credited posthumously.

Abstract

How land-application of digestate sourced from anaerobic digestion (AD) of animal resources influences the functioning of a mixed pasture agroecosystem is not well characterised, particularly with regard to the response of the actively growing microbial community. We studied the impact of the liquid AD digestate on the decomposer community in two different soils, seeded with two different common grassland crops; a mixture of either grass or grass-clover in a greenhouse experiment. We studied bacterial (Leucine incorporation into bacteria) and fungal (Acetate incorporation into ergosterol) growth-responses to AD cattle slurry digestate, undigested cattle slurry, mineral fertilizer (NPK and N) added at a rate equivalent to 150 kg N ha^{-1}, and a no-fertilizer control treatment. Differences in fungal and bacterial growth were evident between the soil and sward types. However, the fertilizers consistently stimulated a higher bacterial growth than the no-fertilizer control, and liquid digestate resulted in a level of bacterial growth higher or equal to that of mineral fertilizer, while undigested slurry resulted in lower bacterial growth. These fertilizer effects on bacterial growth mirrored the effects on plant growth. In contrast, the fungal community responded only marginally to fertilizer treatments. We conclude that the application of digestate stimulates the bacterial decomposer community in a similar way to that of mineral fertilizers. Our results suggest that mineral fertilizer can be exchanged for liquid digestate with limited impact on the actively growing soil microbial community that in turn regulate important soil processes including nutrient cycling in agricultural soils.

Keywords: Animal resources; Biogas; Grassland; Decomposer ecology; Green fertilizer; Legume; Microbial Ecology; Plant nutrition; Soil fertility

5.1 Introduction

Pre-treatment of livestock manure by anaerobic digestion (AD) can reduce the negative side-effects of manure that are used as organic fertilizers. For instance, the AD process reduces odour (Smet et al., 1999) and the net global warming potential (GWP) of manure (Collins et al., 2011), kills weed seeds (Engeli et al., 1993) and reduces pathogen loads (Lund et al., 1996; Sahlstrom, 2003; Kunte et al., 2004). It is for these reasons that AD is becoming increasingly popular on farms for the treatment of manure (Pain and Hepard, 1985; Wilkinson, 2011). The AD process has direct advantages beyond these, however, foremost of which is biogas production for renewable energy, and the enrichment of mineral fractions of N and P during digestion (Field et al., 1984; Masse et al., 2011), resulting in a higher concentration of plant-available nutrients compared with undigested manure and a subsequently elevated plant growth promotion ability, suggested to be similar to mineral fertilizers (Dahlberg et al., 1988; Gutser et al., 2005; Liedl et al., 2006).

The amount and type of organic and mineral fertilizer added to soil is known to directly and indirectly influence the size, activity and structure of the soil microbial community. For example, organic resources can provide a labile substrate, promoting the rapid growth of soil microorganisms whilst sometimes suppressing the growth of others (e.g. mycorrhiza; Egerton-Warburton et al., 2007). Further, increasing the N load to soil, by e.g. fertilizer addition, typically stimulates plant growth, leading to an increase in below ground C inputs via root turnover and exudation, which in turn stimulates fungal and bacterial growth (Knorr et al., 2005; Liu and Greaver, 2010). Consequently, in pasture systems where the majority of top soil is rhizosphere soil, a large impact on the soil microbial community is expected to follow fertilizer applications. Additionally, the relationship between plant productivity and the microbial community is reciprocal in that the balance between fungal and bacterial contribution to decomposition has been linked to an ecosystem’s ability to sequester C (Strickland and Rousk, 2010). This has implications for plant nutrition since the cycling of macronutrients, including e.g. C and N, is linked (Liu and Greaver, 2010), and e.g. increases in C sequestration will also result in reduced nutrient availability and thus affect soil fertility. Consequently, to determine the long-term effects of fertilizers, it is important to determine how they influence the active and growing soil microbial decomposer community.

Organic farmers are prohibited from applying chemical fertilizers to soil, relying largely on the incorporation of legumes into cropping systems or the application of organic resources to supply plant nutrients (Zemenchik et al., 2001). While the benefits of organic farming practices on many aspects of soil health have been studied, the direct impact of AD digestate on actively growing bacteria and fungi communities, particularly in comparison to conventional fertilizers, remains unassessed. In this work, we evaluated the responses of the actively growing soil microbial community to the application of different fertilizer treatments in a 16 week green-house experiment: comparing liquid digestate from an AD unit against mineral fertilizer and undigested slurry and a no-fertilizer control in two different pasture soils with two different common crop mixes.

5.2 Materials and methods

5.2.1 Soil and fertilizer collection

Two soils, both Eutric Cambisols (FAO 1989), were used for experimentation. Rather than comparing different soil management *per se*, two soils were included in the study to increase the statistical power of the fertilizer assessment; if reproducible results could be shown in two independent soils, observed effects are likely general. Moreover, from an applied perspective, both farms were interested in the implementation of AD practise. The first soil was collected from an organic dairy farm (for the past 12 years) under permanent pasture (*Lolium perenne* L., *Trifolium repens* L.) located at Wrexham, NE Wales (53°08’16”N, 2°90’48”W); hereafter termed “organic farmed soil”. The second soil was collected from a conventionally farmed (non-organic) managed sheep farm under permanent pasture located at Abergwyngregyn, NW Wales (53°14’05’’N, 4°00’50’’W); hereafter termed “conventionally farmed soil”. Fresh soil was sieved through an 8 mm sieve and analysed within 24 h of collection, in early May 2010. Undigested slurry (US) and liquid digestate (LD), generated from AD of the slurry were collected from a dairy farm located in Wrexham, NE Wales. The US was collected directly from a cattle housing shed while the LD was collected from a 1000 m^3 mesophilic (38 °C), continually-stirred tank reactor (CSTR) with a retention time of 25 d. Mainly on-farm cattle manure had been fed as feedstock into the CSTR for the previous six months. After digestion, the digestate was mechanically separated into liquid and dry fibre fractions, of which only the liquid fraction was collected for use in the experiment.

5.2.2 Experimental design

The greenhouse experiment was designed to investigate the influence of the fertilizer treatments in soils with various management systems (different crop seed mixes in differently farmed soils). This is a powerful design to assess the universal effects of the fertilizer treatments, but it should be noted that it cannot be used to reliably assess differences in farming practices, since it lacks replication for this. A randomised pot trial experiment was set up in a greenhouse with a mean weekly temperature of 23 ± 3 °C. The plastic pots had a diameter of 150 mm and were filled with 1.3 kg of soil. Two different seed types were applied to each of the two soils, both commonly used in the UK: *Lolium perenne* L. was sown in half the pots at a rate of 40 kg of grass ha^{-1} and a *Lolium perenne* L. and *Trifolium repens* mix was sown in the other half, at a seeding rate equivalent to 40 kg of grass ha^{-1} and 12 kg of clover ha^{-1} (Emorsgate seeds, Norfolk, UK). This separated the trial up into four distinct groups, namely (a) organic farmed soil *Lolium perenne*, (b) organic farmed soil *Lolium perenne-Trifolium repens* mix (c) conventionally farmed soil *Lolium perenne*, (d) conventionally farmed soil *Lolium perenne-Trifolium repens* mix, with five fertilizer treatments and five replicates for each treatment (totalling 4 × 5 × 5 = 100 pots). The five fertilizer treatments consisted of (i) an unamended control (C), (ii) undigested slurry (US), (iii) liquid digestate (LD), (iv) mineral N (NH_4NO_3) fertilizer (GrowHow Ltd., Cheshire, UK), and (v) a mineral NPK fertilizer (N:P:K = 21:8:11) (Yara Ltd., Lincolnshire, UK). Application of fertilizer was normalised for total soluble N. While it is not trivial to standardise synthetic and organic fertilizer additions, using total dissolved N would be more appropriate than total N. In total, an equivalent of 150 kg N ha^{-1} of an available form was applied to all pots except the control treatment. Soil moisture was maintained at 70% field capacity throughout the experiment (monitored gravimetrically, and adjusted as needed). Chemistry and nutrient concentrations of soil and organic fertilizers are provided in Tables 5.1 and 5.2.

After the grass and grass/clover mix had established (4 weeks after planting), 100 kg N ha^{-1} was applied to all treatments except control, and this event defined the initiation of the experiment (time = 0). After 5 weeks (W5) the first herbage harvest was taken as well as soil samples for analysis of the decomposer community. One week later, a further 50 kg N ha^{-1} was applied, and on week 11 (W11) the second herbage harvest was taken and soil collected for analyses. The final herbage harvest and soil samples were collected on week 16 (W16). At the time of each harvest, three randomly selected soil samples were collected from each pot within the top 5 cm. These samples were homogenised and combined to a composite sample that was used to estimate bacterial and fungal growth. For logistical reasons, four randomly selected replicates out of the total five were used for the microbial analyses at week 16.

5.2.3 Plant yield analysis

The herbage was cut approximately 2 cm above the soil surface and subsequently dried at 85 °C for 48 h to determine dry weight. These data were reported in a parallel study (Walsh et al., 2012).

Table 5.1: Physico-chemical properties of both soils used in the study. Values represent means ± SE (*n* = 3) and are expressed in terms of dry weight where applicable.

	Organic farmed soil	Conventionally farmed soil
pH	5.3 ± 0.04	5.5 ± 0.02
EC(µS cm^{-1})	51.0 ± 3.7	44 ± 5.5
Water content (%)	19.5 ± 0.11	21.8 ± 0.27
Total C (mg g^{-1})	20.4 ± 0.4	29.2 ± 0.5
Total N (mg g^{-1})	2.0 ± 0.02	3.1 ± 0.07
C:N	10 ± 0.02	9 ± 0.13
DOC (mg g^{-1})	0.35 ± 0.02	0.11 ± 0.01
NO_3^- (µg g^{-1})	12 ± 2.12	20 ± 1.21
NH_4^+ (µg g^{-1})	10 ± 3.41	9 ±1.34
P (µg g^{-1})	16 ± 2.2	90 ± 5.41
K (µg g^{-1})	14 ± 4.5	30 ± 4.12
Ca (µg g^{-1})	24 ± 2.2	36 ± 3.31
Na (µg g^{-1})	40 ± 3.1	65 ± 6.03

Table 5.2: Physico-chemical properties of the undigested slurry (US) and liquid digestate (LD) used in the study. Values represent means ± SE (n = 3) and are expressed in terms of dry weight where applicable.

	Undigested slurry	Liquid digestate
pH	7.6 ± 0.12	8.6 ± 0.01
EC (mS cm^{-1})	9.0 ± 0.14	12.2 ± 0.10
Water content (%)	85.7 ± 0.26	94.8 ± 0.37
Total C (mg g^{-1})	394 ± 8	274 ± 6
Total N (mg g^{-1})	22 ± 0.4	22 ± 0.8
C:N	18.1 ± 0.26	13.1 ± 0.07
DOC (mg g^{-1})	35 ± 0.2	30 ± 1.0
DON (mg g^{-1})	12 ± 0.14	27 ± 1.3
NO_3^- (mg g^{-1})	0.31 ± 0.15	0.51 ± 0.04
NH_4^+ (mg g^{-1})	6.5 ± 0.25	20.4 ± 0.53
P (mg g^{-1})	11 ± 0.8	1.0 ± 0.28
K (mg g^{-1})	9 ± 0.01	17 ± 0.08
Ca (mg g^{-1})	14 ± 0.13	20 ± 0.10
Na (mg g^{-1})	3.6 ± 0.01	7.3 ± 0.21

5.3 Microbial analyses

5.3.1 Bacterial growth

Bacterial growth was estimated using leucine (Leu; Kirchman et al., 1985) incorporation in bacteria extracted from soil using the homogenization / centrifugation technique (Bååth, 1994) with modifications (Bååth et al., 2001; Rousk and Bååth, 2011). Briefly, 2 µl of radiolabelled Leu ([^{3}H]Leu, 37 MBq ml^{-1}, 5.74 TBq $mmol^{-1}$, Perkin Elmer, UK) combined with non-labelled Leu was added to each tube, resulting in 275 nM Leu in the bacterial suspensions. The amount of Leu incorporated into extracted bacteria per h and g soil was used as a measure of bacterial growth.

5.3.2 Fungal growth and biomass

Fungal growth was assessed using the acetate incorporation into ergosterol method (Newell and Fallon 1991) adapted for soil (Bååth, 2001) with modifications (Rousk et al., 2009; Rousk and Bååth, 2011). Briefly, 1-[^{14}C]acetic acid (sodium salt, 7.4 MBq ml^{-1}, 2.04 GBq mmol^{-1}, Perkin Elmer, UK) combined with unlabelled sodium acetate resulting in a final acetate concentration of 220 µM was added to a soil solution and incubated for 4 h at 22 °C without light. Ergosterol was then extracted, separated and quantified using HPLC equipped with a UV detector (Rousk and Bååth, 2007). The fungal biomass was estimated assuming 5 mg ergosterol g^{-1} fungal biomass (Joergensen, 2000; Ruzicka et al., 2000). The eluent containing the ergosterol peak was collected and the amount of incorporated radioactivity determined. The amount of acetate (Ac) incorporated into fungal ergosterol (pmol h^{-1} g^{-1} soil) was used as a measure of fungal growth.

5.3.3 Soil and fertilizer analysis

Soil and organic fertilizer samples were extracted in deionised water 1:5 (w/v), shaken (250 rev min^{-1}, 1 h, 20°C), centrifuged (4000 *g*, 15 min), and the supernatant filtered (Whatman No. 42). K, Na and Ca were analysed using a model 410 flame photometer (Sherwood Scientific, Cambridge, UK) while nitrate, ammonium and phosphate were determined using the colorimetric methods of Mulvaney, (1996), Miranda et al., (2001) and Murphy et al., (1962), respectively. Total dissolved N (TDN) and dissolved organic N (DON) were determined using a TCN-V analyzer (Shimadzu Corp., Kyoto, Japan) and total C and N were analysed using a CHN2000 elemental analyzer. Samples were oven dried at 105 °C for 24 h to determine gravimetric water content. Electrical conductivity (EC) and soil pH were determined using standard electrodes (EC, Jenway 4010 EC meter; pH, Hanna Instruments pH 209 pH meter), using undiluted samples for US and LD and 1:5 (w/v) distilled water extract for soil.

5.3.4 Statistical analysis

Treatment differences in the microbial variables and the plant yield data were compared by 3-way ANOVAs (JMP 7.0 for Mac, SAS Institute Inc., Cary, NC, USA), using soil (organic farmed or conventionally farmed), crop (grass or grass/clover) and fertilizer (control, US, LD, N, and NPK) as fixed factors. Tukey's HSD pair-wise comparisons ($p < 0.05$) were used to determine differences between fertilizer responses.

5.4 Results

5.4.1 Crop yield

Total cumulative crop yield at the end of the trial from the three harvests combined showed that there were significant effects ($p < 0.001$) of the factors soil, crop and fertilizers (Table 5.3). There was approximately twice the crop yield in the conventionally farmed soil as in the organic farmed soil, and about 25% greater yield in grass-clover than grass treatments. All fertilizer treatments to the soils resulted in higher yield than the control treatment with no differences apparent between the mineral fertilizers and US. In contrast, the LD treatment had the greatest cumulative crop yield and was significantly higher than the other fertilizer treatments ($p < 0.05$). Significant differences occurred between soil ($p < 0.001$) and fertilizer treatments ($p < 0.001$) for all three harvests, however, differences between seed types were only apparent on W5 ($p < 0.001$) and on W11 ($p < 0.05$). At W5 there was approximately 100% greater crop yield from the conventionally farmed soil compared to the organic farmed soil, with about a 70% greater crop yield from grass-clover than from grass. There were differences in yield between fertilizer treatments for all harvests ($p < 0.05$), however, these differences were less pronounced by W16 with both organic and mineral fertilizers still being greater than the control, but with no differences apparent between the fertilizer treatments. These results are provided in greater detail in Walsh et al. (2012).

Table 5.3: Mean crop yield for each harvest, values represent means ± SE (*n* = 5) and are expressed in terms of dry weight, where fertilizer types are control (C), undigested slurry (US), liquid digestate (LD), mineral N and mineral NPK added at a rate of 150 kg N ha^{-1}. Values are from Walsh et al., (2012).

Plant dry weight (g DW pot^{-1})								
	Week 5		Week 11		Week 16		Total cumulative	
Organic farmed								
Grass								
C	0.14	± 0.01	0.15	± 0.01	0.22	± 0.02	0.51	±0.02
US	0.87	± 0.07	0.66	± 0.05	0.74	± 0.10	2.27	±0.13
LD	1.79	± 0.05	1.48	± 0.07	0.68	± 0.06	3.96	±0.13
N	1.65	± 0.03	0.82	± 0.03	0.50	± 0.05	2.98	±0.05
NPK	2.29	± 0.19	1.35	± 0.26	0.69	± 0.07	4.33	±0.46
Clover								
C	0.09	± 0.01	0.14	± 0.01	0.33	± 0.15	0.56	±0.15
US	0.84	± 0.11	0.58	± 0.04	0.61	± 0.04	2.03	±0.16
LD	2.07	± 0.03	1.59	± 0.05	0.79	± 0.12	4.46	±0.13
N	1.43	± 0.04	0.81	± 0.06	0.86	± 0.19	3.10	±0.18
NPK	2.11	± 0.08	1.10	± 0.06	0.73	± 0.07	3.93	±0.16
Conventionally farmed								
Grass								
C	0.15	± 0.03	0.23	± 0.02	0.57	± 0.10	0.95	±0.11
US	0.95	± 0.09	0.66	± 0.04	0.73	± 0.06	2.34	±0.16
LD	2.55	± 0.22	1.76	± 0.05	0.75	± 0.06	5.05	±0.27
N	2.15	± 0.10	0.82	± 0.07	0.95	± 0.06	3.91	±0.14
NPK	2.64	± 0.17	1.08	± 0.08	0.79	± 0.08	4.50	±0.30
Clover								
C	2.44	± 0.08	0.78	± 0.23	0.79	± 0.18	4.01	±0.22
US	4.34	± 0.27	2.39	± 0.48	1.13	± 0.05	7.86	±0.48
LD	5.12	± 0.36	2.02	± 0.14	0.89	± 0.11	8.01	±0.64
N	3.26	± 0.32	0.94	± 0.39	0.66	± 0.09	4.86	±0.71
NPK	3.41	± 0.06	0.89	± 0.02	0.82	± 0.06	5.09	±0.08

Values represent means ± SEM (*n* = 5).

5.4.2 Bacterial growth

Bacterial growth in the control treatment decreased in both soils over time. In one soil (organic farmed) at W5, bacterial growth was about 300 pmol Leu g^{-1} h^{-1} (Fig. 5.1A), decreasing to about 150 pmol Leu g^{-1} h^{-1} (Fig. 4.1B) by W11, and remaining at this level until W16 (Fig. 5.1C), a 50% drop in bacterial growth over time. At W5, the bacterial growth in the other soil (conventionally farmed) was about 200 pmol Leu g^{-1} h^{-1} (Fig. 4.1A), decreasing to about 100 pmol Leu g^{-1} h^{-1} by W11 (Fig. 4.1B) and to approximately 30 pmol Leu g^{-1} h^{-1} at W16 (Fig. 5.1C), a decrease in bacterial growth of about 90% from W5 to W16.

At W5, significant differences emerged between bacterial growth in soils treated with the different fertilizer treatments ($p < 0.001$). LD and mineral fertilizers applied to soil all induced higher bacterial growth than the control treatment, while there were no differences between the control and US treatments to the soils. LD fertilizer application to soils induced a bacterial growth 175% greater than US where bacterial growth was lowest (Fig. 5.1A). There were also minor differences in the two other factors of the experiment, soil and crop type. There were different levels of bacterial growth in the two soils ($p < 0.001$), and between crop types ($p < 0.001$).

The second and third harvests indicated a similar pattern to that observed at the first harvest, only with decreasing overall effect sizes from fertilizer treatments over time. Both for weeks 11 and 16, there were significant effects from the fertilizer treatments ($p < 0.001$) (Fig. 5.1B, C), with the change that the smaller effect sizes of the fertilizers led to reduced differences and only the differences between soils amended with LD and the control treatments remained significant. Soil type still had an effect on the level of bacterial growth throughout the experiment ($p < 0.001$), while differences between the crop types were no longer discernable at week 11 and barely discernable at week 16 ($p < 0.05$).

The effect size of the fertilizer treatments consistently decreased over time (Fig. 4.1), as exemplified by the difference between LD and the control starting at more than 100% increase at W5, to about 25% differences at W16, demonstrating an expected strong interaction between time and response to the studied factors.

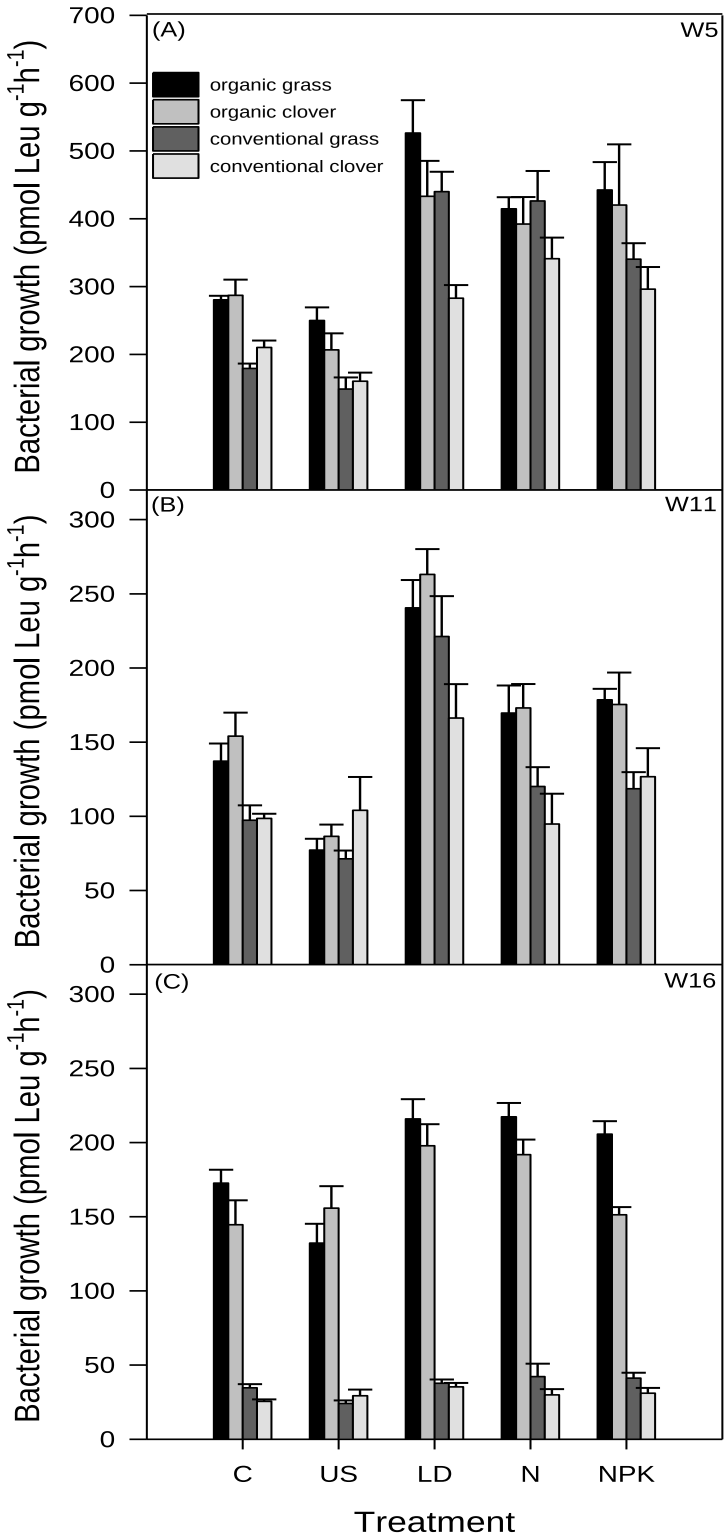

Fig. 5.1: Bacterial growth (leucine incorporation into extracted bacteria) following fertilizer application (150 kg N ha^{-1}): no-fertilizer control (C), undigested slurry (US), liquid digestate (LD), mineral N and mineral NPK in organically and conventionally farmed soils, with grass, or grass /clover crop at three sampling times (Week 5, Panel A; Week 11, Panel B; Week 16, Panel C) post-application. Values represent means with error bars indicating the SE (n = 5 for W5 and W11, and n = 4 for W16). Note difference in y-scales.

5.4.3 Fungal growth

The fertilizer effects on fungal growth were small but detectable throughout the experiment (Fig. 4.2; $p < 0.05$), however, there were no influence of the factors soil type or crop. At week 5, NPK had the highest fungal growth of all treatments, having about 25% greater fungal growth than control (Fig. 5.2A), and was the only treatment to be different from the unamended control treatment ($p < 0.05$), while at W16, the LD treatment to soils induced the highest growth approximately 20% higher than control (Fig. 5.2B). Although there were no statistically significant differences between the soils treated with organic or mineral fertilizers, the soils treated with organic fertilizers tended to be higher.

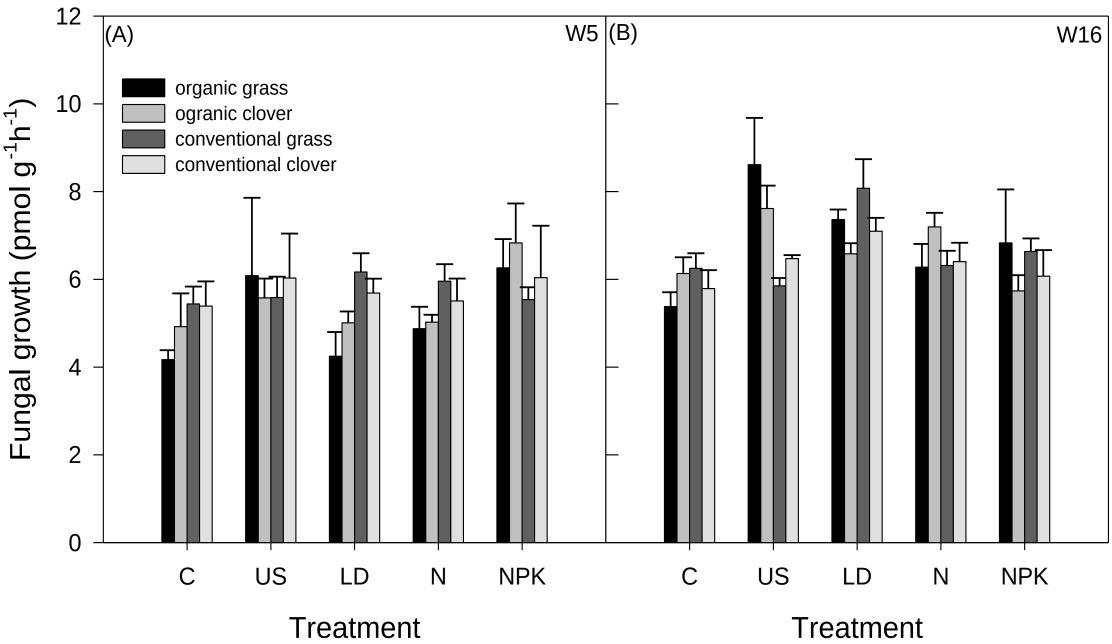

Fig. 5.2. Fungal growth (acetate incorporation into ergosterol) following fertilizer application (150 kg N ha^{-1}): no-fertilizer control (C), undigested slurry (US), liquid digestate (LD), mineral N and mineral NPK in organically and conventionally farmed soils, with grass, or grass /clover crop at two sampling times (Week 5, Panel A; Week 16, Panel B) post-application.

5.4.4 Fungal biomass

There was no change in fungal biomass in the control treatment of one soil (organic farmed) over the duration of the experiment with fungal biomass remaining stable at about 35 µg g^{-1} (Fig. 5.3A, B). However, the fungal biomass decreased over time in the other soil (conventionally farmed) with a 40% decrease in the control treatment between W5 to W16, decreasing from 70 µg g^{-1} to 40 µg g^{-1}.

The fertilizer treatments affected fungal biomass concentrations at week 5 ($p < 0.001$ Fig. 5.3A), with LD and NPK both higher than the control ($p < 0.05$), but differences disappeared by week 16 (Fig. 5.3B). The different soils harboured different concentrations of fungi throughout the experiment ($p < 0.05$; Fig. 3A). While there were no differences between crop types at W5, by W16, differences did emerge ($p < 0.05$).

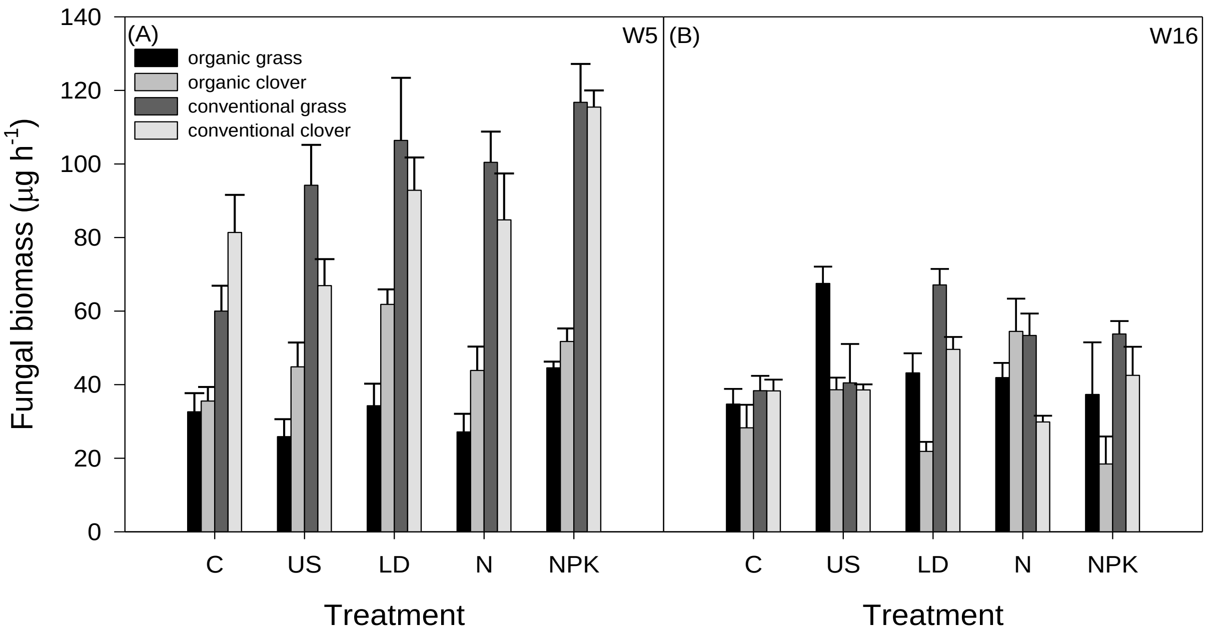

Fig. 5.3. Fungal biomass (ergosterol concentration) following fertilizer application (150 kg N ha^{-1}): no-fertilizer control (C), undigested slurry (US), liquid digestate (LD), mineral N and mineral NPK in organically and conventionally farmed soils, with grass, or grass /clover crop at two sampling times (Week 5, Panel A; Week 16, Panel B) post-application. Values represent means with error bars indicating the SE (n = 5 for W5 and n = 4 for W16).

5.4.5 Fungal:bacterial growth ratio

While effects were small, the fungal-to-bacterial (F:B) growth ratio was affected by fertilizers ($p < 0.001$), and different soils and different crops also harboured different F:B ratios throughout the experiment (all $p < 0.05$).

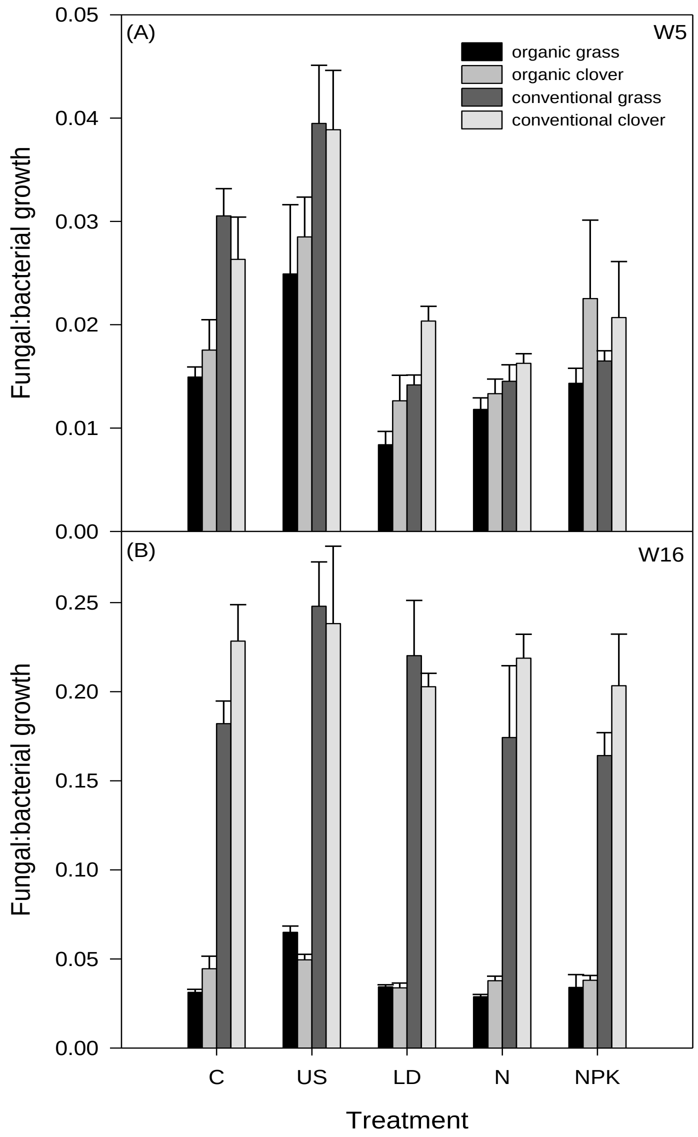

Fig. 5.4: The ratio between fungal and bacterial growth, as an index for the relative dominance of bacteria, following the fertilizer application (150 kg N ha^{-1}): no-fertilizer control (C), undigested slurry (US), liquid digestate (LD), mineral N and mineral NPK in organically and conventionally farmed soils, with grass, or grass /clover crop at two sampling times (Week 5, Panel A; Week 16, Panel B) post-application. Values represent the mean with error bars indicating the SE (*n* = 5 for W5, and *n* = 4 for W16). Fertilizer was added at a rate of 150 kg ha^{-1} over two applications.

5.5 Discussion

As previously reported, the addition of fertilizer increased plant yield in all treatments compared to the unamended control, with greatest yield seen where LD had been applied (Walsh et al., 2012). While crop yields were very different in the two soils, this may be due to the higher fertility of one soil (conventionally farmed) at the onset of the experiment (Table 5.1). Grass-clover had a greater yield than grass, which is economically beneficial to the agricultural sector.

We found a general trend for an increase in the bacterial growth with the addition of N fertilizer, which is in line with work by previous studies investigating manure addition (Böhme et al., 2005; Marschner et al., 2003; Peacock et al., 2001). In the present study, bacterial growth, a more direct measure of the bacterial contribution to resource use and thus to decomposition, responded similarly from LD application as from the mineral fertilizer; while the response to the addition of US was similar to that of the no fertilizer control. This is in contrast to previous studies (Bittman et al., 2005; Sakamoto and Oba, 1994) where evidence for stimulation of bacterial biomass by the application of organic manure compared to mineral fertilizer or no fertilizer has been reported.

A more comprehensive comparison (i.e. replicated) between the history of organically or conventionally farmed (see e.g. Joergensen et al., 2010) is needed before we can assign differences to management type rather than simply differences between individual farms. Therefore, we necessarily need to constrain our conclusions to what we have statistical power to assess, and conclude that the bacterial community reacted differently in different farmed soils, a context dependence that is not surprising. However, with respect to the sown crop, an unanticipated response was the elevated level of bacterial growth in grass compared with grass-clover. This contrasts with previous suggestions that have indicated that N-rich plant materials would tend to stimulate bacterial growth more than N-poor materials (Strickland and Rousk, 2010). However, a previous plant material amendment study suggested that the bacterial growth response following plant material amendment was less related to N richness than the C quality (Rousk and Bååth, 2007).

Crop or soil types were not found to influence fungal growth, differences between fertilizer treatments were subtle and we found no evidence to support the tenet that the fungal community differed between organic or mineral fertilizers over the duration of the experiment. It should be noted that the agricultural soils studied here are typically associated with a bacterial dominated community (Strickland and Rousk, 2010), and it could therefore be argued that the fungal community show only small effect sizes and thus may be relatively poorly resolved. However, this argument would not be supported by the literature since, in other studies of similar agricultural (Rousk et al., 2009; 2010) and grassland (Rousk et al.,

2011) soils, there has been evidence for relatively high fungal growth, and subsequent pronounced responses.

Another possible explanation for the lack of fungal responses to the added fertilizers is a potential interaction between the nutrient effect and a change of the soil pH. Both of the organic fertilizers were slightly alkaline (Table 5.2) and it is possible that this changed the pH of the soil, and favoured bacterial growth while reducing the competitive ability of the fungal community (Rousk et al. 2009; 2010). Soil pH was again measured at the termination of the experiment and remained unchanged in comparison to the initial values. While speculative, a hypothesis yet to explore that could partly explain the observed small fungal responses could be a pH increase (upon application of fertilizer) that would be of a transient nature (not detectable after 16 weeks), which would be consistent with the equally transient microbial responses to the fertilizer additions.

No major changes in fungal growth were apparent throughout the experiment within vegetation cover type, but differences in fungal biomass concentration did emerge; grass had approximately 30% greater fungal biomass than grass-clover in W16. This is in line with work by de Vries et al., (2006) who found that fungal biomass in grass was almost twice as high as in grass-clover. It is surprising that fungal biomass increased over time despite that fungal growth rate was not elevated over the course of the experiment. This discrepancy between fungal biomass and growth could be related to predation. Since we did not assess the level of fungal predation in the present experiment, additional work is needed to verify this hypothesis.

Previous work in soil systems has indicated that there is a large potential for interaction between the major decomposer groups (Rousk et al., 2008; 2010). Furthermore, evidence is accumulating to suggest that bacteria tend to dominate the ecological interactions between these groups, so that fungi grow when conditions are unfavourable for bacteria, while bacteria out compete fungi in conditions of rich resources (Rousk and Bååth, 2011). In line with this, the US treatment, where resources were added in a relatively unavailable form, and where plant growth, and consequently labile rhizodeposits from the plant community, were likely to have been lower, the bacterial community was not favoured, and fungi could freely exploit the available resources. The US introduced higher amounts of C than any other fertilizer treatment and this low-rate addition of a low-quality, fungal promoting resource may provide an explanation as to the increase in fungal biomass over the duration of the experiment in the US treatment.

In general, the highest crop yields also induced the highest overall microbial growth (as a result of unchanging fungi and stimulated bacteria), suggesting that the rhizosphere effect was important in the studied system. Further, plant growth-induced changes in higher quantity and quality of root exudates has also been found to generate shifts in the F:B ratio (Grayston et al., 2001; Mawdsley and Bardgett, 1997). However, previous studies explicitly focused on the influence of concentration of root exudates on the balance of fungal and bacterial decomposers have suggested that higher rates were associated with a shift toward fungi (Griffiths et al., 1999). The lack of systematic pattern highlighted by the inconsistency of these reports suggest that higher crop growth and associated rhizodeposition and its connection bacterial decomposer dominance needs more systematic research attention and that singling out factors for individual study, e.g. type and loading rate of rhizodeposition (e.g. Paterson et al., 2007), are useful paths.

5.6 Conclusions

This trial provides evidence that liquid digestate affected the fungal and bacterial growth in a very similar way to application of mineral fertilizers at comparable rates. Digestate induced a pronounced shift toward a bacterial dominated microbial decomposer community, similar to the effect of mineral fertilizer applications, and effects were consistent in different soils and different sward types. These results can extend work comparing the plant growth promotion of LD vs. mineral fertilizer applications and suggest that mineral fertilizer may be exchanged for LD without affecting plant growth promotion or the actively growing microbial decomposer community. With the microbial decomposer community being the primary providers of functions for plant nutrition and C sequestration in agricultural systems, it is likely that minimal effects on them will translate for equally small effects on soil functioning, although this remains to be explicitly tested. All the above provides increased evidence that AD acts similarly to mineral fertilizer and should be considered as such in its application to land.

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Chapter 6

The effects of applying separated anaerobic digestate, undigested cattle manure and inorganic fertilizer on pasture yield and feed value

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Abstract

On-farm anaerobic digestion (AD) has gained increasing popularity as a means of generating renewable energy. Another product of AD is digestate, which is spread onto land as a soil amendment. Limited field trial data exists on the effect of applying digestate to mixed pasture leys in temperate climates, particularly over more than one growing season. Here we compared yields and forage quality (protein and digestibility) from a field trial of a mixed pasture ley (ryegrass and clover), following the application of five different fertilizer types (liquid digestate generated from anaerobically digested slurry, dry fibre digestate, undigested slurry, ammonium nitrate and a NPK compound fertilizer) in comparison to a no-fertilizer control. Application rates were normalised in terms of dissolved nitrogen (N) and were added as a split dose with 100 kg N ha^{-1} added prior to the first harvest and an additional 50 kg N ha^{-1} supplied after the first harvest, every year for three years. Overall, our results showed that both forms of digestate matched the crop performance obtained with synthetic fertilizers, however, both digestate fertilizers produced greater, although not statistically so, clover yields. No differences were found with regards to digestibility or protein between any treatments. Although the trial was conducted only at one site, the results indicate that synthetic fertilizers can potentially be replaced by digestate without compromising grassland productivity.

Keywords: Animal manure, Biogas, Feed value, Greenhouse gas emissions, Livestock

6.1 Introduction

As global demand for food and energy continue to increase, there is a need to utilize nutrient sources more efficiently. This greater nutrient input is largely provided in the form of synthetic fertilizer. This reliance on synthetic fertilizer usage has an associated increase in energy consumption and greenhouse gas emissions as, during manufacturing, it typically releases 4.96, 1.86 and 0.99 kg of CO_2e per kg of synthetic N, P and K fertilizer, respectively (Econivent, 2007). Due to these economic and environmental concerns, as well as the EU Renewable Energy Directive 2009/28/EC, there is an increasing interest in the use of livestock manure as an alternative source of renewable energy (Demirer & Chen, 2005). Anaerobic digestion (AD) is one such technology that may help increase crop yield without the need for extra synthetic fertilizer, in addition to being both a source of renewable energy and a pollution abatement technology (Walsh et al., 2012a). AD is the decomposition of biologically derived resources in the absence of oxygen (Pain & Hepard, 1985). The products from the AD process include biogas (~70% methane) and a digestate fertilizer which is enriched in N and P relative to the feedstock material (Chadwick et al., 2011; Field et al., 1984; Larsen, 1986; Masse et al., 2007). The biogas can be used as a source of renewable energy to replace fossil fuel, while the digestate is an organic fertilizer which can be applied back to the land. AD may be particularly appealing to livestock farmers as they produce large quantities of animal manures, which are suitable feedstock for AD, and can also provide additional sources of revenue (Demirer & Chen, 2005). Currently there are approximately 100 food- and animal-waste fed AD units in operation in the UK (NNFCC, 2012)' although there is potenital for considerably more (ADAS, 2012). Co-digestion will lead to a greater quantity of organic manures being treated prior to land disposal (Moeller et al., 2011).

Although synthetic fertilizers are often considered to represent a more effective and controllable source of plant nutrients, organic fertilizers offer extra benefits above which synthetic fertilizers are able to provide. These can include: enhancing the microbial activity and biomass of the soil (Garcia-Gil et al., 2000; Powlson et al., 1987); enhancing soil organic matter and consequently improving soil structure, porosity and drainage (Choudhary et al., 1996); and supplying nutrients in a more bioavailable form (Odlare et al., 2011). In addition, compared to undigested animal manures, digestate possesses a relatively low carbon-to-nitrogen ratio, reduced biological and chemical oxygen demand, elevated pH values, higher ammonium (NH_4^+) as a percentage of total nitrogen, and reduced viscosities (Chantigny et al., 2007; Masse et al., 2011; Möller et al., 2008). After digestion, the digestate can be separated into liquid and dry fibre portions, with the dry fibre having a similar texture to compost and a dry matter (DM) content of ~20%. The liquid digestate will have a DM between 4-6%, and is characterized by low P and high N and K contents (Moeller et al., 2010). Thus, the N, P and K are partitioned according to the separated liquid and dry fibre digestate (Bauer et al., 2009; Liedl et al., 2006; Möller et al., 2008).

Although a plethora of information exists in regard to the benefits of biogas from AD (Cavinato et al., 2010; DeVuyst et al., 2011), the agronomic benefits of digestate are less well documented with many contradictory reports present within the literature. Some studies have reported higher crop yields following digestate application (de Boer, 2008; Garg et al., 2005; Pathak et al., 1992; Rubaek et al., 1996; Svensson et al., 2004), while others have reported no difference (Loria & Sawyer, 2005; Möller et al., 2008). Others report the same crop yield as when synthetic fertilizer is applied (Liedl et al., 2006; Walsh et al., 2012a), though some studies witnessed a lower yield than synthetic fertilizer (Quakernack et al., 2012). A search of the literature however, failed to find studies that have investigated the effect of digestate application on forage quality (e.g. protein content and digestibility); even though such information would be useful for efficient grazing management.

The aim of this study was to determine the effects of the repeated application of different fertilizers, including AD digestate, on a mixed pasture ley over three growing seasons. The key indicators used to evaluate treatment performance were dry matter yield, shifts in pasture composition, forage protein and digestibility.

6.2 Materials and methods

6.2.1 Experimental design

The experimental field site was located on freely draining agricultural grassland located in Abergwyngregyn, Gwynedd, North Wales (53°14’05’’N, 4°00’50’’W). The sward contained a mixture of perennial rye grass (*Lolium perenne* L.) and white clover (*Trifolium repens*, L.) and was previously subject to sheep grazing (ca. 15 ewes ha^{-1}). The soil has a clay-loam texture and is classified as a Eutric Cambisol (of the ‘Denbigh’ series) and is derived from mixed glacial till.

Five different fertilizer treatments were applied to 2 × 2 m plots (n = 4), organised in a randomised design. These included: a no fertilizer control (C); undigested cow slurry(US);

the liquid fraction of anaerobically digested cow slurry (liquid digestate, LD); the dry fibre fraction of anaerobically digested cow slurry (dry fibre digestate, DFD); synthetic N 34.5% fertilizer (N; ammonium nitrate) and a synthetic NPK 21.8.11 (NPK) compound fertilizer. Over a three year period, six above-ground vegetation harvests were performed on the plots with two harvests taken per year (May-June and August-September). Weather patterns were recorded over the trial period and total monthly rainfall and mean monthly temperature are reported (Figure 6.1).

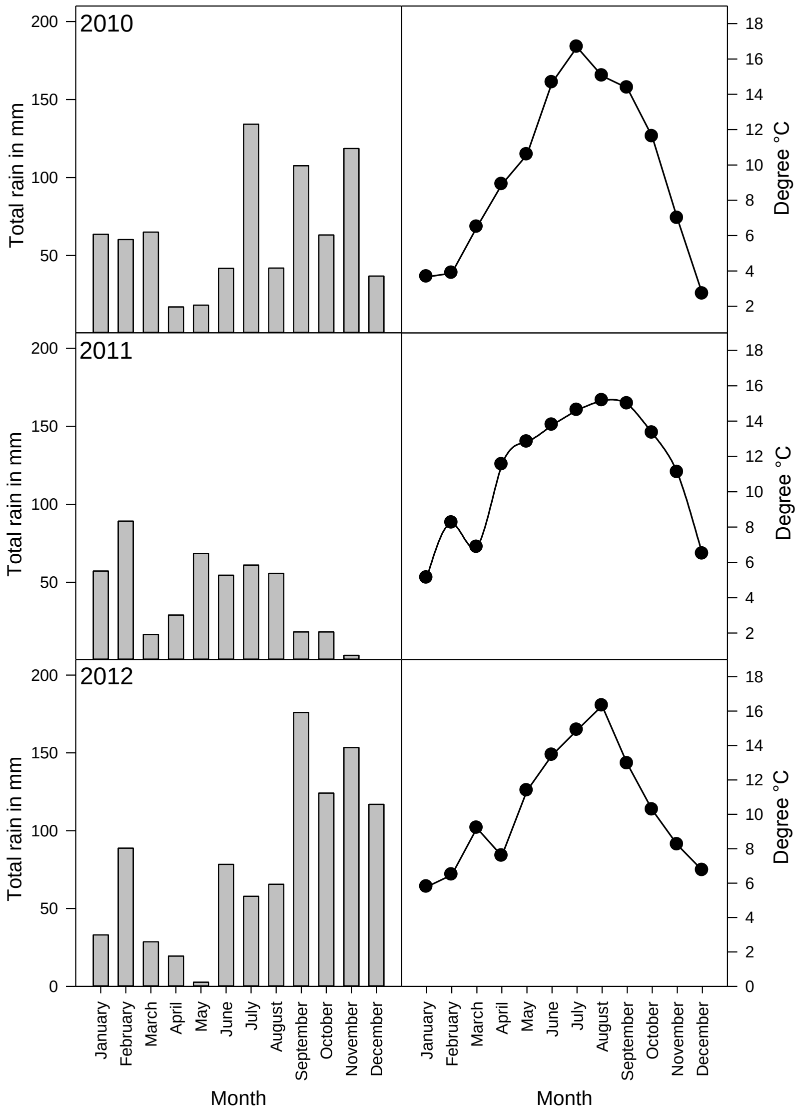

Figure 6.1: Total monthly rainfall and mean monthly temperature over the trial period, with rain in the left column and temperature in the right.

To represent farmer practice, the first harvest was undertaken six weeks after the initial application of 100 kg N ha^{-1} of each fertilizer type. A second fertilizer addition of 50 kg N ha^{-}

^{1} was then applied one week post-harvest and after an additional six weeks, the final harvest of that year was taken. The application rate for each was normalised for nitrogen, based on mineral N (ammonium N) values and total nitrogen content for the synthetic fertilizers. With the exception of the first harvest, the harvested material was manually separated to determine the proportion of grass and clover in the sword. Only the plant biomass within the central 1 m^2 of the plots was quantitatively evaluated to avoid potential edge effects. Soil samples at a depth of 150 mm were taken from each plot at the very beginning of the experiment, after the third harvest and again at the end of the final harvest in year 3. All harvested plant material was weighed wet, and then a 300 g subsample was removed, dried at 85 °C for 48 h, and reweighed. Crop nutrient analysis was undertaken in both harvests in year three of the trial to determine total nitrogen and carbon content of the shoots. Protein content was calculated by multiplying the nitrogen reading by 6.25, which is the industry standard, however, this tends to overestimate the true protein of feedstocks (Sriperm et al., 2011). Digestibility was calculated using the MAD fibre content of each sample (Yara, 2013). The harvesting and drying of the crop can be seen from pictures 6.1 to 6.4.

Picture 6.1: Field trial one day before harvesting

Picture 6.2: Field trial with edge effect waste disregarded, to allow for treatment collection

Picture 6.3: Field trial with treatment harvests in bags for transport to the lab

Picture 6.4: Wet samples (each 300 g from one meter squared plot), dried for further analysis

6.2.2 Soil and fertilizer characterization

Soil (0-15 cm) and organic fertilizer were extracted with deionised water 1:5 (w/v), shaken (250 rev min^{-1}, 1 h, 20 °C), centrifuged (4000 *g*, 15 min) and the supernatant filtered (Whatman no. 42). Major cations (K^+, Na^+ and Ca^{2+}) were analysed using a model 410 flame photometer (Sherwood Scientific, Cambridge, UK) whilst NO_3^-, NH_4^+ and P were determined colorimetrically (Synergy® Microplate Reader; BioTek US, Winooski, VT) using the methods of Mulvaney (1996), Miranda et al. (2001) and Murphy and Riley (1962), respectively. Total dissolved N (TDN) and dissolved organic N (DON) were determined using a TCN-V analyser (Shimadzu Corp., Kyoto, Japan) and total C and N were analysed using a TruSpec® elemental analyser (Leco Corp., St Joseph, MI). Samples were oven dried at 105 °C for 24 h to determine gravimetric water content. Electrical conductivity (EC) and soil pH were determined using standard electrodes in 1:5 (w/v) distilled water extracts.

6.2.3 Predicted ammonia emissions and nitrate leaching

Fertilizer application rate, dry matter content, total nitrogen and total NH_4^+ of the undigested slurry, liquid digestate and dry fibre digestate were inputted into the computer programme MANNER v4.0 (Chambers et al., 1999). MANNER is a software application that allows the user to determine the potential N volatilisation and leaching of organic fertilizer for different regions of the UK. A 3-year average of the fertilizer value (nutrient content) was used (rather than three individual years) to determine what the potential greenhouse gas emission reduction and leaching may have been from all three organic fertilizers over the experimental period.

6.2.4 Statistical analysis

Statistical analysis was performed using SPSS v.18 (IBM UK Ltd., Hampshire, UK). For analysis of crop yield data, total yield from all harvests were used and subject firstly to a one-way ANOVA to determine differences within each sub-group, with treatment as the factor. The same analysis was used for carbon, nitrogen and digestibility tests. Post-hoc tests were carried out on all ANOVAs using Tukey HSD test at the level ($p < 0.05$).

6.3 Results

5.3.1 Soil and fertilizer characterization

Tables 6.1-6.3 report the physico-chemical properties of the three fertilizers used over the duration of the field trial, undigested slurry (US), liquid digestate (LD) and dry fibre digestate (DFD), for each individual year.

Table 6.1: Physico-chemical properties of the organic fertilizers used in year 1 (2010) of the field trial, undigested slurry (US), liquid digestate (LD) and dry fibre digestate (DFD). Values represent means ± SEM (*n* = 3) and are expressed in terms of dry weight where applicable.

	Fertilizer		
	US	LD	DFD
pH	7.55 ± 0.12	8.59 ± 0.11	9.08 ± 0.21
EC (mS cm^{-1})	9.01 ± 0.14	12.2 ± 0.1	20.20 ± 1.47
Dry matter (%)	14.3 ± 0.26	5.2 ± 0.3	22.42 ± 0.31
DOC (mg g^{-1})	35.3 ± 0.2	30.0 ± 0.9	23.04 ± 1.61
DON (mg g^{-1})	11.6 ± 0.1	27.4 ± 1.3	8.67 ± 0.58
NO_3^- (mg g^{-1})	0.31 ± 0.15	0.51 ± 0.04	0.34 ± 0.11
NH_4^+ (mg g^{-1})	6.54 ± 0.25	20.35 ± 0.53	23.55 ± 0.25
P (mg g^{-1})	10.6 ± 0.8	1.0 ± 0.2	4.49 ± 0.76
K (mg g^{-1})	9.1 ± 0.1	16.5 ± 0.0	19.49 ± 4.01
Ca (mg g^{-1})	13.9 ± 0.1	19.5 ± 0.1	2.34 ± 0.64
Na (mg g^{-1})	3.6 ± 0.1	7.2 ± 0.2	7.81 ± 1.39

Table 6.2: Physico-chemical properties of the organic fertilizers used in year 2 (2011) of the field trial, undigested slurry (US), liquid digestate (LD) and dry fibre digestate (DFD). Values represent means ± SEM (*n* = 3) and are expressed in terms of dry weight where applicable.

	Fertilizer		
	US	LD	DF
pH	6.82 ± 0.02	8.21 ± 0.03	8.94 ± 0.02
EC (mS cm^{-1})	3.6 ± 0.85	3.8 ± 0.60	2.4 ± 0.80
Dry matter (%)	17.36± 0.35	3.74 ± 0.05	23.39 ± 0.71
DOC (mg g^{-1})	58.75 ± 0.57	34.81 ± 3.91	23.90 ± 0.24
DON (mg g^{-1})	13.34 ± 0.07	18.79 ± 0.15	7.98 ± 0.02
NO_3^- (mg g^{-1})	0.48 ± 0.01	0.28 ± 0.02	0.14 ± 0.03
NH_4^+ (mg g^{-1})	17.2± 3	37.3 ± 13	23.2 ± 17
P (mg g^{-1})	16.1 ± 0.79	5.8 ± 0.16	8.18 ± 0.5
K (mg g^{-1})	19.17 ± 0.12	13.93± 0.32	12.53 ± 0.59
Ca (mg g^{-1})	5 ± 0.11	3.33 ± 0.21	3.45 ± 0.48
Na (mg g^{-1})	1.17± 0.01	3.34 ± 0.17	5.37 ± 0.15

Table 6.3: Physico-chemical properties of the organic fertilizers used in year 3 (2012) of the field trial, undigested slurry (US), liquid digestate (LD) and dry fibre digestate (DFD). Values represent means ± SEM (*n* = 3) and are expressed in terms of dry weight where applicable.

	Fertilizer		
	US	LD	DF
pH	6.83 ± 0.32	8.51 ± 0.02	8.64 ± 0.10
EC (mS cm^{-1})	4.44 ± 0.76	3.72 ± 0.06	1.76± 0.14
Dry matter (%)	11.66± 0.06	5.84± 0.23	26.95±2.78
DOC (mg g^{-1})	80.41 ± 1.87	76.84 ± 0.44	40.54 ± 0.87
DON (mg g^{-1})	14.62 ± 0.21	35.62 ± 0.32	13.72 ± 0.24
NO_3^- (mg g^{-1})	0.54 ± 0.03	0.26 ± 0.03	12.24± 0.07
NH_4^+ (mg g^{-1})	22.42± 0.83	48.01±2.74	23.99±1.56
P (mg g^{-1})	8.41 ± 0.05	4.28 ± 0.13	5.81 ± 0.09
K (mg g^{-1})	9.31± 0.47	9.81± 0.77	6.18± 0.51
Ca (mg g^{-1})	2.55± 0.10	1.94 ± 0.10	1.22 ± 0.02
Na (mg g^{-1})	3.82 ± 0.04	13.81± 0.21	10.22± 0.16

Tables 6.4-6.6 report the physico-chemical properties from the soil where the field trials took place. Table 6.4 is a mean of soil characteristics from a range of plots before any fertilizer was applied. Tables 6.5 and 6.6 are subdivided into the different treatments, control, pasture applied undigested manure (US), pasture applied liquid digestate (LD), pasture applied dry fibre digestate (DFD), and pasture applied synthetic nitrogen fertilizer as either NPK, or straight N.

Table 6.4: Physico-chemical properties of soil used in the study in year 1 (2010). Values represent means ± SEM (n = 3) and are expressed in terms of dry weight where applicable.

	Soil
pH	5.45 ± 0.02
EC (µS cm^{-1})	44.4 ± 5.4
Dry matter (%)	78.3 ± 0.2
Total C (mg g^{-1})	29.1 ± 0.5
Total N (mg g^{-1})	3.11 ± 0.07
C:N	9 ± 0.13
DOC (mg g^{-1})	0.11 ± 0.01
NO_3^- (µg g^{-1})	20 ± 1.2
NH_4^+ (µg g^{-1})	9 ±1.3
P (µg g^{-1})	90 ± 5
K (µg g^{-1})	30 ± 4
Ca (µg g^{-1})	36 ± 3
Na (µg g^{-1})	65 ± 6

Table 6.5: Physico-chemical properties of soil used in the study in year 2 (2011). Values represent means ± SEM (n = 3) and are expressed in terms of dry weight where applicable.

	Fertilizer					
	C	US	LD	DFD	NPK	N
pH	7.11 ± 0.02	6.96 ± 0.19	6.81 ± 0.23	6.66 ± 0.02	6.52 ± 0.02	6.57 ± 0.18
EC (µS cm^{-1})	82.7 ± 18.5	91.4 ± 15.3	70.1 ± 8.6	128.9 ±12.4	73.2 ±13.3	63.9 ± 3.2
Dry matter (%)	79.18± 0.76	80.27± 0.58	79.31± 0.55	78.91± 0.91	78.96± 0.56	78.63± 0.34
NO_3^- (µg g^{-1})	17.87±2.54	16.12±1.52	17.96±1.65	23.86±1.85	14.71±1.01	23.05 ±1.39
NH_4^+ (µg g^{-1})	16.02± 0.74	13.08± 0.56	16.42±2.16	16.45± 0.55	14.71± 0.08.	14.35 ±3.78
P (µg g^{-1})	84.58±5.99	92.62±3.1	96.35±7.27	90.81±5.6	100.74±10	113.02±9.8
K (µg g^{-1})	27 ± 4.12	39.6 ±5.65	52.2 ± 4.97	84.6±10.4	28.8 ±4.47	23.4 ± 3.01
Ca (µg g^{-1})	7.2 ± 1.55	23.4 ±2.66	23.4 ± 3.45	50.4 ±3.81	5.4 ± 2.46	5.4 ± 1.97
Na (µg g^{-1})	36 ± 6.38	37.8±4.91	72 ± 10.63	68.4 ±6.03	37.8 ±2.74	34.2 ± 1.98

Table 6.6: Physico-chemical properties of soil used in the study in year 3 (2012). Values represent means ± SEM (n = 3) and are expressed in terms of dry weight where applicable.

	soil					
	C	US	LD	DFD	NPK	N
pH	6.40 ± 0.23	6.82 ± 0.18	6.86 ± 0.48	7.34 ± 0.03	6.52 ± 0.14	6.49 ± 0.19
EC (µS cm^{-1})	57 ± 7.87	78.07±9.64	77.26±9.01	92.12±4.49	53.53±5.08	48.12±2.61
Dry matter (%)	78.17± 0.65	76.97± 0.19	72.28±1.31	80.68±4.75	77.21± 0.75	74.87±2.38
NO_3^- (µg g^{-1})	4.75 ± 0.44	6.19 ± 0.51	5.67 ± 0.81	5.31 ± 0.72	5.65± 0.47	5.72 ± 0.45
NH_4^+ (µg g^{-1})	1.48 ± 0.13	1.15 ± 0.04	1.62 ± 0.12	0.91 ± 0.02	1.04 ± 0.04	1.08 ± 0.05
P (µg g^{-1})	24.66± 0.91	16.56± 0.45	24.48± 0.51	47.88± 0.49	9.36 ± 0.12	7.28 ± 0.11
K (µg g^{-1})	5.85 ± 1.63	8.92 ± 1.89	10.69±1.91	8.73 ± 0.57	3.27 ± 0.72	3.18± 0.66
Ca (µg g^{-1})	5.42 ± 0.91	8.44 ± 1.13	9.57 ± 1.11	13.24± 0.64	6.03 ± 0.84	5.53± 0.33
Na (µg g^{-1})	7.31± 0.74	10.84± 0.67	11.03± 0.88	11.02± 0.83	8.84± 0.43	7.68 ± 0.54

6.3.1 Cumulative forage yields

The cumulative forage crop yield across all six harvests showed that plots applied liquid digestate (LD) had the greatest yield; however, it was not significantly different (p > 0.05) from pasture applied undigested slurry (US), digestate fibre (DFD) or standard NPK fertilizer treatments. There was no significant difference (p > 0.05) between the two synthetic fertilizer treatments, and pasture applied synthetic N was not statistically different from the zero amendment (control), which had the lowest overall yield; the reason for which may be due to the greater quantity of clover in the control treatment.

When comparing individual years, there were statistically significant differences between treatments (p < 0.001) and harvest times, except for pasture applied synthetic N. Within the control treatment, year three had the greatest yield in comparison to years 1 and 2 (p < 0.05) which did not differ significantly. Similar results were reported for pasture applied US and LD. Pasture applied synthetic N and DFD yielded most in year three (p < 0.05) and both treatments had their lowest yield in year 2. The crop yield was different in all years (p < 0.05) for NPK, with year two having the lowest yield and year three having the greatest. One constant throughout was that year three had the greatest yield of all treatments above other years harvests.

6.3.2 Impact of fertilizer treatment on sward composition

From harvest 2 onwards, all treatments were separated into grass and clover. A cumulative total from harvest 2 to 6 showed that pasture applied LD had the greatest yield of grass; however, it was not significantly different ($p > 0.05$) from pasture applied US, DFD or NPK treatments (Fig. 6.2). Similarly, there was no significant difference in grass yield between both synthetic fertilizers, while the control treatment had the lowest yield but was not significantly different ($p > 0.05$) from synthetic N. When the cumulative biomass of clover from harvests 2 to 6 was analysed, pasture applied US produced the greatest clover yield of all treatments. However, pasture applied US when compared to other treatments, was only significantly different ($p < 0.05$) than swards amended with synthetic N, and no difference emerged between other treatments. An individual observation of each year can be seen in Fig 6.4 where both harvests from each year were accumulated to show total crop yield in that year. All years followed the same statistical pattern as individual harvests within that year and thus there are no statistical differences shown in Fig. 6.3.

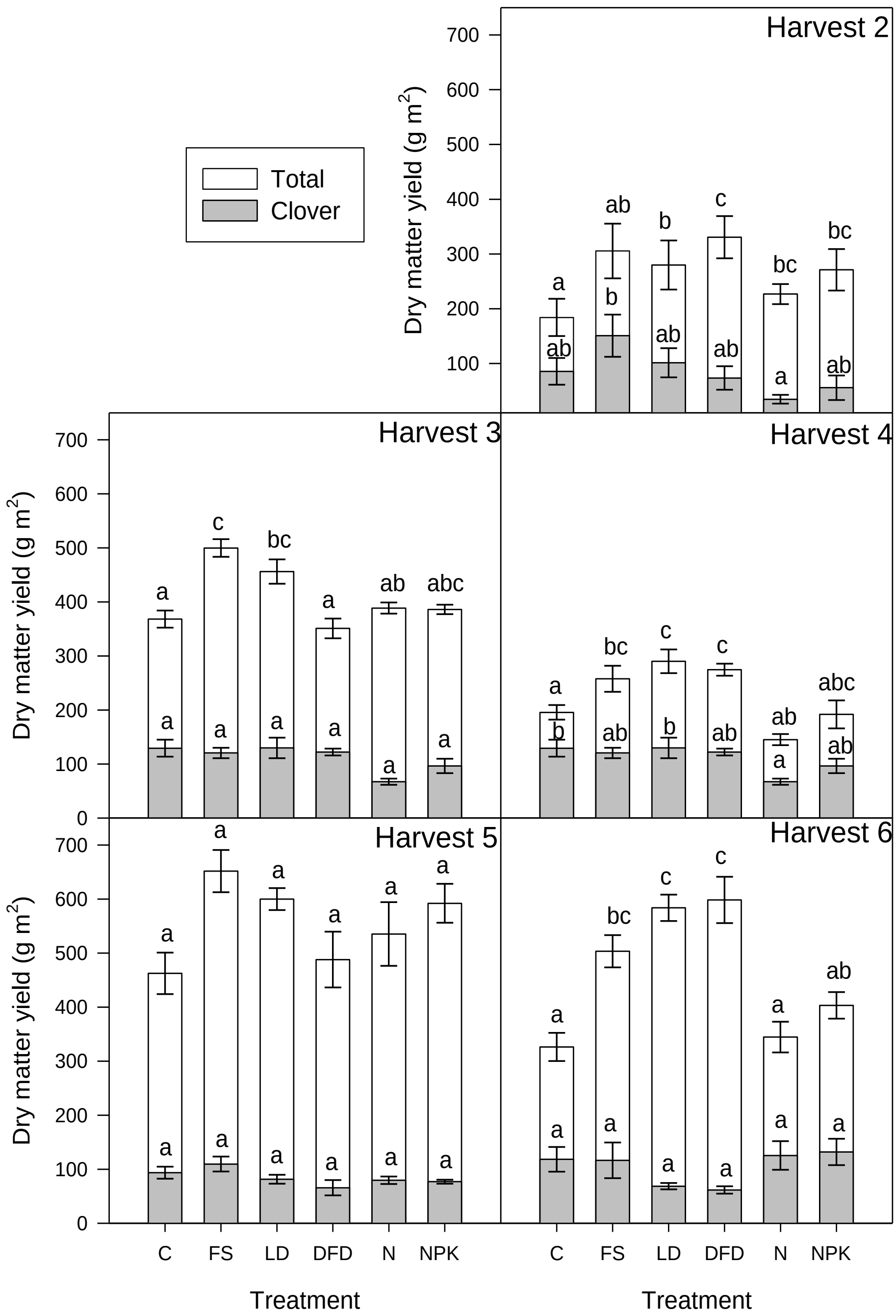

Figure 6.2: Proportion of grass and clover in swards after treatment with fertilizer regimes. Treatments are no-fertilizer control (C), undigested slurry (US), liquid digestate (LD), dry fibre digestate (DFD), mineral nitrogen (N) and mineral NPK (NPK). Values represent the mean ± SEM (n = 4). Lowercase letters within graphs denote differences ($p < 0.05$) between treatments within the same sub-group, for grass and yield quantity in that harvest

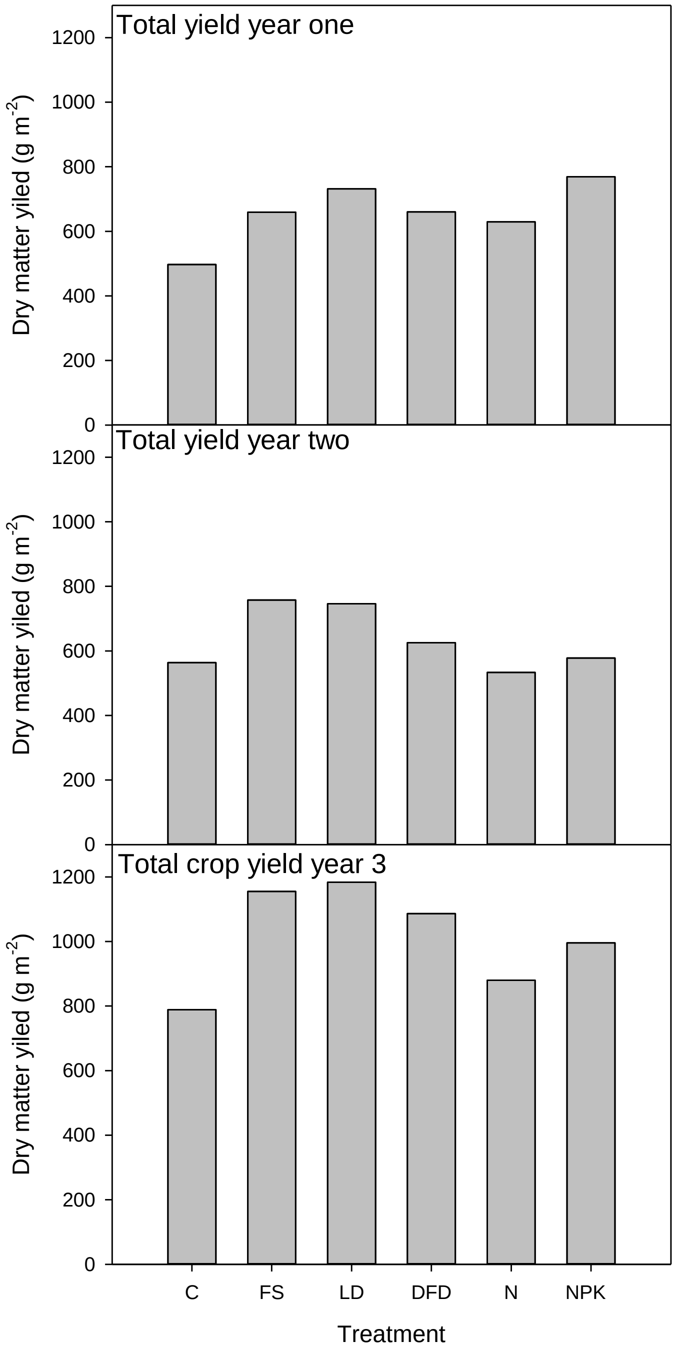

Figure 6.3: Total yearly crop yield after the application of different fertilizer types over three years with two harvests per year: no-fertilizer control (C), undigested slurry (US), liquid digestate (LD), dry fibre digestate (DFD), mineral nitrogen (N) and mineral NPK (NPK).

As a total of the three years harvests, the unamended control had the greatest proportion of clover (of total yield), and was significantly greater ($p < 0.05$) from all other treatments (Table 6.7). Pasture applied FS had the greatest percentage of clover of all treatments applied fertilizer, and was significantly different ($p < 0.05$) from the synthetic fertilizers and DFD. However, no significant difference was revealed between US and LD ($p > 0.05$). After the accumulation of three years' harvests, there were no differences in clover percentage between pasture applied LD and DFD and that applied the two synthetic fertilizers ($p > 0.05$).

Table 6.7: Percentage of clover from total of 5 harvests, values are in dry matter yield per m^2.

Treatment	Total yield ($g\ m^{-2}$)	Grass yield ($g\ m^{-2}$)	Clover yield ($g\ m^{-2}$)	Clover (%)
Control	1849.7 ±108.5	1075.77 ±70.05	644.74 ±85.39	34.9
US	2571.6 ±38.01	1748.38 ±111.39	755 ±101.62	27.2
LD	2661.4 ±75.53	1902.54 ±71.25	587.28 ±88.74	22.1
DFD	2372 ±38.82	1848.65 ±61.37	524.55 ±24.11	22.1
N (34.5%)	2042.7 ±67.94	1413.41 ±86.10	453.91 ±38.13	22.2
NPK	2342.1 ±83.83	1609.2 ±62.67	506.36 ±39.10	21.6

6.3.3 Impact of fertilizer treatment on sward N content

At harvest 5, there were significant differences between foliar nitrogen levels amongst treatments in both separated grass and separated clover ($p < 0.001$). Within grass, all treatments had greater levels of nitrogen ($p < 0.05$) than control, but there were no differences between any of the other fertilizer treatments ($p > 0.05$). With respect to clover, the DFD treatment possessed the greatest amount of nitrogen of all treatments, being significantly different from the synthetic N and NPK treatments which possessed the lowest N levels ($p < 0.05$). Again there was no statistical difference ($p > 0.05$) between the other treatments.

At harvest 6, differences were also apparent in foliar nitrogen content between the different grass treatments ($p < 0.05$), with the control and DFD treatments showing the lowest levels of nitrogen and being significantly different ($p < 0.05$) from the synthetic N which had the greatest levels of nitrogen. No differences emerged between any other treatments. Again with clover, the synthetic N treatment had the lowest level of leaf nitrogen, being significantly different from pasture applied LD and DFD ($p < 0.05$) which had the greatest foliar N concentrations whilst no significant differences emerged between any other treatments.

6.3.4 Forage digestibility

Digestibility of the above-ground foliage was analysed from all treatments for the final two harvests (year three only). Typically, digestibility ranged from 60-70%. LD had the lowest digestibility of all treatments at 60% while all other treatments reported a value of 62%; however, none of these differences were statistically significant ($p > 0.05$).

6.3.5 Prediction of potential N leaching and N volatilisation

The MANNER programme predicted treatment response in relation to NH_3 emissions and NO_3^- leaching from the application of organic fertilizers. The results indicated that US had the lowest potential for leaching at 14 kg N ha^{-1}, and a volatilization rate of 5 kg N ha^{-1}. With digestate, LD reporting the greatest potential to leach at 29 kg N ha^{-1}, and a volatilization rate of 7 kg N ha^{-1}, while DFD had a leaching potential of 22 kg N ha^{-1} and a volatilisation rate of 8 kg N ha^{-1}.

6.4 Discussion

6.4.1 Crop yield and sward composition

The results presented here show that, cumulatively, organic fertilizers derived from anaerobic digestion gave the same forage yield as that provided by synthetic fertilizers over a (medium-term) three year period. This contrasts with results obtained from some previous trials (Möller and Müller, 2012) but supports the findings of others (Morris & Lathwell, 2004). A cause of the disparity in the yield results can be expected due to forage type, and whether the experiment was a field- or pot-scale trial. Generally, pot-scale experiments have reported higher yields, ranging from 10-25% from the application of digestate compared to undigested fertilizer (Bougnom et al., 2012; Morris & Lathwell, 2004; Walsh et al., 2012a). In contrast, field trials rarely demonstrate the same positive growth response (Möller & Müller, 2012), with positive effects from the application of digestate reported in some years, and not in others. However, studies from field trials consisting of more than one growing season are uncommon. A possible explanation for the differences in the field and pot trials may be due to the application method of the fertilizer, being surface applied rather than incorporated within the soil profile (Möller & Müller, 2012), or that pot trials are often set under controlled conditions. In field trials over a longer harvesting period, the extra rooting volume of crops compared to pots may lead to the acquisition of mineralised organic nitrogen from manures by crops (Morris & Lathwell, 2004; Möller et al., 2008; Svensson et al., 2004). In comparison with straight N synthetic fertilizer, organic fertilizers provide additional nutrients in the form of phosphorus and potassium; which may explain the lower yields from pasture applied the former (Fig. 6.2). In the longer term, digestate application, relative to synthetic fertilizer may also increase soil organic matter and hence the retention of nutrients and improve overall soil quality.

The application of nitrogen is frequently reported to suppress clover growth (Hakala et al., 2012; Nesheim et al., 1990). However, Bougnom et al. (2012) reported greater percentage of legumes in soil treated with undigested manure rather than digestate, which corroborates our results (Fig. 6.2C). After three years there is often a decrease in the clover content of a mixed grass clover ley (Mela, 2003). Hakala et al. (2012) reported that at the end of a three year field trial, differences between synthetic and organic fertilizer in clover yield were minimal, but in general clover plants were higher in organic fertilizer compared to synthetic fertilizer, again similar results were seen in this study. At the end of this study, control plots had the greatest percentage of clover, followed by pasture applied with US, which were both higher than all other fertilized treatments (Table 6.7). Thus, although digestate is an organic fertilizer, it may restrict clover growth over time in a similar way to synthetic fertilizer, possible due to the higher amount of plant available nitrogen in digestate fertilizer compared to US. This may further support the concept that LD affects nutrient dynamics within the soil in a similar way to that of synthetic fertilizer (Walsh et al., 2012b).

6.4.2 Forage carbon, nitrogen/protein and digestibility

Foliar nitrogen content correlates with leaf protein at a ratio of 6.25:1 and is the industry standard for converting nitrogen to protein (Sriperm et al., 2011). Protein content is an important parameter of feed value and was calculated in this study. After three years of fertilizer application, little difference in N/protein content was seen between all five fertilizer treatments. However, there was a (statistically insignificant) trend within clover for higher nitrogen levels from organic fertilizers. If further work revealed this to be true, this may prove important for farmers as a means of reducing reliance on synthetic fertilizer, and the importation of concentrate for animals.

In this study, no difference in relation to digestibility between treatments was reported. Digestibility of plant tissue typically increases with N fertilization (Johnson et al., 2001; Messman et al., 1992; Prine & Burton, 1956). The fact that all harvest were taken within 6 weeks of fertilizer application, before digestibility would start to decrease, may explain the lack of treatment effect. It would be beneficial to have a longer harvesting time to determine if differences in digestibility would occur after an additional 2-3 weeks growth.

6.4.3 Gaseous N emissions

With greenhouse gas (GHG) emissions gaining greater prominence in the agriculture sector, it is important to understand the GHG emissions from the application of all fertilizers (organic and inorganic). Methane (CH_4) loss from manure management ranges from 12-41% of total CH_4 emissions from agriculture worldwide (Chadwick et al., 2011), and AD has the potential to reduce the CH_4 losses from manure during storage and application (Sommer & Moller, 2000). During the digestion process, the volatile solids in manures are reduced; this has a knock-on effect of lowering the risk of nitrous oxide (N_2O) emissions from digestate applied to land due to the decrease in microbial demand for oxygen (Chadwick et al., 2011; Petersen et al., 1996). However, digestate has a higher NH_4^+ concentration and a higher pH than undigested manure, and this can lead to greater levels of ammonia losses from digestate compared to undigested manure (Gericke et al., 2012; Möller et al., 2008). However, digestate has a low DM content and facilitates slurry injection, which itself reduces ammonia volatilization by between 47-72% compared to surface application (Rubaek et al., 1996). Results from MANNER show that pasture applied LD and DFD had greater potential levels of N leaching and volatilization than pasture applied US, although the programme is restricted in the GHG emissions it calculates. When all GHG emissions are accounted for, studies have reported that there is approximately a 60% reduction in total GHG emissions per m^3 of cattle manure digested during storage and application compared to US (Amon et al., 2006).

6.5 Conclusion

Although this research was conducted on one crop and soil type and at one geographical location, it demonstrates the potential value of digestate as a fertilizer source for pasture systems. The process of AD appears to increase the agronomic value of manure. Further, the study implies that application of digestate, although an organic fertilizer leads to similar responses in pasture yield as when synthetic fertilizer is applied. Any agronomic benefits of replacing synthetic fertilizer use with digestate should be viewed alongside the long-term wider environmental benefits (e.g. in reducing GHG or loss of N to freshwater). Further work is needed at the field scale to fully explore the agronomic value of digestate under different environmental conditions, and soil and crop types.

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Chapter 7

Valuing the non-market environmental benefits from the anaerobic digestion of livestock resources in the UK

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†G. Edwards-Jones was involved in the original grant proposal and conceptual design of this PhD and therefore is credited posthumously.

Abstract

Anaerobic Digestion (AD) is seen primarily as a source of low-carbon renewable energy. However, the introduction of AD can generate a number of additional positive environmental externalities above those provided by other forms of renewable energy, including greenhouse gas reduction from livestock production systems, closing the nutrient cycle within agriculture; as well as offering an effective replacement for synthetic fertilizer; reducing leaching of nutrients to waterways; reducing the biological and chemical oxygen demand of wastes; pathogen loads; and malodour. This study synthesises the scientific and economic literature on AD to estimate the value of these benefits. We estimate the total environmental non-market benefits from AD of livestock waste to range from £1-5m for each 1% of livestock waste anaerobically digested in the UK, equivalent to £0.03 - £0.15 per kWh of electricity generated from AD. If these non-market benefits were appropriately valued by policy, the UK government's incentive for renewable energy, the Feed-In-Tariffs (FIT) rate should increase to £0.12 - £0.30 per kWh. The findings indicate that current incentives for renewable energy undervalue AD and that energy subsidies should be better aligned to include these wider environmental benefits. The results provide a great deal of variability in valuation, this goes to highlight the current lack of understanding as to the actual level of environmental benefits from the introduction of on-farm AD.

Keywords: Biogas, Biomass, Digestate, Greenhouse gas abatement, Positive externalities, Renewable energy

7.1 Introduction

Anaerobic digestion (AD) is the microbial decomposition of organic material in an oxygen-free environment leading to the production of biogas, predominantly composed of methane (CH_4) to be used as an energy source, and a nutrient-rich digestate (Pain & Hepard, 1985). The biogas produced by AD can be used to replace fossil fuels in energy production, and applying the digestate to land helps close the nutrient cycle, and reduce demand for synthetic fertilizers. AD is particularly suitable for use on livestock farms, due to the large amounts of waste produced. Indeed, approximately 1.5 billion tonnes of animal manure is produced in the EU 27 on a yearly basis (Holm-Nielsen et al., 2009). However, the majority of this manure is currently applied directly to land, meaning that a potentially valuable commodity is not being used to its full potential. AD can add value to animal manure by producing renewable energy, in the form of heat and electricity. The biogas can also be upgraded and used as a transport fuel (Patterson et al., 2011a), allowing AD to contribute to targets set out in the EU Renewable Energy Directive 2009/28/EC and the Fuel Quality Directive 2009/30/EC.

Although the market benefits of AD, i.e. energy from biogas combustion (Capponi et al., 2012; Cavinato et al., 2010; DeVuyst et al., 2011; Moller et al., 2004) and digestate (Masse et al., 2011; Morse et al., 1996) are well reviewed in the literature, values for non-market benefits associated with on-farm AD (i.e. the reduction in negative environmental externalities associated with agriculture) have yet to be fully elucidated. Yiridoe et al. (2009) valued the non-market benefits from AD to the farmer but not to the wider society. Although values were placed on GHG reduction from the displacement of fossil fuels that would be used in the production of electricity, the study did not value the reduction in GHG emissions from applying digestate instead of undigested manure to land. Similarly, Capponi et al. (2012) provide a comprehensive estimate of the carbon dioxide (CO_2) savings from AD, but did not consider other environmental benefits.

In the UK, approximately 20m tonnes of food waste (Defra, 2009) and 85m tonnes of livestock manure is produced per annum (Defra, 2011), suggesting that a sizable quantity of animal manure and other organic feedstocks could be available for AD. ADAS (2012) estimate that there is the potential for 700 food waste and 200 livestock manure fed digesters in the UK. However, despite financial incentives for renewable energy (Feed-In-Tariffs (FIT), Renewable Obligation Certificates (ROC), and Renewable Heat Incentives (RHI)) the uptake of AD remains low in the UK when compared to its European counterparts:

approximately 4,000 digesters exist in Germany (Wilkinson, 2011), compared to approximately 100 in the UK (NNFCC, 2012). Proper valuation of all the externalities from the introduction of AD and comparison with the level of financial incentives provided by the UK government to renewable energy technologies is therefore timely.

We provide an economic valuation of the non-market environmental benefits of on-farm AD and identify knowledge gaps in the scientific and economic literature. This study values the benefits of on-farm AD for every 1% of livestock manure being digested per annum in the UK. We also estimate a value per kWh of electricity produced from the AD of livestock waste in the UK to allow comparison with current environmental subsidies. Percentage of livestock waste is used as it allows policy-makers to determine what a certain percentage target will provide in additional environmental benefits.

7.2 Anaerobic Digestion

7.2.1 Benefits from anaerobic digestion

AD produces two commodities, biogas and digestate; which can be further processed to four saleable commodities: electricity and heat (biogas), and liquid and dry fibre (digestate) (Figure.6.1). The liquid and dry fibre digestate can be applied to land, while the heat and electricity will be used on farm or sold.

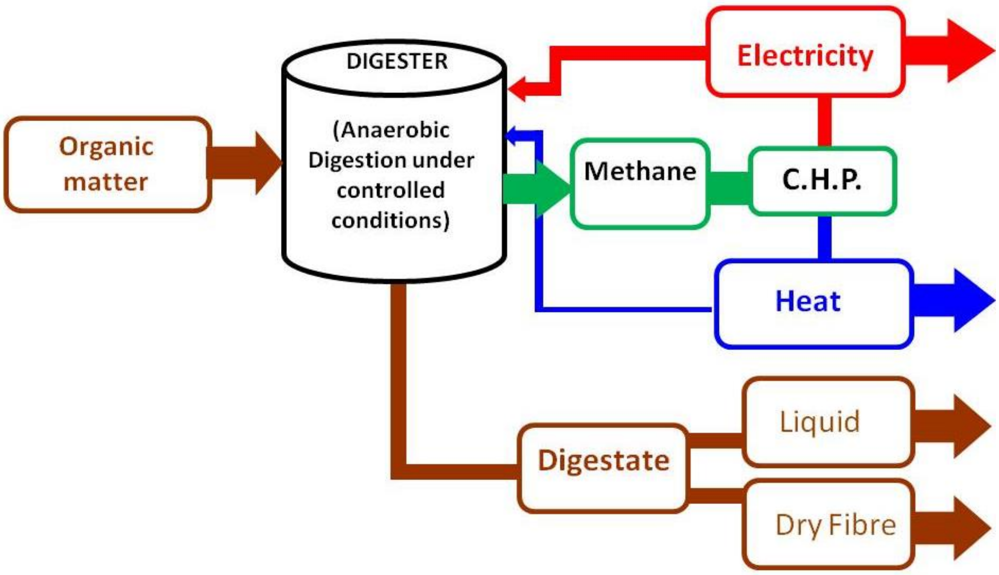

Figure7. 1: Flow diagram of anaerobic digestion (AD) process, showing the organic inputs on the left and the four commodity outputs: electricity, heat, liquid digestate and dry fibre digestate.

The biogas produced from the AD process typically comprises of 50-70% CH_4, 30-45% carbon dioxide (CO_2), ~ 500 ppm hydrogen sulphide (H_2S), and ~100 ppm Ammonia (NH_3) (ADAS, 2012; Mohseni et al., 2012; Probiogas, 2007; Rasi et al., 2007). CH_4 recovery from swine, cattle and poultry manure range from 0.2–0.4, 0.2–0.3 and 0.35–0.6 L CH_4 g^{-1} volatile solids (the part of the feedstock that the microbes get energy from to produce bio-gas), respectively (Masse et al., 2011). Although not considered for valuation in this chapter, food waste biomass is often added to livestock manure for co-digestion, due to its fertilizer value, and its CH_4 potential which ranges from 0.32 – 0.49 CH_4 L g^{-1} of volatile solids (Curry & Pillay, 2012), with approximately 2 kWh of electricity produced per m^3 of biogas (Cuellar & Webber, 2008). On-farm digesters that take in food waste would bring about a further saving of 365 kg CO_2e per tonne of food waste that is diverted from landfill per year (ADAS, 2012).

The by-products from the burning of CH_4 are water (H_2O) and CO_2. The CO_2 may be considered carbon neutral (Caruana & Olesen, 2011; Mohseni et al., 2012) as the crops consumed by the animals absorbed CO_2 during growth, and the nutrients are recycled back to land, whereas specially grown biomass crops have recently been shown to be incorrectly considered carbon neutral (Haberl et al., 2012). The global warming potential of CH_4 is 25 times that of CO_2 (IPCC, 2007) and every kg of CH_4 burned will produce ~ 2.75 kg of CO_2. Therefore, just flaring the CH_4 (without energy recovery) results in a net reduction of 22.25 kg CO_2e for every kg of CH_4 burned instead of being released into the atmosphere from undigested cattle manure.

After digestion, the digestate has an enriched mineral fraction of N and P compared to undigested animal manures, increasing nutrient availability to plants (Field et al., 1984; Masse et al., 2011). Digestate can therefore be used more effectively as a substitute for synthetic fertilizer than undigested cattle slurry (Walsh et al., 2012b).

Table 7.1 is a simplified comparison of the environmental and social benefits from the introduction of various renewable energy technologies, and where AD provides extra benefits. There are three main headings with subgroups below each heading illustrating the extra benefits of each renewable energy technology, above the displacement of fossil fuels in energy conversion. The first of these headings is compatibility, which simply relates to the energy source and its compatibility to electricity demand, broken into two sections: energy storage and dispatchability. Pollution abatement is the environmental benefits that renewable energy technologies can offer above carbon reduction in the replacement of fossil fuel during energy conversion. Finally the last major heading is health and social benefits, which have the sub headings relating to pathogens causing human illness, and odour nuisance. In section 3, all of the extra benefits outlined in Table 7.1 are discussed in detail. Purpose grown energy crops are not considered in this paper for AD, therefore the AD valued does not require additional land which could be used for food production.

Table 7.1: Benefits from the introduction of renewable energy technologies

		Compatibility		Pollution abatement				Health/Social	
Type of renewable energy	**Efficiency during conversion**	Energy storage	Dispatchability	Syntheticfertilizer	Eutrophication	BOD/COD	CH_4	Pathogen	Odour
AD	45-53%[1]	√	√	√	√	√	√	√	√
Hydro	> 90%[1]	√	√						
Biomass	40% [3]	√							
Tidal	80-90%[4]	√							
Photovoltaic	4-22%[1]								
Wind	24-54%[1]								
Wave	80-88%[2]								
Geothermal	10-20%[1]								

Note 1: superscripts denote references; 1. Evans et al. (2009), 2. Dalton et al. (2010), 3. Chen et al. (2010), 4. Payne et al. (2007)
Note 2: wave energy is still firmly in the development stage and therefore we cannot be sure of all its benefits or disadvantages, for further information refer to Dalton et al. (2010, 2012)
Note 3: Energy storage is defined as natural storage as part of the process, not secondary (e.g. hydro with wind). All renewable energies reduce acidification caused by the burning of fossil fuels; therefore it is not mentioned in the table.

7.2.2 Disadvantages of anaerobic digestion

Although on-farm AD is beneficial to both the farmer and the wider environment, there are a number of factors that hinder its uptake. Unlike other forms of renewable energy, AD is a live biological process that requires careful management or the bacterial community may be killed or impeded, resulting in low biogas yields. Consequently, AD is not simply a "plug and wait" technology. There may be traffic nuisance and greenhouse gas emissions if feedstock is transported to the farm (Patterson et al., 2011b). If off-farm waste is co-digested, land application of digestate may be affected by heavy metal build up and/or contaminates (e.g. plastic/glass) in feedstock.

Excess odour from the importation of food waste biomass can be a major social and economic issue. Odour can be eliminated from the importation of food waste if a ‘hub and pod’ system is used, in which the food waste biomass first goes to a central collection point, is macerated and pasteurised and sold to farms that have AD plants. Due to the fact that the biomass is in liquid form it can be pumped into the digester or a separate sealed storage unit on the farm, thus eliminating odour and vermin issues.

As with all renewable energy technologies, AD systems have drawbacks. Table 7.2 provides an overview of the disadvantages associated with renewable energy technologies, divided up into 10 main headings. Where a tick is present under a heading, this is associated to be a negative externality of that particular renewable energy technology. The negative externalities associated with each technology range from minor to major issues associated with that particular renewable energy technology, and the majority are self-explanatory.

Table 7.2: Negative environmental and social impacts associated with the introduction of renewable energy technologies.

Type of renewable energy	Transport by road	Biodiversity loss	NIMBY	Rare earth minerals	Intermittent source	Land requirement	Local environment	Seismic activity	Odour	Air quality particulates
AD (biogas)	✔[8]		✔[7]							✔[10,11]
Hydro		✔[1]								
Biomass	✔[9]		✔[6]			✔[1]				
Tidal		✔					✔			
Photovoltaic				✔[3,4]	✔					
Wind		✔[1,2]	✔[5]		✔		✔			
Wave					✔		✔			
Geothermal							✔	✔[2]	✔[1,2]	

Note 1: superscripts dénotes references; 1. Abbasi & Abbasi (2000), 2. Evans et al. (2009), 3. García-Olivares et al. (2012), 4. Candelise et al. (2011), 5 Pasqualetti (2011), 6. Upreti (2004), 7. Khan (2004), 8. Patterson et al. (2011b), 9. Mahmudi & Flynn (2006), 10. Reid et al. (2005), 11. Hand et al. (2010)

Note 2: NIMBY is an acronym for “not in my back yard” which is often used to describe the negative feelings by residents local to the development and which leads to opposition during the planning stage of renewable energy projects

7.3 Valuing the non-market environmental benefits of AD

In each of the following sections, each non-market benefit of AD is reviewed and the available data used to estimate its value. Data has predominantly been obtained from peer-reviewed sources with additional UK-specific data from government organisations. We estimate the net benefit of livestock manure AD with electricity generation (spreading digestate to fields), relative to spreading the same quantity of manure in undigested form, which is the current practice. Results are presented per m^3 of livestock manure digested[1] and as a total for each 1% of livestock manure potentially anaerobically digested in the UK. We assume that benefits scale linearly with manure quantity. While this is likely to be true for greenhouse gas abatement, local pollution abatement (water, odour) is likely to be strongly scale and context dependent, however, there is little or no evidence on the form of these relationships. Where possible, we also express the non-market benefits of AD per kWh of electricity produced, to permit comparison with environmental subsidies currently paid to AD (e.g. through the UK's FIT). All values are presented in 2012 pounds Sterling: where necessary, estimates from earlier years were inflated using the UK's Consumer Prices Index (Whatsthecost, 2012) and those in other currencies were first converted using annual average exchange rates from Oanda (2012)

GHG reduction can be valued using carbon market prices, such as the European Union Emissions Trading Scheme (EU ETS), marginal abatement costs, or the estimated social damage cost of emitting CO_2. The average EU ETS CO_2 prices for 2011 was £13 per tonne, while marginal abatement costs for the UK in 2010 were estimated to be £52 per tonne of CO_2e abated (DECC, 2011). These are below the mean social damage costs reported by Tol (2005) from peer-reviewed journals of approximately £60 per tonne (February 2013 exchange rate). As there are three figures for per tonne of CO_2e we take the highest (£60; Tol, 2005) and lowest (£13 EU ETS) figures and apply both to the available scientific data to give a range of values.

7.3.1 GHG reductions from the AD of cattle waste

Cattle manure produces the lowest biogas yields of all livestock waste (Masse et al., 2011), producing in the region of 25 m^3 of biogas per tonne (Poeschl et al., 2010; Weiland,

[1] As is standard in agriculture, we treat 1 m^3 of manure/digestate as equivalent to 1 tonne

2010). Approximately 2 kWh of electricity will be produced per m^3 of biogas generated, depending on conversion efficiency (Cuellar & Webber, 2008) and this can be used to displace electricity generated from other sources. Therefore, approximately 50 kWh (25 × 2) of electricity will be produced per tonne of cattle manure. Electricity production from the current mix of technologies in the UK produces approximately 0.547 kg CO_2e per kWh, and as much as 1 kg CO_2e per kWh for coal (Defra, 2012a). The precise effect of electricity production from AD on GHG emissions is complex, and depends on UK and international energy market conditions (price elasticity of supply and demand) and government energy policies. We consider only domestic emissions and simply assume that AD displaces existing energy generation (thus excluding rebound effects). Assuming that AD displaces the current mix of electricity generation, a reduction implies a reduction of CO_2e of 27.35 (0.527 × 50) per m^3 of manure; for coal-fired electricity production, this equates to 50 (1 × 50) kg CO_2e per m^3 of manure. This is worth £354 – 1,641 for the current mix and £605 – 3,000 per kg CO_2e for coal (EU ETS and Tol (2005), respectively). All renewable energies provide a reduction in CO_2e from displacement of fossil fuel; however the current FIT rate is not paid in relation to CO_2e reductions, and thus the values above are not considered in our final calculations of additional pollution abatement brought about by the introduction of AD above other renewable energy technologies.

AD of cattle manure with electricity generation brings an extra reduction in GHG emissions above other renewable energies as it harnesses and uses CH_4 that would otherwise be emitted during storage and application of cattle manure (Amon et al., 2006; Collins et al., 2011). Within cattle production, dairy cows are the largest producers of CH_4; each producing ~16 kg from manure management and ~100 kg through enteric fermentation per year, with non-dairy cattle producing approximately half these amounts (Hynes et al., 2009). Depending on the season, dairy cows in the UK are housed a minimum for four months of the year, making it feasible to collect approximately ⅓ of the manure for AD. CH_4 emissions could therefore be reduced by at least 5% ((16/116))* ⅓), and more if animals are housed for longer.

On-farm AD will also affect other farm GHG emissions. The effect on nitrous oxide (N_2O) and ammonia (NH_3) emissions during storage and application of digestate compared to undigested manure remains uncertain, with some authors reporting an increase (Amon et al., 2006; Clemens et al., 2006; Thomsen et al., 2010), while others have reported lower levels of

N_2O from digestate applied to soils than undigested manure (Bhandral et al., 2009; Petersen, 1999). When all GHG emissions, (CO_2, N_2O, NH_3 and CH_4) associated with the storage and application of manure have been accounted for, there is approximately a 60% reduction in total GHG emissions (54.52 kg CO_2e per m^3 of cattle manure digested) during storage and application, compared to undigested cattle slurry (Amon et al., 2006). Although studies have looked at GHG emissions from different storage systems of pig (Petersen et al., 2013; Prapaspongsa et al., 2010) and poultry (Moore et al., 2011) manure; no studies could be found on the difference in GHG emissions between digested and undigested pig and chicken manure post land-application. Hence pig and poultry wastes were excluded in the evaluation of GHG reductions brought about by the AD of livestock wastes.

In 2010 (latest figures available) 74 m tonnes of solid cattle manure and slurry were produced in the UK (Defra, 2011). Therefore, for every 1% of cattle manure digested in the UK (~740,000 t) there will be a total GHG saving of 38,864 t equivalent to a value of between £505,242 (38,864 × £13; based on EU ETS) and £2,331,840 (38,864 × £60; based on Tol (2005)).

7.3.2 Displacement of synthetic fertilizer

AD increases the availability of macro-nutrients (nitrogen and phosphorous) to crops (Field et al., 1984; Masse et al., 2011) and digestate has been found to either increase crop yields (Holm-Nielsen et al., 2009; Rubaek et al., 1996; Tafdrup, 1995), or to have no negative effect (Chantigny et al., 2008; Loria & Sawyer 2005; Möller et al., 2008; Petersen, 1999; Thomsen et al., 2010) relative to the equivalent quantity of undigested manure. Those studies that report an increase in crop yield witnessed 10 – 25% greater yields when digestate was applied rather than undigested manure (Bougnom et al., 2012; Morris & Lathwell, 2004; Walsh et al., 2012b). The variability in the results may be due to a number of factors, including different crop and soil types used, plus the timing and method of application and whether it was a pot or field trial experiment. Together, these results imply that AD of manure prior to application has the potential to reduce synthetic fertiliser required to meet crop yields, while maintaining yields constant. The rate of substitution between digestate and synthetic fertilizer can be estimated from studies comparing crop yields. Chantigny et al. (2008), Dahlberg et al. (1988), Liedl et al. (2006) and Walsh et al. (2012b) all report no difference in crop yield between digestate and synthetic fertiliser at varying levels of nitrogen application. Although there is a discrepancy in regards to crop yield from digestate trials, there are no reports of application of digestate leading a lower crop yield than undigested or green manure (Gunnarsson et al., 2011; Stinner et al., 2008).

Displacing synthetic fertilizer would lead to an associated reduction of CO_2e (from the reduced manufacture of synthetic fertilizer) of 4.96, 1.86 and 0.99 kg CO_2e per kg of N, P and K displaced, respectively (Econivent, 2007). To provide a range of the potential replacement of synthetic fertilizer demand brought about by the introduction of AD of livestock manure, we chose to use low (10%) and high (25%) estimates from literature reduction in synthetic fertilizer use from the introduction of AD. As digestate has been shown to match crop yield of individual synthetic N and mixed NPK synthetic fertilizer (Walsh et al., 2012b) we give a value for a reduction for all three nutrient types (Table 7.3). Approximately 85 million tonnes of livestock manure are produced and applied to land in the UK every year (Defra, 2011b). Additionally, 1,029,000, 192,000 and 283,000 tonnes of N, P and K in the form of synthetic fertilizer are applied per year in the UK (Defra, 2011). Adopting the low value mentioned previously, a 10% increase in crop yield from the application of digestate above undigested manure applied vis-à-vis would lead to a 10% reduction in the need and thus application of synthetic fertilizer. The rationale behind this assumption is that farmers require a specific nutrient load to produce a total net agricultural crop per year. The nutrient load is achieved through the application of purchased synthetic fertilizers and livestock manures. As the fertilizer market is a free market (with the only restriction on the market being NVZ areas), it can be assumed that total nutrient load is being reached to meet total crop yield demanded by the market. Therefore an increase in crop yield from the digestion of livestock manure, will vis-à-vis lead to an associated reduction in the demand for synthetic fertilizer of the same percentage. The associated benefits from the reduction in demand and manufacture of synthetic fertilizer, and the reduction in CO_2e for every 1% of animal manure being anaerobically digested and applied to land in the UK can be seen in Table 7.3.

Table 7.3: Value of reductions in synthetic fertilizer demand and associated CO_2e decrease in the production of synthetic N, P and K due to the AD of 1% of livestock manure in the UK, using EU ETS and Tol (2005) carbon prices.

Fertilizer	Synthetic manure replacement (t)	Amount of C reduction (t)	EU ETS £ 13	Tol £ 60
		Low 10%		
N	1,029	5,104	66,352	306,240
P	192	357	4,641	21,420
K	283	280	3,642	16,800
		High 25%		
N	7,718	35,568	462,384	2,134,080
P	1,440	2,678	34,814	160,680
K	2,123	2,102	27,326	126,120

Note 1: The same rational described for a 10% reduction in synthetic fertilizer demand was applied to a 25% reduction.

Note 2: A 10% reduction in demand for the 1,029,000, 192,000 and 283,000 t of synthetic N, P and K respectively, currently being used in the UK would equate to a 102,900, 19,200 and 28,300 t reduction if all the 85 million tonnes of livestock manure in the UK was anaerobically digested. More realistically, if 1% of this animal manure would be digested, this would equate to 1029, 192, and 283 tonnes of N, P and K synthetic fertilizer, respectively.

7.3.3 Leaching of nutrients to waterways

Modern agricultural practices have resulted in excess N and P being leached to groundwaters, leading to eutrophication of marine and surface waters (Holman et al., 2010; Howden & Burt, 2009; Weatherhead & Howden, 2009); resulting in fish mortality and plankton build up. During 2010, English rivers had 51% and 32% higher than the recommended level of phosphate and nitrate present, respectively (EA, 2011); with an estimated 60% of all N in inland waterways in England and Wales originating from agriculture (EA, May 2002; Hunt et al., 2004). The hydrological process of nutrient leaching is complex and difficult to quantify, and is dependent upon a number of factors including: soil type, climate, hydrology, topography, land use and time of manure application (Beckwith et al., 1998; Burt et al., 1993; Chalmers, 2001; Chambers et al., 2000; Howden et al., 2011;

Weatherhead & Howden, 2009; Yan et al., 2002). Studies to determine differences in nutrient loss through leaching between organic and synthetic fertilizers have provided mixed results. Some studies have found greater levels of leaching from organic compared to synthetic sources of N (Bakhsh et al., 2005; Basso & Ritchie, 2005; Bergstrom & Kirchmann, 2006), while Bittman et al. (2005) and Di et al. (1999) found the opposite, with Tarkalson et al. (2006) reporting no difference.

Experiments dealing with digestate specifically where normalisation of N took place have also provided mixed results, with Goberna et al. (2011) and Sänger et al. (2010; 2011) reporting higher levels of NO_3^- leaching from digestate compared to undigested manure. These results conflict with those of Möller (2009) who found less leaching of N from digestate, and Lukehurst et al. (2010) and Walsh et al. (2012b) who report no difference between undigested cattle manure and digestate in the volume of N in soil solution after application. Differences in the type of digestion, time of application, whether separated or un-separated digestate is applied, rate of application and the chemical properties of the residue may underlie the conflicting results in the literature (Goberna et al., 2011). In addition, whether nutrient sources were applied to fallow or cropped soils and the root depth of the crop trialled would affect the results. Walsh et al. (2012b) reported that there was approximately 20% less NO_3^- in soil solution from digestate compared to synthetic N fertilizer, the only figures available comparing leaching differences between digestate and synthetic fertilizer attainable at time of writing. There was not enough literature comparing the loss of phosphate from digestate and synthetic fertilizer to enable a scientific or economic valuation, therefore it is not considered in this work.

Although we go on to value potential leaching reductions from the application of digestate compared to synthetic fertilizer, the final value must be considered indicative only as there is very little information in relation to potential leaching of nutrients between digestate and synthetic fertilizer, thus why a large range is provided. Further, the only values available for valuation are for nutrients recovered in soil solution from within the rooting zone of a pot trial by Walsh et al. (2012), and therefore should not be considered leaching per se.

Literature on the costs associated with excess NO_3^- in UK waterways is scarce; however, Pretty et al. (2003) estimated that the annual cost of N leaching to waterways in the UK is between £90 - £134m (2012 £). Replacement of synthetic fertilizer with digestate would reduce N leaching to waterways and hence have a positive environmental and economic benefit in a similar way that a reduction in CO_2e is valued per individual tonne of reduced pollutant.

The total amount of N applied to land in the UK in 2011 was approximately 1,709,000 t; 60% of which (1,029,000 t) was from synthetic fertilizer, and 40% (680,000 t) was from organic sources (Defra, 2012b; Defra, 2011). For valuation purposes, we assume that 60% of N pollution in UK waters from agricultural sources is caused by the application of synthetic N. Due to the fact that 60% of N applied to land is synthetic N, we take 60% of the Pretty et al. (2003) figure (£90 - £134m), resulting in a cost associated with N leaching from synthetic fertilizer to be £54m - 80m. From Table 7.4, it can be seen what the synthetic N replacement may potentially be from the digestion of 1% of livestock manure in the UK (1,029 - 7,718 t of synthetic N). Table 7.4 reports the high and low values are from the potential reduced leaching for every 1% of animal manure digested and applied to land in the UK, and the associated economic savings. As we only have one study in relation to N leaching differences between digestate and synthetic fertilizer, we are restricted to using the Walsh et al. (2012b) 20% figure.

Table 7.4: Value of 20% reduction of N leaching from 1% of animal manure in the UK being anaerobically digested and subsequently displacing synthetic fertilizer

	Tonne of reduced N from introduction of AD	20% of total N reduced in tonne	Price per tonne of N leaching	
			£52.48 low	£77.75 high
Low	1,029	206	£10,811	£16,017
High	7,718	1,544	£81,029	£120,046

Note 1: Walsh et al. (2012b) was a pot trial on one crop type under controlled conditions with shallow roots, and further trials on different crops in different conditions with different root depths will provide varied results. It is expected that Walsh et al. (2012b) will be at the very upper end of leaching differences for this specific reason.
Note 2: Synthetic N has an associated leaching cost per tonne of £52.48 (£54m /1,029,000 t synthetic N; low value) and £77.75 (£80m/1,029,000 t synthetic N; high value). By incorporating the high and low values from Table 7.4 (1,029 t low and 7,718 t high).
Note 3: Pretty et al. (2003) estimated that the annual cost of N leaching to waterways in the UK is between \$105m – \$160m. Included in this valuation is \$7.17m – \$11.19m for GHG associated with eutrophication, though in order to prevent double counting we subtracted the GHG associated figure, thus giving a new value of \$98 – \$148m. At the exchange rate that Pretty (2003) used, and converting to today's values, £90 - £134m (2012£) is the cost attributed to N leaching. It can be assumed that all these costs are attributed to agriculture, as Pretty et al. (2003) valued nutrients to waterways from sewage works separately.

7.3.4 Reduction in biological and chemical oxygen demand

Biological oxygen demand **(**BOD) in waterways refers to the amount of dissolved oxygen required by aerobic microorganisms to break down organic material; the higher the organic pollution the greater the BOD requirement. This reduction of oxygen in the water starves higher organisms, leading to biodiversity loss. The same rationale applies for chemical oxygen demand (COD), and both are often measured as total oxygen demand (TOD). As part of the microbial action during anaerobic digestion, the volatile solids part of the feedstock are converted to CH_4, by so doing there is an associated reduction in the TOD of the digestate compared to undigested manure. The reduction in BOD of animal manure during digestion ranges from 55 to 82% (Anon, 2003; Clemens & Huschka, 2001; Clemens et al., 2006) and a similar COD reduction by between 45 and 90% (Clemens et al., 2006; Canada, 2002).

Livestock manure as well as human and industrial sewage contribute to the majority of BOD in waterways. Sewage treatment and disposal is more highly regulated than that of livestock manure, implying that the latter potentially poses a greater risk of increasing BOD levels to waterways. The main potential organic materials associated with BOD from agriculture include milk, silage effluent and manure. Over the last few decades, government intervention and better farm practices have seen farmers install collection pits to capture silage runoff from farms and milk is only very rarely applied to land (e.g. during protests). It is therefore assumed that the majority of BOD associated with agriculture comes from the application of manure to land.

For this type of analysis, damage costs are the most favourable tool for valuation, with the most up to date available source for the UK being O'Neill (2007), who estimated that between £4m and £5m (in 2012 £) in damage costs can be attributed to BOD in UK waterways. Table 7.5 provides a summation of the sensitivity analysis, combining both the variability in the scientific and economic understanding of the associated value of BOD decrease to UK waters from the AD of livestock wastes. We were unable to obtain any useful economic values for a COD reduction, thus it is not valued here.

Table7.5: The associated BOD reductions if 1 % of animal manure in the UK were to be digested

		Reduction in BOD		
Low and high	Quantity digested (m^3)	55%	74%	82%
Low £4m	850,000	£22,000	£29,600	£32,800
High £5m	850,000	£27,500	£37,000	£41,000

Note: BOD values are obtained by dividing the total damage cost to waterways (£4m; O'Neill, 2007) by the approximate amount of livestock manure produced in the UK (85m t per year). This figure was then multiplied by 1% of manure (850,000 tonnes), thus (£4m/85m tonnes) × (850,000). This value was multiplied by 55%, 74% or 82% (BOD reductions, various references). The same rationale was repeated for the high value of £5m.
Note 2: The rationale behind O'Neill (2007) values is not clear and for our analysis 1/3 of the damage cost associated with "*informal recreation from poor water quality*" from O'Neill (2007) were used for assumed damage caused by BOD. The other ⅔ of the sub heading in O'Neill (2007) "*informal recreation from poor water quality*" are attributed to N and P and have already been valued in Section 7.4.3.

7.3.5 Pathogen reduction

Pathogens from agriculture are a problem for both human and animal health. AD has been shown to destroy viral, bacterial and protozoan pathogens (Cabirol et al., 2002; Lund et al., 1996; Sahlstrom, 2003; Saunders et al., 2012). For example during digestion, coliforms are reduced by 99.9 % (Martin, 2003). This is conditional on a number of factors including: feedstock; temperature; organic matter content; retention time of the manure in the digester; pH and NH_4^+ concentration (Kearney et al., 1993; Ottoson et al., 2008; Sahlstrom, 2003). Temperature is the most important factor in pathogen destruction (Dumontet et al., 1999; Gibbs et al., 1995; Kearney et al., 1993), with *Salmonella* spp. and *M. paratuberculosis* being inactivated within 24 hours at thermophilic temperatures (Olsen et al., 1985; Plymforshell, 1995). Pathogen destruction is also enhanced in multi-stage digestion (Kunte et al., 2004; Sahlstrom, 2003) and during pasteurisation (e.g. pre- or post-digestion). At 70 °C, *Salmonella*, *E. coli* O157 and *Cryptosporidium* are destroyed in less than 1 h (D'Aoust et al., 1988; Mitscherlich & Marth, 1984; Rose, 1997; Ward et al., 2008). It has also been reported that there were fewer pathogens and bacteria found in soil to which digestate was applied than soils applied undigested manure (Goberna et al., 2011; Saunders et al., 2012). In general, on-farm AD is mesophilic and associated with lower pathogen destruction. However, *Salmonella* spp., *E. coli* and *Cryptosporidium parvum* oocysts have been inactivated in mesophilic digestion (Gadre et al., 1986; Kato et al., 2003; Olsen & Larsen, 1987). Although the digestion process kills pathogens, there is the potential for fresh colonisation during storage post-digestion (Clements, 1983; Keller, 1983; Pepper et al., 2006; Sahlstrom, 2003; Sidhu et al., 2001). Therefore proper covered storage post-digestion is advisable to prevent re-inoculation of pathogens; this has the secondary benefits of abating fugitive GHG emissions from storage of the digestate. However, to be certain of pathogen destruction, pasteurisation is recommended.

Although agriculture is associated with pathogen inputs to bathing water, sewage and wildlife also play a part. The impact from wildlife is limited unless large flocks of birds congregate in one small area, and the sewage industry is heavily regulated to prevent contamination of waterways. Therefore we assume that the vast majority of pathogen infection of bathing waters is related to livestock agriculture. The benefit of pathogen destruction from AD will only arise from a reduction of infections contracted from pathogens contained in bathing water or possibly from recreational users on farmland. Mains drinking water is treated by utility companies thus even if all farm manure in the UK was digested, utility companies would still need to treat water for pathogens that may arise from other sources. For this reason, only costs attributed to pathogen infection for bathing waters are considered. Literature on the cost of pathogens from agriculture to waterways is scarce, with most focussing on specific cases with large outbreaks (e.g. Cowden et al., 2001; Grant et al., 2008; Hrudey et al., 2003; Roberts et al., 2000).

Using contingent valuation modelling, Eftec (2002) reported a total UK value of £79m (2012 £) for a 1% chance of each person avoiding stomach upset due to poor bathing water quality from faecal contamination. Due to uncertainty of what percentage of pathogens to waterways are caused by animal manure, we chose a low value of 50% of pathogens in bathing water emanating from animal manure and a high value of 90%. Thus for every 1% (850,000 tonnes) of animal manure in the UK that is digested and pathogens eliminated, the values of reduced pathogen infections range from £380,000 ((£79m×1% livestock manure) × (50% pathogen reduction)) for 50% and £684,000 for 90%.

7.3.6 Odour reduction

A negative externality of livestock agriculture is odour from the storage and handling of manure. The presence of foul odours has a direct effect on quality of life of local residents and an associated negative effect on real estate values within the vicinity of the odour nuisance. There is the potential for the odour problem to worsen due to the increasing densities of livestock (DEFRA, 2010a). AD is a proven and effective technology for reducing odour, especially from animal manure (Lukehurst et al., 2010; Smet et al., 1999; Welsh et al., 1977). However, the reported reduction in odour after manure is digested varies considerably, ranging from 50 - 90% (Lusk, 1998; Pain et al., 1990; Powers et al., 1999). From hedonic valuation studies carried out in Canada and the US, it has been shown that there is a drop in property prices by 4 - 9% within a half mile of a livestock unit due to odour (Herriges et al., 2005; Kim & Goldsmith, 2009; Palmquist et al., 1997; Ready & Abdalla, 2005) regardless of prevailing wind direction (Kim & Goldsmith, 2009).

In this study, to estimate the effect that odour may have on house prices, an average UK house price of £188,640 (index, 2012) is used. Whilst it is acknowledged that house prices vary between regions, this valuation indicates the possible increases in the average house price within a ½ mile radius of a manure storage facility for the UK as a whole (Equation 7.1).

Equation 7.1 The increased value per household by the introduction of AD to reduce the odour from animal manure stores.

$$\text{Eq 1} \qquad \text{increased value} = (D-1) \times H \times X \times Y$$

Where

D = average number of households in area equal to ½ mile radius from farm minus the farmer's property as an increase in the farmer's property is a private benefit, and not a public benefit.

H = average house prices within the locality of the manure storage facility.

X = the percentage drop in house prices associated with odour

Y = the percentage drop in odour

Table 7.6 illustrates the reduction in house prices due to odour from animal manure storage without AD, odour reduction due to manure being digested and finally the house price increase due to the implementation of AD.

Table 7.6: A sensitivity analysis of the average house price in the UK under different odour reduction levels and effected drop in property price due to odour.

Average house price UK	Effect of odour on property price	% drop in property price due to odour	Reduction in odour by the introduction of AD	Price increase per property from the introduction of AD
£188,640	4%	2%	50%	£3,772
£188,640		2.8%	70%	£5,282
£188,640		3.6%	90%	£6,791
£188,640	9%	4.5%	50%	£8,489
£188,640		6.3%	70%	£11,884
£188,640		8.1%	90%	£15,279

With a potential total of 200 farm waste plants in the UK ADAS (2012), we conservatively assume that at least one house on average per AD unit will increase in value (not the farmer’s private residence). At the very lowest possible increase in value of £3,772 and the highest value of £15,279 this gives values ranging from £744,400 (£3,722 × 200) and £3,055,800 (£15,279 × 200) low and high respectively.

7.3.7 Total non-market benefits

The lowest and highest values from each of the non-market benefits presented in the Results section are combined in Table 7.7. Where possible, a FIT value per kWh of electricity produced is reported. Farmers that anaerobically digest livestock manure on their farm currently receive a FIT payment for the electricity produced. This review has shown that AD of livestock manure delivers non-market monetary benefits above the sole value of providing a source of renewable energy; these benefits are currently not valued in the FIT payment. FIT payments are currently issued in 75 jurisdictions around the world (Kim & Lee, 2012), and valuing on a kWh basis will allow for easier extrapolation outside of the UK, as well as highlighting the current undervaluation of electricity produced from the AD of livestock manure. The digestion of 1% of livestock manure in the UK and the potential quantity of biogas produced was used to determine the amount of kWh of electricity produced. Cattle manure was chosen to represent biogas yield and not all animal manure, as it is the most likely form of manure that will be digested in the UK due to its quantity (comprising 87% of total livestock manure; Defra (2010b)). To determine a low and high FIT rate, the total low and high values in Table 7.7 were divided by the total kWh of electricity produced from 1% of cattle manure in the UK being anaerobically digested, yielding 37,000,000 kWh electricity (($25\ m^3 \times 740{,}000$) × (2 kWh)). It can be seen from Table 7.7 that if non-market benefits were taken into account, the current FIT rate of £0.09 - £0.15 per kWh for AD should be increased by between £0.0272 - 0.1520 per kWh of electricity produced. If cattle manure is not digested (i.e. pig or poultry manure digestion only) there will be no decrease in GHG emissions from cattle and thus the environmental benefits are reduced dramatically, reflected in a lower FIT payment of £0.0136 - 0.0889 per kWh. Finally, Table 7.7 shows the continuous and one-off benefits from the introduction of on-farm AD. The only one-off benefit is odour reduction as house prices will be increased once, when the on-farm AD system is implemented; whereas all other environmental benefits continue yearly.

Table 7.7: High and low non-market environmental benefits, with values broken down to a FIT rate, from 1% of livestock manure being AD in the UK

Non-market benefits	Total value in £ for 1%		Value in £ per kWh	
	Low EU ETS	High Tol (2005)	Low	High
GHG from cattle manure	505,242	2,331,840	0.0137	0.0630
Replacing synthetic fertilizer with digestate				
N	66,352	2,134,080	0.0018	0.0577
P	4,641	160,680	0.0001	0.0043
K	3,642	126,120	0.0001	0.0034
Leaching of nutrients to waterways	10,811	120,046	0.0003	0.0032
Biological oxygen demand reduction	22,000	41,000	0.0006	0.0011
Pathogen reduction	395,000	711,000	0.0107	0.019
Odour reduction	744,400	3,055,800	UR	UR
Digestion of cattle manure				
Continuous yearly	1,007,688	5,624,766	**0.0272**	**0.1520**
One off	744,400	3,055,800		
Total	**1,752,088**	**8,680,566**		
Digesting non-cattle manure				
Continuous yearly	502,446	3,292,926	**0.0136**	**0.0889**
One off	744,400	3,055,800		
Total	**1,246,846**	**6,348,726**		

Note 1: UR (unattainable rate), represents where FIT values per kWh were unattainable for odour, as odour reduction will have a one off value of increasing house prices, not a continuously yearly benefit in reduced pollution.

Note 2: FIT values are obtained by dividing the high and low values for 1% of cattle manure anaerobically digested in the UK by the potential quantity of electricity produced (37,000,000 kWh).

7.4 Discussion

More field-scale research on a wide variety of soil and crop types is required to determine the exact effects that digestate application will have on crop yield and nutrient leaching, as well as GHG changes between digested and undigested chicken and pig manure. Scientifically, it is a reasonable assumption to suggest there will be greater crop yield if livestock wastes are digested due to the increased mineralization of N and P during digestion (Field et al., 1984; Masse et al., 2011) and that the effects digestate has on the soil decomposer community is similar to that of synthetic fertilizer (Walsh et al. 2012a). AD of farm manures would allow for better utilization of nutrients and help close the nutrient cycle; in contrast to purpose grown energy crops for biomass which results in the removal of nutrients from the land. AD has benefits above synthetic fertilizer even where crop growth is similar, in introducing carbon and humus to the soil. Average cattle herd size has increased as farmer's marginal returns get smaller and they seek to capitalise on the economies of scale. This intensification will effectively concentrate pollution from agriculture and AD is a technology which can have a major impact on reducing this pollution.

In the UK, approximately 44% of total CH_4 emissions are attributed to agriculture (DECC, 2010). According to Bywater (2011), if all livestock manure in the UK were to be anaerobically digested and the CH_4 utilized, this could potentially produce ~ 10 billion kWh of electricity per year. Electricity produced from AD in the UK currently receives a lower FIT payment (varying between 8.96 p and 14.7 p per kWh) than wind (4.48 p to 35.8 p per kWh) (Ofgem, 2013a), and photovoltaic (7.1 p to 15.44 p per kWh) (Ofgem 2013b), despite AD providing additional pollution abatement that other renewables are unable to provide (Table 7.1) which we estimate to be worth between 2.7-15 p/kWh.

Energy storage is a problem that affects the economic viability of renewable energy providers. AD has excellent dispatchability due to the storability and instant conversion of biogas to electricity. Although photovoltaics, as heat (Madaeni et al., 2012), wind as compressed air (Denholm and Sioshansi, 2009) or pump-storage hydro systems (Sørensen, 1981) can be considered energy storage, these incur extra development costs, and there will be an associated energy loss and/or cost during storage. Due to the low biogas yields solely from cattle manure, farms that do not import other wastes or do not grow crops for co-digestion require large numbers of cattle to ensure sufficient returns on the investment; estimated to be >500 dairy cows in the UK (Fre-Energy, personal communication, 2012). If government financial incentives for AD were increased to take into account the other environmental benefits associated with digestion of cattle manure, this would encourage greater uptake of AD; indeed it is expected that doubling the FIT rate would increase the economic viability of AD plants 4-fold (Bywater, 2011).

Although weed seed destruction during AD has been reported (Engeli et al., 1993; Engler et al., 1999; Jeyanavagam et al., 1984; Westerman et al., 2012a; Westerman et al., 2012b), there was not enough scientific or economic data on weeds associated with UK agriculture available to enable a valuation to take place, thus this area requires further research.

With higher prices for electricity, farmers may sacrifice valuable agricultural land and harvest crops such as maize for digestion, similar to that seen on a large scale in other European countries. This introduces the 'food vs. fuel' debate, and should be avoided. The use of AD to process livestock wastes can bring about a number of environmental benefits in addition to reducing greenhouse gas (GHG) emissions, and AD should be considered a pollution abatement technology as much as a source of renewable energy (Walsh et al., 2012b). Figure 7.2 provides a graphical illustration as to the non-market benefits from the introduction of AD, treating livestock waste and the current value to the UK.

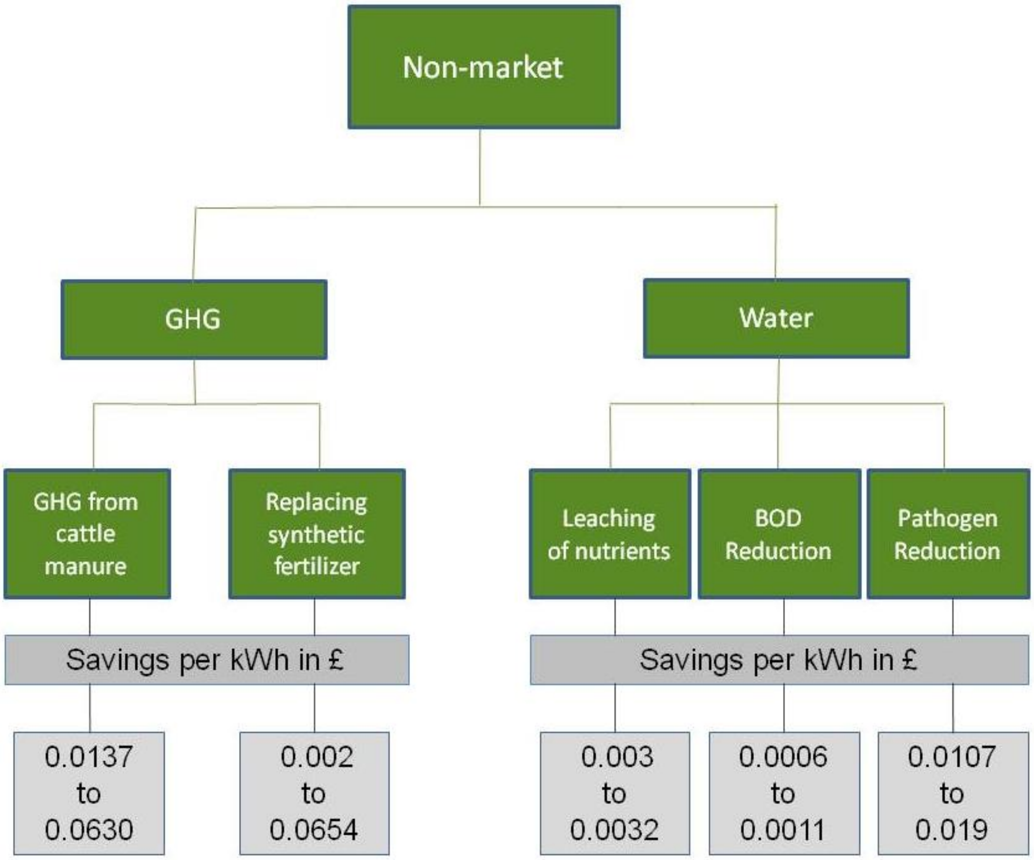

Figure 7.2: Positive environmental externalities of AD and their value per kWh

It may at first seem unusual to attach non-market benefits to a FIT value and not provide government assistance in some other form, such as a capital grant to help farmers reduce the initial cost of on-farm AD investment. There are a number of reasons why a government incentive to aid the uptake of the AD of livestock manure should come in the form of an increase in the FIT payment. Firstly, farmers that have already invested in AD or any renewable energy technology currently receive a government payment per kWh of electricity they produce. Therefore, any change to a current payment to incentivise AD would be simple to implement (i.e. FIT). Secondly, this work has shown that for every tonne of livestock manure AD in the UK there is an environmental benefit. Therefore, if a capital grant was provided for the construction of an AD unit, there are two issues that may arise that would result in the intended environmental benefits not being achieved. This may happen for one of two of the following reasons, or both. Firstly, the farmer may build the AD unit and then decide that after he/she has made the capital repayment to meet the full cost of the construction, that he/she no longer wants the AD unit and may stop using it or may dismantle it and sell it on. There is no incentive to keep digesting livestock waste. The second point which could heavily influence the first is the fact that cattle manure produces such low biogas yield, and thus low electricity, compared to energy crops. If the farmer gets the capital grant he will be incentivised to obtain maximum returns on his investment which will mean highest biogas yields per tonne of biomass feedstock; incentivising farmers to deviate away from the digestion of livestock manure to higher yielding crops. This therefore opens the 'fuel vs food' debate and the associated negative socio-economic and environmental impacts.

With farmers being paid for every kWh of electricity produced, it can easily be determined how much energy a farmer will produce on his/her farm. This can be done by multiplying the amount of manure a famer produces in a day, and then multiply this by the amount of days that the farmer will be able to collect manure for to determine the amount of electricity produced. If for example the figure was 100 kWh, the farmer would get paid the higher FIT rate with all pollution abatement benefits included, and after the farmer reaches the 100 kWh point they would receive the current lower FIT payment. Thus if a farmer produced 150 kWh total per year (the extra 50 kWh through energy crops), they would receive the higher FIT rate for 100 kWh and then a lower current FIT rate for the remaining 50 kWh, in this example. This would eliminate the incentive to cheat as it would not be in their interest to replace livestock manure with energy crops as the economic returns would be the same. In addition, if the farmer was to indeed not digest livestock manure and use energy crops instead, they would have to give up land that they had set aside to feed their animals. In so doing, reducing the number of animals they are able to maintain, and thus reducing the income from one aspect of farming to another. FIT rates could also be adjusted depending on the management of the AD system. For instance, a higher FIT payment could be paid if the farmer employs best practice, such as covering digestate lagoons (which would reduce volatilization of N) and the application of digestate through trailing shoe or injection.

7.5 Conclusions

Although there is uncertainty within the values we estimate, conservative values have been used throughout and ranges are given due to the lack of existing knowledge in available literature. This review has shown that for every 1% of UK livestock manure processed via AD, this would equate to non-market benefits worth £1,007,668 - 5,624,766. Further, there is a potential one-off benefit ranging from £744,400 - 3,055,800 through reduced odour. As the FIT payment does not take into account the value of these non-market benefits, it is estimated that they undervalue electricity generated by AD of livestock manure by £0.03 - 0.15 per kWh. As the AD of livestock manure provides the largest monetary value in terms of non-market benefits, we propose that the current flat-rate FIT payments for AD do not reflect the environmental benefits that it delivers above e.g. AD of energy crops. FIT rates should therefore be re-structured to take such factors into account. In summary, the findings indicate that the FIT rate should be two-fold: a higher rate for electricity production from animal manure and a lower rate for other biomass feedstock.

This valuation has revealed that more work is required to enable a full and accurate valuation of all the benefits that the AD of livestock manure offers. AD has the potential to turn a negative externality of agriculture (pollution from manure) into useful commodities, electricity and digestate. Unless all the environmental benefits are understood and valued, AD will continue to receive a disproportionately low government aid relative to the environmental positive externalities it offers above other renewable energy technologies.

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Chapter 8

Discussion

8.1 AD: renewable energy or pollution abatement technology?

Public awareness of AD ranks quite low in comparison to other renewable energies. Even academia is somewhat guilty of overlooking the technology. An example of such disregard is highlighted by research carried out by Diaz-Rainey and Ashton (2008) in relation to renewable energy in the UK, where AD technology was conspicuous in its absence. As the world continues to deplete non-renewable fossil fuel resources, the role of renewable energies becomes increasingly important, driving both national and international policy. Focused policy objectives such as the Renewable Energy Roadmap, composed by The European Commission, aspire to increase the gross domestic energy consumption from renewable energy sources. The roadmap includes the ambitious target of deriving 20% of energy consumption from renewable sources by 2020. Such aspirations are all the more challenging considering that just 12.4% of EU energy consumption was accounted for by renewables in 2010 (Nkemka & Murto, 2013). Although AD can contribute significantly to the energy supply it is only a small fraction of the whole energy mix, with biomass (AD plus lignin biomass) currently only contributing between 3 - 13% of energy to industrialized nations (De Meester et al., 2012). Indeed, incineration and biodiesel cover most of this supply, whilst biogas from AD contributes only a small fraction (Braun et al., 2009).

AD can be considered a unique technology through the multiple benefits it offers. The research carried out in this PhD study and elsewhere infers that, logically, AD should be considered a source of renewable energy and a pollution abatement technology; although to what extent depends upon the scale and type of feedstock and digester used. To emphasise this point, if a farmer builds an AD unit, and firstly digests cattle manure, the AD unit may be considered a source of income through generation of renewable energy, and a nutrient management regime and pollution abatement technology. However, if the farmer decides to expand the AD unit and import off-farm biodegradable material (e.g. food waste) from the local community, the environmental and economic benefits may be expanded to wider society.

Figure. 8.1 shows how the positive external benefits from the introduction of on-farm AD can benefit society and how economists value such benefits, where the Y axis is the environmental cost of producing a litre/kg of milk/meat and the X axis is the quantity of milk/meat produced. Figure 8.1 is for illustrative purposes only, the exact movement along the marginal benefit (MB) line are not all known at this time. When the pollution abatement benefits of AD are fully accounted for, it can be seen that the production of more animal manure is possible while at the same time reducing the social external cost of farming. Each pollution abatement attribute of AD moves along the MB curve, lowering the environmental cost (from p to p*) associated with increasing quantity of milk/meat produced from q to q*. In summary, greater uptake of on-farm AD would allow the agriculture sector to produce more milk/meat at less environmental cost.

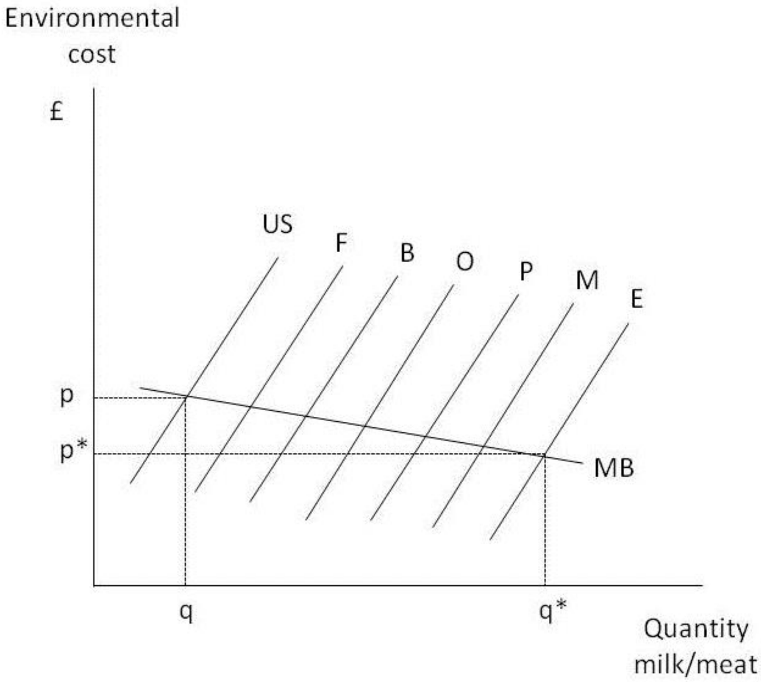

Figure 8.1: Individual environmental benefits from AD, US = undigested cattle manure, F = synthetic fertilizer replacement, B = BOD/COD reduction, O = odour reduction, P = pathogen reduction, M = methane reduction and MB is the marginal benefit the consumer obtains from the consumption of milk and meat. Lower case p and q represent price and quantity, respectfully.

When all of the environmental benefits from Fig. 8.1 are grouped together, as in Fig. 8.2, it can be clearly seen how the social marginal curve (SMC) shifts to the right. The SMC will shift by a large amount from SMC to SMC*, this is due to less pollution; lowering the environmental cost of milk/meat production from P1 to P2 and increasing the quantity at a lower social and private cost from Q1 to Q2.

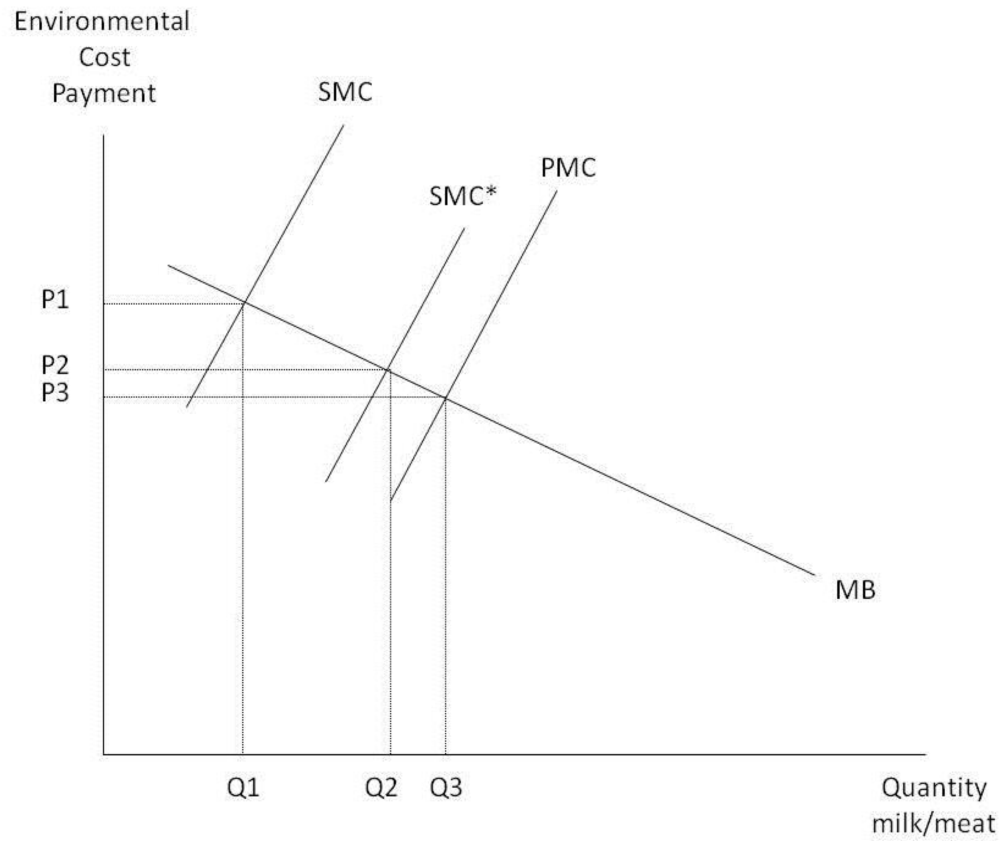

Fig 8.2: Social and private marginal cost curve shifts, where Y is the environmental cost of producing a litre/kg of milk/meat and the X axis is the quantity of milk/meat produced. SMC = the social marginal cost curve, PMC = the private marginal cost cure, and MB = marginal benefit.

However, there are potentially negative environmental issues to consider prior to the installation of on-farm AD systems. Unless the AD unit is located near a main gas line so that the biogas can be directly fed into a gas network, the operator will either have to clean the gas for vehicle fuel or convert to on-site electricity production, which may require investment and disruption (e.g. three-phase transmission lines) by utility companies. Traffic nuisance may also be a problem if feedstock is transported on-farm (Patterson et al., 2011), while vermin may gather if feedstock and digestate are not stored properly. A number of concerns have been raised regarding the environmental and economic impact of growing crops (e.g. maize) specifically for use as feedstocks in on-farm AD systems, as is done in Germany. Heavy metal build up from certain feedstocks can also be an issue as well as contaminates in the feedstock, both chemical and physical (plastic/glass). Additionally, the presence of heavy metals may be a problem in the application of the digestate to land if feedstocks are sourced from off-farm. Such potential negative impacts of on-farm AD must be taken into account when calculating the net environmental benefits of AD systems.

8.2 Further research

Although this work has furthered the knowledge of on-farm AD, the practical experiments were restricted to one digester and digestate type; and the economics chapter has also revealed knowledge gaps. The findings of the work cannot therefore be extrapolated too widely and there is a need for further research.

During the course of the thesis, the aim was to value the non-market benefits of on-farm AD. However, it would be impossible to value all the benefits in the available timescale, therefore two rationales were chosen when deciding which non-market benefits to value. Firstly, that reputable peer-reviewed literature was available allowing for economic valuation of scientific data. Secondly, the largest sources of agricultural pollution that could be mitigated by the introduction of AD were valued first. Accurate valuation of renewable energy is difficult, not only because stated preference techniques are normally used, but rural and urban households place different values on renewable energy, dependent on the type of technology. For instance, urban residents have been seen to prefer large off-shore wind farms, whilst rural residents place importance on the creation of local jobs (Bergmann et al., 2008). Within the UK, 20-35% of people reported to be willing to pay a premium for green energy (Batley et al., 2001; Fouquet, 1998); although more up to date work is required to determine if this remains the case, and how society ranks different renewable technologies against each other.

Important issues were not valued in this work due to lack of evidence; these being the value of reduced herbicide use (due to weed seed kill during AD), and particularly the economic benefits of heat. Although a combined heat and power (CHP) engine will provide the greatest energy recovery from biogas, one issue with on-farm AD is finding uses for the heat. Approximately 10-30% of heat is required for the digester, but this is minimal in the context of the total amount of heat produced. Innovative uses for the heat have been suggested including selling heat to a local school or swimming pool. However, such suggestions may not take into account that farms that have the economies of scale required for investment in on-farm AD are usually too far from households to sell the heat. More practical uses include the use for heating washwater on dairy farms, for heating sheds on poultry units, or for offices. Innovative uses for heat that deserve further research include for greenhouses (where the CO_2 from the CHP engine can also be used for the greenhouse), or evaporating a percentage of water from milk for further dairy processing, etc. Efficient use of heat may considerably aid the uptake of on-farm AD and would build upon payments recently introduced by the UK government for generation of heat (Renewable Heat Incentive).

8.3 Considerations for the future of on-farm anaerobic digestion

8.3.1 Benefits to farmers

There is approximately 1,500 million tonnes of animal manure produced in the EU 27 on a yearly basis and this provides the greatest potential non-crop feedstock source for AD (Holm-Nielsen et al., 2009). As marginal returns diminish in agriculture, farmers will need to increase efficiency of production and size or diversify into products with higher marginal returns. Blank (2001) has shown this to be the case in America, in that farmers have had to move up the value chain of farming, to higher revenue producing crops, while at the same time less marginal land is being farmed. AD may have the potential to increase the productivity of land as well as move farmers up the value chain. Farmers who adopt on-farm AD systems may be able to diversify into organic farming due to the weed seed kill and possible increased crop yield digestate offers above undigested manure.

Farmers have to comply with increasing environmental regulations but may also purposefully reduce pollution so could use AD as a marketing tool to capitalise on environmentally-conscious consumers.

Figure 8.3 illustrates how AD reduces the loss of methane, with the farmer dealing with the pollution on site. This has both social and private benefits, in energy security and less pollution, and the private benefit to farmers of a new revenue stream. Thus this helps farmers to internalize the externality of methane release in the production of milk and meat.

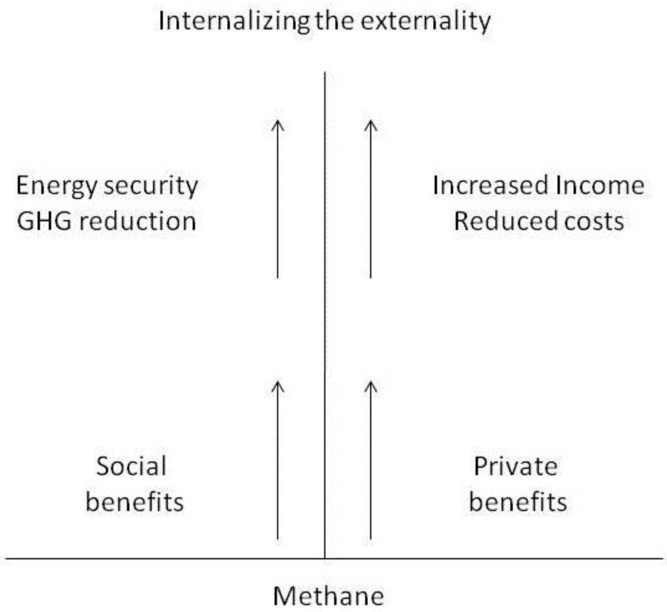

Figure 8.3: Vertical integration of pollution management (methane emissions) through on-farm AD

There is an abundance of winter housing units in the UK due to generous government and EU grants paid out over the last number of decades. The biodegradable commodity for on-farm AD, livestock manure, is already being stored in these winter housing units. Policy-makers should consider this historic investment in both valuation and assistance provided to farmers for future investment in AD.

8.3.2. Points of interest to the anaerobic digestion industry

There are a number of steps that could easily be implemented to aid the uptake of on-farm AD. Whilst the points made below aren't exhaustive, they discuss important constraints to on-farm AD: technical challenges and the lack of information sharing.

Many farmers appear to be hesitant in investing in AD systems due to a number of reasons, one of which is apprehension about the management required. AD companies should therefore include frequent site visits the norm, especially in the first 18 months after digester commissioning. However, remote monitoring through on-site internet connection enables off-site monitoring and early detection of problems (pH, low gas yield, etc.) is now possible and has previously been shown to be successful (Esteves et al., 2000; Spanjers & van Lier, 2006). From the AD unit operator's point of view, consistency and supply of feedstock is of utmost importance; material should be less than 12 mm, be as homogeneous as possible, and should contain the right balance of macro and micro-nutrients, as discussed previously. Such concerns and lack of knowledge may be reduced through greater discussion. Operators, manufacturers and academia should be encouraged to participate in more information transfer (e.g. through online AD discussion boards and social media). This would allow greater sharing of expertise and the extended benefit of increasing social capital within the AD community. Industry bodies such as the Anaerobic Digestion and Biogas Association obviously have a role to play in knowledge-transfer between the public, industry, academia, and policy-makers. AD may also need rebranding: the term digestate needs to be replaced with 'biofertilizer' as the latter may portray a better image to the public, whereas digestate is meaningless. Secondly, the referral of feedstock as 'waste' should cease as it comes with negative connotations; instead it should be called either wet or dry biomass. Finally, AD is a source of renewable energy, however, it provides positive environmental externalities above its sole purpose of producing renewable energy, and these benefits should be reflected through a higher FIT rate paid for electricity generated via AD compared to other renewable energy sources.

8.4 Future research required

As a result of valuing the non-market benefits from the introduction of on-farm AD, this thesis has shown up a vast amount of gaps in the literature that need addressing. This would aid more accurate valuation. The main gaps are listed in subsections below.

8.4.1 Uses for heat

Energy in the form of heat is a major source of renewable energy that is often not used to its full potential within on-farm AD systems. Increasing financial returns could be obtained from heat (e.g. through the RHI) and this would mean that more farmers would cross the threshold of financial gains from investing in AD. The use of heat could further displace burning of fossil fuels and hence would enhance the non-market benefits delivered by on-farm AD.

8.4.2 Destruction of weed seeds

There is insufficient information to evaluate the potential of on-farm AD as a mechanism to destroy weed seeds. Weeds such as docks (*Rumex* spp.) are problematic for the dairy agricultural sector in particular and can considerably limit productivity. Studies aforementioned in this thesis (Chapter 7) have shown AD to destroy weeds, but it is as yet unclear what the critical parameters are within an AD unit to ensure complete destruction. Again, this may allow the non-market benefits of reduced herbicide use to be calculated.

8.4.3 Feedstock

There is a vast array of potential biomass feedstock present in the UK that is not being utilised; including large sectors such as the seafood and alcohol industries. There is very limited peer-reviewed literature in relation to the suitability of a number of wastes that are potentially suitable feedstocks for AD; either when digested alone or co-digested with other feedstocks, in addition to elucidating optimum particle size, etc. The co-digestion of waste biomass from industries outside of agriculture could increase biogas yields and help close the nutrient cycle further; plus increase the economic returns and hence investment in AD. Further work is therefore needed so that the potential is reached.

8.4.4 Losses of nutrients from the application of digestate

The loss of nutrients both above and below ground from the application of digestate is an area of research that requires further experimental research. Gaseous losses of N (e.g. in the form of N_2O or volatilization of NH_3) or via leaching (e.g. NO_3^-) represent a loss of valuable nutrients and also negative environmental impact. Further research into this area is required for the optimization of AD systems and for the proper valuation of on-farm AD.

8.4.5 Anaerobic digestion of chicken manure

Although beyond the scope of this thesis, the use of chicken manure as a feedstock for AD requires further research. Poultry farms tend to produce a lot of manure in a small area of land. If AD could help reduce the environmental impact of poultry farming, this would provide both an economic and environmental benefit to the farmer and community. Areas of research in relation to the digestion of chicken manure include best practice for controlling the C:N ratio and reducing GHG emissions.

8.4.6 Knowledge of the British public in relation to AD

AD is not a well-recognised renewable energy provider outside of those involved in the industry. It would be of benefit to determine what percentage of the British public and indeed farmers are aware of AD. This would help to determine how the British public value AD as a renewable energy and pollution abatement technology. Further, it would help identify the social barriers that frequently impede AD developments, often based on misinformation (e.g. perceived health impacts) and NIMBYISM. The industry and policy-makers could then take positive steps to overcome such barriers.

8.5 Conclusions

One of the aims of this thesis was to narrow the divide that often exists between academia and industry in relation to research, as well as to progress the knowledge of AD. The thesis was written in such a manner that people involved in AD could understand the research, and people new to the area could obtain a good understanding of the benefits of AD, without requiring an in-depth knowledge of the system. Chapter 4 supports the limited number of other studies (Goberna et al., 2011; Möller et al., 2008; Sänger et al., 2011) that report on reduced leaching of nutrients after the application of digestate relative to other organic and synthetic nutrient sources. Chapter 5 has provided interesting findings as to the effects of applying digestate, undigested slurry and synthetic fertilizers has on the soil decomposer community. This showed that, with regards to soil microbial processes, digestate acts more like a synthetic fertilizer than undigested manure when applied to land. This may have implications for nutrient cycling and dynamics in agricultural systems. In addition to showing the improved agronomic value of digested, relative to undigested slurry, the three-year field trial reported in Chapter 6 found that the application of separated digestate to a pasture crop provide the same crop yield as synthetic fertilizer. Digestate also restricted clover growth as synthetic fertilizers do, above that of undigested cattle manure. Chapter 7 is an attempt to value the non-market benefits of AD. It concluded that AD is undervalued by government schemes designed to incentivise the uptake of renewable energy technologies as payment rates do not reflect the added advantages that AD offers in terms of pollution abatement.

This work ultimately discusses the economic benefits of on-farm AD in terms of FITs simply because the FIT is the mode by which the industry is paid by government for the generation of electricity and hence is the whole basis of growth. However, on-farm AD can deliver much more non-market benefits than what is measured through metering the units of electricity generated. Farmers are continually being forced to obtain higher revenues from less land and to reduce their environmental impact. AD has the ability to help farmers achieve both. An increased FIT rate of up to £0.089 per kWh of electricity produced from the AD of livestock manure would be equivalent to the potential saving associated with pollution abatement from the introduction of on-farm AD. This increase in a FIT payment would dramatically increase the uptake of on-farm AD, providing both economic and environmental benefits to the UK and would reflect the non-market benefits that on-farm AD delivers.

In addition, this work indicates that the FIT rate paid for the generation of electricity from AD is currently flawed. This is because to pay a flat-rate FIT to AD units regardless of the source of feedstock does not reflect the non-market benefits that units can offer. Specifically, it is proposed that those units which generate electricity from the digestion of feedstock sourced on-farm (e.g. manures) should receive a higher FIT rate than those AD units which digest purpose-grown crops (e.g. maize). This is because the former will deliver a host of other benefits (in the form of pollution abatement), that the latter will not. Indeed, it may be argued that to grow crops specifically for AD is environmentally unsustainable and hence this should be reflected in a lower FIT rate. We propose therefore that FITs should be thoroughly reviewed and amended accordingly.

Although the findings of pot trials and a field trial with one crop and soil type cannot be extrapolated too widely, the experimental work completed do at least serve as a sound basis for further work or add to the body of knowledge on AD. The initial aim of this thesis was to apply values to the non-market benefits from the introduction of on-farm AD. This was made possible by dedicated experimental trials during the PhD and through drawing findings from other relevant studies. Although there can be great debate on the economic valuations from this work, it has generated a novel chapter which identifies knowledge gaps but also raises important questions to policy-makers as to the validity of current payment rates for renewable energy generation through AD.

8.6 References

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