The Feasibility of Using Marginal Hay Meadow Biomass for Anaerobic Digestion

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Abstract

Marginal ('upland') areas in the UK are enjoyed by many tourists, and provide ecosystem services of clean water, soil carbon storage and agricultural produce (breeding livestock, meat and wool). However, poor financial return from the agricultural land puts it at risk of abandonment. Hay meadows in marginal areas are prized for their biodiversity, which is maintained by their traditional, low-input management including sheep production. If their management was abandoned, their biodiversity would fall. This thesis examined if bioenergy production (by anaerobic digestion (AD)) could provide a sufficiently high income compared to sheep farming, to prevent total abandonment. In addition to encouraging biodiversity, this could mitigate climate change by reducing fertiliser input and producing renewable energy. In marginal areas in northern England, fresh vegetation from five hay fields (not receiving inorganic fertiliser) was compared to five silage fields (receiving inorganic fertiliser), because grass silage is commonly used in AD. Biodiversity; biomass yield; biomethane production by laboratory AD; greenhouse gas (GHG) emissions; and financial returns from sheep farming or using the silage/hay crop in AD were measured. Compared to silage fields, the hay fields had greater biodiversity, but similar biomass yield. Hay had lower GHG emissions but similar biomethane production. And hay was more financially viable in AD than silage, partly due to lower cost. If sheep numbers reduced by 60%, biomethane electricity could meet EU sustainability requirements by saving > 50% GHG compared to fossil fuel electricity. However, in order to make a similar profit per ha to that achieved by sheep production, a farmer would have to sell their hay to a nearby AD plant (such as a dairy farm AD), rather than form a co-operative AD. Bioenergy is commonly perceived as an 'enemy' of biodiversity, but my work shows that bioenergy and biodiversity could co-exist, through AD of hay.

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Author’s Declaration

I declare that this thesis is a presentation of original work and I am the sole author. This work has not previously been presented for an award at this, or any other, University. All sources are acknowledged as References.

Chapter 1: Introduction

Climate change due to increased atmospheric greenhouse gases (GHG) is a threat to the stability of Earth's environment (Rockstrom *et al.* 2009). Its effects are being seen globally, in warmer air and seas, and higher sea levels (IPCC, 2007). The downstream consequences of such changes are wide-reaching, affecting both humans and nature. They include possible effects on the safety and health of humans; increased drought in some regions; increased extreme rain and flooding in others (IPCC, 2007); and loss of biodiversity (Harrison *et al*., 2015). In the future we could see reduced crop yields in southern Europe (Iglesias *et al.,* 2009) and the US (Liang *et al*., 2017), but increased yields in northern Europe (Iglesias *et al.,* 2009). Previously stable carbon may be increasingly released from soils (Hicks Pries *et al.,* 2017), accelerating climate change. These are tremendous challenges, but in addition the Earth faces an increasing population and its need for food, energy and water (Godfray *et al.* 2010). And biodiversity is being lost so rapidly that researchers claim we are in the midst of the sixth mass extinction (Chapin *et al*., 2000; Ceballos *et al*. 2015). Between 1970 and 2012, vertebrate population sizes dropped by 58% (WWF, 2016). In the UK, species population size dropped by on average 16% between 1970 and 2013, due to more species declining than growing in numbers (Hayhow *et al.,* 2016). Since biodiversity is the basis for ecosystem services essential to human well-being, there is strong international support to reduce further biodiversity loss (e.g. Convention on Biological Diversity, 1992 and 2010; Millennium Ecosystem Assessment, 2005).

Unfortunately, attempts to reduce GHG emissions, increase human settlements, increase food production, and preserve biodiversity lead to trade-offs, such as competition for land (Millennium Ecosystem Assessment, 2005; Youngs and Somerville, 2014; Steinhauser *et al*., 2015). Thus, they require very careful planning to reduce unwanted consequences. This research focuses on reducing GHG emissions by producing bioenergy from perennial grasslands. The aim is to assess if bioenergy production can provide an alternative income to farmers, to encourage the maintenance of grassland and its biodiversity, at the same time as generating renewable energy. Bioenergy is commonly seen as an 'enemy' of biodiversity but this work aims to examine if they can co-exist.

1.1 Biodiversity and ecosystem services

Biodiversity is defined as “the variability among all living organisms: within species, between species and of ecosystems” (Millennium Ecosystem Assessment, 2005 p.18). The biggest cause of biodiversity loss is land use, whereby habitats are intensified, changed or degraded (Newbold *et al.*, 2015; WWF, 2016). Grasslands are the most severely affected (Newbold *et al*., 2016). In the UK, agricultural intensification due to policy changes has been the biggest cause of biodiversity loss; although, at the other end of the scale, abandonment of low intensity farmland also causes biodiversity loss (Hayhow *et al*., 2016). The next biggest causes of biodiversity loss are climate change (Harrison *et al*., 2015; Seddon *et al.*, 2015); invasive non-native species; over-use and pollution (Millennium Ecosystem Assessment, 2005). Not all habitats are affected equally by climate change: the habitats where species are most at risk of range loss are upland (Pearce-Higgins *et al.*, 2017), montane, wetlands and coasts (Morecroft, 2017). However, a smaller number of species will actually benefit from climate change (Hayhow *et al*., 2016), particularly if they can withstand the shifts in distribution which are also happening (Pearce-Higgins *et al*., 2017; Pecl *et al.,* 2017). Ecosystems which have recently been disturbed (e.g. by cutting or fire) are more sensitive to climate change (Kroel-Dulay *et al*., 2015), including most agricultural land areas which are regularly disturbed during crop/livestock production. However, an ecosystem with a variety of species present is likely to have higher resilience to environmental changes (such as temperature extremes (Isbell *et al*., 2015)) than a system with fewer species present (Millennium Ecosystem Assessment, 2005), because ecosystems function best when a variety of species are present, particularly under variable environments (Hooper *et al.*, 2005, Isbell *et al*., 2011).

Ecosystems produce services (‘ecosystem services’) which are essential to human well-being. Ecosystem services have different functions, including provision (e.g. of food, fuel, water); regulation (e.g. of climate, water, disease); support (e.g. of soil formation and plant growth) and cultural (e.g. spiritual, recreation, aesthetic) (Millennium Ecosystem Assessment, 2005). Whilst a loss of biodiversity can benefit people through food production (e.g. crops and livestock), there are significant trade-offs (e.g. reduced crop resilience to pests and diseases (Millennium Ecosystem Assessment, 2005) and reduced insect pollination of fruits (Goulson *et al.* 2015)). Overall, loss of biodiversity negatively impacts human well-being (Millennium Ecosystem Assessment, 2005).

The large-scale losses of biodiversity which the world is witnessing at present are causing damage to "nature, society and the economy", with an estimated socioeconomic cost of 50 billion Euros per year (European Parliament, 2016 p. 4). Thus biodiversity must be proactively restored, maintained or enhanced to prevent further loss. Increasing the size of protected areas by only 5% could have substantial positive effects on biodiversity (Pollock *et al*., 2017). In Europe, incentives are paid to farmers to address biodiversity loss through agri-environment schemes, funded by the Common Agricultural Policy. The UK agri-environment schemes aim to increase agricultural biodiversity, and protect soil, water and landscapes (Natural England, 2013a). The schemes have had some success at increasing biodiversity, but more needs to be done to improve their outcomes (Hayhow *et al*., 2016) such as training farmers in implementing them (McCracken *et al*., 2015) and applying them in a more targeted geographical manner (Forestry Commission *et al.,* 2015).

1.2 Hay meadows

Biodiverse conservation targets around the world include areas of high plant and bird endemism, such as tropical islands and mountains (Brooks *et al*., 2006). However, the long history of human management of much of Europe means that many of the most biodiverse habitats in this region are associated with historical land uses, and many of these habitats are threatened because the traditional farming and forestry management that maintained them is no longer economic. One example of such a human-managed, biodiverse habitat under threat in the UK is upland hay meadow (UK Biodiversity Action Plan, 2011). It contains plant species such as wood crane's bill (*Geranium sylvaticum*), great burnet (*Sanguisorba officinalis)* (Joint Nature Conservation Committee, 2014) and globeflower (*Trollius europaeus*) (English Nature, 2001). Species-rich hay meadows also support invertebrates, bats, and birds which are on the UK's highest conservation priority list, including yellow wagtail, twite and curlew (Joint Nature Conservation Committee, 2014; Eaton *et al*., 2015). Therefore they are important sources of biodiversity in marginal, 'upland' areas. As an EU Habitat Directive habitat (and UK Biodiversity Action Plan priority habitat), upland hay meadow is to be conserved, protected and enhanced (UK

Biodiversity Action Plan, 2011). In addition to their biodiversity, they are valued for heritage and beauty (Jefferson, 2005). Once widespread, most species-rich meadows have been lost over the last half-century due to agricultural intensification such as re-seeding, addition of inorganic fertiliser and early harvest for silage production (ADAS, 1996). Furthermore, deposition of nitrogen from the atmosphere (Stevens *et al*., 2004, 2006) and climate change (Harrison *et al.*, 2015) are having negative effects on hay meadow species diversity and composition. There has been much effort to enhance the richness of rare species in existing hay meadows, and to increase species richness in grasslands with potential to be converted to traditional hay meadows (Critchley *et al.* 2007a). For example, the Yorkshire Dales Millennium Trust's Hay Time project has added species-rich hay meadow seed to 279 ha of hay meadows in the Yorkshire Dales (England, UK), and has seen successful establishment of early-successional species (Gamble *et al*., 2012). However, direct conservation efforts are only likely to restore a small percentage of the landscape to species-rich meadows.

It is the traditional management of upland hay meadows which maintains their plant diversity of up to 35 species per field (Jefferson, 2005; Smith *et al*., 2000). Farmyard manure is spread each spring and lime is applied occasionally to maintain a neutral soil pH; inorganic fertiliser is not used (Smith *et al*., 2000; UK Biodiversity Action Plan, 2011). They are usually grazed in early spring and autumn by sheep and some cattle, and are cut for hay after mid-July, which allows seed to set in late-flowering forbs (dicotyledonous flowering plants which are not grasses, sedges or rushes) (Smith *et al.* 1996). The cut grass and forbs are dried and turned in the field, dispersing seed which is an essential element in hay meadow restoration (Smith *et al.* 2008). If the traditional management is disrupted, species-richness can fall (Isselstein *et al.* 2005).

Grasslands support several ecosystem services including food for agricultural livestock, prevention of soil erosion (Hopkins and Holz, 2006) and carbon sequestration (De Deyn *et al*., 2011). Permanent grasslands contain large amounts of soil carbon (Pineiro *et al*., 2009). Grasslands also contribute to biodiversity. For example, at a landscape scale, a mosaic of mainly grassland plus woodland and crop has high biodiversity (Plantereux *et al.*, 2005). Twenty six percent of the global agricultural area is grassland (Panunzi, 2008) and in marginal areas of the UK (of poor agricultural quality, unsuitable for intensive food production), the predominant (58%) agricultural land use is managed grassland (Institute for European Environmental Policy *et al*., 2004). Thus, agriculture can make a large contribution to increasing biodiversity (European Parliament, 2016). However, low farm incomes (Keenleyside and Tucker, 2010), social and environmental issues (Renwick *et al.*, 2013) are resulting in the abandonment of grasslands (Allen *et al*., 2014), which leads in turn to the loss of their biodiversity (see Introduction to Chapter 3). The number of Welsh livestock farms has fallen substantially since 1996, such that the Welsh government has questioned if its uplands should instead be used for water and energy provision (Corton *et al*., 2013). When grasslands are abandoned, invasion by aggressive colonisers e.g. bracken can prevent ecological succession (to scrub and forest) leaving the grasslands in a stable, degraded condition (Cramer *et al*., 2008). Hilly and High Nature Value areas are most at risk of abandonment, but, in models, abandonment is much reduced if there is high production of feedstock for biofuel production (Keenleyside and Tucker, 2010). Increasing the income to farmers in such areas could incentivise them to carry on farming. Therefore financial incentives above those currently paid by agri-environment schemes may be needed to help prevent abandonment and loss of biodiversity.

1.3 Farms on marginal land

'Upland' land in the UK is designated by the European Commission as Less Favoured Area (LFA) land, in recognition of the numerous disadvantages faced by LFA farmers compared to non-LFA farmers: low soil fertility, steep and remote terrain, and a marginal climate (DEFRA, 2002). The LFA designation also originally had the aim of maintaining rural population numbers (Institute for European Environmental Policy *et al*., 2004) because LFA land is at risk of depopulation (European Environment Agency, 2012). However, since the 2013 reform of the Common Agricultural Policy, LFA land has been re-named 'areas with natural or specific constraints'. Concordantly, the primary aim of maintaining the land has changed from financial support of agricultural production to "environmental protection or improvement, maintenance of the countryside, preserving tourist potential" (European Commission, 2015). The term LFA is used throughout this thesis because it is well-known, and it remains as a search term in governmental websites of farm business. Marginal and LFA are used interchangeably in this thesis.

LFA land is found mainly in the north-west of England (covering iconic areas such as the Yorkshire Dales and Lake District), Scotland, the west of Northern Ireland and much of

Wales. There are 2.2 million ha of LFA land in England, 1.6 million of which is Severely Disadvantaged Area (SDA), and the remainder is Disadvantaged Area (Chesterton, 2009). SDA is the most disadvantaged land; the grassland fields I studied were on SDA land. Due to the beautiful, biodiverse nature of the landscape, almost two thirds of the English LFA is designated as national parks and Areas of Outstanding Natural Beauty (DEFRA, 2013b) and nearly 25% are Sites of Special Scientific Interest (SSSI) (English Nature, 2001). LFA land stores the largest amount of soil carbon in England in peat moorland; it is the source of 70% of the UK's drinking water; and it is used for grouse shooting and mineral extraction (DEFRA, 2013b; Galbraith *et al.*, 2013). The major LFA farming activity is extensive sheep and (to a lesser extent) cattle grazing (Institute for European Environmental Policy *et al.*, 2004). Even though farming conditions are harsh, LFA grazing livestock farms produce 44% of England's breeding ewes and 29% of beef cattle (Harvey and Scott, 2016). In 2006-2011, LFA grazing livestock farms made a loss each year when government financial support was excluded (DEFRA, 2011a), and thus they are heavily reliant on public money for their income; but there is considerable public support for preserving the LFA (Commission for Rural Communities, 2010; DEFRA, 2013b). Government funds are paid to farmers to maintain land in good agricultural condition (Basic Payment Scheme), or to forego income in exchange for farming the land in a more wildlife-and environmentally-friendly way (agri-environment payments, which are described in Chapter 6). LFA grazing livestock farms have a third of the farm business income than an average farm outside the LFA; although grazing livestock farms generally have low incomes whether they are inside or outside of the LFA (DEFRA, 2011a). Sixty-two percent of LFA farmers (median age 58) do not know who will succeed them in taking over the farm (DEFRA, 2011a), and hence the future of these farms is uncertain. But LFA farmers are seen as central to holding rural communities together (Parliament, 2010), and they manage the landscape in a way which is enjoyed by many tourists. Therefore, loss of farming in these areas would potentially have negative socio-economic consequences, as well as threatening the biodiversity that is associated with traditional landscape management.

1.4 Climate change and bioenergy

Strategies for managing climate change include adaptation (reduce vulnerability and adjust) and mitigation (tackle the causes by reducing GHG emissions) (IPCC, 2007).

Examples of adaptation are to increase the storage of water; change date of crop harvest; and move human settlements, for example to locations at reduced risk of flooding (IPCC, 2007). Mitigation may be classed as biological, social or technological. Biological examples include increasing soil carbon sequestration (Lal *et al.,* 2007) or afforestation (Woodward *et al.,* 2009). Social examples may include commuters using less energy-intensive modes of transport such as cycling (IPCC, 2007). Technological examples include switching to fuels that minimise GHG emissions; and increasing efficiency in food production and energy use (IPCC, 2007). The EU has set targets of 20% reduction in GHG emissions compared to 1990's levels, and 20% of energy used must derive from renewable sources by 2020 (European Commission, 2009). The UK's target for renewable energy consumption by 2020 is 15%, a large increase from 1.3% in 2005 (European Commission, 2009). Therefore every avenue of renewable energy research must be explored. The EU's target increases to 27% of energy from renewable sources by 2030 (European Commission, 2016a) and a 40% reduction in GHG (European Commission, 2017a). The UK's 2050 target is a very ambitious 80% reduction in emitted carbon (Climate Change Act, 2008). The 2015 Paris Agreement was the first globally-agreed plan on climate change, aiming to restrict global warming to less than 2^{o}C (European Commission, 2016b). The largest contributors to GHG emissions are combustion of fossil fuels (72%, emitting mainly carbon dioxide (CO_2) and some methane (CH_4)), agriculture (12%, emitting methane and nitrous oxide (N_2O)) and land use change (10%, emitting carbon dioxide) (Canadell and Schulze, 2014). The UK has reduced its agricultural GHG emissions by 20% since 1990, mainly due to reduced emissions from soils (DECC, 2012). Agricultural methane is emitted mainly by enteric ruminants (e.g. cows, sheep) and rice cultivation; and agricultural nitrous oxide is largely from fertiliser applied to soil (Canadell and Schulze, 2014), therefore these are targets for GHG reductions (e.g. Hyland *et al*., 2016; Soussana *et al*., 2010; Seitzinger and Phillips, 2017). By using non-fertilised grasslands for bioenergy production, soil emissions from nitrous oxide could be reduced due to the lack of fertiliser; and carbon dioxide could be reduced by replacing fossil fuel combustion with renewable energy.

However, developing sustainable strategies to tackle climate change is an extremely complex task. Biofuels were promoted as sustainable renewable energies which would reduce the largest emitter, fossil fuel use (European Commission, 2009). Yet first generation biofuels (made from edible crops such as maize and rapeseed) had devastating effects, due to the displacement of food production leading to increased food prices; and the loss of biodiverse habitats such as rain forest, due to the advancement of agricultural land area (Renewable Fuels Agency, 2008). The land use change involved in converting grassland to biofuel crops, along with the higher fertiliser applications which may be required for annual crops, also result in increased greenhouse gas (GHG) emissions (Fargione *et al*., 2008; Searchinger *et al.,* 2008). It can take 50 to 93 years to recoup the carbon dioxide released from grassland soils (the 'carbon debt') which are converted to maize for bioethanol production (Fargione *et al*., 2008; Pineiro *et al.* 2009). However, second generation biofuels made from lignocellulosic material such as wastes, grass or wood can have lower GHG emissions and lower general environmental impact (e.g. on biodiversity and water use) than first generation biofuels (Scharlemann and Laurance, 2008). Thus the UK's Gallagher Review (Renewable Fuels Agency, 2008) stated that biofuel production must focus on using waste products, or if dedicated biofuel crops are to be grown, they should use idle and marginal land in order to avoid competition with food production. They must also be as energy efficient as possible (Patterson *et al.*2008). The Gallagher Review concluded that there is probably sufficient land in the UK to meet the needs of food, animal feed and biofuel production until 2020. Within the EU there may be 1 to 1.5 million ha of land which could potentially be planted with bioenergy crops, excluding forests (Allen *et al.*, 2014). Marginal grasslands can themselves be used for bioenergy production, avoiding competition with intensive crop production (Tilman *et al*., 2006).

1.5 Energy crops

The introduction of energy crop plantations such as *Miscanthus* or short rotation coppice (SRC) willow to the uplands could drastically change the look of the landscape, but may encourage maintenance of the land. However, even though SRC willow could be more profitable than sheep faming, Reed *et al*. (2009) report that most LFA farmers would not be interested in bioenergy crop production, even if financial returns were high. Reasons included distance to a bioenergy production plant, and the lack of other producers and knowledge in the local area. SRC willow would require land use change, including ploughing, which can have very negative effects on the carbon balance of bioenergy (Djomo *et al*., 2012). SRC willow would also probably need fertiliser (Reed *et al*., 2009), but despite this its nitrous oxide emissions are 40-99% less than annual energy crops due to lower fertiliser use and higher nitrogen use efficiency (Don *et al.*, 2012). According to Don *et al.*, (2012) converting European grassland to SRC willow or *Miscanthus* may produce neither carbon savings nor a carbon debt. However, mature *Miscanthus* plantations have lower biodiversity than grasslands (Dauber et al., 2015; Donnison and Fraser, 2016), therefore *Miscanthus* is not a suitable crop to be planted in marginal areas if the aim is to maintain biodiversity. To avoid GHG emissions from land use change and the need for increased fertiliser application, the research presented in this thesis will focus on assessing the biofuel potential of plants already growing in the marginal areas (i.e. grasslands).

1.6 Bioenergy production from grass

Grassland biomass can potentially be used for bioenergy production in several ways, including combustion, biogas production (by anaerobic digestion) and ethanol production. Grass for combustion is harvested late in the year (aiming for maximum dry biomass) and receives a low level of fertiliser, which usually supports increased biodiversity on the land, but its combustion is technically more challenging than burning wood (Prochnow *et al.* 2009a; Wachendorf *et al.* 2009). It contains higher concentrations of nitrogen, potassium, magnesium, sulphur and chlorine than wood, leading to the release of nitrous oxide which is a pollutant and GHG; the furnace is corroded by chlorine and sulphur compounds; and a lower ash melting temperature causes the formation of slag, reducing the working life of the processing plant (Prochnow *et al.* 2009a). Harmful particulates are also released into the atmosphere unless complex adaptations are made (Wachendorf *et al.* 2009). Despite this, 404,000 t of straw (from wheat, barley and oats) was burned in UK power stations in 2015 (DEFRA, 2016a).

Anaerobic digestion (AD) of grass, in comparison, has received more government and research interest than combustion of grass (e.g. the UK developed an Anaerobic Digestion Strategy and Action Plan (DEFRA, 2011b)); and AD of grass silage is a well-established technology (Prochnow *et al.* 2009b). Biogas production using wastes (e.g. food waste, the organic fraction of municipal solid waste or manure) could potentially lead to large savings in GHG emissions, when the biogas is used for heat and power production, or as a transport biofuel (European Commission, 2009). Seventy-four percent of feedstocks digested in AD in the UK were wastes in 2014 (Waste and Resources Action Programme, 2017) because the UK government is encouraging AD of organic waste rather than crops

(DEFRA, 2011b). This reduces the amount of waste entering landfill, and is a source of renewable energy (Rural Economy and Land Use Programme, 2011). The Renewable Energy Directive (European Commission, 2009) also states that because biogas plants can be decentralised, they can contribute significantly to sustainable development in rural areas and offer farmers additional sources of income. However, European biofuels originating from human-maintained biodiverse grassland will not qualify for EU financial support, except if it is proven that harvesting is necessary to maintain it as grassland. Neither should biofuels be made from areas designated for nature protection, except if the production of the vegetation does not interfere with this protection (European Commission, 2009). The grasslands examined for bioenergy production in this thesis were located in a National Park, or in an Area of Outstanding Beauty, but grazing and/or cutting is necessary to maintain their species diversity. Therefore, they would qualify for EU financial support. Thus, bioenergy production in diverse upland meadows could potentially receive support from existing, and potentially future, subsidies without harming biodiversity.

1.7 Anaerobic digestion

As a renewable energy technology, anaerobic digestion has some advantages over other renewable energies (DEFRA, 2011b). For example it forms a continuous source of energy (unlike the intermittent power of wind or solar energy); its energy is storable (as biogas); and compressed biomethane can be used in adapted Heavy Goods Vehicles (DEFRA, 2011b). Germany is the largest biogas producer in Europe with over 10,000 AD plants, supported by a substantial subsidy (Prochnow *et al*., 2009b); and the UK is second biggest producer (in the IEA Bioenergy Task 37 group of countries), with 913 AD plants (International Energy Agency, 2015). Anaerobic digestion (AD) occurs when microbes break down degradable organic material in an oxygen-free environment, producing biogas which contains methane. Undigested material remaining at the end of AD (digestate) is rich in nutrients and can be used as a fertiliser (Lukehurst *et al*., 2010). Types of AD plant in Europe range from large-scale commercial plants digesting food waste (DEFRA, 2016a); landfill plants collecting landfill gas; water treatment facilities making biogas from sewage sludge (International Energy Agency, 2015); and small-scale plants on farms digesting cattle slurry. On-farm anaerobic digesters in Germany and Austria tend to digest a mix of crops (e.g. maize silage, grass silage, sugar beet) and agricultural waste (e.g. cattle or pig manure) (Murphy *et al*., 2011). Digesting manure by AD captures the GHG (Petersson *et al*., 2009) which are otherwise released to the atmosphere (as methane) during storage in open tanks (DEFRA, 2011b). Other AD feedstocks may include algae (Murphy *et al*., 2015), abattoir waste (Browne *et al*., 2013) and card packaging (Zhang *et al*., 2012). Thus AD can produce energy from a variety of organic feedstocks, including grass.

Biogas produced during AD contains 50-70% methane, 25-50% carbon dioxide and potentially small amounts of ammonia, hydrogen sulphide, nitrogen, water vapour, oxygen and hydrogen (Anaerobic Digestion and Bioresources Association, 2013, p. 60). It can be burned in a boiler producing heat (Hopwood, 2011), or in a combined heat and power plant producing electricity and heat, although wastage of the heat is common (Rosch *et al.,* 2009). Alternatively, the biogas can be upgraded to biomethane (97% methane) and injected into the natural gas grid, for example to fuel a combined heat and power plant in an area where the heat can be used locally (Redman, 2010). However, biogas upgrading and injection into the gas grid is expensive (Petersson *et al*., 2009) and beyond the reach of the small rural farms studied in this thesis; although a central AD plant with upgrading facility in a rural area of Denmark has started receiving and upgrading biogas from local farm AD plants (International Energy Agency, 2017). Biomethane used as a transport fuel can have a better energy balance than biodiesel or bioethanol (Patterson *et al.*, 2008; Thamsiriroj *et al*., 2011). The EU proposal for renewable energy post-2020 (European Commission, 2016a) aims to increase renewable heating and cooling, which biogas could provide. Such a strategy is likely to continue within UK legislation, under the aegis of the Paris Climate Change Agreement, regardless of the future political and economic relationship between the UK and EU.

1.8 Biochemical reactions in anaerobic digestion

AD comprises a series of enzymatic and microbiological reactions (Angelidaki *et al*., 2009) carried out by a complex community of microbes which are interdependent on one another (Schink, 2008, p. 171) (Figure 1). The first is hydrolysis, where complex molecules (carbohydrates, proteins and lipids) are broken down by bacterial enzymes into monomers (glucose, fatty acids and amino acids). The second stage is acidogenesis, where the monomers are broken down by bacteria into volatile fatty acids (including acetic, propionic and butyric acids) and alcohols, hydrogen sulphide, ammonia and carbon dioxide (Anaerobic Digestion and Bioresources Association, 2013 p. 40). The third stage is acetogenesis, where volatile fatty acids and alcohols are converted into acetate, hydrogen, ammonia and carbon dioxide. Fourthly, hydrogen-using methanogenic archaea convert hydrogen and carbon dioxide to methane; and acetate-cleaving methanogens convert acetate to methane and carbon dioxide (Anaerobic Digestion and Bioresources Association, 2013 p. 40).

When the system is disturbed (by conditions such as feedstock overloading, changes in temperature or substrate, or the presence of toxins), chemical intermediates such as volatile fatty acids, hydrogen and alcohols accumulate (Ahring *et al.*, 1995). Volatile fatty acid accumulation is an indicator of uncoupling between the acid-producing microbes (during acidogenesis) and the acid consumers (the slower-acting methanogens) (Ahring *et al.*, 1995). An acid pH (<6.5) due to volatile fatty acid accumulation inhibits methanogens, reducing methane production (Angenent and Wrenn, 2008 p. 183). Accumulation of hydrogen in the headspace above the digester (a partial gas pressure of more than 10^{-5} atm.) inhibits the conversion of propionic acid to acetate, causing propionate accumulation, which also reduces pH (Angenent and Wrenn 2008, p. 183). At lower feedstock loading rates, most methane is made via acetate, and at higher loading rate most methane is made via hydrogen and carbon dioxide (BMZ, 2012). Initially, AD plants comprised single phase digesters in which all reactions occurred in one reactor, but higher feedstock loading rates have been achieved by separating the hydrolysis and acidogenic phases from the neutral pH-requiring methanogenic phase (Nizami *et al*., 2009).

The flexibility of an established methanogenic system to new feedstocks is somewhat limited, that is, the microbes may require time to adjust (Schink, 2008, p 174). In waste water systems mixed cultures of microbes are used to degrade the complex mix of carbohydrates, proteins and lipids (Angenent and Wrenn, 2008, p. 180). Sugars and starch are more easily broken down than structural carbohydrates (such as ligno-cellulose), proteins and fats (Angenent and Wrenn, 2008, p. 181). Other compounds present in the feedstock, such as polyphenols and furfurals, can inhibit microbial activity (Angenent and Wrenn, 2008, p. 181).

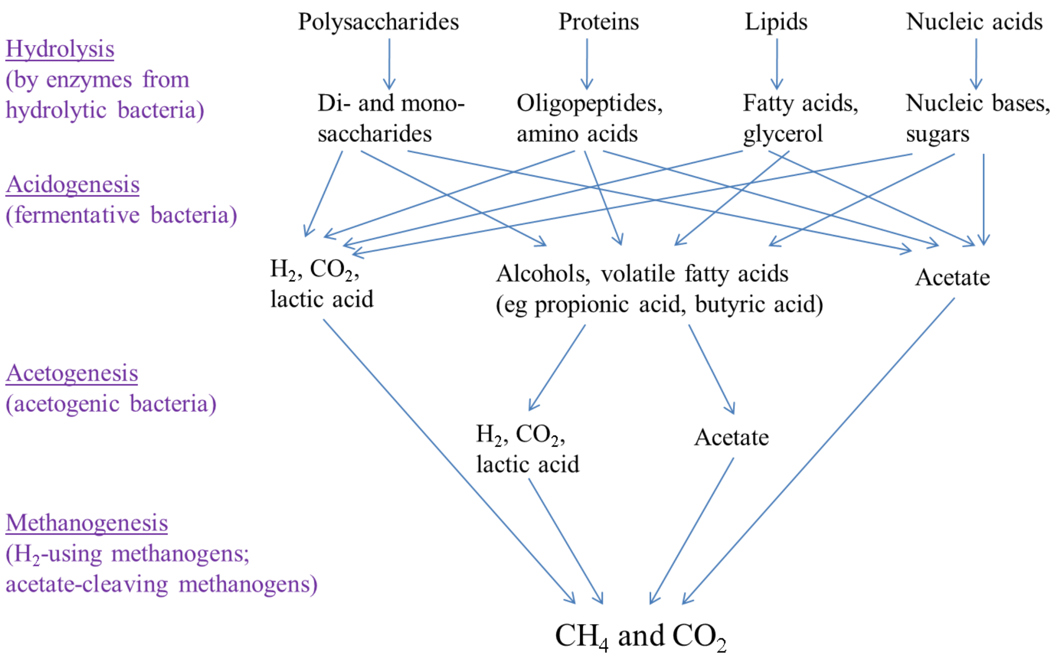

Figure 1. Microbiological processes in methane production by anaerobic digestion. Adapted from Schink (2008) and BMZ (2012). H_2 is hydrogen.

1.9 Financial viability of anaerobic digestion from marginal grasslands

Whether grasslands can be realistically used for bioenergy or not will be determined by the profitability of bioenergy production. This topic is introduced and studied in Chapter 6. The financial returns of the grassland fields as currently managed, for grassland crop and sheep production, were compared with using the grassland biomass in bioenergy production, to determine if it is financially competitive to develop AD production systems in marginal areas. In addition, the economic value of a subset of the social benefits derived from grasslands (biodiversity and GHG saved by not using inorganic fertiliser) was examined. The role of agricultural government payments and energy subsidies were also studied, and potential changes due to the UK leaving the European Union were considered. Only if it is financially viable, will it be possible to maintain and even expand existing traditional hay meadow management to a larger part of the landscape.

1.10 Question asked

It is important to account for all factors affecting a bioenergy production system, because the sustainability and environmental impact of bioenergy production involves much more than just GHG emissions (Gilbert *et al*, 2011). Therefore, this research included studying the biodiversity and financial return, as well as GHG emissions, of bioenergy production from marginal grasslands (which were not manipulated; the fields were managed as normal by the farmers). The overall question asked by this thesis is: can marginal grassland fields produce an anaerobic digestion feedstock, which helps mitigate climate change, provides the farmer with an alternative income and where the farming system is beneficial for wildlife?

I compared biodiverse hay grassland (which did not receive inorganic fertiliser) with species-poor silage grassland (which received inorganic fertiliser), because the latter (silage) is the more typical grass product used in AD. I aimed to determine if biodiversity and bioenergy production could feasibly co-exist.

The hypotheses were that silage (fertilised) fields would have

1) lower plant diversity,
2) higher biomass yield,
3) higher methane production by AD,
4) higher grassland production GHG emissions, and
5) higher financial returns when the silage is used in AD.

If AD of hay was financially and environmentally competitive compared to AD of grass silage, this could (i) prevent abandonment and encourage the creation of increased areas of hay meadow (increasing biodiversity); (ii) reduce GHG emissions by reducing nutrient (inorganic fertiliser) input; (iii) increase the production of renewable energy; (iv) provide an alternative income for farmers; and (v) prevent land use change which increases carbon emissions (Searchinger *et al.* 2008). Few research articles have compared silage and hay grasslands spread across a marginal landscape (compared to studying the effects of different managements within one field). For example, it is unknown how the biodiversity of marginal hay meadows differs compared to marginal silage fields. Additionally, there are relatively few papers studying links between land management intensity, biodiversity and GHG emissions (Waterhouse and Ricci, 2013).

Specifically, in Chapter 3 I studied the biodiversity of grassland; and in Chapter 4 I studied the biomethane production by laboratory AD of grassland samples. In Chapter 5 I estimated GHG emissions from grassland production, and GHG emitted when making electricity from the biomethane (to compare with other sources of electricity). Finally, in Chapter 6, I estimated the financial return of grassland AD, and compared it to the financial return of current field management (grassland crop and sheep production).

Chapter 2: Methods

This Chapter explains the selection, location and management of the fields studied in this thesis. I did not manipulate the management of the fields; they were managed by the farmers under their usual management. Surveying and sampling the fields (described in Chapter 3) was carried out as soon as possible before the farmer's harvest to give an estimation of what farmers are actually producing.

2.1 Finding grassland fields to study

In order to assess the use of biodiverse marginal grasslands in bioenergy production, I had to find biodiverse grassland fields, and compare them to species-poor fields. My supervisors and I had no prior contacts of anyone with such land. The first method of trying to find fields (by looking at botanical surveys of farmland) was not successful and is explained in the Appendix.

The next method of finding land was successful. It involved finding farmers and asking if they had one non-fertilised grassland (cut for hay, which was expected to be more biodiverse); and one fertilised grassland (cut for silage, which was expected to be less biodiverse). A meeting was arranged with Nidderdale Area of Outstanding Natural Beauty, where I was introduced to a local farmer. He had the type of fields required, and agreed to take part in the research. He also recommended several more farmers in Nidderdale, and I visited them. None agreed to take part, although after I rang another farmer in Nidderdale, he agreed to take part. In addition, I approached the Yorkshire Dales Millennium Trust's Hay Time Project because they knew local farmers with hay meadows. I wrote a questionnaire which they sent out to farmers, asking if they had the fertilised and non-fertilised field types, and if they were willing to take part in the research. Amongst a larger group that replied, three suitable farmers were found. This totalled five farms for my research.

2.2 Location and management of fields

The farms were located in hilly areas of Nidderdale and the Yorkshire Dales (which may be referred to as the 'uplands'), in the north of England (Figure 1). The farms had a wet, windy and cool maritime temperate climate (Hensgen *et al.*, 2014) and a mean annual air temperature of 7.2^{o}C in 2010-2011 (UK Meteorological Office, 2013). I surveyed them in 2011 and 2012. They occupied Severely Disadvantaged Area (SDA) land, which is the most disadvantaged land within the larger classification known as Less Favoured Area (LFA) land. LFA land is defined by the European Union as poor agricultural quality land, due to poor climate, soil quality, aspect and relief (DEFRA, 2011a). It can support extensive livestock production but not the production of food crops (DEFRA, 2011a). This is the type of land most likely to be abandoned. In this thesis, LFA land (and the SDA land within it) is called 'marginal'. All the farms produced sheep and beef cattle, which were sold for breeding livestock and meat. Thus the farmers are classed as LFA grazing livestock farmers.

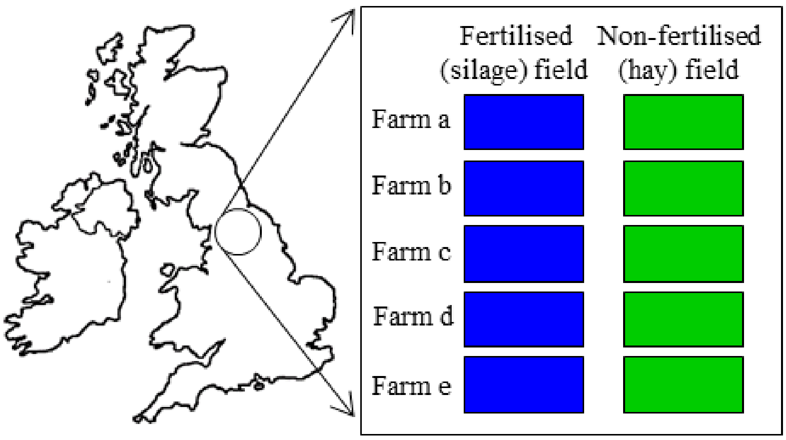

Figure 1. Fields were in the north of England, UK. Each farm (a-e) had a fertilised grassland field (producing silage) and non-fertilised grassland field (producing hay). Each field was sampled by 4 random quadrats.

Each farm had one fertilised field (receiving inorganic fertiliser) and one non-fertilised field (no inorganic fertiliser). Thus there were 5 fertilised and 5 non-fertilised fields. Photos in Figures 2 and 3 show examples of fertilised and non-fertilised vegetation. The fertilised fields were generally cut for silage; and the non-fertilised fields were generally cut for hay. (Throughout this thesis, 'fertilised field' is used interchangeably with 'silage field'; and 'non-fertilised field' is used interchangeably with 'hay field'). In the results Chapters 3 and 4, the fields are referred to as fertilised and non-fertilised because the focus is on the intensity of the land management. In these Chapters, vegetation yield excludes any dry matter losses. However, in reality, dry matter losses occur when processing the cut grass into silage or hay (Section 2.3). Therefore in Chapters 5 and 6, where the focus moves to the use of the processed vegetation (silage or hay) in bioenergy production, for simplicity the fields are referred to as silage and hay fields. Dry matter losses from the vegetation during processing are taken into account in Chapters 5 and 6, to give a more realistic idea of what greenhouse gas emissions and financial returns may be achievable by using silage or hay from marginal grasslands in bioenergy production.

The silage/hay and sheep managements were reported to me by the farmers (Table 1). Both field types received farmyard manure (FYM: a mixture of faeces, urine and straw from cattle housed in winter). The amounts of FYM applied were estimated by the farmers, or unknown. Where unknown, the average was assumed. All fields were grazed and harvested as part of the farmer's usual management. Grazing was mainly by sheep (with a few beef cattle) during autumn and early spring; there was also some winter grazing. Cattle tended to be on the fields for a few days at a time, therefore they were excluded from this research. The numbers and type of sheep grazing the fields are shown in Table 2. Harvesting occurred once per annum except fertilised field (b) which had a second harvest in August; this was not measured. Thus, the fields are suitable for heavy machinery. Grass cut for silage is generally cut earlier in the summer than that for hay. The later cut for hay allows late-flowering plants to release seed, maintaining floral diversity. However, on farm (a) both field types were harvested by the farmer on the same day: he made haylage from the fertilised vegetation (where late-cut vegetation is cut, baled and wrapped like silage). Harvest dates vary depending on the weather, therefore Table 1 shows the farmer's desired harvest date. The silage and hay (forage) is fed to livestock in winter. All non-fertilised fields (and fertilised field (c)) were in European Union agri-environment schemes which pay for income foregone and the costs of carrying out lower input management. Fertilised fields (b) and (e) had been re-sown with productive grasses 19 and 8 years previously, respectively. Some fields received lime recently, whilst others did not receive any at all.

The method of sampling each field using quadrats is described in Section 3.2.2.

Figure 2. Quadrat in a fertilised (silage) field.

Figure 3. Quadrat in a non-fertilised (hay) field.

2.3 Silage- and hay-making in more detail

Grass for silage is cut pre-flowering, when its sugar levels are adequate, around June. It contains more protein and energy at this stage than older grass (Agriculture and Horticulture Development Board, 2015). Before cutting, grass has a dry matter content of around 18% (Nix, 2015): when making silage the cut grass is left to wilt (dry slightly) in the field to around 25% dry matter content. It's then chopped, pressed into bales and wrapped in plastic, to exclude air and start anaerobic acid production which preserves the grass (Agriculture and Horticulture Development Board, 2015). Grass for hay is cut after flowering, from mid-July; it is left on the field to dry for longer than silage, and is turned several times over several days to aid drying. It is dried to around 86% dry matter content, then it is baled and stored dry. As well as water loss which occurs from transforming fresh grass to silage or hay, losses of organic matter occur. These occur during harvest, drying, transport and storage: in this thesis 18% loss in dry matter when making silage was assumed, and 36% loss of dry matter when making hay was assumed (Buhle *et al*., 2012).

Table 1. Field management in 2011.

Farm	Field Type	Grid reference	Aspect	Altitude (m)	Inorganic N (kg N/ha)	FYM N (kg N/ha)	Total N (kg N/ha)	Year lime applied
a	Fert.	391545 485649	Flat	330	25	67	92	1990
b	Fert.	414729 466288	Flat	140	50	59	109	2011
c	Fert.	370971 486749	N and S	140-170	53	67	120	1980
d	Fert.	407667 474456	NE	270-310	50	67	117	-
e	Fert.	405796 484354	SE	280-290	50	74	124	-
a	Non-fert.	391351 485411	NW	310-340	0	67	67	1990
b	Non-fert.	409518 474584	E	240-270	0	59	59	2011
c	Non-fert.	371043 486761	N	140-160	0	67	67	1980
d	Non-fert.	407798 474381	Flat	270	0	67	67	2010
e	Non-fert.	404234 484127	S	400-410	0	74	74	2000

FYM has a mean nitrogen (N) content of 6.0 kg t^{-1} (DEFRA, 2015b).

Table 1 continued.

Farm	Field Type	Grazing	Re-seeded	Farmer cuts from	End-use	Agri-env. scheme
a	Fert.	Sheep, a few cattle: autumn, early spring.	-	25 July	Haylage	-
b	Fert.	Sheep: autumn-early spring, not spring 2011.	1992	10 June	Silage	-
c	Fert.	Sheep: summer-spring. Cattle: July, Aug.	-	mid-June	Silage/hay	ELS
d	Fert.	Sheep: Aug-autumn, early spring.	-	20 July	Silage	-
e	Fert.	Sheep: Aug-Nov, early spring. Cattle: autumn	2003	late June	Silage	-
a	Non-fert.	Sheep, a few cattle: autumn, early spring.	-	25 July	Hay	HLS
b	Non-fert.	Sheep: autumn, most of winter, spring.	-	mid-July	Hay/silage	CS
c	Non-fert.	Sheep: late summer-spring. Cattle: July, Aug.	-	mid-July	Hay	HLS
d	Non-fert.	Sheep: autumn, some winter, early spring.	-	31 July	Hay	CS
e	Non-fert.	Sheep: Aug-Oct, early spring.	-	late July	Hay	CS

Cattle tended to be on fields for a few days at a time therefore were not included in this research.

ELS is Entry Level Stewardship

HLS is Higher Level Stewardship

CS is the old Countryside stewardship

Table 2. Sheep numbers, and length of time present, on the fields.

Farm	Sheep type	Fertilised (silage) field		Non-fertilised (hay) field	
		No. sheep/ha	No. months	No. sheep/ha	No. months
a	Ewes	10	2.5	10	2.5
	Lambs	8	4	8	4
b	Ewes	26	5	6	6
	Lambs	0	0	12	2
c	Ewes	12	6.5	12	6.5
	Lambs	15	3.5	15	3.5
d	Ewes	9	4.5	11	3.5
	Lambs	15	3	21	3
e	Ewes	8	1.5	4	1.5
	Lambs	15	2.5	9	2.5

Chapter 3: Biodiversity, yield and land management of marginal agricultural grasslands

ABSTRACT

Bioenergy production should ideally be compatible with the maintenance of biodiversity. Non-fertilised agricultural grasslands (receiving no inorganic fertiliser) growing in marginal (poor quality) areas are important contributors to agrobiodiversity and ecosystem services. However, they are being abandoned, intensified or their land use changed, leading to biodiversity loss. This might be prevented if they were used for bioenergy production. In this chapter, I assess the plant biodiversity, biomass yield and soil characteristics of fertilised (silage) and non-fertilised (hay) marginal grassland fields in the UK. Non-fertilised fields had higher plant species richness, a significantly higher number of conservation indicator species, and significantly lower grass cover than fertilised fields (which receive higher N input through inorganic fertiliser use). However, they had similar mean harvested and total annual biomass yield. Therefore, non-fertilised grassland fields in marginal areas have potential to provide bioenergy feedstock which is more environmentally sustainable than the more commonly used species-poor grasslands. There may also be greenhouse gas and economic benefits arising from the lack of inorganic fertilisers in non-fertilised grassland (studied in Chapters 5 and 6 of the thesis).

These results may be specific to the soil and management of these farms (the fields were not manipulated), but they nonetheless give an indication of the potential sustainable use of biodiverse grasslands in marginal areas for bioenergy production, without changing their management. If not used for a purpose other than sheep farming, these fields could potentially be abandoned or changed to alternative land uses in the future, and their biodiversity and associated conservation priority species threatened.

3.1 INTRODUCTION

Climate change mitigation efforts to replace fossil fuel are causing rapid expansion of bioenergy production (Dornburg *et al.*, 2010). Biodiversity has already been greatly impacted by intensive agriculture and forestry (Tscharntke *et al.*, 2005; Foley *et al.*, 2011), causing important ecosystem service declines (Bjorklund et al., 1999; Millenium Ecosystem Assessment, 2005), for example in crop pollination and crop disease resistance. Biodiversity losses are likely to increase further if there is additional land conversion (Tilman *et al.*, 2001; Newbold *et al.*, 2015) and intensification for bioenergy production systems. A major challenge, therefore, is to develop bioenergy systems that are compatible with other demands on the land (Smith *et al.*, 2013; Canadell and Schulze, 2014), such as biodiversity and food production, as well as other ecosystem service provision.

Biodiversity conservation and bioenergy production are potentially compatible on marginal grassland (of poor agricultural quality) (Tilman *et al.*, 2006; Fargione *et al.*, 2009; Stoof *et al.*, 2015), avoiding land use change (Searchinger *et al.*, 2008). Many traditionally-managed marginal grasslands have been lost (Isselstein *et al.*, 2005) due to (i) intensification to increase productivity (including the addition of inorganic fertilisers, which incur costs and potentially generate additional greenhouse gases (Gilbert *et al.*, 2011)); (ii) land use change (e.g. to forestry); or (iii) abandonment (where agricultural management is withdrawn) (Allen *et al.*, 2014). The causes of abandonment are multifaceted but it often occurs due to low vegetation productivity and low economic return (Keenleyside and Tucker, 2010). Abandonment is occurring in the EU (where 600,000 ha of agricultural grassland were abandoned in 2009-2012 (Allen *et al.*, 2014)), North America and East Asia (Secretariat of the Convention on Biological Diversity, 2014). Abandonment leads to (i) ecological succession and reduced numbers of meadow-associated plants (Losvik, 1999; Isselstein *et al.*, 2005), orthoptera (Marini *et al.*, 2009) and birds (Werling *et al.*, 2014), and/or (ii) lower ecosystem service value (Cramer *et al.*, 2008). Proponents of 're-wilding' may encourage grassland abandonment, but it can actually lead to reduced species richness if dominant plants such as bracken or *Molinia caerulea* invade and exclude other species (Hajkova *et al.*, 2009; Fraser, 2014). This is already happening in old abandoned fields in Central America, Australia, and the Mediterranean, where ecological succession has halted and the fields are in new, stable, degraded conditions (Cramer *et al.*, 2008).

The production of bioenergy from at-risk grasslands may reduce their chance of abandonment by farmers if they increase financial returns from the land. Bioenergy production might be achieved most efficiently by planting dedicated bioenergy crops, but with the potential drawback that it may reduce biodiversity and incur a carbon debt due to land use change (Fargione *et al*., 2008; Searchinger *et al.*, 2015). Alternatively, use of low intensity, traditionally-managed grassland could maintain higher biodiversity levels, although this may come at the cost of a lower bioenergy yield. Grassland yields may be potentially increased by fertiliser: 58% of Great Britain's grazing grassland receives inorganic fertiliser (DEFRA, 2015b). Higher N in a grassland system usually leads to increased dry matter yield (Schellberg *et al.*, 1999; Scotton *et al*., 2014), due to changes in plant composition from forbs (defined in this study as all non-grasses) to faster-growing grasses. This generally reduces plant species diversity (Smith *et al*., 2003; Culman *et al.*, 2010), although there are exceptions (Jarchow and Liebman, 2013).

This Chapter evaluated plant biodiversity and biomass yield of non-manipulated grassland fields, receiving different levels of N input. I asked whether plant biodiversity, biomass yield and soil characteristics differ between non-fertilised fields (receiving no inorganic fertiliser) and fertilised fields (receiving higher N input through inorganic fertiliser) on marginal land. It was expected that non-fertilised fields would have higher plant diversity and lower biomass yield, associated with lower N input.

The study area was marginal ('upland') farms in the north of England, UK (a map is shown in Chapter 2, Figure 1). The farms were representative of the types of grasslands that are being abandoned (Secretariat of the Convention on Biological Diversity, 2014). All fields were classified as 'permanent' grassland (> 5 years old), grazed and harvested annually as per the farmer's usual management. Fields varied in management and soils, and were non-manipulated, and thus the result pertain specifically to the fields and farms under consideration (further work will be needed to identify the extent to which the results can be generalized further). Nonetheless, they do address the general issue of whether biodiverse grassland fields can be compatible with bioenergy production.

3.2 MATERIALS AND METHODS

3.2.1 Field sites and management

Details of how sites were selected, and how they were managed, and their location are given in the Methods Chapter (Chapter 2, Section 2.1 and 2.2). In summary, there were 5 fertilised grassland fields (receiving inorganic fertiliser) and 5 non-fertilised grassland fields (no inorganic fertiliser). Both field types received farmyard manure. All fields were grazed by sheep and were cut in summer for forage production. Fertilised fields were cut for silage, generally earlier in the summer; and non-fertilised fields were cut for hay, generally later in the summer.

3.2.2 Quadrat sampling and analysis

Each field's vegetation and soil was sampled in 2011 using 4 randomly positioned quadrats (each quadrat 0.28 m^2) per field. To determine a quadrat position within a field, first the width and length of the field was measured in number of steps, starting from the corner of the field which was to the left of the gate (the start point). A step was always the length of one edge of the quadrat. I returned to the start point then determined a set of co-ordinates using the random number function of a calculator. These corresponded to (i) the number of steps forwards, and (ii) number of steps to the right from the start point. After walking to this position in the field, random numbers were again generated on a calculator for co-ordinates within a 10 x 10 m grid (measured out using the edge of the quadrat). The quadrat was then placed at this position within the grid, and sampling then occurred. I returned to the start point (field corner) and repeated the random number generation for positioning each new quadrat.

Quadrat sampling occurred as soon as possible before the farmer's harvest date. This provided a realistic representation of the farmer's harvested biomass (Table 1; and the farmer's desired harvest dates are shown in Table 1 in the Methods Chapter).

Table 1. Sampling dates for 2011.

Farm	Fertilised	Non-fertilised
a	05/07/11	05/07/11
b	08/06/11	19/07/11
c	08/06/11	11/07/11
d	28/06/11	19/07/11
e	15/06/11	04/07/11

The following biodiversity, biomass and soil variables were measured in each quadrat: number of plant species (species richness); number of plant indicator species (associated with semi-improved grassland, and upland hay meadow); biomass yield; Ellenburg N (a measure of plant species association with soil fertility), Ellenburg F (species' association with soil moisture) and Ellenburg R (species' association with soil pH) (Hill *et al.*, 2004); soil pH; soil organic carbon; and soil moisture. Quadrat samples were analysed and stored individually.

To measure biomass yield, vegetation was cut at approximately 5 cm stubble height (Hensgen *et al.*, 2012) using grass shears, and air-dried at 30^{o}C until constant weight. Biomass analyses specific to AD (that is: chopping, oven-drying and milling vegetation in order to measure volatile solids content and carbon:nitrogen ratio) are described in Chapter 4, Section 4.2.2. Two soil samples were taken per quadrat, to 14 cm, or the depth of the soil when less: they were bulked and well-mixed to provide one sample of soil for each quadrat. Roots and shoots were removed. pH was measured on fresh soil (stored for one day at 4^{o}C) at room temperature. The soil was shaken vigorously with an equal volume of 0.01 M $CaCl_2$, allowed to settle for 10 minutes, then the pH read after the pH probe had been in the supernatant for 1 minute (pH 210, Hanna Instruments, UK). Fresh soil was dried at 70^{o}C until constant weight for moisture measurement. Soil organic carbon was measured on oven-dried soil by loss on ignition (Tipping *et al*, 2003) at 450^{o}C for 6 hours. Percentage loss on ignition was multiplied by 0.58 (Rodegheiro *et al*, 2009) to convert it to soil organic carbon.

In order to remove day of sampling as a variable, because fertilised fields were sampled before non-fertilised due to being cut earlier by the farmer, the fertilised and non-fertilised quadrat sites on a farm were re-surveyed in 1 day in 2012, as soon as possible before the fertilised harvest (Table 2). Therefore non-fertilised vegetation was surveyed earlier in 2012 than in 2011, and hence it was shorter, which may have facilitated the identification of the smaller plants. Species richness, number of indicator species and Ellenburg N values were recorded in 2012; and % area cover by each plant species was added. Biomass yield was also measured but because non-fertilised was sampled early, only fertilised yield is comparable between the years.

Table 2. Sampling dates for 2012.

Farm	Fertilised	Non-fertilised
a	26/06/2012	26/06/2012
b	12/06/2012	12/06/2012
c	11/06/2012	11/06/2012
d	09/07/2012	09/07/2012
e	15/06/2012	15/06/2012

When anaerobic digestion (AD) was later performed (Chapter 4), each AD bottle contained vegetation from one quadrat, thus strengthening interpretation of the effect of biodiversity and yield on AD.

3.2.3 Estimation of biomass eaten by grazing sheep

Biomass harvested in Section 3.2.2 had been grazed by sheep earlier in the year, therefore in order to assess total annual biomass yield, the amount eaten by grazing sheep was estimated (Appendix Table 3E shows the number of sheep and lambs grazing per ha in each field). Amount of grass grazed was estimated using standardised data on daily dietary dry matter (DM) requirements for ewes and lambs. Ewes' DM requirements vary throughout the year depending on their stage of productivity (pre-pregnancy, pregnancy, lactation etc). It was assumed that the ewes in this research were 'hill' ewes, weighing approximately 60 kg which is 10 kg lower than lowland ewes (Department of Agriculture, Environment and Rural Affairs, 2012). The lower weight of hill ewes is due to (i) the harsher environmental conditions in the uplands and hills, and (ii) the hardy breeds of ewe which are farmed here, and are most adapted to these conditions. The amount of vegetation DM eaten/day by a 60 kg ewe was assumed to be 1.5% of bodyweight (0.9 kg) when the ewe is dry to mid-pregnancy (approximately mid-July to Jan) (Agriculture & Horticulture Development Board, 2015a). In late pregnancy (Feb to mid-April) daily DM dietary requirement was assumed to be 2% of bodyweight (1.2 kg); and from early to late lactation (mid-April to mid-July) it was 2.75% of bodyweight (1.65 kg) (Agriculture & Horticulture Development Board, 2015a). At weaning, in approximately mid-June, lamb weight may be 18 to 21 kg (Laws and Genever, 2014) therefore the lower estimate of 18 kg was used in this research to reflect the lower weight of their ewe mothers. From mid-June to mid-Dec, a lamb was assumed to eat 4% of bodyweight (0.72 kg) DM per day (Laws and Genever, 2014). From mid-Dec to mid-April, sheep are fed hay and it was assumed that they received all their DM needs from hay, and not from grazing. If any older lambs were present in winter (because they were not being sold in Oct), they were assumed to meet their DM requirements from hay. Using daily dietary DM requirements, number of sheep per ha, length of time on the field, and the particular stage of productivity of the ewes, the annual amount of grass grazed was calculated for each field. The dry matter content of grazed vegetation was adjusted from 100% to 90% to reflect the DM content of air-dried biomass which I measured at harvest. Then the grazed yield was added to the field's harvested biomass to give an estimation of total annual yield.

3.2.4 Statistics used

Statistical analysis was used to examine differences between fertilised fields (n = 5 fields, with 4 quadrats each) and non-fertilised fields (n = 5 fields, with 4 quadrats each). The quadrats were nested within field (allowing the use of individual quadrat data and avoiding pseudoreplication) in the context of a nested ANOVA (IBM SPSS Statistics for Windows, Version 24.0, 2016). The quadrat data provided replication within the random factor 'field', which was nested within the fixed factor 'fertiliser'. Nested ANOVA was used for testing differences between fertilised and non-fertilised fields in species richness, semi-improved grassland indicator species and upland hay meadow indicators (which were all log- or square-root transformed to allow use of a parametric test). Percent forb cover, Ellenburg N and R, and soil pH were also log- or square-root transformed to give homogeneous variances where necessary. If quadrat data variances remained non-homogeneous after transformation, 1-way ANOVA was performed using the average quadrat value per field (this value was also log- or square-root transformed if variances were not homogeneous). Variables tested by 1-way ANOVA were upland hay meadow indicator species, % grass cover, % legume cover, biomass yield, Ellenburg F, soil moisture and soil organic carbon. Non-parametric Mann-Witney *U* tests were used to assess differences in legume cover because variances were not homogeneous. In order to control for factors which may influence biomass yield (species richness, soil pH, altitude, number of semi-improved grassland indicator species and % grass cover), an analysis of covariance (ANCOVA) was performed using the average quadrat value per field. Count data (species richness and number of indicator species) were first transformed to allow use of a parametric test. Paired t-test was used to compare biomass yield between 2011 and 2012, after homogeneity of variances was checked. Tests of relationship were carried out by correlation or regression using the average quadrat value per field as each data point. Due to the nested structure of quadrats within each field, 'mean fertilised' or 'mean non-fertilised' data were calculated in two steps: first, the average quadrat value per field was calculated; second, an average was taken of these values.

3.3 RESULTS

3.3.1 Plant species: biodiversity and nutrient tolerance

As anticipated, plant biodiversity differed between the fertilised and non-fertilised fields (Table 3; details are shown in Tables 3A-3D in the Appendix). Specifically, mean plant species richness per quadrat (i.e. per 0.28 m^2) in non-fertilised fields was 48% higher than in fertilised fields in 2011 (Fig. 1a), which was of borderline significance. When re-measured in 2012, species richness was 63% higher in non-fertilised fields, which was significant (Table 3). Upland hay meadow is a rare grassland habitat and semi-improved grassland, which is similar to the non-fertilised fields studied here, may have been derived from and be restorable to such BAP habitat (Natural England, 2010). Thus, fields containing these indicator species are of interest to conservationists: and the non-fertilised fields contained significantly more species of both indicators (Table 3; Fig. 1b). Indicator species are listed in bold in the Appendix Tables 3A and 3B.

The variables which are associated with higher soil nutrient input were also significantly higher in fertilised fields: (i) % grass cover; and (ii) Ellenburg N, which is a species' association with soil fertility (Table 3). The higher % grass cover in fertilised fields was due to higher cover of perennial ryegrass (*Lolium perenne)* (Table 4; Fig. 1c), which has high dry matter yield and high digestibility in ruminants (and in anaerobic digestion, the bioenergy process which is examined in Chapter 4). Perennial ryegrass was also the dominant species in non-fertilised fields, but at approximately half the cover of fertilised (20% versus 38% respectively). Non-fertilised forb (non-grass) cover was nearly twice that in fertilised fields (52% and 27% respectively). Legume cover was not significantly different between fertilised and non-fertilised fields (Table 3), although the non-significant trend (19% higher in fertilised fields) may contribute to a slightly higher N input into fertilised fields.

Table 3. Comparison of fertilised and non-fertilised fields in biodiversity, biomass yield and soil.

	Variable	Year	Mean (SE)		% Difference		Statistics:	$F_{1,8}$	P
			Fert.	Non-fert.	Fert.	Non-fert.	ANOVA		
Biodiv-ersity	Species richness	2011	9.6 (1.6)	14.1 (1.0)		+ 48	nested	5.182	0.052
		2012	10 (1.6)	16.2 (0.6)		+ 63	nested	9.056	0.017*
	Semi-imp. indicators$	2011	1.1 (0.5)	3.7 (0.8)		+ 236	1-way	8.48	0.02*
		2012	1.4 (0.5)	4.1 (0.5)		+ 204	nested	13.873	0.006*
	UHM indicators$	2011	0.05 (0.05)	0.8 (0.3)		+ 1500	1-way	7.171	0.028*
		2012	0.05 (0.05)	1.3 (0.29)		+ 2500	1-way	18.12	0.003*
	Grass cover (%)	2012	73 (10)	48 (4.4)	+ 52		1-way	5.796	0.043*
	Forb cover (%)	2012	27 (10)	52 (4.4)		+ 92	nested	4.09	0.078
	Legume cover (%)	2012	11.5 (5.1)	9.7 (1.9)	+ 19		MWU†	N/A	0.840
Biomass ($t\ ha^{-1}$)	Harvested yield	2011	4.2 (0.7)	4.3 (0.2)		+ 4	1-way	0.152	0.706
		2012	3.5 (0.4)β						
	Grazed yield	2011	2.4 (0.4)	1.9 (0.5)	+ 21		1-way	0.459	0.517
	Total annual yield	2011	6.5 (0.4)	6.3 (0.6)	+ 4		1-way	0.112	0.746
Soil	Ellenburg N	2011	5.2 (0.2)	4.7 (0.1)	+ 12		nested	8.295	0.02*
		2012	5.2 (0.1)	4.8 (0.1)	+ 10		nested	13.86	0.006*
	Ellenburg R	2011	5.9 (0.1)	5.7 (0.1)	+ 3		nested	1.986	0.196
	Ellenburg F	2011	5.3 (0.1)	5.3 (0.1)		+ 0.4	1-way	0.097	0.764
	Soil pH	2011	5.2 (0.2)	4.7 (0.2)	+ 11		nested	2.655	0.142
	Soil moisture (%)	2011	34 (1.9)	37 (1.6)		+ 9	1-way	2.751	0.260
	Soil organic C (%)	2011	9.2 (0.6)	11 (3.8)		+ 21	1-way	0.255	0.627

SE is standard error (n = 5 fields). *means a significant difference between fertilised and non-fertilised fie

$Indicators is no. of indicator species: semi-imp is semi-improved grassland; UHM is upland hay meadow.

†MWU is Mann-Witney-U

βIn 2012, non-fertilised biomass yield was measured earlier than in 2011, therefore is not comparable.

Table 4. Mean percentage cover of plant species in fertilised and non-fertilised fields. (Only species covering >3% are listed.)

Species	Mean cover (%)	
	Fertilised	Non-fertilised
Perennial ryegrass (*Lolium perenne*)	38	20
Yorkshire fog (*Holcus lanatus*)	17	10
White clover (*Trifolium repens*)	13	6
Rough meadow grass (*Poa trivialis*)	9	
Creeping buttercup (*Ranunculus repens*)	5	7
Soft brome (*Bromus hordeaceous*)	4	
Meadow fox-tail (*Alopecurus pratensis L*.)	4	
Crested dog's tail (*Cynosurus cristatus*)		8
Sweet vernal grass (*Anthoxanthum odoratum*)		7
Yellow rattle *(Rhinanthus minor)*		7
Meadow buttercup (*Ranunculus acris*)		7
Marsh-marigold (*Caltha palustris*)		6
Common sorrel *(Rumex acetosa)*		6
Daisy (*Bellis perennis*)		4
Red clover *(Trifolium pratense)*		4
Red fescue (*Festuca rubra L*.)		4

Soil pH, organic carbon and moisture were not significantly different between fertilised and non-fertilised fields (Table 3), although pH was 11% higher in fertilised fields due to more liming, and organic carbon was 21% higher in non-fertilised fields, but this was because one non-fertilised field was on peat soil which was next to moorland.

Higher species richness was correlated with lower % grass cover (Spearman rank order, using average per quadrat for each field: n = 10 fields; r = -0.869, P 0.001) (Fig. 2). Thus, compared to fertilised fields, non-fertilised fields had higher plant species richness, greater number of conservation indicator species (of semi-improved grassland and upland hay meadow habitat), lower grass cover and lower Ellenburg N.

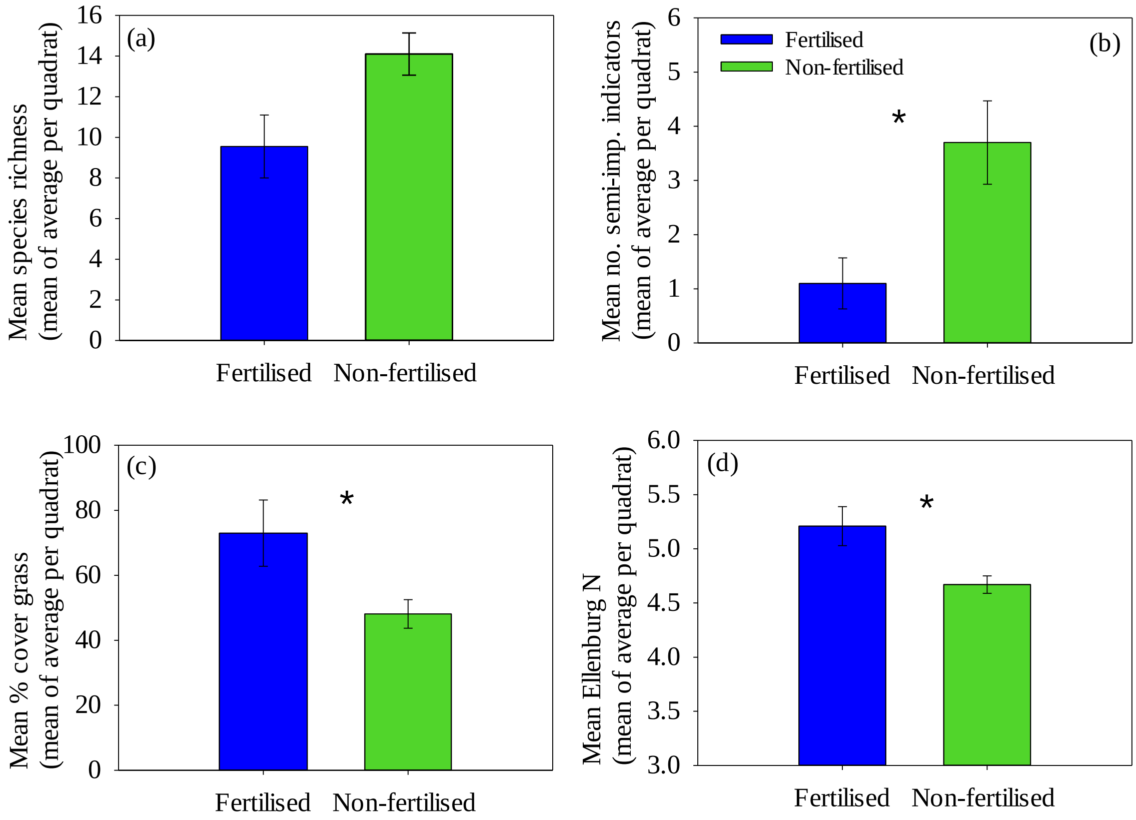

Fig. 1. Mean plant characteristics in fertilised and non-fertilised fields in 2011. (a) species richness (which was significantly different when re-measured in 2012), (b) number of semi-improved grassland indicator species, (c) % grass cover, (d) Ellenburg N. Error bars are SE (n = 5 fields). * indicates significant difference.

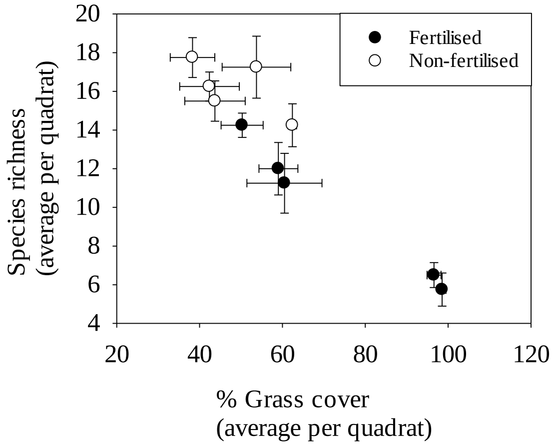

Figure 2. Relationship between species richness and % grass cover in all the fertilised and non-fertilised fields (Spearman rank order r = -0.869, *P* 0.001). Each data point is average per quadrat in a field.

3.3.2 Harvested biomass yield

Mean air-dried biomass yield (containing 90% DM) was very similar between fertilised and non-fertilised fields (4.2 and 4.3 t ha^{-1} respectively) (Table 3 and Fig. 3a; individual field yields are shown in Table 3F, Appendix). This yield is the gross yield (in later Chapters, I assume that it is reduced due to losses during processing of fresh vegetation into hay (containing 96% DM) or silage (containing 25% DM)). My estimates of biomass yield were validated by Farmer ‘d’ who reported a silage yield on their fertilised field in 2015 of 9.0 t silage ha^{-1}. This is similar to my estimate three years previously of 10.3 t silage ha^{-1} for that field: biomass yield is known to vary between years (for example, the yield of dried harvested biomass in fertilised fields was 4.2 t ha^{-1} in 2011 and 3.5 t ha^{-1} in 2012 (Table 3)). Therefore the estimates of yield in this research appear reasonable.

Yield across all fields in 2011 showed a significant positive regression with harvest date when the outlying high-yielding field was excluded (shown in Fig. 3b) (linear regression equation, n = 9 fields, using mean $field^{-1}$: yield = 0.04 x day of harvest + 3.00. r^2 = 0.810, F = 29.812, P = 0.001).

Yield remained not significantly different between fertilised and non-fertilised even after species richness, number of semi-improved grassland indicator species, altitude, soil pH and % grass cover were controlled for using ANCOVA. Yield was not correlated with rates of inorganic N application (reported by the farmers), species richness, number of semi-improved grassland indicator species, %grass cover, altitude or soil pH (Fig. 3A in Appendix). Field aspect was not clearly linked to biomass yield, although the highest yielding field faced south-east, and would therefore have received more of the available sunlight.

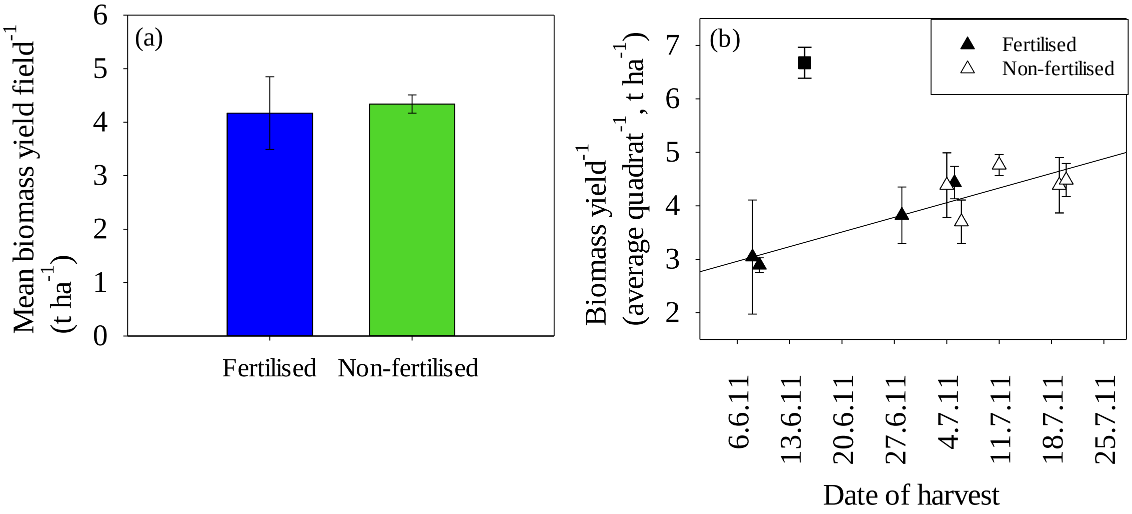

Fig. 3. (a) Mean biomass yield from fertilised and non-fertilised fields (extrapolated to t ha^{-1}). Due to the nested structure of quadrat within field, variation due to quadrats is lost. Error bars are SE (n = 5 fields). (b) Effect of sampling date on biomass yield of each field. Each data point is average per quadrat in a field. Error bars are SE (n = 4 quadrats). For linear regression equation see text above.

Thus, fertilised and non-fertilised fields had very similar mean harvested biomass yield, and date of harvest had a strong effect on yield of all fields.

Harvested dry biomass yield was measured again in 2012: fertilised yield was lower than, but not significantly different between, 2011 and 2012 (Table 3. Paired t-test P 0.439). Non-fertilised fields were cut earlier in 2012 than 2011, so their 2012 harvested yield was not included in Table 3 (it was not comparable between years).

3.3.3 Estimation of grazed biomass yield, to give total annual yield

Annual dry biomass yield per field was estimated by adding estimated grazed yield to harvested yield (Appendix, Table 3F). Grazed yield varied by a similar amount within each field type (1.3-3.3 t ha^{-1} in fertilised fields; 0.8-3.3 t ha^{-1} in non-fertilised fields) but mean grazed yield was 21% higher in the fertilised fields (2.4 vs 1.9 t ha^{-1} in non-fertilised fields: Table 3). Mean total annual yield was similar between fertilised and non-fertilised fields, with fertilised ranging from 5.9-8.0 t ha^{-1}, and non-fertilised ranging from 5.1-8.1 t ha^{-1}. However, in fertilised fields, grazed yield had a significant negative effect on harvested yield (Fig. 4) (linear regression: harvested yield = -1.777 x grazed yield + 8.337. P = 0.023. r^2 = 0.860). Non-fertilised fields showed no relationship between grazed and harvested yield.

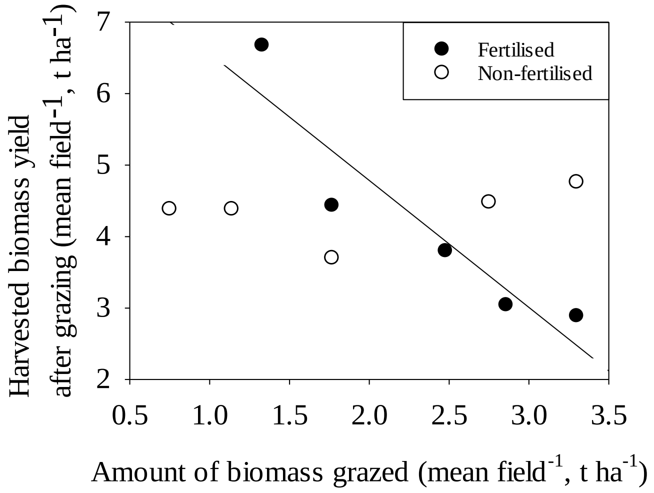

Figure 4. Amount of air-dried biomass grazed and harvested. The regression line applies only to the fertilized fields (see text above for regression equation).

3.3.4 Variability within field type

Variability was higher within a field type than between field types, for many of the variables (Table 3 in this Chapter shows differences between fertilised and non-fertilised; and Table 3G in the Appendix shows variables that were significantly different among fertilised fields; or among non-fertilised fields). Fertilised fields were more variable than non-fertilised. For example, harvested biomass yield, species richness (in 2011), soil pH, soil moisture and Ellenburg R were significantly different among fertilised fields, but not between field types. Ellenburg N, % grass cover and semi-improved indicator species were also significantly different among fertilised fields, but also between field types. Non-fertilised fields were less variable than fertilised: species richness, soil pH and soil organic carbon were significantly different among non-fertilised fields, but not between field types. The number of indicator species was significantly different among non-fertilised fields, and between field types.

3.4 DISCUSSION

Biodiversity, biomass yield and soil characteristics were studied in ten marginal grass fields, to evaluate whether they could potentially be used for bioenergy production; in order to avoid land abandonment and degradation of grassland biodiversity. The overall results show that biodiversity is higher in non-fertilised fields, yet these same biodiverse fields generate as much biomass as fertilised ones. This suggests that biomass production and biodiversity conservation are compatible in this system.

The non-fertilised fields were grazed, harvested and received no inorganic fertiliser, yet they achieved a similar harvested dry biomass yield to the fertilised fields (which were also grazed and harvested, but received higher N input through inorganic fertiliser). Annual biomass yield (including that estimated to be grazed by sheep) was also similar between fertilised and non-fertilised fields. The non-fertilised fields contained 48-63% higher plant species richness; double the forb cover; and they contained significantly more indicator species, characteristic of semi-improved grassland and EU-protected upland hay meadow habitat. This indicates that farmers may be able to use less fertiliser without compromising yield; and achieve higher biodiversity and improve other ecosystem services if they used non-fertilised fields for bioenergy production. (Bioenergy production from fertilised and non-fertilised vegetation is reported in Chapter 4). Livestock grazing may be helpful to maintain plant species richness (Smith and Rushton, 1994) in a bioenergy production system, but this is not always necessary (Schellberg *et al.*, 1999). There may also be greenhouse gas (Chapter 5) and economic benefits (Chapter 6) from non-fertilised grassland, arising from the lack of applying inorganic fertilisers.

3.4.1 Plant biodiversity and fertiliser addition

The rate of (farmer-reported) inorganic fertiliser application in the fertilised fields (25-73 kg N ha^{-1}) was lower than the national average for marginal grasslands (75 kg N ha^{-1} a^{-1}; DEFRA, 2015b), but nonetheless its presence was linked with lower biodiversity. This observation agrees with that of Smith *et al.* (2008), who showed that 25 kg inorganic N fertiliser ha^{-1} a^{-1} and 12 t ha^{-1} a^{-1} farmyard manure led to a reduction in species richness and increase in grass abundance compared to non-fertilised areas, in similar marginal grassland (fields in the present study received approximately 10-12 t farmyard manure ha^{-1}

a^{-1}). Smith *et al.* (2008) and Hensgen *et al*. (2016) also found a higher Ellenburg N (that is, greater species association with higher nutrient levels) in fertilised fields, as was found here. Thus, the lower nutrient input of non-fertilised fields was likely to be a factor in their higher biodiversity, which included greater number of conservation species. Harvest date can also affect biodiversity, with later harvest date allowing later-flowering forbs to set seed and perpetuate (Jefferson, 2005). Typically, fertilised fields are harvested earlier (for silage, which is made from younger plants) and non-fertilised fields are cut later (for hay, which is made from more mature vegetation). Hay is often the preferred forage for sheep on the studied farms (*pers. comm*. Andrew Hattan and Stephen Ramsden). If non-fertilised fields were used for bioenergy production under current management (albeit with fewer grazing sheep because their winter forage would be used instead for bioenergy production), this would promote biodiversity conservation. Grazing is discussed in Section 3.4.3.

3.4.2 Biomass yield

Contrary to expectation, we found no difference in mean harvested yield, and mean annual dry biomass yield, between fertilised and non-fertilised fields. This was unexpected because, although increasing plant species diversity can lead to higher productivity (Tilman *et al.*, 2006; Bullock *et al.*, 2007), higher N input often increases yield (e.g. Schellberg *et al*., 1999). However similar yields between fertilised and non-fertilised grasslands have also been demonstrated in experimental plots in Germany (Weigelt *et al.*, 2009); and between switchgrass (*Panicum virgatum*) and species-rich grasslands in USA (Werling *et al.*, 2014; Dickson and Gross, 2015). Fraser (2015) also demonstrated that grassland productivity can increase with species richness, peaking at around 30 species m^{-2} before falling again. Therefore similar yield between fertilised and non-fertilised fields has been seen elsewhere. It is also possible that the thin, ‘hungry’ soils (*pers. comm*. Andrew Hattan) in these marginal areas require higher levels of nutrient input to optimally stimulate plant productivity. In the present study, later harvest date led to higher harvested yield, when the high-yielding outlier was excluded (Fig. 3b). Therefore the lack of difference in harvested yield is also partly due to farmer behaviour, given that farmers adjust cut dates to when the field is ‘ready’ for its particular forage to be made.

By definition, marginal grasslands are relatively low-yielding, and the harvested DM yields were in the same range as similar grasslands elsewhere: 4.4 - 4.9 t ha^{-1} for the first cut on single-cut lowland and mountain hay meadow in Germany (Hensgen *et al.*, 2014); 4.1 – 4.4 t DM ha^{-1} on single-cut mountain hay meadows in the Italian Alps (Scotton *et al.*, 2014); and 3.7 – 5.0 t DM ha^{-1} for the first cuts on unfertilised and un-grazed roadsides in Belgium (Van Meerbeek *et al.*, 2015). The outlying, high-yielding fertilised field (e) (Fig. 3b) had been re-seeded with high-yielding grasses 8 years previously (giving 99% grass cover, see Appendix Fig. 3A); anecdotal evidence suggests re-seeding is not frequently carried out in marginal areas (*pers. comm.* Andrew Hattan). The field was also south-east facing (Appendix Fig. 3A), received the highest amount of organic and inorganic N (124 kg ha^{-1} a^{-1}) and was the least grazed fertilised field (Appendix Table 3F). Presumably a combination of these factors contributed to its higher harvested and annual yield. In the remaining fields, the cool climate and poor soils may also be more important constraints on biomass yield than nutrient input or species richness. Within the group of fertilised fields, there was large variation in harvested yield, 86% of which was due to sheep grazing, which is discussed below.

Total annual biomass yield (harvested plus grazed biomass) was similar between fertilised and non-fertilised fields but variability was quite large, ranging from 5 to 8 t ha^{-1} in both fertilised and non-fertilised fields. Carlsson *et al.* (2017) also reported lower variation between fertilised and non-fertilised biomass than between sites (and years). Roadsides in Belgium produced 6.9-8.1 t DM ha^{-1} annually (Van Meerbeek *et al.*, 2015) and mountain hay meadows in the Italian Alps produced 4.1-10.5 t DM ha^{-1} annually (Scotton *et al.*, 2014). Therefore, my results overlap with previous research. The amount of biomass eaten by grazing sheep was an estimate, based on the DM feeding requirements of ewes and lambs at different points of their life cycle. Fertilised fields had 21% more biomass removed from them by grazing than did non-fertilised fields; and a higher grazed yield in fertilised fields also led to lower harvested yield. Perhaps the longer re-growth period after fields were 'shut up' from grazing sheep contributed to the lack of relationship in non-fertilised fields. However total annual biomass was very similar between fertilised and non-fertilised fields, suggesting that non-fertilised fields were had comparable productivity to fertilised fields, and inorganic fertiliser may not be necessary to boost yield if species richness is sufficiently high. Not using inorganic fertiliser could save the farmer money (Chapter 6) and reduce greenhouse gas emissions (Chapter 5).

If sheep continue to be produced on the land (with winter feed produced elsewhere – discussed in Chapters 5 and 6), the amount of biomass available for bioenergy production would be the harvested yield measured in the fields (mean fertilised 4.2, and non-fertilised 4.3 t DM ha^{-1}); whereas if sheep were to be removed from the land, either because of threatened abandonment or to maximise bioenergy feedstock production, the amount of biomass available would be the total annual yield (mean fertilised 6.6, and non-fertilised 6.4 t DM ha^{-1}). Low soil pH in the fertilised fields (mean 5.2) may have contributed to a low fertilised yield (i.e. similar to non-fertilised yield) if it led to leaching of inorganic fertiliser, and less N being taken up by the vegetation than expected. However the species richness (in 2012), number of conservation indicator species, Ellenburg N and percentage grass cover differed significantly between fertilised and non-fertilised fields, suggesting that there was an effect of higher N input in the fertilised fields.

3.4.3 Effects of modifying grazing

The financial implications of using the harvested volume of biomass in bioenergy production are studied in Chapter 6: if sheep were grazed less often on the land, or not at all, the higher harvestable yield could benefit the financial returns of bioenergy. However, grazing has complex effects on biodiversity, nutrient cycling and soil organic carbon, and hence its cessation may not be desirable. In terms of biodiversity, reduced grazing pressure on marginal moorland between 1998 and 2007 allowed an increase in larger, faster-growing plants (Martin *et al*., 2013), which can reduce the abundance of smaller diverse plants by blocking out light and reducing seedling germination (Corton *et al.*, 2013). Light grazing can be beneficial for plant biodiversity and reduce soil erosion compared to heavier grazing, but a complete lack of grazing can reduce biomass yield (Martin *et al*., 2013) and could reduce biodiversity (Boatman *et al.*, 2010). The exact level of grazing necessary to maintain biodiversity and yield varies, however: light grazing (to 5-6 cm sward height) can maintain plant biodiversity in UK upland hay meadows, but the optimal timing of grazing depends on spring temperature (Pinches *et al.*, 2013). Yang *et al*. (2016) found that grassland species richness in China was highest under low grazing intensity, and biomass yield was highest with no grazing. UK moorland biodiversity is most optimally maintained under moderate grazing (Martin *et al*., 2013). In Sweden, cutting grassland maintained its species richness better than grazing (Hansson and Fogelfors, 2000), and across Europe, the biodiversity of NATURA 2000 grasslands is being maintained by cutting because grazing has reduced (Donnison and Fraser, 2016). Cutting increased species richness in abandoned European grassland (Hajkova *et al.*, 2009). Therefore harvesting the biomass would help maintain biodiversity (by removing nutrients, shade and litter which smother smaller, slower-growing plants) but some form of grazing alternative to sheep may also be needed.

The farmers included in the present study produce cattle in addition to sheep, but cattle graze the studied fields for only a short time each year, therefore they were not included in this thesis. Farmers could increase the length of time non-fertilised fields are grazed by cattle, which may help maintain species richness if sheep were absent. Cattle productivity can be maintained for short periods on such higher diversity grass (Fraser *et al*., 2014). Cattle on the studied farms were fed silage (produced from fertilised fields) in winter, so the diversion of hay biomass to bioenergy production may have little effect on cattle winter forage supply. With regards to soil carbon and nutrient cycling, lack of grazing can increase soil organic carbon, but reduce assimilation of atmospheric CO_2 (Martin *et al.*, 2013). The corollary has been shown, whereby removal of vegetation by harvesting and grazing can reduce grassland soil carbon (Soussana *et al*., 2007). Moderate grazing can increase soil microbial biomass and its associated nutrient cycling (Martin *et al.*, 2013). Therefore the effects of changing grazing may need to be monitored to ensure no detrimental effect on the habitat.

3.5 CONCLUSION

Non-fertilised (hay) marginal grass fields had the biodiversity benefits of higher plant species richness, significantly higher number of conservation indicator species, and significantly lower grass cover than fertilised (silage) fields; yet they also had similar mean biomass yield to fertilised fields. Importantly, several other ecosystem services may additionally be improved in biodiverse non-fertilised grassland, including improved bee health and pollination of nearby crops (Kremen *et al,* 2004; Goulson *et al*., 2015; Pywell *et al*., 2015), number of insect predators of crop pests (Pywell *et al*,. 2015), water management, recreation, aesthetic services, and plant and bird biodiversity (Newton, 2004; Pinches *et al.*, 2013). Therefore, as a feedstock for anaerobic digestion (the bioenergy system examined in this thesis), non-fertilised (hay) grass fields in temperate marginal areas have potential to provide a feedstock which is more biodiverse yet similar-yielding to the more commonly used grass silage (which is species-poor), at least for the types of farms and fields studied here. If not used for another purpose other than sheep farming, these fields could be abandoned in the future and their biodiversity and conservation species lost.

Chapter 4: Anaerobic digestion of fertilised and non-fertilised marginal grassland

ABSTRACT

Marginal grasslands could potentially provide abundant biomass for bioenergy production. Following the observation in Chapter 3 that marginal grassland not receiving inorganic fertiliser (hay fields) had higher biodiversity but similar biomass yield to grassland receiving inorganic fertiliser (silage fields), here, I examine if there are any differences in the efficiency of using biomass from non-fertilised and fertilised land for biomethane production (by anaerobic digestion, AD). For these experiments the biomass samples were harvested as part of the normal farming cycle. Fresh grass was cut, dried and chopped before being used in AD. Biomethane production (by batch AD) of 5 fertilised and 5 non-fertilised agricultural marginal grasslands were studied (total 40 samples). There were no significant differences between biomethane produced per unit weight of vegetation and per hectare between field types. This suggests that non-fertilised grassland has potential as an AD feedstock within marginal regions, where land management and bioenergy feedstock choices are limited. Lower input farming methods not only favour higher biodiversity, but are also likely to have a lower carbon footprint and management cost than high input land (studied in Chapters 5 and 6), suggesting that bioenergy production might prove an alternative use for marginal grassland areas.

4.1 INTRODUCTION

In Chapter 3, I examined biomass yield and biodiversity in a range of fertilised (silage) and non-fertilised (hay) grassland fields in farms in the poorly productive area of the Yorkshire Dales. The findings of that work were that the unfertilised fields maintained greater biodiversity and that there was no significant differences in biomass yield between the two field types. This suggests that using biomass from unfertilised fields might provide a way of providing sustainable bioenergy feedstock while maintaining high biodiversity in marginal areas. Across Europe there are large areas of extensively-managed, species-rich grasslands which could be used as a source of biomass for bioenergy (Smyth *et al.* 2009, Blokhina *et al.* 2011) by anaerobic digestion (AD). The marginal land studied in this thesis was Severely Disadvantaged Area (SDA) land which is of poor agricultural quality (DEFRA 2011). SDA land is the most disadvantaged land out of the larger classification, Less Favoured Area (LFA) (Harvey and Scott, 2016). Using the biodiverse, perennial grasslands already present in marginal areas for bioenergy promotes wildlife and protected species conservation (Fargione *et al.,* 2009). Other bioenergy grass species such as *Miscanthus* can be grown on marginal land but its dense canopy suppresses other plant growth, reducing biodiversity (Dauber *et al.*, 2015; Donnison and Fraser, 2016). Farmers are already familiar with grass cultivation, harvest and storage as compared to *Miscanthus* husbandry (Smyth *et al*., 2009). In addition, in contrast to *Miscanthus*, permanent grassland does not require ploughing or land use change, which increase CO_2 losses from soils (Searchinger *et al.* 2008). The supply of other high-yielding bioenergy feedstocks, such as food waste, is limited in these rural areas compared to cities because, for example, only 4% of the population of England lives in LFAs (DEFRA, 2011a). On the other hand, grass is an abundant feedstock covering on average 86 ha per LFA farm in England (FBS 2016). Grass can be combusted (Prochnow *et al*. 2009a, Stoof *et al.*, 2015), fermented to bioethanol (Adler *et al*., 2009) or pyrolsed to char and oil (Corton *et al.*, 2016). Here, anaerobic digestion (AD) was chosen as the bioenergy method to evaluate the biogas production of harvested fertilised and non-fertilised grass biomass (from Chapter 3), primarily because grass is a proven feedstock in AD (Prochnow et al., 2009b), and grass silage is a popular feedstock in on-farm anaerobic digesters in Germany and Austria (Rosch *et al.* 2009).

Grass feedstocks for AD tend to comprise intensively-managed, inorganically fertilised, perennial ryegrass (*Lolium perenne*) swards which are highly digestible in ruminants and can have high biomethane yield (Asam *et al.*, 2011; Nizami *et al*., 2012). Such grassland has low biodiversity and is harvested at an immature (pre-flowering) stage to be preserved as silage (Nizami *et al.*, 2009). In contrast, traditionally-managed, low-input grassland (which does not receive inorganic fertiliser and is harvested later, after flowering, for hay) has greater plant biodiversity but reduced ruminant digestibility (Isselstein *et al*., 2005). This is partly due to the lower leaf:stem ratio, and therefore higher structural carbohydrate content (lignin, hemicellulose and cellulose) of post-flowering vegetation (Tallowin and Jefferson, 1999; Bruinenberg *et al.*, 2002). There has been comparatively less research on AD of non-fertilised, species-rich grasslands, the higher forb and lower grass content of which could affect the specific methane production (per unit weight of biomass) (Prochnow *et al.*, 2009b).

Grasses can be digested freshly cut or after ensiling, which offers a method of preservation and year-round supply of feedstock (Nizami *et al.* 2009). Ensiling has been shown to improve biogas production compared to fresh grass (Nizami *et al.* 2009), but this is not always the case (Kreuger *et al.*, 2011). However, grass can cause problems inside anaerobic digesters by aggregating and floating on top of the liquid if not sufficiently agitated (Thamsiriroj *et al*. 2010). Co-digestion of grass with manure increases stability of conditions within the anaerobic digester such as pH, volatile fatty acid and hydrogen concentration (Nizami *et al.* 2009, Ahring *et al.* 1995). Dried grass (such as tall fescue (*Festuca arundinaceae*) and timothy (*Phleum pratense*)) can yield 253-394 m_N^3 CH_4 t^{-1} of grass volatile solids (VS, a measure of organic content) (m_N^3 is m^3 at standard temperature and pressure) (Seppala *et al.* 2009). Dried cocksfoot (*Dactylis glomerata*) and dried perennial ryegrass (*Lolium perenne*) can give 207 and 229 m_N^3 CH_4 t^{-1} VS respectively (McEniry and O'Kiely, 2013). German semi-natural grassland silage can produce 158-268 m_N^3 CH_4 t^{-1} VS (Richter *et al.* 2009). For comparison, maize silage can produce 268-365 m_N^3 CH_4 t^{-1} VS and manure produces up to 166 m_N^3 CH_4 t^{-1} VS (Amon *et al.* 2007a). Thermochemical pre-treatments such as alkali can also increase methane yield, for example from 326 to 453 m_N^3 CH_4 t^{-1} grass silage VS (Xie *et al.* 2011). Many other factors also affect biogas yield, including: 1) grassland conditions such as type of grass, biomass yield, soil type, geographical area and climate; 2) agricultural management including amount of fertilisation, number of harvests (the first harvest yields more methane than subsequent cuts) and silage quality; 3) the AD technology itself including pre-treatment and type and efficiency of the digester (Prochnow *et al.* 2009b, Nizami *et al.* 2009).

The objective of my research was to investigate if biomethane production by AD is compatible with the higher biodiversity of the non-fertilised marginal grasslands sampled in Chapter 3, when cut at the time of the farmer's usual harvest. The question posed in this chapter is what are the effects of grassland management on biomethane production by batch AD? I anticipated that the non-fertilised biomass would have lower specific methane production, associated with the presence of less digestible plant species in these fields and a later harvest date, giving higher lignocellulose content. Lower specific methane production would thereby result in lower methane production per unit area. However, (i) the biomass yield was comparable in fertilised and non-fertilised fields (Chapter 3), (ii) the level of fertilisation (in fertilised fields) was low relative to intensive agriculture, and (iii) harvest dates overlapped slightly between fertilised and non-fertilised fields. Therefore, it was not clear if a difference in biomethane production would be observed.

4.2 MATERIALS AND METHODS

4.2.1 Biomethane assay development

Details of method development to produce a biomethane assay for anaerobic digestion are reported in the Appendix (Appendix to Chapter 4, Sections 4.1 and 4.2).

4.2.2 Preparation of biomass sampled from fertilised and non-fertilised grasslands.

Biomass samples used in AD were those which had been taken just before the farmers' harvest (in 2011; described in Chapter 3, Section 3.2.2), so it was representative of what farmers could actually achieve if management was not changed. The selection of fertilised (n = 5) and non-fertilised (n = 5) marginal grassland fields is explained in Chapter 2, Section 2.1. Harvesting, air-drying and measurement of biomass yield from each of the four quadrats per field is described in Chapter 3, Section 3.2.2. Biomass harvested from a quadrat was a single sample: after air-drying, it was chopped to 0-5 mm length using a Cucina coffee mill (Philips, Guildford, UK), and any longer lengths which remained after chopping were cut to 0-5 mm with scissors. Chopped material was then transferred to a plastic bag and thoroughly mixed by hand to ensure even distribution of all species throughout. Volatile solids content (VS) (a measure of organic content defined as percentage loss of mass from oven-dry to ashed biomass) was determined for the biomass. Subsamples of chopped material were oven-dried at 70^{o}C until constant weight, then burned in a furnace to ash at 450^{o}C for 6 hours. Further subsamples of chopped, air-dried material were oven-dried at 105^{o}C for 3 hours, then milled for carbon:nitrogen (C:N) analysis on an elemental analyser (EAS32, Costech International, S.p.A., Italy). Control grass was harvested in September 2011 from a fertilised lawn comprising 95% perennial ryegrass (*Lolium perenne*). It was dried and analysed as above. Cell wall composition (lignin, cellulose and hemicellulose) was analysed by Rachael Hallam (CNAP, Department of Biology, University of York) as per Ercolano *et al.* (2015). Lignin was measured using the acetyl bromide soluble lignin method (Ercolano *et al*., 2015); which hydrolyses lignin into S, G, H subunits and allows the absorbance of the solubilised phenols to be measured on a spectrophotometer.

4.2.3 Anaerobic digestion

Air-dried, chopped (0-5 mm), thoroughly-mixed biomass (Section 4.2.2) was used in AD batch experiments. One AD experiment was set up for each quadrat's biomass. Two grassland fertiliser levels (fertilised and non-fertilised) x 5 fields x 4 samples (quadrats) per field, equalled 40 field biomass AD experiments. All the AD experiments were carried out at the same time in one incubator. Inoculum was collected from 3 depths (0.5, 2.3 and 4.2 m) of an anaerobic digester at a sewage treatment plant (Yorkshire Water, Naburn, York, UK), then mixed and stored anaerobically at 37^{o}C for 4 days until used. It had been used in two previous experiments and had been shown to digest grass (Appendix Section 4.2). VS (a measure of organic matter content) of field vegetation was 92-95% of total solids (Table 1). VS and total solids content of inoculum were 1.4% and 2.1% of fresh matter, respectively. Inoculum pH was 8 at the start of the experiment. Three inoculum-only AD were included to assess methane production in the absence of feedstock.

To maximise methane production, all the dried field vegetation samples were pre-treated. Preserving grass by making silage (Chapter 2, Section 2.3) is a common pre-treatment and storage method used in anaerobic digestion crops, but attempts at making silage in the lab from fresh grass failed due to fungal growth. Therefore alkaline thermo-chemical pre-treatment of the dried grass was performed because it often increases methane production (Himmel *et al.*, 2007; Hendriks *et al.*, 2009). Pre-treatment comprised 10%:90% (w/v) air-dried vegetation:NaOH (0.3 M), at 100^{o}C for 2.5 hours (modified from Xie *et al.*, 2011). 0.7 g VS of vegetation was used. After cooling, samples were neutralised with 0.3 M HCl then all samples had water added, where necessary, to make them up to the same volume. The mixture was poured into a 100 ml Wheaton bottle (Supelco, Bellefonte, PA, USA) inside an anaerobic chamber containing 95% nitrogen and 5% hydrogen. Three pre-treated and 3 non-treated dried control grass samples were included, to assess the effect of pre-treatment. Control grass VS was 88% of total solids.

AD experiments were adapted from Angelidaki *et al.* (2009). 50.4 ml inoculum (containing 0.7 g VS) was added to the pre-treated vegetation inside the anaerobic chamber (1:1 feedstock:inoculum ratio of VS). All bottles were made up to the same volume with water (around 60 ml). The bottles were sealed with a butyl rubber bung and crimp seal, and agitated in a shaking incubator at 140 rpm, 37°C (Fig. 1). Headspace pressure was monitored each day using a hand-held pressure gauge (Digi Gauge, Rototherm, Port Talbot, UK) with a needle inserted through the bung. When the pressure exceeded 25 psi in any bottle, the gas was sampled in all bottles (using a 5 ml syringe and needle) then excess gas was released from the AD bottles to a headspace pressure of 2 psi; then the headspaces were sampled again to be able to calculate accumulated methane. When gas production had slowed down, it was sampled once a week in all bottles, as above. The gas samples were diluted 1/400 in nitrogen to allow methane measurement by gas chromatography (Autosystem XL, Perkin Elmer Arnel, Waltham, MA, USA). Methane production by inoculum-only bottles was subtracted from the methane production of vegetation samples. Biomethane production was expressed either per amount of feedstock (specific methane production, in m_N^3 CH_4 t^{-1} VS) or per area (area methane production, in m_N^3 CH_4 ha^{-1}). Area methane production depends on specific methane production and biomass yield.

Fig. 1. AD bottles in the shaking incubator, digesting 40 air-dried vegetation samples from 5 fertilised grasslands and 5 non-fertilised grasslands (4 quadrats per field, put through AD individually).

Note that the methane yields reported here may be lower in absolute values (but not in proportional differences between fertiliser inputs) than would be generated under optimal conditions, because the inoculum digester had broken down several weeks previously. Inoculum from the same digester had been used in two previous pilot studies using control grass (Appendix Section 4.2); the present inoculum contained 33-35% less VS than previously, and produced on average 24% less accumulated methane and 27% less m_N^3 CH_4 t^{-1} VS from control grass. Therefore methane production for the studied field vegetation may be higher if this experiment was repeated with optimal inoculum.

4.2.4 Statistical analysis

Statistical analysis was performed in the same was as described in Chapter 3 (Section 3.2.4). Nested ANOVA (using each quadrat's data) was used to test the difference between fertilised (n = 5 fields) and non-fertilised (n = 5 fields) biomass for lignin, cellulose, hemicellulose, total cell wall content and VS destruction. There were 4 quadrats per field. One-way ANOVA was used to examine differences between fertilised and non-fertilised grasslands using the mean value per field in specific and area methane production. Mean value per field was the average across the 4 quadrats. One-way ANOVA was also used to test methane production by pre-treated and untreated control grass (n = 3 of each). Tests of relationship were carried out by correlation or regression using the mean value per field (average of 4 quadrats) as each data point.

4.3 RESULTS

4.3.1 Chemical and cell wall composition analysis

Before testing the value of different grass batches in AD, I carried out a compositional analysis of the biomass from each set of field samples. Cell wall composition did not differ significantly between fertilised and non-fertilised biomass (either in lignin, cellulose, hemicellulose or total cell wall content (i.e. the sum of lignin, cellulose and hemicellulose)) (Tables 1 and 2). C:N was also not significantly different between fertilised and non-fertilised biomass. However, both marginal field types (fertilised and non-fertilised) were relatively similar to one another, compared to the control rye grass lawn clippings, which had a much higher N content (and hence lower C:N), and lower hemicellulose, lignin content and % cell wall (Table 1).

Table 1. Mean (SE) chemical and cell wall composition of biomass from fertilised and non-fertilised fields; and control grass (lawn clippings, 95% perennial ryegrass)

	Fertilised	Non-fertilised	Control grass
VS (% of TS)	93 (0.6)	92 (0.2)	88 (4.1)
Ash (% of TS)	6.8 (0.6)	7.7 (0.2)	12 (4.1)
C (%)	44 (0.2)	44 (0.2)	43
N (%)	2.1 (0.1)	1.8 (0.1)	4
C:N	22 (1.5)	25 (0.8)	11
Lignin (%)	27 (0.6)	28 (0.8)	21 (7.4)
Cellulose (%)	19 (0.8)	18 (1.0)	17 (0.7)
Hemicellulose (%)	14 (0.7)	15 (0.8)	9 (1.6)
Total cell wall (%)	60 (0.7)	62 (2.4)	47 (8.3)

VS, volatile solids; TS, total solids; C, carbon; N, nitrogen; total cell wall is sum of lignin, cellulose and hemicellulose content.

Table 2. Test of difference between fertilised and non-fertilised field biomass in chemical and cell wall composition, using nested ANOVA.

Variable	$F_{1,8}$	*P*
Lignin	1.649	0.234
Cellulose	0.533	0.486
Hemicellulose	2.618	0.144
Total wall†	0.646	0.443
C:N	2.985	0.121

†The sum of lignin, cellulose and hemicellulose

Lignin, cellulose, hemicellulose and total cell wall content were tested for correlations with species richness, number of semi-improved grassland indicator species, percentage grass cover and harvest date (using mean per field as each data point, n = 10 fields) (Table 3). Increasing species richness was marginally positively correlated with lignin content; harvest date had a marginal positive effect on hemicellulose; but harvest date showed no effect on lignin content. No other relationships were apparent.

Table 3. Correlations between cell wall components and other variables

Variables correlated		r†	r^{2*}	*P*
Lignin	Species richness	0.611		0.061
Lignin	Harvest date		0.031	0.932
Hemicellulose	Harvest date		0.356	0.068

†Spearman rank-order correlation was used for count data
*Pearson correlation was used.

4.3.2 Anaerobic digestion

4.3.2.1 Anaerobic digestion of fertilised and non-fertilised biomass

During anaerobic digestion, fertilised and non-fertilised biomass accumulated methane at similar rates, and both at a much higher rate than the inoculum-only control, with a high proportion of biogas generated within the first 20 days of digestion (Fig. 2a). Methane content was around 50% of the biogas, with very little difference between the field types (Fig. 2b). Specific methane production (Fig. 3a) and area methane production (Fig. 3c) were not significantly different between fertilised and non-fertilised biomass (Table 4). Table 4C in the Appendix shows specific and area methane production per field. VS destruction was similar between fertilised and non-fertilised biomass (Table 4). At the end of the experiment, the pH was an average of 7.4 for fertilised and 7.35 for non-fertilised AD.

The area methane production is an estimate of the production that could be achieved per ha from the harvested biomass, but this excludes vegetation eaten during the rest of the year by grazing sheep. Chapter 3 (Table 3) reported that total annual yield of biomass (an estimation of grazed yield eaten by sheep, plus harvested yield) was 6.5 t dry matter ha^{-1} (SE 0.4) for fertilised fields, and 6.3 t dry matter ha^{-1} (SE 0.6) for non-fertilised fields. Therefore, if fields were not grazed, I estimate that the total annual area methane yield would be 1237 and 1141 m_N^3 CH_4 ha^{-1} for fertilised and non-fertilised fields, respectively; assuming that the ‘grazed’ component of the vegetation in each field has the same biogas productivity as the ‘cut’ component (annual area values in Table 4). Although this is not significantly different between fertilised and non-fertilised fields, it constitutes a 52%

increase in area methane production for fertilised fields; and a 40% increase for non-fertilised fields.

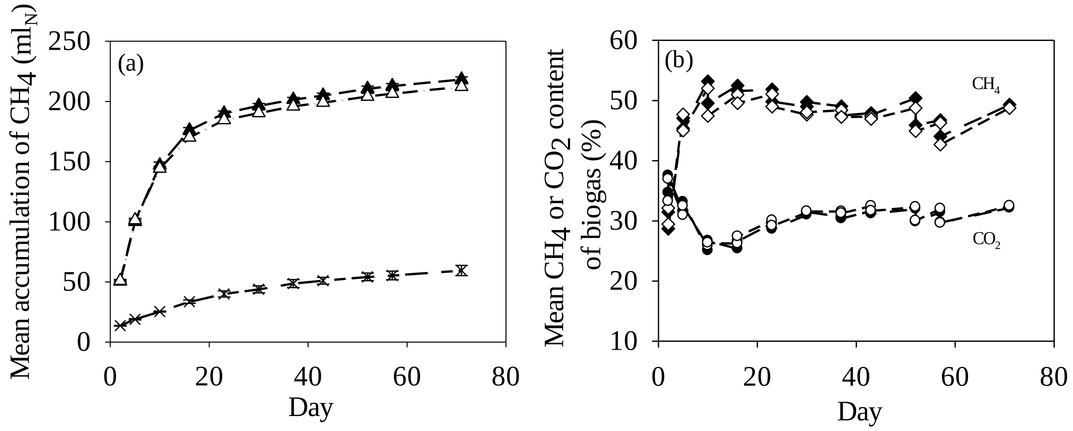

Fig. 2. (a) Mean accumulation of methane by AD of fertilised grassland biomass (black triangle) and non-fertilised grassland (white triangle). Error bars are SE (n = 5 fields of each type). Inoculum-only control (x symbol. Error bars are SE, n = 3). (b) Mean CH_4 and CO_2 content (%) of fertilised and non-fertilised biogas. Black symbols are fertilised; white symbols are non-fertilised.

Table 4. Methane production and VS destruction by fertilised and non-fertilised biomass in AD

Variable	Unit	Mean (SE)		% Difference		Type of	$F_{1,8}$	*P*
		Fertilised	Non-fert.	Fert.	Non-fert.	ANOVA		
Specific CH_4 prod.	Nm^3 CH_4 t^{-1} VS	227 (3)	218 (4)	+ 4		1-way	3.534	0.097
Area CH_4 prod.	Nm^3 CH_4 ha^{-1}	813 (123)	817 (41)		+ 0.5	1-way	0.001	0.976
Annual area CH_4 prod.	Nm^3 CH_4 ha^{-1}	1237 (64)	1141 (114)	+ 8		1-way	0.539	0.484
VS destruction	%	61.2 (1)	60.5 (0.4)	+ 1.2		nested	0.447	0.522

Pre-treatment did not significantly increase specific methane production of the control grass: 1-way ANOVA, n = 3 replicates, $F_{1,8} = 0.337$, $P = 0.593$. Mean specific methane production for pre-treated and non-treated control grass was 213 (SE 7) and 207 (SE 8) m_N^3 CH_4 t^{-1} VS respectively.

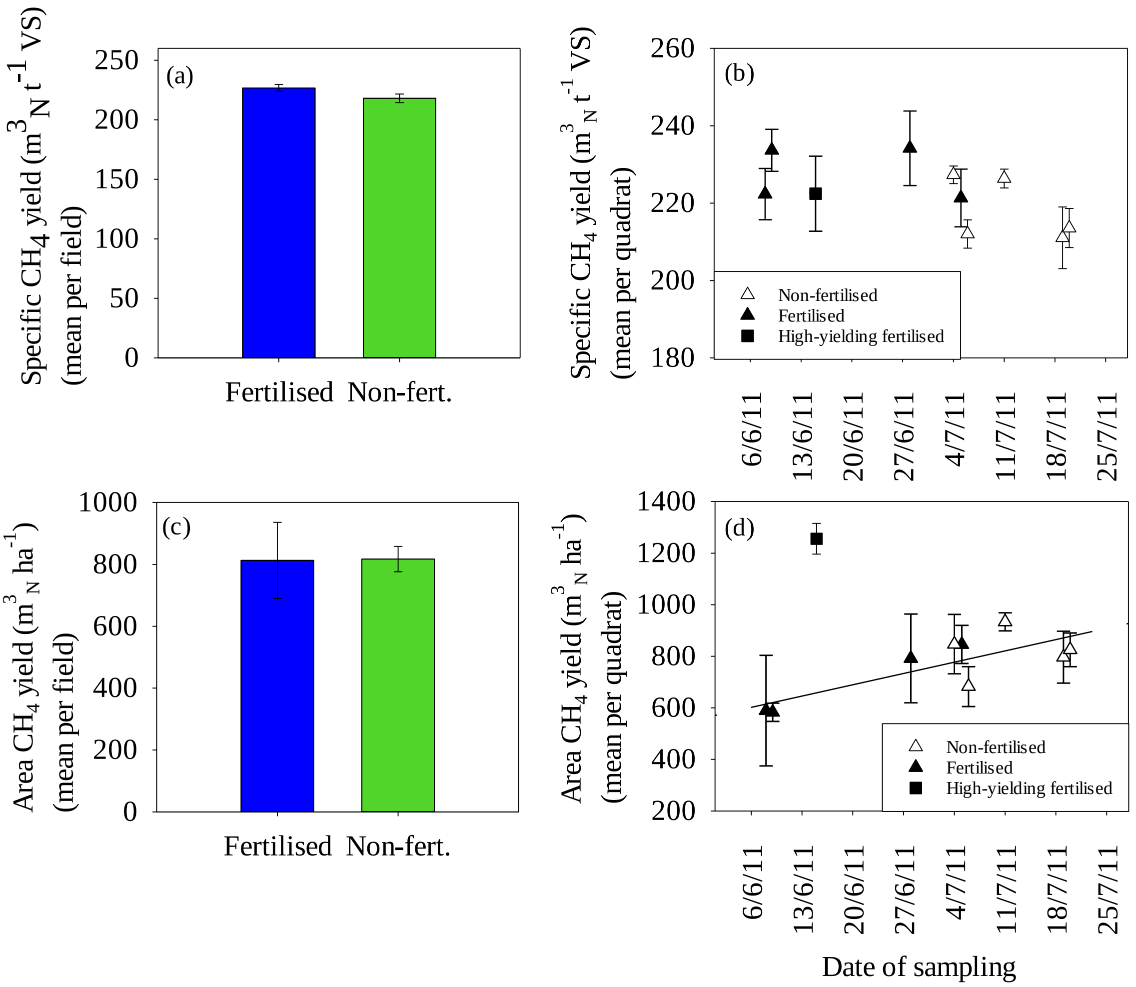

Fig. 3. (a) Mean specific methane production and (c) mean area methane production. Error bars are SE (n= 5 fields) (due to the nested structure of quadrats within field, variation due to quadrats is lost). (b) and (d) show effect of harvest date on specific methane production, and area methane production respectively. Each symbol is the average of 4 quadrats. Error bars are SE (n = 4 quadrats). The black square is an outlying high-biomass yield field. Linear regression equation: see Section 4.3.2.2.

4.3.2.2 Influences on specific methane production and area methane production

There was possibly a trend for later-harvested fields to have a lower specific methane production but there was no significant correlation or effect of field type (Table 5 and Fig. 3b). However, area methane production was positively correlated with harvest date (R^2 = 0.631, d.f. = 7, P = 0.011, excluding the high-biomass outlier; Fig. 3d) because of the increased biomass yield from field that were harvested at a later date. Given that specific methane production did not vary among field types or with harvest date, biomass yield was the primary determinant of area methane production (r^2 = 0.976: Table 5 and Fig. 4). This is why non-fertilised fields (which are generally cut later, once biomass has accumulated) had similar mean area methane production to fertilised fields. Neither specific methane production nor area methane production were correlated with plant species richness, percentage grass cover (Fig. 4C in the Appendix) or altitude (Table 5). Specific methane production and area methane production were not correlated with each other.

Table 5. Correlations between methane production and field/chemical variables.

Correlation with	Specific CH_4 production (m^3_N t^{-1} VS)			Area CH_4 production (m^3_N ha^{-1})		
	$r^{\dagger}$	$r^{2\$}$	*P*	$r^{\dagger}$	$r^{2\$}$	*P*
Species richness	0.15		0.676	0.078		0.829
% Grass cover		0.028	0.644		0.076	0.443
Biomass yield		0.039	0.751		0.976	< 0.001*
Altitude		0.04	0.577		0.096	0.383
Harvest date		-0.321	0.088		0.630	0.011*
Cellulose		0.152	0.265		0.0003	0.965
Hemicellulose		-0.425	0.041*		0.110	0.351
Lignin		-0.084	0.416		0.031	0.624

†Spearman rank-order correlation was used for count data, otherwise Pearson$ correlation was used.
d.f. = 8

Hemicellulose content had a significant negative effect on specific methane production (specific methane production = -2.769 x hemicellulose content + 262.585, and R^2 = -0.425, P = 0.041) (Fig. 5), but it showed no association with area methane production (Table 5). Lignin, cellulose and total cell wall content showed no relationship with specific methane production or area methane production (Table 5). Vegetation C:N content was marginally negatively correlated with specific methane production (Pearson r^2 = -0.386, P = 0.055).

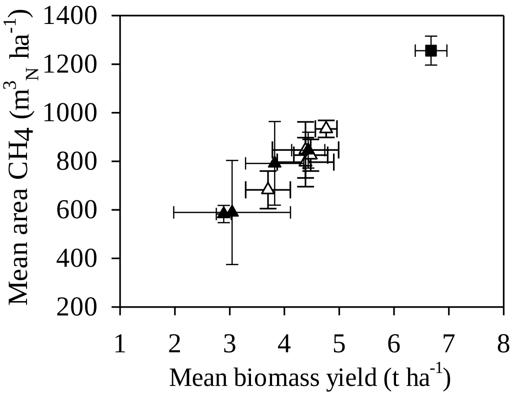

Fig. 4. Relationship between mean area methane production and biomass yield of all the fertilised and non-fertilised fields. Symbols are average per quadrat, extrapolated to t ha^{-1}. Error bars are SE (n = 4 quadrats). Black triangles are fertilised; white triangles are non-fertilised. For correlation see Table 5. Black square is the fertilised outlier field.

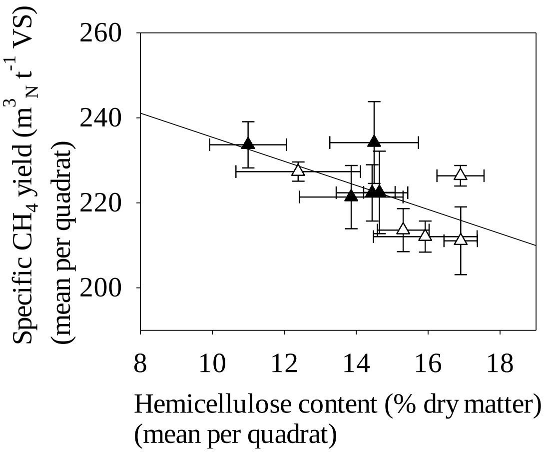

Fig. 5. Relationship between hemicellulose content and specific methane production of fertilised and non-fertilised biomass (mean per quadrat). Error bars are SE (n = 4 quadrats). Black triangles are fertilised; white triangles are non-fertilised. For linear regression see Section 4.3.2.2.

4.4 DISCUSSION

The findings presented here generally support the idea that using marginal grasslands for biomethane production by AD is compatible with biodiversity. The non-fertilised marginal grasslands achieved as much specific methane production and area methane production as fertilised marginal grasslands, yet had 48% higher biodiversity (in 2011 when the biomass was harvested for AD; and 63% higher when re-surveyed earlier in the summer in 2012) (Chapter 3).

I found no significant difference in specific methane production (non-fertilised mean 218, fertilised mean 227 m_N^3 CH_4 t^{-1} VS), despite our hypothesis that it would be higher for fertilised vegetation due to earlier harvest (and lower cell wall content), and greater cover of digestible grasses. Others have found comparable specific methane productions in similar grasslands. For example: intensive permanent grassland silage in an Austrian valley site produced 190-392 m_N^3 CH_4 t^{-1} VS (Amon *et al.*, 2007b); low-input permanent hill grassland silage in Austria yielded 128-221 m_N^3 CH_4 t^{-1} VS (Amon *et al.*, 2007b); semi-natural grassland silage in Germany gave 218 m_N^3 CH_4 t^{-1} VS (Richter *et al.*, 2009); roadside grassland silage under a 2-cut regime in Germany produced 211-221 m_N^3 CH_4 t^{-1} VS (Piepenschneider *et al.*, 2016); and low-input meadow foxtail silage from Germany produced approximately 190 and 200 m_N^3 CH_4 t^{-1} VS in June and July (Herrmann *et al.*, 2013).

Later harvest usually has a negative effect on grass specific methane production due to increase in lignocellulosic material (Prochnow et al., 2009b; Melts and Heinsoo, 2015). I observed a significant negative effect of hemicellulose content on specific methane production (due to a slight, but not-significant, increase of hemicellulose with harvest date; this relationship was significant when observed by Amon *et al.* (2007a). However, overall lignin, cellulose, hemicellulose and total cell wall content did not differ between fertilised and non-fertilised biomass, perhaps because of the slight overlap in harvest dates between field types, due to being cut at the time of the farmer's harvest. Total cell wall content (60% for fertilised, 62% for non-fertilised biomass) was similar to mature grass (Nizami *et al.*, 2009). It has been suggested that high forb cover (and therefore low grass cover) could affect specific methane yield (Prochnow *et al.*, 2009b) but % grass cover showed no effect on specific methane yield. Field vegetation C:N showed a slight negative trend with specific methane production, thus, lower N led to lower methane production, as may be expected due to a lower N supply for microbes. It has been suggested that species-diverse grassland contains plants at different developmental stages, which may also contribute to digestibility being not as reduced as expected (Tallowin and Jefferson, 1999). This may help explain the lack of difference in cell wall content observed in this study, which may have contributed to the lack of difference in specific methane production, despite the differences in species diversity and composition. It should be remembered that both field types are in marginal land, and were therefore relatively similar to one another, despite the difference in fertiliser regime; the biomass from both field types had much lower N content and higher structural content than immature lowland lawn grass (Table 1).

The efficiency of VS destruction was similar to that reported in previous work (Lehtomaki *et al*., 2008). Ensiling is a common pre-treatment for energy crops in AD, but due to contamination with fungus during ensiling, thermochemical pre-treatment was instead performed, but had no significant effect on methane yield. Lack of an effect of thermochemical pre-treatment on dried grass has been seen before (Fernandes *et al*., 2009; Menardo *et al*., 2012), possibly due to lignin redistribution and condensation, and increasing cellulose stability (Hendriks and Zeeman, 2009). Pre-treatment was therefore considered to have not influenced specific methane production of field samples. Previous experiments had shown that digestion of grass by the sewage digestate inoculum was reproducible when collected fresh each time (Appendix Section 4.2.2). A previous experiment using the same inoculum, control grass and AD assay conditions as the current conditions (Appendix Table 4B, treatment 6) had produced specific methane yields of 280 m_N^3 CH_4 t^{-1} VS. Therefore methane yields were substantially lower in this experiment (207 m_N^3 CH_4 t^{-1} VS), most likely due to the lower activity of the inoculum after a previous break-down of the source digester (Section 2.3). Thus methane yields of the field grass may be substantially higher when the inoculum is working optimally.

We expected higher area methane production in fertilised fields, due to higher specific methane production and higher biomass yield. However, we found no difference between field types (low-input 817, high-input 813 m_N^3 CH_4 ha^{-1}) and our values are a little lower but similar to previous estimates for marginal low-input grasslands (for example 919 m_N^3 CH_4 ha^{-1} (Richter *et al.* 2009) and 910 m_N^3 CH_4 ha^{-1} $annum^{-1}$ (Amon *et al.*, 2007b)). If the total annual biomass yield (including that eaten by grazing sheep) is considered, area methane production increases substantially to 1141-1237 m_N^3 CH_4 ha^{-1}. These values are similar to a hill site with 2-3 harvests per year in Austria, but less than an intensively managed valley site (Amon *et al.*, 2007b), reflecting the more extensive nature of the fertilised fields in this study. The high correlation between area methane production and biomass yield shows that 98% of variation in area methane production was due to vegetation yield rather than specific methane production, which is commonly observed (Prochnow *et al.*, 2009b). The increasing biomass yield with harvest date compensated for the slight, not-significant fall in specific methane production, leading to similar area methane production between later-cut non-fertilised fields and earlier-cut fertilised fields.

4.5 CONCLUSION

Non-fertilised grassland (at risk of abandonment and loss of 'traditional' agricultural biodiversity) appears to have potential as a feedstock for AD within marginal regions where other options of bioenergy production are limited. This conclusion is likely to extend to other marginal grasslands in the cooler parts of Europe, and supports Amon *et al*'s (2007b) proposal that AD crops must be grown for sustainability as well as yield; reiterated by the European Commission's sustainability requirements for solid and gaseous biomass (European Commission, 2010). Agriculture is the second largest GHG producer globally, partly due to fertiliser use (Candell and Schulze, 2014). GHG emissions for marginal grassland AD may be lower for low-input than high-input grassland, for the same biomethane yield, due to absence of inorganic fertiliser (Clayton *et al.*, 1997; Gilbert *et al.*, 2011). This is examined in Chapter 5. Applying digestate (the nutrient solution remains of AD) to land also reduces fertiliser use and recycles nutrients (Lukehurst *et al.*, 2010).

Overall, the results suggest that the two field management types are equally suitable for methane production, given that the two types generate comparable specific methane production (Fig. 3a) and per-unit-area methane production (Fig. 3c). Therefore, using non-fertilised marginal grasslands for bioenergy applications may be compatible with their higher biodiversity, when cut at the time of the farmer's usual harvest. Farmers could potentially use less fertilizer and, if livestock numbers fell, use their harvested biomass for AD rather than for feeding animals, without changing other management practices. The economic viability of such a strategy is assessed in Chapter 6.

Chapter 5: Greenhouse gas emissions from anaerobic digestion of hay or silage

ABSTRACT

As reported in Chapters 3 and 4, hay grasslands in marginal areas (which did not receive inorganic fertiliser) had higher biodiversity, but similar biomethane production by anaerobic digestion (AD) to silage grasslands (which received inorganic fertiliser). Due to poor growing conditions and productivity, such marginal grassland fields are at risk of abandonment, and loss of biodiversity, if an alternative use of their biomass is not found such as using the silage or hay for bioenergy production, via AD. Biomethane can be burned to produce electricity and heat. However, in order to qualify for financial support in the EU since January 2017, electricity from new bioenergy plants must achieve GHG savings of at least 50% compared to fossil fuel electricity. Therefore, in the context of using marginal grassland crop for AD, this Chapter asked whether hay or silage (plus sheep production in both field types) had the lower GHG emissions (i) per ha of land; (ii) per tonne dry matter of silage/hay; and (iii) per unit of electricity (kWhe) generated from biomethane. It also asked whether electricity from silage or hay biomethane had sufficiently reduced GHG emissions to meet the EU sustainability requirements. Empirical farm data were entered into a carbon calculator (Cool Farm Tool) to estimate GHG emissions. Data on electricity and heat production, from burning the biomethane, were taken from financial models of AD in Chapter 6. Sheep had a major impact on GHG emissions from silage/hay fields, such that if they were to remain on land growing bioenergy feedstock, there would be no GHG savings from biomethane electricity (compared to fossil fuel electricity). This would prohibit EU financial support. If farmers decided to reduce sheep numbers by 60% on the studied land, biomethane electricity could produce GHG savings of > 50% compared to using fossil fuel electricity, which would meet EU sustainability requirements. And if there were no sheep on the land, GHG savings of up to 96% were feasible. Hay consistently produced significantly lower GHG emissions than silage (per ha; per t dry matter; and per kWh of electricity). And hay produced higher GHG savings from biomethane electricity than silage. In terms of GHG savings, the commonly used AD feedstock, species-poor grass silage, could be swapped for hay and biodiversity would be maintained in marginal areas.

5.1 INTRODUCTION

Mitigating climate change requires substantial reductions in global-warming greenhouse gas (GHG) emissions. Globally, agriculture is the second largest emitter of GHG (producing mainly methane and nitrous oxide) after fossil fuel combustion (Canadell and Schulze, 2014). Thus, if biorenewable energy (which displaces fossil fuel combustion) is produced using a low GHG-emitting agricultural feedstock, it could mitigate climate change by reducing emissions of carbon dioxide (CO_2), nitrous oxide (N_2O) and methane (CH_4). This Chapter describes the GHG emissions from marginal grassland, and the use of its biomass in bioenergy.

Grasslands cover 26% of the Earth's land area (Foley *et al.,* 2011) and are increasingly becoming abandoned by agriculture (Secretariat of the Convention on Biological Diversity, 2014; Stoate *et al*., 2009). In response to this, traditionally-managed, biodiverse perennial grasslands could potentially be used for bioenergy production without competing for intensive food production (Tilman *et al.,* 2006; Fargione *et al*., 2009). Because abandonment of farmland can lead to reduced biodiversity due to loss of grassland plant and bird species (Stoate *et al.,* 2009), a co-benefit of bioenergy production is that maintenance for bioenergy production could protect the biodiversity which is essential for climate and pest resilience, as well as maintaining ecosystem services (Frison *et al.*, 2011). This research has already established that hay land has higher biodiversity than silage land due to lower nutrient input (no inorganic fertiliser) on hay land, later harvest of vegetation and no re-sowing with high-yielding grasses. Permanent grassland represents a bioenergy crop without the impacts of land use change, which have been shown to be so deleterious to bioenergy GHG savings (Djomo and Ceulemans, 2012; Ruan and Robertson, 2013). Silage is widely used in on-farm AD; hay has been studied for combustion (Rosch *et al.,* 2009) and grass has been studied for transport fuel production via Fischer-Tropsch conversion (Corton *et al*., 2013), but AD was chosen as the bioenergy process for study here because it is already used on-farm. Wastes (agricultural, food or municipal) perform better than crops at GHG reduction in AD (Styles *et al.,* 2016), but in marginal regions access to sufficient food and municipal wastes may be difficult. Therefore this research focuses on the feedstocks which may be available to farmers in marginal areas should some of their land be abandoned for livestock production; that is silage and hay.

First generation (food-based) bioenergy crops, have been widely criticised for their impacts on land use change, increased food prices and even increased GHG emissions compared to fossil fuel energy (e.g. Searchinger *et al.*, 2008, Fargione *et al.*, 2008). Therefore the sustainability of bioenergy feedstocks has become a focus of research and government policy in recent years. This includes, for example, the European Union's (EU) aims to increase energy efficiency, and protect highly biodiverse habitats and carbon stocks when sourcing solid and gaseous biomass; as well as to reduce GHG emissions (European Commission, 2010). Greenhouse gases contribute to climate warming, therefore potential reductions in GHG emissions by producing biorenewable energy, rather than burning fossil fuels, are widely investigated. But environmental sustainability of bioenergy encompasses many more impacts than climate warming potential; including ozone depletion, human toxicity, acidification, eutrophication, ecotoxicity and resource depletion (Gilbert *et al.,* 2011; Poeschl *et al*., 2012). These impacts can be calculated using Life Cycle Assessment (LCA) and it has been shown that bioenergy systems, such as anaerobic digestion (AD), may be beneficial in one impact but detrimental in others (Mezzullo *et al.*, 2013). For example, AD of agricultural waste such as cattle manure/slurry can be beneficial by providing significant savings in fossil fuel depletion, by burning the biogas (rather than heating oil) to provide heat on-farm (Mezzullo *et al.*, 2013). This contributes to mitigation of climate change. The use of digestate as a fertiliser also helps to mitigate climate change, having lower CH_4 emissions than livestock slurry (Waste and Resources Action Programme, 2016); and its use reduces fossil fuel resource depletion (Mezzullo *et al.*, 2013). However, it is possible to have negative impacts from AD (Mezzullo *et al.*, 2013) and biofuel production (Elshout *et al.,* 2015): these need to be prevented, including production of harmful respiratory inorganics (from open storage of digestate and combustion of biogas/other fuels); acidification (of soil, groundwater and rain); and eutrophication (of soil and water) (Styles *et al.*, 2016). Capturing multiple impacts such as those listed above requires large datasets, can be relatively resource intensive, or can require specialist costly LCA software. Since the legislation affecting bioenergy is concerned primarily with calculating GHG emissions (European Commission, 2009), the research presented in this chapter used a carbon calculator (Cool Farm Tool; Hillier *et al.*, 2011) to identify the global warming impact (i.e. GHG emissions) of bioenergy feedstock production and electricity production in marginal areas.

A LCA is structured in a different way to standard Biology research papers. It is traditionally made up of 4 component parts: 1) goal and scope, 2) life cycle inventory, 3) life cycle impact assessment, 4) interpretation (ISO14040, 2006). The following sections will describe each of these steps for this research.

This Chapter examines greenhouse gas emissions from the fertilised and non-fertilised fields examined in Chapters 3 and 4. However, the focus is now on the use of the fields' processed biomass (i.e. silage and hay). Therefore, for simplicity, the fertilised fields are now labelled 'silage fields', and the non-fertilised fields are labelled 'hay fields'. There is no change in their management. One factor previously ignored (dry matter loss during processing of the cut grass) is now included, in order to give a more realistic estimate of GHG emissions. This is explained more in Chapter 2, Section 2.2 and 2.3.

5.2 SCOPE

This Section covers lots of general information on the inputs and outputs from the system being studied, which in this case is silage/hay production (including sheep farming) and its use in AD to produce electricity. Specifically, the scope includes the goal of the research, what products were examined and how they were made. The scope defines the processes which will be included in the LCA (the system boundary). It describes the units in which the LCA results will be expressed (the functional units); and it describes the scenarios of electricity production from AD of silage/hay. The procedure for dividing the GHG emissions between the products of AD is covered (the allocation procedures), as well as the research questions being asked. The Cool Farm Tool carbon calculator is also described.

5.2.1 Goal

The first goal of this Chapter was to estimate GHG emissions from hay and silage production ('carbon footprint') on sheep farms in a marginal area, to find out whether silage or hay land had lower emissions. The second was to estimate GHG emissions from using the silage/hay in AD and burning the biomethane for electricity, to determine if GHG savings could be made compared to EU fossil fuel electricity, or the UK's electricity mix (which includes renewables and nuclear). In order to receive financial support in the EU and count towards emissions reduction targets as of January 2017, new bioenergy installations need to demonstrate GHG savings of at least 50% compared to EU fossil fuel electricity, and 60% savings from January 2018 (European Commission, 2010). This work may be of interest to those investigating alternative uses of the grassland biomass produced in marginal areas e.g. policy makers or farmers, who are interested in climate change and how to reduce their GHG emissions.

In order to achieve the goals, the questions asked were: which fields' management (silage or hay, including sheep) had the lower GHG emissions (i) per ha of land; (ii) per tonne of silage/hay DM; or (iii) per unit of electricity generated from biomethane? And did electricity from biomethane have sufficiently reduced GHG emissions to meet the EU sustainability requirements? I answered these questions by undertaking LCA using a carbon calculator called the Cool Farm Tool (Hillier *et al.*, 2011), described in Section 5.2.6.

5.2.2 The products being examined

The agricultural products being examined were silage and hay, from the 5 silage (fertilised) fields and the 5 hay (non-fertilised) fields examined in the previous Chapters. They were on 5 grazing livestock farms (producing mainly sheep but also some beef cattle) in the north of England (UK). Each farm had 1 silage (fertilised) field and 1 hay (non-fertilised) field, and they were all grazed by sheep. The silage/hay is produced as forage for livestock to eat in winter. All fields received farmyard manure (FYM; cattle manure mixed with straw bedding and urine, collected when cattle are housed for the winter. It is used as organic fertiliser). Some fields received lime. The methods of making the silage and hay, as well as the location and selection of the fields, are explained in the Methods Chapter (Chapter 2). An overview comparing the generalised management of silage/hay fields is shown in Figure 1. Vegetation loses dry matter during silage/hay-making: dry matter losses used in this thesis were 18% loss while making silage, and 36% loss while making hay (Buhle *et al.*, 2012). Therefore all silage/hay yields reported in this Chapter have been reduced from the gross yields reported in Chapter 3. They represent the actual amount available to the farmers. The time period examined was 1 year (2011): yield can vary between years but the silage and hay fields had been managed by grazing and cutting for years.

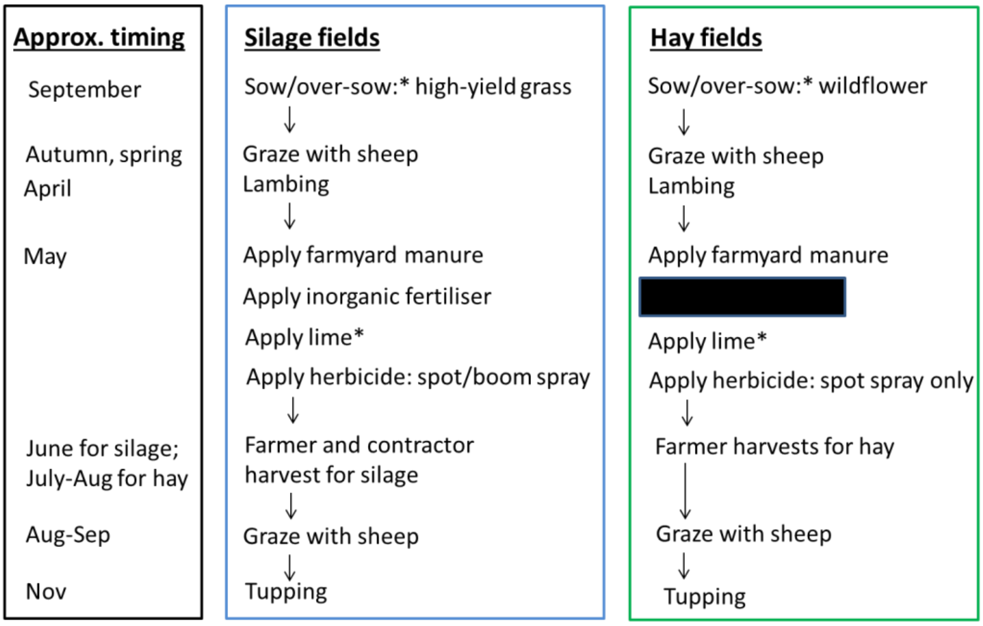

Figure 1. Schematic comparing silage and hay field management. * means not done every year.

The bioenergy products being examined were electricity and heat, produced from the burning of biomethane (made by AD). The amount of electricity and heat that could be made by AD of silage or hay was estimated in Chapter 6 (which examines the finances of AD). This is explained in the next Section. The feedstocks digested were silage or hay, plus manure.

5.2.3 System boundary

The system boundary defines which manufacturing / crop management / livestock farming (etc.) processes are included in the calculations of GHG emissions in the LCA (Figure 2). GHG emissions were calculated for each of the 10 fields. The processes 'upstream' of the fields which were included were manufacture of the crop and sheep inputs. Inside the field, the processes included were: fossil fuel burned by tractors during e.g. sowing seed, spreading fertiliser and cutting vegetation; emissions which occur naturally from the soils (background soil emissions); soil emissions due to application of inorganic fertiliser, FYM, lime; and so on as shown in Figure 2. Because sheep are present on the land, emissions from 'sheep' were added to emissions from 'crop' (as defined in Figure 2). Sheep are moved to different areas of a farm throughout the year, therefore sheep inputs and emissions were only considered when the sheep were present on the fields. The allocation Section (5.2.5) explains how emissions from sheep were removed in order to simulate the land being abandoned by livestock. GHG due to manufacture of buildings and infrastructure was not included, as is often the case (Jury *et al.,* 2010).

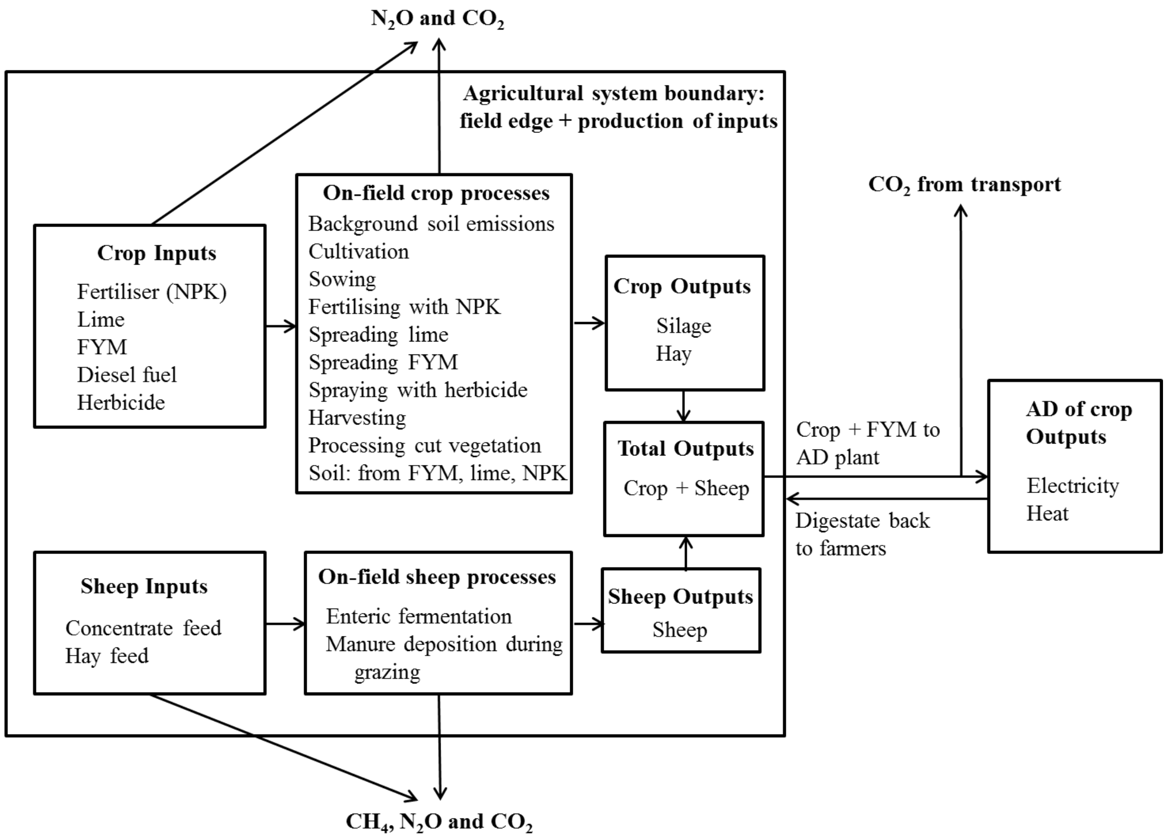

Figure 2. System boundary of management inputs, processes and outputs from crop (silage/hay) and sheep production systems. Crop transport to the AD plant includes FYM which is co-digested. Transport also includes digestate transported back to the farmers. Note that hay fields do not receive fertiliser (NPK), and FYM is farmyard manure.

The electricity and heat that could be produced by AD of silage or hay was estimated for 10 different scenarios of AD in Chapter 6 (described briefly in Table 2 of Chapter 6; and in detail in Sections 6.3.2.1, 6.3.2.2 and 6.3.2.3). The AD scenarios are summarised here. There were 4 scenarios of a co-operative of sheep farmers running an AD plant (the fields studied in this thesis were from sheep farms). There were 6 scenarios of a dairy farmer running his/her own AD plant. The reason for including the two different types of AD business (co-operative and dairy farm) was purely financial. All AD scenarios digested hay or silage, plus manures. The manures were included to provide long-term microbial stability (Thamsiriroj *et al.,* 2012), as is commonly done in AD in reality. The manure used in co-operative AD was FYM, and the manure used in dairy AD was cattle slurry. The purpose of the AD scenarios was to compare silage and hay as AD feedstocks. The amount of silage/hay fed to each scenario differed, to allow investigation of different amounts of feedstock in AD (Chapter 6, Table 2). In scenarios where the amount of silage/hay is that currently grown by sheep farmers, they are labelled 'hay' or 'silage'. In scenarios where more hay or silage was digested than is currently produced by sheep farmers (requiring conversion of land to produce the extra), they are labelled 'increased hay' or 'increased silage'. Where the dairy farmer grows their own crop on-farm, it is labelled 'own hay' or 'own silage'. Specifically the 4 co-operative scenarios digested silage (scenario 2a), hay (2b), increased silage (2c) and increased hay (2d) (grown by the sheep farmers). The 6 dairy scenarios digested silage (scenario 3a), hay (3b), increased silage (3c) and increased hay (3d) (grown by the sheep farmers), own silage (3e) and own hay (3f) (grown by the dairy farmer) (Chapter 6, Table 2).

When calculating GHG emissions per unit of electricity from biomethane, emissions from the transport (of feedstock and/or digestate) were added to the silage/hay production emissions (Figure 2). All co-operative AD scenarios included transport of feedstock and digestate, between farms and the AD plant. The first four dairy AD scenarios (digesting silage, hay, increased silage or increased hay) included transport of feedstock to the AD plant, but no digestate transport because digestate would be used on-farm. Dairy AD scenarios digesting 'own' hay or silage had no transport included, because all feedstocks and digestate were produced and used on-farm. The amount of electricity generated per year is shown in Tables 6 and 7 in Chapter 6. The biomethane (as raw biogas) was assumed to be burned in a Combined Heat and Power plant with an assumed 33% electricity and 42% heat production efficiency. The heat was used to heat the digester and the farmhouse, but some heat was unused. The effect of the amount of heat used (and therefore included in the allocation of GHG) is investigated in the sensitivity analysis (described in Section 5.2.5).

Some GHG were not included: processing feedstock (European Commission, 2010) at the anaerobic digestion plant; reduced fossil fuel depletion by replacing inorganic fertiliser with digestate (Buhle *et al.,* 2012); and leakage of methane from the AD plant which would have deleterious effects on GHG savings (Bacenetti and Fiala, 2015).

5.2.4 Functional unit

The units in which the results of the LCA are going to be expressed are defined here (called ‘functional units’). The impact of interest for each functional unit is GHG emissions, since this is the metric favoured by EU legislation for bioenergy production. A functional unit is “the value associated with the function of the system” (Caffrey and Veal, 2013). Function can incorporate several features including quality, time, area, etc. The function of the fields considered in this Chapter was to produce (i) silage or hay, and (ii) sheep. (Hay fields also had the function of maintaining/enhancing biodiversity in return for agri-environment payments from the UK government which is not examined further here, but is studied in Chapter 6.) There are different ways of expressing agricultural GHG emissions (e.g. per ha, per kg of product) highlighting that agricultural systems are complex (Caffrey and Veal, 2013) and can require more than one functional unit (Nemecek *et al.,* 2015) for full understanding of the system. Therefore several functional units were used to compare GHG from silage and hay land in a marginal area of the north of England:

(i) Land management function (Nemecek *et al.*, 2015): GHG emissions from 1 ha of silage or hay land (measured in kg CO_2e ha^{-1}). This provides understanding of the intensity of land use per area with its potential associated environmental effects.
(ii) Productivity function (Nemecek *et al.*, 2015): GHG emissions from 1 t of dry matter (DM) silage or hay (measured in kg CO_2e t^{-1} DM silage or hay). This was calculated by dividing GHG per ha by the dry matter yield of silage or hay. Biomass yield per ha can thus affect GHG per t DM.
(iii) Electricity function: GHG emissions from 1 kWh of electricity (kWhe) made by burning the biomethane from AD (including emissions from crop production, sheep and transport; measured in kg CO_2e $kWhe^{-1}$).

5.2.5 Allocation

Because sheep are on the studied fields, emissions from sheep were included in (‘allocated’ to) crop production and thus in the functional units GHG per t DM and per ha. This crop was then used in AD, thus emissions from electricity production included emissions from sheep. However, the context of this research is that sheep may fall in number or be removed entirely from such marginal grasslands. Therefore sensitivity analyses were performed for each functional unit (GHG per ha, per t DM, per kWh electricity), to examine the effect of removing sheep from the land. Sensitivity analyses were performed for a single silage field (d) and a single hay field (b), which were chosen because they had median silage and hay land management emissions per ha. The emissions from enteric fermentation, sheep manure management, concentrate feed production and hay feed production together accounted for emissions from sheep. Thus in sensitivity analyses where there were zero sheep, these emissions were excluded. Sensitivity analyses also included changing all other inputs to determine their effect on emissions.

The AD scenarios produced both electricity and heat, therefore GHG emissions due to electricity production (explained in Section 5.2.3) were allocated between them according to their energy content (such that 1 kWh of electricity = 1 kWh of heat). GHG were also allocated according to the % of heat used, such that the more heat that was used, the more GHG were allocated to heat, and the lower the GHG per unit of electricity. The effect of using different amounts of heat was examined in a sensitivity analysis, which also examined changes in other inputs. Because there were 10 different AD scenarios, which was too many to perform sensitivity analyses on, two AD scenarios were chosen for the sensitivity analysis: co-operative AD increased silage; and co-operative AD increased hay. They were chosen because they had the highest transport requirements (therefore they could show the effect of maximal and zero transport), and the hay scenario was financially interesting (Chapter 6).

5.2.6 The carbon calculator tool used

The Cool Farm Tool (Hillier *et al.,* 2011) carbon calculator tool was used to calculate emissions of CH_4, N_2O and CO_2. It was considered *"the highest-scoring, publically-available, free GHG accounting tool"* in a review by Whittaker *et al.* (2013) of 11 UK farm-based and bioenergy-based carbon calculator tools; and has been used by Sozanska-Stanton *et al.* (2016) and Styles *et al*. (2015b) to calculate GHG emissions. It is a UK-based tool which increases relevance of the analysis because emissions data can vary widely across the world. There are two versions of the Cool Farm Tool: the online version was used to calculate GHG due to crop production (silage and hay), and the Excel version was used for sheep emissions (because the online version did not provide all options required for the calculation of emissions from sheep). They used the same methods except that I corrected the global warming potential of N_2O in the Excel livestock tool from 296 to 298, to match the online Tool.

5.3 LIFE CYCLE INVENTORY

The life cycle inventory describes the actual data used in the LCA in order to be transparent. The data are those inputs into, and outputs from, the system being studied. In this research, that means all inputs into silage/hay production and sheep farming. Outputs being studied in this research are the silage/hay produced by farming, and electricity and heat produced by AD. These were described in general above, but are now described in detail. Also included are the 'emission factors' used by the carbon calculator to calculate GHG.

5.3.1 Silage/hay production data

The first process examined was silage/hay production. Data used were either empirical data measured on the fields (E); reported by the farmers (R); or standard data (default data contained in the Cool Farm Tool, or from another publicly-available source as specified) (S). Table 1 shows the data for silage/hay production which were used in the online Cool Farm Tool for each of the 5 silage fields, and each of the 5 hay fields. 'Vegetation type' (E) was grass-clover if total clover cover was ≥ 10%, or perennial grass if < 10% (plant cover is reported in Chapter 3): the two vegetation types were very similar in emissions ha^{-1}. Fresh-weight yields of silage (E) and hay (E) were derived from the dried biomass yield measured in the fields (Chapter 3). In order to convert dried yield ha^{-1} to fresh yields of silage and hay, DM losses in making silage (18%) and hay (36%) were applied (Buhle *et al.*, 2012). Then the DM content of the remaining biomass was converted to 25% DM content to give yield of fresh silage, and 86% DM content to give yield of fresh hay. Sheep return to the fields within an average of 2.6 weeks after harvest, therefore it was assumed that any silage/hay losses which occurred on the fields were consumed by sheep, and are included in the livestock enteric fermentation GHG emissions. Inorganic fertiliser (NPK) rates (R) were reported by the farmers. NPK comprised a mix of N (assumed to be divided equally into ammonium-N and nitrate-N); P_2O_5 and K_2O. As a waste product, per EU sustainability GHG accounting (European Commission, 2010), emissions from the production of FYM (S) were assumed to be zero. Soil dynamics can be very important in assessing carbon footprint (Buhle *et al.*, 2012), and they are taken into account by the Cool Farm Tool. Soil texture (S) and drainage (S) for each site were estimated using Cranfield Soil and Agrifood Institute's Soilscapes website (Cranfield University, 2017). Soil pH (E), moisture (E) and soil organic carbon (SOC) (E) had been previously measured for each field (Chapter 3). Herbicide (S) was spot-sprayed annually on both field types, and its production emissions were estimated at 9.1 kg CO_2 ha^{-1} using data from Nix and Redman (2015) and the Cool Farm Tool.

The Tool provided default values for the amount of fuel (diesel) (S) used by tractors in cultivation and harvesting operations. These Cool Farm Tool default values were used for all operations except for spreading FYM (S) (9.7 l ha^{-1}, sourced from Whittaker *et al.*, 2013); shallow rotavation of fertilised fields (S) (13 l ha^{-1}, sourced from Witney (1988), p 147); heaping silage bales in the field (R) (0.25 litre diesel per bale of 500kg) (*pers comm.* Andrew Hattan); and loading hay bales onto a trailer (R) (0.01 litre diesel per 20 kg hay bale) (*pers comm.* Andrew Hattan). Boom spraying on one fertilised field; and cultivation, spreading seed and spreading lime on other fields do not occur every year, therefore the fuel and amount of product used in these processes was annualised.

Table 1. Annual silage/hay production data for each field (inputs and yield of crop)

Annualised field management		*	Silage fields a	b	c	d	e	Hay fields a	b	c	d	e
Field details												
Vegetation type		E	Grass-clover	Perenn. grass	Grass-clover	Grass-clover	Perenn. grass	Perenn. grass	Grass-clover	Grass-clover	Grass-clover	Perenn. grass
Fresh weight yield (silage/hay) ($t\ ha^{-1}$)[#]		E	13.1	8.9	8.5	11.2	19.7	2.5	2.9	3.2	3	2.9
Inorganic fertiliser rate ($kg\ ha^{-1}$)		R	125	250	213	250	250	0	0	0	0	0
NPK type		R	20:10:10	20:10:10	25:05:05	20:10:10	20:10:10					
FYM rate ($t\ ha^{-1}$)[$]		R	11	10	11	11	12	11	10	11	11	12
Limestone rate ($t\ ha^{-1}$)		R	0.15	0.62	0.15	1.25	0	0.15	0.62	0.15	0.83	0.45
Herbicide amount		S	9.1	9.1	9.1	9.1	9.1	9.1	9.1	9.1	9.1	9.1
Soil	Texture	S	Coarse	Coarse	Medium	Medium	Coarse	Medium	Coarse	Medium	Medium	Medium
	Moisture	E	Moist	Moist	Moist	Moist	Moist	Moist	Moist	Moist	Moist	Moist
	Drainage	S	Good	Good	Good	Poor	Good	Poor	Good	Good	Poor	Poor
	pH	E	4.94	5.84	5.05	4.83	5.36	4.43	5.28	4.2	4.99	4.7
	SOC (%)	E	11.51	8.35	9.35	8.44	8.19	7.99	6.64	6.78	7.94	26.12
Diesel fuel use ($l\ ha^{-1}$)												
Cultivation		S	0	0.43	0	0	0.43	0	0	0	0	0.01
Spread seed		S	0	0.25	0	7.6	0.25	0	0	0	0	0.304
Spot spray herbicide		S	0.62	0.62	0.62	0.62	0.62	0.62	0.62	0.62	0.62	0.68
Spread FYM*		S	9.7	9.7	9.7	9.7	9.7	9.7	9.7	9.7	9.7	9.7
Spread inorganic fertiliser		S	7.6	7.6	7.6	7.6	7.6	0	0	0	0	0
Spread lime		S	0.25	1.9	0.25	1.9	0	0.25	1.9	0.25	1.27	0.76
Harvesting	Mow	S	1.66	5.82	1.66	1.66	1.66	1.66	1.66	1.66	1.66	1.66
	Turn	S	2.21	2.21	4.42	2.21	2.21	6.63	4.42	6.63	8.84	6.63
	Rake	S	2.21	2.21	2.21	2.21	2.21	2.21	2.21	2.21	2.21	2.21
	Bale	S	11.35	8.57	8.23	10.08	14.08	9.87	11.23	11.98	11.43	11.23
	Wrap	S	11.35	8.57	8.23	10.08	14.08	0	0	0	0	0
	Heap silage bales	R	6.54	4.49	4.27	5.61	9.85	0	0	0	0	0
	Load hay on trailer	R	0	0	0	0	0	1.24	1.47	1.59	1.5	1.47

*Source of data: E is empirical, from experimental data gathered in the fields; R is reported by farmers; S is standard data. In order to use standard data in diesel fuel use, field management information was sought from farmers.

[#]Yield was adjusted from that reported in Chapter 3, for dry matter losses incurred in producing silage (18% loss) and hay (36% loss). Silage has 25% DM content, hay 86%.

[$]FYM is farmyard manure, organic fertiliser produced from cows

5.3.2 Sheep production data

The next process examined was the production of sheep. Sheep data which were used to calculate emissions from sheep are shown in Table 2. The data were reported by the farmers (R), or standard data (default data contained in the Cool Farm Tool, or from another publicly-available source as specified) (S). The data comprised number of different sheep type per ha^{-1} (R) (juvenile, productive adult, non-productive adult), length of time present (R), amounts of concentrate feed (R and S) and hay feed eaten (R and S). Concentrate feed is supplementary food for livestock, given in times of high energy and protein requirements such as late pregnancy and early lactation. The concentrate feed (85% DM content, 16% crude protein) (S) was assumed to be a mix including cereals, soy hulls and oilseed meal (O'Brien *et al.,* 2014). The amount of concentrate given (S) may vary by farmer, but it was assumed that all farmers gave their ewes concentrate in the last 6 weeks of pregnancy (March-April), at an estimated daily intake of 0.4 kg fresh weight of feed day^{-1} (Hybu Cig Cymru, 2006). Lactating ewes on silage field 'a' and hay field 'a' also received concentrate for an assumed 4 weeks in April-May (R), at approximately 0.9 kg fresh weight of feed day^{-1} (S) (Hybu Cig Cymru, 2006). During winter, sheep were assumed to receive hay feed produced from the farm's hay field. Therefore on hay fields, hay feed production emissions were zero, having been incorporated into the crop emissions footprint. The amount of hay feed eaten when sheep are on the field between mid-December and lambing (0.917 kg hay per ewe per day) was an average of farmer-reported (R) (*pers. comm*. Andrew Hattan) and standard data (S) (Department of Agriculture and Rural Development, 2012). Additional conditions selected in the Cool Farm Tool were: manure management was by grazing (S) (i.e. sheep manure was deposited on the ground during grazing); and all grazing vegetation was classed as medium quality pasture (S).

Table 2. Sheep farming inputs for each field. Ewes are divided into 'productive' and 'non-productive' depending on if they are pregnant/lactating or not, whilst on the field†.

		Sheep type	Number animals	Time on field (months)	Conc. feed eaten (freshweight kg ha^{-1})$	Hay feed eaten on silage fields (kg ha^{-1})♦
Data source*		R	R	R	R and S	R and S
Silage fields	a	Juvenile#	8	4		
		Productive adult†	10	2.5	278	0
		Non-productive adult!	0	0		
	b	Juvenile	0	0		
		Productive adult	0	0	0	1347
		Non-productive adult	26	5		
	c	Juvenile	15	7.5		
		Productive adult	12	6	223	1320
		Non-productive adult	12	0.5		
	d	Juvenile	15	3		
		Productive adult	9	3.4	42	223
		Non-productive adult	9	1.1		
	e	Juvenile	15	2.5		
		Productive adult	8	1.5	0	0
		Non-productive adult	0	0		
Hay fields	a	Juvenile	8	4		
		Productive adult	10	2.5	278	
		Non-productive adult	0	0		
	b	Juvenile	12	2		
		Productive adult	6	6	109	
		Non-productive adult	0	0		
	c	Juvenile	15	7.5		
		Productive adult	12	6	223	
		Non-productive adult	12	0.5		
	d	Juvenile	21	3		
		Productive adult	11	3.4	49	
		Non-productive adult	11	0.1		
	e	Juvenile	9	2.5		
		Productive adult	4	1.5	0	
		Non-productive adult	0	0		

#Juvenile sheep is a lamb which is destined for sale at < 1 year old

!Non-productive adult is sheep over 1 year old (including ewes) which is not pregnant or lactating.

$Concentrate feed is fed to productive adults in March to May only

♦Only hay feed consumed on silage fields was considered because they don't produce hay. Hay fields produced more hay than was consumed on them: their hay feed emissions are embedded in crop emissions.

*Source of data: R is reported by farmers; S is standard data.

5.3.3 Data used to calculate emissions from biomethane electricity

Transport was assumed to be required for an 8-mile round trip to 7 farms in the co-operative, when transporting feedstock and digestate. (Assumptions of distance between AD plant and farm are shown in Chapter 6, Section 6.2.2.2.) GHG emissions due to transport were estimated using the online version of the Cool Farm Tool. Dairy AD required transport of feedstock to the AD plant (an 8-mile round trip); except, when digesting 'own' crops, dairy AD required no off-farm transport. The amount of electricity produced in each of the AD scenarios is shown in Tables 6 and 7 in Chapter 6.

5.3.4 Emission factors used by the Cool Farm Tool

The GHG emissions were calculated by the Cool Farm Tool using IPCC emission factors (Table 3) (van Tonder and Hillier, 2014): either Tier 1 global emission factors, or country-specific Tier 2 emission factors (applicable to the UK). Specifically, the inorganic fertiliser production emission factors are the results of LCAs of European fertiliser production (van Tonder and Hillier, 2014). Tier 1 emission factors were used to calculate sheep enteric fermentation as is currently done in the UK National Inventory Report (Brown *et al*., 2016). Emissions from soils were split into (i) background emissions and (ii) emissions due to fertiliser, FYM and lime application. Soil emissions were calculated by the Tool using Tier 1 and Tier 2 emission factors and equations, and Bouwman *et al*. (2002) and Food and Agriculture Organisation *et al*. (2001) models. Background soil emissions (N_2O) depended on vegetation type, soil pH, soil moisture, soil texture, soil drainage and soil organic carbon content. Inorganic fertiliser produced soil emissions of N_2O (directly and via volatilisation of NH_3 and NO_x, and from leached N); FYM released N_2O and CH_4; and lime caused the release of CO_2 (van Tonder and Hillier, 2014). GHG emitted by transport of feedstocks/digestate between the farms and AD plant was included in biomethane electricity production.

Table 3. Emission factors used in calculations of GHG emissions (as used by the Cool Farm Tool, except for emissions from concentrate feed and hay feed production – see notes below Table)

Input		Unit	Emission factor
Inorganic fertiliser production	NPK$ 20:10:10	kg CO_2 kg^{-1}	0.79
	NPK$ 25:5:5	kg CO_2 kg^{-1}	0.94
Limestone production	55% $CaCO_3$ / 29% CaO	kg CO_2 kg^{-1}	0.06
Fuel combustion	Diesel	kg CO_2 $litre^{-1}$	2.68
Sheep enteric fermentation		kg CH_4 $head^{-1}$ $year^{-1}$	8
Concentrate feed production*	16% crude protein	kg CO_2 kg^{-1} DM	0.34
Hay feed production$		kg CO_2e t^{-1} fresh hay	133
Transport	HGV	kg CO2e t-1 mile-1	0.2

$N comprises equal amounts of ammonium-N and nitrate-N

*From O'Brien *et al*. (2014)

$From this research

5.4 LIFE CYCLE IMPACT ASSESSMENT

The life cycle impact assessment traditionally describes the conversions of individual GHGs to carbon dioxide equivalent and the results of the LCA. I am also going to include the results of the sensitivity analyses in this Section. Usually they may go in the Interpretation (the last part of the LCA), but the effect of sheep on each functional unit needs to be examined (especially since sheep may be removed from the land) before the functional units can be understood. Therefore I have included the sensitivity analyses here.

5.4.1 Conversion factors to carbon dioxide equivalent

The Cool Farm Tool reports emissions for the three GHGs most relevant to agriculture: CH_4, N_2O and CO_2 (van Tonder and Hillier, 2014). In order to assess the impact of their different global warming impacts, they were converted into a common unit, CO_2 equivalent (CO_2e) using the IPCC (2007) global warming potential values from a 100 year time period: 1 kg CO_2 = 1 kg CO_2e, 1 kg CH_4 = 25 kg CO_2e and 1 kg N_2O = 298 kg CO_2e (IPCC, 2007).

5.4.2 GHG emissions due to land management (per ha)

Mean land management emissions (which were due to silage/hay and sheep production, measured per ha) were 17% higher in silage fields than hay fields, but the difference was not significant (Table 4). The higher emissions from silage land were due to both higher sheep emissions and silage production (which has higher inputs than hay). Land management emissions were mainly due to sheep enteric fermentation (producing CH_4) and sheep manure (deposited during grazing, producing N_2O and CH_4). Together these contributed 88% and 93% of land management emissions in silage and hay fields respectively (Fig. 3). Total emissions from sheep (which included the low level CO_2 emissions from production of concentrate feed and hay feed) varied widely across silage and hay fields due to the variation in sheep age (because lambs received less feed than adult ewes), number of sheep ha^{-1}, stage of ewe pregnancy (because ewes received more feed in later pregnancy), and length of time on the field $year^{-1}$.

Table 4. Comparison of silage and hay emissions, including sheep (for land management and crop productivity).

Emissions (kg CO_2e)	Mean		Min, Max		SE		Silage vs hay†	
	Silage	Hay	Silage	Hay	Silage	Hay	$F_{1,9}$	*P*
Land management (per ha)	6042	5170	3186, 8870	2555, 8411	1019	1308	0.276	0.613
Crop productivity (per t DM* silage or hay)	2314	2023	875, 4160	969, 3171	632	460	0.139	0.719

†Statistical test: 1-way ANOVA

*Adjusted for dry matter (DM) losses in silage- and hay-making (18% and 36% respectively).

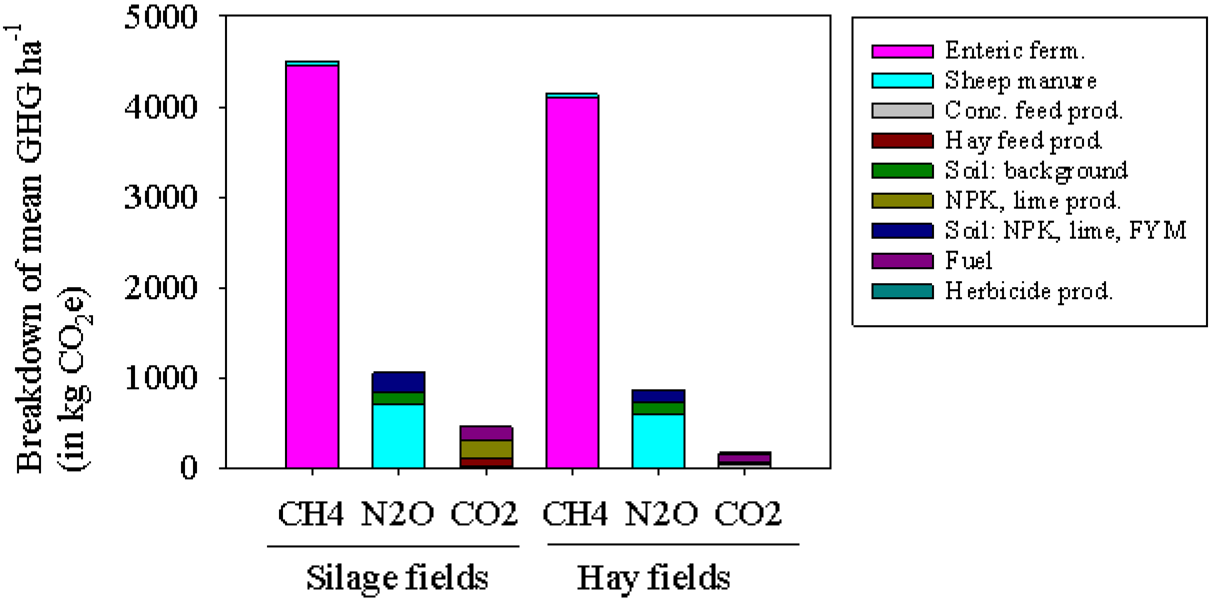

Figure 3. Sources of GHG emissions ha^{-1} in silage fields and hay fields including sheep, showing CH_4, N_2O and CO_2 converted into CO_2 equivalent. Enteric ferm is enteric fermentation in sheep; sheep manure is deposited on soil during grazing; conc is concentrate; prod is production; background soil emissions are due to soil type, soil organic matter, soil pH, moisture, drainage and crop growing; soil: NPK, lime, FYM is soil emissions after their application.

5.4.2.1 Sensitivity analysis of land management emissions

Sensitivity analysis of a single silage field and a single hay field showed that sheep numbers had the largest effect on land management emissions (Table 5). If sheep were removed from the land, *mean* land management emissions of all silage fields and all hay fields would drop to 719 (SE 25) and 385 (SE 20) kg CO_2e ha^{-1} respectively (compare to Table 5, see Figure 4). The silage and hay emissions without sheep were significantly different to each other (1-way ANOVA, $F_{1,9} = 109.5$, $P < 0.001$). Fertiliser had the second strongest effect on land management emissions: inorganic NPK in the silage field, and organic FYM in the hay field. A change in silage/hay yield had little effect on emissions per ha because it only affected the amount of fuel used during baling. Table 5A in the Appendix shows the breakdown of GHG from land management per field.

Table 5. Sensitivity analysis showing GHG from land management per ha (kg CO_2e ha^{-1}) for one silage field (Table i) and one hay field (Table ii) when a single input is increased or decreased by 10%, 20% or 100%.

(i) Silage field

Change	Sheep number	Inorg. Fertiliser (NPK)	Farmyard manure	Soil organic matter	Lime	Soil pH	Silage yield	Sheep conc. feed
+ 100%	12750	7039	6888	6831	6827	6798	6775	6763
+ 20%	7994	6807	6778	6749	6765	6772	6757	6752
+ 10%	7260	6778	6764	6749	6757	6749	6753	6751
0%	6749	6749	6749	6749	6749	6749	6749	6749
- 10%	6016	6721	6737	6749	6742	6749	6745	6748
- 20%	5505	6692	6723	6749	6734	6749	6741	6746
- 100%	772	6466	6615	6671	6671	6749	6709	6735

(ii) Hay field

Change	Sheep number	Farmyard manure	Soil organic matter	Soil pH	Lime	Sheep conc. feed	Hay yield
+ 100%	7308	3994	3943	3915	3912	3910	3886
+ 20%	4658	3896	3873	3893	3881	3881	3877
+ 10%	4229	3884	3873	3893	3877	3877	3876
0%	3873	3873	3873	3873	3873	3873	3873
- 10%	3512	3860	3873	3873	3870	3870	3871
- 20%	3298	3848	3873	3873	3865	3866	3869
- 100%	396	3752	3843	3873	3835	3836	3850

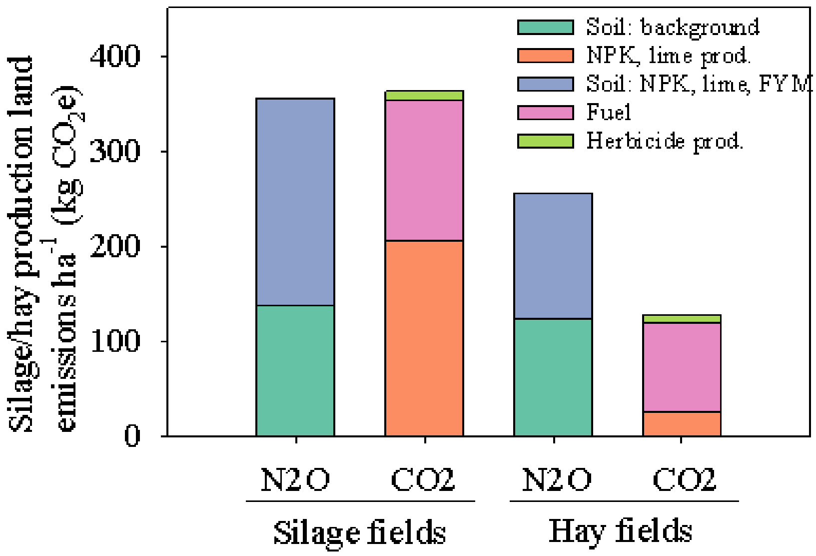

Figure 4. Breakdown of GHG from land management (mean) on silage and hay land, when sheep have been removed, showing N_2O and CO_2 expressed as CO_2 equivalent. Prod is production; background soil emissions are due to soil type, soil organic matter, soil pH, moisture, drainage and crop growing; soil: NPK, lime, FYM is soil emissions after their application.

If sheep were removed, silage land management emissions were evenly split between N_2O and CO_2; whereas hay land emitted less N_2O (due to absence of NPK on soil), and even less CO_2 (due to absence of NPK production) (Figure 4).

5.4.3 Silage and hay productivity emissions (per t dry matter)

Of course, it is important to explore the benefits of inorganic fertiliser (applied to the silage land) on yield. Although not significantly different, productivity emissions (t^{-1} DM) silage were 14% higher than hay when sheep were included (Table 4).

5.4.3.1 Sensitivity analysis of productivity emissions

Sensitivity analysis of silage field and hay field showed that silage productivity emissions were most sensitive to sheep numbers (Table 6). If sheep were removed from the land, *mean* productivity emissions of all silage fields and all hay fields would fall to 255 and 155 kg CO_2e t^{-1} DM silage and hay, respectively (compare to Table 4); a 9- and 13-fold reduction for silage and hay respectively. These were significantly different (1-way ANOVA, $F_{1,9} = 7.322$, P 0.027). Changing the amount of DM lost whilst making silage had the second largest effect on silage productivity GHG. Higher yield of silage, made from the same starting amount of fresh grass (by losing less DM during processing), gave lower emissions t^{-1} DM. Hay productivity sensitivity was slightly different, at least for the single hay field analysed: DM loss was more influential than sheep numbers, probably because hay-making incurs a much higher DM loss (36%) than silage-making (18%). For both silage and hay, fertiliser was the third most influential variable: inorganic fertiliser in silage fields, and FYM in hay fields.

Table 6. Sensitivity analysis showing productivity emissions per t dry matter (kg CO_2e t^{-1} DM) for one silage field (Table i) and one hay field (Table ii) when a single input is increased or decreased by 10%, 20% or 100%.

(i) Silage field

Change	Sheep numbers	DM loss	Inorg. Fertiliser (NPK)	Farmyard manure	Soil organic matter	Lime	Soil pH	Sheep conc feed
+ 100%	4546	3083	2510	2456	2436	2434	2424	2412
+ 20%	2850	2517	2427	2417	2407	2412	2415	2408
+ 10%	2589	2461	2417	2412	2407	2410	2407	2407
0%	2407	2407	2407	2407	2407	2407	2407	2407
- 10%	2145	2355	2396	2402	2407	2404	2407	2406
- 20%	1963	2305	2386	2397	2407	2401	2407	2406
- 100%	275	1973	2306	2359	2379	2379	2407	2402

(ii) Hay field

Change	DM loss	Sheep numbers	Farmyard manure	Soil organic matter	Soil pH	Lime	Sheep conc feed
+ 100%	3505	2893	1582	1561	1550	1549	1548
+ 20%	1728	1844	1543	1534	1541	1537	1536
+ 10%	1625	1674	1538	1534	1541	1535	1535
0%	1534	1534	1534	1534	1534	1534	1534
- 10%	1452	1391	1528	1534	1534	1532	1532
- 20%	1378	1306	1523	1534	1534	1530	1531
- 100%	981	157	1485	1522	1534	1518	1519

5.4.4 GHG emissions of 1kWh electricity from biomethane

This Section first describes the effect of sheep on GHG per unit of biomethane electricity, and then goes on to report emissions when sheep are assumed to have been removed from the land.

When the current numbers of sheep on silage/hay land were included, GHG from biomethane electricity saved only 3-4% GHG compared to EU fossil fuel electricity (Table 7). This is well below the (2017) EU sustainability requirement of a minimum of 50% GHG saving on electricity produced by bioenergy compared to fossil fuel electricity.

Table 7. Sensitivity analysis of the effect of changing sheep numbers, on GHG from biomethane electricity (in kg CO_2eq $kWhe^{-1}$) from two AD scenarios (co-operative increased silage; and co-operative increased hay). The Table also shows the % GHG saved compared to EU fossil fuel electricity (assuming 100% heat use).

Change in sheep numbers	Co-operative AD scenario		% GHG saving compared to EU fossil fuel elec.*	
	Increased silage	Increased hay	Increased silage	Increased hay
+ 100%	1.29	1.32	-81	-86
+ 20%	0.81	0.82	-13	-14
+ 10%	0.75	0.75	-5	-6
0%	0.69	0.69	4	3
- 10%	0.63	0.63	12	12
- 20%	0.57	0.56	20	21
-30%	0.51	0.50	29	30
-40%	0.45	0.44	37	39
-50%	0.39	0.37	46	48
-60%	0.33	0.31	54	57
-70%	0.27	0.25	63	66
-80%	0.21	0.18	71	74
-90%	0.15	0.12	79	83
- 100%	0.09	0.06	88	92

*GHG from EU fossil fuel electricity = 0.713 kg CO_2eq kWh $elec^{-1}$ (EC COM(2010))

However, if sheep numbers were reduced by 60% or they were removed entirely from the land, sufficient GHG savings could be made (Table 7). Therefore the rest of this section explores GHG of electricity from biomethane assuming that sheep are removed from the silage/hay land.

When all AD scenarios were examined excluding sheep (Figure 5), GHG per unit of electricity from biomethane was significantly lower for hay than silage, in like scenarios (Figures 5 and 7) (1-way ANOVA, $F_{1,9} = 9.408$, P 0.015). GHG savings of > 50% were possible in every AD scenario (compared to EU fossil fuel electricity (0.713 kg CO_2eq $kWhe^{-1}$ (European Commission, 2010), and compared to the UK's electricity mix (coal, gas, renewable, nuclear: 0.22 kg CO_2eq $kWhe^{-1}$ (Department for Business, Energy and Industrial Strategy, 2017)) if sheep were removed from the silage/hay land (Figure 6). Electricity from hay biomethane had greater GHG savings than electricity from silage, in like scenarios. Biomethane electricity emissions were also lower than UK coal electricity

(0.88 kg CO_2e $kWhe^{-1}$) and UK gas electricity (0.33 kg CO_2e $kWhe^{-1}$) (Department for Business, Energy and Industrial Strategy, 2017).

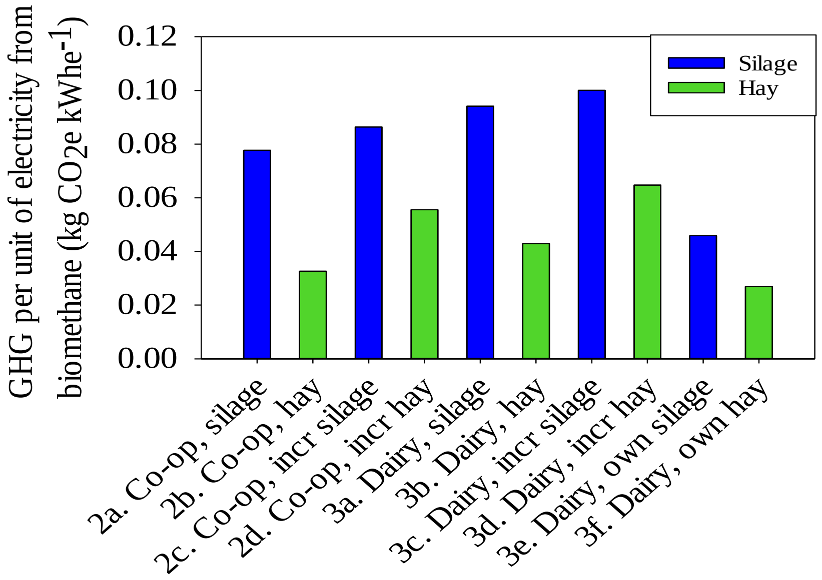

Figure 5. GHG emissions per unit of electricity (kWhe) from biomethane, for each anaerobic digestion scenario. Emissions from sheep were excluded.

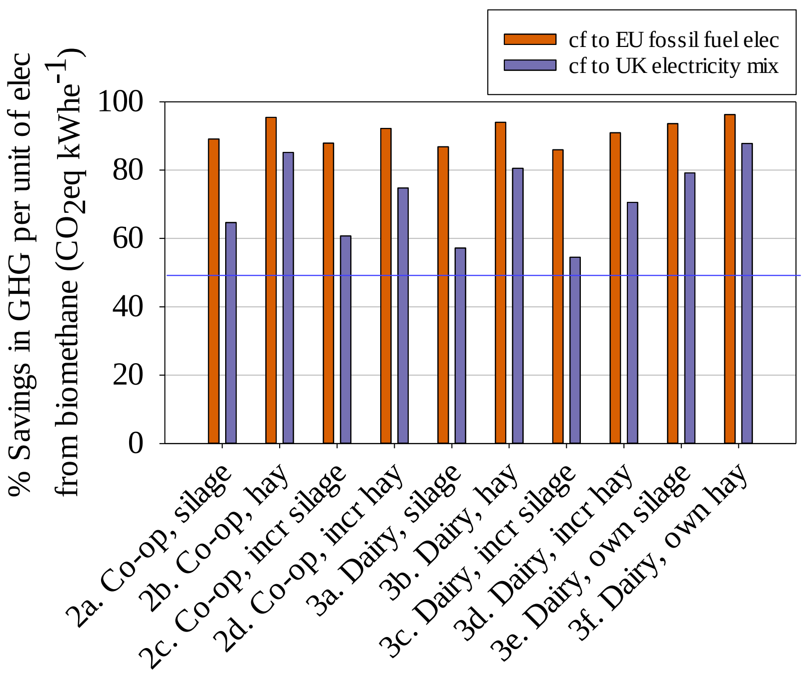

Figure 6. Percentage GHG savings made by electricity from biomethane compared to EU fossil fuel electricity (0.713) and the UK electricity grid mix (0.22) , for each anaerobic digestion scenario. Blue line on graph shows 50% GHG savings compared to EU fossil fuel electricity. Emissions from sheep were excluded.

5.4.4.1 Sensitivity analysis of GHG per unit of electricity from biomethane (emissions from sheep excluded)

The above Section (5.4.4) assumed that GHG were allocated between electricity and all the heat produced; but in the models of AD only 36-56% of heat was used. Sensitivity analysis showed that using no heat (and therefore allocating all GHG to electricity) more than doubled GHG per unit of electricity (Table 8) (emissions from sheep were excluded). Inorganic fertiliser (in silage AD) and FYM (in hay AD) were the second most influential variables. Transport was the least influential variable, despite the scenarios in the sensitivity analyses having larger transport inputs than other scenarios (Table 5B in the Appendix).

Table 8. Sensitivity analysis showing GHG per unit of electricity from biomethane (kg CO_2e $kWhe^{-1}$) for a silage AD scenario (Table i) and a hay AD scenario (Table ii) when a single input is changed by 10%, 20% or 100%. Uses mean silage/hay production data; and assumes sheep are excluded.

(i) Co-operative AD, increased silage

Change	% Heat used	Inorg. Fertiliser (NPK)	Farmyard manure	Soil organic matter	Lime	Transport
+ 100%	0.09	0.12	0.10	0.10	0.09	0.09
+ 20%	0.09	0.09	0.09	0.09	0.09	0.09
+ 10%	0.09	0.09	0.09	0.09	0.09	0.09
0%	0.09	0.09	0.09	0.09	0.09	0.09
- 10%	0.09	0.08	0.09	0.09	0.09	0.09
- 20%	0.10	0.08	0.08	0.09	0.08	0.09
- 100%	0.20	0.06	0.07	0.08	0.08	0.08

(ii) Co-operative AD, increased hay

Change	% Heat used	Farmyard manure	Soil organic matter	Lime	Transport
+ 100%	0.06	0.07	0.06	0.06	0.06
+ 20%	0.06	0.06	0.06	0.06	0.06
+ 10%	0.06	0.06	0.06	0.06	0.06
0%	0.06	0.06	0.06	0.06	0.06
- 10%	0.06	0.05	0.06	0.06	0.06
- 20%	0.06	0.05	0.06	0.05	0.05
- 100%	0.13	0.04	0.05	0.05	0.05

Table assumes GHG are allocated between electricity and heat used (1 kWhe = 1kWh thermal). Baseline heat use is 100% heat used, thus increasing % heat used does not affect GHG $kWhe^{-1}$.

5.4.5 GHG per unit of heat

Although not a functional unit studied in this research, GHG emissions of AD were allocated to heat as well as electricity, when heat was used. Because GHG emissions were allocated by energy content, emissions from 1 kWh of heat were the same as that from 1 kWh of electricity. Emissions per unit of electricity/heat are shown in Table 5B (Appendix) for each AD scenario. A similar pattern of GHG savings was observed in GHG per unit of heat, as was seen in GHG per unit of electricity: when all heat was used, heat from biomethane emitted 68-91% less GHG than heat from EU fossil fuel heat (0.3132 kg CO_2e kWh^{-1} heat; European Commission, 2010. And in all AD scenarios, GHG from heat was below the maximum limit to be eligible for the UK's heat subsidy, Renewable Heat Incentive (0.1253 kg CO_2e kWh^{-1} heat). Heat from hay biomethane also had higher GHG savings than heat from silage biomethane.

5.4.6 Summary of comparison between hay and silage

In summary, when sheep are excluded from the field, hay has significantly lower GHG emissions than silage in all the functional units examined: per t of dry matter; per ha of land; and per kWh of biomethane electricity produced by AD (Figure 7a and b).

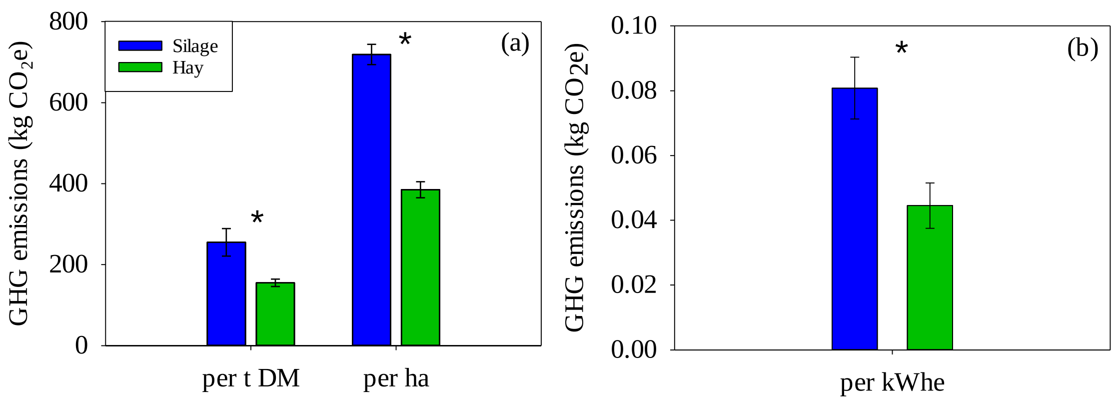

Figure 7a and b. Mean GHG emission of each functional unit of silage and hay, excluding sheep. (a) shows mean GHG emissions for the 5 silage fields, and 5 hay fields, per t DM and per ha; (b) shows mean GHG per kWh of biomethane electricity of the 5 AD scenarios digesting silage, and the 5 AD scenarios digesting hay. Error bars are SE (n = 5). *denotes significant difference.

5.5 INTERPRETATION

The Interpretation Section discusses the results, and what they mean.

GHG emissions from land management (per ha) and productivity (per t DM) were non-significantly 17% and 14% higher for silage (than hay) when sheep were included. In both land and productivity GHG, sheep contributed most of the GHG; such that if sheep were removed from the silage/hay land, land management and productivity emissions dropped substantially, and hay emissions became significantly lower than hay. Hay emissions per t DM were also significantly lower than for silage. If sheep were maintained on silage/hay land whilst the crop was used for bioenergy production (by anaerobic digestion which produces biomethane), GHG of electricity from biomethane were only 3-4% lower than emissions from EU fossil fuel electricity. However if the farmers wanted to remove the sheep, GHG per unit of electricity from biomethane could provide 86-96% GHG savings compared to EU fossil fuel electricity. Emissions per unit of biomethane electricity made from hay were also significantly lower than electricity from silage, with biomethane from hay providing larger electricity GHG savings than that from silage. Therefore the commonly-used anaerobic digestion feedstock, grass silage, could be swapped for hay, and biodiversity would be maintained. The method of GHG allocation between electricity and heat had a large effect on GHG per unit of electricity, such that when all GHG were allocated to electricity (assuming none of the useable heat produced by the combined heat and power plant is used), GHG per unit of electricity doubled compared to when all the heat is used (where GHG are allocated between electricity and all heat).

5.5.1 GHG emissions due to land management (per ha)

Emissions from sheep varied widely across fields, and they contributed a much higher proportion of GHG than the inputs into silage or hay production. When sheep were removed in the sensitivity analysis, land management GHG became significantly higher on silage than hay land, reflecting the higher intensity of land use. Silage land received NPK (emitting CO_2 during production, and N_2O after application to soil) which was the second most influential variable in GHG per ha. More fuel was also used on silage land (emitting CO_2) for spreading NPK and wrapping silage bales. This may have implications for other environmental impacts, such as leaching of N (eutrophication), acidification or respiratory inorganics; but these were not studied here. Tian (2014) predicted that, due to increasing temperature and fertiliser use, soil emissions of N_2O could increase the most in the future from crops grown in higher altitude (such as the fields studied here). Therefore encouraging reduced nutrient input into grasslands (by encouraging more species-rich hay land) could help reduce the escalation in N_2O release. The association of lower intensity land use and lower GHG emissions per area has been observed by Goglio *et al.* (2012) in Mediterranean arable crops; by Casey and Holden (2006) in Irish beef production; and by Mueller *et al.,* 2014 in Swedish organic milk farming. The result also supports the findings of Goglio *et al*. (2014) that a reduction in N-fertilisation has a substantial impact on GHG emissions, but can have little impact on crop yield. There are few studies on GHG emissions from silage or hay production on marginal land with which to compare, because most marginal land LCAs focus on livestock production and their GHG emissions are usually presented per kg animal product. However, extensive hay production in Switzerland produced 228 kg CO_2e ha^{-1} (Nemecek *et al*., 2011), but no manure was added to the land which may explain why it is lower than the average non-fertilised field in this study (385 kg CO_2e ha^{-1}). Silage production on a less-intensively managed Welsh sheep farm (Edwards-Jones *et al.*, 2009) emitted a median of 274 kg CO_2e ha^{-1} (range 103-1459) excluding emissions from sheep. This range encompasses both the silage fields (mean 719 kg CO_2e ha^{-1}, range 657-772) and hay fields (mean 385 kg CO_2e ha^{-1}, range 338-446) in this study when emissions from sheep are excluded. But comparisons with published papers should be viewed with caution due to differences in emission factors and LCA methods used.

At the time of this research, the studied silage and hay fields were managed for sheep farming and silage/hay production, but globally livestock are being removed from grasslands (Secretariat of the Convention on Biological Diversity, 2014). If that trend was to hit the Yorkshire Dales, and the farmer was forced to change land management, for example to bioenergy production with no or fewer sheep, it would substantially reduce GHG emissions per ha (discussed further in Section 5.5.6).

5.5.2 Productivity emissions (per tonne dry matter of silage or hay)

Productivity GHG (per tonne dry matter of silage or hay, including DM losses during silage- or hay-making) allowed any benefits of inorganic fertiliser on yield to be examined, but a similar picture emerged as was seen in land management emissions. When emissions from sheep were excluded, the slightly higher productivity of silage fields (due to lower DM losses than when making hay) did not outweigh the higher silage emissions (due to the inclusion of inorganic fertiliser), thus silage dry matter had significantly higher GHG emissions than hay.

Productivity GHG were also sensitive to the amount of DM lost during silage- or hay-making. Hay GHG was particularly sensitive, probably because hay has bigger DM losses than silage. DM losses used in the literature vary: for silage, 5% (Jury *et al.*, 2010), 12% (Richter *et al.*, 2011) and 18% as used in this work (Buhle *et al.*, 2012); and for hay, 12, 18, 30% (Richter *et al.*, 2010) and 36% as used in this work (Buhle *et al.*, 2012). In this research, higher yield of silage (by losing less DM) gave lower emissions t^{-1} DM, which is commonly reported. For example, Gerber *et al*. (2011) reported that emissions per unit of product can fall with increasing productivity per ha; and high-input farming (with high productivity) can compensate for the higher emissions from fertiliser inputs (Elshout *et al.*, 2015).

If sheep were removed from the land, biomass yield would increase due to less being eaten during grazing. This would have little effect on GHG per ha (Table 5) but because the amount of dry matter produced would increase, GHG per t DM would decrease. However cutting N-fertilisation rate may be necessary to reduce emissions per ha in arable crops, but it mustn't be cut by too much otherwise emissions per tonne of crop can increase by reducing yield per ha (Nemecek *et al.*, 2015; Kulak *et al.*, 2013). Therefore the relationship between emissions per unit and productivity per ha is not always clear-cut, and is based on a complex balance between inputs and yield (Nemecek *et al.*, 2011). The GHG emissions per tonne DM reported here are lower than those calculated by Styles *et al*. (2016) (390 kg CO_2e t^{-1} DM for silage; 255 in the present study) and Styles *et al*. (2015a) (528 kg CO_2e t^{-1} DM for hay; 155 in the present study) presumably because the Styles *et al.* papers assumed fertiliser and management inputs appropriate to an (intensive) arable or dairy farm, which would have higher emissions due to higher fertilisation rate and number of cuts per year.

5.5.3 GHG emissions per unit of electricity from biomethane

Withdrawal of livestock from, and abandonment of, agricultural land is an increasing problem globally (Secretariat of the Convention on Biological Diversity, 2014; Stoate *et al.* 2009), and this research works on the premise that abandonment of the marginal areas examined is a possibility, with consequences which would be detrimental to grassland biodiversity (Isselstein, 2005). When sheep were included, GHG per unit of electricity from biomethane was almost the same as GHG from fossil fuel electricity. But, when it was assumed that livestock were removed from the land, biomethane electricity provided substantial savings in GHG compared to fossil fuel electricity, well in excess of the 50% and 60% required by the EU in 2017 and 2018, respectively (European Commission, 2010). All the AD scenarios also exceeded 50% savings compared to the UK's electricity mix (which includes nuclear and renewables) when sheep were removed. As reported in Chapter 6, silage produced slightly more electricity per t DM (872 kWhe) than hay (835 kWhe t^{-1} DM). However, silage produced proportionally more GHG per t DM than hay, therefore hay provided slightly higher GHG savings in biomethane electricity than silage.

5.5.4 Use of the heat produced

In the models of AD, heat produced by burning biomethane was assumed to be used to heat the AD tank and the farmhouse. This equated to only 36-56% of heat produced depending on the AD scenario. Even though the sensitivity analysis (Table 8) showed that emissions per unit of electricity were very sensitive to heat use, if no heat was used and all GHG were allocated to electricity, the scenarios examined here could still meet the EU's sustainability requirements. However, emissions would be too high to meet the requirements, and funding, of the UK's heat subsidy. If heat, rather than electricity, production was the main purpose of a renewable energy system, combustion of hay provides larger savings in GHG than anaerobic digestion, but it does not provide recycling of nutrients via digestate (Buhle *et al.*, 2012). Rosch *et al.* (2009) recommend that burning hay is better for biodiversity conservation then digesting grass silage, due to the higher species richness in hay land. Year-round uses of the heat would be needed, however, which may be difficult to find in isolated marginal areas. Users of heat could include horticultural greenhouses, farm cottages, or homes in a nearby village.

5.5.5 Comparison to other renewable energies

The 86-96% GHG savings by biomethane electricity (from silage and hay) are comparable to 90% and 95% savings achievable by burning Miscanthus or wheat straw, respectively (European Commission, 2010). Bacenetti and Fiala (2015) reported that, compared to Italian grid electricity (which emitted 0.5417 kg CO_2eq $kWhe^{-1}$), on-farm AD of crops and/or manures could save 0.263 to 0.476 kg CO_2eq $kWhe^{-1}$ when GHG credits for digesting manures were not applied. When the biomethane electricity emissions reported here were compared to Italian grid electricity, the savings were similar (0.44 to 0.51 kg CO_2e $kWhe^{-1}$). Thus anaerobic digestion of marginal grassland crop could compete, on a GHG basis, with established bioenergy processes. However, bioenergy has higher GHG emissions per unit of electricity produced than other renewable/non-fossil fuel energies. Gibon *et al*. (2017) compared GHG emissions of electricity production from renewable (e.g. bioenergy, solar, wind) and nuclear generation technologies (Table 9). They reported that bioenergy production (by gasification of woody energy crops and forest residue) has the highest electricity emissions of all the technologies examined. GHG from biomethane electricity (reported here) was within their range of bioenergy GHG, although they included GHG from the manufacture of each technology, which if included in this research would have increased GHG by an undetermined amount.

Table 9. GHG emissions per unit electricity of different renewable and nuclear electricity generation technologies, as assessed by Gibon *et al*. (2017)

	GHG per unit electricity (kg CO_2e kWh^{-1})
Biomass burning	0.04 - 0.2
Photovoltaics	0.015 - 0.08
Wind, offshore	0.015 - 0.02
Wind, onshore	0.008 - 0.025
Geothermal	0.008 - 0.012
Nuclear	0.006 - 0.01
Hydroelectric	0.005 - 0.23
Biomethane*	0.027-0.100

*Electricity from biomethane was estimated in this research

5.5.6 Removing sheep from the land

Sozanska-Stanton *et al.* (2016) recommend removing or reducing livestock numbers on biodiverse grasslands to reduce livestock and soil emissions; but they recognise that this may have adverse effects on biodiversity (considered in Chapter 3). Livestock compress soil and deposit N-containing excreta which increase denitrification (producing N_2O); and the manure contains methanogens causing increased soil CH_4 production (Sozanska-Stanton *et al.* 2016). If a reduced number of sheep were to remain on the land, and the harvested crop was used partly for bioenergy production, and there was sufficient remaining for their winter feed, the GHG for silage/hay production could be considered 'free' in bioenergy production since silage/hay production would be happening for the animals anyway. If sheep were to remain on the land in current numbers, the amount of harvestable biomass would remain the same. However production of the sheep's winter feed would have to be displaced elsewhere. If produced somewhere else on the farm without land use change, the electricity GHG savings could be maintained, but if more concentrate feed was imported (for winter feed) the GHG savings could be obliterated, as reported by Styles *et al.* (2015a) and cost to the farmer would increase. If the farmer decided to remove all sheep from the land, GHG from biomethane electricity would substantially reduce, allowing it to qualify for financial support under EU sustainability rules (European Commission, 2010).

However, the process of allocating GHG to different outputs is subjective. Thus, an alternative to allocating emissions from sheep to the crop and AD, would be to allocate by economic value of the sheep and the biomethane electricity, such that if sheep were worth e.g. twice as much as electricity, they would be allocated twice the GHG as the electricity. This is called economic allocation. However, the current method of allocating sheep emissions to crop (and thus AD), and the use of sensitivity analyses, brought an understanding of the effect of the emissions from sheep. In future work, emissions from sheep could be allocated entirely separately to sheep (although there are overlaps in inputs to grassland production and sheep, e.g. FYM is needed to give an adequate grass yield for both grazing by sheep, and harvested crop production). Alternatively, there could be economic allocation of emissions between sheep and electricity.

5.5.7 Limitations

If a holistic understanding of the process was desirable, an LCA should also include impacts other than global warming potential. Due to using a carbon calculator, the GHG emissions reported here are estimates only, based on a mix of empirical farm data and default data provided by the Cool Farm Tool. It is beneficial that empirical data are included in the analysis because land management can vary widely between farms, as was indeed observed; for example in the number of sheep per ha which had large effects on emissions. The data represent one year only, but field management and crop yields can change over time especially on fields not governed by agri-environment schemes. However Hyland *et al.* (2016) found no significant differences when carbon footprints of sheep and/or beef farming were repeated over time.

5.5.8 Conclusion

When it was assumed there were no sheep on the land, hay had significantly lower emissions than silage in every functional unit examined, i.e. lower GHG per ha, lower GHG per t of dried crop, and lower GHG per kWh of electricity produced by AD. Hay also produced higher GHG savings in electricity production than silage. Biomethane electricity could also provide sufficient GHG savings compared to fossil fuel electricity to meet EU sustainability requirements, if sheep numbers were reduced or removed. In terms of GHG emissions from using hay or silage in AD in marginal areas, species-poor grass silage could be swapped for hay and biodiversity would be maintained, whilst GHG emissions are reduced.

Chapter 6: The profitability and policy of maintaining biodiverse hay land through different uses of the biomass

ABSTRACT

In this Chapter I examined the financial implications of a range of land use options for farmers in marginal areas which could encourage maintenance of high biodiversity grasslands. The aim of the work was to investigate whether different bioenergy scenarios could provide farmers with an alternative to low-income sheep farming, and avoid land abandonment and resulting biodiversity loss. The economic potential of biodiverse hay land was compared with that of higher GHG-emitting, lower biodiversity silage land in three different models: (1) sheep farming (the current land management, which includes silage or hay production); (2) co-operative anaerobic digestion (AD) (produce silage/hay for use in a co-operative AD where co-owner farmers share profits and digestate as fertiliser); and (3) dairy farm AD (dairy is the recommended farm-type for low-cost AD). I found that hay performed better financially than silage in AD, particularly when hay production was increased. Before public money for land subsidy/payments was included, digestion of hay in the dairy AD model could provide a means to maintain biodiverse hay grassland, although it was too low to secure funding. (Public money for energy subsidies was included in every AD scenario).

The picture changed when land subsidy and environmental payments were included, such that the co-operative AD model became more profitable than dairy AD. When agri-environment payments were replaced by societal benefits on hay land (of higher wildlife and GHG saved), co-operative AD of increased hay was more profitable (per ha of land growing the AD feedstock) than selling hay or producing sheep on hay land at current land subsidy/payment levels. Current agri-environment payments for hay land are lower than the societal valuation of species-rich grassland (Boatman *et al.* 2010) justifying an increase. However hay AD was less cost-effective than other bioenergy production systems (although GHG savings from renewable energy and displaced inorganic fertiliser were not included). Before land subsidy/payments were included, financial return of AD was most sensitive to electricity price. Therefore the level and targeting of land subsidy, environmental payments (which could also incorporate societal values) and electricity

subsidies after BREXIT will have a crucial effect on whether AD is a viable option for maintaining biodiverse land.

6.1 INTRODUCTION

In the light of falling biodiversity, efforts are under way in the EU to preserve and increase species-rich grasslands found on marginal (poor quality) agricultural land (Natural England, 2013b), with their associated benefits of improved ecosystem services (Goulson et al., 2015; Carvell *et al*., 2017). Ecosystem services are defined as the “benefits people obtain from ecosystems” (Millennium Ecosystem Assessment, 2005 p. v); these services are derived from a stock of e.g. land, forests and rivers (Mayer, 2016). These grasslands are of international conservation importance (European Communities, 1992), but are increasingly under threat of abandonment (Keenleyside and Tucker, 2010; Abolina and Luzadis, 2015) and degradation (Cramer *et al.*, 2008). Biodiversity underpins ecosystem services (Millennium Ecosystem Assessment, 2005) and if lost, the flow of ecosystem services is reduced (Bjorklund *et al.*, 1999; Biesmeijer *et al.*, 2006; Isbell *et al*., 2011). Although abandonment may benefit landscape biodiversity in some areas, by increasing the mosaic of habitats, it would be detrimental to the biodiverse grasslands studied here because important conservation species would be lost (Keenleyside and Tucker, 2010). Abandonment may occur since the productivity and financial returns from the land are too low (Keenleyside and Tucker, 2010). Helm (2016) describes farming on marginal land as a “precarious economic proposition” and the numbers of farmers doing so have been falling for decades without an obvious answer to reverse the trend.

The work presented in this Chapter asks whether bioenergy production from the grassland crop could provide a viable income for the marginal farmers compared to livestock rearing. Sheep and some cattle production is the mainstay of farmers in marginal areas, but the number of (particularly) sheep in marginal areas have dropped substantially since 2000 (DEFRA, 2011a) (with a peak before 2003 when headage payments per animal had encouraged overproduction (Fraser, 2008)). It has already been shown in this thesis that hay grassland (which is non-fertilised and biodiverse, and thus a target for conservation) can produce the same amount of bioenergy per hectare, by anaerobic digestion, as silage grassland (which is fertilised and more species-poor) (Chapters 3 and 4). The issue investigated here is whether such a farming system is financially viable.

Here, I compare income from sheep farming per ha of silage/hay land with income from AD of the silage or hay (assuming that there are no sheep on the land). It is more common to express sheep in terms of profit per animal (e.g. per breeding ewe) than per ha of land (e.g. Agricultural Budgeting and Costing, 2015; Nix and Redman, 2015). However an area-based financial assessment allows links with valuations of biodiversity, biomethane and greenhouse gas emissions to be investigated, as these were measured per area of land (in Chapters 3, 4 and 5). It is also relevant to area-based farm subsidies.

Farmers currently receive two main types of land-based subsidy/payments from the Common Agricultural Policy: (1) basic payment scheme (BPS), a direct payment received by every farmer for maintaining the land in sound agricultural condition, which supports their income (called 'subsidy' in this research); and (2) additional voluntary agri-environment scheme payments (based on an estimate of income foregone) for farming in a more environmentally-friendly manner than required by the BPS (called 'payments' in this research). However, agricultural land subsidy/payments will most likely change (Helm, 2016; Cressey, 2017) after the UK leaves the EU, bringing great uncertainty. This opens up the possibility that more emphasis might be placed on payments that are intended to deliver environmental benefits to areas with poor agricultural productivity but high cultural, biodiversity or landscape value to society (Helm, 2016), to prevent land abandonment and loss of such benefits. If BPS is no longer paid, it is possible that agri-environment payments may be increased. The level of payment that government is willing to put into maintaining, increasing or restoring species-rich grasslands highlights the national and international importance of these habitats (Critchley *et al*., 2007a). Given how the two payment types may operate in the future (i.e. a possibly reduction or cessation of BPS and a possible continuation of agri-environment schemes), these subsidy/payments were added into the land management and AD scenarios in this study in a step-wise manner (agri-environment scheme only, then agri-environment scheme plus BPS). Financial studies of crop anaerobic digestion (AD) vary on whether they include land subsidy/payments (e.g. Prochnow *et al*., 2009b; Blokhina *et al*., 2011; Blumenstein, *et al*., 2012), or exclude them (e.g. Redman, 2010; Bywater, 2011; Hopwood, 2011); this study examines both.

The silage and hay fields which were studied are currently managed by sheep farming (with silage/hay production for on-farm consumption by livestock), which made it possible to calculate the production cost of silage and hay (whether for livestock or feedstock production costs in AD). They are perennial grasslands which, when extensively managed, have been shown to have greater environmental and financial performance than annually-sown grasses (Golkowska *et al.*, 2016).

In this chapter, I compared three business models of silage or hay land use which could occur in a marginal area:
1) Sheep: the studied land is currently managed for sheep, plus silage or hay production that is consumed by livestock on the same farm;

2) Co-operative: a co-operative of sheep farmers who would collectively own an AD plant and provide it with silage or hay at production cost, as well as farmyard manure for co-digestion which would aid microbial stability;

3) Dairy: a dairy farmer would individually own an AD plant and digest silage or hay bought at market price from the sheep farmers, as well as their own slurry for co-digestion, which would aid microbial stability. The dairy AD model includes further feedstock scenarios whereby the dairy farmer would grow their own silage or hay for AD. Dairy farm AD is recommended as a low-cost model of farm AD (Bywater, 2011).

All details are explained in Methods. Single ownership AD plants on sheep farms were not considered because the grass production from any individual farm would be inadequate to sustain it. The financial return of each AD business was measured by its internal rate of return (IRR, which measures rate of return of an investment and is used to measure and compare profitability of potential investments (Investopaedia, 2017).

The aims of an AD plant should be made clear (Methanogen, 2010). Here, the aspirations would be to:

- produce renewable energy. The biogas would be burnt in a combined heat and power plant (CHP) to make electricity and heat. The electricity would supply the 'parasitic' electricity needs of the AD plant, and displace the farmhouse's electricity use; while the excess electricity could be sold to the national grid. The heat would heat the digester and displace the farm house's heat requirements.

- produce an income. Seek profit per ha of silage or hay feedstock land, which could provide an alternative income to the farmer, reducing the risk of land abandonment;
- produce an organic fertiliser. This would be the digestate (i.e., the fibrous/liquid residue left after AD) which contains nutrients that are more biologically available than undigested manure;
- promote biodiversity. Ongoing management of biodiverse hay land (for AD feedstock) would aim to prevent biodiversity and ecosystem service loss.

Economic valuation of natural capital and ecosystem services may be needed to identify trade-offs between commercial land use and biodiversity (Bateman *et al*., 2014). Future low carbon energy production must take into account its effects on ecosystem services and the wider environment (Holland *et al.*, 2016), because if benefits such as biodiversity are ignored, they may be irretrievably damaged and their stock of natural resources destroyed (HM Treasury, 2011). As well as the market value of the products from the silage/hay land (sheep and wool from sheep farming; and electricity, heat and displaced inorganic fertiliser from AD), species-rich grassland (similar to the hay land) provides other ecosystem services which are of importance to society, but for which there are no market ('sellable') values. These include wild species diversity, pollination, water quality, soil quality, cultural, climate and hazard regulation and air quality services (UK National Ecosystem Assessment, 2011). In this thesis, hay land had higher plant biodiversity and lower GHG emissions than silage land (Chapters 3 and 5): these were assigned economic values (to society), even though estimates of the value of biodiversity to society are particularly contentious (Helm and Hepburn, 2012, Bateman *et al*., 2014). Methods for valuing biodiversity and GHG are described in Section 6.3.3.

This Chapter specifically asks –

i) Which grassland feedstock is more financially viable in AD: silage or hay?

ii) Which business model of AD ownership is more financially viable: a co-operative of sheep farmers, or a single dairy farmer?

iii) Would a sheep farmer make more money per ha by sheep farming; being part of a co-operative AD; or by selling silage/hay to a dairy AD?

iv) What is the public value of biodiversity and GHG benefits from silage-hay conversion?

6.2 METHODS

6.2.1 Sheep: current land management of silage and hay fields

I estimated a partial budget of current land management (sheep farming and silage/hay production) per ha of each of the silage and hay fields studied in this thesis, for 2011, the year that vegetation samples were taken for anaerobic digestion and plant biodiversity was measured. The fields' locations and managements are described in detail in the Chapter 2. The five silage (fertilised) fields and five hay (non-fertilised) fields were on Less Favoured Area (LFA) land, thus the farmers are classed as LFA grazing livestock farmers. Both the silage fields and hay fields were farmed for sheep, plus silage or hay. The silage and hay was consumed on-farm by cattle and sheep respectively in winter, and hence the value of harvested grass was not included, because agricultural budgeting books (Nix and Redman (2015); and Agricultural Budgeting and Costing (2015)) do not include it in budgets of sheep farming. Cattle were present on the silage/hay fields for only a few days per year, therefore were not included in this research. The budget was partial because it excluded unpaid labour performed by the farmer and their family, but it included all other inputs. Making silage on these small farms requires the hire of a contractor, whose costs include labour (included in the partial budget). However, because this research is studying the budget per ha, and not per agricultural enterprise, as is more commonly done, expected annual income from the sheep was reduced in proportion to the time the sheep were off the field (per year for ewes, per 6 months for lambs). The lack of grazing at those times allows plant biomass to grow for harvest.

General field, and sheep, management details are shown in Tables 1 and 2 of Chapter 2, as reported by the farmers. Grassland management costs included land cultivation, seeding, fertilisation, liming, herbicide spraying, harvesting and processing into silage or hay. These costs were an average of values from Nix and Redman (2015) and Agricultural Budgeting and Costing (2015); however the average cost of harvesting also included costs from one farmer; and fertiliser cost was taken directly from a local agricultural supplier (W.E. Jameson & Son Ltd, Masham). Sheep management costs included concentrate feed, vet, medicine, shearing, scanning and market expenses: costs were taken from Agricultural Budgeting and Costing (2015). Fixed costs of an average grazing livestock farm on Less Favoured Area land were also included (again, an average of Nix and Redman (2015) and

Agricultural Budgeting and Costing (2015)): they are static per ha of farm land, and include land rent, interest on loans, utilities, machinery costs and depreciation, and were similar between field types.

Fuel prices for each step in grassland management were taken from Agriculture & Horticulture Development Board (2016), and fuel consumption from the Cool Farm Tool (Hillier *et al*., 2011) (the carbon calculator tool used in Chapter 5). Income came from sheep sales and government subsidy/payments (BPS and agri-environment schemes). Sheep sold were classed into 4 categories: cast ewes (older ewes which are sold to farmers on lower ground for further breeding); store lambs (for fattening on a lowland farm), finished lambs (for slaughter) and mule lambs (for cross-breeding on a lowland farm). Sheep prices were an average of values reported by one farmer and the prices for hill store lamb or upland breeding stock in Agricultural Budgeting and Costing (2015); except the price of cast ewe which was an average of one farmer's values and a local livestock auction mart (Hawes Auction Mart, September 2015). The value of each sheep type is different, thus the approximate proportions of each sheep type sold from each field were estimated from information given by the farmers (Table 1). The quite high variability in proportions of sheep type between farms showed that each farm ran a slightly different system of sheep production. For simplicity, it was assumed that the proportion of different sheep types remained constant throughout the year on each field.

Table 1. Proportions of sheep sold per field, and mean price per animal. a to e are the different farms.

	Silage or hay field					
Proportion of	a	b	c	d	e	Price per animal (£)
ewes that are cast ewes	0.18	0.21	0.18	0.15	0.18	28.25
lambs that are stores	0.60	0.80	0.53	0.64	0.00	38.00
lambs that are mules	0.00	0.00	0.26	0.06	0.20	89.00
lambs that are finished	0.20	0.00	0.00	0.12	0.60	53.13

BPS was paid to all silage and hay fields at £170 ha^{-1}. In addition to BPS, the hay fields received agri-environment payments (under the 2007-2013 Rural Development Programme for England). Four hay fields received agri-environment payments for maintenance/restoration of plant species diversity (Higher Level Stewardship schemes HK6 and HK7 respectively; both paid at £200 ha^{-1}) plus hay-making (Higher Level

Stewardship scheme HK18, paid at £75 ha^{-1}). Thus total land subsidy/payments for these four hay fields were £275 ha^{-1} from agri-environment payments plus £170 ha^{-1} from BPS. The fifth hay field, and one silage field, were in Entry Level stewardship, paid at £62 ha^{-1}, thus their total land subsidy/payments were £170 ha^{-1} from BPS plus £62 ha^{-1} from agri-environment payments. The remaining four silage fields received only BPS (£170 ha^{-1}), and no agri-environment payments. Table 6A (Appendix) shows the budgets of individual fields.

6.2.2 Anaerobic digestion scenarios

6.2.2.1 General details

As described in the Introduction, two business models (a co-operative AD model, and a dairy farm AD model) were used to assess hay or silage in AD. Farmyard manure from the cattle kept by the sheep farmers, and slurry from the dairy farmer, were included in the co-operative and dairy AD respectively, because co-digestion of grass with manure improves microbial stability in AD and captures GHG otherwise released by the manures. Both AD models were assumed to be located on LFA land. It was assumed that the harvested yield of silage/hay was used in AD, and that no sheep were on the land.

The National Non Food Crop Centre (NNFCC)'s AD Cost Calculator (2013) was used to calculate financial returns for each AD scenario, with kind permission from David Turley and Lucy Hopwood (NNFCC). It has been used by, for example, Jones and Harris (2010) to outline a business case for AD. Wet mesophilic AD was assumed, where the dry matter content of the feedstock mix was < 15% to make it pump-able, with a temperature of around 37^{o}C, because wet AD is more commonly employed in the UK than dry AD. Simple one-digester systems were assumed with a retention time of 40 days to allow maximal digestion of the vegetation (Hopwood, 2011). 10% of digestate was assumed to be returned to the digester to recycle AD microbes and reduce the size of the digestate storage tank. The AD calculator reduced theoretical biogas production by 10% due to losses and inefficiencies.

The biogas was assumed to be burned in a combined heat and power plant (CHP) with an energy efficiency of 33% electricity and 42% heat production. The electricity produced would power the AD plant and displace farm house electricity use (receiving the UK renewable electricity subsidy Feed-in Tariff (FiT) at 8.21 p $kWh^{-1}e$), with surplus electricity sold to the national grid (receiving FiT and an additional export tariff of 4.91 p $kWh^{-1}e$ (Office of Gas and Electricity Markets, 2016b)). 15,010 kWh electricity was assumed to be used in displacing farmhouse electricity (Business electricity prices, 2017), and parasitic electricity demand of the AD plant was set by the AD calculator at 6 kWh electricity per tonne of feedstock. The CHP also produced heat which heated the digester (not eligible for the UK heat subsidy, Renewable Heat Incentive (RHI)) and displaced farmhouse heating (eligible for the RHI, paid at 5.9 p kWh^{-1} thermal (Office of Gas and Electricity Markets, 2016a)), with the rest wasted. Parasitic heat demand of the digester was set by the AD Calculator at 33% of heat produced; and 20,000kWh thermal of the heat heated the farmhouse (Department of Energy and Climate Change, 2015) which displaced oil heating in the farmhouse at 3.5 p kWh^{-1} thermal (Lukehurst and Bywater, 2015).

6.2.2.2 Co-operative: co-operative model of anaerobic digestion

The co-operative AD plant was a shared centralised plant. It was assumed that eight LFA grazing livestock farmers each had a financial share in the business, supplying grassland crops (silage (25% DM) or hay (86% DM)) plus farmyard manure (25% DM) to be digested, then receiving their share of profits, and digestate which was used as fertiliser. The AD plant was assumed to be located on one of the livestock farms. The number of suppliers was chosen as eight because there are 15 farms surrounding two of the studied farms (in Nidderdale, north England) and it was hypothesised that around half of those farms may participate. Other AD co-operatives have a wide-ranging number of farm suppliers e.g. 20, 66, 79 (Redman, 2010; Anaerobic Digestion and Bioresources Association, 2014).

Maximum distance to the AD plant was 4 miles as assessed using a national telephone directory for farm details (Yellow Pages, 2016) and Google maps for distances. A round trip of 8 miles (12.9 km) was therefore used in calculating transport costs, though this is an overestimate as most farms are closer. Poeschl *et al.*, (2012) estimate that manure feedstock and digestate can travel 64 km and 95 km respectively before positive environmental impacts of AD are reversed. Corn silage feedstock (which receives higher inputs than the grass silage examined here) can travel 53 km, and its digestate can travel 19 km before positive environmental impacts of AD are reversed (Poeschl *et al.*, 2012). Therefore the maximum round trip of 12.9 km is within reasonable limits. GHG from transport were not included in this analysis.

Table 2. Feedstock details of each co-operative and dairy AD scenario. There were 4 grass crop scenarios for the co-operative AD, and 6 grass crop scenarios for the dairy AD. Manure-only scenarios were also included. FYM is farmyard manure. Incr is increased.

Co-operative model

Scenario	**2a**	**2b**	**2c**	**2d**	**2e**
Grassland Feedstock	Silage	Hay	Incr. Silage	Incr. Hay	N/A
Manure Feedstock	FYM	FYM	FYM	FYM	FYM-only
Grassland Feedstock area (ha)	93	36	129	129	N/A

Dairy model

Scenario	**3a**	**3b**	**3c**	**3d**	**3e**	**3f**	**3g**
Grassland Feedstock	Silage	Hay	Incr. Silage	Incr. Hay	Own Silage	Own Hay	N/A
Manure Feedstock	Slurry	Slurry	Slurry	Slurry	Slurry	Slurry	Slurry-only
Grassland Feedstock area (ha)	93	36	129	129	16	16	N/A

The following co-operative AD scenarios were modelled (Table 2) (sheep farming was option 1 therefore co-operative and dairy AD were options 2 and 3 respectively):

- 2a) Silage
    - from 93 ha (the current production area of silage land per LFA grazing livestock farm (11.6 ha) x 8 farms). 93 ha x 12.3 t ha^{-1} = 1,142 t silage;
- 2b) Hay
    - from 36 ha (the current production area of hay land per LFA grazing livestock farm (4.5 ha) x 8 farms). 36 ha x 2.9 t ha^{-1} = 105 t hay;
- 2c) Increased silage
    - from 129 ha (the total silage and hay land if hay was converted to silage (11.6 + 4.5 ha) x 8 farms). 129 ha x 12.3 t ha^{-1} = 1,583 t silage;
- 2d) Increased hay
    - from 129 ha (the total silage and hay land if silage was converted to hay (11.6 + 4.5 ha) x 8 farms). 129 ha x 2.9 t ha^{-1} = 379 t hay;
- 2e) farmyard manure-only (explained below)
    - 0.036 t FYM produced per day (DEFRA, 2013) x 200 days housed x 30 cows x 8 farms = 1,740 t FYM.

Farmyard manure was included in all co-operative scenarios. It was a mixture of manure, urine and straw bedding from the average 30 beef cattle per farm (Farm Business Survey, 2016) housed 200 days over winter. Silage/hay was supplied at production cost (Table 3, as calculated in the budget of current land management, Section 6.4.1) because the farmers would have a share in the business and want to reduce costs. The fresh yields of silage (12.3 t ha^{-1}) and hay (2.9 t ha^{-1}) are very different to each other due to the large difference in water content, and higher DM losses when making hay (36% versus 18% losses in making hay and silage respectively (Buhle *et al.*, 2012)). Scenarios 2a and 2b assume silage and hay production at current production areas of LFA grazing livestock farms (Harvey and Scott, 2016; DEFRA, 2015b). Scenarios 2c and 2d ('increased' silage and hay) assumed that the total silage and hay grassland per farm (16.1 ha) was converted to either all silage, or all hay; thus increasing the area of production to 129 ha (Table 2). This also allowed comparison when digesting silage or hay from equal areas of land. A FYM-only AD scenario was also included for comparison.

Table 3. Feedstock data used in AD calculator.

Feedstock	DM (%)	Cost per fresh tonne (£)*	Cost to produce per ha (£)	Biogas yield (m^3 t^{-1})
Silage	25	Production cost: 26 Market price: 36	316	109
Meadow hay	86	Production cost: 59 Market price: 72	170	359
Farmyard manure	25	0		45
Slurry	8.5	0		20

*Co-operative AD used production costs; dairy farm AD used market price in scenarios 3a to 3d, and production cost in 3e and 3f.

The nutrients in digestate are in a more biologically available form than undigested manure (Anaerobic Digestion and Bioresources Association, 2013), and can be used as a fertiliser. This reduces the costs of bought-in fertiliser, so its net value was included in all the AD models. The concentration of K and N in digestate was lower than that needed on average silage/hay land, but P concentrations were higher, limiting the amount of digestate that could be spread per area. In the co-operative model, more digestate was produced than could be spread on silage/hay land (requiring 3-13 ha for excess spreading) but this was easily accommodated by the 24 ha of remaining 'inbye' land which was assumed to be pasture and can be fertilised. Therefore all digestate could be used on-farm. It was assumed that the cost of spreading digestate would be the same as spreading FYM.

Costs incurred only by the co-operative model (and not the dairy model) included transport (of feedstock to and digestate from the AD plant); purchase of a tank big enough to store 5 months' digestate (Agricultural Budgeting and Costing, 2015; DEFRA, 2013a) (which could be performed by a slurry tank in the dairy AD model) and labour for administration and running of the centralised plant. Capital and operating costs are detailed in Section 6.3.2.4.

6.2.2.3 Dairy: dairy farm model of anaerobic digestion

In the dairy farm AD model, the AD plant was on a dairy farm on LFA land (although they are less numerous than sheep farms, there are still 1154 dairy farms on mainly LFA land in England (Farm Business Survey, 2016)). It was assumed that the dairy farmer was the sole owner. The following dairy farm AD scenarios were modelled (Table 2):

- 3a) Silage, bought from the 8 grazing livestock farmers
    - same amount as co-operative 1a, Section 6.3.2.2 (93 ha, 1,142 t silage);
- 3b) Hay, bought from the 8 grazing livestock farmers
    - same amount as co-operative 1b, Section 6.3.2.2 (36 ha, 105 t hay);
- 3c) Increased silage, bought from the 8 grazing livestock farmers
    - same amount as co-operative 1c, Section 6.3.2.2 (129 ha, 1,583 t silage);
- 3d) Increased hay, bought from the 8 grazing livestock farmers
    - same amount as co-operative 1d, Section 6.3.2.2 (129 ha, 379 t hay);
- 3e) Own silage, produced on dairy farm, used at production cost
    - from 16 ha (16 ha x 12.3 t ha^{-1} = 198 t)
- 3f) Own hay, produced on dairy farm, used at production cost
    - from 16 ha (16 ha x 2.9 t ha^{-1} = 47t)
- 3g) Slurry-only
    - 0.062 t slurry produced per day (DEFRA, 2013) x 200 days housed x 135 cows (FBS, 2017) = 1,684 t slurry.

Slurry (8.5% dry matter) was included in all dairy scenarios, instead of farmyard manure. It is a mixture of manure and urine from the average 135 dairy cattle per farm housed over winter; and it is wetter than farmyard manure therefore less water was needed to make the feedstock mix pump-able. In scenarios 3a to 3d silage/hay was bought in at market price (i.e. higher than the price that farmers produce it) (Table 3) from the same 8 LFA grazing livestock farmers modelled in the co-operative model. As in the co-operative's first two scenarios, dairy scenarios 3a and 3b digest silage or hay at the current amounts produced by the livestock farmers; and scenarios 3c and 3d digest 'increased' amounts of silage or hay produced by the livestock farmers (described in Section 6.3.2.2). After it was shown that the dairy farm AD model was more financially viable than the co-operative AD, two additional dairy AD scenarios were added, where the dairy farmer grows silage (scenario 3e) or hay (3f) on their own farm. This silage and hay is grown on equal areas of land (16 ha, chosen because it is the same area per farm as the scenarios with increased silage or hay production; it equates to 13% of an average LFA dairy farm (Farm Business Survey, 2016)) and is supplied to the dairy AD business at production cost. A slurry-only AD scenario was also included for comparison.

The dairy farmer was assumed to keep all the profits and digestate (which could all be spread on the dairy farm). Transport, labour and digestate storage costs were not included (contrary to the co-operative model) because the market price of the silage/hay was assumed to include delivery, and digestate did not have to be transported off-farm; labour was assumed to be carried out by existing farm staff who feed it when they feed the cattle (Hopwood, 2011); and a pre-existing slurry tank was assumed to be used for digestate storage. Capital and operating costs are detailed in Section 6.3.2.4.

6.2.2.4 Biogas yields and costs

All costs and subsidies were at 2016 prices. Feedstock data are shown in Table 3 and biogas yields assumed 60% methane content of biogas. Hay and silage biogas yields were deduced from the lab-scale AD of dried, fresh-cut vegetation (reported in Chapter 4, Table 4; no ensiling was performed). One tonne of hay DM was therefore assumed to produce the same amount of biogas as one tonne of silage DM, thus assuming that ensiling does not significantly increase biogas yield per tonne DM. Although a commonly practiced preservation and pre-treatment method in agricultural AD, ensiling does not always increase biogas production when losses of volatile compounds are taken into account (Kreuger *et al.*, 2011). The silage biogas yield was very similar to the AD Calculator's default value (which was 106 $m^3\ t^{-1}$), whilst the hay biogas yield was lower than the AD Calculator's default (which was 426 $m^3\ t^{-1}$) (biogas yields shown in Table 3).

AD plant capital costs vary quite widely and are difficult to obtain for the UK (*pers comm*: Lucy Hopwood; Lukehurst and Bywater, 2015). Plus they may be quoted as total costs, so sometimes it is not clear what is included and what is not. Data from Jain (2013) were used to estimate CHP cost. An average cost for the digester tank was taken from Redman (2010) and Jain (2013); plus either Bywater (2011) or Hopwood (2011), depending on which most closely matched the scenario. Where sources quoted total price for CHP and digester, CHP price was deducted to give digester price separately. The value estimated for digester cost was assumed to include connection costs to the electricity grid, although only Hopwood (2011) definitely included connection in capital costs, therefore the actual cost of connection is probably higher than assumed here. It was assumed that 80% of the capital would be funded by a bank over a 20 year term (Lukehurst and Bywatwer, 2015) at a loan rate of 4% (Agricultural Budgeting and Costing, 2015; Lukehurst and Bywater, 2015). Digestate storage tank costs were calculated for the co-operative model using costs for a slurry lagoon (Agricultural Budgeting and Costing, 2015).

Operating costs included a) insurance: assumed to be £2000 (AD calculator default) for a 540 m^3 digester, which was modified in proportion to other digester sizes; b) labour for the co-operative model only, at 2 hours per day (Lukehurst and Bywater, 2015) at £8.70 per hour (Agricultural Budgeting and Costing, 2015) for a 540 m^3 digester scenario: this was modified in proportion to the amounts of feedstock in other scenarios; c) transport for the co-operative model only: an 8 mile round trip using a tanker for digestate at £2.80 per m^3, and lorry for feedstock at £2 per m^3 (Waste and Resources Action Programme, 2013); and d) water added to reduce total DM of feedstock to $< 15\%$ at £1 per m^3 (Jones and Harris, 2010).

6.2.2.5 Financial returns

Financial returns were measured as return on capital (ROC: a profitability ratio where profit or loss is divided by the capital costs, showing the efficiency with which capital is employed) and internal rate of return (IRR, explained in the Introduction). Profit and ROC were measured by the AD Calculator when finance was 50% repaid (which was between years 12 and 13); internal rate of return was measured 20 years after set-up of the AD plant. Financial returns in AD always included energy subsidies (for renewable electricity and heat produced by the CHP). Sensitivity analysis was performed by the AD Calculator: it helps determine in which aspects the business is most vulnerable to future changes.

In order to obtain finance from a lender, the internal rate of return (IRR) of an agricultural business should be a minimum of 10% (Bywater, 2011). It is reasonable to include land subsidy (BPS) and land payments (agri-environment scheme payments) in AD scenarios where farmers grow their own silage/hay for AD (co-operative scenarios 2a-2d, and dairy AD scenarios 3e and 3f). Agri-environment payments cover up to 100% of a typical farm's costs and income assumed to be lost ('foregone') by farming in a more environmentally friendly way (European Commission, 2017b; Hasund and Johansson, 2016). However, in England the income foregone estimate is based on the "typical" farm. These estimates may be higher or lower than the actual costs for a farm. The agri-environment payments were the average presently received by the studied farmers (under the 2007-2013 Rural Development Programme for England): £12 ha^{-1} for silage fields and £232 ha^{-1} for hay fields and BPS payments were £170 ha^{-1}. After BREXIT, the future of land subsidy/payments is highly uncertain, therefore they were added in a step-wise manner (agri-environment only; then agri-environment plus BPS). The FiT is the electricity subsidy paid on all electricity generated, at a rate chosen by the UK government; whereas electricity producers can choose to opt out of the export tariff to negotiate a higher deal with power companies to whom they sell the electricity. Therefore the level of FiT required to give an IRR of 10% in each scenario was calculated.

6.2.2.6 Comparison of profit per ha

To allow comparison of sheep farming, co-operative AD and selling the silage/hay as land management options for the sheep farmer, co-operative AD scenarios were also expressed as profit ha^{-1} for a grazing livestock farmer (this was not done for a dairy farmer because a budget of current management of the dairy farm had not been performed). This was done by dividing AD profit by the area of land used to grow the silage or hay, then subtracting fixed costs of land management per ha (Equation 1). Fixed costs were £252 and £233 ha^{-1} for sheep farming on silage and hay land respectively, but in situations where sheep were assumed to be removed from the land (i.e. co-operative AD, and silage/hay being sold), fixed costs were £228 and £217 ha^{-1} for silage and hay land because ram depreciation was removed. Due to the uncertain future of land subsidy/payments after BREXIT, profit ha^{-1} was calculated (i) without land subsidy/payments, (ii) with agri-environment payments only, and (iii) with both BPS and agri-environment payments

Profit ha^{-1} in AD = (profit of AD / no. ha of grassland feedstock) – fixed cost of land per ha

Equation 1. Calculation of profit ha^{-1} in AD. Profit of AD is profit at 50% finance repaid. For fixed cost, see text.

6.2.3 Estimating non-market value of carbon dioxide and ecosystem services

6.2.3.1 Carbon value

I estimated the value to society of GHG reductions by producing hay rather than silage. In order to incentivise reductions of greenhouse gas (GHG) emissions, the EU and other countries have developed carbon trading programmes whereby carbon dioxide equivalent (CO_2e) (the sum of GHG - including carbon dioxide, methane and nitrous oxide - expressed as a common unit). However, GHG reductions by moving from silage to hay are not tangible in such markets but they are assigned a non-market monetary value, to reflect the value to society of reducing GHG emissions (HM Treasury, 2011; Green, 2017). These CO_2e values are based on an estimate of the future abatement costs that will need to be incurred in order to meet specific emissions reduction targets (Department of Energy and Climate Change, 2009). In 2016 (the year that prices in this thesis were calculated) the value assigned for non-traded carbon was £62 t^{-1} (Bateman *et al.*, 2014).

6.2.3.2 Biodiversity value

Biodiversity is fundamental to the flow of ecosystem services but their estimation is exceptionally difficult. Boatman *et al.* (2010) provide one such estimate that relates to the higher biodiversity of hay fields. They estimated the value to society of “wildlife and landscape (birds, insects and plants, water quality and air quality)” benefits of land in UK agri-environment schemes, compared to land not in a scheme. The wildlife and landscape benefits were assigned non-market monetary values, based on people’s ‘willingness to pay’ for them (Boatman *et al.*2010). The hay fields in this study were in agri-environment schemes (Higher Level Stewardship (HLS) or the equivalent: detailed in Section 6.3.1) which aim to maintain or restore species-rich grassland by traditional management. Most (4/5) of the fertilised fields were not in agri-environment schemes, therefore it was assumed that they represented the situation that would occur without an agri-environment scheme. This allows the wildlife and landscape benefits of hay fields to be compared directly to the silage fields. According to Boatman *et al.* (2010), 15% of land in HLS (i.e. 165,000 ha) would be under such species-rich grassland schemes. The total low, mid and high-range benefits of these species-rich grassland schemes as valued by Boatman *et al.* (2010) were £36 million, £56 million and £84 million respectively. On a per hectare basis, the estimated non-market value of the wildlife and landscape benefits of hay land over silage land was £219-£511 ha^{-1} (mid-range £342 ha^{-1}). The average agri-environment payment actually paid by the UK government on the hay land was £232 ha^{-1} (Table 4) which is £110 less per ha than its 'worth' as estimated by willingness to pay i.e. the societal value created by the management is estimated to be greater than the cost to the public purse.

6.2.3.3 Cost-effectiveness of reducing GHG by AD

The cost-effectiveness of reducing GHG by AD was calculated: the difference in profit per ha for AD scenarios 'co-operative increased hay' and 'co-operative increased silage' was divided by the GHG saved per ha by digesting hay (Equation 2). Only feedstock production GHG was included; other GHG savings from AD, including renewable energy production and displaced inorganic fertiliser production, were not taken into account.

Cost-effectiveness = difference in AD profit ha^{-1}/difference in GHG emitted by land management ha^{-1}

Equation 2. Cost-effectiveness of GHG reduction per tonne by AD.

The cost-effectiveness of using public money to reduce GHG was calculated (Equation 3). The costs were energy subsidies (Feed-in Tariff, export tariff and Renewable Heat Incentive) per tonne of GHG saved. Again, only feedstock production GHG was included;

other GHG savings from AD, including renewable energy production and displaced inorganic fertiliser production, were not taken into account.

Cost-effectiveness = Total energy subsidies paid per hay AD scenario/tonnes of GHG saved by hay production per AD scenario.

Equation 3. Calculation of cost-effectiveness of using public money (energy subsidies) to reduce GHG.

6.2.4 Statistics used

Statistical analysis was used to examine differences in current management between silage fields (n = 5) and hay fields (n = 5), using one-way ANOVA (IBM SPSS Statistics for Windows, Version 24.0, 2016). If data failed the Levene test of homogeneity of variance, they were transformed (log or square-root). To examine relationships between costs and return on capital in the anaerobic digestion scenarios (n = 10 scenarios), linear regressions were performed.

6.3 RESULTS

6.3.1 Sheep: current sheep farming of silage and hay fields

On average, sheep and silage/hay production in marginal areas made a loss ha^{-1} before land subsidy (whole farm payment called the Basic Payment Scheme, BPS) or agri-environment payments were included (Figure 1 and Table 4). Gross margin (income minus variable costs) was positive for hay fields and negative for silage fields due to the lower variable costs of hay fields (hay fields do not require payment of an external contractor, or the cost of inorganic fertiliser). Furthermore, hay fields receive larger agri-environment payments, and once they were included, hay fields made a larger profit while silage fields still made a loss (Table 4). This £220 difference in the agri-environment payments between the two options reflects the additional work required for hay.

The mean income from sheep was similar between silage and hay fields but there was considerable variation in sheep sales on both field types due to the large variation among farms in: numbers of sheep per ha, length of time spent on the field per year, and proportion of different types of lamb (of different values) (Table 6A, Appendix). The differences between silage and hay fields were significant (Table 5).

Table 4. Budget of sheep farming (plus silage/hay production) in silage fields and hay fields; units £ ha^{-1} $year^{-1}$. Unpaid labour by the farmer and their family is not included. Sheep types are defined in Section 6.3.1.

		Silage fields			Hay fields		
		Mean	Min	Max	Mean	Min	Max
Sales							
Sheep sales	Cast ewes	26	5	65	16	3	34
	Lambs - stores	96	0	181	136	0	255
	Lambs - mules	72	0	212	66	0	212
	Lambs - finished	61	0	201	48	0	115
	Wool sold	7	1	16	5	1	10
Total sheep sales		**262**	81	437	**270**	146	437
Variable costs							
Cultivation and seeding	Permanent pasture grass	6	0	13	9	0	25
Herbicide spraying	Spot spraying	8	8	8	7	6	9
Making silage or hay	Farmer and contractor costs	231	183	328	146	131	164
Fertiliser - delivery, spreading	NPK	62	41	70	0	0	0
Lime - delivery, spreading	Calcium lime	9	0	25	9	3	17
Sheep costs	Purchased concentrate feed	21	0	64	26	0	64
	Purchased hay feed	1	0	6	2	0	8
	Vet, shearing, scanning, transport	66	13	150	43	7	92
Total variable costs		**405**	384	430	**240**	190	289
Gross margin		**-143**	-350	42	**30**	-65	148
Fixed costs							
Machinery	Depreciation, fuel, general contract	131	128	135	121	120	123
Overheads	Farm maintenance, utilities, misc.	68	68	68	68	68	68
Rent & interest	Rent & interest	29	0	58	29	0	58
Livestock depreciation	Ram depreciation	24	5	55	16	3	34
Total fixed costs		**252**	208	282	**233**	199	263
Profit before subsidy/payments		**-395**	-558	-216	**-203**	-278	-103
Subsidy/payments							
Basic payment scheme		170	170	170	170	170	170
Agri-environment scheme		12	0	62	232	62	275
Total subsidy/payments		**183**	170	232	**403**	232	445
Profit after total subsidy/payments		**-212**	-388	17	**200**	5	342

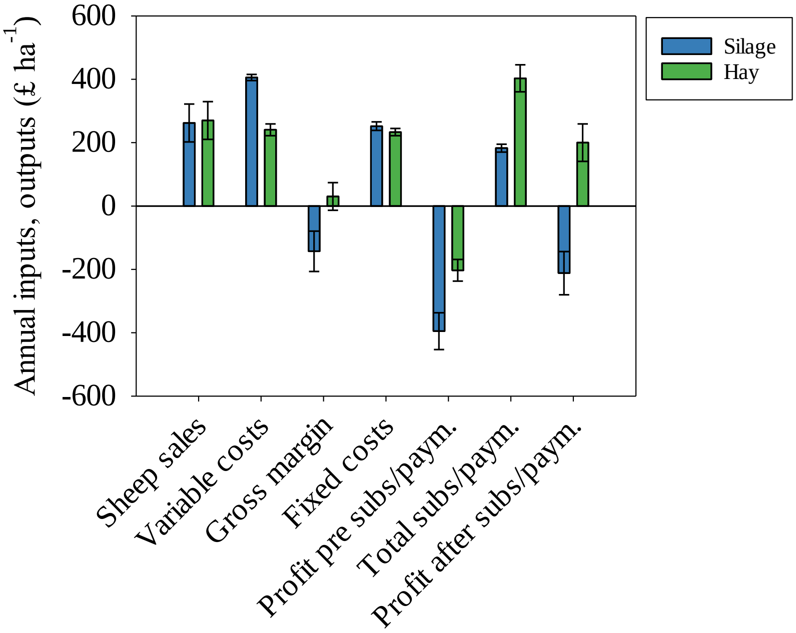

Figure 1. Mean budget of sheep farming (plus silage/hay production) in silage fields and hay fields; Error is SE. Subs/paym is Basic Payment Scheme (BPS) plus agri-environment payments.

Table 5. Statistical tests comparing costs and income for sheep farming on silage fields and hay fields; performed by 1-way ANOVAs.

	$F_{1,8}$	*P*
Variable costs	62.64	<0.001
Gross margin	5.02	0.055
Profit before total subsidy/payments	8.18	0.021
Total subsidy/payments	24.59	0.001
Profit after total subsidy/payments	20.97	0.002

Total subsidy/payments is Basic Payment Scheme (BPS) plus agri-environment scheme payments.

6.3.2 Anaerobic digestion

6.3.2.1 Comparing co-operative AD and dairy AD (land subsidy/payments not included)

None of the co-operative scenarios (digesting silage or hay) made a positive financial return, measured using the internal rate of return (IRR), whereas the dairy scenarios were profitable when used with hay-based feedstock (Tables 6 & 7, Fig. 2, and Appendix Table 6B & 6C). The additional transport, capital (digestate storage tank, larger digester to handle watered-down farmyard manure) and labour costs (see Sections 6.3.2.2 and 6.3.2.3) incurred by the co-operative model made it uncompetitive, compared to the dairy model. Even though total biogas production was higher in the co-operative scenarios, because farmyard manure has approximately double the biogas production of slurry, the co-operative costs were too high to produce a profit. The highest financial return was the dairy scenario that imported 'increased' hay from the sheep farmers (Scenario 3d; Fig. 2 and Table 7).

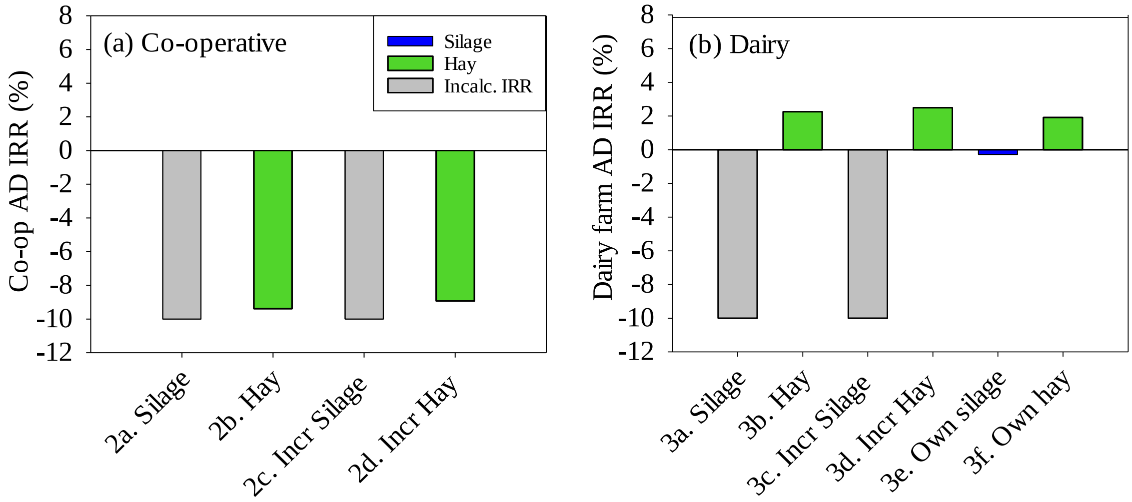

Figure 2. Internal rate of return (IRR) for the (a) co-operative AD scenarios and (b) dairy farm AD scenarios before land subsidy/payment was included. Incr is increased. Incalc means incalculable IRR because returns were all negative. All AD scenarios include energy subsidies.

6.3.2.2 Comparing hay and silage feedstocks (land subsidy/payments not included)

Hay consistently performed better financially than silage as an AD feedstock (when compared under like scenarios: Tables 6 and 7): the cost of feedstock production per GJ of biomethane produced in AD was lower for hay (£9) than silage (£15) (all dry matter losses included). This was because, even though hay AD plants had lower biogas production and incomes than silage AD (and produced slightly less electricity per t DM than silage: 835 vs 872 kWhe t^{-1} DM), they had proportionally lower costs due to smaller digesters and lower transport cost, because hay is more energy dense than silage (silage is 75% water, hay is only 14% water). Hay was also cheaper to produce and buy (per t DM): production cost was £68 and £103 t^{-1} DM for hay and silage respectively; and market price was £84 and £144 t^{-1} DM for hay and silage respectively.

When farmyard manure (FYM) was digested alone in co-operative AD (no crops added), it produced an IRR of -6.83% (Table 6, scenario 2e). Adding silage/hay to co-operative AD reduced IRR (when land subsidy/payment was not included). Digesting slurry alone in dairy AD produced an IRR of -0.53% (Table 7, scenario 3g). Adding hay to dairy AD increased the IRR. Adding bought-in silage reduced the IRR in dairy AD, and adding own-grown silage had little effect compared to slurry alone.

Table 6. Summary of results of the co-operative AD scenarios before land subsidy/payment was included. 2a and 2b digest silage/hay from current area of production on livestock farm; 2c and 2d digest silage/hay from increased area of production. FYM is farmyard manure.

Scenario	**2a**	**2b**	**2c**	**2d**	**2e**
Grassland Feedstock	**Silage**	**Hay**	**Incr. Silage**	**Incr. Hay**	**N/A**
Manure Feedstock	**Farmyard manure**		**Farmyard manure**		**FYM only**
Grassland feedstock area (ha)	93	36	129	129	N/A
Digester size (m^3)	540	390	620	570	330
Total Biogas (m^3 $year^{-1}$)	202778	115995	250847	214361	78300
Electricity generation (kWh $year^{-1}$)	405556	231990	501694	428722	156600
CHP elec. capacity (per hour) (kWe)	46	26	57	49	18
Capital cost of set-up (£)	308169	236211	372732	318276	209531
Unit cost (Capital/CHP elec.) (£ kWe^{-1})	6699	9085	6539	6495	11641
Total Operational Costs (£ $year^{-1}$)	87331	48028	109266	81444	33077
Total Revenue (£ $year^{-1}$)	64684	37988	79481	68091	25941
Return on capital (ROC) (%)	**-5.75**	**-2.69**	**-6.39**	**-2.60**	**-1.81**
Internal rate of return (IRR) (%)	**-**	**-9.38**	**-**	**-8.93**	**-6.83**

Table 7. Summary of results of the dairy farm AD scenarios before land subsidy/payment was included. Dairy farmer buys silage/hay from the sheep farmers in scenarios 3a-3d. 3a and 3b digest silage/hay from current area of production; 3c and 3d digest silage/hay from increased area of production. In 3e and 3f the dairy farmer grows their own silage/hay, on areas of the same size.

Scenario	**3a**	**3b**	**3c**	**3d**	**3e**	**3f**	**3g**
Grassland Feedstock	**Silage**	**Hay**	**Incr. Silage**	**Incr. Hay**	**Own Silage**	**Own Hay**	**N/A**
Manure Feedstock	**Slurry**		**Slurry**		**Slurry**		**Slurry-only**
Grassland feedstock area (ha)	93	36	129	129	16	16	N/A
Digester size (m^3)	330	200	410	350	210	190	190
Total Biogas (m^3 $year^{-1}$)	158158	71375	206227	169741	55262	50553	33680
Electricity generation (kWh $year^{-1}$)	316316	142750	412454	339482	110524	101106	67360
CHP elec. capacity (per hour) (kWe)	36	16	47	39	13	12	8
Capital cost of set-up (£)	202290	140697	237273	209011	136655	128427	125369
Unit cost (Capital/CHP elec.) (£ kWe^{-1})	5619	8794	5048	5359	10512	10702	15671
Total Operational Costs (£ $year^{-1}$)	64570	22855	85252	52698	19644	16394	12871
Total Revenue (£ $year^{-1}$)	50751	23967	65455	54469	18762	17224	12111
Return on capital (ROC) (%)	**-5.23**	**2.39**	**-6.74**	**2.45**	**0.95**	**2.25**	**0.99**
Internal rate of return (IRR) (%)	**-**	**2.26**	**-**	**2.50**	**-0.28**	**1.92**	**-0.53**

6.3.2.3 Sensitivity analyses of all co-operative and dairy scenarios (land subsidy/payments not included)

Sensitivity analyses showed the effect on return on capital when key parameters of the AD business were independently increased or decreased (Table 8). All co-operative scenarios (digesting hay or silage) maintained negative return on capital when AD parameters were increased or decreased. Dairy AD digesting hay maintained positive financial returns when the parameters were increased or decreased; but returns were low and are particularly sensitive, in order, to falling electricity price (greatest effect), biogas yield and feedstock volume, and increasing capital and feedstock costs (smallest effect).

Table 8. Sensitivity analysis of the return on capital (ROC) of co-operative and dairy farm AD scenarios before land subsidies were included. (a) Increase parameters by 5%, (b) decrease parameters by 5%. The change in ROC is shown: positive/negative numbers are increase/decrease respectively. Incr is increased; feed is feedstock.

(a)

			Change in ROC after INCREASE parameters by 5%					
Scenario		ROC	Elec. value	Biogas yield	Feed. vol.	Heat value	Feed. cost	Capital cost
1a	Co-op, silage	-5.7%	0.9	0.9	0.6	0.01	-0.5	-0.2
1b	Co-op, hay	-2.7%	0.6	0.6	0.7	0.01	-0.1	-0.3
1c	Co-op, incr silage	-6.4%	0.9	0.9	0.5	0.01	-0.5	-0.1
1d	Co-op, incr hay	-2.6%	0.9	0.9	0.7	0.01	-0.3	-0.3
2a	Dairy, silage	-5.2%	1.0	1.0	0.2	0.02	-1.0	-0.2
2b	Dairy, hay	2.4%	0.7	0.6	0.6	0.02	-0.3	-0.6
2c	Dairy, incr silage	-6.7%	1.1	1.2	0.2	0.01	-1.2	-0.1
2d	Dairy, incr hay	2.4%	1.1	1.1	0.7	0.02	-0.7	-0.6
2e	Dairy, own silage	1.0%	0.5	0.5	0.5	0.03	-0.2	-0.5
2f	Dairy, own hay	2.2%	0.5	0.5	0.6	0.03	-0.1	-0.5

(b)

			Change in ROC after DECREASE parameters by 5%					
Scenario		ROC	Feed. cost	Capital cost	Elec value	Biogas yield	Feed. vol.	Heat value
1a	Co-op, silage	-5.7%	0.5	0.2	-0.9	-0.9	-0.6	-0.01
1b	Co-op, hay	-2.7%	0.1	0.3	-0.6	-0.6	-0.7	-0.01
1c	Co-op, incr silage	-6.4%	0.5	0.1	-0.9	-0.9	-0.5	-0.01
1d	Co-op, incr hay	-2.6%	0.3	0.3	-0.9	-0.9	-0.7	-0.01
2a	Dairy, silage	-5.2%	1.0	0.2	-1.0	-1.0	-0.2	-0.02
2b	Dairy, hay	2.4%	0.3	0.6	-0.7	-0.6	-0.6	-0.02
2c	Dairy, incr silage	-6.7%	1.2	0.1	-1.1	-1.2	-0.2	-0.01
2d	Dairy, incr hay	2.4%	0.7	0.6	-1.1	-1.1	-0.7	-0.02
2e	Dairy, own silage	1.0%	0.2	0.5	-0.5	-0.5	-0.5	-0.03
2f	Dairy, own hay	2.2%	0.1	0.6	-0.5	-0.5	-0.6	-0.03

6.3.2.4 Including land subsidy/payments (BPS and agri-environment scheme)

Step-wise manner addition of agri-environment payments and BPS are shown in Figure 3, and in Tables 6D and 6E (Appendix) (energy subsidies were included in all AD scenarios). Scenarios were included where the AD owner grows the feedstock. When only agri-environment payments were included (because BPS could be stopped after BREXIT), two AD scenarios produced positive IRRs: the highest was co-operative digesting increased hay; and the second highest was dairy farm AD digesting own-grown hay. Including BPS as well as agri-environment payments predictably increased IRR further leading all AD scenarios digesting hay to produce positive IRRs. With full land subsidy/payments, silage

AD produced a positive IRR but only when own-grown and used in dairy AD; it was lower than all the hay scenarios. The co-operative silage AD scenarios remained loss-makers. Obviously, these land based payments are independent of the AD scenarios which remain intrinsically only weakly viable. However, the agri-environment payments are intended to achieve biodiversity conservation ends and might be thought of as supportive of hay production for AD purposes.

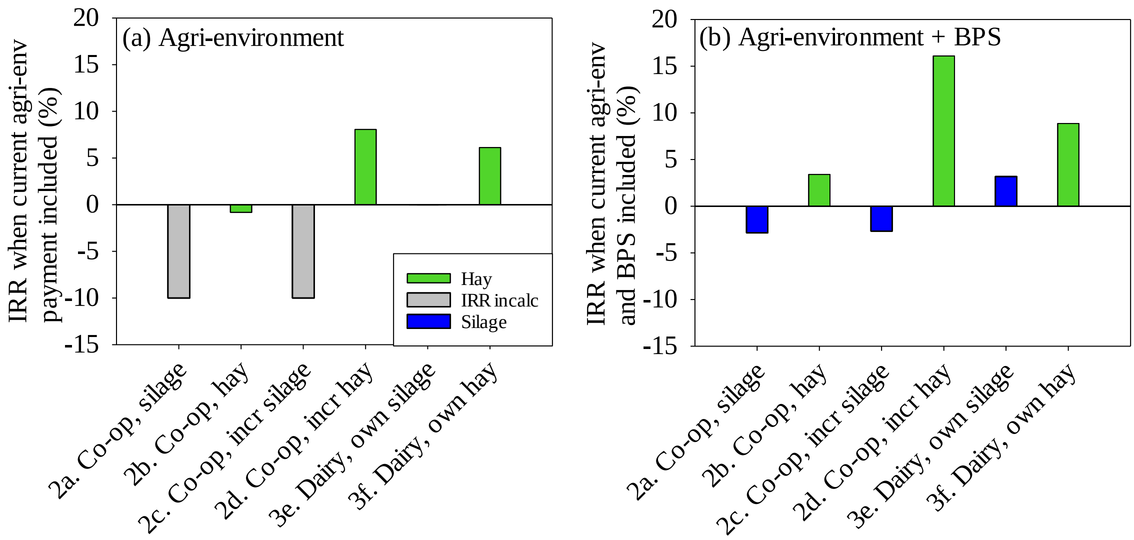

Figure 3. IRR of AD scenarios when different land subsidy/payments were included (where feedstock was grown by the AD owner): (a) agri-environment payments only; (b) agri-environment plus basic payment scheme (BPS). Incalc means incalculable IRR because returns were all negative. Incr is increased. Energy subsidies are included in all AD scenarios.

6.3.2.5 Level of agri-environment scheme payment required to give an IRR of 10%

Given future political uncertainties over BPS subsidy levels, it is useful to evaluate the level of agri-environment payment required to give an economically-viable IRR of 10% in AD (Fig. 4, and Table 6F in the Appendix). The co-operative scenario digesting increased hay required the lowest new agri-environment payment (272 ha^{-1}), a £40 increase on the average £232 ha^{-1} currently paid. The silage AD scenarios and other hay scenarios required much larger payments to produce an IRR of 10%. Thus, an increased payment in this case could support a greater area of hay production.

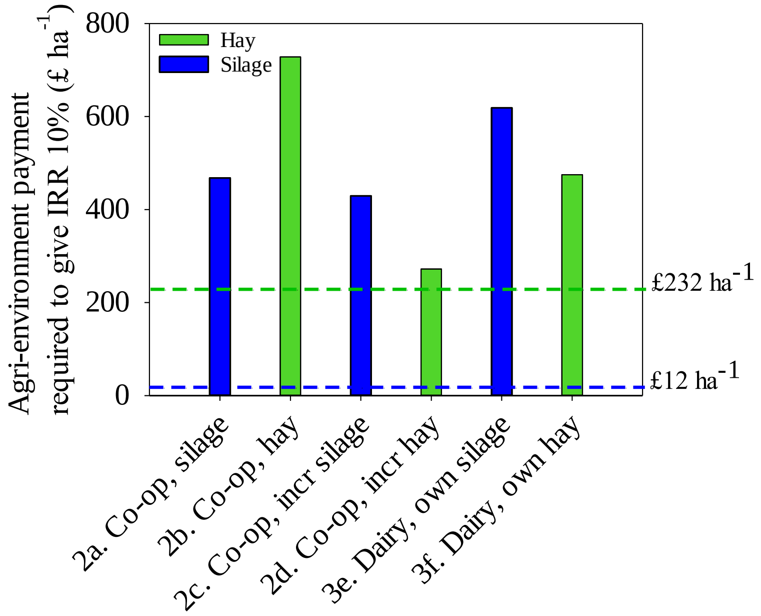

Figure 4. New agri-environment payments required to give an IRR of 10% (in AD scenarios where feedstock is grown by the AD owner). Blue and green dashed lines are the current mean silage and hay agri-environment payments (respectively) received by the studied farmers.

6.3.2.6 Level of renewable electricity subsidy required to give an IRR of 10%

All AD scenarios in this research included energy subsidies: electricity subsidies (FIT and export tariff) and a renewable heat incentive (RHI, which had very little effect on profitability and is not considered further). The value of electricity had a strong effect on return on capital (Table 8) therefore the level of Feed-in Tariff required to give an IRR of 10% in each scenario was calculated (Fig. 5 and Table 6G in the Appendix), with no land subsidy/payments included. The minimum increase in Feed-in Tariff required was from 8.21 to 11.65 p $kWhe^{-1}$ in the dairy scenario digesting increased hay.

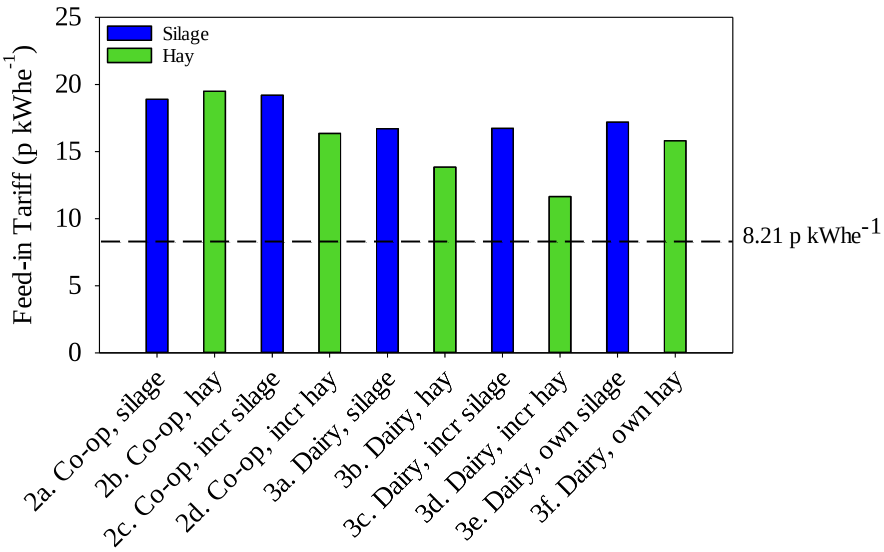

Figure 5. Electricity subsidy (Feed-in Tariff) required to give an IRR of 10% in each AD scenario. No land subsidy/payments were included. Current FIT (8.21 p kWhe^{-1}) is shown by the dotted line.

6.3.3 Values to society

6.3.3.1 GHG saved and higher biodiversity (when digesting hay rather than silage)

Hay production had mean GHG savings of 334 kg CO_2e ha^{-1} compared to silage (mean 385 versus mean 719 kg CO_2e ha^{-1} respectively, reported in Chapter 5 Section 5.4.2.1, excluding sheep), valued by the UK Treasury at £62 t^{-1}. Therefore annual GHG savings by making hay rather than silage were worth an average £20.65 ha^{-1}. The aggregate value of CO_2e saved in each AD scenario by using hay for AD rather than silage was £3277 when produced at current amounts in the co-operative system; £2656 when increased hay is produced in the co-operative system (compared to increased silage); and £330 when a dairy farmer grows their own hay for AD (compared to own-grown silage) (Figure 6).

Wildlife benefits of hay land were £219-£511 ha^{-1} (mid-range £342 ha^{-1}) greater than silage land (Section 6.3.3.2). Thus the combined value to society of GHG saved and biodiversity per hectare was £239-£532 ha^{-1} (mid-range £363 ha^{-1}) on hay land.

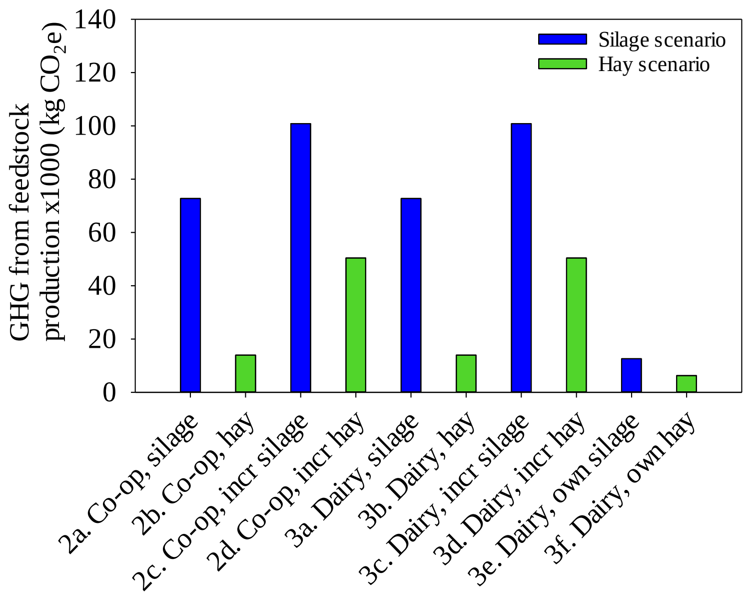

Figure 6. Total GHG emissions from feedstock production (silage or hay) for each AD scenario.

Per AD scenario, the wildlife and GHG benefits of using hay rather than silage were, in aggregate, worth £8,566-19,067 (mid-range £13,010) in scenarios where current amounts of hay were digested; £30,745-68,436 (mid-range £46,696) in scenarios where increased amounts of hay were digested; and £3,824-8,512 (mid-range £5,808) in the dairy AD scenario growing own hay. These societal benefits of hay AD would be worth more than the private profit of the AD business (cf. Tables 6B and 6C in the Appendix).

6.3.3.2 Replacing hay agri-environment payments with social values in AD

In hay AD, where the AD owner grew their own feedstock, agri-environment payments were replaced by social values (of greater wildlife and GHG saved) (Table 9) because agri-environment payments include valuation of the higher biodiversity, therefore including both social values and agri-environment payments would be double-counting. Social values (£363 ha^{-1}) were higher than agri-environment payments (£232 ha^{-1}) therefore returns were higher with social values than with agri-environment payments. BPS was included (and energy subsidies were always included in AD scenarios). The co-operative (digesting increased hay) produced the highest IRR (21.72%, scenario 2d), dairy digesting own-grown hay had the second highest IRR (10.84%, scenario 3f), and co-operative digesting hay (at current production rate) had the third highest IRR (6.2%, scenario 2b).

Table 9. IRR of co-operative and dairy AD scenarios including different levels of land subsidy/payments. In 'BPS + (AES or social)', silage fields receive BPS and AES, while hay fields receive BPS and social values. Only scenarios which grew their own AD feedstock were assumed to receive land or social payments. Energy subsidies were included in all AD scenarios. Incr is increased.

	Co-operative				Dairy					
Scenario	**2a**	**2b**	**2c**	**2d**	**3a**	**3b**	**3c**	**3d**	**3e**	**3f**
Grassland feedstock	**Silage**	**Hay**	**Incr. Silage**	**Incr. Hay**	**Silage**	**Hay**	**Incr. Silage**	**Incr. Hay**	**Own Silage**	**Own Hay**
No land subsidy/payments	-	-9.38	-	-8.93	-	2.26	-	2.50	-0.28	1.92
BPS + AES	-2.86	3.40	-2.68	16.09	-	2.26	-	2.50	3.19	8.87
BPS$ + (AES or social*)	-2.86	6.20	-2.68	21.72	-	2.26	-	2.50	3.19	10.84

BPS is basic payment scheme (£170/ha on all field types)
AES is agri-environment scheme payment (£12/ha on silage land; £232/ha on hay land)
*Social is value of wildlife and GHG saved on hay land. Total £363/ha (hay land). No social value for silage land.
$BPS plus whichever figure is higher out of AES or social value. On silage land: AES. On hay land: social value.

6.3.3.3 Cost effectiveness of reducing GHG by AD

The cost-effectiveness of reducing GHG by hay AD was £413 t^{-1} CO_2e when the co-operative scenarios 'increased hay' and 'increased silage' were compared. The cost-effectiveness of using public money (i.e. energy subsidies Feed-in Tariff, export tariff and Renewable Heat Incentive) to reduce GHG by hay AD was £1288 t^{-1} CO_2e for the co-operative digesting increased hay; £1028 t^{-1} CO_2e for the dairy digesting increased hay; and £2477 for dairy digesting own hay compared to own silage.

6.3.4 Comparison of profit per ha for different uses of silage/hay land on sheep farm

Expressing the co-operative system as annual profit ha^{-1} (which was not possibly for dairy) permitted comparison with sheep farming (Fig. 7 and Table 6H in the Appendix). This emphasises the higher financial viability of hay over silage (silage was only profitable when sold on, and with full land subsidy/payments (Fig. 7a)). When receiving agri-environment and BPS payments, hay land was profitable for sheep production (£200 ha^{-1}), but was 12% more profitable if sheep were removed and the hay was sold (£224 ha^{-1}). If BPS is discontinued but current agri-environment payments continue, hay land would be marginally profitable under current sheep management (£29 ha^{-1}). The options of co-operative AD digesting increased hay, and selling silage, both produced £82 ha^{-1} (with no sheep) when current land subsidy/payments were included (energy subsidies were included in all AD scenarios). The IRR of the co-operative AD scenario (digesting increased hay, including all land subsidy/payments) was 16.1%, but a profit of £82 ha^{-1} is low, especially compared to current management of hay land (£200 ha^{-1}). Thus, if a land payment was created to encourage hay AD, it would have to be £118 higher per ha than total current land subsidy/payments to match the profit ha^{-1} of sheep farming on hay land. Replacing agri-environment payments with social values (GHG saved and higher biodiversity) on hay land pushed up all hay profits: hay AD, using current production rates of hay (£212 ha^{-1}), became slightly more profitable than sheep farming receiving current land subsidy/payments (£200 ha^{-1}).

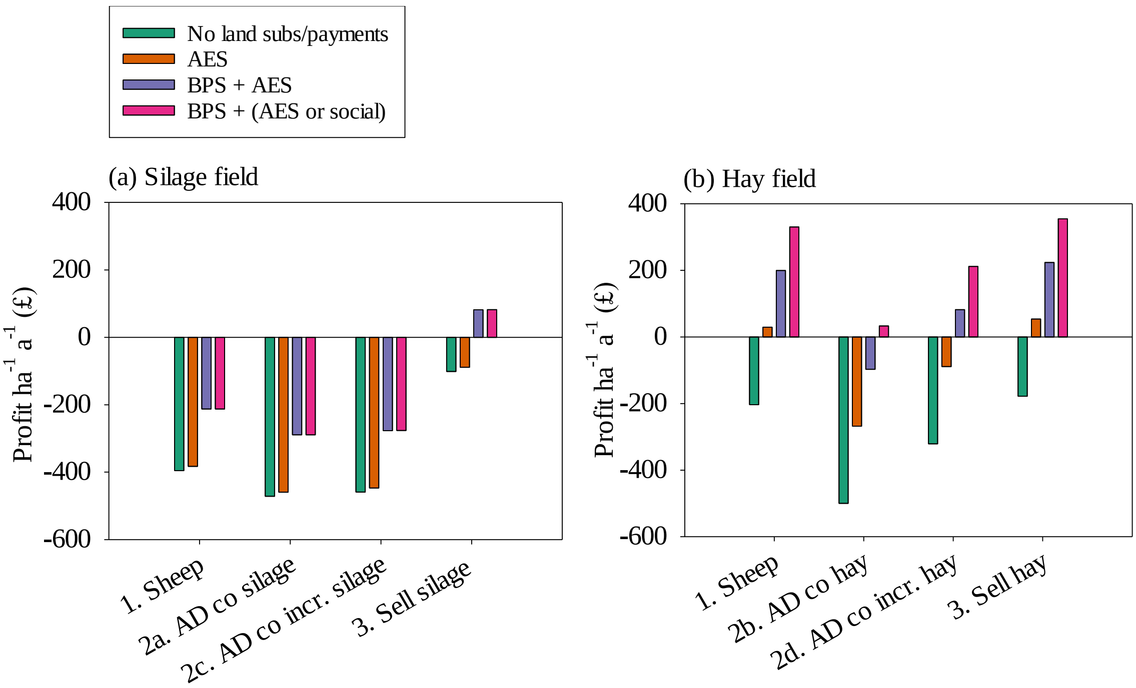

Figure 7. Profit per ha for different land management options for a sheep farmer including different levels of land subsidy/payments. (Energy subsidies were included in all AD scenarios.) (a) Silage field; (b) hay field. In 'BPS + (AES or social)', silage fields receive BPS and AES, while hay fields receive BPS and social values. AES is agri-environment payments. Sheep is sheep farming; co is co-operative; incr is increased silage/hay production for AD. Purple bars show the current situation.

6.4 DISCUSSION

All of the financial scenarios that I examined emphasise the marginal nature of the farming land, and that the land subsidy/payments are crucial to maintain such land management practices. Further, indirect societal benefits are potentially large and need to be taken into account when considering the future of the farming systems studied. As such, payments to facilitate perceived social 'goods' (increased wildlife, traditional landscape, increased energy production, reduced GHG emissions) have the power to alter farmer behaviour and may affect decision-making.

I found that hay performed better financially than silage in AD, particularly when hay production was increased. Before public money for land subsidy/payments was included, digestion of hay in the dairy AD model could provide a means to maintain biodiverse hay grassland (although public money for energy subsidies was included in every AD scenario).

The picture changed when land subsidy and environmental payments were included, whereby the co-operative AD model became more profitable than dairy AD, but it was still less profitable on a land area basis (per ha of land growing the AD feedstock) than selling hay or producing sheep. Current agri-environment payments for hay land are lower than the estimated willingness to pay valuation of species-rich grassland (Boatman *et al*., 2010) therefore an increase may be justified which may make hay AD competitive with sheep farming on an area basis. For the most profitable AD feedstock (increased hay) to become a reality, each sheep farmer would have to convert 11 ha of their farm to species-rich hay production which they may deem too risky. However hay AD was less cost-effective than other biomass production systems (although GHG savings from renewable energy and displaced inorganic fertiliser were not included). Before land subsidy/payments were included, financial return of AD was most sensitive to electricity price. Therefore the level and targeting of land subsidy, environmental payments (which could also incorporate societal values) and electricity subsidies after the UK leaves the EU will have a crucial effect on whether AD is a viable option for maintaining biodiverse land.

6.4.1 Which grassland feedstock is more financially viable in AD: silage or hay?

Hay performed better financially than silage in AD in all comparable scenarios. Silage is more commonly used in AD: silage production costs were similar to Hopwood (2011) (£25 t^{-1} fresh weight), and at the low end of silage from landscape management grass in Germany (£26-33 t^{-1}) (Blokhina *et al.,* 2011). The production cost of silage GJ^{-1} biomethane was £2.81 higher than that found by Gissen *et al.* (2014) for intensively managed grass silage (£11.93), which may be explained by the lower biomass yield and lower specific methane yield in this study. Improving biogas yield (per unit vegetation) would affect returns positively (Table 8). As found here, Nolan *et al.* (2012) found that AD of grass silage with pig manure was unviable in Ireland partly due to large silage costs.

'Increased' hay had potential as an AD feedstock, but achieving the increase in production would require the conversion of all silage land (11.6 ha per sheep farm) to hay. In reality it is unlikely that the 8 co-operative farmers would convert their prime (silage) land to hay, unless there were large incentives or changes to the running of the farm, such as reduced livestock numbers. Larger co-operatives could benefit from economies of scale, however. The UK agri-environment facilitation fund could pay for a co-ordinator to convert silage land to biodiverse hay land across several farms (DEFRA, 2016b). However, increasing grassland species richness can take many years, plus seed addition, if soil nutrient levels are high (Pywell *et al.*, 2007; Smith *et al.*, 2008), although in some cases species diversity recovers spontaneously after withdrawal of N (Storkey *et al*, 2015). Changes to farm management can have many implications, however; for example reducing forage production for livestock can lead to increasing imports of concentrated animal feed, with financial and environmental costs (Styles *et al.*, 2015) thus change needs to be properly planned. However, Blumenstein *et al*. (2012) point out that hay-making is a risky business in the temperate climate of the regions studied due to high rainfall, which can prevent the cut vegetation drying quickly enough (to around 86% DM). Silage-making, on the other hand, is less risky because it requires less drying (to around 25% DM) and is performed more quickly, by contractor. It is possible to make silage from later-cut grassland (Blumenstein *et al.,* 2012; Herrmann *et al.,* 2013) which would maintain plant species richness by allowing late-flowering plants to set seed. However, any increase in feedstock production cost due to the need for specialist silage machinery or silage additive reduces the returns from AD. Climate change impacts in the UK are generally predicted to include drier summers which may benefit hay-making, but sporadic heavy rain events are also likely to increase (IPCC, 2013), therefore it is not clear if hay-making will become more or less risky in the future.

6.4.2 Which business model of AD ownership is more financially viable: a co-operative of sheep farmers, or a single dairy farmer?

The dairy farm AD model was less reliant on public money (land subsidy/payments) than the co-operative, making it more attractive at this time of UK political uncertainty (although all AD scenarios included energy subsidies). It is also simpler to administrate than a co-operative because the co-operative would require shareholders to agree on e.g. how the plant is run, the use of the biogas and heat, the location of the plant, etc.; and digestate would be sent out to farmers. Despite the suggestion by Blumenstein *et al.*, (2012), that co-operation between farms may increase the feasibility of bioenergy production systems, the higher costs of co-operative AD were also observed by Gutierrez *et al.*, (2016) in Mexico. However, a larger co-operative with more farmers may benefit from economies of scale, and be more profitable. But dairy AD does not meet the IRR required to secure funding even when current land subsidy/payments are included. If society is willing to pay for hay AD (via land subsidy/payments and social benefits), and farmers want to modify their farm management to accommodate growing increased hay for AD and not for animal feed, co-operative AD is the more profitable model.

Then number of dairy herds in the UK has reduced dramatically in number over the last few decades (Agriculture and Horticulture Development Board, 2017) therefore the dairy farmer may need to consider other uses for their land, such as bioenergy production. However marginal sheep farms cover a much larger area than marginal dairy farms (68% and 4% of SDA land respectively) (DEFRA, 2011a), therefore sheep farms have greater potential for increasing the area of biodiverse hay land. Both co-operative and dairy models were most sensitive to changes in electricity value and biogas yield. Electricity income was mainly due to the Feed-in Tariff (paid on all electricity generated) therefore the financial success of an AD plant could be dependent on any governmental changes to electricity subsidies, which are unrelated to land subsidy/payments.

6.4.3 Which land management option is most profitable for a sheep farmer per ha?

Selling hay or sheep farming on hay land were more profitable than co-operative AD per ha. The sheep farming budget was similar to the Farm Business Survey (FBS) in England for Less Favoured Area grazing livestock farms (Harvey and Scott, 2016) and in Wales for Wales for Severely Disadvantaged Area hill sheep farms (Farm Business Survey Wales, 2016) (Table 10). Income from sheep sales in this research was between the English and Welsh FBS results. Variable costs for the silage land were higher than the FBS data because silage has the highest input of land on sheep farms, and the FBS averages data over the whole farm area including less productive land. Variable costs for the hay fields were similar to the English and Welsh FBS results, due to the average lower input. Thus profit from the hay fields in this research (before land subsidy/payments) was the same as the English FBS (-£203 ha^{-1}), but profit from silage land was much lower (£395 ha^{-1}) due to the higher variable costs. The profitability of hay land after inclusion of land subsidy/payments suggests that investment in hay land may be worth it if it continues to benefit from governmental financial support.

Table 10. Comparison of sales and costs between this research and Farm Business Survey (FBS) results for England and Wales (2015/2016). All values are mean £ ha^{-1}.

	This research Field type		FBS	FBS
	Silage	Hay	England	Wales
Sheep sales	262	270	229	368
Variable costs	405	240	224	266
Fixed costs	252	233	208	273
Profit, before subs/payments	-395	-203	-203	-157

Energy production from grassland can perform better than current livestock farming when measured on an IRR basis (Blumenstein *et al*., 2012), but Hopwood (2011) recommended that the returns from AD should also be examined at the farm-scale (for example as profit ha^{-1} of land growing the feedstock). At current financial support levels, maintenance of biodiverse hay land by AD was either similar to sheep farming on an area basis (i.e. if hay was sold by sheep farmers to a dairy AD); or uncompetitive (i.e. co-operative AD). But the replacement of agri-environment payments with societal benefits on hay land made co-operative AD digesting hay slightly (6%) more profitable per ha than sheep farming (at current financial support). Societal values were also worth more than AD profits, thus they would be very influential in the viability of biodiverse AD.

In other studies, financial return of AD could be improved, and therefore profit per ha increased, by reducing costs of feedstock production and capital, increasing biomass yield and increasing biogas yield (Prochnow *et al.*, 2009; Blokhina *et al.*, 2011). If animals didn't graze the land, total annual biomass yield (grazed plus harvested yield) was estimated to be 6.5 and 6.3 t DM ha^{-1} for silage and hay fields (Chapter 3, Table 3) which is 56% and 45% higher than average harvested silage and hay DM yield, positively influencing returns (Table 8). Fixed costs of AD land may fall further, and profits improve, if there were fewer sheep on the farm. Although returns were not sensitive to heat (due to large wastage) selling more heat can improve financial return (Rosch *et al.*, 2009); or even be integral to profitability of late-cut conservation grass AD (Blokhina *et al.*, 2011).

One recently developed, cheaper and lower-GHG-emitting technology for grass AD is the integrated generation of solid fuel and biogas from biomass (IFBB) (Blumenstein *et al.*, 2012; Buhle *et al.*, 2012). It burns the fibrous part of compressed semi-natural grassland silage, and digests the liquor in AD. However the scale of the IFBB system reported by Blumenstein *et al*. (2012) was much larger than the AD plants studied here (requiring a CHP size of 144 kWe, compared to the studied scenarios of 16-57 kWe CHP); and it is not clear if it would be financially viable if scaled down, therefore it may not be an appropriate system for the studied farms. Replacing improved grassland with dedicated bioenergy grasses such as *Miscanthus* is an alternative land management option (McCalmont *et al*., 2017), but this would change the look of the landscape and ultimately reduce biodiversity (Donnison and Fraser, 2016). It would also prevent livestock production, whereas the AD scenarios examined here could still allow animal grazing (but not winter forage production).

6.4.4 How much would the public need to pay, through subsidies/payments, to achieve financial viability of high biodiversity AD?

Ecosystem services such as pollination and carbon sequestration are being eroded by large-scale bioenergy production in Germany (Lupp *et al.*, 2015). In contrast, increasing hay production for AD would bring societal benefits and improved ecosystem services of greater biodiversity (with its associated benefits for pollinators (Goulson *et al.,* 2015)); lower GHG emissions; and the highest financial returns in AD. Public money payable for AD includes renewable energy subsidies and land subsidy/payments for the growers of the AD crops. The latter can incentivise prevention of land abandonment and could be utilised to ensure uptake of AD to increase the benefits of hay. The non-market valuation by Boatman *et al*. (2010) of biodiverse grasslands was £110 ha^{-1} higher than the average agri-environment payments currently received on hay land by the studied famers, and could justify higher payments. Income foregone in a new agri-environment scheme could include the risks of making hay in wet weather e.g. increased cost of grass turning to dry it. It is recognised that valuing biodiversity is difficult (Helm and Hepburn, 2012): it may not be robust and it tends to be difficult to generalise to other situations (Bateman *et al.,* 2014). However the valuation of Boatman *et al*., (2010) was used because, although it was based on people's willingness to pay, it relates directly to the system being analysed here since it was based on being in a UK agri-environment scheme or not. In the UK, beyond the EU agri-environment schemes that are applied with a number of restrictions, incorporating the value of the natural environment in land management is at a very early stage. Several large privately-run estates of land have started to account for the value of their natural capital (The National Trust's Wimpole Estate (Economics for the Environment Consultancy, 2015) and the Duchy of Cornwall (Duchy of Cornwall, 2016)) to help monitor the condition of the natural capital, detect changes due to new land management, and to inform future management decisions.

The cost-effectiveness of reducing GHG by hay AD (£413 t^{-1} CO_2e) was higher than the cost-effectiveness for producing power from biomass in a power plant (approximately £70 - £380 t^{-1} CO_2e) (Department of Energy and Climate Change, 2012), which is not surprising. However, other GHG savings from AD were not taken into account here, including renewable energy production and displaced inorganic fertiliser production, which could improve hay AD's cost-effectiveness. The cost-effectiveness of using public money to reduce GHG by hay AD was 17- to 40-fold higher than the societal value of CO_2e (£62 t^{-1}), suggesting that AD has high costs, but again renewable energy production and displaced inorganic fertiliser production were not taken into account, which could reduce the cost per GHG saved.

If hay land was to be used for bioenergy production, some grazing by livestock may need to be carried out in spite of the assumption that sheep have been removed from the land, because the agri-environment schemes for maintenance/enhancement of grassland species diversity in SDAs require that present management is continued (Natural England, 2013b). Complete lack of grazing can be detrimental to biodiversity (Evans *et al.*, 2006). It could be grazed by beef cattle in autumn if the land is not too wet (*pers. comm.* Andrew Hattan) then cut in summer for AD; and the silage land would continue to provide forage for the cattle in winter.

6.4.5 Returns from anaerobic digestion compared with the published literature

European semi-natural grassland (from Germany, Wales and Estonia), similar to the hay fields studied here, yielded 3.8 t DM ha^{-1} gross yield (about 10% lower than in this research) and produced an IRR of 6.22% when used in dry AD with all land subsidy/payments included (335 Euros yr^{-1}) (Blumenstein *et al.*, 2012). Reducing feedstock volume by 10% in the dairy model digesting own-grown hay (including land subsidy/payments), gave an IRR of 6.6%, thus they are similar. Hopwood (2011) estimated that AD digesting slurry from 130 dairy cows and grass silage from 15 ha in the UK produced an IRR of -3.29%. The dairy AD scenario digesting own-grown silage (from 16 ha) and slurry (from 135 cows) in this research also produced a loss (IRR -0.28%). Hopwood (2011)'s capital costs were higher than the averaged values used here, explaining the lower IRR. Unit cost of electricity (capital cost of the AD plant divided by electricity production per hour) (£5,048 – 10,702 kWe^{-1}) was between that of Jain (2013) (£3,000 – 6915 kWe^{-1}) and Hopwood (2011) (£8,621 – 18,750 kWe^{-1}) because their data were used to estimate capital costs. Thus there are similarities between the findings of this research and the literature.

6.4.6 Implications: effect on food production

Because more than half of all UK sheep production occurs in the LFAs (Stoate *et al*., 2009) and LFAs supply lambs to lowland farms (for finishing before slaughter), and ewes (for further breeding), loss of LFA sheep would impact UK breeding sheep and lamb production. This research does not promote reducing livestock numbers (e.g. sheep) in marginal areas in favour of bioenergy production, because of the livestock's role in food production (Ceotto, 2008); their use of land which could otherwise not be used for food production without adverse effects (Godfray *et al.*, 2010); maintenance of farming incomes and communities; and, in the absence of harvesting, maintenance of grassland biodiversity (Galbraith *et al*., 2013). However this thesis does note that sheep have high GHG footprints (Chapter 5), and sustainable food production, which can feed the increasing world's population whilst trying to reduce GHG emissions, may need to include reduced meat production particularly in developed countries (Godfray *et al.*, 2010).

6.4.7 Limitations

The budget of current land management of silage and hay fields did not include the income from beef cattle, because most (4/5) of the farms had cattle on the studied fields for only a few days per year. However it could be considered that this undervalues the silage land, which is producing forage for these cattle. Silage can also be fed to sheep which would remove this limitation to the study, but hay was more commonly fed to sheep than silage on the studied farms.

6.4.8 Conclusion

In conclusion, hay was a more financially viable AD feedstock than silage in marginal areas. 'Increased' hay (which would require land conversion to hay) was the most profitable AD feedstock: it would lead to increased species-rich grassland area in marginal areas, and reduced GHG emissions (from land management, displaced inorganic fertiliser and renewable energy production). Dairy AD of hay could provide a means to encourage maintenance of biodiverse hay grassland before public money for land subsidy/payments was included, although returns were too low to secure funding. But if land subsidy/payments were included, co-operative AD (of increased hay) became the most profitable model; and when agri-environment payments on hay land were replaced with societal benefits, it was more profitable per ha than sheep farming with current financial support. Agri-environment payments for hay land are lower than the willingness to pay valuation of species-rich grassland by Boatman *et al.*, (2010), justifying an increase. Therefore the level of agricultural land subsidy/payment after the UK leaves the EU, and inclusion of societal values, will have a crucial effect on whether AD in a marginal area is viable or not.

Chapter 7: Discussion

In addition to agricultural production, it is generally recognised that marginal (‘upland’) areas of the UK have value to society through their beauty, rich cultural heritage and provision of recreation (Institute for European Environmental Policy *et al*., 2004). Since 1946, the difficult economic conditions endured by marginal farmers have been recognised, and financial support provided, through various government funds (DEFRA, 2002). In recent years, the focus of financial support has moved to include nature conservation and the environment (Galbraith *et al.,* 2013). Most recently, the concept that marginal areas provide ecosystem services, which can benefit people well beyond the fringes of marginal land, has become more prevalent (Galbraith *et al*., 2013). Thus marginal areas are now formally recognised for their management of water, biodiversity and carbon (Galbraith *et al*., 2013). However, the increasing abandonment of such poor agricultural quality land (Allen *et al.*, 2014; Secretariat of the Convention on Biological Diversity, 2014) is the motivating force behind the research presented here. This thesis examines if an alternative use of the marginal grassland biomass (i.e. bioenergy production) could encourage farmers to continue managing the land for the biodiversity and social benefits it provides. The bioenergy technology chosen was anaerobic digestion, because grass silage is commonly digested in on-farm anaerobic digesters in the UK (DEFRA, 2016a).

The overall question asked in this PhD research was: can marginal grassland fields produce an anaerobic digestion feedstock, which helps mitigate climate change, provides the farmer with an alternative income and where the farming system is beneficial for wildlife?

7.1 Summary of thesis findings

Grassland fields in a marginal area (Less Favoured Area) of the north of England, UK, were examined, as potential sources of biomass for bioenergy production. In order to be easily accessible for harvesting for bioenergy, the fields that were studied were silage and hay fields, i.e. they are routinely cut by the farmers each year. Comparing hay and silage production allowed the comparison of different levels of input into bioenergy production because silage fields received inorganic fertiliser, and hay fields did not. Hay fields exhibited higher plant biodiversity than silage fields, with almost double the forb cover and higher levels of conservation indicator species. Hay production was shown to produce lower greenhouse gas (GHG) emissions than silage production, suggesting production of hay would help mitigate climate change better than silage. Silage and hay fields had similar harvested biomass yield, and similar annual yield, estimated from the total biomass harvested plus that grazed by sheep. Dried, fresh vegetation from silage and hay fields also had similar biomethane production by anaerobic digestion (AD), per g of vegetation and per ha. In financial models of AD, electricity made by burning the biomethane could save up to 96% GHG compared to fossil fuel electricity, when hay was digested. Hay emitted significantly lower GHG per ha of land, per t dry matter and per unit of biomethane electricity than silage. Hay was more profitable than silage in models of AD (measured as internal rate of return of the AD business); and hay also produced a higher profit per ha than silage. Selling hay to a nearby dairy AD plant could produce a similar profit per ha to sheep farming, incentivising maintenance of hay grassland.

This work is interesting because it suggests that, within the constraints of a rural area with limited choice of bioenergy feedstock and poor agricultural land, the better AD feedstock out of silage and hay is hay, which is cheaper, more biodiverse and emits less GHG. This may also apply to on-farm AD plants in more productive agricultural areas, where bigger hay yields and lower costs may also produce higher profits than silage. Using more hay for AD (rather than silage) would encourage an increase in biodiversity across a landscape; help mitigate climate change by reducing GHG emissions from the land by using less fertiliser, and producing more renewable energy; provide an alternative income for farmers if selling hay to a nearby AD plant; and prevent possible land use change which contributes to climate change. The Yorkshire Dales, where the work was carried out, is an interesting area to apply to other marginal areas with temperate a climate and lots of small farms.

7.2 Where the results of this research could be applied

This Chapter discusses the implications of these findings if they were extrapolated to all LFA land in England, as an example of what may be achievable in other countries where abandonment by livestock is more of an issue, such as Wales (Corton *et al*, 2013). This research was performed in England because, before this study began (pre-2009), livestock numbers in England had been falling and land was at some risk of abandonment. However, since 2011 the number of cattle has remained steady; and, after previously falling, sheep numbers have been generally rising since 2007 (DEFRA, 2017). Therefore the results of this research may be best applied to other areas with temperate grasslands and similar climate, where livestock numbers have already fallen. The financial analysis of anaerobic digestion undertaken in this work is specifically applicable to the UK, but in countries where renewable energy subsidies are more substantial (such as Germany; Prochnow *et al.*, 2009b), returns from AD may in fact be more favourable than those reported here.

7.3 Biodiversity, yield and nitrogen

The similar total annual dry matter yield of silage and hay fields, and higher biodiversity in hay fields, suggests that if farmers increased the species richness of their grassland fields, they could apply less fertiliser but maintain yield of bioenergy feedstock. As the source of over 70% of the UK's drinking water (Watts *et al.*, 2001), marginal ('upland') areas have important functions in maintaining water quality (Beharry-Borg *et al.*, 2013). As well as the potent climate warming effect of nitrous oxide released from fertilised soils, leaching of reactive nitrogen compounds from fertilised land into water causes environmental damage by eutrophication (Seitzinger and Phillips, 2017) and reduced quality drinking water. Added to this, the volatilisation of nitrogenous compounds from fertilisers into the atmosphere has a fertilisation effect when they deposit on vegetation, reducing plant biodiversity (Stevens *et al.*, 2006). Hence, reducing nitrogen pollution by the agricultural use of fertilisers, whilst increasing the efficiency of fertiliser use, is a worldwide aim (Seitzinger and Phillips, 2017). Marginal areas are not large sources of nitrogen leachate into water (Chesterton, 2009) but reducing nutrient leaching by reducing fertiliser use would only be beneficial to water quality. The higher species richness of the hay fields (aside from the lack of inorganic fertiliser) may also help reduce nitrogen leaching into soil water, particularly if legumes are not present (Leimer *et al.,* 2015). The similar biomass and biomethane production of silage and hay fields would hypothetically allow bioenergy production and biodiversity to co-exist in hay fields. But in order to meet EU sustainability requirements of GHG from biorenewable electricity, and to avoid indirect land use change, reduced grazing would be necessary on the fields examined in this thesis. In areas where livestock numbers have already dropped, no change in grazing may be necessary. The level of grazing would need to be sufficient to maintain biodiversity, but as has been discussed in Chapter 3 (Section 3.4.3) and Chapter 5 (Section 5.5.6), sometimes cutting and removing the vegetation is, by itself, sufficient to maintain biodiversity (Donnison and Fraser, 2016).

7.4 Converting silage land to hay: effects on ecosystem services, greenhouse gas emissions and bioenergy production

Hay was also more financially attractive than silage in models of AD. If farmers wanted to supply hay to an AD plant, they may want to increase the amount of hay they produce, because this was shown to produce higher financial returns than the current hay production. In the marginal areas of England there are 6,577 LFA grazing livestock farms producing sheep and beef cattle (Harvey and Scott, 2016), each with an average of 11.6 ha of silage land (DEFRA, 2015b). This totals 76,293 ha. If this was converted to species-rich hay production, the total area of hay land in English marginal areas would be 105,758 ha, with a 48-63% higher biodiversity. This is approximately 18% of the managed grassland in LFA areas (Institute for European Environmental Policy, 2004). The links between biodiversity and ecosystem services are complex (Turner *et al.*, 2007), but the increase in grassland biodiversity could potentially produce positive outcomes in several ecosystem services. For example, as well as a producing an annual biomass yield comparable to fertilised silage fields, there may be a greater number of pollinators in better health (Goulson *et al.,* 2015; Carvell *et al*., 2017) which would benefit any nearby cropland (Kremen *et al*., 2004; Pywell *et al*., 2015). Increasing grassland biodiversity can also lead to greater soil carbon sequestration (De Deyn *et al*., 2011), improve drinking water quality as discussed in the last Section (Beharry-Borg *et al.*, 2013) and improve the landscape's aesthetics (Millennium Ecosystem Assessment, 2005).

The move from moderate-input to low-input farming would lead GHG emissions to fall by 334 kg CO_2e ha^{-1}, if farmers changed land from silage to hay production (assuming there were no sheep on the land). English agricultural GHG emissions would reduce by 25,482 tonnes CO_2e if all 6577 LFA grazing livestock farms changed their silage land to hay production. Because hay produced higher GHG savings than silage from biomethane electricity, it should be the preferable AD feedstock. This is pertinent because the EU will require greater GHG savings each year from renewable electricity (50% saving in 2017; 60% saving in 2018) (European Commission, 2010). This has implications for all on-farm

AD plants in the UK digesting grass silage: if they swapped from grass silage (plus manure) to hay (plus manure) they could reduce GHG emissions per unit of electricity by 35-41% (Table 5B in the Appendix). In the UK, on-farm AD plants digest mainly crops (such as maize and grass silage) and manures (Waste Resources and Action Programme, 2014) but unfortunately the UK does not record the amount of grass silage digested (DEFRA, 2016a). Therefore it is not possible to estimate the total amount of GHG which may be saved in the UK if AD plants which currently digest grass silage swapped to hay. However, it is clear from my results that if AD plants were to be established in marginal areas, digesting hay from current production area (4.5 ha of hay per farm) from all the English LFA grazing livestock farms (totalling 85,701 t hay), 16,630 houses could be supplied with electricity and 6,010 with heat. If silage land on those marginal farms was converted to hay (giving a total hay area per farm of 16 ha) and the hay (307,605 t) was digested in AD, 59,690 houses could be supplied with electricity and 21,580 supplied with heat. Manure was not included in these estimates, but it would increase the energy produced. However, reducing the area of silage land could have negative consequences on the number of cattle a farmer could keep, because cattle eat silage in winter. If silage production was increased elsewhere to feed the same number of cattle (constituting indirect land use change), GHG savings acquired by increasing the area of low-input land may be lost (Styles *et al*., 2015a). Therefore the production of bioenergy from grassland crops may have to be limited to areas where livestock numbers have already fallen, or excess grassland is available.

7.5 Effect of sheep on greenhouse gas emissions

As was seen in Chapter 5 (Tables 5, 6, 7), the number of sheep on the hay or silage land had a major effect on GHG emissions. If the farmers decided to produce biomethane from their hay or silage fields as currently managed, sheep numbers (or length of time on the field) would need to be reduced by 60% and 70% to meet the 2017 and 2018 (respectively) EU sustainability requirements described in the previous Section (Table 5C in the Appendix). This suggests that, in terms of GHG emissions at least, some sheep could still remain on the field to graze, and the electricity produced from biomethane electricity could still qualify for renewable energy subsidies. Because the harvested yield would be higher due to less grazing, it is possible that the farmer could keep enough hay to feed the sheep, and use the rest for AD. Otherwise the sheep's hay feed would need to be produced elsewhere, which could be either beneficial if an intensively managed area of land is converted to low-input hay production; or alternatively it could be detrimental (resulting in indirect land use change (Styles *et al*, 2015a)) if the land converted to hay production was previously unharvested and did not usually receive manure or lime. Therefore the production of bioenergy and livestock would not suit all farming systems, but could be applicable on those farms where there is more grassland than is needed by livestock. It is interesting that, theoretically, sheep and bioenergy production could co-exist in a marginal landscape which presents limited options for providing an income. Doing this would reduce the negative consequences of removing sheep entirely from the land (such as reduced biodiversity, stopping food production, changing the way of life for the farmer, and reducing the income stream achieved from livestock). However, having lower sheep numbers would also have the positive effect of reducing agricultural emissions due to ruminant enteric fermentation. If people adjust their diets to include less meat, it could aid greater health benefits and sustainable food production (Rockstrom *et al*., 2017). The effects of reduced sheep numbers on food production were discussed in Chapter 6, Section 6.4.6.

7.6 Answering the overall question asked in this thesis

The question whether marginal grassland fields can produce an anaerobic digestion feedstock, which helps mitigate climate change, provides the farmer with an alternative income and where the farming system is beneficial for wildlife can thus be answered. Yes, it is potentially feasible for biodiverse marginal grasslands to produce GHG-saving renewable energy, and provide the farmers with an alternative income, if farmers sell their hay (to a nearby AD plant, for example on a dairy farm), and if sheep numbers on the hay land have reduced. This could produce a profit per ha of land which is competitive with sheep farming and could help reduce the risk of land abandonment.

However, obviously the option of selling hay for AD requires a nearby AD plant. Dairy farm AD was suggested for this because in the financial models of AD, dairy AD could produce a small positive return even in the absence of land subsidy/payments. If the hay was alternatively sold for other purposes such as livestock feed, this would encourage maintenance of the hay land, but there would be no climate change mitigation. If it became financially feasible for a group of sheep farmers to set up a co-operative AD business, and it was accepted by the local community, this could greatly benefit the farmers and community (European Commission, 2009). Such an approach might help reduce social isolation and depopulation of the marginal area by bringing different stakeholders together, bringing in jobs, and producing electricity and heat energy for the local community. However, the difficulties of achieving planning permission (Bywater, 2011), coupled with reductions in government financial support for renewable energy, have reduced the number of new community energy projects in the UK (Harvey, 2016). If it was easier to get planning permission, and financial viability was increased through economies of scale (Style *et al.*, 2015a) by increasing the number of farmers, co-operative AD may become feasible.

Several negative points about moving to AD could include an increase in road transport which could disrupt the local area. Additionally, eutrophication and acidification increase via loss of ammonia from digestate, but these can be reduced if digestate stores are well sealed and digestate is injected into soil (Styles *et al*., 2015b). Whilst AD of food waste and burning of *Miscanthus* pellets produce the greatest GHG savings through biorenewable energy when modelled on an arable farm (Styles *et al.*, 2015b), they each have disadvantages. Importing food waste onto a farm produces biohazard risks, and requires more planning and permitting than simple crop and manure AD (Hopwood, 2011), which small farmers may be unwilling to undertake. The distance from food waste source to the AD plant may also be large in marginal areas. Once *Miscanthus* plantations are established and yielding well, they reduce biodiversity compared to improved grassland (Dauber *et al.*, 2015; Donnison and Fraser, 2016) which is counter to the aim of this research. AD of energy crops, such as maize, also has negative environmental effects including poorer water and soil quality (Styles *et al.* 2015b). In contrast, using a pre-existing perennial grassland crop such as hay for AD, which does not receive inorganic fertiliser, does not hold these specific risks.

Producing renewable energy by anaerobic digestion is just one option of energy production in marginal areas, although the direct use of the hay biomass in the energy production does provide a clear need to maintain the hay land. An alternative renewable energy in marginal areas could be wind power, because the hilly, exposed land in LFA areas may lend itself to creating electricity from wind (no analysis was performed on this). Through installing wind turbines on hay land, sheep farming could continue in areas where livestock numbers have not fallen, without any consequence to livestock feed production. However, this would not promote maintenance of biodiverse grasslands. Diversifying into tourism in an area with biodiverse meadows may increase farming income and reduce land abandonment.

7.7 Future work

It would create greater confidence in the results, particularly in biodiversity and biomass yield, if this study was repeated with more fields. They could be spread over several different marginal areas within the UK because each area is distinctive; and it could include sites (e.g. in Wales) where abandonment is already happening. A more in-depth study of the financial return from AD could be performed, to include specifics on capital and running costs rather than the general data which, by necessity, were used here.

7.8 Conclusion

Marginal hay grasslands can indeed provide biodiversity, and bioenergy feedstock, but the most financially viable system of encouraging their maintenance is selling the hay. If sold to a nearby AD plant (e.g. on a dairy farm) it would help mitigate climate change by substantially reducing GHG emissions from electricity production.

APPENDICES

APPENDIX TO CHAPTER 2.

The first method of trying finding species-rich and species-poor marginal grassland fields.

First I looked for data on species richness of grasslands in marginal (‘upland’) areas. I went through large files of botanical data of grasslands collected by ADAS for Natural England between 1987 and 2002 in the Pennine Dales Environmentally Sensitive Area (published as Critchley *et al.*, 2007b). I found the most biodiverse land surveyed in 2002. Natural England kindly contacted these land managers on my behalf, and several gave me permission to harvest some vegetation. Unfortunately, though they did not have a species-poor field with which to compare. Therefore I could not use these sites.

APPENDIX TO CHAPTER 3.

Table 3A. 2011 Fertilised fields: species richness, Ellenburg N and number of semi-improved grassland indicator species (marked in bold).

		Species richness																				Ellenburg N																			
	Fertilised Field →	a				b				c				d				e				a				b				c				d				e			
2011	Quadrat →	1	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4
Annual meadow-grass	*Poa annua* L.	1						1									1					7						7									7				
Broad-leaved dock	*Rumex obtusifolius*									1																				9											
Cock's foot	*Dactylis glomerata* L.	1	1	1	1		1			1					1	1						6	6	6	6		6			6					6	6					
Common bent	*Agrostis capillaris* L.		1	1	1									1		1	1						4	4	4									4		4	4				
Common chickweed	*Stellaria media*								1	1			1																7	7			7								
Common mouse ear	*Cerastium fontanum*		1	1	1	1	1	1	1		1	1	1	1	1	1	1	1					4	4	4	4	4	4	4		4	4	4	4	4	4	4	4			
Common nettle	*Urtica dioica*																				1																				8
Common sorrel	***Rumex acetosa***			**1**						**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**							4						4	4	4	4	4	4	4	4				
Creeping bent	*Agrostis stolonifera* L.		1	1	1												1						6	6	6												6				
Creeping buttercup	*Ranunculus repens*	1	1	1	1					1	1	1	1		1							7	7	7	7					7	7	7	7		7						
Crested dog's-tail	*Cynosurus cristaus* L.				1	1					1														4	4					4										
Daisy	*Bellis perennis*										1	1																			4	4									
Dandelion	*Taraxacum* agg.				1		1			1	1														6		6			6	6										
Meadow buttercup	***Ranunculus acris***				**1**			**1**		**1**	**1**	**1**	**1**				**1**								4			4		4	4	4	4				4				
Meadow fox-tail	*Alopecurus pratensis* L.								1				1		1	1	1	1	1		1								7				7		7	7	7	7	7		7
Perennial rye-grass	*Lolium perenne* L.	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6
Pignut	*Conopodium majus*										1																				5										
Red clover	***Trifolium pratense***				**1**																				5																
Red fescue	*Festuca rubra* L.					1		1	1		1	1			1											5		5	5		5	5			5						
Ribwort plantain	***Plantago lanceolata***										**1**																				4										
Rough hawkbit	*Leontodon hispidus*						1				1																3				3										
Rough meadow-grass	*Poa trivialis* L.	1	1	1			1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	6	6	6			6	6	6	6	6	6	6	6	6	6	6	6	6	6	6
Sheep's fescue	*Festuca ovina* L.				1	1	1	1	1	1			1												2	2	2	2	2	2			2								
Smooth meadow-grass	*Poa pratensis* L.	1		1		1		1		1			1									5		5		5		5		5			5								
Soft brome	*Bromus hordeaceus* L.				1		1			1	1	1	1					1	1						4		4			4	4	4	4					4	4		
Sweet vernal-grass	*Anthoxanthum odoratum* L.										1	1	1		1	1															3	3	3		3	3					
Timothy-grass	*Phleum bertolonii* L.				1																				6																
White clover	*Trifolium repens*	1		1	1					1	1	1			1		1					6		6	6					6	6	6			6		6				
Yarrow	***Achillea millefolium***				**1**		**1**			**1**			**1**												4		4			4			4								
Yellow oat-grass	*Trisetum flavescens* (L.) Beauv.				1									1		1									4									4		4					
Yorkshire fog	*Holcus lanatus* L.		1	1	1	1	1		1		1	1	1	1	1	1	1	1	1	1	1		5	5	5	5	5		5		5	5	5	5	5	5	5	5	5	5	5
Total species richness; or mean Ellenburg N		7	8	11	17	7	10	8	8	14	17	12	14	7	11	10	11	6	5	3	5	6.1	6	5	5	4.4	5	5	5	5.4	5	5	5	4.7	5	5	5	5.3	6	6	6
Mean quadrat^{-1}		10.8				8.3				14.3				9.8				4.8				5.5				4.8				5.0				5.1				5.8			
SE quadrat^{-1}		2.25				0.63				1.03				0.95				0.63				0.25				0.20				0.16				0.18				0.24			

Total no. semi-improved grassland indicator species		0	0	1	3	0	1	1	0	3	3	2	3	1	1	1	2	0	0	0	0
Mean quadrat^{-1}		1.0				0.5				2.8				1.3				0.0			
SE quadrat^{-1}		0.71				0.29				0.25				0.25				0.00			

Table 3B. 2011 Non-fertilised fields: species richness, Ellenburg N and number of semi-improved grassland indicator species (marked in bold).

Species richness

	Non-fertilised Field →	a				b				c				d				e			
2011	Quadrat →	1	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4
Autumn hawkbit	***Scorzoneroides autumnalis***					**1**	**1**	**1**	**1**												
Black medick	***Medicago lupulina***	**1**	**1**	**1**								**1**	**1**								
Broad-leaved dock	*Rumex obtusifolius*								1												
Cock's foot	*Dactylis glomerata* L.	1	1		1													1			
Common bent	*Agrostis capillaris* L.	1	1	1		1	1	1	1	1	1				1		1	1		1	
Common eyebright	*Euphrasia nemorosa*					1	1	1													
Common knapweed	*Centaurea nigra*											1	1								
Common mouse ear	*Cerastium fontanum*	1	1	1	1	1	1		1	1	1	1	1	1	1	1	1	1	1	1	1
Common sorrel	***Rumex acetosa***	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**
Creeping bent	*Agrostis stolonifera* L.													1					1		
Creeping buttercup	*Ranunculus repens*	1	1	1	1	1	1		1	1	1	1	1	1	1	1	1		1		
Creeping soft-grass	*Holcus mollis* L.															1					
Crested dog's-tail	*Cynosurus cristaus* L.	1			1	1	1		1	1	1	1	1					1	1	1	1
Daisy	*Bellis perennis*				1				1	1	1		1						1	1	1
Dandelion	*Taraxacum* agg.																1				
Germander speedwell	***Veronica chamaedrys***				**1**												**1**				
Lesser trefoil	***Trifolium dubium***							**1**		**1**			**1**								
Marsh-marigold	*Caltha palustris*													1							
Meadow buttercup	***Ranunculus acris***	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**
Meadow fox-tail	*Alopecurus pratensis* L.	1							1						1						
Perennial rye-grass	*Lolium perenne* L.	1	1	1	1	1	1		1	1	1	1	1	1	1	1	1	1	1		1
Pignut	*Conopodium majus*				1					1			1								
Red clover	***Trifolium pratense***				**1**	**1**	**1**	**1**	**1**	**1**	**1**		**1**								
Red fescue	*Festuca rubra* L.							1											1		1
Ribwort plantain	***Plantago lanceolata***					**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**								
Rough hawk's-beard	*Crepis biennis*											1									
Rough meadow-grass	*Poa trivialis* L.	1		1	1	1			1			1	1			1			1	1	1
Selfheal	***Prunella vulgaris***						**1**	**1**													
Sharp-flowered rush	*Juncus acutiflorus*							1													
Sheep's fescue	*Festuca ovina* L.	1													1		1	1	1	1	1
Small-leaved Timothy-grass	*Phleum pratense* subsp. *bertolonii*													1							
Smooth meadow-grass	*Poa pratensis* L.	1											1	1			1		1		
Soft brome	*Bromus mollis* L.	1							1	1	1		1					1	1		1
Sweet vernal-grass	*Anthoxanthum odoratum* L.	1				1	1	1		1	1	1	1		1		1	1		1	1
Timothy-grass	*Phleum pratense* L.																		1		
White clover	*Trifolium repens*	1	1		1		1		1	1	1	1	1	1	1	1	1	1	1	1	1
Yarrow	***Achillea millefolium***									**1**			**1**								
Yellow oat-grass	*Trisetum flavescens* (L.) Beauv.									1			1				1				
Yellow-rattle	*Rhinanthus minor*	1	1	1	1	1	1	1			1	1									
Yorkshire fog	*Holcus lanatus* L.	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Total species richness; or mean Ellenburg N		18	11	10	15	15	16	14	17	18	15	15	21	11	11	9	14	12	16	11	13
Mean quadrat^{-1}		13.5				15.5				17.3				11.3				13.0			
SE quadrat^{-1}		1.85				0.65				1.44				1.03				1.08			
Total no. semi-improved grassland indicator species		3	3	3	4	5	6	7	5	6	4	4	7	2	2	2	3	2	2	2	2
Mean quadrat^{-1}		3.3				5.8				5.3				2.3				2.0			
SE quadrat^{-1}		0.25				0.48				0.75				0.25				0.00			

Ellenburg N

	Non-fertilised Field →	a				b				c				d				e			
2011	Quadrat →	1	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4
Autumn hawkbit	***Scorzoneroides autumnalis***					4	4	4	4												
Black medick	***Medicago lupulina***	4	4	4								4	4								
Broad-leaved dock	*Rumex obtusifolius*								9												
Cock's foot	*Dactylis glomerata* L.	6	6		6													6			
Common bent	*Agrostis capillaris* L.	4	4	4		4	4	4	4	4	4				4		4	4		4	
Common eyebright	*Euphrasia nemorosa*					4	4	4													
Common knapweed	*Centaurea nigra*											5	5								
Common mouse ear	*Cerastium fontanum*	4	4	4	4	4	4		4	4	4	4	4	4	4	4	4	4	4	4	4
Common sorrel	***Rumex acetosa***	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4
Creeping bent	*Agrostis stolonifera* L.													6					6		
Creeping buttercup	*Ranunculus repens*	7	7	7	7	7	7		7	7	7	7	7	7	7	7	7		7		
Creeping soft-grass	*Holcus mollis* L.															3					
Crested dog's-tail	*Cynosurus cristaus* L.	4			4	4	4		4	4	4	4	4					4	4	4	4
Daisy	*Bellis perennis*				4				4	4	4		4						4	4	4
Dandelion	*Taraxacum* agg.																6				
Germander speedwell	***Veronica chamaedrys***				5												5				
Lesser trefoil	***Trifolium dubium***							5		5			5								
Marsh-marigold	*Caltha palustris*													4							
Meadow buttercup	***Ranunculus acris***	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4
Meadow fox-tail	*Alopecurus pratensis* L.	7							7						7						
Perennial rye-grass	*Lolium perenne* L.	6	6	6	6	6	6		6	6	6	6	6	6	6	6	6	6	6		6
Pignut	*Conopodium majus*				5					5			5								
Red clover	***Trifolium pratense***				5	5	5	5	5	5	5		5								
Red fescue	*Festuca rubra* L.							5											5		5
Ribwort plantain	***Plantago lanceolata***					4	4	4	4	4	4	4	4								
Rough hawk's-beard	*Crepis biennis*											6									
Rough meadow-grass	*Poa trivialis* L.	6		6	6	6			6			6	6			6			6	6	6
Selfheal	***Prunella vulgaris***						4	4													
Sharp-flowered rush	*Juncus acutiflorus*							2													
Sheep's fescue	*Festuca ovina* L.	2													2		2	2	2	2	2
Small-leaved Timothy-grass	*Phleum pratense* subsp. *bertolonii*													4							
Smooth meadow-grass	*Poa pratensis* L.	5											5	5			5		5		
Soft brome	*Bromus mollis* L.	4							4	4	4		4					4	4		4
Sweet vernal-grass	*Anthoxanthum odoratum* L.	3				3	3	3		3	3	3	3		3		3	3		3	3
Timothy-grass	*Phleum pratense* L.																		6		
White clover	*Trifolium repens*	6	6		6		6		6	6	6	6	6	6	6	6	6	6	6	6	6
Yarrow	***Achillea millefolium***									4			4								
Yellow oat-grass	*Trisetum flavescens* (L.) Beauv.									4			4				4				
Yellow-rattle	*Rhinanthus minor*	4	4	4	4	4	4	4			4	4									
Yorkshire fog	*Holcus lanatus* L.	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5
Total species richness; or mean Ellenburg N		4.7	5	5	5	4.5	5	4	5	4.6	5	5	5	5	5	5	5	4.3	5	4	4
Mean quadrat^{-1}		4.9				4.6				4.6				4.8				4.4			
SE quadrat^{-1}		0.06				0.21				0.06				0.09				0.15			

Table 3C. 2012 Fertilised fields: species richness, Ellenburg N and number of semi-improved grassland indicator species (marked in bold).

		Species richness																			
	Fertilised Field →	a				b				c				d				e			
2012	Quadrat →	1	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4
Broad-leaved dock	*Rumex obtusifolius*						1									1	1		1		
Changing forget-me-not	*Myosotis discolor*				1																
Cock's foot	*Dactylis glomerata* L.				1										1		1				
Common bent	*Agrostis capillaris* L.	1	1	1					1						1	1	1				
Common chickweed	*Stellaria media*						1			1						1	1				
Common mouse ear	*Cerastium fontanum*	1		1	1	1	1	1		1	1	1	1	1	1	1	1	1			
Common sorrel	***Rumex acetosa***		**1**	**1**						**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**				
Creeping bent	*Agrostis stolonifera* L.		1	1										1		1	1				
Creeping buttercup	*Ranunculus repens*	1	1	1	1						1	1	1		1	1	1				
Creeping soft-grass	*Holcus mollis* L.				1				1						1						
Crested dog's-tail	*Cynosurus cristaus* L.		1		1						1	1	1								
Cuckooflower	***Cardamine pratensis***		**1**	**1**	**1**													**1**			
Daisy	*Bellis perennis*		1							1	1	1	1								
Dandelion	*Taraxacum* agg.									1	1		1								
Germander speedwell	***Veronica chamaedrys***							**1**													
Marsh fox-tail	*Alopecurus geniculatus* L.	1												1		1			1		
Meadow buttercup	***Ranunculus acris***		**1**		**1**					**1**	**1**	**1**	**1**			**1**					
Meadow fox-tail	*Alopecurus pratensis* L.						1		1	1					1	1	1	1	1	1	
Perennial rye-grass	*Lolium perenne* L.	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Pignut	*Conopodium majus*									1											
Red clover	***Trifolium pratense***		**1**								**1**										
Red fescue	*Festuca rubra* L.													1			1				
Ribwort plantain	***Plantago lanceolata***										**1**										
Rough hawksbeard	*Crepis biennis*										1	1									
Rough meadow-grass	*Poa trivialis* L.	1	1	1	1	1	1	1	1	1	1	1	1		1	1	1	1	1	1	1
Sheep's fescue	*Festuca ovina* L.			1											1						
Soft brome	*Bromus hordeaceus* L.				1	1	1	1	1	1	1	1	1					1	1	1	1
Sweet vernal-grass	*Anthoxanthum odoratum* L.				1						1	1	1						1		
White clover	*Trifolium repens*	1	1	1	1					1	1	1	1	1	1	1	1				
Yarrow	***Achillea millefolium***									**1**		**1**									
Yellow oat-grass	*Trisetum flavescens* (L.) Beauv.														1						
Yorkshire fog	*Holcus lanatus* L.		1	1	1	1	1	1	1	1	1	1	1	1	1	1			1	1	1
Total species richness; or mean Ellenburg N		7	13	11	14	5	8	6	7	14	16	14	13	8	13	14	13	6	8	5	4
Mean quadrat^{-1}		11.3				6.5				14.3				12.0				5.8			
SE quadrat^{-1}		1.55				0.65				0.63				1.35				0.85			
Total no. semi-improved grassland indicator species		0	4	1	1	0	0	0	0	3	4	3	2	2	0	3	1	0	1	0	0
Mean quadrat^{-1}		1.5				0.0				3.0				1.5				0.3			
SE quadrat^{-1}		0.87				0.00				0.41				0.65				0.25			

		Ellenburg N																			
	Fertilised Field →	a				b				c				d				e			
2012	Quadrat →	1	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4
Broad-leaved dock	*Rumex obtusifolius*						9									9	9		9		
Changing forget-me-not	*Myosotis discolor*				3																
Cock's foot	*Dactylis glomerata* L.				6										6		6				
Common bent	*Agrostis capillaris* L.	4	4	4					4						4	4	4				
Common chickweed	*Stellaria media*						7			7						7	7				
Common mouse ear	*Cerastium fontanum*	4		4	4	4	4	4		4	4	4	4	4	4	4	4	4			
Common sorrel	***Rumex acetosa***		4	4						4	4	4	4	4	4	4	4				
Creeping bent	*Agrostis stolonifera* L.		6	6										6		6	6				
Creeping buttercup	*Ranunculus repens*	7	7	7	7						7	7	7		7	7	7				
Creeping soft-grass	*Holcus mollis* L.				3				3						3						
Crested dog's-tail	*Cynosurus cristaus* L.		4		4						4	4	4								
Cuckooflower	***Cardamine pratensis***		4	4	4													4			
Daisy	*Bellis perennis*		4							4	4	4	4								
Dandelion	*Taraxacum* agg.									6	6		6								
Germander speedwell	***Veronica chamaedrys***							5													
Marsh fox-tail	*Alopecurus geniculatus* L.	6												6		6			6		
Meadow buttercup	***Ranunculus acris***		4		4					4	4	4	4			4					
Meadow fox-tail	*Alopecurus pratensis* L.						7		7	7					7	7	7	7	7	7	
Perennial rye-grass	*Lolium perenne* L.	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6
Pignut	*Conopodium majus*									5											
Red clover	***Trifolium pratense***		5								5										
Red fescue	*Festuca rubra* L.													5			5				
Ribwort plantain	***Plantago lanceolata***										4										
Rough hawksbeard	*Crepis biennis*										6	6									
Rough meadow-grass	*Poa trivialis* L.	6	6	6	6	6	6	6	6	6	6	6	6		6	6	6	6	6	6	6
Sheep's fescue	*Festuca ovina* L.			2											2						
Soft brome	*Bromus hordeaceus* L.				4	4	4	4	4	4	4	4	4					4	4	4	4
Sweet vernal-grass	*Anthoxanthum odoratum* L.				3						3	3	3						3		
White clover	*Trifolium repens*	6	6	6	6					6	6	6	6	6	6	6	6				
Yarrow	***Achillea millefolium***									4		4									
Yellow oat-grass	*Trisetum flavescens* (L.) Beauv.														4						
Yorkshire fog	*Holcus lanatus* L.		5	5	5	5	5	5	5	5	5	5	5	5	5	5			5	5	5
Total species richness; or mean Ellenburg N		5.6	5	5	5	5	6	5	5	5.1	5	5	5	5.3	5	6	6	5.2	6	6	5
Mean quadrat^{-1}		5.0				5.3				4.9				5.5				5.4			
SE quadrat^{-1}		0.20				0.25				0.08				0.23				0.14			

Table 3D. (Continued on next page). 2012 Non-fertilised fields: species richness, Ellenburg N and number of semi-improved grassland indicator species (marked in bold).

		Species richness																				Ellenburg N																			
	Non-fertilised Field →	a				b				c				d				e				a				b				c				d				e			
2012	Quadrat →	1	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4
Annual meadow grass	*Poa annua* L.								1																				7												
Autumn hawkbit	***Scorzoneroides autumnalis***					1	1	1	1																	4	4	4	4												
Cat's ear	*Hypochaeris radicata*	1	1														1					3	3														3				
Changing forget-me-not	*Myosotis discolor*	1			1	1	1	1								1	1					3			3	3	3	3								3	3				
Cock's foot	*Dactylis glomerata* L.		1		1							1	1										6		6							6	6								
Common bent	*Agrostis capillaris* L.														1		1																		4		4				
Common chickweed	*Stellaria media*														1				1	1															7				7	7	
Common eyebright	*Euphrasia nemorosa*					1	1	1	1					1	1		1									4	4	4	4					4	4		4				
Common mouse ear	*Cerastium fontanum*	1		1	1		1		1	1		1	1	1	1	1	1	1	1	1	1	4		4	4		4		4	4		4	4	4	4	4	4	4	4	4	4
Common sorrel	***Rumex acetosa***	1	1	1	1	1		1	1	1	1	1	1	1	1	1	1	1	1	1	1	4	4	4	4	4		4	4	4	4	4	4	4	4	4	4	4	4	4	4
Creeping bent	*Agrostis stolonifera* L.													1	1		1																	6	6		6				
Creeping buttercup	*Ranunculus repens*	1	1	1	1		1	1	1	1	1	1	1	1	1	1	1					7	7	7	7		7	7	7	7	7	7	7	7	7	7	7				
Crested dog's-tail	*Cynosurus cristaus* L.	1	1		1	1	1	1	1	1	1	1					1	1	1	1	1	4	4		4	4	4	4	4	4	4	4					4	4	4	4	4
Cuckooflower	***Cardamine pratensis***													1		1	1	1	1															4		4	4	4	4		
Daisy	*Bellis perennis*	1	1	1	1		1	1		1	1	1	1					1	1		1	4	4	4	4		4	4		4	4	4	4					4	4		4
Dandelion	*Taraxacum* agg.	1						1			1		1	1	1		1	1	1			6						6			6		6	6	6		6	6	6		
False oat grass	*Arrhenatherum elatius* (L.) Beauv.			1																				7																	
Germander speedwell	***Veronica chamaedrys***				1																				5																
Lesser trefoil	***Trifolium dubium***	1		1			1	1														5		5			5	5													
Marsh fox-tail	*Alopecurus geniculatus* L.						1	1	1					1	1	1	1										6	6	6					6	6	6	6				
Marsh-marigold	*Caltha palustris*													1		1																		4		4					
Meadow buttercup	***Ranunculus acris***	1		1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	4		4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4
Meadow fox-tail	*Alopecurus pratensis* L.		1													1							7													7					
Oval sedge	*Carex ovalis*					1																				4															
Pearlwort	*Sagina* sp.							1																																	
Perennial rye-grass	*Lolium perenne* L.	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6
Pignut	*Conopodium majus*		1		1					1	1												5		5					5	5										
Red clover	***Trifolium pratense***	1		1	1	1	1	1	1	1	1	1	1				1					5		5	5	5	5	5	5	5	5	5	5				5				

Table 3D. (Continued). 2012 Non-fertilised fields: species richness, Ellenburg N and number of semi-improved grassland indicator species (marked in bold).

Red fescue	*Festuca rubra* L.				1													1	1	1	1
Ribwort plantain	***Plantago lanceolata***				**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**								
Rough hawk's-beard	*Crepis biennis*	1										1									
Rough meadow-grass	*Poa trivialis* L.		1	1			1		1			1		1		1			1	1	1
Rush species	*Juncus* sp.					1		1													
Selfheal	***Prunella vulgaris***	**1**				**1**	**1**									**1**					
Soft brome	*Bromus mollis* L.	1		1	1					1	1	1	1			1		1	1		1
Soft rush	*Juncus effusus*																		1		
Sweet vernal-grass	*Anthoxanthum odoratum* L.	1	1		1	1	1	1	1	1	1	1	1	1			1	1	1	1	1
Wall speedwell	*Veronia arvensis*	1		1										1							
White clover	*Trifolium repens*	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Wild carrot	*Daucus carota* ssp. *Carota*				1																
Yarrow	***Achillea millefolium***											**1**	**1**								
Yellow oat-grass	*Trisetum flavescens* (L.) Beauv.																	1	1	1	
Yellow-rattle	*Rhinanthus minor*	1	1	1	1	1	1	1	1	1	1	1						1			
Yorkshire fog	*Holcus lanatus* L.	1	1	1	1	1	1	1		1	1	1	1	1	1	1	1	1	1	1	1
Total species richness; or mean Ellenburg N		20	14	15	20	16	19	20	16	15	15	18	17	16	13	15	18	15	17	12	13
Mean quadrat^{-1}		17.3				17.8				16.3				15.5				14.3			
SE quadrat^{-1}		1.60				1.03				0.75				1.04				1.11			
Total no. semi-improved grassland indicator species		5	1	4	4	5	8	7	5	4	4	5	5	3	3	3	4	2	2	1	2
Mean quadrat^{-1}		3.5				6.3				4.5				3.3				1.8			
SE quadrat^{-1}		0.87				0.75				0.29				0.25				0.25			

Red fescue	*Festuca rubra* L.				5													5	5	5	5
Ribwort plantain	***Plantago lanceolata***				4	4	4	4	4	4	4	4	4								
Rough hawk's-beard	*Crepis biennis*	6										6									
Rough meadow-grass	*Poa trivialis* L.		6	6			6		6			6		6		6			6	6	6
Rush species	*Juncus* sp.																				
Selfheal	***Prunella vulgaris***	4				4	4									4					
Soft brome	*Bromus mollis* L.	4		4	4					4	4	4	4			4		4	4		4
Soft rush	*Juncus effusus*																		4		
Sweet vernal-grass	*Anthoxanthum odoratum* L.	3	3		3	3	3	3	3	3	3	3	3	3			3	3	3	3	3
Wall speedwell	*Veronia arvensis*	5		5										5							
White clover	*Trifolium repens*	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6
Wild carrot	*Daucus carota* ssp. *Carota*				3																
Yarrow	***Achillea millefolium***											4	4								
Yellow oat-grass	*Trisetum flavescens* (L.) Beauv.																	4	4	4	4
Yellow-rattle	*Rhinanthus minor*	4	4	4	4	4	4	4	4	4	4	4	4					4			
Yorkshire fog	*Holcus lanatus* L.	5	5	5	5	5	5	5		5	5	5	5	5	5	5	5	5	5	5	5
Total species richness; or mean Ellenburg N		4.6	5	5	5	4.3	5	5	5	4.6	5	5	5	5	5	5	5	4.5	5	5	5
Mean quadrat^{-1}		4.8				4.6				4.7				5.0				4.6			
SE quadrat^{-1}		0.13				0.13				0.04				0.13				0.08			

Figure 3A. Relationship between biomass yield and other field variables.

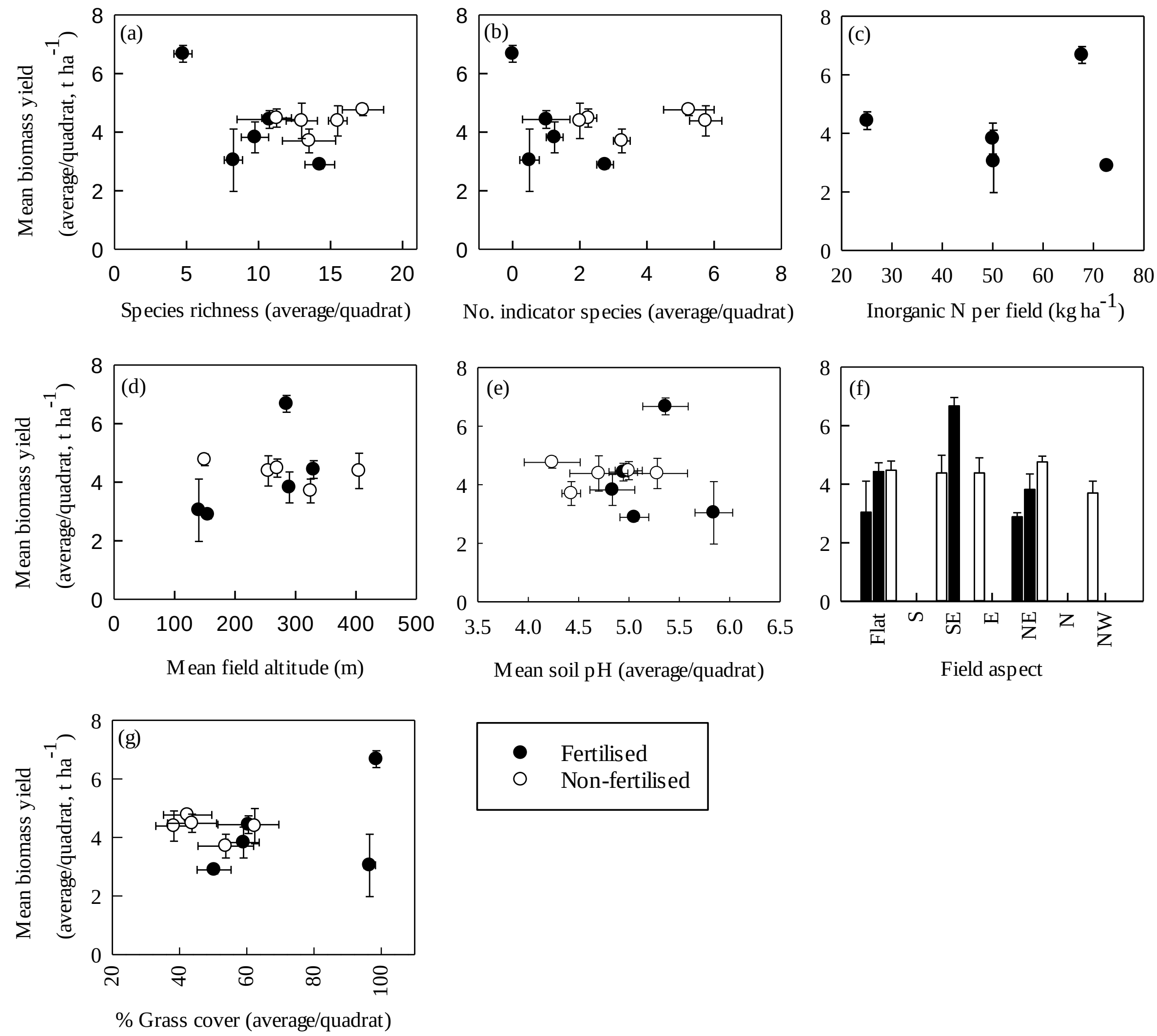

Table 3E. Number of ewes and lambs per ha which graze grass.

(a) Number of ewes/ha

Fertilised	Jan	Feb	Mar	April	May	June	July	Aug	Sep	Oct	Nov	Dec
a	0.0	0.0	0.0	12.4	12.4	0.0	0.0	0.0	0.0	0.0	6.2	0.0
b	0.0	0.0	0.0	16.3	0.0	0.0	0.0	0.0	26.1	26.1	0.0	13.1
c	0.0	0.0	0.0	9.3	14.9	0.0	0.0	0.0	0.0	0.0	21.2	4.5
d	0.0	0.0	0.0	5.4	8.6	8.6	0.0	0.0	0.0	0.0	10.3	5.2
e	0.0	0.0	0.0	7.6	7.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Non-fertilised												
a	0.0	0.0	0.0	12.4	12.4	0.0	0.0	0.0	0.0	0.0	6.2	0.0
b	0.0	0.0	0.0	3.6	5.0	0.0	0.0	0.0	0.0	0.0	7.2	3.6
c	0.0	0.0	0.0	9.3	14.9	0.0	0.0	0.0	0.0	0.0	21.2	4.5
d	0.0	0.0	0.0	6.8	10.8	0.0	0.0	0.0	0.0	0.0	10.8	5.4
e	0.0	0.0	0.0	4.1	4.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0

(b) Number of lambs/ha

Fertilised	Jan	Feb	Mar	April	May	June	July	Aug	Sep	Oct	Nov	Dec
a	0.0	0.0	0.0	14.8	14.8	0.0	0.0	1.9	3.7	3.7	0.0	0.0
b	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
c	0.0	0.0	0.0	17.9	8.9	0.0	0.0	0.0	22.3	22.3	0.0	5.4
d	0.0	0.0	0.0	6.0	13.7	12.9	0.0	0.0	0.0	25.8	0.0	0.0
e	0.0	0.0	0.0	4.6	9.1	0.0	0.0	20.6	0.0	0.0	0.0	0.0
Non-fertilised												
a	0.0	0.0	0.0	14.8	14.8	0.0	0.0	1.9	3.7	3.7	0.0	0.0
b	0.0	0.0	0.0	0.0	10.0	0.0	0.0	0.0	14.3	0.0	0.0	0.0
c	0.0	0.0	0.0	17.9	8.9	0.0	0.0	0.0	22.3	22.3	0.0	5.4
d	0.0	0.0	0.0	10.8	18.9	0.0	0.0	0.0	0.0	27.0	27.0	0.0
e	0.0	0.0	0.0	2.5	5.0	0.0	0.0	12.2	0.0	0.0	0.0	0.0

Numbers are red where sheep are present but they are fed hay therefore number of sheep grazing grass is zero, even if they are on the field; or numbers are red where sheep graze for part of the month.

Table 3F.Mean grazed, harvested (gross yield) and total annual biomass yield per field (air-dried t/ha, 90% DM).

	Field	Fertilised	Non-fertilised
Grazed	a	1.77	1.77
	b	2.86	1.14
	c	3.30	3.30
	d	2.48	2.75
	e	1.33	0.75
Harvested (gross)	a	4.43	3.70
	b	3.04	4.38
	c	2.89	4.76
	d	3.80	4.48
	e	6.68	4.39
Total annual (grazed + harvested)	a	6.20	5.47
	b	5.90	5.52
	c	6.19	8.06
	d	6.28	7.23
	e	8.00	5.13

Table 3G. Variables that were significantly different among fertilised fields; or among non-fertilised fields. NS is not significant.

Variable	Among fertilised		Among non-fertilised	
	Statistical test	*P*	Statistical test	*P*
Harvested biomass yield	Kruskal-Wallis	0.014		NS
Species richness	1-way ANOVA	<0.001	1-way ANOVA	<0.036
Semi-imp. indicator species	Kruskal-Wallis	0.013	Kruskal-Wallis	0.002
Soil pH	1-way ANOVA	0.048	1-way ANOVA	0.012
Ellenburg N	1-way ANOVA	0.025		NS
Ellenburg R	1-way ANOVA	0.012		NS
Soil moisture	1-way ANOVA	<0.001		NS
Soil organic carbon		NS	Kruskal-Wallis	0.022
% Grass cover	1-way ANOVA	<0.001		NS

APPENDIX TO CHAPTER 4.

4. Method development for a biomethane assay (by anaerobic digestion)

4.1 Using manure as an inoculum

In attempting to develop a biomethane assay, I first performed three AD experiments using cow manure as an inoculum (collected from local common grassland), and mature grass cut from the university campus as the feedstock (containing mainly Yorkshire fog (*Holcus lanatus*), with small amounts of false oat-grass (*Arrhenatherum elatius*), rough meadow grass (*Poa trivialis*) and cock's foot (*Dactylis glomerata*)). The fresh grass was chopped finely in a coffee grinder then used in AD. Volatile solids (VS) content of manure and grass was determined (as shown in the Methods Section of Chapter 4) to ensure their correct ratio. VS is a measure of organic content. All experiments were performed in 100 ml Wheaton bottles (Supelco, Bellefonte, PA, USA), at 37^{o}C as per the methods in Chapter 4 (Section 2.3). These experiments gave me useful information on conditions to use in AD in 100 ml bottles, summarised below.

4.1.1 The first experiment aimed to investigate which final reaction volume (50, 60 or 70 ml with water, performed in triplicate) produced the most methane, using a feedstock to inoculum volatile solids' content (F/I) ratio of 1.5, as used by Liu *et al.* (2009). It lasted 20 days. I used 6g of mature grass from the University of York campus and 5.1 ml of manure. At day 19, the 50 ml reactions produced the most methane (i) in the manure-only controls (5.03 Nml), and (ii) in the manure plus grass reaction (0.029 Nml). However, I had made an error in calculating the amount of grass to use: 6g was too much and this led to a high rate of hydrolysis, producing excessive volatile fatty acid and a fall in pH (to pH 5), which inhibits methanogenesis (Schink 2008). The recalcitrance of the mature vegetation may also have hindered hydrolysis. A final AD reaction volume of 50 ml was determined to be the best volume, and this was used in subsequent experiments.

4.1.2 A second experiment tested whether water or phosphate buffer (pH 6.8) was a better back-ground medium in AD of grass and manure. 5 ml manure and 1.5 g chopped, frozen and defrosted lawn grass were used in a final volume of 50 ml, at a F/I ratio of 1.0. It lasted 147 days. In the AD with water, one grass replicate produced methane (154 Nml CH_4g^{-1} grass VS), while the two other grass replicates did not; but grass AD in buffer produced no more methane than the control. Therefore, I had shown that (i) methane was produced from one grass replicate (compared to the complete failure of the first experiment); and (ii) subsequent experiments should be performed in water.

4.1.3 A third AD experiment using manure and grass was performed (lasting 113 days), but it did not add any useful information.

4.2 Using digestate from a sewage anaerobic digester as inoculum

4.2.1 The aim of this preliminary experiment with sewage digestate was to determine (i) if it could produce methane from grass, and (ii) to compare if 'anaerobic grass' (flushed with nitrogen for 3 hours before being added to the inoculum) produced more methane than 'aerobic grass' (which was left open to the air). 0.5 litre of digestate was collected from each of 3 depths (0.5 m, 2.3 m and 4.2 m) of an anaerobic digester treating sewage at the local sewage treatment plant (Naburn, York). The digestate (pH 8, 2.16% VS of fresh weight) was mixed in an anaerobic chamber. This experiment was set up in larger bottles (3 x 1 litre, containing 330 ml digestate in each) than previously because methane production can be lower in smaller AD vessels compared to larger vessels (Nizami 2012). Treatments were 1 x aerobic grass, 1 x anaerobic grass and 1 x digestate-only control. The grass was frozen, defrosted, chopped lawn clippings (21.3% total solids (TS), 17.6% VS of fresh weight)) which were then ground in a coffee grinder. It was decided to add the grass in three batches to each bottle to avoid overloading the microbial system with feedstock it had not encountered before; methanogenic systems usually require time to adjust to new feedstocks. The overall F/I ratio was 0.5 and 5.99g grass was added 3 times at 3-week intervals, up to a total of 17.97 g grass. All additions of grass occurred in the anaerobic chamber. The control bottle was also opened at the same time, but no grass was added. Methane produced by the digestate control was subtracted from methane produced by the grass experiments.

Adding grass approximately doubled the methane production compared to digestate alone (Fig 4A). The digestate pH remained high (7.5) at the end of the experiment, indicating a balanced microbial system with no acid accumulation. There had been 42% volatile solids destruction in both aerobic and anaerobic grass reactions. Aerobic grass produced 284.2 Nml CH_4g^{-1} VS, and anaerobic grass produced 258.8 Nml CH_4g^{-1} VS. Therefore sewage digestate was a suitable inoculum for grass AD, and because aerobic grass produced more methane than anaerobic grass, it was not necessary to make grass anaerobic before AD.

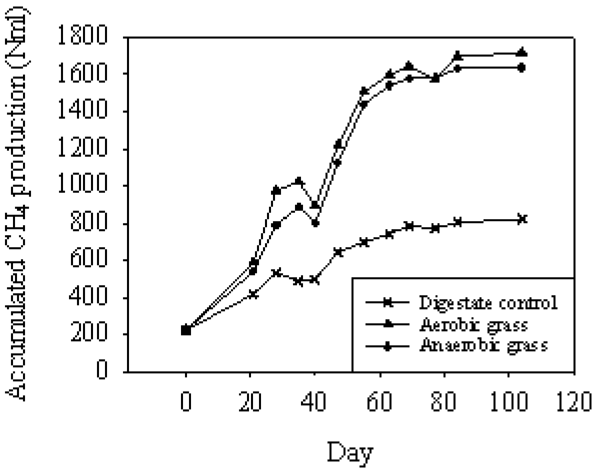

Figure 4A. AD using sewage digestate and grass. Grass was added on day 0, 21 and 47.

4.2.2 The use of sewage digestate as an inoculum was optimised in this final method development experiment. The questions asked were:

a) Is the preliminary sewage digestate experiment (4.2.1) reproducible using freshly collected inoculum?
b) Does AD in 100 ml bottles give the same results as 1 litre bottles?
c) Which F/I ratio is best: 0.5 or 1?
d) Can grass be added all at the beginning or must it be divided into batches?

The experiment's treatments are shown in Table 4A. Digestate-only controls were included. Question a) was answered by Treatment 1; question b) was answered by Treatment 2; question c) was answered by comparing Treatment 2 to 5, 3 to 6, 4 to 7; and question d) was answered by Treatments 2-4 for F/I ratio 0.5, and Treatments 5-7 for F/I ratio 1.

The inoculum was the same as the previous experiment (4.2.1) (sewage digestate from an anaerobic digester at Naburn Sewage works), but it was collected fresh. All reactions were set up in the anaerobic chamber. The digestate pH was 8 and it contained 2.10% VS of fresh weight. The same grass feedstock was used (4.2.1), and all treatments were performed in triplicate. In 1 litre bottles, 330 ml digestate was used and in 100 ml bottles 50 ml digestate was used. Bottles were put in a shaking incubator at 37°C and gas pressures and/or gas samples taken every day in the first week, then less often as the rate of CH_4 production reduced. Bottles and their digestate controls were opened in the anaerobic chamber when grass was added to a treatment. When grass was added in 3 batches, there was a gap of 3 weeks between additions (as was approximately done in 4.2.1); when grass was added in 2 batches, there was a gap of 2 weeks between batches. Gas was released from treatment bottles when headspace pressure rose above approximately 28 psi: in the fume hood, a needle was inserted into the bottle bung and removed quickly when the hissing noise slowed. Pressure and gas samples were taken before and after release of pressure.

Table 4A. Conditions tested in optimisation of the AD assay using sewage digestate as inoculum and lawn grass as the feedstock.

Treatment	Bottle volume	No. batches grass	F/I ratio	Grass added per batch (g)	Total grass added (g)
1	1 litre	3	0.5	6.38	19.13
2	50 ml	3	0.5	0.97	2.9
3	50 ml	1	0.5	2.9	2.9
4	50 ml	2	0.5	1.45	2.9
5	50 ml	3	1	1.933	5.8
6	50 ml	1	1	5.8	5.8
7	50 ml	2	1	2.9	5.8

Results

- *Question a)* Is the preliminary sewage digestate experiment (4.2.1) reproducible?

Yes, the preliminary experiment in 1 litre bottles was reproducible, producing similar accumulate and specific CH_4 yields (per g grass VS). Methane accumulation was 1713 Nml CH_4, compared to the previous experiment's 1638-1718 Nml CH_4; and mean specific CH_4 production was also similar: 279 Nml CH_4g^{-1} VS in this experiment (Table 4B); and 259-284 in the preliminary experiment.

- *Question b)* Does AD in 100 ml bottles give the same results as 1 litre bottles?
- *Question d)* Can grass be added all at once or must it be divided into batches?

Figure 4B shows methane accumulation in treatments 2-4 which answer these questions. When grass was added in 1 batch on day 0, CH_4 production increased very rapidly, which may be due to the presence of grass-like vegetable-degrading microbes in human guts. The more grass added at the beginning of the experiment, the higher the rate of accumulation of CH_4. When the grass was added in 1 batch, CH_4 production plateaued after approximately 70 days, though it reached near-peak production after 30 days. After the final addition of grass in the 3-batch treatment, CH_4 accumulation quickly reached the same level as the other treatments because the total amount of grass added was the same. The controls accumulated more CH_4 when they were opened more.

Answering question b), mean specific methane production in 100 ml bottles (with grass added in 3 batches) was substantially lower (222 Nml CH_4g^{-1} VS) (Table 4B) than that achieved in 1 litre bottles (279 Nml CH_4g^{-1} VS), supporting the findings of Nizami (2012) that methane potential is smaller if performed in smaller vessels.

Answering question d), when the F/I ratio was 0.5, the grass could all be added at the beginning of the experiment (in 1 batch) because there was no evidence of inhibition, and final methane yield was reached more quickly than when grass was added in 3 batches.

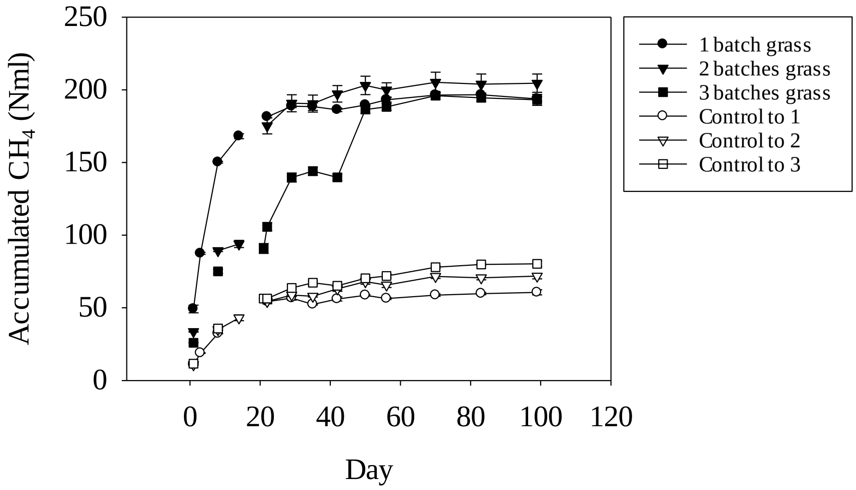

Figure 4B. AD of grass (F/I ratio 0.5) in 100 ml bottles. Grass was added in 1, 2 or 3 batches (treatments 3, 4, 2 respectively). Controls were opened each time a treatment bottle was opened. Error bars are SE of the triplicates per treatment.

- *Question c)* Which F/I ratio is best: 0.5 or 1?
- *Question d)* Can grass be added all at once or must it be divided into batches?

When the F/I was 1, methane accumulated in a very similar pattern to the lower F/I but because 100% more grass was used, final accumulated CH_4 yields were 70-92% higher. Specific methane yields were 280-287 Nml CH_4g^{-1} VS when the F/I was 1 (Table 4B).

Answering question c), a F/I ratio of 1.0 produced higher specific methane yields than when the F/I was 0.5 (Table 4B), although headspace pressure grew too high and had to be released 3 times from each treatment. When the F/I is 1, the answer to question d) is that grass can be added in one batch at the start of the experiment without significantly impacting specific methane yield.

Specific methane yields of all the treatments

Mean specific methane yields of all the treatments (Table 4B) were significantly different to each other ($p < 0.001$, one-way ANOVA). Post-hoc LSD test showed that treatment 2 was significantly lower than all other treatments, and the highest yielding treatments 5, 6 and 7 were not significantly different to each other.

Table 4B. Specific CH_4 yields (CH_4 production/g grass VS added) for all the treatments (listed in Table 4A). Triplicates were performed per treatment.

Treatment	Mean specific CH4 yield ($Nml\ CH_4\ g^{-1}\ VS$)	SE
1	279	4.0
2	222	7.4
3	261	3.4
4	261	12.4
5	286	5.8
6	280	3.1
7	287	2.6

Treatment 6 had the highest % CH_4 content (55.5%) in its biogas. The others ranged from 50.8 – 55.5 % CH_4.

Conclusion

In conclusion, sewage digestate was a reliable inoculum, which could digest grass with reproducible results when collected from the sewage plant at different times. The conditions chosen for the AD assay, which was to be used to digest biomass from the fertilised and non-fertilised fields, were those used in treatment 6. That is, add all the grass in one batch at the start of the experiment; use a F/I of 1; but also be aware that the small (100 ml bottles) produced a lower specific methane yield than 1 litre bottles. The small bottles had to be used in the digestion of the field biomass because 49 samples needed to be digested in one experiment. Because the controls accumulated more methane when they were opened during the experiment, control headspace pressure would also be released when the pressure was released from the field biomass bottles.

4.3 Data from digesting fertilised and non-fertilised field biomass

Table 4C. Mean biomethane production data per field (average of 4 AD experiments per field).

		Methane production	
		Specific (SE)	Area (SE)
Field type	Farm	(Nm^3 CH_4 t^{-1} VS)	(Nm^3 CH_4 ha^{-1})
Fertilised	a	221 (7)	846 (74)
	b	222 (7)	589 (214)
	c	234 (5)	582 (36)
	d	234 (10)	792 (173)
	e	222 (10)	1256 (60)
Non-fertilised	a	212 (4)	683 (77)
	b	211 (8)	797 (101)
	c	226 (2)	933 (35)
	d	214 (5)	826 (65)
	e	227 (2)	847 (115)

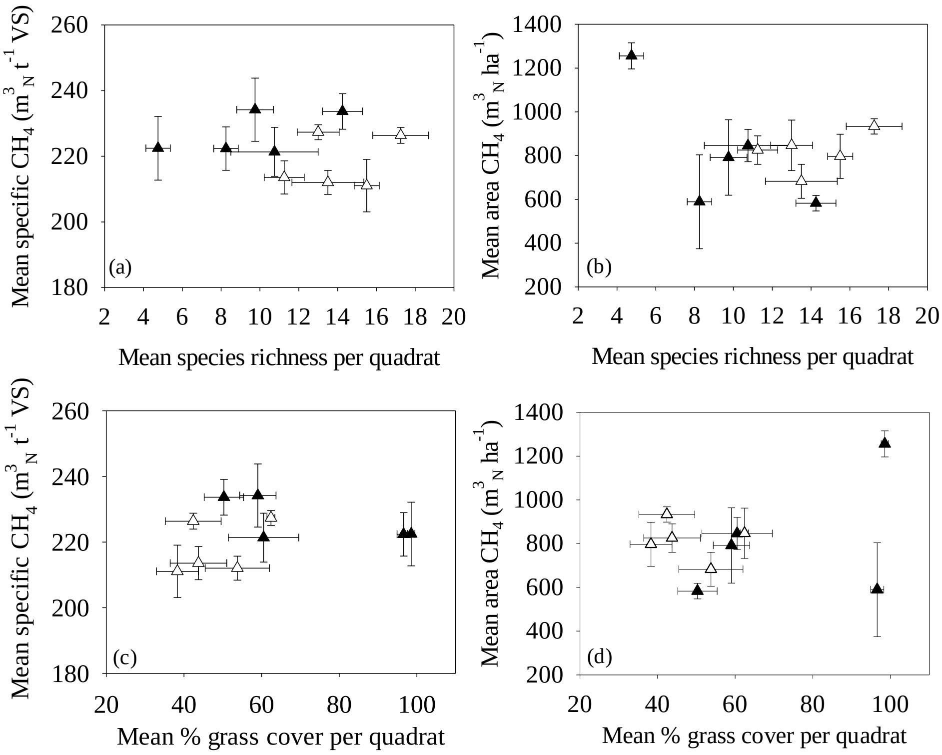

Fig. 4C. Effects of species richness and % grass cover on specific methane and area methane production. Error bars are SE (n = 4 quadrats). Black shows fertilised fields, white shows non-fertilised.

APPENDIX TO CHAPTER 5

Table 5A. Breakdown of annual GHG emissions per ha of each silage field and hay field.

Source of emissions		GHG ha^{-1} (kg)	Silage fields a	b	c	d	e	Hay fields a	b	c	d	e
Silage/hay production	Background soil	CO2										
		N2O	0.7	0.5	0.3	0.4	0.5	0.4	0.5	0.3	0.4	0.6
		CH4										
		as CO2e	200	145	100	113	134	109	138	100	113	165
		Mean CO2e	138					125				
	Soil: NPK, lime, FYM†	CO2	0.0	0.2	0.0	0.3	0.0	0.0	0.2	0.0	0.2	0.1
		N2O	0.7	0.8	0.7	0.7	0.8	0.4	0.4	0.4	0.4	0.5
		CH4										
		as CO2e	196	223	217	215	238	130	122	127	130	151
		Mean CO2e	218					132				
	NPK, lime production	CO2	109	237	210	276	199	9	38	9	51	28
		N2O										
		CH4										
		as CO2e	109	237	210	276	199	9	38	9	51	28
		Mean CO2e	206					27				
	Herbicide production	CO2	9.1	9.1	9.1	9.1	9.1	9.1	9.1	9.1	9.1	9.9
		N2O										
		CH4										
		as CO2e	9.1	9.1	9.1	9.1	9.1	9.1	9.1	9.1	9.1	9.9
		Mean CO2e	9.1					9.3				
	Fuel use	CO2	143	140	127	159	168	86	89	93	100	93
		N2O										
		CH4										
		as CO2e	143	140	127	159	168	86	89	93	100	93
		Mean CO2e	147					92				
Sheep production	Enteric fermentation	CO2										
		N2O										
		CH4	81	208	264	212	128	81	120	264	281	76
		as CO2e	2017	5200	6600	5306	3200	2017	3000	6600	7019	1900
		Mean CO2e	4465					4107				
	Sheep manure	CO2										
		N2O	1	3	4	2	1	1	1	4	2	1
		CH4	1	2	3	1	1	1	1	3	2	0
		as CO2e	418	953	1397	627	363	418	440	1397	744	209
		Mean CO2e	752					642				
	Conc. feed production	CO2	95	0	76	14	0	95	37	76	17	0
		N2O										
		CH4										
		as CO2e	95	0	76	14	0	95	37	76	17	0
		Mean CO2e	37					45				
	Hay feed production	CO2	0	59	42	11	0	0	0	0	0	0
		N2O	0	0	0	0	0	0	0	0	0	0
		CH4										
		as CO2e	0	180	136	29	0	0	0	0	0	0
		Mean CO2e	69					0				
TOTAL GHG PER HA		CO2e	3186	7088	8870	6748	4310	2872	3873	8411	8183	2555
		Mean CO2e	6040					5179				
		SE	1019					1292				

†Soil emissions after application of these

Table 5B. GHG emitted from each anaerobic digestion scenario. Transport and feedstock GHG; GHG per unit of electricity; % contribution of transport to GHG per unit of electricity; % reduction in GHG per unit of electricity when digesting hay rather than silage.

	Anaerobic digestion scenario	Transport GHG (kg CO_2e)	Feedstock GHG (kg CO_2e)	Total GHG (kg CO_2e)	Total kWh elec and heat produced	GHG per unit of elec (kg CO_2e $kWhe^{-1}$)	Contribution of transport to GHG per unit of elec (%)	% Reduction in GHG per unit of elec when digesting hay
2a	Co-op, silage	4856	66677	71533	920612	0.0777	6.8	
2b	Co-op, hay	3356	13815	17171	526617	0.0326	19.5	58
2c	Co-op, increased silage	5919	92428	98347	1138845	0.0864	6.0	
2d	Co-op, increased hay	4467	49588	54055	973199	0.0555	8.3	36
3a	Dairy, silage	898	66677	67575	718037	0.0941	1.3	
3b	Dairy, hay	83	13815	13898	324043	0.0429	0.6	54
3c	Dairy, increased silage	1245	92428	93673	936271	0.1000	1.3	
3d	Dairy, increased hay	298	49588	49886	770624	0.0647	0.6	35
3e	Dairy, own silage	0	11496	11496	250889	0.0458	0.0	
3f	Dairy, own hay	0	6168	6168	229511	0.0269	0.0	41

Assumes 100% heat used. Co-operative scenarios included emissions from transport of the crop, FYM and digestate. Dairy scenarios which buy in feedstock (3a-3d) included emissions from transport of the crop to the AD plant.

APPENDIX TO CHAPTER 6

Table 6A. Financial budget of current land management by farmers (growing grass crops and producing sheep) for each fertilised field (n = 5) and non-fertilised field (n = 5); units £ ha^{-1} $year^{-1}$. Unpaid labour by the farmer and their family is not included. Farms (a) and (c) had the same number of sheep per ha on silage and hay fields.

		Silage fields Each field					Hay fields Each field				
		a	b	c	d	e	a	b	c	d	e
Sales											
Sheep sales	Cast ewes	11	65	34	15	5	11	18	34	13	3
	Lambs - stores	123	0	181	177	0	123	123	181	255	0
	Lambs - mules	0	0	212	38	112	0	0	212	54	64
	Lambs - finished	57	0	0	45	201	57	0	0	65	115
	Wool sold	3	16	10	5	1	3	5	10	5	1
Total sheep sales		**195**	**81**	**437**	**280**	**319**	**195**	**146**	**437**	**392**	**183**
Variable costs											
Cultivation and seeding	Permanent pasture grass	0	8	0	13	9	0	0	0	25	18
Herbicide spraying	Spot spraying	8	8	8	8	8	6	6	6	6	9
Making silage or hay	Farmer and contractor costs	239	183	193	213	328	139	131	151	164	147
Fertiliser - delivery, spreading	NPK	41	70	63	70	70	0	0	0	0	0
Lime - delivery, spreading	Calcium lime	3	12	3	25	0	3	12	3	17	9
Sheep costs	Purchased concentrate feed	64	0	36	7	0	64	20	36	9	0
	Purchased hay feed	0	0	0	6	0	0	0	0	8	0
	Vet, shearing, scanning	29	150	92	48	13	29	42	92	43	7
Total variable costs		**384**	**430**	**395**	**391**	**428**	**241**	**210**	**289**	**272**	**190**
Gross margin		**-189**	**-350**	**42**	**-110**	**-108**	**-46**	**-65**	**148**	**119**	**-7**
Fixed costs											
Machinery	Depreciation, fuel, contract	131	130	128	134	135	120	121	121	123	121
Overheads	Farm maintenance, utilities	68	68	68	68	68	68	68	68	68	68
Rent & interest	Rent & interest	29	0	29	58	29	29	0	29	58	29
Livestock depreciation	Ram depreciation	55	11	34	18	5	15	11	34	16	3
Total fixed costs		**282**	**208**	**258**	**276**	**236**	**232**	**199**	**251**	**263**	**220**
Profit before subsidy/payments		**-471**	**-558**	**-216**	**-387**	**-345**	**-278**	**-264**	**-103**	**-144**	**-227**
Subsidy/payments											
Basic payment scheme		170	170	170	170	170	170	170	170	170	170
Agri-environment scheme		0	0	62	0	0	275	275	275	275	62
Total subsidy/payments		**170**	**170**	**232**	**170**	**170**	**445**	**445**	**445**	**445**	**232**
Profit after total subsidy/payments		**-301**	**-388**	**16**	**-217**	**-175**	**168**	**182**	**342**	**301**	**5**

Table 6B. Results of the co-operative AD scenarios (8 sheep farmers) before land subsidy/payments were included. 2a and 2b digest silage/hay from current area of production on livestock farm; 2c and 2d digest silage/hay from increased area. 2e is farmyard manure (FYM).

Scenario	**2a**	**2b**	**2c**	**2d**	**2e**
Grassland Feedstock	**Silage**	**Hay**	**Incr. Silage**	**Incr. Hay**	**N/A**
Manure Feedstock	**Farmyard manure**		**Farmyard manure**		**FYM only**
Grassland feedstock area (ha)	93	36	129	129	N/A
Amount hay (86% DM) (t year^{-1})	0	105	0	379	N/A
Amount silage (25% DM) (t year^{-1})	1,142	0	1,583	0	N/A
Total no. beef cows (supplying FYM)	240	240	240	240	240
Amount FYM (25% DM) (t year^{-1})	1,740	1,740	1,740	1,740	1,740
% feedstock from FYM (DM)	60	83	52	57	100
% feedstock from hay, silage (DM)	40	17	48	43	0
Added water (m^3 year^{-1})	2,000	1,700	2,300	3,000	1,200
Total feedstock (incl. water) (t year^{-1})	4,882	3,545	5,623	5,119	2,940
Digester					
Digester size (m^3)	540	390	620	570	330
Digestate produced (t or m^3 year^{-1})	4,640	3,370	5,340	4,860	2,790
Biogas Production					
Biogas per t hay or silage (m$_N^3$ t^{-1})	109	359	109	359	N/A
Biogas per t FYM (m$_N^3$ t^{-1})	45	45	45	45	45
Biogas per t total feedstock* (m^3 t^{-1})	42	33	45	42	27
Total Biogas (m^3 year^{-1})	202,778	115,995	250,847	214,361	78300
Electricity Production					
Electricity generation (kWh year^{-1})	405,556	231,990	501,694	428,722	156,600
CHP elec. capacity (per hour) (kWe)	46	26	57	49	18
Capital Cost					
Capital cost of set-up (£)	308,169	236,211	372,732	318,276	209,531
Unit cost (Capital/CHP elec.) (£ kWe^{-1})	6,699	9,085	6,539	6,495	11,641
Operational costs (£ year^{-1})					
Feedstock	29,349	6,150	40,683	22,198	0
Maintenance of digester and CHP	9,023	6,159	11,131	9,416	5,028
Depreciation of digester and CHP	18,399	14,022	21,987	19,006	12,297
Transport (feedstock and digestate)	15,278	10,256	17,590	13,961	6,152
Labour	6,351	4,612	7,315	6,659	3,825
Insurance, water, interest	8,931	6,923	10,560	10,203	5,774
Total Costs (£ year^{-1})	87,331	48,028	109,266	81,444	33,077
Revenue					
Total elec income, incl. FiT (£ year^{-1})	52,713	30,335	65,108	55,683	20,622
Total heat income, incl. RHI (£ year^{-1})	1,879	1,883	1,875	1,878	1,881
Total net value of digestate (£ year^{-1})	10,092	5,769	12,498	10,530	3,438
Total Revenue (£ year^{-1})	64,684	37,988	79,481	68,091	25,941
Financial Summary					
Return per t total feedstock* (£ t^{-1})	-4.64	-2.86	-5.30	-2.61	-2.43
Profit or loss (£ year^{-1})	-22,647	-10,135	-29,785	-13,352	-7136
Return on capital (ROC) (%)	**-5.75**	**-2.69**	**-6.39**	**-2.60**	**-1.81**
Internal rate of return (IRR) (%)	**-**	**-9.38**	**-**	**-8.93**	**-6.83**
Grassland feedstock	Silage	Hay	Incr. Silage	Incr. Hay	FYM only

*Total feedstock includes added water

Table 6C. Results of the dairy farm AD scenarios before land subsidy/payments were included. Dairy farmer buys silage/hay from 8 livestock farmers in scenarios 3a-3d. 3a and 3b digest silage/hay from current area of production on livestock farm; 3c and 3d digest silage/hay from increased area. In 3e and 3f the dairy farmer grows their own silage/hay, on the same area each.

Scenario	**3a**	**3b**	**3c**	**3d**	**3e**	**3f**	**3g**
Grassland Feedstock	**Silage**	**Hay**	**Incr. Silage**	**Incr. Hay**	**Own Silage**	**Own Hay**	**N/A**
Manure Feedstock	**Slurry**		**Slurry**		**Slurry**		**Slurry-only**
Grassland feedstock area (ha)	93	36	129	129	16	16	N/A
Amount hay (86% DM) (t year^{-1})	0	105	0	379	0	47	N/A
Amount silage (25% DM) (t year^{-1})	1,142	0	1,583	0	198	0	N/A
Total no. dairy cows (supplying slurry)	135	135	135	135	135	135	135
Amount slurry (25% DM) (t year^{-1})	1,684	1,684	1,684	1,684	1,684	1,684	1,684
% feedstock from slurry (DM)	33	61	27	31	74	78	100
% feedstock from hay, silage (DM)	67	39	73	69	26	22	0
Added water (m^3 year^{-1})	100	0	400	1,100	0	0	0
Total feedstock (incl. water) (t year^{-1})	2,926	1,789	3,667	3,163	1,882	1,731	1,684
Digester							
Digester size (m^3)	330	200	410	350	210	190	190
Digestate produced (t or m^3 year^{-1})	2,780	1,700	3,480	3,000	1,790	1,640	1,630
Biogas Production							
Biogas per t hay or silage (m_N^3 t^{-1})	109	359	109	359	109	359	N/A
Biogas per t slurry (m_N^3 t^{-1})	20	20	20	20	20	20	20
Biogas per t total feedstock* (m^3 t^{-1})	54	40	56	54	29	29	20
Total Biogas (m^3 year^{-1})	158,158	71,375	206,227	169,741	55,262	50,553	33,680
Electricity Production							
Electricity generation (kWh year^{-1})	316,316	142,750	412,454	339,482	110,524	101,106	67,360
CHP elec. capacity (per hour) (kWe)	36	16	47	39	13	12	8
Capital Cost							
Capital cost of set-up (£)	202,290	140,697	237,273	209,011	136,655	128,427	125,369
Unit cost (Capital/CHP elec.) (£ kWe^{-1})	5,619	8,794	5,048	5,359	10,512	10,702	15,671
Operational costs (£ year^{-1})							
Feedstock	41,112	7,560	56,988	27,288	5,089	2,753	0
Maintenance of digester and CHP	6,158	3,557	7,660	6,479	3,225	2,992	2,707
Depreciation of digester and CHP	12,742	8,745	14,889	13,191	8,365	7,890	7,454
Transport (feedstock and digestate)	0	0	0	0	0	0	0
Labour	0	0	0	0	0	0	0
Insurance, water, interest	4,559	2,992	5,715	5,740	2,964	2,759	2,710
Total Costs (£ year^{-1})	64,570	22,855	85,252	52,698	19,644	16,394	12,871
Revenue							
Total elec income, incl. FiT (£ year^{-1})	41,581	19,144	53,976	44,551	14,889	13,698	9,284
Total heat income, incl. RHI (£ year^{-1})	1,888	1,875	1,881	1,880	1,880	1,881	1,880
Total net value of digestate (£ year^{-1})	7,282	2,948	9,598	8,038	1,993	1,646	947
Total Revenue (£ year^{-1})	50,751	23,967	65,455	54,469	18,762	17,224	12,111
Financial Summary							
Return per t total feedstock* (£ t^{-1})	-4.72	0.62	-5.40	0.56	-0.47	0.48	-0.45
Profit or loss (£ year^{-1})	-13,819	1,112	-19,797	1,771	-882	830	-760
Return on capital (ROC) (%)	**-5.23**	**2.39**	**-6.74**	**2.45**	**0.95**	**2.25**	**0.99**
Internal rate of return (IRR) (%)	**-**	**2.26**	**-**	**2.50**	**-0.28**	**1.92**	**-0.53**
Grassland feedstock	Silage	Hay	Incr. Silage	Incr. Hay	Own Silage	Own Hay	Slurry-only

*Total feedstock includes added water

Table 6D. Agri-environment payments were included in AD scenarios where feedstock was grown by the AD owner. A dash means incalculable IRR because returns were all negative. FYM means farmyard manure. 2a to 2d are co-operative AD; 3e and 3f are dairy AD.

Scenario	2a	2b	2c	2d	3e	3f
Grassland feedstock	**Silage**	**Hay**	**Increased Silage**	**Increased Hay**	**Own Silage**	**Own Hay**
Manure feedstock	**FYM**		**FYM**		**Slurry**	
Ave agri-environment payment (£ ha^{-1} $year^{-1}$)	12	232	12	232	12	232
Area of silage or hay land (ha)	93	36	129	129	16	16
Total agri-env. payment per scenario (£ $year^{-1}$)	1,151	8,329	1,595	29,896	198	3,718
Financial Summary						
Return per tonne total feedstock (£ t^{-1})	-4.64	-2.86	-5.30	-2.61	-0.47	0.48
Profit/loss (£ $year^{-1}$)	-21,491	-1,806	-28,190	16,544	-684	4,548
Return on capital (%)	**-5.37**	**0.84**	**-5.96**	**6.80**	**1.10**	**5.14**
IRR (%)	**-**	**-0.83**	**-**	**8.06**	**-0.03**	**6.12**

Table 6E. Agri-environment payments plus basic payment scheme (BPS) were included in AD scenarios where feedstock was grown by the AD owner. FYM means farmyard manure. 2a to 2d are co-operative AD; 3e and 3f are dairy AD.

Scenario	2a	2b	2c	2d	3e	3f
Grassland feedstock	**Silage**	**Hay**	**Increased Silage**	**Increased Hay**	**Own Silage**	**Own Hay**
Manure feedstock	**FYM**		**FYM**		**Slurry**	
BPS + agri-environment (£/ha/year)	183	403	183	403	183	403
Total BPS + agri-env. of all grazing farmers (£/year)	16,963	14,436	23,515	51,816	2924.72	6448
Financial Summary						
Return per tonne total feedstock (£/t)	-4.64	-2.86	-5.30	-2.61	-0.47	0.48
Profit/loss (at 50% finance repaid) (£/year)	-5,679	4,301	-6,270	38,464	2,043	7,278
Return on capital (%)	**-0.24**	**3.42**	**-0.08**	**13.69**	**3.10**	**7.27**
IRR at 20 years (%)	**-2.86**	**3.40**	**-2.68**	**16.09**	**3.19**	**8.87**

Table 6F. New agri-environment payments required to give an IRR of 10% in AD scenarios where feedstock is grown by the AD owner. FYM means farmyard manure. 2a to 2d are co-operative AD; 3e and 3f are dairy AD.

Scenario	2a	2b	2c	2d	3e	3f
Grassland feedstock	**Silage**	**Hay**	**Increased Silage**	**Increased Hay**	**Own Silage**	**Own Hay**
Manure feedstock	**FYM**		**FYM**		**Slurry**	
Total new agri-env. payments, all grazing farmers (£/year)	43400	26100	55200	35000	9900	7600
New agri-env. payments (£/ha/year)	**468**	**728**	**429**	**272**	**619**	**475**
Current average agri-env. payment (£/ha/year)	12	232	12	232	12	232
Increase above current payment (£/ha/year)	455	496	417	40	607	243
Fold increase	38	3	35	1	52	2
Return on capital (%)	8.34	8.36	8.42	8.40	8.21	8.22

Table 6G. Electricity subsidy (Feed-in Tariff) required to give an IRR of 10% in each AD scenario. No land subsidy/payments were included. FYM means farmyard manure. 2a to 2d are co-operative AD; 3a to 3f are dairy AD.

Scenario	**2a**	**2b**	**2c**	**2d**	**3a**	**3b**	**3c**	**3d**	**3e**	**3f**
Grassland feedstock	**Silage**	**Hay**	**Incr. Silage**	**Incr. Hay**	**Silage**	**Hay**	**Incr. Silage**	**Incr. Hay**	**Own Silage**	**Own Hay**
Manure feedstock	**FYM**		**FYM**		**Slurry**		**Slurry**		**Slurry**	
New FIT required (p/kWhe)	**18.90**	**19.50**	**19.20**	**16.35**	**16.70**	**13.85**	**16.73**	**11.65**	**17.20**	**15.80**
Current FIT (p/kWhe)	8.21	8.21	8.21	8.21	8.21	8.21	8.21	8.21	8.21	8.21
Increase above current FIT (p/kWhe)	10.69	11.29	10.99	8.14	8.49	5.64	8.52	3.44	8.99	7.59
% increase on current	**230**	**238**	**234**	**199**	**203**	**169**	**204**	**142**	**210**	**192**
Return on capital (%)	8.32%	8.40%	8.40%	8.37%	8.04%	8.11%	8.07%	8.03%	8.226	8.2218

Table 6H. Profit per ha for the sheep farmer, for different land managements after inclusion of land subsidy and payments; and replacement of agri-environment payments with social values on hay land (estimated from Boatman *et al*. (2010) for GHG saved and higher biodiversity). Energy subsidies were included in all AD scenarios. Incr is increased. Average agri-environment payment for silage and hay land is £12 and £232 ha^{-1} respectively; BPS is £170 ha^{-1} for all fields; social values were valued at £363 ha^{-1}. Numbers are rounded.

Land management options	Mean profit per ha (£) Silage	Hay
1) Sheep farming		
No land subsidy/payment	-395	-203
Agri-environment	-383	29
Agri-environment + BPS	-212	200
(Silage=agri-env; hay=social) + BPS	-212	330
2) AD co-operative: current production of silage/hay (scenarios 2a, 2b)		
No land subsidy/payment	-472	-500
Agri-environment	-459	-268
Agri-environment + BPS	-289	-97
(Silage=agri-env; hay=social) + BPS	-289	33
2) AD co-operative: increased silage/hay (scenarios 2c, 2d)		
No land subsidy/payment	-459	-321
Agri-environment	-447	-89
Agri-environment + BPS	-276	82
(Silage=agri-env; hay=social) + BPS	-276	212
3) Sell silage/hay to dairy AD		
No land subsidy/payment	-101	-178
Agri-environment	-89	54
Agri-environment + BPS	82	224
(Silage=agri-env; hay=social) + BPS	82	355

ABBREVIATIONS

Acronyms and abbreviations

AD	Anaerobic digestion
AES	Agri-environment scheme
Agri-env	Agri-environment scheme
BPS	Basic Payment Scheme
BREXIT	The United Kingdom leaving the European Union
CHP	Combined heat and power plant
Co-op	Co-operative
DEFRA	Department for Environment, Food and Rural Affairs
DM	Dry matter
ELS	Entry Level Stewardship
EU	European Union
FBS	Farm Business Survey
FiT	Feed-in Tariff
FYM	Farmyard manure
GHG	Greenhouse gases
HLS	Higher Level Stewardship
HM Treasury	Her Majesty's Treasury
Incr	Increased
LFA	Less Favoured Area
LUC	Land use change
IPCC	Intergovernmental Panel on Climate Change
IRR	Internal rate of return
RHI	Renewable heat incentive
ROC	Return on capital
SDA	Severely Disadvantaged Area
Subs	Subsidy
UK	United Kingdom
VS	Volatile solids

Units of measurement and chemical nomenclature

CH_4	Methane
CO_2	Carbon dioxide
CO2e	Carbon dioxide equivalent
GJ	Gigajoule
Ha	Hectare
Kg	Kilogram
K	Potassium
kWe	Kilowatts of electricity produced per hour
kWh	Kilowatt hour (unit of energy)
kWhe	Kilowatt hour of electricity (unit of electricity)
N	Nitrogen
N_2O	Nitrous oxide
P	Phosphorus
T	Tonne

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