Harper Adams University

Proceedings of the 6th Symposium on Agri-Tech Economics for Sustainable Futures

18 – 19th September 2023, Harper Adams University, Newport, United Kingdom.

Global Institute for Agri-Tech Economics,

Food, Land and Agribusiness Management Department,

Harper Adams University

Valorisation of Poultry Litter: A socio-environmental cost-benefit comparison of traditional land application and anaerobic digestion

Deborah Hall[A], Karl Behrendt[A], Stephen Woodgate[B], Simon Jeffery[A] and Marie Kirby[A]

[A] *Harper Adams University, Newport, Shropshire, United Kingdom*

[B] *Beacon Research Limited, Clipston, Northamptonshire, United Kingdom*

Abstract

Traditional land application of poultry litter (PL) as a fertiliser has led to numerous environmental issues, including eutrophication and soil acidification. An alternative valorisation option is, therefore, sought. Anaerobic digestion (AD) of PL is an emerging field that shows promise and benefits from both energy and fertiliser production. This study aimed to compare the economic, environmental, and social costs and benefits of land application and AD of PL using a modified economic life cycle assessment (LCA) approach. Using economic data from literature and industry reports, a model for each method was created to calculate key economic markers, including net present value (NPV). LCA was incorporated into the model with the environmental emissions of each method being calculated for Global Warming Potential (GWP), Acidification Potential (AP), Freshwater Eutrophication (FE), Photochemical Ozone Potential (POP), and Particulate Matter Formation Potential (PMFP) impact categories. The social value of these impact categories was applied to the emissions data to calculate a socio-environmental cost (or benefit) for each method. Using Monte Carlo simulation, the model shows that AD performs worse when focusing purely on the economic category with an NPV of £707.17 per tonne of PL, compared to £1838.36 per tonne for land application of fresh PL. However, when factoring in the environmental costs, both methods generated a negative NPV. However, AD is shown to be less environmentally damaging than direct land application with an NPV of -£1354.17 per tonne of PL compared to -£5788.34 for direct land application. Furthermore, the model showed that it is possible to optimise the AD process to generate a positive economic and socio environmental NPV, through operational control of biogas and energy production. Further research is needed in this area to determine the optimal parameters to operate a PL mono-digestion AD process for economic and socio-environmental gain.

Keywords

Poultry litter, Valorisation, Life Cycle Assessment, Techno-economic

Presenter Profile

Deborah Hall is a PhD researcher at Harper Adams University, Shropshire, focusing on the valorisation of poultry litter through the use of anaerobic digestion.

Introduction

Greenhouse gas (GHG) emissions from poultry litter (PL) are of significant concern. The high nitrogen concentration in PL, and its volatilisation through conversion to ammonia gas (NH_3), has notable detrimental effects on the environment due to its involvement in the production of acid rain (Choi and Moore Jr, 2008). When ammonia enters the atmosphere and condenses, the resultant rainwater has a higher pH, giving it a greater ability to dissolve sulphur dioxide (Nahm, 2005). The sulphur dioxide and ammonia form into ammonium sulphate, and, when entering the soil, oxidise and release sulphuric and nitric acids (Pote and Meisinger, 2014). In addition, the volatilisation of ammonia significantly increases the atmospheric fallout of nitrogen, which adds to the eutrophication of waterbodies (Nahm, 2005). Cabrera *et al.* (1993) explain that up to 50% of the nitrogen within PL is emitted as either ammonia or nitrous oxide gas, particularly when PL is land spread. This loss of nitrogen not only reduces the value of the fertiliser, but also has notable impacts on the atmosphere. Forster *et al.* (2007) states that the global warming potential (GWP) of nitrous oxide is around 300 times that of carbon dioxide (298 kg of CO2-equivalents per kg). Methane is reported to have a global warming potential 24 times higher than carbon dioxide (Forster *et al.,* 2007). Ahn *et al.* (2011) estimates that 62 megatons of CO_2 equivalents of methane and nitrous oxide have been emitted globally from animal manures since the start of the industrial age.

Good PL management is, therefore, a necessity, from environmental, economic, and nutrient recycling viewpoints. The traditional use of PL has been land application to recycle nutrients, predominantly N, P and K (Lorimor and Xin, 1999). Due to the high costs associated with transportation, the majority of fresh PL is spread within a 5km radius of the poultry production facility. Furthermore, until recently, application rates were calculated to meet the N requirement of the crop, which often results in an elevated and unnecessary P application. This means that much of the land surrounding intensive poultry facilities have reached their agronomic and regulatory threshold for soil P. This threshold means that the farmer gains notable agronomic benefits with higher crop yields, whilst excess P (and N) is leached into nearby watercourses and impacts water quality (Harmel *et al.*, 2009). Therefore, it is necessary to adopt an improved assessment of PL usage and processing within agro ecosystems to determine optimal applications for soil health, agricultural yield, and water quality. This assessment also needs to consider the economic impact of utilising PL as a fertiliser or diverting its use into biomass for energy production.

To reduce the economic and environmental impact of waste disposal options for PL, an alternative option to land spreading, which has been considered more frequently over recent years, is the valorisation of this waste stream. There are numerous strategies for poultry manure valorisation currently described in the literature. These include composting of PL (Vandecasteele *et al.* 2014), thermal energy recovery using pyrolysis (Kim *et al.* 2009), combustion (Lynch *et* al. 2013) and gasification (Palma and Martin, 2013) or biological energy recovery through the use of technologies such as anaerobic digestion (AD) (Rao *et* al, 2013). This research will complete a social cost benefit and life-cycle comparison between traditional land spreading of PL with AD coupled with energy recovery technology.

Aim and Objectives

The primary aims and objectives of this research are shown in Table 1; in brief the research compares the use of AD with combined heat and power (CHP) energy recovery and organic fertiliser production with traditional land spreading of PL; this includes an assessment of economic viability and social and environmental impact evaluation.

Primarily, the theoretical system is modelled using baseline data from previously published literature along with calculated data from Ecoinvent and includes a 'hot spot' analysis assessment to determine the significant parameters within the foreground system that have the largest social and environmental impacts. In the sensitivity analysis, some of these parameters are varied to determine their effect on the overall results. As previously explained, the system has been expanded to include background processes to allow for the impact of energy and materials recovery from the waste to be considered; this includes the production of inorganic fertiliser through the production and use of digestate, along with the displacement of fossil fuel produced electricity and heat energy using CHP energy recovery.

Ideally, site-specific data would be used for the foreground processes; however, the use of AD for PL valorisation is a relatively new and complex topic with limited practical application; therefore, theoretical and average data from the literature has been used. As Heijungs and Guinée (2007) caution that LCA studies sometimes produce conflicting results, all assumptions made in this research are described as succinctly as possible to enable a reproduction of the analysis. The study focuses on five LCA impact categories that have been chosen due to their environmental significance. These are global warming potential (GWP) as an indicator of climate change and greenhouse effect; acidification potential (AP) as an indicator of the production and impact of acid rain; freshwater eutrophication (FE) as an indicator of the eutrophication impact of nitrate and phosphate leaching into freshwater; photochemical ozone potential (POP) from the emission of NOx and the creation of photo-smog, and particulate matter formation potential (PMFP) from the emission of small particulate matter ($PM_{2.5}$) and its effect on human health. These impact categories are internationally accepted through ISO 14044 recommendations (ISO, 2006). A 100-year time horizon assessment has been used as per the IPCC recommendations.

Table 1. Project aims

Aim	Purpose
Assess Economic Viability	Determine the economic feasibility of anaerobic digestion as a valorisation method compared to traditional land spreading
Environmental Impact Evaluation	Evaluate the potential environmental benefits and drawbacks of each option in terms of emissions and eutrophication
Social Cost-Benefit Analysis	Perform a comprehensive social cost-benefit analysis (SCBA) to capture both economic and social implications of each method

Table 2. Project objectives

Objective	Purpose
Cost Analysis	a. Calculate the initial investment costs for AD, including construction, equipment, and setup expenses. b. Estimate operational costs for AD, considering OPEX and maintenance costs c. Compute the net present value (NPV), benefit-cost ratio (BCR), and payback period for AD and land spreading.
Environmental Impact Assessment	a. Quantify emissions from AD and traditional land spreading. b. Estimate the social cost of emissions using appropriate valuation methods. c. Assess the cost of eutrophication caused by nitrate and phosphate leaching and compare it across different methods.
Sensitivity Analysis	a. Conduct sensitivity analyses to assess the impact of changing key parameters on the economic indicators and environmental outcomes.

Methods

Whilst there are numerous strategies that have been developed and presented in the literature that attempt to integrate LCA into process design and optimisation frameworks, the majority of these are within the chemical process design field. Indeed, Grossmann and Guillén-Gosálbez (2010) stated that the major limitation of LCA application to process systems is the lack of systematic methodology for melding the LCA impacts with good economic performance. This was largely addressed by the multi-objective optimisation approach presented by Gerber, Gassner and Marechal (2011), which focused on environmental, economic, and thermodynamic impacts on life cycle performance. However, these authors were focused on just one product, that of electricity from biowaste. In this study, two products are considered: energy and organic fertiliser production. As such, a modified LCA and techno-economic approach have been used.

The economic evaluation follows a cost-benefit analysis approach. This model focuses on the estimate of capital and operational costs and the associated calculation of the net present value, benefit cost ratio and payback period. The estimate of capital costs is based on equipment and installation cost estimates provided by IRENA (2012). A financial spreadsheet was designed and utilised to incorporate the costs and benefits of each valorisation method. The capital cost was estimated based on the required digestor size to treat the total mass of feedstock (20,000 tonnes of PL). A dynamic interest rate function is applied to appropriately discount future cash flows considering changing economic conditions. A constant salvage value is considered for the AD to account for asset value at the end of its life. Net present value (NPV) and benefit-cost ratio are calculated using time series data and dynamic interest rate. The formula for calculating NPV is as follows:

$$NPV = \frac{R_t}{(1+i)^t}$$

where NPV = Net present value; R = net cash flow at time t; i = discount rate and t = time of the cash flow.

Benefit cost ratio (BCR) is calculated as follows:

$$BCR = \frac{\sum_{t=0}^{n} \frac{CFt\ [Benefits]}{(1+i)^t}}{\sum_{t=0}^{n} \frac{CFt\ [Costs]}{(1+i)^t}}$$

where CF = cash flow; i = discount rate; n = number of periods; and t = time of the cash flow. OPEX costs incorporate the annual running costs of the plant and are split into fixed and variable costs. Fixed costs include labour, scheduled maintenance, routine component replacement and insurance, whilst variable costs include non-biomass fuel costs, unplanned maintenance, equipment replacement and incremental servicing costs. These OPEX costs are estimated using estimates provided by IRENA (2012). All costs are presented on a 2023 basis and the main financial assumptions are tabulated in Section 3.

For the environmental impact, a modified LCA approach has been utilised. Clift (2013) explains that life cycle assessment (LCA) is one of the most significant and widely utilised tools for assessing and comparing the environmental impact of alternative technologies. The tool enables the quantification of energy and materials within a complete supply chain, or life cycle, of services or goods, whilst also identifying wastes and/or emissions from these life cycles (Azapagic *et al*., 2003). Furthermore, Azapagic *et al.,* (2003) explain that LCA enables the identification of system 'hot spots', which are the areas that exert the most significant impacts on the environment, thereby enabling the modification of systems to more sustainable approaches. However, from this it is necessary to determine a rational approach to allocate the environmental costs or impacts of each of the processes. This allocation issue has been debated by Clift *et al*. (2000) and Heijungs and Guinée (2007) but ably clarified by Eriksson *et al.* (2007) who support the broadening of the system boundaries to account for the environmental benefits of recovered resources whilst including the avoided burdens associated with conventional systems. An avoided burden is effectively a saved impact that arises from the reuse, recycling or energy generation from waste and is generally subtracted from the categorised impacts to generate a reduced overall environmental impact. This is the approach that is applied throughout this research. As such, 'foreground' and 'background' processes are initially identified as per the Integrated Waste Management approach defined by Clift *et al.* (2000). The 'foreground' processes are those that are directly influenced by study-based decisions, whilst the 'background' processes are those that interact with the foreground processes through the supply or receipt of energy or materials.

There is always an element of uncertainty with LCA, particularly when theoretical, rather than site specific, data are used. For this study, average data is used that has been obtained and collated from previously published, peer-reviewed literature. In addition, calculated emission data published by Ecoinvent has been used where available.

To improve the robustness of the process, a sensitivity analysis is performed by running Monte Carlo simulations of the model's 14 variables:

These variables are:

- Biogas yield (m^3)
- Biogas potential (l/kg)
- Biogas conversion efficiency (%)
- Methane content (%)
- Total energy production (kWh)
- Electrical conversion efficiency (%)
- Heat conversion efficiency (%)
- Total electricity production (kWh)
- Total heat production (kWh)
- Parasitic load percentage (%)
- Mineral fertiliser cost (£)
- Fixed O&M costs (£)
- Variable O&M costs (£)
- Capital cost (£)

These variables were given range values that were determined from previously published literature. The results were studied under two different levels of analysis; Analysis 1 considers purely economic parameters (CAPEX, OPEX, yield, energy production, etc) whilst Analysis 2 considers all these costs plus the costs of environmental emissions (GWP, AP, FE, etc). All financial parameters were calculated over a 20-year life span, considered to be feasible for an AD plant. For the land application method, mineral fertiliser cost is the only considered variable.

For the Monte Carlo simulation, all of the variables were considered at once; therefore, multiple regression analysis was performed in order to determine the most influential variables on NPV. Limitations include reliance on input data quality, potential uncertainties due to changing economic conditions, and evolving technology performance.

Assumptions

Biogenic CO2

Emissions of biogenic CO2 are defined by the US EPA (2011) as "emissions from a stationary source directly resulting from the combustion or decomposition of biologically based materials other than fossil fuels". In line with the approach used by Christensen *et al.* (2009) and Manfredi *et al.* (2011), this research considered biogenic CO2 emissions as neutral with regards to global warming, as they are a part of the natural carbon cycle. Therefore, for the purpose of this study, the biogenic carbon within the organic matter (PL or digestate) is sequestered into the soil and removed from the atmosphere, therefore its characterisation factor from organic sources is considered to be zero throughout the study.

Transportation

As this research is following a comparative LCA approach, processes that are identical within each alternative are omitted as they are not considered to impact on the overall results (Finnveden, 2008). This includes the transportation of PL between stages, from the PL house to the storage tank or stockpile and from here to the field. The cost of spreading of the PL and digestate is also valued equally, despite potentially different distances being covered from stockpiles and the treatment plant. As explained by Patterson *et al*. (2011), the environmental impact of transportation distances on LCA results is arbitrary and therefore is unlikely to impact on the overall result.

System expansion for electricity, heat and fertiliser production

To ascertain the impact of the background process, it is necessary to apply a system expansion approach. This involves the identification of the type and quantity of the product, i.e., energy and digestate / organic fertiliser, that is replaced by the technology (Fruergaard and Astrup, 2011). Consequential LCA studies often use marginal technology data and are focused on the significances of policy or broader changes; conversely, attributional LCA studies are used to describe a proposed or specific current process and often use average technology data to calculate the avoided burdens linked with the system expansion (Fruergaard *et al.*, 2009). As this research focuses on a specific, though assumed, process, it is a form of attributional analysis, thereby allowing the use of average data for organic fertiliser and energy production to be used.

The avoided burdens of electricity and heat export to the National Grid have been collated from data presented by Evangelisti *et al*., (2014) and utilises an average UK mix of technologies and fuels. The results of the avoided burdens per kWh of energy produced (heat and electricity) are shown in Table 3 and reported for four of the five environmental impact categories considered (no data was reported for particulate matter formation).

Table 3. Avoided burdens per kWh of energy for substitution of heat and electricity produced (Evangelisti et al, 2014).

Impact category	National Grid mix UK	Thermal energy natural gas
Global warming potential (kg CO_2 eq)	0.167	0.004
Acidification potential (kg SO_2 eq)	0.00058	0.00001
Photochemical oxidant potential (kg NOx eq)	0.000032	0.000001
Nutrient enrichment (eutrophication) potential (kg NO_3 eq)	0.00051	0.000008

In order to calculate the avoided burdens through the substitution of PL or digestate as an organic fertiliser on inorganic fertiliser production and use, the nutrient availability to the crops has been used. Inorganic fertiliser substitution has been discussed previously in the literature (Bernstad and la Cour Jansen, 2011; Moller *et al.*, 2009). Moller *et al*. (2009) supported the use of average burden calculations for the production of N, P and K fertilisers. As such, for the purpose of this study, average data from a fertiliser life cycle assessment by Skowroñska and Filipek (2014) has been used, as shown in Table 4. Table 5 shows the economic LCA values per kg of each gaseous emission for each impact category.

Table 4. Avoided burdens associated with digestate or PL use compared to inorganic fertiliser (from Skowroñska and Filipek, 2014).

Parameter	Unit	Avoided burden
GWP	kg CO_2 eq/kg fertiliser produced	1.79
AP	kg SO_2 eq/kg fertiliser produced	6.07
FE	kg PO_4 eq/kg fertiliser produced	0.53

Table 5. Economic LCA values for each environmental impact category.

LCA Values	GBP (£)
1kg CO2 eq	0.11
1kg SO2 eq	7.99
1kg PM2.5	31.96
1kg PO4 eq	4.31
1kg NMVOC eq	4.58

Use of digestate as a replacement fertiliser

Whilst digestate is a by-product from AD, its use as a substitute for inorganic fertilisers is becoming more mainstream. For the purpose of this study, it is assumed that all produced digestate is spread on the land as a fertiliser with the quality of the digestate mirroring that of the feedstock. This follows the method proposed by Moller *et al.* (2009) whereby it is assumed that the AD process results in no net loss of nitrate, phosphate, or potassium. Application of the digestate and PL as organic fertiliser has been assumed on a rate of 120:60:40, N:P:K, respectively, which is considered suitable for standard maize cultivation and complies with the use of agricultural fertilisers within the UK (UK Government, 2008).

Case study scenario

Tables 6 – 9 provide the data and source of assumptions and values used in the model along with the theoretical farm situation. Table 6 describes the theoretical farm situation, detailing volume of PL, NPK application rate, electricity and heat usage in the poultry house along with key parameters that are included or omitted from the study.

Table 6. Theoretical Farm Situation

Assumption	Value	Notes
Poultry litter quantity	20,000 tonnes	Broiler poultry farm housing 14,500 birds
Land application radius	5km	Surrounding land of 6250 hectares
NPK application rate	120:60:40	Standard maize cultivation
Electricity consumption	20,341 kWh/yr	For poultry unit operations
Heat consumption	140,000 kWh/yr	For poultry unit operations
Scale of valorisation options	Small-scale on-farm technology	Technologies built on farm premises

Transportation and emissions, excluding spreading costs	Not included	Transport between farm and field considered to be similar for each method so not included
GHG Emissions from poultry unit	Not included	Assumed to be the same for both methods
Cost and emission differences	Calculated	Between fertiliser / digestate / and PL applications
Economic analysis		
Spreading costs	£10.15 per tonne	Literature values range from £9.76 /t (Vervoort and Keeler, 1999) to £10.54 /t (Huijsmans *et al.,* 2004) adjusted for inflation and exchange rate.

Table 7 outlines a number of assumptions for each of the valorisation methods that were considered likely to impact emissions within the model.

Table 7. Assumptions for Different Valorisation Methods

Assumption	**Land spreading**	**Anaerobic digestion**
Energy source for poultry house (Electricity	National Grid	CHP
Electricity cost	£0.34 per kWh	
Energy source for poultry house (Heat)	LPG	CHP
Heat cost	£0.10 per kWh	
Poultry litter storage	Field windrow	Bunded, covered tank
Maximum storage period	6 months	N/A

Table 8 details the digestor size and assumed hydraulic retention time used for the case study.

Table 8. Anaerobic Digestor Assumptions (fixed)

Assumption / Calculation	**Value**	**Reference / Notes**
Digester size calculation	Size (m^3) = Flow rate (m^3) x Hydraulic retention time (days)	
Flow rate	125 m3/day	Assumption: Given maximum daily feedstock flow rate
Hydraulic retention time	40 days	Mahdy *et al.,* 2020
Digester size	5000m3	

Table 9 provides the variable ranges that are included in the Monte Carlo simulation for the model sensitivity analysis.

Table 9. Anaerobic Digestor Assumptions (variables)

Assumption / Calculation	**Value**	**Reference / Notes**
Capital cost	£4,188,990 - £9,934,275	IRENA, 2012
OPEX	Range 2.1-7% of installed cost	IRENA, 2012
Biogas potential	Range 88 -226 l/kg fresh weight.	Jurgutis *et al.,* 2020
Biogas production efficiency	45 - 60%	
Methane content of biogas	48 - 62%	Assumption
Electrical conversion efficiency	30 - 50%	
Heat conversion efficiency	30 - 50%	
Parasitic load (electricity)	Range 4% - 31.4% of the electrical energy production	Gikas, 2014; Murphy and Power, 2006; Murphy and Thamsiriroj, 2013; Walker *et al*., 2017
Parasitic load (heat)	22.65% (Range 15% - 30.3%) of the heat energy production	Aui, Li and Wright, 2019; Walker *et al.,* 2017

Results

Analysis 1. Economic Analysis

The average net present value (NPV), benefit cost ratio (BCR), payback period and modified internal rate of return for the two options were calculated from a purely economic viewpoint and are recorded in Table 10. Figure 1 shows the NPV range for the two methods calculated through Monte Carlo analysis. Comparison of key financial parameters between land application and AD are shown in Figure 2.

Table 10. Net Present Value (NPV) for 1 tonne of poultry litter (economic comparison)

Valorisation Method	NPV (£/t)	BCR (return per £ invested)	Payback period	MIRR (%)
Land spreading	1838.36	12.55	n/a	n/a
Anaerobic Digestion	707.17	1.63	5.34	6.86

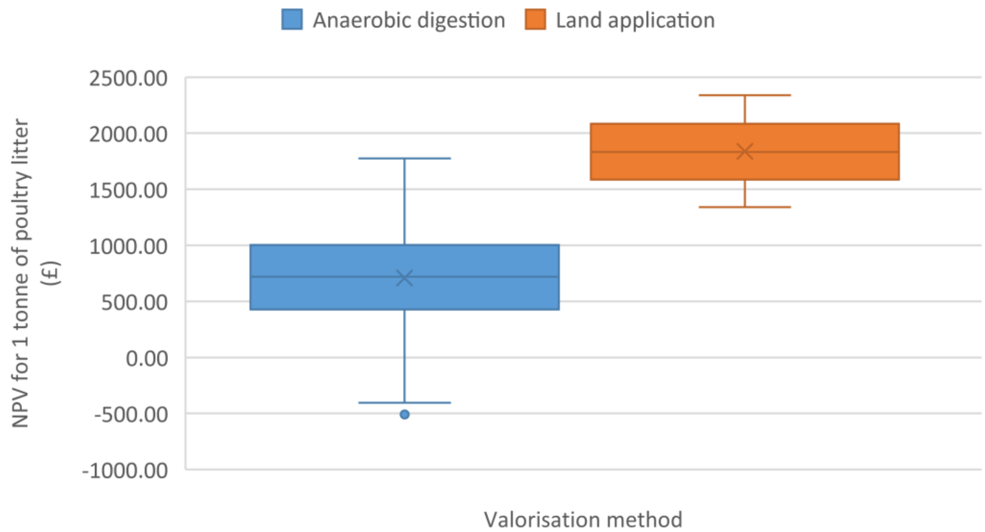

Figure 1. Sensitivity analysis results comparing NPV values for anaerobic digestion and land application.

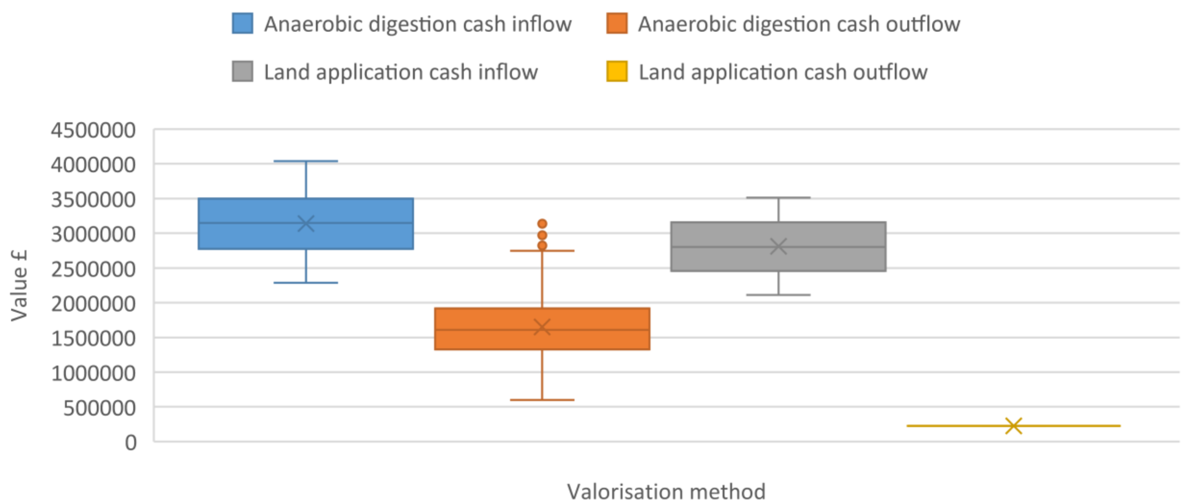

Figure 2. Comparison of the cash inflow and cash outflow of anaerobic digestion and land application.

Analysis 2. Economic and environmental analysis

Average NPV and BCR were calculated for both methods using the Monte Carlo simulation. Due to the negative results, payback period and MIRR are not calculated. Table 11 presents the results.

Table 11. Net Present Value (NPV) and BCR for 1 tonne of poultry litter (economic and environmental comparison)

Valorisation Method	NPV (£)	BCR (return per £ invested)
Land spreading	- £5788.34	0.33
Anaerobic Digestion	-£1354.17	0.74

The avoided burdens of fertiliser (4687 tonnes) and heat used (140,000kWh) and electricity produced (average 3648416.8kWh) were subtracted from the environmental emissions data to calculate and compare overall environmental costs for each method (Figure 3). In Figure 3, negative figures denote a net environmental gain when comparing each method against the use of equivalent mineral fertilisers and through National Grid electricity and heat substitution from the AD.

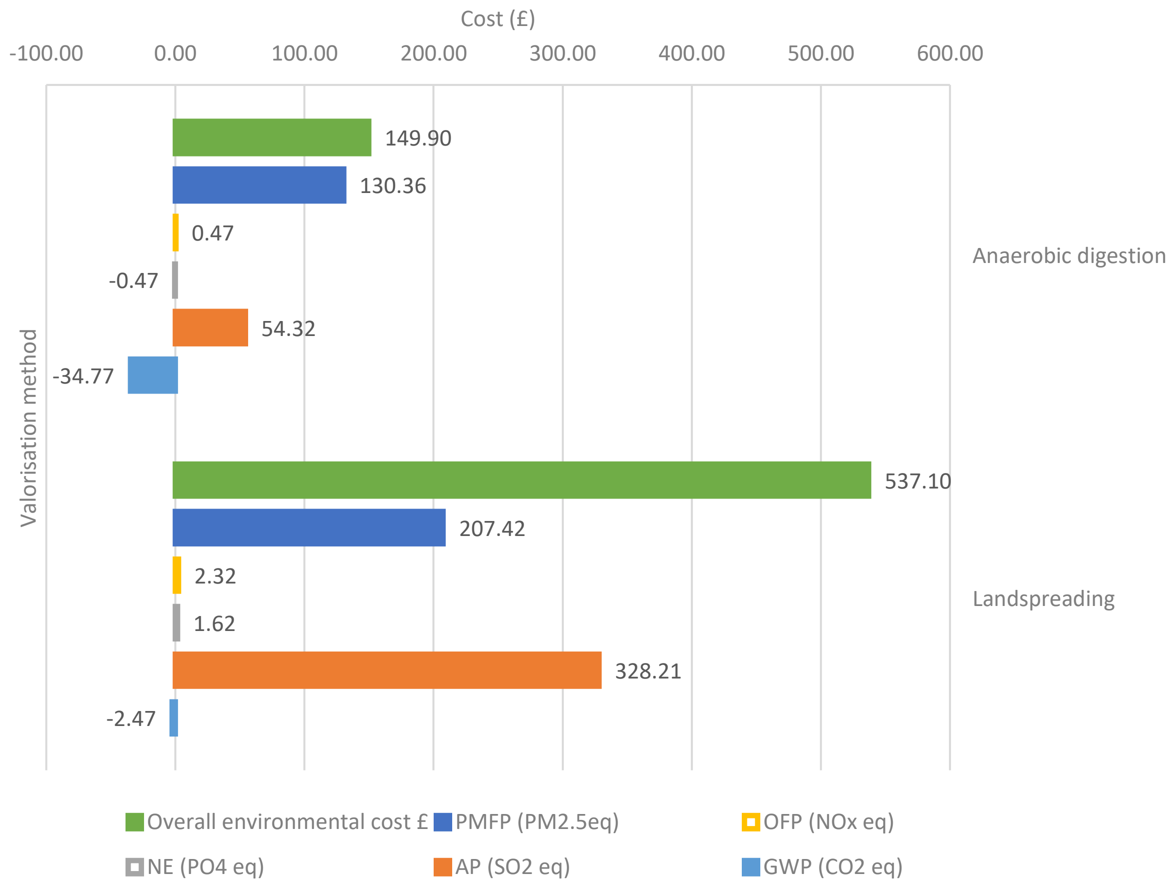

Figure 3. Environmental cost comparison between land application and anaerobic digestion per tonne of poultry litter

The mean, standard deviation, min, and max data were collated from the Monte Carlo simulations of all ten variables and analysed alongside the NPV, BCR and payback period. This sensitivity analysis utilised all collated data, therefore, was focusing on the overall costs of each method, inclusive of environmental costs. Figure 4 shows the NPV comparison from this sensitivity analysis.

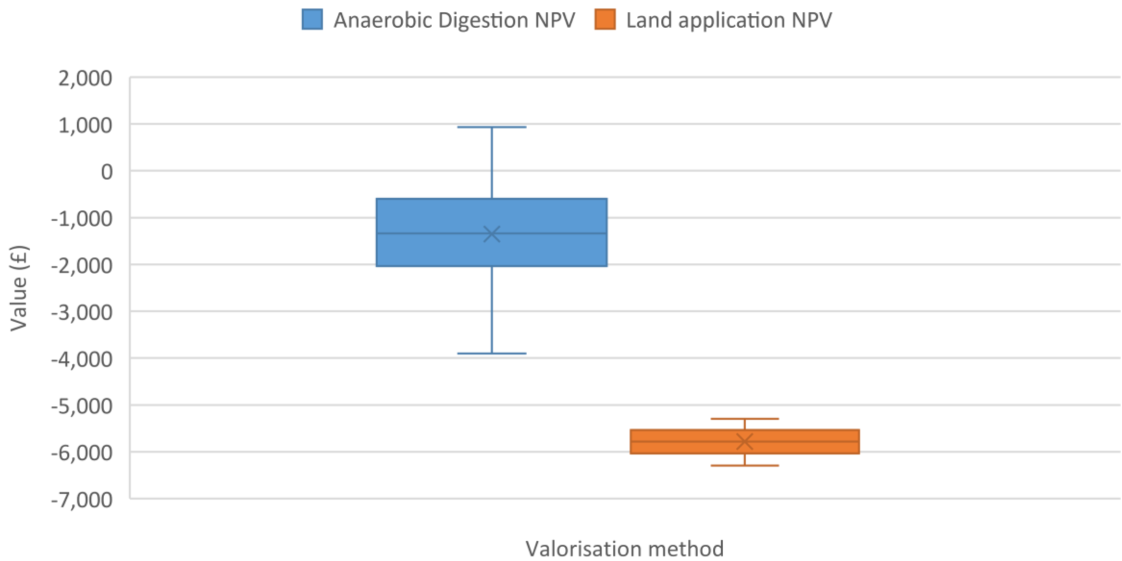

Figure 4. NPV comparison between AD and land application from Monte Carlo sensitivity analysis

This comparison displayed expected results, with the AD option showing a higher, therefore better, NPV than that of traditional land application. Whilst the comparison between the two options is notable, the range within the AD NPV is worthy of further investigation. A negative NPV value was expected from this analysis, as the emissions from the construction of the AD plant and associated operational emissions will have negative impacts on the environment; however, the sensitivity analysis showed that in certain scenarios, a positive NPV was gained. By ranking the data sets using NPV value, and selecting all of those with a positive NPV, further analysis was undertaken.

Multiple regression analysis was carried out to determine the relationship between NPV and the independent variables. The correlation matrix is shown in Table 12.

Table 12. Multiple regression analysis – Correlation matrix (pearson)

	NPV	Biogas Potential	Biogas efficiency	Methane content	Electrical conversion	Heat conversion	Parasitic load value	Mineral fertiliser cost	Biogas yield	Total energy production	Fixed O&M Cost	Variable O&M cost	Capital cost	Total electricity	Total heat production
NPV	1	-0.1701	-0.1152	-0.1883	0.02732	-0.1294	-0.0288	0.27831	-0.2376	-0.3456	-0.3034	-0.2981	-0.1966	-0.2049	-0.3177
Biogas Potential	-0.1701	1	-0.2671	-0.278	0.10736	-0.0349	-0.2422	0.23951	0.81004	0.65647	-0.1387	-0.1015	0.04585	0.5145	0.42507
Biogas efficiency	-0.1152	-0.2671	1	-0.0356	-0.0073	0.12687	-0.3824	0.06564	0.34463	0.33524	0.09169	-0.0614	-0.1143	0.20128	0.31594
Methane content	-0.1883	-0.278	-0.0356	1	0.04353	-0.0381	0.00404	0.20108	-0.2944	0.2638	0.21191	-0.097	0.09162	0.17673	0.14199
Electrical conversion efficiency	0.02732	0.10736	-0.0073	0.04353	1	0.07198	-0.2231	0.22982	0.10275	0.11987	0.14258	0.1088	0.09557	0.767	0.15055
Heat conversion efficiency	-0.1294	-0.0349	0.12687	-0.0381	0.07198	1	-0.129	-0.1127	0.04269	0.01397	-0.024	0.17942	0.05872	0.07491	0.73087
Parasitic load value	-0.0288	-0.2422	-0.3824	0.00404	-0.2231	-0.129	1	-0.0585	-0.4555	-0.4655	-0.0979	-0.0766	0.03548	-0.4588	-0.4129
Mineral fertiliser cost	0.27831	0.23951	0.06564	0.20108	0.22982	-0.1127	-0.0585	1	0.2795	0.394	0.36008	0.05256	-0.1597	0.40494	0.19779
Biogas yield	-0.2376	0.81004	0.34463	-0.2944	0.10275	0.04269	-0.4555	0.2795	1	0.84165	-0.0833	-0.1273	-0.0218	0.62414	0.60643
Total energy production	-0.3456	0.65647	0.33524	0.2638	0.11987	0.01397	-0.4655	0.394	0.84165	1	0.0407	-0.1876	0.01934	0.72345	0.68678
Fixed O&M Cost	-0.3034	-0.1387	0.09169	0.21191	0.14258	-0.024	-0.0979	0.36008	-0.0833	0.0407	1	0.04273	-0.0977	0.11633	0.01501
Variable O&M cost	-0.2981	-0.1015	-0.0614	-0.097	0.1088	0.17942	-0.0766	0.05256	-0.1273	-0.1876	0.04273	1	0.08229	-0.0297	0.00791
Capital cost	-0.1966	0.04585	-0.1143	0.09162	0.09557	0.05872	0.03548	-0.1597	-0.0218	0.01934	-0.0977	0.08229	1	0.08189	0.06677
Total electricity production	-0.2049	0.5145	0.20128	0.17673	0.767	0.07491	-0.4588	0.40494	0.62414	0.72345	0.11633	-0.0297	0.08189	1	0.55749
Total heat production	-0.3177	0.42507	0.31594	0.14199	0.15055	0.73087	-0.4129	0.19779	0.60643	0.68678	0.01501	0.00791	0.06677	0.55749	1

By plotting the absolute correlation coefficients on a tornado chart, the variables with the strongest influence can be determined (Figure 5).

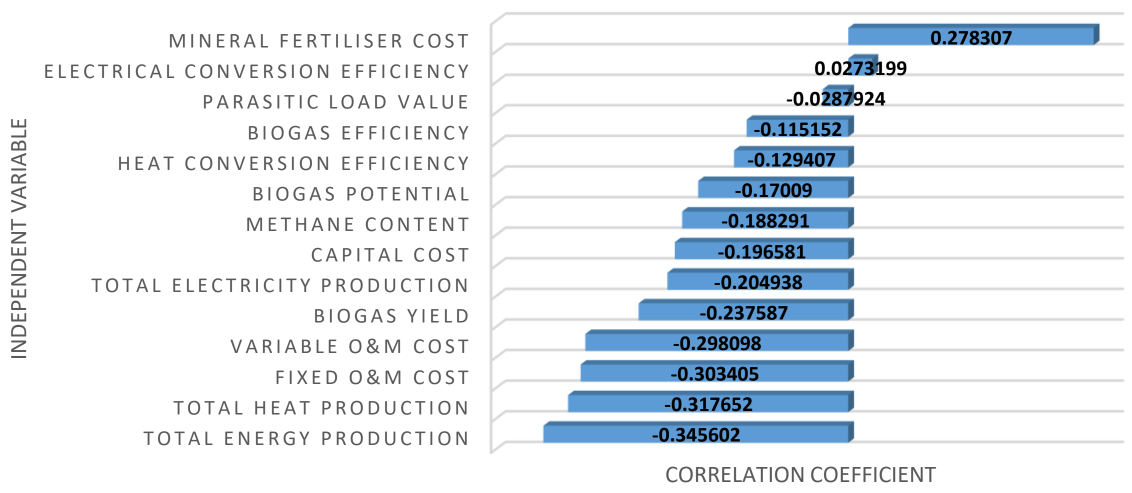

Figure 5. Correlation coefficients for the 14 independent variables and their influence on NPV.

Discussion

Table 10 provided the NPV, BCR, payback period and MIRR for AD compared with traditional land application from a purely economic viewpoint. The payback period of 5.34 years for the AD plant is similar to the findings of Kabir *et al*. (2015) and Orive *et al.* (2016), who reported <8 years and 6.7 years, respectively, in their techno-economic assessments of AD performance. Whilst neither of these studies considered PL as a feedstock, the digestor size and feedstock volume are comparable to the scenario described in this study. In addition, the NPV of £707.17 per tonne of feedstock is comparable to that reported by Li *et al.* (2021) ($972.7 per tonne of feedstock) but far higher than those reported by Li *et al*. (2020) ($75 per tonne feedstock). However, the latter study compared 2 systems in 3 different scenarios, all of which were purchasing a lower methane potential feedstock, therefore, resulting in low gas production and low product sale revenue. Both aforementioned studies focused on small-scale, on farm AD plants, suggesting they are directly comparable with our study.

When including the environmental costs, the NPV of both AD and land application became negative, suggesting that neither option is economically and environmentally viable. This finding is supported by Bora *et al*. (2020) in their comprehensive LCA and TEA study focusing on multiple valorisation options for PL. There are, however, several key differences between this study and that of Bora *et al.* (2020). Firstly, the Bora study uses a real scenario in the USA, rather than a theoretical UK scenario. Secondly, the volume and composition of the PL differ considerably in our study, thereby altering the required digestor size, capital and operational costs, biogas yield and associated energy production. Furthermore, Bora *et al*. (2020) did not use PL as a direct AD feedstock but used hydrothermal liquefaction as a pretreatment step.

The five environmental impact categories considered in the LCA component of this study have varying influence on the economic feasibility of the valorisation methods. There is an overall difference of £387.20 per tonne of PL between AD (£149.9) and land application (£537.1), with particulate matter formation and acidification potential exerting the most influence with values of £207.42 and £328.21, respectively, for land application, and £130.36 and £54.32, respectively, for AD. These high values were expected for the traditional land application approach, due to high ammonia and hydrogen sulphide emissions during land application. There is a notable difference in these environmental impacts when comparing land application to AD, with a reduction of £49.42 and £273.89 per tonne of PL for particulate matter formation and acidification potential, respectively. However, despite the reduction, these two impact categories are still relatively high in AD and are the main contributors to the economic infeasibility of the scenario. With 1kg of particulate matter and 1 kg of sulphur dioxide being priced at £31.96 and £7.99, respectively, they are the two highest costing impacts out of the five considered. In order to understand the reasons for this, one needs to recognise the notable levels of hydrogen sulphide and ammonia that are emitted during the degradation of biomass during the AD process. Ammonia is easily oxidised to NO_2, then hydrated to nitric acid, whilst hydrogen sulphide is oxidised to SO_2 then hydrated to sulphuric acid, thereby both contributing the acidification potential. With regards to particulate matter formation, again ammonia is key as $PM_{2.5}$ constitutes high levels of ammonium ions formed when ammonia gas reacts with NO_x and SO_x in the atmosphere. These levels can be reduced in AD through the upgrading of biogas to biomethane by implementing membrane separation, or water scrubbing, followed by the use of a thermal power plant. This process would effectively lower the ammonia and hydrogen sulphide emissions to levels that are on a par with natural gas.

Within AD, the GWP impact is also of note with a negative value, suggesting that AD has a positive impact on the environment. This negative figure is largely due to the avoided burdens of both fertiliser production and electricity production afforded by the use of AD. Indeed, whilst the GWP impact costs for AD amount to £12.82 per tonne of PL, the avoided burdens through energy and fertiliser productions amount to £47.59 per tonne of PL, thereby resulting in an enviro-economic gain of £34.77.

With regards to the small number of positive NPV results arising from the Monte Carlo simulation, results of the multiple linear regression indicated that there was a very strong collective significant effect between the Biogas Potential, Biogas efficiency, Methane content, Electrical conversion efficiency, Heat conversion efficiency, Parasitic load value, Mineral fertiliser cost, Biogas yield, Total energy production, Fixed O&M Cost, Variable O&M cost, Capital cost, Total electricity production, Total heat production, and NPV, ($F(7, 65) = 893.38$, $p < .001$, $R^2 = 0.99$, $R^2_{adj} = 0.99$). The individual predictors were examined further and indicated that Biogas Potential (t = -3.256, p = .002), Biogas efficiency (t = -37.564, p < .001), Methane content (t = 59.002, p < .001), Electrical conversion efficiency (t = -13.358, p < .001), Heat conversion efficiency (t = -44.538, p < .001), Parasitic load value (t = -41.758, p < .001) and Mineral fertiliser cost (t = 2.927, p = .005) were significant predictors in the model.

The tornado chart in Figure 5 showed that mineral fertiliser cost is strongly positive, suggesting that the higher the price of mineral fertiliser, the more cost-effective AD becomes. This is obvious as the average avoided financial burden of fertiliser purchase from the Monte Carlo simulation is in excess of £2.7m. Conversely, biogas potential, biogas efficiency, methane content, heat conversion efficiency, and parasitic load are strongly negative, suggesting that the more energy that is produced, the lower the NPV becomes. This is surprising; however, this may be due to higher environmental impact emissions being associated with higher energy production.

Within our study, it is assumed that there is no cost for feedstock or no financial value for the digestate fertiliser; these are scenarios that could have a significant impact on the economic feasibility of the AD plant. Furthermore, Renewable Heat Incentive payments, and other government initiatives, are also not considered. As such, further research should consider these aspects to determine the economic impact that these scenarios would have on the feasibility of AD use.

Conclusion

In conclusion, AD shows notable promise as a cost effective and environmentally beneficial valorisation option for PL. Whilst the average NPV from the Monte Carlo simulation for both land application and AD was negative, the simulation showed that it is possible to generate a positive NPV for AD with a favourable payback period through the operational optimisation of the technology. Surprisingly, the optimisation involves reducing the biogas and energy yield. Further research into this is required to fully understand the optimisation process. However, it should also be noted that the positive socio-environmental-economic NPV is also largely determined by the cost of mineral fertiliser as an avoided burden as the average cost of fertiliser needed exceeds £2.7 million and is the highest variable cost. Further research is also required to compare this valorisation method with other options, such as gasification, pyrolysis and incineration. By using the model created in this study, this comparison would provide a comprehensive socio-enviro-economic model to evaluate the costs and benefits of these valorisation methods with AD, in order to determine the optimal technology.

References

Ahn, H.K., Mulbry, W., White, J.W. and Kondrad, S.L. (2011) 'Pile mixing increases greenhouse gas emissions during composting of dairy manure', Bioresource technology, 102(3), pp. 2904-2909.

Azapagic, A. (2003) 'Systems approach to corporate sustainability: a general management framework', Process Safety and Environmental Protection, 81(5), pp. 303-316.

Bernstad, A. and la Cour Jansen, J. (2011) 'A life cycle approach to the management of household food waste–a Swedish full-scale case study', Waste Management, 31(8), pp. 1879-1896.

Bora, R.R., Lei, M., Tester, J.W., Lehmann, J. and You, F. (2020) 'Life cycle assessment and technoeconomic analysis of thermochemical conversion technologies applied to poultry litter with energy and nutrient recovery', ACS Sustainable Chemistry & Engineering, 8(22), pp. 8436-8447.

Cabrera, M.L., Chiang, S.C., Merka, W.C., Thompson, S.A. and Pancorbo, O.C. (1993) 'Nitrogen transformations in surface-applied poultry litter: Effect of litter physical characteristics', Soil Science Society of America Journal, 57(6), pp. 1519-1525.

Choi, I.H. and Moore Jr, P.A. (2008) 'Effect of various litter amendments on ammonia volatilization and nitrogen content of poultry litter', Journal of Applied Poultry Research, 17(4), pp. 454-462.

Christensen, T.H., Gentil, E., Boldrin, A., Larsen, A.W., Weidema, B.P. and Hauschild, M. (2009) 'C balance, carbon dioxide emissions and global warming potentials in LCA-modelling of waste management systems', Waste Management & Research, 27(8), pp. 707-715.

Clift, R. (2013) 'System approaches: life cycle assessment and industrial ecology', Pollution 5th Edition: Causes, Effects and Control, 5, pp. 385.

Clift, R. (2003) 'An introduction to life cycle assessment', IB Revija (Ljubliana), 37(4), pp. 70-79.

Eriksson, O., Finnveden, G., Ekvall, T. and Björklund, A. (2007) 'Life cycle assessment of fuels for district heating: A comparison of waste incineration, biomass-and natural gas combustion', Energy Policy, 35(2), pp. 1346-1362.

Evangelisti, S., Lettieri, P., Borello, D. and Clift, R. (2014) 'Life cycle assessment of energy from waste via anaerobic digestion: a UK case study', Waste Management, 34(1), pp. 226-237.

Finnveden, G. (2008) 'A world with CO 2 caps: Electricity production in consequential assessments', The International Journal of Life Cycle Assessment, 13, pp. 365-367.

Forster, P., Ramaswamy, V., Artaxo, P., Berntsen, T., Betts, R., Fahey, D.W., Haywood, J., Lean, J., Lowe, D.C. and Myhre, G. (2007) 'Changes in atmospheric constituents and in radiative forcing. Chapter 2'Climate change 2007. The physical science basis.

Fruergaard, T. and Astrup, T. (2011) 'Optimal utilization of waste-to-energy in an LCA perspective', Waste Management, 31(3), pp. 572-582.

Fruergaard, T., Astrup, T. and Ekvall, T. (2009) 'Energy use and recovery in waste management and implications for accounting of greenhouse gases and global warming contributions', Waste Management & Research, 27(8), pp. 724-737.

Gerber, L., Gassner, M. and Maréchal, F. (2011) 'Systematic integration of LCA in process systems design: Application to combined fuel and electricity production from lignocellulosic biomass', Computers & Chemical Engineering, 35(7), pp. 1265-1280.

Gikas, P. (2014) 'Electrical energy production from biosolids: a comparative study between anaerobic digestion and ultra-high-temperature gasification', Environmental technology, 35(17), pp. 2140-2146.

Grossmann, I.E. and Guillén-Gosálbez, G. (2010) 'Scope for the application of mathematical programming techniques in the synthesis and planning of sustainable processes', Computers & Chemical Engineering, 34(9), pp. 1365-1376.

Guinée, J.B. and Heijungs, R. (2007) 'Life Cycle Assessment, Kirk-Othmer Chemical Technology and the Environment', Wiley-Interscience Publication, 1, pp. 20-46.

Harmel, R.D., Smith, D.R., Haney, R.L. and Dozier, M. (2009) 'Nitrogen and phosphorus runoff from cropland and pasture fields fertilized with poultry litter', Journal of Soil and Water Conservation, 64(6), pp. 400-412.

Huijsmans, J., Verwijs, B., Rodhe, L. and Smith, K. (2004) 'Costs of emission-reducing manure application', Bioresource technology, 93(1), pp. 11-19.

Jurgutis, L., Slepetiene, A., Volungevicius, J. and Amaleviciute-Volunge, K. (2020) 'Biogas production from chicken manure at different organic loading rates in a mesophilic full scale anaerobic digestion plant', Biomass and Bioenergy, 141, pp. 105693.

Kabir, M.M., Rajendran, K., Taherzadeh, M.J. and Horváth, I.S. (2015) 'Experimental and economical evaluation of bioconversion of forest residues to biogas using organosolv pretreatment', Bioresource technology, 178, pp. 201-208.

Kim, S., Agblevor, F.A. and Lim, J. (2009) 'Fast pyrolysis of chicken litter and turkey litter in a fluidized bed reactor', Journal of Industrial and Engineering Chemistry, 15(2), pp. 247-252.

Li, Y., Han, Y., Zhang, Y., Luo, W. and Li, G. (2020) 'Anaerobic digestion of different agricultural wastes: A techno-economic assessment', Bioresource technology, 315, pp. 123836.

Li, Y., Qi, C., Zhang, Y., Li, Y., Wang, Y., Li, G. and Luo, W. (2021) 'Anaerobic digestion of agricultural wastes from liquid to solid state: Performance and environ-economic comparison', Bioresource technology, 332, pp. 125080.

Lorimor, J.C. and Xin, H. (1999) 'Manure production and nutrient concentrations from high-rise layer houses', Applied Engineering in Agriculture, 15(4), pp. 337-340.

Mahdy, A., Bi, S., Song, Y., Qiao, W. and Dong, R. (2020) 'Overcome inhibition of anaerobic digestion of chicken manure under ammonia-stressed condition by lowering the organic loading rate', Bioresource Technology Reports, 9, pp. 100359.

Manfredi, S., Tonini, D. and Christensen, T.H. (2011) 'Environmental assessment of different management options for individual waste fractions by means of life-cycle assessment modelling', Resources, Conservation and Recycling, 55(11), pp. 995-1004.

Moller, I.S., Kichey, T., Ingvordsen, C.H. and Schjoerring, J.K. (2009) 'Foliar nitrogen application in wheat: the effects on grain N content, recovery of fertilizer and the response of cytosolic glutamine synthetase'.

Murphy, J.D. and Power, N.M. (2006) 'A technical, economic and environmental comparison of composting and anaerobic digestion of biodegradable municipal waste', Journal of Environmental Science and Health Part A, 41(5), pp. 865-879.

Murphy, J.D. and Thamsiriroj, T. (2013) 'Fundamental science and engineering of the anaerobic digestion process for biogas production'The biogas handbook Elsevier, pp. 104-130.

Nahm, K.H. (2005) 'Environmental effects of chemical additives used in poultry litter and swine manure', Critical Reviews in Environmental Science and Technology, 35(5), pp. 487-513.

Orive, M., Cebrián, M. and Zufia, J. (2016) 'Techno-economic anaerobic co-digestion feasibility study for two-phase olive oil mill pomace and pig slurry', Renewable Energy, 97, pp. 532-540.

Palma, C.F. and Martin, A.D. (2013) 'Inorganic constituents formed during small-scale gasification of poultry litter: A model based study', Fuel Processing Technology, 116, pp. 300-307.

Patterson, T., Esteves, S., Dinsdale, R. and Guwy, A. (2011) 'Life cycle assessment of biogas infrastructure options on a regional scale', Bioresource technology, 102(15), pp. 7313-7323.

Pote, D.H. and Meisinger, J.J. (2014) 'Effect of poultry litter application method on ammonia volatilization from a conservation tillage system', Journal of Soil and Water Conservation, 69(1), pp. 17-25.

Rao, A.G., Gandu, B., Sandhya, K., Kranti, K., Ahuja, S. and Swamy, Y.V. (2013) 'Decentralized application of anaerobic digesters in small poultry farms: Performance analysis of high rate self mixed anaerobic digester and conventional fixed dome anaerobic digester', Bioresource technology, 144, pp. 121-127.

Skowroñska, M. and Filipek, T. (2014) 'Life cycle assessment of fertilizers: a review', International Agrophysics, 28(1).

Vandecasteele, B., Reubens, B., Willekens, K. and De Neve, S. (2014) 'Composting for increasing the fertilizer value of chicken manure: effects of feedstock on P availability', Waste and Biomass Valorization, 5, pp. 491-503.

Vervoort, R.W. and Keeler, A.G. (1999) 'The economics of land application of fresh and composted broiler litter with an environmental constraint', Journal of environmental management, 55(4), pp. 265-272.

Walker, M., Theaker, H., Yaman, R., Poggio, D., Nimmo, W., Bywater, A., Blanch, G. and Pourkashanian, M. (2017) 'Assessment of micro-scale anaerobic digestion for management of urban organic waste: A case study in London, UK', Waste Management, 61, pp. 258-268.