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Potential Greenhouse Gas Mitigation from Utilising Pig Manure and Grass for Hydrothermal Carbonisation and Anaerobic Digestion in the UK, EU, and China

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Abstract: Pig manure currently results in sizeable greenhouse gas emissions, during storage and spreading to land. Anaerobic digestion and hydrothermal carbonisation could provide significant greenhouse gas mitigation, as well as generate renewable heat and power (with anaerobic digestion), or a peat-like soil amendment product (with hydrothermal carbonisation). The greenhouse gas mitigation potential associated with avoidance of pig manure storage and spreading in the UK, EU, and China, as well as the potential to provide heat and power by anaerobic digestion and soil amendment products by hydrothermal carbonisation was herein determined. In each case, the mono-conversion of pig manure is compared to co-conversion with a 50:50 mixture of pig manure with grass. Anaerobic digestion displayed a greater greenhouse gas mitigation potential than hydrothermal carbonisation in all cases, and co-processing with grass greatly enhances greenhouse gas mitigation potential. China has the largest greenhouse gas mitigation potential (129 MT CO_2 eq), and greatest mitigation per kg of pig manure (1.8 $kgCO_2$/kg pig manure volatile solids). The energy grid carbon intensity has a significant impact on the greenhouse gas mitigation potential of the different approaches in the different regions. Pig manure is generated in large amounts in China, and the energy generated from biogas offsets a higher carbon intensity grid. Greenhouse gas savings from the anaerobic digestion of pig manure and grass have been calculated to provide a significant potential for reducing total greenhouse gas emissions representation in China (1.05%), the EU (0.92%), and the UK (0.19%). Overall, the utilisation of pig manure could bring about substantial greenhouse savings, especially through co-digestion of pig manure with grass in countries with large pig farming industries and carbon intense energy mixes.

Keywords: hydrothermal carbonisation; anaerobic digestion; pig manure; grass; peat substitution; GHG mitigation

1. Introduction

Background

Vast quantities of manure are generated annually, with pig manure (PM) making up a large proportion of the total. Approximately 122 million tonnes of pork is consumed globally every year, with pork accounting for 35% of total meat consumption [1]. This consumption is expected to increase to 138 million tonnes by 2030, and 149 million tonnes by 2050, when compared to 2020 levels [2]. Consequentially, the pig farming industry generates around 1.7 billion tonnes of PM per year, which is predicted to increase to over 2 billion tonnes per year by 2050 [1]. The majority of PM is stored in open tanks for long periods of time and then applied directly to land, leading to significant greenhouse gas (GHG) emissions [3], potential nutrient leaching, and the release of pathogens [4]. Li et al. [3] conducted a Life Cycle Assessment (LCA) of the GHGs associated with pig farming, and found the management of manure to be responsible for the majority of emissions.

The two regions with the largest pig numbers and subsequent PM generation are China and the EU, respectively, while the UK generates comparatively lower quantities [5,6]. The

UK generates approximately 0.7 million tonnes of PM per year, whereas the EU generates considerably higher quantities, at almost 23 million tonnes of PM per year (Table 1). This is dwarfed by the approximately 72 million tonnes of PM per year generated in China (Table 1).

To reduce GHG emissions associated with storage and spreading, the PM can be processed by technologies, such as anaerobic digestion (AD) or hydrothermal carbonisation (HTC). HTC involves the conversion of organic biomass in water at elevated temperatures and pressures ranging between 180–260 °C and 20–40 bar pressure. The process generates a solid carbonaceous product known as hydrochar (HC), a process water (PW) containing soluble organic and inorganic components, and a gaseous fraction consisting primarily of CO_2 [7]. HTC can be applied to many organic feedstock types; however, HTC is typically best suited for the processing of high-moisture feedstock types, such as wet wastes, sludges, and manures. The moisture content of HTC reactions typically ranges from 70–90 wt%, with retention times varying from minutes to hours [7].

During HTC, manures can be processed alone or co-processed with alternative sources of biomass, such as lignocellulosic biomass and green wastes. Vast quantities of green waste, such as grasses are generated in different regions and sites including farms, parks, and roadside verges. This waste is typically left unutilized to decompose or composted [8]. Therefore, grass represents a potentially interesting co-feedstock for both AD and HTC. The utilisation of the resulting HC has largely focused on the production of a peat-like soil amendment product or for energy generation as a solid fuel (bio-coal). Whereas, the PW can be anaerobically digested to generate energy and digestate, which could contribute to powering the HTC process. The extraction of peat and its usage for horticultural practices is a significant contributor to global GHG emissions [9]. Roy et al. [9] investigated the GHG mitigation potential of using HTC to create a substitute 'peat-like' soil amendment product. Substantial GHG mitigation opportunities associated with HTC and peat substitution were identified. Not all of the nutrients (e.g., nitrogen and phosphorus) in the feedstock are released into the PW. Only approximately 40% of the nitrogen (N) [10] and 19% of the phosphorous (P) are passed into the PW and resultant digestate, which can be applied to land as a biofertiliser [11]. This results in a loss of fertiliser substitution associated with HTC, compared to the direct land application of PM, where all of the nutrients can be utilised to substitute chemical fertilisers. Alternatively, if processed by HTC, the HC can be used as a solid fuel. Co-processing of PM with lignocellulosic biomass has advantages, such as improving the calorific value of the resulting HC and improving combustion behaviour [12,13].

Another approach for processing PM is biomethane generation through AD. Generally, PM needs to be co-digested with other feedstocks to ensure a suitable C:N ratio and avoid inhibition. The co-digestion of PM with agri-residues and grasses is feasible and can enhance biomethane yields [1]. AD is generally considered to be the most suitable approach for generating energy from PM [14], and is the most mature technology. The biomethane produced can be used to generate heat and power, and the digestate can be used as a biofertiliser. This can be directly applied to land for recovering the majority of nutrients from the original feedstocks [1]. Zhang et al. [1] found substantial GHG mitigation opportunities from the AD of PM, with these opportunities being greatly enhanced by co-digestion with grass. Furthermore, studies by Dargahi et al. [15], Shokoohi et al. [16], and Almasi et al. [17] have identified significant GHG mitigation opportunities from AD of organic wastes.

The co-processing of PM with feedstocks, such as grass by HTC or AD has clear advantages for mitigating GHG emissions. The GHG mitigation from the different processing options is influenced by the carbon intensity of the grid in operation in the different regions. Grid energy carbon intensity refers to the equivelent CO_2 emissions associated with using a set amount of main energy and is usually measured in $kgCO_2/kWh$. Therefore, the carbon intensity of grid energy required for heating the HTC process, the emissions offset by generating energy from AD, and GHG mitigation associated with chemical fertiliser substitution by direct spreading of PM or digestate to land will vary considerably in the

different regions [18–21]. As a result, the carbon intensity will affect the choice of mitigation technology in the different regions, and thus, it is useful to compare and contrast how the choice of technology for treating PM (HTC or AD) would differ in regions with different grid carbon intensities. However, a comparison of these technologies and the effect of grid carbon intensity has yet to be made.

The UK has a relatively low grid energy carbon intensity of roughly 0.17 $kgCO_2/kWh$, as well as relatively low GHGs associated with chemical fertilisers, 1.39 kg CO_2eq per kg N and 1.08 kg CO_2eq per kg P (Table 1). The UK would have lower GHGs associated with net energy inputs, such as from HTC, as well as a lower GHG mitigation from net energy generation and high chemical fertiliser generation, such as with AD.

Overall, the EU has a slightly higher grid energy carbon intensity when compared to the UK (0.28 $kgCO_2/kWh$), as well as the same relatively low GHGs associated with chemical fertilisers, 1.39 kg CO_2eq per kg N and 1.08 kg CO_2eq per kg P (Table 1). This results in slightly higher GHGs associated with net energy inputs and a slightly higher GHG mitigation from net energy generation, when compared to the UK. However, mitigation from chemical fertiliser substitution would remain the same per kilogram generated. China has a significantly higher grid energy carbon intensity of 0.58 $kgCO_2/kWh$, as well as higher GHGs associated with chemical fertilisers containing nitrogen at 3.04 kg CO_2eq per kg N; however, GHG mitigation for phosphorous is the same as for the UK and EU at 1.08 kg CO_2eq per kg P (Table 1).

Table 1. Pig manure estimation and product emission/mitigation potential in each country.

	UK	EU	China	References
Feedstock quantities				
Pig number (millions)	4.1	141.7	449.22	[5,6]
Tonne volatile solid (VS)/pig	0.16	0.16	0.16	[21]
Manure total per year (tonne of VS)	0.66	22.67	71.88	
GHG emission/mitigation figures				
Energy intensity ($kgCO_2/kWh$)	0.168	0.28	0.58	[18,19]
N substitution mitigation per kg N	1.39	1.39	3.04	[20]
P substitution mitigation per kg P	1.08	1.08	1.08	[21]

This research investigates the practical implications of both co-digestion of PM and grass by AD and HC utilisation and peat substitution, such as the impact of HC quality and differences between peat and HC. This paper, for the first time, compares the GHG mitigation potential for AD and HTC of pig manure with grass, in China, EU, and UK, and the effect of energy intensity. To the authors knowledge, no study has specifically assessed the comparative GHG mitigation potential of the different approaches in the selected regions. The assessment can be used to help policy makers understand the theoretical opportunities for HTC and AD in the different regions.

2. Materials and Methods

2.1. Methods Outline

A quantitative assessment of the total mitigation potential (total MT CO_2eq/year), as well as the average mitigation potential per tonne of PM volatile solids (tonne of CO_2eq/tonne of PM VS) was carried out across each region: UK, EU, and China. The HC was assumed to substitute horticultural peat use, while the PW was anaerobically digested to generate energy to power the HTC process, as well as contributing to nutrients in the digestate. Second, the co-digestion of PM with grass via AD is assumed to produce biogas to generate heat, in order to substitute natural gas and electricity to substitute grid energy, while the digestate is used to substitute chemical fertilisers.

2.2. GHG Mitigation Scenarios

Figure 1 illustrates the two GHG mitigation scenarios explored. The HTC scenario involves the treatment of PM, as well as the co-processing of PM and grass. This scenario considers GHG mitigation from atmospheric emissions avoidance of PM storage, as well as GHG mitigation from using HC generated to substitute peat usage in horticulture. This scenario includes the GHG emissions associated with the net energy input requirements for HTC, as well as the reduced biofertiliser production compared to the business as usual (BAU) scenarios of direct land application of PM. The AD scenario includes the digestion of PM, as well as the co-processing of PM and grass. This scenario considers GHG mitigation by avoiding atmospheric emissions from PM storage, as well as GHG mitigation from generating heat and power, with the energy generated substituting grid energy in the region. In each case, the carbon intensities of grid energy for power generation and production of chemical fertilisers are considered for each of the different regions.
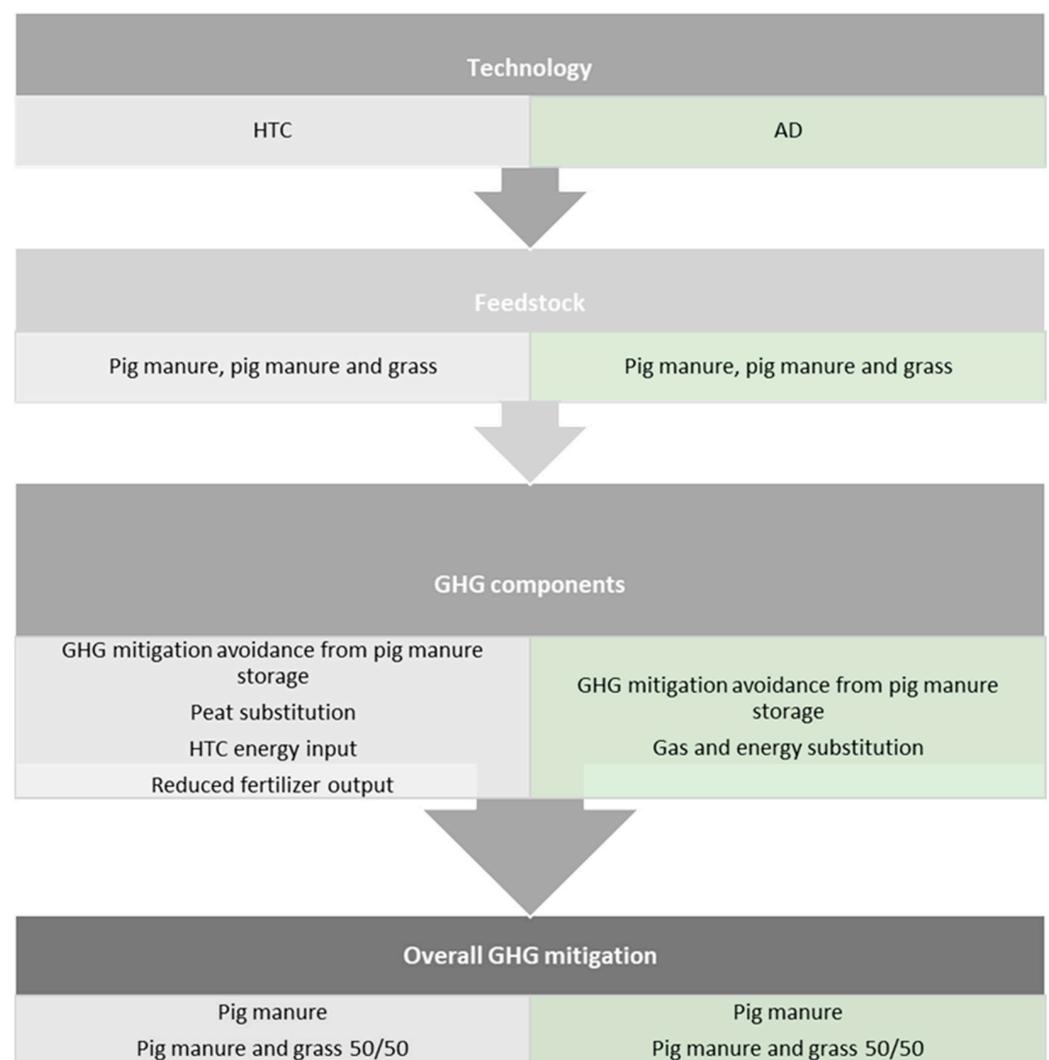
Figure 1. GHG mitigation scenarios explored.

For the AD scenario, it is assumed that 1.5% of N is lost during the AD process, and fugitive NH_3 and N_2O emissions are negligible [1]. For both scenarios, it is assumed that all of the PM storage emissions could be avoided by continuously feeding the HTC unit or the anaerobic digester. Moreover, it is assumed that there would be sufficient quantities of grass available to co-feed the HTC and AD units with a 50:50 mixture (on a VS basis).

Total quantities of PM generated in each region were estimated from the literature, based on pig numbers and typical VS quantities per pig, per year (Table 1).

2.3. PM Storage Emission Avoidance

The mitigation from avoided PM storage was calculated to be 0.71 TCO$_2$/TPMVS, as obtained from a study by Zhang et al. [1]. This figure is based on the atmospheric emissions of methane and nitrous oxide. The PM methane production capacity was assumed to be 0.26 m^3 CH$_4$/kg VS, with the factor converting CH$_4$ from m^3 to kg as 0.67 [22,23]. The N$_2$O emission factor from raw manure storage was 0.38% of the total N, while the N$_2$O emission factor from atmospheric deposition was 0.01 kg N$_2$O–N/kg N and from leaching/runoff was 0.0075 kg N$_2$O–N/kg N [1].

2.4. Description of the HTC Peat Substitution Scenario

The HTC scenario assumes that all of the PM generated in each region is processed by HTC alone or co-processed with equal amounts of grass (on a VS basis).

Two HTC temperatures (200 and 250 °C) are considered, each with a reaction time of 60 min. The feedstocks were processed as received with a moisture content of 75% and 81% for grass and PM, respectively. The HC is utilised as a soil amendment product in horticulture to substitute the current peat usage and mitigate 830.52 kgCO$_2$/THC [9], while the digestate is used as a biofertiliser to substitute chemical fertiliser usage onsite, with a mitigation per tonne N varying in each country [20]. HC and carbon yields were obtained from the experimental work. The net energy inputs were calculated using models proposed by Borbolla-Gaxiola et al. [24], based on subtracting the energy generated from AD of the PW from the energy required to hydrothermally carbonise the PM and grass. HTC energy inputs have been calculated using Equation (1), while the PW energy output was calculated using Equation (2). Moreover, it was assumed that 40% of the N [10] and 19% of the P [11] would partition into the PW.

$$\text{Energy input (KJ)} = -1970 + 8.4\ T\ (°C) - 0.2\ RT\ (min) + 47\ MC\ (\%)0.0393\ T\ (°C){}^*T\ (°C)0.0541\ RT\ (min){}^*RT\ (min)0.46\ MC\ (\%){}^*MC\ (\%) + 0.0986\ T\ (°C){}^*RT\ (min) + 0.080\ T\ (°C){}^*MC\ (\%)0.002\ RT\ (min){}^*MC\ (\%) \quad (1)$$

$$\text{PW energy(KJ)} = -1389 + 5.51T(°C) - 1.3\ RT(min) + 21.6MC(\%) + 0.0060\ T(°C){}^*T\ (°C) + 0.0430RT(min){}^*RT\ (min) - 0.005\ MC(\%){}^*MC\ (\%) - 0.0218\ T(°C){}^*RT\ (min) - 0.0912T(°C){}^*MC(\%) + 0.032RT(min){}^*MC\ (\%) \quad (2)$$

In Equations (1) and (2), T is temperature, RT is retention time, and MC is moisture content.

2.5. Description of the AD Heat and Power Generation Scenario

The AD scenario assumes that all of the PM generated in each region is digested alone or co-digested with equal amounts of grass (on a VS basis). The biogas generated is used for heat and power (using a CHP boiler) and the digestate is used to substitute chemical fertiliser use onsite. The biomethane potential for mono-digestion of PM was estimated to be 154 mL CH$_4$/g VS, rising to 251 mL CH$_4$/g VS when PM was co-digested with grass on a 50:50 mixture on a VS basis [25]. A CHP boiler would be used to generate heat and power with respective efficiencies of (55%) for heat and (30%) for power. The parasitic loads were 19% for heat and 20% for energy. The biomethane content of the biogas was assumed to be 70% and the methane calorific value was assumed to be 9.8 kWh/m^3, based on data provided by an AD supplier. It was assumed that the fugitive CH$_4$, NH$_3$, and N$_2$O emissions from anaerobic digesters are negligible, and thus, were discounted [26].

2.6. Grid Energy Use/Substitution and Heat Substitution

GHG mitigation figures for grid energy substitution were calculated for the different regions, where the energy generated from AD would be used to contribute directly toward the national grid energy supply, while the net energy required for the HTC process would

be obtained directly from the national grid supply. The grid energy use and substitution was performed by multiplying the net energy generation (GWh) by the grid carbon intensity (Kg CO_2 eq/kWh) (displayed in Table 1), as shown in Equation (3).

$$GWF \text{ (grid)} = E(total) \times CIgrid \tag{3}$$

In Equation (3), GWF (grid) is the total CO_2 eq mitigation from grid substitution/usage, E(total) is the total energy generated (in GWh, after efficiency losses), and CIgrid is the carbon intensity of the grid (in g CO_2 eq/kWh).

Heat generated by AD would be used to substitute natural gas in the different countries, with natural gas having a carbon intensity of 0.185 kgCO_2/kWh [27].

2.7. Fertiliser Substitution

While it was assumed that digestate from the AD scenario would retain the same quantity of N and P for synthetic fertiliser substitution as the direct land application of PM, in the HTC scenerio, some of the N and P would be retained in the HC and would not be available as a biofertiliser. It was assumed that 40% of the N [10] and 19% of the P [11] would be released into the PW, leading to a 60% loss in biofertiliser substitution potential of N and an 81% loss of P. In reality, the HC would contain the rest of the nutrients and would be applied to land; however, it would be used to replace peat, which has similar levels of N and P, and thus, the associated GHG mitigations are accounted for with peat substitution. The reduced mitigation potential from chemical fertiliser substitution was calculated as detailed in Equation (4).While it is likely that P fertiliser emissions would differ in the different regions, a lack of literature comparing the carbon intensity of P fertiliser production in the UK, EU, and China indicates that a single figure is needed to be applied for all regions.

$$GWF \text{ (fertiliser)} = (N(total) \times CinN) + (P(total) \times CinP) \tag{4}$$

In Equation (4), GWF (fertiliser) is the total CO_2eq mitigation from fertiliser substitution, N(total) is the total reduction in N generated (in biofertiliser form), CinN is the carbon intensity of chemical N fertiliser, P(total) is the total reduction in P generated (in biofertiliser form), and CinP is the carbon intensity of chemical P fertiliser.

2.8. Mitigation Per Tonne of PM VS

Total GHG mitigation figures (MT CO_2 eq) were converted into GHG mitigation per tonne of PM VS (TCO_2/T PM) by dividing the total mitigation (MT CO_2 eq) by the quantities of PM VS generated in each region, with the PM quantities (Table 1). The calculation is shown in Equation (5), where grass is co-processed with PM, the mitigation per tonne of PM VS includes the net mitigation from avoided PM storage and spreading from 1 tonne of PM VS, as well as the mitigation from the HTC or AD of both 1 tonne of PM VS and 1 tonne of grass VS.

$$MpPM = GWF \text{ (total)} \div PMvs \tag{5}$$

In Equation (5), MpPM is the CO_2 eq mitigation per tonne of PM (TCO_2/T PM), GWF (total) is the total GHG mitigation for the different mitigation options (MT CO_2 eq), and PMvs is the total PM (in VS) for the PM in the different regions (MT).

3. Results

The results first include regional-specific findings for the UK, the EU, and then China and include the individual contributions of the GHG mitigation components: (i) GHG mitigation from avoided GHGs from PM storage; (ii) GHG mitigation through the energy generated from PM and PM-grass HTC to substitute peat; and (iii) heat and power mitigation from AD of PM and PM-grass. Both technologies are compared for each of the regions.

3.1. UK Greenhouse Gas Mitigation Potential

The GHG reductions per tonne of total PM VS for the different GHG mitigation scenarios have been calculated for HTC (Table 2) and AD (Table 3). Avoided PM storage and spreading has the greatest GHG mitigation, representing 0.37 MT of equivalent CO_2 and reducing the GHGs associated with UK PM by 0.56 tonnes of CO_2 eq for every tonne of PM VS generated in the country. HTC of PM with subsequent peat substitution had a total mitigation of 0.14 MT of CO_2 eq and 0.21 tonnes of CO_2 eq per tonne of VS when processed at 200 °C for 1 h. This reduces by slightly over a third (35.7%) to 0.09 MT of CO_2 eq and 0.14 tonnes of CO_2 eq tonne of VS when the process temperature is increased to 250 °C, due to a greatly increased net energy input and a slight reduction in peat substitution. These figures assume a like-for-like substitution of peat for hydrochar, which assumes high humic content in the hydrochar.

Table 2. UK GHG mitigation from HTC of PM and PM-grass blend.

Feedstock and HTC Temperature	Pig Manure 200 °C	Pig Manure 250 °C	Grass and Pig Manure Blend 200 °C	Grass and Pig Manure Blend 250 °C
Net HTC GHG mitigation (TCO_2eq/TVS)	0.21	0.14	0.20	0.13
Net GHG mitigation from avoided PM storage and spreading emissions per tonne of pig manure VS (TCO_2eq/TVS)	0.56	0.56	0.56	0.56
Total Net GHG mitigation per tonne of VS PM (TCO_2eq/TVS)	0.77	0.70	0.96	0.82
Manure total per year (million tonnes of VS)	0.66	0.66	0.66	0.66
Total GHG mitigation from HTC (million tonnes of CO_2eq/year)	0.14	0.09	0.26	0.17
Total GHG mitigation from avoided storage (million tonnes of CO_2eq/year)	0.37	0.37	0.37	0.37
Total GHG mitigation/year (million tonnes of CO_2eq/year)	0.51	0.46	0.63	0.54

Table 3. UK GHG mitigation from AD of PM and PM-grass blend.

Feedstock and HTC Temperature	Pig Manure	Grass and Pig Manure Blend
Net AD GHG mitigation (TCO_2eq/TVS)	0.19	0.30
Net GHG mitigation from avoided PM storage and spreading emissions per tonne of pig manure VS (TCO_2eq/TVS)	0.71	0.71
Total Net GHG mitigation per tonne of VS PM (TCO_2eq/TVS)	0.90	1.31
Manure total per year (million tonnes of VS)	0.66	0.66
Total GHG mitigation from AD (million tonnes of CO_2eq/year)	0.12	0.40
Total GHG mitigation from avoided storage (million tonnes of CO_2eq/year)	0.47	0.47
Total GHG mitigation/year (million tonnes of CO_2eq/year)	0.59	0.86

The addition of grass to PM results in a near-doubling in the total GHG mitigation, from 0.14 to 0.26 tonnes of CO_2 eq tonne of VS (200 °C for 1 h) and from 0.09 to 0.17 tonnes of CO_2 eq tonne of VS (250 °C for 1 h), which is largely due to the increased HC yields. This implies that the co-processing of PM with feedstocks that increase HC yields (such as woody biomass), may increase GHG mitigation further. The net GHG mitigation per tonne of VS for the PM-grass blend was slightly lower, compared to processing PM alone. This is due to a slightly higher net energy input requirement for HTC, as well as a slight decrease in the quantity of peat substitution per tonne of VS processed. Altogether, the maximum total GHG mitigation for the HTC of PM (200 °C for 1 h), associated with avoided storage,

corresponds to 0.51 MT of equivalent CO_2, rising to 0.63 MT eq CO_2 when co-processed with grass.

The overall GHG mitigation potential from AD is higher than HTC in the UK (Table 3). The key explanation for the enhanced net GHG mitigation results from the net avoided PM storage and spreading, which is 27% higher than HTC (0.47 MT CO_2eq/year and 0.77 TCO_2eq/TVS). This is mainly due to the digestate retaining all of the nutrients from the feedstock, as opposed to HTC, where there is nutrient loss. AD of PM generates a GHG mitigation of 0.12 MT CO_2eq/year, compared to 0.14 MT CO_2eq/year with HTC. Conversely, when co-digested with grass, AD of PM had a greater GHG mitigation than HTC (0.40 MT CO_2eq/year with AD, when compared to 0.26 MT CO_2eq/year with HTC). This is due to the addition of grass increasing biomethane yields.

3.2. EU Greenhouse Gas Mitigation Potential

Higher PM quantities result in higher GHG mitigation potentials than the UK, while higher grid carbon intensities had significant impacts on both HTC (Table 4) and AD (Table 5). As with the UK, avoided PM storage and spreading has the greatest GHG mitigation, representing 12.8 MT CO_2eq and 0.56 tonnes of CO_2 eq for every tonne of PM VS generated (same as the UK). Moreover, HTC of PM and peat substitution had a total mitigation of 4.39 MT of equivalent CO_2 and 0.19 tonnes of CO_2 eq tonne of VS when processed at 200 °C for 1 h. This reduced by almost half (47.4%) to 2.29 MT of equivalent CO_2 and 0.10 tonnes of CO_2 eq tonne of VS when increasing the process temperature to 250 °C for 1 h. This reduction was higher for the EU than the UK, due to the EU's higher grid carbon energy intensity and higher GHGs associated with the increased energy input requirements. As with the UK, the addition of grass to the PM results in almost a doubling of GHG mitigation from HTC. Total GHG mitigation increased from 4.39 to 8.12 tonnes of CO_2 eq tonne of VS when processed at 200 °C for 1 h (2.29 to 4.23 tonnes of CO_2 eq tonne of VS) when increasing the process temperature to 250 °C for 1 h. Altogether, the maximum total GHG mitigation for HTC of PM at 200 °C for 1 h, associated with avoided storage corresponds to 17.16 MT of equivalent CO_2, rising to 20.89 MT of equivalent CO_2 with the addition of grass.

Table 4. EU GHG mitigation from HTC of PM and PM-grass blend.

Feedstock and HTC Temperature	Pig Manure 200 °C	Pig Manure 250 °C	Grass and Pig Manure blend 200 °C	Grass and Pig Manure Blend 250 °C
Net HTC GHG mitigation (TCO_2eq/TVS)	0.19	0.10	0.18	0.09
Net GHG mitigation from avoided PM storage and spreading emissions per tonne of pig manure VS (TCO_2eq/TVS)	0.56	0.56	0.56	0.56
Total Net GHG mitigation per tonne of VS PM (TCO_2eq/TVS)	0.76	0.66	0.92	0.75
Manure total per year (million tonnes of VS)	22.67	22.67	22.67	22.67
Total GHG mitigation from HTC (million tonnes of CO_2eq/year)	4.39	2.29	8.12	4.23
Total GHG mitigation from avoided storage (million tonnes of CO_2eq/year)	12.77	12.77	12.77	12.77
Total GHG mitigation/year (million tonnes of CO_2eq/year)	17.16	15.06	20.89	17.00

The overall GHG mitigation potential from AD is substantively higher than HTC for the EU (Table 5). Mono-digestion of PM generated a slightly higher GHG mitigation than HTC of PM of 5.08 MT CO_2eq/year (when compared to 4.39 MT CO_2eq/year with HTC). AD in the EU had a greater GHG mitigation per tonne of VS than the UK, due to an enhanced GHG mitigation from AD with a higher carbon intensity of grid energy. Moreover, when co-digested with grass, AD of PM had a substantively greater GHG mitigation than HTC (16.55 MT CO_2eq/year with AD, when compared to 8.12 MT CO_2eq/year with HTC).

Table 5. EU GHG mitigation from AD of PM and PM-grass blend.

Feedstock and HTC Temperature	Pig Manure	Grass and Pig Manure Blend
Net AD GHG mitigation (TCO_2eq/TVS)	0.22	0.37
Net GHG mitigation from avoided PM storage and spreading emissions per tonne of pig manure VS (TCO_2eq/TVS)	0.71	0.71
Total Net GHG mitigation per tonne of VS PM (TCO_2eq/TVS)	0.93	1.45
Manure total per year (million tonnes of VS)	22.67	22.67
Total GHG mitigation from AD (million tonnes of CO_2eq/year)	5.08	16.55
Total GHG mitigation from avoided storage (million tonnes of CO_2eq/year)	16.10	16.10
Total GHG mitigation/year (million tonnes of CO_2eq/year)	21.18	32.65

3.3. China Greenhouse Gas Mitigation Potential

Increased quantities of PM in China result in higher overall GHG mitigation potential than both the UK and EU. Additionally, substantively higher carbon intensities associated with grid energy and chemical fertilisers have a major impact on both HTC (Table 6) and AD (Table 7). In the HTC scenario, avoided PM storage has the greatest mitigation, totalling 32 MT of equivalent CO_2. On the other hand, net GHG mitigation per tonne of VS associated was slightly lower than the UK and EU (0.44 compared to 0.56 tonnes of CO_2 eq per tonne of PM VS). This was due to the fact that less biofertiliser was generated in the HTC scenario, and the synthetic nitrogen fertiliser was more carbon-intense in China. Moreover, HTC of PM and peat substitution had a mitigation totalling 10.43 MT of equivalent CO_2, but only 0.15 tonnes of CO_2 eq tonne of VS when processed at 200 °C for 1 h. When processing at 250 °C for 1 h, GHG emissions from HTC energy inputs outweighed the GHG mitigation from peat substitution, leading to net emissions of 0.16 MT of equivalent CO_2. The addition of grass to the PM results in an increase in the net GHG mitigation from HTC at 200 °C for 1 h (from 10.43 to 18.29 tonnes of CO_2 eq tonne of VS). On the other hand, co-processing with grass at 250 °C for 1 h results in a net increase in GHG emissions (from -0.16 MT of equivalent CO_2 to -1.67 MT of equivalent CO_2). Altogether, per quantity of feedstock, HTC performed less well in China than the other regions, due to high grid carbon intensity levels, making HTC a carbon intense process, particularly at higher processing temperatures. Overall, there was a maximum total GHG mitigation (from peat substitution and avoied storage), with HTC at 200 °C for 1 h of 42.13 MT of equivalent CO_2, rising to 49.99 MT of equivalent CO_2 when co-processing with grass.

Table 6. Chinese GHG mitigation from HTC of PM and PM-grass blend.

Feedstock and HTC Temperature	Pig Manure 200 °C	Pig Manure 250 °C	Grass and Pig Manure Blend 200 °C	Grass and Pig Manure Blend 250 °C
Net HTC GHG mitigation (TCO_2eq/TVS)	0.15	0.00	0.13	−0.01
Net GHG mitigation from avoided PM storage and spreading emissions per tonne of pig manure VS (TCO_2eq/TVS)	0.44	0.44	0.44	0.44
Total Net GHG mitigation per tonne of VS PM (TCO_2eq/TVS)	0.59	0.44	0.70	0.42
Manure total per year (million tonnes of VS)	71.88	71.88	71.88	71.88
Total GHG mitigation from HTC (million tonnes of CO_2eq/year)	10.43	−0.16	18.29	−1.67
Total GHG mitigation from avoided storage (million tonnes of CO_2eq/year)	31.70	31.70	31.70	31.70
Total GHG mitigation/year (million tonnes of CO_2eq/year)	42.13	31.54	49.99	30.03

Table 7. Chinese GHG mitigation from AD of PM and PM-grass blend.

Feedstock and HTC Temperature	Pig Manure	Grass and Pig Manure Blend
Net AD GHG mitigation (TCO$_2$eq/TVS)	0.33	0.55
Net GHG mitigation from avoided PM storage and spreading emissions per tonne of pig manure VS (TCO$_2$eq/TVS)	0.71	0.71
Total Net GHG mitigation per tonne of VS PM (TCO$_2$eq/TVS)	1.04	1.8
Manure total per year (million tonnes of VS)	449.22	449.22
Total GHG mitigation from AD (million tonnes of CO$_2$eq/year)	24.04	78.36
Total GHG mitigation from avoided storage (million tonnes of CO$_2$eq/year)	50.82	50.82
Total GHG mitigation/year (million tonnes of CO$_2$eq/year)	74.86	129.18

The GHG mitigation potential from AD is higher than HTC for China (Table 7). Avoided storage and spreading emissions were considerably higher, due to the increase in biofertiliser generation (compared to HTC) and the high carbon intensity of chemical fertiliser in China. Moreover, specific GHG mitigation from AD of PM generated more than double the GHG mitigation than the HTC of PM (24.04 MT CO$_2$eq/year, compared to 10.43 MT CO$_2$eq/year). Moreover, when co-digested with grass, AD of PM had a GHG mitigation many times higher than HTC (78.36 MT CO$_2$eq/year with AD, when compared to 18.29 MT CO$_2$eq/year with HTC).

3.4. Comparison of Greenhouse Gas Mitigation Potential in the UK, EU, and China

GHG mitigation from the HTC and AD scenarios showed real differences in the UK, EU, and China (Table 8). Differing carbon intensities of grid energy and fertilisers, as well as the total PM generation levels in the countries had key implications on the findings.

Table 8. Comparing total GHG mitigation from HTC and AD in the UK, EU, and China.

Feedstock and HTC Temperature	Pig Manure HTC 200 °C	Pig Manure HTC 250 °C	Grass and Pig Manure Blend HTC 200 °C	Grass and Pig Manure Blend HTC 250 °C	Pig Manure AD	Pig Manure and Grass Co-Digestion AD
UK Total Net GHG mitigation per tonne of VS PM (TCO$_2$eq/TVS)	0.77	0.70	0.96	0.82	0.90	1.31
EU Total Net GHG mitigation per tonne of VS PM (TCO$_2$eq/TVS)	0.76	0.66	0.92	0.75	0.93	1.45
China Total Net GHG mitigation per tonne of VS PM (TCO$_2$eq/TVS)	0.59	0.44	0.70	0.42	1.04	1.8
UK Total GHG mitigation/year (million tonnes of CO$_2$eq/year)	0.51	0.46	0.63	0.54	0.59	0.86
EU Total GHG mitigation/year (million tonnes of CO$_2$eq/year)	17.16	15.06	20.89	17.00	21.18	32.65
China Total GHG mitigation/year (million tonnes of CO$_2$eq/year)	42.13	31.54	49.99	30.03	74.86	129.18

Considering GHG mitigation per tonne of PM VS, HTC performed the best in the UK, due to lower energy inputs associated with less carbon intense grid energy. On the other hand, AD performed the least well in the UK, due to lower GHG mitigation from

energy generation associated with the low grid energy carbon intensity. Conversely, the high carbon intensity of grid energy and fertilisers in China indicated that it had the lowest GHG mitigation from HTC (due to high energy inputs) and greatest GHG mitigation from AD (due to high mitigation substituting grid energy and fertiliser). HTC at 200 °C had a greater GHG mitigation than at 250 °C in all countries, primarily due to higher energy input requirements.

Regarding the overall GHG mitigation potential, China has a significantly greater PM generation than the EU and UK, leading to substantially higher GHG mitigation opportunities. Overall, China could mitigate 75 MT CO_2eq/year with PM AD, rising to 130 MT CO_2eq/year when co-digesting PM with grass. GHG mitigation levels in the EU were significantly lower at 21 MT CO_2eq/year with PM AD, rising to 33 MT CO_2eq/year when co-digesting PM with grass. This was due to the lower total PM generation levels and a lower level of GHG mitigation from the AD scenarios. Nevertheless, UK GHG mitigation levels are still lower (0.59 MT CO_2eq/year with PM AD, rising to almost 0.86 MT CO_2eq/year when co-digesting PM with grass). This was resultant from the lower PM generation and GHG mitigation per tonne of PM VS when anaerobically digested.

4. Implications of Current Trends and Future Policy

4.1. Current Trends in Pig Manure Generation in the UK, EU, and China

In the UK and EU, domestic pork industries and related PM decreased in 2022, due to a decline in pig numbers (8% and 4% reduction in the UK and EU, respectively) [28,29]. On the other hand, pig numbers and associated PM generation levels are increasing in China [9% in 2022]. These changes indicate a decrease in GHG mitigation opportunities in the UK and EU from the PM treatment, but an increase in opportunities in China.

4.2. Relevant Policies in the UK, EU, and China

Between 2000 and 2020, the carbon intensity of grid energy reduced by half in the UK [30], almost half in the EU [31], and by over a third in China, with reductions set to increase into the future. Moreover, the UK and EU have set a target to reach net zero CO_2 eq emissions by 2050, while China has targeted net zero by 2060 [30,31]. Energy decarbonisation has been identified as a key contributer to this target in all regions, through improving energy efficiency and using cleaner energy sources [30,31]. With this in mind, co-digestion of PM and grass could play a role in decarbonising the energy sector of the different regions. Conversely, the GHG mitigation benefits related to AD in the UK, EU, and China will likely decrease in the future based on current trends, due to the rapid decarbonisation of the energy mix in the respective regions.

The UK and EU are currently trying to phase out peat usage in horticulture, with the UK, Ireland, Germany, and Norway in the EU targeting a 100% reduction by 2030 [32,33]. Therefore, the generation of a peat-substitute product through HTC of PM could greatly assist in the phasing out of peat use, by offering the potential for a plentiful supply of peat-like soil amendment product [9]. Moreover, net GHG mitigation levels from HTC of PM would increase in the future, due to a decarbonisation of grid energy and resultant decrease in the GHGs associated with energy input requirements.

4.3. Contrast among EU Countries

While conclusions can be drawn for the EU as a whole, the EU is a very diverse union of countries, with vastly different pig manure generation rates and energy intensities (Table 9). The country with the highest PM generation (Table 9) is Germany, with over 26 million tonnes per year, almost 5 times the EU average. Moreover, Germany displays a relatively high grid energy carbon intensity of 406 gCO_2/kWh (41% higher than the EU average). Therefore, Table 9 suggests that Germany is highly suitable for the utilisation of PM through AD, which would offset its relatively carbon intense grid energy. Conversely, France has the third highest PM generation in the EU (14.4 million tonnes of PM slurry annually) along with a low grid carbon intensity of 54 gCO_2/kWh (Table 1). Therefore,

France's low carbon energy makes it suitable for processing PM through HTC, in order to generate a soil amendment product to substitute peat usage.

Table 9. PM opportunities in different EU countries.

Country	EU Average	Germany	Spain	France	Poland	Denmark	Sources
Pig manure generation (slurry 1000 tonnes/year)	5500	26,073	25,494	14,362	13,847	11,195	[33]
Carbon intensity of grid energy (gCO$_2$/kWh)	287	406	276	54	789	189	[17]

5. Practical Considerations for the Utilisation of Pig Manure

5.1. Practical Issues of Hydrochar as a Soil Amendment Product

The theoretical GHG mitigation benefits from using HTC of PM to generate a peat-like soil amendment product are clear; however, in reality, HC has very different properties to peat. HTC is known to promote the production of 'humic-like substances' in HC; however, the yields and properties of these materials vary across feedstocks and processing conditions. As a result, the assumption of a like-for-like replacement is not yet proven. HCs have been proven to be beneficial in terms of aiding plant growth through the slow release of nutrients and stimulation of soil microbes. Moreover, HC may increase water retention in soils, aid carbon sequestration [34], and reduce N$_2$O and CH$_4$ emissions during crop production [35]. HC has previously been identified as a suitable material for substituting a proportion of peat in horticultural growth mediums [36,37]. However, these studies suggest that HC should be combined with peat, in order to form a high-quality soil amendment product, and thus, cannot fully replace peat alone [36,37]. Furthermore, HC has been proven to be phytotoxic, and thus, is problematic as a soil amendment product without further treatment [38].

Various solutions have been indentified to address problems pertaining to HC phytotoxicity. The simplest solution is natural ageing, where HC is added to the soil for a minimum of 9 weeks before crops are planted in the HC-amended soil. During this time, all phytotoxic properties were mitigated [38]. HC can also be washed with water or organic solvents (e.g., acetone) to reduce phytotoxicity levels between 79% and 96%, through the reduction in harmful compounds, such as HMF, furfural, catechol, and cresol [39]. Another solution to overcome HC phytotoxicity includes composting with microbally-active compost [40]. Roehrdanz et al. [41] found that HC composted for 12 weeks performed only as well as peat-based gardening products. Composting of HC results in some losses in C and N and a reduction in overall mass. Perhaps the quickest option for reducing the phytotoxicity is the pyrolysis of HC. Heating the HC (200–600 °C) under oxygen-free conditions between 1 and 5 h leads to a removal of the majority of phytotoxic elements [42]. However, the process is very energy intensive [40], and thus reduces the net GHG mitigation potential from the soil amendment production from PM.

5.2. Practical Issues Associated with Anaerobic Digestion of Pig Manure and Grass

While it is clear that there would be a huge potential for GHG mitigation from the AD of PM, in reality, PM is a potentially problematic feedstock for AD. PM contains numerous pathogens, which require sterilisation [43]. Thermophilic AD of PM is preferable due to its capacity for sanitising the PM. On the other hand, thermophilic AD of PM is known to be highly problematic due to the high ammonia content inhibiting biomethane production [44]. Moreover, PM has high sulphur content, leading to the stimulation of bacteria that feed off sulphates, with these bacteria competing against methanogens, reducing biomethane yields, and leading to increasing H$_2$S formation [43]. Furthermore, PM has a below optimal C/N ratio (of around 13:1) [45], when compared to the optimal C/N ratio for AD (of around 30:1) [46]. The low C/N ratio of PM suggests that the digestion of PM alone would not be suitable [44].

Numerous studies have looked at methods to tackle the challenges for AD of PM, with the co-digestion of PM with grass seen as a possible solution. Co-digestion of PM with grass cuttings can assist in reducing ammonia and sulphate generation, as well as reducing the associated methanogen inhibition [46]. Additionally, co-digestion can improve the C/N ratio of the feedstock [47], increasing biomethane yields [46].

In addition to feedstock-related issues, AD as a technology has certain risks associated with its operation. There is potential for methane to be released from AD into the atmosphere through leakages, contributing to GHGs in the form of fugitive emissions, rather than mitigating GHG emissions. Methane is a potent GHG, contributing to climate change at a rate that is 25 times higher than CO_2 per quantity of gas released (British Standards Institution, 2011). Fugitive emissions can be negligible (as assumed in this assessment), and are often assumed to be around 1.5% of the total CH_4 production [48]. However, fugitive emissions can increase to over 15% of the total CH_4 generation [49], depending on the quality of the equipment and operation management. Management approaches, such as feedstock feeding rate, can also influence biogas composition [50]. In this study, it was assumed that the AD process would be optimized, in order that methane would make up 62% of the total biogas content [25]; however, the proportion of methane in biogas can drop below 50% if the AD process is not optimised [50]. Therefore, if the proposed AD system is not run effectively (as assumed in the assessment), then the associated GHG mitigation levels would be substantially lower than calculated.

In addition, ammonia leaching from the digestate applied to land could be a considerable issue associated with AD. Ammonia leaching is an issue, due to the fact that it can cause acidification and eutrophication of sensitive ecosystems. Moreover, particulate NH_3 can negatively impact human health, and ammonia leaching can eventually lead to the release of GHGs through secondary conversion to other particles [51]. Zhang et al. [1] estimated that almost 22 kg of ammonia would be leached from digestate into land for every tonne of PM VS digested.

6. Potential of Hydrothermal Carbonisation and Anaerobic Digestion of Pig Manure and Grass for Greater Decarbonisation

Table 10 shows large variations in the scale of GHG mitigation for using PM in HTC and AD across each region. China displays the greatest total GHG mitigation across all scenarios, although it also has the greatest national GHG emissions. China has the greatest proportion of national GHGs that could be mitigated through AD, with AD of all of China's PM having the potential to mitigate 0.61% of China's total GHG emissions, rising to 1.05% if co-digested with grass. Alternatively, Table 10 shows that the AD of PM and grass could mitigate 0.92% of total EU emissions and 0.19% of GHG emissions in the UK.

Table 10. Total and proportional GHG mitigation from PM and grass HTC, and in the UK, the EU, and China.

	Pig Manure HTC 200 °C	Grass and Pig Manure Blend HTC 200 °C	Pig Manure AD	Pig Manure and Grass Co-Digestion AD	Total Annual GHGs (MT CO_2eq) [52,53]
Percentage of UK GHGs mitigated	0.11	0.14	0.13	0.19	447.9
Percentage of EU GHGs mitigated	0.48	0.59	0.59	0.92	3567
Percentage of Chinese GHGs mitigated	0.34	0.40	0.61	1.05	12,355

When considering GHG mitigation opportunities from HTC, the EU has the greatest potential for GHG mitigation, with the possibility to mitigate 0.48% and 0.59% of regional emissions when utilising PM and PM with grass, respectively. Additionally, HTC of PM and grass could mitigate 0.40% of total Chinese emissions, but only 0.14% of UK emissions.

Altogether, it is clear that overall and proportional GHG mitigation potential from AD and HTC of PM and grass are higher in China and the EU, when compared to the UK. This is due to the fact that they are larger regions, with significantly greater numbers of pigs (and associated PM), as well as significantly larger, established pig farming industries on a regional level, when compared to the UK [5,6].

While there is a lack of literature to quantify the GHG mitigation from AD and HTC of PM that is directly applied to land and unutilised grass, Li et al. [3] identified the enormous GHG mitigations that are associated with methane and nitrous oxide emitted during the storing and spreading of pig manure. In another study, Zhang et al. [1] found that the AD of PM could greatly reduce the atmospheric emissions associated with the direct application of PM to land, and the heat and energy generated could greatly enhance GHG mitigation. Moreover, Zhang et al. [1] found that co-digestion of PM with grass enhanced the GHG mitigation associated with AD. Similar findings were identified by Mehta et al. [54], while investigating the GHG mitigation associated with PM and grass co-digestion. Additionally, Amado et al. [55] found that utilising PM for AD to generate energy and biofertiliser resulted in significant GHG reductions, and proved highly cost-effective. Furthermore, a study by Roy et al. [9] found that treating organic wastes through hydrothermal carbonisation and utilising the HC as a soil amendment product for the substitution of peat could have the potential to bring about significant reductions in GHG emissions.

When comparing the GHG mitigation potentials of various organic treatment strategies, Salemdeeb et al. [56] and Kim et al. [57] found that AD has the greatest GHG mitigation, due to its net energy outputs and biofertiliser generation. Moreover, a study by Davison et al. [13] found that organic waste treatment strategies with energy generation associated had a greater GHG mitigation potential from energy generation in countries with higher grid energy carbon intensity levels, such as China when compared to the EU, as shown in this study with the AD scenario.

While it is clear that AD is the best performing technology, and that the addition of grass enhances the GHG mitigation associated with AD more than HTC, it may be possible to utilise other resources to greatly increase the opportunities associated with HTC. There is a plentiful supply of unutilised lignocellulosic biomass, such as agricultural residues, garden waste, and woody biomass that are typically unsuitable for AD, due to recalcitrant structures that often require pre-treatment [58]. However, these feedstocks could be hydrothermally carbonised to increase the feedstock supply chain availabiltity and associated GHG mitigation [9]. Moreover, co-processing PM with lignocellulosic biomass would increase HC yields and humic content, improving its properties as a soil amendment product [58]. On the other hand, the addition of lignocellulosic material may decrease the biodegradability and biomethane generation potential of the PW [12], and in turn, increase the net energy input required to power the HTC process. Furthermore, it is clear that reducing the process temperature of HTC from 250 to 200 °C substantively improved the GHG mitigation potential, through increased HC yields and reduced net energy requirements. Nevertheless, operating the HTC process at a lower temperature would likely increase the associated GHG mitigation, with a temperature of 180 °C often being used when generating soil amendment products [58]. Moreover, the lower temperature conditions have the added benefit of improving the quality of the HC by reducing the phytotoxicity of the HC [59].

7. Conclusions

This study explored the benefits and limitations associated with HTC and AD of PM, which was currently directly applied to land across the UK, EU, and China. Options were explored for processing PM alone and co-blending with grass cuttings. China had the greatest total GHG mitigation potential and the greatest GHG mitigation potential per quantity of PM treated. AD had a greater GHG mitigation than HTC in all cases. Co-processing with grass enhanced the GHG mitigation associated with both AD and HTC. Altogether, co-digestion of PM and grass could play a role in GHG mitigation in China,

the EU, and to a lesser extent in the UK (around 1% of China's and EU's GHGs could be mitigated, but only 0.2% of UK's GHGs).

The implications of future changes to the energy and horticultural industries in the different regions, and practical implications associated with the proposed AD and HTC scenarios have been examined. In regard to the practical implications of HC replacing peat, this study has identified that the phytotoxicology of the HC would be an issue, and methods, such as washing, composting, or pyrolysis of the HC would likely be required for the HC to be a suitable soil amendment product. In regard to issues associated with PM HTC, it was identified that PM has numerous problematic qualities that negatively affect biomethane production when digested alone. Furthermore, it was identified that co-digestion with grass would mitigate the majority of these issues.

References

1. Zhang, Y.; Jiang, Y.; Wang, S.; Wang, Z.; Liu, Y.; Hu, Z.; Zhan, X. Environmental sustainability assessment of pig manure mono- and co-digestion and dynamic land application of the digestate. *Renew. Sustain. Energy Rev.* **2021**, *137*, 110476. [CrossRef]
2. McLeod, A. *World Livestock 2011-Livestock in Food Security*; Food and Agriculture Organization of the United Nations (FAO): Rome, Italy, 2011.
3. Li, Q.; Gao, M.; Li, J. Carbon emissions inventory of farm size pig husbandry combining Manure-DNDC model and IPCC coefficient methodology. *J. Clean. Prod.* **2021**, *320*, 128854. [CrossRef]
4. Fangueiro, D.; Surgy, S.; Napier, V.; Menaia, J.; Vasconcelos, E.; Coutinho, J. Impact of slurry management strategies on potential leaching of nutrients and pathogens in a sandy soil amended with cattle slurry. *J. Environ. Manag.* **2014**, *146*, 198–205. [CrossRef]
5. DEFRA. Livestock Populations in England at 1 June 2022. Available online: https://www.gov.uk/government/statistics/livestock-populations-in-england/livestock-populations-in-england-at-1-june-2022#:~:text=The%20female%20breeding%20flock%20increased,2.6%25%20to%20almost%2033%20million (accessed on 10 September 2022).
6. Statista Number of Pigs Worldwide in 2022, by Leading Country (in Million Head). Available online: https://www.statista.com/statistics/263964/number-of-pigs-in-selected-countries/#:~:text=Globaloverview,tonsofporkeachyear (accessed on 10 September 2022).
7. Xiong, J.-B.; Pan, Z.-Q.; Xiao, X.-F.; Huang, H.-J.; Lai, F.-Y.; Wang, J.-X.; Chen, S.-W. Study on the hydrothermal carbonization of swine manure: The effect of process parameters on the yield/properties of hydrochar and process water. *J. Anal. Appl. Pyrolysis* **2019**, *144*, 104692. [CrossRef]
8. Brown, A.E.; Ford, J.S.; Bale, C.S.; Camargo-Valero, M.A.; Cheffins, N.J.; Mason, P.E.; Price-Allison, A.M.; Ross, A.B.; Taylor, P.G. An assessment of road-verge grass as a feedstock for farm-fed anaerobic digestion plants. *Biomass Bioenergy* **2020**, *138*, 105570. [CrossRef]
9. Roy, P.; Dutta, A.; Gallant, J. Evaluation of the life cycle of hydrothermally carbonized biomass for energy and horticulture application. *Renew. Sustain. Energy Rev.* **2020**, *132*, 110046. [CrossRef]

10. Ekpo, U.; Ross, A.; Camargo-Valero, M.; Williams, P. A comparison of product yields and inorganic content in process streams following thermal hydrolysis and hydrothermal processing of microalgae, manure and digestate. *Bioresour. Technol.* **2016**, *200*, 951–960. [CrossRef]

11. Heilmann, S.M.; Molde, J.S.; Timler, J.G.; Wood, B.M.; Mikula, A.L.; Vozhdayev, G.V.; Colosky, E.C.; Spokas, K.A.; Valentas, K.J. Phosphorus Reclamation through Hydrothermal Carbonization of Animal Manures. *Environ. Sci. Technol.* **2014**, *48*, 10323–10329. [CrossRef]

12. Paul, S.; Dutta, A. Challenges and opportunities of lignocellulosic biomass for anaerobic digestion. *Resour. Conserv. Recycl.* **2018**, *130*, 164–174. [CrossRef]

13. Davison, N.; Gaxiola, J.B.; Gupta, D.; Garg, A.; Cockerill, T.; Tang, Y.; Yuan, X.; Ross, A. Potential Greenhouse Gas Mitigation for Converting High Moisture Food Waste into Bio-Coal from Hydrothermal Carbonisation in India, Europe and China. *Energies* **2022**, *15*, 1372. [CrossRef]

14. Zhou, J.; Zhang, R.; Liu, F.; Yong, X.; Wu, X.; Zheng, T.; Jiang, M.; Jia, H. Biogas production and microbial community shift through neutral pH control during the anaerobic digestion of pig manure. *Bioresour. Technol.* **2016**, *217*, 44–49. [CrossRef] [PubMed]

15. Dargahi, A.; Mohammadi, M.; Amirian, F.; Karami, A.; Almasi, A. Phenol removal from oil refinery wastewater using anaerobic stabilization pond modeling and process optimization using response surface methodology (RSM). *Desalination Water Treat.* **2017**, *87*, 199–208. [CrossRef]

16. Shokoohi, R.; Rahmani, A.; Asgari, G.; Dargahi, A.; Vaziri, Y.; Abbasi, M.A. Evaluation of Autothermal Thermophilic Aerobic Digester Performance for the Stabilization of Municipal Wastewater Sludge. *Pak. J. Biol. Sci.* **2017**, *20*, 260–266. [CrossRef]

17. Fazlzadeh, M.; Almasi, A.; Dargahi, A.; Amrane, A.; Soltanian, M.; Hashemian, A. Effect of molasses addition as biodegradable material on phenol removal under anaerobic conditions. *Environ. Eng. Manag. J.* **2018**, *17*, 1475–1482. [CrossRef]

18. IEA. Development of CO2 Emission Intensity of Electricity Generation in Selected Countries, 2000–2020. Available online: https://www.iea.org/data-and-statistics/charts/development-of-co2-emission-intensity-of-electricity-generation-in-selected-countries-2000-2020 (accessed on 12 October 2021).

19. EEA. Greenhouse Gas Emission Intensity of Electricity Generation in Europe. Available online: https://www.eea.europa.eu/data-and-maps/indicators/overview-of-the-electricity-production-3/assessment#:~:text=TheGHGemissionintensityoftotalelectricityg enerationinthe,onaverage%2Cinthattime (accessed on 11 October 2021).

20. Jenssen, T.K.; Kongshaug, G. Energy consumption and greenhouse gas emissions in fertiliserproduction. In Proceedings of the IFA Technical Conference, Merrakech, Morocco, 28 September–1 October 1998.

21. Wood, S.; Cowie, A. A review of greenhouse gas emission factors for fertiliser production. In *Technical Report for IEA Bioenergy Task*; IEA: Paris, France, 2004; Volume 38, 20p.

22. Vechi, N.T.; Jensen, N.S.; Scheutz, C. Methane emissions from five Danish pig farms: Mitigation strategies and inventory estimated emissions. *J. Environ. Manag.* **2022**, *317*, 115319. [CrossRef] [PubMed]

23. Dennehy, C.; Lawlor, P.G.; Croize, T.; Jiang, Y.; Morrison, L.; Gardiner, G.E.; Zhan, X. Synergism and effect of high initial volatile fatty acid concentrations during food waste and pig manure anaerobic co-digestion. *Waste Manag.* **2016**, *56*, 173–180. [CrossRef] [PubMed]

24. Borbolla-Gaxiola, J.E.; Ross, A.B.; Dupont, V. Multi-Variate and Multi-Response Analysis of Hydrothermal Carbonisation of Food Waste: Hydrochar Composition and Solid Fuel Characteristics. *Energies* **2022**, *15*, 5342. [CrossRef]

25. Xie, S.; Lawlor, P.G.; Frost, P.; Dennehy, C.D.; Hu, Z.; Zhan, X. A pilot scale study on synergistic effects of co-digestion of pig manure and grass silage. *Int. Biodeterior. Biodegrad.* **2017**, *123*, 244–250. [CrossRef]

26. Liebetrau, J.; Reinelt, T.; Clemens, J.; Hafermann, C.; Friehe, J.; Weiland, P. Analysis of greenhouse gas emissions from 10 biogas plants within the agricultural sector. *Water Sci. Technol.* **2013**, *67*, 1370–1379. [CrossRef] [PubMed]

27. Emissions from Home Energy Use. Available online: https://www.carbonindependent.org/15.html#:~:text=TheCO2 generatedbyburning,kg%2FkWh%5B9%5D (accessed on 14 October 2022).

28. AHDB. Pork Market Outlook. Available online: https://ahdb.org.uk/pork-market-outlook (accessed on 12 October 2022).

29. AHDB. USDA Meat Production Forecasts Remain in Growth. Available online: https://ahdb.org.uk/news/usda-meat-production-forecasts-remain-in-growth (accessed on 12 October 2022).

30. Data, O.W. In Carbon Intensity of Electricity, 2000 to 2021. Available online: https://ourworldindata.org/grapher/carbon-intensity-electricity?tab=chart&country=~GBR (accessed on 13 October 2022).

31. Fragkos, P.; van Soest, H.L.; Schaeffer, R.; Reedman, L.; Köberle, A.C.; Macaluso, N.; Evangelopoulou, S.; De Vita, A.; Sha, F.; Qimin, C.; et al. Energy system transitions and low-carbon pathways in Australia, Brazil, Canada, China, EU-28, India, Indonesia, Japan, Republic of Korea, Russia and the United States. *Energy* **2021**, *216*, 119385. [CrossRef]

32. Hirschlet, O.; Osterburg, B.; Weimar, H.; Glasenapp, S.; Ohmes, M.F. *Peat Replacement in Horticultural Growing Media: Availability of Bio-Based Alternative Materials*; Thünen Working Paper, No. 190; Johann Heinrich von Thünen-Institut: Braunschweig, Germany, 2022.

33. Foged, H.L.; Flotats, X.; Blasi, A.B.; Palatsi, J.; Magri, A.; Schelde, K.M. *Inventory of Manure Processing Activities in Europe*; European Union: Brussels, Belgium, 2012.

34. Chen, D.; Zhou, Y.; Xu, C.; Lu, X.; Liu, Y.; Yu, S.; Feng, Y. Water-washed hydrochar in rice paddy soil reduces N2O and CH4 emissions: A whole growth period investigation. *Environ. Pollut.* **2021**, *274*, 116573. [CrossRef]

35. Álvarez, M.L.; Gascó, G.; Plaza, C.; Paz-Ferreiro, J.; Méndez, A. Hydrochars from Biosolids and Urban Wastes as Substitute Materials for Peat. *Land Degrad. Dev.* **2017**, *28*, 2268–2276. [CrossRef]

36. Farru, G.; Dang, C.H.; Schultze, M.; Kern, J.; Cappai, G.; Libra, J.A. Benefits and Limitations of Using Hydrochars from Organic Residues as Replacement for Peat on Growing Media. *Horticulturae* **2022**, *8*, 325. [CrossRef]

37. Celletti, S.; Bergamo, A.; Benedetti, V.; Pecchi, M.; Patuzzi, F.; Basso, D.; Baratieri, M.; Cesco, S.; Mimmo, T. Phytotoxicity of hydrochars obtained by hydrothermal carbonization of manure-based digestate. *J. Environ. Manag.* **2021**, *280*, 111635. [CrossRef] [PubMed]

38. Melo, T.M.; Bottlinger, M.; Schulz, E.; Leandro, W.M.; Filho, A.M.D.A.; Wang, H.; Ok, Y.S.; Rinklebe, J. Plant and soil responses to hydrothermally converted sewage sludge (sewchar). *Chemosphere* **2018**, *206*, 338–348. [CrossRef]

39. Al-Wabel, M.I.; Rafique, M.I.; Ahmad, M.; Ahmad, M.; Hussain, A.; Usman, A.R. Pyrolytic and hydrothermal carbonization of date palm leaflets: Characteristics and ecotoxicological effects on seed germination of lettuce. *Saudi J. Biol. Sci.* **2019**, *26*, 665–672. [CrossRef] [PubMed]

40. Busch, D.; Stark, A.; Kammann, C.I.; Glaser, B. Genotoxic and phytotoxic risk assessment of fresh and treated hydrochar from hydrothermal carbonization compared to biochar from pyrolysis. *Ecotoxicol. Environ. Saf.* **2013**, *97*, 59–66. [CrossRef]

41. Roehrdanz, M.; Greve, T.; de Jager, M.; Buchwald, R.; Wark, M. Co-composted hydrochar substrates as growing media for horticultural crops. *Sci. Hortic.* **2019**, *252*, 96–103. [CrossRef]

42. Hitzl, M.; Mendez, A.; Owsianiak, M.; Renz, M. Making hydrochar suitable for agricultural soil: A thermal treatment to remove organic phytotoxic compounds. *J. Environ. Chem. Eng.* **2018**, *6*, 7029–7034. [CrossRef]

43. Hansen, K.H.; Angelidaki, I.; Ahring, B.K. Improving thermophilic anaerobic digestion of swine manure. *Water Res.* **1999**, *33*, 1805–1810. [CrossRef]

44. González-Fernández, C.; García-Encina, P.A. Impact of substrate to inoculum ratio in anaerobic digestion of swine slurry. *Biomass Bioenergy* **2009**, *33*, 1065–1069. [CrossRef]

45. Wu, S.; Shen, Z.; Yang, C.; Zhou, Y.; Li, X.; Zeng, G.; Ai, S.; He, H. Effects of C/N ratio and bulking agent on speciation of Zn and Cu and enzymatic activity during pig manure composting. *Int. Biodeterior. Biodegrad.* **2017**, *119*, 429–436. [CrossRef]

46. Wijesinghe, D.T.N.; Dassanayake, K.B.; Sommer, S.G.; Scales, P.; Chen, D. Biogas Improvement by Adding Australian Zeolite During the Anaerobic Digestion of C:N Ratio Adjusted Swine Manure. *Waste Biomass Valorization* **2019**, *10*, 1883–1887. [CrossRef]

47. Sievers, D.M.; Brune, D.E. Carbon/Nitrogen Ratio and Anaerobic Digestion of Swine Waste. *Trans. ASAE* **1978**, *21*, 0537–0541. [CrossRef]

48. Vo, T.T.; Rajendran, K.; Murphy, J.D. Can power to methane systems be sustainable and can they improve the carbon intensity of renewable methane when used to upgrade biogas produced from grass and slurry? *Appl. Energy* **2018**, *228*, 1046–1056. [CrossRef]

49. Flesch, T.K.; Desjardins, R.L.; Worth, D. Fugitive methane emissions from an agricultural biodigester. *Biomass Bioenergy* **2011**, *35*, 3927–3935. [CrossRef]

50. Babaee, A.; Shayegan, J. Effect of Organic Loading Rates (OLR) on Production of Methane from Anaerobic Digestion of Vegetables Waste. In Proceedings of the World Renewable Energy Congress, Linköping, Sweden, 8–13 May 2011; pp. 411–417. [CrossRef]

51. Webb, J.; Broomfield, M.; Jones, S.; Donovan, B. Ammonia and odour emissions from UK pig farms and nitrogen leaching from outdoor pig production. A review. *Sci. Total Environ.* **2014**, *470–471*, 865–875. [CrossRef]

52. BEIS. 2020 UK Greenhouse Gas Emissions, Final Figures. Available online: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/1051408/2020-final-greenhouse-gas-emissions-statistical-release.pdf (accessed on 14 October 2021).

53. Bank, W. Total Greenhouse Gas Emissions (kt of CO2 Equivalent)—China, India, European Union. 2021. Available online: https://data.worldbank.org/indicator/EN.ATM.GHGT.KT.CE?end=2018&locations=CN-IN-EU&start=1970&view=chart (accessed on 14 October 2022).

54. Mehta, N.; Anderson, A.; Johnston, C.R.; Rooney, D.W. Evaluating the opportunity for utilising anaerobic digestion and pyrolysis of livestock manure and grass silage to decarbonise gas infrastructure: A Northern Ireland case study. *Renew. Energy* **2022**, *196*, 343–357. [CrossRef]

55. Amado, M.; Carrasco, J.; Ochoa, L.D.; Rangel, C.J.; Becerra, A.P.; Cabeza, I.O.; Acevedo, P.A. Technical and Environmental Analysis of Large-scale Pig Manure Digestion Through Process Simulation and Life Cycle Assessment. *Chem. Eng. Trans.* **2021**, *87*, 439–444. [CrossRef]

56. Salemdeeb, R.; zu Ermgassen, E.K.; Kim, M.H.; Balmford, A.; Al-Tabbaa, A. Environmental and health impacts of using food waste as animal feed: A comparative analysis of food waste management options. *J. Clean. Prod.* **2017**, *140*, 871–880. [CrossRef]

57. Kim, M.-H.; Song, Y.-E.; Song, H.-B.; Kim, J.-W.; Hwang, S.-J. Evaluation of food waste disposal options by LCC analysis from the perspective of global warming: Jungnang case, South Korea. *Waste Manag.* **2011**, *31*, 2112–2120. [CrossRef] [PubMed]

58. Parmar, K.R.; Ross, A.B. Integration of Hydrothermal Carbonisation with Anaerobic Digestion; Opportunities for Valorisation of Digestate. *Energies* **2019**, *12*, 1586. [CrossRef]

59. Fiori, L.; Basso, D.; Castello, D.; Baratieri, M. Hydrothermal carbonization of biomass: Design of a batch reactor and preliminary experimental results. *Chem. Eng. Trans.* **2014**, *37*, 55–60. [CrossRef]