Supplemental Information for “Regional Economic Aspects of Carbon Markets and Anaerobic Digesters in the United States: The Case of Swine Production”

January 26, 2024

This document accompanies the manuscript *Regional Economic Aspects of Carbon Markets and Anaerobic Digesters in the United States: The Case of Swine Production* and contains additional information and data regarding the analysis. Note that in the main manuscript as well as in this document, the carbon price is in $ per metric ton of CO_2-equivalent.

Contents

1 Environmental Effects and GHG Emissions from Manure

Due to its nutrient content, almost all manure is ultimately applied to cropland for fertilization. The long-term storage of liquid manure without applying it to land contributes significantly to GHG emissions. Hog producers in the Southeast use lagoons and apply manure with an irrigation system to cropland. This leads to a higher volatilization of N_2O, but more manure can be applied to cropland. GHG emissions from manure application to cropland also depend on the amount and timing (Aguirre-Villegas and Larson, 2017).

The application of manure to cropland has complex environmental implications. Varma et al. (2021) provide an extensive review of the various aspects of manure management systems. Nutrient management plans match the nutrient requirement of crops to the nutrients discharged, which leads to lower pollution. Since 2011, the EPA requires Concentrated Animal Feeding Operations (CAFO) to have a nutrient management plan under the Clean Air Act. There are also differences in the nutrient uptake of crops depending on the regions. For example, the amount of nitrogen excreted in North Carolina may be the same as in Iowa but the uptake per acre may be lower because crop yields in North Carolina are lower.

There is a large literature assessing the emissions from manure management systems with results depending on whether manure is treated as waste or analyzed from a life cycle perspective in which it is a co-product from livestock production that can substitute for mineral fertilizer (Corbalá-Robles et al., 2018; Leip et al., 2019). An integrated crop and livestock system allows for the production of manure and subsequent application to cropland without large transportation distances to haul the manure to the fields. Through consolidation of hog operations in recent years, the number of hogs is concentrated on a smaller amount of land (i.e., the inventory per farm has increased), which has led to various hurdles regarding the integration of crop and livestock production (Key et al., 2011).

Contractual agreements allow hog producers to concentrate on one phase of hog production, which lead to economies of scale and an increase in the animal inventory per farm (Key et al., 2011). There are two important aspects given this development. First, an increase in inventory per farm leads to a decline in cropland per animal unit, i.e., there is potentially not enough cropland to spread the manure (USDA ERS, 2023). Second, because there is an increase in the number of animals per farm, the surrounding cropland may not be sufficient to produce the feed required. However, higher feed productivity may reduce both, the required cropland for feedstock production and manure spreading.

The application of manure to cropland may also depend on the price of fertilizer and crops. Some farmers in the Southeast plant Bermuda Grass because it has a higher nitrogen uptake than other crops (Key et al., 2011). But an increase in commercial fertilizer prices reduces the amount of Bermuda grass planted. In Iowa, fertilizer price increases have also been met with increased use of manure (Pudenz and Schulz, 2022). Other side effects of hog manure management are ammonia emissions. There is also a trade-off between water and air pollution. Livestock operators may store manure in uncovered lagoons to reduce the nutrient load to meet water quality standards but at the same time, they are increasing air pollution (Key et al., 2011).

Wu et al. (2013) consider land application and gasification as the two manure management pathways and find a large difference in net GHG emissions from land application (119 kg CO_2-

e) and gasification (-643 kg CO_2-e). Manure land application avoids (life-cycle) emissions from mineral fertilizer, and gasification produces not only syngas as an energy source but also biochar, which can be used as a soil amendment. Biochar can have beneficial effects in terms of soil water retention and yield improvements but displays variations in quality (Wu et al., 2013; Kauffman et al., 2014; Dokoohaki et al., 2019; Dumortier et al., 2020). Aguirre-Villegas et al. (2014) and Aguirre-Villegas and Larson (2017) assess the life-cycle GHG emissions from solid/liquid separation, anaerobic digestion, and a combination of the two systems and compare it to the base-case (i.e., manure collection, storage, and subsequent land application) for dairy farms in Wisconsin. They conclude that the anaerobic digestion system reduces GHG emissions by 48% compared to the base-case. Theory and practice suggest that once the anaerobic digester is constructed, various biogas end uses are possible, which result in a decrease in GHG emissions.

Digestate is a by-product of ADs. Focusing on small dairy farms (100–250 cows) in the U.S., Klavon et al. (2013) find that the revenue (measured in \$ cow^{-1} $year^{-1}$) is highest for using digested solids for bedding (\$100), followed by heat, and/or electricity production (\$47–\$70). Using it as an organic fertilizer also reduces synthetic fertilizer cost and emissions.

2 Carbon Pricing in the U.S. and European Context

The manuscript assesses the effects of carbon payments and policy on the profitability of anaerobic digesters and biogas production for swine producers in the United States. Currently, carbon policies vary significantly across the world and within the United States. According to data collected by the World Bank, there are 70 carbon pricing initiatives implemented with a large variation in carbon prices ranging from a few cents to over \$120 (see World Bank Carbon Pricing Dashboard). Four carbon markets currently exist in the U.S. comprising fourteen states (California, Oregon, Washington, and the Regional Greenhouse Gas Initiative (RGGI) comprising eleven states in the Northeast), the largest of which, California, collects CH_4, in part, from dairy. Key and Sneeringer (2011) estimate that dairy and hog operations could offset about 62% of their GHG emissions at a carbon price of \$15. ICF (2013) estimates that the break-even carbon price to switch from an anaerobic lagoon to an AD in the Appalachia region is \$1–\$15 depending on the digester design for a 2,500 head swine farm. The Appalachia region contains North Carolina, as the second largest swine-producing state after Iowa. According to USDA (2019), 93.4% of U.S. hogs and pigs are on farms with a herd size of more than 2,000. Those values suggest a relatively low break-even carbon price to trigger investment. Given alternative investment opportunities, a modestly positive net present value and return may not be sufficient for investment.

The situation is different in Europe where ADs are broadly used, with approximately 20,000 biogas plants in 2020 (Key and Sneeringer, 2011; IEA, 2020; EBA, 2021). Whiting and Azapagic (2014) report that European countries with a high number of ADs also benefit from financial incentives in the sale of electricity ranging from EUR 0.085 to EUR 0.25 per kWh. Higher electricity prices increase the adoption of ADs for two reasons. First, it becomes profitable to substitute on-farm generated electricity for grid electricity. Second, the surplus electricity generated can be sold back to the utility at a higher price if net-metering is in place. More importantly, long-term contracts of 20 years are in place that reduce the revenue uncertainty associated with ADs. The

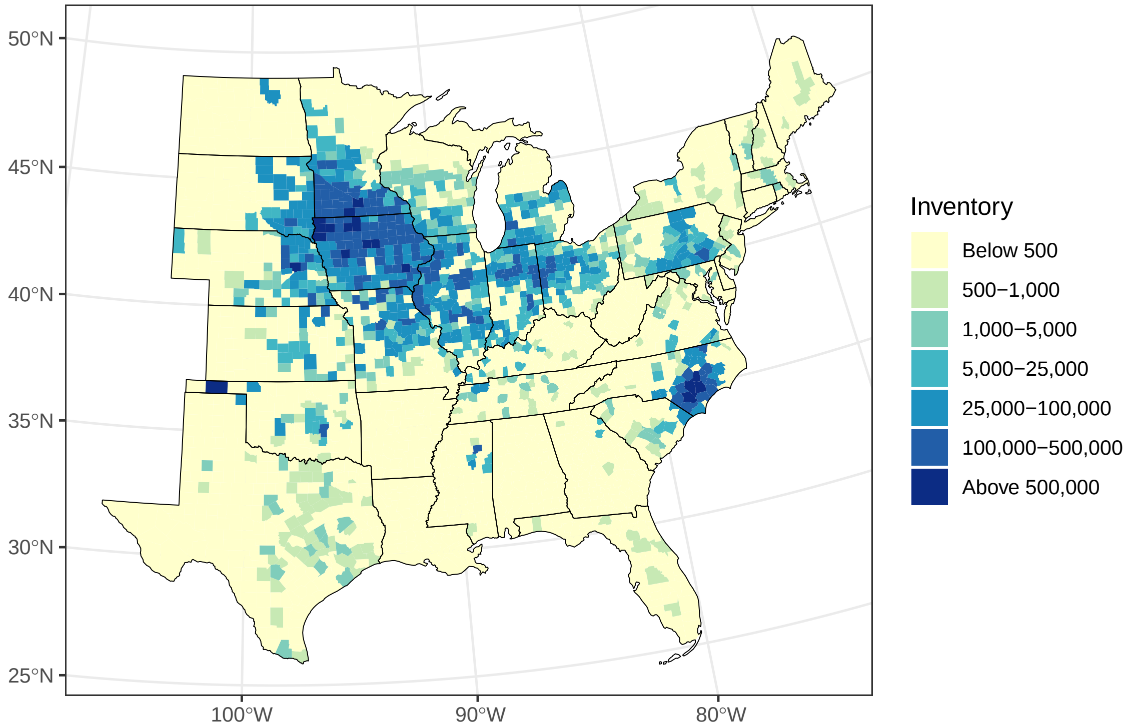

Figure 1. Distribution of the number of hogs per county according to the 2017 Census of the U.S. Department of Agriculture (USDA).

United Kingdom has seen an increase in the number of ADs following the implementation of feed-in tariffs (i.e., long-term contracts to purchase electricity above market price) in 2010 (Whiting and Azapagic, 2014). In Germany, farmers receiving subsidies are not allowed to use the produced electricity on-farm and must feed all of it into the power grid. Although farmers in Europe do not benefit directly from carbon prices, there is an indirect effect of higher electricity prices through the carbon price prevailing in the EU Emission Trading System. Since electricity and natural gas prices will likely remain decoupled from carbon payments in the U.S., more options in terms of biogas use and government support are possible.

3 Hog and Temperature Distribution

The viability of anaerobic digesters (AD) depends on the ambient temperature. Figures 1 and 2 depict the county level distribution of hogs and temperatures in the United States.

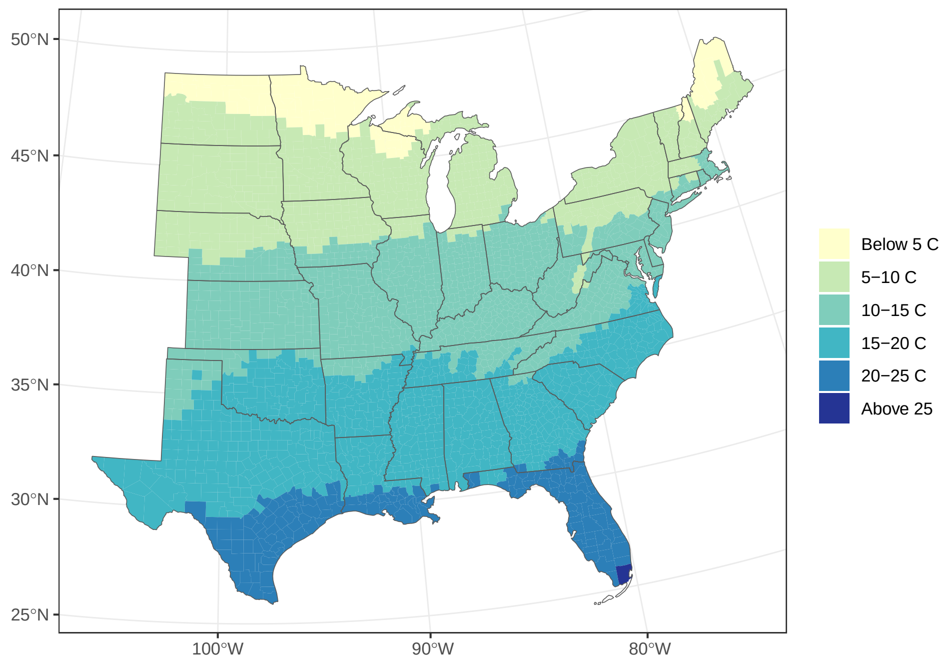

Figure 2. Mean county temperature (2012–2021) based on daily PRISM climate data.

4 Number of Farms by Inventory

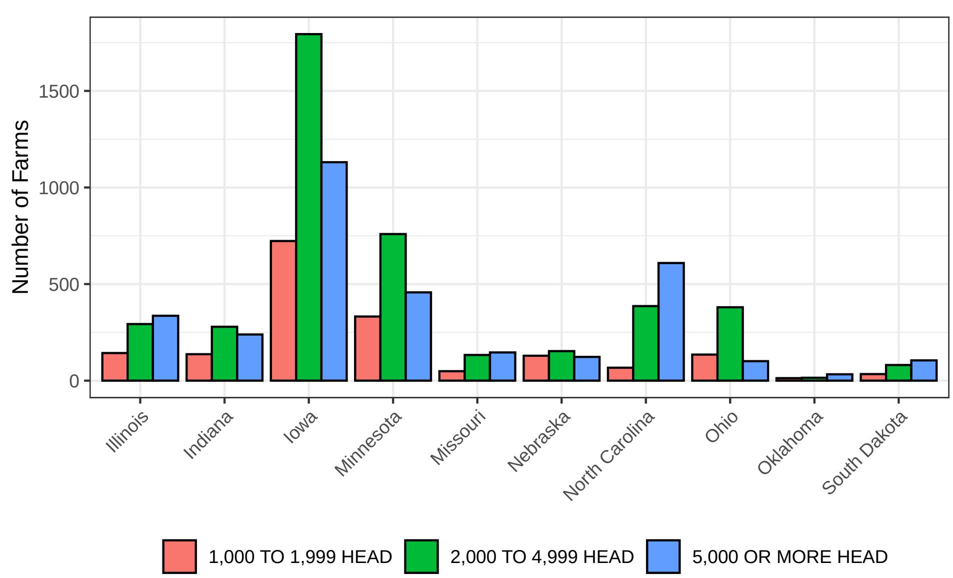

Figure 3. Number of farms by inventory size based on 2017 U.S. Department of Agriculture Census.

State	Co-digestate	Slurry/Liquid	Anaerobic lagoon	Deep pit	Pit (<1 month)
Illinois	Corn stover	15%	7%	71%	5%
Indiana	Corn stover	3%	12%	78%	7%
Iowa	Corn stover	10%	4%	80%	5%
Minnesota	Corn stover	3%	2%	88%	5%
Missouri	Corn stover	16%	33%	34%	15%
Nebraska	Corn stover	9%	22%	49%	19%
North Carolina	Corn stover	33%	49%	1%	16%
Ohio	Corn stover	10%	9%	67%	13%
Oklahoma	Wheat straw	11%	53%	3%	32%
South Dakota	Corn stover	17%	11%	57%	14%

Table 1. Manure management system use according to EPA (2022) in the ten largest U.S. states in terms of swine inventory. The table also contains the most common co-digestate based on the state's crop production.

5 Emission Calculations

In this analysis, we cover methane (CH_4) emissions from manure management and assess the effect of installing an anaerobic digester (AD). Since the focus of the analysis is on farmers' incentives to construct ADs in the U.S., we do not cover CH_4 emissions from enteric fermentation and nitrous oxide (N_2O) emissions from manure management. The calculations closely follow EPA (2022). The Global Warming Potential (GWP) used are 28 and 265 for methane and nitrous oxide, respectively. The swine inventory from USDA NASS is subdivided into five categories (1) breeding, (2) market under 50 pounds (under 23 kg), (3) market 50–119 pounds (23–54 kg), (4) market 120–179 pounds (54–81 kg), and (5) market 180 pounds and over (greater than 81 kg).

5.1 Manure Management System Use in the U.S.

Table 1 lists the manure management system distribution of the ten largest swine producing states. The table also includes the co-digestate for the state-specific analysis in the main manuscript.

5.2 Methane Emissions from Manure Management

The U.S. EPA specifies four relevant manure management systems (MMS) for swine: (1) liquid/slurry, (2) anaerobic lagoon, (3) deep pit, and (4) deep pit (less than 1 month). The methane conversion factors (MCF) for the four MMSs considered are taken from (CARB, 2014) and were approximated using functional forms to reduce the mean squared error over the temperature range 10–28°C. The MCF remains constant at 0.03 for manure being stored for less than a month in a deep pit. For liquid/slurry (SLU), anaerobic lagoon (ANL), and deep pit (PIT), the MCFs are as follows: $MCF_{SLU} = 0.1576 \cdot \exp(0.0884 \cdot t)$, $MCF_{ANL} = 0.6493 + 0.0517 \cdot \ln(t)$, and $MCF_{PIT} = 0.1614 \cdot \exp(0.0866 \cdot t)$ where $t = T - 9$ and T representing the average temperature.

Methane emissions by animal type (i) and MMS (j) can be calculated as follows (IPCC, 2019; EPA, 2022):

$$E_{MMS} = \sum_{i,j} \left(N_{i,j} \cdot DVS_{i,j} \cdot B_0 \cdot MCF_j(t) \cdot 0.662 \cdot 365.25 \right) \tag{1}$$

where $N_{i,j}$ is the number of animals of type i in MMS j. $DVS_{i,j}$ represents the daily production of volatile solids (Table 2). For the swine population and the manure systems considered, the methane generation potential (B_0) is constant at 0.48 m^3 CH_4 kg^{-1} VS added (EPA, 2022). The value of 0.662 is the CH_4 density at 25°C. Given farm inventory, farm type (i.e., breeding or market swine), manure management system, and temperature, E_{MMS} represents the baseline emissions for a swine farm in the absence of an AD.

5.3 Methane Emissions from Anaerobic Digestion

The emissions calculations from an AD are based on EPA (2022). In a first step, the methane production from a particular AD system is calculated:

$$P = \sum_{i} \left(N_i \cdot DVS_{i,j} \cdot B_0 \cdot 0.662 \cdot 365.25 \cdot 0.9 \right) \tag{2}$$

where 0.9 is the methane production factor for AD systems. The other components are similar to Equation (1) except that the subscript for various MMS is dropped. Based on the methane production, the emissions for AD type k are calculated as follows:

$$E_{AD} = P \cdot CE_k \cdot (1 - DE_k) + P \cdot (1 - CE_k) \tag{3}$$

where the destruction efficiency (DE) is set to 0.98 for all digester types. The collection efficiency (CE) is 0.75 for covered lagoons and 0.99 for complete mix and plug flow digesters. Given farm inventory, farm type (i.e., breeding or market swine), E_{AD} represents the baseline emissions for a swine farm in the presence of an AD.

Animal Type	TAM (kg)	VS (kg)	Daily VS (kg)	CH_4 Yield (m^3 kg^{-1} VS)
Gestating Sow	200	2.30	0.460	275
Lactating Sow	192	5.40	1.037	356
Boar	200	1.70	0.340	356
Market (under 23 kg)	13	8.80	0.114	356
Market, (23–54 kg)	39	5.40	0.211	356
Market, (54–81 kg)	68	5.40	0.367	356
Market, (over 81 kg)	91	5.40	0.491	356

Table 2. Waste characteristics for swine: TAM represents the total animal mass in kilogram. VS and Daily VS represent the daily volatile solids produced per 1000 kg of animal mass and per animal, respectively.

6 Biogas Use Cost Calculations

The cost calculations in this section are based on EPA (2016) but were converted from Standard Cubic Feet per Minute (SCFM) and British thermal units (BTU) in 2016 USD to GJ in 2021 USD. Conversion to 2021 USD was made using the GDP deflator. For the production of pipeline-quality natural gas, the cost were adjusted downward because EPA (2016) account for higher cost in California by using a multiplier. The analysis includes increases in efficiency for large biogas plants. This efficiency increase manifests itself at very high biogas flows, which are outside the range of our analysis. Thus, the linear functional forms used in the main part of the paper appropriately represent the cost and product flow based on biogas input flow (Table 3).

Biogas Flow (m^3 year^{-1})	Capital Cost ($)	O&M Cost ($)	Output (GJ year^{-1})
Microturbine (MTU)			
194,970	231,677	14,354	1,004
392,918	374,936	22,353	2,115
1,093,918	984,906	51,728	6,655
1,339,492	1,083,396	58,273	8,149
1,739,851	1,343,053	75,184	10,992
Reciprocating Engine (REC)			
595,330	459,996	43,656	3,343
833,461	651,381	58,329	5,071
1,012,060	797,997	68,914	6,157
1,146,009	903,203	76,534	7,240
1,488,324	1,186,364	97,013	9,751
1,934,821	1,566,895	126,680	13,582
2,678,983	2,137,693	169,167	19,433
3,571,978	2,730,875	216,201	26,747
4,316,140	3,256,904	269,406	33,329
6,250,961	4,577,572	366,073	51,195
7,888,117	5,573,670	475,148	66,449
11,311,262	7,688,979	578,484	97,933
Compressed Natural Gas (CNG)			
744,162	1,421,398	98,491	7,314
1,488,324	1,813,122	153,332	14,627
2,976,648	2,451,072	247,346	29,254
Renewbale Natural Gas (RNG)			
1,082,012	3,077,172	249,962	14,584
3,952,989	6,268,898	699,717	53,280
9,249,934	11,252,678	1,341,416	124,674

Table 3. Biogas input and product output flow as well as capital and operating & maintenance (O&M) cost associated with various biogas uses based on EPA (2016).

7 CNG Stations and Natural Gas Pipelines

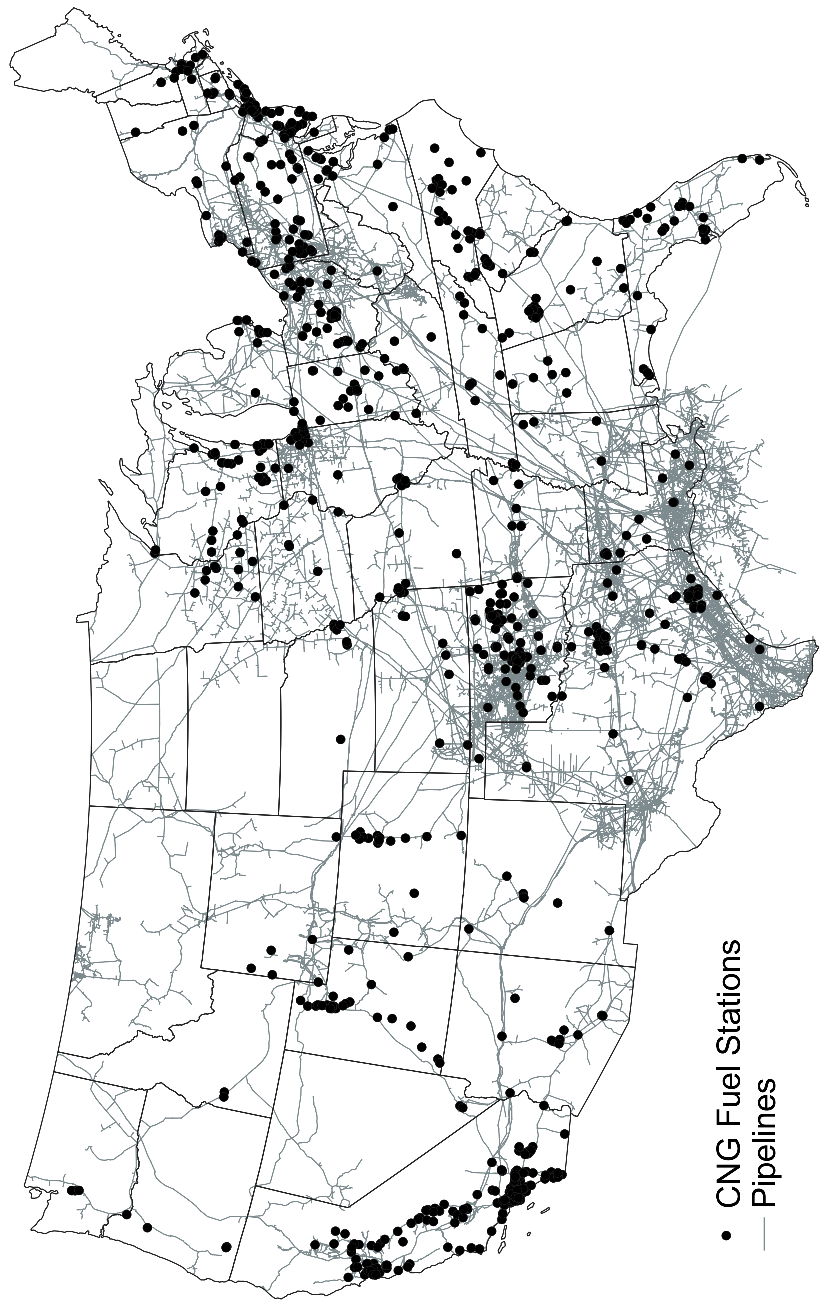

Figure 4. Compressed natural gas fueling stations and natural gas pipeline network in the United States. Source: AFDC Natural Gas Fueling Station Locations and U.S. Energy Atlas Natural Gas Interstate and Intrastate Pipelines.

References

Aguirre-Villegas, H. A. and Larson, R. A. (2017). Evaluating greenhouse gas emissions from dairy manure management practices using survey data and lifecycle tools. *Journal of Cleaner Prododuction*, 143:169–179. DOI: 10.1016/j.jclepro.2016.12.133.

Aguirre-Villegas, H. A., Larson, R. A., and Reinemann, D. J. (2014). From waste-to-worth: energy, emissions, and nutrient implications of manure processing pathways. *Biofuels, Bioproducts and Biorefining*, 8(6):770–793. DOI: 10.1002/bbb.1496.

CARB (2014). *Compliance offset protocol livestock projects: Capturing and destroying methane from manure management systems*. California Environmental Protection Agency - Air Resources Board.

Corbalá-Robles, L., Sastafiana, W. N., van Linden, V., Volcke, E. I., and Schaubroeck, T. (2018). Life cycle assessment of biological pig manure treatment versus direct land application: a trade-off story. *Resources, Conservation and Recycling*, 131:86–98. DOI: 10.1016/j.resconrec.2017.12.010.

Dokoohaki, H., Miguez, F. E., Laird, D., and Dumortier, J. (2019). Where should we apply biochar? *Environmental Research Letters*, 14(044005):1–9. 10.1088/1748-9326/aafcf0.

Dumortier, J., Dokoohaki, H., Elobeid, A., Hayes, D. J., Laird, D., and Miguez, F. E. (2020). Global land-use and carbon emission implications from biochar application to cropland in the United States. *Journal of Cleaner Production*, 258(120684):1–10. DOI: 10.1016/j.jclepro.2020.120684.

EBA (2021). Annual report 2020. Annual report, European Biogas Association.

EPA (2016). Evaluating the air quality, climate & economic impacts of biogas management technologies. Technical Report EPA/600/R-16/099, U.S. Environmental Protection Agency.

EPA (2022). Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2020. Report EPA 430-R-22-003, United States Environmental Protection Agency.

ICF (2013). Greenhouse gas mitigation options and costs for agricultural land and animal production within the United States. Report prepared for U.S. Department of Agriculture, ICF International.

IEA (2020). Outlook for biogas and biomethane: Prospects for organic growth. World energy outlook special report, International Energy Agency.

IPCC (2019). *2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories*, chapter Emissions from Livestock and Manure Management, pages 10.1–10.171. Intergovernmental Panel on Climate Change.

Kauffman, N., Dumortier, J., Hayes, D. J., Brown, R. C., and Laird, D. A. (2014). Producing energy while sequestering carbon? the relationship between biochar and agricultural productivity. *Biomass and Bioenergy*, 63:167–176. DOI: 10.1016/j.biombioe.2014.01.049.

Key, N., McBride, W. D., Ribaudo, M., and Sneeringer, S. (2011). Trends and Developments in Hog Manure Management: 1998-2009. Economic Research Report 81, U.S. Department of Agriculture Economic Research Service.

Key, N. and Sneeringer, S. (2011). Climate change policy and the adoption of methane digesters on livestock operations. Economic Research Report 111, U.S. Department of Agriculture Economic Research Service.

Klavon, K. H., Lansing, S. A., Mulbry, W., Moss, A. R., and Felton, G. (2013). Economic analysis of small-scale agricultural digesters in the United States. *Biomass & Bioenergy*, 54:36–45. DOI: 10.1016/j.biombioe.2013.03.009.

Leip, A., Ledgard, S., Uwizeye, A., Palhares, J. C., Aller, M. F., Amon, B., Binder, M., Cordovil, C. M., De Camillis, C., Dong, H., Fusi, A., Helin, J., Hörtenhuber, S., Hristov, A. N., Koelsch, R., Liu, C., Masso, C., Nkongolo, N. V., Patra, A. K., Redding, M. R., Rufino, M. C., Sakrabani, R., Thoma, G., Vertès, F., and Wang, Y. (2019). The value of manure - manure as co-product in life cycle assessment. *Journal of Environmental Management*, 241:293–304. DOI: 10.1016/j.jenvman.2019.03.059.

Pudenz, C. C. and Schulz, L. L. (2022). Challenges and opportunities for a manure market in Iowa. Agricultural policy review, Center for Agricultural and Rural Development, Iowa State University.

USDA (2019). 2017 Census of Agriculture - United States Summary and State Data. Geographic Area Series AC-17-A-51, U.S. Department of Agriculture.

USDA ERS (2023). Increasing the value of animal manure for farmers. Administrative Publication 109, U.S. Department of Agriculture (USDA) Economic Research Service (ERS.

Varma, V. S., Parajuli, R., Scott, E., Canter, T., Lim, T. T., Popp, J., and Thoma, G. (2021). Dairy and swine manure management: Challenges and perspectives for sustainable treatment technology. *Science of The Total Environment*, 778(146319):1–18. DOI: 10.1016/j.scitotenv.2021.146319.

Whiting, A. and Azapagic, A. (2014). Life cycle environmental impacts of generating electricity and heat from biogas produced by anaerobic digestion. *Energy*, 70:181–193. DOI: 10.1016/j.energy.2014.03.103.

Wu, H., Hanna, M. A., and Jones, D. D. (2013). Life cycle assessment of greenhouse gas emissions of feedlot manure management practices: Land application versus gasification. *Biomass and Bioenergy*, 54:260–266. DOI: 10.1016/j.biombioe.2013.04.011.