Evaluation of Manure Management Systems (MMS) and Cost-Share Programs for Mitigating Livestock Environmental Impacts

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ACADEMIC ABSTRACT

Animal agriculture is a climate-exposed industry, creating the need to implement climate-smart manure management practices to alleviate manure's pollutive potential through greenhouse gas emissions (GHGe) and excess nutrient management. Manure management systems (MMS) require high initial investment for implementation and are often incentivized by cost-share programs that assume part of the implementation cost. The objective of Chapter 3 was to quantitatively summarize the literature on GHGe and manure nutrient composition in response to MMS use. Included studies provided data on the following: system size (L), species type, days stored, manure type (i.e. whole slurry, digestate, fractions, etc.), MMS type (anaerobic digestion, solid-liquid separation, covered and uncovered storage, composting systems, or mixed MMS use), and manure emissions and nutrient composition pre- and -post MMS use. The data were used to derive emissions coefficients and explore pollution reduction correlations between MMS with different pollution targets. A key takeaway is that there is inadequate data for MMS efficacy when used across species and location. This limits the accuracy of predictions made with the derived coefficients and limits the accuracy of which cost-share programs can be designed to achieve pollution reductions. Chapter 2 had the objective of exploring the strengths and weaknesses of cost-share programs. The complimentary analysis of historical cost-share data coupled with a stated-preference survey identified prioritization of MMS targeting GHGe as the most efficient and effective use of cost-share funding.

The analysis also revealed that producer willingness to pay (WTP) has not changed over the decade, but that cost-share program structure should be equipped to account for higher pollution reduction prices while promoting MMS longevity. The pursuit of sustainability relies on the continuation of cost-share programs and MMS that focus on all facets of pollution reduction.

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GENERAL AUDIENCE ABSTRACT

Livestock contribute to climate change by emitting greenhouse gas emissions (GHGe) and excess nutrients from their manure, impacting air and water quality. Manure management systems (MMS) further process manure for alternative uses while minimizing environmental harm. These systems are expensive and often require incentives for farmers to integrate them into their operations. Cost-share programs provide incentive by sharing a portion of MMS implementation cost. To ensure these programs utilize funding efficiently, agencies must be informed on the efficacy of each MMS and understand the influence different species manure and location can have on its performance. In response, our work aimed to identify the strengths and gaps within MMS literature and determine MMS ability to reduce pollution. Furthermore, we explored opportunities to improve existing cost-share programs, gauged the ability of MMS to maintain co-benefits between air and water quality reductions, and used current farmer perspectives to inform cost-share program structure. Towards that goal, a literature search of MMS efficacy was conducted, and studies were included if they provided data regarding MMS type (anaerobic digester, solid-liquid separation, composting, covered/uncovered storage, and mixed-system use), system size (L), species type, manure type (i.e. manure consistency and state), days stored, and the emissions or nutrient composition of manure before and after MMS use. A key finding from this work was that there is not enough data on MMS as they are used across species and location to

accurately inform cost-share programs. Additionally, an analysis of historical cost-share data from the Department of Conservation and Recreation (DCR) and a survey gauging producer attitude about cost-share programs was also conducted. This work highlighted that MMS targeting GHGe or both pollution types should be implemented, and incentivizing those MMS may require more funding than before. To continue to combat climate change, current and accurate estimates of MMS capabilities are needed, and cost-share programs need to reorient focus towards implementing GHGe focused MMS or those addressing all areas of manure pollution.

DEDICATIONS

I would like to dedicate this work to my husband, family, and friends who are family for their support over this past year in helping me to accomplish so much in such a short amount of time. Without Jake sacrificing our ability to be close and choosing to support my goals, my family constantly encouraging me, and my friends being there along the way I would not have been able to successfully complete this degree.

I would also like to use this dedication to point to God and the mercy and grace He has given to me in pursuit of this degree so that I may better serve His people in my career.

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ABBREVIATIONS

WP-4: Animal waste control Facilities
WP-4B: Dairy loafing lot management system
WP-4C: Composter facilities
WP-4E: Animal waste structure pumping equipment
WP-4F: Animal mortality incinerator
WP-4FP: Feeding pad
WP-4LC:Animal waste control facility for confined livestock operations
WP-4LL: Loafing lot management system with manure management (excluding bovine dairy)
WP-4SF: Seasonal feeding facility with attached manure storage
IFSM: Integrated Farm System Management
DCR: Department of Conservation and Recreational
USDA: United States Department of Agriculture
NRCS: Natural Resources Conservation Service
EPA: Environmental Protection Agency
GHGe: Greenhouse Gas Emissions
MMS: Manure Management System
WTP: Willingness to Pay
MN: Minnesota
VA: Virginia
EQIP: Environmental Quality Incentives Program
AgSTAR: Agricultural Conservation Technologies for Reducing Pollution
BMP: Best Management Practices
REAP: Rural Energy for America Program
VACS: Virginia Agricultural Best Management Practices Cost Share
CH_4: Methane
N_2O: Nitrous oxide
CO_2: Carbon dioxide
Non-CO_2: Non-Carbon CO_2
CO_2-eq: Carbon dioxide equivalents
N: Nitrogen
P: Phosphorous
SLS: Solid-liquid separation
AD: Anaerobic digester
COMP: Composting
USL: Uncovered storage: liquid
USS: Uncovered storage: solid
CSL: Covered storage: liquid
CSS: Covered storage: solid
M: Mixed MMS use
NEPA: National Environmental Policy Act
CWA: Clean Water Act
IPCC: Intergovernmental Panel on Climate Change
UN: United Nations
UNFCCC: The United Nations framework Convention on Climate Change

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GENERAL INTRODUCTION

Agriculture's contribution to climate change makes it instrumental in the task of reducing greenhouse gas emissions (GHGe) (Key, 2023). This role presents the opportunity to implement climate-smart technologies to further improve the industry's sustainable impact. Given the evidence of an exponentially growing population (Morse, 2023), the need for technologies that promote a robust global food supply (Sadik, 1991) is justified by the environment's role in food production. Effectively supporting the future food supply is dependent on the implementation of agricultural practices that better steward the environment and available resources (FAO, 2017). In 2021, agriculture accounted for 10.6% of total domestic GHGe (Sands, 2023). This contribution of GHG was part of the 6.9% increase in GHGe from 1990 to 2021 (Sands, 2023). To significantly decrease global warming impacts to 50-52% below the 2005 levels by 2030 (USDE, 2022), climate-smart technologies must be adopted with a focus towards improving the industry's efficiency (FAO, 2017) through the consolidation of inputs and mitigation of pollutive emissions (IPCC, 2023).

Identifying areas of relative contribution to environmental change within the agriculture industry is key to implementing adequate solutions. Agricultural emissions are determined from crop and livestock production both within and external to the farm gate (FAO, 2020). The GHGe from each of these components factor into agriculture's total emissions (FAO, 2020). Compared with CO_2 emissions, greater emphasis has been placed on non-carbon (non-CO_2) emissions due to their concentrated warming potentials (EPA, 2012). Non-CO_2 emissions from agriculture are the result of methane (CH_4) and nitrous oxide (N_2O) emitted through management techniques and natural processes (EPA, 2012).

As of 2018, crop and livestock activities within the farm gate generated approximately 5.3 Gt CO_2 equivalent emissions globally (FAO, 2020). Livestock management is associated with CH_4 emissions, due to the process of enteric fermentation from ruminants and manure management practices (Sands, 2023). In a special report, the IPCC highlighted that through improved livestock management, utilizing available technologies, there was a 0.2 to 2.4 Gt CO_2-eq mitigation potential annually (Mbow et al., 2019). This sets the trajectory of animal agriculture to focus on strategies that promote GHG neutrality and target the reduction of non-CO_2 emissions from agriculture. Greenhouse gas neutrality is defined as a state where the rate of emissions entering the atmosphere equals the rate of emissions removed from the atmosphere (Liu, Proudman, & Mitloehner, 2021). When GHG neutrality is successful there is no additional atmospheric warming and when technologies are leveraged for greater removal, atmospheric cooling is experienced (Liu, Proudman, & Mitloehner, 2021). If livestock CH_4 emissions can be efficiently managed to promote GHG neutrality, there is a greater probability of hitting GHG mitigation potentials and experiencing annual atmospheric cooling (i.e., abating climate change).

An avenue to address CH_4 emissions from livestock is through manure management. Historically, manure has been seen and used as a natural resource due to its high nutrient contents and fertilizer potential (EPA, 2015). Technologies today still acknowledge manure's favorable characteristics as a fertilizer, however the more recent innovations in manure management focus on GHG mitigation and the possibility of biogas production. Engaging manure as a renewable energy source has influenced the industry to consider how manure management technologies can better support CH_4

consolidation and recovery (EPA, 2023). Ambrose et al.'s study highlights manure storage techniques that can improve CH_4 recovery, including floating lagoon covers, natural crusts, solid-liquid separation, frequent slurry removal, aeration, and pasteurization (Ambrose et al., 2023). When paired with processing technologies that harness biogas, manure management can be dually purposed to better steward the environment and produce energy. However, cumulative technology use has created convolution as to how much GHG mitigation can be attributed to each respective technology.

This convolution presents an economic barrier to properly subsidizing these technologies for producers based on the degree of GHG mitigation they are achieving. For example, anaerobic digesters average a capital investment of 1.5 million US dollars (Klavon et al., 2013), consolidating financial viability to large scale producers (EPA, 2018). To experience the true mitigation potential of anaerobic digesters and other capital-intensive technologies, there needs to be a large-scale adoption across livestock producers. Programs like the Agricultural Conservation Technologies for Reducing Pollution (AgSTAR) from the Environmental Protection Agency (EPA) (EPA, 2018) and Rural Energy for American Program (REAP) by the United States Department of Agriculture (USDA) (Sam, Bi, & Farnsworth, 2017) have been developed to help aid in making the initial investment of anaerobic digesters affordable to producers, but these programs are not designed to incentivize the greatest level of GHG reduction. Since GHG reduction is directly influenced by the manure management technologies and strategies employed, the degree to which existing programs have supported GHG reductions is largely unknown.

The literature provides evidence that differences in mitigation potentials are observed in diverse manure management systems. Periera et al.'s study examined the changes in CH_4 reduction when implementing co-digestion strategies of three different additives, which resulted in the further processed solid and liquid fraction treatments exhibiting 45% less CH_4 emissions than the unprocessed anaerobic digestate (Periera et al., 2022). Similarly, Hansen et al. looked at the effect that covering stored manure had on GHG emissions prior to further digestion or processing. It was found that covering stored manure reduced cumulated CH_4 emissions by 88% (Hansen et al., 2006). The literature also discusses the economic barriers and revenue opportunities of different types of anaerobic digesters (Klavon et al., 2013). Cost-share programs are commonly identified as an opportunity to overcome these economic challenges. However, even with cost sharing, no positive cash flows are guaranteed from manure management systems adoption (Klavon et al., 2013). Given these practical challenges, the overarching goal of this work is to explore the existing manure management systems (MMS) research to more effectively estimate the associated emissions reduction from the applied technologies, and to explore how incentive structures influence economic viability and environmental effectiveness of manure management system adoption. To address the goal of summarizing existing MMS research, we conducted a literature search on MMS and their GHG emissions reduction. For studies to be included they must meet the following inclusion criteria and provide: system size (L), species manure, days stored, and manure type (i.e. whole slurry, digestate, fractions, etc.) for MMS categorized as anaerobic digestion, solid-liquid separation, covered and uncovered storage, composting systems, or mixed MMS use. Furthermore, the studies will need to include measurements for dry matter (DM), organic matter (OM), nitrogen (N), phosphorus (P), carbon dioxide (CO_2), CH_4, and N_2O for influent and effluent loads respectively. This data will be used to determine emissions coefficients attributed to the MMS used and if it was influenced by other characteristics of the operation. The data will also be used to identify synergies or existing relationships between MMS.

To address the goal of evaluating suitability of incentive structures, we analyzed a dataset of instances where MMS was cost-shared across the past decade from the Department of Conservation and Recreation (DCR). This analysis was coupled with a stated-preference survey targeting livestock producers with the goal of using their attitudes and experiences with cost-share programs to determine a program structure that aligns with producer willingness to pay (WTP).

Our studies found that programs and policies supporting climate-smart initiatives are ill-informed by the existing understanding of emissions reductions associated with MMS use. Currently, policies push water-quality improvements as their main environmental agenda, however our research suggests changing focus to emissions reductions is a more worthwhile use of resources. This will require more research delving into how MMS efficacy is influenced by operational characteristics of farms and the restructuring of cost-share programs in a way that supports GHGe reductions.

CHAPTER 1: Literature Review

Jillian B. Hammond and Robin R. White

1.0 Sustainability and Climate-smart Agriculture

The Intergovernmental Panel on Climate Change (IPCC) defines climate change as the movement away from average climate conditions over time in response to natural and anthropogenic processes that reduce air and water quality (IPCC, 2022). The United Nations framework Convention on Climate Change (UNFCCC) frames climate change more specifically as the deviation away from the average climatic conditions directly or indirectly related to human activities (IPCC, 2022). This focus on human impact on the environment formed the concern that the current state of society and industries do not promote the longevity of the natural environment. Due to the integral role the natural environment plays in food production and recreational well-being, policies like the National Environmental Policy Act (NEPA) of 1969 (EPA, 2024e) and the Clean Water Act (CWA) of 1972 (EPA, 2024d) were passed to focus environmental preservation and conservation as national priorities. The NEPA held Federal agencies accountable for evaluating proposed policies' impact on the environment. This included keeping record of potentially harmful impacts the policy could have on the environment and human well-being. The NEPA was the first form of legislation that brought environmental conservation to the forefront of political decision-making (CEQ, n.d.). Similarly, the Clean Water Act promoted environmental transparency by establishing regulations of waterway pollutants through the implementation of wastewater standards and permit programs (EPA, 2024d). However, the current idea of sustainability did not surface until 1987 with the Brundtland Report after the Second World War. This report offered the perspective that climate change and other “wicked problems”, being problems that are complex, dynamic, and interdependent on other societal issues (UNEP, 2017), are the common future of our world (Brundtland, 1987). Brundtland (1987) highlighted how the problems society faces are interlinked and will require a global attitude of sustainability to remedy.

Sustainability can be defined by the IPCC as ensuring the persistence of natural and human systems (IPCC, 2022), or by the United Nations (UN) as the ability to meet the needs of the current population without compromising the needs of future generations (EPA, 2024e). The growing global population, projected to reach 9.7 billion in 2050 (UN, n.d.), influences the ability to achieve sustainability targets. Meeting sustainability targets requires evaluating industries and identifying where they either meet or neglect opportunities to adopt sustainable practices. The act of identifying areas of environmental damage within an industry or system and adapting them to hold a climate-conscious perspective is known as climate-smart (World Bank Group, 2024).

In response to the increase in extreme weather events and changing climate conditions that make the production of food more challenging (Nardone et al., 2010), climate-smart agriculture has emerged as a viable solution. In light of the changes required of agriculture, climate-smart agriculture also presents opportunities to influence progress in economic development and poverty reduction in rural areas (Steenwerth et al., 2014). The USDA defines climate-smart agriculture as the increase or maintenance of productivity and yield to enhance resilience to environmental changes and reduce GHGe (USDA, n.d.-a). The USDA is specific to define climate-smart agriculture as not just the introduction of new practices but the adaptation of existing practices. One way this is accomplished is through improving the productivity of existing systems. Agricultural productivity measures the efficiency of system inputs used to produce the desired outputs, and its efficacy is measured by total factor productivity which aims to increase the amount of outputs produced without increasing inputs (USDA, n.d.-c).

2.0 Climate-exposed Industries

Climate change holds consequences for all industries, with the height of economic damage felt in climate-exposed industries like animal agriculture (IPCC, 2023). Climate-exposed industries are those whose longevities are reliant on the natural environment (Smith & Gregory, 2013) and are subject to the adverse effects of climate change (Shaw et al., 2021). Financial security in climate-exposed industries is largely dependent on stochastic variables, and climate change adds another layer of variability and risk (Thai et al., 2023). Animal agriculture is a climate-exposed industry and is significant because of its influence on the global food supply, contributing 40% of agriculturally valued outputs worldwide (World Bank Group, 2021). Efficient animal agriculture production depends on providing animals with optimal environmental conditions (Johnson et al., 2018).

2.1 Heat Stress

Common production animals, like beef and dairy cattle, poultry, and swine can maintain ideal productivity in temperatures between 4.4 to 37.8 degrees Celsius (Johnson et al., 2018). Climate change, specifically atmospheric warming, makes it more common for animals to experience temperatures that surpass the threshold for optimal production. When animals are exposed to high temperatures for extended periods, they experience heat stress. This is consequential because during heat stress animals prioritize thermoregulation and release the stress hormone cortisol (Bett et al., 2017). Collectively, this redirection of physiological efforts neglects productive processes like lactation, reproduction, and growth (Baumgard et al., 2012; Thatcher et al., 2018). Furthermore, heat stress also suppresses immunological responses that can increase herd susceptibility to disease (Bett et al., 2017; Stott et al., 1976). In all livestock, especially dairy and beef cattle, heat stress can lead to mastitis and alter the efficacy of colostrum's passive immunity characteristics, negatively impacting newborn viability (Rowlinson et al., 2008; Thatcher et al., 2018). Collectively, heat stress causes whole-system impairments in key physiological activities essential to maintaining and improving productivity of animal agriculture. Therefore, increased incidences of heat stress caused by climate change will negatively impact productivity and opportunities to work toward sustainability goals.

2.2 Disease

Atmospheric warming also influences disease prevalence and patterns in livestock (Torremorell & Leman, 2010). Increased surface temperatures have resulted in extreme weather events occurring more often and outside of their typical season. De La Rocque et al. (2008) reports that changes in temperatures will impact the geographical range of diseases and the change in seasonality of weather patterns will influence the intensity of disease outbreaks. Moderate climates usually equipped with colder seasons to manage disease transmission become more vulnerable to disease outbreaks (Rojas-Downing et al., 2017). Disease vectors are dependent on ambient temperatures, with warmer temperatures providing a conducive environment for metabolism and pathogen replication (Baumgard et al., 2012). As vectors move into new regions, they are also more likely to diversify (Bett et al., 2017). Given these data, climate change presents a threat to the productivity of livestock production with the increased risk of prevalent diseases (De La Rocque et al., 2008).

2.3 Feed Availability

Feed shortages and availability are also challenges presented by climate change. As the climate changes, farmers are faced with new environmental norms that make the production of consistent crop yields challenging (Becker, 2008). Since forages and cereal grains are strained to produce the quantities needed to supply both human food and animal feed, the livestock industry is likely to experience heightened vulnerability (Hidosa & Guyo, 2017). Feed has historically been one of the largest production expenses in raising livestock, making up about 70% of all costs (Becker, 2008; Makkar, 2018). This cost is expected to go up as there is more demand for feed products and more risk assumed in crop production (Wheeler & Reynolds, 2013). Due to increased competition for an increasingly limited supply, livestock producers might have to choose between feed quality and quantity (Hidosa & Guyo, 2017; Makkar, 2018). To continue to receive the quality feed needed for optimal production, producers will have to assume a higher feed cost for a more largely demanded product. In response, Wheeler and Reynolds (2013) predicted that animal production is likely to switch to less digestible feeds that are not as competitive with crops desirable for human consumption. This switch will impair productivity of animals and increase emissions from enteric fermentation and manure, creating challenges in meeting sustainability goals.

3.0 Animal Agriculture Environmental Impact

3.1 GHGe Contribution

Animal agriculture is a known contributor of greenhouse gas emissions (GHGe) responsible for 14.5% of global anthropogenic emissions (FAO, 2017). Of those emissions, 37% and 65% are CH_4 and N_2O (Steinfeld & Wassenaar, 2007) that hold warming potentials 27 and 273 times more potent than CO_2 respectively (IPCC, 2024; Liu & Liu, 2018; Protocol', 2024). Within animal agriculture, GHGe can originate from natural or anthropogenic sources. Natural sources are those stemming from animal’s biogenic and physiological processes (Takahashi, 2006). In ruminant animals, part of their contribution of CH_4 emissions stem from enteric fermentation. This digestive process produces CH_4 as a by-product, which is released during eructation or the regurgitation of ruminant cud (Buccioni et al., 2015).

Anthropogenic sources of GHGe are those influenced directly or indirectly by people (USGS, 2015). The production of livestock to fulfill food demands and the manure produced from those livestock are indirect extensions of human influence. Manure CH_4 emissions come from manure decomposing anaerobically, and N_2O emissions are generated by nitrification-denitrification processes within manure (Jun et al., 1996a). Unlike enteric fermentation and other natural sources of GHGe, anthropogenic sources of GHGe are more accessible avenues for implementing mitigation strategies.

3.2 Water Quality Contribution and Nutrient Pollution

In addition to the challenges linking livestock to GHGe, livestock also present a threat to the integrity of aquatic ecosystems. Manure is a nutrient-rich resource, commonly carrying high concentrations of N and P. High concentrations of N and P in manure stem from diet surpluses and from natural physiological inefficiencies (Powell et al., 2010). When these nutrients enter waterways in surplus, it can lead to heightened algal growth that depletes the ecosystem of oxygen and harms much of the wildlife (NOAA, n.d.). A common source of nutrients entering these ecosystems is through the leaching of manure after it has been spread on cropland as fertilizer (EPA, 2015). Excess nutrients can also volatilize into the atmosphere and later enter aquatic ecosystems through rainfall (Tarawali & de Haan, 2010). Since the CWA was passed, government agencies and policies have supported water quality improvements, specifically through best management practices that minimize the amount of livestock contact with waterways and prevent excess nutrients from entering waterways.

4.0 Manure Management

4.1 Manure Management Systems (MMS)

Manure management systems allow for the use of manure while minimizing its pollutive potential. Manure management is the collective set of processes in which manure is captured, stored, treated, and used (USDA, 2023). In the United States alone, 1.4 billion tons of manure are produced annually (Pagliari et al., 2020). The global goal of achieving net-zero emissions by 2050 raises GHG neutrality as a primary focus for climate-adaptation strategies like the adoption of MMS. Subsequently, MMS act as GHGe sinks. This means that more GHGe get absorbed by MMS than emitted into the atmosphere (EPA, 2004, 2024c), which aids in the potential of experiencing atmospheric cooling. In 2022, manure management emitted 64.7 MMT CO_2-eq as CH_4 and 17.0 CO_2-eq as N_2O (EPA, 2024c). The IPCC predicts a 0.2-2.4 Gt CO_2-eq annual mitigation potential through improved management of livestock and by extension livestock manure (Mbow et al., 2019). Furthermore, Szogi et al. (2015) calls for the implementation of technologies accounting for the farm losses in manure N and P. Proper MMS use provides an avenue to address the two major pollutants stemming from animal agriculture and aid in achieving global sustainability goals.

4.2 Manure Management System Tradeoffs

Although MMS provide avenues to address both nutrient and GHG associated pollution from manure, current systems favor the reduction of one or the other. This prioritization creates a tradeoff, where the untargeted form of pollution is more likely to be increased (Niles et al., 2022). Despite the literature supporting the potential for synergies (Lorimor et al., 2006; Qu et al., 2025), MMS are currently not employed based on their ability to address all facets of pollution. For example, Chadwick (2005) demonstrated how pollution prioritization has one-sided benefits as the study showed that covering stored manure substantially reduced GHGe but had little impact on the amount of N lost. In a different instance, fugitive emissions or unintended emissions, can stem from all types of MMS but especially those that prioritize reductions in nutrient pollution (Flesch et al., 2011). Due to MMS tradeoffs, there is a need for improved estimates of nutrient and GHG emissions to understand how MMS can be employed most effectively across farm sizes and animal species to support concurrent emissions reductions (Wightman & Woodbury, 2016).

5.0 Manure Management System Types

To explore the potential for MMS to simultaneously address the goals of water quality improvement and GHG emissions reductions, it is useful to review the function and known environmental outcomes associated with MMS used in today's agricultural industry.

5.1 Anaerobic Digestion

Originally anaerobic digestion was used as a means to reduce the associated odor with livestock waste (Velsen, 1981). However, its coupled ability to produce renewable energy through the breakdown of pollutive organic material (i.e., manure) has made it a viable MMS. Anaerobically based processing is the most promising strategy for biogas production as it is energy generative rather than consumptive (Awasthi et al., 2019) and can reduce livestock CH_4 emissions by 85% annually (AGSTAR, n.d.). The biochemical process driving anaerobic digestion employs anaerobic bacteria to breakdown organic matter and produce biogas through methanogenesis (Meegoda et al., 2018; Yao et al., 2020). Before achieving the final phase of methanogenesis, organic material has three prior phases of degradation, those being: hydrolysis, acidogenesis, and acetogenesis. These processes create the substrates and environment needed to catalyze methanogenesis (Venkiteshwaran et al., 2015). The methanogenesis process is performed by archaea bacteria that are sensitive to pH and pressure fluctuations (Ziganshin et al., 2013). Likewise, loading the digester with manure influent from different species can influence the microbial community present. This sensitivity makes the selection of digester size and influent manure important for optimal biogas production (Venkiteshwaran et al., 2015).

Continued study has allowed agricultural producers to better implement anaerobic digesters into their operations accompanied by digester additives and manure storage strategies that reduce their GHGe. Anaerobic digesters are commonly used on dairy and swine operations due to the liquid slurry consistency of their manures (EPA, 2024a). Although digesters can be used for all waste types, pre-processing steps may be required to ensure the manure is ready for digestion. A barrier to wide-scale adoption of anaerobic digesters is the cost of implementation. The average investment to implement a digester is roughly 1.5 million USD (Klavon et al., 2013). However, their ability to reduce emissions by 14.8 MMT of CO_2-eq (EPA, 2024a) could outweigh the costs of implementation especially with the incentive of cost-share programs. It is important to note that anaerobic digestion produces little N_2O emissions as the focus is on CH_4 production for biogas generation and the nitrification-denitrification process requires an aerobic environment (Jun et al., 1996a). Although much is known about anaerobic digestion, some apparent gaps in the literature include: anaerobic digestion's efficacy across multi-specie use and its impact on the consolidation of manure nutrients.

5.2 Solid-Liquid Separation

Solid-liquid separation (SLS) or mechanical separation is a process where manure is split into its solid and liquid fractions further condensing manure nutrients (Perazzolo et al., 2015). Separation technologies can take varying forms with the most common being: centrifugation, sedimentation, screening, and pressurized or non-pressurized filtration (Hjorth et al., 2011). The basis of these technologies is to separate the solid and liquid fractions of manure based on density or particle size (EPA, 2024a). Variation in these technologies makes them affordable and easily employable (Rico et al., 2012). Solid-liquid separation is often used as a pre- or post-treatment process to be coupled with other MMS or another variation of SLS (Cocolo et al., 2012).

Zhang et al. (2022) notes that SLS shows both increases and reductions in emissions depending on its use and coupled use with other MMS. For example, an increase in N_2O during solid fraction storage was observed because N had been consolidated into that fraction and exposed to oxygen spurring the nitrification-denitrification process. However, decreases in N_2O and CH_4 from the liquid fractions of manure have been observed due to storage in anaerobic environments or the reduction of organic matter from manure. Regarding nutrient pollution, SLS improves plant uptake of nutrients by priming manure into fertilizer and reducing the amount of nutrients leached from manure spreading (Zhang et al., 2022). The adaptability of SLS to be used with other MMS are promising, however more data standardizing those interactions are needed for predictive accuracy.

5.3 Covered and Uncovered Storage

Manure storage systems are the most common form of manure management due to their encompassing definition. Manure storage systems can be short or long term, covered or uncovered, and used for any manure type (EPA, 2024a). Like SLS, storage systems can and are often used alongside other MMS. Whether a storage system is covered or not can play a large role in its mitigative potential. Uncovered storage systems expose manure to weather and oxygen which can contribute to the leaching of nutrients or the nitrification-denitrification process emitting N_2O (Harter et al., 2014). Covered storage facilities provide manure protection from the weather and help trap GHG from escaping into the atmosphere (Work, 2011). Forms of coverings stored manure can range from manure crusts to man-made lagoons and pit covers (Samer, 2015). Manure consistency also influences the structure and pollution reduction abilities of the storage facility and is demonstrated in liquid manure typically requiring more extensive infrastructure than solid manure. Liquid manure is naturally an anaerobic environment that would limit N_2O production but produce CH_4, while solid manure presents a greater threat as a nutrient pollution source (Petersen et al., 2013).

The attributed pollution reduction for storage depends on its coverage status and partitions used to separate manure from the ground. For example, Chadwick (2005) measured up to 36% in total GHGe reductions from covering solid stored manure (Petersen et al., 2013). When used in conjunction with other MMS like SLS (Fangueiro et al., 2008) or anaerobic digestion (Clemens et al., 2006), reductions in GHGe were also notable. Storage facilities can also minimize nutrient losses from manure as they keep manure from being leached into water sources and volatilized into the atmosphere (Ali et al., 2019). The variation in environmental effects that stored manure can have, calls for the standardization of efficacy across its coupled use with other MMS and identifying if species or location of storage influences its efficacy.

5.4 Composting Systems

Composting is the controlled biological decomposition of organic matter. Two main types of composting are static or in-vessel and windrow-based composting. Both processes require manure to be piled so that microorganisms can decompose the existing carbon and oxygen and establish an internal temperature of 131 to 170 degrees Fahrenheit. The difference between these two processes is static composting requires only 3 days for decomposition while windrow composting requires 15 days. Due to the length of time required for windrow composting, the piles must be turned or aerated to maintain aerobic conditions within the pile (Chen et al., 2011; Hao et al., 2001).

Greenhouse gas emissions associated with composting stem from the inability to aerate the pile. If the pile is not aerated, it becomes anaerobic and facilitates the production of CH_4 (Fukumoto et al., 2003). Composting also has the potential to produce N_2O, as aeration of piled manure exposes manure N to O facilitating the nitrification-denitrification process (Hao et al., 2001). A general trend observed is that as compost piles increase in size, so too does their relative emissions (Fukumoto et al., 2003). Abd El Kader et al. (2007) also notes that species of manure can influence emissions as the composition can be affected. The variability of pile size, manure species, and composting style paired with limited composting GHG inventories, makes predicting attributed GHG difficult (Sommer & Møller, 2000).

5.5 Mixed MMS Use

There are several instances throughout the literature where MMS are used together to produce greater GHGe reductions than sole MMS use. Pereira et al. (2022) examines the effects of additives on the ability of solid-liquid separation efficiency and co-digestion to mitigate GHGe. This study found a 45% decrease in CH_4 emissions compared to manure that had not undergone further processing. Clemens et al. (2006) observed fewer fugitive emissions when stored digestive slurry was stored with an impermeable cover. Likewise, Hansen et al. (2006) observed an 88% reduction in cumulative CH_4 emissions when covering slurry prior to being fed as influent to an anerobic digester. There is plenty of evidence that mixed MMS use is an efficient way to minimize emissions from manure; however, more accurate emissions coefficients for each individual MMS are needed for robust predictions.

6.0 Manure Management System Accessibility and Adoption

6.1 Barriers to Adoption

Widescale adoption of MMS throughout the U.S. is hindered by various barriers to entry. The most notable of these barriers being high implementation costs. The idea of climate-smart practices allows for the integration of MMS into existing operations; however, the integration of MMS is an added expense on top of an operation's normal operating expenses. To the average farmer the assumption of another cost is not just a barrier to implementation but also a barrier to motivating climate-smart changes in general. The size of the system needed is dependent on the size of the operation, amount of manure needing managed, and the goals of the operation. Prices can also vary drastically, ranging from $2 billion to implement an anaerobic digester (Cowley & Brorsen, 2018) to $30,000 to implement a basic storage system (UM, 2003). This investment of capital along with annual operating expenses of the MMS itself, keep producers from implementing with MMS (Klavon et al., 2013). Animal agriculture will only be able to adopt climate-smart practices to the capacity desired when the financial barrier is alleviated. There is a need to explore how reducing implementation costs, but also maintenance costs will increase the adoption and longevity of MMS use.

Another barrier to entry less quantifiable is farmer knowledge of manure management practices (Teenstra et al., 2014). Without a basic understanding of manure's impact on the environment, there is limited reason to engage with manure management on the basis of environmental conservation. This makes Extension education an important factor in manure management adoption, especially as national policies continue to push for environmental conservation in agriculture (Levins et al., 1996).

6.2 Cost-share Programs

Government agencies have worked to create cost-share programs that alleviate initial capital investments. These programs are structured around sharing a percent of the cost needed to implement an MMS. Typically, these programs follow an application process that ensures the operation receiving funding meets their desired characteristics. Existing examples of these programs are the Environmental Quality Incentives Program (EQIP) through the USDA (USDA, n.d.-b), Virginia Agricultural BMP Cost-Share Program (VACS) through the DCR (DCR, n.d.), and the Agricultural Conservation Technologies for Reducing Pollution (AgSTAR) program through the EPA (EPA, n.d.). Historically these programs have been focused on the implementation of MMS instead of MMS efficacy after implementation, and have been justified based specifically on the need for water quality improvements. Weersink et al. (2001) highlights that implementation alone will not ensure MMS maintenance unless there are additional benefits to MMS use (i.e. fertilizer use, energy generation, etc.). Similarly, Houston and Sun (1999) name efficient abatement of MMS as impossible without measurement or monitoring of MMS function. The current climate trajectory requires full efficacy of MMS for both air and water quality improvements, and that will require exploration of a new cost-share structure to incentivize not only MMS implementation but optimal MMS use.

7.0 Conclusion

The existing literature consistently demonstrates the need for manure management in animal agriculture to reduce environmental impact and effectively utilize manure. Given the pressing challenges linked with the changing climate, there is a need for accurate data to help support the context-specific adoption of appropriate MMS, and to inform cost-share program structures which enable that adoption. As priorities for pollution abatement change, understanding cost-share program structures that select operations for participation, promote optimal uses of MMS, and the maintenance of MMS are also needed. Future work should be directed towards identifying accurate emissions coefficients for each MMS and synergies between MMS in hopes of reducing pollution tradeoffs. Further exploration into cost-share program structures that are more dynamic than just MMS implementation is also an area warranting research.

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Evaluation of Manure Management Systems (MMS) Cost-Share Programs for GHGe and Nutrient Pollution Reduction

CHAPTER 2

Jillian B. Hammond and Robin R. White

1.0 Abstract

Cost-share programs to offset expenses associated with implementing manure management systems[1] (MMS) help to improve adoption rates of these expensive technologies. Manure management systems have historically been cost-shared to support minimizing nonpoint source pollution into surface waters; however, more recently, there is interest in leveraging these programs to help reduce greenhouse gas emissions[2] (GHGe). This study's objective was to explore the strengths and opportunities of MMS cost share programs for achieving joint reductions in nutrient and GHGe. Data used in this analysis included a historical data analysis and a survey of operations managing manure. Historical cost-share data sourced from the Virginia Department of Conservation and Recreation (DCR) were analyzed to determine how payment amounts were related to demographic, infrastructure, and environmental project attributes. Complementing the historical analysis, a survey of operations managing manure in Minnesota (MN) and Virginia (VA) was conducted to characterize producer attitudes toward and experiences with cost-share programs. The survey also sought to explore stated and revealed producer views on cost share pricing structure. This analysis reveals that existing cost share programs for MMSs are adaptive to key farm-scale factors, such as farm size, location,

[1] Manure management systems
[2] Greenhouse gas emissions
[3] Best management practices
[4] Willingness to pay

and economic conditions. Funding from agencies like the NRCS and USDA is correlated with proxies for the effectiveness of best management practices (BMPs), such as the amount of manure treated. However, the cost for producers to implement BMPs has remained relatively constant over time and does not vary based on the quantity of manure managed. Survey data indicate that producers would be slightly more willing to adopt BMPs under new cost share programs, showing a preference for 65% cost share (compared to 75% historically requested), though this might reflect specific attitudes of surveyed producers. Overall, the findings suggest that developing dedicated cost share programs focused on GHGe reduction, rather than relying on water quality programs, may be more effective and financially efficient in achieving GHGe goals in manure management.

Keywords

Animal agriculture, climate change, cost-share, manure management system (MMS), greenhouse gas emissions (GHGe), manure

2.0 Introduction

Agriculture is a climate-exposed industry due to its reliance on environmental quality for successful commodity production (Agrawala et al., 2011; Tingey-Holyoak et al., 2024). The changing climate alters global capacity to produce food due to outcomes like environmental degradation, changing weather patterns, and an increase in extreme weather events (IPCC, 2012 & Farooq et al., 2022). As the global population grows, climate impacts on the food supply may create challenges in meeting the growing food demand (IPCC, 2023) and negatively impact economic resilience of the 42% of livelihoods worldwide that rely on agriculture (Aznar-Sánchez et al., 2019). The IPCC reports warming trends with high probability of surpassing a 1.5°C increase in surface temperatures in the 21st century (IPCC, 2023). In addition to climate change, many agricultural regions around the world are under pressure to improve local water quality and abate historical water quality degradation (EPA, 2023; FAO, n.d.). Collectively, this necessitates adoption of management practices that contribute to reduced GHGe, and improved water quality. Animal agriculture has been identified as a major contributor of GHGe and excess nutrient leaching, with manure management being an important source of both. In 2022, domestic manure management contributed 64.7 million metric tons (MMT) CO_2eq as CH_4 and 290.8 MMT CO_2eq as N_2O to the atmosphere (EPA, 2024). Similarly, by 2050 manure is expected to increase N and P surpluses by 23% and 54% respectively (Bouwman et al., 2013).

Manure management systems present the opportunity to support simultaneous reductions in GHGe and in nutrient surpluses (USDA, 2024). To support these environmental goals, best management practices (BMPs) for the design and operation of

MMS have been developed (Lorimor et al., 2006). However, widescale adoption of manure management BMPs has been limited by their high initial investment, requiring producers to assume a new cost without market-driven incentive to help offset that expense. Government organizations have created cost-share programs to alleviate the financial risk for producers by sharing a portion of the implementation costs for manure management BMPs. Programs like the Environmental Quality Incentives Program (EQIP) and Agriculture Strategic Targeted Animal Residuals (AgSTAR) are existing examples which provide financial assistance to incorporate MMS into livestock operations (EPA, 2024; USDA, 2024). These programs have historically focused on BMPs which support water quality benefits; however, to meet the 2030 goal of reducing US GHGe by 50-52% below 2005 levels (USDS, 2024), there is increasing interest in expanding cost share programs to incentivize BMPs focusing on GHGe.

Toward that goal, this study explored the strengths and opportunities associated with leveraging manure management BMP cost share programs for GHGe reduction. Analysis of historical cost-share records gave perspective of what has been done over the past decade to achieve water quality improvement goals and estimate collateral impacts on GHGe. A survey supplemented this historical analysis to explore stated and revealed producer views on cost share programs. We hypothesized that cost share programs designed for water quality goals would not necessarily contribute to greenhouse gas (GHG) benefits, and that producers would have limited willingness to pay (WTP) for BMPs which reduce GHGe.

3.0 Methods

3.1 Historical Cost-Share Data

Historical cost-share program records (n=483) were sourced from the Virginia Department of Conservation and Recreation (DCR). These records described animal waste MMS implemented in Virginia over the last decade (2014 to 2024). The dataset was inclusive of 9 MMS, as described by the 2024-2025 Virginia Agricultural Cost Share Manual (DCR, 2024). Table 2-1 summarizes the historical data, including details on the location, mitigative potential, and financial investment in the varying MMSs represented within the dataset. Each record within the dataset provided values for the total cost of the project, the proportion of the cost sourced from different funding actors, and any tax incentives which offset expenses. These data were used to calculate 6 key response variables that were analyzed:

Total Cost = Project Expenses – Tax Incentive

where *Total Cost* is the expense of the project after accounting for tax incentives, and the project expense is the total expense paid to complete the project.

Total Funding = USDA + NRCS + Other

where, *Total Funding* is the total amount of cost share assistance received, *USDA* is the cost share assistance received from the United States Department of Agriculture, *NRCS* is the cost share assistance received from the Natural Resources Conservation Services, and *Other* is the cost share assistance received from other funding sources, which could include foundations, non-profit organizations, and other funding agencies.

The share of project expenses realized by the producer was defined as:

Producer Share = Total Cost – Total Funding

For the purpose of analysis, *Total Cost* and *Total Funding* were analyzed as calculated, using USD as the units for expressing these costs. The share of expenses attributed to producers, USDA, NRCS and other agencies were expressed as proportional to the total funding to explore how these agencies differed in their relative expenditures.

3.2 DCR Data Analysis and Model Comparisons

Prior to analysis, data were cleaned for analysis by removing entries lacking values for total cost, tax credit, or amount of waste treated. The primary response variables analyzed included: Total Cost, Total Funding, as well as the proportion of costs paid by the producer, USDA, NRCS, and other funding sources. Two types of models were explored. These models differed in explanatory variables considered and were designed to inform different research questions.

In the first analysis, we explored factors influencing payment structure. The explanatory variables included in the first analysis were: practice code, completion fiscal year, hydrologic unit priority, drainage, total animal units serviced, waste treated, and county. Practice codes included: WP-4, WP-4B, WP-4C, WP-4E WP-4F, WP-4FP, WP-4LC, WP-4LL, WP-4SF as defined by NRCS practice standards (DCR, 2024). Completion fiscal year reflected the NRCS fiscal year during which the project was completed. The hydrologic unit priority could be categorized as high, medium, or low, reflecting the USDA-assigned priority for nutrient mitigation in sensitive waterways adjacent to the proposed BMP. Drainage could be categorized as Southern Rivers or Chesapeake Bay, denoting the watershed the BMP was implemented in. Total BMP animal units reflected that NRCS-calculated animal units serviced by the installed manure management BMP. Waste treated (tons/year) was determined as the manure produced based on the amount of daily manure production of the number and type of animals present on the operation as provided by the dataset. County reflected the VA county in which the project was completed. For each response variable, a linear model was used to explore how these factors influenced project costs or costs per unit of manure treated.

The second analysis specifically explored how payments aligned with the masses of nutrients treated within the MMS. Nutrient concentrations were derived from averaged daily total manure production values for each animal species by the Midwest Plan Service (Lorimor et al., 2004). Total manure was estimated based on the total animal units and the average daily manure production of different animal types. This value was used to calculate nutrient concentrations by dividing the mass of nutrients within the manure, also sourced from the Midwest Plan Service, by the average total manure production. In this analysis, the same response variables were regressed against practice code, completion fiscal year, hydrologic unit priority, drainage, total BMP animal units, county and the concentration of nitrogen (N), phosphorous (P), potassium (K), volatile solids (VS), or total solids (TS) within the treated manure. Due to collinearity among the nutrients, one model was fit considering each nutrient concentration. The intent behind this analysis was to explore the degree to which payment scaled with the nutrients treated by the incentivized MMS. A significant relationship between payment amount and manure nutrient concentration would reflect payment structures that provide monetary support for treatment of manure nutrients, while non-significant associations suggest payment amounts that are unlikely influenced by the nutrients treated.

All statistical analyses and model derivations were conducted using the R Software version 2024.04.2+764 (R Core Team, 2024). The statistical significance of the relationships was determined by analysis of variance and declared at *P*-value ≤ 0.05. Tendencies were considered at 0.05 < *P* < 0.10.

3.3 Extending DCR Data to Infer Estimated Potential Greenhouse Gas Emissions Reduction

In addition to exploring how existing cost share programs align payment structures with water quality mitigation efforts, we wanted to explore whether there was evidence that these programs may have supported improved GHGe as well. Toward that goal, the DCR data were divided into subsets to include only those projects which were completed on beef and dairy operations. For each project, the IFSM (Rotz et al., 2022) was used to simulate farm GHGe. The IFSM program is suited for the modeling of beef and dairy operations, therefore only these records were evaluated through IFSM. The IFSM program provides a comprehensive look at integrated crop and livestock production systems by using programmed equations to account for production interactions over time. In this analysis, we leveraged the reference file "VA grass-fed beef.frm" as a baseline to represent VA beef operations, and the reference file "VA 200 cow dairy.frm" as a baseline to represent VA dairy operations. Each file can be altered in the following subsections: crop and soil, grazing, machine, tillage and planting, harvest, storage, animal and feeding, manure, and economic information (Rotz et al., 2012).

The BMP scenarios were created to represent how farm-level manure was managed through the implemented practice. Farms were run through two simulations. In each simulation, total farm system GHGe and manure GHGe were recorded. Through this approach each farm was modeled to calculate emissions with and without the use of the incentivized MMS, creating a generalized file for the animal waste practice and its counterpart baseline to isolate the effect of the MMS on total farm emissions. Because very little information was provided about each farm, these simulations reflect potential GHG benefits and should not be used to infer true changes in farm-scale GHGe while implementing these MMS.

Farm specific demographics were changed as necessary to accurately represent each data entry. Data entries provided animal species and total BMP animal units that were used in simulation creation. Due to a lack of specification in animal type, the animal type proportion provided in the pre-downloaded farm file was applied to the total BMP animal units. All other applicable information to the MMS that was not specifically denoted in the dataset was assumed as the state average. Simulations were run based on the estimated practice lifespan of the MMS so the applicable weather data could be considered. Based on the DCR data, the animal units, animal facilities, grazing area, grazing costs, miscellaneous machine information specific to manure pump/agitators and feed/manure loaders, and manure information for primary handling were changed to reflect the unique conditions on each farm.

Baseline simulations present a major limitation of this analysis in which GHGe were estimated based on limited information available about each farm implementing manure management BMPs, and no information was available to accurately define baseline. We selected a top-loaded earthen basin, no storage of manure, or no manure collection method as a baseline MMS for beef and dairy to minimize the number of assumptions used in the study, and to give the greatest likelihood of predicted reduction in GHGe. Exception was made to operations implementing WP-4SF and WP-4E since these were not representative of a complete MMS and their baselines were changed to recognize when a covered housing or pumping equipment was not used.

Preliminary visualizations directed analysis toward finding environmentally relevant comparison points specific to water quality and GHGe. The percent change in farm GHGe between the BMP scenario and the baseline scenario was calculated and used as the response variable in a linear regression. A separate model was run for each nutrient concentration to avoid collinearity and determine its individual influence on financial response variables. The explanatory variables included total cost, practice code, amount of waste treated, and their respective interactions, along with completion year and county. Statistical significance of relationships within these models was again determined by analysis of variance. The significance of manure nutrient concentration was summarized using an ANOVA table and declared at P-value of <0.05. Estimated marginal means (Lenth, 2024) were calculated from models to explore targeted significant relationships revealed by the analysis.

3.4 Animal Waste Survey

To complement the historical data analysis, an online survey was distributed to animal feeding operations in Virginia and Minnesota. The intent of the survey was twofold: 1) it was used to solicit producer interest in participating in a manure management cost-share program administered by the Virginia Tech Alliance to Advance Climate-Smart Agriculture; and 2) it was used to explore producers stated and revealed views on manure management cost share programs. The survey was designed and implemented using the QuestionPro Research Edition from Austin, Texas and approved by the Virginia Tech Institutional Review Board (IRB; Protocol #24-330) prior to administration. The survey was advertised through state and local commodity organizations (e.g., VA Cattlemen's Association, VA State Dairymen's Association, VA Poultry Federation, VA Pork Council, VA Forage and Grassland Council, MN Farmers Union, MN State Cattlemen's Association, MN Milk Producers Association, MN Beef Council, and MN Cattlewomen), and targeted producers operating animal feeding operations where MMS were employed. Minnesota and Virginia were the target states due to the focus of the Alliance to Advance Climate Smart Agriculture in these states. The survey was opened May 30th, 2024 and closed July 16th, 2024.

The survey was structured around 6 blocks of information and questions, which included: participant consent, an introduction to MMS and cost-share programs, previous cost-share experience, stated preference for cost-share program characteristics, stated attitudes towards environmental conservation, and demographic identification. The survey administered to participants has been included as a supplementary material. A total of 82 responses were collected, of which 38 were complete (46% completion rate). Participation had significant potential financial benefit to producers, as completion of the survey constituted producers expressing interest in being considered for cost-share funding through the Alliance to Advance Climate Smart Agriculture livestock sub-pilot. Due to the focus on producers operating farms where manure management was implemented, the survey was structured to terminate if the participant did not utilize manure storage or treatment systems as defined by the NRCS (NRCS, 2024). This restriction of participation accounted for 44 dropout responses.

3.5 Animal Waste Survey Data Analysis

Survey data were extracted in aggregate and summarized to yield descriptions of the participant demographic information and responses about cost-share program structure questions. Table 2-2 summarizes demographic data of respondents, highlighting state, county, primary species type, and current MMS used on responding operations. Anticipated cost and requested cost share percent were determined by prompting participants to estimate the value of MMS and cost-share assistance. These values were reliant on the producer's previous experience with MMS and cost-share programs. Positionality statements were used to determine participant attitudes towards responsibility of agriculture for environmental stewardship (Table 2-3). The population percent strongly agreeing with the positionality statements were recorded and extracted directly from the QuestionPro results analysis. Similarly, Likert scale ranking questions were used to gauge the importance of different cost-share attributes (Table 2-4). The average rank of each attribute across three influential categories (i.e. MMS Prospective Interest, Decision-making Factors, and Important Program Attributes) were used to demonstrate their value to participants, where a value of 1 indicated most important and the highest value indicated the least important factor.

4.0 Results & Discussion

4.1 Cost-share funding allocation

Table 2-1 summarizes attributes of operations historically receiving funding within the dataset. Cost-share programs have historically favored incentivizing widescale adoption of technologies and enrolling operations with the highest pollutive potential (Weersink et al., 2001). This intent is demonstrated within the data by producers covering an average of 19% of costs for implementing MMS, while funding organizations covered

74%. The residual 7% was made up by tax incentives received by the producer, reflecting a cost the producer would have to cover upfront but would eventually be recouped. Minimized initial costs have boosted producer interaction and adoption of MMS over time (Cooper & Keim, 1996). The average project summarized in this dataset cost $101,340, with median costs ($80,864) suggesting the distribution is slightly skewed to smaller projects. Poultry projects were most frequently funded, likely reflecting the population of birds within Virginia (NASS, 2022), the economic importance of the poultry industry to the state (ERS, 2023), and the environmental impacts of poultry litter (Bolan et al., 2010; Seidavi et al., 2019). Beef and dairy projects were also frequently funded, again reflecting population (NASS, 2022) and environmental impacts of these animals within Virginia (Place & Mitloehner, 2010; Rotz, 2024; Tamminga, 1998), while goats, horse, and sheep projects were less frequent. Over 75% of projects were funded within the Chesapeake Bay, reflecting the national significance of this watershed (NWF, n.d.) and the scope of environmental impact mitigation efforts (Voyles, n.d.).

The most frequently supported projects included WP-4, WP-4B and C, while E, F, FP, LC, LL, and SF were less frequently supported. The practice standard for WP-4 is inclusive of most manure storage facilities, including solid and liquid storage, while the various lettered applications reflect more specific manure storage (loafing lots, compost, pumps, etc.) reflect specific additions to or applications of manure storage. Thus, it is not surprising that these WP-4 accounted for the vast majority of projects as the development of storage infrastructure is a critical first step toward nutrient containment (Millner, 2009).

Table 2-5 summarizes significance of relationships identified in models exploring how the response variables like total cost, total funding, and the proportion of funding by the producer, USDA, NRCS, and other funding sources were influenced by explanatory variables like practice code, completion fiscal year, hydrologic unit priority, drainage, animal units, waste treated, and county. Total cost and total funding, which were expressed in USD, were responsive to MMS type, year of completion, amount of waste treated (t/yr), and the operation's residing county. Notably, both total funding and total cost were inversely associated with total BMP animal units indicating that as more animal units were treated, the total expense decreased. We expect this may be due to a prevalence of poultry operations within the data, which have a very high number of animal units, but use MMS with lower cost compared with beef, dairy, swine, and other livestock systems. Total cost and total funding also indicated a strong positive association with waste treated, meaning more capital is allotted for projects managing a greater amount of waste through cost-share programs. The directionality of this association suggests clear alignment between funding allocation and the program goal of managing manure nutrients.

The relationships for each response variable against completion fiscal year suggest that existing cost share programs have flexibility to adapt to year-on-year changes in project costs (e.g., inflation). Furthermore, programs can respond to the cost of implementing different practices and to locally driven economic variation. It is interesting to note that total funding, but not total cost was significantly influenced by hydrologic unit priority and tended to be influenced by drainage. By comparing these two response variables, we might infer that funding agencies value more highly prioritized hydrologic units or drainages over those with lower priority. This prioritization of certain hydrologically sensitive areas suggests cost share programs are also flexible to environmental needs, prioritizing focus on areas where changes in nutrient might have the greatest potential impact.

When compared on a proportional basis, many of these trends shift and unique nuances are identified among funding sources. Producers' payment proportion differed with different practice codes and counties, likely reflecting the true cost differences of implementing various MMS in different regions of the state. The USDA and NRCS payment proportions were not affected by practice code, suggesting that the mass of manure treated by different practices is a primary determinant of their value to these agencies. Agency payment proportions scaled with year and differed by county, likely reflecting local economic variation (Lichtenberg et al., 1993) and responsiveness to rising costs and inflation (BLS, n.d.) occurring over the studied time period. Producer payments, however, were unaffected by year.

Without cost-share programs, MMSs are not financially viable due to insufficient returns (Aigner et al., 2003), and this static payment contribution from producers is likely reflective of saturated willingness to pay for these practices. USDA payments, but not those administered through NRCS programs, differed with drainage and total BMP animal units, suggesting that their programs focus on alleviating the amount of manure, inferred by animal BMP units, and subsequently the amount of excess nutrients, entering waterways within specific drainage systems. Collectively, the analysis of relationships between project attributes and the proportional funding from different agencies suggests that flexibility exists within the funding landscape to support a diverse of projects. This flexibility is important for producers, as farms are heterogeneous both in focus and in financial structures.

Cost-share programs have a finite amount of funding to support the implementation of MMS on livestock operations. Fleming et al. (2018) discusses that the shift in agricultural conservation efforts from land retirement to incentivizing conservation practices has led to the development of cost-share programs and the establishment of necessary funding. Fixed funding requires organizations to efficiently use available capital to support maximum pollution reduction (Claassen et al., 2008; U.S. Congress, 1996). Overall, our analyses suggest that financial support provided by funding organizations is responsive to the goal of improving water quality using MMS implementation and geographical targeting as primary strategies, which was also identified by Cattaneo et al. (2005). The data also suggest, however, that producer payments for MMS are less directly tied with water quality improvements, and may be motivated by other factors such as convenience, production goals, etc. (Bopp et al., 2019; Piñeiro et al., 2020).

Table 2-6 summarizes how the addition of manure nutrient concentration as an explanatory factor causes relationships with financial response variables to change. Total cost and total funding remain responsive to year of project completion and county and were strongly associated with the amount of waste treated and concentration of nutrients treated within manure. However, it is important to note that payment proportions across financial response variables were not affected by manure nutrient concentrations. This suggests that the significance of manure nutrient concentrations in driving total project funding and costs may simply reflect the masses of manure treated. Like Table 2-5, the directionality of waste treated upon further analysis of manure on nutrient concentration was maintained as positive where manure nutrient concentrations held a negative slope. This means that as more manure is produced, more capital will be invested in those projects, however regulated by manure nutrient concentrations. As such, the evidence that cost share programs are targeting water quality degrading nutrients, specifically, is limited at present, as the data suggest that these programs primarily rely on the mass of manure treated as a proxy for the mass and concentration of nutrients treated.

4.2 Water Quality Focused Cost Share Programs and Potential Greenhouse Gas Emissions Changes

Of the 189 projects explored in the GHGe analysis through IFSM, only 49 were estimated to result in emissions reductions. In this subset of projects, the median cost of mitigating manure GHGe with manure management was \$34.45 per kg of CO_2e (Figure 2-1). Comparatively, the Virginia Regional Greenhouse Gas Initiative (RGGI) prices carbon at \$17.64 per ton of carbon or \$0.02 per kg (World Bank, 2024). These data suggest a substantial mismatch among costs realized when implementing MMS when compared to costs paid by carbon incentive programs. Specifically, the comparison suggests that that the true cost of mitigation may be much greater than the prices garnered within incentive programs like RGGI. Claassen et al. (2008) suggests the prioritization of practices that offer co-benefits for both climate and water quality. The selection of MMS with co-benefits is important because benefits cannot be assumed (Babcock et al., 1997). Although there may be MMS with important co-benefits to water quality and GHGe mitigation, those evaluated within the present analysis suggest high costs associated with GHGe mitigation. This is consistent with the understanding that most MMSs target either

GHGe reduction or excess nutrient management (De Vries et al., 2015) and finding strategies to achieve both goals may require programs specifically designed for that purpose. While co-benefits have been explored in the literature, lenient policy frameworks have allowed for climate initiatives to be more easily integrated into water quality programs (Löjönen et al., 2020).

Table 2-7 shows the relationships (over the entire 189 projects for which GHGe were calculated) between the estimated potential changes in farm carbon footprint, manure emissions, or manure CH_4 emissions and funding variables, practice code, mass of waste treated, completion fiscal year, and county. Total costs for manure management projects were strongly associated with changes in emissions relative to baseline, with greater costs creating larger reductions (i.e., more negative) in emissions. However, the slight positive slope of this association infers that increasing or decreasing total cost past the initial implementation cost does little to reduce or increase emissions, supporting the need to reallocate funding that exceeds the necessary implementation cost to more beneficial avenues.

The proportion of funding provided by NRCS programs tended to be significantly associated with emissions reductions; however, greater proportional funding from NRCS was associated with positive changes in emissions (increases) relative to baseline. Congruent with IPCC projections (IPCC, 2023), costs associated with GHGe mitigation have increased over time (Figure 2-5). The limited number of projects where GHGe were reduced through manure management, along with the high cost of mitigation, suggest that although spending in water quality BMP programs aligns with GHGe reductions, these programs may be an inefficient way to achieve cost-effective GHGe reductions on farms.

Programs specifically designed to mitigate GHGe are needed for more cost-effective and productive outcomes.

As such, these data should be viewed as an incomplete estimation of the potential GHGe reduction that could be achieved during practice installation on average farms implementing these practices. The assumptions will favor larger GHGe realizations than may be observed in real situations. Future work should explore the efficacy of water quality BMPs in reducing GHGe through measured changes on farms.

4.3 MMS survey results

Cost-share program success is reliant on producer participation. Table 2-2 presents demographic data summarizing the type of producer interacting with the MMS survey. Responses commonly originated from Virginia dairy, beef, and poultry operations, located in agriculturally affluent counties, and who currently leveraged solid manure storage facilities, or other storage and composting facilities. By design, all participants of the survey practiced a type of manure management with 41 indications of their current use across the 38 total survey responses. Of those participants, 26% had previously participated in cost-sharing programs to support installation of MMSs.

Table 2-2 also provides perspective on the financial assistance requests supplied by producers indicating interest in implementing varying forms of manure management over the next 24 months. The median desired cost share requested was 65% of total costs, which was slightly lower than the median cost share received within the historical data (75%) but was within a similar range. Rather than reflecting that producers have greater stated willingness to pay for manure management BMPs than was revealed within the historical data, we believe that collectively the data suggest there is considerable variation among producers in their willingness to incur expenses associated with manure BMP adoption. The majority of producers participating in the present study stated that they strongly agreed environmental stewardship is their responsibility (Table 2-3). Norton et al. (1994) and Weaver (1996), discuss that producers valuing environmental quality will not require full abatement and will be more forthcoming with their willingness to pay. This trend is further supplemented by Caswell et al. (2001) which suggest that altruism towards the interactions between agriculture and the environment, when equipped with the proper education of pollution reduction technologies, demonstrates an increased voluntary effort even when profits are jeopardized. Participants' altruistic views towards the environment may have supported the lowered request for cost share assistance observed within the survey population relative to the historical cost share dataset.

When asked about the attributes of cost share programs that are most important, the total project expense was the most important factor affecting decision making for most respondents, and producers ranked technical assistance and installation support and financial maintenance support as the most critical attribute not currently supported by programs (Table 2-4). Participant responses indicate that producer access to education materials is not inhibitory to MMS optimal use, and that factors like insurance or support for ongoing maintenance were less critical for design of future cost-share programs.

Survey participation holds inherent self-selection bias as it was promoted as an application for a pilot cost-share program through Virginia Tech's Alliance for the Advancement of Climate-Smart Agriculture to receive funding for an MMS. Producers who chose to participate in the survey represent the preferences of that self-selected group, as their initiative and desire to participate separates them from those who chose not to. This self-selection has the potential to influence the perceived importance of MMS types and cost-share program characteristics, because producers who did not participate in the survey may have dramatically different views and preferences compared with those who did participate (Fleming et al., 2018).

5.0 Conclusion

Analysis of historical data suggests that existing cost share programs to support adoption of MMSs are responsive to important farm-scale sources of variation, including farm size, location, and current economic conditions. Although funding from agencies like NRCS and USDA is associated with proxies for BMP efficacy like the mass of manure treated, producer costs incurred when implementing BMPs are independent of amount of manure treated and static through time. Producer surveys focused on hypothetical and future cost share programs show moderately higher willingness to pay for BMPs (65% cost share request versus 75% in historical data); however, this is likely reflective of the attitude of the sample of producers responding to the survey. Collectively, the analyses suggest that cost share programs focused specifically on GHGe reduction may be essential, and that reliance on water quality focused programs to achieve GHGe goals may be financially and functionally inefficient.

6.0 Declaration of generative AI and AI-assisted technologies in the writing process

During the preparation of this work the authors used ChatGPT 4.0 in order to edit the manuscript for clarity while maintaining technical language, writing style, and voice.

After using this tool, the authors reviewed and edited the content as needed and take full responsibility for the content of the published article.

TABLES

Table 2-1. Summary of key variables within the historical cost-share dataset

Categorical Variables									
Practice Code	WP-4	WP-4B	WP-4C	WP-4E	WP-4F	WP-4FP	WP-4LC	WP-4LL	WP-4SF
Count	340	37	80	5	2	0	5	2	8
Primary Animal Type	Beef	Poultry	Dairy	Goat	Horse	Sheep			
Count	177	209	86	1	4	2			
Hydrologic Unit Priority	High	Low	Medium						
Count	243	108	128						
Drainage	Chesapeake Bay		Southern Rivers						
Count	374		105						

Continuous Variables	**Mean**	**Median**	**Minimum**	**Maximum**
Fiscal Completion Year	2019	2019	2014	2024
Total BMP Animal Units	1,674	150	2.64	33,000
Waste Treated[a]	703	273	1.00	9,999[d]
Total Actual Instance Cost[b]	101,340	80,864	0	588,812
Producer Paid[b]	20,257	14,838	-218,099	313,500
Total Funding[b]	75,348	59,362	0	588,812
DCR[c]	153	144	1.00	379
USDA[c]	7.52	0	0	267
NRCS[c]	15.9	1.00	1.00	120
Other[c]	1.32	0	0	239

[a] Units expressed as tons per year
[b] Units expressed in USD
[c] Units express the proportion of cost covered by the actor in USD
[d] Values should be interpreted as infinite estimates
WP-4: Animal waste control facilities, WP-4B: Dairy loafing lot management system, WP-4C: Composter facilities, WP-4E: Animal waste structure pumping equipment, WP-4F: Animal mortality incinerator facilities, WP-4FP: Feeding pad, WP-4LC: Animal waste control facility for confined livestock operations, WP-4LL: Loafing lot management system with manure management (excluding bovine dairy), WP-4SF: Seasonal feeding facility with attached manure storage

Table 2-2. Summary of key variables within the MMS survey

Categorical Variables								
State	VA	MN						
Count	19	8						
County	Rockingham	Amelia	Kandiyohi	Albemarle	Chesapeake	Fauquier		
Count	5	4	4	3	3	3		
Indicated Species Type	Dairy	Beef	Poultry	Swine	Horse	Other[a]		
Count	17	13	12	4	6	5		
Current Manure Management System	SM	C	L	RC	RCF	CL	SLS	AD
Count	11	8	6	5	6	1	3	1

Continuous Variables	**Mean**	**Median**	**Minimum**	**Maximum**
Anticipated Cost ($)	84,950	12,500	500	800,000
Requested Cost Share (%)	62.3	65	0	100

[a] Inclusive of sheep, alpacas, and rabbits
SM: Solid manure storage facility; C: Composting systems; L: Lagoon or liquid manure storage facility; RC: Roofs and covers for solid manure storage; RCF: Roofs and covers for animal feeding or loafing areas; CL: Covers for lagoons or liquid manure storage; SLS: Solid-liquid separation; AD: anaerobic digestion

Tables 2-3. Producer environmental stewardship attitudes

Survey Questions	**Strongly Agree (%)**
Agriculture operations have the responsibility for promoting environmental stewardship through the implementation of nutrient management practices.	69.6
Financial support is essential for agricultural operators to effectively implement environmental stewardship through the implementation of nutrient management practices.	77.3
Environmental stewardship through the implementation of nutrient management practices supports improved agricultural productivity.	68.2
Environmental degradation is not the responsibility of individual agricultural producers.	13.6

Table 2-4. Likert scale ranking indicating producer preferences for cost-share program characteristics

MMS Prospective Interest[a]	**Avg. Rank**
Solid Manure Storage Facility	2.67
Lagoon/liquid Manure Storage Facility	3.00
Composting Systems	3.50
Roofs & Covers for Solid Manure Storage	3.56
Roofs & Covers Animal Feeding/Loafing Areas	4.43
Covers for Lagoons/liquid Manure Storage	5.62
Solid-Liquid Separation	6.07
Anaerobic Digestion	6.57
Decision-making Factors[a]	
Total Project Cost	1.25
Project Timeline	3.9
Labor Availability	3.5
Influence on Operation’s Returns	3.35
Creates Access to New Markets and Revenue Streams	4.15
Access to Relevant Extension Materials	4.60
Important Program Attributes[a]	
Technical Assistance and Installation	2.26
Maintenance Support	3.39
Financial Maintenance Support	2.20
Insurance	4.17
Ease of Application	2.79

[a.]Likert scale questions were ranked with a score of 1 being most desirable and the highest number being the least desirable.

Table 2-5. Significant factors influencing financial response variables

	Intercept[d]	Practice Code[e]	Drainage[e]	Hydrologic Unit Priority[e]	Completion Fiscal Year[d]	Total BMP Animal Units[d]	Waste Treated[ad]	County[e]
Total Cost[b]	-1.381×10^{7} (<0.05)	<0.05	0.397	0.161	-6.921×10^{3} (<0.05)	-0.8467 (0.249)	26.86 (<0.05)	<0.05
Total Funding[b]	-1.15×10^{7} (<0.05)	<0.05	0.064	<0.05	5.73×10^{3} (<0.05)	-0.4187 (0.516)	25.16 (<0.05)	<0.05
Producer Payment[c]	-1.67×10^{6} (0.170)	<0.05	0.447	0.561	8.54×10^{2} (0.156)	-0.351 (0.393)	1.112 (0.483)	<0.05
USDA[c]	87.34 (<0.05)	0.225	<0.05	0.232	-4.318×10^{-2} (<0.05)	-7.480×10^{-6} (<0.05)	-1.818×10^{-7} (0.983)	<0.05
NRCS[c]	-57.37 (<0.05)	0.436	0.284	0.462	2.829×10^{-2} (<0.05)	-1.206×10^{-6} (0.755)	2.687×10^{-5} (0.069)	<0.05
Other[c]	9.839 (<0.05)	0.613	0.139	0.142	-4.881×10^{-3} (0.03)	9.294×10^{-6} (<0.05)	-8.443×10^{-7} (0.885)	0.999

[a.] Units expressed as tons per year
[b.] Units expressed in USD
[c.] Units express the proportion of cost covered by the actor in USD
[d.] Continuous variables listing estimated slope and the attributed estimate *P*-value in parentheses
[e.] Categorical variables list the attributed P-value for the model
Total cost is the total expense for system implementation that will be broken into different share of funding
Total Funding is the collective funding received by the producer for system implementation and is inclusive of DCR, USDA, NRCS, and Other funding sources
Other funding sources represents any other sources outside of the DCR, NRCS, or USDA that contributed to the funding of system implementation

Table 2-6. Variable *P*-value influencing financial response variables on a nutrient basis

	Nutrient	Intercept[e]	Completion Fiscal Year[f]	Hydrologic Unit Priority[f]	Drainage[f]	Total BMP Animal Units	Waste Treated[be]	[Treated][ae]	County[f]
Total Actual Instance Cost[c]	N	-1.856×10^{7} (<0.05)	<0.05	0.456	0.616	-1.003 (0.228)	29.03 (<0.05)	-7.010×10^{6} (<0.05)	<0.05
	P	-1.634×10^{7} (<0.05)	<0.05	0.454	0.615	-0.414 (0.627)	27.80 (<0.05)	-1.218×10^{7} (<0.05)	<0.05
	K	-1.675×10^{7} (<0.05)	<0.05	0.476	0.623	-1.710 (<0.05)	29.85 (<0.05)	-1.55×10^{7} (<0.05)	<0.05
	VS	-1.796×10^{7} (<0.05)	<0.05	0.452	0.614	0.2568 (0.772)	29.81 (<0.05)	-5.876×10^{5} (<0.05)	<0.05
	TS	-1.896×10^{7} (<0.05)	<0.05	0.441	0.610	0.412 (0.636)	28.68 (<0.05)	-4.328×10^{5} (<0.05)	<0.05
Total Funding[c]	N	-1.51×10^{7} (<0.05)	<0.05	0.145	0.429	-0.397 (0.585)	27.0 (<0.05)	-5.57×10^{6} (<0.05)	<0.05
	P	-1.33×10^{7} (<0.05)	<0.05	0.144	0.429	4.42×10^{-2} (0.953)	26.1 (<0.05)	-9.50×10^{6} (<0.05)	<0.05
	K	-1.36×10^{7} (<0.05)	<0.05	0.158	0.438	-0.970 (0.183)	27.7 (<0.05)	-1.20×10^{7} (<0.05)	<0.05
	VS	-1.47×10^{7} (<0.05)	<0.05	0.141	0.427	0.643 (0.406)	27.6 (<0.05)	-4.76×10^{5} (<0.05)	<0.05
	TS	-1.56×10^{7} (<0.05)	<0.05	0.133	0.421	0.782 (0.304)	26.7 (<0.05)	-3.53×10^{5} (<0.05)	<0.05
Producer Paid[d]	N	-2.249×10^{6} (0.102)	0.794	0.319	0.139	-0.499 (0.270)	1.55 (0.347)	-1.23×10^{6} (<0.05)	<0.05
	P	-1.87×10^{6} (0.158)	0.794	0.318	0.139	-0.387 (0.405)	1.34 (0.417)	-2.19×10^{6} (<0.05)	<0.05
	K	-2.00×10^{6} (0.142)	0.794	0.320	0.139	-0.613 (0.168)	1.673 (0.308)	-2.99×10^{6} (0.087)	<0.05
	VS	-1.97×10^{6} (0.147)	0.794	0.320	0.139	-0.371 (0.444)	1.736 (0.290)	-8.09×10^{4} (0.091)	<0.05
	TS	-2.12×10^{6} (0.121)	0.794	0.320	0.139	-0.341 (0.479)	1.57 (0.339)	-6.108×10^{4} (0.057)	<0.05
USDA[d]	N	80.41 (<0.05)	<0.05	0.487	<0.05	-7.51×10^{-6} (<0.05)	2.747×10^{6} (0.731)	-0.147 (0.960)	<0.05
	P	80.39 (<0.05)	<0.05	0.487	<0.05	-7.56×10^{-6} (<0.05)	2.65×10^{-6} (0.743)	-0.508 (0.920)	<0.05
	K	80.99 (<0.05)	<0.05	0.487	<0.05	-7.602×10^{-6} (<0.05)	2.98×10^{-6} (0.710)	2.027 (0.815)	<0.05
	VS	78.23 (<0.05)	<0.05	0.486	<0.05	-6.35×10^{-6} (<0.05)	2.11×10^{-6} (0.791)	-0.280 (0.236)	<0.05
	TS	78.37 (<0.05)	<0.05	0.486	<0.05	-6.56×10^{-6} (<0.05)	1.84×10^{-6} (0.818)	-0.160 (0.313)	<0.05
NRCS[d]	N	-59.38	<0.05	0.480	0.164	3.37×10^{-7}	2.56×10^{-5}	-13.97	<0.05

		(<0.05)				(0.931)	(0.067)	(<0.05)	
	P	-50.25 (<0.05)	<0.05	0.488	0.168	-1.19×10^{-6} (0.770)	2.82×10^{-5} (<0.05)	-6.22 (0.481)	<0.05
	K	-57.52 (<0.05)	<0.05	0.481	0.165	-8.21×10^{-7} (0.830)	2.66×10^{-5} (0.057)	-38.40 (<0.05)	<0.05
	VS	-55.42 (<0.05)	<0.05	0.481	0.165	1.41×10^{-6} (0.737)	2.80×10^{-5} (<0.05)	-0.832 (<0.05)	<0.05
	TS	-58.38 (<0.05)	<0.05	0.481	0.165	2.33×10^{-6} (0.575)	2.57×10^{-5} (0.066)	-0.728 (<0.05)	<0.05
Other[d]	N	7.19 (0.123)	0.228	0.119	<0.05	9.59×10^{-6} (<0.05)	-5.46×10^{-6} (0.318)	-2.66 (0.189)	1.00
	P	8.55 (<0.05)	0.229	0.120	<0.05	9.51×10^{-6} (<0.05)	-5.35×10^{-6} (0.333)	-2.61 (0.449)	1.00
	K	7.78 (0.091)	0.229	0.119	<0.05	9.34×10^{-6} (<0.05)	-5.18×10^{-6} (0.343)	-6.27 (0.288)	1.00
	VS	6.92 (0.131)	0.228	0.118	<0.05	1.03×10^{-5} (<0.05)	-5.31×10^{-6} (0.329)	-0.284 (0.079)	1.00
	TS	6.99 (0.130)	0.228	0.118	<0.05	1.01×10^{-5} (<0.05)	-5.61×10^{-6} (0.305)	-0.167 (0.124)	1.00

[a.] Manure nutrient concentration calculated from the total waste treated (ton/yr) by the system
[b.] Units expressed as tons per year
[c.] Units expressed in USD
[d.] Units express the proportion of cost covered by the actor in USD
[e.] Continuous variables listing estimated slope and the estimate's attributed *P*-value in parentheses
[f.] Categorical variables list the attributed P-value for the model
Total cost is the total expense for system implementation that will be broken into different share of funding
Total Funding is the collective funding received by the producer for system implementation and is inclusive of DCR, USDA, NRCS, and Other funding sources
Other funding sources represents any other sources outside of the DCR, NRCS, or USDA that contributed to the funding of system implementation

Table 2-7. Summary of models exploring alignment between manure management system installation expenses, cost share sources, and changes in GHGe

Variable	Change in Carbon Footprint, % difference from baseline	Change in Manure Emissions, % difference from baseline	Change in Manure Methane Emissions, % difference from baseline
Intercept[d]	$1.06x10^{3}$ (0.166)	$2.64x10^{3}$ (0.082)	$2.73x10^{3}$ (0.142)
Total Cost[b]	$4.33x10^{-5}$ (<0.05)	$6.55x10^{-5}$ (<0.05)	$8.69x10^{-5}$ (<0.05)
USDA[c]	-6.85 (0.275)	3.20 (0.796)	9.88 (0.517)
NRCS[c]	7.79 (0.194)	28.2 (<0.05)	28.39 (0.053)
Producer[c]	0.597 (0.926)	11.9 (0.354)	4.65 (0.767)
Practice Code[e]	<0.05	<0.05	<0.05
Waste Treated[ad]	$4.16x10^{-3}$ (<0.05)	$2.36x10^{-3}$ (0.459)	$1.79x10^{-3}$ (0.646)
Completion Fiscal Year[e]	-0.524 (0.166)	-1.32 (0.080)	-1.361 (0.139)
County[e]	0.302	0.095	0.074

[a.] Units expressed as tons per year
[b.] Units expressed in USD
[c.] Units express the proportion of cost covered by the actor in USD
[d.] Continuous variables listing estimated slope and the attributed estimate *P*-value in parentheses
[e.] Categorical variables list the attributed P-value for the model
Total cost is the total expense for system implementation that will be broken into different share of funding
Total Funding is the collective funding received by the producer for system implementation and is inclusive of DCR, USDA, NRCS, and Other funding sources
Other funding sources represents any other sources outside of the DCR, NRCS, or USDA that contributed to the funding of system implementation

FIGURES

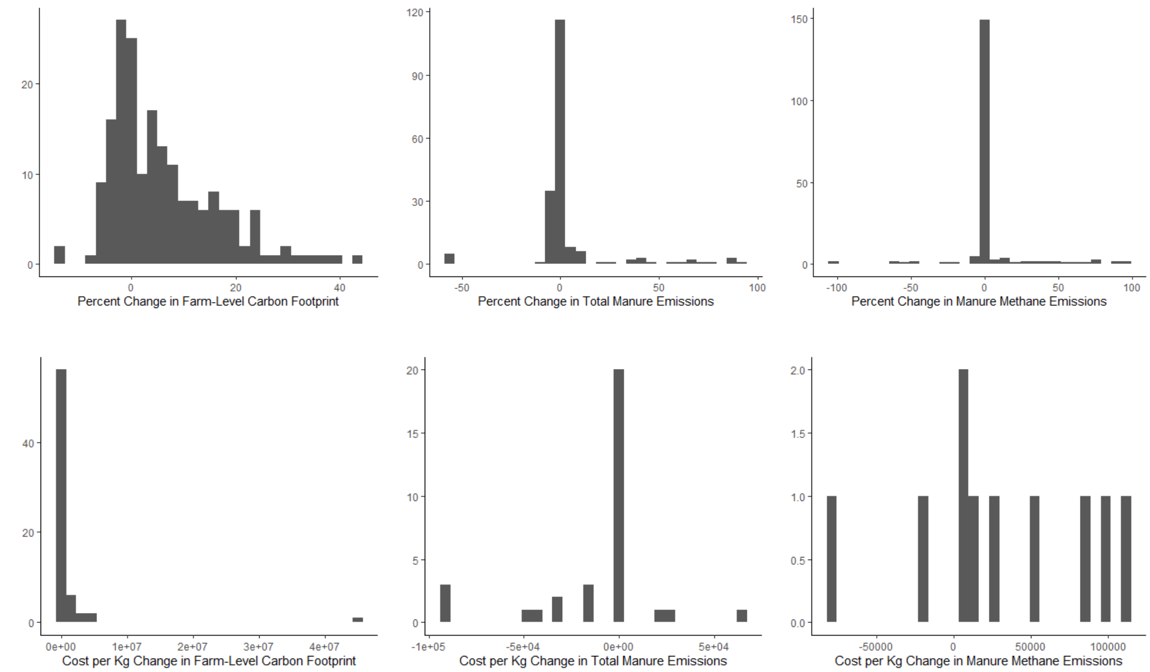

Figure 2-1. Percent change and cost per unit of change in GHGe environmental metrics. The units for cost expressed as USD. The y-axis is the frequency in which the x-axis quantitative variable occurs.

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A Quantitative Summary of Livestock Manure Management Systems (MMS) Pollution Mitigation Efficacy

CHAPTER 3

Jillian B. Hammond[a] and Robin R. White[a1]

1.0 Abstract

The persistence of climate change makes the utilization of manure management systems (MMS) for the reduction of greenhouse gas emissions (GHGe) and excess nutrient management a critical priority to minimize the environmental impact of livestock. Our objective was to quantitatively summarize current literature reporting the impact of MMS on GHGe and manure nutrient composition. A literature search was conducted to identify studies measuring MMS effects on GHGe and manure nutrient composition. The keywords “greenhouse gas” and “livestock” along with MMS-specific keywords (i.e. anaerobic digester, lagoon cover, solid-liquid separation, storage, or compost) were used to search Google Scholar and Agricultural Online Access (AGRICOLA) for relevant works. Papers were included if they met the following inclusion criteria: manure was treated by desired MMS, influent and effluent measurements were taken for manure composition, and effluent measurements were provided for GHGe. Linear, mixed-effects models were derived to explore how environmental outcomes were influenced by manure and system characteristics. Response variables included CH_4 and N_2O emissions per kg manure dry matter (DM), and the proportional change in N and P occurring between influent and effluent of MMS. Explanatory variables included species, MMS, location, and the 2- and 3-way interactions among these variables. Data were often a limiting factor in model derivation and a need for more comprehensive testing of MMS effectiveness was identified for several MMS and species combinations. For the systems with available data, MMS type and the species from which manure originates significantly influenced emissions and nutrient concentrations. Although emissions per kg manure correlated well between N_2O and CH_4, emissions and changes in nutrient composition were not correlated in this dataset. A major conclusion from this work is the need for a broader diversity of experimental evidence to underpin accurate and representative emissions factors for MMS application across livestock species.

Keywords

Livestock, Manure, Greenhouse Gas Emissions (GHGe), Manure Management System (MMS), Nutrient Pollution

2.0 Introduction

Livestock manure is a known contributor to nutrient sourced pollution and greenhouse gas emissions (GHGe) (Chadwick et al., 2011; O'Mara, 2011), with manure management accounting for 59.6% of CH_4 emissions and 19.7% of N_2O emissions in the US as of 2020 (EPA, 2024c). Similarly, aquatic ecosystems are threatened by manure nutrient surpluses, with livestock manure contributing roughly one-third of N and nearly half of the P in the U.S. (Del Rossi et al., 2023). Much of these nutrients are likely to aid in the 41% and 46% of streams and rivers experiencing consequential levels of these nutrients respectively (Metaxoglou & Smith, 2022). Continued agricultural production is reliant on the preservation of the environment and natural resources (Aillery et al., 2005; EPA, 2024a), establishing the need for best management practices (BMPs) and manure management systems (MMS) to direct conservation efforts (IPCC, 2022). The IPCC reports that MMS could benefit adaptation and sustainable development goals with the potential to mitigate 3.9 gigatons (Gt) of CO_2 equivalents every year (IPCC, 2022).

Estimating the impact of MMS on manure nutrient excretion and GHG is critical to support regulatory activities. Environmental regulations largely leverage the language of NRCS practice standards. Tools like the CarbOn Management & Emissions Tool (COMET) (USDA & CSU, n.d.) and the Integrated Farm System Modeling (IFSM) (Rotz et al., 2022) can be used to estimate the impact of some of these practice standards on livestock operations. However, these models often rely on emissions factors (emissions per unit of manure treated) to estimate environmental impacts, and the data underlying these emissions factors may be somewhat limited (Rotz et al., 2022; USDA

& CSU, n.d.). Furthermore, there is misalignment between NRCS practices and the MMS simulated in these software tools. For example, the COMET software successfully simulates storage, anaerobic digestion, and different housing for most species; however, it lacks specification of storage coverage, composting, and time manure was stored for some species (USDA & CSU, n.d.). Additionally, although IFSM simulates manure storage and anaerobic digestion, it lacks specification of storage coverage, and species and geographic variation (Rotz, 2024). To enable more precise evaluation of how NRCS practices influence GHGe and manure N and P excretion, it is critical to evaluate the data underpinning emissions factors specific to NRCS practices and to summarize average environmental outcomes associated with practice implementation.

Toward that goal, the objectives of this work were twofold: 1) to identify strengths and gaps within the MMS literature; and 2) to derive species-specific emissions factors to represent expected emissions associated with MMS use. We hypothesized that changes in GHGe production could be estimated by MMS, manure characteristics, and that minimal co-benefits (i.e., simultaneous improvements in GHGe and nutrient emissions) would be achieved by some MMS but not others. Identifying MMS that hold co-benefits to reduce both pollution types will minimize the instances where pollution tradeoffs occur and help guide policy programs to expand use of NRCS practice standards to support GHGe reductions.

3.0 Methods

3.1 Data Collection

A literature search was conducted across the Google Scholar and AGRICOLA databases to collect data on the efficacy of manure management technologies. Each database was searched using the phrases “greenhouse gas” and “livestock” in conjunction with “anaerobic digester”, “lagoon cover”, “solid-liquid separation”, “storage” or “compost”. The search was guided by the Preferred Reporting Items for Systematic Reviews and Meta-Analyses (PRISMA) inclusion methodology, as shown in Figure 3-1. Literature reviews, textbooks, and field guides were excluded from the literature search as only original research studies reporting pre- and post-MMS treatment effects for manure nutrient concentrations and greenhouse gas emissions were included in the final dataset. Studies were included for analysis if they provided details on the days manure was stored, treatment mass (kg) or volume (L), system volume (L), manure species, and characteristics of manure type (i.e. whole slurry, digestate, compost, etc.). Studies were also required to report influent and effluent data on dry matter (DM) or total solids (TS), and effluent data on the various forms of N and the following GHGe: CH_4, N_2O, and less stringently CO_2. Other data recorded were the influent and effluent values for organic matter (OM), volatile solids (VS), N, P, ammonium (NH_4), ammonium nitrogen (NH_4-N), ammonia (NH_3), ammonia nitrogen (NH_3-N), total Kjeldahl nitrogen (TKN).

The MMS selected for observation were anaerobic digesters, solid-liquid separation technologies, composting systems, covered and uncovered storage, and combinations of these systems. These systems were selected to correspond to NRCS practice codes No. 366, No. 632, No. 317, No. 313, No. 318, and No. 591 (USDA, n.d.). In many cases, papers did not expressly report whether storage was covered, and in these situations, we assumed manure was in uncovered storage. Some studies also tested combinations of more than one MMS, and these studies were broadly classified as mixed use. Data on manure nutrient composition was collected to account for the varying dietary nutritional contents, which were expected to vary with species and geographic location. The units reported in individual publications were normalized to kilograms (kg) through the use of manure (Lorimor et al., 2004) and GHG densities (Air Liquide, 2024). A summary of the resulting dataset is included in Table 3-1.

3.2 Data Analysis and MMS Comparisons

All statistical analyses and modeling were conducted using R software version 2024.04.2+764 (R Core Team, 2024). Preliminary analyses and relationship visualizations were run using the ggplot2 (Wickham, 2016) functions to identify outlying data through density plots. Of the articles chosen for inclusion (n=61), seven were excluded as outliers after preliminary analysis (n=54). These outliers reported values that were several-fold higher or lower than most of the data and could not obviously be assigned to a unit reporting error (e.g., g versus kg).

Data were analyzed using linear, mixed-effects models as described previously (Bates et al., 2015). Response variables included CH_4 and N_2O emissions per kg of manure DM treated, as well as the proportional change in N and P concentrations between influent and effluent, calculated as ([effluent] – [influent])/[influent]. Response variables were log-transformed prior to analysis. Explanatory variables included factors representing manure system type, species, and location. Location of studies was categorized into broader continental groups, including Europe (encompassing studies from Spain, Denmark, Portugal, Norway, Italy, Sweden, Germany, Austria, France, Netherlands, United Kingdom, and Hungary); Asia (Indonesia, Japan, China, and Korea); and North America (Canada and the United States). Initial models for CH_4 and N_2O included manure system type, species, location, and the two- and three-way interactions among these variables. Models were derived using backward stepwise elimination, where variables or interaction terms were sequentially removed from the initial model. At each step in the backward stepwise elimination process, terms with the highest *P*-value were selected for removal; however, non-significant main effects that were part of significant interaction terms were retained during the stepwise elimination process. Once a model was identified that included only significant terms, or main effects that were part of significant interactions, variance inflation factors (VIF) were checked (Wickham et al., 2019). For main effects, VIF < 10 was targeted. For interactions, VIF were expected to be higher due to correlation by calculation, and values of < 500 were accepted (Akinwande et al., 2015).

After a final model was identified that met these criteria, the model alignment with the data was evaluated through calculating the concordance correlation coefficient (CCC)(Lin, 1989), the root mean square errors (RMSE, % mean), the sigma hat errors ($\hat{\sigma}_e$), and corrected Akaike information criterion (AICc)(Hurvich & Tsai, 1993).

The *emmeans* (Lenth, 2024) package was used to calculate relevant estimated marginal means considering significant interaction terms identified for each model. These means were back transformed out of log scale to allow for exploration of expected emissions per unit of manure DM. Variation around these emission factors, as well as completeness of the data for estimating emissions factors were analyzed to identify gaps in the current literature.

3.3. Alignment of Environmental Impacts

Increasingly there is interest in management systems that simultaneously address water quality and GHG objectives; however, the ability of diverse systems to reduce excess nutrients and GHG have not been widely explored. To explore the alignment between emissions and nutrient removal, we calculated Pearson's Correlation Coefficients among environmental measures of system performance. Pairwise correlation plots were visualized through the *ggpairs* function from the *GGally* package (Schloerke et al., 2024). Each scatterplot within the matrix represents the pairwise comparison of two of the four variables to identify the additive value between two MMS.

4.0 Results

Table 3-2 shows the median GHG per mg DM per day across manure management systems and species and highlights the number of observations included in generating each median. This summary of the raw data is critical to identify gaps in the literature. Although manure from beef, dairy, and swine have been evaluated in each of the explored manure management systems, some systems are more broadly studied in specific species. The data also highlight that CH_4 was more commonly reported than N_2O. Covered manure storage was widely studied in beef systems, composting was widely studied in dairy systems, and anaerobic digestion was commonly studied for swine manure. The only poultry litter management studies identified from the literature search focused on anaerobic digestion and composting.

Methane emissions per kg manure DM were significantly affected by manure system and location, as well as the interaction between species and manure system, and the interaction between species and location. The significance of individual model terms, the slopes and standard errors derived for each term, and the fit statistics for this model are reported in Table 3-3. The model was able to explain the majority of the variation in the data (CCC = 0.974); however, the comparison of adjusted and unadjusted fit statistics suggests that substantial variation among studies was apparent within the data. Emissions estimated by the model for different combinations of manure management system and species (dairy) are summarized in Figure 3-2.

Table 3-4 summarized the parameter estimates, standard errors, and fit statistics for the model exploring factors affecting N_2O emissions. Location and species were the primary drivers of N_2O emissions, and manure system was not included in the final model. The N_2O models also explained considerable variation within the data (CCC=0.968), with unadjusted values indicating considerable study-to-study variability.

The change in N and P concentration between the influent and effluent was affected only by manure system type. The differences in N concentration due to manure management systems are summarized in Figure 3-3. Pearson correlation coefficients calculated among environmental measures of manure systems suggest limited alignment between emissions intensity and changes in nutrient concentrations (Figure 3-4). Across all manure management systems, only CH_4 and N_2O were significantly associated with one another. Although data within some specific manure management systems showed greater alignment between emissions and nutrient concentrations (e.g., covered storage for N_2O and change in N concentration), within-system correlations tended to align with cross-system correlations. Figure 3-4 also demonstrates the lack of studies measuring MMS influence on manure P concentrations, despite most MMS being P limited.

5.0 Discussion

5.1. Strengths and gaps within the MMS literature

Our literature summary suggested that some MMS are more broadly studied than others and helped identify gaps in data on specific MMS/species combinations that limit our capacity to derive species-specific emissions factors for different MMS. The literature is more saturated with studies focused around the GHG mitigation effects of MMS on beef, dairy, and swine operations. Both CH_4 and N_2O (kg DM/d) are generally well represented across all MMS for the previously mentioned species, with some exceptions. Notably, limited N_2O measurements for anaerobic digestion systems have been reported, possibly because anaerobic digesters rely on biogas fermentation (Venkiteshwaran et al., 2015) and produce very little N_2O (Jun et al., 1996b). Poultry and mixed species manure use in MMS are less represented across the literature, possibly due to poultry litter's common use as a fertilizer in crop production (MacDonald et al., 2009).

Studies evaluating composting were primarily employed in dairy MMS, with limited studies reporting applications for treating beef and swine manure. Liquid manure covered storage was reported in beef, dairy, and swine MMS; however, solid manure covered storage was solely reported in beef MMS. This could be due to the liquid nature of dairy and swine manure that make it less likely to be categorized into solid storage (Lorimor et al., 2004). Uncovered storage for both solid and liquid manure was well represented across beef, dairy, and swine. However, uncovered solid manure storage was primarily reported for dairy MMS, which could be attributed to the use of another MMS like SLS that minimizes the air quality impact by separating manure into its fractions (EPA, 2024b; Zhang et al., 2024). Although some of these gaps may reflect the varying applicability of MMS in typical housing systems for different species, the summary of data highlights data gaps that could help direct future studies.

5.2. Emissions factors for MMS

Although the data were limited in the coverage of measurements across all MMS for all species, we were able to do some targeted comparisons of well-studied MMS and species. In addition to MMS and species, location of study was an important factor in both the CH_4 and N_2O models. Both Europe and North America had emissions that were lower than Asia, with some species-specific differences in ranking between estimated emissions coming from European and North American systems. This location-driven shift in emission potential of systems may reflect broader management differences among locations affecting manure composition or suitability of MMS.

Species was also important in estimating emissions factors for both CH_4 and N_2O. Beef, poultry and mixes of species' manure resulted in greater emissions of N_2O, while dairy and swine manure resulted in lesser N_2O emissions. This could be explained by swine and dairy manure typically being stored in anaerobic conditions that minimize N_2O emissions by not facilitating aerobic storage conditions (EPA, 2024b; Hatfield et al., 1998). The emissions factors for poultry manure were smaller than expected when compared to daily values found in Ro et al. (2017) and Dunkley and Dunkley (2013), but do reflect that poultry litter contains high N, which provides substrate for processes like nitrification or denitrification to produce N_2O. Likewise, cattle manure is commonly used as a fertilizer for crop and forage production (Loss et al., 2019) due to its dense N and P nutrient profiles. However, the spreading of manure, when not properly incorporated into the soil through injection or tillage, can produce more N_2O emissions because manure N is available to be volatilized into the atmosphere (Webb et al., 2010).

Anaerobic digestion systems were expected to have among the lowest CH_4 emissions potential for managing manure. The purpose of anaerobic digestion is to facilitate anaerobic conditions so OM can be converted into biogas through microbial breakdown (Ziganshin et al., 2013). The transition of OM into biogas (i.e. CH_4) is sensitive to manure species type as each species offers a different OM load and microbial community, providing explanation to the variation of MMS abilities across manure species type. The low CH_4 production from dairy manure reflects the benefits of anaerobic digestion in limiting CH_4 escape to the environment.

Composting systems for dairy operations were the only studies with enough data to derive species-specific coefficients regarding composting. Composting has been identified as a strategy to manage manure with limited CH_4 production because its aerobic conditions inhibit methanogenesis (Brown et al., 2008).

Systems that relied on solid-liquid separation, covered or uncovered storage, or mixes of MMS had relatively consistent emissions compared to composting and AD. These trends likely reflect the anaerobic nature of these production systems. For example, in SLS, the concentration of solid and liquid manure fractions minimizes CH_4 emissions from the solids but creates a more anaerobic environment for the liquids (EPA, 2024b; Zhang et al., 2024). Anaerobic conditions facilitate microbial growth supportive of catalyzing the methanogenesis pathway, producing greater CH_4 emissions (Mutungwazi et al., 2021). The use of SLS is common in swine systems, where anaerobic storage conditions are used to degrade most of the solid fraction and conditioning it for preferred anaerobic digester use (Hatfield et al., 1998). Both composting and SLS demonstrate that both dairy and swine play a role in measured CH_4 change.

5.3. Multi-Benefit Systems

Figure 3-3 shows positive changes in manure N from influent and effluent concentrations for covered storage and SLS, where anaerobic digestion, composting, uncovered storage, and mixed MMS use showed negative changes, or a loss of N. The observed positive changes for covered storage and SLS reflect the ability of these MMS to retain N by protecting N from exposure to the atmosphere while SLS consolidates manure N to one fraction. Wan et al. (2021) highlighted that different influent manures will alter the expected microbial communities present for degradation in storage systems. Comparatively, MMS like composting and uncovered storage would be instances that manure N loss is likely to the environment from a lack of protection against leaching or exposure to nitrification and volatilization (Petersen et al., 1998).

Correlation analyses suggest that few MMS are associated with concurrent improvements in nutrient concentrations and GHG. This misalignment in environmental objectives could reflect the fact that MMS, although used for environmental mitigation purposes, are also used to refine manure composition into a better fertilizer for field application (DeLuca & DeLuca, 1997; Goldan et al., 2023). Increasing the accessibility of N in manure is favorable for fertilizer as it is a valuable nutrient for producing robust crop yields (Campbell et al., 1995; Hirel et al., 2011; Olson & Kurtz, 1982).

Environmental impacts from MMS rarely aligned such that minimizing GHG also resulted in proportional decreases in manure N or P content. The only correlation significant across MMS was that between CH_4 and N_2O emissions. Webb et al. (2012) provides evidence to this correlation by highlighting the similarities of where these gasses are sourced from in livestock housing. In investigating the MMS-specific correlations calculated, AD and CS showed inverse associations between CH_4 and N_2O emissions.

The anaerobic digestion process promotes methanogenesis and increases CH_4 (Venkiteshwaran et al., 2015), while anaerobic conditions may have decreased the likelihood of N_2O emissions in response to the bacterial community of that digester (Harter et al., 2014) and species manure (Ziganshin et al., 2013). Covered storage not only presented a negative correlation, because of the favorable conditions for CH_4 production compared to N_2O, but also was presented as a significant indicator that change in GHGe would occur with its use. Both uncovered storage and the use of multiple MMS displayed strong positive correlations, meaning that the increase or decrease of one GHGe would result in the increase or decrease of the other. Uncovered storage likely showed a strong, positive correlation because of its exposure to atmospheric conditions, allowing for more CH_4 and N_2O to be emitted. However, using multiple MMS likely indicates a positive correlation towards decreasing emissions because manure is further processed with the goal of minimizing GHGe.

With the exception of covered storage, the change in manure N was not associated with CH_4 or N_2O emissions. Covered storage showed negative correlations between N losses and N_2O emissions, meaning that reducing one will result in the increase of the other. This association suggests that minimizing N loss in covered storage systems comes at the expense of N_2O emissions, likely because denitrification is increased under anaerobic conditions (Harter et al., 2014). The change in P was inversely associated with CH_4 emissions in anaerobic digesters, perhaps due to harvesting of CH_4 during anaerobic digestion and the limited impact on P contents. Overall, very few of the manure management systems showed capacity to simultaneously benefit both nutrient emissions and GHGe objectives.

6.0 Conclusion

This study aimed to identify relationships between species and MMS use by deriving species-specific emissions coefficients and analyzing data from the literature. For a more comprehensive analysis and to conclude meaningful inferences from the data, more data regarding MMS use and species, MMS N_2O emissions, and manure P concentrations are needed. Although the study theoretically identifies opportunities for MMS to be used for concurrent air and water quality improvements, the general assumption cannot be made that all MMS will present co-benefits. Continued progress in agricultural sustainability will require leveraging MMS synergistic reduction abilities to address all facets of manure pollution.

TABLES

Table 3-1. Key variables and calculations

Categorical Variables								
Manure System	Anaerobic Digestion	Composting	Solid-Liquid Separation	Mixed	Covered Storage: Liquid	Covered Storage: Solid	Uncovered Storage: Liquid	Uncovered Storage: Solid
Count	17	6	15	8	8	4	10	7
Species	Cattle	Dairy	Swine	Poultry	Mixed	Other		
Count	13	21	26	4	4	1		
Manure Type	Whole Slurry	Liquid Fraction	Solid Fraction	Digestate	Compost			
Count	35	13	12	9	6			
Continuous Variables								
	Mean	**Min**	**Max**					
System Volume (L)	80,218	0.500	2,630,000					
Treatment Volume (L)	6,200	0.200	150,000					
Manure Mass (kg)	24,873	0.200	1,130,000					
Days Stored	79.8	0.000	487					
	Influent			**Effluent**				
	Mean	**Min**	**Max**	**Mean**	**Min**	**Max**		
Dry Matter (kg)	3,250	0.02	81,900	1,380	0.02	61,100		
Organic Matter (kg)	3,870	0.01	54,700	2,000	0.01	33,900		
N (kg)	119	0.000	6,670	190	0.000	16,800		
NH_4 (kg)	5.48	0.000	58.4	0.0271	0.0158	0.0497		
NH_4-N (kg)	2.38	0.00009	238	119	0.000	9,450		
NH_3 (kg)	5.42	0.000	49.4	1.03	0.000	26.5		
NH_3-N (kg)	0.197	0.0001	0.510	0.0501	0.0001	0.470		
TKN (kg)	73.2	0.000	6,840	2.09	0.0002	20.5		
Effluent GHGe								
	Mean	**Min**	**Max**					
CH_4 (g/kg initial mass/d)	1.100	0.000	64.4					
CH_4 Ln (kg DM/d)	-17.4	-28.5	-2.74					
N_2O (mg/kg initial mass/d)	0.340	0.000	23.7					
N_2O Ln (kg DM/d)	-24.2	-35.8	-10.7					

Table 3-2. Median GHGe production for interactions between species and manure management types

CH_4 (mg DM/d)	Species				
MSID	Beef	Dairy	Swine	Poultry	Mixed
Composting	2.86 (2)	8.72 (16)	10.9 (2)	-	-
Covered Storage: Liquid	7.42 (26)	5.44 (2)	2.16 (1)	-	-
Covered Storage: Solid	4.12 (4)	-	-	-	-
Uncovered Storage: Liquid	0.295 (6)	3.76 (8)	0.00935 (1)	-	72.6 (1)
Uncovered Storage: Solid	3.80 (2)	0.231 (7)	13.8 (4)	-	-
Solid-Liquid Separation	228 (5)	21,200 (7)	836 (2)	-	63.8 (2)
Anaerobic Digestion	4,670 (4)	1.64 (5)	2,740 (32)	11,100 (12)	62,000 (11)
Mixed		1.03 (4)	5.01 (7)	-	-
N_2O (mg DM/d)	**Species**				
MSID	Beef	Dairy	Swine	Poultry	Mixed
Composting	0.0506 (4)	0.0648 (16)	11.5 (2)	0.0276 (4)	-
Covered Storage: Liquid	0.0143 (20)	0.0356 (2)	-	-	-
Covered Storage: Solid	0.0885 (4)	-	-	-	-
Uncovered Storage: Liquid	0.0336 (6)	0.00324 (7)	0.000000389 (1)	-	22.9 (1)
Uncovered Storage: Solid	0.0197 (2)	0.0196 (5)	0.00291 (2)	-	-
Solid-Liquid Separation	3.55 (3)	0.0161 (3)	-	-	50.1 (2)
Anaerobic Digestion	-	0.00615 (2)	0.0915 (6)	-	-
Mixed	-	0.00412 (3)	0.00247 (6)	-	-

Number of observations in the determination of median values are denoted in parentheses.

Table 3-3. Model estimating methane emissions per kg manure per day

Common Terms	**Parameter Estimate**
Intercept	-16.5 (4.52)*
MSID – COMP	8.54 (6.32)
MSID – CSL	-0.513 (4.58)
MSID – CSS	0.607 (3.72)
MSID – M	5.19 (3.52)
MSID - SLS	4.43 (4.75)
MSID – US	6.26 (4.96)
MSID – USL	-0.555 (4.63)
MSID – USS	0.724 (3.50)
SPEC – D	6.00 (5.13)
SPEC – M	6.94 (2.85)*
SPEC – S	-5.38 (3.81)
L1 – Europe	-3.57 (3.75)
L1 – North America	-11.7 (3.57)*

			Species		
Species-Specific Terms	D	M	O	P	S
MSID – COMP	-13.6 (6.88)$^{+}$				-4.95 (6.38)
MSID – CSL	1.94 (4.86)				
MSID – M	-3.26 (3.95)				
MSID - SLS	-4.64 (5.01)	-5.11 (3.06)			
MSID – USL	0.950 (4.84)				6.44 (4.97)
MSID – USS	-6.02 (3.89)				
L1 – Europe	-7.70 (4.33)$^{+}$				
L1 – North America	5.01 (4.13)				

Fit Statistics	
N	96
RMSE (%)	-6.06
uRMSE (%)	-10.0
CCC	0.932
uCCC	0.819
AICc	375
Sigma-e	1.47

Standard errors are denoted in parentheses.
* denotes significance at <0.05
+ denotes a trend between 0.05 and 1

Table 3-4. Model estimating nitrous oxide emissions per kg manure per day

Variable	Parameter Estimate	SE
Intercept	-15.6*	2.36
Dairy	-3.73+	1.84
Mixed	2.64*	0.898
Poultry	-0.151	0.945
Swine	-5.68*	2.08
Europe	-7.02*	2.11
North America	-8.79*	2.18
Fit Statistics		
N	102	
RMSE (%)	-3.98	
uRMSE (%)	-13.0	
CCC	0.968	
uCCC	0.603	
AICc	389	
Sigma-e	1.11	

* Denotes significance at <0.05
+ Denotes a trend between 0.05 and 1

FIGURES

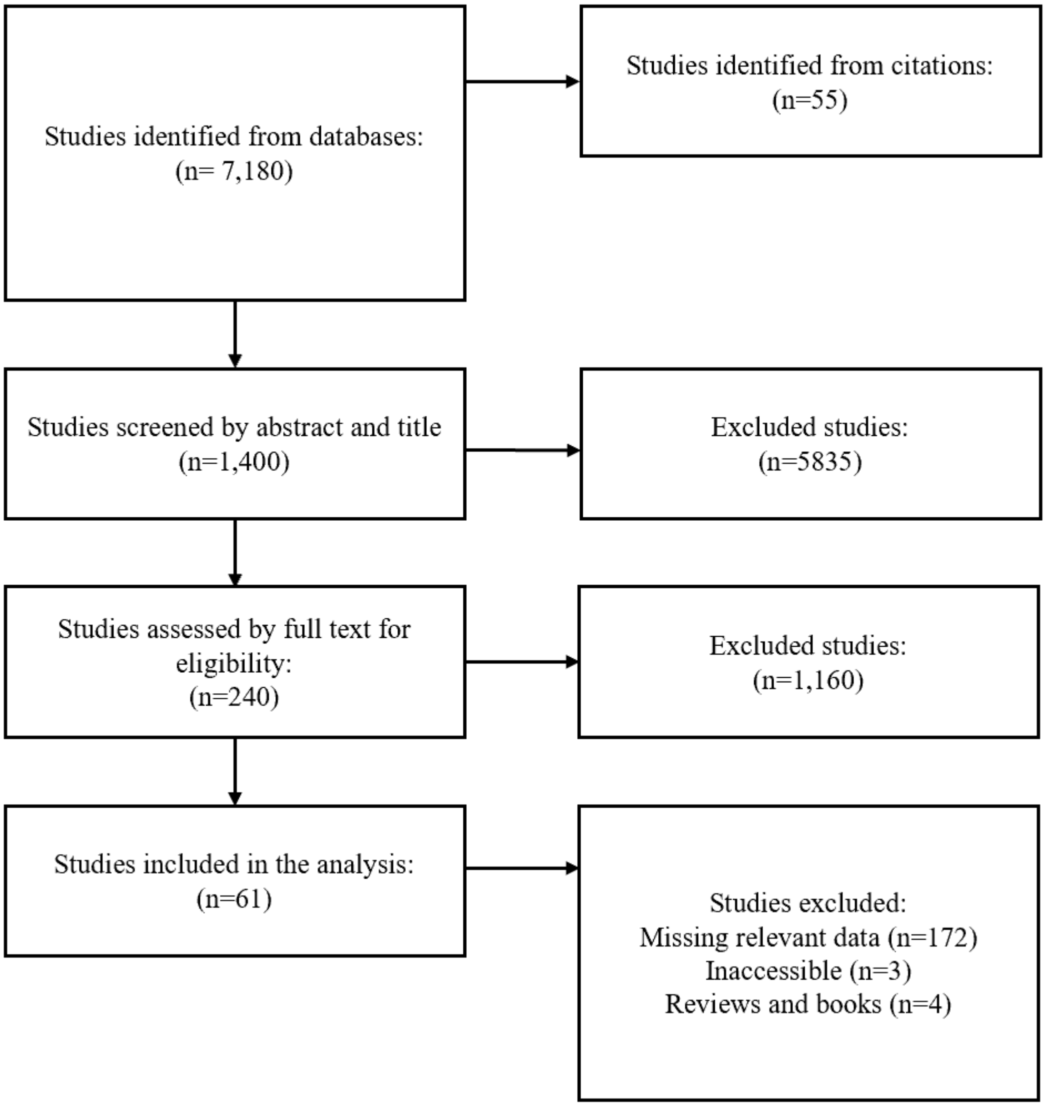

Figure 3-1. PRISMA literature selection method

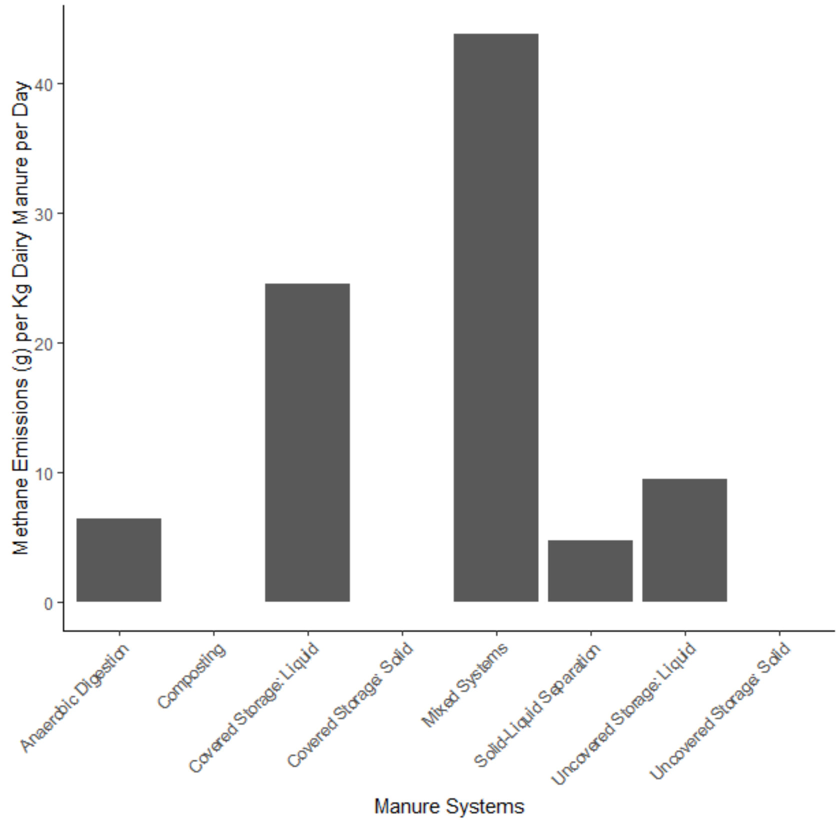

Figure 3-2. Methane emission per kg of manure per day estimated by the model summarizing the relationships between manure management system, species, and emissions.

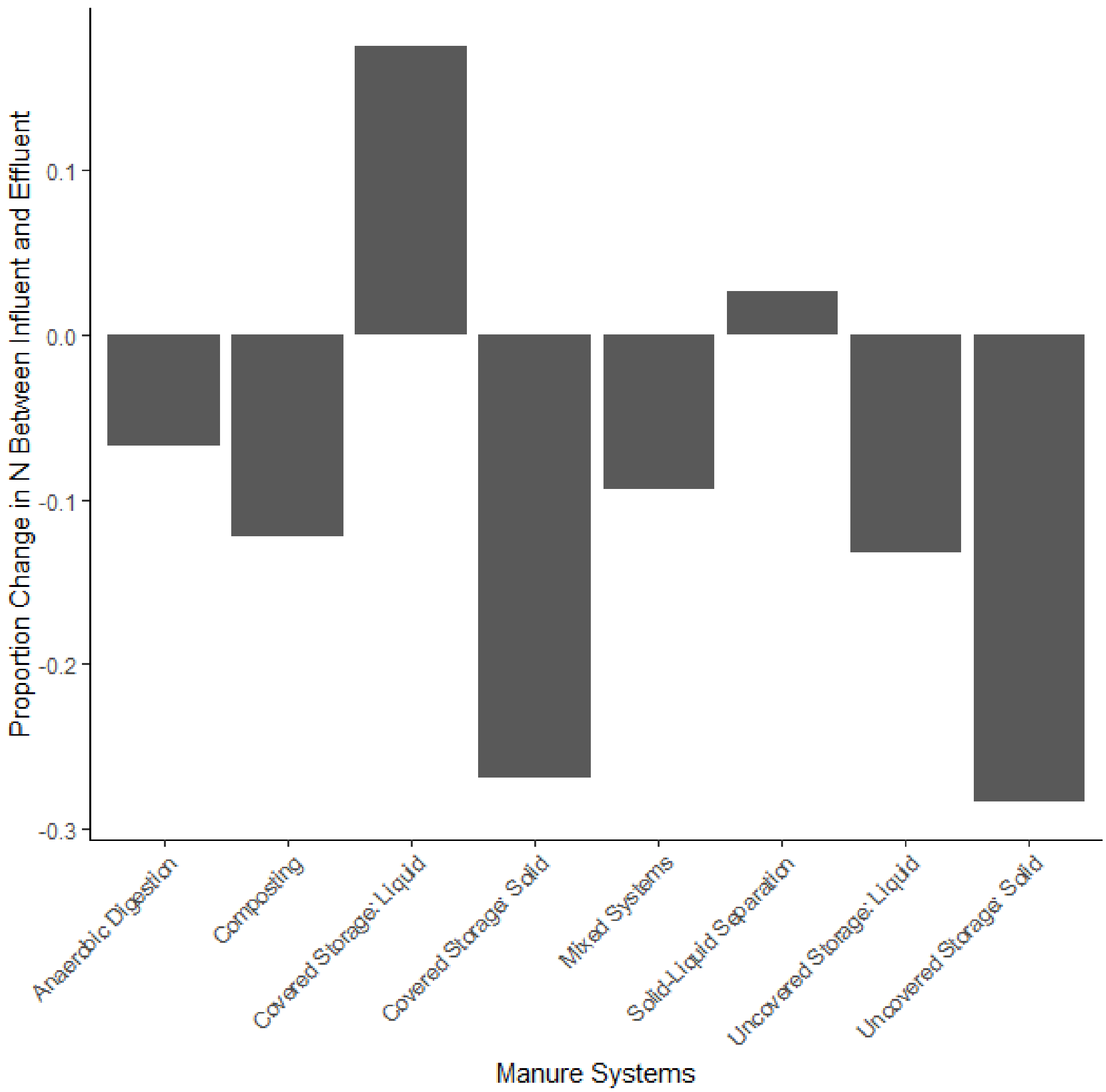

Figure 3-3. Proportional change in N measured in effluent versus that measured in influent. Positive values reflect an increase in N (often due to codigestion or costorage) while negative numbers reflect a loss of N. Mixed systems refers to the use of more than one of the denoted MMS.

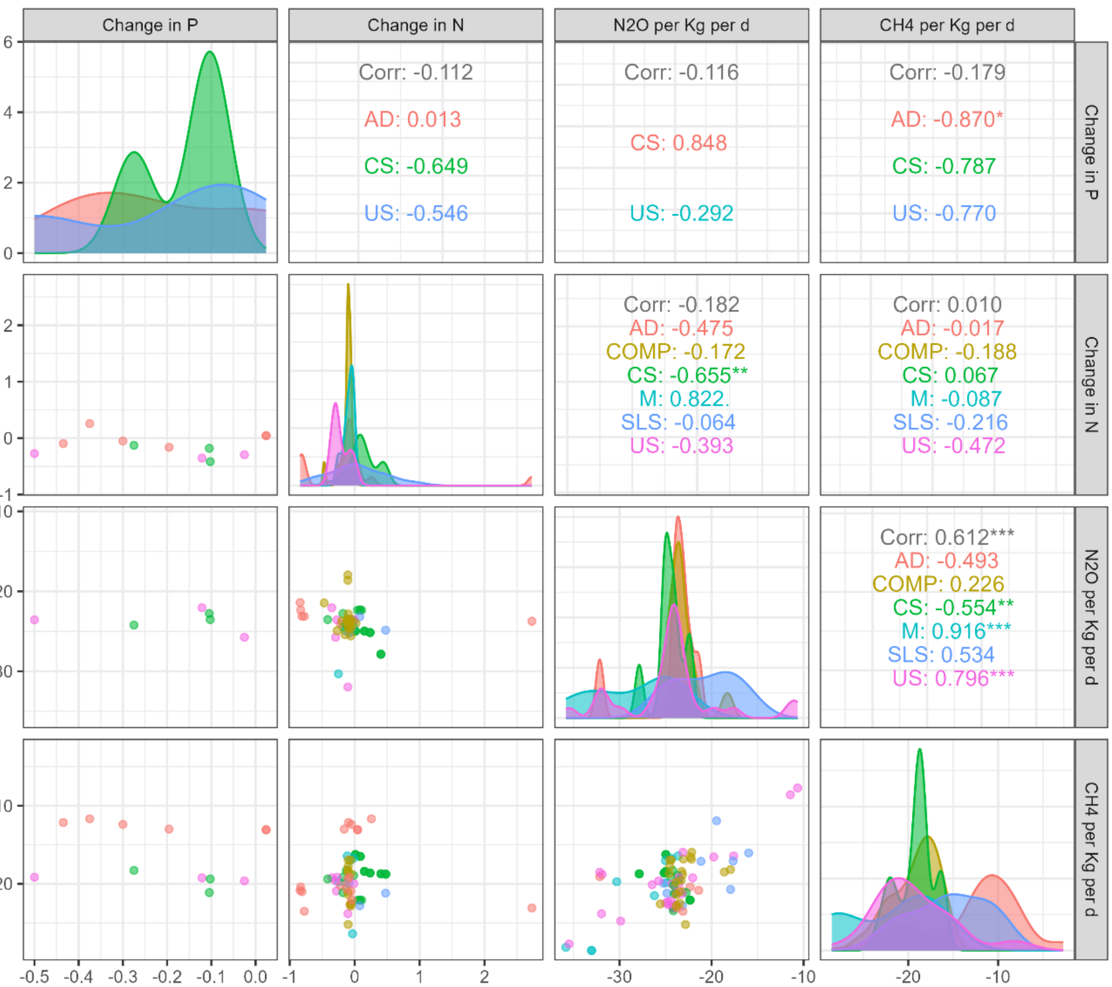

Figure 3-4. Correlation between the percent changes in P and N, and the changes in N_2O and CH_4 for each MSID denoting significance in dually mitigating ability via asterisks. MMS are represented by colors in each density plot to define the correlations. The color assignments are as follows: correlations coefficient in black, anerobic digestion in red, composting in yellow, covered storage in green, mixed use in teal, solid-liquid separation in blue, and uncovered storage in pink. The exception to this distinction is found in the correlation between N_2O per Kg per d and the change in P, where covered storage is red and uncovered storage is teal.

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GENERAL CONCLUSION

In light of the pressing challenges presented by the expanding population and changing climate, animal agriculture needs to adapt to preserve the production capacity while minimizing its impacts on the natural environment. Toward this goal, our study focused on the exploration of MMS efficacy and the design of cost-share programs at supporting MMS adoption on livestock operations. The past decades have been integral in demonstrating the urgency of climate change and creating the definition of climate-smart agriculture. The goal of this work was to explore how attributed MMS pollution reductions could be accurately estimated and identify how cost-share programs influence MMS adoption and long-term efficacy.

Available data in the literature were used to determine emissions coefficients that could be individually attributed to MMS on the basis of species and location. The data pool was largely incomplete and will require the collection of comprehensive MMS efficacy data across varying species and locations is needed for accurate estimates. A key takeaway from this analysis is that MMS hold theoretical synergistic reduction potentials that could combat against pollution tradeoffs; however, this synergy is unlikely to be realized without specific emphasis and incentive.

The analysis of historical cost-share data demonstrated that MMS currently implemented as a part of water quality improvement programs do not often result in efficient reductions in GHGe. This may be due to the outdated MMS reduction potentials in the literature informing MMS implementation decisions for cost-share programs; however, it is more likely due to the fact that existing programs have not focused on GHGe benefits. For MMS to improve both air and water quality targets, programs must be specifically designed to work toward these goals concurrently and account for the likely increased mitigation price associated with implementing nuanced MMS.

Continued progress towards GHGe neutrality and industry sustainability will require more work highlighting effective MMS and creating robust cost-share programs built for MMS longevity as the price of pollution reduction increases. Future work will need to redirect efforts towards the integration of climate-smart practices into animal agriculture accounting for the unique differences of each MMS.