ABSTRACT

AGHDAM, SOMAYEH. Environmental Impact Assessment (LCA) and Techno-Economic Assessment (TEA) of Struvite Recovery in Swine Manure (Under the direction of Dr. Mahmoud Sharara).

North Carolina is the 3rd U.S. swine production state with more than 2,500 swine farms. These farms produce a significant amount of manure. To make swine production more environmentally sustainable, adopting alternative manure management technologies is essential to reduce nutrient accumulation and runoff in dense production regions. This study evaluates the environmental impacts of swine manure nutrient recovery in the form of struvite ($NH_4MgPO_4 \cdot 6H_2O$) under North Carolina production conditions using life cycle assessment (LCA) methodology. Also, techno-economic assessment was conducted to investigate the economic feasibility of adopting this technology using the net present value (NPV) analysis. This evaluation assessed struvite recovery as a proposed nutrient concentration strategy at two different farm scales, i.e., small (3,000 head farm) and large (6,000 head farm) to reflect the size distribution of N.C. swine farms. Uncertainty analyses were performed to evaluate the impacts of different production and technology parameters and operating variables on performance of proposed struvite recovery process when compared to conventional management. In addition, geospatial analysis employed to identify farms that could benefit more from implementing the technology. The environmental assessment indicated that adopting struvite recovery technology reduced the eutrophication impacts of swine manure management while simultaneously increasing other environmental impacts (namely global warming potential, resource depletion, and acidification). However, implementing this technology still can be beneficial for phosphorus- sensitive regions. Economic assessment showed that under current pricing and market, adopting the technology is not feasible, but several modifications can be done to improve the condition. Also, under nutrient trade programs the technology can be offered as a phosphorus reduction and export tool.

Keywords: Swine manure, Lagoon-Sprayfield System, Phosphorus, Struvite, Life Cycle Assessment (LCA), Techno-economic Assessment (TEA)

Environmental Impact Assessment (LCA) and Techno-Economic Assessment (TEA) of Struvite Recovery in Swine Manure

by
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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

1. INTRODUCTION

1.1. Background

North Carolina is the third US state in swine production after Iowa and Minnesota. In December 2020, there were more than 8.8 million pigs in production on 2,500 swine farms, with farm capacities ranging from 700 to 10,000 swine (DEQ, 2021; USDA NASS, 2020). An expansion primarily took place between 1990 and 1995 with an increase of the NC hog population between those years from 2.8 million to 8.3 million head (Furuseth, 1997). This growth was primarily centered on few counties in Eastern North Carolina: Duplin, Sampson, Bladen, and Robeson counties (DEQ, 2021). This expansion relied on the lagoon spray field system to store, treat, and utilize the generated manures. As a result of the combined production density and manure disposal method, North Carolina's swine producers are facing several social and environmental issues. These problems are primarily associated with manure impacts including odor (Horton et al., 2009; Schiffman et al., 2000; Vukina et al., 1996), air emission, and water contamination with nutrients and bacteria/pathogens. It is expected that these issues will intensify as climate changes. In particular, since most North Carolina swine operations are within the coastal plains regions which is susceptible to hurricanes and intense rainfall, the risk of flooding and inundation is particularly high (Schaffer-Smith et al., 2020).

1.1.1. NC Swine Manure Management

Considering the amount of manure each animal generates, which is 20-30 liters daily, the importance of adopting sustainable manure management practices is clear. In North Carolina, swine manure management relies on anaerobic lagoons (open lagoons). The excreted manure (both solid and liquid portions) is transferred to the lagoon by flushing animal houses with recycled liquid through flushing pipelines. In the lagoon, the solid part settles by gravity and starts to break down while the supernatant (liquid portion) is used as a source of nutrient through land application on farms' cropland. Anaerobic decomposition of manure in open lagoon is a treatment practice since it reduces organic matter concentration in the slurry. Also, the nitrogen content of manure is reduced through ammonia volatilization from lagoon open surface. The phosphorus, majority of which is bound in manure solids, settles into the sludge zone of the lagoon. The sludge zone grows over time with the lagoon use and stays for 25 to 30 years until it is cleaned out. As a result, the lagoon lowers the concentration of nitrogen and phosphorus in the effluent from the originally excreted manure. In addition, since the lagoon supernatant is low in solids (less than 1%), it is typically irrigated on fields using spray irrigation systems. This further reduces the amount of nitrogen that reaches the soil and can be useable by growing crops or grasses.

While the current practice of treating swine manure is economical and simple to operate, it has been associated with several environmental and social concerns such as odor, insects and rodents, greenhouse gas emissions, pathogens, and contamination of the surface and groundwater due to rainfall and flooding (Amini et al., 2017; Vanotti et al., 2009). These problems are more notable in regions with a higher concentration of pig production.

1.1.2. Problems

In this section, key environmental and social concerns associated with the current manure management practices are briefly described.

1.1.2.1. Environmental concerns

A key pathway that animal manure contributes to air pollution is through the volatilization of free ammonia from swine barns, manure storage (lagoon), and during application to fields.

Concentrated animal feeding operations (CAFOs) are considered one of the major sources of ammonia emission (Hristov et al., 2011). During storage in the lagoon, the urea in the animal urine decomposes forming ammonia (NH_3), while the carbohydrates in the manure are hydrolyzed into volatile fatty acids (VFAs) that further decompose forming methane (CH_4) and carbon dioxide (CO_2) (Dong et al., 2006). Although other gases like nitrous oxide (N_2O), hydrogen sulfide (H_2S), and volatile organic compounds (VOCs) are emitted from the lagoon surface, both CO_2 and NH_3 are the main gaseous emissions from the surface of the lagoons (Liu et al., 2014). Several of the lagoon emission gases, i.e., CH_4, N_2O, and CO_2 are considered greenhouse gases (GHG), which are associated increased radiative forcing (the change in energy flux in the atmosphere) resulting climate change. The amount of methane emission in North Carolina's swine farm was estimated to be about 33.4 kg head^{-1} year^{-1} (Dong et al., 2006).

Nitrogen and Ammonia Emissions

The rate of NH_3 emissions from an open swine lagoon can vary drastically (Arogo et al., 2003). In their review of ammonia emission studies, Arogo et al. (2003) reported swine lagoon ammonia emissions ranging from 0.6 to 158 kg of N-NH_3 per hectare of lagoon area per day. Ammonia gas has a strong, unpleasant smell and in combination with H_2S and other volatile organic compounds (VOCs), is responsible for the malodorous emissions associated with swine farms. In addition to odor emissions, ammonia is associated with formation of fine particulate matter ($PM_{2.5}$) which impacts visibility and has severe negative health impacts. Different models to simulate lagoon ammonia emissions have been developed (Aneja et al., 2001; Montes et al., 2009; Visscher et al., 2002). Models indicate an increase in ammonia emission fluxes with increased lagoon temperature, pH, total ammoniac nitrogen (TAN), and wind speed. As a result, lagoon ammonia emissions exhibit a time-based pattern that reflects the seasonal weather changes.

On average, only 50% of lagoon supernatant (effluent) used in crop irrigation is taken up by the crop, with the remainder either being lost to the atmosphere as NH_3 and other nitrogen forms (N_2O, dinitrogen (N_2)) or lost to water primarily as nitrates (NO_3) (Langevin et al., 2010). The estimated amount for emissions are about 24%, 5%, and 8% of total nitrogen for NH_3, NO, and N_2O, respectively. These emissions are hazardous for ecosystems and human health (Galloway and Cowling, 2002; Smith et al., 2000; Buckley et al., 2009).

Most crops need nutrients in ratios different from their concentration in manures. In particular, phosphorus-to-nitrogen ratio (N:P) in animal manure are typically higher than that required by crops. According to (Shehu et al., 2019) N:P ratio needed in a typical crop like maize is 6: 1, while the average N:P ratio in swine manure is 2:1 to 4: 1. As a result, nitrogen-based manure application leads to excess phosphorus application (D. Cordell et al., 2011). Over-applying nutrients is likely to occur particularly in areas with high manure availability (dense animal production regions) such as North Carolina.

Outside of animal production systems, crop phosphorus demand is met by phosphorus fertilizers derived from phosphate rock. This resource is a non-renewable resource, as it relies on a mined resource, which may be depleted for 50–100 years while phosphorus global demand is increasing (Dana Cordell et al., 2009). Although swine manure can be used as fertilizer, intensive swine operations limit this opportunity since areas with higher manure production have less land available to efficiently utilize the manure as fertilizer (Loyon, 2017). In regions with intensive swine operations like North Carolina, excess manure needs to be transported to farther cropland for efficient use. In a recent study, Spiegal et al. (2020) modeled the size of region required to efficiently utilize manure nutrients and showed the Carolinas (North and South Carolina) to require the largest transportation distance to efficiently distribute regional manure phosphorus. Because of the cost associated with manure transportation, particularly as a liquid source, and the associated odor issues, manure is not transferred sufficiently far from the originating farm (Taos, Fattah, & Huchzermeier, 2016). Effective use of manure as a nutrient source is an opportunity to reduce commercial fertilizer consumption. In this case, not only a scarce resource such as phosphorus rocks is preserved, but also several negative environmental impacts associated with overapplying manure-based P and manufacturing commercial P fertilizers can be avoided.

Water pollution, both ground and surface waters, is another environmental concern associated with the current practice. Manure organic matter and nutrients can be delivered to waterbodies through different mechanism: runoff of liquid manure over soil surface, sediment erosion, leaching through soil profile to groundwater, aerosol drift, and both wet and dry deposition of volatilized ammonia and particulate matter. Manure organic matter loading to water bodies, as either dissolved or particulate matter, reduce oxygen levels in waterbodies. This can shift aquatic environments to anaerobic conditions leading to odorous emissions and fish kills.

Increased concentration of phosphorus and nitrogen in surface waters can increase eutrophication risks (Pedizzi et al., 2018; Nelson et al., 2003).In addition, excess nutrients applied can contribute to groundwater contamination. Nitrate (NO_3) is one of the most critical contaminants in groundwater. Studies have shown that 4.4% of private wells in the United States had nitrate concentrations greater than the EPA maximum contaminant level (10 mg l^{-1} as N). This percentage is significantly higher (22%) in private wells near agricultural areas (Dubrovsky, 2010). Nitrate is a common contaminant of drinking water in many agricultural areas and it is linked to several health problems such as cancers and specific birth defects (Zirkle et al., 2016).

1.1.2.2. Social concerns

Recently, environmental nuisance issues such as malodorous emissions, pest, and insect, increased complaints and lawsuits by neighbors. These complaints are not only related farms' activities affecting their health and welfare, but also the value of their properties decrease considering the unpleasant odor and its attached concerns. In North Carolina, lawsuits have resulted in judgments for millions of dollars awarded to neighboring communities in compensation and damages with other lawsuits pending (Associated Press, 2020).

Adopting new manure treatment technologies and systems that convert manure phosphorus and nitrogen into a valuable product could improve social and environmental impacts of swine production. Among the technologies investigated in this regard is recovering nutrients (mostly phosphorus and nitrogen) from liquid portion of manure in the form of struvite. This technology has been selected for the focus of this thesis investigation since swine farms in North Carolina generate large volumes of manure primarily as liquid (excreted manure is 92% water, in addition to rainfall dilution of lagoon liquid). Considering the limitation in global natural resources of phosphate rock, reusing, and recovering phosphorus is a sustainable (Linderholm et al., 2012). As mentioned earlier, using lagoon effluent for crop irrigation, usually results in phosphorus loss. Reusing this valuable nutrient will not only prevent environmental impacts downstream, but also could reduce (or avoid) occurring a variety of environmental impact at upstream by avoiding extraction of finite phosphorus rocks and impacts related to manufacturing the commercial phosphorus fertilizers. At the same time, the need for mineral nitrogen and phosphorus fertilizers and their production cost is steadily growing, which could encourage the agricultural community to get more involved to recover and reuse nutrients from manure (Pedizzi et al., 2018). Struvite is a promising slow-release fertilizer, with higher purity and lower heavy metal content than commercial phosphate fertilizers.

Although using struvite recovery technology is a rare option in the agricultural industry sector, it is a well-established technology for removing struvite from municipal wastewater treatment plants MWWTP. For instance, a full- scale facility to treat municipal wastewater was put into operation in Oregon in 2009, and the produced fertilizer product is currently being marketed throughout North America (Ostara Nutrient Recovery Technologies).

1.1.3. Struvite Technology

Struvite (magnesium ammonium phosphate, $NH_4MgPO_4 \cdot 6H_2O$) is an orthophosphate mineral, containing magnesium, ammonium, and phosphate in equal molar concentrations Struvite forms in a 1:1:1 molar ratio following the equation (11) (with n= 0, 1, or 2)(Jordaan et al., 2010; K. Le Corre et al., 2009; Md Mukhlesur Rahman et al., 2014; Shih & Yan, 2016):

$$Mg^{2+} + NH_4{}^{+} + H_n\, PO_4{}^{3-n} + 6H_2O \rightarrow MgNH_4PO_4.6H_2O + nH^{+} \qquad (11)$$

Struvite is a slow-release fertilizer with no odors. Since it has low solubility in water compared to other phosphorus fertilizers, it can significantly reduce field losses of phosphorus (Zhang et al., 2012)

A continuous flow crystallizer is used to combine soluble phosphorus, nitrogen, and magnesium in the manure into struvite crystals. In this method, a bed of struvite seed crystals is initially added to the reactor. Increasing pH is necessary to encourage struvite deposition around struvite seeds, which is achieved by adding magnesium (Mg) or other bases like sodium hydroxide (NaOH) (Westerman et al., 2009). This is a great manure management alternative when there is not enough farmland available to assimilate manure nutrients or when manure application fields are in watersheds with phosphorus loading problems (freshwater eutrophication). Nutrient capture rate in struvite is 70-95 % for phosphorus and 70- 90 % for nitrogen (Amini et al., 2017). Nutrient-rich product (struvite) is economical to transport into long distances and be used efficiently.

1.2. Rational

Regarding struvite recovery system, this technology has been successfully operated in municipal waste water treatment plant (MWWTP), considering the similarity in wastewater and manure contents, the assumption is that it could be suitable for animal manure treatment as well. Also, it has been successfully installed and operated for recovering struvite from swine manure in Japan since 1998. In Europe, a full-scale plant in Belgium has also been installed and the struvite successfully recovered from treating anaerobic lagoon effluent and approved for agricultural use (Moerman et al., 2009). The motivation behind this project is that although this technology is mature, it is not being used in North Carolina's swine farms because of insufficient knowledge about economic and environmental impacts associated with its adoption. This study aimed at filling this knowledge gap.

1.3. Research Objectives

The overall goal of this research is to assess the feasibility and environmental impacts of struvite recovery as a manure management technology on North Carolina swine farms and identify critical factors that impact adoption**.** This research assumes that: (1) lack of complete data related on impacts of technology implementation in North Carolina swine farms is a barrier for its adoption; (2) implementing this technology can reduce adverse environmental impacts without a significant economic burden; and (3) the main parameters required for decision-making process (e.g. regulations, price of fertilizer and equipment, etc.) are changing constantly and need to be integrated into the technology assessment.

2. MATERIALS AND METHODS

2.1. Environmental Impacts Assessment (Life Cycle Assessment (LCA))

Life cycle assessment (LCA) is a method used to quantify the environmental impacts of a product or process throughout its entire life cycle (Hauschild, 2018). The LCA method consists of four stages: [1] goal and scope definition, [2] inventory analysis, [3] impact assessment, and [4] interpretation. This method has been used to compare technologies and processes in numerous sectors including construction, bioprocessing, waste processing, cropping, and animal production sectors. The following sections will present, in details, the scenarios and modelling approach used to study the struvite production technology.

2.1.1. Goal and Scope Definition

The goal of this study is to evaluate the comparative environmental impacts associated with two manure management options (scenario) on North Carolina swine farms. The considered scenarios focus primarily on nutrient recovery and utilization.

2.1.1.1. Evaluation Scenarios

The first scenario (baseline scenario) (**Figure 1**) represents current practices on swine farms, which consist of manure storage and treatment in an open lagoon where the lagoon supernatant is land applied to spray fields for beneficial use by growing plants. The second scenario (struvite scenario), entails adding a phosphorus recovery (struvite precipitation) unit to the lagoon treatment system while maintaining the rest of the baseline scenario. Considering the particularities of NC swine sector, each of these two scenarios have been assessed with two cases: [1] where commercially valuable crop (cash crop) is grown on the spray field, and [2] Bermudagrass Hey is grown solely for its high nutrient intake. For case [1], manure is credited with replacing commercial fertilizers (will be labelled *As Fertilizer* or (AF) sub-scenario), while in the second sub-scenario where no fertilizer replacement is credited (will be labelled *Not As Fertilizer* (NAF) sub-scenario). The system boundary encompasses "gate-to-grave" covering production, storage, and utilization of the manure. The boundary of the scenarios with inputs and outputs of the systems is depicted in **Figure 1**. Feed and animal production-related impacts and credits are not included in the scope of this assessment.

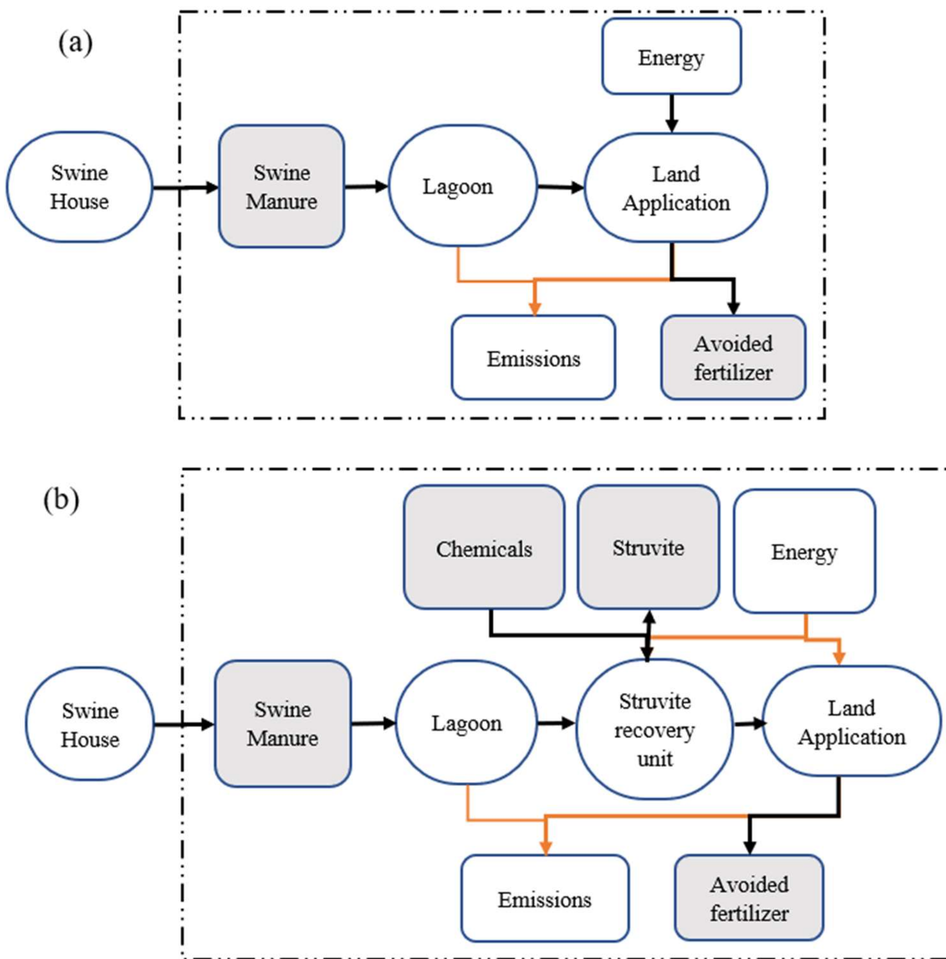

Figure 1. System boundary for (a) the baseline scenario (b) the struvite scenario

2.1.1.1.1. Farm scales selected for study

According to the North Carolina Department of Environmental Quality (N.C. DEQ), there are 2,634 permitted swine facilities in the state including 1,249 feeder-to-finish farms (47%). **Figure 2** shows location of feeder-to-finish swine farms in North Carolina and **Figure 3** illustrates the cumulative distribution of feeder-to-finish farms by number of raised animals (size). Approximately 20% of feeder-to-finish farms have between 2,250 to 3,000 heads animal. Also, 3,000 heads swine farm represents the 45th percentile for farm size in NC, so it was taken as a proxy for small farms. On the other hand, a 6,000 heads farm size, which corresponds to the 80th percentile, was used to represent large farms.

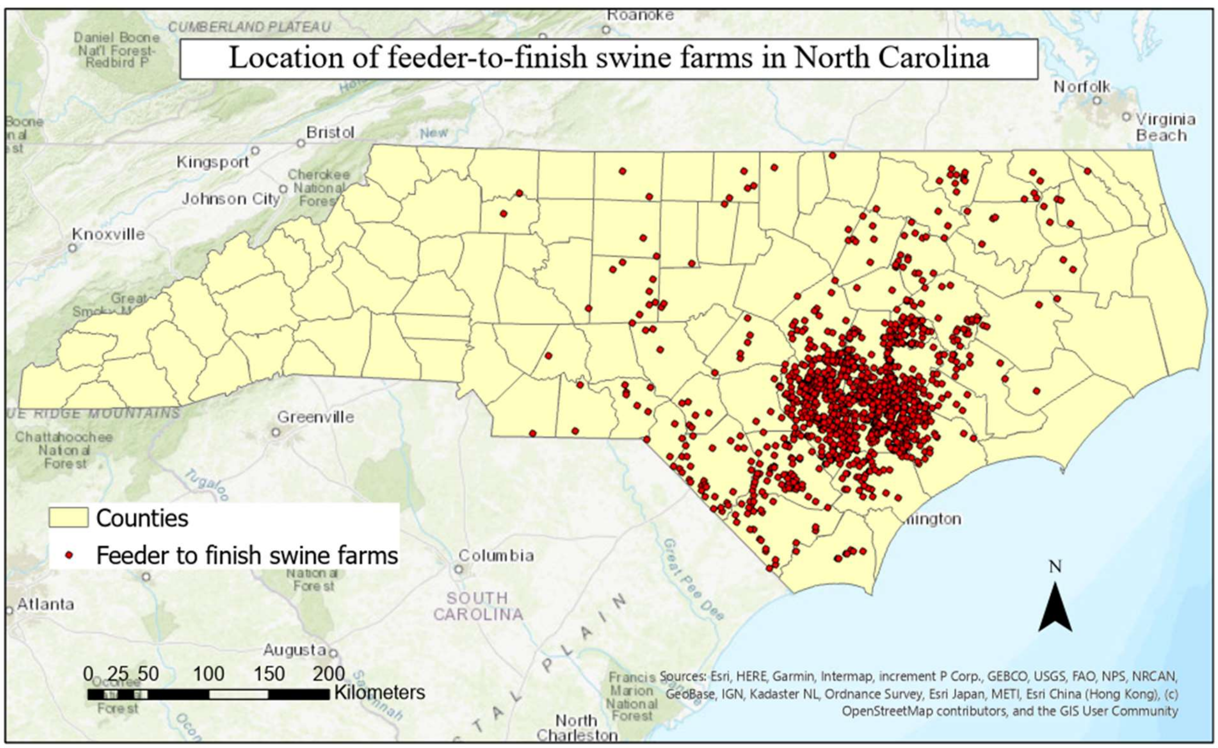

Figure 2. Location of feeder-to-finish swine farms in North Carolina

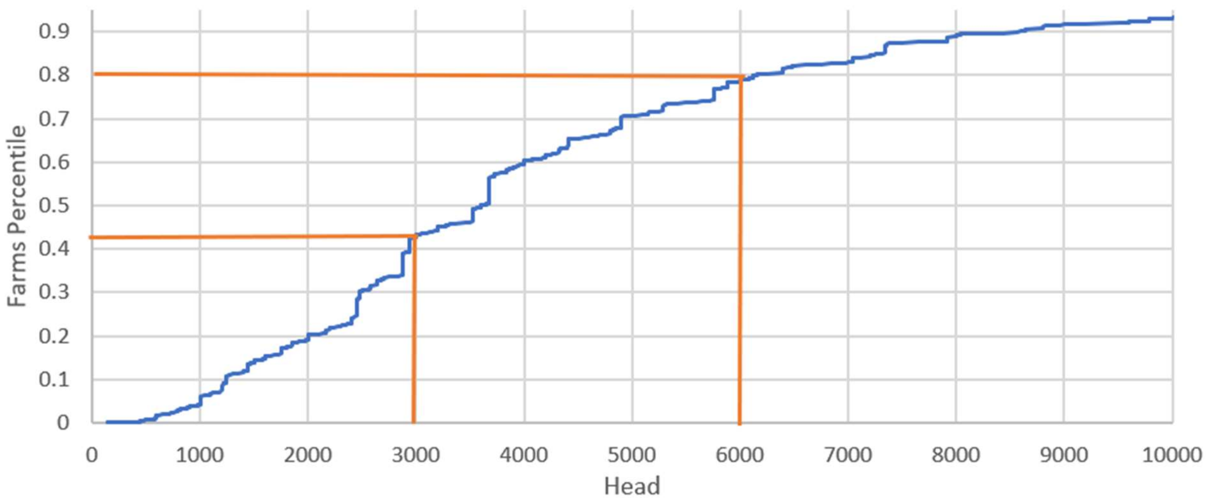

Figure 3. Cumulative distribution of finishing swine farm size in NC

2.1.1.2. Functional Unit and Reference Flow

The system function is managing swine waste on a feeder-to-finish (23 kg to 127 kg). Management here refers to storage, treatment, and utilization of manure generated on farm during one calendar year. Therefore, the functional unit (FU) is unit volume of manure managed in one farm over one year. In this phase, we assumed that a group of swine (referred to as a turn) will be placed for 120 days, which corresponds to 3 turns per year (ASAE, 2000). The amount of manure considered as reference flow is calculated using equation (21)

$$RF = \frac{\#\, finished\ animals}{farm} \times \frac{\#\, turn}{year} \times \frac{\#\, days}{turns} \times \frac{Manure\ (liter)}{finished\ animal\ \times day} = \frac{liter}{farm \times year} \tag{21}$$

Manure production for feeder-finisher hog varies from 4.5 to 8.3 liter (L) per animal per day. This variability is attributed to diet variations as well as age and data source, i.e., ration-based models or field data (ASAE, 2000; Amini, 2014; Barker et al., 2002; and Wiens et al., 2008). In this study, a manure production volume of 7-liter per animal per day will be used as reference flow:

$$RF\ (small\ farms) = 3{,}000 \times\ 3 \times\ 120 \times 7 = 75.6 \times 10^5\ L$$

$$RF\ (large\ farms) = 6{,}000 \times\ 3 \times\ 120 \times 7 = 151.2 \times 10^5\ L$$

2.1.2. Inventory Analysis

An Inventory of all inputs and outputs associated with manure management in each scenario was developed. Required data in this section was divided into two categories: <u>(1) systems' input and output along with their quantity</u>, <u>(2) the environmental impact associated with each input and output.</u> Data required for inventory development was compiled and generated using peer-reviewed literature and existing databases such as ecoinvent through the LCA software (Simapro). An overview of system inputs and outputs is provided in the following sections.

2.1.2.1. Manure storage and treatment

Manure characteristics, such as nutrient concentration, vary widely based on diet, stage of life cycle, manure management system, and weather conditions. Nutrient concentration is important for this study as it is critical to estimating potential yield of struvite product. **Table 1** shows a summary of nutrient concentrations in swine manure as characterized in different studies.

Table 1. Summary of literature data on concentration of nitrogen and phosphorus in swine manure at different production stages

Manure type	Phosphorus (mg l^{-1})	Nitrogen (mg l^{-1})	Reference
As excreted	1,356	8,387	(ASAE, 2000)
Liquid manure- after solid-Liquid separation	182	1,212	(Huang et al., 2011)
Raw manure	50-75	NA	(Amini, 2014)
	380-1,680	2,300-5,700	(Piveteau et al., 2017)
	169	5,000	(Bayo et al., 2012)
	990	NA	(Prapaspongsa et al., 2010)
	NA	3600	(Chadwick et al., 2011)
	89-189	1380-2000	(Zhang et al., 2012)
Screened manure	145	532	(Suzuki et al., 2007)
Flushed manure	368	1066	(Hill & Baier, 2000)
	270	1293	(Vanotti et al., 2009)
	66	673	(Vanotti et al., 2009)
	52	587	(Philip W Westerman et al., 2008)
	572	NA	(Burns et al., 2001)
Liquid manure	524-540	1380-1440	(Çelen et al., 2007)
	60	3809	(Rahman et al., 2011)

As evident from **Table 1**, the nutrient concentration in swine manure vary noticeably. Even for the same farm, manure composition varies seasonally and with the shift in diet and animal age. To overcome this challenge, manure composition data representative of N.C. production conditions is critical. In addition to nutrient concentration (nitrogen and phosphorus), the manure pH and Magnesium (Mg) concentration are essential to study struvite recovery systems and estimate required reagents for the precipitation process. **Table 2** presents two datasets representative of NC swine farm manure composition. The first set is from a study conducted by Westerman et al. (2010) where they investigated nutrient concentration variations in anaerobic lagoons through sampling 20 farms, including 8 finishing farms, over a 4-year period from 2002 to 2005. The second dataset was provided by the Agronomic Division of the North Carolina Department of Agriculture and consumer services (NCDA&CS), representing the period from 2013 to 2019. This dataset (about 70,000 samples) encompasses analysis results for samples submitted by producers and consultants for analysis to guide the application rates of swine lagoon supernatant, i.e., nutrient management planning (NMP).

Table 2. Means and standard deviations (between brackets) for swine manure nutrient concentration and pH

Characteristics	(Westerman et al., 2010)	NCDA&CS Analyses
Nitrogen (mg l^{-1})	587 (170)	343 (254)
Phosphorus (mg l^{-1})	51.7 (15.2)	51.7 (244)
Mg (mg l^{-1})	45 (11)	43.7 (130)
NH3-N (mg l^{-1})	544 (181)	NA
pH (-)	7.75 (0.18)	7.5 (0.4)

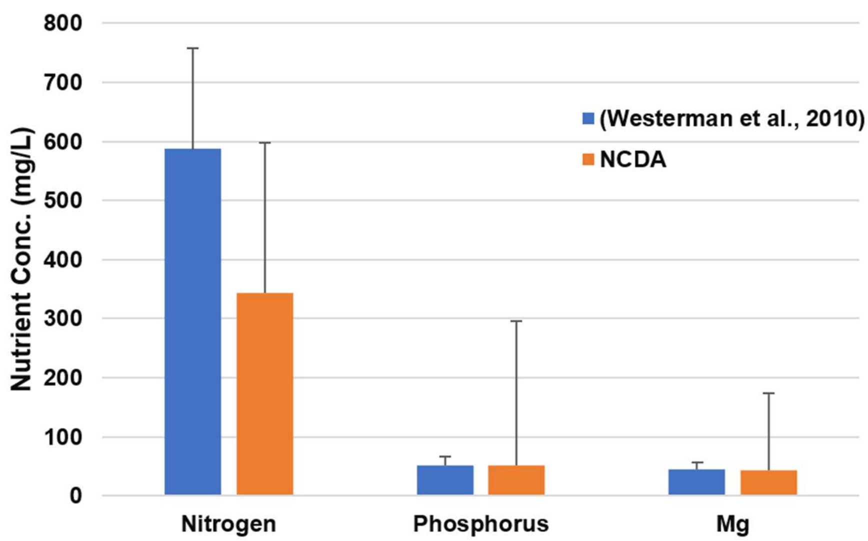

Figure 4. Comparison of nutrient concentration in swine manure in two datasets

In both datasets, the mean P concentration was 51.7 mg l^{-1}, while the mean Mg concentration was slightly higher in the Westerman et al. (2010) dataset (about 3%). In the case of nitrogen, the Westerman et al. (2010) dataset showed a higher concentration that could be attributed to farm types represented in the manure samples submitted to NCDA&CS labs, while the Westerman et al. (2010) dataset is exclusively from finishing farms. Since our study focused on NC finishing swine farms, the data reported by Westerman et al. (2010) will be utilized for further calculations and assumptions. However, the NCDA&CS dataset served as a quality check to confirm the temporal representativeness of the Westerman et al. (2010) dataset.

2.1.2.1.1. Air Emission

Methane (CH_4) emissions: Liquid manure undergoes anaerobic decomposition during lagoon storage and treatment, releasing significant amounts of methane (CH_4) and carbon dioxide (CO_2), in addition to ammonia (NH_3) and other volatile organic compounds (VOCs) (Hongmin Dong, Joe Mangino, 2006). The CH_4 emission factor from manure in open lagoons was calculated based on the Tier 2 method from the equation (22) adopted from the latest updated IPCC report (2019).

$$CH_4 - EF_{(T)} = \left(VS_{(T)} * 365\right) * \left[B_{o(T)} * 0.67\frac{kg}{m^3} * \sum_{S,k}\frac{MCF_{S,k}}{100} * AWMS_{(T,S,k)}\right] \tag{22}$$

where CH_4-$EF_{(T)}$ is the annual CH_4 emission factor for livestock category *T* (kg CH_4 animal $^{-1}$ yr^{-1}), $VS_{(T)}$ is daily volatile solid excreted for livestock category *T* (kg dry matter animal^{-1} day^{-1}), *365* is the basis for calculating annual VS production (day. year^{-1}), $B_{o(T)}$ is maximum CH_4 producing capacity for manure produced by livestock category *T* (m^3 CH_4 kg^{-1} of VS excreted), *0.67* is the density of CH_4 in units of m^3 per kg of CH_4, $MCF_{(S,k)}$ is CH_4 conversion factors for each manure management system, *S*, by climate region, *k* (%), and $AWMS_{(T,S,k)}$ is a fraction of livestock category "s manure handling using a manure management system, *S*, in climate region k, dimensionless.

For this study, *VS* was calculated using equation (23) (IPCC, 2019).

$$VS_{(T,P)} = \left(VS_{rate(T,P)} \times \frac{TAM_{\ T,P}}{1000}\right) \times 365 \tag{23}$$

where $VS_{(T,\ P)}$ is annual VS excretion for livestock category T with system productivity of P (kg VS animal $^{-1}$ yr^{-1}), $VS_{rate\ (T,\ P)}$ is default VS excretion rate for livestock category T with system productivity of P (kg VS (1,000 kg animal mass)$^{-1}$ day^{-1}), and $TAM_{(T,\ P)}$ is typical animal mass for livestock category T with productivity of P (kg animal $^{-1}$). Based on range presented in the report, the TAM considered 75 kg animal $^{-1}$ and VS rate is 3.9 kg (1000 kg animal mass)$^{-1}$ day^{-1}. Thus, VS for North Carolina finishing swine farms is 107 kg animal^{-1} yr^{-1} or 0.293 kg animal^{-1} day^{-1}.

In equation (22), B_0 assumed 0.48 m^3 CH_4 kg_{VS}^{-1} excreted and MCF assumed to be 73%. Since the anaerobic lagoons are the only manure management method in farms, the AWMS equals to 1. Finally, the CH_4 emission factor was calculated as follow:

$$EF_{(T)} = (0.29 * 365) * \left[0.48 * 0.67 \frac{kg}{m^3} * \frac{73}{100}\right] = 25.06 \ \frac{Kg\ CH_4}{animal.year}$$

<u>Nitrous Oxide (N_2O) direct emission:</u> The production of direct N_2O emissions from livestock manure depends on the composition of the manure (manure includes both feces and urine) (EPA, 2019). The amount of direct N_2O emission is calculated based on the equation (4) (IPCC, 2019).

$$N_2O(direct) - EF_{(T)} = \left[\sum \left[\sum_{T,P} \left(\left(N_{(T,P)} \times \; Nex_{\;(T,P)} \right) \times AWMS_{(T,S,P)} \right) + N_{cdg(s)} \right] \times EF_{3(S)} \right] \times \frac{44}{28} \qquad (4)$$

where, $N_2O(direct)\text{-}EF_{(T)}$ is direct emission from manure management in the country (kg N_2O yr $^{-1}$), $N_{(T,\,P)}$ is the number of head of livestock species /category T in the country, for productivity P, Nex $_{(T,\;P)}$ is annual average N excretion per head of species/category T in the country, for productivity P (kg N animal $^{-1}$ yr $^{-1}$), $N_{cdg(s)}$ is annual nitrogen input via co-digestate in the country (kg N yr^{-1}), AWMS is a fraction of total annual nitrogen excretion for each livestock species/category, $EF_{3(S)}$ is emission factor for direct N_2O emissions from manure management system S in the country (kg N_2O-N (kg N)$^{-1}$) in manure management system S, S is manure management, and T is species. Category of livestock.

The N_2O emission factor (EF) for uncovered anaerobic lagoon is 0, resulting in zero direct N_2O emission from lagoon operation. Also, Pratt et al. (2015) and Wang et al. (2017) estimated that the N_2O emission from lagoon liquid is negligible.

N_2O indirect emission (NH_3 and NO_x emission): The amount of indirect N_2O emission due to the volatilization of N from manure management is calculated using the equation (5) (IPCC, 2019).

$$N_2\,O(indirect) - EF = \left(N_{valatilization\_MMS} \times \; EF_{\;4} \right) \times \frac{44}{28} \qquad (5)$$

where, *$N_2O(indirect)$-EF* is the emission factor for the indirect N_2O emissions due to N volatilization and transformation after manure spraying and field application (kg N_2O yr^{-1}), EF_4 is emission factor for N_2O emission from atmospheric deposition of nitrogen on soil and water surfaces (kg NH_3-N + NO_x-N volatilized)$^{-1}$, and $N_{volatilization\_MMS}$ is the amount of manure nitrogen that is lost due to volatilization of NH_3 and NO_X (kg N yr^{-1}). According to table 11.3, EF_4 is 0.01. The $N_{volatilization\_MMS}$ is calculated using equation (6).

$$N_{volatilization\_MMS} = \left[\sum\left[\sum_{T,P}\left(\left(N_{(T,P)} \times Nex_{(T,P)}\right) \times AWMS_{(T,S,P)}\right) + N_{cdg(s)} \times Frac_{GasMS(T,S)}\right]\right] \quad (6)$$

where, $N_{volatilization\_MMS}$ is the amount of manure nitrogen that is lost due to volatilization of NH_3 and NO_x (kg N yr^{-1}), $N_{(T,P)}$ is the number of livestock species/category, for productivity system P, $Nex_{(T,P)}$ is annual average N excretion per head of species/category T, for productivity system P (kg N $animal^{-1}$ yr^{-1}), $N_{cdg(s)}$ is the amount of nitrogen from co-digestates added to biogas plants such as food waste or purpose-grown crops, P is productivity class, high or low, $AWMS_{(T,P)}$ is the fraction of total annual nitrogen excretion for each livestock species/ category T that is managed in manure management system S (dimensionless), and $Frac_{gasMS(T,P)}$ is the fraction of managed manure nitrogen for livestock category T that volatilizes as NH_3 and NO_X in the manure management S.

$Frac_{gasMS(T,P)}$ for uncovered anaerobic lagoons is equals to 0.4 according to table 10.22 and $Nex_{(T,P)}$ could be calculated using the equation (7):

$$Nex_{(T)} = \left(N_{intake\,(T)} \times N_{retention\,(T)}\right) \times 365 \quad (7)$$

For this study, the default amount of nitrogen excretion rate is 0.46 kg N (1,000 kg animal mass)$^{-1}$ day^{-1}. By considering 75 kg average weight for each animal, the excretion rate becomes 12.6 N animal $^{-1}$ year $^{-1}$. The amount of manure nitrogen that is lost due to volatilization of NH_3 and NO_X is 5.037 kg N animal $^{-1}$yr^{-1}. Finally, by adding findings into the equation (5), the total N_2O indirect emission is 0.079 kg N_2O animal $^{-1}$yr^{-1}.

$$N_2O = (5.037 \times\ 0.01) \times \frac{44}{28} = 0.079 \frac{kg\ N_2O}{animal.year}$$

2.1.2.2. Struvite Recovery Unit

Struvite can be synthesized in continuous or batch reactors (**Figure 5**). In general, batch reactors are simple to operate and they allow reactions to proceed homogeneously until completion. The continuous reactors do not ensure a complete reaction of all reactants. However, these reactors are more suitable to process a large volume of slurry such as wastewater or large swine farms (Lin, 2012). In the case of struvite recovery, continuous mode systems are generally preferable to batch systems because size, and hence capital cost, is small per unit struvite recovered. Batch systems may be better if the reaction rate is slow, which does not apply here (Bowers, 2002).

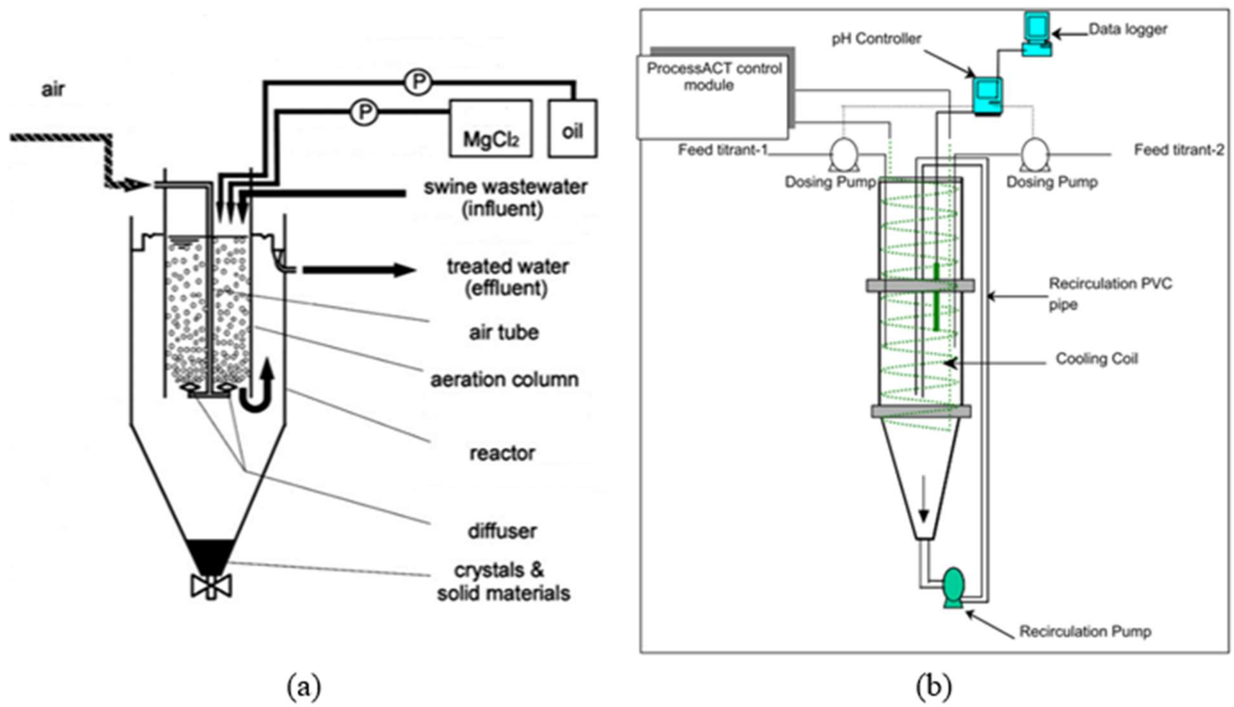

Figure 5. Struvite recovery reactor (a) continuous flow (Suzuki et al., 2007) (b) batch (Ali & Schneider, 2008)

For struvite precipitation, a fluidized bed reactor is more suitable because the reaction rate is high enough and it is easier to manage this type in the farm environment. For struvite precipitation process, a cone-shaped reactor is more suitable, with the increase in diameter with height, this shape results in a higher velocity of flow at the bottom in comparison to flow at the top (K. Bowers, 2002). This prevents sweeping very small bed particles with the exiting liquid (attrition) yet provides sufficient velocity to maintain larger particles in a fluidized state (K. Bowers, 2002) . A cone-shaped fluidized bed crystallizer was found to provide a high surface area into a small space and allows varying flow rate which is important for operate different rates and resistance to upsets caused by temporary high flow or loss of flow (Philip W. Westerman et al., 2009). The product collection is a critical part of the struvite production system. Simplicity is very important for reducing labor costs while ensuring high recovery rates.

The reactor volume is determined based on the design flow rate and required hydraulic retention time (HRT). The HRT is the length of time liquids stay in a reactor, equal to the reactor volume divided by the volumetric flow rate (K. Bowers, 2002). The HRT varies between studies from 10 minutes to 4 hours in studies at lab-scale (Burns et al., 2001; Çelen et al., 2007; Nelson et al., 2003; Md M. Rahman et al., 2011; Song et al., 2011), in compare with pilot-scale studies that were tested in the field. For instance, Amini (2014) run his tests under 8 minute HRT, while Westerman et al. (2009) designed the reactor based on 5 minute HRT.

Westerman et al. (2009) considered 6-minute residence time for the field-scale system. Scaling up the system increases in residence time. When scaling up the system, the cross-sectional area of the reactor (cone shape) needs to increase proportionally with the volumetric flow rate to ensure flow velocity range remains unchanged. This means the reactor diameter must be increased proportionally to the square root (the½2 power) of the volumetric flow rate while the height of the bed must increase proportionally with the diameter (the square root of the volumetric flow rate). As a result, the volume of the reactor needs to increase by the 1.5 power of the volumetric flow rate. Therefore, HRT (reactor volume/flow rate) increases with the square root of the flow rate (K. E. Bowers & Westerman, 2005). So, the HRT for this study is calculated as below:

The flow rate in reference study: 5 l min^{-1}, HRT in reference study: 6 min, flow rate in this study:14 l min^{-1}. Therefore, HRT in this study equals to $((14/5)^{0.5} * 6) = 10.85$ min.

The other aspects of the unit design are listed in the **Table 3** (Amini, 2014; Bowers, 2002; Westerman et al., 2009).

Table 3. Struvite construction LCI

Item	Unit	Value
Volume of reactor	m^3	0.156
Crystallizer (carbon steel)	kg	0.26
Stairs (steel)	kg	212
Catwalk /access platform (steel)	kg	2653
Connections, elbow, pipe, valve (Polyvinyl Chloride (PVC))	number of items	10
pH meter	number of items	1
Flowmeter	number of items	1
Concrete	m^3	0.25
Pump	number of items	2

2.1.2.2.1. Operation Factors

Several Physico-chemical factors impact the struvite precipitation process including the concentration of Mg and P (Mg:P molar ratio) in raw manure, pH, flow rate, aeration rate, and presence of Calcium (K. Le Corre et al., 2009). However, increasing the operational pH values and adjusting the molar ratios of magnesium and phosphate are the most important factors (Song et al., 2011). These factors will be reviewed in more detail in the following sections.

pH: Struvite precipitation is sensitive to pH since the activities of both NH_4 and PO_4^{3-} are pH-dependent (Nelson et al., 2003). Generally, with increases in pH, the solubility of struvite increases resulting in supersaturation conditions which subsequently raise the growth rate (K. Le Corre et al., 2009). Several studies investigated the effects of pH on struvite crystallization. Kabdaşlı et al. (2017) reported that decreasing pH reduces the speed of struvite particles forming and may affect the quality of formed particles. Furthermore, Matynia et al. (2006) indicated that pH can influence struvite crystal physical characteristics. They have shown that increasing pH from 8 to 11 could decrease struvite mean crystal size in synthetic solutions by 5 times. Finally, several studies proposed pH range between 8 to 9 to be optimal for the precipitation rate of struvite (Burns et al., 2001; Çelen et al., 2007; Jordaan et al., 2010; Stratful et al., 2001; Suzuki et al., 2007). Different approaches are used to control reaction pH including the addition of bases such as sodium hydroxide (NaOH) or ammonia (NH_3), or by CO_2 striping through aeration.

NaOH: Adding NaOH is the most common method for adjusting pH in the process of struvite precipitation. Çelen et al. (2007) Used NaOH for adjusting the pH with and without Mg addition. They showed that by increasing $MgCl_2$ for achieving a higher Mg:P molar ratio, the NaOH requirement increases as well. They used 560 mg l^{-1} in a sample that was set to 1:1 molar ratio in comparison to 777 mg l^{-1} in a sample with a 2:1 molar ration. In both samples, the initial pH was 6.8 and the goal was to achieve pH of 8. In another study, while the Mg:P ratio were constant, the tests were conducted to achieve different final pH levels (from initial 7.5 to 8.4, 8.7 and 9) resulting in increases in NaOH requirement to be 4, 7, and 10 ml per 1100 ml of sample manure (Nelson et al., 2003). Furthermore, Burns et al. (2001) added 30 and 50 ml NaOH to 400 ml sample for primary and secondary anaerobic swine pond respectively to increase pH to 9, While the initial pH was the same (6.9) and the same Mg:P ratio (1.6:1) were desired. The mentioned studies show that the amount of NaOH added varies with (a) different manure types, (b) initial pH and desired pH, and (c) the quantity of additional chemicals needed for adjusting Mg concentration. Although many studies that investigated NaOH usage for adjusting pH are lab-scale, there is a possibility that the consumption of NaOH will be economically feasible in a full-scale farm recovery process (Burns et al., 2003).

CO_2 striping: One of the reasons that struvite formation occurs at a higher rate in wastewater treatment plants is because of CO_2 stripping (Neethling & Benisch, 2004). Due to the aeration steps that increase CO_2 volatilization rate leading to a CO_2 stripping from the solution. Song et al. (2011) used this method to adjust the pH and indicated this method to provide a more controllable pH adjustment. Therefore, CO_2 stripping not only eliminates the need for NaOH addition but also reduces Mg^{2+} addition. Amini (2014) studied two different systems for adjusting pH in swine manure, adding NaOH and aeration. They reported the same struvite production rate with both methods. Although aeration reduced chemical use, the capital cost was higher due to both the larger reactor capacity and increased energy consumption. Besides, aeration could reduce the NH_4 component in the final product since a percentage (40-90%) of the nitrogen will be volatilized through air stripping as ammonia (NH_3) (Song et al., 2011).

Ammonia: Westerman et al. (2009) investigated the effects of adding anhydrous ammonia and NaOH to raise pH. They reported struvite yield, as mean reductions of substrate phosphorus, to be similar in both approaches. Using ammonia resulted in an increase in TAN value (9%) in the effluent while using NaOH reduced it by 9% in the effluent. Using NH4 can be contradictory in agricultural wastewater management contexts where the goal is to concentrate P, and to a lesser extent N, in the formed struvite.

Regardless of the pH adjustment approach, an optimal point needs to be defined based on manure characteristics to avoid excess chemical or aeration use as well as adverse impacts of elevated pH. One factor that should be considered when determining the optimum pH is the purity of the recovered crystal. Studies suggested that at higher pH, while the highest phosphorus removal occurs, the struvite purity declines, which will result in less commercial value for the product as fertilizer. Thus, defining the optimal pH is dependent on a reasonable trade-off between struvite purity and quantity (Jordaan et al., 2010).

A summary of the relation between the P removal rate and pH in different studies is presented in **Figure 6**. No clear statistical trends are discernable for P removal in this figure. All studies were conducted on swine manure in lab-scale experiments. As shown, most reductions happen in pH between 7.5-9.5.

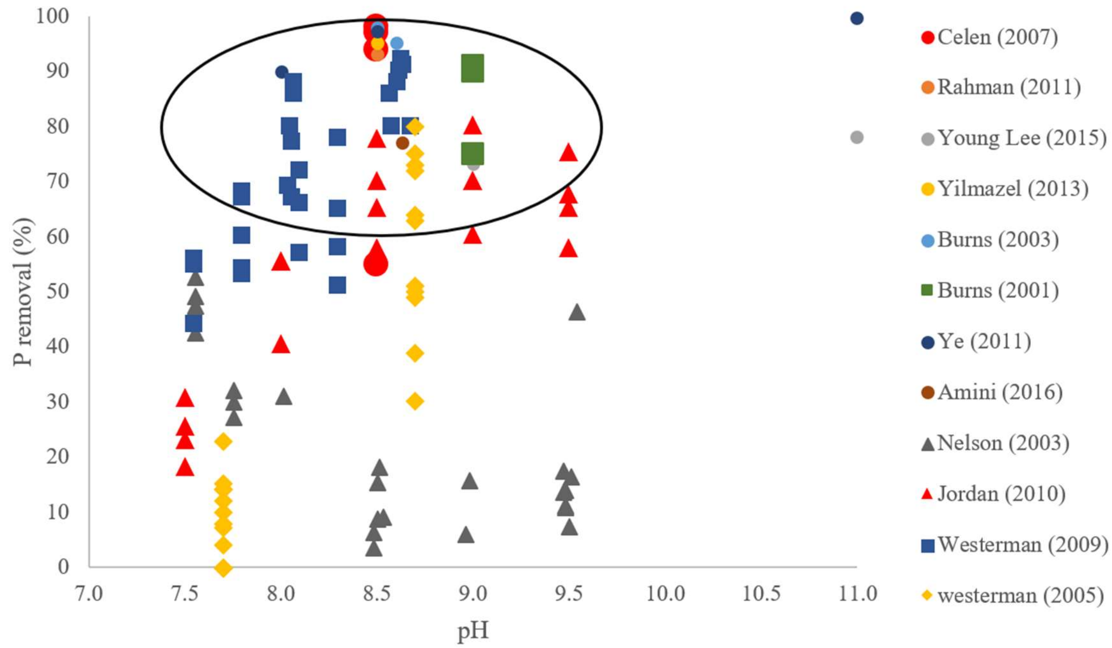

Figure 6. Summary of literature on P removal rate in different pH

According to Westerman et al. (2009), the amount of ammonia required to increase the slurry pH by 0.01 unit is 2.6 mg l^{-1}. The initial pH in the current study is 7.75, higher than initial pH of 7.3 in Westerman et al. (2009). Adding 13 $mg_{ammonia}$ l^{-1} could achieve pH of 7.8. Based on this assumption, the total amount of required ammonia for small farms would be 98 kg (reference flow)$^{-1}$. Westerman et al. (2009) reported in their study that using ammonia and NaOH will have the same result on increasing the pH but using ammonia will increase TAN concentration in the effluent. For preventing higher N concentration in effluent, NaOH was selected as the pH adjustment reagent. The NaOH requirement is not an independent parameter as it depends on Mg addition requirement as well. Çelen et al (2007) showed both by model and in the lab that achieving the same pH will require a different amount of NaOH for the same slurry based on Mg and P concentrations. However, in case that increasing pH is desirable, the related calculation can be carried out as shown in **Table 4** below to estimate amount of reagent required (Ndam, 2017).

Table 4. Additional ammonia required to increase pH (Ndam, 2017)

Variable	Amount	Units
V1 (solution to adjust pH)	20,712.33	$l\ d^{-1}$
Original pH	7.75	[-]
Target pH	8	[-]
[H+]_1	1.778E-08	ions
Normality	5	N
C1	8.89E-08	-
pOH_2	6	-
[OH-]_2	1.000E-06	-
C2	5.000E-06	-
V2	368.3	NaOH
NaOH molar mass	40	$g\ mol^{-1}$
NaOH concentration	14.73	$kg\ d^{-1}$
NaOH mass	711.31	$mg\ l^{-1}$

Mg-P Molar Ratio: Adding Mg is one approach to increase the struvite yield. In some cases, low Mg^{2+} concentration in the wastewater stream, including swine supernatant, can reduce struvite formation (Çelen et al., 2007; Nelson et al., 2003). Also, the addition of NaOH rapidly increases the solution pH and results in a sudden surge of magnesium precipitation in other forms besides struvite thus reducing struvite formation(Zeng & Li, 2006). Therefore, the actual required Mg^{2+} is usually higher than the theoretical values. Magnesium addition is often required to provide for at least 1:1 of Mg:P so that most of the P can be recovered (Amini, 2014). However, since the chemicals for Mg additions are expensive, many studies investigated struvite precipitation without Mg addition. For instance, Song et al. (2011) reported that although additional P can be removed with adding Mg, , the increase in P recovery would not justify the additional Mg cost. The reason is that the P removal rate does not increase linearly with an increase in Mg:P molar ratio. In some cases that initial manure has a higher Mg concentration, adding Mg is not necessary and a reasonable P removal rate could be achieved. Amini (2014) achieved 77% P removal without additional chemical for adjusting molar ratio. Musvoto et al. (2000) reported that overdosing Mg, combined with a high concentration of P could also form trimagnesium phosphate ($Mg_3(PO_4)_2.22H_2O$) or $Mg_3(PO_4)2.8H_2O$), and reduce the amount of struvite precipitated.

Common forms of magnesium that could be used in the process include $MgCl_2$, $MgSO_4$, Mg $(OH)_2$, and MgO. Adding chemicals to increase Mg concentration is considered the most significant operational expense of struvite precipitation process and it is estimated to contribute up to 75% of overall production costs (Amini, 2014). Lin (2012) performed an analysis of the economic feasibility of magnesium addition, He found that $MgCl_2$ addition is not economically feasible, while MgO addition was economically more favorable when the desired ratio of Mg: P was between 1.30-1.78. Burns et al. (2001) found that $MgCl_2$ was a better Mg^{2+} source than MgO for swine wastewater since it has a typical Mg^{2+} concentration of 1250 mg l^{-1}. They achieved 70% P removal from biosolids by using seawater as an Mg^{2+} source and on experiments on swine wastewater, they reported 81% OP removal at pH 10.

In different studies, the desired molar ratio was defined based on different forms of phosphorus. For instance, Mg: OP (orthophosphate phosphorus) molar ratios were investigated in Westerman et al. (2009) study, while Burns et al. (2001) and Nelson et al. (2003) attempted to reach the molar ratio of 1.6:1 of Mg: TP (total phosphorus). Çelen et al. (2007)used the ratio of Mg: PO_4 as an indicator to study the effect of reaching three ratio levels at 1, 1.5, and 2 on the P removal rate. In most studies, the $MgCl_2$ was used as a source of Mg for increasing the molar ratio. The amount of chemicals depends on the concentration of Mg and P in raw manure. While researchers tried to determine an optimal ratio for highest struvite production rate, this value varied across studies. For instance, it is identified as 1.5:1 (Çelen et al., 2007), 1.3:1 (Philip W.

Westerman et al., 2009; Yilmazel & Demirer, 2013), and 1.6:1 in (Burns et al., 2001, 2003; Nelson et al., 2003). **Figure 7** shows the P removal rate in different studies based on the molar ratio.

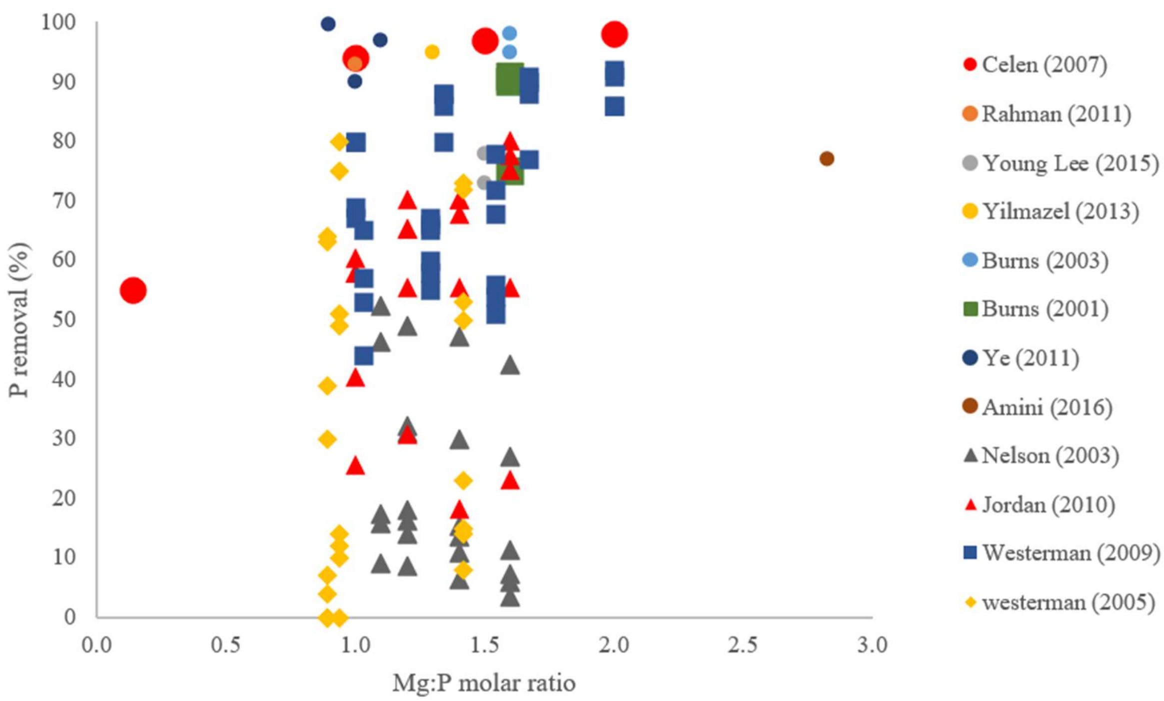

Figure 7. Summary of literature on P removal rate in different Mg:P molar ratio

Estimating required chemicals for adjusting the Mg:P molar ratio is important for this study but since in most studies, the amount of $MgCl_2$ consumption was not presented or because of the differences in concentrations of Mg and P in raw manure, the numbers are not applicable. However, by having an estimation of initial Mg and TP in raw manure, it is possible to calculate the required $MgCl_2$, for achieving different levels of desired molar ratio. The initial Mg: TP molar ratio in NC finishing farms was calculated as below:

Table 5. Initial manure Mg:P molar ratio for this study

Item	Unit	Amount
Mg molar mass	mg mol^{-1}	24.3
Mg concentration	mg l^{-1}	45
Mg molarity	moles l^{-1}	1.85
TP molar mass	mg mol^{-1}	31
TP concentration	mg l^{-1}	52
TP molarity	moles l^{-1}	1.68
Raw manure Mg: TP molar ratio	-	1.1

The values for the raw manure P and Mg contents were developed using a large dataset of sample analyses, spanning multiple years, from NC Department of Agriculture and Consumer Services (NCDA&CS). The raw manure in this study has a relatively high Mg: TP molar ratio, which can be attributed to age of the lagoons (and amount of sludge in the lagoons) which increases the concentration of TP and Mg. These results indicate that less $MgCl_2$ consumption is needed to adjust the ratio for a higher TP removal rate.

Based on the molar mass of Mg and TP, the required amount of $MgCl_2$ for increasing molar ratio was calculated as presented in the **Table 6**. As shown, about 31 mg l^{-1} $MgCl_2$ is required to achieve the optimal molar ratio at 1.3.

Table 6. Additional $MgCl_2$ required to increase the molar ratio

Variable	Amount	Units
TP	52	mg l^{-1}
Mg	45	mg l^{-1}
Molar mass P	31	mg mol^{-1}
P Normality	1.68	moles l^{-1}
Molar mass Mg	24.3	mg mol^{-1}
Mg Normality	1.85	moles l^{-1}
Initial Mg: TP molar ratio	1.104	
Target Molar ratio	1.3	
Required Mg	0.33	mole l^{-1}
Required Mg	7.99	mg l^{-1}
Molar mass $MgCl_2$	94	mg $mole^{-1}$
Required $MgCl_2$	30.91	mg l^{-1}

Production Yield

P removal: The P removal rate depends on adjustments to pH and Mg: P molar ratio. For instance, P removal was observed to be 79% after only pH adjustment using NaOH, which then increased to 91% after adding $MgCl_2$ (Burns et al., 2001). In general, eliminating $MgCl_2$ addition (mostly for economic reasons) results in less P removal rate. In one study the P reduction rate jumped from 55% into 94% after adding $MgCl_2$ and improving the Mg:P molar ratio toward 1:1 (Çelen et al., 2007). Interestingly, results in the same study showed that increasing the molar ratio to 2:1 resulted in a 98% P removal rate, which may not be beneficial economically since twice as much chemicals are needed for achieving 4% increase in P removal. While most lab-scale studies reported the P removal rate, they might not be valid for estimating the removal rate in field conditions. Westerman et al. (2009) examined the P removal rate in both lab and field conditions and initial field tests showed lower reductions in phosphorus in comparison to laboratory tests. The TP reduction rate in the baseline condition (no additional chemicals for adjusting molar ratio and pH) was about 60% based on lab tests, while in the same condition the reduction rate was zero for field tests.

To describe the relationship between the P removal rate and effective factors, the following correlation (equation (8)) was developed from literature data using the R program (Orthogonal Polynomials, ($R^2 = 0.94$)).

$$\text{TP removal rate} = -31.1\,(\text{pH})^2 + 76.7\,(\text{pH}) + 38\,(\text{Mg:P ratio}) + 49.7 \tag{8}$$

N removal: In the struvite precipitation process, the NH4-N removal occurs in two ways, i.e., NH_3 stripping under relatively high pH over 9.0 and struvite formation. Usually, the raw manure contains a much higher amount of NH4-N than other elements (Mg and P) for struvite formation, and most NH4-N can be removed by NH_3 stripping with aeration equipped in the struvite crystallization reactor. So, mostly the NH4-N removal is because of NH_3 striping, not the struvite formation and this process is pH dependent. The change in carbonate ions (CO_3^{+2}) to CO_2 caused by aeration increased the solution pH, thus converting stable NH_4^+ ions in the solution to NH_3 that is volatilized to the air (Shim et al., 2020).

Struvite removal rate: Predicting struvite production rate can be challenging as it is controlled by a combination of factors including the crystal state of initial compounds, thermodynamic of liquid-solid equilibrium, phenomena of matter transfer between solid and liquid phases (Jeong & Kim, 2001). The main obstacle in predicting struvite formation rate is that many ionic species (e.g. Ca^{+2}, K, and CO_3^{-2}) can influence the saturation of struvite by reacting with its component ions, Mg^{2+}, PO^{3-} and NH_4 (K. Le Corre et al., 2009). While in most studies the focus is on determining the P removal, and the result of adjusting different effective factors (such as pH and Mg: P molar ratio) is presented as to how they affect the P removal rate, in reality, the P removal is not always equal to struvite production. For instance, Song et al. (2011) showed that although P removal rate was up to 85%, only 30 % of P was recovered as struvite. In another study, the molar ratio of Mg^{2+}:NH_4: PO_4^{3-} of the product precipitate was 1:0.74:3.2, which means the produced phosphorus precipitate was not pure struvite, as the molar ratio of pure struvite is 1:1:1. The formed precipitate is enriched with phosphorus from the formation of other phosphate-containing compounds that may have been formed but not yet identified (Burns et al., 2003). A high concentration of calcium in raw manure could result in the formation of calcium phosphates or calcium carbonate instead of struvite. If calcium phosphates were formed, the purity of struvite recovered would decline, which means a lower value of product fertilizer. If calcium carbonate formed during the recovery, it could reduce rate of struvite formation but would not compete for phosphorus with struvite (Le Corre et al., 2005). This reduction in struvite purity is also illustrated by the molar ratios of the precipitate component concentrations (Jordaan et al., 2010). If the general goal is to recover phosphorus, not produce pure struvite, these results may be still favorable, but the theoretical purity of struvite product based on the weight of N, P, and Mg is 5.71 %, 12.62 %, and 9.90 %. However, the concentrations of N, P and Mg in the struvite product are normally less than theoretical since there is about 1.3 to 2.2 % Calcium (Philip W. Westerman et al., 2009). Estimating the struvite production rate is a key parameter for this study for assessing both environmental and economic impacts, but there are a few studies that presented the result of phosphorus recovery as struvite production, and they mostly focused on P removal rate since the experiments were las-scale. There are many other products other than struvite that could be formed because of P removal. Mathematical models and software, such as Visual MINTEQ and PHREEQC, have been employed to predict some possible precipitates during P precipitation, and these are some possible products: magnesium hydrogen phosphate (MgH_4PO_4), magnesium phosphate ($Mg_3(PO_4)_2 \cdot 8H_2O$), trimagnesium phosphate ($Mg_3(PO_4)_2 \cdot 22H_2O$), hydroxyapatite ($Ca_5(PO_4)_3OH$), tricalcium phosphate ($Ca_3(PO_4)_2$), etc. (Amini, 2014).

In some pilot-scaled studies, the struvite yield rate was reported. For instance, In a field-scale study conducted by Westerman et al. (2009), a recovery rate of 120-130 g $hour^{-1}$ (flow rate equal to 5.4 liters min^{-1}), which is equal to 0.385 g l^{-1}, was reported. This result corresponded to 55-60% P removal through increasing pH by 0.5 points (aiming for 7.8) and adjusting Mg: TP molar ratio of 1.3:1. In another pilot-scaled study for swine CAFOs in Florida, the production rate was reported as 0.48 g l^{-1} manure (Amini, 2014). The significance of Amini's study was not using any additional chemical for increasing Mg concentration (Mg and soluble P concentration were 71 ± 58 mg l^{-1} and 62 ± 13 in raw manure). In some lab-scaled studies, the struvite yield rate varied from 0.077 to 1.64 mg l^{-1} manure (Çelen et al., 2007; Song et al., 2011; Suzuki et al., 2007).

Using a 70% TP reduction rate for this study at the following condition (pH = 8 and molar ratio Mg:P =1.29), the struvite production rate was predicted using regression analysis (Equation 8) to be 0.3 g l^{-1} (12% of struvite by weight is P). Thus, the total amount of struvite production for small farms considered to be 2,277 kg (reference flow)$^{-1}$.

According to these assumptions, the effluent from the struvite recovery unit has 15.86 mg l^{-1} TP and 521 mg l^{-1} TAN concentration in case of using NaOH for adjusting the pH. If ammonia is used for this purpose, the TAN concentration will increase to 640 mg l^{-1}. The operation factors are reported in the table below in summary for small farms at optimal struvite removal.

Table 7. Struvite recovery unit operation LCI

Item	Units	Amount	Amount rf^{-1}
Initial pH		7.75	
Target pH		8	
Initial Mg:P molar ratio		1.1	
Target Mg:P molar ratio		1.3	
HRT	min	10.9	
Flow rate	liters min^{-1}	14	7,5600,000 liters
Electricity	kwh l^{-1}	0.00201	15,200 kwh
Required NaOH	kg d^{-1}	14.73	
Required $MgCl_2$	mg l^{-1}	30.9	233.7 kg
TP reduction	%	69.5	
Struvite production	g l^{-1}	0.301	2,277 kg
TP concentration in effluent	mg l^{-1}	15.9	120 kg
N concentration in effluent	mg l^{-1}	521	3,937 kg

2.1.2.3. Land Application

2.1.2.3.1. Energy Consumption

Manure irrigation systems include center pivot, linear move, traveling gun, and solid set. Spray irrigation systems have a range of operating pressures, ranging from 1 to 100 pounds per square inch (psi). High-pressure systems generate small droplets and can achieve a wider coverage but require more energy to operate. Since lagoon supernatant contains suspended and dissolved solids, a high-pressure system is suitable to prevent nozzles clogging (Porter & Perry, 2015).

For calculating the energy consumption in the irrigation process, the first step is calculating the total power needed for manure application (equation (9)) (Porter & Perry, 2015).

$$\mathrm{HP} = (\mathrm{Q} * \mathrm{H}) / (3{,}960 * \mathrm{E}) \quad (9)$$

where *HP* is required power (horsepower), *Q* is the system flow rate (gallons per minute (gpm)), *H* is total pressure head (ft), and *E* is operating efficiency of pump (as a ratio).

Pressure head for traveling gun irrigation systems typically falls between 60 and 120 psi (NRCS, 2010; Porter & Perry, 2015; USDA, 1997). For this study, the total pressure head was assumed to be 120 psi (277 ft of water). Irrigation pump efficiency typically varies between 45% and 75%, for this study a pump efficiency of 55 % was assumed.

To determine the operating flow rate (Q), the following equation, equation (10), was employed (Washington State University Extension).

$$Q_n = 28.9 * D^2 * \sqrt{P} \quad (10)$$

where *Q* is flow rate from nozzle (gpm), *P* is pressure at nozzle (psi), and *D* is nozzle diameter (inch).

Nozzle diameter in high- pressure systems could vary from 0.5-1.5 inches (Porter & Perry, 2015). For this study D was modeled as one inch (1 inch) and P was 120 psi.

Accordingly, the flow rate was calculated as follow:

$$Q = 28.9 * 1^{2} * \sqrt{120} = 317\ gpm \quad (11)$$

By fitting assumptions and findings into equation (9), total required power is equal to:

(12)

$$\text{HP} = (317 * 277) / (3{,}960 * 0.55) = 40\ \text{HP} = 30\ \text{kW}$$

By obtaining the required power for irrigation, the energy consumption for small and large farms could be calculated by multiplying the required duration for spreading manure by the total irrigation power.

The time required for spreading manure can be calculated by dividing manure volume (gallon) to the flow rate (gpm). For small farms total manure based on reference flow is 1,997,140 gallons, by considering 317 gpm flow rate, the required duration is 105 hours per year. The duration becomes 210 hours for large farms (considering 3,994,281 gallons manure generated). This calculation is conservative as it assumes all the manure volume excreted during this year is applied. In reality, a small share of this volume collects as sludge in the lagoon bottom.

Using the procedure outlined earlier, the annual irrigation energy consumption for small and large swine operations is determined to be 3,159 and 6,318 kWh, respectively.

2.1.2.3.2. Avoided Fertilizers

The term mineral fertilizer equivalent (MFE) is usually used to describe the potential of alternative nutrient source, including manure, to provide nutrients for crops in units of typical mineral fertilizer source. Empirical estimation of MFE is carried out using field studies and they typically benchmark performance against an established reference fertilizer. Potential MFE for N for swine slurry is about 40-70%. The actual first-year N-MFE value of a particular manure application depends not only on the application method, but also on the crop, environmental conditions such as soil and weather conditions, and the application timing (Jenson, 2013). Lopez-

Ridaura et al. (2008) and Langevin et al. (2010) suggested N-MFE value to fall between 65% and 60% of total N for manure application respectively.

The P in manure is mostly in inorganic form (80–90%) (Huang et al., 2011), and it has a higher MFE value compare to N. Jenson (2013) and Bayo et al. (2012) suggested that the P-MFE value of manure in around 100%. This number was reported slightly less in other references as 95% and 80% (Lopez-Ridaura et al., 2008) and (Minnesota University extension).

In this study, the MFE values for N, P_2O_5 were 65% and 95% respectively. Accordingly, the avoided N and P fertilizers could be calculated as follows:

$$Avoided\ N\ fertilizer\ =\ Total\ N\ (kg)\ *\ N-MFE\ =\ 4{,}438 *\ 0.65 \\ =\ 2{,}884\ kg\ N \tag{13}$$

$$Avoided\ P\ fertilizer \\ =\ Total\ P\ (kg)\ *\ P-MFE\ *\ 2.29\ (kg\ P\ kg\ P2O5-1) \\ =\ 391 *\ 0.95 *\ 2.29\ =\ 850\ kg\ P2O5 \tag{14}$$

Note: the 2.29 is coefficient to convert units from kg P to kg P_2O_5.

2.1.2.3.3. Air emissions

Most manure-related gaseous emissions post land application are NH_3 and N_2O emissions, in addition to CO2, CH_4, and small concentrations of volatile fatty acids (VFA) and odorous compounds (Mohankumar Sajeev et al., 2018). In general, 50% of applied manure N is taken by crops, while the rest is lost in different forms as ammonia (NH_3), nitrogen oxides (nOx), nitrous oxide (N2O) or dinitrogen (N2), or leaches to groundwater as nitrate (NO3) (Langevin et al., 2010). Manure-related emissions are strongly dependent on the manure application method.

CH_4 emissions: Manure CH_4 emissions result from the decomposition of organic compounds under anaerobic conditions during storage and after land application (Montes et al., 2013). the amount of CH_4 emission from land application phase considered negligible due to (1) the low C concentration in liquid manure, and (2) aerobic condition during land application (Bayo et al., 2012; Prapaspongsa et al., 2010; Pratt et al., 2015; Wesnæs et al., 2009; Wesnæs et al., 2013). Accordingly, the CH_4 emission was deemed negligible in our study.

N_2O emissions: Two pathways have been identified for N_2O emissions after land application: (1) direct N_2O from soils after land application and (2) indirect N_2O emissions through two different pathways. The first is N volatilization as ammonia and oxides of N (NO_x), and the second is the leaching and run-off from land (IPCC, 2019). The following sections briefly describe the estimation of the N_2O magnitudes attributable to each pathway.

Direct N_2O emission: N_2O emissions are initially low, one study showed that it dropped to background concentration levels after 90 days from application (swine slurry), and the highest emission was measured after rainfall events (Montes et al., 2013). Sherlock et al. (2002) estimated the total N_2O emissions as only 2.1% of the N applied. However, since N_2O is a potent global warming gas, with 1 kg N_2O is equivalent to 298 kg of CO_2, it is considered the primary GHG gas species in this phase.

Several studies measured N_2O direct emissions. Corbala-Robles et al. (2018) estimated N_2O direct emissions to be 1.57% total applied N, while Brockmann et al. (2014) determined N_2O emission factor as 1.3% TN, and Ten Hoeve et al. (2014) reported N_2O emissions about 2% of total N. In their meta-analysis of manure management emissions, Wang et al. (2017) showed how manure application methods impact N_2O emissions. Their review suggested the highest emission rate occurred during rapid incorporation and the lowest emission happened when the band-spreading system was utilized (3.65% and 0.74% respectively).

According to the IPCC report (2019), the emission factor value for N_2O resulting from manure land application (EF_1) is 0.01 kg N_2O-N kg N $^{-1}$ (or 1% of total N). Thus, the amount of N_2O emissions resulting from the field application can be calculated as below.

$$\mathrm{N_2O} = \mathrm{N} * \mathrm{EF_1} * 44/28 \tag{15}$$

where N_2O is direct N_2O emissions produced from managed soils (kg), N is the amount of N content in manure (kg), EF_1 is emission factor for N_2O emissions from N inputs (kg N_2O–N (kg N input)$^{-1}$) (table 11.1), and (44/28) is conversion coefficient of N_2O–N emissions to N_2O emissions. Based on this equation, the direct emission of N_2O for small farms during the baseline scenario is equal to 69.7 kg per reference flow for the baseline scenario.

$$\begin{aligned} N_2O\ direct\ emissions \\ &= 4{,}438\ (kg\ N) * 0.01\ (kg\ N_2O - N\ kg\ N - 1) * (44/28) \\ &= 69.7\ kg\ N_2O/\ rf \end{aligned} \tag{16}$$

<u>Indirect N_2O emission (N leaching and runoff):</u> As indicated earlier, indirect N_2O emissions occur through two pathways. As such, estimating indirect N_2O emissions includes two emission factors: one associated with volatilized and re-deposited N (EF_4), and the second associated with N lost through leaching/runoff (EF_5) (IPCC, 2019). Indirect N_2O emissions are typically lower than direct N_2O emissions and varies between studies from 0.31% (Corbala-Robles et al., 2018) to 0.13% TN (Brockmann et al., 2014).

<u>Volatilization</u>: N_2O from atmospheric deposition of N volatilized from managed soils could be calculated using equation (17).

$$N_2O - N = F_{ON} * Frac_{GASM} * EF_4 \tag{17}$$

where, N_2O-N is the annual amount of N_2O-N attributed to atmospheric deposition of N volatilized from managed soil (kg N_2O-N yr^{-1}), F_{ON} is the annual amount of managed animal manure and other organic N additions applied to soils (kg N), $Frac_{GASM}$ is the fraction of applied organic N fertilizer materials that volatilizes as NH_3 and NO_x (kg of N applied or deposited)-1, and EF_4 is emission factor for N_2O emissions from atmospheric deposition of N on solid and water surface (kg N_2O–N (kg NH_3–N + NO_X–N volatilized)$^{-1}$). The amount of $Frac_{GASM}$ and EF4 were obtained from table 11.3 as 0.21 and 0.014 respectively. The organic N in lagoon supernatant ranges between 7 and 19 % of total N (Blunden & Aneja, 2008; Shah et al., 2009). For this study, organic N is considered to be 10% of the total N in lagoon supernatant.

By using equation (17) and considering the coefficient of N_2O–N emissions to N_2O emissions (44/28), the total amount of indirect N_2O emissions in the form of volatilization in small farms is calculated as below.

$$\begin{aligned} &N_2O\ indirect\ emissions \\ &\quad = 4{,}438\ (kg\ N)\ *\ 0.1\ kg\ NO\ /kg\ N\ *\ 0.014\ (kg\ N_2O - N\ kg\ N \\ &\quad - 1)\ * 0.21 *\ (44/28)\ =\ 2.05\ kg\ N_2O/\ rf \end{aligned} \tag{18}$$

Leaching/run-off: N_2O from N leaching/runoff from managed soils could be calculated using equation (19)

$$N_2O - N = F_{ON} * Frac_{lea} \quad * EF_5 \tag{19}$$

where *N_2O-N* is the annual amount of N_2O–N produced from leaching and runoff of N additions to managed soils in regions where leaching/runoff occurs, kg N_2O–N yr-1, F_{ON} is the annual amount of managed animal manure, compost, sewage sludge and other organic N additions applied to soils in regions where leaching/runoff occurs, kg N yr^{-1}, $Frac_{leach}$ is the fraction of all N added to managed soils in regions where leaching/runoff occurs that is lost through leaching and runoff, kg N (kg of N additions)$^{-1}$, and EF_5 is emission factor for N_2O emissions from N leaching and runoff, kg N_2O–N (kg N leached and runoff). The amount of $Frac_{leach}$ and EF_5 were obtained from table 11.3 as 0.24 and 0.011 respectively. The total amount of indirect N_2O emissions in the form of leaching and run-off in small farms is calculated as below.

$$N_2O\ indirect\ emissions = 4{,}438\ (kg\ N) * 0.1\ kg\ NO\ /kg\ N * 0.011\ (kg\ N_2O - N\ kg\ N - 1) * 0.24 * (44/28) = 1.84\ kg\ N_2O/\ rf \quad (20)$$

The table below presents the total amount of N_2O emission through the land application phase.

Table 8. N_2O emissions from different pathways for baseline scenario (small farms)

Emission pathway		Amount (kg N_2O rf^{-1})
Direct emission		69.70
Indirect emission	Volatilization	2.05
	Leaching/run-off	1.84
Total emission		73.59

Ammonia emissions: NH_3 devolatilization after manure land application is a major pathway of N losses in the agricultural sector (Misselbrook et al., 2002). Studies suggest the ammonia emission rate during and after application to strongly depend on application methods (Hou et al., 2015; Langevin et al., 2010; Powell et al., 2011; Y. Wang et al., 2017) as well as soil and climate conditions (Langevin et al., 2010). Sherlock et al. (2002) measured CH_4, NH_3, and N_2O emissions immediately after land application of swine slurry and up to 90 days afterward.

They reported a high NH_3 emission rate immediately after application that decreased rapidly (22.5% of the applied N overall). Incorporating manure can reduce NH_3 emissions but can increase the potential for N_2O emission through nitrification and denitrification (Montes et al., 2013). In one study, Powell et al. (2011) examined the NH_3 emission potential of three irrigation treatments, i.e., (1) surface broadcast, (2) surface broadcast followed by partial incorporation using an aerator implement, and (3) injection. Slurry total N loss as ammonia was 20.5%, 12.0%, and 4.4% respectively. The table below summarizes ammonia emission rates associated with manure land application determined in different studies.

Table 9. Summary of literature values of ammonia emission associated with manure land application

Value	Unit	Manure application method	Reference
25	% TN		(Wiens et al., 2008)
18	% TN		(Brockmann et al., 2014)
30	% TN	surface broadcasting	(Y. Wang et al., 2017)
1	% TN	Injection	(Y. Wang et al., 2017)
10	% TN	rapid incorporation	(Y. Wang et al., 2017)
8.5	% TN	band spreading	(Y. Wang et al., 2017)
27	% TN	surface broadcast	(Powell et al., 2011)
23	% TN	surface broadcast followed by partial incorporation	(Powell et al., 2011)
9	% TN	injection	(Powell et al., 2011)
24	% TN		(Langevin et al., 2010)
12	% TN		(Marianne Wesnæs et al., 2013)
48	% TAN*	surface spreading	(Hou et al., 2015)
21	% TAN	sand spreading, incorporation	(Hou et al., 2015)
11	% TAN	Injection	(Hou et al., 2015)
20	% TAN		(Sharara, 2015)
0.88	kg m^{-3} raw sludge		(Bayo et al., 2012)
12	% TN		(Ten Hoeve et al., 2014)
20-21	% TN		(Prapaspongsa et al., 2010)
47	% TAN	band spreading	(Misselbrook et al., 2002)
48	% TAN	shadow injection	(Misselbrook et al., 2002)
17	% TAN	surface incorporation	(Huijsmans et al., 2003)
68	% TAN	surface spreading	(Huijsmans et al., 2003)
48	% TAN		(Corbala-Robles et al., 2018)

* TAN: Total ammonical nitrogen

The TAN is 75% - 79% of the total N in swine manure (Buckley et al., 2009; Wesnæs et al., 2013). In this study, the NH_3 emission factor was taken as 20% of total N in the manure. So the emission for small farms is as below.

$$NH_3\ emissions\ =\ N\ (kg)\ *\ NH_3\ EF\ =\ 4{,}438\ (kg\ N)\ *\ 0.2\ =\ 888\ kg \qquad (21)$$

2.1.2.3.4. Leaching and run-off

Leaching rates depend on the availability of mineral N and P in soils, the water balance (rainfall and irrigation vs. evapotranspiration), and soil characteristics (depth and texture). Soils with fine-texture (high clay) are in general less vulnerable to leaching than sandy-textured soils because water permeability is much lower (FAO, 2017).

P leaching: The factors affecting P losses are (1) soil physical and chemical properties, (2) application approach, and (3) climatic and environmental conditions (rainfall, drought, erosion, etc.). Dissolved and particulate P (eroded soil particles) are the forms of P that are most probable to be lost from the soil via leaching (FAO, 2017). The amount of P loss from different fields varies since it is driven by the complex interaction of soil P levels and forms, climate, topography, soil conditions, and crop type (Bergström & Kirchmann, 2006). In several studies, 10% of the total P applied to the field was considered as P loss as leaching (Wesnæs et al., 2013; Wesnæs et al., 2009; Sharara, 2015). Using the same ratio, the amount of P leaching in small farms is equal to 39 kg per reference flow.

$$P\ leached\ (kg)\ =\ 391 \times\ 0.10 =\ 39\ kg\ P \qquad (22)$$

N leaching: Nitrogen leaching occurs after manure application because of the vertical movement of nitrogen in the soil profile. Nitrate leaching was estimated as a fixed ratio of the total amount of nitrogen applied, and direct losses of NH_3 and N_2O in Langevin et al. (2010)study. Sørensen & Rubæk (2012) reported that the amount of nitrate leaching depends on the application timing and it rises after fall application, and they estimated that the nitrate leached was equal to 23–35% of total manure N at the highest time. On thorough LCA reports conducted by Wesnæs et al. (Wesnæs et al., 2013; Wesnæs et al., 2009), NO_3 loss was reported 35%. (Liu et al., 2019) conducted a series of experiments from 2011 to 2014 to estimate the N loss through surface drainage (free drainage) under the North Carolina climate condition. The input of total Nitrogen during tests was 390 kg N ha^{-1}. They reported that N loss through leaching was equivalent to about 10% of total N input (36.8 kg N ha^{-1}). 84% of the Nitrogen was in the format of Nitrate-N format (30.8 kg N ha^{-1}), which means about 8% of total applied N leached as Nitrate-N. For this study, 8% of total N is considered as NO3 leaching. Therefore, the total amount of NO3 leached is calculated as below for small farms under the baseline scenario.

$$\begin{aligned} NO_3\ leached\ (kg) &= Manure\ N\ (kg\ N) * (8/100) * 4.43\ (kg\ NO_3\ kg\ N-1) \\ &= 4{,}438 * 0.08 * 4.43 = 1{,}573\ kg\ NO_3/rf \end{aligned} \tag{23}$$

Note: the 4.43 is coefficient for kg N to kg NO_3 conversion

Phosphorus run-off: P runoff occurs due to transport of both soluble and particulates (sediments) P from the soil. The amount of runoff varies based on field slope, rainfall pattern, soil properties, and infiltration rates (FAO, 2017; Miller et al., 2011). Even under the same climate, runoff magnitude vary with initial field P concentration, application method, and field topography. These two factors determine both the amount and form P loss and the types of P in runoff. The amount of P loss is lower when manure is incorporated upon application (Gessel et al., 2004) since it decreases the surface P concentration resulting in lower P runoff. In an experiment conducted by Kleinman et al. (2002) tests were conducted using soils with different initial P concentrations to estimate the magnitude of P loss in different conditions. The results showed that in low-P soils, the difference between surface application and mixing averaged 6.0 mg dissolved reactive phosphorus (DRP) l^{-1} and 7.2 mg TP l^{-1}. In high-P soils, the difference averaged 10.8 mg DRP l^{-1} and 11.3 mg TP l^{-1}. Also, the results showed the impact of different application approaches as mixing application (incorporation), DRP accounted for 9% of runoff TP, while DRP accounted for 64% of TP in runoff from surface application method. The final results demonstrated that the P loss was 2 to 5% of applied TP through surface application and 0.5 to 2% through the incorporated application. The higher range in both application methods was for high-P soils and the lower range was for low-P soils (Kleinman et al., 2002). Another study suggested P runoff as 0.2- 1.5% of total P (Wang et al., 2019). Utilizing the injection method for application results in even less P loss via runoff. Daverede et al. (2004) reported that injection of swine manure compared with surface application resulted in the reduction of runoff P by 93% and 82% for dissolved reactive P (DRP) and total phosphorus (TP).

Regarding the Type of P in the runoff, Miller et al. (2011) reported that dissolved P fractions in runoff were linked to annual manure P applied while the total and particulate P were related to cumulative manure P applied. This means that dissolved P is more likely to be lost during recent manure applications. Miller et al. (2011) suggested a linear relationship between TP in manure and runoff for one year as below:

$$TPr = 7.1\,Pa - 11.2 \tag{24}$$

where *Pa* equals annual total P applied in manure and *TPr* is total P in runoff.

For this study, the amount of P runoff is considered 5% of total P in applied manure as suggested by Kleinman et al. (2002). Thus, the amount of P runoff for the small farm through the baseline scenario is calculated as below.

$$P\ runoff\ (kg) = \ Manure\ P\ (kg\ P) \times \left(\frac{5}{100}\right) = \ 391\ \times\ 0.05 = \ 19.55\ \frac{kg\ P}{rf} \qquad (25)$$

2.1.3. Impacts Assessment

Impact assessment method chosen for this study is the Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts (TRACI), developed by the US Environmental Protection Agency (EPA). TRACI is chosen because it was developed by the US EPA with impact assessment methodologies applicable to North America. The impact categories considered in this study include: global warming potential (GWP), acidification, eutrophication, and resource depletion.

2.2. Uncertainty & Sensitivity analysis

Many variables have been included in the model such as emission factors, nutrients concentration, energy consumption etc. Each of these variables are defined in a range, which would add to uncertainty of model. Thus, uncertainty analysis seems to be necessary to investigate the uncertainty of variables which are our model inputs. Uncertainty analysis determine the accuracy of model outcomes by quantifying the variability of the output due to the variability of the input. To calculate uncertainty associated with the model, the Crystal Ball software (Oracle, Austin, TX, Release 11.1.2.4.850) was used which was Microsoft Excel Add-Ins. This software uses the Monte Carlo simulation which rely on repeated random sampling to calculate range of model output.

For uncertainty analysis, the first step was identifying the model input parameters that are subject to uncertainty, then these variables were assigned for an appropriate distribution pattern (normal, triangle, uniform etc.) based on the nature of data and their range. The model was run for 1000 trials and the distribution of model output frequency was plotted.

Also, the sensitivity of the model was evaluated. Sensitivity analysis reveals the extent of impact model input may have on the outputs. A model is sensitive toward a parameter if a small change in this parameter will result in a large change in the model output, whereas a model is insensitive toward a parameter if any change in this parameter will have negligible effect on the model output. Sensitivity analysis usually is presented by tornado charts to show both direction and magnitude of each variable's impact. The Crystal Ball software also provides the sensitivity chart of the model outputs and makes it possible to see which inputs had the biggest impact on outputs.

2.3. Economic Assessment

Very limited information is available on cost data for struvite precipitation reactors and processes in the literature because most designs are proprietary. The costs change based on different designs and technologies and choosing the proper technology depends on users' priorities. Some would rather implement more expensive technologies to produce a more uniform precipitate that is more easily approved for fertilizer sale and marketed as a high-quality product. Thus, tradeoffs are depending on the expected use of the final product (Amini, 2014).

In the following, an estimation of costs and revenues associated with adopting struvite recovery technology in NC swine farms is reported.

2.3.1. Net present value (NPV)

Cost-benefit analysis (CBA) is a procedure of detecting, quantifying, and comparing the benefits and costs of an investment project or program (Campbell, H. F. and Brown, 2016).

The CBA generally attempts to put all related costs and benefits on a common temporal basis, by using the time value of money calculations. This is often done by converting the future expected streams of costs and benefits into a present value amount with a discount rate by using the net present value (NPV) method. The net present value (NPV) or net present worth (NPW) applies to a series of cash flows occurring at different times. The present value of a cash flow depends on the period between now and the cash flow. It also depends on the discount rate. The NPV accounts for the time value of money. It provides a method for evaluating and comparing capital projects or financial products with cash flows spread over time, as in loans, investments, payouts from insurance contracts plus many other applications. The following equation was used to calculate NPV:

$$NPV = \sum_{t=1}^{n} \frac{R_t}{(1+i)^t} \tag{26}$$

where R_t equals net cash inflow outflows during a single period t, the i equals to the discount rate and t time of the cash flow.

A positive NPV indicates that the projected earnings generated by a project or investment exceeds the anticipated costs. It is assumed that an investment with a positive NPV will be profitable, and an investment with a negative NPV will result in a net loss.

2.3.2. Capital and operating cost

2.3.2.1. Investment (capital) cost

For this report, three main categories were considered for estimating the total investment (capital) cost including direct costs, indirect costs, and working capital.

Direct Cost (DC): The direct cost includes equipment, material, installation, and construction cost. By using equation (27) costs adjustment was performed for different scales.

$$\mathrm{Cost}_2 = \mathrm{Cost}_1 \left(\frac{Size_2}{Size_1}\right)^a \tag{27}$$

where, $cost_1$ is cost associated with $size_1$, $cost_2$ is cost associated with $size_2$ and a is an exponent between 0.5 and 1 with an average value for vessels of around 0.6.

This equation presents the scaling law (also known as the "0.6 rule"). According to this rule, when the size of a vessel doubles, its cost will increase by a factor of $(2/1)^{0.6}$, or approximately 52% (Petrides, 2013).

Westerman et al. (2009) estimated direct capital cost for a farm with 1000 animals in two sections including equipment and material ($44,000), installation, and construction ($43,000). The total direct capital cost is $87,000. Based on the inflation rate of 2%, the cost for the year 2020 equals to $105,000 (by considering the cumulative rate of inflation of 21.3%) (https://www.usinflationcalculator.com)

By using equation (27), the direct cost for small (3,000 animals) and large farms (6,000 animals) in NC was calculated. The a (the equation's exponent number) was considered 0.65 for the equipment and material costs (the number for crystallizers) (Amini, 2014) and 0.6 for the equipment and material costs (Petrides, 2013).

Indirect Cost (IC): Indirect costs extend beyond the expenses for creating a product to include the costs involved with maintaining and running a project. These overhead costs are the ones left over after direct costs have been calculated. The indirect cost included engineering and supervisors (5% DC), contractor fee (2% DC), and contingency (6% DC) (Rafie et al., 2013).

Working Capital (WC): Working capital accounts for cash that must be available for investments in on-going expenses and consumable materials. The required amount of working capital for a process is usually 10 to 20% of the DC (Petrides, 2013). For this project, we assumed that the working capital is 10% fixed capital investment (FCI) (Rafie et al., 2013). The FCI equals the sum of direct and indirect capital costs.

Total capital investment: Finally, the total capital investment (TCI) was calculated by using equation (28) (See the excel file).

$$\mathrm{TCI} = \mathrm{DC} + \mathrm{IC} + \mathrm{WC} \tag{28}$$

2.3.2.2. Operating cost

Two factors primarily affect the operating cost, the flow rate of the system, and the concentrations of Mg (Amini, 2014). This is because the Mg concentration determines the amount of required $MgCl_2$. Operating costs for this project include labor cost, utility (electricity), required chemical costs (additional NaOH and $MgCl_2$), and maintenance.

2.3.2.3. Revenue

The revenue for this project comes from the product (struvite) sale. The struvite is a slow-release fertilizer with nutrient content of approximately 5% nitrogen, 12% phosphorus (28% as

P_2O_5), and 10% magnesium in high- quality products (such as the commercial production of Ostara company known as Crystal Green). Determining the exact value of struvite in compare to other commercial fertilizers is not straightforward since not only the quality of struvite varies in different technologies, also the different price is suggested in literature for the product. For instance, some experiments suggest struvite has a relative fertilizer efficiency of 64%-134% of triple superphosphate (TSP) fertilizer (Perez et al., 2009). If we consider it as effective as TSP, based on the latest data (https://www.indexmundi.com/), the product could be sold at $282 per metric ton ($0.28 kg^{-1}). Another study suggests that when struvite is applied in the form of granules, its behaviors in the soil may bear more similarity with rock phosphate than soluble mineral P fertilizer (Vries et al., 2017).

Amini et al. (2017) assumed that struvite can sell for the same cost as Diammonium phosphate (DAP) at approximately 0.37 $ kg^{-1}. In another example, the recovered struvite has provided an increasing source of revenue for Madison Metropolitan Sewerage District MMSD, with a guaranteed price of at least $300 ton^{-1} of struvite per its agreement with Ostara (Kucek et al., 2017). In this case, struvite recovering also reduced MMSD's operating and maintenance costs since before struvite harvesting, nuisance struvite was a major source of maintenance cost included struvite scaling in draft tube mixers, heat exchangers, heat recirculation pumps, and sludge transfer lines.

2.4. Site Selection

In this section the question is where are the locations that could benefit the most from implementing struvite recovery technology regarding environmental aspects. Several environmental criteria have been defined to select high-risk locations related to eutrophication impact. The reason is that struvite recovery technology mostly focuses on phosphorus (key element responsible for eutrophication) recovery.

2.4.1. The site selection criteria

1. Number of animals on farm: 6,000 or greater heads of animal
2. Located within a basin with nutrient-sensitive water resource: there are three main nutrient sensitive water resources in the state including Cape Fear (N and P sensitive), Neuse (N sensitive), and Tar Pamlico (N and P sensitive). Generally, livestock is one of the main resources for nutrient (N and P) load in these basins. For instance, sources of N and P for Tar Pamlico basin reported as **Table 10** based on a report North Carolina Department of Agricultural and resource economics.

Table 10. Sources of N and P in Tar Pamlico system

Source	% Nitrogen	% Phosphorus
Point sources	25	8
Agriculture and livestock	44	44
Forestry	5	9
Urban area	3	2
Wetlands	2	4
Water (atmospheric deposition)	17	32

3. Proximity to rivers: the impact of selecting different size stream buffers (0.1 to 7 miles) on number of selected farms is shown in **Figure 8**. For this research 0.3 miles proximity has been selected.

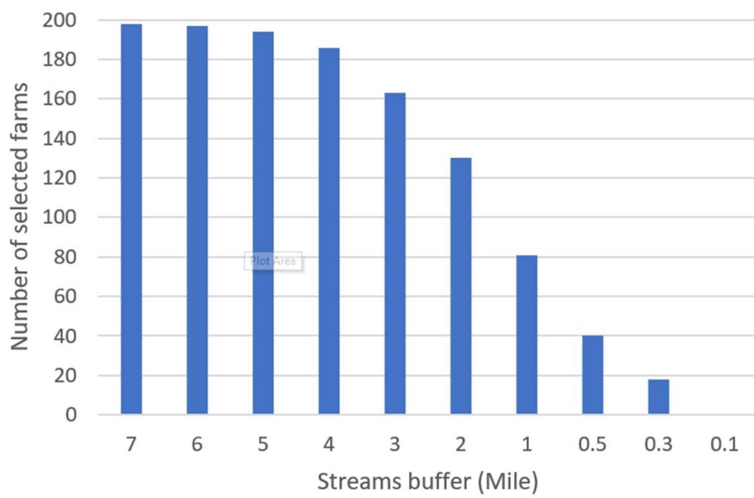

Figure 8. The impact of the buffer size on the number of swine farms in NC that fall within this buffer

4. Soil P level: being in counties with very high soil P index: Higher P concentration in soil could increase the chance of P runoff or leaching, North Carolina's department of agriculture and customer service indicated P index to classify the counties regarding soil P level ass shown in **Table 11**. Based on this definition, the average soil PI is illustrated in the **Figure 9**. For this research locations with P index higher than 100 has been selected.

Table 11. Soil phosphorus level based on soil phosphorus index

Soil Test P Level	Soil Test P index
Very high	>100
High	51-100
Medium	26-50
Low	11-25
Very low	0-10

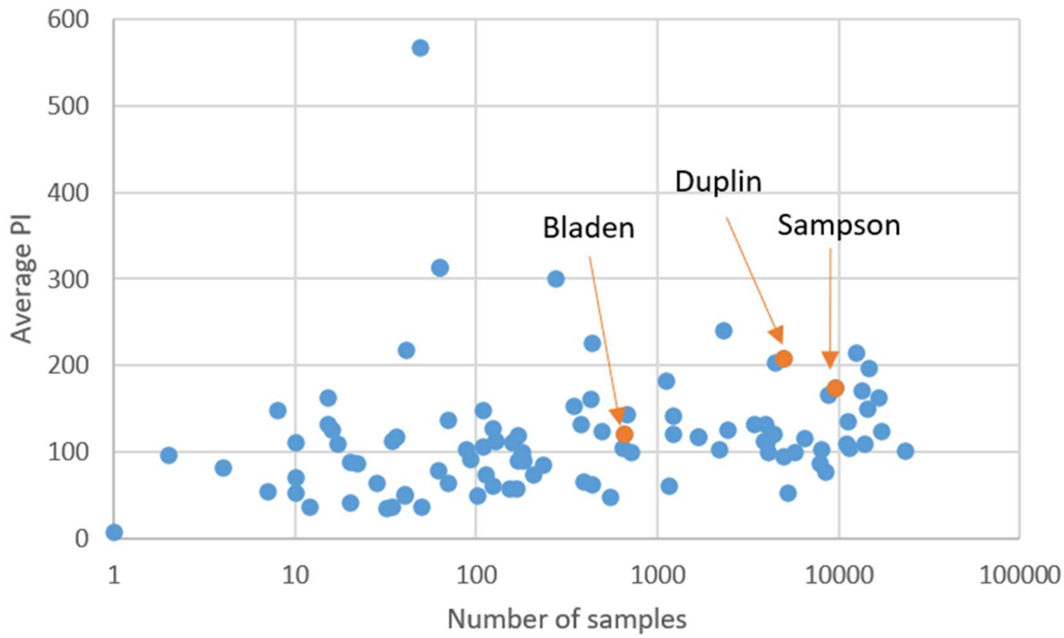

Figure 9. Average Phosphorus index (PI) in NC counties based on number of samples in each county (red dots present three counties with highest swine production)

2.4.2. GIS tools

The ArcGIS pro software has been used for selecting suitable sites for implementing struvite recovery units. The **Figure 10** illustrates the model used for this purpose.

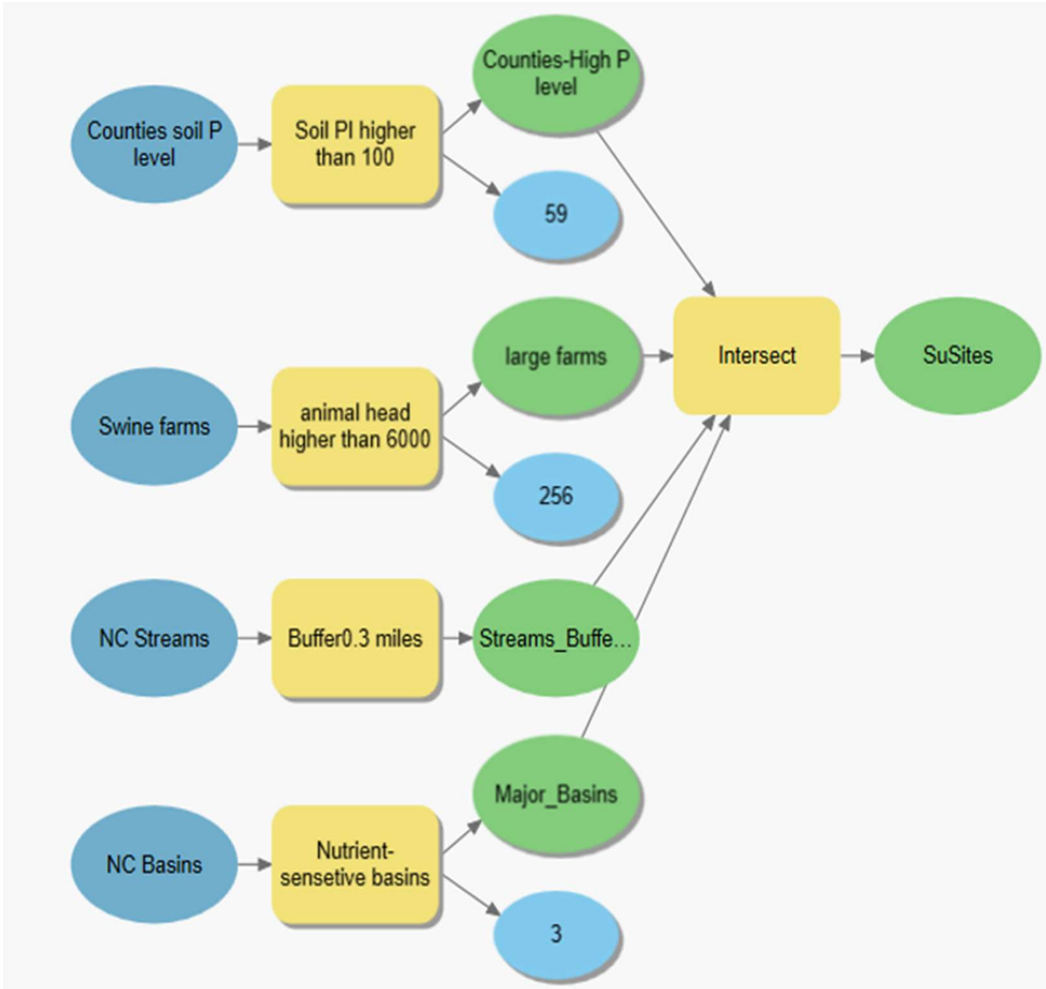

Figure 10. A graphical representation of the screening tool developed to identify sites to prioritize struvite technology implementation

3. RESULTS AND DISCUSSION

This thesis investigated environmental impacts and economic feasibility of adopting struvite technology as an alternative manure management for recovering nutrients (mostly P) from NC swine farms. The findings in this thesis fill the data gap related to cost and impacts of technology adoption. These findings are not only beneficial for other researchers in this field but also for farmers, investors, and authorities to make informed decisions moving toward more sustainable livestock industry. A researcher can study the process with new reagent (pH adjustment chemical or aeration) or using a solar-powered equipment or a passive in-lagoon design that can significantly improve process efficiency for the same price or reduce cost for the same capacity. A grower might consider this if an incentive structure is offered to target P for nutrient planning instead of N. Currently, all swine and poultry growers in the state develop use N-based plans for manure application. Regulators can add struvite technology with complete information to the list of options they can recommend, or potentially cost-share. Studying subscenarios AF and NAF provided data that could be beneficial for authorities by providing data on how much environmental benefit can be achieved by optimizing usage of manure (10% less eutrophication potential, nine times less acidification potential, and 15 time less resource depletion potential). This might encourage them to move toward providing opportunities for farmers to use manure as a fertilizer not as waste.

3.1. Life Cycle Assessment (LCA)

The results indicate that the effect of different scales (small and large farms) is less than 0.01% in investigated environmental impacts when comparing two size farms (not shown here). These results suggest that the environmental impacts per animal head are almost identical regardless of the farm size. Struvite technology is a chemical/physical process which rely on chemicals, such as NaOH and $MgCl_2$, for each processed unit volume of effluent. The production and use of these chemicals are linked to significant environmental impacts that are directly proportional to number of swine. However, Amini (2014) concluded that the large scale system was more environmentally friendly than medium scale and created a "economies of scale" effect with environmental impact. The reason might be that in his study, struvite unit was in line with other treatment units (anaerobic digestion and iron exchange units) for recovering biogas and nitrogen.

Therefore, results are presented on a per head animal per year basis throughout this section. Regarding environmental assessment, the main results indicate that adopting struvite technology will reduce negative water quality impacts (eutrophication). This comes at a cost of an increase in other impact categories (e.g., GWP, acidification, and resource depletion). The struvite technology reduces eutrophication by reducing P amount in lagoon effluent and thus the potential for runoff or leaching. On the other hand, the increase in GWP, acidification, and resource depletion impacts are associated with increased use of chemicals (NaOH and $MgCl_2$) and electricity to power the crystallizer when only product is P fertilizer (as fertilizer). The environmental benefits from avoided consumption of commercial P fertilizer do not exceed the negative impacts associated with on-farm chemical production and energy consumption for operating the system. Amann et al. (2018) suggested that implementing struvite recovery for wastewater showed a different result and reduced the GWP, which can be due to different characteristics of wastewater (manure vs. municipal wastewater). However, the struvite technology can be advantageous in nutrient sensitive areas, especially phosphorous-impaired regions, such as watershed with reported eutrophication impacts (algal blooms, elevated P levels in lakes) or in counties with high concentration of P in their soil.

In the following section, the environmental impacts assessment results for each scenario are presented before comparing two scenarios together.

3.1.1. Baseline scenario environmental impacts

The results indicate the following:

Magnitude of all impact categories (GWP, eutrophication. acidification and resource depletion) increased when manure was not utilized beneficially (NAF). However, the resource depletion and acidification categories show a greater increase rate (10 and 9 times, respectively) in comparison to (GWP) (almost 2%). When swine manure was assumed to not offset commercial fertilizer (NAF option), even though all impact categories increased, the resource depletion impact category increased by highest percentage. On the other hand, GWP did not change significantly from AF to NAF scenario (for either the baseline or the struvite recovery) since methane emissions from the lagoon are the largest contributor to GWP. Since lagoon conditions did not change across those two scenarios, it was reasonable to observe little change in GWP. For decreasing this impact, other measures should be considered such as covering lagoons or installing digesters. Over the past few years, adopting anaerobic digesters on NC swine operations have steady increased particularly with the opportunity to collect and inject recovered methane into pipeline as renewable natural gas (RNG). These technologies allow for energy recovery from manure which could be utilized onsite to offset the energy demand associated with nutrient concentration (through struvite precipitation or any other comparable technology).

Investigating environmental impacts based on system phases, i.e., lagoon and land application stages show that: (a) under AF sub-scenario, other than eutrophication impact category, lagoon operation has a higher contribution. (b) Through the NAF option, the land application phase plays a negative role in all four impact categories, and other than GWP, in other impact categories, the land application phase has a higher contribution to the overall outcome.

Investigating environmental impacts suggests that: (a) in AF option (**Table 12**), lagoon methane emissions was the largest contribution to GWP as NAF option (97% and 95% respectively). For the acidification category, energy consumption during the land application has a higher contribution. For eutrophication, P leaching during land application was the largest contributor to eutrophication impacts (52%), and for resource depletion consumed energy on the land application phase has the highest impact (**Figure 11**).

Table 12. Environmental impacts associated with the baseline scenario based on flows (AF option)

Impacts category	Reference unit	Mass/energy flow									
		Lagoon	Land application								
		CH_4	N_2O	N_2O	Ammonia	Electricity	Avoided P fertilizer	Avoided N fertilizer	N (leaching)	P (leaching)	P (runoff)
Global Warming	kg CO_2 eq	626.61	23.59	7.31	-	0.60	-0.50	-10.13	-	-	-
Acidification	g SO_2 eq	-	-	-	-	5.15	-5.34	-43.19	-	-	-
Eutrophication	g N eq	-	-	-	35.09	0.07	-5.31	-16.70	28.01	94.98	47.79
Resource depletion	MJ surplus	-	-	-	-	0.58	-0.90	-7.76	-	-	-

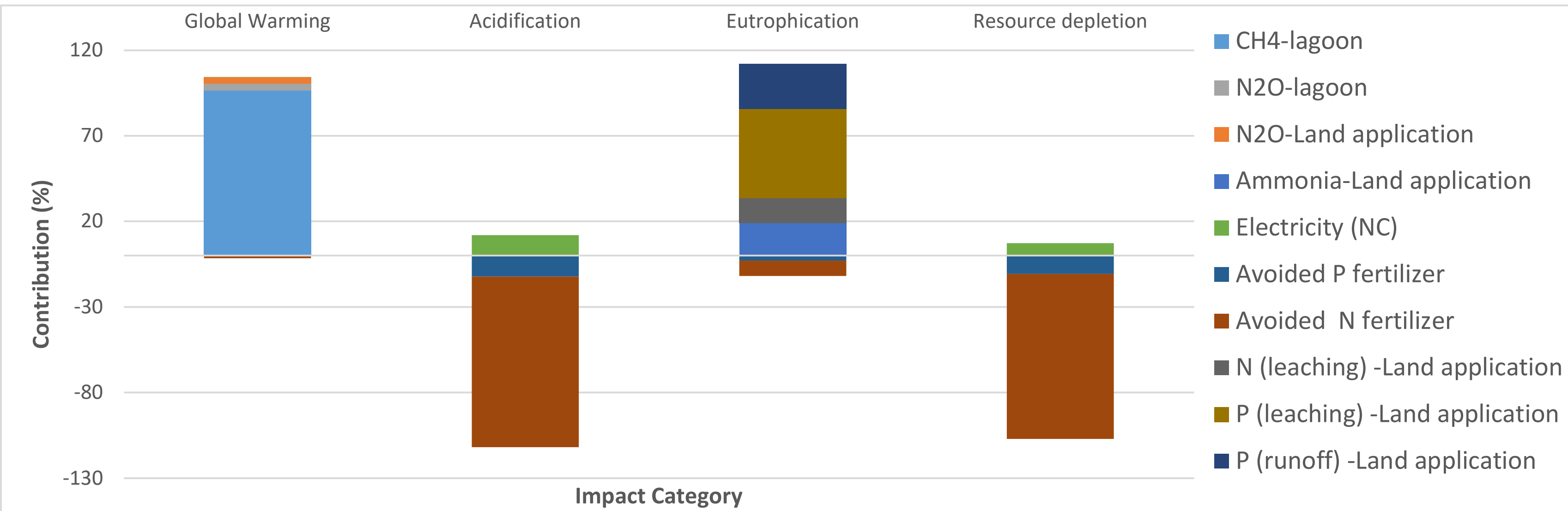

Figure 11. Mass/energy flows contribution in environmental impacts (the baseline scenario (AF))

3.1.2. Struvite precipitation scenario

The results show that adopting struvite precipitation led to a slight increase in global warming potential (1% increase) when compared to the baseline scenarios (both AF and NAF). Acidification and resource depletion impact categories after struvite technology adoption showed a significant increase, 600% and 150%, respectively, when compared to AF and NAF options. **Figure 12** shows the differences between impact categories for AF and NAF options for this scenario. In both AF and NAF options, GWP is the main environmental concern. Investigating environmental impacts based on mass/energy flows (**Table 13**) shows that in both AF and NAF options, methane emission from lagoon has the highest role in GWP and ammonia emission from land application phase, P leaching, and N leaching have the highest contribution in eutrophication. Regarding acidification impact, electricity consumption during struvite precipitation operation is the responsible flow. This flow is also accountable for resource depletion impacts along with consumption of NaOH.

Avoiding consumption of commercial N fertilizer had the highest positive contribution to the struvite assessment, although the focus was capturing P from manure by precipitating struvite. the same results have been revealed when implementing struvite unit for anaerobically digested dairy manure (Temizel-Sekeryan et al., 2021). This primarily due to the dependence on natural gas to produce commercial ammonia fertilizer.

Results indicate that energy consumption in form of electricity had the highest contribution in environmental impacts for struvite operation in all impact categories (60%- 75%). So, relying on solar power might be an option for reducing impacts which could be evaluated in future research. This is particularly relevant as swine operations have wide open spaces (including the lagoon surface itself) that could be utilized to install solar photovoltaic cells. The second large contribution to environmental impacts is NaOH consumption for pH adjustment. An alternative method could be aeration to strip CO_2. The same result reported by (Sena et al., 2021) about contribution of inputs in impact categories. They evaluated environmental impacts when utilizing struvite technology for treating wastewater.

Results showed that operating the struvite precipitation unit has significantly higher environmental impacts compared to the construction phase associated with the unit. Thus, it is more important to focus on improving the operation condition rather than construction aspects of the unit. Amini (2014), who investigated more complex treatment scenarios including anaerobic digestion, came to the same conclusion. This means that the system performance is more sensitive to the control of operation parameters including reagents use, power consumption and influent composition.

Table 13. Environmental impacts (per head animal) associated with the struvite scenario based on flows (AF option)

Impacts category	**Reference unit**	**mass/energy flows**													
		Lagoon		Struvite unit				Land application							
		CH_4	N_2O	Avoided P fertilizer	Electricity	Steel	NaOH	Avoided P fertilizer	Avoided N fertilizer	N (leaching)	P (leaching)	P (runoff)	N_2O	Electricity	Ammonia
Global Warming	kg CO_2 eq	626.61	23.58	-3.09	0.60	0.06	0.84	-0.16	-8.98	-	-	-	6.49	2.88	-
Acidification	g SO_2 eq	-	-	-36.66	5.15	0.17	7.19	-1.63	-38.32	-	-	-	-	24.78	-
Eutrophication	g N eq	-	-	-3.23	0.07	-0.28	0.13	-1.62	-14.80	24.85	28.97	14.48	-	0.33	31.13
Resource depletion	MJ surplus	-	-	-0.52	0.58	0.03	1.64	-0.26	-6.88	-	-	-	-	2.79	-

3.1.3. Comparing scenarios

Environmental assessment result for the whole system: Figure 12 shows the overall environmental impacts for two main scenarios, baseline and after struvite adoption, and sub scenarios options (AF and NAF). The results indicate that even though the struvite precipitation scenario (AF option) has lower impacts for eutrophication (56% less), the baseline scenario is a superior choice under the remaining impact categories (GWP is almost the same and we can see an increase in acidification and resource depletion impacts by 9% and 67%).

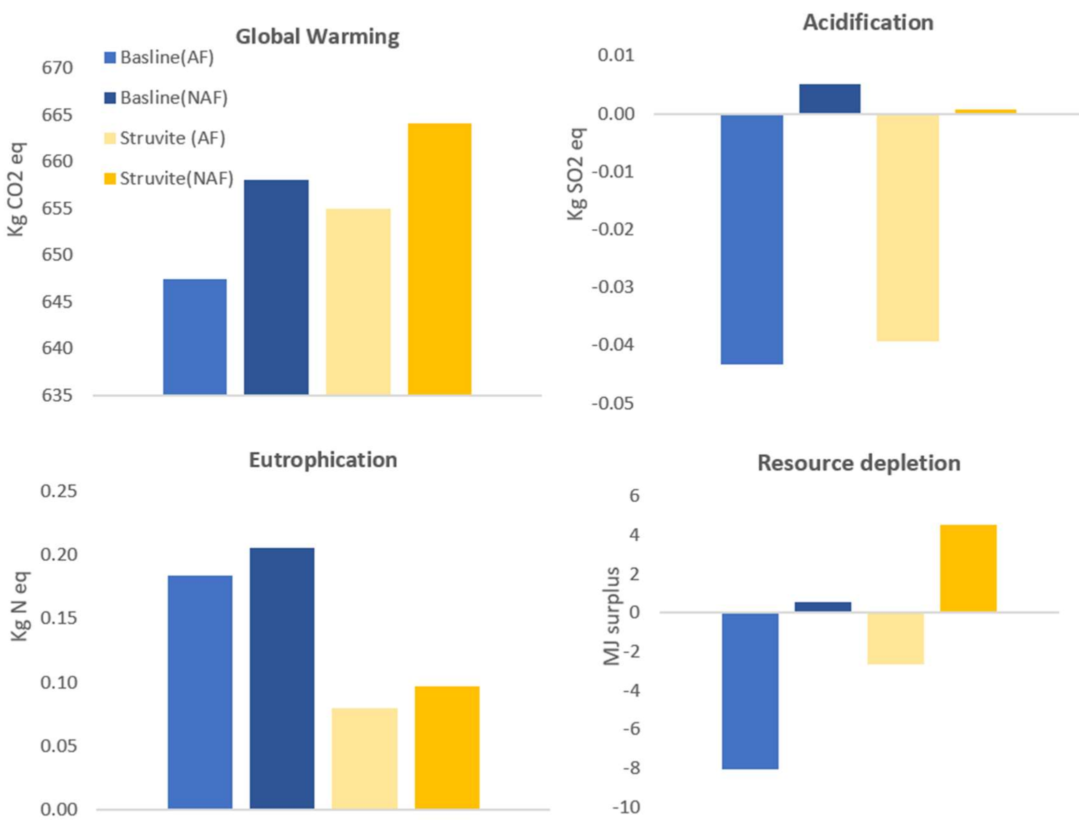

Figure 12. Environmental impacts associated with different scenarios

The magnitudes of environmental impacts are shown in **Table 14**. In the case of the NAF option, eutrophication impact decrease by almost 50%, acidification declines by almost 90%, GWP remains almost the same and we can see a large increase in resource depletion impact (5 times greater).

Table 14. Environmental impacts associated with different scenarios

Impacts category	Reference unit	Baseline		Struvite precipitation	
		AF	NAF	AF	NAF
Global Warming	kg CO_2 eq	647.5	658.2	655.0	664.2
Acidification	g SO_2 eq	-43.40	5.15	-39.28	0.66
Eutrophication	g N eq	183.63	205.63	80.04	96.47
Resource depletion	MJ surplus	-7.9	0.73	-2.64	4.51

Environmental impacts per stage: the baseline scenario includes the lagoon operation and land application stages. In the struvite precipitation scenario, these two stages are included in addition to the struvite recovery unit construction and operation. In this section, the magnitude of each environmental impact is presented. As shown in **Table 15**, in the case of the AF option, GWP in both scenarios is mostly the result of the lagoon phase. In the baseline scenario, the land application phase reduces the global warming impacts. The reason for this observation is that the effluent contains a substantial amount of N and P that can displace commercial fertilizers and the emissions involved in their synthesis. In the struvite scenario, since the amount of N and P in the lagoon effluent are reduced (through being recovered as struvite), the land application phase is the second great source for producing CO_2 eq.

Table 15. Environmental impacts associated with different scenarios (AF option)

Impacts category	Reference unit	Baseline		Struvite		
		Lagoon	Land application	Lagoon	Struvite Recovery unit	Land application
Avoided Fertilizer (AF) scenarios						
Global Warming	kg CO_2 eq	650.20	-3.16	650.20	0.69	-2.05
Acidification	g SO_2 eq	-	-0.05	-	68.82	-39.28
Eutrophication	g N eq	-	0.18	-	3.44	83.07
Resource depletion	MJ surplus	0.156	-8.46	-	4.97	-6.57
Not avoided Fertilizer (AF) scenarios						
Global Warming	kg CO_2 eq	650.20	7.48	650.20	0. 69	7.08
Acidification	g SO_2 eq	-	1.43	-	68.82	5.15
Eutrophication	g N eq	-	205.58	-	3.44	99.50
Resource depletion	MJ surplus	-	0.16	-	4.97	0.58

3.1.4. Model uncertainty and sensitivity analysis

To assess the uncertainty associated with the LCA results, IBM crystal ball software was used to capture the contribution of variables uncertainty to overall model outcomes. This software uses the Monte Carlo simulation method to conduct uncertainty analysis (Gonzalez et al., 2005). This method relies on a computational algorithms that conducts repeated random sampling from distributions of input variables to obtain the probability distribution of the resulting variables/outcomes.

For this stage, all variables were assigned an appropriate distribution based on relevant literature. Subsequently, a number of simulations runs (1,000 runs) were used to estimate the distribution of the environmental impact outcomes by sampling each of the uncertainty variables from their respective distributions. For each simulation run, the difference between baseline and alternative management (struvite precipitation) was calculated for each impact category. The distribution of these differences was used to assess confidence interval in the relative performance of assessed technologies. **Figure 13** shows the uncertainty forecast for the differences of GWP impact category between struvite and baseline scenarios. **Figure 14** shows the magnitude of contributions by key variables to the differences of GWP impact category between struvite and baseline scenarios as a tornado plot. **Table 16** shows the differences between impact categories for two scenarios (S2-S1) considering uncertainty in model. Negative numbers mean that the baseline scenario has greater environmental impact.

Table 16. Differences between environmental impacts for two main scenarios considering model uncertainty

Impact category	**Reference unit**	**AF**		**NAF**	
		Mean	SD	Mean	SD
Global Warming	kg CO_2 eq	7.35	0.16	5.99	0.24
Acidification	g SO_2 eq	3.17	1.69	-4.76	0.02
Eutrophication	g N eq	-103.62	28.05	-108.85	29.08
Resource depletion	MJ surplus	5.14	0.29	3.78	0.00

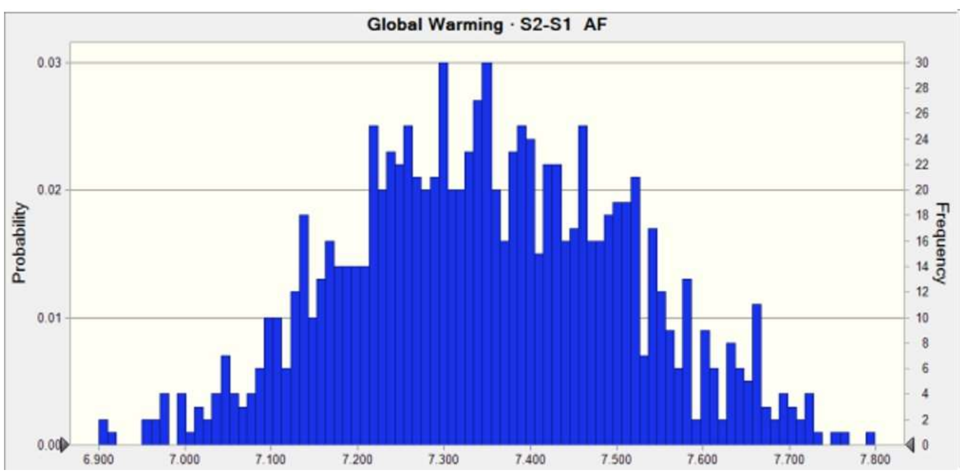

Figure 13. Distribution of difference of global warming potential (GWP) between struvite and the baseline scenario

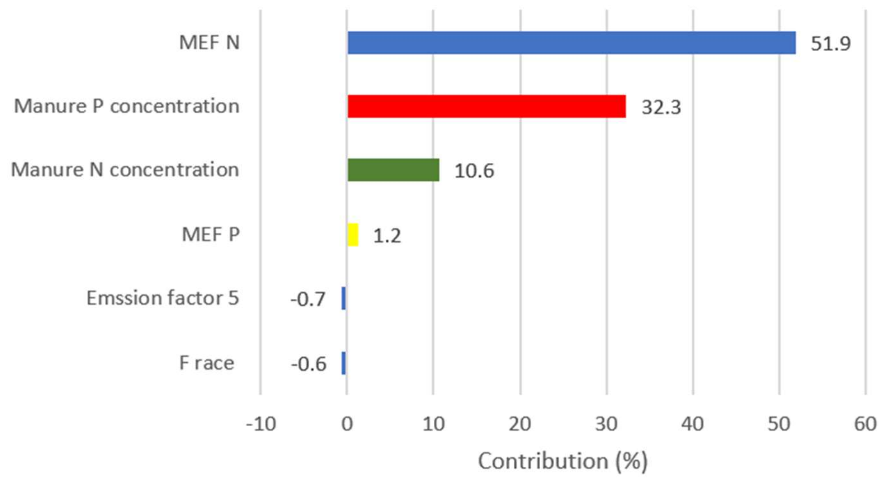

Figure 14. Sensitivity analysis chart for difference of global warming potential GWP between struvite and the baseline scenario

3.2. Economic Assessment

Evaluating the economic feasibility of the struvite recovery showed that under current pricing and cost structure, this technology is not feasible. These findings are contrary to Yetilmezsoy et al. (2017) findings, which suggested this technology is economically feasible in fertilizer industry. Two reasons might explain this disagreement; first, the volume assessed in Yetilmezsoy's study was much greater than that assessed in our study ($500 m^3\ d^{-1}$ vs. $21\ m^3\ d^{-1}$). **Figure 18** illustrates the decline in capital and operating cost per unit struvite by increasing the system scale (through increasing number of animals and thus the manure volume). The second reason might be related to different market condition, including labor and electricity costs, in two regions (USA vs. Turkey).

3.2.1. Capital and operating costs for small and large farms

For this section, data were obtained from the literature. The direct investment adjusted for given scales by using equation (27). Initially costs adjustment was performed for different scales (3,000 and 6,000 herd size) (**Table 17**).

Table 17. Capital and operating cost (in 2020 US Dollars) for two swine farm scales

Items	Small farm	Large farm
		Capital Cost
Direct cost (DC)	42,201	66,221
Indirect cost (IC)	9,142	13,857
Fixed capital investment (FCI)	51,344	80,078
Working capital	5,134	8,008
Total capital Investment	56,478	88,086
		Operating Cost
Total labor cost	16,200	24,300
Electricity annual cost	1,657	3,314
Annual NaOH cost	7,529	15,057
Annual $MgCl_2$ cost	151	301
Maintenance	2,567	4,004
Total operating cost	28,103	46,976

Figure 15 shows the contribution of different cost components (capital and operating) per kilogram of recovered struvite product in small farms. The cost associated with labor and purchasing NaOH accounted for majority of the cost of struvite production (75% of all costs).

Labor cost had the highest contribution to overall unit cost of struvite (%51). Reducing this cost category through empowering the producers or current operators to oversee this process or increasing the level of automation, or both, reduce that cost. The cost of the alkaline agent (NaOH) represented 25% of the unit struvite cost. Although this chemical can be replaced with ammonia (a cheaper alternative), we did not consider that as it increases the N in manure output which can increase environmental concerns. Westerman et al. (2010) used ammonia for pH adjustment, however, the project was not financially feasible. Relying on aeration to shift the pH to the basic range, through CO_2 removal, can potentially overcome this challenge provided a renewable, low-cost aeration design is implemented.

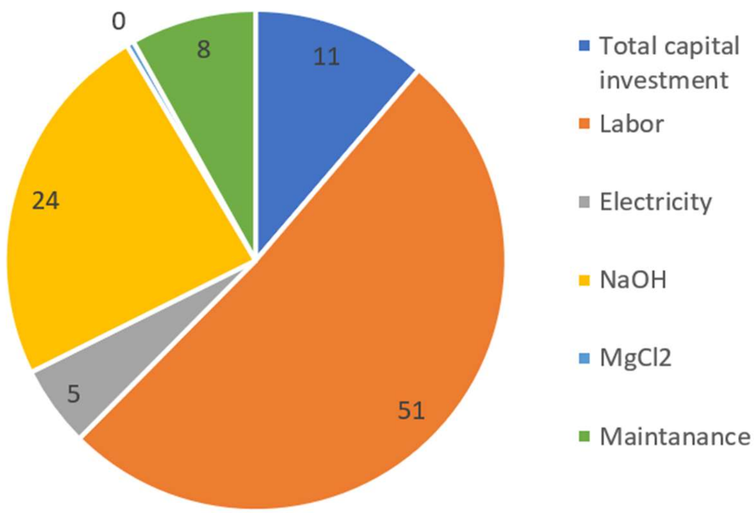

Figure 15. Cost contribution percentages in small farms

3.2.2. Net present value (NPV)

Both the cost and revenue of the system need to be considered to quantify the net present value (NPV) of the project. Product (struvite) sale price was assumed as \$0.5 /kg struvite. Assuming a project lifetime of 15 years, a 6% discount factor, and inflation rate of 2%. Using the cost estimates along with the revenue associated with the struvite sale, the NPV was estimated. **Figure 16** shows a histogram of NPV values generated using a Monte Carlo sampling (1000 iterations) for input variables. The sources of uncertainty in the histogram are both from struvite production and cost of inputs. The forecast chart groups output values into intervals (or bins) for ease of analysis. The bottom axis shows the range of NPV values. The results show that over the project life-time for prices and values of inputs and outputs, the investment does not generate revenue (cash-negative).

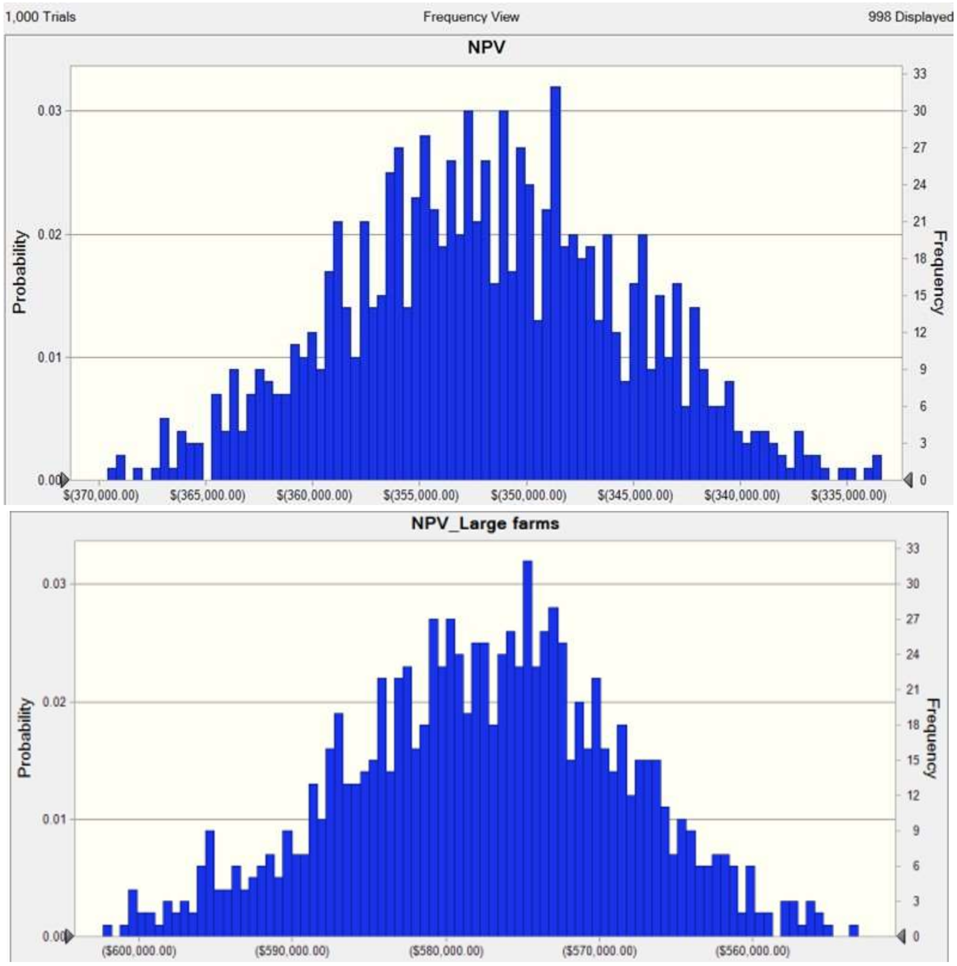

Figure 16. Distribution of net present values (NPV) associated with installing and operating a struvite production system on small swine operations (top) and large swine operations (bottom).

To understand how much a given variable impacts the NPV, a sensitivity analysis has been compiled (**Figure 17**). This chart allows demonstrates quantity and direction of impact for key variables in the struvite production. As shown in the figure below, hourly rate for labor, P content in manure, the discount factor assumed, Mg content in manure, and NaOH cost are the key factors that are impacting the NPV forecast.

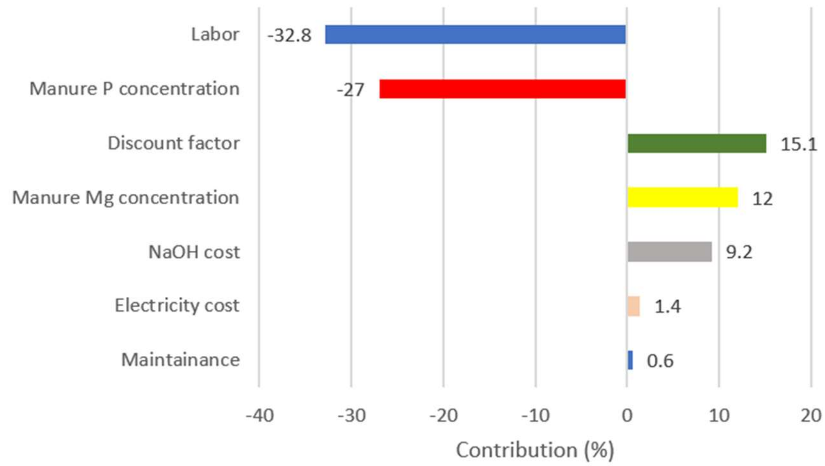

Figure 17. Sensitivity analysis of net present value (NPV)

The initial results from calculating NPV implies that adopting struvite technology is not economically feasible in small and large farm considering current production conditions.

The minimum sale price (MSP) was calculated to determine the break-even point for struvite sale each scale farm scale. The break-even point occurs under conditions that NPV is equal to zero. At this point, the technology adoption does not result in economic loss either have any profit. The MSP per 1 kilogram of struvite was found to be \$14.61 for small swine farms and \$13.21 for large swine farms.

3.2.3. Capital and operating cost per head animal

The capital cost per head animal was slightly lower for large farms (\$18.83 vs \$14.68, almost 30% less). Also, in the case of operating cost, almost the same pattern can be seen, \$9.37 annual operating cost for small farms declines into \$7.83 for large farms (20% decrease). The results indicate that struvite technology can achieve economies of scale advantage through adoption on larger operations. **Figure 18** shows the changes in capital cost, operating cost, and

MSP as a result of technology upscaling. As shown, the benefits gained from scale gradually diminish beyond a certain farm size (12,000 to 15,000 heads).

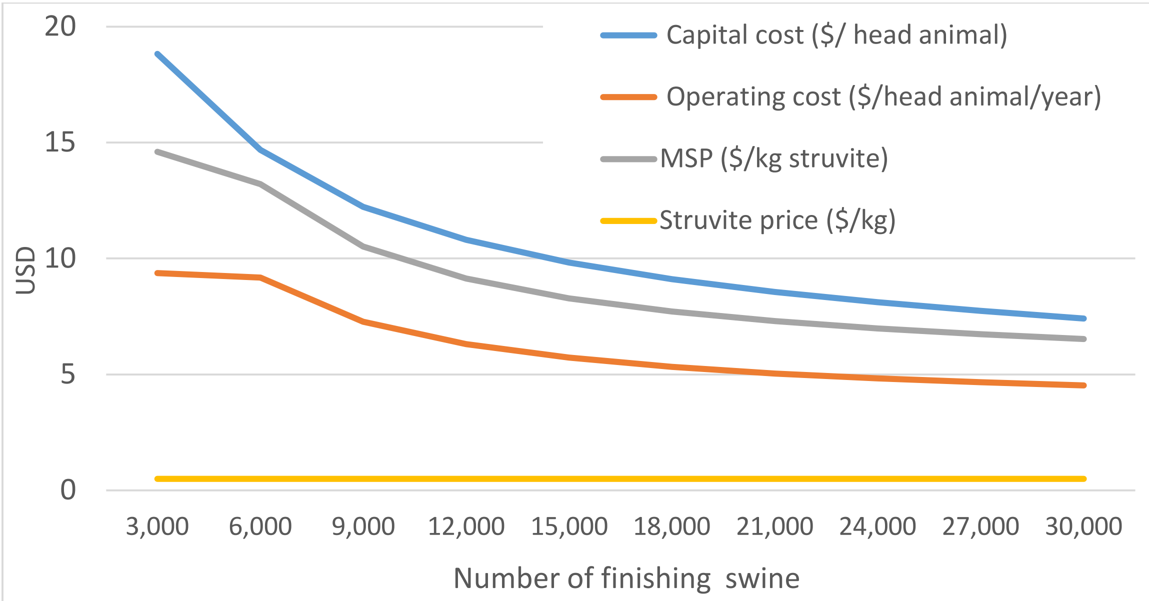

Figure 18. The impact of farm size on struvite technology adoption economics: per animal capital and operating cost, and minimum sale price (MSP) of unit struvite

Although the maximum number of animals on a farm, as depicted in **Figure 18** is assumed to be 30,000 animals, the 99th percentile farm size is 20,000 head animals. In addition, economic and environmental considerations associated with liquid manure transportation to central facilities further reduce struvite recovery performance. This means there is a limit to the economies of scale" that could be realized for struvite recovery on swine operations.

Although some improvements are possible to reduce the capital and operating costs, they are not sufficient to move the technology performance towards positive cash flow. For instance, considering a scenario in which producers assume the labor duties and therefore eliminate labor cost, the minimum sale price (MSP) for a 30,000 head farm size is reduced from $6.53 to $5.11. Similarly, assuming government grants providing 50% of the capital cost required, the MSP would be decreased from $5.11 to $4.64. While the cumulative impact of these adjustment does not deem it economical, continued technology improvement to reduce chemical inputs coupled with environmental credit programs to support nutrient concentration technologies can help uptake of struvite recovery technology.

Overall, although the adopting this technology is not currently profitable, it can be a valid option to reduce non-point source (NPS) phosphorus risk in areas with eutrophication risk by employing pollution trading program. Nutrient trading programs seek to attain environmental goals in the most cost-effective way by using market forces to achieve highest reduction in nutrient contamination avoidance. Through these programs, authorities set an upper limit on the amount of pollutant allowed among a group of stakeholders (including water treatment utility districts, cities, and industrial entities). Each stakeholder can then buy or sell allocations of pollutant emission with an advantage to entities that can capably reduce pollutant level through selling unused pollution credits to those for whom cleaning up would be more expensive. Under a nutrient trading program, this study can provide insights into pricing phosphorus mitigation through struvite production by using minimum sale price (for the technology scale desired) and the impacted region that can benefit from its implementation.

3.3. Site selection

Figure 19 shows the counties soil phosphorus index (PI) level and selected farms based on considered criteria explained earlier in this report. The PI has been identified as average amount of P in soil samples. The result of applying the criteria, limits farms to 18 farms as illustrated in **Figure 19**. Suggested farms are between 0.1 to 0.3 miles away from waterbodies. The median number of animals in these farms is 7,260; Average is 7,978.

On average, manure TP and N concentration in one of the farms is about 3 and 33 kg day^{-1} respectively, which means 6 tones yr^{-1} struvite production in each farm is expected. Producing this amount of struvite, prevents P leaching by 73 kg yr^{-1} and P runoff by 37 kg yr^{-1}. It also means that 115 kg N yr^{-1} can be captured and reduce in runoff.

Considering NC conditions, results showed that installing struvite technology in either of suggested farms can result in reducing eutrophication potential by average 828 kg N eq yearly (600-1,500 kg N eq). Considering about 8,000 swine on average for suggested farms and 0.18 kg N eq per head animal per year as potential of eutrophication, adopting this technology will result in reducing eutrophication potential by about 56% compared to the baseline scenario. The exact benefit to a specific watershed can be assessed after quantifying all sources of P loading to waterbodies and the value of mitigating swine-driven P loading.

Figure 19 also illustrates that Sampson and Duplin counties are location for more than half of these 18 farms that meet the site selection criteria (**Figure 10**). PI for each county is labeled and green points present 18 farms that meet the site selection criteria.

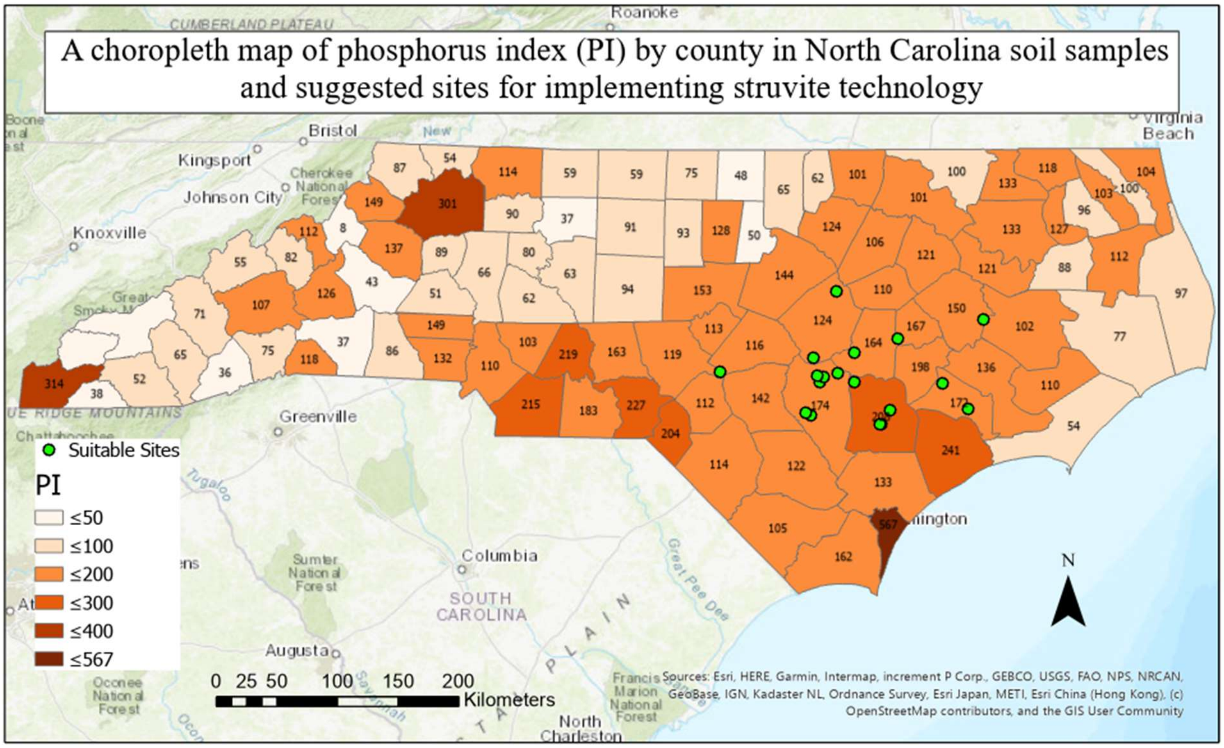

Figure 19. A choropleth map of phosphorus index (PI) by county in North Carolina soil samples submitted for analysis to NCDA&CS laboratory between years 2019 and 2020.

Suggestions for future research are summarized in the following:

- Investigating the environmental impacts of adopting the technology when using alternative operating methods such as (1) aeration or use of other less expensive chemicals to adjust pH instead of using NaOH (2) solar-based electricity from local on-farm microgrid instead of utility purchased electricity.
- Studying the impact of covering the lagoon in addition to adding struvite unit to investigate how much covering a lagoon can reduce the GWP and how that can play role in other phases (struvite and land application) and other impact categories.
- Conducting environmental impacts and economic assessment for other alternative manure management technologies especially by focusing on treating solid portion of manure. This can be helpful to understand which fraction of manure in the current manure management

(lagoon) should be prioritized for alternative processing solid-portion (sludge) or liquid portion (supernatant). The liquid-portion is more frequently cycled by irrigation so a reduction can help control current P and N emissions. Sludge, on the other hand, is a reserve of P and N that will have to be managed at some point to ensure compliance.

3.4. Conclusions

Environmental impacts assessment:

- Environmental impacts amount per head animal is same for large and small farms, which means scaling up will not be helpful to reduce the environmental impacts.
- The Struvite precipitation scenario is a better choice for reducing the eutrophication impact category while in the other three categories, the baseline scenario has lower impacts.
- In both scenarios, not utilizing lagoon effluent as efficient fertilizer (NAF option), results in a surge in environmental impacts (all four categories).
- The GWP (as one of the most challenging environmental impacts globally) in both scenarios is mostly because of methane emissions from the lagoon (95% to 97%) and adopting a struvite precipitation system is not beneficial in this case.
- Struvite precipitation scenario reduces eutrophication impact by almost 50%. Adopting this technology is specifically advantageous for areas with eutrophication potential or risk (near to nutrient-sensitive watersheds or area that soil has a high density of phosphorus)

Economic feasibility assessment:

- The capital and operating costs per head animal decline as the herd size increases.
- The labor cost and NaOH purchasing costs have the highest contribution on final costs per kg product (struvite).

- Considering $0.5/ kg struvite, the NPV value is negative for both farm sizes, which means under given assumptions adopting this technology is not feasible.
- The MSP value for the break-even point is $14.6 and $13.21 for small and large farms.
- Although making some adjustments such as scaling up, cutting labor costs by training producers, and providing grants to cover a portion of capital cost could improve the condition, none of them (or a combination of them) are enough to change the results and make adopting the technology feasible under current fertilizer pricing.
- Nutrient trading programs can be a potential solution to overcome economic barriers for adopting this technology especially in locations with nutrient-sensitive watersheds or high P density soils.

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