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Ecological Impacts of Food Waste Digestate Management in an Agricultural Watershed

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A Thesis Submitted in Partial Fulfillment of the Requirements for the Degree of Master of Science in Environmental Science

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Abstract

Globally, more than two billion tons of food are wasted each year, creating significant economic and environmental issues. Anaerobic digestion (AD), a process to convert organic waste to biogas, is a potential waste-to-energy alternative to landfilling wasted food (FW). Liquid digestate, a by-product of AD, is typically stored on-site in ponds prior to field application as a fertilizer, presenting risks of greenhouse gas (GHG) emissions and nutrient release to sensitive waterways. To assess this risk, we used a co-digestion (FW plus manure) facility and associated row-crop disposal system in an agricultural watershed in western New York, USA as a case study. A literature review of gaseous N and nitrate losses was complemented by targeted empirical measurements. We developed a mass balance model of nitrogen (N) across the digestate disposal pathway (storage and field application) and assessed nitrate and GHG losses relative to traditional manure and inorganic fertilizer practices. Sensitivity scenarios evaluated the volume of FW processed, crop type, and spreading practices. We validated results using the geospatial Soil and Water Assessment Tool (SWAT) and found good agreement between approaches. Digestate N content and seasonal variation in storage volume and application rate controlled nitrous oxide release at both stages. Ammonia volatilization was the dominant gaseous loss pathway, with nitrate leaching as the highest overall loss for digestate N. Field level losses for digestate were greater than stored manure or inorganic fertilizer, and increased significantly with higher application rates. However, at the watershed scale, current and two-fold greater FW processing levels did not substantially increase nitrate loss or global warming potential, as long as the field application rate remains constant. These findings suggest that sustainable diversion of FW from landfills to AD includes a decentralized strategy, with smaller digesters and sufficient storage and adjacent cropland.

Keywords: Digestate, mass balance model, SWAT

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List of acronyms

AcoD: Anaerobic Co-digestion

AD: Anaerobic Digestion

CAFO: Concentrated Animal Feeding Operations

CH_4: Methane

CO_2: Carbon dioxide

EF: Emission Factor

FW: Food waste

GIS: Geographical Information System

K: Potassium

LSM: Land Suitability Modeling

LTHIA: Long-term Hydrological Impact Assessment Tool

N: Nitrogen

N_2O: Nitrous oxide

NH_3: Ammonia

NH_4^+: Ammonium

NO_3: Nitrate

NPK: Nitrogen, Phosphorus and Potassium in fertilizers

P: Phosphorus

PO_4^{3-}: Phosphate

SWAT: Soil and Water Assessment Tool

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1. Introduction

Wasted food is a global problem, posing an increasing management crisis in industrialized countries (Melikoglu et al., 2013; Bao et al., 2015). The Food and Agricultural Organization (FAO) defines food waste (FW) as food resources in the supply chain that are safe for human consumption but discarded nonetheless, also defined as “wasted food” by EPA (EPA, 2019; FAO, 2014). Globally, more than two billion MT of food are wasted from farm-to-fork, according to a recent report by the World Wildlife Fund (2021). In the United States, where 35 million MT of food waste are generated annually (EPA, 2015), wasted food is akin to about one-third of the calories each American consumes daily (Conrad et al., 2018). Roughly 40% of food produced or imported in the US is landfilled (Dana Gunders and Getting, 2015). Economically, this waste was valued at $936 billion at the retail level and $165.6 billion at the consumer level in 2008 (Buzby and Hyman, 2012). The world’s FW production is projected to increase threefold, i.e., to 10 million MT daily, by 2100, as per ‘the business-as-usual’ projection (Hoornweg et al., 2013). Globally, 30 to 40% of wasted food (1.3 billion metric tons [MT] per year) is landfilled (Godfray et al., 2010; Gustavsson et al., 2011) (Godfray et al., 2010; Gustavsson et al., 2011; Hoornweg et al., 2013; Melikoglu et al., 2013; Bao et al., 2015); this number is more concerning in the US, where FW occupies 21% of landfill space (USDA, 2016) and only 5.3% of FW is diverted for valorization through other pathways (EPA, 2015). Currently, greenhouse gas (GHG) emissions associated with FW rank third-highest after the country-level emissions of the US and China (FAO, 2011) with wasted food accounting for 3% of the global GHG emissions during landfilling alone (Papargyropoulou et al., 2014). Management of the food waste stream presents significant challenges beyond resource and economic losses. Particularly concerning environmental impacts are caused by increasing waste volume, disease transmission, pollution of water, air, and land (FAO, 2011). However, regulatory policy, mechanisms for valorization, and understanding of the implications of by-products of valorization are lacking (Clarke et al., 1999; Ravindran and Jaiswal, 2016). The immediate environmental concern, however, is posed by challenges in food waste management at regional and local levels.

One approach to food waste management already taken in some regions involves exploiting food industry waste for high-value products, including bioenergy (Ravindran and Jaiswal, 2016). For instance, European countries committed to reducing 40% of GHG emissions by 2030, compared to the baseline of 1990, by mandating 20% renewable energy sources (e.g., animal and food waste) (European Council, 2014). In the United States, only 8% of biomass potential has been exploited (EPA, 2019). Still, bioenergy valorization of FW through processes such as anaerobic digestion (AD) can generate 18.5 billion cubic meters of biogas per year (USDA, EPA, 2014) and help to achieve renewable energy targets while reducing landfilling. By 2030, the EPA and USDA plan to divert 50% of wasted food away from the landfill, suggesting the need to evaluate potential impacts of alternative disposal pathways. Specifically, in New York State (NYS) a Commercial Organics Law, the Food Donation and Food Scrap Recycling Act, passed in 2017 bans landfilling of organic waste from any commercial food waste generators that produce 1.81 or more MT of FW weekly (Senate Bill S2995). Together, the largest food generators in New York like supermarkets, restaurants, and hospitals produce roughly 360,000 MT of wasted food and food scraps per year, that must be diverted from the landfill to composting or anaerobic digester facilities (except for New York City) (Cole Rosengren, 2017; DEC, 2021). However, we need to reconcile the trade-offs linked to such valorization technologies with the management of byproducts and release of GHG and nutrients to ecosystems that occur, even with state-of-the-art waste management models.

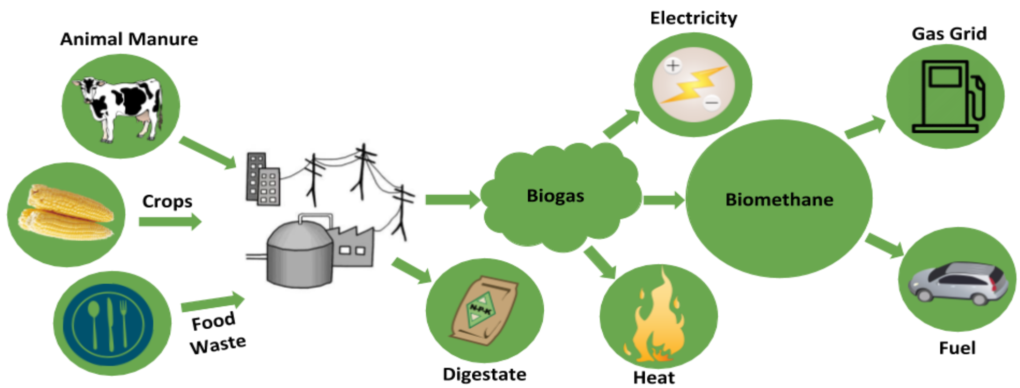

Figure 1: Schematic diagram of Anaerobic Co-Digestion (AcoD) Process.

Anaerobic digestion is a commercial valorization technology used to convert organic waste to energy to minimize the hazards and costs of open disposal or landfilling. Microorganisms, primarily acidogenic and methanogenic bacteria breakdown organic matter in the absence of oxygen and generate methane as a byproduct (Chen et al., 2007)(Figure 1). In the past, AD facilities were used primarily for odor reduction and nutrient management of large-scale livestock operations (Binkley et al., 2013; Gould, 2015; Powers et al., 1999). However, AD may also minimize additional environmental problems by converting bio-wastes into two potentially useful byproducts: (1) biogas, a form of renewable energy, and (2) anaerobic digestate. The first product, biogas, is comprised of methane (55-75%), carbon dioxide (25-50%), water, oxygen, and trace gases such as hydrogen sulfide (Wellinger et al., 2013). The second byproduct, the nutrient-rich residual liquid referred to as digestate, has a wide range of potential applications, including bedding for livestock, soil amendments, and fertilizers.

Potential feedstock (material input) for AD are derived from a variety of sources, including (1) agriculture (animal wastes including dairy manure, pig slurries, wastewater, bedding, and cleaning overflow; or crop residues, such as unused energy crops, stalks, straws), (2) municipal waste (sewage/waste/sludge), (3) industry (food waste, paper, and pulp, textile and petrochemical refineries industrial wastes), and (4) post-consumer or residential waste (wasted food, paper scraps, etc.) (Chen et al., 2007). The biogas yield may be higher when varieties of feedstocks (e.g., livestock manure plus waste food) are combined in co-digestion (AcoD) (Chen et al., 2007), potentially by balancing material C:N with cellulase activity (Idris et al., 2004; Ward et al., 2008). However, in some cases, feedstock availability may be a concern for large-scale, centralized digesters. Recently, however, commercial FW has emerged as a readily available feedstock in NYS, motivated by the restrictions on organic waste landfilling and the potential for valorization (Cole Rosengren, 2017).

Awareness of resource recovery and economic benefit has increased interest in AD facility establishment (Banks et al., 2011), with a focus on the benefits of AcoD such as biogas generation capacity, digestate as potential fertilizers, and cost-benefit analysis of chemical fertilizers versus organic digestate (e.g., Holm-Nielsen et al., 2009; Zamalloa et al., 2011). Much less attention has been paid to the logistical challenges, policy barriers and possible environmental threats that occur at both the storage and field application phases at centralized AD facilities (Alburquerque et al., 2012; Edwards et al., 2015; Möller and Müller, 2012; Möller and Stinner, 2009, Tel-Tek, 2013; Ebner et al., 2015). Overflow of storage ponds may create significant downstream ecological problems (Alexander et al., 2007). Further, because digestate is heavy and requires large trucks to transport material from the storage facility to the field, there is a limit to practical transportation distance and thus crop land must be readily available (Armington, 2019). As a result, farm managers must weigh the risk of continued storage versus field demands and the limitations of spreading posed by distance and weather. Previous work on manure, energy crops, and pig slurry feedstock can be used to shed light on the management implications of food-waste based AcoD. For instance, Nkoa (2014) addressed agronomic benefits and environmental and health risks of anaerobic digestate management, which occur at two crucial phases: storage and field application (Figure 2). However, only a few studies have evaluated the uncertainties and risks associated with storage or land application of FW digestate (Möller, 2009; Nkoa, 2014; Rehl and Müller, 2011), and no research to date has studied the potential risks of FW digestate disposal from a watershed or geospatial perspective. These poorly understood risks will be exacerbated as regional digestate volume increases (NYS DEC, 2019).

The direct GHG emissions from agriculture are approximately 10% of total US emissions (Bellarby et al., 2008; Heller and Keoleian, 2015; USEPA, 2019), with roughly 14% of agricultural emissions resulting from manure management (US EPA, 2019). In the case of N_2O, with a global warming potential of 298 relative to the CO_2 (Petersen, 2018), agriculture contributes 70% of total emissions (Tenuta et al., 2001). GHG emissions associated with manure may be significantly reduced through the AD process (Burg et al., 2018); however, the remaining N in digestate (especially mineral nitrogen) can be released in the gaseous form i.e., (N_2, N_2O and NH_3). GHG production and nutrient release to waterways may occur across the digestate life cycle, but especially during the prolonged storage (often many months) and field application stages (Hobson and Wheatley, 1994)(Figure 2). In the case of wasted food, which is typically rich in organic components like protein and fats, GHG emissions may be higher as the AcoD process may fail to stabilize the food waste constituents (De la Rubia et al., 2010), but research is lacking in this area. While GHG emissions associated with the AD storage phase from feedstock like energy crops or animal slurries have been measured (e.g., Hansen et al., 2006; Menardo et al., 2011) or modeled (Tel-Tek, 2013; Ebner et al., 2015), few studies have empirically evaluated the loss of GHG from storage ponds from food-waste based digestate and compared these rates to relative losses across the life cycle.

Following storage, the most common end-of-life scenario for digestate is application to fields as fertilizer (Pivato et al., 2016) where residual N and P, along with other essential macro and micronutrients (Coruzz and Bush, 2001) in digestate may displace inorganic fertilizer use. Farmers have been spreading livestock manure on fields for centuries, and more recently, have adopted AD as a potential value-added process that retains the fertilizer value while generating biogas. While there is some debate regarding the relative quality of AD, manure, and chemical fertilizers (Alburquerque et al., 2012b; Möller and Müller, 2012; Pezzolla et al., 2012), the anticipated increase in AD may necessitate the replacement of chemical fertilizer with AD in some regions. While feedstock characteristics determine the digestate attributes and fertilizer value (Comino et al., 2010), the liquid effluent is rich in nutrients like nitrogen (both inorganic and organic), phosphorus, and potassium, making it an excellent potential fertilizer (Möller et al., 2009; Nkoa, 2014) that adds micronutrients, and enhances soil density, organic matter, water holding capacity, texture, and pH for both field and greenhouse cultivation (Garg et al., 2005).

Inorganic fertilizer supplies readily available N to the soil for plant uptake, while organic fertilizers, such as digestate or manure, provide both available inorganic (nitrate, nitrite, and ammonium) and organic (amino acids, protein, nucleic acids, urea, etc.) nutrients that must be converted to bioavailable forms in the soil prior to plant uptake. After spreading, these interactions compete with plant uptake and lead to immobilization and transformation, with some N released to the atmosphere as N_2, NO_x, and NH_3. Ammonification of organic nitrogen produces NH_4^+ that may volatilize to the atmosphere as NH_3 (a short-lived GHG) or undergo nitrification to NO_3^- and subsequent denitrification to N_2 prior to release to the atmosphere. Incomplete denitrification produces the GHG N_2O. The relative emissions of these three gases in AD application is poorly understood, especially relative to manure or traditional fertilizer application.

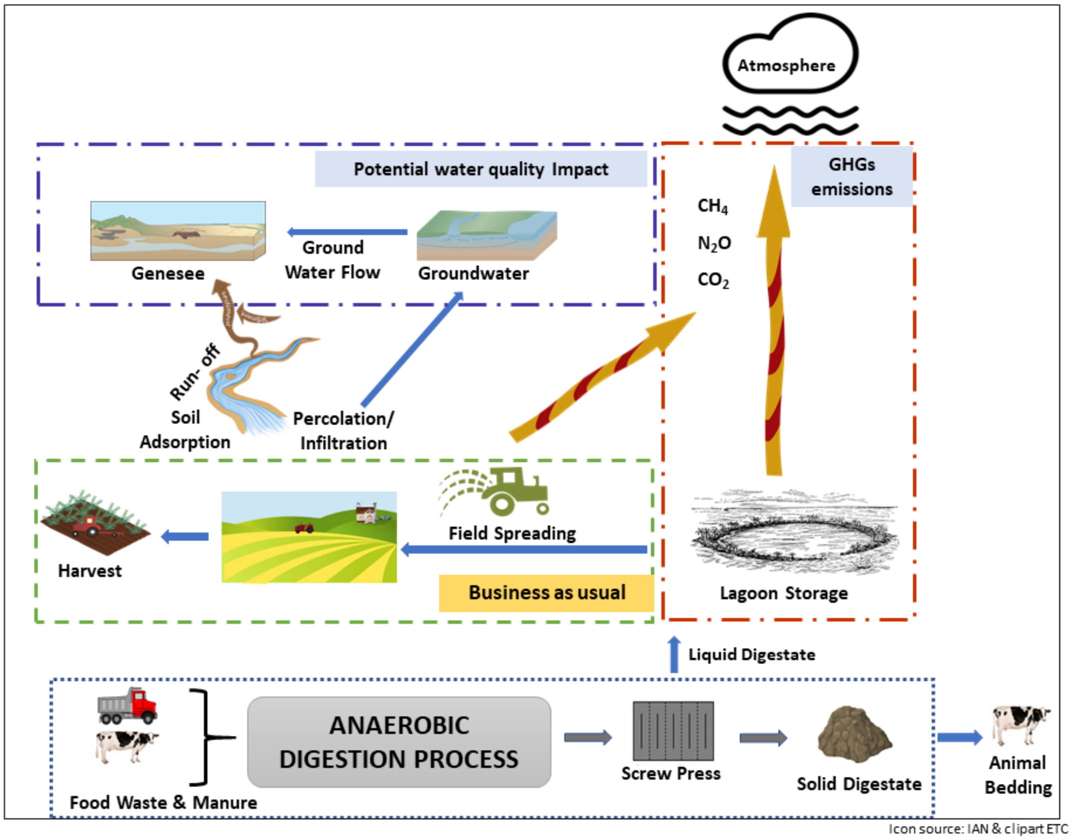

Figure 2: The business as usual scenario of food waste AD and subsequent disposal of the digestate is illustrated on the left side of the diagram, where liquid digestate is stored in open lagoons and then opportunistically spread on nearby agricultural fields, with some risk to local waterways and emission of high global warming potential greenhouse gases.

Excess N not taken up by crops or released to the atmosphere is either adsorbed onto soil or leached to groundwater and surface water. The excessive use of fertilizers and digestate, exceeding the potential soil adsorption capacity, may release nitrogen from the soil into streams (Yao et al., 2012). Nitrate (NO_3^-) release from agricultural fields to water bodies is a long-standing ecological concern (Casalí et al., 2008; Soldat and Petrovic, 2008) because of the risk of eutrophication (Correll, 1998). In general, N loss to water bodies depends upon factors like current soil N level, slope, soil types, and climate (Oenema et al., 2003). New York's state-level policy addresses uncertainties of digestate application and prohibits field application if rain is in the forecast within 48 hr when the agricultural land is waterlogged, and in case of cracked down soils (DEC, 2019). Further, because of potential toxicity risks associated with digestate, it is typically spread only on bare fields to avoid direct contact with plants (Alburquerque et al., 2012; Nkoa, 2014; Owamah et al., 2014; Pivato et al., 2016). Application of organic fertilizers, such as digestate, typically leads to N immobilization, increasing soil N content in the long run (Gutser et al., 2005), which may decrease runoff to waterways if inorganic fertilizers are replaced with digestate.

The balance between nutrient retention by soil, plants, or microbes, runoff or leaching to groundwater, and emission to the atmosphere will depend on the volume of digestate applied to fields along with a variety of local environmental factors. Because digestate application may rise with landfill diversion to AD facilities, in turn, runoff of both nutrients and carbon may increase, leading to stream water quality impairment (USEPA, 2015; Walsh et al., 2012). However, there is great uncertainty in the relative magnitude of N loss to the atmosphere and waterways under different management scenarios, including substitution of AD for inorganic fertilizer and an increase in AD application volume over current levels. The difference and variability in the composition of digestate relative to inorganic fertilizer make it challenging to predict the dynamics at the field level. Although regulations are in place to minimize potential loss to waterways, the potential for a significant release of nutrients remains but is poorly understood compared to use of inorganic fertilizers and/or fresh manure.

Consumers in the US have access to food grown locally, nationally, and internationally. With an impending ban on FW disposal in landfills, food waste resources may be concentrated in large, centralized digester facilities in agricultural areas. In regions where there is both local agricultural production and import of food products, the generation of food waste represents a significant influx of nitrogen to the regional ecosystem. Previous research has highlighted the benefits of anaerobic digestion as a quick fix for waste management and AD technologies might succeed in partial valorization through the generation of electricity and natural gas. It is not clear, however, that the volume of digestate can be accommodated without posing additional environmental risk to sensitive stream and lake habitats and enhanced release of GHG. Food waste-based digestate disposal and ecological risk thus stretch across geographical boundaries.

Considering the limited information regarding food waste and manure co-digestion, this work contributes a novel perspective on food waste-based digestate disposal and gives a holistic overview of nutrient loss across disposal pathways. The potential issues associated with ramping up food waste diversion to AcoDs in agricultural areas are inherently regional. As such a case study approach assessing nitrogenous greenhouse gas and runoff emissions was conducted using an agricultural watershed in Western New York State, USA. This work attempts to fill the literature gap about the potential ecosystem risks of increasing food waste based digestate disposal in the Great Lakes region and incorporates a comparison to traditional manure management and use of inorganic fertilizer. An assessment of potential nutrient loss across each phase of digestate management will promote environmentally conscious decision making and identify sustainable food waste management options.

2. Methodology

A regional commercial-scale anaerobic digester in a predominantly agricultural watershed in Western New York State, USA, was selected for this case study. We applied a mass balance approach to current and future management scenarios using parameters derived from empirical data, current management practices and literature sources. Estimated non-point source N losses to the atmosphere and waterways were projected to the watershed level, and we validated results at the sub-basin level using a geospatial hydrological model, the Soil and Water Assessment Tool (SWAT). Operational parameters for digestate and manure management came from a regional co-digestion facility and dairy farm along with the associated crop farm that manages the AD field application. We used 2019 as the base year for the model.

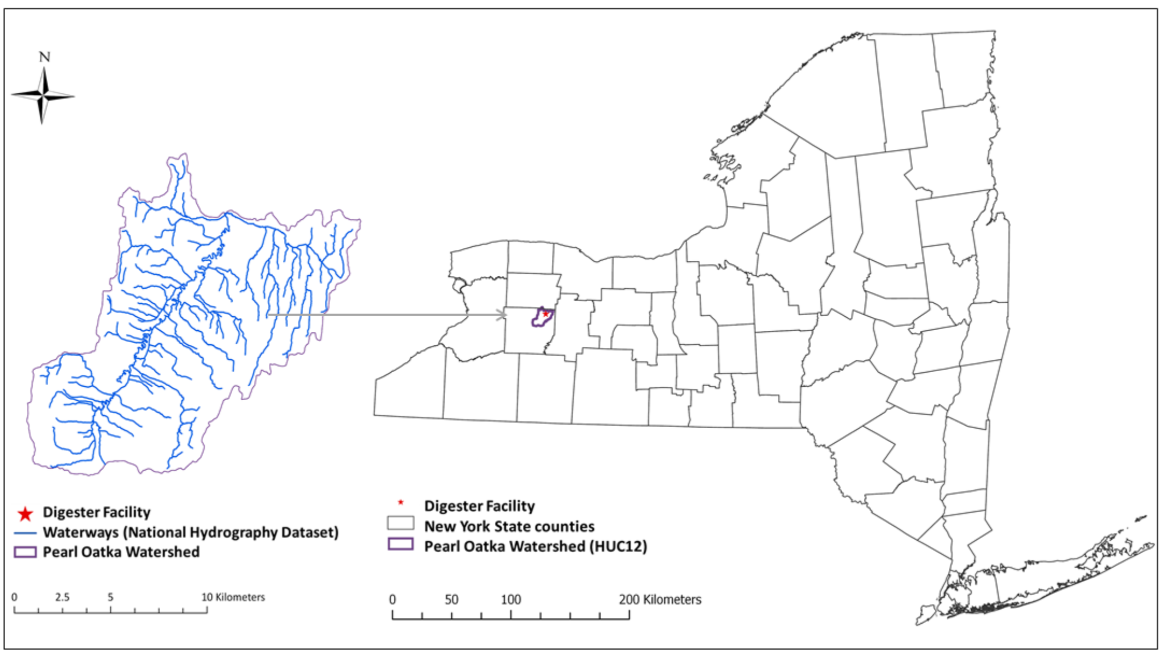

Figure 3: Location map of digester facility and Pearl Oatka Watershed

Study Area and Facility Description

The digester facility and associated dairy farm lie in the Pearl-Oatka Creek watershed (14,700 ha) (Figure 3), a sub-basin of the Oatka Creek watershed in the Lower Genesee River basin that channels through Wyoming and Genesee counties. The watershed is comprised primarily of agricultural land (51%, mostly row crops), forests (27%), wetlands (8%), and <2% developed.

There are five registered dairy concentrated animal feeding operations (CAFOs) within the watershed boundaries, with a total of 4833 cattle and 2,233 heifers (Organic Resource Locator (NYS Pollution Prevention Institute, 2017). Of these, 2100 mature cows were co-located with the digester. The Genesee River is a major river network contributing to Lake Ontario. The climate of Western New York is influenced by Lake Erie and Lake Ontario, two of the Laurentian Great Lakes, with long and cold winters and relatively warm summers. Similar neighboring watersheds within the region also contain the Finger Lakes, eleven glacially formed lakes that are a significant freshwater and economic resource. The area has an elevation of around 283.8 – 545.5 m (US Census Bureau) with an average annual precipitation of 87 cm of rainfall, 251 cm of snowfall and average humidity of 81% (morning) and 61% (evening) (US Climate Data).

Data acquisition for mass balance

We estimated losses to waterways and the atmosphere at each step of the digestate and manure management cycle from the storage phase through field application (Figure 4). The input to the model is the total nutrient content of the digester output plus the associated manure stored in the same pond system. The equation's right side includes the potential nutrient loss pathways as inputs to the atmosphere, ground- or surface water, and through crop harvest. P_N is loss at the digestate storage pond phase through gaseous emissions. At the field level, we use the approach of (Oenema et al., 2003) to estimate the budget at the farm-gate, with input of N estimated from crop fertilizer recommendations and output through the soil surface (atmospheric flux, A_N), leaching to groundwater, and overland run off (W_N). Additional farm-gate losses such as crop harvest (C_N), or retention in the field through adsorption or incorporation into soil organic matter (S_N), were not estimated and for the purposes of this study, we focus on release of gases to the atmosphere and leaching and runoff to local waterways.

Literature-based data were obtained from scientific journal articles accessed using search platforms including Google Scholar, Wiley Online Library, and the Web of Science, government reports from, e.g., the United States Environmental Protection Agency, American Biogas Council, etc. Search terms included: Anaerobic Digesters, GHG emissions during manure/digestate storage, GHG emissions in agriculture, nutrient leaching from fields, effects of digestate disposal, life cycle assessment of AD, FW based anaerobic digestion, and agricultural (N) runoff from anaerobic digestate. We selected literature based on cattle manure as feedstock (digestate vs. traditional manure), climate of the study area, management decisions at storage and field application phases, and for nations with similar economic development to the USA. As much as possible, field studies, rather than laboratory incubation studies, were chosen. Digestate-based scenarios were limited to AD operated on commercial FW and livestock waste, excluding studies conducted of household, WWTP, and industrial capacity.

The commercial biogas facility studied (Appendix, Figure A.3), is New York's largest biogas plant, with annual energy generation of approximately 1.4 MWh. The target feedstock composition is 70% wasted food and 30% manure, with an import of approximately 52,000 m^3 of FW from a variety of commercial food processing operations. The mean daily digester output is approximately 170 $m^3\,d^{-1}$. Manure is sourced from the on-site CAFO. Dairy and process waste (e.g., cheese, whey, tomatoes, and soup), and fat, oil, and grease (FOG) are collected from a radius of roughly 100 km. Solids are removed from the feedstock at the inlet to the digester. The facility has three on-site digestate and manure storage ponds and one remote lagoon connected by a buried pipeline (Appendix, Figure A.4 and Figure A.5) that in total provide storage for an annual production of 62,100 m^3 of liquid digestate (in 2019), along with excess fresh manure. Fresh digestate and manure are pumped into Pond 1 and distributed to Ponds 2, 3, and the satellite pond; digestate is extracted from Pond 1 or the satellite pond for field spreading.

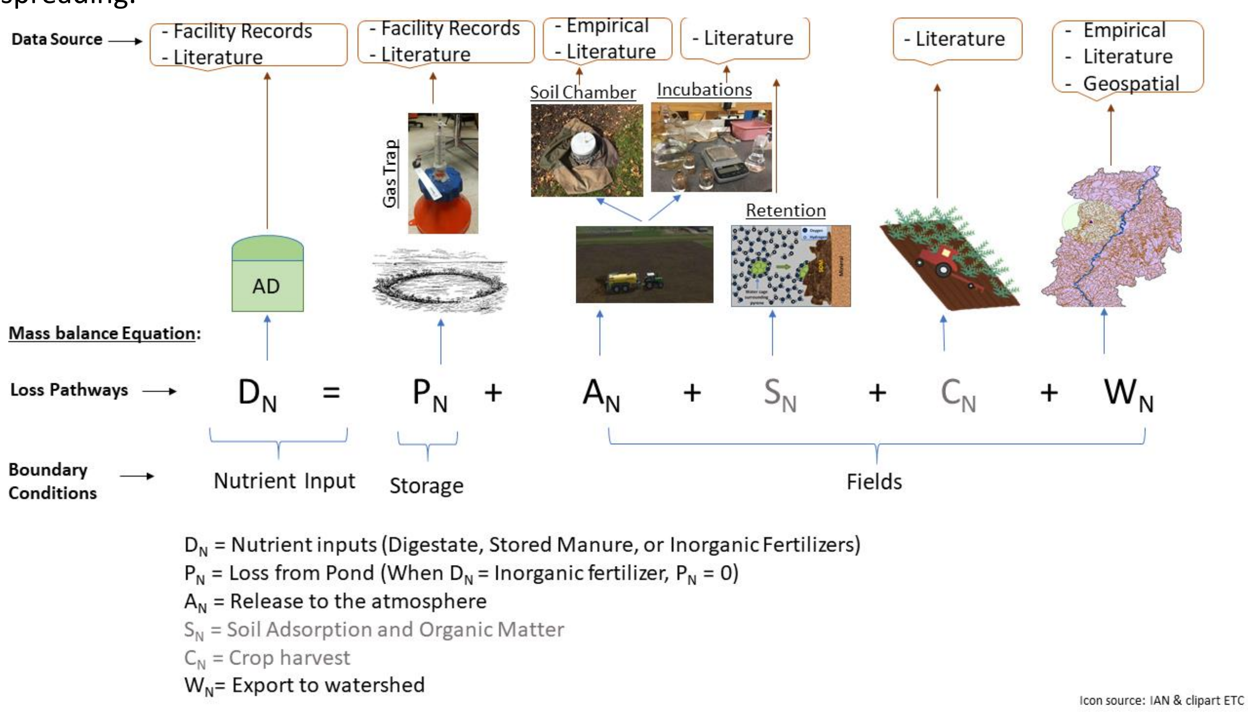

Figure 4: Nitrogen mass balance approach to understand the potential ecological impacts of food waste digestate management, showing each potential loss pathway and indicating the source of data used to parameterize the model: empirical measurements, literature review, facility records, and geospatial model predictions. Values in gray text were not included in the mass balance model.

2.1 Properties of digestate and manure

We sampled digestate at the outlet of the digester and from the primary storage pond roughly monthly during 2019. These samples were sent to Agro-One (Ithaca, NY) for analysis of nutrient content. The mean composition of manure collected from 5 manure-only storage ponds at random intervals in 2018 and 2019 (n=2-4 per pond) was analyzed by Dairy One (n=14) and provided by the farm manager. The N content of fresh manure was obtained from Jokela et al. (2010) and is the mean value of >2,300 dairy cow manure samples. The concentration of N in fresh digestate and fresh manure were used to estimate the mass of N entering the storage ponds. The concentration of stored digestate and manure were used to parameterize the total availability of liquid N resources at the watershed level and the field spreading sub-model and the geospatial (SWAT) analysis.

2.2 Digestate and manure generation

At the digester, storage ponds contain a mixture of digested food waste and fresh manure, a common practice for co-located facilities. We used a constant input of digestate to the pond (170 m^3 d^{-1}) based on facility records. Roughly 38 m^3 d^{-1} of fresh manure was sent to digester, with the remainder of onsite manure entering the storage ponds directly ([2,100 milking cows x 0.17 m^3 manure cow^{-1} d^{-1}] – 38 m^3 d^{-1} to digester = 88 m^3 manure d^{-1}). The total volume of manure generated in the watershed outside of the co-located CAFO was based on the number of mature (2,833) and juvenile (2233) dairy cows and an estimated production rate of 68 and 22 kg manure d^{-} for cows and heifers, respectively (0.17 m^3 d^{-1} and 0.09 m^3 d^{-1}; assuming density = 1,000 kg m^{-3}) (ASABE 2005). The remaining manure in the watershed was assumed to be stored in the other ponds in the watershed. We estimated the total storage capacity of the digester ponds and storage ponds at the four additional CAFOs in the watershed using Google Earth imagery to measure area, and an assumed depth of 3.66 m (a standard depth for storage ponds). This led to an estimated watershed storage capacity of approximately 81,000 m^3 at the digester and 80,000 m^3 elsewhere in the watershed.

All ponds in the study were uncovered, and thus subject to precipitation inputs and evaporative losses. The annual rainfall (0.98 m) and snowfall (1.83 m converted to liquid water using a factor of 10) data for 2019 were obtained at the county level from NOAA (1.16 m total liquid). We did not include run off from the surrounding landscape, potentially causing an underestimate of the total liquid entering each pond. Evaporation from the ponds was calculated based on the area of the pond and an estimated rate of 1.9 mm d^{-1} (Ham, 2002). Because gas fluxes from the ponds are volume-dependent estimates, we scaled the rates based on an estimated volume of material in storage on a daily time step using calculated inputs of digestate, fresh manure, and precipitation, and outputs for field application and evaporation. Ponds are emptied to the greatest degree possible during the fall to prepare for the winter accumulation period when field application is prohibited. We thus started with the pond volume at 25% of total capacity on January 1, and then generated the volume of material in the pond on a daily time step over an annual cycle. We used empirical removal records provided by the farm manager to develop a removal scheme, and assumed that there was no spreading from December 1 through March 31, and also during the peak growing season from June 1 – August 15 as spreading on growing crops is not practiced. The daily removal on days for which withdrawal occurred was estimated at 1,500 m^3, which is the approximate capacity of a standard spreader and represents the maximum daily load. We mirrored this withdrawal paradigm for other manure slurry storage ponds in the watershed.

To evaluate the potential impact of increased anaerobic digestion of wasted food in New York State on N loss to the atmosphere and waterways, we also created a scenario for increased import of FW to the watershed. The estimated increase in the supply of FW for AD as a result of the prohibition of landfilling by large producers is roughly two-fold. This scenario is based on an estimate that current AD of FW in NYS is approximately 158,000 MT per year (Shahid and Hittinger, 2021) and that there is roughly 360,000 MT of wasted food generated by large producers subject to the policy threshold (>104 US tons per year) (EPA, 2021.). We thus assumed that to accommodate this increased diversion of FW from landfills, roughly a doubling of digester capacity is required. We therefore increased the current FW import two-fold, and assumed a similar co-digestion scenario. This scenario thus decreases the amount of fresh manure entering storage and increases the amount of co-digestate.

2.3 Storage emissions

GHG emissions from digestate storage ponds (P_N) were measured in Ponds 1 and 2 roughly monthly from May (Pond 1 only) through the end of October 2019. We deployed an inverted funnel (24 cm diameter) fitted with a 60-cc syringe to trap both diffusive and ebullitive gas flux (Appendix, Figure A.5). A styrofoam ring was fitted to the funnel outlet for floatation. Traps were deployed by inverting the funnel below the pond's surface, filling it with digestate, and righting the funnel. Three to four traps were spread around the pond to capture the spatial heterogeneity inherent in ebullition. Traps were attached to stakes at the edge of the pond and deployed for 24 hrs. The temperature was noted, and total gas volume accumulated measured. A subsample was stored in an evacuated serum vial and later analyzed for N_2O, CH_4 and CO_2 using a gas chromatograph (Shimadzu 2014 Greenhouse Gas Analyzer). N_2O release is typically dominated by diffusive fluxes (Baulch et al., 2011) across a free surface. Because CH_4 ebullitive fluxes were quite high, the free surface for diffusive flux varied significantly as CH_4 gas built up in the funnel over the 24-hr deployment. To accommodate for this, we assumed a constant rate of gas flux and scaled the free surface area over time to calculate the total N_2O flux. The difference between Pond 1 and Pond 2 was evaluated using a t-test and the effect of season was assessed using a one-way analysis of variance (ANOVA).

To broaden the range of emission factors for the model, we obtained values for N_2O, N_2 and NH_3 fluxes for both digestate and stored manure from the literature for uncovered ponds. The literature values were restricted to field measurements for dairy manure or food waste digestate. Laboratory based measurements were largely excluded because of the typical measurement of potential flux rates in this setting. Because we were not able to locate measurement of N_2 emission for FW-based digestate, the value for dairy manure was used for both scenarios. In cases where the literature values were presented as an areal flux, a volumetric flux was calculated using an assumed total pond depth of 3.66 m and that ponds are maintained at roughly 50% capacity (1.83 m) during the warmer months of the year when the greatest fluxes to the atmosphere occur.

Estimated emissions from the digester ponds and the watershed manure-only ponds were calculated on a daily time step using the estimated volume in the ponds on each day calculated as described above. The emissions were then calculated based on the emission rates in Table 2 and the volume of material in storage in the watershed on a daily time step for one year. Based on VanderZaag et al., (2010) and Park et al. (2006), we assumed that NH_3 and N_2O emissions decrease to zero when the air temperature is below 0^0C. Since the N_2 emissions are dependent on the same microorganisms, we assumed the same cut off for N_2. To estimate the potential variability in these flux rates, we conducted a Monte Carlo simulation (n=1,000 iterations) using the mean and standard deviation for each emissions factor.

2.4 Field application of organic and inorganic fertilizer

To evaluate the relative release of N to the atmosphere and waterways associated with FW digestate disposal, we developed a model to estimate loss at the field level for three baseline fertilization scenarios: inorganic fertilizer (IG), digestate (DI), and stored manure (SM) and an increased digestate scenarios: 50% increase (DI +50) in field application rate. These comparisons allow for evaluation of the relative loss of N to the environment for DI relative to traditional fertilization practices, and also the impact of increasing the rate of DI application per ha, a likely scenario if cropland is limiting disposal of DI. These first models assess the impact per unit area. Below we describe an additional set of scenarios where we assess the impact of replacing inorganic fertilizer with digestate at the watershed scale. In the study watershed, alfalfa and corn are the two dominant crops, occupying 46% and 41% of the agricultural area, respectively (Figure 9); as such, we focused on these two crops. The initial rate of field application for each crop and scenario was estimated based on crop recommendations provided by the farm manager (Table 1). For corn, inorganic fertilizer applications occurred prior to planting, at planting, and as a side dressing in mid-summer. We assumed that digestate and manure are typically supplemented with inorganic fertilizer for corn, but when manure or digestate slurry is applied to alfalfa that it replaces the inorganic fertilizer. In practice, the volume of digestate applied is greater than for manure to accommodate the lower nutrient content, as such we used an initial application rate of +25% for digestate relative to manure. For these rates, this translated to the baseline application of 30.3 and 37.9 m^3 ha^{-1} for manure and digestate, respectively (roughly 8,000 and 10,000 gal/ac). For corn, this was split equally into a spring (prior to planting) and fall (post-harvest) application. For alfalfa, liquid fertilizer was applied only in August prior to planting. For alfalfa, which is typically on the field for three to four years over which the recommended rate of N application varies, we estimated the N applied as either digestate, manure, or inorganic fertilizer separately for each of the three years of growth before the final harvest. These rates were then scaled by emissions factors to estimate total gas losses on an areal basis.

Table 1: Recommended fertilizer application for fields in the Pearl-Oatka watershed under three different fertilization scenarios: Digestate plus inorganic fertilizer (DI), stored manure plus inorganic fertilizer (FM) and inorganic fertilizer only (IG). Nitrogen values are the total annual mass applied (kg N ha^{-1} yr^{-1}) based on the measured concentration of N in each material. For corn, which is planted annually, each year is the same and, in all scenarios, the liquid digestate or manure is supplemented with inorganic fertilizer. Alfalfa typically remains on the field for three years, with higher applications of fertilizer in Yr 1 than in Yr 2 and 3. For the DI +50 scenario, both the spring and fall applications of DI were increased by 50%, respectively; inorganic fertilizer application was held constant.

Corn			Alfalfa		
				Year 1	Year 2 and 3
DI	Digestate (spring)	89	DI	177	89
	Inorganic (planting)	55			
	Inorganic (side-dressing)	75			
	Digestate (fall)	89			
FM	Manure (spring)	94	FM	187	94
	Inorganic (planting)	55			
	Inorganic (side-dressing)	75			
	Manure (fall)	94			
IG	Inorganic (planting)	133	IG	12	20
	Inorganic (side-dressing)	75			

2.5 Losses of N during field application

Gas flux measurements: Soil N_2O, CH_4, and CO_2 fluxes from agricultural fields were measured before and after (~2 hrs) digestate application roughly monthly from June to October 2019 using the static soil chamber method. The method of digestate application varied between top-spreading and injection. The chamber-based flux methodology is a common, inexpensive, sensitive, and unbiased technique adopted from Parkin and Venterea (2010) and has been used to evaluate GHG flux in various systems, including grasslands (Pezzolla et al., 2012). The chamber was constructed from a plastic bucket (headspace = 15 cm high x 24 cm diameter) fitted with a sampling septum and thermometer. A rubber-coated fabric skirt was glued to the chamber and splayed out around the chamber base after insertion into the soil; the skirt was held in place with a heavy gauge chain to prevent lateral gas exchange. The chamber was placed to adequately represent the field, covering both the row and inter-row area of the plot (Parkin and Venterea, 2010) (n=3 per time point). Samples were taken every 15 minutes for a 45 min period (4-time points including an initial point). Prior to the extraction of each sample, the headspace was mixed by filling and evacuating a 20 ml syringe three times to ensure a homogeneous sample. Gas samples were stored in evacuated vials until analysis using a gas chromatograph (Shimadzu 2014 Greenhouse Gas Analyzer). Emissions for the two primary digestate application seasons, early summer (prior to planting) and fall (post-harvest but prior to the over-winter crop planting) were pooled, and results were analyzed using a full-factorial three-way analysis of variance (ANOVA) with treatment (before and after application), method (top-spreading or injection), and season as fixed factors. Soil was collected before and after digestate application (n=3) during each measurement cycle. Organic matter content was assessed based on loss on combustion at 550^0C (Heiri et al. 2001).

In September 2019, fluxes were evaluated before and at 6, 21, 27, 168 and 336 hr after application to obtain the total emission of GHG following application. We calculated the cumulative gas emission over a 14-d period to estimate the total N_2O released prior to the return to baseline conditions. The flux over time was calculated by fitting a curve to the emission over time, segmenting the time series into two parts – 0-21 hr and 21-336 hr – and calculating the cumulative emission over 400 hr. The sampling duration was deemed sufficient based on prior work (Ellis et al. 1998). This value was used among other literature values to calculate the fraction of total applied N lost to the atmosphere as N_2O (emission factor) and parameterize the mass balance's A_N term (Figure 4 and Appendix Table A. 4).

Development of emissions factors and N loss following field application: Emission factors of three gases (N_2, N_2O, and NH_3) at the field level were assessed based on literature values for inorganic fertilizer, dairy manure and digestate application to fields. Nutrients like nitrate, ammonium, and phosphate from digestate applied to fields can interact with the soil through sorption or leach through the soil and enter groundwater or exit the field via stormwater runoff during heavy rainfall events. To represent this loss pathway, we used literature values for leaching of NO_3^-. Leaching of ammonium is typically very low and estimates of organic N leaching from application of organic fertilizers were scarce, so we did not include these in solute losses. We limited this review to the temperate zone and to field measurements. Emissions factors were calculated as a proportion of the total N applied. Details of literature review of fluxes are listed in Appendix Table A. 4, Table A. 5 and Table A. 6 respectively.

To estimate the loss of N during field application, each fertilizer application scenario was multiplied by the corresponding emission factor. For cases where multiple fertilizer applications occurred, the emissions associated with each application were summed to determine the total expected emission associated with each fertilization scenario and crop type. To estimate the potential variability in these flux rates, we conducted a Monte Carlo simulation (n=1,000 iterations) using the mean and standard deviation for each emissions factor.

Watershed level N losses following field application: To estimate the impact of food waste digestate on potential N losses to waterways and the atmosphere at the watershed scale (i.e., A_N and W_N in mass balance), we used an extended watershed as described in the geospatial modeling section below, this resulted in a watershed area of 16,378 ha (original Pearl-Oatka watershed is 14,700 ha). The relative percentages of each land use type remained the same. We simplified the watershed by assuming that all agricultural fields were planted with either corn or alfalfa (>85%). Given that these two crops occupy the majority of the agricultural area this is an adequate assumption. Agricultural fields were then split into three categories, depending on the type of nutrient management practice: FW Digestate (DI), Stored manure (SM), and Inorganic fertilizer (IG). For the baseline scenario, we used the total volume of digestate and manure determined previously as the amount of material available prior to using inorganic fertilizer. We used the subbasin approach described below to assign (1) the crop (and within subbasins designated for alfalfa, fields in year 1, year 2 and year 3 of planting were assigned equally), and (2) the fertilizer treatment. The first series of subbasins were designated for the DI fertilization scenario until all available material was consumed, followed by the SM scenario, and finally all remaining subbasins were designated for IG fertilizer.

For each fertilization scenario, losses were calculated per area of each crop using the emission factors in Appendix Table A. 4, Table A. 5 and Table A. 6 under the current availability of FW digestate and manure in the watershed. To assess the potential impact of increasing the digestate availability without increasing the areal spread rate on individual fields, we used the volume of material generated under the doubled import of FW scenario described above. Using a similar method of assigning subbasins for DI application until all available material is consumed, followed by manure and inorganic fertilizer, the impact of increased FW availability was assessed. This resulted in fewer subbasins treated with only inorganic fertilizer and provides an estimate of the total impact of increasing FW import to the watershed.

2.6 Geospatial modeling (Hydro-ecological model)

To validate the mass balance developed above, we used a geospatial modeling approach to estimate N losses from the watershed under the different fertilization scenarios described above. Environmental parameters like land use (residential/commercial, agricultural, forested, etc.), slope, elevation, climate, and topography determine the runoff pattern in a watershed. Since an empirical analysis of landscape-scale runoff and leaching is challenging at a watershed scale, many studies have previously used geospatial modeling to study the pollution load into streams for changing environmental conditions, including the GIS-NPS model of urban land-use change (Bhaduri et al., 2000), models of environmental impacts of agricultural chemicals (Verro et al., 2002), and the runoff depth change on the Kissimmee River basin (Melesse and Shih, 2003). Recent advances in geospatial technology and computing capacity have improved the ability to estimate nutrient release at the watershed level.

For our assessment of relative nutrient loss under the different fertilization scenarios described above, we used the Soil and Watershed Assessment Tool (SWAT), a commonly used tool in agricultural watershed modeling (Oeurng et al., 2011). SWAT, initially developed in 1998, provides a watershed modeling approach to study the water quality impacts of nutrient, pesticide, and sediment management practices (Manguerra, 1999), and water provisioning services (Karabulut et al., 2016), with greater accuracy than a traditional hydrological model, as SWAT considers factors like slope, weather (temp, relative humidity, sunlight, wind speed), and farming practices (application rate and fertilizer concentration) to parameterize the model. SWAT demonstrates how each sub-watershed (Hydrologic Response Unit (HRUs) reacts to a combination of land use, soil, and slope with a more detailed analysis than Long-term Hydrological Impact Assessment (LTHIA), although both based on curve number approach. Once the SWAT model is set up, it is easier to calibrate parameters like fertilizer application (type and quantity), and to calibrate based on river baseflow, etc.

ArcSWAT version 2012.10.21, a freely available ArcGIS-ArcView extension for SWAT developed by Texas A&M University, was used in this work. GIS data (land use, soil categories, and slope) shown in Figure 5, non-GIS weather data (precipitation, temperature, wind speed, solar radiation), management data (use of fertilizer and application rate), N content of digestate and manure, and fertilizers application rate were integrated into the analysis (see Appendix, Figure A. 9 for model flow).

Model setup: Pearl Oatka (HUC12) watershed, 132.2 km^2 in the area surrounding the facility was selected for assessing water quality impact. However, the SWAT was set up in a larger basin (163 km^2), surrounding the Pearl Oatka to avoid the isolation of some connected streams (Figure 5A). The watershed delineation was automated based on the projected National Elevation Dataset-Digital Elevation Model (30m NED DEM) dataset, with a filled sink. A minimum area of 45 ha i.e. (500 pixels, each 30 x 30m raster), a reasonable boundary for a small watershed, was set as the sub-watershed threshold. We performed an automatic watershed delineation to categorize stream segments (reaches or sub-basins) based on the similarity of discharge, depth, area, and slope, which generated monitoring/outlet points in the watershed, as shown in Figure 6. The land-use categories were input as described previously for the watershed. The (Soil Survey Geographic Database (SSURGO), a detailed map of soil data (scale 1:12,000 to 1:63,360), was chosen for this analysis, used widely by county, township, and local users due to fine resolution. A map unit key (MUKEY) of the SSURGO identifies soil in the area for soil classification, a unique soil identifier for a hydrologic group (HYDGRP) of the soils (A, B, C, D) in curve number calculations. Slope data were derived from the DEM, classified into five ideal classes for the agricultural watershed as per the SSURGO soil survey.

The primary model inputs: Land use (NLCD2016), Soil Data (SSURGO), and Slope (based on DEM), as shown in Figure 5, were fitted for HRU. An ideal threshold percentage for HRU delineation (20% for land cover, 20% for slope, and 10% for soil) for each parameter was defined. Based on the above preliminary watershed and HRU delineation, the SWAT created 209 sub-basins and 1941 HRUs with the homogenous land use, soil types and slope. The weather data, precipitation (in mm), temperature (in ^{0}C), relative humidity (in fraction), solar radiation (MJ m^{-2}), and wind speed (m s^{-1}), were simulated from two nearby weather stations.

Model Parameterization (Management Operations): We executed SWAT simulations for 2018 to 2020 in the extended Pearl Oatka watershed using a series of management inputs parameterized to align with the fertilization scenarios described above for development of the field-based mass balance model. The 209 sub-basins of the Pearl Oatka were divided randomly into equal numbers of corn and alfalfa crop types. The sub-basins designated for alfalfa were divided equally between the three crop growth years, as the amount of fertilizer applied varied based on the year of planting (see Table 3). The model was then parameterized for several different fertilization scenarios. In the first set, all subbasins received the same treatment (DI, FM, IG, or DI +50). These scenarios allow for a comparison of the impact of digestate application relative to more traditional fertilizers (SM or IG), and also with the impact of using a higher rate of digestate application on the individual fields (DI +50). The second set of models followed an approach similar to that described above for the field mass balance model: We randomly selected sub-basins to receive first digestate, alternating between corn and alfalfa, until the available supply was depleted. Then manure was applied to subsequent subbasins until depleted, and remaining subbasins were fertilized with inorganic fertilizer. This approach was then followed for the increased volume of digestate produced in the FW x2 scenario developed above to assess the impact of increasing the total area receiving digestate.

The model validation was confirmed using an ArcSWAT extension called SWATCheck to identify potential model errors (“SWAT Soil & Water Assessment Tool,” 2016), that provides users with a budget summary (of hydrology, sediment, N cycle, P cycle, plant growth, landscape nutrient losses, land use summary, instream processes, point sources, and reservoirs) and timely warning through graphics. Watershed level annual N inputs were analyzed using SWAT Output Viewer, a tool to quickly assess the SWAT simulations output.

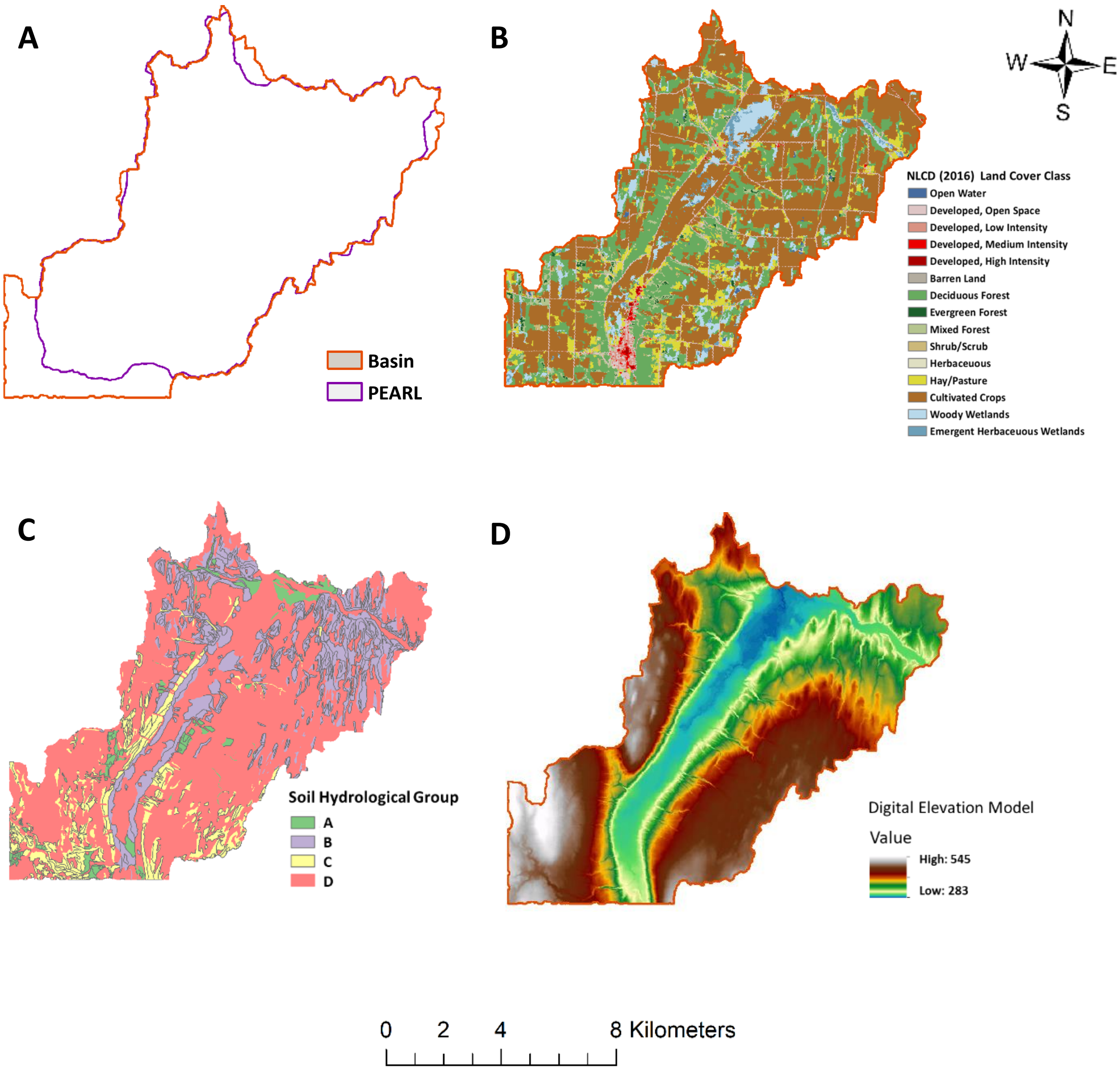

Figure 5: Input parameters for the SWAT model include (A) Boundary of Pearl Oatka watershed and SWAT basin, (B) land use land cover, (C) soils groups, and (D) the slope percent.

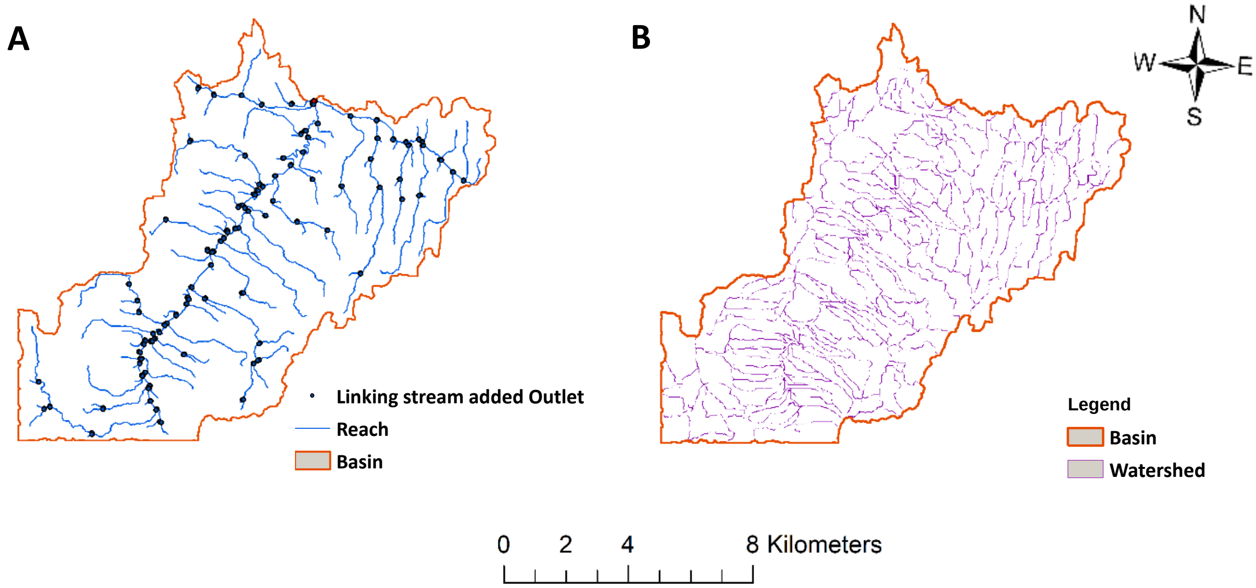

Figure 6: Watershed Delineation in SWAT model (A) Stream networks and monitoring points, (B) Sub-basins definition of SWAT.

3. Results

3.1 Properties of digestate

The total N, K, and P in fresh digestate were higher than stored digestate, reflecting losses during the storage phase (Table 1). The nutrient value in fresh manure was higher than for either digestate or stored manure. Additional characteristics of digestate are in the Appendix (Appendix Table B. 1).

Table 2: Characteristics (mean % +/- SE) of Liquid Digestate collected at the exit of the digester (n = 10), in the storage Lagoon (n = 10), Fresh Manure (n = 3) sampled in 2019 and 2020.

	Fresh Digestate			Stored Digestate			Fresh Manure*			Stored Manure		
Moisture	97.89	±	1.02	96.69	±	1.62		--		87.10	±	9.33
Dry Matter	2.11	±	1.02	3.31	±	1.62		--		12.90	±	9.33
Phosphorus	0.04	±	0.03	0.03	±	0.0	0.04	±	0.02	0.09	±	0.05
Potassium	0.09	±	0.02	0.09	±	0.05	0.19	±	0.08	0.27	±	0.16
Total N	0.22	±	0.02	0.19	±	0.05	0.27	±	0.11	0.23	±	0.12
Ammonia	0.12	±	0.02	0.09	±	0.06	0.13	±	0.05	0.08	±	0.04
Organic N	0.10	±	0.04	0.09	±	0.04	0.14	±	0.06	0.15	±	0.09

*from: Jokela et al., 2010

3.2 Storage emissions

Nitrous oxide emission: Gas emitted from digestate storage lagoons contained <1% nitrous oxide (Details of CH_4 and CO_2 can be found in Appendix – C.1 Carbon emissions). The ebullition (gas bubbling) was visibly heterogeneous in summer across the lagoon resulting in more than two-fold variability in flux rates across chambers (as shown in Appendix, Figure A.6). Daily areal release rates of nitrous oxide were typically very low and ranged from 0.07 ± 0.01 mg N_2O m^{-2} d^{-1} in July to 0.26 ± 0.10 mg N_2O m^{-2} d^{-1} in October (Figure 7), and were very low compared to literature values (Appendix Table A. 2). Values in October were highly variable. Our one-way ANOVA with month as fixed factor showed no significant differences among the dates ($F=1.8$, $p=0.16$). Thus, we used the global mean and standard deviation (0.15 ± 0.13 mg N_2O m^{-2} d^{-1}) and converted to a volumetric basis using a mean pond depth of 1.83 m.

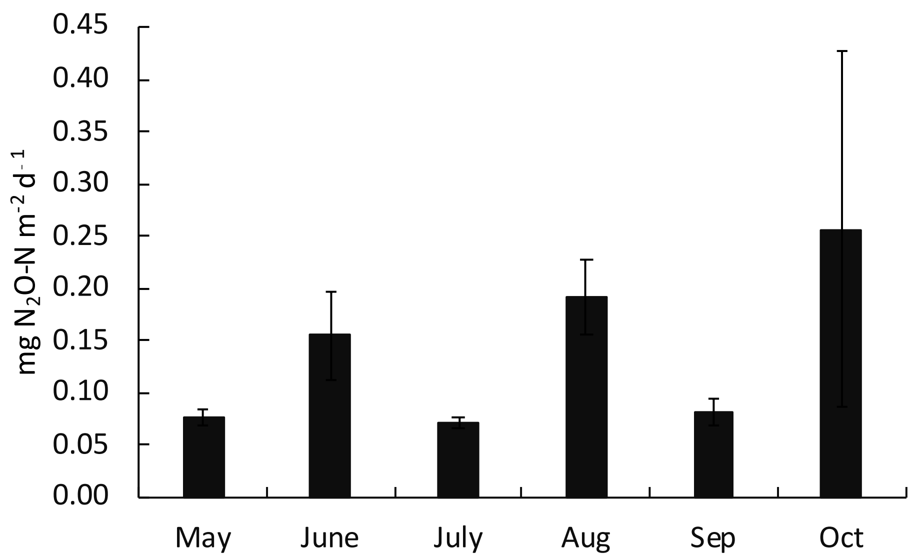

Figure 7: Daily release of nitrous oxide (mean +/- SE) from the two primary storage ponds at the digester facility measured using gas traps in 2019 (n=4 for each pond).

Table 3: Storage flux parameters (mg N m^{-3} d^{-1}) derived from literature values and this study (for digestate N_2O flux). Details of the literature used can be found in Appendix A.

		mg N m^{-3} d^{-1}			Range		
NH_3	**Manure**	471	±	116	338	-	624
	Digestate	1004	±	1336	82	-	2924
N_2O	**Manure**	171	±	124	42	-	375
	Digestate	144	±	129	0	-	292
N_2	**Manure**	1048	±	1261	156	-	1939

Storage emission factors: The final model input value was the mean +/- standard deviation for all literature and empirical (this study) measurements for each gas (Table 4). All values were highly variable across literature sources, but the highest variability was for digestate, reflecting the smaller number of studies and the very high variability in both methods of data collection and digestate composition and storage. Mean release of NH_3 was greater for digestate, but with very high variation, whereas the release of N_2O was more similar for the two materials.

3.3 Storage level N losses:

We estimated that the annual volume of material entering storage at the digester facility was 105,000 m^3, and was comprised of 62,000 m^3 digestate, 32,000 m^3 manure, and 11,000 m^3 of net precipitation (precipitation – evaporation). Outside of the digester, an additional 86,000 m^3 of manure was generated with an additional net influx of 7,000 m^3 of precipitation (93,000 m^3 total). Doubling the total volume of FW entering the watershed resulted in a 45% increase in the total volume of digestate (plus manure; 154,000 m^3 total). This production results in an estimated N availability of 228 MT, 223 MT, and 323 MT for digestate, manure and digestate 2x FW (Table 4).

Applying the emission factors to these volumes results in losses that scale appropriately with the volume of FW and the estimated EF for each material. At the watershed level, the storage losses of NH_3 are substantially higher for digestate, reflecting the greater EF. When double the volume of the FW enters the watershed, all emissions are increased by approximately 45%.

Table 4: Estimate of total watershed N generation and loss of FW digestate and manure during storage in gaseous form under the current fertilization scenario, and under projected increases in 2x FW digestion. All values are in MT of N per year and the error estimate is the standard deviation based on Monte Carlo simulation.

	Baseline		2x FW
	Manure	**Digestate + Manure**	**Digestate + Manure**
N Produced	228.1	222.6	323.2
Losses			
NH_3	4.2 ± 1.1	8.9 ± 12.7	13.0 ± 18.6
N_2O	1.1 ± 0.8	0.9 ± 0.9	1.3 ± 1.3
N_2	6.3 ± 8.1	6.5 ± 8.4	9.5 ± 12.3
Total	11.5 ± 9.9	16.3 ± 22.0	23.8 ± 32.1
N at Spreading	214.7	197.3	288.0

3.4 Field application of organic and inorganic nutrients and loss pathways

Nitrous Oxide: Our nitrous oxide emission average in Spring for injection and field spreading were 9.7×10^{-6} to 8×10^{-5} g N_2O-N m^{-2} hr^{-1} before and after digestate application, respectively in 2019. Similarly, the Fall emissions were 7.3×10^{-5} and 9.7×10^{-4} g N_2O-N $m^{-2} hr^{-1}$ for the pre- and post-digestate application, respectively. The soil flux results show that digestate application significantly increases the GHG emissions compared to the pre-application conditions (Figure 8), with significant differences in the rate measured in early summer relative to the fall. There was a significant interaction among all three fixed factors in the ANOVA, suggesting that the impact of digestate application is dependent on both season and on the mode of application (Table 5). The greatest increase in N_2O release over baseline was measured in the fall for the injection method (Figure 8). The N_2O emissions peaked within 48 hr of application and returned to baseline within two weeks following digestate application in September (Figure 9). The cumulative emission of N_2O-N was 0.4 g N m^{-2}, which was approximately 2.1% of the total N applied as digestate (approximately 177 kg N ha^{-1}). This was somewhat higher than other reported measurements (Table 6). Soil organic matter increased significantly post-digestate application, shown in the Appendix B. 4.

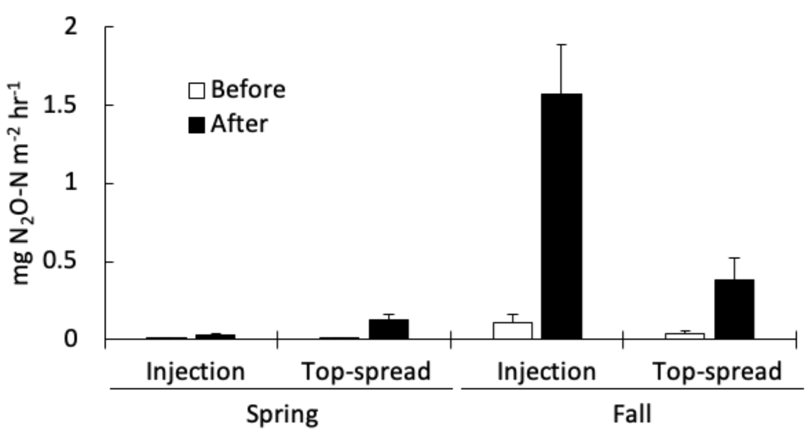

Figure 8: Field N_2O-N fluxes (mean ± SE) before and 30-120 min after application of co-digestate at the beginning of the growing season (Spring) and prior to planting fall crops (Fall) using two methods of application (Injection and Top-spreading) (n=3 for all except for Spring top-spreading, where n=9.

Table 5: Results of three-way ANOVA evaluating the impact of Treatment (before and after digestate application), Method (injection or top-spread), and Season (summer and fall) on nitrous oxide emission from soil.

Treatment	$F_{1,43} = 58.64$	$p < 0.0001$
Method	$F_{1,43} = 28.79$	$p = 0.0099$
Season	$F_{1,43} = 80.13$	$p = 0.0020$
Season*Method	$F_{1,43} = 39.15$	$p = 0.0062$
Season*Treatment	$F_{1,43} = 42.93$	$p < 0.0001$
Method*Treatment	$F_{1,43} = 16.17$	$p = 0.0004$
Season*Method*Treatment	$F_{1,43} = 22.76$	$p < 0.0001$

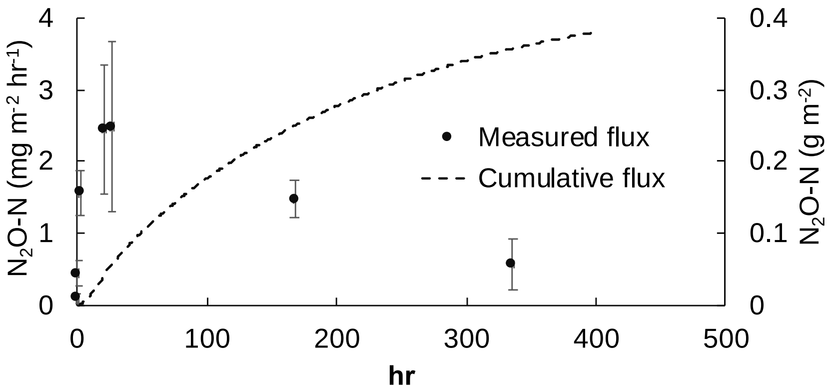

Figure 9: N_2O-N emitted in agricultural fields (mean +/- SE) from digestate application (injection) over two weeks.

A summary of the literature values and empirical measurements for gaseous N and NO_3^- release following field application is found in Table 6. Based on the literature review, for manure, loss of N by ammonia volatilization is a substantially greater loss pathway than through either denitrification or N_2O loss. Digestate NH_3 volatilization ranged from 7.3 – 40 % while that of manure ranged from 4.7 – 27.5 % and inorganic ranged from 0.3 – 21.9 % (Details in Literature Review, Table A. 4, Table A. 5 and Table A. 6). As with the storage emission parameters, there is substantial variability in the reported values, especially for digestate. NO_3^- release following inorganic fertilizer application is substantially higher than for digestate or manure, indicating potential benefits of digestate application.

Table 6: Emission factors (EF, as a percent of total N applied) for gaseous emissions (NH_3, N_2O, and N_2) and dissolved NO_3^-. Values are the mean of literature values. Details on the studies used can be found in Appendix C. N_2O release obtained in this study is included in the mean for this factor.

		EF	Range
NH_3	Manure	12.5 ± 9.0	4.7 - 27.5
	Digestate	18.2 ± 14.8	7.3 - 40.0
	Inorganic	11.2 ± 10.2	0.3 - 21.9
N_2O	Manure	0.8 ± 0.6	0.1 - 1.6
	Digestate	0.6 ± 0.7	0.1 - 2.1
	Inorganic	1.1 ± 0.6	0.1 - 1.9
N_2	Manure	2.6 ± 1.3	1.1 - 3.6
	Inorganic	3.8 ± 2.4	2.1 - 5.5
NO_3^-	Manure	9.7 ± 10.7	1.3 - 29.3
	Digestate	16.6 ± 15.0	2.8 - 32.5
	Inorganic	27.4 ± 16.3	9.7 - 41.7

**no values for digestate for N_2; substituted with manure values

3.5 Field and watershed level N losses following application:

The three fertilization scenarios lead to substantially different N losses, with the lowest losses for the inorganic fertilizer (Table 7). In all cases, the greatest gaseous loss of N is for NH_3, and the lowest for N_2O. Loss of N_2 is 2-3 fold higher than N_2O. The higher loss rate for DI reflects both the higher EF and the greater amount of DI applied to the fields. Alfalfa has lower losses overall than corn, reflecting the lower fertilizer demand. It suggests, though, that this is a poor disposal pathway because less material is consumed. With 50% more digestate applied per ha, the overall N loss impact increases about 30%, reflecting the increased digestate but constant inorganic fertilizer also applied. We note that the losses calculated here per ha are for the cropland only.

Under the current availability of FW, there was sufficient FW to treat approximately 1,250 ha of cropland (mix of alfalfa and corn) in the modeled watershed. At this level, the overall N losses reflect the heterogeneous pattern of fertilization, and are roughly 10% greater than the IG alone scenario in per ha N loss. The losses of N_2O, NH_3 and NO_3^- attributed to the fields receiving DI under this scenario are 1.7, 39.8, and 46.8 MT yr^{-1}, respectively. Similarly, when scaled to the watershed, doubling of FW digestion results in roughly 20% greater loss overall. These calculations ignore potential run off or emissions from non-cropland, which may contribute to the overall N released to the atmosphere and waterways at the watershed level.

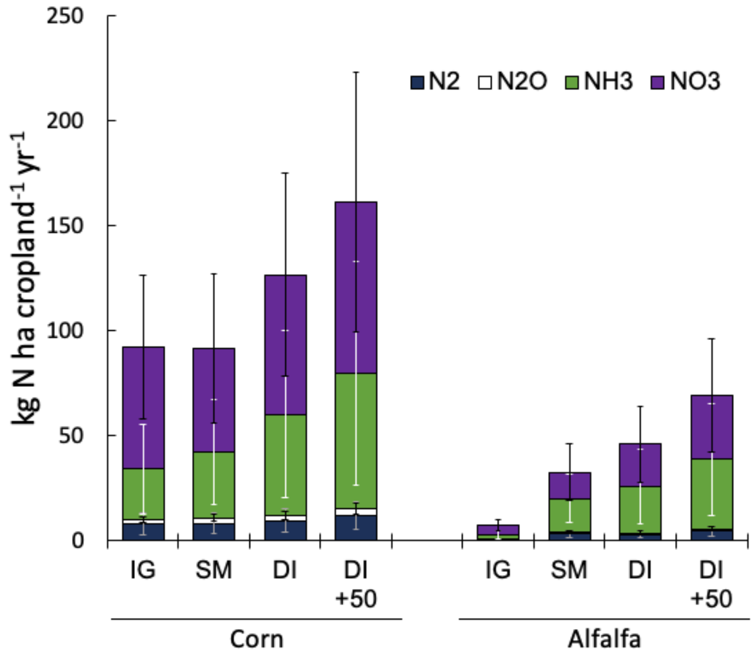

Figure 10: Potential gas emissions and leaching after field application of inorganic fertilizer (IG), stored manure (SM), digestate (DI) and a 50% increase in the rate of digestate application (DI +50).

Table 7: Estimate of total watershed N loss from cropland in the Pearl-Oatka watershed in gaseous form or through leaching under three current fertilization scenarios (inorganic only [IG], stored manure + inorganic [SM], digestate + inorganic [DI], and under projected increases in digestate application of 50% to individual fields (DI +50%). Application of all digestate and manure produced in the watershed along with inorganic fertilizer application to remaining fields under the current rate of production (MULT) and a doubling of FW import to local digesters (MULT 2X FW) was assessed at the watershed scale.

			IG	SM	DI	DI +50%	MULT	MULT 2X FW
Mass Balance Approach	**Areal Loss Rate (kg N ha^{-1} yr^{-1})**	N_2-N	2.0 ± 1.2	2.8 ± 1.5	3.0 ± 1.6	4.0 ± 2.1	2.2 ± 1.3	2.3 ± 1.4
		N_2O-N	0.5 ± 0.3	0.8 ± 0.5	0.8 ± 0.6	1.0 ± 0.9	0.6 ± 0.4	0.6 ± 0.4
		NH_3-N	5.8 ± 5.0	11.5 ± 8.5	17.0 ± 13.4	23.8 ± 18.7	7.9 ± 6.8	9.3 ± 7.7
		NO_3^--N Leached	14.0 ± 8.0	14.5 ± 11.3	20.5 ± 15.2	26.7 ± 20.6	14.3 ± 9.5	15.5 ± 10.4
	Watershed Loss Rate (MT N yr^{-1})	N_2-N	32 ± 20	46 ± 25	50 ± 27	65 ± 35	35 ± 21	38 ± 23
		N_2O-N	9 ± 5	14 ± 9	12 ± 11	16 ± 14	10 ± 6	10 ± 7
		NH_3-N	96 ± 82	189 ± 139	278 ± 219	390 ± 306	129 ± 111	153 ± 126
		NO_3^--N Leached	230 ± 132	237 ± 185	335 ± 249	438 ± 337	235 ± 156	254 ± 171
Geospatial Approach	**Areal Loss Rate (kg N ha^{-1} yr^{-1})**	N_2-N	1.0	1.5	1.6	2.5	1.0	1.0
		NH_3-N	2.9	3.7	4.3	5.6	3.3	3.3
		NO_3^--N Overland	1.0	1.4	1.4	1.8	1.1	1.1
		NO_3^--N Leached	7.1	14.1	15.0	24.9	9.3	9.3
	Watershed Loss Rate (MT N yr^{-1})	N_2-N	16	25	27	40	16	16
		NH_3-N	47	60	70	92	53	54
		NO_3^--N Overland	16	23	23	30	18	18
		NO_3^--N Leached	116	231	245	408	153	153

3.6 Geospatial Modeling

The output from the geospatial model for all N loss pathways falls within the range of estimates for the mass balance model at the watershed level (Table 7). At the field level (per ha), the values are also within the same range, but we note that the mass balance approach presents only the losses for cropland, whereas the SWAT output includes other land use types and thus we would anticipate lower average areal release from the SWAT model. The NO_3 leached, i.e., percolation past the bottom of soil profile in the watershed, incorporates lateral and groundwater yield and is comparable to the leached value for the mass balance approach. The values for inorganic fertilizer are within the range of the mass balance estimate, but somewhat less, possibly reflecting the use of only ammoniacal N in the SWAT input. However, the output for both DI and SM are very similar to one another, with slightly greater loss for DI. SWAT allows for an estimate of NO_3^- run off overland as well, which was small (<15%) of the NO_3^- lost through leaching. NO_3^- represents the largest loss pathway for all scenarios and is greater than all gaseous emissions combined. Again, NH_3 represents the largest gaseous N loss pathway, followed by N_2. Applying 50% more digestate to individual fields results in substantially higher release of all N compounds. However, increasing the number of fields to which digestate is applied results in very little detectable change overall in the gaseous emissions or NO_3^- release at the field or watershed scale.

3.6 Greenhouse gas emissions

We calculated the potential Global Warming Potential (GWP) for four scenarios, as shown in **Error! Reference source not found.**. In all cases, the majority of the GWP is generated at the field phase, with the storage phase contributing less than 25% of the total. The lowest GWP is associated with application of only inorganic fertilizer in the watershed. We added a scenario where all manure generated in the watershed is simply stored and spread on fields, with no AD (including the manure generated by the digester CAFO), which had roughly 30% greater GWP than inorganic fertilizer alone. The addition of the digester facility to the watershed at current and 2-fold FW processing were >1,000 MT CO2 e greater than the inorganic fertilizer alone, but only 6% and 10% more GWP than undigested manure plus fertilizer.

Table 8: Global warming potential associated with storage and field application for four scenarios in the Pearl-Oatka watershed: only inorganic fertilizer is used on all crops (Inorganic); all manure generated in the watershed is stored in ponds and spread on fields with no AD (Manure no AD; includes manure currently generated at digester facility); the current scenario where digestate and manure are applied to

fields, along with inorganic fertilizer (MULT); and a similar scenario but with two-fold import of FW to the watershed (MULT 2X FW). All units in MT CO_2 e.

		Contribution From			
	Source	Inorganic	Manure	Digestate + Manure	Total
Inorganic	Field	2540	--	--	2540
Manure no AD	Storage		453	--	
	Field	1967	928	--	3348
MULT	Storage	--	324	288	
	Field	1765	672	509	3557
MULT 2X FW	Storage	--	324	418	
	Field	1459	665	804	3670

4. Discussion

The approach used in this study, combining empirical measurements with literature values, along with a geospatial modeling approach, generated a complete picture of the potential risks of increased anaerobic digestion of FW in an agricultural watershed. Evaluation of results suggests clear management opportunities to maximize the value of AD and minimize risk across the disposal life cycle of digestate. Significant environmental impacts may be minimized by increasing storage availability and using management techniques to reduce GHG (ie, covered ponds). Likewise, by ensuring that adequate crop land is available for field application, over-application on a smaller area of crops, which results in significant increases in run off and GHG, can be minimized.

4.1 Properties of digestate:

Digestate typically has higher ammonium and total nitrogen to carbon ratio than original feedstock (Tampio et al., 2016) and chemically represents the composition and ratio of the feedstock. For instance, feedstock with low N value (e.g., silage) will generate digestate of lower N quality than feedstock with high N value (cereals, pig slurries, and poultry) (Möller and Müller, 2012). The composition of manure varies over the course of the year, but is relatively consistent (Rico et al., 2011). However, the composition of the FW-based digestate depends upon the food waste input into the digester, which varies over the course of the year, lending heterogeneity to the nutrient value of digestate (Appendix, Table B. 2) for the digestate characterization data). N transformation in the storage pond depends upon factors like manure (quantity and quality of animal diet) and storage conditions (temperature, aeration, and compaction), which may be widely variable (Dämmgen and Hutchings, 2008)(See summary in Appendix, Table B. 3). The somewhat lower nutrient content of AD relative to stored manure (Table 2), suggests that practices that increase the rate of AD application to fields may be warranted to ensure an equivalent N delivery to fields. While some loss may occur through particulate settling or leaching from the pond, this is typically <4% of total (Petersen et al., 1998) and gaseous emissions are the primary loss. It appears that the majority of the loss over the storage period is in the inorganic component (ammonia), suggesting that practices to minimize this loss will preserve the nutrient value.

4.2 Storage emissions

Storage emissions are higher when the pond operates at full capacity than partially filled due to increased volume. Thus, the total volume of material entering storage ponds is a critical parameter, and also a potential leverage point. Depending on management practices, additional liquid may enter storage ponds in the form of run off from the pond watershed, or from wash water used in milking houses and cow barns. Milk house wash water (0.029 m^3 cow-1 d; Krauß et al., 2016) can be a significant year-round liquid source enhancing the total volume of liquid that must be spread. In addition, the removal rate depends upon field management like the timing of crops, precipitation, spreader/truck limit, irrigation, and policy limits on timing of application. The temporal dynamics (especially storage temperature rather than air temperature) have a significant role in the AD storage phase that may alter the emissions (Tel-Tek, 2013). Because storage N fluxes are significantly lower below freezing (VanderZaag et al., 2010), accumulation of material in ponds is less critical during winter from the perspective of gas emissions (Srinivasan et al., 2006). Thus, manure handling determines N transformation (Petersen, 2018) and as previously stressed by Baral et al., (2017) and Menardo et al. (2011), storage emissions mitigation is an important component of manure management chains to minimize environmental impacts and preserve nutrients for field applications. It is clear that storage-phase digestate management is an opportunity for a digester facility to preserve fertilizer value and displace inorganic fertilizer (Ebner et al., 2015).

Nitrous oxide emissions: The overall GHG emissions at a storage pond are variable and depend upon environmental parameters (lagoon temperature, wind) and storage conditions (pond lining, cover, duration the digestate at storage, agitation, and emptying the lagoon). In unvegetated digestate storage ponds, ebullition and diffusion are the two dominant pathways of gas release to the atmosphere (Bastviken et al., 2004). Ebullition is an important pathway for methane transport while it is negligible for nitrous oxide transport into the air (Baulch et al., 2011) and agitation of the pond does not affect N_2O and NH_3 fluxes but increases CH_4 and CO_2 fluxes due to formation of bubbles and dissolved gases (VanderZaag et al., 2010). Previous studies with manure (fresh or composted beef manure or poultry manure) suggest that N_2O emissions are typically low, as pond conditions tend to have high ammonium that favors nitrification and denitrification, leading to proportionally higher N_2 emission relative to N_2O (Amon et al., 2005), as reflected in the overall storage parameters generated in this literature review (Table 3).

The N_2O emissions observed from gas traps in this study were at the low end of the range previously measured (see Appendix). This may be due to the lower N content of liquid digestate, the very high heterogeneity of the pond, and potentially to the formation of crusts. , Usually, crusts aren't formed in the digestate storage pond because of the high proportion of liquid to solids (Tel-Tek, 2013), but other factors may influence crust formation like digestate agitation, manure percentage, etc. Crust formation was periodically observed on the study ponds, but was removed from the gas traps, which may have led to lower N_2O emissions as crust may provide aerobic microsites where denitrification is incomplete (Aguerre et al., 2012).

Amon et al., (2006) highlighted the temporal and spatial variability of N_2O release during storage, which we also observed (Figure 7). GHG emissions are generally higher during spring turnover, which occurs when the pond temperatures exceed 39.5°F and the densest material comes to the surface (Nicolai et al., 2004). GHG emissions may be high at early stages of storage in lagoons but decrease upon aging of digestate as labile material and nitrate are exhausted. In an empirical comparison of untreated manure to digested manure. Amon et al., 2005) found that for manure, 84.2% of N_2O loss takes place during storage, with 15.8% during field application; this fraction increases to 91.3% for digestate at storage and 8.7% during field application. Our model results suggest that a higher proportion of N_2O is lost during storage, with roughly 30% of the loss occurring at the storage phase (based on the emissions associated with DI fields only in the MULT scenario). Thus, management aimed to reduce these emissions will be a key towards sustainable increase in AD.

Ammonia volatilization and release: In general, the AD process doesn't influence NH_3 (Amon et al., 2006), leading to high NH_3 concentrations in storage lagoons. While some studies have suggested that the rate of NH_3 emission is higher for fresh manure relative to digestate because of the lower pH of FW digestate induced by the variety of feedstock. However, our literature review indicates that the range of values for NH_3 emission is quite high (Table 3), with an average value for digestate that is two-fold greater than manure-based systems. The range of values is also extremely high, representing the wide variety of environmental conditions and measurement techniques, and also the need for better constrained estimates of gas fluxes from storage ponds. During the storage phase, the ideal condition for NH_3 volatilization is dry, warm, sunny, and windy conditions that enhance the volatilization for an open pond. Similarly, NH_3 emissions may be increased by 77% when aerated (Amon et al., 2006). However, the NH_3 losses may be much greater in uncovered ponds (Clemens et al., 2006 and Tel-Tek, 2013), as the cumulative NH_3 emissions are reduced in the covered ponds (Chadwick, 2005), suggesting an opportunity to reduce N loss through this pathway. Roughly 17% of the total emissions associated with digestate disposal occurred at the storage phase, suggesting that management at the storage phase is critical to prevent decline in nutrient value and fugitive emission of volatile N that may impact adjacent ecosystems.

Overall, we were able to account for the majority of the estimated N loss during storage for both digestate and manure, when comparing the initial N produced to the available N after prolonger storage. The small amount of "missing" N (5-10% of the original N content), may be attributed to sedimentation in the pond, which can account for a small amount of the total loss (Petersen et al., 1998).

4.3 Field application of organic and inorganic nutrients and loss pathways

Nitrous Oxide and Dinitrogen Emissions: N_2O emission following digestate application is higher than pre-treatment values as a function of both the digestate application and tillage (Mutegi et al., 2010), where the digestate disturbs the soil surface and may create water-filled pore spaces that favor incomplete denitrification and release of N_2O. We observed significant temporal and management-practice variability in measured N_2O release following field spreading. Previous studies suggest a 2 to 20-fold higher N_2O emission in top-spreading than injection (Adair et al., 2019), but our data are less conclusive, likely because of the lack of contemporaneous measurements. The field level N_2O emissions were spatially variable, likely due to uneven digestate application by spreader trucks, with repeat application on some rows, or dumping of remaining digestate prior to refilling that create hotspots (Peterson, 2018). Additional factors including ambient temperature, soil type, waterlogged conditions, and tillage contribute to heterogeneity. One study found that only 3.2% of the farm areas had contributed to 9.4% of the total farm N_2O emissions because of emission hotspots in the field (Luo et al., 2017) and thus the uniform application of digestate on the field presents an opportunity to reduce GHG emissions. Our result of the N_2O-N emission over the time of two weeks suggests relatively rapid response of the microbial community, but that the available N is rapidly stabilized or lost from the system. The combined N loss was approximately 3% of the total N applied to the field during this period, which is somewhat higher than some literature values for digestate (e.g., Nicholson et al., 2017) or manure (Ellis et al., 1998; Nicholson et al., 2017; Van Groenigen et al., 2004), but higher than others (Tiwary et al., 2015), ranging 4-10% of N applied. Values were significantly greater than the IPCC Tier 1 value (<1% of N applied).

The N_2O release per unit N applied to fields for manure and digestate were similar, and both lower than for inorganic fertilizer (Table 6), suggesting an opportunity for a decrease in GHG. However, when scaled to the total N typically applied for each type of fertilizer, the release of N_2O is about 60% higher per unit area because of compensation for potentially lower nutrient value in organic fertilizers (Table 7). In contrast, the release of N_2 increases only about 30%, suggesting that the N loss pathway shifts towards N_2O with organic fertilizers, creating an associated risk for enhanced GHG production. A 50% increase in digestate application rate caused a similar increase in N_2O release, but these estimates don't take into account the increased waterlogging and changes in water-filled pore spaces associated with liquid fertilizer application, and as such may underestimate the actual increase in N_2O relative to N_2 production when more digestate is applied to an individual field.

Ammonia Volatilization: NH_3 emissions from fields typically peak in the first week of field application, and gradually stabilizes in the following weeks (Bustamante et al., 2012). From our literature study, the EF for digestate was higher than for manure and inorganic fertilizer, but was also highly variable (Table 4). This higher mean value led to greater loss estimates per area for DI, but with substantial variability. This is counter to studies suggesting that the higher pH of manure favors NH_3 loss at the field stage (the NH_3 is optimal at pH > 8) (Nicholson, 2017), but may represent the high variability of soil conditions under which the various literature studies were conducted. The injection technique of application may be helpful in reducing the NH_3 volatilization relative to top spreading, as the fertilizer exposure to atmosphere is minimal (Tiwary et al., 2015). However, this technique may have no effect on the FW-based digestate, likely because of the properties of digestate (Nicholson, 2017) and may act to exacerbate N_2O release due to soil disturbance and the supply of N and C below the soil surface (injection blades deposit material ~6 to 8 inches below the surface). However, there is a clear need for more estimates of gas production following application of FW digestate to assess the role of digestate composition and how potential increased application rate (per ha) will impact total emission.

NO_3^- leaching: NO_3^- leaching from agricultural fields depends upon the crop type due to difference in root depth, the soil type, the depth of the soil profile, and availability of NO_3^-. NO_3^- is usually stored in the soil until rainfall events trigger displacement below the root zone (Rimski-Korsakov et al., 2004), but once transported below 150 cm becomes less available to plants and more likely to leach to aquifers (Rimski-Korsakov et al., 2004). While plants prefer NO_3^-, it is also the most soluble and mobile form of N that is easily leached, so, leading to aquifer contamination and surface runoff. Thus, mineral fertilizers with a high proportion of nitrate may lead to greater risk of contamination of waterways relative to organic forms that contains more reduced species of N that are more readily adsorbed to soil particles (Nkoa, 2014). Thus organic fertilizers with very low NO_3^-:NH^+ present an opportunity to minimize loss of inorganic N to waterways (Sogn et al., 2018) and enhance soil fertility by promoting immobilization of N in the soil (Gutser et al., 2005).

Indeed, the proportion of N lost as NO_3^- through leaching for digestate was about 60% of the loss for inorganic fertilizer, and even lower for manure (Table 6). This translated to similar rates of N loss for IG and SM, in spite of the much higher application of N to fields for the SM scenario. The release for DI was somewhat higher, but less in proportion to the increase for a comparable N application rate of inorganic fertilizer (Table 7). The geospatial model predicted a relatively higher NO_3^- loss at the field level for both SM and DI relative to relative to IG. The similarity in run off between SM and DI in the SWAT model is likely due to the lower N content of DI, in spite of the application of 25% greater volume (Table 1 and Table 2). Thus, the FW import to the watershed replaces an inorganic fertilizer and its potential nitrate leaching risks. However, with a 50% increase in digestate application per area, there is a substantial increase in NO_3^- loss to the watershed of 30% (mass balance prediction) to 65% (SWAT prediction). This may be an underestimate of the total loss, as leaching (and overland runoff) losses are likely to increase as the soil is increasingly saturated with N following repeated high rate of N application (Yao et al., 2012).

4.4 Watershed level losses of N

We would anticipate that the Pearl Oatka watershed, a rural watershed dominated by agriculture, has high potential for N leaching at the watershed level. Oatka Creek's Wyoming Road segment, the closest downstream monitoring station to our study area, lies within the larger watershed's headwaters, where the small tributaries, including Pearl-Oatka Creek, merge into the main stem. Concentrations of NOx, and NH_4^+ at the Wyoming Road and Garbutt stations generally meet water quality standards, and are higher in winter than summer, due to increased runoff and slower crop uptake during the colder months (Commission, 2002).

We compared our watershed parameters and primary results (curve numbers, N loss, and NOx components) with Pettenski's (2012) study of the Oatka Creek Watershed. We limit our comparison to the Wyoming Road section, which incorporates a comparable area. As the number of CAFOs and agricultural fields in the region is high (highest agricultural production of all NYS counties), with eight CAFOs within a radius of 40 miles, N availability is twice as high as other sections of Oatka Creek and the measured water concentrations of nitrate and loading per unit area were comparably elevated relative to downstream at the Garbutt, NY, gauging station (Pettenski, 2012). For instance, the loading at the Wyoming Road segment was 27.2 kg $ha^{-1}yr^{-1}$ relative to downstream at Garbutt where loading was 21.3 kg $ha^{-1}yr^{-1}$. These loading rates are somewhat higher (1.5-3 fold) than all of our SWAT export estimates for NO_3^- leaching plus overland flow, except for the scenario with 50% higher digestate application rates. It should be noted that these two monitoring stations collect the water sample drained from a larger creek than the Pearl Oatka watershed alone. We do not incorporate the potential release of N associated with CAFOs aside from manure spreading, but this discrepancy suggests that we may underestimate the actual application of fertilizer to fields.

The main goal of the SWAT analysis was to check if the N loss behavior derived from our mass-balance model was realistic at the watershed scale. The general patterns between results from the two methods were remarkably similar, with generally similar patterns of gaseous and dissolved N loss across scenarios and in all cases the SWAT output was within the overall range of variability of the mass balance estimate. The greatest differences at the per ha estimate were for the nitrate leaching, where the SWAT values were significantly lower for the IG and DI scenarios, but remarkably similar for the manure. This is likely a result of differences in the model assumptions and working environment. For instance, the mass-balance N losses for the IG treatments were derived from a compilation of available literature for a variety of inorganic fertilizer types, whereas SWAT simulated a single type of inorganic fertilizer (anhydrous ammonia) for all scenarios. Thus, the mass balance model is very sensitive to variability in the EF, as shown by the large error estimates. Further, SWAT incorporates simulated weather data of 2018 – 2020 for precipitation, and from this calculates runoff, surface, and lateral flow, etc., which the mass-balance model doesn't directly incorporate. From our geospatial analysis, we can infer that the digestate pathway mass-balance model is verifiable within a watershed scale, especially for an agricultural watershed.

When we projected each scenario across the watershed, utilizing the available digestate and manure to completion, and then applying inorganic fertilizer to complete the balance, only 15% of the crop area was needed to consume all of the digestate at the baseline application rate (1,250 of 8,157 ha cropland). An additional 16% was covered by manure generated outside of the CAFO sited at the digester, and the remainder was treated with inorganic fertilizer only. Increasing the FW import to the watershed two-fold resulted in 45% increase in digestate production which was sufficient to treat 24% of the crop area; the area treated with inorganic fertilizer was reduced accordingly and the manure treatment area was the same.

Under the current production of digestate (MULT, Table 7) the release of both N_2 (for both the mass balance and the SWAT prediction) and N_2O (mass balance result only; SWAT does not predict) is similar to the scenario where only inorganic N (IG) is applied. This suggests that at the current production rate, long-lived N-based GHG production does not increase, and land application may not carry increased risk as digestate replaces the application of some inorganic fertilizer and carries a lower EF. There were increases in NH_3 release for the MULT scenario (more for the mass balance estimate than the SWAT estimate), however, suggesting increased risk for deposition of reactive N in waterways or adjacent ecosystems following atmospheric transport. The NO_3^- leaching was similar for the DI and MULT scenarios in the mass balance, but about 32% higher for the SWAT estimate, likely reflecting the difference in EF for both IG and DI incorporated into the geospatial model and the more local soil conditions. For both the mass balance and SWAT predictions, increasing the digestion of FW and spreading of the generated material at the current levels as a partial substitute of inorganic fertilizer does not pose a substantially greater risk of GHG or NO_3^- export to local waterbodies relative to the use of inorganic fertilizer alone at the field stage, suggesting that any increase in FW processing must be accompanied by an emphasis on maintaining low rates of application across a larger number of fields and better management at the storage phase. Remedying these potential risks assumes responsibility for changes to digestate storage capacity, digestate transportation (spreader capacity, leakage risks, and CO_2 emissions risks during transport), and logistical/management practices (ownership of digester vs. fields).

Our field-based watershed-scale result showed that ecological (N loss) risks to waterways are greater at the field (per ha) level for digestate use relative to manure or inorganic fertilizer, primarily because of the higher rate of application of digestate relative to manure. When viewed at scale, with the limited number of crop area receiving digestate, the net impact of digestate on N loss to waterways declines. Thus, in a highly agricultural watershed, the net increase of installing a co-AD FW processing facility is relatively small. Similarly, the GWP for inorganic fertilizer use alone is the lowest of all scenarios evaluated (**Error! Reference source not found.**), and increases roughly 30% when manure is included at both the storage and field phases. But incorporating FW at current or even double rates does not substantially increase the overall GWP, primarily because the base rate for current agricultural operations is so high.

4.5 Policy Implications

As pointed out across literature, and throughout this thesis, digestate management is often neglected during site selection for Anaerobic Digesters. Poor siting means that the high volumes of digestate generated continuously must be managed and transported to fields within the periphery, increasing the risk of GHG and soluble N release, particularly when storage space is limiting. Increasing application rates to fields in the immediate vicinity of the digester exacerbates the ecological and GHG risks over the long term, negating the positive economic benefit of biogas generation. Thus, decentralization of digester siting, where smaller volumes are generated and applied to fields within a reasonable transportation distance, may provide greater economic benefits and minimized ecological risks. Best practices will include a closed and cement-lined storage operation of sufficient size to store digestate over many months, frequent cleaning to pond to avoid sedimentation, tapping of fugitive emissions, digestate transportation strategies to ensure decentralized field application, etc. While construction of right-sized storage facilities (especially, cemented and covered), can be expensive, the additional storage capacity helps ensure field application at an appropriate time.

Economic incentives or regulatory policies to promote resource stewardship may provide a greater economic value to digestate, thereby avoiding the risks of over-application. To stabilize nutrients and prevent eutrophication after digestion and before storage, identifying the feasible routes of optimal nutrient recovery following digestion (like Ammonia Stripping) can be helpful. For instance, an algorithm by Vaneeckhaute et al., (2017) suggests an alternative route to choose for nutrient recovery based on the N and P content, pH, alkalinity, etc. of the digestate, which helps to accommodate for the high variability of digestate composition. Furthermore, there are methods of adding value to the digestate, like algal growth, pychars as soil amendments/bio-adsorbents, etc. with an associated reduction in ecological risk associated with each technique (Monlau et al., 2015).

Prior to digester establishment, an assessment of ecological risks with geospatial models (such as the SWAT used here) by construction firms may assess the nutrient load each watershed can handle in order to plan for decentralizing the digestate in the farm base. Most importantly, knowledge and information dissemination among operating ADs and research labs/universities can create a stronger knowledge base regarding FW-based digestate risks and best practices. For the FW based nutrient recovery and management at a state level, a more detailed projection of future FW availability and composition, along with manure and farm base availability, may aid in visionary planning and effective implementation.

5. Conclusion

This study investigated the ecological risk of food-waste based co-digestate through a series of N loss estimates during storage and field application. We focused on AcoD, considering its reputation as a sustainable FW valorization technique, and found that adjusting the digestate management (storage to distribution) practices will ensure fewer ecological threats at the post-digestion phase. Our empirical GHG emissions at storage and field phases of the digestate life-cycle showed heterogeneity over seasons and mode of field applications. The mass-balance model and geospatial model showed that the volume of digestate in continual storage and the rate of application to fields are the key drivers of emissions and leaching of N to waterways and thus represent the two primary leverage points for a sustainable WtE FW industry.

The variables that affect N loss during digestate storage most significantly in our limited study were the digestate N content and volume of digestate in storage at any one time, which is linked to the field application timeline as production of digestate is relatively constant. Due to the time and resource limitations, assumptions on input parameters were made to simplify the mass-balance modeling, and the compiled literature values showed a great deal of variability primarily because of the lack of relevant data. Even with these limitations, our model showed that N loss at the watershed scale is accounted for mostly by the volume of digestate stored during the warmest months, which is thereby impacted by seasonal variations in demand for fertilizer and limitations imposed by weather and crop phenology. Additional variables affecting N loss during application were digestate N content, type of crops grown, the frequency, quantity, and methods of field application, soil characteristics, history of fertilizers applied previously, and rainfall intensity. Together, these sources of risk-enhancement point to the need for greater care in facility siting, and the need to ensure, especially, adequate access to crops for disposal.

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Appendices

Appendix A. Supplemental Figures and Tables

Figures

Figure A.1: Digestate application in the field.

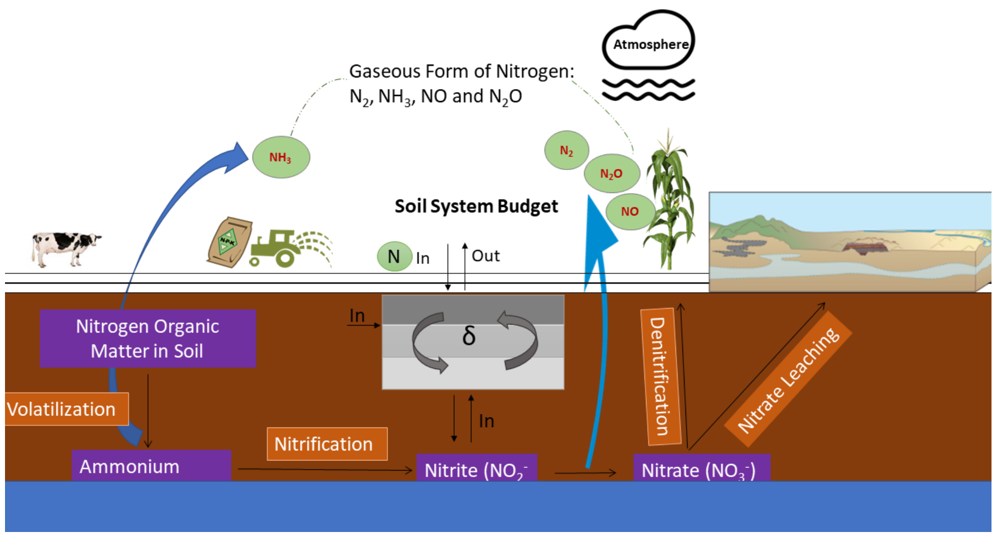


Figure A.2: Nitrogen cycle in agricultural fields illustrating N transformation from fertilizer into the atmospheric form and associated leaching risks to ecosystem.

Figure A.3: Digester facility, illustrating the location of the three digestate storage ponds.

Figure A.4: On-site and satellite digestate storage ponds.

Figure A.5: Gas trap designed to measure GHG emissions from digestate storage pond.

Figure A.6: The gas trap submerged on the edge of the lagoon pond.

Figure A.7: Soil chambers for flux measurements

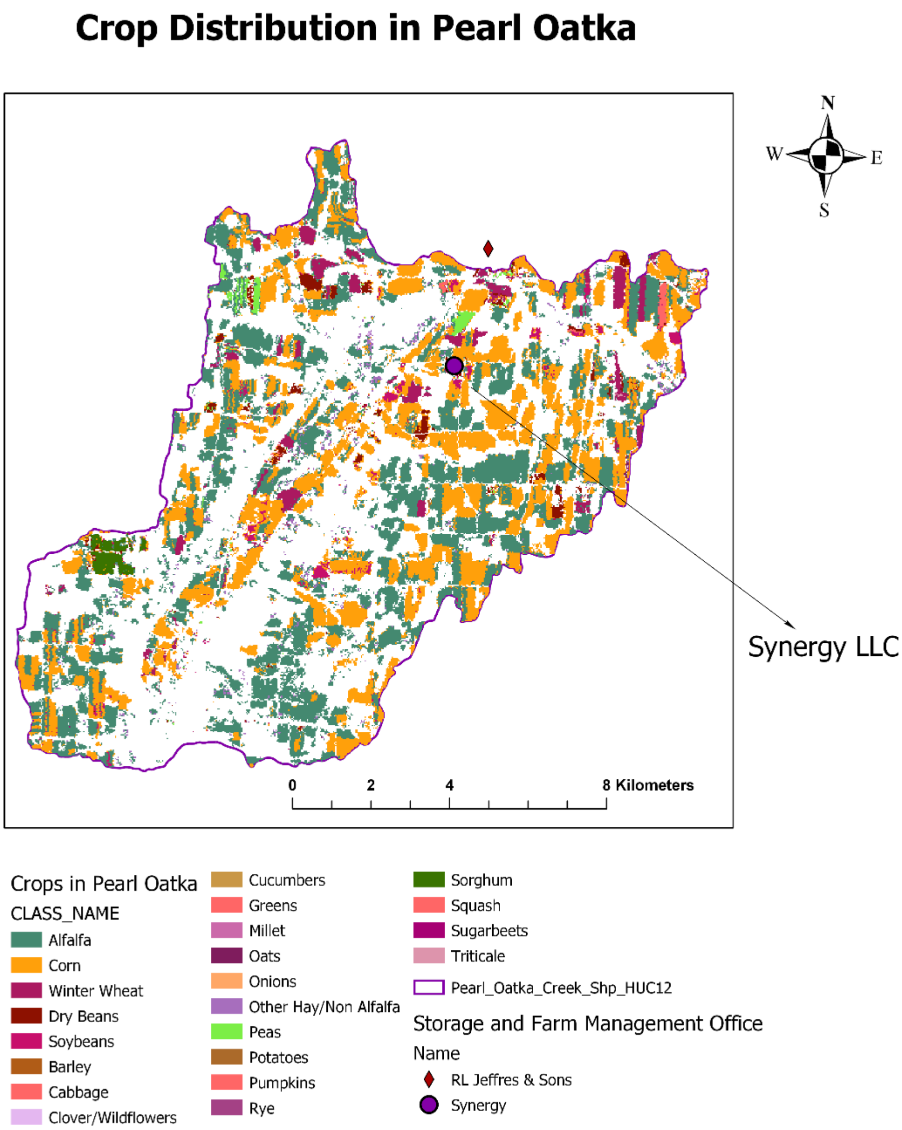


Figure A. 8: Crop distribution in Pearl Oatka Watershed

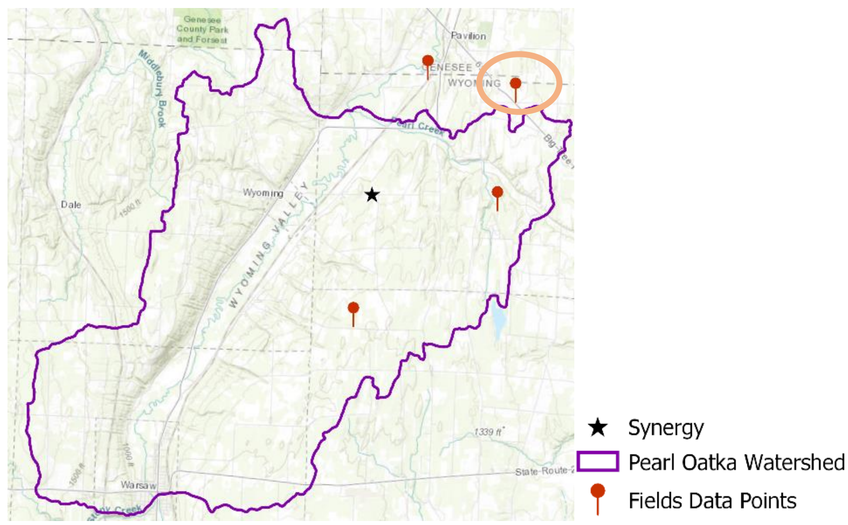


Figure A.9: Field sites for measuring GHG emissions of FW digestate application (circled indicates location where multiple measurements were made over time).

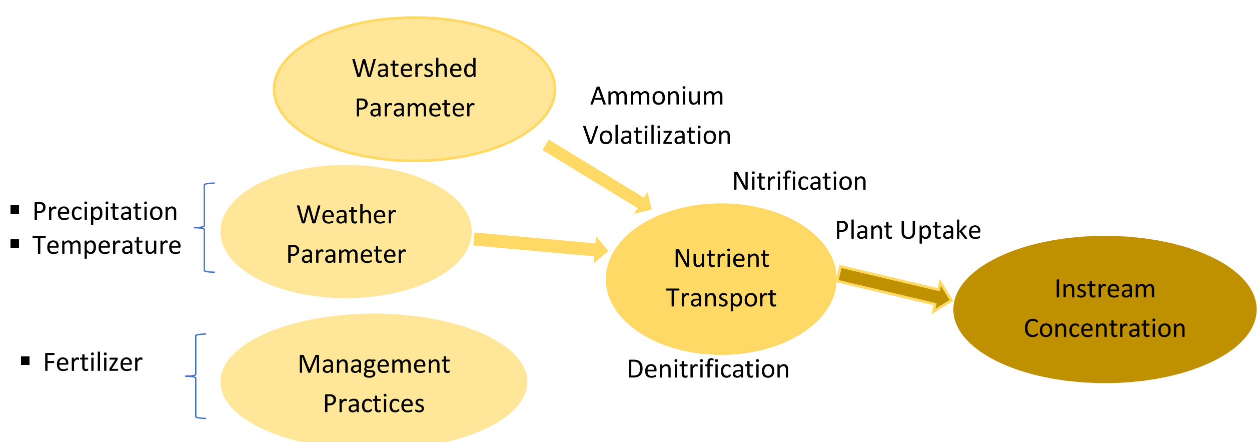


Figure A. 9: Sequential steps of SWAT model.

A. Literature Review

Storage Emissions

Table A. 1: Literature values for N_2 emissions (g N_2-N/m^3/d) during storage of digestate and manure slurry.

Fertilizer Type	Feedstock	Season	Study Design	Region	g N_2-N/m^3/d	Source
Manure	Dairy cattle farmyard manure	Early July	Field experiment (stored tank)	Devon, UK	1.94	Moral et al., 2012
	Dairy Cattle	September and October	Field experiment (pile-covered)	Guelph, Ontari	0.16	Tenuta et al., 2001
Mean					1.05 ± 1.26	

Table A. 2: Literature values for N_2O emissions (g N_2O-N/m^3/d) during storage of digestate and manure slurry.

Fertilizer Type	Feedstock	Season	Study Design	Region	g N_2O-N/m^3/d	Source
Manure	Dairy Cattle	Summer	Field experiment (stored tank)	Austria	0.16	Amon et al., 2006
	Slurry with potato	Summer and winter	Field experiment (stored tank)	Germany	0.25	Clemens et al., 2006
	Dairy Cattle	Early July	Field (stored concrete tank)	Austria	0.13	Moitzi et al., 2007

	Manure of cattle (mainly), pigs and mink,	Summer and Autumn	Field experiment (stored tank)	Denmark	0.04	Baral et al., 2017
	Dairy cattle farmyard manure	Early July	Field experiment (stored tank)	Devon, UK	0.37	Moral et al., 2012
	Dairy Cattle	September and October	Field experiment (pile-covered)	Guelph, Ontari	0.07	Tenuta et al., 2001
	Mean				0.17 ± 0.12	
Digestate	Dairy Cattle	Summer	Field experiment (stored tank)	Austria	0.23	Amon et al., 2006
	Slurry with potato	Summer and winter	Field experiment (stored tank)	Germany	0.29	Clemens et al., 2006
	Dairy Cattle	Early July	Field (stored concrete tank)	Austria	0.18	Moitzi et al., 2007
	Manure of cattle (mainly), pigs and mink,	Summer and Autumn	Field experiment (stored tank)	Denmark	0.02	Baral et al., 2017
	Co-fermented (FW and manure)	Summer-Fall	Field Experiment (Gas Trap)	Wyoming County, NY, USA	$5.08*10^{-5}$	This study
	Mean				0.14 ± 0.13	

Table A. 3: Literature values for NH3 emissions (g NH_3-N/m^3/d) during storage of digestate and manure slurry.

Fertilizer Type	Feedstock	Season	Study Design	Region	g NH_3-N/m^3/d	Source
Manure	Dairy Cattle	Summer	Field experiment (stored tank)	Austria	0.42	Amon et al., 2006
	Slurry with potato	Summer and winter	Field experiment (stored tank)	Germany	0.62	Clemens et al., 2006
	Dairy Cattle	Early July	Field (stored concrete tank)	Austria	0.34	Moitzi et al., 2007
	Manure of cattle (mainly), pigs and mink,	Summer and Autumn	Field experiment (stored tank)	Denmark	0.42	Baral et al., 2017
	Dairy cattle farmyard manure	Early July	Field experiment (stored tank)	Devon, UK	0.55	Moral et al., 2012
	Mean				0.47 ± 0.12	
Digestate	Dairy Cattle	Summer	Field experiment (stored tank)	Austria	0.10	Amon et al., 2006
	Slurry with potato	Summer and winter	Field experiment (stored tank)	Germany	0.91	Clemens et al., 2006
	Dairy Cattle	Early July	Field (stored concrete tank)	Austria	0.08	Moitzi et al., 2007
	Manure of cattle (mainly), pigs and mink,	Summer and Autumn	Field experiment (stored tank)	Denmark	2.92	Baral et al., 2017

	Mean	1.00 ± 1.34	

Field Emissions

Table A. 4: Literature values for N_2 emission factors (EF; as %N lost to the atmosphere) following field spreading of digestate, manure slurry, or inorganic fertilizer.

Fertilizer Type	Feedstock	Spreading Method	Season	Study Design	Region	EF	Source
Manure	Cattle	Top-spread and injection	Feb-Mar	Field experiment	Hampshire, UK	3.60	Ellis et al., 1998
	Cattle	Top-spread	Summer	Greenhouse experiment	British Columbia, Canada	1.61	Paul et al., 1998
Mean						2.61 ± 1.40*	
Digestate	Note: EF of manure was used for digestate*						
Inorganic	NH_3NO_3	Top-spread and injection	Feb-Mar	Field experiment	Hampshire, UK	2.10	Ellis et al., 1998
	Inorganic-urea-300 and 600	Top-spread	Growing season	Field experiment (Randomized block experiment)	Eastern China	5.55	Cao et al., 2006
Mean						3.28 ± 2.44	

Table A. 5: Literature values for N_2O emission factors (EF; as %N lost to the atmosphere) following field spreading of digestate, manure slurry, or inorganic fertilizer.

Fertilizer Type	Feedstock	Spreading Method	Season	Study Design	Region	EF	Source
Manure	Dairy Cattle	Injection	August	Field experiment	Austria	0.07	Amon et al., 2006
	Cattle slurry	Top-spread	May	Field experiment	Netherlands	1.21	Van Groenigen et al., 2004
	Dairy cattle slurry + potato starch	Top-spread	Summer and winter	Field experiment	Germany	0.1	Clemens et al., 2006
	Dairy Cattle	Top-spread or shallow disk injection	Annual	Field experiment	Pennsylvania	1.3	Duncan et al., 2019
	Livestock slurry	Injection	Spring and autumn	Field experiment	England	0.45	Nicholson et al., 2017
	Dairy cattle slurry	Top-spread	Spring and autumn average	Field experiment	Ontario, Canada	1.6	Schwager et al., 2016
	Dairy cattle slurry	Top-spread	Summer	Model based on Europe	Denmark	0.65	Sommer et al., 2004
	Organic amendments – liquid and	Not mentioned	149 observations	Meta-analysis	Global	0.96	Charles et al. 2017

	solid manure, compost, crop residue, wastewater, biosolid etc.						
Mean						0.79 ± 0.56	
Digestate	Food waste	Injection	August	Field experiment	Austria	0.07	Amon et al., 2006
	Food waste	Top spread (using hand and bottle in experimental plot)	May, June and September	Field experiment	Devon, UK	0.25	Pezzolla et al., 2012
	Livestock slurry	Injection	Spring and autumn	Field experiment	England	0.45	Nicholson et al., 2017
	Dairy cattle slurry + potato starch	Top-spread	Summer and winter	Field experiment	Germany	0.08	Clemens et al., 2006
	Silage maize and addition of a nitrification inhibitor (Piadin)	Injection	Full year	Field experiment	Central Germany	0.21	Wolf et al., 2014
	Dairy cow slurry with organic household wastes	Co-fermented	spring-summer	Injection	Bonn, Germany	0.17	Wulf et al., 2002

	Dairy cattle slurry	Top-spread	Spring and autumn average	Field experiment	Ontario, Canada	1.25	Schwager et al., 2016
	Cattle slurry	Top-spread	Summer	Model based on Europe	Denmark	0.30	Sommer et al., 2004
	Organic amendments – liquid and solid manure, compost, crop residue, wastewater, biosolid etc.	Non-mentioned	10 observations	Meta-analysis	Global	0.92	Charles et al., 2017
	Silage maize	Not mentioned (Top spread?)	annual	Field experiment	Germany	0.32	Dicke et al., 2015
	Co-fermented (FW and manure)	Top-spread and injection	Summer-Fall	Field Experiment	Wyoming County, NY, USA	2.1	This study
Mean						0.59 ± 0.65	
Inorganic	Calcium ammonium nitrate	Top-spread	May	Field experiment	Netherlands	1.18	Van Groenigen et al., 2004
	Inorganic-urea-300 and 600	Top-spread	Growing season	Field experiment (Randomized block experiment)	Nanjing, China	1.32	Cao et al., 2006

	Dairy cattle slurry + potato starch	Top-spread	Summer and winter	Field experiment	Germany	0.5	Clemens et al., 2006
	Dairy cattle slurry	Top-spread	Spring and autumn average	Field experiment	Ontario, Canada	1.90	Schwager et al., 2016
	Organic amendments – liquid and solid manure, compost, crop residue, wastewater, biosolid etc.	Injection	99 observations	Meta-analysis	Global	1.34	Charles et al. 2017
	Mineral fertilizer (MIN)	Injection	Full year	Field experiment	Central Germany	0.12	Wolf et al., 2014
Mean						1.06 ± 0.64	

Table A. 6: Literature values for NH_3 emission factors (EF; as %N lost to the atmosphere) following field spreading of digestate, manure slurry, or inorganic fertilizer.

Fertilizer Type	Feedstock	Spreading Method	Season	Study Design	Region	EF	Source
Manure	Livestock slurry	Top-spread and injection	Spring and autumn	Field experiment	England	27.5	Nicholson et al., 2017

	Dairy cattle	Surface, incorporated, injected - using mean of all modes	Annual	Field experiment	Wisconsin	11.5	Powell et al. 2011
	Dairy cattle	Top-spread or shallow disk injection	Annual	Field experiment	Pennsylvania	12.1	Duncan et al., 2019
	Dairy cattle slurry + potato starch	Top-spread	Summer and winter	Field experiment	Germany	6.5	Clemens et al., 2006
	Cattle	Injection	August	Field experiment	Austria	4.71	Amon et al., 2006
Mean						12.46 ± 8.98	
Digestate	Cattle	Injection	August	Field experiment	Austria	7.31	Amon et al., 2006
	Livestock slurry	Injection	Spring and autumn	Field experiment	England	40.00	Nicholson et al., 2017
	Dairy cattle slurry + potato starch	Top-spread	Summer and winter	Field experiment	Germany	11.8	Clemens et al., 2006
	Silage maize and addition of a nitrification inhibitor (Piadin). Value w	Injection	Full year	Field experiment	Central Germany	13.67	Wolf et al., 2014

	inhibitor not used						
Mean						18.20 ± 14.78	
Inorganic	Mineral fertilizer (MIN)	Top-spread	Full year	Field experiment	Central Germany	0.33	Wolf et al., 2014
	Dairy cattle slurry + potato starch	Top-spread	Summer and winter	Field experiment	Germany	5.00	Clemens et al., 2006
	Mineral fertilizer (MIN)- Urea	Top-spread	Cropping season	Review article	N America	17.50	Pan et al., 2016
	Inorganic-maize-integrated horizontal flux (L)	Top-spread and injection	June-October	Field experiment	North China	21.87	Pacholski et al., 2008
Mean						11.17 ± 10.17	

B. Tables

B. 1: Range of the physical-chemical characteristics of the fresh liquid digestate and in lagoon ponds (based on 'as fed' and 'dry matter' basis) in 2019's monthly samples.

Measurement	**Range**					
	Liquid digestate		**Lagoon 1**		**Lagoon 2**	
	As fed	DM	As fed	DM	As fed	DM
Moisture (%)	95.17-98.83	0	93.48-98.64	0	97.76-99.05	0
Dry Matter (%)	1.17-4.83	0	1.36-6.52	0	0.95-2.24	0
Phosphorus (%)	0.002-0.076	0.10-3.57	0.01-0.05	0.22-3.53	0.01-0.07	0.28-5.07
Phosphorus, as P_2O_5 (%)	0.05-0.173	0.24-11.54	0.02-0.11	0.54-8.09	0.02-0.16	0.66-11.54
Potassium (%)	0.03-0.09	0.97-6.07	0.03-0.22	0.49-16.10	0.05-0.26	1.20-19.12
Potassium, as K_2O (%)	0.038-0.111	1.16-7.31	0.03-0.26	0.57-19.41	0.05-0.31	1.44-23.01
Total Nitrogen (%)	0.11-0.27	4.45-17.52	0.11-0.24	2.73-8.75	0.09-0.22	1.43-8.44
Ammonia (%)	0.03-0.16	0.66-11.88	0.04-0.14	0.70-5.59	0.06-0.12	1.06-4.85
Organic N (%)	0.05-0.18	3.60-9.17	0.04-0.18	1.09-4.65	0.01-0.101	0.37-3.95
Carbon (%)	0.42-0.76	33.11-36.89	0.57-2.71	41.56-45.86	0.15-0.46	4.29-15.86
C/N ratio	2.22-3.17	2.22-3.17	5.04-15.22	5.04-15.22	1.67-3.01	1.67-3.19

B. 2: Digestate characteristics comparing characteristics of our liquid digestate samples of 2019) with available literature.

Parameter	**FW digestate concentration (Bimonthly 2018)**	**FW digestate concentration (Monthly 2019)**	**Values from literature**	**References**
DM (%)	2.1 – 7	1.17 – 4.83	4.5 – 6.6	(Bauer et al., 2009; Möller et al., 2009)
Total N (% DM)	4.44-10.51	4.45 – 17.52	7.7 – 9.2	(Kirchmann and Witter, 1992)
Total C content (% DM)	0.77 – 50	33.1 – 36.89	48	
C: N ratio (as DM)	2.06 – 11.25	2.22 – 3.17	3.7 – 4.8	(Möller et al., 2009; Möller and Müller, 2012)
Total P content (% DM)	0.73 – 2.22	0.10 – 3.57	0.4 – 0.7	Möller & Müller, 2012; Möller et al., 2009)
Potassium (% DM)	1.89 – 5.05	0.97-6.07	3.9	Möller & Müller, 2012; Möller et al., 2009)

B. 3: Factors affecting emissions at the storage phase (Data derived from Tel-Tek, 2013).

Factor	Nitrous Oxide	Ammonia	Methane
pH	Optimum at 6 (negligible at <5 and >8)	Increase	Optimum at 7 (50% at 6.5 and 8.3)

Temperature	Summer emissions are 2x greater than winter.	Summer emissions are approx. 4x greater than winter.	No emissions below 0^0C, and low below 15^0C, with exponential increase above 15^0C. Summer emissions are increased by approx. ten times than winter.
Crust and covers	Increase (occurs at biofilms)	Reduce	Reduce

B. 4: Organic matter analysis at each site collected before and after digestate application. Values are the percent organic matter expressed as mean (standard deviation) for n=3 samples.

	Before	After
April	4.22 ± 0.15	7.03 ± 0.09
May	6.79 ± 0.12	7.39 ± 0.27
07-Jun	4.67 ± 0.07	4.66 ± 0.0
10-Jun	3.80 ± 0.07	5.35 ± 0.07
27-Jun	9.73 ± 0.31	8.29 ± 1.13
Aug	6.26 ± 0.06	6.17 ± 0.13
Sept	4.85 ± 0.04	4.89 ± 0.07
10-Oct	ND	5.34 ± 0.08
30-Oct	8.80 ± 0.04	9.17 ± 0.07

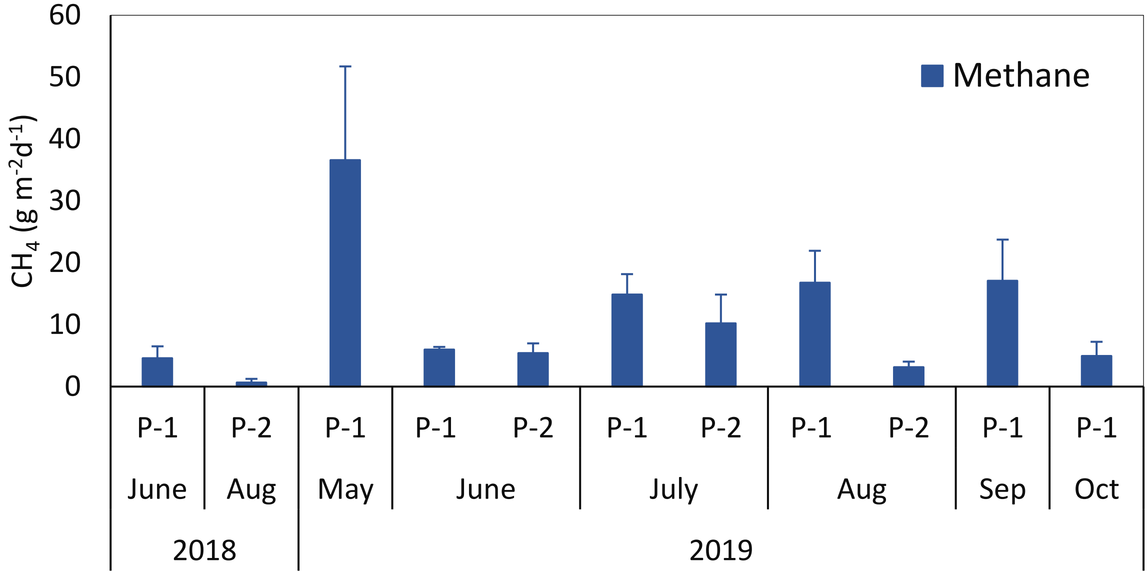
Appendix C. Supplementary Data

Supplementary Information

C.1. Carbon Emissions:

Storage emissions:

Mean daily release of methane from of stored digestate ranged from 4.6 to 36.0 g CH_4 $m^{-2}d^{-1}$ in Summer and 4.9 – 17.1 g CH_4 $m^{-2}d^{-1}$ in Fall and 0.01 – 55.0 g CO_2 $m^{-2}d^{-1}$ in Summer and 4.0 – 17.6 g CO_2 $m^{-2}d^{-1}$ in Fall for methane and carbon dioxide. Literature values for AD storage emission of methane ranged widely from 81 CH_4 g/m^3 to 1343 CH_4 g/m^3, varied seasonally and were dependent upon the cover of the lagoon storage ponds (Table A.1). Methane production is significantly reduced during digestion (Amon et al., 2006). For digestate, the net CH_4 loss mostly takes place at storage (by 99.9%) relative to field during application (0.4%) (Amon et al., 2006). In the case of untreated manure, 100% of CH_4 loss occurs at storage. For methane, the summer fluxes for uncovered digestate are higher than the covered storage ponds, however, there is no difference in the winter (Rodhe et al., 2015). Methane production depends upon surface temperature, but a detailed study on daily surface temperature, wind speed, and direction, agitation, etc. is needed to provide a complete synopsis of methane emissions. Thus, C transformation (like N transformation) also depends upon the feedstock composition and digestate handling by the AD facility. Capping of primary digestate storage ponds has the potential to significantly reduce GHG release and to provide a mechanism for re-capture of fugitive methane emissions.



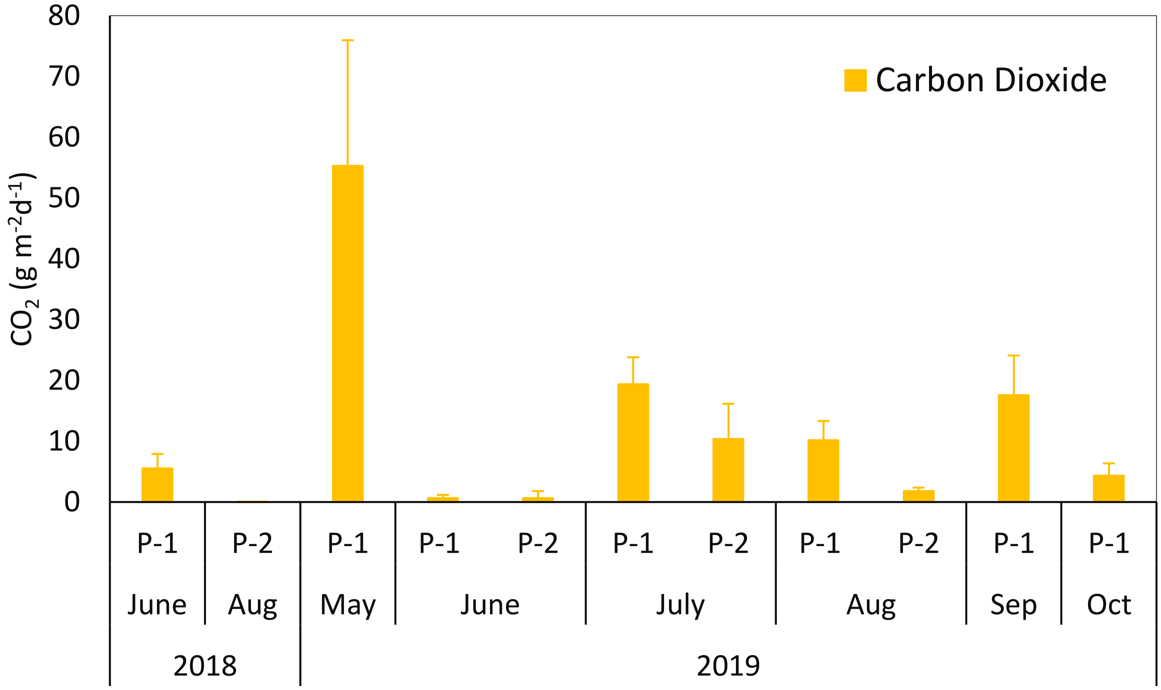


Figure B.1: Mean daily release of methane (mean +/- SE) (top panel), carbon dioxide (bottom panel). Emission of GHG from digestate storage ponds during the summers of 2018 and summers and falls in 2019.

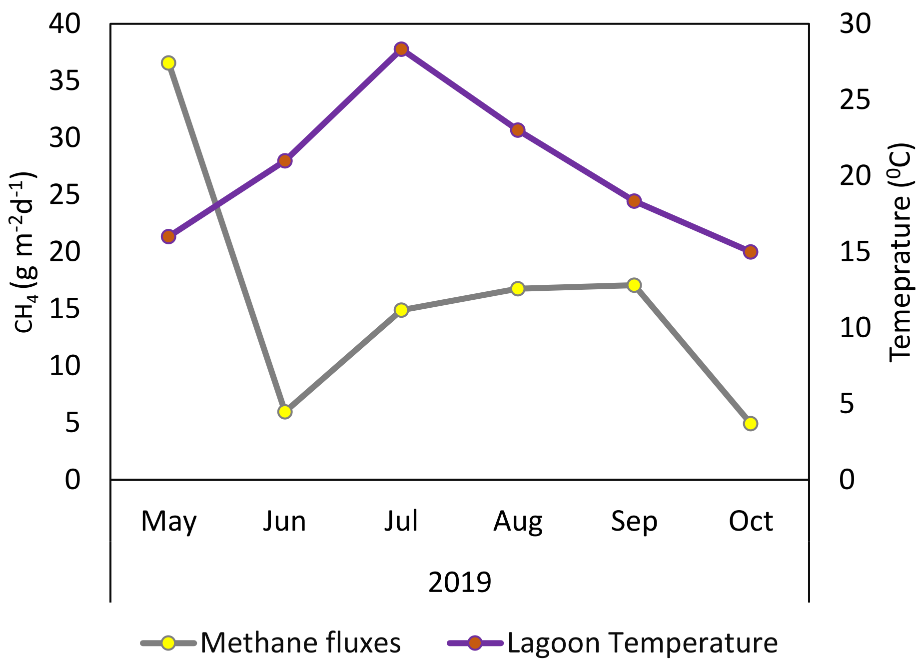


Figure B.2: Mean daily fluxes of methane (in g CH_4 m^{-2}d^{-1}) and lagoon surface temperature (^{0}C).

Field application

The post-digestate emissions were observed to be always greater than pre-digestate emissions for the same field condition and temperature, which are contributed by the CH_4 and CO_2 constituents in the digestate Figure B.2. The field CH_4 and CO_2 don't show have any difference over the summer and fall of 2019. These emissions level stabilized after a few weeks of digestate application, as shown in Figure B.4.

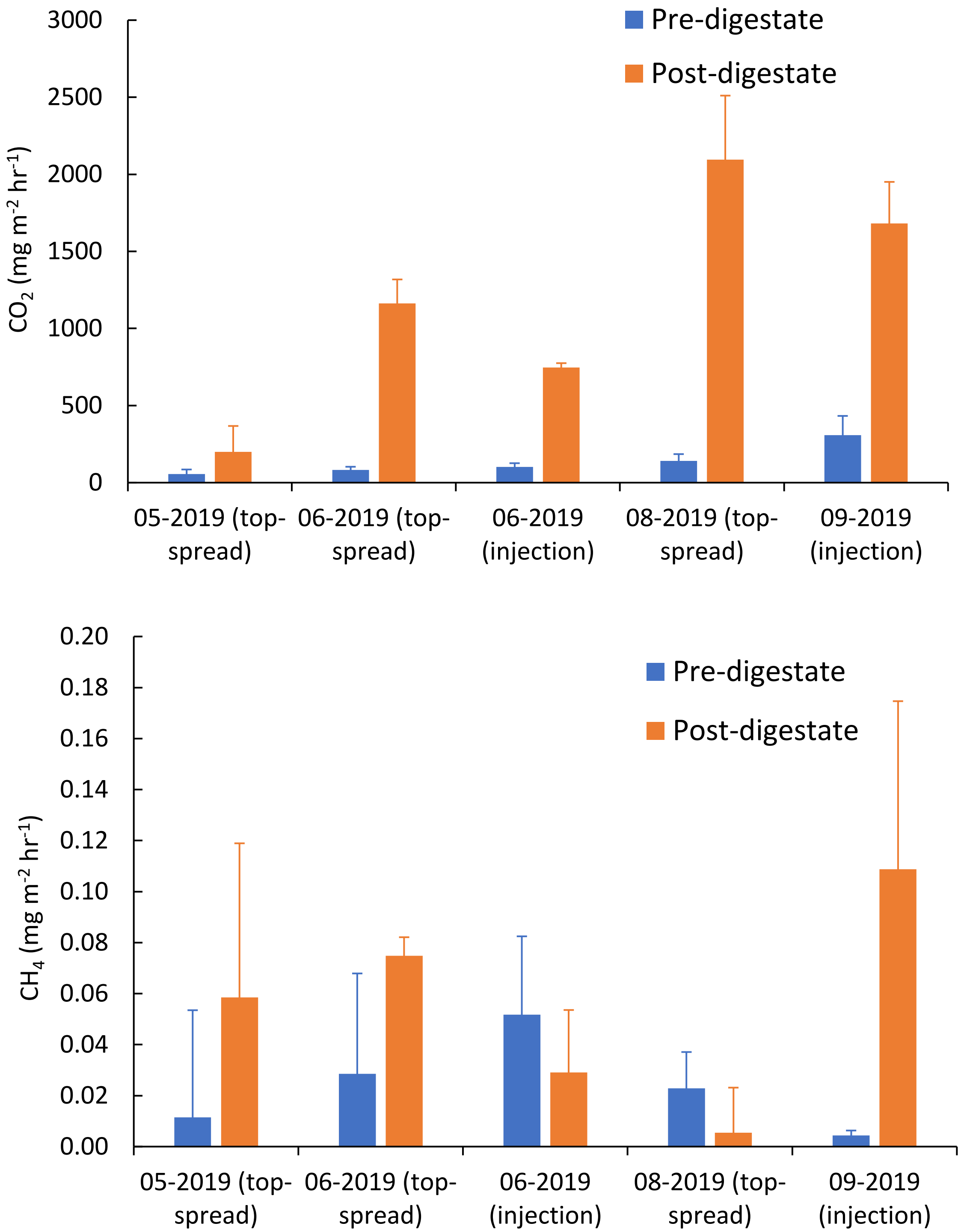


Figure B.3: CH_4 (top-panel) and CO_2 (button-panel) emissions from fields (mean +/- SE) measured in Summer and Fall of 2019.

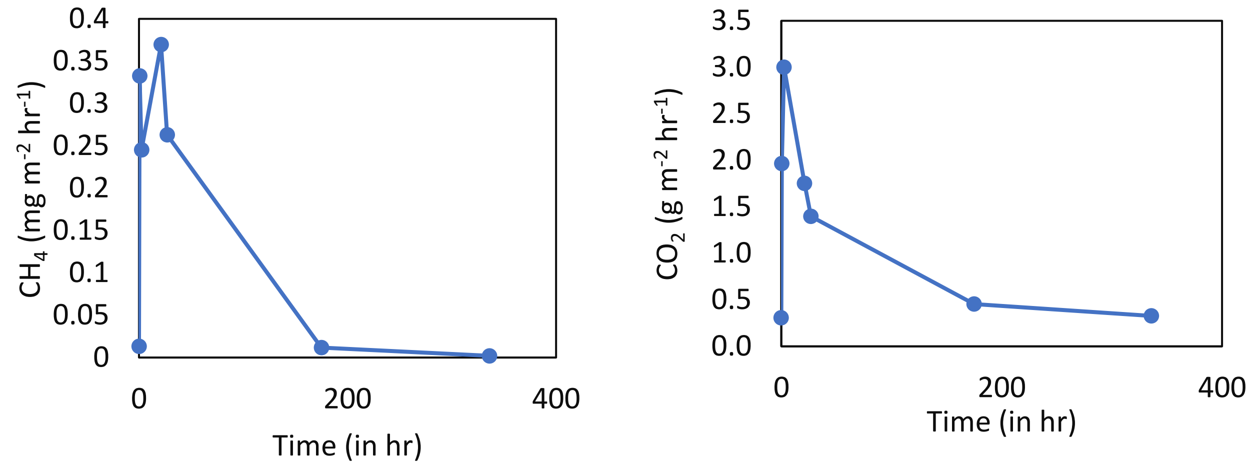


Figure B.4: CH_4 and CO_2 emitted in agricultural fields from digestate application (injection) over a two-week period.