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ADDRESSING THE NITROGEN CHALLENGE: FOOTPRINT TOOLS AND ON-FARM SOLUTIONS

BY

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DISSERTATION

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DEDICATION

I dedicate this dissertation to my late grandfather, John Gensior, whose encouragement and support made it all possible.

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TABLE OF CONTENTS

CHAPTER 3: CHARACTERIZING COMPOST EXHAUST GAS FROM AERATED STATIC PILE HEAT RECOVERY COMPOSTING

LIST OF TABLES

CHAPTER 1: AN INTEGRATED TOOL FOR CALCULATING AND REDUCING INSTITUTION CARBON AND NITROGEN FOOTPRINTS

CHAPTER 2: HOW AERATED STATIC PILE HEAT RECOVERY COMPOSTING AFFECTS AN ORGANIC DAIRY FARM'S NITROGEN BUDGET

CHAPTER 3: CHARACTERIZING COMPOST EXHAUST GAS FROM AERATED STATIC PILE HEAT RECOVERY COMPOSTING

LIST OF FIGURES

CHAPTER 1: AN INTEGRATED TOOL FOR CALCULATING AND REDUCING INSTITUTION CARBON AND NITROGEN FOOTPRINTS

CHAPTER 2: HOW AERATED STATIC PILE HEAT RECOVERY COMPOSTING AFFECTS AN ORGANIC DAIRY FARM'S NITROGEN BUDGET

CHAPTER 3: CHARACTERIZING COMPOST EXHAUST GAS FROM AERATED STATIC PILE HEAT RECOVERY COMPOSTING

ABSTRACT

ADDRESSING THE NITROGEN CHALLENGE: FOOTPRINT TOOLS AND ON-FARM SOLUTIONS

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Nitrogen management presents a unique dilemma: We must use nitrogen to grow our food and sustain life on earth, but excess reactive nitrogen that accumulates in the environment contributes to a cascade of negative impacts to human and ecosystem health. Addressing this nitrogen challenge will require a suite of solutions. This dissertation presents and explores three nitrogen management strategies: 1) The first ever integrated carbon and nitrogen footprint tool for campus sustainability management; 2) Exporting compost to improve a farm's nitrogen efficiency; and 3) Methods for reducing gas emissions from aerated static pile heat recovery composting.

Nitrogen footprint tools connect our everyday choices with the associated nitrogen pollution to the environment. The campus-level nitrogen footprint tool has been particularly successful at both communicating the nitrogen story and encouraging real change with nitrogen footprint reduction goals. However, it is important to assess environmental impacts together to identify management strategies and avoid trade-offs. In this paper, the development and methodology behind the first ever integrated carbon and nitrogen footprint tool for campuses is presented. Comparisons of campus carbon and nitrogen footprints show that the footprints correlate strongly, and scenario analyses indicate benefits to both footprints from a range of management strategies. Integrating the carbon and nitrogen footprints into a single tool for campuses facilitates more comprehensive and integrated management of campus sustainability.

Food production is a significant source of nitrogen pollution, and new and improved farm nitrogen management practices are necessary to reduce nitrogen losses. In this study, aerated static pile heat recovery composting is considered as a nitrogen management strategy. To assess its potential, the nitrogen budget of an organic dairy farm was first assessed, where it was found that organic practices led to the cycling of substantially more nitrogen on the farm property than was imported or exported. Some of the potential farm nitrogen loss pathways were characterized, including gas emissions from the compost facility (ammonia, carbon dioxide, methane), but future research should characterize other nitrogen loss pathways to assess the balance between storage and environmental loss. Management strategies for reducing greenhouse gas and ammonia emissions from the compost facility were identified. Scenario analysis found that exporting finished compost was a viable strategy for improving the farm’s nitrogen use efficiency as long as enough nitrogen is retained on-site to support crop production.

INTRODUCTION

Nitrogen management presents a unique dilemma: We must use reactive nitrogen to grow our food and sustain life on earth, but excess reactive nitrogen that accumulates in the environment contributes to a cascade of negative impacts to human and ecosystem health (Galloway et al. 2003; Sutton et al. 2011; Erisman et al. 2013). Reactive nitrogen (all species N except the unreactive N_2 that makes up most of the atmosphere) takes many forms, including an essential building block of cells (e.g., amino acids, DNA, RNA); a nutrient for agriculture; a greenhouse gas (nitrous oxide, N_2O); a local air quality pollutant (ammonia, NH_3; nitrogen oxides, NOx); and a local water quality pollutant (nitrate; NO_3^-; ammonium, NH_4^+). All of these forms of reactive N are part of the N cascade, which is the concept that a single atom of reactive N can move through different environmental spheres—and cause multiple environmental impacts in sequence—before being converted back to unreactive N_2 through denitrification (Galloway et al. 2003).

1. The nitrogen challenge

The benefits from reactive N are clear: Reactive N is necessary for life. All organisms on earth need some form of N to survive, and almost all organisms require a reactive form of N (Galloway et al. 2014). Humans consume their reactive N as protein in food. The only organisms that do not need reactive N are specialized microorganisms that are able to fix N_2 from the atmosphere into a usable form. These microorganisms play a key role in the N cycle, and they were responsible for fixing most of the reactive N on earth before anthropogenic sources became dominant (Galloway et al. 2014). A minor natural source of reactive N is lightning. The heat and energy of lightning are strong enough to break the N_2 triple bond and form reactive N species.

Anthropogenic reactive N creation rates began to increase dramatically in the 20th century due to the broad use of the Haber Bosch process and, to a lesser degree, the increase in fossil fuel combustion (Erisman et al. 2008; Galloway et al. 2014). The Haber Bosch process is the synthetic combination of unreactive N_2 and hydrogen (H_2) into NH_3 under intense heat and pressure. Originally developed by German scientists Fritz Haber and Carl Bosch as a feedstock for munitions for World War I, the Haber Bosch process is mainly used today to make synthetic fertilizer and for industrial products (Smil 2001; Galloway et al. 2014). The synthetic fertilizer produced from the Haber Bosch process has essentially provided an unlimited supply of reactive N—formerly a limiting nutrient—for food production. In fact, almost 50% of the global population is supported by food produced from Haber Bosch fertilizer (Erisman et al. 2008). This means that without the Haber Bosch process, we would not have enough reactive N to grow food for 50% of the global population. Given this, the Haber Bosch process was arguably one of the most impactful inventions of the 20th century.

Although it has had significant benefits for food production, the anthropogenic creation of reactive N has also led to the accumulation of reactive N in the environment. N is a very leaky and mobile element (Galloway et al. 2003). Most of the reactive N used to produce food (e.g., as fertilizer, feed) is not contained in the final food product (Leach et al. 2012; Lassaletta et al. 2014). Pathways of N loss include fertilizer leaching and runoff, crop residue, manure, food waste, and more. Some of this N is recycled back into the food production chain, but most of it is lost to the air, water, and soils. On average worldwide, only a fraction of the reactive N used to grow food is contained in crops (<50%) and meat and animal products (<20%) (Lassaletta et al. 2014, 2016; Zhang et al. 2015). The global N cycle has become so unbalanced from anthropogenic activities that humans are now creating 5 times as much reactive N as is created through natural terrestrial processes (Vitousek et al. 1997, 2013).

‘Planetary boundaries’ is a methodology that identifies the level of risk for different Earth systems (Rockström et al. 2009; Steffen et al. 2015). Nine Earth systems—one of which is the nitrogen and phosphorus cycles—were identified and categorized as being at a safe level, a zone of increasing risk, or a critical zone of high risk. Because of the accumulation of N and phosphorus in the environment, the N and phosphorus cycles were identified as one of only two Earth systems that are in a ‘high risk’ zone. Of the nine Earth systems, the only other one deemed to be in a critical zone is genetic diversity. The remaining systems are in zones of increasing risk (climate change, land-system change), at a safe level (stratospheric ozone depletion, ocean acidification, freshwater use), or not yet quantified (novel entities, atmospheric aerosol loading).

Once in the environment, reactive N affects water quality, air quality, ecosystem services, and human health. Negative impacts to water quality include freshwater eutrophication, algal blooms, and hypoxic and dead zones (Galloway et al. 2003; Erisman et al. 2013). High concentrations of nitrates and nitrites in drinking water cause methemoglobinemia, which is a condition that reduces blood transport of oxygen and can lead to infant death (Knobeloch et al. 2000). Reactive N contributes to air quality problems like smog, particulate matter, and ground-level ozone. N_2O is a greenhouse gas that has a global warming potential 265 times more potent than CO_2 (IPCC 2014), and it also causes stratospheric ozone depletion (Erisman et al. 2013). Excess reactive N contributes to acid rain and forest N saturation, which can lead to forest decline (Aber et al. 1989). Biodiversity loss, the second planetary boundary that is in a ‘high risk’ zone, is also worsened by excess N (Rockström et al. 2009; Steffen et al. 2015). The N cascade is the concept that each atom of reactive N can cause each of these impacts in sequence before it is converted back to unreactive N_2 (Galloway et al. 2003). The combination of large quantities of excess reactive N with the N cascade has led to widespread impacts to the environment.

This brings us to our nitrogen challenge: We as a society must determine how to optimize the beneficial uses of N while minimizing its negative consequences. There are several broad approaches that can be taken to address the N challenge: 1) Policy, 2) Education and outreach, and 3) Technology. In this dissertation, strategies using the latter two approaches are presented.

2. Policy solutions for nitrogen management

Policy has been successful at reducing nitrogen oxide (NOx) emissions from fossil fuel combustion in the US (Galloway et al. 2015). Since 1970, NOx emissions in the US have decreased 2-fold, largely due to the US Clean Air Act and the development of pollution control technologies for point sources of NOx. The US Clean Water Act has also targeted point sources of water pollution, such as wastewater effluent to waterways. Federal guidelines limit the nitrate concentrations in drinking water through the National Primary Drinking Water Regulations (EPA 2018a). Other efforts have attempted to limit the N delivered to bodies of water, such as the Total Maximum Daily Load policies established by the EPA in the Chesapeake Bay region for pollution control and drinking water quality (EPA 2018b). However, regulations have not yet been very successful at managing nonpoint N pollution sources like agriculture (Galloway et al. 2015). To fully address the N challenge, new policy solutions will likely be necessary to both reduce reactive N losses and to manage the damage already caused by reactive N.

3. Education and outreach on the nitrogen story

Consumers are a key stakeholder in N management. Consumer choices drive the types of food and energy produced, and they can drive policy decisions. Historically, global consumer demand has shifted us towards N-intensive diets that are both increasing in total per capita food consumption and in the proportion of the diet made up of meat and animal products (Lassaletta et al. 2016; Godfray et al. 2018). With collective action, consumers could reduce N pollution by choosing renewable energy sources and more sustainable food. However, communicating the importance and consequences of N use has been challenging due to the complexity of the N cycle.

N footprint tools are a novel strategy for communicating the N story to consumers. N footprint tools connect our consumption choices with the associated N pollution (Leach et al. 2012; Galloway et al. 2014). N footprint tools have been developed at the consumer level for 7 countries (Galloway et al. 2014); at the campus level (Leach et al. 2013, 2017); at the watershed level for the Chesapeake Bay (CBF 2018); and at the urban level for the city of Baltimore, Maryland (Milo 2018). To help inform consumer food choices, environmental impact food labels combining the carbon, nitrogen and water footprint were developed (Leach et al. 2016) and have been used in social psychology studies to understand how environmental food labels impact consumer decisions (Piester et al. in preparation).

The campus-level N footprint tool has been particularly successful at both communicating the N story and encouraging real change with N footprint reduction goals (Castner et al. 2017). Originally developed in 2009 at the University of Virginia (Leach 2009), the campus N footprint tool was used to establish the first-ever N reduction goal at the University of Virginia in 2013. After pilot testing by a network of over a dozen colleges and universities in the US and abroad (Castner et al. 2017), the campus N footprint was integrated with the campus carbon footprint (Cleaves et al. 2009) at the University of New Hampshire Sustainability Institute. This integrated tool is called SIMAP (Sustainability Indicator Management and Analysis Platform; www.unhsimap.org). Since its launch in November 2017, this web-based tool is now used by over 500 colleges and universities as their official footprint tracking method (personal communication, Yulia Rothenberg, October 2018). In **Chapter 1** of this dissertation, we present the motivation and methods behind the integration of the campus carbon and nitrogen footprint tools into SIMAP. This chapter also explores the impact of food and energy management strategies on both the carbon and nitrogen footprints.

4. Technology solutions for nitrogen management

Technological solutions can more efficiently convert reactive N sources into products. The rate at which food N inputs are converted into food N products is called the N use efficiency (de Klein et al. 2017; Erisman et al. 2018). Improved N management practices and new technologies have led to increased N use efficiencies during food production in many countries (e.g., Netherlands, France, Germany; Lassaletta et al. 2014; Zhang et al. 2015). However, excess fertilizer application has also decreased many countries’ food N efficiency over the last several decades (e.g., China, India, Brazil). This disparity underscores the challenge of managing N in food production: Higher rates of N application improve yields and food security, but it also leads to higher N losses and consequences to environmental and human health. As countries experience economic growth, they tend to apply more N fertilizer to increase food production, which reduces their N use efficiency. Then at a certain point, they ‘turn the corner’ and begin improving their N use efficiency. Zhang et al. (2015) refer to this as the N use efficiency curve, which is analogous to the environmental Kuznets curve (i.e., with economic growth, countries experience increased and then decreased N pollution; Grossman et al. 1995). Zhang et al. propose that we use lessons learned and new technologies to ‘tunnel through’ the N use efficiency curve to avoid the lower levels of N use efficiency as countries develop.

Projections suggest that we will need to grow 25% to 70% more food to meet crop demands in 2050 (Hunter et al. 2017). With growing demand for N-intensive meat, especially in developing countries, the reactive N losses will only increase. Novel solutions are necessary to further understand, measure, and improve farm N flows and efficiency. To help 'tunnel through' the N use efficiency curve in growing countries, it is especially important to develop effective, affordable, and transferrable solutions for improved N management.

Composting is an old technology that has been receiving more attention in recent years as a strategy for managing farm waste streams while also recycling nutrients back to the fields (Smith & Aber 2018). Composting is the breakdown of organic materials into a stable soil amendment through aerobic microbial activity (Rynk et al. 1992; Haug et al. 1993). Historically, composting methods have been on a smaller scale, such as windrow piles that must be turned periodically (Misra 2003). However, with growing agricultural by-product waste streams and increasing restrictions on processing those wastes, there has been new interest in industrial-scale composting methods (Haaren et al. 2010). Anaerobic digesters have grown in popularity as an industrial-scale waste treatment method that also obtain value by capturing and using methane as a fuel source (Moser et al. 1998). A novel composting technology that simultaneously processes waste streams and captures the heat generated by microbial activity during decomposition is aerated static pile (ASP) heat recovery composting (Smith & Aber 2014, 2018). This waste management method can be implemented at a wide range of scales, processes agricultural by-products into a stable soil amendment, and captures heat for on-farm energy needs.

In this dissertation, I explore the potential for composting as an N management strategy at an organic dairy farm in New Hampshire. The University of New Hampshire Organic Dairy Research Farm is home to the only commercial-scale ASP heat recovery compost facility for research, which provides tremendous opportunity for obtaining valuable knowledge for practitioners. Constructed in 2011, the Joshua Nelson Energy Recovery Compost Facility processes agricultural by-products (e.g., manure, spent bedding, waste baleage) from the Organic Dairy Research Farm. Studies to date have explored the design and construction of a commercial-scale ASP heat recovery compost facility (Smith & Aber 2014; Smith 2016) and its the energy capture potential (Smith et al. 2017).

The potential for ASP heat recovery composting as an N management strategy is addressed in two chapters of this dissertation. First, the effect of an ASP heat recovery compost facility on the farm's overall N budget was assessed (**Chapter 2**). This chapter quantifies the N budget and N flows on the UNH Organic Dairy Research Farm, and it then considers how the farm's N efficiency would be affected by exporting compost. Then, gas fluxes (NH_3, CO_2, CH_4) from a commercial-scale ASP heat recovery compost facility are characterized (**Chapter 3**). This chapter also presents a method for predicting gas fluxes using an indicator variable and identifies ASP heat recovery composting management strategies for reduced gas fluxes.

5. Synthesis and conclusions

Addressing the complex N challenge will require an array of solutions. Because N is released through so many different pathways and at different stages of supply chains, the solutions must involve a range of strategies and stakeholders. The three chapters in this dissertation present and discuss N management solutions that inform consumer decisions through N footprint tools and manage agricultural waste streams with composting.

Consumers are key stakeholders in addressing the N challenge because their food and energy choices drive N pollution. In the first chapter, we present an integrated carbon and nitrogen footprint tool for campuses to calculate, track, and manage their footprints together. The two footprints compare strongly in most sectors, and scenario analysis indicates benefits to both footprints from a range of reduction strategies. The calculation of campus carbon and nitrogen footprints can both help campuses improve their own sustainability and educate a community of students who can bring their sustainability knowledge to their future careers.

Food production is a significant source of N pollution, and new and improved farm N management practices are necessary to reduce N losses. In the second and third chapters of this dissertation, aerated static pile (ASP) heat recovery composting is considered as an N management strategy. To assess its potential, the N budget of an organic dairy farm was first assessed, where it was found that organic practices led to the cycling of substantially more N on the farm property than was imported or exported. Some of the potential farm N loss pathways were characterized, including gas emissions from the compost facility (chapter 3), but future research should characterize other N loss pathways to assess the balance between storage and environmental loss. Scenario analysis found that exporting finished compost was a viable strategy for improving the farm's N use efficiency as long as enough N is retained on-site to support crop production.

In summary, this dissertation presents N management solutions for educating consumers and improving on-farm N efficiency. These two strategies can contribute to the body of research and action that strives to reduce N pollution and improve N sustainability.

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CHAPTER 1: AN INTEGRATED TOOL FOR CALCULATING AND REDUCING INSTITUTION CARBON AND NITROGEN FOOTPRINTS

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Abstract

The development of nitrogen footprint tools has allowed a range of entities to calculate and reduce their contribution to nitrogen pollution, but these tools represent just one aspect of environmental pollution. For example, institutions have been calculating their carbon footprints to track and manage their greenhouse gas emissions for over a decade. Here we present an integrated tool that institutions can use to calculate, track, and manage their nitrogen and carbon footprints together. This paper will present (1) the methodology for the combined tool, (2) a comparison of institution nitrogen and carbon footprint results by several metrics, and (3) management strategies that reduce both the nitrogen and carbon footprints. The data requirements for the two tools overlap substantially, although integrating the two tools does necessitate the calculation of the carbon footprint of food. Comparison results for five institutions suggest that the institution nitrogen and carbon footprints correlate strongly, especially in the utilities and food sectors. Scenario analyses indicate benefits to both footprints from a range of utilities and food footprint reduction strategies. Integrating these two footprints

into a single tool will account for a broader range of environmental impacts, reduce data entry and analysis, and promote integrated management of institutional sustainability.

1. Introduction

Institutions of higher education provide an ideal setting to measure, analyze, and improve sustainability performance. They have the potential to make significant improvements to their sustainability given the span and impact of their overall activities and their ability to make management decisions both from the top-down (e.g., by the administration) and bottom-up (e.g., through student initiatives). Institutions of higher learning can also be used as a ‘learning laboratory’ to both test sustainability strategies and to educate large populations of students about the importance of managing and reducing their environmental impact.

The interest and potential for institutions of higher education to lead in sustainability initiatives has been demonstrated by the success of the Campus Carbon Calculator^{TM}, a carbon footprint tool for institutions to track and manage their carbon footprint (Cleaves et al. 2009). More than 90% of the colleges and universities that report their carbon footprint for the Second Nature Carbon Commitment (formerly known as the American College & University Presidents Climate Commitment) use the Campus Carbon Calculator^{TM}.

1.1. Integrating environmental footprints

Multiple footprints have been established to calculate a consumer’s contribution to environmental pollution, such as the ecological footprint (Rees 1992), carbon footprint (Pandey et al. 2011), water footprint (Hoekstra & Mekonnen 2012), and nitrogen footprint (Leach et al. 2012). The many footprints can be confusing to consumers, which has prompted a new interest in tools that combine footprints (Galli et al. 2012; Leach et al. 2016).

The only environmental footprint tool currently available to institutions is the carbon footprint. An institution-level nitrogen footprint tool has been developed, piloted, and tested by participants in a project of the Nitrogen Footprint Tool Network (Leach et al. 2013; Castner et al. 2017a), but adding a second separate footprint tracking tool would be cumbersome for institutions and would not capture any potential synergies and trade-offs between footprint management strategies. Therefore, the overarching goal of this paper is to present a new integrated carbon and nitrogen footprint tool for institution-level sustainability management. These footprints were selected because they represent two important areas of environmental concern and they are the two environmental footprints for which institution-level footprint tools are already available.

1.2. Nitrogen footprint

A nitrogen (N) footprint is a measurement of the amount of reactive N (all species of N except N_2) released to the environment as a result of an entity's resource consumption (e.g., food, utilities, transit) (Leach et al. 2012). Although it is necessary for food production and to support life, excess reactive N can cause a cascade of detrimental impacts to ecosystem and human health (Galloway et al. 2003; Erisman et al. 2013). The N footprint aims to reduce the loss of reactive N through both education and the elucidation of possible management scenarios for reducing reactive N losses.

The N footprint methodology was first developed at the consumer level for the United States and the Netherlands (Leach et al. 2012). The tool has since been applied in the United Kingdom, Germany, Austria, Japan, Australia, and Tanzania and is in development for Denmark, China, and Taiwan (Galloway et al. 2014). A nitrogen footprint tool was then developed for a different type of entity: an institution (Leach et al. 2013). First developed and applied at the

University of Virginia, the tool accounts for N losses associated with food purchases, utilities usage, transport, fertilizer application, research animals, and agricultural activities. The N footprint includes the different forms reactive N released from institution activities (e.g., NOx, N_2O, total N), which are converted to and reported as the total weight of N.

1.3. Carbon footprint

The carbon (C) footprint is based on the total greenhouse gas emissions associated with a product, service, or other entity (Pandey et al. 2011; Röös et al. 2013). The C footprint typically includes the six major greenhouse gases: carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), hydrofluorocarbons (HFCs), perfluorocarbons (PFCs), and sulfur hexafluoride (SF_6) (Galli et al. 2012). These greenhouse gases are reported together based on their global warming potential in units of CO_2-equivalents. Major sectors that emit greenhouse gases (GHG) include fossil fuel combustion, land conversion, livestock production, and crop production (Hertwich et al. 2013).

The Campus Carbon Calculator^{TM} is an institution-level C footprint tool that has been used by thousands of institutions worldwide and is the standard tool for managing institution GHG emissions in the United States (Cleaves et al. 2009). The tool was originally developed in 2001 in partnership between the University of New Hampshire Sustainability Institute (UNHSI) and the private non-profit Clean Air Cool Planet (CACP). Following established best practices for carbon accounting, the C footprint is reported in three categories of "scopes," which reflect how institutional decisions are capable of directly influencing C emissions (see **Table 1** for more information on scopes) (WRI 2004).

1.4. Objectives

Here we present a newly developed integrated tool that allows institutions to track and manage their C and N footprints together. To be released in 2017, this tool combines the institution-level Nitrogen Footprint Tool (Leach et al. 2013) and the University of New Hampshire Sustainability Institute (UNHSI) Carbon Management and Analysis Platform (CMAP), an online platform that uses the Campus Carbon Calculator™ methodology (www.campuscarbon.com). Combining these two tools expands the ability of institutions to account for a wider range of environmental impacts.

The objectives of this paper are:

1. Present the integrated nitrogen and carbon footprint tool for institutions;
2. Compare the nitrogen and carbon footprints of institutions by several metrics; and
3. Identify reduction strategies that will reduce both the nitrogen and carbon footprints.

2. Methods

2.1. Integrating the nitrogen and carbon footprints

Combining the distinct institution-level nitrogen and carbon footprint tools requires four phases: 1) comparing data requirements and addressing gaps; 2) integrating the calculations; 3) identifying how to report the results; and 4) incorporating projections and management scenarios. The first three will be complete when the first version of the integrated tool is launched in 2017, and projections and scenarios will be incorporated in a future version of the tool.

2.1.1. Comparing data requirements

The sectors included in the carbon and nitrogen footprint calculations were compared, and any differences in the sectors included were identified. For example, the sector "refrigerants" is part of the carbon footprint but not the nitrogen footprint. All sectors in each stand-alone footprint tool are included in the integrated tool. For any sector that was in one tool but not the other, a review was conducted to determine if that sector should be added to the other footprint tool. For example, refrigerants have a negligible nitrogen footprint.

2.1.2. Integrating the calculations

The methods and equations for the two footprints were aligned for consistency and comparability in the integrated tool. The calculations were aligned by first ensuring that the data input describing resource consumption (e.g., the amount of fuel consumed) was the same for the two footprints. Any conversions necessary to calculate the total resource consumption (e.g., assumptions about commuting) were also kept consistent. For most sectors, the only difference in the two footprint calculations is the emissions factors used (e.g., for utilities, transportation). However, the calculations for the carbon and nitrogen footprint do diverge for food consumption and food production because of the different pathways through which greenhouse gases and nitrogen pollution are released from these sectors. Equations for calculating the carbon and nitrogen footprints for on-site stationary combustion, public transit, purchased electricity, food production, and food consumption/wastewater are given in **Appendix A**. Complete documentation for the C and N footprints can be found in each tool's user's guide (Leach et al. 2013, 2016b; CACP 2016).

The food sector will be added to the C footprint using the N footprint methods for estimating the weight of food purchases (Leach et al. 2013, 2016b). Briefly, the food weights are calculated using purchase records for an entire year or for a subset of the year or locations and then scaled. Food weights can be scaled based on the percent of purchases or percent of weight represented in the subset of data. Each food product is placed in a food category based on up to three ingredients, and the weight is distributed evenly across those ingredients. Guidance for assigning food categories is provided in the Nitrogen Footprint User's Manual (Leach et al. 2016b). The C and N footprint calculations differ for both food production and food consumption. For food production, the C footprint is calculated by multiplying a weight of food by a greenhouse gas emissions factor (Heller & Keoleian 2014), whereas the N footprint has several components that are summed: virtual N (calculated by multiplying the weight of food N by a virtual N factor; Leach et al. 2012), wasted food N, and transport emissions (See **Table A1** in **Appendix A**). For food consumption/wastewater, the C footprint is calculated by multiplying the volume of wastewater processed by a greenhouse gas emissions factor for a given wastewater treatment system, whereas the N footprint calculation multiplies the amount of N consumed (which ultimately enters the sewage stream) by one minus the N removal rate at the local wastewater treatment facility. See **Appendix A, Equations 5-8**, for more information about the food calculations.

2.1.3. Identifying how to report the results

Because the two footprints mostly represent different environmental impacts, the footprints will be reported separately as the C footprint (units of metric tons CO_2-eq) and the N footprint (units of metric tons of N). It should be noted that there is one area of overlap: nitrous oxide (N_2O) is both a greenhouse gas and is part of the N footprint. However, N_2O will be included in both footprints because of its contribution to the N cascade (e.g., global warming; stratospheric ozone depletion; Galloway et al. 2003) and because the two footprints are not additive and will be presented separately. The geographic scale for the two footprints also differs. Greenhouse gas emissions are well mixed and contribute to global climate effects regardless of where they are emitted. Nitrogen losses can have local to global effects, depending on the type of N released.

The C and N footprints will each be reported on a total basis, on a per capita basis, by sector, and by scope. The results will be reported as the total C footprint and total N footprint. The per capita C footprint and per capita N footprint will be reported to normalize to each institution's population. The per capita footprints are calculated using full-time equivalents (FTE), which consider how often different populations (e.g., part-time students, full-time students, faculty, staff) are at the institution. The footprints will also be presented by sector (food consumption/wastewater, food production, utilities, transport, and research & agriculture; see more information in **Table A2** in **Appendix A**) and by scope (scope 1, 2, 3). Scope 1 includes on-site stationary combustion, fleet vehicles, and research animals; scope 2 is purchased electricity; and scope 3 includes commuting, air travel, food production, wastewater, and feed for research animals (WRI 2004). In the integrated online tool, additional comparison and normalization metrics (e.g., per gross square footage) will also be available.

2.2. Comparing preliminary footprint results

Although the Campus Carbon Calculator™ has been used by thousands of institutions, the nitrogen footprint tool has been pilot tested by ~20 institutions. Results for the integrated C and N footprints are presented here for the following five institutions as a case study: Eastern Mennonite University, Dickinson College, University of New Hampshire, Colorado State University, and University of Virginia. Nitrogen footprint results were obtained from Castner et al. (2017a), and C footprint results were obtained directly from each institution. Additional offsets (e.g., purchased Renewable Energy Credits) and non-additional offsets (e.g., sold Renewable Energy Credits) were not included in this comparison so that the sources and emissions for the C and N footprints could be directly compared. The calculation year is fiscal year 2014.

The total footprints were compared by sector and by scope. The footprints were also compared on a per capita basis for the total footprint, on a per capita basis by sector (utilities), and the footprint per kg of food purchased (food). Additional comparison metrics for the N footprint are explored in Castner et al. (2017b). Linear regressions between the C and N footprints will be used to show how the two footprints relate at the institution scale, and p-values will be presented to determine if correlations are significant.

2.3. Identifying integrated management strategies

The effect of management strategies on the C and N footprints was explored for the five institutions presented as case studies in this paper. The management strategies analyzed were energy scenarios (purchase 25% renewable energy, improve energy efficiency by 10%, and replace all purchased electricity with renewables) and food scenarios (replace 25% of beef purchases with chicken, replace 25% of meat protein with vegetable protein, and reduce food waste by 25%). These scenarios do not include projections of changes in population because they aim to show the direct effect of specific changes in practices. However, when institutions are setting C and N footprint reduction goals, projections must be included.

3. Results and discussion

3.1. Integrating the nitrogen and carbon footprints

A review of the data inputs required for the existing C and N footprints identified substantial overlap in the utilities and transport sectors (**Table 1**; See **Table A3** in **Appendix A** for a complete list of data inputs). In these sectors, the C footprint incorporates more options (e.g., more fuel types), and the N footprint is being expanded to fill in these gaps. The C footprint does not currently include a major sector of the nitrogen footprint: food. As part of this integration, the C footprint of food will be incorporated into the combined C and N footprint tool.

Table 1. Comparison of the C and N footprint data requirements. Comparisons are organized by scope for the Campus Carbon Calculator™ and Nitrogen Footprint Tool. See **Table A3** in **Appendix A** for a more detailed comparison.

Scope	Data category	Carbon footprint	Nitrogen footprint
Scope 1	On-campus stationary sources	Yes	Yes[a]
	Direct transportation sources	Yes	Yes[a]
	Refrigerants & chemicals	Yes	No
	Agriculture sources	Yes[b]	Yes[b]
Scope 2	Electricity, steam, chilled water	Yes	Yes
Scope 3	Commuting	Yes	Yes[a]
	Directly financed outsourced travel; Study abroad; Student travel to/from home	Yes	Yes
	Solid waste	Yes	To be added
	Wastewater	Yes	Yes
	Paper	Yes	Under review
	Food purchases	To be added	Yes
Offsets	Offsets with additionality	Yes	Yes[c]
	Non-Additional Renewable Energy Certificates (RECs)	Yes	Under review

[a] Additional fuel types will be added for the nitrogen footprint
[b] Animal types will be added for the C footprint (research animals) and N footprint (research farms)
[c] Additional offsets may be added for the nitrogen footprint.

3.2. Comparing preliminary footprint results

The size of the total C and N footprints, which range from 6,560 to 337,000 metric tons CO_2-eq and 11 to 444 metric tons N per year, are likely driven by the institutions' populations (**Figure 1A, 1C**). However, when footprints are compared on a per capita basis, the effects of different practices begin to emerge (**Figure 1B, 1D**). Across the C and N footprints, the two largest sectors are food and utilities. Food production makes up the largest proportion of total institution N footprints (34-78%) while it makes up a smaller proportion of total C footprints (6-17%). On the other hand, utilities are the largest contributor to the total C footprint (41-83%) and typically a smaller contributor to the total N footprint (8-52%).

The food production C and N footprints are driven by the types and amounts of food purchased by an institution. For example, Dickinson College has larger food footprints because nearly all students eat most meals on campus and the campus hosts summer programs that include meals in its dining services, which is not the case for the other universities in the comparison. The utilities footprints differ across institutions based on the total energy consumption and the types of fuel used. For example, the University of New Hampshire has small utilities C and N footprints because its energy is derived from an on-campus cogeneration facility that uses processed methane generated at the local landfill. The University of Virginia has a larger utilities footprint because its campus includes a hospital and because most of its electricity is purchased and the electricity fuel mix has a high percentage of coal (Leach et al. 2013).

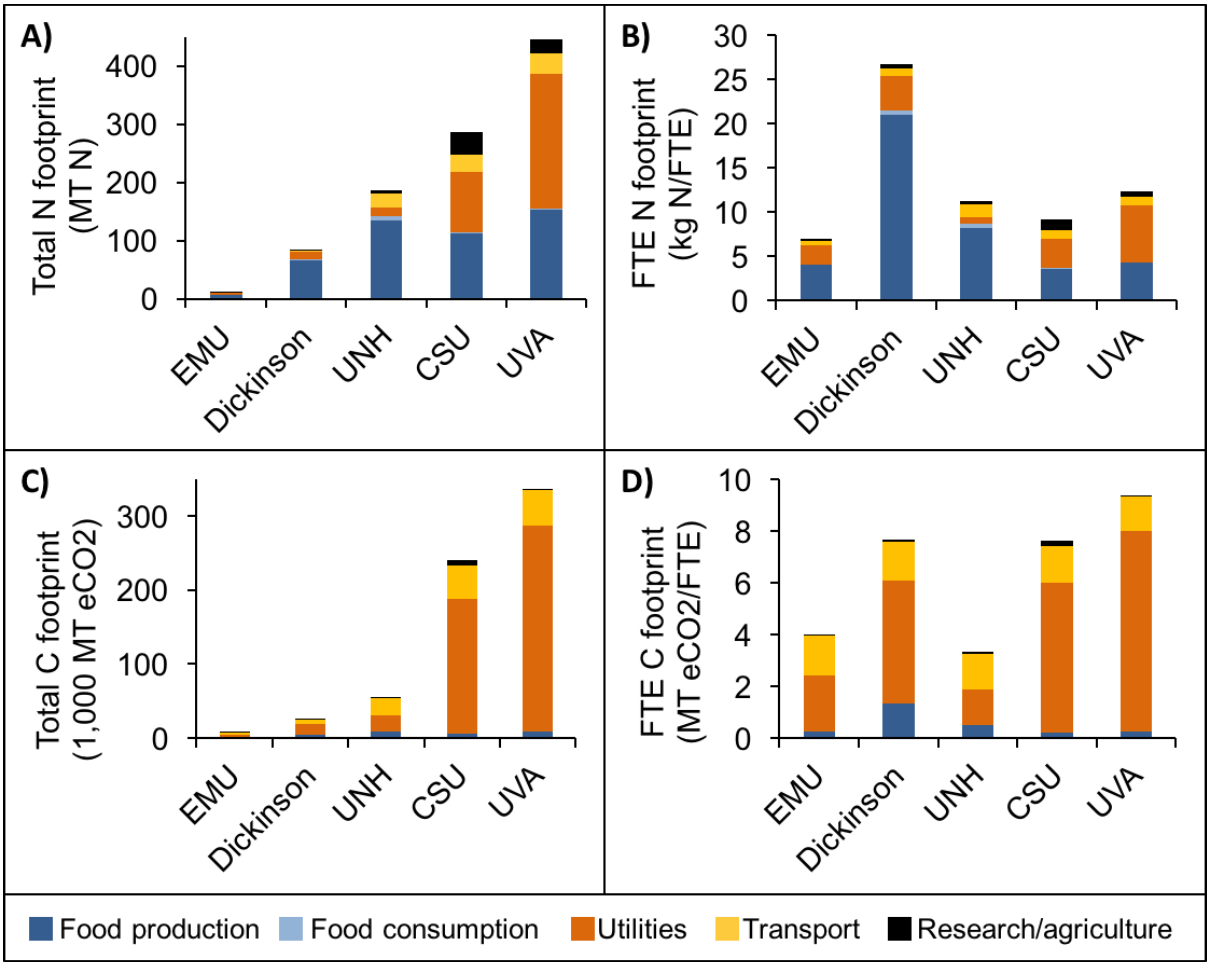

Figure 1. Institution nitrogen footprints by sector. Footprints shown as A) the total institution nitrogen footprint, B) the nitrogen footprint per full-time equivalent population (FTE), C) the total institution carbon footprint, and D) the carbon footprint per FTE. Footprints are shown for Eastern Mennonite University (EMU, population 1,648), Dickinson college (population 3,174), University of New Hampshire (UNH, population 16,548), Colorado State University (CSU, population 31,409), and University of Virginia (UVA, population 35,894).

C and N footprint results can also be presented by scopes, which describe how directly emissions are related to institution activities (scope 1 is the most direct; scope 3 is the least; **Figure 2**). Both scope 1 and 2 contribute a large proportion of the C footprint, whereas the largest scope for the total N footprint is typically scope 3 (43-88%). This means that most C emissions occur closer to the institution, while most N losses occur elsewhere. Greenhouse gas emissions contribute to the global greenhouse effect regardless of where they are emitted. Conversely, N losses have more local pollution effects for most forms of nitrogen, such as local water quality and air quality effects. Given this, institutions may consider implementing two N footprint reduction goals: a goal for scope 1 (with a focus on local N pollution) and a goal for the overall N footprint. Many of the benefits from an overall N reduction goal could occur in ecosystems far removed from the institution itself, but those environmental impacts are still the responsibility of the institution.

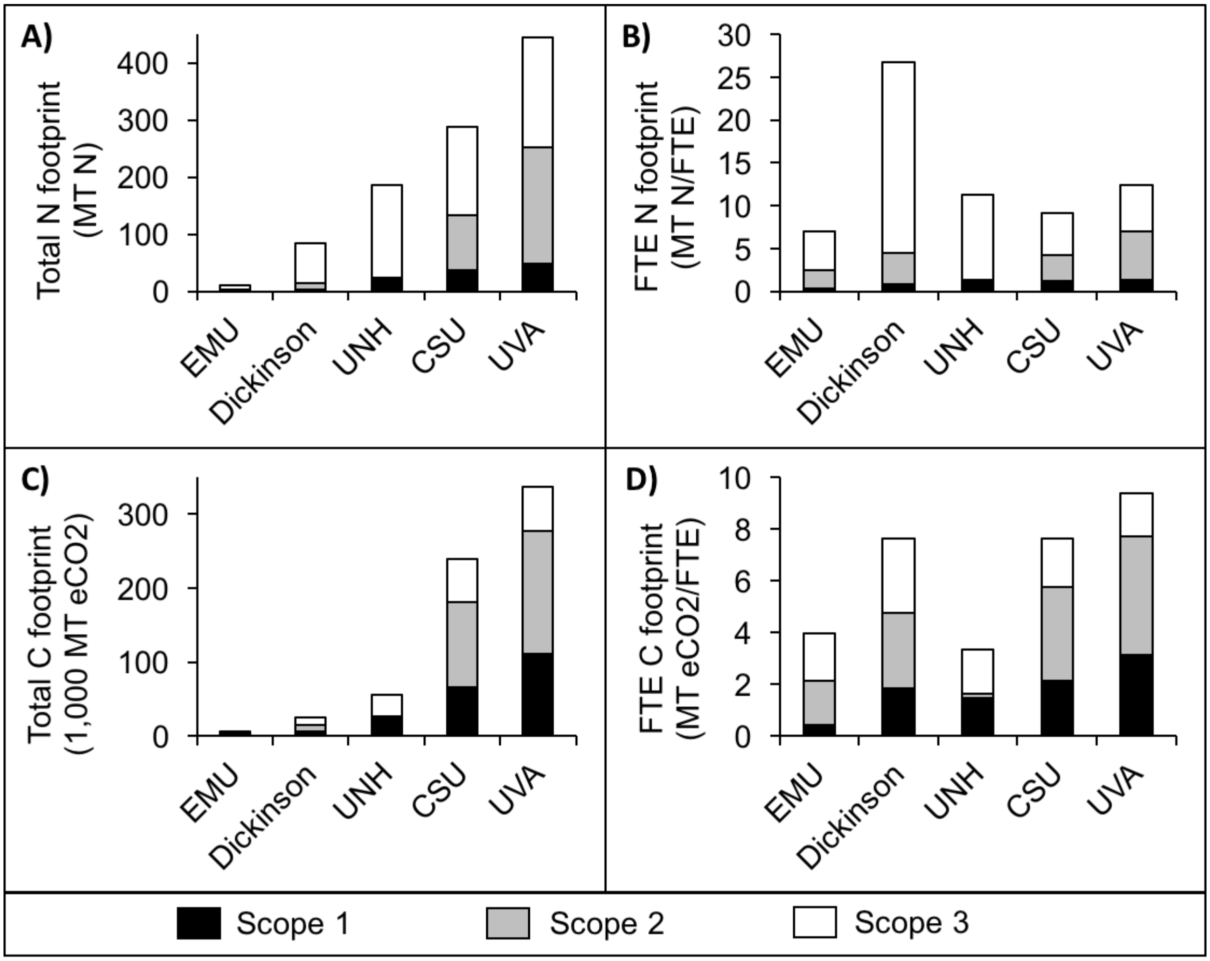

Figure 2. Institution nitrogen footprints by scope. Footprints shown as A) the total institution nitrogen footprint, B) the nitrogen footprint per full-time equivalent population (FTE), C) the total institution carbon footprint, and D) the carbon footprint per FTE. Footprints are shown for Eastern Mennonite University (EMU, population 1,648), Dickinson college (population 3,174), University of New Hampshire (UNH, population 16,548), Colorado State University (CSU, population 31,409), and University of Virginia (UVA, population 35,894).

The five institutions' C and N footprint results were compared (**Figure 3**). The total C and N footprints correlate strongly (R^2=0.92, p-value = 0.009; **Figure 3A**), which suggests they may have similar drivers. Regressions comparing each of the total footprints to gross square footage for each campus found a significant correlation (R^2>0.95, p-value<0.005), suggesting that institution size is a driving factor for the total C and N footprints (regressions not shown). However, the comparison of per capita C and N footprints was not significant (R^2=0.14, p-value = 0.5; **Figure 3B**), likely due to differences in sector-specific institution activities. For example, Dickinson has a large food N footprint because 94% of students have meal plans and a moderate per capita C footprint. On the other hand, UVA has a large C footprint due to its research facilities and fuel mix and a moderate N footprint. Due to the differences in institution activities, the footprints should be explored on a sector-specific basis.

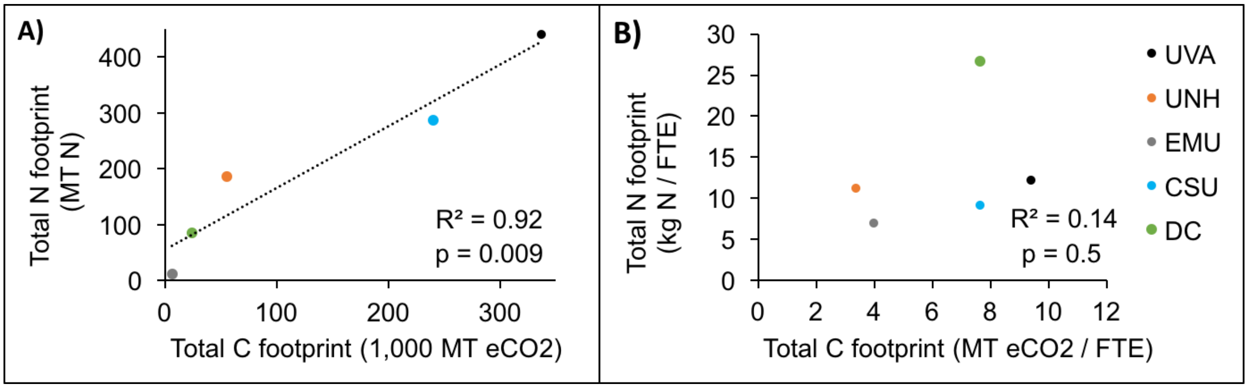

Figure 3. Linear regressions of institution C and N total footprints. A comparison of institution carbon (C) and nitrogen (N) footprints in terms of A) the total C and N footprint and B) the total C and N footprint by full-time equivalent population (FTE). Footprints are shown for the University of Virginia (UVA), University of New Hampshire (UNH), Eastern Mennonite University (EMU), Colorado State University (CSU), and Dickinson College (DC).

A linear regression between the per capita utilities C and N footprints found a significant correlation (R^2=0.89, p-value = 0.02), which is likely because of the similar relative magnitude of C and N footprints for different fuel types (**Figure 4A**). The linear regression for the C and N food footprints per-kg food was also significant, which reflects the consistency in the relative impacts of different food products for the C and N footprints (R^2=0.95, p-value = 0.005; **Figure 4B**; Leach et al. 2012; Heller & Keoleian 2014). The C and N footprints for other sectors (e.g., transportation) and normalizations (e.g., per gross square foot) did not exhibit correlations.

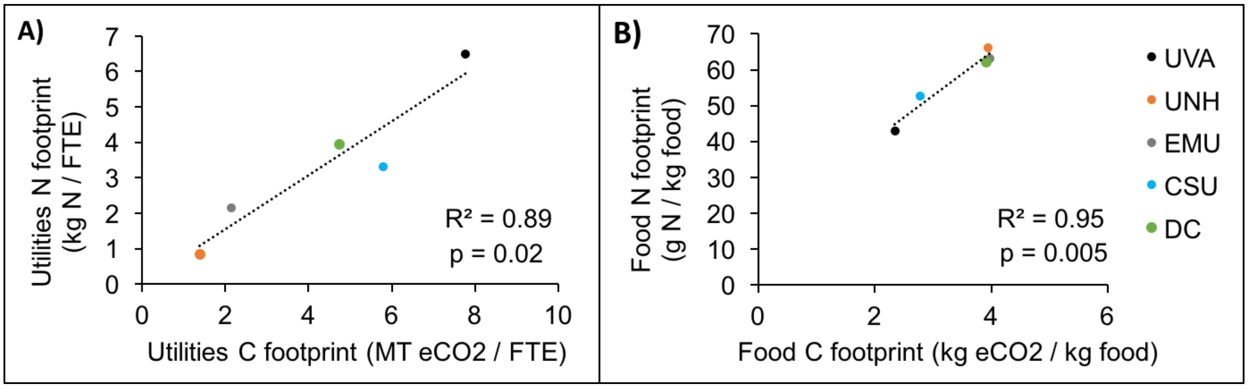

Figure 4. Sector-specific linear regression of institution C and N footprints. Comparisons are in terms of A) the utilities N and C footprint by full-time equivalent population (FTE) and B) the food production N and C footprint per kg of food purchased. Footprints are shown for the University of Virginia (UVA), University of New Hampshire (UNH), Eastern Mennonite University (EMU), Colorado State University (CSU), and Dickinson College (DC).

3.3. Identifying integrated management strategies

The effects of a variety of food and energy management strategies were reviewed for five institutions (**Table 2**). Of the food scenarios analyzed, the most impactful was replacing 25% of meat purchases with vegetable purchases. Within the food sector, this scenario resulted in a reduction of 16-21% for the food C footprint and 7-18% for the food N footprint. However, when presented in the context of the total footprint, the reductions were just 0.4-4% for C and 3-14% for N. Generally, the food scenarios had a smaller impact on the total C footprint than the N footprint because food makes up a smaller percentage of the overall C footprint. The utilities management strategies had a larger impact on both footprints. Replacing all purchased electricity with a renewable energy source has the potential for substantial reductions: 5-49% for the total C footprint and 0.2-46% for the total N footprint. However, the size of the potential reduction is determined by the percent of total electricity usage that is from purchased electricity versus on-campus stationary combustion sources.

Table 2. The range of reductions from food and utilities scenarios. Reduction ranges include results from five campus carbon and nitrogen footprints.

Scenario		Carbon footprint reduction[a,b]		Nitrogen footprint reduction[a,b]	
		Within sector	*For total footprint*	*Within sector*	*For total footprint*
Food	Replace 25% of beef purchases with chicken	5-9%	0.1-2%	2-5%	1-3%
	Replace 25% of meat protein with vegetable protein	16-21%	0.4-4%	7-18%	3-14%
	Reduce food waste by 25%	4-5%	0.1-1%	4-5%	1-3%
Utilities	Purchase 25% renewable energy	3-21%	1-12%	1-35%	0.04-15%
	Improve energy efficiency by 10%	3-9%	1-5%	1-10%	0.04-5%
	Replace all purchased electricity with renewables	11-85%	5-49%	2-99%	0.2-46%

[a] Results are given both within the sector of interest (food, utilities) and for the total footprint.

[b] The results presented are an average for the University of Virginia, University of New Hampshire, Eastern Mennonite University, Colorado State University, and Dickinson College.

All scenarios analyzed found reductions for both the C and N footprint, and other studies assessing the effects of campus sustainability initiatives on both footprints have had similar findings (Barnes et al. 2017). Energy scenarios were more effective for reducing the total C footprint, whereas the most effective strategies for the N footprint vary by institution. The energy scenarios are successful because the entire utilities footprint can be offset with renewable energy, which has a minimal C and N footprint (Schlömer et al. 2014). The same cannot be accomplished for food purchases because all methods of food production for all types of food release both greenhouse gases and nitrogen pollution. As a result, achieving nitrogen footprint neutrality is difficult without additional offsets, such as the purchase of Renewable Energy Credits (Leip et al. 2014). Despite this, important reductions in the food footprints can and should still be achieved by shifting towards less impactful sources of protein (e.g., chicken, vegetable protein), choosing foods from more sustainable farms, and reducing food waste.

4. Next steps and summary

The integrated carbon and nitrogen footprint tool will be publicly launched in 2017. A subsequent version of the online tool will include the ability to analyze projections and scenarios and perhaps even include other footprints, such as phosphorus or water. Offsets for N footprints will be explored more, especially since N footprint neutrality is not possible without offsets. Other ways of presenting the footprints will also be explored, such as linking the footprints to social and economic costs (Compton et al. 2017).

Here we present an integrated tool that institutions can use to calculate, track, and manage both their nitrogen and carbon footprints together. The data requirements for the two tools overlap substantially, although integrating the two tools will add a calculation of the carbon footprint of food. Institution nitrogen and carbon footprints compare strongly in most sectors, and scenario analysis indicates benefits to both footprints from a range of reduction strategies. Integrating these two footprints into a single tool will account for a broader range of environmental impacts, reduce data entry and analysis, and promote integrated management of institutional sustainability.

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Appendix A: Data and documentation for the integrated carbon and nitrogen footprint tool

Documentation for the C and N footprints

A complete listing and explanation of C and N footprint methodologies and equations are available in the current documentation for each of the tools. An updated user's manual will be released with the new integrated tool.

The carbon footprint tool is available as a web-based tool (www.campuscarbon.com) and through Microsoft Excel (http://sustainableunh.unh.edu/calculator). A user's guide is available for both the web-based CarbonMAP and the Excel-based Campus Carbon Calculator (CACP 2016).

The nitrogen footprint tool is available for download through the US Environmental Protection Agency (Leach et al. 2016). The user's manual, titled 'How to calculate your institution's nitrogen footprint,' provides complete documentation and equations for the stand-alone nitrogen footprint tool.

Methods

Example equations are given to demonstrate how the carbon footprint and nitrogen footprint calculations compare. The sectors shown are on-site stationary combustion and transit (scope 1; **Equation 1, 2**), purchased electricity (scope 2; **Equation 3, 4**), and food production and wastewater (scope 3; **Equation 5, 6, 7**). The equations for the C and N footprints are very similar for scopes 1 and 2 but diverge for scope 3. The equations presented below would use emissions factors for specific gases and are the first step for calculating each footprint. For the C footprint, these emissions include carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). For the N footprint, these losses include nitrogen oxides (NOx), nitrous oxide (N_2O), and units of N for food production N losses. All of these emissions must then be converted to a normalized unit. The C footprint is normalized to CO_2-equivalents using the global warming potential of the different greenhouse gases. The N footprint is normalized to units of N based on atomic weight. The final footprints are reported in these normalized units.

Scope 1: On-site utilities and transit

Eq. 1. $N_i = F_i * EF_{ni}$

Eq. 2. $C_i = F_i * EF_{ci}$

> where N_i is the nitrogen footprint result for a given fuel i, C_i is the carbon footprint result for fuel i, F_i is the fuel consumption for fuel i, and EF_{ni} and EF_{ci} are the appropriate nitrogen and carbon emissions factor for that fuel, respectively.

Scope 2: Purchased electricity

Eq. 3. $N_e = E * EF_{ne}$

Eq. 4. $C_e = E * EF_{ce}$

> where N_e is the nitrogen footprint result for purchased electricity, C_e is the carbon footprint result for purchased electricity, E is the electricity consumption, and EF_{ne} and EF_{ce} are the appropriate nitrogen and carbon emissions factors, respectively, for electricity in the institution's region.

Scope 3: Food production

Eq. 5. $N_f = S_f * V_f + W_f + T_f * M_f * EF_t$

> where N_f is the nitrogen footprint result for food production for food category f, S_f is the food N supply for food category f, V_f is the virtual N factor for food category f, W_f is the wasted food N for food category f, T_f is the number of trips required to transport a given weight of food f (calculated as the total food weight divided by a truck's cargo capacity), M_f is the average food miles for food f, and EF_t is the nitrogen emissions factor for transporting food in a truck.

Eq. 6. $C_f = \sum(S_{fw} * EF_f)$

> where C_f is the total carbon footprint result for food production, S_{fw} is the food supply by weight for food category f, and EF_f is the greenhouse gas emissions factor for food category f.

Scope 3: Food consumption / wastewater

Eq. 7. $N_w = S * (1 - R)$

where N_w is the nitrogen footprint result for wastewater, S is the total food N supply, and R is the nitrogen removal and reduction credit at the local sewage treatment facility.

Eq. 8. $C_w = W_w * EF_{ww}$

where C_w is the total carbon footprint result for wastewater, W_w is the volume of wastewater generated by the institution, and EF_{ww} is the greenhouse gas emissions factor for a given wastewater treatment method.

Table A1. Factors used to calculate the C and N footprints of food. Carbon footprint factors were collected from Heller & Keoleian et al. 2014, and N footprint factors were collected from Castner et al. 2017.

Food category	C footprint factor	N footprint factor
	kg eCO_2/ kg food	*kg N lost / kg food*
Poultry	5.1	2.7
Bovine	26.5	6.9
Pigmeat	6.9	3.8
Milk	1.3	3.6
Cheese	9.8	3.6
Eggs	3.5	3.8
Fish	3.8	2.4
Cereals	0.6	0.6
Fruits	0.4	7.7
Pulses	0.8	0.4
Starchy roots	0.2	0.8
Vegetables	0.7	7.7
Stimulants	0.7	7.7
Oilcrops	1.6	7.7
Sugarcrops	0.7	7.7
Nuts	1.2	0.4
Spices	0.7	7.7
Beverages	0.7	7.7

Table A2. Assignment of footprint categories for the C and N integration. Footprint-specific sectors as assigned to comparable sectors for the carbon and nitrogen footprint.

Sector	Carbon footprint categories	Nitrogen footprint categories
Food consumption/ wastewater	Wastewater	Food consumption
Food production	Food production	Food production
Utilities	Co-gen electricity	Co-gen electricity
	Co-gen steam	Co-gen steam
	Other on-campus stationary	Other on-campus stationary
	Purchased electricity	Purchased electricity
	Purchased steam / chilled water	Purchased steam / chilled water
	Scope 2 T&D losses	n/a
Transport	Direct transportation	Direct transportation
	Faculty / staff commuting	Faculty / staff commuting
	Student commuting	Student commuting
	Directly financed air travel	Directly financed air travel
	Other directly financed travel	Other directly financed travel
	Study abroad air travel	Study abroad air travel
	Student travel to/from home (optional)	n/a
Research and agriculture	Fertilizer	Fertilizer
	Agriculture	Agriculture
	Refrigerants & chemicals	n/a
	Solid waste	n/a
	Paper	n/a
Additional offsets	Not included in this comparison[a]	
Non-additional offsets		

[a] Additional offsets (e.g., purchased renewable energy credits) and non-additional offsets (e.g., sold renewable energy credits) were excluded so that direct emissions from campus activities could be compared before adjustments due to the purchase or sale of offsets.

Results

A complete comparison of the data inputs for the Campus Carbon Calculator$^{\text{TM}}$ and the Nitrogen Footprint Tool was conducted (**Table A3**). The comparison found substantial overlap in the data requirements, and gaps were identified that will be filled in the integrated tool (e.g., some fuel sources for the nitrogen footprint, food for the carbon footprint).

Table A3. Comparison of the C and N footprint data requirements. The data requirements for the Campus Carbon Calculator and Nitrogen Footprint Tool are organized by scope. For each sector, it is noted whether that sector is already included for each footprint, will be added, or is under review.

	Category	Data inputs	Carbon footprint	Nitrogen footprint
Scope 1	On-campus stationary sources	Residual Oil (#5-6), Distillate oil (#1-4), Natural gas, LPG (propane), Coal (Steam Coal)	Yes	Yes
		Incinerated waste, wood chips, wool pellets, grass pellets, residual bioheat, distillate bioheat, attributable solar - electric, attributable soil - thermal, attributable wind	Yes	To be added
	Direct transportation sources	Gasoline fleet, diesel fleet, natural gas fleet	Yes	Yes
		E85 fleet, B5 fleet, B20 fleet, B100 fleet, hydrogen, other fleet fuel, electricity fleet	Yes	To be added
	Refrigerants & chemicals	HFC-134a, R-404a, HCFC-22, HCFE-235da2, HG-10, Other	Yes	No
	Agriculture sources	Synthetic fertilizer applied and % nitrogen, organic fertilizer applied and % nitrogen	Yes	Yes
		% fertilizer applied to general landscape or experimental farm, by type	To be added	Yes
		Dairy cows, beef cows, swine, goats, sheep, horses, poultry, other	Yes	To be added
		Research animals (e.g., mice, rats)	To be added	Yes
Scope 2	Electricity, steam, chilled water	Electricity, steam, chilled water	Yes	Yes
Scope 3	Commuting	Carbon-free modes	Yes	To be added
		Automobile, bus, light rail, commuter rail	Yes	Yes
	Directly financed	Faculty/staff, students, train, taxi/ferry/rental car, bus, alternative fuel	Yes	To be added

	outsourced travel	bus, cycling, personal mileage reimbursement		
	Study abroad travel	Air travel - study abroad	Yes	Yes
	Student travel to/from home	Automobile, carpool, bus, train, air, cycling	Yes	Yes
	Solid waste	Mass burn, refuse derived fuel (RD), no CH4 recovery, CH4 recovery and flaring, CH4 recovery and electric generation	Yes	To be added
	Wastewater	Septic system, aerobic, anaerobic, aerobic digestion	Yes	Yes
		Sludge fate (land applied, landfill, incineration, other use)	To be added	To be added
	Paper	0 lb., 0.25 lb., 0.5 lb., 0.75 lb., 1 lb.	Yes	Under review
	Food purchases	Number of meals served, meal plans, % organic food, % local food	To be added	Yes
		All food purchases, categorized by food type, and any scaling information	To be added	Yes
Offsets	Offsets with Additionality	On-campus composting	Yes	Yes
		Forest preservation, retail offsets, other	Yes	Under review
	Non-Additional Renewable Energy Certificates	Green power certificates, retail offsets (high and low end), other	Yes	Under review

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CHAPTER 2: HOW AERATED STATIC PILE HEAT RECOVERY COMPOSTING AFFECTS AN ORGANIC DAIRY FARM'S NITROGEN BUDGET

Abstract

As dairy farms become larger and generate more by-products per farm, novel waste management strategies are needed to reduce nitrogen pollution and promote the recycling of nutrients. Aerated static pile (ASP) heat recovery composting is a manure management method that processes agricultural by-products, generates a stable soil amendment, and captures heat. In this study, we explored how ASP heat recovery composting affects an organic dairy farm's nitrogen budget and nitrogen use efficiency. The nitrogen budget of the University of New Hampshire Organic Dairy Research Farm (UNH ODRF) was calculated at four scales: The whole farm, the crop system, the animal system, and the compost facility. Through its organic practices and compost facility, the farm cycles substantially more N on its property (18,500 kg N) than it imports (7,080 kg N) or exports (1,740 kg N). The whole farm N surplus (5,340 kg N) and nitrogen use efficiency (NUE; 25%) can be explained by the crop system (N surplus of 3,650 kg N; NUE of 70%) and the animal system (N surplus of 11,370 kg N; NUE of 13%). Although the UNH ODRF has a lower whole farm NUE (25%) than other dairy farms in the literature (32% average), its N surplus per unit area is also much lower (66% less) than the average. This means that N losses are released over a much larger area and are less likely to contribute to local negative environmental impacts.

The compost facility processed 2,600 kg N of feedstock in 2014 and generated 2,300 kg N finished compost. Exporting just 20% of that finished compost off-site would increase the

whole farm NUE to 31% (nearly the dairy farm average) while still retaining enough N for the farm’s N balance and pasture productivity. However, exporting too much compost could put the farm at risk for soil N mining. Producing and selling compost with an ASP heat recovery compost facility is a viable strategy for both improving a farm’s N efficiency and adding an additional revenue stream. However, a farm’s N balance must first be assessed to confirm that the farm will still have adequate N stores for pasture productivity.

1. Introduction

Nitrogen pollution from manure contributes to a cascade of negative impacts to human and ecosystem health (Galloway et al. 2003; Sutton et al. 2011), and it is expected to increase as global demand for meat and animal products grows over the coming decades (Pelletier & Tyedmers 2010; Steinfeld & Gerber 2010). This presents an opportunity and a need to explore novel methods to decrease nitrogen (N) pollution from manure. A new method of manure management that could help recycle available N and reduce N pollution is aerated static pile (ASP) heat recovery composting (Smith & Aber 2018). The effectiveness of this method of composting for N management can be assessed using a farm N budget and a compost export scenario (Watson & Atkinson 1999; Oenema et al. 2003; de Klein et al. 2017).

1.1. Composting for manure management

Over the last several decades, the number of dairy farms in the US has been decreasing while the number of cows per farm has been increasing (MacDonald & Newton 2014). From 1992 to 2012, the median number of dairy cows per farm increased from 100 to over 900. The approximately 1,800 farms that each house over 1,000 cows contain 50% of the US dairy cow population. As dairy operations have become more concentrated, the amount of manure generated in a small land area has increased, necessitating novel manure management methods.

The selection of a manure management system can depend on many factors, including the livestock type, farm size, land availability, infrastructure costs, labor costs, environmental pollution, and the farmer's goals (Ogejo 2009). Common dairy farm types include confined dairy systems, open feedlot dairy systems, and pasture dairy systems. Confined dairy systems house dairy cows in an enclosed structure, which facilitates the collection of manure for treatment. In open feedlot and pasture dairy systems, a portion of the manure is deposited directly to the land, making collection and management more challenging. However, some of that manure can still be collected from the farm's barn.

Common methods for managing collected dairy manure are storage in a lagoon, land application, anaerobic digestion, and composting (Van Horn et al. 1994). A lagoon is usually a pit designed to store manure, and some denitrification can occur in these lagoons. Pollution control measures can be implemented, such as installing a liner and cover for the lagoon. A holding lagoon can be an intermediate step before another manure management method, such as a constructed wetland for treatment. Land application is the spreading of manure and other excrement directly to the land. This method promotes nutrient recycling by returning manure to crop fields, but it can lead to significant N losses during application, especially when the product has a high moisture content (Bussink & Oenema 1998). Anaerobic digestion is the process in which waste is collected in a vessel and processed by anaerobic microorganisms (Dong et al. 2006). Methane (CH_4) is a by-product of anaerobic digestion that can be captured and used for heating and power. The leftover digestate from the anaerobic digester can be used as a fertilizer. The installation of anaerobic digesters can require a significant capital investment, but farms often see benefits like on-site energy production, improved odor control, and reduced costs for manure application (Moser et al. 1998).

Composting is the breakdown of organic materials into a stable soil amendment by aerobic microorganisms (de Bertoldi et al. 1983; Rynk et al. 1992; Haug 1993). Composting methods differ in how they circulate oxygen through the compost pile. For windrow composting, organic materials are combined in a long row (i.e., a windrow) that is turned or agitated periodically to introduce oxygen (Misra et al. 2003). Windrow composting is typically less expensive than other forms of composting because the systems are usually outside and require reduced capital expenditures. However, windrow composting requires more labor for turning the piles periodically. Environmental pollution from windrow composting can be reduced by covering the piles to prevent volatilization of gases and lining the piles to prevent leaching of nutrients and pollutants. However, windrow composting sometimes does not achieve the minimum temperature necessary to kill pathogens and weed seeds.

In-vessel composting encompasses any method in which the compost is contained in a structure (Misra et al. 2003). Benefits of in-vessel composting include reduced effects from the weather, containment of odors, improved temperature control, and improved control over waste streams (e.g., leachate, exhaust gas). In-vessel composting can either use mechanical methods of aeration (e.g., turning, agitation) or forced aeration.

Aerated static pile (ASP) composting forces air through a stationary compost pile, usually in a vessel or structure (Misra et al. 2003). Because of the higher temperatures achieved from ASP composting, this method can reduce the amount of time required for the composting process. Environmental pollution could be reduced because the system is in a contained structure, and pollution streams can be managed. Labor requirements are minimal because the piles do not need to be turned. However, constructing a facility for ASP composting can initially be capital-intensive.

An addition to ASP composting that could make the process more cost-effective is heat recovery (Smith & Aber 2014, 2018). The heat generated by microorganisms during decomposition is usually released to the atmosphere as a by-product of composting. ASP heat recovery composting can capture some of that heat and use it for other on-farm heat and energy needs, reducing other energy costs on the farm. The potential for commercial-scale ASP heat recovery composting to improve a farm's overall N balance has not yet been assessed.

1.2. Farm nitrogen budgets

A farm N budget determines the efficiency with which a farm uses N by quantifying the N inputs and outputs (Dalgaard et al. 1998, 2012; Watson & Atkinson 1999; Oenema et al. 2003; Oenema 2006; Leip et al. 2011; de Klein et al. 2017; **Figure 1**). This method can be used to determine the effect of various pollution management strategies on the farm's overall N balance and performance indicators. Examples of farm N inputs include fertilizer, feed, purchased livestock, bedding, and biological nitrogen fixation (BNF). The farm N outputs are the farm products (e.g., milk, meat). Other unintended N loss pathways from the system (e.g., volatilization, leaching, erosion) can also be quantified as part of an N budget. The results of a farm N budget can be used to identify points of inefficiency, educate farmers, and inform policymakers (Schröder et al. 2003). The efficiency of farms that vary in land area and production level can be compared with this approach (Sassenrath et al. 2012). The farm N budget approach has been used extensively in the literature (e.g., de Klein et al. 2017).

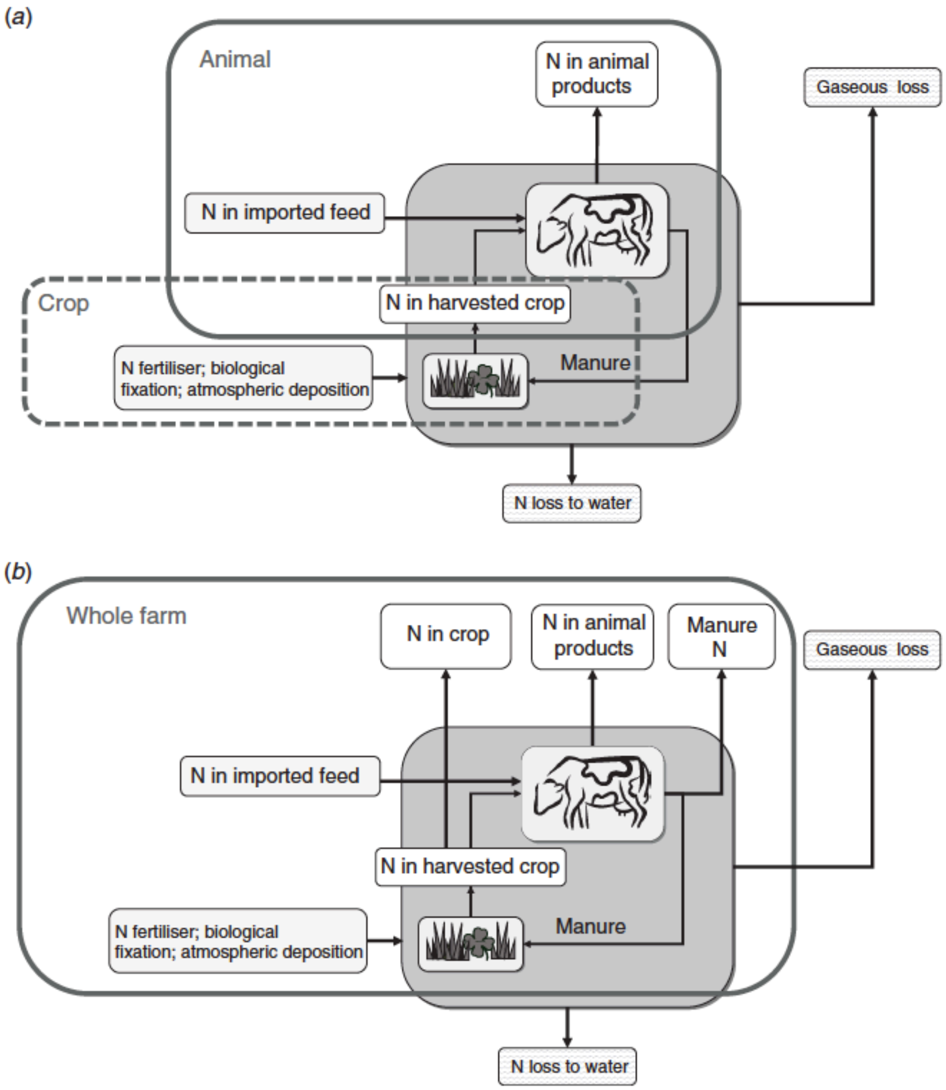

Figure 1. Dairy farm nitrogen inputs and outputs at three nitrogen budget scales. The budget scales are: a) Animal and crop; and b) whole farm. The whole farm scale assesses nitrogen imports to and exports from the farm property (e.g., grain imported, milk exported), whereas the crop and animal scales include both farm imports/exports as well as on-farm cycling of nitrogen (e.g., nitrogen harvested in baleage, manure deposited to the fields). *Source: de Klein et al. 2017, reproduced with permission.*

Farm N budgets can vary in complexity and system bounds. At the simplest level, farm N budgets compare the N that enters and exits the property (Oenema et al. 2003). At the most complex level, a farm N budget quantifies all N flows into, within, and from the farm property. This detailed farm N budget is the only type that identifies the fate of any excess N. Possible fates of excess N are storage on the farm property (e.g., in soil organic matter, vegetation) or loss from the farm (e.g., leaching, volatilization, erosion, denitrification). However, these calculations require extensive monitoring data sets that are not available on most farms. Efforts have been made in the literature to identify clear and comprehensive N budget types and metrics for assessing a farm's N efficiency with more commonly available data sets (de Klein et al. 2017).

On a dairy farm, three types of N budgets can be calculated together to explore how N is cycling through the connected farm systems: Crop, animal, and the whole farm (**Figure 1,** de Klein et al. 2017). The crop N budget assesses the balance between N soil additions (e.g., fertilizer, BNF) and the crops that are harvested or grazed (e.g., harvested baleage, pasture). The animal N budget assesses the balance between what the cows consume (e.g. pasture, baleage, grain) and the products (e.g., milk, meat). Finally, the whole farm N budget assesses the balance between imports to the farm property (e.g., grain, purchased baleage, BNF, atmospheric deposition) and the products exported from the farm (e.g., milk, meat).

1.3. Farm nitrogen performance indicators

The potential for N loss and the N efficiency of a dairy farm can be calculated and compared using two performance indicators: N surplus and N use efficiency (NUE) (Oenema et al. 2003; de Klein et al. 2017). Both performance indicators can be calculated at the crop, animal, and whole farm scale for a dairy farm. The N surplus is the difference between N inputs and outputs (Oenema et al. 2003; Oenema 2006; Leip et al. 2011; de Klein et al. 2017). This indicator determines how much of the N that was used to make a product is not contained in that final product. A lower N surplus means more of the N inputs are converted into product and less excess N remains for environmental loss. The fate of the N surplus is not always clear depending on the level of detail in the budget calculations. Potential fates for the N surplus include storage on the property (e.g., biomass uptake) or loss to the environment (e.g., leaching, volatilization).

NUE is the percent of N invested into farm production that makes it into the intended product (e.g., crops, milk) (Oenema 2006; Erisman et al. 2018). NUE is calculated as a ratio of the products (e.g., crops, milk) over the inputs (e.g., fertilizer, feed, BNF).

The N performance indicators can be plotted on a conceptual diagram of N targets specific to the farm type (**Figure 2**). The conceptual framework brings together the N inputs, outputs, surplus, and NUE. Because of the mobility of N and the inherent N losses during food production, the target NUE should always less than 100% (de Klein et al. 2017). An NUE of 100% would assume that all N inputs to a system are converted into N outputs. However, because N is lost at every stage of the food production supply chain (e.g., fertilizer runoff, processing waste, manure losses), an NUE of 100% would actually pull N from other sources (e.g., soil reserves), which decreases the fertility of the system over time. The NUE target ranges reported in the literature take these expected N losses into account to avoid N soil mining, which is the depletion of N from soil reserves.

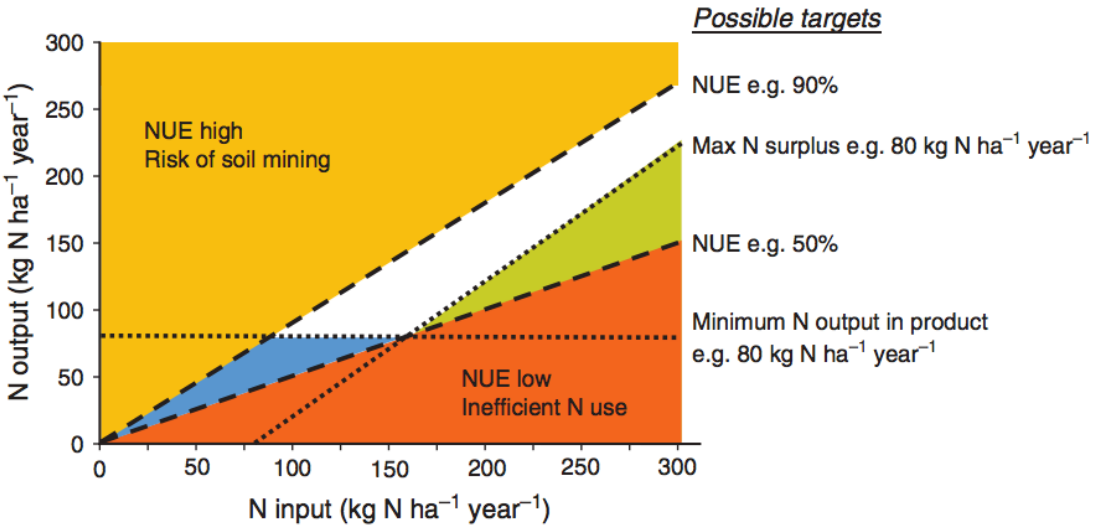

Figure 2. Example conceptual diagram for farm NUE targets. This conceptual diagram for a cropping system shows how the farm N inputs and outputs can be plotted relative to NUE targets. The framework was developed by the EU Nitrogen Expert Panel. *Source: de Klein et al. 2017, reproduced with permission.*

1.4. Research objectives

To explore whether ASP heat recovery composting can be an effective strategy for managing farm N pollution, the overarching question of this study is: **How is a farm's N budget affected by ASP heat recovery composting?**

The specific objectives addressed are:

1. Determine the whole farm, animal, and crop N budgets of the UNH ODRF.
2. Determine the N budget of the ASP heat recovery compost facility at the ODRF.
3. Calculate N performance indicators (N surplus and N use efficiency) at the whole farm, animal, and crop scales for two scenarios:
    a. The ODRF as it currently operates, and
    b. Exporting 20% and 50% of the compost as a product.
4. Compare the ODRF farm N budget to other dairy farms in the literature.

2. Methods

2.1. Study site

The dairy farm study site was the UNH ODRF in Lee, New Hampshire. The ODRF has a herd of Jersey cows made up of approximately 50 dairy cows and 50 replacement cows (heifers and calves) (**Figure 3, Appendix A Table A1**). The farm property spans 120 ha, which is made up of 55 ha for crop and forage production, 15 ha for pasture, and 50 ha of woodland (Smith 2016). The farm is a USDA-certified organic dairy farm. The cattle graze at least one third of the year, following USDA organic guidelines. The cattle diet is made up of a mixture of imported feed grain, forage, and baleage, most of which is produced on-site or on nearby fields. The farm property has an open bedded pack barn for the cattle, storage barns, a step-up milking parlor, and a composting facility.

Figure 3. Images of the UNH Burley-Demeritt Organic Dairy Research Farm. Images show Jersey cows A) grazing and B) in the bedded pack barn. *Source for images: UNH 2012; Personal communication, Matthew Smith, 2016.*

Agricultural by-products from the ODRF are processed at the on-site Joshua Nelson Energy Recovery Compost Facility and later land-applied to the ODRF fields. The compost facility uses ASP heat recovery composting, and it is the only commercial-scale research facility of its kind. The compost facility is a pole barn structure that is 30 m x 15 m x 7 m (Smith & Aber 2014, 2018). It consists of 8 bays into which compost feedstock materials are loaded. The piles are aerated with a fan that pulls air through the piles via perforated pipes in the facility floor. Heat is captured in a water tank with an isobar array (Smith & Aber 2014, 2018). Feedstock materials include cow manure, bedded pack, waste baleage, and wood chips. Most feedstock materials are from the ODRF property. Gas emissions from the piles are routed through a biofilter, and leachate and condensate that drains from the piles are collected in a storage tank and spread on adjacent fields.

2.2. Farm nitrogen budget

The ODRF N budget flows were identified and grouped into those entering the property, cycling on the property, and exiting the property (**Figure 4**). Additional monitoring data sets were collected to asses on-farm N cycling and to identify potential N loss pathways. The N flow data sets and methods are described below.

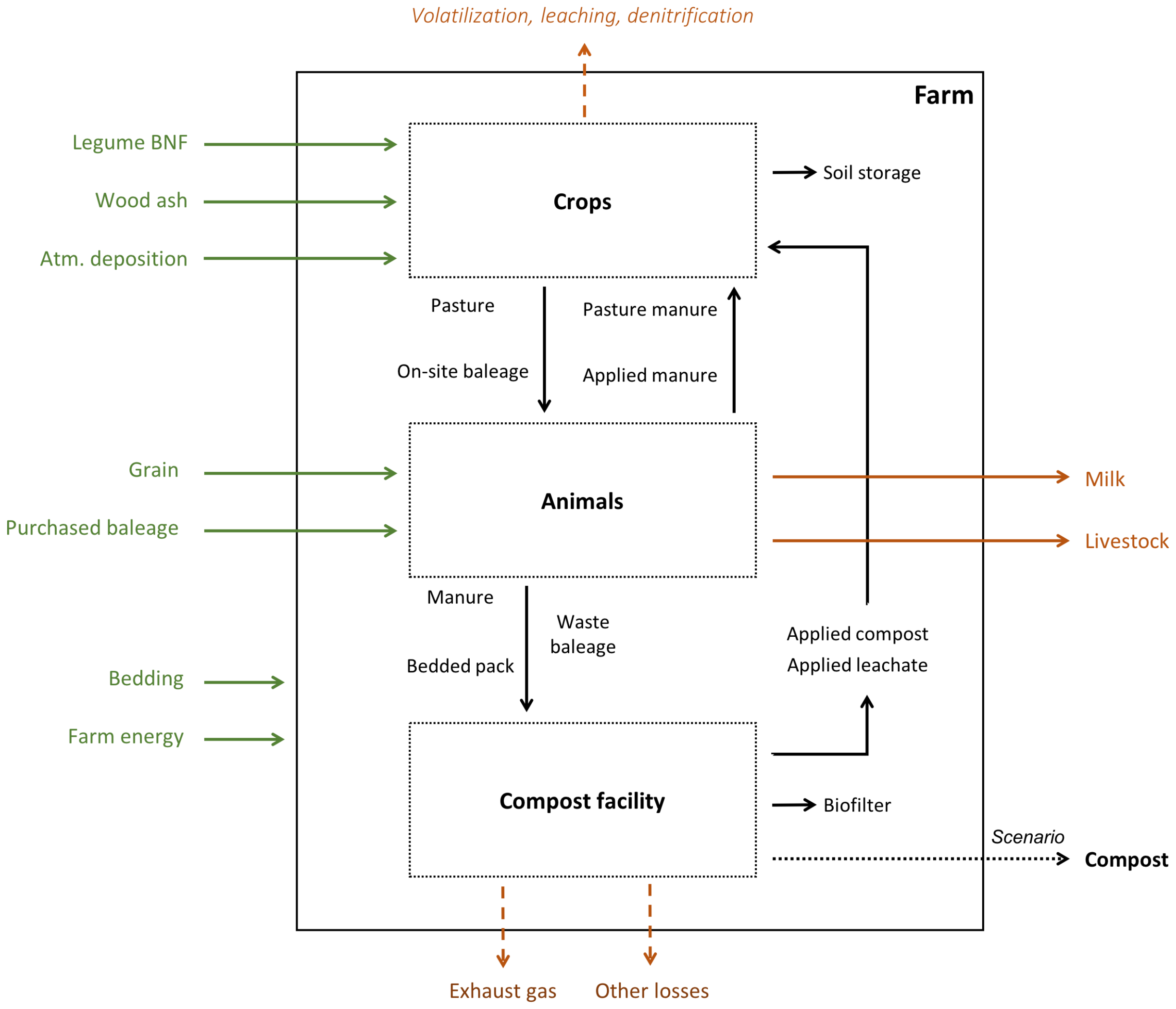

Figure 4. Farm nitrogen inputs, outputs, and on-farm cycling for the UNH ODRF. The solid box indicates the farm's geographic boundary, and inputs to and from the farm property are depicted. Solid green arrows are nitrogen inputs (e.g., feed), solid orange arrows are products (e.g., milk), and solid black arrows are on-farm nitrogen flows (e.g., manure to the fields). Dashed arrows are unintended fluxes (e.g., ammonia volatilization from manure). The on-farm nitrogen cycling is shown for three connected systems: The crop system, the animals, and the compost facility. All flows depicted were quantified except soil storage, volatilization, leaching, and denitrification. BNF is biological nitrogen fixation.

The N inputs to the farm property are feed grain, purchased baleage, cattle, bedding, BNF, atmospheric deposition, and energy use (**Figure 4**). The products or exports from the farm property are milk and sold cattle. On-farm N cycling processes included in the crop and animal N performance indicators are the grazed pasture and on-site baleage production. The potential N loss pathways explored in this study are soil fluxes from manure application and stream and groundwater N concentrations. The compost facility N budget flows are the inputs to the facility (feedstock materials), the intended products (compost), and the measured N losses (gas emissions, leachate, and condensate). See **Section 2.**4 for the methods for each of these N flows.

Data were collected from three sources: ODRF records, completed and ongoing monitoring studies, and new research (**Table 1**). Data sets were collected for calendar year 2014.

Table 1. Data sources for the ODRF nitrogen budget. The data sources include farm records, monitoring studies, and new research at the UNH ODRF in Lee, New Hampshire.

Data source	Farm nitrogen budget data sets
Farm records: Data acquisition from farm records	• Annual milk production, protein content, fat content • Livestock counts, types, and ages • Feed grain purchases • Baleage (on-site production and purchased) • Pasture grazed by cattle • Purchased and sold livestock • Bedding (purchased) • Manure production and management • Energy use (e.g., farm vehicles, electricity)
Monitoring studies: Data collected from complete/ongoing monitoring studies at the University of New Hampshire	• Atmospheric deposition [a] • Pasture vegetation surveys [b] • Groundwater and stream water nitrogen concentrations [c] • Compost feedstock materials [d]
New research: Data collected from new research at the compost facility	• Compost exhaust gas emissions • Leachate/condensate [d] • Compost production [d]

[a] Atmospheric deposition from monitoring station at Thompson Farm from PREP 2017

[b] Pasture vegetation surveys from Antaya 2016 and Green 2011

[c] Groundwater and stream water nitrogen data are collected by the UNH Water Quality Analysis Laboratory 2016

[d] Compost experiment records from Matthew Smith, personal communication

2.3. Farm nitrogen performance indicators

The farm N budget results were used to calculate two N performance indicators: N surplus and NUE (**Table 2**). The N surplus was calculated as the difference between inputs and outputs (**Equation 1**; see **Figure 5** for a summary of inputs and outputs) (Oenema et al. 2003; de Klein et al. 2017). The NUE was calculated as the ratio of N outputs (e.g., milk) to farm N inputs (e.g., feed, bedding, BNF) (**Equation 2**). The N performance indicators were calculated at the whole farm, crop, and animal scale.

To assess how the compost facility affects the ODRF's N balance, the N performance indicators were calculated in two ways: 1) With the current management system (compost land application), and 2) with a percentage (20% and 50%) of the finished compost exported.

Table 2. Farm nitrogen performance indicators. These two nitrogen performance indicators were calculated for the UNH Organic Dairy Research Farm at the whole farm, crop, and animal scale. ECM = energy-corrected milk.

Farm metric	Budget level	Units
Nitrogen surplus	Whole farm; Crop; Animal	kg N kg ECM^{-1}
Nitrogen use efficiency	Whole farm; Crop; Animal	%

Equation 1. Nitrogen surplus

$$Nitrogen\ Surplus = \sum Inputs - \sum Outputs$$

Where *Inputs* (kg N kg ECM^{-1}) describe the inputs to the farm (e.g., feed, legume biological N fixation, bedding, energy use) and *Outputs* (kg N kg ECM^{-1}) describe the N contained in the farm products or in gaseous fluxes from the property. *Nitrogen Surplus* is in units of total kg N kg ECM^{-1}.

Equation 2. Nitrogen Use Efficiency (NUE)

$$NUE = \frac{N\ outputs}{N\ inputs}$$

Where *N outputs* (kg N kg ECM^{-1}) describe the N contained in the farm products (milk); and *N inputs* (kg N kg ECM^{-1}) describe the N inputs to the farm (e.g., feed, legume BNF, bedding, energy use). *NUE* can be a ratio or percentage (%).

For comparison with other farms, the weight of milk should be standardized to units of energy-corrected milk (ECM) based on its fat and protein content (Kimming et al. 2014). The average fat and protein content of milk are 3.5% and 3.2%, respectively (Maulfair et al. 2011). All N performance indicators for the ODRF were calculated and compared in terms of ECM.

2.4. Farm nitrogen flows

The whole farm N budget assesses how effectively the intended farm N inputs were converted into the farm N products (**Figure 5**). The inputs for the whole farm N budget are feed grain, purchased baleage, bedding, legume BNF, atmospheric deposition, purchased wood ash, and energy use. The exports from the farm are milk and sold cattle. The crop scale indicators assess the balance between soil N additions and crops produced on-site. The inputs for the crop N budget are the soil N additions: Manure application, finished compost application, compost leachate application, BNF, and atmospheric deposition. The output is the on-site baleage production and pasture for grazing. The animal scale indicators assess the balance between N in the feed and the farm products. The inputs for the animal N budget are the feed consumed by the animals: Grain, baleage (purchased and on-site production), and pasture. The output is the milk produced and livestock sold. Potential N loss pathways (e.g., trace gas fluxes, stream water concentrations) were explored to consider how and where the whole farm N surplus could be lost or stored.

It was assumed that all purchased products were used in the same year that they were purchased. Although some purchased products (e.g., grain) were likely stored and used in a following year, the adjustment calculations based on farm records (e.g., for purchased baleage) made all flows specific to the year 2014.

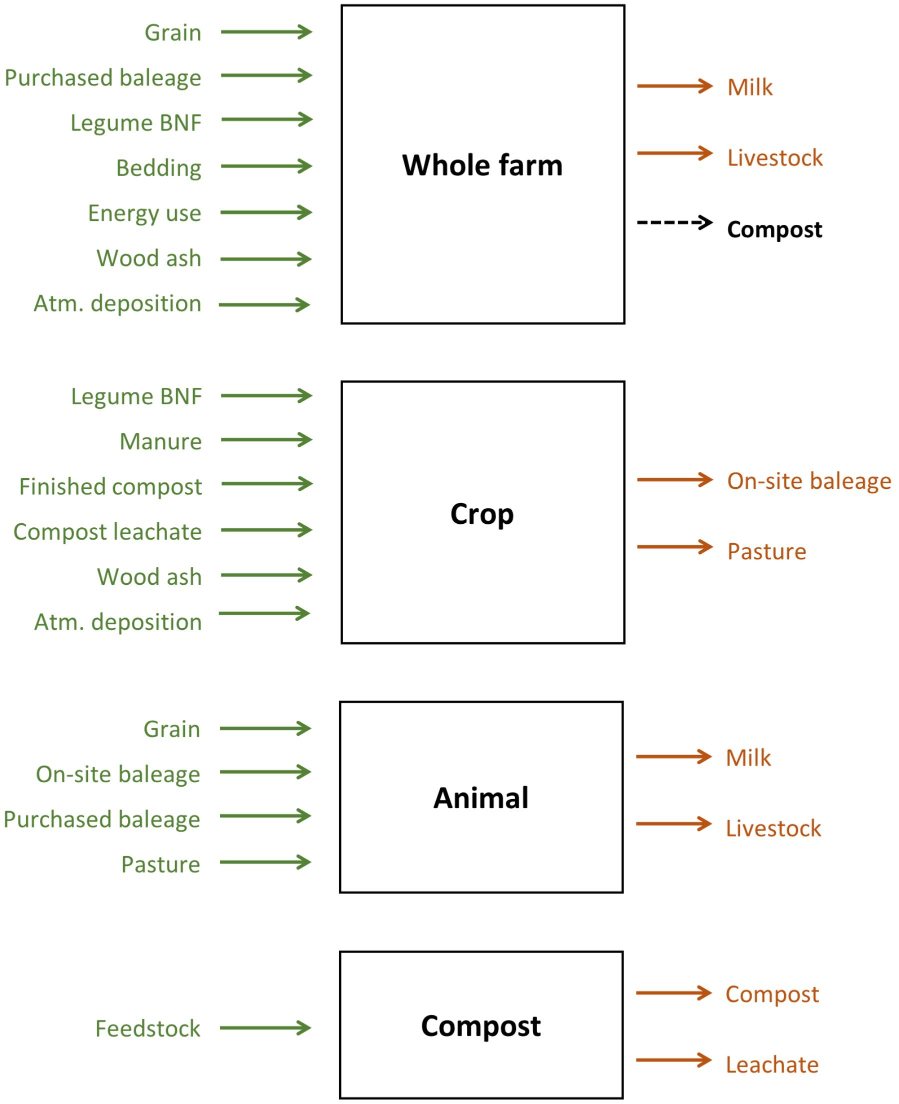

Figure 5. UNH ODRF nitrogen inputs and outputs at the whole farm, crop, and animal scale. These nitrogen flows were used to calculate the whole farm, crop, and animal nitrogen performance indicators: Nitrogen use efficiency and nitrogen surplus. The dashed line for compost in the whole farm budget illustrates a scenario analyzed in this study.

2.4.1. Feed: Grain, baleage, pasture

The N imported in feed grain purchases was calculated using several types of farm records: purchase records from the UNH Business Services Center (UNH BSC 2014); feed logs (milking cows only; UNH ODRF 2014a), total mixed ration (TMR) recipe builder spreadsheets (milking cows only; Poulin Grain 2014a), and personal communications (Nicole Guindon

October 2015; Ryan Courtright October 2018) (**Appendix A Table A2**). The purchase records reported the total quantity of feed purchased by type. The milking cow feed logs show the amount of grain fed to the milking cows each day. The TMR recipe builder spreadsheets report the milking cow dietary mix throughout the year, including grain, baleage, and grazing. Personal communications with the ODRF farmers were used to estimate the average grain consumed by non-milking cows (dry cows, heifers, yearling heifers, calves) because daily feed records and TMR spreadsheets are not kept for non-milking cows. The multiple data sources were necessary to confirm the actual feeding rate. The milking cow feed logs were the primary source used for milking cow feed grain, and personal communications were used to estimate the annual feed grain for non-milking cows. The purchase records and TMR spreadsheets were used as checks. The protein content of the feed grain was obtained from feed tags (grain for non-milking cows) and the TMR spreadsheets (specialized grain mix for milking cows that varies in protein content throughout the year). The protein content was used to calculate the N content, assuming the protein is 16% N.

The on-site baleage production was calculated using farm records (UNH ODRF 2014b; Poulin Grain 2014b) (**Appendix A Table A3**). These records reported the number of bales produced and the quality analysis for each of those bales (e.g., moisture content, protein content). It was assumed that baleage produced on-site in 2014 was fed to the cows in 2014. However, some of the baleage produced the previous year would be fed in 2014, and some of the 2014 baleage would be fed the following year. Despite this, the assumption is reasonable because the total baleage fed (on-site production + purchases) was calculated based on 2014 feed logs and farm records.

Purchased baleage imports were calculated using the milking cow feed logs (UNH ODRF 2014a), purchase records, (UNH ODRF 2018a) on-site baleage production records (UNH ODRF

2014b), and personal communications (**Appendix A Table A3**). The purchased baleage records were available for 2015 and 2016 but not 2014. Given this, the purchased baleage requirement was calculated as the difference between the calculated total herd baleage intake and the on-site baleage production. The total herd baleage intake was calculated using the milking cow feed logs for annual baleage intake and farmer estimates of daily baleage intake by non-milking cows (personal communication, Ryan Courtright, October 2015). The 2015/2016 purchased baleage records were used as a check to confirm the calculated 2014 purchased baleage was in a reasonable range.

The pasture harvested through grazing was calculated using dry matter intake (DMI) calculation worksheets from farm records (UNH ODRF 2014c) (**Appendix A Table A4**). The DMI worksheets are calculation templates for estimating the daily feed mix of grain, baleage, and pasture per cow for organic certification. These records are kept for each cow type (milking cows, dry cows, heifers, yearling heifers). The pasture intake is calculated as the difference between the dry matter demand (DMD) by cow type and the reported feeding rates for grain and baleage. The DMD values are standards for Jersey cows at different life stages from the National Research Council (UNH ODRF 2014c). The N content of the pasture was calculated from literature values representing a weighted average of grasses (bluegrass, fescue, orchardgrass) and legumes (alfalfa, red cover) (IPNI 2014) (**Appendix A Table A5**). The weighted average was based on the mixture of grasses and legumes in a vegetation survey by Antaya (2016) (**Appendix A Table A6**). It was assumed that cows graze 21 weeks per year, from early May through October (personal communication, Ryan Courtright, November 2017). The DMI worksheets were not used to estimate grain and baleage intake because their estimates represented just a single point in time over the summer, which was not accurate for the winter feeding rates.

2.4.2. Soil nitrogen inputs: BNF, deposition, manure

The pasture BNF was calculated using a vegetation survey conducted by Antaya (2016) (**Appendix A Table A6**). The vegetation survey reported the average percent dry matter by type (57% grasses, 17% legumes, 25% other), but it did not identify specific legume species. Based on site observations, the legume species composition was assumed to be 50% white clover, 25% red clover, and 25% alfalfa based on observations by Green (2011). BNF rates were collected from the literature for clover and alfalfa (Ball et al. 2007; Brady 1982; Carlsson & Huss-Dannell 2003; Havlin et al. 1999; Johnson et al. 1997; **Appendix A Table A7**). It was assumed that N fixation from trees and shrubs would be negligible. There are not any records of N-fixing trees on the property, but N-fixing shrubs (e.g., autumn olive) have been observed on the pasture edges (Eisenhaure 2016). Because the shrub N fixation would be small and records on coverage are not available, N fixation from shrubs and trees was not included in the budget.

Atmospheric N deposition was calculated using deposition rates measured at a monitoring station at the nearby Thompson Farm, which is located 5 km from the ODRF (**Appendix A Table A8**). The wet and dry N deposition rates at Thompson Farm are reported as annual averages. The N deposition rates in 2014 and the ODRF land area were used to calculate total annual N deposition (PREP 2017).

The total amount of manure N produced by the herd in 2014 was calculated based on the number of the cows and the average rates of fecal and urine N production per day (**Appendix A Table A9**). The average grazing manure N application rates for adult Jersey cows are 162 g fecal N/cow/day and 161 g urine N/cow/day (Knowlton et al. 2010). This total manure N was then allocated across three different farm flows: 1) manure collected from the bedded pack barn that goes to the compost facility, 2) manure applied directly to the pasture during grazing, and 3) manure collected from the bedded pack barn that is land-applied. The manure that goes to the compost facility was calculated based on feedstock loading records (see **Section 2.5**). The manure that is applied directly to the pasture during grazing was calculated based on the grazing days per year and the amount of grazing time spent on pasture. The cows graze for 21 weeks per year. Non-milking cows (heifers, yearling heifers, dry cows) spend all of this time on pasture. Milking cows spend 94% of this time on pasture, and the remaining 6% of time (1.5 hours/day) in the barn for milking (personal communication, Ryan Courtright, November 2018). Finally, the amount of manure N that is collected at the bedded pack barn and then applied to the land was calculated by difference.

2.4.3. Farm products: Milk and cows

The weight of milk produced each month in 2014 was reported in farm records (UNH ODRF 2018b) (**Appendix A Table A10**). The weight of milk was converted to ECM using farm records on the butterfat and protein content. The protein content was used to calculate the N content.

The cow population, imports, and exports were totaled through farm records (**Appendix A Table A11**). The only cows added to the herd in 2014 were those born on the farm property. Female calves born on the property were not considered imports because they were born from the existing herd. Cows leaving the farm were sold as bull calves, sold as adult cows, or died on the farm. Cows that died on the farm are composted on the property and were therefore not farm N outputs. The N in the cows leaving the farm was calculated based on the cow weight (UNH ODRF 2014c) and the N content of the live animal (NRC 2003) since the cows were not slaughtered at the ODRF.

2.4.4. Other farm nitrogen flows

The weight of imported bedding (pine wood shavings) was recorded from purchase records from the UNH Business Services Center (**Appendix A Table A12**). Literature values were used for the bedding N content (Rynk et al. 1992).

The purchased wood ash was calculated based on farm records (UNH BSC 2014) and a literature value for the N content of wood ash (Risse & Gaskin 2002) (**Appendix A Table A12**).

Farm records on energy usage were used with emissions factors from the literature to estimate the N emissions from on-farm energy use (personal communication, Tom Oxford, October 2015) (**Appendix A Table A13**). The types of on-farm energy usage include electricity, diesel, and propane. The nitrogen oxide (NOx) and nitrous oxide (N_2O) emissions factors for electricity were obtained for the New England regional grid average from the US EPA eGRID database for 2014 (US EPA, 2017). The NOx and N_2O emissions factors for on-farm diesel and propane were collected from the US EPA Inventory of Greenhouse Gas Emissions and Sinks (US EPA 2016).

2.4.5. Potential nitrogen loss pathways

The N not incorporated into products can either be stored on the farm or lost to the environment through several pathways. The following data sets from the literature and existing on-farm monitoring studies at the ODRF were used to explore the potential for N losses: Trace gas N fluxes; compost leachate N concentrations; and stream and groundwater N concentrations.

2.5. Compost facility nitrogen budget

The N budget of the on-site Joshua Nelson Energy Recovery Compost Facility was calculated separately from the rest of the ODRF N budget to facilitate scenario analysis. The N inputs and outputs were calculated for each ‘batch’ of compost. A batch of compost is a single load of feedstock materials of about 107 m^3, and the feedstock is divided evenly across two bays in the compost facility. The composting process typically lasts 21-60 days. The N inputs and outputs for all compost batches loaded in 2014 were summed (**Appendix A Table A14**).

The main input to the compost facility N budget was the feedstock material (cow manure, bedded pack, waste baleage, wood chips). All the compost feedstock materials in 2014 originated on the farm property. Data sets on the types and amounts of feedstock material were obtained from compost facility records (personal communication, Matthew Smith, June 2018). The N content of these feedstock materials was calculated based on average N contents measured for each feedstock material type (Smith 2016). The energy used by the compost facility (electricity) was included the whole farm N budget and was very small (<60 kg N, or <1% of the compost facility N inputs), so it was not included in the compost facility N budget.

Measured outputs from the compost facility were the by-products of the composting process, including gas emissions, leachate, and condensate. The N-based gas emissions generated during aerobic composting include ammonia (NH_3) and (N_2O). Other common gas emissions include carbon dioxide (CO_2) and methane (CH_4). Average ASP heat recovery composting NH_3 emissions factors (kg NH_3/kg feedstock) were available from a study at this compost facility (See **Chapter 3 of this dissertation**). N_2O emissions were not measured in Chapter 3 of this dissertation because colorimetric gas detection tubes are not available for N_2O. Literature values for N_2O emissions from ASP composting were used and applied specifically to the weight of manure feedstock because the emissions factor is in terms of N excreted (kg N_2O-N/kg N excreted; Hao et al. 2001). Starting in 2015, the compost exhaust gas was routed through a simple biofilter consisting of woodchips. Preliminary studies on this woodchip biofilter suggest it captures at least 80% of NH_3 emissions (Williamson et al. in preparation).

Leachate and condensate from the entire compost facility collect in a single storage tank. The N contained in the leachate and condensate was calculated by multiplying the total volume pumped from the storage tank each year with the average total dissolved nitrogen (TDN) content of that leachate/condensate (Smith 2016) (**Appendix A Table A15**). Leachate/condensate collected in the storage tank is applied on an adjacent field on the ODRF property. The point source collection of leachate/condensate is a benefit of the ASP compost facility because the farmer has control over when and how the leachate/condensate are used. The leachate/condensate was an input to the ODRF fields.

The product from the compost facility is the finished compost, which is applied at the ODRF and other nearby fields used for ODRF baleage production. The amount of finished compost N was calculated by difference based on the weight of the feedstock materials, the measured gaseous losses, the measured leachate/condensate losses, and an assumed percentage of other losses not quantified. It was assumed that an additional 5% of the feedstock N was lost through other pathways not measured in this study. This value was selected because it amounts to half of the total measured N loss pathways (exhaust gas and leachate/condensate). Because exhaust gas and leachate/condensate are the major loss pathways from the compost facility, assuming an additional 50% in losses is conservative and is likely an overestimate of losses from the system. Other potential N loss pathways not measured in this study include passive emissions from the piles between aeration and peak gas fluxes not recorded during the exhaust gas sampling schedule.

2.6. Literature comparisons for dairy nitrogen budgets

Dairy farm N budgets were collected from the literature for comparison and context. These budgets were for dairy farms only—mixed farms were not included. Both organic and dairy farm N budgets were included. The N performance indicators (farm N surplus, NUE) were recorded at the whole farm, crop, and animal scale when available. de Klein et al. (2017) plotted dairy farm N budgets from the literature on a conceptual diagram of NUE targets for dairy farms. To compare the ODRF to other dairy farms in the literature, the ODRF N budget was plotted on this NUE conceptual diagram.

3. Results

3.1. Farm nitrogen budget

The imports to the ODRF in 2014 were 127 t feed grain, 160 bales of purchased baleage, 135 t bedding, and 100 t wood ash for field application (**Table 3**). The exports from the farm were 264 t milk, 14 adult Jersey cows, and 23 bull calves. The products that were produced and used on-farm were 778 bales of baleage, 360 t pasture for grazing, and 340 t finished compost.

Table 3. Farm inputs, outputs, and on-farm products for the UNH ODRF in 2014.

Flow type	Category	Details
Inputs	Feed grain purchases	127 t grain (113 t for milking cows, 8 t for heifers, 6 t for calves)
	Baleage: Purchased	~160 bales, or 100 t baleage wet weight
	Legume coverage	17% legume coverage
	Bedding (wood shavings)	135 t bedding
	Energy consumption	3200 gal diesel; 762 gal gasoline; 97,650 kwh electricity
	Wood ash	100 t applied to fields
Outputs/ Products	Milk sales	264 t milk or 320 t ECM
	Livestock sold	14 adult cows; 23 bull calves
On-farm	Baleage: On-site production	778 bales, or 500 t wet weight
	Pasture grazed	360 t wet weight
	Finished compost	340 t applied to fields

ECM = energy-corrected milk
t = metric ton = 1,000 kg

The total N inputs to the ODRF (7,100 kg N) were four times greater than the N outputs (1,700 kg N) (**Figure 6, 7**). The major N imports to the property were grain (3,000 kg N), legume BNF (1,800 kg N), purchased baleage (1,500 kg N), and atmospheric deposition (420 kg N). Smaller nitrogen imports were wood ash for field application (150 kg N), bedding (120 kg N), and N emissions from on-farm energy use (60 kg N). The N outputs were two products: milk (1,600 kg N) and sold cows (150 kg N).

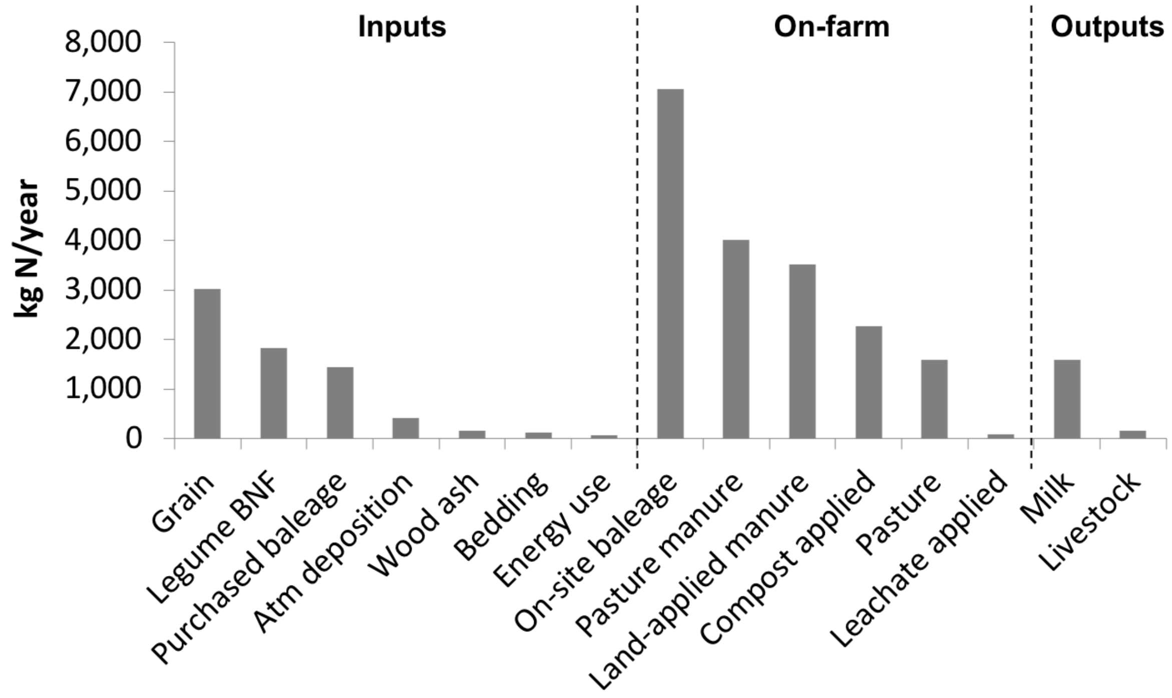

Figure 6. Nitrogen flows at the UNH ODRF in 2014. The nitrogen flows are organized into property inputs, on-farm cycling, and outputs. See Figure 7 for more information on how these nitrogen flows relate to the farm systems (whole farm, crop system, animal system).

The N cycling on the farm greatly exceeded the N inputs and outputs (**Figure 6, 7**). In total, 18,500 kg N of flows originated on and cycled through the farm property. The largest flows were forage for cows (baleage produced on-site: 7,100 kg N; grazed pasture: 1,600 kg N) and on-farm N applied to the fields (manure excreted on the pasture: 4,000 kg N; manure collected from the bedded pack and applied to the land: 3,500 kg N; finished compost applied to the land: 2,300 kg N). Smaller on-farm N flows were compost leachate applied to the fields (90 kg N) and female calves born and raised on the farm (40 kg N).

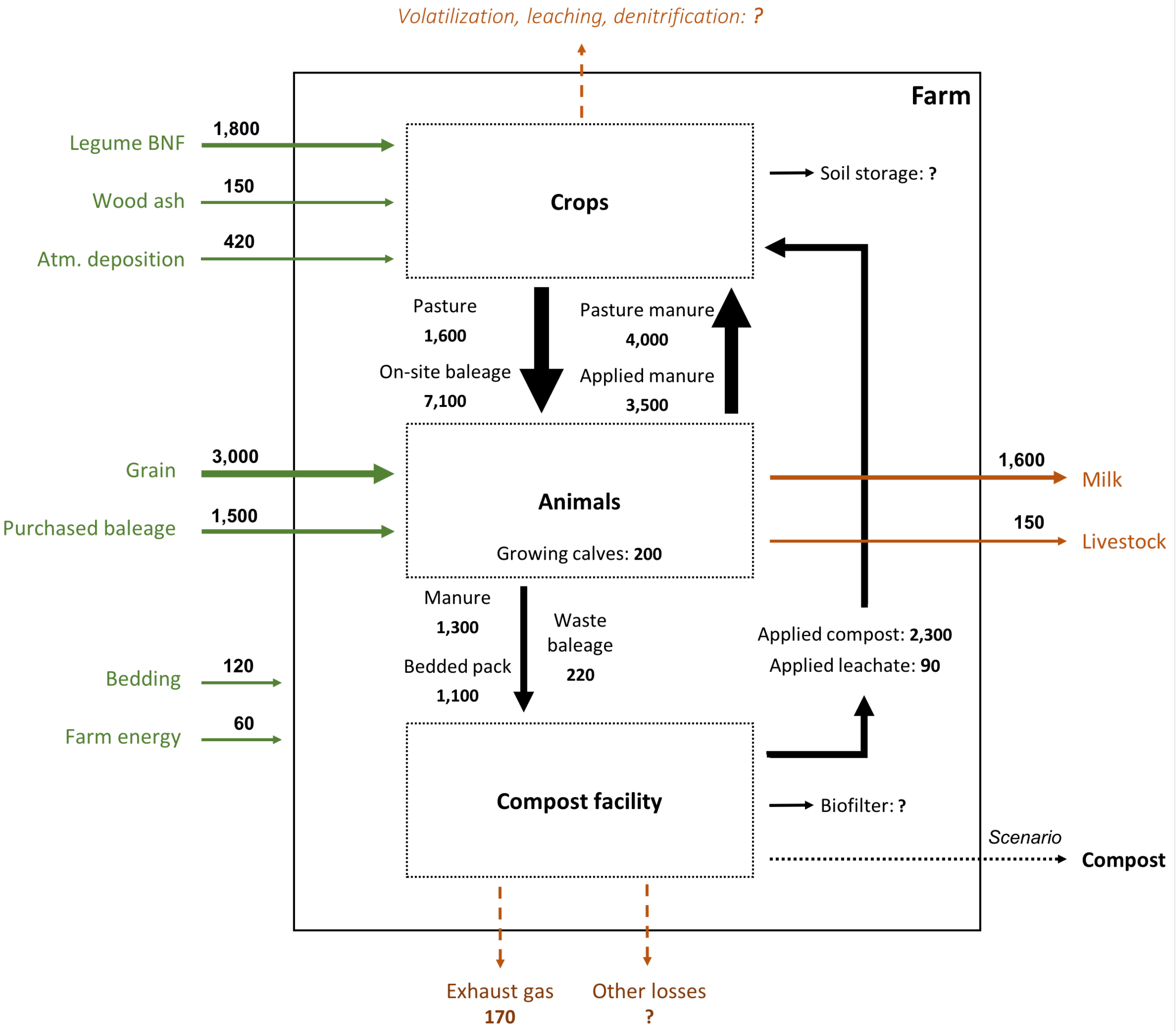

Figure 7. Detailed farm nitrogen budget for the UNH ODRF in 2014. Farm nitrogen flows are in kg N. The solid black line delineates the farm property boundary, and green arrows into the farm property are inputs of nitrogen. The dotted boxes indicate on-farm systems (crops, animals, compost facility) that have nitrogen inputs and outputs. Solid black arrows within the farm boundary are nitrogen flows within the farm. Solid orange lines leaving the property are products. Dashed orange lines leaving the farm are farm N loss pathways. The dotted black line leaving the property is a compost export scenario analyzed in this study.

The detailed farm N budget (**Figure 7**) illustrates how N flows into, within, and out of the ODRF. Large N flows occur between the on-farm systems: crops, animals, and the compost facility. In particular, 8,700 kg N flows from the crops to the animals in the form of pasture and on-site baleage production, and the animals return 7,500 kg manure N directly to the fields. Other inputs to the animal system are imported feed grain (3,000 kg N) and purchased baleage (1,500 kg N). The remaining animal N surplus goes to the compost facility as manure, bedded pack, and waste baleage (2,600 kg N); milk that is exported from the farm (1,600 kg N); growing calves and exported cattle (350 kg N); and other unidentified losses or sinks (1,000 kg N). Other N flows that were not quantified in this study are shown, including losses (e.g., leaching, volatilization), storage on the property (e.g., soil organic matter, compost biofilter uptake), and denitrification.

The whole farm N surplus was 5,300 kg N, or 76 kg N/ha/yr for the 70 ha at the ODRF in active management (e.g., pasture, baleage fields) (**Figure 8, Table 4**). The whole farm NUE was 25%. The crop system had an N surplus of 3,600 kg N and NUE of 70%. The crop N surplus is either lost to the environment (e.g., leaching, volatilization, denitrification) or stored on the farm property. The animal system had an N surplus of 11,400 kg N/yr, most of which is manure that is transferred to other systems on the property like the crop fields and the compost facility. The animal NUE was 13%.

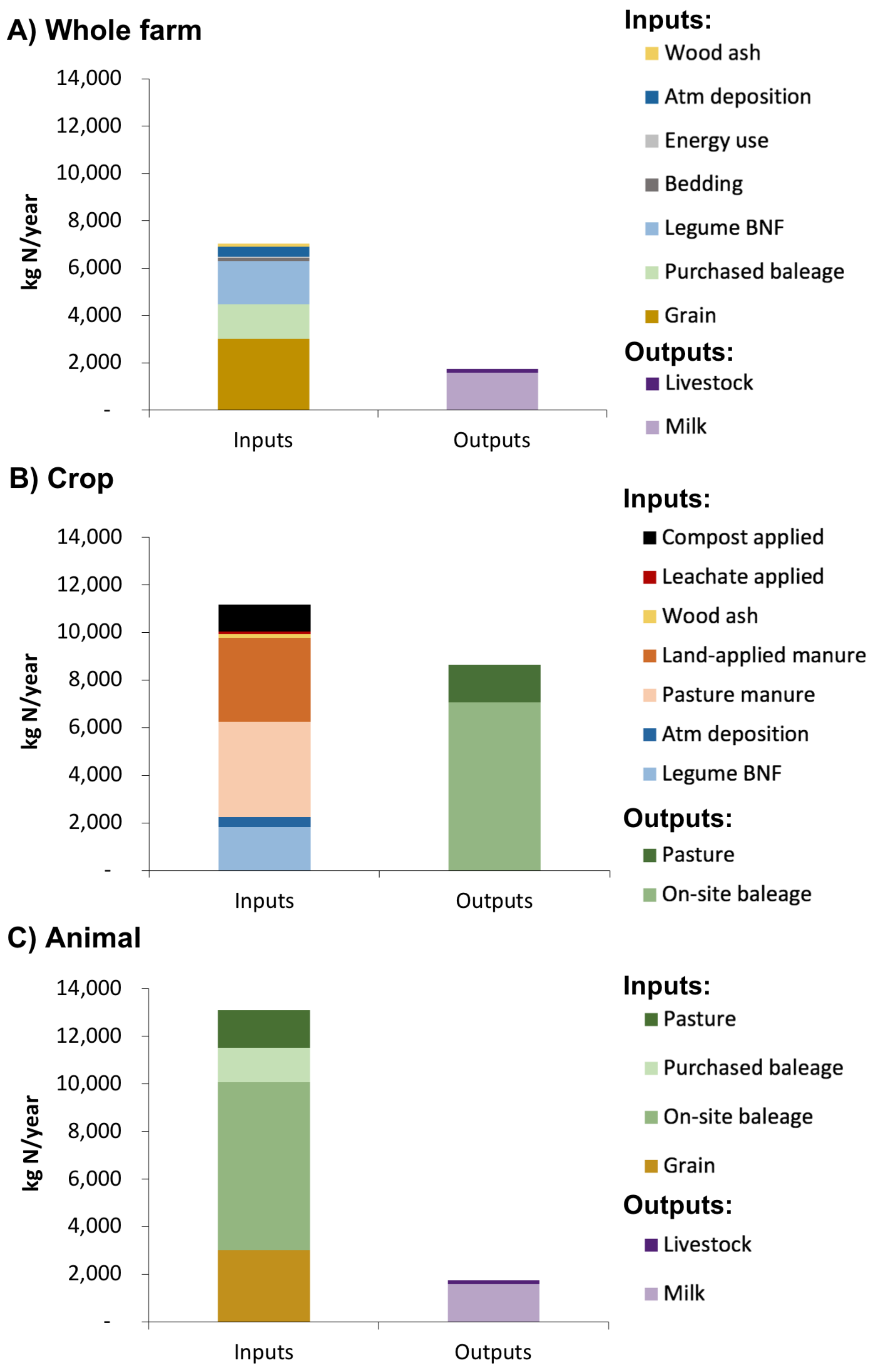

Figure 8. Nitrogen inputs and outputs for three systems at the UNH ODRF in 2014. A) Whole farm, B) Crop, and C) Animal. The difference between the nitrogen inputs and outputs is the nitrogen surplus for that system, and the outputs divided by the inputs is the nitrogen use efficiency. See **Figure 7** for a conceptual diagram showing how the systems connect.

Table 4. Nitrogen performance indicators for the UNH ODRF in 2014. The nitrogen performance indicators are shown for the whole farm, crop, and animal scales on a total basis and per unit area. The land area (70 ha) includes only the actively managed farm fields (pasture and baleage fields) but not the woodlot, which is not grazed or used for forage production.

Budget scale	Indicator	Total farm	Per unit area
Whole farm	N inputs	7,080 kg N/yr	101 kg N/ha/yr
	N outputs	1,740 kg N/yr	25 kg N/ha/yr
	N surplus	5,340 kg N/yr	76 kg N/ha/yr
	NUE	25%	n/a
Crop	N inputs	12,290 kg N/yr	176 kg N/ha/yr
	N outputs	8,640 kg N/yr	123 kg N/ha/yr
	N surplus	3,650 kg N/yr	52 kg N/yr/yr
	NUE	70%	n/a
Animal	N inputs	13,110 kg N/yr	187 kg N/ha/yr
	N outputs	1,740 kg N/yr	25 kg N/ha/yr
	N surplus	11,370 kg N/yr	162 kg N/ha/yr
	NUE	13%	n/a

3.2. Compost facility nitrogen budget

In 2014, there were 7 batches of compost feedstock loaded into the compost facility (**Table 5**). The average 2014 feedstock mixture was made up of 39% cow manure, 40% bedded pack, 20% waste baleage, and 1% wood chips. Each batch loaded to the facility was 107 m^3, but the bulk density, moisture content, and C:N ratios varied based on the specific feedstock mixture. The total wet weight of feedstock loaded in 2014 was 580 t, which contained 2.6 t N and 69 t C.

The only major input to the compost facility N budget is feedstock material (2,600 kg N, **Figure 9**). The largest N output from the compost facility is finished compost (2,300 kg N). The measured N outputs were exhaust gas (170 kg N) and leachate/condensate (90 kg N). The remaining N losses not measured could include passive emissions from piles while in the facility and gas fluxes not recorded during the sampling schedule.

Table 5. Characteristics of feedstock loaded into the UNH compost facility in 2014. The 2014 average characteristics and annual totals for feedstock material loaded are shown. *Source: Smith 2016*

Fiscal year	**2014**
Number of batches	7
Feedstock mixture	39% cow manure, 40% bedded pack, 20% waste baleage, 1% wood chips
Moisture content (%)	73%
N content (% DM)	1.7%
C content (% DM)	45%
C:N ratio	32:1
Bulk density (kg/m3)	774
Volume (m^3)	107
Total volume (m^3)	749
Total wet weight (t)	580
Total dry weight (t)	154
Total N content (t N)	2.6
Total C content (t C)	69

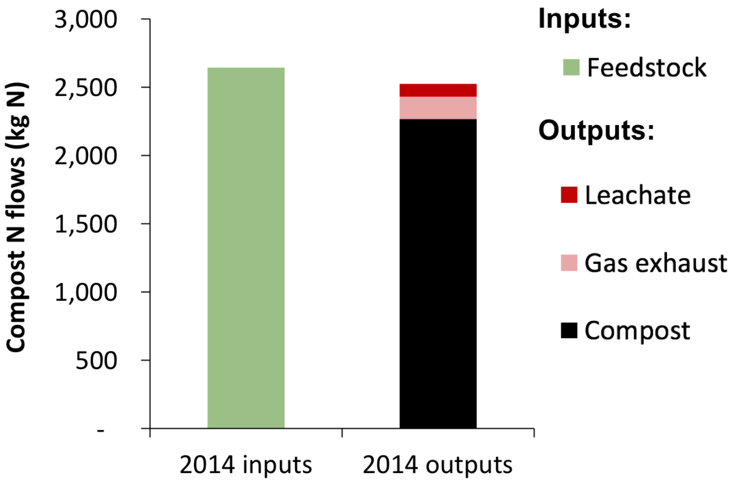

Figure 9. Nitrogen budget of the UNH compost facility in 2014. Feedstock inputs include all feedstock loaded (cow manure, bedded pack, waste baleage, and wood chips).

3.3. Scenario on exported compost

Two compost scenarios were analyzed: 1) Exporting 20% of compost (68 t wet weight, 450 kg N); and 2) Exporting 50% of compost (170 t wet weight, 1,130 kg N) (**Table 6, Figure 10**). The 20% compost scenario is more conservative because it accounts for a small percentage (12%) of the crop N surplus, and there would still be sufficient N for pasture and baleage production. The 50% compost export scenario is a more ambitious scenario (31% of crop N surplus) to assess the effect of a larger N export on the farm N budget. The exported compost would not be applied to the crop fields and would be a farm output, which affects the whole farm and crop N performance indicators. The animal N performance indicators are unaffected.

Table 6. Compost export scenario results for the UNH ODRF nitrogen budget. Whole farm and crop N performance indicators showing the effect of exporting 20% and 50% of the compost. The calculation for each N budget scenario is shown; the final results are in bold.

		Scenario: 20% compost export		**Scenario: 50% compost export**	
Budget scale	**Indicator**	**Total farm** kg N/yr	**Per unit area** kg N/ha/yr	**Total farm** kg N/yr	**Per unit area** kg N/ha/yr
Whole farm	N inputs	**7,080**	**101**	**7,080**	**101**
	N outputs	1,740 Compost: +450 **2,190**	25 Compost: +6 **31**	1,740 Compost: +1,130 **2,870**	25 Compost: +16 **41**
	N surplus	5,340 Compost: -450 **4,890**	76 Compost: -6 **70**	5,340 Compost: -1,130 **4,210**	76 Compost: -16 60
	NUE	25% Compost: +6% **31%**	n/a	25% Compost: +16% **41%**	n/a
Crop system	N inputs	12,290 Compost: -450 **11,840**	176 Compost: -9 **167**	12,290 Compost: -1,130 **11,160**	176 Compost: -16 **160**
	N outputs	**8,640**	**123**	**8,640**	**123**
	N surplus	3,650 Compost: -450 **3,200**	52 Compost: -6 **46**	3,650 Compost: -1,130 **2,520**	52 Compost: -16 **36**
	NUE	70% Compost: +3% **73%**	n/a	70% Compost: +7% **77%**	n/a

With 20% of compost exported, the whole farm N surplus decreased by 450 kg N to 4,890 kg N, and the whole farm NUE increased by 6% to 31% (**Table 6, Figure 10**). The crop system N surplus likewise decreased by 450 kg N to 3,200 kg N, and the crop NUE increased by 3% to 73%. With 50% of compost exported, the whole farm N surplus decreased by 1,130 kg N to 4,210 kg N, and the whole farm NUE increased by 16% to 41%. The crop system N surplus likewise decreased by 1,130 kg N to 2,520 kg N, and the crop NUE increased by 7% to 77%.

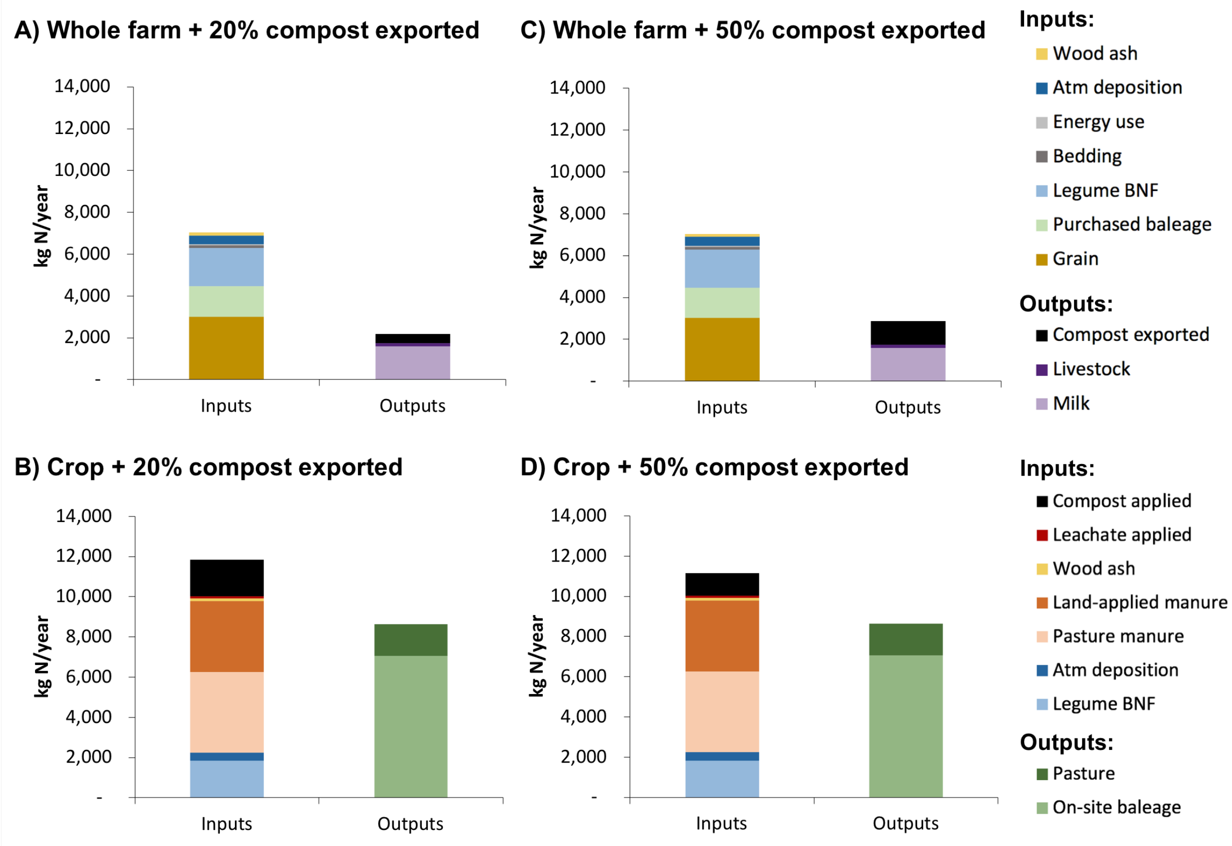

Figure 10. Compost export scenarios for the UNH ODRF in 2014. The figures show the effect of exporting the compost produced on-site at two levels: 20% compost export for A) the whole farm budget and B) the crop system budget, and 50% compost export for C) the whole farm budget and D) the crop system budget.

3.4. Dairy nitrogen budgets from the literature

Dairy farm N performance indicators were compiled by de Klein et al. (2017) (**Table 7**). A total of 17 studies were compiled, which include over 250 dairy farm N budgets at different scales. There was variation in both the countries represented (USA, Netherlands, Chile, Australia, Denmark, Ireland, New Zealand) and the types of farming systems (commercial and research; conventional and grazed). However, only 1 study in this data set focused on an organic dairy farm (Dalgaard et al. 1998).

The literature dairy farm crop NUE ranged from 16-91% (average: 61%), and the crop N surplus ranged from 25-229 kg N/ha/yr (average: 129 kg N/ha/yr). The animal NUE ranged from 15-36% (average: 25%), and the animal N surplus ranged from 110-308 kg N/ha/yr (average: 210 kg N/ha/yr). The whole farm NUE ranged from 8-56% (average: 32%), and the whole farm N surplus ranged from 40-700 kg N/ha/yr (average: 236 kg N/ha/yr). The grazed and organic systems typically had both lower N surpluses and lower NUE.

Table 7. Dairy farm nitrogen budgets from the literature and the UNH ODRF. Farm nitrogen budgets are shown at three scales: Crop, animal, and whole farm. The nitrogen use efficiency (NUE), nitrogen surplus (kg N/ha/yr), country, farming method (e.g., conventional, commercial, grazed), and reference are given. The UNH ODRF nitrogen budget results are shown in bold. *Source: Updated from de Klein et al. 2017, reproduced with permission.*

	NUE (%)	**N surplus** (kg N/ha/yr**)**	**Country; brief description**	**Reference**
Crop NUE	16-57	n/a	USA; NUE from manure	Beegle et al. 2008
	59-77	85-184	Netherlands; research farm	Aarts et al. 2000
	56-91	25-229	Netherlands; 16 commercial farms	Oenema et al. 2012
	61-71	112-136	Chile; three grazed systems	Núñez et al. 2010
	70	**52**	**USA; organic dairy research farm**	***This study***
	73	**46**	**USA; organic dairy research farm + 20% compost exported**	***This study***
	77	**36**	**USA; organic dairy research farm + 50% compost exported**	***This study***

Animal NUE	15-35	120-320	Australia; 17 commercial grazed systems	Gourley et al. 2012a
	17-34	110-125	USA; 12 commercial grazed and confinement systems	Gourley et al. 2012a
	22-24	275-308	Netherlands; research farm	Aarts et al. 2000
	21-36	n/a	USA; commercial dairy herds	Chase 2004
	22-27	n/a	Netherlands; 16 commercial farms	Oenema et al. 2012
	18-33	n/a	USA; 54 commercial dairy farms	Powell et al. 2006
	13	**162**	**USA; organic dairy research farm**	***This study***
	13	**162**	**USA; organic dairy research farm + 20% compost exported**	***This study***
	13	**162**	**USA; organic dairy research farm + 50% compost exported**	***This study***
Whole-farm NUE	8-55	40-700	Australia; commercial dairy systems	Ovens et al. 2008
	14-50	47-601	Australia; 43 commercial grazed systems	Gourley et al. 2012b
	17-42	121-358	EU; high and low N dairy systems	Castillo et al. 2000
	16	160-380	Denmark; 14 conventional dairy farms	Dalgaard et al. 1998
	18-20	231-277	Ireland; 21 commercial dairy farms	Treacy et al. 2008
	21-39	124-259	New Zealand; commercial grazed systems in five catchments	Monaghan and de Klein 2014
	22-36	174-275	Ireland; commercial dairy farm	Huebsch et al. 2013
	21	85-155	Denmark; 16 organic dairy farms	Dalgaard et al. 1998
	27-35	140-198	Netherlands; research farm	Aarts et al. 2000
	24-42	116-409	New Zealand; four grazed farmlet systems	Ledgard et al. 1999
	25-64	140-314	USA; eight commercial dairy farms	Hristov et al. 2006
	29-42	98-252	Netherlands; 16 commercial farms	Oenema et al. 2012
	35-56	n/a	USA; high and low stocking rates	Powell et al. 2010
	25	**76**	**USA; organic dairy research farm**	***This study***
	31	**70**	**USA; organic dairy research farm + 20% compost exported**	***This study***
	41	**60**	**USA; organic dairy research farm + 50% compost exported**	***This study***

The ODRF N performance indicators compared favorably to dairy farms from the literature (**Table 7**). In general, the ODRF NUE indicators were lower (~20% less) than dairy farm averages, but its N surplus per unit area was much lower (~70% less). This means that although the ODRF has a lower N efficiency, its N losses are released over a larger area. The crop NUE (70%) was greater than the literature average (61%) and the crop N surplus (52 kg N/ha/yr) was less than half the literature average (129 kg N/ha/yr). The ODRF animal NUE (13%) was lower than all reported literature values (15-36%), but its animal N surplus (162 kg N/ha/yr) was 25% less than the literature average (210 N/ha/yr). The ODRF whole farm NUE (25%) was less than the average of the literature values (32%), but its whole farm N surplus (76 kg N/ha/yr) was also much lower than the dairy farm average of 236 kg N/ha/yr.

The compost export scenarios for the ODRF increased both the crop and whole farm NUE and decreased the crop and whole farm N surplus (**Table 7**). The 20% compost export scenario increases the ODRF whole farm NUE to 31%, which is nearly the average of the reported literature values (32%). The 50% compost export scenario further increases the whole farm NUE to 41%, which is on the high end of the dairy literature values and exceeds the dairy farm NUE target range.

When plotted on the dairy farm NUE target conceptual diagram, the ODRF falls within the target NUE range of 20-40% (**Figure 11**). The dairy farm NUE target range aims to meet adequate dairy farm N efficiencies (i.e., minimum of 20%) while avoiding the soil mining that typically occurs with higher dairy farm N efficiencies (i.e., NUE greater than 40%). Higher NUE values are not currently possible without mining N from the dairy farm system due to the inherent efficiencies of animal production systems (e.g., N uptake rates for pasture, the conversion of feed into milk).

Although the ODRF falls within the target NUE range, its N outputs (25 kg N/ha/yr) are less than the commercial N product target of 80 kg N/ha/yr. With its larger land area per unit product than the commercial/conventional farms, the ODRF also had one of the lowest N surpluses of any of the plotted dairy farms. The compost export scenarios put the ODRF closer to the production target, although the change is small relative to the range of other farms shown on the diagram.

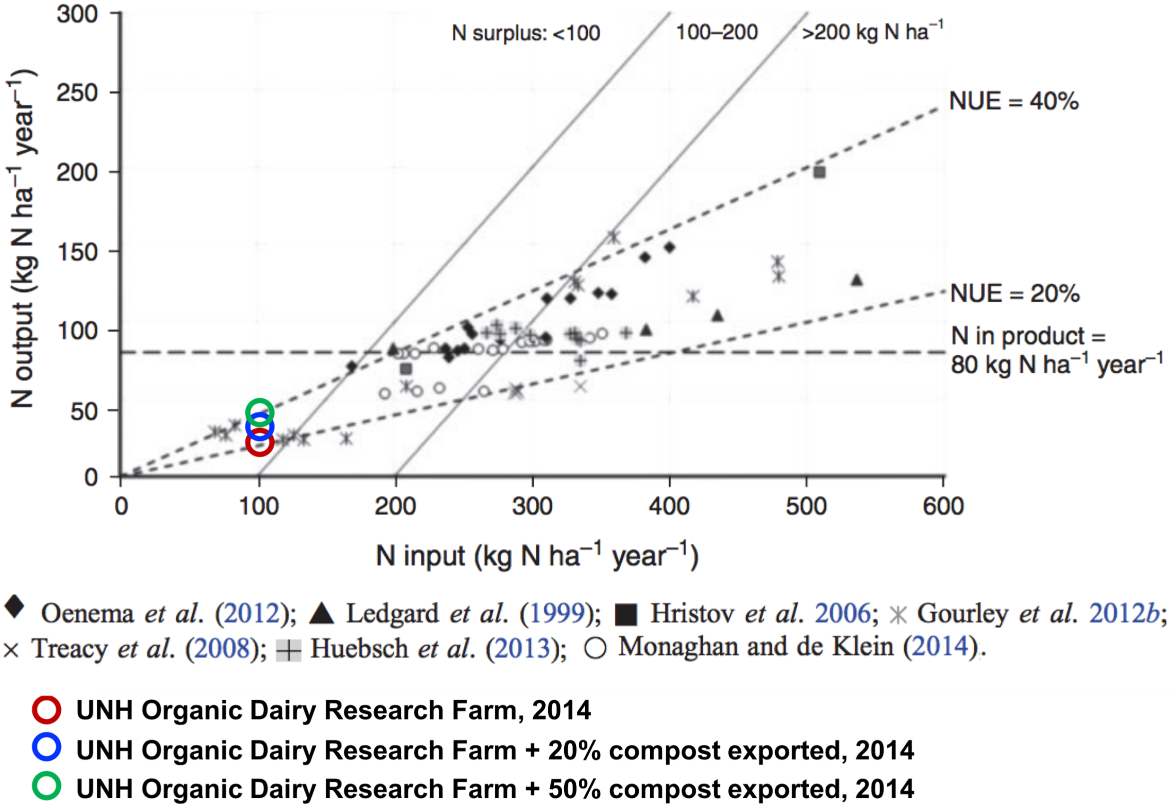

Figure 11. Dairy farm NUE conceptual diagram with the UNH ODRF. The UNH ODRF farm nitrogen budget is plotted with its baseline nitrogen budget in 2014 (red open circle) and two scenarios where compost is exported at rates of 20% (blue open circle) and 50% (green open circle). Farms are plotted based on their nitrogen inputs and outputs. The horizontal dashed line indicates a production target for commercial-scale dairy farms (80 kg N/ha/yr). The diagonal dashed lines show a target range for nitrogen use efficiency (NUE). The diagonal solid lines show the whole farm nitrogen surplus. The ideal target range is above the production target, between the NUE lines, and at the lower end for N surplus. See Table 7 for details on the plotted dairy farms from the literature. *Source: Updated from de Klein et al. 2017, reproduced with permission.*

4. Discussion

4.1. Farm nitrogen surplus: Potential losses and sinks

4.1.1. Crop nitrogen surplus

The crop system N surplus of 3,650 kg N/yr (52 kg N/ha/yr) has several potential loss or storage pathways (**Figure 7**). Loss pathways include leaching, runoff, volatilization, erosion, and denitrification. Because N is a very leaky element, it is likely that some of the crop N surplus is going to all of these pathways (Galloway et al. 2003; Sutton et al. 2011; Yang et al. 2011). With the exception of denitrification, all of these N loss pathways could contribute to environmental impacts in the short and/or long term. Long term monitoring studies and literature values help explain the likely fate of the crop N surplus.

Long term stream and groundwater sampling records at the ODRF have recorded N concentrations on the property and exiting the property (UNH Water Quality Analysis Laboratory 2016). Groundwater N concentrations in 2014 are highest near the barn (20-50 mg N/L), old outdoor compost piles (10-15 mg N/L), and an old pig lagoon (10-20 mg N/L). Most of these high N concentrations are due to N inputs from years earlier when the farm property was under different management, and these N concentrations are declining over time (UNH Water Quality Analysis Laboratory 2016). The pasture and downstream N concentrations exiting the property in 2014 are all within drinking water standards (1-10 mg N/L). This difference could be due to groundwater N losses, denitrification, or sedimentation on the property. Because of the mobility of N, it is likely that some of the N surplus is going towards all of these loss pathways. However, more research into the groundwater hydrology of the ODRF is needed to identify the magnitude of N loss to the different water pathways.

N volatilization (NH_3, N_2O) occurs after soil amendments are applied, and volatilization is especially high for less stable soil amendments like manure and slurry. N_2O losses make up a small percentage of manure N volatilization, but they are important to consider in a greenhouse gas budget because of the high global warming potential of N_2O (Byrne et al. 2007). NH_3 losses from fecal matter and urine excreted during grazing can range from 0-28% of the N excreted depending on the rate of manure application, the weather, and the soil cation exchange capacity (Bussink & Oenema 1998). When manure is collected and applied to the land, 1-13% of N can volatilize as NH_3 after application (Bussink & Oenema 1998). NH_3 volatilization losses can also occur during manure storage, ranging from 0-20% N. Taken together, these estimates suggest that up to 30% of applied manure N could be immediately lost and unavailable for uptake. Yang et al. (2011) found that 26% of applied manure N was lost during storage and land application, and the remaining manure N was immediately available to crops (36%) or stored and available the following year (39%).

Findings from other studies suggest that N volatilization losses from land application of manure could be as high as 30% (Bussink & Oenema 1998; Yang et al. 2011). If 30% of applied manure N at the ODRF (2,260 kg N or 60% of the crop N surplus) were immediately lost and removed from the crop system N inputs, then the crop NUE would increase to 86% and the crop N surplus would decrease to 1,390 kg N/yr. This lower crop N surplus would limit the amount of compost N available for export, but it would still support the export of 20% of finished compost (450 kg N).

The crop N surplus could also be stored on the farm property in the soil and through biomass uptake. Organic practices and specifically compost amendments have been shown to build up soil organic matter (Leifeld et al. 2009; Ryals et al. 2014), which would also increase N storage and improve the productivity of the fields in future years. For example, Ryals et al. (2014) found an increase of 9-30% in soil N content following organic matter amendments. A detailed soil quality survey was conducted at the ODRF in 2006 by the Maine USDA ARS office, which was shortly after the ODRF was established in 2005. A follow-up soil survey could determine whether the soil organic matter and N storage have increased since the farm transitioned to organic management practices 13 years ago. This long term record and comparison would be very valuable in the literature to show how organic dairy farm grazing practices can affect soil organic matter over the course of a decade.

One solution for more effectively and efficiently using the available N inputs at the farm is to compost more of the collected manure in the ASP heat recovery compost facility. This would create a more stable soil amendment that would make more N available for pasture productivity.

4.1.2. Animal nitrogen surplus

The animal N surplus is the difference between feed intake and product (i.e., milk, meat). Most of the animal N surplus (11,370 kg N/yr, 162 kg N/ha/yr) goes towards manure, which then either is applied to the fields or used as a feedstock in the compost facility (**Figure 7**). However, manure loss pathways account for only 9,900 kg N of the animal N surplus. Some of the remaining 1,470 kg N is due to cow weight gain (200 kg N accumulated by growing female calves) and feed refusals, which is the feed not consumed by cows (220 kg N waste baleage was used as compost feedstock). The remaining 1,000 kg N could be due to additional feed refusals and/or uncertainty in the calculations.

Feed refusals were likely higher than the 220 kg N waste baleage used as compost feedstock, and additional refusals could have been disposed of. The feed logs for the milking cows indicated refusals 60% of days, with over 25% of days having ‘medium’ or ‘a lot’ of refusals. These refusals would go towards the heifers, but the heifers could also have refused this extra feed. Records were not kept on heifer feed refusals. The herd was fed a daily weighted average of 1,450 kg DM feed, but the dry matter demand for the herd was actually only 1,130 kg DM feed (UNH ODRF 2014c). This extra feed would either be consumed and go towards manure, or it would be refused and would be wasted.

Uncertainty in the animal feed and manure production calculations could also explain the remaining 1,000 kg N in the balance (see **Section 4.5** for more information on data uncertainty and a sensitivity analysis). The calculated animal feed N could have been an overestimate. There is higher certainty in the feed grain calculations, which are based on the grain weight purchased, grain weight fed to cows, and the reported protein content. There is more uncertainty in the baleage and pasture intake calculations. The baleage intake calculations are based on the daily milking cow feed logs and average feeding baleage feeding rates to non-milking cows. On days when milking cow feed refusals were higher, the non-milking cows may not have received additional fresh baleage beyond the refusals. The pasture intake was calculated by difference based on the daily dry matter demand from the NRC and reported grain and baleage intake rates by cow type. If the grain or baleage were underreported on the DMI worksheets, then the cows may not have needed as much pasture as calculated. Finally, underestimating the manure production calculations could also explain the remaining animal N surplus. The manure production was calculated based on average rates of fecal and urine N produced per Jersey cow per day. If the feeding rate was higher than average, then the cows would have produced more manure than calculated.

4.1.3. Whole farm nitrogen surplus

The whole farm N surplus (5,340 kg N/yr, 76 kg N/ha/yr) can be explained by the crop and animal systems within the farm (**Figure 7**). This whole farm N surplus is almost fully accounted for by the N surplus from the crop system (3,600 kg N/yr), which remains on the property or is lost to the environment, and the unexplained animal N surplus (1,000 kg N/yr). The remainder of the whole farm N surplus (700 kg N/yr) is contained in the significant N cycling that occurs between the crop, animal, and compost systems within the ODRF property.

4.2. Dairy farm nitrogen use efficiency and land area

Although the ODRF has a lower NUE (25%) than the literature average for dairy farms (32%), its N surplus is much lower and released over a larger area (76 kg N/ha/yr) than the literature average (236 kg N/ha/yr). Most dairy farm NUE values from the literature are for conventional farms (de Klein et al. 2017), which emphasize production and have a higher concentration of animals per unit land area. This concentration of animals means that a large amount of N can be generated in agricultural by-products—especially manure—which can have significant environmental consequences when released over a small area of land. Although the ODRF has a lower total N production and NUE, its environmental impacts are likely much lower because its N surplus is released over a larger area of land, where it is put towards pasture and baleage production.

4.3. ASP heat recovery compost facility: Compost export scenario

4.3.1. Improved nitrogen efficiency

A farm can improve its NUE and reduce its N surplus by exporting finished compost from the property. However, it is important for a farm considering exporting finished compost to first assess its N budget to ensure its on-site crop production will not be impacted.

When exporting only 20% of its compost, the ODRF's whole farm NUE increased from 25% to 31% and its N surplus decreased from 76 to 70 kg N/ha/yr. When exporting 50% of its compost, the ODRF's whole farm NUE increased even further to 41% and its N surplus decreased to 60 kg N/ha/yr. This substantial increase in NUE could not be easily achieved by other on-farm methods, such as reducing feeding or increasing on-site baleage production. This method of improving N efficiency can also be profitable if the compost is sold. However, exporting compost is only viable when a farm has enough N for its pasture and baleage production. Otherwise, the fields will begin depleting N from the soil reserves, which will reduce the overall farm's productivity and could require more nutrients (e.g., fertilizer) to be imported (de Klein et al. 2017). The 50% compost export scenario may remove too much of the farm N inputs and lead to soil mining, so a more conservative approach is advisable.

Exporting compost from the ODRF property could also free up space in the compost facility for loading more manure. The composting process must last at least 21 days to meet commercial guidelines, but compost often stays in the ODRF facility several months before it is unloaded and spread on the fields. If some of this compost were instead exported from the property after 21 days, then more manure from the ODRF could be processed in the compost facility. In the spring, manure accumulates too quickly at the ODRF for loading into the compost facility and is instead land-applied. Because manure is a less stable soil amendment, more N is lost to the environment from manure application. Processing more manure in the compost facility would generate a more stable soil amendment, ensuring that more of the applied N is available for forage production.

4.3.2. Potential for additional revenue from compost

The construction of an ASP heat recovery compost facility can be costly, but its expense can be balanced by selling compost, offsetting energy inputs through onsite production, and reducing labor requirements (Smith et al. 2017). Because the compost piles are static and have forced aeration, they do not need to be turned, which is one of the largest labor costs for composting.

In addition to improving farm N efficiency, selling compost can also create a new stream of revenue for a farm. For example, in the seacoast region of New Hampshire, finished compost sells for \$20-\$35 per m^3 (personal communication, Matthew Smith, November 2018). For large conventional dairy farms with less land area, the substantial amount of manure produced could be converted into large quantities of value-added compost. The compost is value-added because it is a more stable soil amendment that can be transported longer distances than manure due its reduced water weight.

4.3.3. Additional pathways for nitrogen pollution avoidance

The ASP heat recovery method of composting also avoids some N pollution pathways due to its enclosed design. Most of the composting by-products leaving the facility go through specific outlet points, which allows for them to be controlled. Common N losses from composting are exhaust gas emissions, leaching, and runoff.

The exhaust gas in a compost facility with forced aeration is piped through a single exit point where a biofilter can be installed. A biofilter is a simple waste capture technology that is made up of layers of wood chips and finished compost (Pagans et al. 2005). When the exhaust gas is routed through a biofilter, the biofilter takes up NH_3 through microbial activity and NH_3 condensing out into solution due to the high temperature and humidity of the incoming exhaust gas. Lab-scale studies have observed biofilter NH_3 removal rates over 90% (Pagans et al. 2005), and preliminary studies at the commercial-scale ODRF compost facility biofilter have found NH_3 removal rates exceeding 80% (Williamson et al. in prep).

The leachate and condensate from compost piles at the ODRF collects in a storage tank. Because it is collected, the leachate and condensate are not immediately lost to the environment as they would be in a windrow composting system or from direct land application of manure. The leachate and condensate can therefore be managed with wastewater treatment or used as a nutrient-rich soil amendment.

4.4. Other manure management methods

Common dairy manure management methods include holding lagoons, land application of slurry, anaerobic digestion, and windrow composting. Of these methods, the only ones that would create a value-added product that could be exported from the farm are windrow composting and anaerobic digestion. However, the infrastructure required for anaerobic digestion can require a large capital investment (Moser et al. 1998). Windrow composting has larger labor needs for turning the piles, and depending on the pile design, it can lead to gas volatilization and leaching. Windrow composting also requires more time than ASP heat recovery composting, and the windrow compost piles do not always achieve the minimum temperatures necessary for selling the compost, especially during the winter when the piles are exposed to freezing temperatures. Holding lagoons promote some denitrification, but they are primarily a waste storage/disposal method and they do not improve the N efficiency of a farm. Land-applying slurry does use the N as a soil amendment, but the volatilization and leaching losses from liquid slurry application are substantial since the slurry is in liquid form (Bussink & Oenema 1998). ASP heat recovery composting is unique among manure management methods in that it has lower labor and infrastructure costs, captures environmental losses, generates a value-added product, and recovers heat from composting.

4.5. Data quality, uncertainty, and sensitivity analysis

There is a high level of certainty for most of the major farm N flows. Several of the inputs and outputs were documented by purchase and sales records, and the protein content was often reported in quality analysis reports (e.g., grain purchases, milk sales, livestock sales, wood ash purchases, bedding purchases). The atmospheric deposition was calculated using a deposition rate from a monitoring facility just 5 km away. The energy use was recorded from farm records, and the N emissions factors were regional or national averages. The amount of compost produced was based on detailed records of feedstock materials and measured N losses.

There are four large farm N flows with a lower level of certainty: Legume BNF, purchased baleage, pasture intake, and pasture and land-applied manured. A sensitivity analysis was conducted for each of these N flows.

The legume BNF estimate had a moderate level of certainty, but its result has only a moderate effect on the farm NUE and N surplus. The legume coverage was obtained from a vegetation survey at the ODRF that assessed the percent coverage by vegetation type (Antaya 2016). However, the legume coverage could have changed since the vegetation survey was conducted. Literature values were used for N fixation rates, but there is a large range in N fixation rates depending on the N availability in a system: Legumes will fix less N if there are stores of N available in the soil. If the total legume BNF were decreased by 50%, then the whole farm NUE would increase from 25% to 28%, and the crop NUE would increase from 71% to 77%. The NUE metrics would decrease by the same amount if legume BNF were instead increased by 50%.

Because purchased baleage records were not available for 2014, the purchased baleage was calculated by difference between reported feeding rates (milking cow feed logs, farm records) and the on-site production of baleage. However, purchased baleage could have been overestimated if large amounts of milking cow feed refusals were given to heifers, which could have reduced baleage requirements for heifers. If the purchased baleage were reduced by 50%, then the animal NUE would increase from 13% to 14% and the whole farm NUE would increase from 25% to 27%. Although there is uncertainty in this variable, the farm system N budgets are not greatly affected by it.

Pasture intake was calculated based on daily reported rates by cow type in DMI worksheets and the amount of time the cows spent grazing. The rates reported in the DMI worksheets were calculated by difference from the NRC-reported dry matter demand and average feeding rates of grain and baleage. Because the farm balance indicates that the cows may have received more feed than needed, they also might not have consumed as much pasture as calculated in the DMI worksheets. If the pasture intake were decreased by 50%, then crop NUE would decrease from 71% to 64% and the animal NUE would increase from 13% to 14%. Although the crop NUE change is fairly substantial, it is unlikely that the pasture intake was overestimated by 50%. In addition, it should be noted that any overestimate of baleage and pasture intake would be linked through the DMI worksheets, and any overestimate in intake would be distributed across both feed types.

The amount of manure production was based on the size of the herd and average daily fecal and urine N production rates of Jersey cows. However, these rates can vary based on the type and amount of feeding. If the pasture- and land-applied manure were increased by 13% (or 1,000 kg N, which is the amount of the animal N surplus that is unexplained), then the crop NUE would decrease from 71% to 65%.

This sensitivity analysis shows that the overall findings of the farm N budget would not be affected by substantial shifts in variables that have a lower level of certainty. The crop NUE was the most affected indicator by changes to the variables with lower levels of certainty. With the exception of legume BNF, the expected direction of the uncertainty (higher levels of N inputs and lower levels of pasture/baleage intake) all made the crop NUE lower, suggesting that there may be even more surplus N applied to crop fields than estimated in this study. Given this, it is clear that there is enough N available to the crop system to allow for 20% of the finished compost to be exported from the property.

4.6. Applicability to other farms

Exporting compost is an effective strategy for improving a farm's NUE and reducing its N surplus. However, a farm should first assess its N balance to confirm that it has surplus N to export. Otherwise, exporting compost could lead to soil mining of N and reduced pasture productivity. ASP heat recovery composting is a less expensive alternative than other manure management methods that generate a saleable, value-added product (e.g., anaerobic digestion).

5. Summary and next steps

Through its organic practices and compost facility, the UNH ODRF cycles substantially more N on its property (e.g., harvested baleage, land application of manure) than it imports (e.g., feed grain). The UNH ODRF has a lower whole farm NUE (25%) than other dairy farms (32% average). However, its N surplus per unit area is also much lower (66% less) than the average dairy N surplus, meaning that any N losses are released over a much larger area and are less likely to contribute to local negative environmental impacts. The farm NUE could be improved by exporting compost from the farm. Exporting just 20% of the finished compost would increase the whole farm NUE to 31% (nearly the dairy farm average) while still retaining enough N for the farm's N balance and pasture productivity. Producing and selling compost with an ASP heat recovery compost facility is a viable strategy for both improving a farm's N efficiency and adding an additional revenue stream. However, a farm's N balance must first be assessed to confirm that the farm will still have adequate N stores for pasture productivity.

To better understand the UNH ODRF's N budget and how the compost facility affects it, two key areas of future research should be explored. First, the N loss pathways not quantified in this study (leaching, runoff, volatilization, denitrification, storage) should be characterized to assess the balance between farm N storage and losses. Second, other potential pollution management strategies associated with ASP heat recovery composting (biofilter, leachate/condensate collection, fossil fuel replacement, soil C sequestration) should be studied together to determine their cumulative effect on reducing environmental pollution.

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Appendix A: Data for the UNH Organic Dairy Research Farm nitrogen budget in 2014

Table A1. Cow population by type at the UNH ODRF in 2014. Herd average across the year. Source: UNH ODRF 2014c

Cow type	Milking cows	Dry cows	Heifers	Yearling heifers	Calves	Total*
Age	18+ months	18+ months	6-12 months	12-18 months	0-4 months	n/a
Count	46	14	10	16	21	107

*Total population is an overestimate because it includes female calves born on the farm in 2014.

Table A2. Grain fed to cattle at the UNH ODRF in 2014. The type of grain, total weight, protein content, and vendor are shown. Source: UNH BSC 2014

	Weight (t)	% Crude protein, as fed	Company
UNH Organic Meal [a]	114.6	14.5%	Poulin Grain
Organic 20% calf starter	4.6	21%	Green Mountain Feeds
Organic dry cow pellets	3.1	21%	Green Mountain Feeds
Organic 16% high energy dairy pellet	4.4	16%	Green Mountain Feeds

[a] UNH Organic Meal mix varies throughout the year. The protein content is a weighted average of the purchases in 2014.

Table A3. On-site and purchased baleage at the UNH ODRF in 2014. The weight and quality information are given for the baleage. Source: UNH ODRF 2014b, 2018a

	Number of bales	Weight per bale (kg)	Wet weight (t)	Dry matter %	Dry weight (t)	Crop protein, DM (%)
On-site production [a]	778	640	497	55%	272	16%
Purchased [b]	160	640	137	55%	123	16%

[a] On-site baleage quality information is a weighted average of all cuts at the UNH Organic Dairy Farm in 2014.

[b] Purchased baleage records were not available for 2014, so it was assumed that the quality of the purchased baleage was equivalent to the on-site production.

Table A4. Dry Matter Intake (DMI) worksheet summary for the UNH ODRF in 2014.
Average daily DMI during the summer by cow type. Source: UNH ODRF 2014c

	Dry cows	**Heifers 12-18 mo.**	**Heifers 6-12 mo.**	**Lactating cows**
Date	1-Jun	1-Jun	1-Jun	1-Jun
Number of animals	14	16	10	46
Average weight (lb)	800	650	500	950
DMD (Dry Matter Demand, lb)	20.7	16.9	13.4	37.5
Other feed source 1	None	Baleage	None	Baleage
lb, as fed		20.0		18.0
x % DM of feed source		53.0%		45.3%
= DMI, lb	0.0	10.6	0.0	8.2
Other feed source 2				Grain
lb, as fed				14.0
x % DM of feed source				90.0%
= DMI, lb	0.0	0.0	0.0	12.6
Other feed source 3				
lb, as fed				
x % DM of feed source				
= DMI, lb	0.0	0.0	0.0	0.0
Total DMI from feed sources, lb	**0.0**	**10.6**	**0.0**	**20.8**
% DMI from feed sources	0.0%	62.7%	0.0%	55.4%
Pasture DMI, lb	**20.7**	**6.3**	**13.4**	**16.7**
% DMI from pastures	100.0%	37.3%	100.0%	44.6%

Table A5. Nitrogen content of pasture grasses and legumes. The types of grasses and legumes included were those observed on the UNH Organic Dairy Research Farm. Source: IPNI 2012.

Type of grass	**N (lb/unit)**	**Unit (ton DM)**	**%N**
Bluegrass (DM)	30	2000	1.5%
Fescue (DM)	37	2000	1.9%
Orchardgrass (DM)	36	2000	1.8%
Grass average N content			**1.7%**
Type of legume	**N (lb/unit)**	**Unit (ton DM)**	**%N**
Alfalfa	51	2000	2.6%
Red clover	45	2000	2.3%
Legume average N content			**2.4%**

Table A6. Vegetation survey for the UNH ODRF. The vegetation types were categorized into % dry matter by vegetation type: Grass, legume, weed, dead. Source: Antaya 2016.

	Period 1	Period 2	Period 3	Period 4	Average
	July	August	September	October	July - October
Grass	63.77%	51.18%	66.90%	47.25%	57.28%
Legume	24.84%	17.69%	9.64%	13.85%	**16.51%**
Weed	9.26%	15.44%	9.60%	11.93%	11.56%
Dead	1.08%	11.15%	13.86%	26.97%	13.27%

Table A7. Biological nitrogen-fixation rates of legumes. The legumes included were those observed at the UNH ODRF. The maximum N-fixation rate is an average of reported maximum values in a range, the minimum N-fixation rate is an average of reported minimum values in a range, and the average is an overall average of all data points.

Species name	**Common name**	**Type**	**Nitrogen-fixation rate (kg N ha^{-1} yr^{-1})**			**Source**	**Number of studies**
			Avg.	**Max.**	**Min.**		
Medicago sativa [b]	Alfalfa	Perennial	169	234	88	1,2,3,4,5	13
Trifolium repens [d]	White clover	Perennial	155	240	67	1,2,3,4,5	22

[b] *Studies took place in the US (New York, Kentucky, Minnesota, Alaska), Australia, Austria, Canada, and Sweden.*

[d] *Studies took place in the US (Georgia, Kentucky, Minnesota), Denmark, Netherlands, New Zealand, Switzerland, and the UK.*

Sources: (1) Ball et al. 2007; (2) Brady 1982; (3) Carlsson & Huss-Dannell 2003; (4) Havlin et al. 1999; (5) Johnson et al. 1997

Table A8. Atmospheric nitrogen deposition rates in 2014 at Thompson Farm, which is located 5 km from the UNH Organic Dairy Research Farm. Source: PREP 2017.

Wet total dissolved N dep	**Dry deposition**	**Total deposition (wet + dry)**	**ODRF land area**	**Atm dep**
kg N/ha/yr	*kg N/ha/yr*	*kg N/ha/yr*	*ha*	***kg N/yr***
3.76	2.18	5.94	70	**416**

Table A9. Manure nitrogen production rates for Jersey cows. Source: Knowlton et al. 2010.

Cow type	Wet feces excretion (kg/cow/day)	Urine excretion (kg/cow/day)	Fecal N (g N/cow/day)	Urine N (g N/cow/day)
Adult Jersey	33.6	16.3	162	161

Table A10. Milk production and quality at the UNH ODRF in 2014. Source: UNH ODRF 2018b

	Milk production (kg)	% Butter fat	% Protein	Energy corrected milk (kg)
January	23,405	4.85%	3.75%	28,387
February	22,449			27,228
March	26,595			32,256
April	28,833			34,970
May	28,136			34,125
June	20,710			25,118
July	19,498			23,648
August	19,782			23,992
September	17,008			20,629
October	16,432			19,930
November	18,515			22,456
December	22,668			27,493
Total	**264,031**			**320,233**

Table A11. Cows that left the herd at the UNH ODRF in 2014. Cows that died on the property were composted on the property and were not exports from the farm budget.

Type	Category	Count	Live animal weight (kg/cow) [a]	Live animal % N [b]
Adult jersey	Died	2	430	2.2%
Adult jersey	Culled	13	430	2.2%
Adult jersey	Sold	1	430	2.2%
Bull calf	Sold	23	30	2.9%

[a] Source: UNH ODRF 2014c

[b] Source: NRC 2003

Table A12. Bedding and wood ash imported to the UNH ODRF in 2014.

	Amount purchased	Source	Nitrogen content	Source
Bedding	135 t	UNH Business Services Center	0.09%	Rynk et al. 1992
Wood ash	100 t	UNH Business Services Center	0.15%	Risse & Gaskin 2002

Table A13. Fuel and electricity use at the UNH ODRF in 2014, and the nitrogen emissions factors for those fuel sources.

Type	Value	NOx emissions factors	N_2O emissions factors
Diesel [a]	3200 gallons	0.05146 kg NOx/gal	0.00026 kg N_2O/gal
Gasoline [a]	762 gallons	0.01477 kg NOx/gal	0.00062 kg N_2O/gal
Electricity [b]	97654 kWh	0.0002111 kg NOx/kwh	0.00001 kg N_2O/kwh

[a] Source: US EPA Inventory of US Greenhouse Emissions and Sinks: 1994-2014

[b] Source: US EPA eGrid 2014 database

Table A14. Compost feedstock batch characteristics at the UNH compost facility in 2014. Source: Personal communication, Matthew Smith, June 2018.

Loading Date	**Feedstock mixture, by volume***	**C:N Ratio**	**Moisture Content %**	**Bulk density** (kg/m^3)	**Volume** (m^3)
1/15/14	40% M, 40% BP, 20% WB	31.1	81%	951	107
1/17/14	40% M, 40% BP, 20% WB	31.1	81%	951	107
5/23/14	40% BP, 30%M, 10% WC, 20% WB	34.3	72%	693	107
5/23/14	40% M, 40% BP, 20% WB	31.1	65%	691	107
10/29/14	40% M, 40% BP, 20% WB	31.1	70%	712	107
11/20/14	40% M, 40% BP, 20% WB	31.1	70%	712	107
12/4/14	40% M, 40% BP, 20% WB	31.1	70%	712	107

*Where M = cow manure, BP = bedded pack, WB = waste baleage, and WC = wood chips

Table A15. Leachate pumping records and nitrogen and carbon content. Pumping records report the volume pumped each date the leachate/condensate tank was empty. Nitrogen and carbon contents were measured from leachate/condensate samples from the UNH compost facility.

Date tank was pumped	Leachate and condensate pumped [a] (liters)	Total dissolved nitrogen [b] (mg N/L)	Dissolved organic carbon [b] (mg C/L)
2/2/14	6,511	1,473	8,414
3/16/14	5,697		
4/5/14	7,571		
5/20/14	5,860		
6/2/15	7,650		
6/6/14	3,744		
6/16/14	5,209		
7/10/14	4,720		
8/25/14	6,511		
11/1/14	7,813		

[a] Source: Personal communication, Matthew Smith, June 2018.

[b] Source: Carbon and nitrogen content analysis completed by the William McDowell Lab at the University of New Hampshire.

CHAPTER 3: CHARACTERIZING COMPOST EXHAUST GAS FROM AERATED STATIC PILE HEAT RECOVERY COMPOSTING

Abstract

Aerated static pile (ASP) heat recovery composting, a novel method for processing agricultural wastes and by-products, has great potential for pollution capture and management due to its closed system and the addition of heat recovery. However, the gas emissions generated during the composting process—a potentially large pollution source—had not been well-quantified. In this study, we report carbon dioxide (CO_2), ammonia (NH_3), and methane (CH_4) gas concentrations from a commercial-scale ASP heat recovery compost facility under standard operating conditions and for three case studies (blocked airflow, delayed microbial activity, and low temperature). In all trials, the gas concentrations peaked early in the composting cycle during transition from the mesophilic to thermophilic phase, and then decreased steadily after the easily digestible material was decomposed. The case studies highlighted the importance of regular aeration and pile temperature management for reduced gas emissions. For example, higher rates of CO_2 and CH_4 emissions were observed with reduced aeration, and higher temperatures were associated with higher gas concentrations. Under standard operating conditions, exhaust gas temperature is a reasonable predictor of NH_3 emissions (R^2=0.75) and CO_2 emissions (R^2=0.55) with an exponential relationship. The ASP heat recovery composting approach can promote pollution control pathways that should be studied further. These include capturing emissions with the use of a biofilter; utilizing the exhaust gas to heat and fertilize a greenhouse; using recovered heat and therefore avoiding fossil fuel energy emissions; and increasing carbon sequestration through compost land application.

1. Introduction

The food production process generates agricultural waste at each step of the supply chain (Tilman et al. 2002). Agricultural wastes are the by-products of production and processing at agricultural operations; examples include crop residue, manure, spent bedding, waste feed, and food waste. When they are not used, these agricultural by-products can then contribute to nutrient pollution (e.g., nitrate runoff), poor air quality (e.g., ammonia emissions), and climate change (e.g., nitrous oxide, carbon dioxide emissions) (Tilman et al. 2002; Pelletier & Tyedmers 2010; Steinfeld & Gerber 2010). However, these agricultural wastes contain many of the same nutrients that are necessary inputs to food production. Methods for reusing and recycling these agricultural wastes can improve the efficiency and economics of food production and help avoid environmental pollution. Composting is one method that is gaining renewed popularity for processing and reusing agricultural by-products.

1.1. Aerated static pile heat recovery composting

Composting is the process by which organic materials are broken down by microorganisms into a stable, pathogen-free, humus-like product (de Bertoldi et al. 1983; Rynk et al. 1992; Haug 1993). Composting requires a carbon source (e.g., plant litter, crop residue, wood chips), a nitrogen source (e.g., animal manure, human waste, food waste), and microorganisms that then decompose the feedstocks (Ryckeboer et al. 2003; Misra et al. 2003). One advantage of compost over other soil amendments is that the nutrients are in a more stable form. The land application of compost enhances soil organic matter, which improves the soil's water retention capacity and reduces the rate of nutrient losses (Reeves 1997; Rivero et al. 2004). Land application of compost has also been shown to promote carbon storage in both agricultural soils (Ryals & Silver 2013) and urban environments (Renforth et al. 2011).

The composting process is controlled by oxygen availability, the material's moisture content, ratio of carbon to nitrogen, particle size, temperature, pH, bulk density, and the microorganisms present (de Bertoldi et al. 1983; Rynk et al. 1992; Pace et al. 1995; Ryckeboer et al. 2003).

The composting process encompasses three unique stages, which each have distinct microbial communities and physical characteristics (Cornell 2015; Peigné & Girardin 2004; Tuomela et al. 2000). The three stages are an initial mesophilic stage, a thermophilic stage, and a second mesophilic or maturation phase. During the initial mesophilic stage, temperatures increase and range from 20-40°C (Pace et al. 1995; Ryckeboer et al. 2003; Cornell 2015). The thermophilic phase starts a few hours to days after the initial mesophilic phase, and it is characterized by a spike in microbial activity and temperatures greater than 40°C. The high temperatures during the thermophilic phase are necessary to kill pathogens and seeds in the feedstock materials (Rynk et al. 1992). The second mesophilic or maturation phase is a period of decreasing microbial activity and temperature (<40°C) after the easily digestible feedstock material has been decomposed. Different communities of bacteria thrive in the mesophilic and thermophilic phases (Haug 1993).

Aerated static pile (ASP) composting, which pulls air through a stationary pile of composting feedstocks, can reduce the amount of time required to fully decompose the organic material (Rynk et al. 1992). Constructing a facility for ASP composting can be costly, but because the piles do not need to be turned, labor costs are lower throughout the composting process. An approach that could make ASP composting more economical and could lead to greater pollution reduction is the addition of heat recovery (Smith & Aber 2014, 2018). Heat generated by microbial activity during decomposition can be captured and used for other purposes, such as heating a building or greenhouse. ASP composting also makes the exhaust gas a point source for greenhouse gas and ammonia emissions, which can then be captured (e.g., with a biofilter) or utilized (e.g., for plant growth in a greenhouse) (Pagans et al. 2005). ASP heat recovery composting can require higher start-up costs for construction, but it could be more economical due to reduced labor costs and offset costs from heat recovery (Smith et al. 2017).

1.2. Compost exhaust gas

The decomposition of organic matter generates gases, heat, and water vapor, which are emitted in the compost exhaust gas. Compost pile and exhaust gas temperatures typically peak a few days into the composting process when microbial activity is at its peak during the thermophilic phase. Compost must reach a minimum temperature of 55°C to eliminate seeds and pathogens (Haug 1993). Smith and Aber (2018) found that the temperature in ASP composting piles at the University of New Hampshire (UNH) heat recovery composting facility ranged from 20-70°C. The temperature of the exhaust gas leaving the facility is lower than the pile temperatures because heat is captured from the exhaust gas in a heat exchanger and water tank.

Gases generated during the aerobic composting process include carbon dioxide (CO_2), ammonia (NH_3), methane (CH_4), and nitrous oxide (N_2O) (Beck-Friis et al. 2001; Fukumoto et al. 2003; Ahn et al. 2011; Szanto et al. 2007; Shen et al. 2011). The exhaust gas concentrations are driven by the quality and characteristics of the feedstock materials, oxygen availability, and microbial activity. CO_2 is a by-product of aerobic microbial respiration, and its emission is therefore a necessary by-product of decomposition and composting. The CO_2 and NH_3 concentrations in compost exhaust gas are much higher than ambient levels. Ambient CO_2 is currently 400 ppm (0.04%) and ambient NH_3 is 1 ppb (0.001 ppm) (NADP 2012; NOAA/ESRL 2016), whereas studies on compost exhaust gas at ASP composting facilities have found NH_3

concentrations in excess of 4,000 ppm (Pagans et al. 2005, 2006) and daily CO_2 fluxes over 17% CO_2-C of initial carbon (Beck-Friis et al. 2001).

Most studies on compost exhaust gas concentrations have occurred at the lab scale and have relied on expensive lab testing equipment (e.g., Beck-Friis et al. 2001; Pagans et al. 2006; Shen et al. 2011). However, lab-scale results may not be scalable to commercial-scale facilities due to the insulation effects of larger piles and the potential for heterogeneity (Pagans et al. 2006). In addition, there have been no studies addressing gas concentrations at ASP facilities with heat recovery. To assess the potential for ASP heat recovery composting as a viable method for processing agricultural wastes at a large scale, studies at the commercial scale are necessary.

1.3. Potential for pollution avoidance

ASP heat recovery composting facilities offer great potential for pollution capture and avoidance because they allow for more process control and manipulation. For example, leachate and condensate gas are collected in storage tanks rather than lost to waterways, and exhaust gas that is pulled through the piles exits the facility through specific pipes that can be managed. Environmental pollution can be reduced through a variety of pathways, such as collecting and using manure for compost rather than land-applying it (e.g., Rynk et al. 1992); capturing emissions with the use of a biofilter (Pagans et al. 2005); utilizing the exhaust gas to heat and fertilize a greenhouse (e.g., Mortensen 1987; Smith & Aber 2018); and using recovered heat and therefore avoiding fossil fuel energy emissions (Smith & Aber 2014, 2018).

1.4. Research objectives

ASP heat recovery composting is a waste management method that has great potential for pollution diversion and capture (Smith & Aber 2014, 2018; Smith 2016), but the exhaust gas concentrations from ASP heat recovery composting have not yet been studied at the commercial scale. Therefore, the overarching questions of this paper are: **What are the trace gas emissions and leaching losses from a commercial-scale aerated static pile heat recovery composting facility? How can an ASP heat recovery compost facility reduce or avoid pollution?**

The following specific research objectives were addressed:

1. Measure the CO_2, NH_3, CH_4, O_2, and temperature of the exhaust gas from an ASP heat recovery compost facility over the 21-day composting cycle.
2. Determine whether exhaust gas temperature could be used to estimate compost exhaust gas concentrations.
3. Discuss how the environmental pollution and offsets from ASP heat recovery composting could be compared to other manure management methods.

2. Material and methods

2.1. Study site

The study site for the compost exhaust gas measurements is the University of New Hampshire Joshua Nelson Energy Recovery Compost Facility located in Lee, New Hampshire. This compost facility uses ASP composting with heat recovery. The facility, which was constructed in 2013, is a pole barn structure that is 30 m x 15 m x 7 m (**Figure 1**). The facility is located at the UNH Burley-Demeritt Organic Dairy Research Farm (ODRF), which has 100 head of dairy cattle and spans over 120 ha (55 ha crop and forage production, 50 ha forest, 15 ha pasture). The compost feedstock is primarily from the dairy farm property. The main nitrogen source is dairy manure, and the carbon source is a combination of spent bedding from the bedded pack barn, waste baleage, and wood chips from the farm property.

Figure 1. Photos of the Joshua Nelson Energy Recovery Compost Facility. The images show A) the exterior of the facility where compost feedstock is loaded, B) piles of composting material in the facility, and C) the back of the facility where the compost exhaust gas is piped. *Sources for images: UNH 2014; Personal communication, Matthew Smith, 2016.*

The composting facility is divided into eight bays where compost feedstock can be loaded (**Figure 2**). The floor of the facility has embedded perforated pipes through which air is pulled using a fan system to aerate the compost. This facility is the only commercial-scale research facility for ASP heat recovery composting, providing tremendous opportunity for exploring the potential of this method of composting.

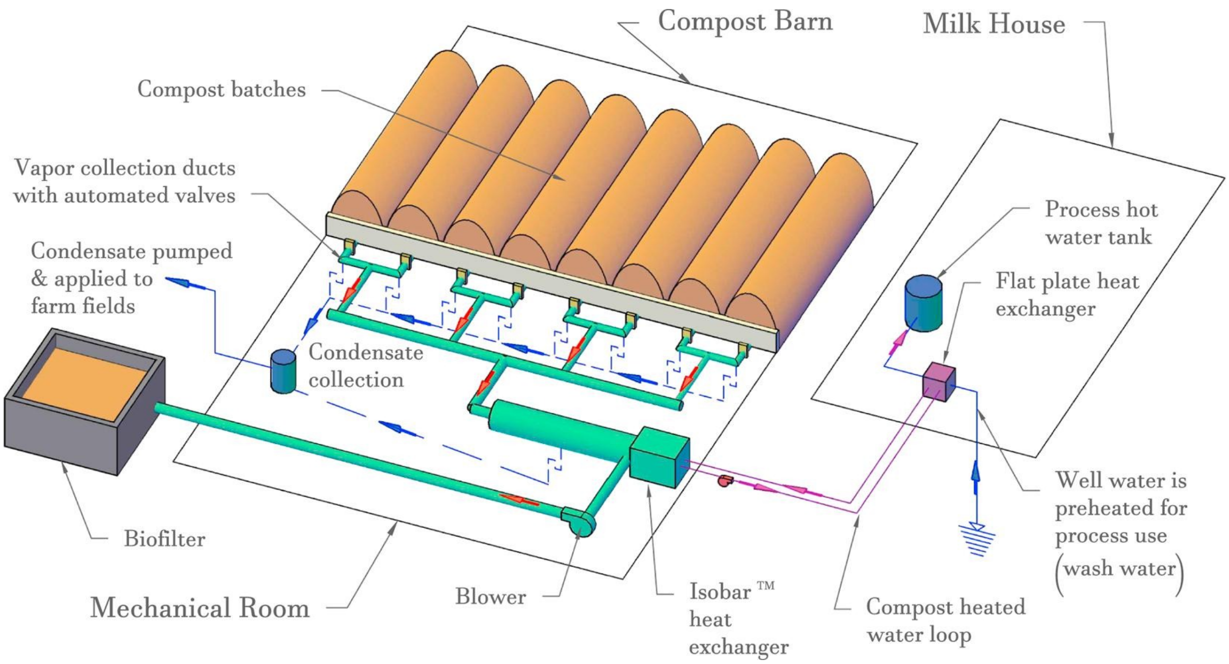

Figure 2. Conceptual diagram of the UNH ASP heat recovery composting facility. This diagram is an aerial view that shows the direction of airflow from the piles to the heat exchanger to the facility exhaust. Exhaust gas samples were taken from the pipes immediately exiting the bays. *Source: Smith and Aber 2018, Reproduced with permission.*

2.2. Characterizing compost exhaust gas

The compost exhaust gas was sampled for gas concentrations and temperature. The gases measured were NH_3 (an air quality pollutant and product of microbial activity), CO_2 (a greenhouse gas and product of microbial activity), and CH_4 (a greenhouse gas and product of anaerobic microbial activity, which means its presence should be limited in aerobic composting). O_2 was also measured to monitor compost aeration.

2.2.1. Exhaust gas measurements

Although a variety of gas sampling methods are available for lab-scale ASP compost studies, those methods cannot be easily applied to a commercial-scale facility and have sometimes produced unreliable results due to the high temperature and humidity conditions of composting (Beck-Friis et al. 2001). Methods for sampling compost exhaust gas concentrations include gas detectors, gas detection tubes, and passive gas sampling. Gas detectors have gas sensors that can provide real-time and continuous concentration measurements (RKI Eagle 2013). However, few have the high concentration, temperature, and humidity ranges necessary for sampling exhaust gas at a compost facility. Gas detection tubes are single-use glass tubes containing a chemical reagent (RAE Systems 2013). An air sample is pulled through a tube, and the reagent changes color when the target gas is present. Gas detection tubes can typically be used at both high temperature and high humidity (Paillat et al. 2005). Finally passive gas sampling involves collecting a gas sample and analyzing it in a lab for its gas concentration, such as with gas chromatography (NADP 2012; Scholtens et al. 2004). Passive gas sampling is most effective when both gas samples and condensate are collected and analyzed because water soluble gases condense out of the high-humidity and high-temperature exhaust gas (Beck-Friis et al. 2001). For all gas sampling methods, the accuracy, ease, and affordability for each method should be considered so that it could be repeated by compost facility operators at commercial-scale facilities.

We used two gas sampling methods: Single-use colorimetric gas detection tubes (NH_3, CO_2) and a gas detector (CH_4, O_2). Colorimetric gas detection tubes were used to record measurements for CO_2 and NH_3 (RAE Systems 2013; Dräger 2018). These detection tubes were selected given their wide concentration range (RAE Systems: 25-1000 ppm NH_3, Dräger: 1000-5000 ppm NH_3, RAE Systems: 0.25-20% CO_2) and accuracy at high temperature and humidity. Gas detection tubes are also relatively inexpensive and can be used by compost facility operators. See **Appendix A** for detailed specifications for the gas detection tubes. A gas detector (RKI Eagle Portable Gas Detector) was used for CH_4 measurements because quantitative gas detection tubes for CH_4 are not available. CH_4 is measured using a catalytic combustion sensor with a measurement range of 0-100% of the lower explosive limit (or 0-5% by volume). A 1:1 dilution fitting was used to dilute the incoming air sample because of the limited relative humidity (0-95%) and temperature (-10 to 40°C) range tolerated by the gas detector. Although the RKI Eagle can also measure NH_3 and CO_2, the measurements were not used because the NH_3 concentration range was inadequate (0-400 ppm). See **Appendix A** for detailed specifications for the RKI Eagle Portable Gas Detector.

O_2 measurements from the RKI Eagle were used to monitor aeration before an O_2 sensor (SST Sensing Screwfit zirconium dioxide oxygen sensor) was installed in the compost facility. The RKI Eagle uses an electrochemical gas sensor with a measurement range of 0-40% O_2. The O_2 measurements from the RKI Eagle were corrected using a consistent and linear relationship between the RKI Eagle and facility sensor O_2 measurements. The facility O_2 sensor was installed after this study concluded. It was considered more reliable because the gas detector consistently had O_2 readings that exceeded expected O_2 concentrations—often in the ambient range (See **Appendix B** for more information on the O_2 correction).

The exhaust gas temperature was measured with sensors in each of the eight bay exhaust pipes, and samples were recorded continuously (once per minute) using a Web Energy Logger (WEL). This record was processed to determine the average exhaust gas temperature associated with each sampling event.

For each sampling event, gas concentrations were recorded for the two paired bays in a given trial. To replicate the compost facility's regular aeration schedule, gas concentrations were only recorded after the aeration fan had been off for 2 hours. The aeration fan was then turned on for the bays of interest and run for 5 minutes (See **Appendix C** for details on the methods tests used to identify the sampling approach). Gas measurements were taken through a small sampling port in the pipe exiting the bays while air was flowing through the pipes (**Figure 2**). First, gas detection tube measurements (NH_3, CO_2) were taken simultaneously for both bays, with replicate measurements taken regularly. Next, the gas detector was used to record gas measurements (CH_4, O_2) every 30 seconds for at least 1.5 minutes or until the readings had stabilized. The stabilized reading was recorded and used.

2.2.2. Compost sampling trials

Gas sampling was conducted for four separate compost batches, each of which had two paired bays with the same feedstock material (**Table 1**). These individual bays are not true replicates because there is not a divider between bays, which can allow aeration for one bay to also pull some air through an adjacent bay. However for this study, each of the eight individual bays will be treated as a sample unit.

Table 1. Descriptive information for four compost sampling periods.

Start date	Bays	Total number of days	Daily sampling for:	Number of sampling events	Name of case study
September 2015	1 & 2	83	4 days	32	Blocked airflow
October 2015	3 & 4	53	6 days	23	Delayed microbial activity
August 2016	3 & 4	61	22 days	39	Standard conditions
November 2016	7 & 8	40	14 days	23	Low temperature

Each sampling period was identified by the starting month and year (e.g., September 2015). All measurements took place in the summer and fall, but different bays were used based on which were available for loading. Daily exhaust gas samples were taken at the start of the sampling period before transitioning to sampling 3 days per week. The daily samples aimed to capture the peak microbial activity, temperatures, and gas fluxes during the transition from the first mesophilic phase to the thermophilic phase. All sampling periods lasted at least 40 days, with the longest extending for 83 days. The minimum number of days before compost is considered ‘mature’ is 21 days.

Other data sets were collected on a limited basis for monitoring and to help understand the exhaust gas results. Data sets on feedstock quality characteristics (moisture content, carbon to nitrogen ratio, bulk density) were collected for all trials (Smith 2016). The feedstock quality characteristics were determined based on lab measurements of the individual feedstock materials (e.g., dairy manure, bedding, hay). The known ratio of mixed feedstock materials was used to determine a weighted average for the quality characteristics of a given mixture. Leachate (liquid that drains through composting feedstock) and condensate (liquid that condenses from the compost airflow) samples were analyzed for their carbon (dissolved organic carbon) and nitrogen (total dissolved nitrogen) content for a single trial (UNH Water Quality Analysis Laboratory 2018). The samples were only analyzed once because the leachate and condensate collect together in a single tank for all bays at the compost facility

Given the substantial variation in compost exhaust gas emissions and trends across the four paired bay trials, the results in this paper will first be presented for the August 2016 data set, which reflects standard operating conditions. The results of the other three trials will be presented as case studies to show how changes in feedstock materials, management practices, weather, and technical functioning can affect the final compost product and associated emissions. The scenarios for the three case study trials are blocked airflow (September 2015), delayed microbial activity (October 2015), and low temperature (November 2016).

2.2.3. Compost exhaust gas data analysis

The gas concentration and temperature measurements were plotted over time for each bay and sampling period. The NH_3, CO_2, CH_4, and O_2 concentrations were compared within a given sampling period, with a paired bay, and across different sampling periods to identify trends and ranges in concentrations.

The total flux of NH_3, CO_2, and CH_4 emitted over the composting cycle was calculated for each bay. The compost facility aeration schedule (aeration on and off time throughout life cycle) and airflow rate (measured every minute on sensors) were used for these calculations. The total daily gas flux from each bay was calculated (**Equation 2**) using the aeration schedule, measured gas concentrations, and airflow rate (**Equation 1**). For days without gas measurements, the gas concentration on the most recent sampling day was used. All daily gas emissions were summed to determine the total gas flux emitted over the composting life cycle.

Equation 1. Air flux calculation.
Air flux (m^3/day) = Aeration on time (min) * No. of aeration cycles per day * airflow rate (m^3/min)

Equation 2. Gas flux calculation.
Gas flux (kg/day) = Air flux (m^3/day) * Gas concentration (kg/m^3)

The gas fluxes and leachate/condensate were also reported as the percentage of the original feedstock C and N content for comparison to lab-scale studies reported in the literature. To calculate this metric, the total C and N content of the original feedstock mixture was calculated using the volume, bulk density, moisture content, and C and N content of the feedstock on a dry matter basis (**Equation 3**). The gas fluxes were converted to their C and N equivalents using their atomic weights (e.g., CH_4-C, CO_2-C, and NH_3-N).

Equation 3. Feedstock N calculation example for nitrogen.
Feedstock N (kg N) = Volume loaded (m^3) * Bulk density (kg/m^3) * (1 - % moisture content) * % N

> Where variables (bulk density, moisture content, feedstock % N) are weighted based on the feedstock volume ratios, and feedstock % N is on a dry weight basis.

The C and N content of the collected leachate/condensate was also calculated as an additional loss pathway. Because the leachate and condensate from the entire compost facility drains into a single storage tank, average ratios of leachate/condensate generation, C content, and N content were used to estimate the leachate/condensate C and N produced (**Equation 4**). The C and N content were measured as the average dissolved organic carbon (DOC) and total dissolved nitrogen (TDN) content of the leachate/condensate (UNH Water Quality Analysis Laboratory 2018).

Equation 4. Leachate calculation example for nitrogen.
Leachate N (kg N) = Feedstock wet weight (metric tons) * (Liters leachate / metric ton wet weight) * TDN (g N/L)

> Where the (Liters leachate / metric ton wet weight) is an average across all leachate collected in 2016, and the TDN concentration is based on measurements from two batches of compost in October 2017.

Finally, the % initial C and % initial N lost were calculated for the measured gases and leachate/condensate on a dry weight basis (**Equation 5**); the remaining % C and % N are contained in the final compost or is lost via another pathway not quantified in this study (see discussion section).

Equation 5. % initial N calculation for NH_3-N
% initial NH_3-N = kg NH_3-N / feedstock metric tons N dry weight

The total gas emitted was normalized per kg dry weight of compost feedstock for comparison across trials and with literature values.

Finally, the compost gas concentration results were analyzed to identify potential predictor variables to help compost facility managers estimate gas concentrations. Following Pagans et al. 2006, exponential regressions were performed between exhaust gas temperature (easy for facility managers to measure) and the exhaust gas concentrations (NH_3, CO_2, CH_4) in Microsoft Excel.

2.3. Estimating avoided pollution

The closed system and addition of heat recovery in an ASP heat recovery composting facility could prevent or avoid environmental pollution. A full life cycle comparison should be conducted between this method of composting and other methods of manure management. The factors that should be considered in this comparison are discussed.

3. Results

The results presented below reflect one compost trial that followed standard commercial operating conditions as an example of expected trends during the composting process. Beginning in August 2016, this compost trial had well-mixed feedstock (40% dairy manure, 30% horse manure imported from the UNH Equine Facility, 20% bedded pack, 10% waste baleage); a moisture content of 74%; and no technical difficulties (**Table 2**). The three remaining compost trials are presented as case studies to show how changes in feedstock, management practices, weather, and technical difficulties can affect the composting process and the associated exhaust gas concentrations.

Table 2. Compost feedstock mixture characteristics for four compost trials.

Start date	**Name of case study**	**Mixture***	**C:N Ratio**	**Moisture content** (%)	**Bulk density** (kg/m3)	**Volume** (m3)	**Wet weight** (metric tons)	**Dry weight** (metric tons)
Sept 2015	Blocked airflow	40% M, 40% BP, 20% WB	31	70%	712	107	76	23
Oct 2015	Delayed microbial activity	40% M, 40% BP, 20% WB	31	70%	712	107	76	23
Aug 2016	Standard conditions	40% M, 20% BP, 10% WB, 30% HM	26	74%	604	107	65	17
Nov 2016	Low temperature	40% M, 40% BP, 20% WB	31	65%	715	107	77	27

*M = cow manure, BP = bedded pack, WB = waste baleage, W = wood chips, HM = horse manure

3.1. Characterizing compost exhaust gas

3.1.1. Exhaust gas measurements

Compost exhaust gas concentrations reached an early peak and then decreased slowly throughout the trial (**Figure 3**). The results on the first day of sampling are a near minimum for NH_3, but a maximum result for both CO_2 and CH_4. These concentrations are likely due to a buildup of gases that accumulated while the piles were anaerobic before loading into the compost facility and before aeration was turned on. On day one, the O_2 concentrations averaged 14%. After just one day of aeration, the NH_3 concentrations quickly peaked to a maximum reading of 3000 mg/m^3 and a temperature of 60°C. At the same time, CO_2 and CH_4 concentrations both started to decline. For the remainder of the composting period, the CO_2, NH_3, and CH_4

concentrations continued to decline, while O_2 increased to 17% with more airflow through air pockets. Throughout the first half of the trial, bay 3 had higher concentrations than bay 4.

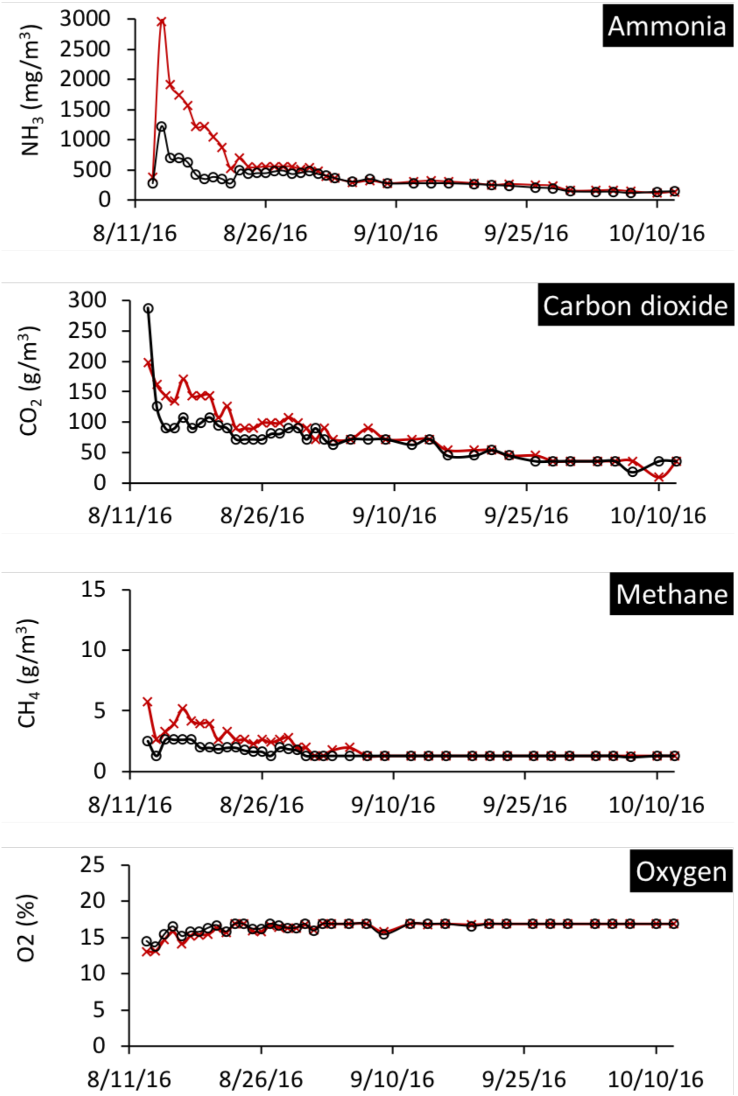

Figure 3. Compost exhaust gas trends for August 2016 trial. The exhaust gas concentrations are shown over a full composting trial beginning in August 2016 for two composting piles: Bay 3 (red line with X marker) and Bay 4 (black line with circle marker). Exhaust gas concentrations are shown for ammonia, carbon dioxide, methane, and oxygen (for tracking pile aeration).

The exhaust gas temperature of the August 2016 compost pile ranged from 39°C to 63°C, exceeding the minimum target temperature of 55°C to kill pathogens, weeds, seeds, etc. (**Figure 4**). When aeration first began, the exhaust gas temperature was at a near minimum because the compost feedstock had been sitting for several weeks before mixing and loading. Microbial activity commenced quickly after aeration and the pile entered the thermophilic phase within 1 day, as reflected by the temperatures reaching over 60°C within two days of aeration. After peaking, temperatures of both bays began to slowly decline, finally returning to temperatures around 40°C near the end of the study. The pile did not return to the mesophilic stage (<40°C) until the last few days of the trial. Throughout the first half of the trial, bay 3 had exhaust gas temperatures slightly greater than bay 4. When the temperatures in bay 3 were greater than bay 4, the gas concentrations were also higher in bay 3.

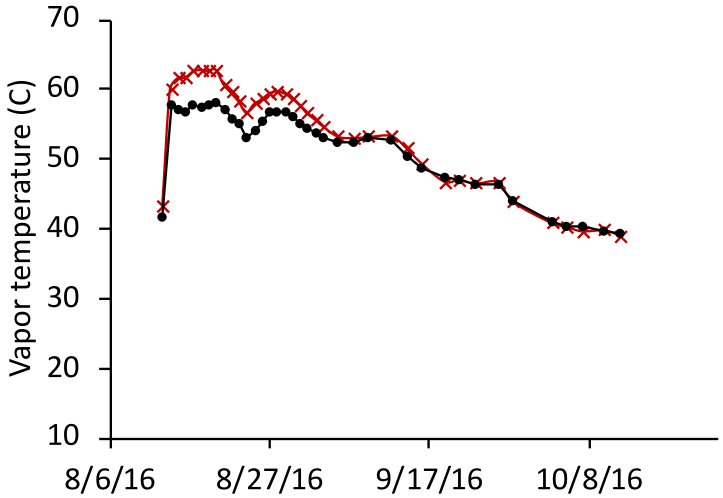

Figure 4. Exhaust gas temperatures for August 2016 trial. The exhaust gas temperatures are shown over a full composting trial beginning in August 2016 for two composting piles: Bay 3 (red line with X marker) and Bay 4 (black line with circle marker).

3.1.2. Total gas fluxes and normalizations

During the first 21 days of sampling, the daily gas fluxes across both bays averaged 0.15 metric tons CO_2/day, 3.2 kg CH_4/day, and 0.93 kg NH_3/day. All three gases had a peak flux in the first few days of the trial: CO_2 (0.72 kg/day) and CH_4 (14 kg/day) on the first day, and NH_3 (5.5 kg NH_3/day) on the second day. These peak flux days represent a large percentage of the total 21-day cycle emissions for CO_2 (19%), CH_4 (15%), and NH_3 (19%). After these peaks, the fluxes all decreased and leveled off over time.

The total gas fluxes for the two bays in the August 2016 trial over the 21-day sampling period were 41 kg NH_3, 6 metric tons CO_2, and 142 kg CH_4 (**Figure 5**). All three gas fluxes were higher in bay 3 than bay 4 throughout the sampling period.

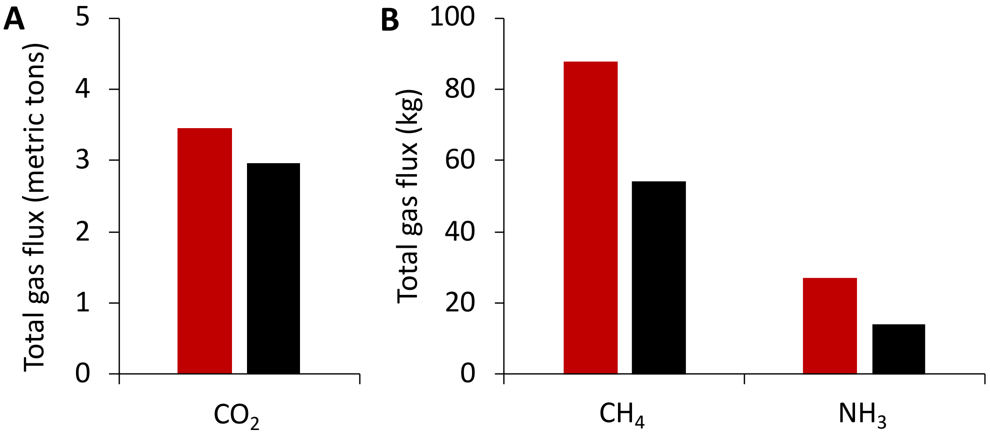

Figure 5. Total gas flux for first 21 days of August 2016 trial. The total gas flux is shown for two compost piles: Bay 3 (red bar) and bay 4 (black bar). A) carbon dioxide (metric tons), B) methane and ammonia (kg).

The measured loss pathways account for 24-28% of the initial carbon in the feedstock and 10-15% of the initial nitrogen in the feedstock for the August 2016 trial (**Figure 6**). Most of the C loss is as CO_2-C (22-25%), followed by CH_4-C (1-2%) and leachate/condensate-C (1%). Most of the N loss is as NH_3-N (6-12%), followed by leachate/condensate-N (3%). Other potential loss pathways not quantified in this study include N_2O-N and passive gas fluxes. The remaining C and N is contained in the final compost product.

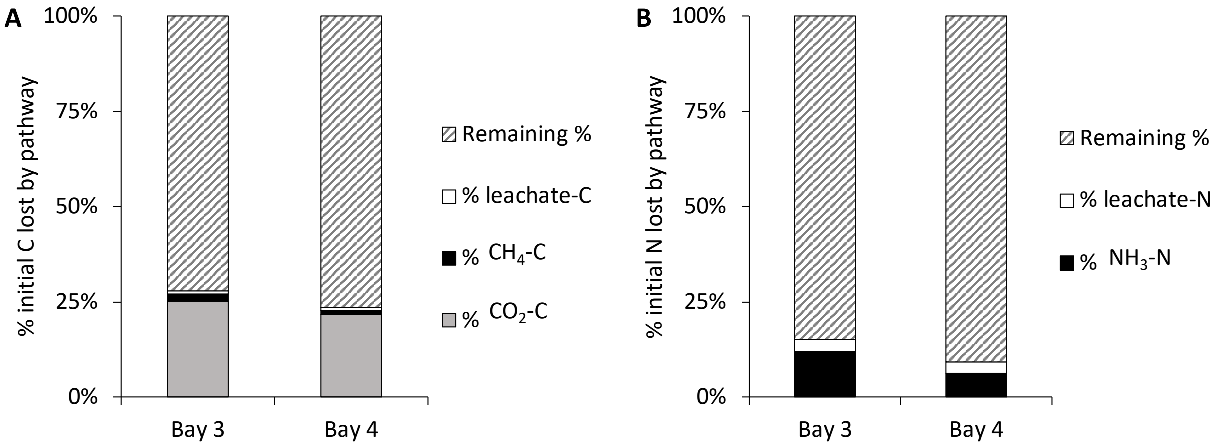

Figure 6. Percent loss of initial carbon and nitrogen during August 2016 trial. Loss pathways for (A) initial carbon and (B) initial nitrogen were gas emissions and collected condensate and leachate. Results are from the August 2016 compost trial for first 21 days for bays 3 and 4. Remaining % carbon and % nitrogen is contained in the pile or lost (e.g., before loading or from passive emissions).

The total gas fluxes were normalized to the dry weight of compost feedstock. Over the 21-day composting period for the August 2016 trial, 1 metric ton of dry weight compost feedstock emitted an average of 380 kg CO_2, 8 kg CH_4, and 2 kg NH_3.

Table 3. Gas flux per metric ton of compost feedstock for August 2016 trial. The gas fluxes include the first 21 days of the August 2016 trial.

Bay number	Dry weight of compost feedstock *(metric tons)*	CO_2 *(kg CO_2/metric ton dry weight)*	CH_4 *(kg CH_4/metric ton dry weight)*	NH_3 *(kg NH_3/metric ton dry weight)*
Bay 3	8	410	10	3
Bay 4	8	350	6	2

3.1.3. Predictive variables for gas concentrations

Regressions between exhaust gas temperature and NH_3 and CO_2 exhibited strong correlations using an exponential trend line (**Figure 7**). NH_3 correlated strongly with exhaust gas temperature (R^2=0.75), although the relationship appears to be weaker for the highest NH_3 concentrations. CO_2 also correlated strongly with exhaust gas temperatures (R^2=0.55). CO_2 had a single major outlier due to a high concentration at low temperature on the first day of sampling from off-gassing after anaerobic conditions. A weaker correlation was observed between exhaust gas temperatures and CH_4 concentrations (R^2=0.37, graph not shown).

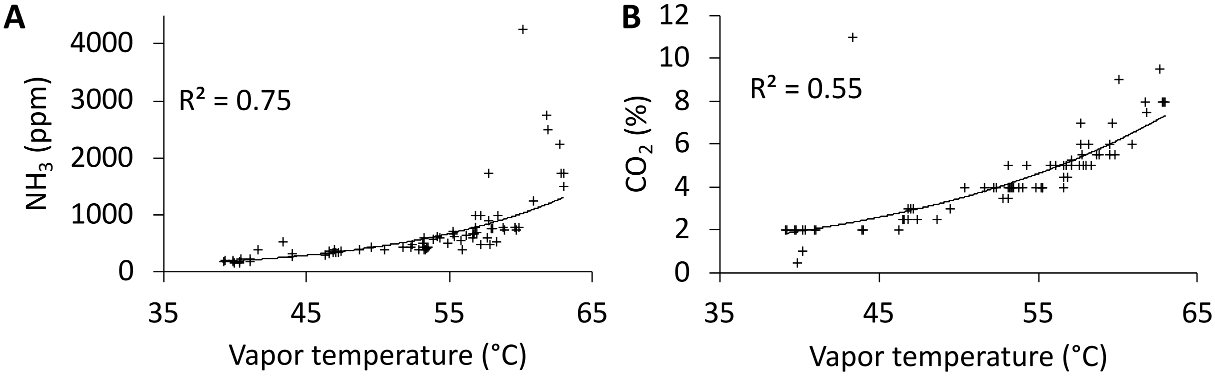

Figure 7. Regressions of exhaust gas temperature and concentrations for August 2016 trial. Regressions are shown for A) ammonia concentrations and B) carbon dioxide concentrations for all sampling results from bays 3 and 4 in the August 2016 trial. Trend lines are exponential trend lines.

3.2. Case studies for different compost trials

Three compost trials had different management conditions and, in some cases, were characterized by technical challenges. These case studies resulted in unexpected but informative results for managing compost piles at a commercial-scale ASP compost facility. Those trials occurred in September 2015 (‘Blocked airflow’), October 2015 (‘Delayed microbial activity’), and November 2016 (‘Low temperature’) (**Figure 8**). See **Appendix D** for complete results and figures for each case study.

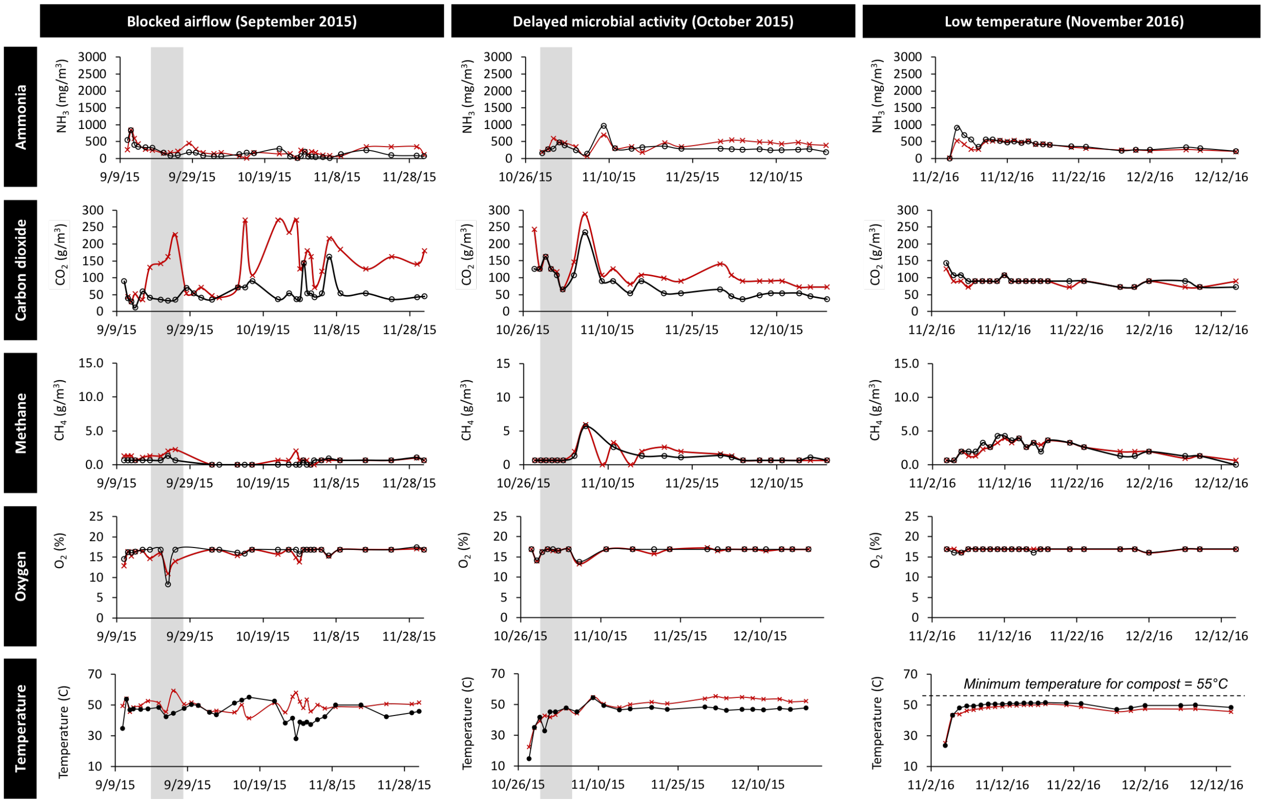

Figure 8. Compost exhaust gas trends for three case studies. The full composting trial results are shown for two bays for each of the following case studies: ‘Blocked airflow’ in September 2015 (bays 1 and 2), ‘Delayed microbial activity’ in October 2015 (bays 3 and 4), and ‘Low temperature’ in November 2016 (bays 5 and 6). The sections highlighted in grey show when the trial’s specific management condition occurred, and the minimum compost temperature is shown on the ‘low temperature’ case study graph. The first bay listed is denoted by the red line with X marker, and the second bay is denoted with the black line with circle marker. Exhaust gas concentrations are shown for ammonia, carbon dioxide, methane, and oxygen (for tracking pile aeration). Y-axis ranges are set equal to those in **Figure 3** for direct comparison.

3.2.1. Blocked airflow

The September 2015 trial was marked by technical difficulties. The aeration valve for bay 1 stuck repeatedly throughout the composting process, which led to under-aeration throughout the study. Whenever the valve was stuck for more than one day, the effect was observable in the gas concentrations: CH_4 and CO_2 concentrations would increase, while O_2 would decrease (**Figure 8**). For example, on day 12 of the study, O_2 concentrations dropped to just 9% after the aeration valve was stuck for one day. At the same time, CO_2 and CH_4 concentrations both increased. These results suggest that regular aeration is necessary to reduce greenhouse gas emissions during aerobic composting.

3.2.2. Delayed microbial activity

Early in the October 2015 trial, the temperature of the pile was not increasing to the target temperature necessary to kill pathogens and weed seeds. To reach this temperature, the pile aeration was turned off for two days, which allowed the pile to heat up. The lack of aeration also led to the beginning of anaerobic decomposition and corresponding elevated concentrations of CH_4 and CO_2. After aeration was resumed, the pile briefly reached the target temperature. Gas concentrations stayed low throughout the composting process, suggesting that lower temperatures may be preferable for reduced gas emissions. This is especially true for NH_3 emissions, which have an exponential relationship with pile temperature (Pagans et al. 2006).

3.2.3. Low temperatures

The November 2016 trial never reached the target temperature of 55°C despite proper aeration and comparable feedstock to other trials. Instead, the pile maintained temperatures of 50°C or less. This trial took place during colder weather than the other two trials; the monthly average temperature was just 5°C. Although the facility is enclosed, the exterior temperature can impact the ability of piles to heat up sufficiently. Management practices may be necessary to encourage piles to reach target temperatures over the winter, such as ensuring a lower starting moisture content, recirculating warm exhaust gas into a newly loaded bay, or slowly ramping up aeration to allow frozen feedstocks to thaw evenly. This batch of compost would be ineligible for commercial sale because it did not reach the minimum temperature of 55°C.

3.3. An approach for estimating avoided pollution

During ASP heat recovery composting, there are multiple sources of pollution and several points where pollution can be avoided or offset (**Table 4**). After feedstock is loaded into the compost facility, the major sources of pollution are gaseous emissions from composting (NH_3, CO_2, CH_4, N_2O); leachate and condensate that collects during composting; and losses that occur after the compost is land-applied. Pollution can be avoided or offset from ASP heat recovery composting through offset fossil fuel energy requirements (e.g., greenhouse gas and NOx emissions); the capture of NH_3 emissions in a biofilter; the ability to collect and treat or use collected condensate and leachate; and soil C sequestration after compost is applied to the land.

Table 4. ASP heat recovery composting: Pollution sources and potential for avoidance. The pollution pathways from ASP heat recovery composting are paired with potential strategies for managing that pollution stream. Any relevant completed and ongoing studies at the UNH compost facility are noted.

ASP heat recovery composting step	Pollution source	Pollution management strategy	UNH study
Microbial activity	Gas emissions (NH_3, CO_2, CH_4, N_2O)	Biofilter to capture NH_3	*This study** Williamson et al. in preparation
Leaching and runoff	Leachate and condensate	Collected leachate/condensate can be treated or used	Smith et al. in preparation
Heat capture	N/A	Offset fossil fuel energy needs through captured heat	Smith & Aber 2018
Land application of compost	Runoff, leaching, volatilization	Increased soil C sequestration	n/a**
Exporting compost	n/a	Export compost to improve farm N efficiency	**Chapter 2** of this dissertation

*This study quantified the gaseous emissions from ASP heat recovery composting. Williamson et al. (in preparation) will report the NH_3 capture from a biofilter.

**Studies in the literature have explored the C sequestration benefits from land application of compost (e.g., Ryals & Silver 2013), but there are currently no plans for a similar study at UNH.

Ongoing studies at the UNH ASP heat recovery compost facility are exploring other pollution avoidance and management strategies. Smith & Aber (2018) quantified the emissions avoidance from replacing fossil fuel energy with heat recovery. This study measured the exhaust gas concentrations during composting. Williamson et al. (in preparation) examined the NH_3 removal efficiency of different biofilter designs—the first study of its kind at a commercial-scale facility. Smith et al. (in preparation) assessed whether applying leachate to composting feedstock can jumpstart the composting process, especially during cold months when it is difficult for piles to meet minimum temperature requirements. **Chapter 2** of this dissertation calculated the effect of exporting compost on the overall ODRF's N budget and N efficiency.

4. Discussion

4.1. Gas concentrations during ASP heat recovery composting

The standard commercial operating conditions trial (August 2016) maintained high temperatures in the thermophilic stage throughout most of the 21-day trial. However, the exhaust gas concentrations peaked early and then declined substantially. The early large flux of emissions indicates the importance of emissions management early in the composting process, such as with a biofilter (e.g., Pagans et al. 2005) or cooling, condensing, and collecting gaseous emissions in solution (e.g., Beck-Friis et al. 2001).

The compost trials with technical challenges (blocked airflow, delayed microbial activity, low temperature) show the large fluctuations in emissions that are possible under a wide range of management conditions. In two of the case studies, the piles spent time under anaerobic conditions, which led to higher emissions of CO_2 and CH_4 but lower emissions of NH_3 until aeration resumed. For an ASP heat recovery compost facility, the forced aeration should encourage regular airflow that promotes aerobic microbial activity. These case studies made clear the importance of careful management of an ASP heat recovery compost facility for optimal composting conditions and lower air pollutant (NH_3) and greenhouse gas (CO_2 and CH_4) emissions.

The compost trials took place in the summer and fall months (August through December), and most of the trials used similar feedstock materials from the organic dairy farm where the facility is located. For a more complete picture of exhaust gas concentrations at a range of conditions, future work should measure gas concentrations for compost feedstock materials with different characteristics (e.g., range of moisture contents, carbon to nitrogen ratios, feedstock materials). Studies should also be completed during different times of the year, especially over the winter due to the management challenges of below-freezing temperatures.

4.2. Comparisons with lab-scale studies in the literature

Previous studies have assessed emissions of NH_3, CO_2, and CH_4 at ASP compost facilities, but none of those facilities had a heat recovery component and all were conducted at a lab scale. Lab scale reactors are much smaller than commercial scale facilities. Lab scale reactors contain 30 to 600 L of feedstock, whereas the commercial-scale facility in this study processes 50,000 L of feedstock material in each of the 8 bays in the facility. Scale is important, as the emissions and heat generation from microbial activity does not necessarily scale linearly from a lab-scale reactor to a commercial-scale facility processing 100 times as much material. Commercial-scale facilities must have the capacity to process large amounts of material to be profitable. However, commercial-scale facilities face additional challenges not observed by lab-scale reactors, such as temperature differences throughout the piles and pockets with different types and sizes of materials. Our study is one of the first to study the temperature profile and gas emissions directly at a commercial-scale facility to understand the potential pollutants and challenges that can occur.

Beck-Friis et al. (2001) measured gaseous emissions (CO_2, NH_3, N_2O) from a lab-scale forced air composting reactor with organic household waste feedstock by cooling the exhaust gas and analyzing the collected condensate. The total amount of initial C and N emitted as CO_2-C (67%) and NH_3-N (24%) over a 22-day trial was much greater than the average losses recorded in this study (17% of CO_2-C and 5% of NH_3-N). The lower C and N losses in this study could be due to differences in the experimental design, such as: the facility design of Beck-Friis et al. (200 L sealed airtight container) versus this study (large commercial-scale facility often open to the outside); the more frequent watering regime of Beck-Friis et al. (constant moisture content regulation at 65% by adding water to replace moisture lost via airflow) versus this study (once daily); and the method of sample collection in Beck-Friis et al. (condensed gas emissions) versus this study (direct measurement of exhaust gas). In addition, the characteristics of the feedstock itself could affect the results, such as: the lower C:N ratio of organic household waste (22:1) in Beck-Friis et al. versus agricultural waste (30:1) in this study; and the processing of the feedstock material into uniform pieces in Beck-Friis et al. versus manual mixing of the feedstock in this study. Beck-Friis et al. observed that N_2O emissions contributed a very small percentage (2%) of total N emissions; N_2O was not measured in this study.

Pagans et al. (2006) compared the temperature and NH_3 emissions profile from a range of feedstock materials composted in a lab-scale (30 L) aerobic composting reactor. The feedstock materials most similar to the agricultural wastes used in this study were the organic fraction of municipal solid waste, raw sludge, and anaerobically digested sludge. All three feedstock materials had comparable peak temperatures (58-60° C) but a range of peak NH_3 emissions (100-660 mg NH_3/m^3). This study had peak temperatures of 52-63° C and peak NH_3 concentrations of

840-3000 mg NH_3/m^3. The differences in results are likely due to the very different feedstock material: The C:N ratio of the feedstock materials in Pagans et al. were all less than 17:1. Pagans et al. also observed a strong exponential correlation between exhaust gas temperature and NH_3 emissions in four out of five trials. In addition, they observed that the strength of the correlation improved when the thermophilic stage and the final mesophilic stage of composting were analyzed separately. These observed correlations over a range of temperatures and NH_3 concentrations further indicate the potential for temperature to be used as a predictor for NH_3 concentrations.

Shen et al. (2011) compared the gas emissions (NH_3, CH_4, N_2O) for low aeration and moderate aeration rates from a lab-scale (60 L) aerobic compost reactor with poultry manure feedstock. As observed in this study, the CH_4 emissions increased to levels as high as 9% with lower aeration rates. Results from both this study and Shen et al. (2011) suggest that maintaining frequent and sufficient aeration is necessary to minimize greenhouse gas emissions during aerobic composting.

4.3. Relationship between exhaust gas temperature and gas concentrations

This study found exhaust gas temperature to be a reasonable predictor of NH_3 and CO_2 exhaust gas concentrations under standard operating conditions. Relatively easy to measure, exhaust gas temperature is an indicator of the level of microbial activity: with more microbial activity during aerobic decomposition, the exhaust gas temperature will increase. It then follows that both NH_3 (a by-product of mineralization and nitrification when nutrients are in excess) and CO_2 (generated from microbial respiration) would also be elevated with a higher temperature (Buscot & Varma 2005). The relationship between exhaust gas temperature and NH_3 and CO_2 was exponential, but there are upper limits for temperature, NH_3, and CO_2 during composting. If temperatures exceed 60°C, then the thermophilic bacteria key to aerobic composting become dormant (Peigné & Girardin 2004). At temperatures above 80°C, compost piles can spontaneously combust. Pagans et al. (2006) also found an exponential relationship between temperature and NH_3 emissions from the thermophilic composting stage for a range of feedstock materials: raw sludge, municipal solid waste, digested sludge, and animal by-products. However, a linear relationship was observed between temperature and NH_3 emissions during the final mesophilic maturation phase for all feedstock materials except raw sludge, which had an exponential relationship.

NH_3 and CO_2 emissions are limited by the amount of feedstock material and its C:N ratio. Microorganisms require carbon for their energy supply and growth, and they use nitrogen for protein production and reproduction (Ryckeboer et al. 2003). The ideal C:N ratio for composting is 25:1-35:1, but composting can be completed with a broader range (e.g., 20:1-40:1) (Pace et al. 1995). For example, if the C:N ratio exceeds 40:1, then there is not enough nitrogen for composting to be completed. Alternatively, if the C:N ratio is less than 25:1, then there is not enough carbon and excess nitrogen. Microbes release this extra nitrogen via mineralization in the form of ammonium (NH_4^+), which can then volatilize to NH_3 gas (Buscot & Varma 2005). Microbial activity slows down substantially when microorganisms deplete the available supply of either carbon or nitrogen in the ratio at which they use carbon and nitrogen.

NH_3 and CO_2 vapor concentrations can be estimated from exhaust gas temperature under certain conditions. First, aerobic conditions must be maintained. If the piles become anaerobic, then the aerobic relationship between temperature and the gas concentrations no longer holds, such as in the case studies (See **Appendix D**). Second, the temperatures can best predict gas concentrations at a range of 40–60°C, or the thermophilic phase. At temperatures less than 40°C, the level of microbial activity is not generating temperatures and gas concentrations high enough to fall in the predictive range. At temperatures greater than 60°C, the thermophilic bacteria become dormant. Future research should prioritize a set of exhaust gas temperature prediction curves for commercial-scale facilities that reflect a range of moisture contents and C:N ratios under well-managed aerobic conditions.

4.4. Potential for pollution avoidance from ASP heat recovery composting

4.4.1. Comparing to other manure management methods

A comprehensive comparison should be conducted between the ASP heat recovery compost facility and other manure management methods. A preliminary literature survey suggests that land application of manure could lead to lower greenhouse gas emissions but higher N losses than the ASP heat recovery compost facility (Dong et al. 2006; Yang et al. 2011; Aguirre-Villegas & Larson 2017). However, a complete comparison must take into account every step of the ASP heat recovery composting process to ensure all pollution sources and management strategies are considered (See **Table 4**).

In comparison to other manure management methods, aerobic composting reduces the percent of carbon emitted as CH_4 versus CO_2. Because CH_4 is a by-product of anaerobic microbial activity, its emission is reduced in aerobic systems like the ASP heat recovery compost facility. Conversely, CH_4 emission is the main carbon emission following the storage and land application of manure (Amon et al. 2006; Smith et al. 2008). Emitting carbon as CO_2 instead of CH_4 can reduce the contribution to the global warming effect because CH_4 has a 28 times greater global warming potential than CO_2 (IPCC 2014).

4.4.2. Other pollution avoidance pathways

Because of how an ASP heat recovery composting system is designed, there are a variety of options for capturing and treating waste streams, thereby reducing pollution. These strategies can be used to manage pollution from the air exhaust (e.g., with a biofilter or greenhouse) and leachate and condensate collection (e.g., with capture and treatment). ASP heat recovery composting can also reduce greenhouse gas emissions (e.g., through carbon sequestration from land application of compost and replacing fossil fuel heating energy with captured heat).

Gaseous emissions from ASP composting are often emitted directly to the atmosphere. Biofilters, which use living material to capture pollution, could reduce emissions of NH_3 (Pagans et al. 2005; Rosenfeld et al. 2004) and CH_4 (Gebert & Gröngröft 2006; Melse & Van Der Werf 2005). For example, a biofilter with a carbon source (e.g., wood chips, finished compost) can capture NH_3. Lab-scale studies have found NH_3 removal efficiencies over 90% (Pagans et al. 2005), but these studies have occurred in a controlled environment with lower NH_3 concentrations (less than 200 ppm) than those observed at our commercial-scale facility (often in excess of 1000 ppm). Preliminary results at our study site have found average NH_3 removal efficiencies exceeding 80% for a biofilter made of wood chips and finished compost (Williamson et al. in preparation). Although biofilters prevent NH_3 from being immediately emitted to the atmosphere, biofilters are still a relatively short term sink for N.

The nutrient-rich exhaust gas from composting could be used to heat and fertilize crops in a greenhouse. Plants generally respond positively to some increased concentrations of CO_2 and NH_3 (Krupa 2003; Long et al. 2004), but they begin to experience adverse effects and eventually toxicity at high concentrations (Mortensen 1987; Fangmeier et al. 1994; Van Der Eerden et al. 1998; Krupa 2003), which necessitates the use of filtration.

Leachate from animal excreta can enter waterways and contribute to water pollution. An ASP compost facility collects this leachate throughout the composting process, preventing its immediate loss to the environment. This leachate can then either be reapplied to the composting piles (which can facilitate composting of low nitrogen feedstock due to its high nitrogen content), processed with a wastewater treatment method, or applied to the land as a fertilizer. All methods give the facility manager more control over the fate of the leachate. Further research is needed to better understand the potential for leachate to facilitate the composting process of different feedstocks (Smith et al., in preparation).

Applying compost to the land can increase soil organic matter and water retention, and it can even promote carbon storage. Ryals and Silver (2013) found that a single application of compost to rangeland both increased the net primary productivity and the net ecosystem carbon storage in all trials except one that was water-limited. Future research should assess the carbon storage potential from applying compost to New England soils.

Using recovered heat from composting could reduce the need for fossil fuel energy and the associated emissions (Smith & Aber 2018). The heat generated from the microbial activity during composting can be captured and used directly on the farm for heating needs, replacing the need for fossil fuel heating. Heat capture from composting can offset the emissions that would otherwise be generated from fossil fuel heating.

The potential for avoiding pollution through these strategies has not yet been explored for a commercial-scale facility using an integrated pollution mitigation strategy approach. In addition, most studies to date have been lab-scale studies. An important next step for this research is to quantify the potential pollution avoidance associated with a series of strategies at a commercial-scale ASP heat recovery composting facility.

4.5. Data quality

4.5.1. Operational considerations and limitations

The gas sampling methods used in this study do not address all potential loss pathways from composting feedstocks. There are three specific fluxes that are not quantified in this study: emissions from the feedstock before loading, passive emissions from the piles, and the peak fluxes not during sampling times. In addition, leachate/condensate carbon and nitrogen content were not analyzed for all trials.

The feedstock materials can accumulate for several days to weeks before mixing and loading into the composting facility. Due to the labor costs associated with loading, compost feedstock is only loaded when enough material for a complete batch is available. The length of time depends on the availability of feedstock. For example, while cattle graze during the summer, most manure is deposited to the field and cannot be easily collected. Over the summer, manure piles accumulate more slowly and have the potential to generate more emissions before enough material is available to load into the compost facility. This is a drawback when compost facilities are located on smaller farms that do not generate as much manure; larger farms would have ample feedstock for frequent loading.

Each compost pile is not constantly aerated in the compost facility. Instead, each pile is aerated in sequence for a period of time (10-45 minutes, depending on the composting stage), followed by a period without aeration. The entire aeration cycle can extend for two to three hours. In between aeration periods, passive emissions could be released from the piles. Some of these emissions would be pulled through the pipes during the next aeration cycle, but some of those emissions could exit the facility and be lost directly to the atmosphere. These passive pile emissions were not measured in this study and should be studied in future experiments.

Because gas sampling only occurred once per day, peak gas fluxes may have been missed. This would be especially important early in the composting process as temperatures and gas concentrations are increasing; later in the compost cycle, gas concentrations stabilize and decline at a steady rate (see **Figure 3**). Ideally, a future study would have sensors installed in the system that can take continuous measurements. However, single-use gas detection tubes were selected for this study due to their accuracy at high heat and temperature and their ease of use at other comparable facilities.

Leachate can collect in two ways in an ASP heat recovery compost facility. First, excess moisture in the piles can leach and be collected. Second, condensation can collect in the piping. The relative humidity of the gas effluent is 100%, which promotes condensation in the piping as the gases exit the compost and move through a piping system. NH_3 is very soluble and dissolves easily into water to form ammonium (NH_4^+). When NH_3 is measured as both gas and condensate from a compost facility, the NH_3-N can make up as much as one third of the total initial N by mass (Beck-Friis et al. 2001). CO_2 does not dissolve as readily into solution. There was uncertainty around the leachate/condensate results because the specific leachate/condensate samples associated with a particular batch of compost could not be isolated. Instead, the leachate/condensate from all 8 bays collected in a single collection tank.

4.5.2. Nitrous oxide concentrations

Nitrous oxide (N_2O), which is released during the composting process, was not measured in this study. The gas measurements prioritized in this study were NH_3 and CO_2, which were expected to be higher with aerobic composting, and CH_4 to monitor for the presence of anaerobic pockets. The measurement methods we selected did not support N_2O testing. Previous studies have measured N_2O using collection traps later analyzed with gas chromatography (Szanto et al.

2007; Shen et al. 2011) and a photoacoustic multi-gas monitor (Beck-Friis et al. 2001). Beck Friis et al. found that N_2O was a relatively small fraction of all N emissions (<2%, with NH_3 making up the remaining 98%), suggesting its inclusion would not greatly affect the N balance. However, the strong global warming potential of N_2O could mean that a small emission of N_2O could still be a large contributor to total greenhouse gas emissions from ASP heat recovery composting. A future study should include N_2O measurements using one of these methods for a more complete assessment of gas emissions from ASP heat recovery composting.

4.5.3. Gas measurement uncertainty

The gas sampling results have uncertainty due to limitations for the methods selected, the types of emissions measured, specifications for the measurements, and gas sampling conditions. The following emissions were not measured in this study because they were outside the scope of the analysis: emissions from the feedstock before loading, passive emissions from the piles, and the peak fluxes not during sampling times (see **Section 4.5.1** for more information).

The compost exhaust gas sampling methods used in this study were selected because they capture average fluxes over time with a method that is both easy to use and affordable. These methods do have the potential to miss peak fluxes because the sampling is not continuous, but most of the composting cycle has relatively small changes in concentrations under standard operating conditions (see **Figure 3**). The largest fluxes occur as temperatures are increasing early in the composting trial during the transition from the first mesophilic phase to the thermophilic phase. If temperature is tracked continuously with sensors, then practitioners will know when to expect higher fluxes and when to sample more frequently. Higher frequency measurements early in the composting process, especially as temperature is still increasing, would adequately capture most peak composting fluxes.

The gas detection tubes can be used under high humidity and high temperature conditions. The reported precision or relative standard deviation was ≤ 10% for CO_2 and 12% for NH_3 (RAE tubes for 12-1000 ppm), and 10-15% for NH_3 (Dräger tubes for >1000 ppm). The humidity effects are minimal (no effect for up to 90% and 100% relative humidity for NH_3 and CO_2, respectively). The NH_3 gas detection tubes do require a 1.2 correction factor for temperatures greater than 40°C, which was applied to the results. The catalytic combustion sensor for CH_4 measurement has a reported accuracy of ±5% of the reading, and it can be used at up to 95% relative humidity and 40°C. A 1:1 dilution fitting was used for the high temperature and humidity conditions. See **Appendix A** for detailed specifications of the sampling methods.

The sampling conditions at a compost facility can be affected by the high temperature, high humidity, and the presence of multiple gases in the compost exhaust. Exhaust gas temperatures often exceed the upper limit of many instrument sampling ranges (>40°C), and the relative humidity is usually as high as 100%. The gas detection tubes were the most accurate and reliable of the methods that could be applied at other facilities. Future studies should pair gas detection tubes with gas chromatography to confirm their accuracy during composting conditions.

4.6. Applicability to other compost facilities

The proposed compost exhaust gas sampling methods can be applied at other ASP compost facilities relatively easily, given the low cost and ease of use of the single-use gas detection tubes. In addition, other facilities could take considerably fewer measurements by targeting the peak fluxes identified in this study (i.e., during peak temperatures) and by adjusting the prediction curves with spot measurements. The prediction curves can be adjusted with moderate scaling to assess approximate total gas fluxes. However, the prediction curves for the fluxes across the sampling cycle and correlations with exhaust gas temperature should only be used when a facility has comparable feedstock material and conditions.

The exhaust gas results and prediction curves can be applicable to other facilities using similar feedstock materials and facility designs. The feedstock materials used in this study were primarily from an organic dairy farm (cow manure, waste hay, bedded pack, wood chips), with imported horse manure used when adequate supplies of cow manure were not available. The moisture content and C:N ratio are particularly important for assessing whether the exhaust gas results and prediction curves can be used. For example, cow manure has a higher average moisture content (88%) than poultry manure (75%) but is comparable to swine (90%) and horse (80-86%) manure (Rynk et al. 1992; Lorimor et al. 2004). The size of the facility is also important, as composting is not scalable to size due to the insulating effect of additional material and the pore space compaction that can result from too much material.

5. Summary and next steps

ASP heat recovery composting is a novel waste management strategy for processing agricultural by-products on a commercial scale. In this study, the gas emissions (NH_3, CO_2, CH_4) from an ASP heat recovery compost facility were characterized for the first time. Case studies with different operating conditions demonstrated the importance of regular aeration and careful pile temperature management for reducing gas emissions. Under standard operating conditions, exhaust gas temperature was found to be a reasonable predictor of NH_3 and CO_2 concentrations, which can be difficult and costly to measure. Future studies should characterize gas emissions under a range of climate conditions and with different feedstock material mixtures to better understand how management practices influence gas emissions at a commercial-scale facility.

The ASP heat recovery composting approach can promote pollution control pathways that should be studied further. These include capturing emissions with the use of a biofilter; utilizing the exhaust gas to heat and fertilize a greenhouse; using recovered heat and therefore avoiding fossil fuel energy emissions; and increasing carbon sequestration through compost land application. A full life cycle analysis of the pollution sources and management strategies of ASP heat recovery composting should be compared to other manure management methods.

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Appendix A: Compost exhaust gas sampling method specifications

The specifications for the compost exhaust gas sampling instruments are reported below. The gas sampling instruments used were RAE and Dräger colorimetric gas detection tubes (**Table A1**) and the RKI Eagle Portable Gas Detector (**Table A2**).

Table A1. Specifications for RAE Systems and Dräger gas detection tubes. RAE Systems gas detection tubes were used for carbon dioxide and ammonia (<1000 ppm NH_3), and Drager gas detection tubes were used from ammonia (>1000 ppm NH_3). *Sources: RAE Systems 2013, Dräger 2018*

	Carbon dioxide	**Ammonia: RAE**	**Ammonia: Dräger**
Standard range	1-20%	25-500 ppm	0.5 to 10% by volume
Extended range	0.25-20%	12-1000 ppm	
Precision (relative standard deviation)	≤ ± 10%	12%	± 10 to 15 %
Accuracy (20-100% full scale)	10%	10% OR 15%	
Accuracy (<20% full scale)	12%	12% OR 20%	
Humidity	No effect at 5-100% relative humidity	No effect at 10-90% relative humidity. At <5% RH multiply reading by 0.8	3 to 12 mg H_2O/L
Temperature range	0-40°C (32-104°F)	0-40°C (32-104°F) > 40°C, use correction factor of 1.2	10-30°C
Reaction principle	$CO_2 + H_2NNH_2 \rightarrow H_2NNHCO_2H$	$3NH_3 + H_3PO_4 \rightarrow (NH_4)_3PO_4$	

Table A2. Specifications for RKI Eagle Portable Gas Detector. The gas detector was used for oxygen and methane concentrations. The oxygen concentrations recorded with the gas detector were updated using a correction factor with readings from a newly installed oxygen sensor (See **Append B** for more information). *Source: RKI Eagle 2013*

	Oxygen	**Methane**
Standard range	0-40%	0-100% LEL [b]
Extended range [a]	0-40%	0-100% LEL [b]
Accuracy	±0.5% of reading	±5% of reading or ±2% LEL, whichever is greater
Humidity	0-95% relative humidity	0-95% relative humidity
Temperature range	-10-40°C (14-104°F)	-10-40°C (14-104°F)
Type of sensor	Electrochemical gas sensor	Catalytic combustion

[a] The extended range uses a 1:1 dilution fitting.

[b] % LEL is the percent of the lower explosive limit. For methane, the lower explosive limit is 5%. Therefore, the actual methane concentration range is 0-5% by volume.

Appendix B: Gas detector oxygen correction factor

Background

Compost exhaust gas measurements were taken using RAE and Drager colorimetric gas detection tubes (NH_3, CO_2) and an RKI Eagle Portable Gas Detector (CH_4, O_2). The RKI Eagle provides continuous, real time gas concentration readings. However, the RKI Eagle O_2 concentrations were higher than expected ranges from aerobic composting, likely due to the 1:1 dilution fitting used to protect the sensors from the high heat and high humidity compost exhaust gas. After a new inline O_2 monitor (SST Sensing Screwfit zirconium dioxide oxygen sensor) was installed, we observed consistently lower O_2 reading than those from the RKI Eagle. These lower O_2 readings were considered to be more accurate and reliable for several reasons. First, the O_2 sensor measured O_2 concentrations in an expected range, such as readings of <10% O_2 when new feedstock materials were loaded after they were in storage and were likely anaerobic. Second, the 1:1 dilution fitting on the RKI Eagle appeared to pull in more ambient air than sample air because the concentrations recorded by the RKI Eagle were also lower than other measurement methods for CO_2 and NH_3. For these reasons, a comparison between the O_2 sensor and RKI Eagle readings was conducted to attempt to correct the RKI Eagle O_2 readings.

Methods

A series of 20 minute trials was performed for each of the 8 composting bays. Simultaneous O_2 readings were recorded for the O_2 sensor and the RKI Eagle every 2 minutes. The paired sets of data were then plotted on a scatter plot, and a linear regression was performed to assess the strength of the relationship.

Results

A significant correlation was found between the RKI Eagle O_2 measurements and the O_2 sensor measurements (R^2=0.71, p-value=1.01x10^{-132}, **Figure B1**). The RKI Eagle O_2 measurements were consistently less than the O_2 sensor measurements. The readings taken with the RKI Eagle before the O_2 sensor was installed were corrected with **Equation B1**.

Equation B1. Correction for RKI Eagle O_2 readings

$$C = 0.9025A - 2.0129$$

Where C = O_2 monitor reading/corrected O_2 value; A = RKI Eagle reading (uncorrected)

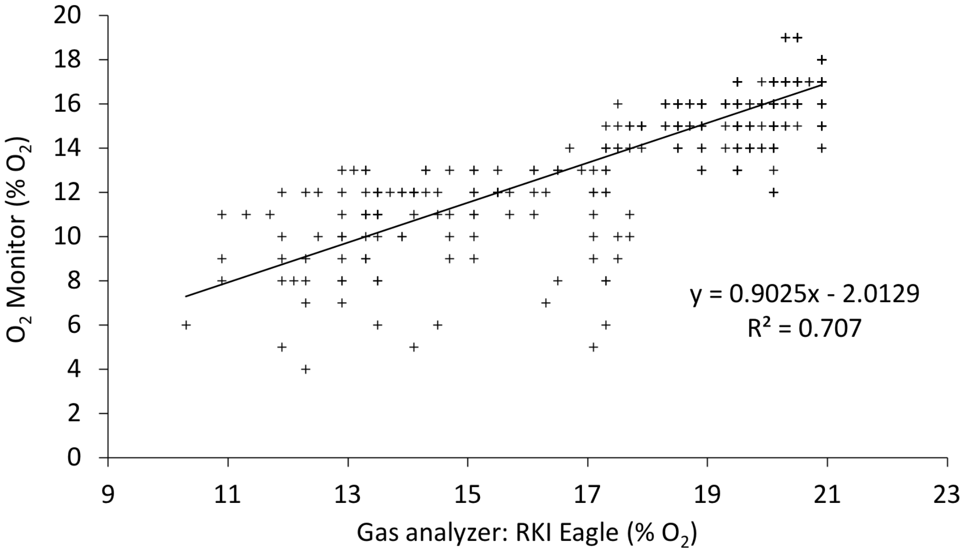

Figure B1. Linear regression for oxygen readings from two methods: the RKI gas detector and an SST Sensing Screwfit zirconium dioxide oxygen sensor. Oxygen measurements taken with the RKI gas detector were corrected based on this linear regression.

Appendix C: Methods testing for compost exhaust gas

Preliminary measurements of compost exhaust gas concentrations indicated that the timing of gas sampling relative to the aeration schedule can affect the gas concentrations. Gas samples are taken when the aeration fan is on to ensure that measurements reflect the composting pile. To promote aerobic decomposition, each composting pile is aerated for 15-45 minutes once every 2-4 hours, depending on how old the composting pile is. New piles are aerated for longer periods of time (up to 45 minutes) to promote decomposition, whereas older piles are only aerated for 15 minutes (personal communication, Matthew Smith, 2017). The aeration schedule is adjusted when pile temperatures begin to decrease, indicating that microorganisms have broken down the easily digestible material and the pile is moving from the thermophilic phase to the final mesophilic phase of composting.

Two questions were addressed to identify when compost exhaust gas sampling should be conducted relative to the aeration schedule:

1. How long should the aeration fan be on before sampling?
2. How long should the aeration fan be off before sampling events?

1. Methods test 1: Aeration fan on time

To determine how long the aeration fan should be on before sampling, frequent gas concentration samples were taken for 1 hour after the aeration fan was turned on. Samples were recorded at the following time points after the aeration fan was turned on: 0.5, 5, 10, 15, 20, 30, 40, and 60 minutes. Gas samples were taken using the RAE colorimetric gas detection tubes. This methods test was repeated 6 times in June 2016. The results were analyzed to identify when gas samples should be taken relative to how long the fan has been on.

Over 6 trials, it was observed that NH_3 concentrations increased with longer aeration times, while CO_2 concentrations decreased with longer aeration times (**Figure C1**). The linear regressions for the 6 trials found a negative correlation for CO_2 (R^2=0.54) and a positive correlation for NH_3 (R^2=0.32). The decreasing CO_2 concentrations and increased NH_3 concentrations are due to the increased aeration. With continuous aeration, any CO2 released from microbial activity would be diluted. Conversely, with increased aeration, piles would dry out and the NH_4^+ dissolved in solution would volatilize to NH_3.

The ideal sampling time was identified as 5 minutes after the aeration was turned on (**Figure C1**). This time point allows for adequate aeration to ensure that concentrations are not just reflecting a buildup of gases in the piles, while it also occurs before the NH_3 and CO_2 concentrations begin to be affected by longer term aeration.

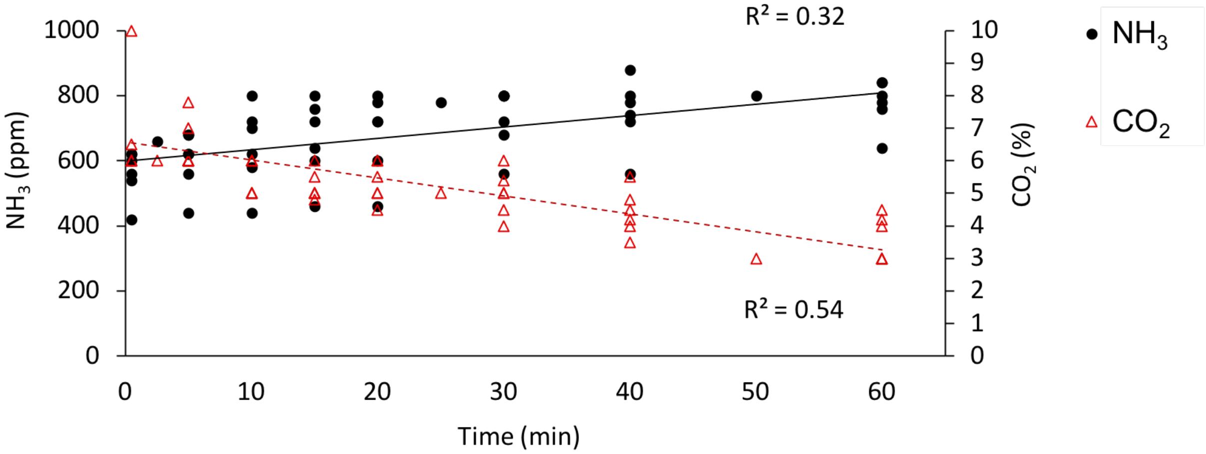

Figure C1. Methods test 1 results: Aeration fan on time. Ammonia (NH_3) and carbon dioxide (CO_2) compost exhaust gas methods test results for the length of time the fan should be on before sampling should occur. The aeration fan was on for the entire period displayed, and the points indicate sampling events.

2. Methods test 2: Aeration fan off time

To determine how long the aeration fan should be off before exhaust gas sampling events, gas concentration samples (NH_3, CO_2) were taken after the fan was off after a range of fan off times. The aeration fan was first turned off for 2 hours, which is the average 'off' time for each bay for the regular aeration schedule. The fan was turned on, and gas concentration measurements were recorded. The fans were then turned off for 5 minutes, and samples were taken again. This process was repeated with the fan turned off for the following lengths of time: 10 minutes, 30 minutes, 60 minutes, 90 minutes, 120 minutes, 180 minutes, and 240 minutes. This methods test was repeated four times in June 2016 with different compost piles to ensure a range of gas concentrations.

The results were compared to identify how long the fan should be off before sampling events. The NH_3 concentrations were variable with shorter fan off times (5-100 minutes), but they appeared to stabilize with aeration times of 120 minutes or greater (**Figure C2**). The CO_2 concentrations also indicated variability with shorter aeration times, but they stabilized more quickly after less than 50 minutes of aeration time (**Figure C3**). This minimum "fan off" time was used for all following gas concentration measurements.

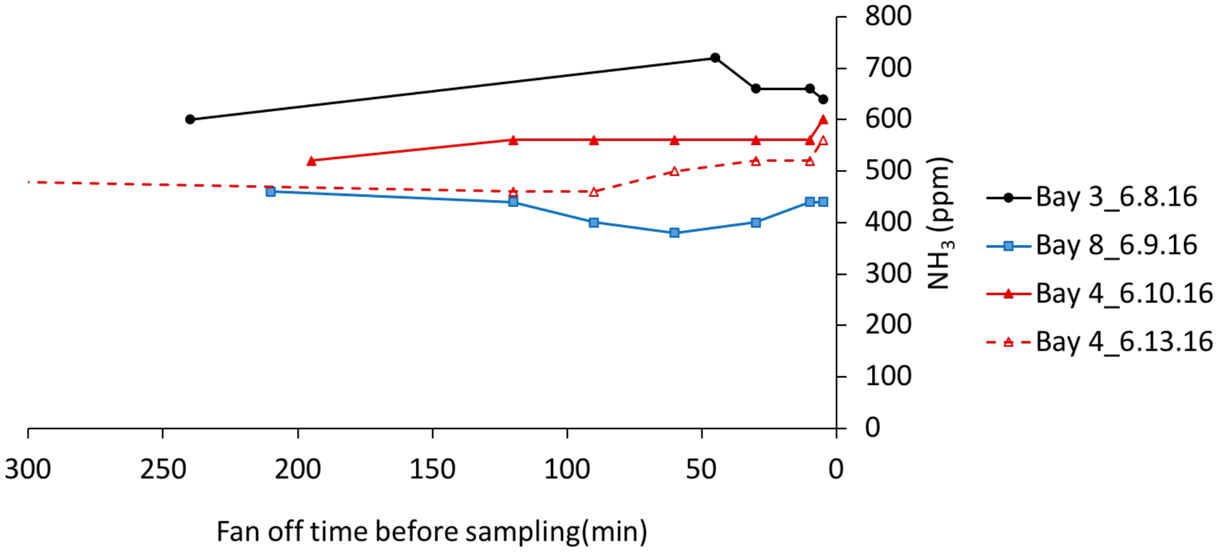

Figure C2. Methods test 2 results for ammonia: Aeration fan off time. Ammonia (NH_3) compost exhaust gas methods test results for the length of time the fan should be off before sampling should occur. The points indicate the sampling events, and the length of time between each sampling event (minutes) indicates how long the aeration fan was turned off between sampling events. The measurements are shown in the opposite order in which they were recorded.

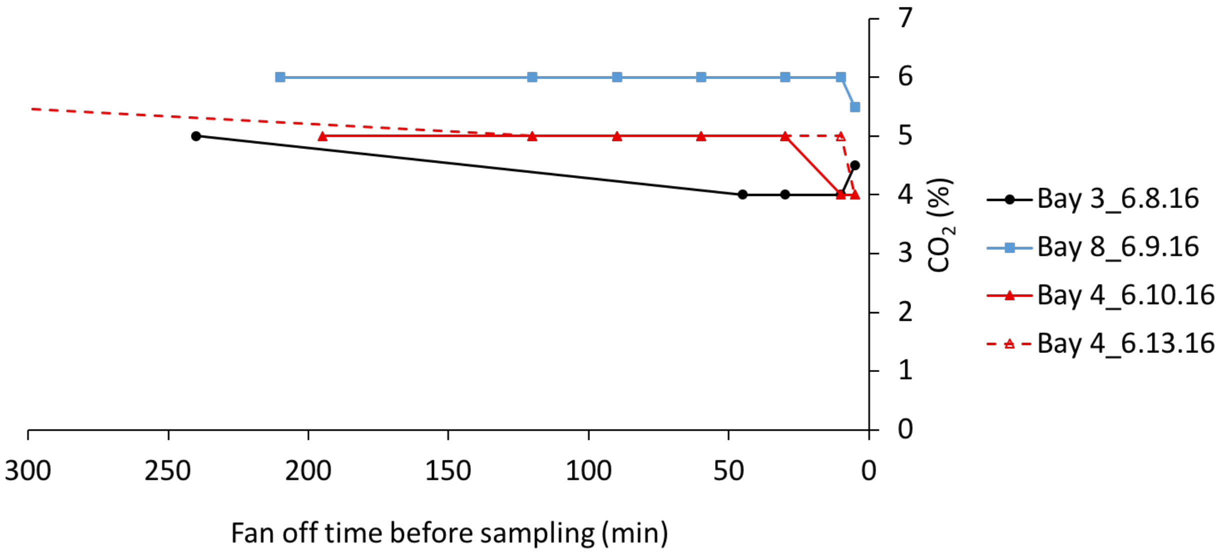

Figure C3. Methods test 2 results for carbon dioxide: Aeration fan off time. Carbon dioxide (CO_2) compost exhaust gas methods test results for the length of time the fan should be off before sampling should occur. The points indicate the sampling events, and the length of time between each sampling event (minutes) indicates how long the aeration fan was turned off between sampling events. The measurements are shown in the opposite order in which they were recorded.

Appendix D: Compost exhaust gas case study results

Detailed results are presented below for each of the three case studies: Blocked air flow (September 2015), delayed microbial activity (October 2015), and low temperature (November 2016). For each case study, figures are shown for the total gas flux, the percent loss of initial carbon and nitrogen in the feedstock, and regressions of exhaust gas temperature with carbon dioxide and ammonia gas concentrations. See the main text for case study results on gas concentrations (ammonia, carbon dioxide, methane) and exhaust gas temperatures.

CASE STUDY 1: Blocked airflow, September 2015

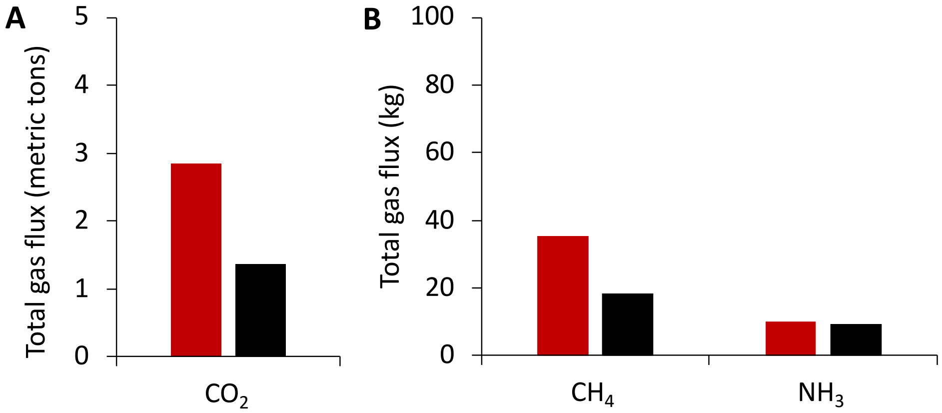

Figure D1. Total gas flux for first 21 days of September 2015 trial. The total gas flux is shown for two compost piles: Bay 1 (red bar) and bay 2 (black bar). A) carbon dioxide (metric tons), B) methane and ammonia (kg).

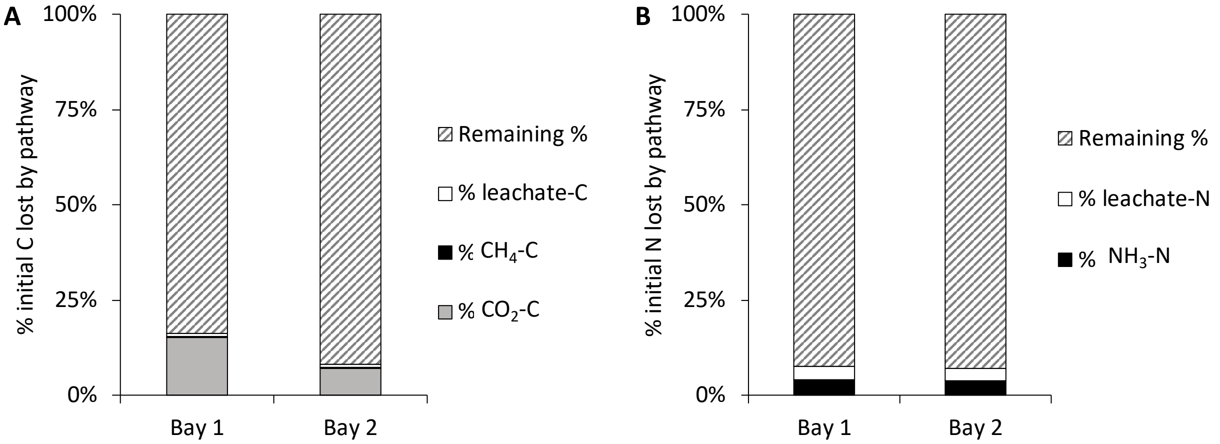

Figure D2. Percent loss of initial carbon and nitrogen during September 2015 trial. Loss pathways for (A) initial carbon and (B) initial nitrogen were gas emissions and collected condensate and leachate. Results are from the September 2015 compost trial for the first 21 days for bays 3 and 4. Remaining % carbon and % nitrogen is contained in the pile or lost (e.g., before loading or from passive emissions).

Table D1. Gas fluxes per metric ton of compost feedstock for September 2015 trial. Gas fluxes include the first 21 days of the compost trial and are per metric ton of dry weight feedstock loaded to the compost facility.

Bay number	Dry weight of compost feedstock *(metric tons)*	CO_2 *(kg CO_2/metric ton dry weight)*	CH_4 *(kg CH_4/metric ton dry weight)*	NH_3 (*kg NH_3/metric ton dry weight)*
Bay 1	11	249	3	0.9
Bay 2	11	119	2	0.8

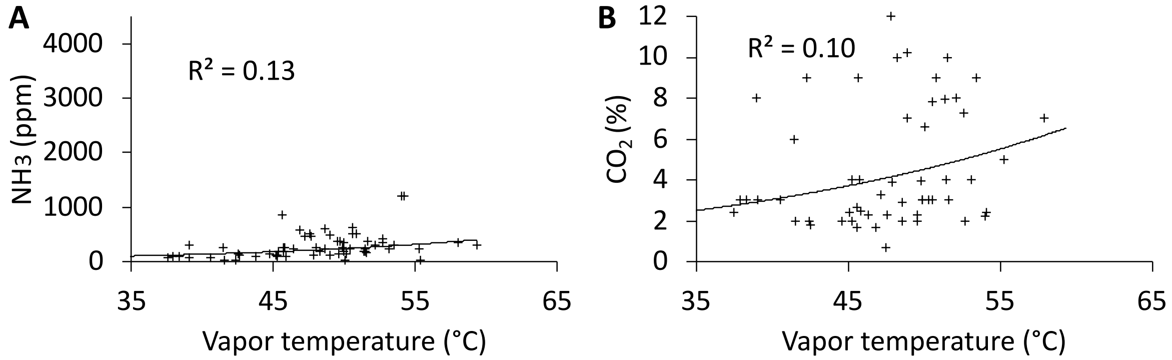

Figure D3. Regressions of exhaust gas temperature and concentrations for Sept. 2015 trial. Exhaust gas temperature regressions for case study 1: Blocked airflow. Regressions shown between exhaust gas temperature and A) ammonia concentrations and B) carbon dioxide concentrations for all sampling results from bays 1 and 2 in the September 2015 trial. Trend lines are exponential trend lines and are shown for demonstration; correlations are not significant.

CASE STUDY 2: Delayed microbial activity, October 2015

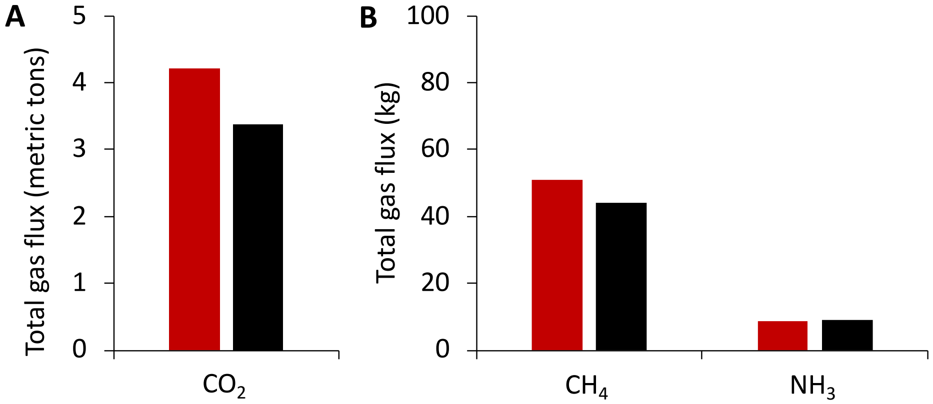

Figure D4. Total gas flux for first 21 days of October 2015 trial. The total gas flux is shown for two compost piles: Bay 3 (red bar) and bay 4 (black bar). A) carbon dioxide (metric tons), B) methane and ammonia (kg).

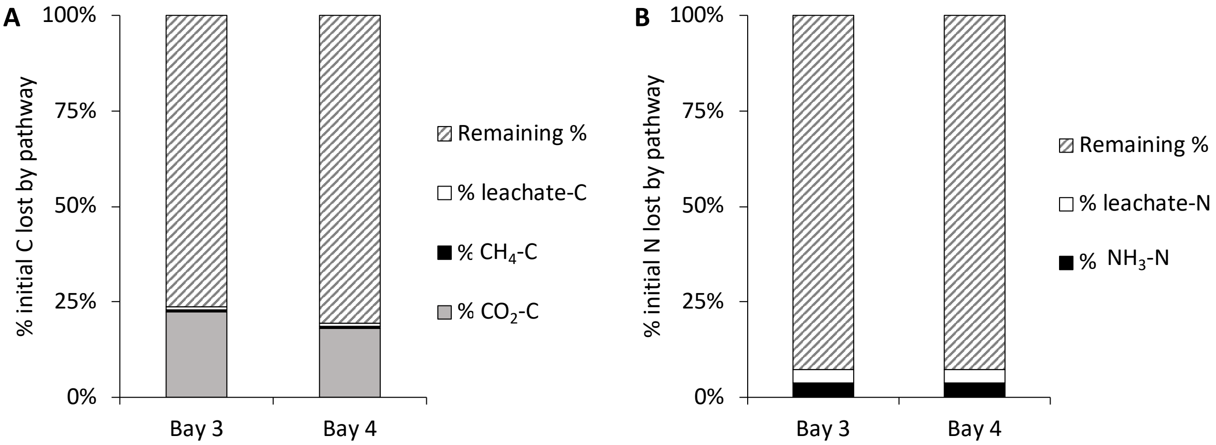

Figure D5. Percent loss of initial carbon and nitrogen during October 2015 trial. Loss pathways for (A) initial carbon and (B) initial nitrogen were gas emissions and collected condensate and leachate. Results are from the October 2015 compost trial for the first 21 days for bays 3 and 4. Remaining % carbon and % nitrogen is contained in the pile or lost (e.g., before loading or from passive emissions).

Table D2. Gas flux per metric ton of compost feedstock for October 2015 trial. Gas fluxes include the first 21 days of the compost trial and are per metric ton of dry weight feedstock loaded to the compost facility.

Bay number	Dry weight of compost feedstock *(metric tons)*	CO_2 *(kg CO_2/metric ton dry weight)*	CH_4 *(kg CH_4/metric ton dry weight)*	NH_3 *(kg NH_3/metric ton dry weight)*
Bay 3	11	367	4	0.8
Bay 4	11	297	4	0.8

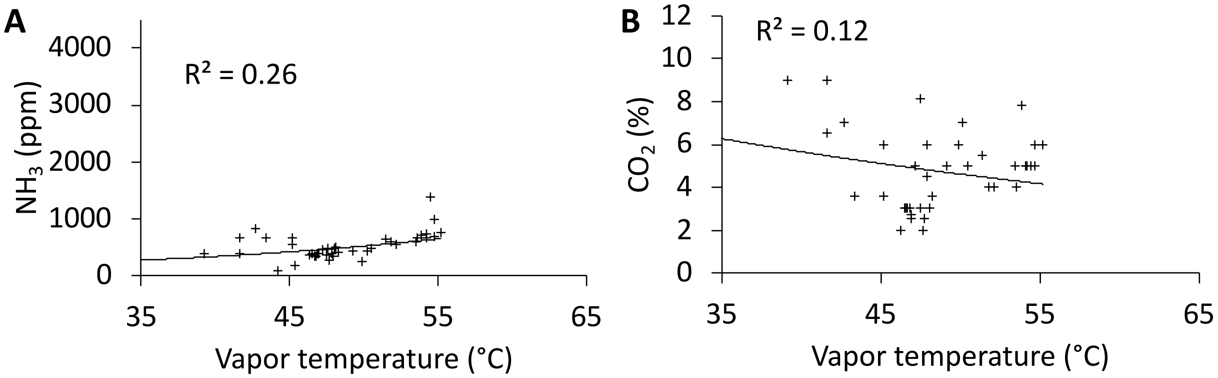

Figure D6. Regressions of exhaust gas temperature and concentrations for Oct. 2015 trial. Regressions are shown for A) ammonia concentrations and B) carbon dioxide concentrations for all sampling results from bays 1 and 2 in the October 2015 trial. Trend lines are exponential trend lines and are shown for demonstration; correlations are not significant.

CASE STUDY 3: Low temperature, November 2016

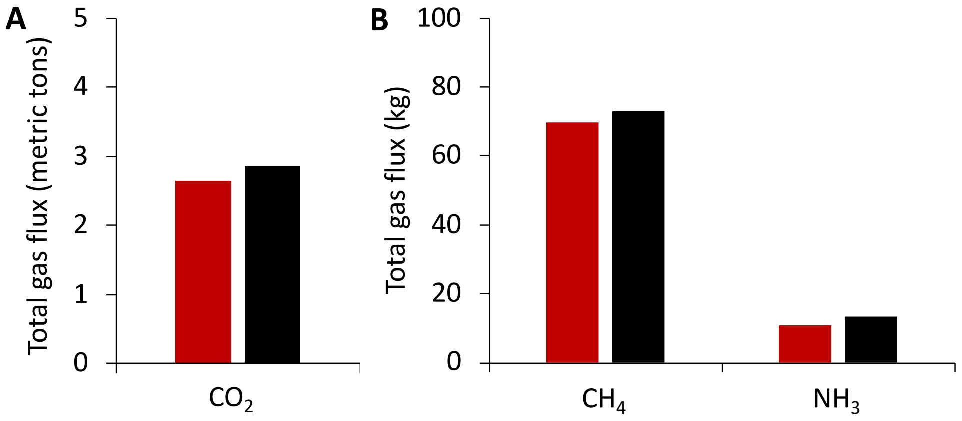

Figure D7. Total gas flux for first 21 days of November 2016 trial. The total gas flux is shown for two compost piles: Bay 5 (red bar) and bay 6 (black bar). A) carbon dioxide (metric tons), B) methane and ammonia (kg).

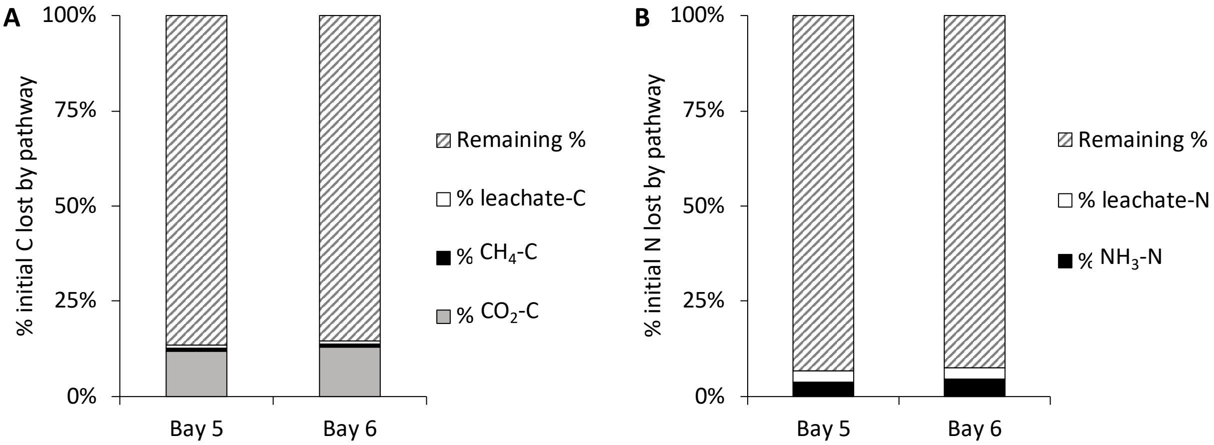

Figure D8. Percent loss of initial carbon and nitrogen during November 2016 trial. Loss pathways for (A) initial carbon and (B) initial nitrogen were gas emissions and collected condensate and leachate. Results are from the November 2016 compost trial for first 21 days for bays 3 and 4. Remaining % carbon and % nitrogen is contained in the pile or lost (e.g., before loading or from passive emissions).

Table D3. Gas fluxes per metric ton of compost feedstock for November 2016 trial. Gas fluxes include the first 21 days of the compost trial and are per metric ton of dry weight feedstock loaded to the compost facility.

Bay number	Dry weight of compost feedstock *(metric tons)*	CO_2 *(kg CO_2/metric ton dry weight)*	CH_4 *(kg CH_4/metric ton dry weight)*	NH_3 *(kg NH_3/metric ton dry weight)*
Bay 5	13	198	5	0.8
Bay 6	13	214	5	1.0

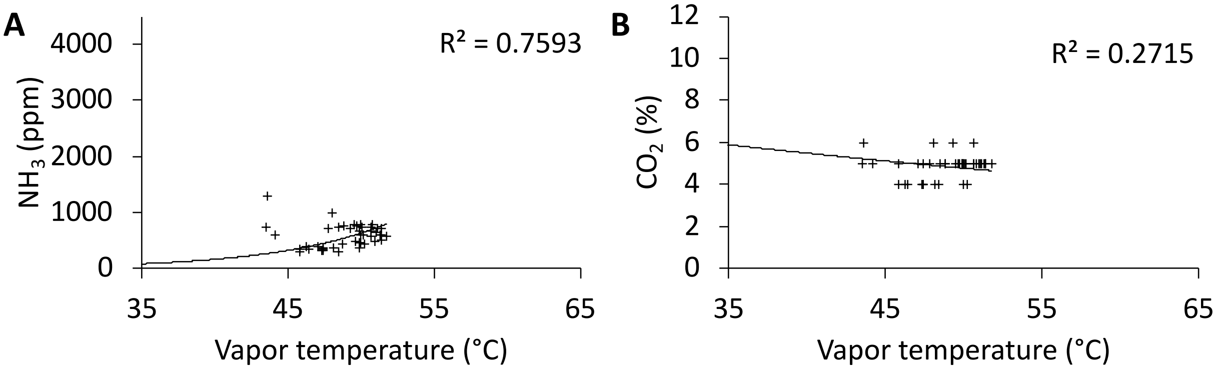

Figure D9. Regressions of exhaust gas temperature and concentrations for Nov. 2016 trial. Regressions are shown for A) ammonia concentrations and B) carbon dioxide concentrations for all sampling results from bays 5 and 6 in the November 2016 trial. Trend lines are exponential trend lines and are shown for demonstration; correlations are not significant.