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Quantifying Soil Greenhouse Gas Emissions And Soil Carbon Storage To Determine Best Management Practices In Agroecosystems

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QUANTIFYING SOIL GREENHOUSE GAS EMISSIONS AND SOIL CARBON STORAGE TO DETERMINE BEST MANAGEMENT PRACTICES IN AGROECOSYSTEMS

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ABSTRACT

Intensive agriculture, coupled with an increase in nitrogen fertilizer use, has contributed significantly to the elevation of atmospheric greenhouse gases (GHGs), including carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). Rising GHG emissions usually mean a decrease in soil carbon. Currently, soil C is twice that of all standing crop biomass, making it an extremely important player in the C cycle. Fortunately, agricultural management practices have the potential to reduce agricultural GHG emissions whilst increasing soil C. Management practices that impact GHG emissions and soil C include various tillage practices, different N fertilization amounts and treatments (synthetic N, cattle manure, or a combination of both), the use of cover crops, aeration, and water levels. Employing agricultural best management practices (BMPs) can assist in the mitigation and sequestration of CO_2, N_2O and soil C. Measuring soil carbon storage and GHG emissions and using them as metrics to evaluate BMPs are vital in understanding agriculture's role in climate change. The objective of this research was to quantify soil carbon and CO_2 and N_2O emissions in agroecosystems (dairy, crop, and meat producing farms) under differing management practices.

Three farms were selected for intensive GHG emissions sampling: Shelburne Farm in Shelburne, VT, a dairy in North Williston, VT, and Borderview Farm in Alburgh, VT. At each site, I collected data on GHG (CO_2 and N_2O) emissions and soil carbon and nitrogen storage to a depth of 1 meter. Soil emissions of CO_2 and N_2O were taken once every two weeks (on average) from June 2015 through November, 2015 using static flux chambers and a model 1412 Infrared Photoacoustic Spectroscopy (PAS) gas analyzer (Innova Air Tech Instruments, Ballerup, Denmark). Fluxes were measured on 17 dates at Shelburne Farms, 13 dates at the Williston site, and 13 dates in the MINT trial. Gas samples were taken at fixed intervals over a 10-14 minute time frame, with samples normally taken every one or two minutes. I also measured soil carbon to a depth of 1m in six BMPs at Borderview Farm.

Overall, I found that manure injection increased N_2O and CO_2 emissions, but decreased soil C storage at depth. Tillage had little to no impact on N_2O emissions, except at Shelburne Farms, where aeration tillage decreased N_2O emissions (marginally significant, $P < 0.1$). No-till did, however, decrease CO_2 emissions relative to other conservation tillage practices (strip and vertical tillage) but we were unable to detect a significant change in soil C due to tillage practices. At Borderview farm, N_2O emissions increased with soil NO_3 and soil moisture, while CO_2 emissions increased with soil temperature and nitrate. At Williston, CO_2 emissions only increased with temperature; at Shelburne CO_2 emissions increased with nitrate. N_2O fluxes at Shelburne and Williston were not associated with any of the measured covariates.

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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

Figure Page

CHAPTER 1: COMPREHENSIVE LITERATURE REVIEW

1.1. Introduction

With human population projection estimates pointing to nine billion by year 2050, the importance of maintaining Earth's basic ecosystem services has quickly become increasingly important. Supporting this expanding population with enough food, fiber, and fuel has intensified demands on agricultural land and other natural resources (Haile-Mariam et al., 2008). Intensive agriculture, coupled with an increase in nitrogen (N) fertilizer use, has contributed significantly to the elevation of atmospheric greenhouse gases (GHGs), including carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O)(Haile-Mariam et al., 2008). These three GHGs differ noticeably in their atmospheric concentrations, residence time in the atmosphere, and global warming potential (GWP) (Leibig et al., 2012). Of the three GHGs, N_2O is present in the lowest atmospheric concentrations but has the greatest GWP, at 298 times that of CO_2 (IPCC, 2007). While the agricultural sector accounts for a negligible amount of global CO_2 emissions (not accounting for land-use change or secondary energy emissions) and approximately 34% of CH_4 (mostly from enteric fermentation and manure management) (DRAFT Inventory of U.S. GHG Emissions and Sinks, 2015), it is the largest anthropogenic source of N_2O in the U.S., accounting for 69% of the total N_2O emitted, and 92% of agricultural N_2O emissions are derived from soil management practices (Liebig, et al., 2012).

Overall, the agricultural sector may be a large GHG source, and management practices can substantially reduce (or increase) agricultural GHG emissions. For example, applying N fertilizer in the spring, when plant demand for nutrients is high, rather than in the fall can substantially reduce N_2O emissions (Millar et al. 2010). Agricultural management can also decrease soil emissions of CO_2 as well as maximize the storage of atmospheric carbon in crop biomass and eventually in soil organic matter (Johnson, et al, 2007). Such GHG emissions reductions can be accomplished using a bevy of agricultural best management practices (BMPs) including conservation tillage or no-till, use of nitrogen fixing cover crops, reduced soil compaction, reduction of synthetic N fertilizer, and better manure management (Hatfield and Sauer, 2011). Because of agriculture's significant role in GHG emissions, implementing BMPs on agricultural systems has the potential to prevent thousands of tons of GHGs from entering the atmosphere (Johnson, et al, 2007).

1.2 Drivers and impacts of agricultural management practices on GHG emissions

1.2.1 Carbon Dioxide & soil C storage

Carbon dioxide emissions from soil is a natural component of the carbon cycle. In total, more than twice as much carbon is stockpiled in the world's soil than in the vegetation or atmosphere combined (Ciais et al, 2013). Of the carbon stored in soil, soil organic carbon (SOC) makes up about 50% of all soil organic matter (SOM) (Pribyl 2010).

Soil CO_2 flux is primarily the result of a combination of microbial decomposition of SOM and plant root respiration (Savage et al., 2014). The main drivers of soil CO_2

flux are soil temperature, soil moisture, and substrate carbon (C) availability (Raich & Schlesinger 1992; Lloyd & Taylor 1994; Raich & Tufekciogul 2000; Rustad, et al., 2000; Hogberg, et al. 2001; Scott-Denton et al. 2006).

Temperature affects CO_2 flux by speeding up the rate of microbial decomposition when soils are warm and water is not limiting (Wan et al. 2007; Lloyd & Taylor 1994). Although rising temperatures cause an increase in CO_2 flux rate from soils, in some parts of the world there are no clear trends of decreasing soil carbon with increasing mean annual temperature (Thornley and Cannell, 2001). This is due, partly, because of competing processes within the system, such as soil carbon increasing due to increased primary productivity as a result of better water and nutrient availability, but decreased by increased respiration (Thornley and Cannell, 2001). While in the short-term warming does deplete soil carbon, in the long-term, carbon losses by accelerated microbial respiration may be equalized by increases in carbon inputs to the soil owed to increased net primary production, as well as any acceleration of soil physico-chemical 'stabilization' reactions (Thornley and Cannell, 2001). Additionally, changes in microbial community composition or declines in the temperature sensitivity decomposition processes may reduce the response of microbial respiration to increasing temperature over time (i.e., thermal acclimation (Wallenstein et al, 2011, Wei et al. 2014).

Cold soil and air temperatures have the opposite effect on CO_2 flux rate, causing it to slow down. Even though slowed, soil microorganisms maintain both catabolic (CO_2 production) and anabolic processes (biomass synthesis) under frozen conditions (R.K. Shrestha et al, 2013). Because of this, gaseous exchange between the atmosphere and soil does not stop even under frozen soil, resulting in the accumulation of CO_2 during winter and its release into the atmosphere during spring thaw events. (R.K. Shrestha et al, 2013).

Another of the dominant factors controlling the net exchange of GHG's is soil moisture, which can vary dramatically over time and space (Savage et al., 2014). The production and transport of GHG's in soil is strongly affected by changes in soil moisture through diel cycles, wet-up and dry-down events, management practices, seasonal patterns, and interannual variation in climate (Borken et al, 2006). Overall, when water is limiting, plant and microbial availability increase with soil moisture, thereby increasing soil CO_2 flux directly by alleviating plant and microbial desiccation stress and indirectly by increasing substrate availability (via higher rates of plant growth, photosynthesis, belowground C allocation) and microbial access to substrate (e.g., increased C diffusion through soil water; Wan et al. 2007).

Finally, respiration generally increases with C availability. Plant respiration is largely dependent on C from current photosynthetic activity (Hogberg et al. 2001) and, under non-limiting soil temperatures and moisture availabilities, microbial respiration increases with labile C availability (Hungate et al. 1997). Thus, soils with high organic matter inputs and stocks, like those found near the equator, means greater C substrate availability, which is synonymous with greater flux (Thornley and Cannell, 2001).

Depth and placement of soil carbon is yet another factor to consider when attempting to make precise conclusions about CO_2 flux. For example, in agroecosystems, the bulk of SOM is within the top 10 cm of the soil surface. Because of this, temporal dynamics of CO_2 flux are more intimately related to air temperature than to soil temperature (Parkin and Kaspar, 2003). Also, it is known that the respiration rates of many soils are strongly linked with the amount of carbon not intimately associated with minerals. Mineral soil occurs below the litter and organic layer, where soil carbon may be closely associated with mineral particles—accounting for over 60% of carbon in most forest soils (Parkin and Kaspar, 2003). Liski et al. (1999) and Giardina and Ryan (2000) proposed that the decomposition/respiration rate of mineral soil carbon is relatively insensitive to temperature (Thornley and Cannell, 2001). This is because the carbon located here may be protected from microbial mineralization by stabilization mechanisms, such as occlusion in soil aggregates (physical protection) or interactions with mineral surfaces (chemical sorption to mineral surfaces (O'Brien and Jastrow, 2013).

1.2.2 Nitrous Oxide

Nitrous oxide contributes significantly to atmospheric warming because it is a powerful GHG that can persist for up to 150 years and has approximately 300 times the effective heat trapping capability of CO_2 (IPCC 2007). It is also a potent stratospheric-ozone-depleting chemical, further compounding global warming concerns (Thomson et al, 2012). Because agriculture is the largest anthropogenic source of N_2O, much research centers around determining the drivers of soil N_2O emissions and pinpointing strategies to reduce N_2O emissions from agricultural soils (Venterea 2014).

Both nitrification and denitrification contribute to N_2O emissions. Nitrification transforms ammonium to nitrite (NO_2^-) and then nitrate (NO_3^-), which is frequently considered a limiting substrate for denitrification (Xue et al, 2013). In anaerobic conditions, denitrification, a process that involves four enzymatically catalyzed reductive steps (Wallenstein et al, 2006) converts soil NO_3^- into NO_2^-, NO, N_2O, and finally N_2 (Inselbacher et al, 2011). Denitrification is a facultative anaerobic microbial process that involves a diverse group of phylogenetically unrelated bacteria, including members of the Aquificae, Deinoccoccus-Thermus, Firmicutes, Actinobacteria, Bacteroides and Proteobacteria phyla (Wallenstein et al, 2006). Fungi and Archaea are also capable of denitrification (Wallenstein et al, 2006). It is the varied composition of denitrifiers across different soil environments that make concrete determinations about N loss a challenging process.

Nitrification and denitrification are regulated by many soil factors, including soil texture, water content, soil temperature, aeration, the amount of soluble organic carbon, soil pH, and the communities of soil microbes present (Granli and Bockman, 1994). Key factors impacting N_2O emissions specifically are temperature, soil water regime and availability of C and N substrates (Xue et al, 2013). O_2 partial pressure and soil pH are also important (Richardson et al, 2009).

Temperature regulates microbial activity, and N_2O emissions have an exponential association with increasing temperature when substrate and moisture availability are not limiting factors (Xue et al, 2013). Upward mass flow of N_20 as warming soil air expands has also been observed (Richardson et al, 2009).

Moisture, or soil water, is another leading controlling factor in N_2O emissions from soil. Relatively large denitrification rates typically occur under anoxic soil conditions when C and N substrates are abundant. The presence of anaerobic conditions is the key driving factor in allowing denitrifiers to use the supply of carbon as an energy source and nitrate as an electron acceptor (Xue et al, 2013).

While soil ammonium (NH_4^+) and NO_3^- have both been found to influence rates of nitrification, denitrification, and N_2O emission (Baggs and Blum, 2004), Venterea (2014) found that soil nitrite (NO_2^-) levels had the strongest correlation with amount of N_2O emitted from soil. Venterea found that neither soil NO_3^- nor NH_4^+ levels had similar correlations with N_2O.

Though N_2O emissions are influenced by many variables, the soil microbial community controls an immense stake in the processes of soil N_2O emissions. According to Inselbacher et al. (2011), soil microbes cannot be treated as a uniform pool in the soil. For example, the denitrifying bacteria *Paracoccus denitrificans* has a unique sequence of triggers for enzyme production that results in early, high levels of N_2O reductase and only trace emissions of N_2O during denitrification (Bakken et al. 2012). In contrast, *Agrobacterium tumefaciens,* another bacterial denitrifier, does not produce nitrous oxide reductase and is therefore unable to reduce N_2O to N_2 (Bakken et al. 2012). However, the diversity of microbes involved combined with the species specific approaches to denitrification mean that a predictive linkage between microbial community composition and N_2O flux rates (or to $N_2O/(N_2O + N_2)$ efficiencies) remains problematic at best (Bakken et al. 2012).

The fact that so much N_20 is produced from bacterial denitrification indicates that the enzyme nitrous oxide reductase (NOR), which is responsible for the final step of denitrification, or the bacterial population as a whole do not always carry out the final step either proficiently or in synchrony with upstream parts of the pathway (Richardson et al, 2009). N_2O emissions could be drastically reduced if there was an easy means of ensuring that N_2O would be reduced into N_2. One key to the puzzle is realizing that the bacterial group responsible for executing the final step of converting N_2O to N_2 may depend on a co-factor: copper. In a study done by Richardson et al. (2009), removing copper ions from a bacterial culture while it was carrying out denitrification caused N_2O emissions to rise, whereas adding copper to the growth medium caused N_2O emissions to drop and N_2 emissions to increase. Because N_2O reductase is a copper enzyme, biological N_2O consumption is an obligatory copper-dependent process, and though a copper deficient bacterial community is expected to remain viable, it will continue to emit N_2O.

1.2.3 Agricultural Management Impacts on Soil Carbon and Greenhouse Gas Emissions

Globally, we stand on the brink of some major opportunities in agriculture and food production for lowering the production of GHG (Richardson et al, 2009). There are several BMP's to consider when addressing agriculture's role in sequestering GHG's, but the American Society of Agriculture, Crop Science Society of America, and Soil Science Society of America's (ASA-CSSA-SSSA) Greenhouse Gas Working Group has provided five wide-ranging strategies for mitigating agricultural GHG emissions (Greenhouse Gas Working Group, 2010):

1. Enhance soil C sequestration;
2. Improve N-use efficiency;
3. Increase ruminant digestion efficiency;
4. Capture GHG emissions from manure and other wastes, and;
5. Reduce fuel consumption.

Of the five, several are particularly relevant to agriculture as practiced in the NE US: improving N-use efficiency and enhancing soil C sequestration via tillage and manure management practices. The improvement of N fertilizer use-efficiency was listed as one of the primary modes in which to reduce GHG emissions from a known source (Liebig, et al., 2012). This improvement of efficiency involves making N available to plants in the amount needed at the correct time to meet plant demand (Liebig, et al., 2012). Major N losses often occur during the first week after applying N fertilizer and manure, with additional elevated N losses normally continuing over the following three weeks (Inselbacher et al, 2011). Improving N fertilizer and manure efficiency and application techniques may successfully result in less reactive N available for potential conversion to N_2O.

Throughout the US, the application of fertilizer in the form of manure is a common practice (Greenhouse Gas Working Group, 2010). Nutrients available in manure consist of nitrogen, phosphorus, potassium, calcium, magnesium, zinc, iron, manganese, copper, sulfur, and boron (Eghball et al, 2002).

Proper timing of manure application is an important way to reduce potential losses. Vermont has a prohibition on manure application between December 15th and April 1st; many farms have chosen to focus their efforts on herd and crop management and hire "custom operators" to spread manure in the spring, once during the summer and again in the fall. Typically, custom operators arrive at a farm according to a set schedule and empty lagoons regardless of the weather (vtwaterquality.org).

Application methods also impact N losses and N_2O emissions. Manure injection is expected to reduce N losses via ammonia (NH_3) volatilization, but it may increase N_2O

emissions (Maguire et al, 2011). Nitrous oxide emissions in agricultural settings vary widely across the landscape, however, and in some cases, N_2O fluxes may be more closely related to soil properties than to the timing and method of N fertilizer sources applied (Haile-Mariam et al., 2008).

Timing, amount, application method, and form of N fertilization may interact with soil properties to affect N losses and N_2O emissions, and while N fertilization may influence all soil microbes, effects can vary between different soils and/or different communities of micro-organisms (Cavagnaro et al., 2008 and Wallenstein 2006). Because of this, fertilizer induced changes in soil processes and GHG emissions may also be dependent on soil properties and the characteristics and activities of the soil microbial community (Inselbacher et al., 2011).

Conservation tillage, a soil management practice regaining favor in the NE and throughout the US reduces loss of soil and water relative to conventional tillage (Brady and Weil, 2010). There are several types of conservation tillage methods, including minimum till, mulch till, ridge till, strip till, and no-till. In contrast, conventional tillage is usually thought of as a tillage practice that encourages the turning of soil completely in order to prepare the seedbed, as well as a means for weed control (Brady and Weil, 2010). Conventional tillage disrupts soil structure, exposing previously protected soil organic matter to decomposition (O'Brien and Jastrow 2013). This disturbance stimulates soil microbial activity (i.e. respiration) by increasing the availabilities of both oxygen and soil organic matter for microbial decomposition (Brady and Weil, 2010). Turning the soil also moves non-grain crop residue from the soil surface to underground, leaving the soil less protected from wind and water erosion (Brady and Weil, 2010). The increase in respiration by the soil microbes correlates to a direct increase in the amount of GHG's being emitted by that system.

The systems being studied for this research are under a combination of conventional and conservation tillage management practices common in the northeastern US. Conservation tillage regimes usually mean that next years' corn crop is planted directly into a seedbed not tilled since harvest of the previous crop. The primary advantage of no-till over conventional tillage is that it does not disturb the soil habitat and leaves anywhere from 50 to 100% of the soil surface covered with non-grain crop residues (Brady and Weil, 2010). Reduced soil disturbance is thought to result in far lower GHG emissions in comparison to conventional tillage, but reduction in climate change forcing through carbon sequestration under no-tillage can potentially be offset by increases in soil N_2O emissions (Richardson et al, 2009). This is because higher soil carbon levels and smaller porosity in soils under no-tillage often induce higher denitrification rates and N_2O losses (Richardson et al, 2009). However, when the additional benefit of decreasing the use of tractor fuel compared to conventional tillage is considered, the benefits of reduced tillage outweigh conventional tillage again, with less net GHG's emitted (Powlson et al, 2012).

Changes in the profile distribution of soil C stocks for conventional versus no-tillage can also affect N_2O losses (Xue et al, 2013). Under conventional tillage, large N_2O losses may occur due to the combination of greater soil C content at deeper layers (ploughed soils) and moist profiles after N fertilizer application (humid regions; Xue et al, 2013). Additionally, deep N placement (via manure injection) appears to aggravate rather than ameliorate these concerns and inverted C profiles create larger N_2O

emissions, presumably because of the greater C contents where soil conditions are wetter (Xue et al, 2013). Still, the consensus is that the benefits of a no-till system outweigh the previously mentioned concerns, thus the amount of acres being managed as no-till have expanded to nearly all regions of the United States including Vermont (H. Darby personal communication, 2014), and are now implemented in some form on almost half of all conservation tillage acres (Brady and Weil, 2010).

Aside from tillage practices alone, water management also plays an important role in the potential minimization of GHG's from agroecosystems. According to Xue et al (2013), soil moisture content influenced CO_2 flux during the growing season, but not in the dormant season. It is widely recognized that delivering water to crops in precise doses with minimal loss is one way to increase water and nutrient-use efficiency (Delgado et al., 2011).

In Vermont, aeration tillage is also considered to be a type of conservation tillage (agriculture.vermont.gov). Aeration tillage is defined as a minimum tillage technique that is used in conjunction with conventional liquid manure application on perennial croplands such as pasture or hay fields (agriculture.vermont.org). Aeration tillage is used to combat soil compaction in permanent hay fields, which can result in anaerobic conditions and poor water infiltration; aeration can increase water infiltration and reduce erosion and runoff of water, N and P (DeLaune and Sij 2012, DeLaune et al. 2013). It may also decrease losses of manure when aeration is followed by manure application (Harrigan et al, 2006). However, there is very little or no information regarding the impact of aeration tillage on GHG emissions. Furthermore, while tillage and manure application methods may each have impacts on C storage and GHG fluxes, even less is known about how these methods interact to impact C and GHG fluxes.

Finally, it can be extremely difficult to parse out each GHG process and treat it as independent, when in fact, all of the processes are intimately connected. One example of this difficulty arises when looking at the role methanotrophs play. Methanotrophs significantly contributed to nitrification in the rhizosphere, while the contribution of nitrifiers to CH_4 oxidation was insignificant. This indicates that the beneficial effect of methanotrophs on GHG balance could be reduced by the production of NOx (Le Mer and Roger, 2001).

As the sources of atmospheric GHG's are closely related to human activities, it is theoretically possible to control them (Le Mer and Roger, 2001). However, it is unlikely that it will ever be possible to develop farming practices that completely eliminate denitrifier-N_2O emissions from agriculture. Only when we have greater understanding of the production and reduction of N_2O will it be possible to provide farmers with more precise prescriptions to minimize N_2O emissions for, say, application of nitrogenous or copper fertilizer, SOM management and, where necessary, liming of crops or grasslands with specific characterized carbon and nitrogen traits (Richardson et al, 2009).

1.3 Research Goals and Hypothesis

The objective of this research was to investigate the impact of current agricultural BMPs on GHG emissions in the NE US. To this end, I analyzed soil carbon storage and soil CO_2 and N_2O emissions in various agroecosystems (dairy, crop, and meat producing farms) under differing management practices, including various tillage practices, different N fertilization amounts and treatments (synthetic N, cattle manure), the use of cover crops, aeration, and water levels. I expected that employing agricultural BMPs in temperate agricultural systems would assist in the sequestration of carbon in soils and mitigation of CO_2 and N_2O emissions.

Overall, we expected that: (1) relative to conventional tillage, conservation tillage practices would increase soil C storage and decrease CO_2 fluxes, but increase N_2O fluxes by increasing soil structure and the opportunity for low O_2 and high soil moisture conditions; and (2) while methods that incorporate manure below ground in the absence of conventional tillage (e.g., aeration or injection) may decrease C and N losses via runoff, NH_3 loss, and CO_2 flux, these methods may increase N_2O fluxes by creating high N and high moisture microsites that promote denitrification.

To investigate these hypotheses we quantified GHG emissions and soil C storage at three sites:

1) Shelburne Farm: The sites being studied here are two perennial hay fields, under differing management practices, with almost identical sizes, slopes and drainage patterns. One field is being managed under an aeration practice (four anchors), while the other is not being aerated (four anchors). This pairwise study will be ideal for looking at the difference aeration causes in GHG emissions between hay fields.

2) N. Williston Cattle Company: These sites are two corn plots under differing management practices. One plot is under a conservation tillage practice with manure being injected into the soil. A cover crop was left over winter and was not removed or killed prior to the corn planting date (Field 1; four anchors).

The second plot is managed under a conventional tillage practice with manure being broadcast and left on the surface (Field 2; four anchors).

3) MINT trial at Borderview Farm: This trial is in a continuous corn system. There are three tillage treatment plots (vertical till, strip till, no till) that are 40 feet wide by 192 feet long, with 40 feet buffers between them. Within each tillage plot there are two manure application methods: broadcast and injected. Each tillage and manure treatment combination is replicated four times (i.e., each manure treatment is replicated four times within each tillage plot; 28 anchors). Measurements taken from these treatments will be compared to measurements taken in conventional agricultural (control) management plots: conventional tillage with broadcast manure (four replicates). Within each treatment combination we will measure soil C and N, mineral N, soil moisture, temperature and GHG emissions (CO_2, N_2O and CH_4).

Based on our overarching hypotheses, we predict that:

1) The aerated field at Shelburne Farms will exhibit greater cumulative CO_2 emissions due to the soil microbes having access to greater oxygen levels at depth. Also predicted is that N_2O emissions will be lessened in the aerated field. This is because denitrification is an anoxic process, so the availability of increased oxygen in the aerated field suggests that N_2O emissions will be muted.

2) The hypothesis for N. Williston is that Field 1 will have less CO_2 emissions than Field 2, but will perhaps show an increase in N_2O emissions, since the N will be placed in a more anoxic profile location within the soil in Field 1 (via injection).

Another reason that N_2O emissions can be expected to be greater in Field 1 is because some of the N being broadcast on Field 2 will be lost as ammonia (NH_3), reducing the total amount of N available for transformation into N_2O.

3) The hypothesis for the MINT trial site is that the control plots will have elevated CO_2 emissions in comparison with the conservation tillage plots. This is due to conventional plowing, which increases oxygen availability to the soil microbes, increases available carbon by breaking up soil aggregates, and possibly also because the C substrate available for respiration will be placed deeper in the soil profile (versus remaining on the surface), providing an increased C stock for the soil microbes present there. Differences in CO_2 emissions between different types of conservation tillage replicates is difficult to predict, but the no-till plots should have the lowest CO_2 emissions due to less oxygen available for respiration, and less soil disturbance. It's hypothesized that N_2O emissions will be greatest in the no-till plots where manure is injected. This is because no-tilled soils exhibit greater water retention capabilities, allowing denitrification to take place, and with the N being placed within the soil profile rather than being left on the surface, the N will be placed in a more favorable position within the soil profile for denitrification.

CHAPTER 2: GREENHOUSE GAS EMISSIONS IN VERMONT SOILS: HOW AGRICULTURAL PRACTICES IMPACT FLUX RATES

2.1 Abstract

Three farms were selected for intensive GHG emissions sampling: Shelburne Farm in Shelburne, VT, a dairy in North Williston, VT, and Borderview Farm in Alburgh, VT. At each site, I collected data on GHG (CO_2 and N_2O) emissions. Soil emissions of CO_2 and N_2O were taken once every two weeks (on average) from June 2015 through November, 2015 using static flux chambers and a model 1412 Infrared Photoacoustic Spectroscopy (PAS) gas analyzer (Innova Air Tech Instruments, Ballerup, Denmark). Fluxes were measured on 17 dates at Shelburne Farms, 13 dates at the Williston site, and 13 dates in the MINT trial. Gas samples were taken at fixed intervals over a 10-14 minute time frame, with samples normally taken every one or two minutes. Our results indicate that manure injection increases both CO_2 and N_2O emissions relative to broadcasting manure. Aeration decreased N_2O emissions in comparison to non-aeration. No-till produced a significant decrease in CO_2 fluxes, but both strip-till and vertical-till caused an increase in CO_2 emissions. The increases in CO_2 emissions seen in the vertical and strip-till managements were only significant when soil temperature was not accounted for. When soil temperature was taken into account, the tillage practices were no longer significant, implying that the higher soil temperatures in those plots were due to higher soil temperatures instead.

2.2 Introduction

Soils have been a major source of atmospheric CO_2 and other GHGs (CH_4, N_2O) ever since the beginning of settled agriculture (Ruddiman 2003, 2005, Haile-Mariam et al. 2008). The magnitude of CO_2-C emission from soil to the atmosphere since the industrial revolution (~1750 AD) is estimated at 78 ± 12 Pg (Lal, 1999) and most soils managed under agricultural practices are depleted in soil organic carbon (SOC; Singh et al. 2011). Although N_2O is present in the lowest atmospheric concentrations, it has the greatest GWP, at 298 times that of CO_2 (IPCC, 2007). The agricultural sector is the largest anthropogenic source of N_2O in the U.S., accounting for 69% of the total N_2O emitted and 92% of agricultural N_2O emissions are derived from soil management practices (Liebig, et al., 2012).

Because agricultural SOC pools are largely depleted and N_2O emissions are heavily dependent on fertilization and management practices, there lies significant potential for agricultural best management practices (BMPs) to mitigate future GHG emissions. Some well known agricultural BMPs include use of conservation agriculture with crop residue mulch and cover cropping, integrated nutrient management with liberal use of compost and manure in conjunction with chemical fertilizers and organic amendments, and cropping/farming systems involving forages and agroforestry (Singh et al., 2011). Such management practices have great potential to reduce GHG emissions. For example, agricultural management can decrease soil emissions of CO_2 and maximize the storage of atmospheric carbon in crop biomass and eventually in soil organic matter (Johnson, et al, 2007). N_2O emissions may be substantially reduced by applying N fertilizer in the spring, when plant demand for nutrients is high, rather than in the fall (Millar et al. 2010). Because of agriculture's significant role in GHG emissions, implementing conservation practices – or BMPs – on agricultural systems may potentially prevent thousands of tons of GHGs from entering the atmosphere (Johnson, et al, 2007).

Agricultural management likely impacts CO_2 and N_2O emissions by altering one or more of the main drivers of these fluxes. For CO_2 emissions this includes soil temperature, soil moisture, and C substrate availability (Raich & Schlesinger 1992; Lloyd & Taylor 1994; Raich & Tufekciogul 2000; Rustad, et al., 2000; Hogberg, et al. 2001; Scott-Denton et al. 2006). If water is not limiting, CO_2 emissions generally increase with temperature (Wan et al. 2007; Lloyd & Taylor 1994). When water is limiting, water availability increases plant and microbial activity and thus CO_2 emissions directly by alleviating plant and microbial desiccation stress and indirectly by increasing substrate availability (via higher rates of plant growth, photosynthesis, belowground C allocation) and microbial access to substrate (e.g., increased C diffusion through soil water; Wan et al. 2007). Finally, respiration increases with C availability; when temperature and water are not limiting, microbial respiration increases with labile C availability (Hungate et al. 1997) and plant respiration is largely dependent on C from current photosynthetic activity (Hogberg et al. 2001).

Both nitrification and denitrification contribute to N_2O emissions. Nitrification transforms ammonium to nitrite (NO_2-) and then nitrate (NO_3-), which is frequently considered a limiting substrate for denitrification (Xue et al, 2013). In anaerobic conditions, denitrification, a process that involves four enzymatically catalyzed reductive steps (Wallenstein et al, 2006) converts soil NO_3- into NO_2^-, NO, N_2O, and finally N_2 (Inselbacher et al, 2011). Nitrification and denitrification are regulated by many soil factors, including soil texture, water content, soil temperature, aeration, available soluble organic carbon, soil pH, and the communities of soil microbes present (Granli and Bockman, 1994). Key factors impacting N_2O emissions specifically are temperature, soil water regime and availability of C and N substrates (Xue et al, 2013). O_2 partial pressure and soil pH are also important (Richardson et al, 2009). In general, N_2O emissions increase with temperature and soil moisture, but (Xue et al, 2013) the presence of anaerobic conditions is the key driving factor in allowing denitrifiers to use the supply of carbon as an energy source and nitrate as an electron acceptor (Xue et al, 2013).

The improvement of N fertilizer use-efficiency was listed as one of the primary modes in which to reduce GHG emissions from a known source (Liebig, et al., 2012). This improvement of efficiency involves making N available to plants in the amount needed at the correct time to meet plant demand (Liebig, et al., 2012). Major N losses often occur during the first week after applying N fertilizer and manure, with additional elevated N losses normally continuing over the following three weeks (Inselbacher et al, 2011). Improving N fertilizer and manure efficiency and application techniques may successfully result in less reactive N available for potential conversion to N_2O.

Proper timing of manure application is an important way to reduce potential losses. Vermont has a prohibition on manure application between December 15th and April 1st. Manure application methods also impact N losses and N_2O emissions. Manure injection is expected to reduce N losses via ammonia (NH_3) volatilization, but it may increase N_2O emissions (Maguire et al, 2011).

Timing, amount, application method, and form of N fertilization may interact with soil properties to affect N losses and N_2O emissions, and while N fertilization may influence all soil microbes, effects can vary between different soils and/or different communities of micro-organisms (Cavagnaro et al., 2008 and Wallenstein 2006). Because of this, fertilizer induced changes in soil processes and GHG emissions may also be dependent on soil properties and the characteristics and activities of the soil microbial community (Inselbacher et al., 2011).

The primary advantage of no-till over conventional tillage is that it does not disturb the soil habitat and leaves anywhere from 50 to 100% of the soil surface covered with non-grain crop residues (Brady and Weil, 2010). Reduced soil disturbance is thought to result in far lower GHG emissions in comparison to conventional tillage, but reduction in climate change forcing through carbon sequestration under no-tillage can potentially be offset by increases in soil N_2O emissions (Richardson et al, 2009). This is because higher soil carbon levels and smaller porosity in soils under no-tillage often induce higher denitrification rates and N_2O losses (Richardson et al, 2009). However, when the additional benefit of decreasing the use of tractor fuel compared to conventional tillage is considered, the benefits of reduced tillage outweigh conventional tillage again, with less net GHG's emitted (Powlson et al, 2012).

Changes in the profile distribution of soil C stocks for conventional versus no-tillage can also affect N_2O losses (Xue et al, 2013). Under conventional tillage, large N_2O losses may occur due to the combination of greater soil C content at deeper layers (ploughed soils) and moist profiles after N fertilizer application (humid regions; Xue et al, 2013). Additionally, deep N placement (via manure injection) appears to aggravate rather than ameliorate these concerns and inverted C profiles create larger N_2O emissions, presumably because of the greater C contents where soil conditions are wetter (Xue et al, 2013). Still, the consensus is that the benefits of a no-till system outweigh the previously mentioned concerns, thus the amount of acres being managed as no-till have expanded to nearly all regions of the United States including Vermont (H. Darby personal communication, 2014), and are now implemented in some form on almost half of all conservation tillage acres (Brady and Weil, 2010).

Quantification of the impacts of agricultural management on GHG emissions are an understudied topic with much potential to grow with technological advancements. Conservation tillage, a soil management practice regaining favor in the NE and throughout the US reduces loss of soil and water relative to conventional tillage (Brady and Weil, 2010) and maintains soil structure (e.g., soil aggregates; O'Brien and Jastrow 2013). Reduced soil disturbance is thought to result in far lower GHG emissions in comparison to conventional tillage, but reduction in climate change forcing through carbon sequestration under no-tillage can potentially be offset by increases in soil N_2O emissions (Richardson et al, 2009). This is because higher soil carbon levels and smaller porosity in soils under no-tillage often induce higher denitrification rates and N_2O losses (Richardson et al, 2009).

Recent technological advances allow farmers to "inject" manure into fields, regardless of tillage practice (Maguire et al. 2011), but there is little information available on how this practice affects GHG emissions. While manure injection is expected to reduce N losses via ammonia (NH_3) volatilization, it may increase N_2O

emissions relative to broadcast application by increasing available C and N in belowground anaerobic microsites (Maguire et al. 2011). Furthermore, it is not clear how this manure application method will interact with different tillage practices to impact GHG emissions.

2.3 Research Goals and Hypothesis

The objective of this research was to investigate the impact of current agricultural BMPs on GHG emissions in the NE US. To this end, I analyzed and soil CO_2 and N_2O emissions in various agroecosystems (dairy, crop, and meat producing farms) under differing management practices, including various tillage practices, different N fertilization amounts and treatments (synthetic N, cattle manure), the use of cover crops, aeration, and water levels. I expected that employing agricultural BMPs in temperate agricultural systems would assist in the sequestration of carbon in soils and mitigation of CO_2 and N_2O emissions.

Overall, we expected that: (1) relative to conventional tillage, conservation tillage practices would decrease CO_2 fluxes, but increase N_2O fluxes by increasing soil structure and the opportunity for low O_2 and high soil moisture conditions; and (2) while methods that incorporate manure below ground in the absence of conventional tillage (e.g., aeration or injection) may decrease C and N losses via runoff, NH_3 loss, and CO_2 flux, these methods may increase N_2O fluxes by creating high N and high moisture microsites that promote denitrification.

2.4 Methods

2.4.1 Site locations

Farms were selected based on several criteria: farms were meat, dairy, or vegetable producers; used one or more best management practices (i.e., cover crops, conservation tillage, wetlands conservation, storm water run-off management, or rotational grazing); grossed more than 10K/year; and was willing to host research on their land. The BMPs for this study were selected from an extensive literature review of already practiced and accepted forms of agricultural BMPs, along with an agricultural survey given to farmers, (Schattman, 2013). Of the selected farms, we designated three for intensive GHG emissions sampling: Shelburne Farm in Shelburne, VT, North Williston Cattle Co., in Williston, VT, and Borderview Farm in Alburgh, VT.

Table 2.1: Names, locations, soil management, and cropping system characteristics of farms being sampled.

Farms	Lat./Long.	Soil Management	Cropping System
Borderview Farm	45.01/-73.31	Various: No-Till, Strip-Till, Vertical-Till, Conventional Tillage	Perennial Corn with Rye Cover Crop
N. Williston Cattle Co.	44.47/-73.05	No-Till/Conventional	Perennial Corn System
Shelburne Farm	44.39/-73.26	Aerated/Non-Aerated	Hay/Grass Field for Dairy Cattle

Shelburne Farms

Shelburne Farms is a 1,400-acre working farm, forest, and National Historic Landmark, located on the shores of Lake Champlain in Shelburne, Vermont (figure 2.1; shelburnefarms.org). The Shelburne (SHE) study sites are composed of field-scale paired watersheds on an operating dairy farm. The SHE study watersheds (Figure 2.1) are currently in permanent hay production and are comprised of Covington soil (90% of SHE1) and Vergennes soil (100% of SHE2), both clay textured soils (hydrologic soil group D). SHE1 is 6.75 acres, while SHE2 is 5.79 acres. Slope at both sites is 3%.

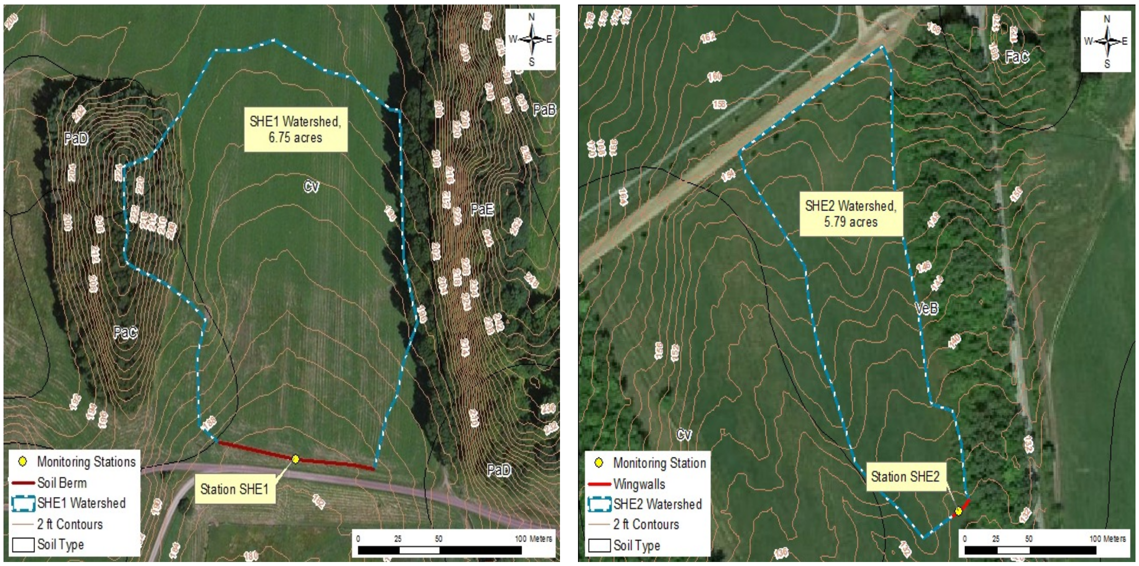

Figure 2.1: Field scale watersheds at Shelburne Farms. SHE1 (left) and SHE2 (right). Red solid lines are wingwalls built for field runoff collection and sampling. Yellow dots are edge of field monitoring stations.

During our study, both SHE1 and SHE2 were fertilized via liquid manure applications, but SHE1 was managed under an aeration tillage practice, while SHE2 was not aerated (i.e., no till). The aeration was done via an "Aerway" plow, which lifts the soil like a small spade, fracturing the compacted soil and introducing oxygen back into the soil profile (Aerway.com). Manure was added to both fields on July 28, 2015 at a rate of ~30,550 liters hectare^{-1}.

Within each watershed we measured mineral N (NH_4+ and NO_3-), soil moisture, temperature and GHG emissions (CO_2 and N_2O). Both hay fields also had weather monitoring and edge of field equipment installed, which provided data on precipitation temperature, runoff water, sediment and nutrients, and more. This pairwise study allowed us to examine how aeration tillage impacts GHG emissions and soil carbon storage in hay fields.

Table 2.2: Shelburne and N. Williston soil information.

Watershed	Area (acre)	Mean slope %	Aspect	Soil Type	Hydrologic Soil Group
SHE1	6.75	2.7	SW	Covington silty clay 89.4% Palatine silt loam 10.6%	D C
SHE2	5.79	3	S	Vergennes clay 100%	D
WIL1	4.27	0.12	S	Limerick silt loam 85.9% Hadley very fine sandy loam 7% Winooski very fine sandy loam 7%	C B B
WIL2	2.01	0.06	N	Limerick silt loam 34.6% Winooski very fine sandy loam 65.3%	C B

North Williston Cattle Co.

Located in Williston, Vermont (Figure 2.2), N. Williston Cattle Co. is a family owned dairy farm that raises all its own replacement animals and grows all its own forages (workinglands.vermont). Like the Shelburne site, the Williston (WIL) study site has field-scale paired watersheds on an operating dairy farm. The WIL paired watersheds are adjacent to one another in a field used for corn silage production with very low topographic relief (0.1%; Figure 2.2). The WIL1 and WIL2 watersheds are partially defined by a soil berm on their southwestern boundary, which was constructed to establish a consistent watershed boundary. Limerick silt loam comprises 86% of the WIL1 watershed, whereas the dominant soil in the WIL2 watershed is Winooski very fine sandy loam (65%), followed by Limerick silt loam (35%). Limerick silt loam is classified as hydrologic soil group C and Winooski very fine sandy loam is in hydrologic soil group B. WIL1 is 4.27 acres, while WIL2 is 2.01 acres. However, soil texture analysis indicated that soils in both fields are approximately 30% sand, 60% silt and 10% clay (silt loam; Stone Environmental 2013).

The WIL1 watershed is under a conservation tillage practice with manure being injected into the soil. WIL2 is being managed under a conventional tillage practice with manure being broadcast and left on the surface (See Table 2.3).

Table 2.3: N. Williston Cattle Co. manure and tillage dates

Field	Manure Method and Date Applied	Tillage Method and Date Performed
WIL1	Injected on 05/10/2015	No-Till/Rolled on 05/15/2015
WIL2	Broadcast on 05/10/2015	Disc Harrowed on 05/15/2015

Within each watershed we measured mineral N, soil moisture, temperature and GHG emissions (CO_2 and N_2O). As for SHE, both WIL watersheds have weather monitoring and edge of field equipment installed, which provide added data on precipitation, temperature, runoff water, sediment and nutrients, and more.

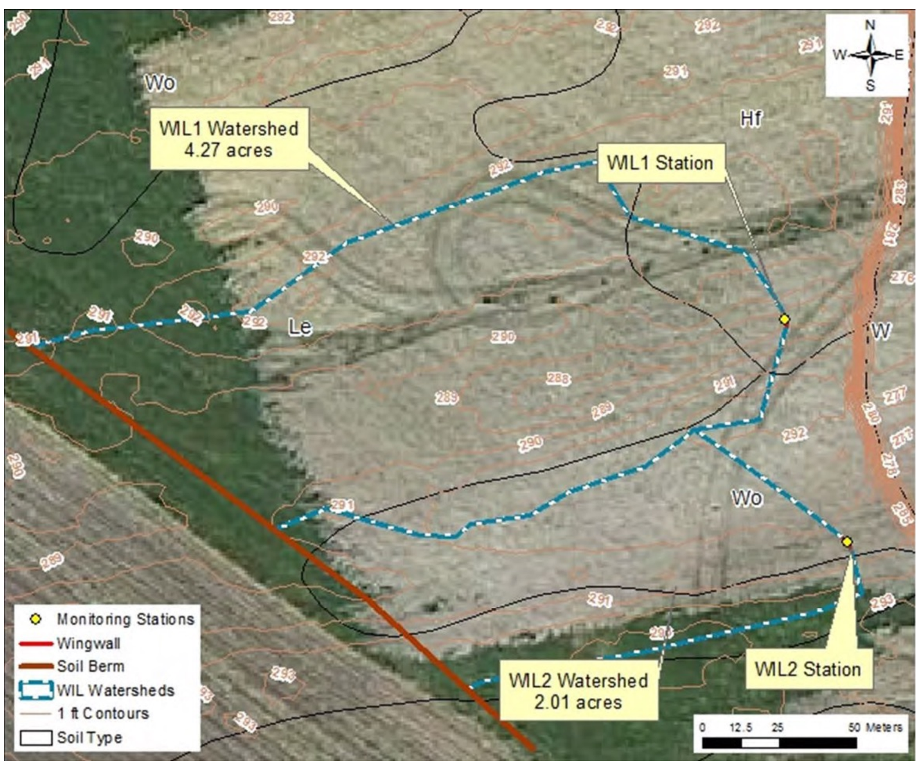

Figure 2.2: WIL1 and WIL2 watersheds at the WIL study site. Dark red solid line is a soil berm built for field runoff collection and sampling. Yellow dots are edge of field monitoring stations.

Manure Injection No Till (MINT) trial at Borderview Farm

The Manure Injection No Till (MINT) farm trial located at Borderview Farm in Alburgh, VT was established in May of 2014 (figure 2.3). This trial is in a continuous corn system. For the measurement year, the corn was planted on May 18th, 2015 with a 10-20-20 (N-P-K) fertilizer at 280 kg ha^{-1}. The soils at this site are classified as a Benson rocky silt loam (Darby, personal communication). There are three tillage treatment plots (vertical till, strip till, no till) that are 40 feet wide by 192 feet long, with 40 foot buffer strips between them. Strip tillage cultivates a 4-6" strip of soil along both sides of the planted row. Strip tillage allows the soil in close proximity to the seed to dry out and warm up faster than it would without tillage. It also deeply tills the soil (8-10 inches) where the crop is planted. No-till implements do not till the soil, but rather use metal coulters to cut the soil and plant seed into the slot created by the coulters (disk openers). An attachment on the back of the planter closes the slot and maximizes seed to soil contact to facilitate germination. Vertical tillage is a tillage system that lightly tills the top 2-3 inches of the soil to prepare a smooth seedbed. Within each tillage plot there are two manure application methods: broadcast and injected. Manure was applied at a rate of 15,500+ liters ha^{-1} on May 14th and 15th, 2015.

Each tillage and manure treatment combination is replicated four times (i.e., each manure treatment is replicated four times within each tillage plot; 28 plots). Within each treatment combination we measured mineral N, soil moisture, temperature and GHG emissions (CO_2 and N_2O). The corn crop was harvested on September 30, 2015, with a cover crop of rye planted two days after.

Figure 2.3: Borderview Farm map showing experimental design.

2.4.2 Site measurements

At each site, I collected data on GHG (CO_2 and N_2O) emissions. Soil emissions of CO_2 and N_2O were taken once every two weeks (on average) from June 2015 through November, 2015 using static flux chambers and a model 1412 Infrared Photoacoustic Spectroscopy (PAS) gas analyzer (Innova Air Tech Instruments, Ballerup, Denmark). Fluxes were measured on 16 dates at Shelburne Farms, 12 dates at the Williston site, and 13 dates in the MINT trial. Gas samples were taken at fixed intervals over a 10-14 minute time frame, with samples normally taken every one or two minutes. Borderview Farm had a total of 28 static flux chambers present, with 4 chambers per treatment (3 different tillage regimes*2 manure treatments, and one "conventional" treatment). Both Shelburne and North Williston had 8 chambers total, with 4 each per watershed/treatment. All chambers were installed using a stratified random sampling protocol, split between high and low elevations, and then two chambers were randomly placed in each area to ensure the absence of bias,

Flux chamber collars were PVC (polyvinyl chloride) pipe with a diameter of 0.3m (inner diameter) and height 0.15m. The collars were pushed into the soil to a depth of 0.11m so that the height remaining above the soil surface was 0.04m. During gas measurements, a vented PVC lid (0.095m inner height and 0.3m inner diameter) was placed on a chamber collar, sealed and connected in a closed-loop system with the PAS gas analyzer. The PAS measures concentrations nondestructively so any gas passed by the detector is returned to the chamber with unaltered gas concentrations. Gas concentrations ($\mu L\ L^{-1}$) are reported by the instrument at standard temperature (20°C) and pressure (101.325 kPa).

Fluxes of CO_2 and N_2O were computed by fitting a linear regression of gas concentration against time after chamber closure. Small chambers and long measurement times can lead to high chamber gas concentrations that alter soil–atmosphere diffusion gradients (Venterea, 2009), but our chamber size and sampling duration maintained low chamber gas concentrations and changes in concentration over time were linear. The time period used for flux rate calculations was the 2- to 10-min time segment (i.e., excluding the first measurement). Fluxes of CO_2, and N_2O were calculated as:

$$F = \frac{\Delta C}{\Delta t} * \frac{V}{A} * \rho * \alpha$$

where F is the gas production rate for CO_2 (mg CO_2–C m^{-2} h^{-1}), or N_2O (µg N_2O-N m^{-2} h^{-1}), $\Delta C/\Delta t$ is the change in gas concentration in the chamber (10^{-6} L L^{-1} h^{-1}), V is the chamber volume (0.00954 m^3), A is the chamber surface area (0.0707 m^2), ρ is the density of gas at 20°C and 0.101 MPa (1 mole per 24.04 m^3), and α is a conversion coefficient (28/44 for N_2O; 12/44 for CO_2). Here, the density of gas was calculated based on 20°C and not the actual air temperature because the PAS instrument calculates the concentration of each gas at 20°C.

During each gas measurement, I also collected both soil and air temperature data, soil moisture data, and, often, soil inorganic N. Soil temperatures were taken adjacent to the gas collecting chambers so as not to disturb the soil within the chamber. Air temperatures were taken once at the beginning of gas sampling and once more at the end of gas sampling. Water content of soil was measured using a soil moisture probe. Available inorganic N was measured by taking one soil sample (0-10cm) using a 2 cm diameter soil core. Soils were homogenized in the lab and extracted using 2 M KCl and extract was analyzed for ammonium and nitrate on a Lachat Flow Injected Analyzer.

2.4.3 Data analysis

Daily emissions data from the MINT trial was first analyzed using a repeated measures ANOVA with chambers nested within manure and/or tillage treatments as a random effect (JMP 11.2.0; SAS Institute, Cary, NC, USA). All treatments were considered fixed effects. We then used the same structure to perform a repeated measures ANCOVA with soil temperature, soil moisture, and soil nitrate as covariates (without interactions). Daily emissions data were analyzed similarly for Shelburne and Williston, but chamber was nested only within the one treatment present at each site. The "Day of Year" variable for Borderview Farm was entered as a nominal value, whereas for WIL and SHE day of year was considered continuous. This was due to the Williston and Shelburne sites having a smaller sample size.

2.5 RESULTS

2.5.1 Borderview Farm MINT Trial

Within the MINT trial, no-till decreased CO_2 emissions relative to vertical and strip tillage on many, but not all days (tillage by day interaction, P=0.005; Figure 2.4), while manure injection tended to increase CO_2 emissions compared to broadcast manure application (marginally significant manure effect, P = 0.059; Figure 2.5). The repeated measures ANOVA explained 77% of the variation in the CO_2 emissions data, but the ANCOVA, which included soil moisture, soil temperature, and soil nitrate concentration as covariates, explained 81% of the variation in the CO_2 emissions data. In the ANCOVA, day, soil temperature, and soil moisture were all significant (P<0.05; Table 1). However, manure and tillage treatments were no longer significant (although the manure treatment and day by manure by tillage treatments were marginally significant, P<0.1; Table 1). Soil CO_2 emissions increased with temperature and soil nitrate concentrations (Figures 2.6 and 2.7).

Manure injection increased N_2O emissions on many, but not all days (manure treatment by day interaction, P<0.0001; Figure 2.8). Tillage treatments had no impact on N_2O emissions (Table 2). The repeated measures ANOVA explained 51% of the variation in N_2O flux rates, while the ANCOVA explained 60% of the variation in flux rates (Table 2). In the ANOVA only day and the day by manure interaction effects were significant (P<0.0001). In the ANCOVA, day, soil nitrate concentration, and soil moisture were significant (P<0.05). Soil N_2O fluxes increased with increasing levels of soil nitrate and soil moisture (figures 2.9 and 2.10).

Table 2.4: Borderview ANOVA & ANCOVA; Response Kgs Lost CO_2-C hectare^{-1} day^{-1}. (* indicates significance, ** indicates marginal significance).

	CO_2-C ANOVA				CO_2-C ANCOVA			
Dependent Variable		**DF**	**F Ratio**	**P**		**DF**	**F Ratio**	**P**
R^2	0.7695				0.8148			
Day of Year		12	56.3013	<.0001*		9	9.7696	<.0001*
Tillage		2	0.0566	0.9451		2	0.0857	0.9182
Day of Year*Tillage		24	1.9905	.0054*		18	1.311	0.1884
Manure		1	4.0795	.0586**		1	3.148	0.0933**
Day of Year*Manure		12	0.8594	0.5893		9	1.1465	0.3338
Tillage*Manure		2	0.6536	0.5321		2	0.9034	0.4232
Day of Year*Tillage*Manure		24	1.4933	.0715**		18	1.6159	0.0626**
Soil Moisture (VWC)		-	-	-		1	0.0119	0.9131
Soil Temp. (Celsius)		-	-	-		1	12.7162	0.0005*
NO_3-N (mg N/Kg)		-	-	-		1	4.8683	0.0288*

Table 2.5: Borderview ANOVA & ANCOVA; Response Kgs Lost N_2O-N hectare^{-1} day^{-1}. (* indicates significance).

	N_2O-N ANOVA				N_2O -N ANCOVA			
Dependent Variable		**DF**	**F Ratio**	**P**		**DF**	**F Ratio**	**P**
R^2	0.5137				0.6006			
Day of Year		12	20.2795	<.0001*		9	5.9236	<.0001*
Tillage		2	0.9136	0.4191		2	1.7596	0.1983
Day of Year*Tillage		24	0.5643	0.951		18	0.9177	0.5583
Manure		1	2.8984	0.1061		1	0.8744	0.3634
Day of Year*Manure		12	3.699	<.0001*		9	0.7564	0.6568
Tillage*Manure		2	0.2333	0.7943		2	0.1137	0.8932
Day of Year*Tillage*Manure		24	0.4402	0.9901		18	0.4163	0.9828
Soil Moisture (VWC)		-	-	-		1	11.714	0.0009*
Soil Temp. (Celsius)		-	-	-		1	1.9181	0.1679
NO_3-N (mg N/Kg)		-	-	-		1	9.1611	0.0029*

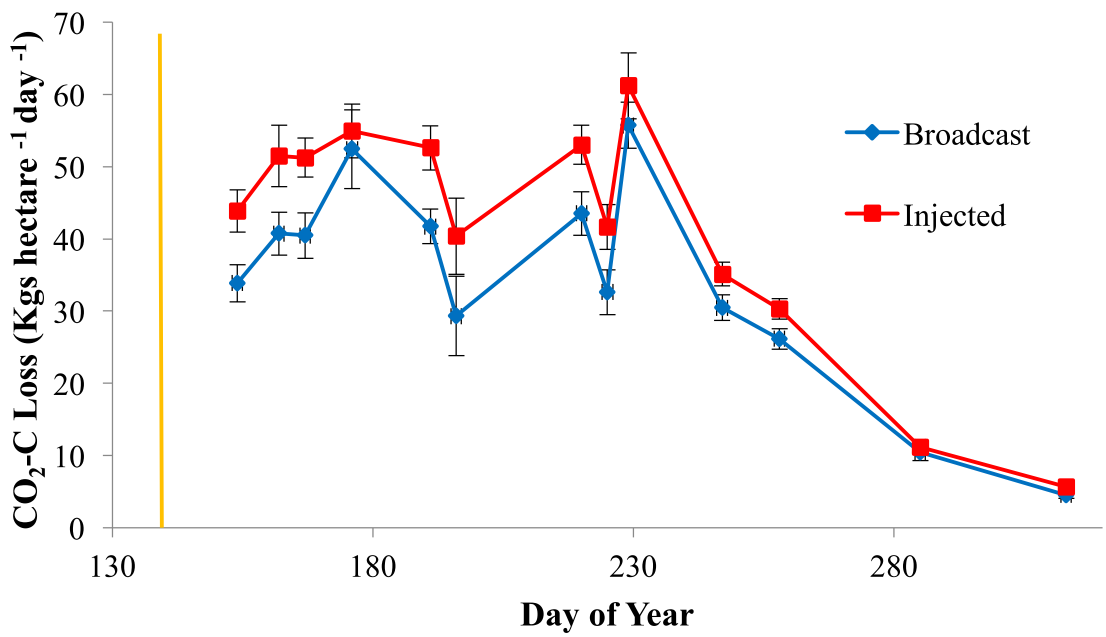

Figure 2.4: CO_2-C Lost Kgs hectare $^{-1}$ day $^{-1}$ vs. Day of Year, by Manure treatment (p-value= .059); Borderview Farm. Orange vertical line represents manure application date.

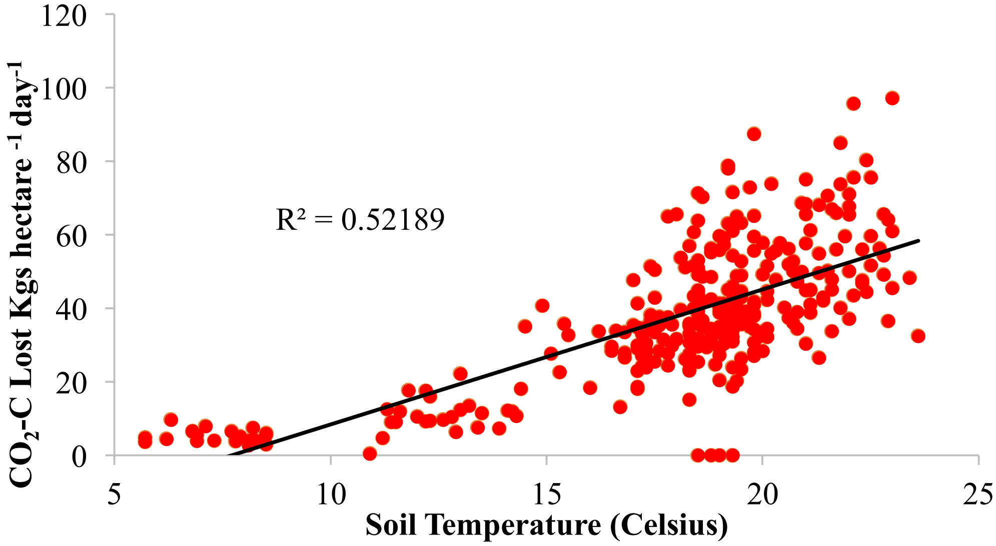

Figure 2.5: CO_2-C Lost Kgs hectare $^{-1}$ day^{-1} vs. Soil Temperature (p-value= .0005)

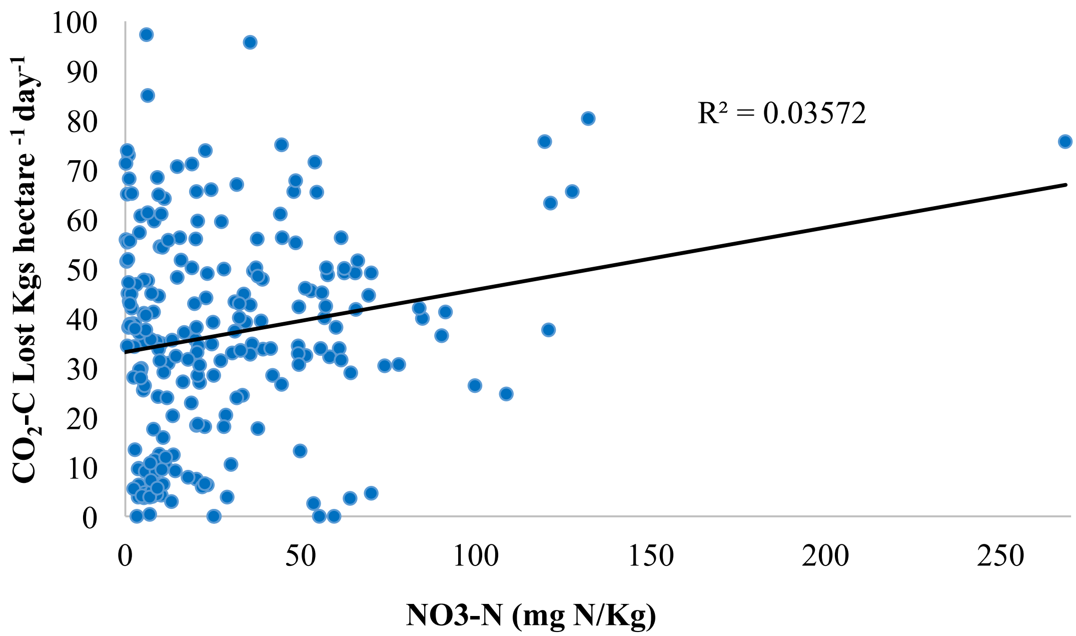

Figure 2.6: CO_2-C Lost Kgs hectare $^{-1}$ day^{-1} vs. NO_3-N (mg N/Kg) (p-value= .0288)

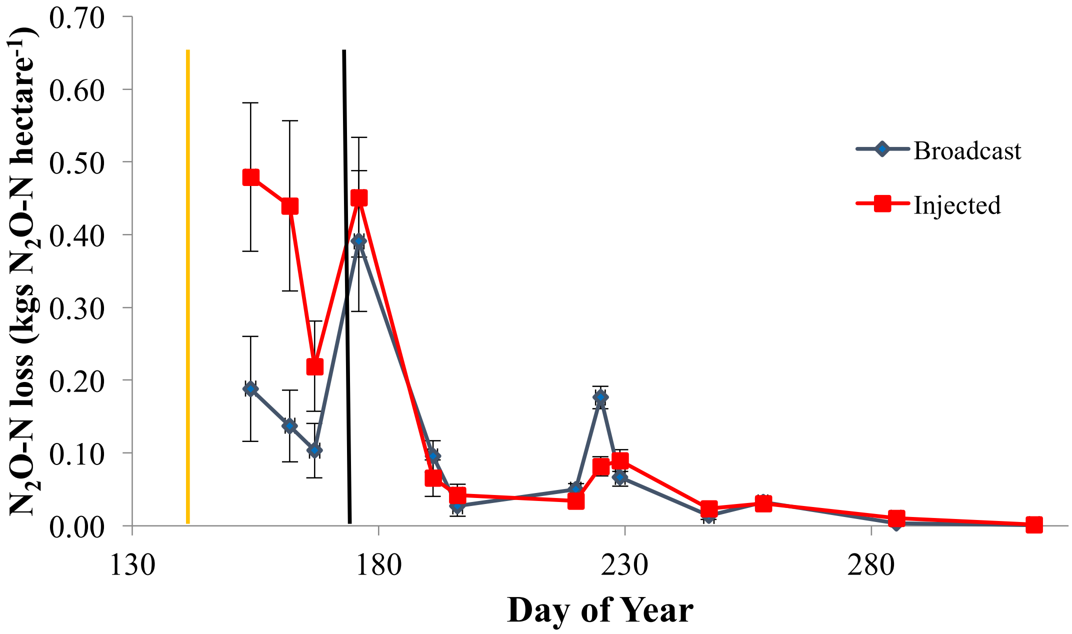

Figure 2.7: N_2O-N loss (Kgs N_2O-N hectare^{-1}) vs. Day of Year by manure treatment (p-value < .0001); Borderview Farm. Solid vertical black line represents a rain event (1cm). Orange vertical line represents manure application date.

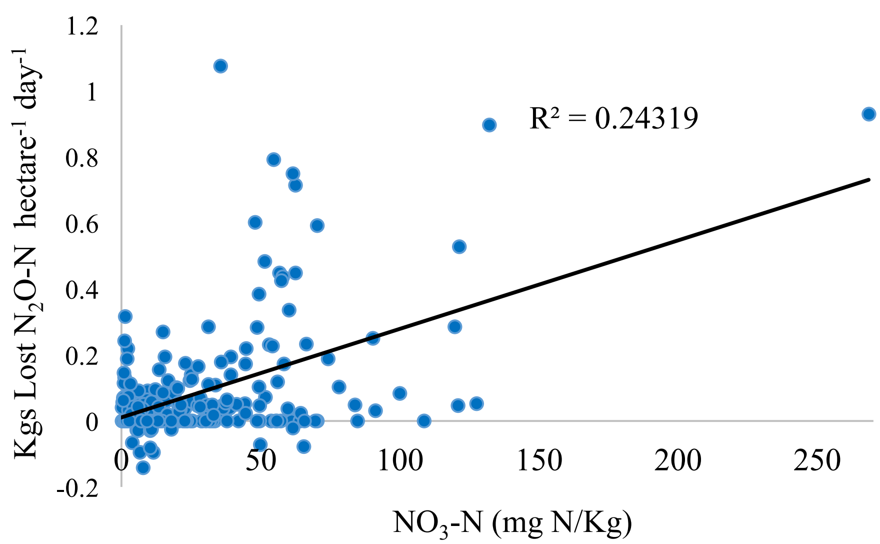

Figure 2.8: Kgs N_2O lost (hectare^{-1} day^{-1}) vs. NO_3-N present (p-value = .0029); Borderview Farm

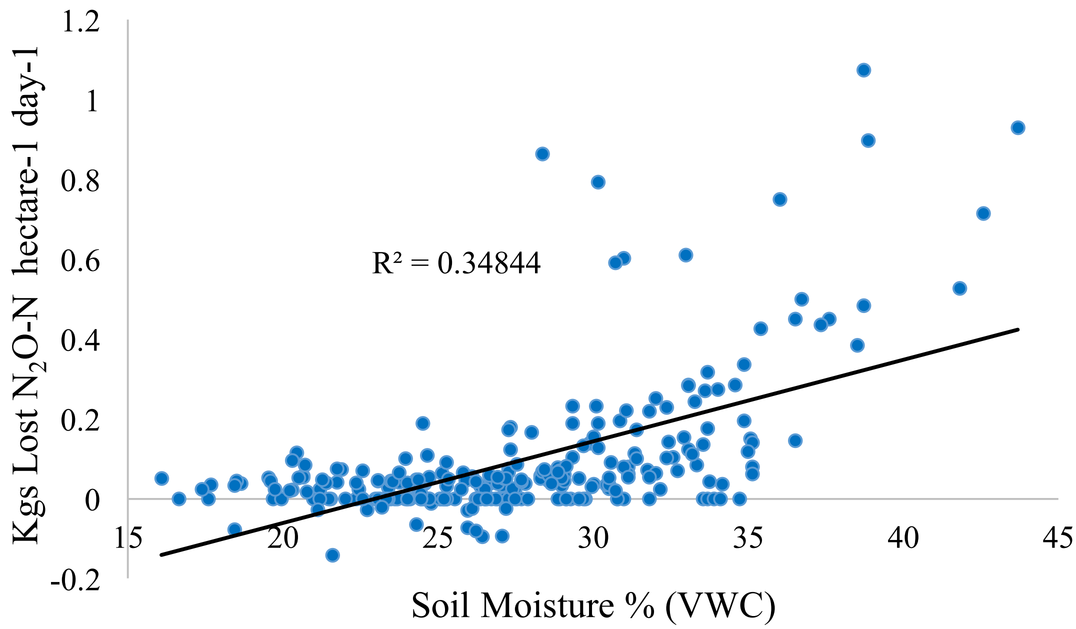

Figure 2.9: Kgs N_2O lost ($hectare^{-1}$ day^{-1}) vs. volumetric soil moisture (%); (p-value = .0009); Borderview Farm.

2.5.2 North Williston Cattle Co.

At the WIL sites, only day of year significantly impacted CO_2 fluxes. The two treatments, with the WIL1 watershed being managed under a conservation tillage practice with manure being injected into the soil, and WIL2 being managed under a conventional tillage practice with manure being broadcast and left on the surface, had no impact on CO_2 flux. Indeed, the ANOVA model was a poor fit to the data (Table 2.3). However, the ANCOVA explained 19% of the variation in CO_2 flux, with treatment and soil temperature significantly impacting CO_2 flux (Table 2.3, Figure 2.11).

Both the ANOVA and the ANCOVA fit the N_2O flux data poorly (Table 2.4; $R^2 < 0$). However, the ANOVA showed both Day of Year and treatment to be significant and the ANCOVA found treatment and only treatment to be significant (Table 2.4).

Table 2.6: N. Williston ANOVA and ANCOVA; Response Kgs Lost CO_2-C hectare^{-1} day^{-1} (*indicates significance).

	CO_2-C ANOVA				CO_2-C ANCOVA			
Dependent Variable		**DF**	**F Ratio**	**P**		**DF**	**F Ratio**	**P**
R^2	-0.0351				0.1947			
Day of Year		1	12.6537	0.0006*		1	0.0811	0.7771
Treatment		1	0.0329	0.8618		1	9.0268	0.0173*
Day of Year*Treatment		1	0.2965	0.5876		1	0.2455	0.6231
Soil Moisture (VWC)		-	-	-		1	1.7906	0.1881
Soil Temp. (Celsius)		-	-	-		1	4.1953	0.0467*
NO_3-N (mg N/Kg)		-	-	-		1	0.1298	0.7212

Table 2.7: N. Williston ANOVA and ANCOVA; Response Kgs Lost N_2O-N hectare^{-1} day^{-1}. (* indicates significance)

	N_2O-N ANOVA				N_2O-N ANCOVA			
Dependent Variable		**DF**	**F Ratio**	**P**		**DF**	**F Ratio**	**P**
R^2	-0.0202				-0.2297			
Day of Year		1	5.0428	0.0276*		1	1.5994	0.2128
Treatment		1	10.6231	0.0162*		1	7.0169	0.0265*
Day of Year*Treatment		1	0.2668	0.6070		1	0.1005	0.7529
Soil Moisture (VWC)		-	-	-		1	0.0056	0.9407
Soil Temp. (Celsius)		-	-	-		1	0.8495	0.3618
NO_3-N (mg N/Kg)		-	-	-		1	0.5285	0.4721

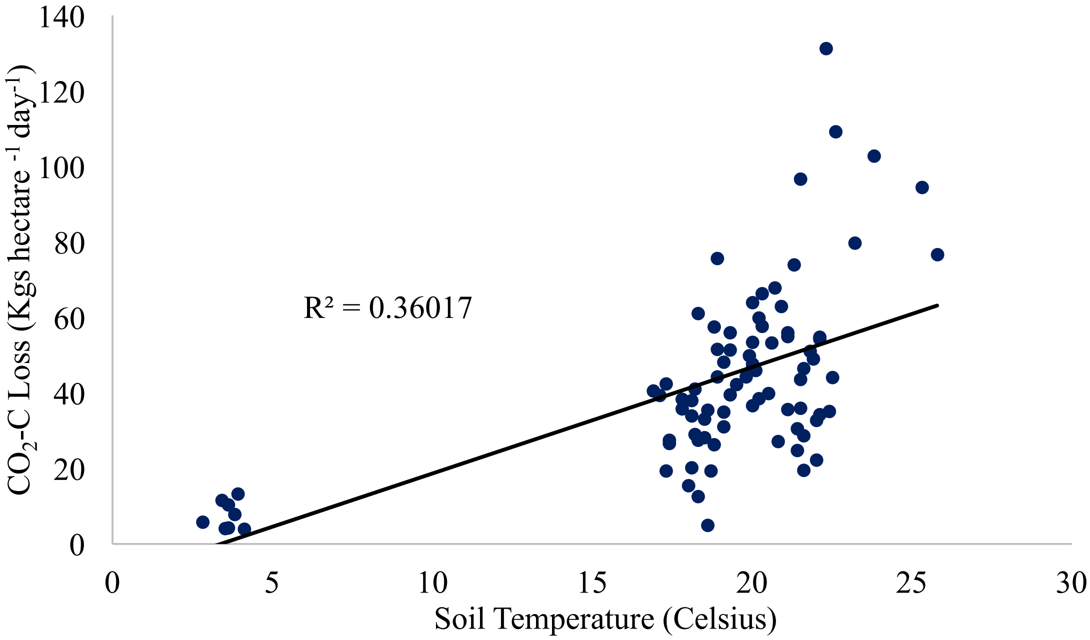

Figure 2.10: CO_2-C Loss (Kgs hectare $^{-1}$ day^{-1}) vs. Soil Temperature (p-value .0085); North Williston Cattle Co.

2.5.3 Shelburne

At Shelburne, the watershed treatments did not impact CO_2 emissions. The ANOVA explained 11% of the variation in CO_2 emissions, but only day of year was found to be significant (Table 2.5). The ANCOVA, however, explained 35% of the variation in CO_2 emissions, which found day of year, along with soil nitrate to be significant, showing that higher NO_3-N correlated with elevated CO_2 emissions (Table 2.5).

In contrast, aeration (treatment) had a marginally significant impact on N_2O emissions, with aeration decreasing N_2O emissions (Table 2.6; Figure 2.12). The ANOVA explained 5.7% of the emission variation, while the ANCOVA explained 9.2%

(Table 2.6). Both analyses found Day of Year to be significant, but only the ANOVA found treatment to be significant.

Table 2.8: Shelburne Farms ANOVA & ANCOVA;
Response Kgs Lost CO_2-C hectare^{-1} day^{-1} (* indicates significance).

	CO_2-C ANOVA				CO_2-C ANCOVA			
Dependent Variable		**DF**	**F Ratio**	**P**		**DF**	**F Ratio**	**P**
R^2	0.1117				0.3480			
Day of Year		1	26.2116	<.0001*		1	54.6582	<0.0001*
Treatment		1	0.0343	0.8588		1	1.4902	0.2549
Day of Year*Treatment		1	0.1208	0.7288		1	0.0541	0.8167
Soil Moisture (VWC)		-	-	-		1	2.8209	0.1076
Soil Temp. (Celsius)		-	-	-		1	0.308	0.5808
NO_3-N (mg N/Kg)		-	-	-		1	6.9132	0.0103*

Table 2.9: Shelburne Farms ANOVA and ANCOVA; Response Kgs Lost N_2O-N hectare^{-1} day^{-1} (* indicates significance, ** indicates marginal significance).

	N_2O-N ANOVA				N_2O-N ANCOVA			
Dependent Variable		**DF**	**F Ratio**	**P**		**DF**	**F Ratio**	**P**
R^2	0.0574				0.0920			
Day of Year		1	7.9578	<.0057*		1	7.766	0.0067*
Treatment		1	3.8264	0.0953**		1	1.087	0.3284
Day of Year*Treatment		1	0.708	0.4019		1	0.5983	0.4416
Soil Moisture (VWC)		-	-	-		1	0.8835	0.3519
Soil Temp. (Celsius)		-	-	-		1	0.0476	0.8278
NO_3-N (mg N/Kg)		-	-	-		1	0.0186	0.8918

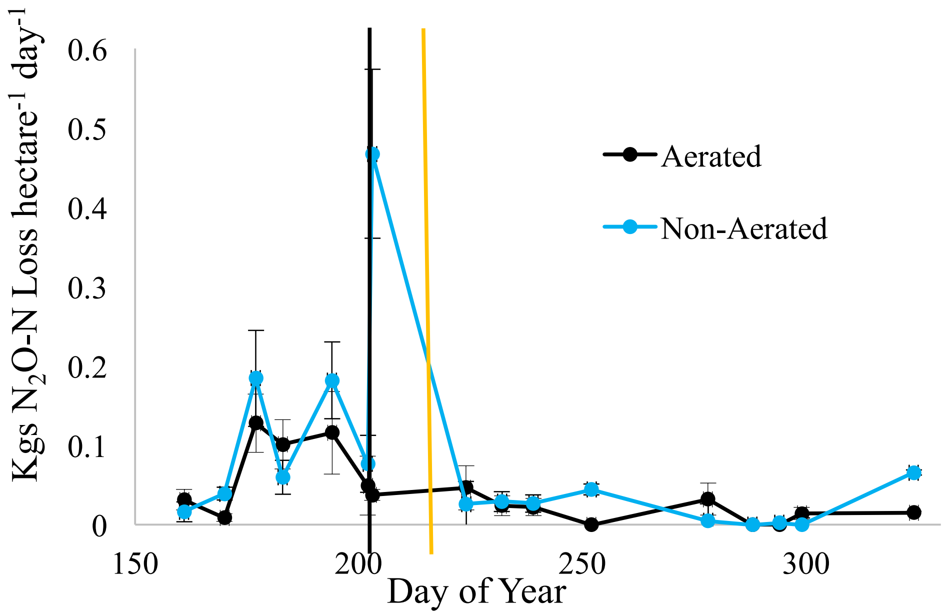

Figure 2.11: Kgs N_2O-N Loss hectare^{-1} day^{-1} vs. Day of Year; Shelburne Farm. Solid vertical black line represents a rain event (2cm). Orange vertical line represents manure application date.

2.6 DISCUSSION

As hypothesized, we found that manure injection tended to increase N_2O fluxes, but that aeration decreased N_2O fluxes associated with manure spreading. Additionally, consistent with my hypothesis and current literature, no-till tended to decrease CO_2 fluxes versus other forms of tillage.

In the MINT trial, manure injection increased both CO_2 and N_2O fluxes, while tillage only affected CO_2 fluxes, with no-till decreasing CO_2 fluxes relative to the other tillage treatments. Although manure injection and vertical and strip tillage significantly increased CO_2 fluxes, these treatments were no longer significant once soil temperature and soil nitrate concentrations were included in the (i.e., ANOVA vs ANCOVA results).

This suggests that the main impacts of tillage and manure treatments may have been to increase soil temperature (perhaps via tillage and loss of cover crops) and/or increase soil nitrate concentrations (via manure application and increased rates of nitrification).

Tillage practices can also decrease the physical protection of carbon against decomposition by breaking up macroaggregates (Post and Kwon, 2000). Additionally, tillage mechanically mixes aboveground inputs and a majority of roots into the surface layer. Together, these factors affect decomposition, and hence, GHG emissions, by exposing carbon to soil organisms and altering the degree of contact of SOM with mineral soil and microbes (Post and Kwon, 2000). Thus, tillage increases C availability to microbes and thus CO_2 emissions. Tillage can also change soil moisture and temperature conditions (Post and Kwon, 2000). Indeed, soil temperatures throughout the MINT study were higher in the strip-till and vertical-till plots versus the no till plots ($P < 0.0001$; ANOVA as described in the methods; data not shown) and soil temperature was positively correlated with CO_2 fluxes ($P < 0.05$; linear regression).

While no-till may increase emissions of N_2O due to compaction and the lack of both soil disturbance and residue incorporation (Ball et al., 1999), tillage had no impact on N_2O emissions in the MINT trial. N_2O emissions are normally associated with N (as fertilizer or manure) application under wet conditions and CO_2 emissions with aerobic respiration, which is often stimulated by tillage (Ball et al., 1999). The production, consumption and transport of N_2O and CO_2 are strongly influenced by the changes in soil structural quality and in water content associated with tillage and compaction (Ball et. al, 1999).

N_2O fluxes were only increased by manure injection (at WIL and in the MINT trial), which I hypothesized would increase N_2O fluxes by introducing carbon and nitrogen into the soil where anaerobic microsites promote denitrification and losses of N via N_2O (Xue et al, 2013). Accordingly, I found that manure treatments in the MINT trial were no longer significant after I added soil moisture and soil nitrate concentrations to the analysis, suggesting that manure injection increased N_2O fluxes by increasing NO_3^- availability and soil moisture (perhaps by adding liquid manure below the surface). Application of manure to cropland increases soil OM, microbial biomass, and mineralization rate and improves a number of soil properties including soil tilth, water-holding capacity, oxygen content, and fertility; it also reduces soil erosion, restores eroded croplands, reduces nutrient leaching, and can increase crop yields (Montes et al, 2013). Most of the N_2O resulting from manure is produced in manure-amended soils through microbial nitrification under aerobic conditions and partial denitrification under anaerobic conditions, with denitrification generally producing the larger quantity of N_2O (Montes et. al, 2013). Thus, while applying manure increases C and N availability, injecting manure likely increases it near anaerobic microsites, where denitrification is more likely than nitrification.

At the Williston site, the manure injected and conservation tillage treatment (WIL1) increased CO_2 fluxes (but only after soil temperature was accounted for) and N_2O fluxes. At WIL, adding soil temperature, moisture and nitrates into the analysis did not change the significance of manure injection for N_2O fluxes.

At the Shelburne site, aeration had no impact on CO_2 fluxes, but tended to decrease N_2O fluxes relative to broadcasting. This suggests that aeration may be a beneficial management strategy for incorporating manure without increasing CO_2 emissions. Unlike injection, aeration may reduce the abundance of anaerobic microsites, decreasing N_2O emissions via denitrification. However, there has been very little research on the impacts of aeration on soil properties, and more research is needed to define the impacts of this management technique on agricultural soils.

Overall, our results suggest that manure injection increases both CO_2 and N_2O emissions relative to broadcasting manure. Aeration decreased N_2O emissions. No-till caused a significant decrease in CO_2 fluxes, but both strip-till and vertical-till caused an increase in CO_2 emissions. The increases in CO_2 emissions seen in the vertical and strip-till managements were only significant when soil temperature was not accounted for. When soil temperature was taken into account, the tillage practices were no longer significant, suggesting that the higher soil temperatures in those plots were due to higher soil temperatures instead.

CHAPTER 3: SOIL CARBON STORAGE AND AGRICULTURAL MANAGEMENT PRACTICES

3.1 Abstract

Borderview Farm, Shelburne Farm, and N. Willison Cattle Co. were all studied for soil C. Results showed tillage to be significant at 25cm and 80cm mean depth, with vertical till having the highest amounts at 25cm and strip-till having the most at 80cm. Soil C stocks are twice that of all standing crop biomass, making it a major player in the C cycle, with lots of potential to help mitigate future GHG emissions.

In order to manage agricultural soils for C storage and climate change mitigation, researchers need to quantify soil C at depths up to 100 cm. Many studies have routinely taken 30cm deep cores when measuring stored SOC, though significant C is stored far below that level. Future research should create an industry standard soil sampling depth, which should include agreeing on a deeper soil core sampling depth.

3.2 Introduction

Terrestrial carbon (C) stocks have been altered by increasing atmospheric CO_2 concentrations and N deposition, as well as by land use change (Matson et al., 2002). There are four main global sinks for these emissions: the atmosphere, the oceans, tropical, temperate and boreal vegetation, mainly forests, and soils (Tamm et al. 1982, Post et al. 1990). Because standing stocks of soil carbon are twice as large as the standing crop biomass of all terrestrial biomes combined, with potentially much longer residence times (Post et al., 1990; Anderson, 1992), much research has focused on increasing soil C storage.

Standard agricultural practices such as complete-inversion (conventional) tillage and the addition of agro-chemicals have degraded soils, contaminated the atmosphere, and led to a decrease in the soil's capacity to store soil organic matter (Adviento-Borbe, et al., 2007) Extensive cultivation has led to the loss of upwards of 40% of original soil surface layer C via mineralization to CO_2 (Coleman, Crossley, and Hendrix, 2004). Since the beginning of settled agriculture, soils have been a major source of atmospheric CO_2 and other GHGs (CH_4, N_2O) (Ruddiman 2003, 2005, Haile-Mariam et al. 2008), with soil C emissions to the atmosphere since the industrial revolution (~1750 AD) estimated at 78 ± 12 Pg (Lal, 1999, 2004). Thus, most soils being managed under agricultural practices contain lower SOC pools than their natural/undisturbed ecosystem counterparts. There are several explanations for this: (1) lower inputs of biomass and detritus material to soils, (2) higher decomposition rates due to changes in soil temperature and moisture regimes, (3) increased leaching losses of dissolved organic C (DOC), and (4) severely increased losses by accelerated wind and water erosion (Singh et al., 2011). Thus, most cropland soils have lost 25-75% of their original SOC pool (Singh et al., 2011).

While the conventional wisdom is that conservation tillage increases soil carbon storage (Baker et al., 2007), a recent review of carbon storage in agricultural soil management, especially as it pertains to tillage practices, suggests that conservation tillage may have less impact on soil carbon storage than previously believed. According to Baker et al. (2007), conservation tillage increases soil carbon in the top 0-20 cm when compared to conventional tillage, but deeper in the soil profile, increases are null or absent altogether. In several instances, conventional tilled soils exhibit a greater amount of stored carbon than its conservation tilled counterparts. This is due to the physical placement of soil organic matter deeper in the soil profile by the inversion that takes place when soils are conventionally tilled (Powlson et al., 2011). The issue of SOC then becomes one of depth distribution rather than a more/less dynamic. Still, small net accumulations of SOC under no-till managements have been noted if no-till was continued for at least 10-15 years (Angers and Eriksen-Hamel, 2008).

Fertilizer/manure additions to soil are an important nutrient source for the crops being grown. Because the inputs being added to the crop system are external, the crop system is the recipient of added carbon, nitrogen, and other elemental nutrients not previously present. This added material has but a few fates: it will leave the system via volatilization or as biochemically assisted gas emissions, will leave the system with water runoff or erosion, will be assimilated into biomass, or remain in the soil (Eghball et al., 2002).

Our goal was to determine how various tillage and manure application management practices impacted soil C storage. We expected that no-till or conservation tillage agriculture would increase soil C storage relative to other tillage methods, particularly in the top 0-20 cm of soil. Broadcast fertilizer/manure additions were also expected to increase soil C, especially in the top 20 cm of the soil profile. Injected manure was originally expected to increase soil C, but due to the priming effect, where the added labile carbon is placed deep in the soil profile allowing it to stimulate soil microorganisms and causing higher respiration and mineralization rates, the expectations were amended to believe that soil C would, in fact, be lowered.

3.3 Methods

3.3.1 *Site Description*

The Manure Injection No Till (MINT) farm trial located at Borderview Farm in Alburgh, VT was established in May of 2014 (figure 2.3). This trial is in a continuous corn system. For the measurement year, the corn was planted on May 18th, 2015 with a 10-20-20 (N-P-K) fertilizer at 280 kg ha^{-1}. The soils at this site are classified as a Benson rocky silt loam (Darby, personal communication). There are three tillage treatment plots (vertical till, strip till, no till) that are 40 feet wide by 192 feet long, with 40 foot buffer strips between them. Strip tillage cultivates a 4-6" strip of soil along both sides of the planted row. Strip tillage allows the soil in close proximity to the seed to dry out and warm up faster than it would without tillage. It also deeply tills the soil (8-10 inches) where the crop is planted. No-till implements do not till the soil, but rather use metal coulters to cut the soil and plant seed into the slot created by the coulters (disk openers). An attachment on the back of the planter closes the slot and maximizes seed to soil contact to facilitate germination. Vertical tillage is a tillage system that lightly tills the top 2-3 inches of the soil to prepare a smooth seedbed. Within each tillage plot there are two manure application methods: broadcast and injected. Manure was applied at a rate of 15,500+ liters ha^{-1} on May 14th and 15th, 2015.

Each tillage and manure treatment combination is replicated four times (i.e., each manure treatment is replicated four times within each tillage plot; 28 plots). Within each treatment combination we measured soil C and N, mineral N, soil moisture, temperature and GHG emissions (CO_2 and N_2O). The corn crop was harvested on September 30, 2015, with a cover crop of rye planted two days after.

3.3.2 *Sampling and laboratory analyses*

I measured soil C and N in two of the four replicates for each tillage-manure treatment combination to a depth of 1 meter. Soil cores were 3.81cm in diameter. I took 14 cores in total (2 no-till/broadcast, 2 no-till/injected, 2 vertical-till/broadcast, 2 vertical-till/injected, 2 strip-till/broadcast, 2 strip-till/injected, 2 conventional tillage). Soil cores were sectioned into 0-10, 10-20, 20-30, 30-60, and 60-100 cm as detailed in the GRACEnet soil sampling protocol (Parkin and Venterea, 2010). Core sections were homogenized, dried in the oven at 60 degree C, sieved (4 mm), and subsampled to measure total C and N by combustion (Flash EA).

3.3.3 *Statistical analysis*

Total carbon and nitrogen (%) from the MINT trial was analyzed using an ANOVA for each core section (JMP 11.2.0; SAS Institute, Cary, NC, USA). All treatments were considered fixed effects.

3.4 Results

Only soil carbon showed a significant response to the tillage and/or manure treatments. At 20-30 cm depth, tillage practice was marginally significant, with vertical-till carbon almost twice that of strip-till, and more than 2.5 times that of no-till (Table 3.2, Figure 3.2). At 60-100 cm, manure treatment was significant with broadcast manure having almost 2.5 times more carbon than injected manure (Table 3.3, Figure 3.1). At this same depth, tillage was marginally significant, with carbon in strip-till greater than in no-till, and almost 3 times greater than in vertical-till (Table 3.3, Figure 3.2).

In contrast to my expectations, manure application and tillage method had no significant impacts on soil N.

Table 3.1: Whole model response to percent carbon by soil depth (25cm) by treatment. (** indicates marginal significance)

	Carbon Midpoint (20-30 cm) ANOVA			
Dependent Variable		**DF**	**F Ratio**	**P**
R^2	0.3247			
Manure		1	0.1761	0.6894
Till		2	4.6436	0.0605**
Manure*Till		2	0.4126	0.6794

Table 3.2: Whole model response to percent carbon by soil depth (80cm) by treatment. (* indicates significance, ** indicates marginal significance)

	Carbon (60-100 cm) ANOVA			
Dependent Variable		**DF**	**F Ratio**	**P**
R^2	0.696			
Manure		1	12.236	0.0173*
Till		2	4.874	0.0669**
Manure*Till		2	1.265	0.4019

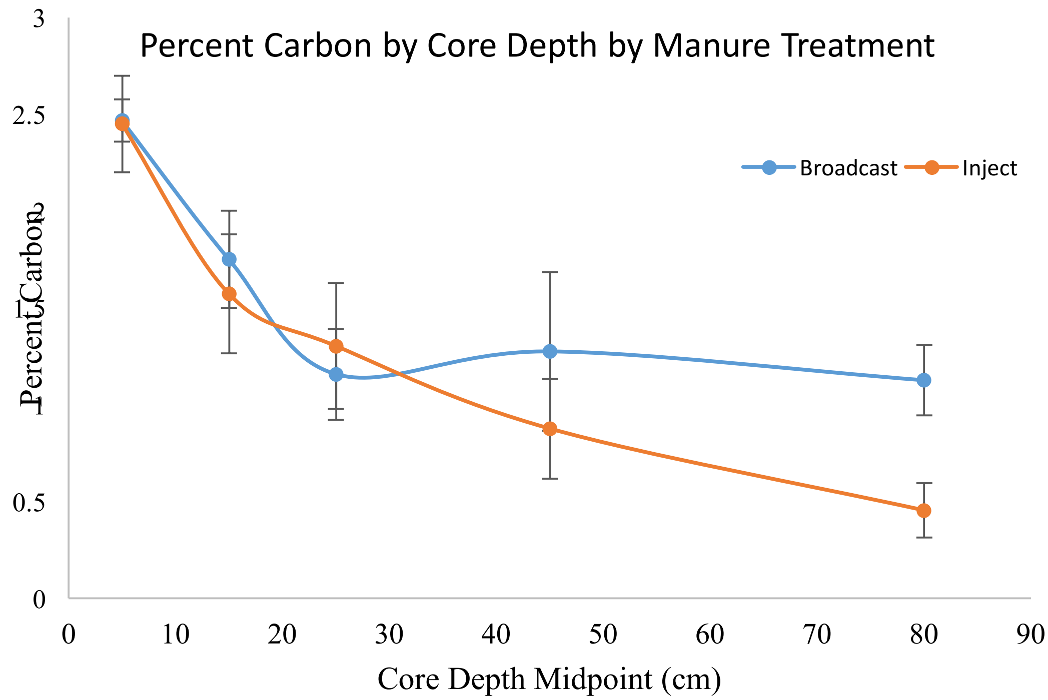

Figure 3.1: Borderview Farm; Percent C by soil core depth mean (cm) by manure treatment.

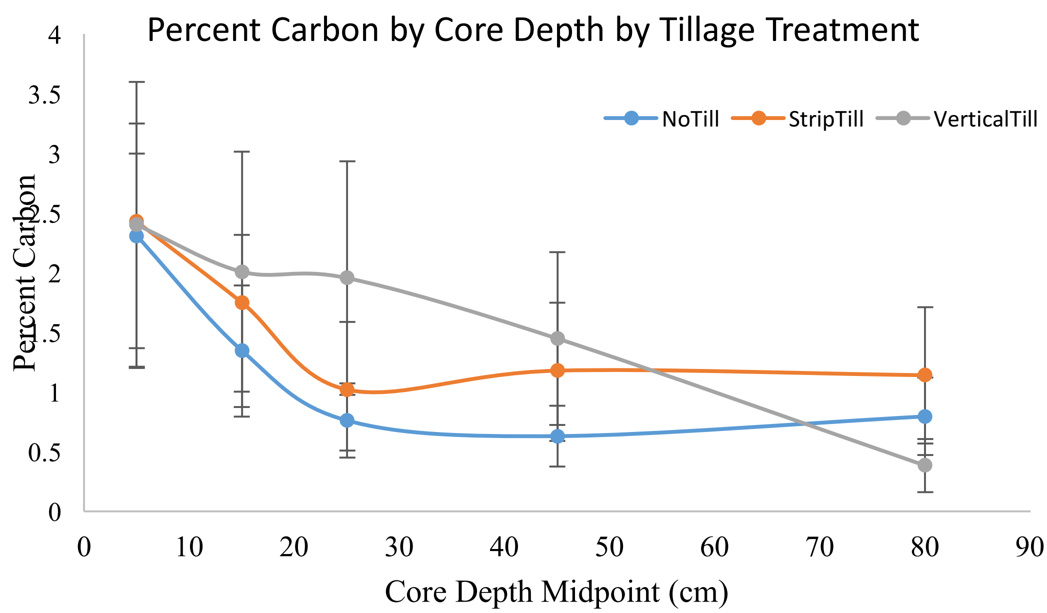

Figure 3.2: Borderview Farm; Percent C by soil core depth mean (cm) by tillage treatment.

3.5 Discussion

In contrast to my expectations, I did not see a significant difference in soil C in the top 20 cm of soil, but rather found that injecting manure decreased soil C at depth within the profile.

The loss of C at the 80cm depth in the injection treatment may be at least partially due to the priming effect. The priming effect is the idea that placing fresh organic matter (FOM) at depth, via manure injection or soil inversion from tillage, gives soil microorganisms at depth access to an energy source that was not previously present, causing them to increase their rates of mineralization, and thus, respiration rates (Fontaine et al., 2003). Soil C is the driving force of most microbially mediated processes, especially soil respiration and N mineralization (Fontaine et al., 2003). More recent literature proposes that in the absence of FOM, the stability of organic matter is maintained (Fontaine et al, 2007), and this seems to coincide with the data we collected. This suggests that an absence of fresh carbon may prevent the decomposition of the organic carbon pool in deep soil layers even if future temperatures rise (Fontaine et al, 2007). The opposite is also true: any change in management of agricultural soils that distributes FOM at depth could stimulate the ancient carbon, causing it to be lost (Fontaine et al, 2007), highlighting the importance of implementing BMP's on agricultural land and scientist's role in helping farmers to determine what those BMP's are.

Also in contrast to our expectations, C storage in the soil profile was either unaffected or not increased by no till management. While no till has been found to increase soil C in surface soils (Powlson et al, 2011), other studies have found no till to have minimal impacts (Baker et al, 2007). However, the impacts of tillage practices in this study may become clearer as the study continues beyond two years.

Our results suggest that, in order to manage agricultural soils for C storage and climate change mitigation, researchers need to quantify soil C at depths up to 100 cm. Many studies have routinely taken only 30cm deep cores when measuring stored SOC though stored C is far below that level, and cumulatively makes up a significant portion of total C. Corn roots grow more than 2m down into the soil, leaving OM at depths that are rarely examined. New manure application methods also inject carbon and other nutrients deeper into the soil, with unknown impacts for C and nutrients at greater depths. These results are in line with the results of Powlson et al., (2011) who also found that agricultural practices (such as no till) impacted not only C storage in surface soils, but those at depth. Future research should create an industry standard soil sampling depth, which should include agreeing on a deeper soil core sampling depth.

CHAPTER 4: REFLECTIONS AND IMPLICATIONS FOR FUTURE RESEARCH

4.1 Pros and Cons of On-Farm Research

By nature, on-farm research can be complicated by numerous peripheral factors. Because real life working farms are just that, coordinating when and when not to be in their fields can be problematic without proper communication with the farmer. Any time major field work needs to be performed by the farmer, it is imperative that GHG sampling chambers be removed as to not have them run over with equipment, and then consequently, they need to be promptly reinstalled after the work is done. On one occasion, we did have our chambers run over by a tractor late in the season, and in another our chambers were left in the field during both the manure application and the aeration plow, causing the farmer to go around our chambers, which of course defeats the purpose since our goal was to capture the emissions from those varying practices. Chambers were moved using the same stratified random sampling protocol as previously done to catch those GHG fluxes moving forward from that time. Weather and other natural disturbances (rabbits pooping in chambers) can also affect GHG sampling data.

Some positive aspects of doing research on working farms is that the data collected reflects real, in-practice, management systems. Though less controlled than in a lab environment, the data gathered is genuine, which lends itself to a truer representation of the factors at play concerning in-field dynamics. While it can be nice to control for factors in a laboratory setting, perhaps the control itself can lead to skewed results.

Building relationships with farmers is also a positive feature of on-farm research. Bridging the gap between farmers and academia is an important first step in building the farmers' trust and confidence with an on-site researcher, hopefully leading to greater reception on the farmer's part of BMP recommendations made by the scientist.

4.2 Farm Management Implications

A lot of what we are addressing here in this research, aside from the GHG measurements themselves, is nitrogen use efficiency (NUE). NUE is the concept of placing N in the correct amounts and at the best time to ensure as little N loss as possible. Not only how much and when to apply N, but also which types of N to apply is of concern. Synthetic N is very energy intensive to produce, using high pressures and temperatures during production, in what is known as the Haber-Bosch process. While this revolution in N fixation was a boon for agriculture, causing yields to increase dramatically in what is known as *The Green Revolution*, and had the side benefit of allowing populations to grow exponentially, it has now led us to pushing the upper limits, perhaps, of Earth's carrying capacity for humans, and creating unforeseen complications when regarding natural resource allocation.

Conservation tillage practices, while great for reducing the number of passes farm equipment makes on the field, and thus reducing C emissions, have exhibited mixed results concerning yield production early on in the conversion phase from conventional to conservation. In the long term, conservation tillage practices improve soil health by increasing the soil's water holding capacity, increasing organic matter, and reducing erosion, but in the short term, yields can suffer for several years, before rebounding (and often surpassing) the previous yield amounts (Brown et. al, 1989). This is something the soil scientist needs to consider and be aware of in order to properly inform the farmer.

Economic concerns was outside the scope of this research, but one that obviously plays an important role. No recommendations for implementing BMPs matter whatsoever if the farm is unable to stay profitable as a result of a change in management practice. The farms we do our research on are these farmer's livelihoods. They take their farms' business very seriously as any business should.

Works Cited

Adviento-Borbe, M. A. A., et al. (2007). "Soil greenhouse gas fluxes and global warming potential in four high-yielding maize systems." Global Change Biology **13**(9): 1972-1988.

Anderson, J.M. (1992). Responses of soils to climate change. *Adv. Ecol. Res.* 22, 163-210.

Angers, D. A., & Eriksen-Hamel, N. S. (2008). Full-Inversion Tillage and Organic Carbon Distribution in Soil Profiles: A Meta-Analysis. *Soil Science Society of America Journal, 72*(5), 1370-1374.

Baggs, E., & Blum, H. (2004). CH_4 oxidation and emissions of CH_4 and N_2O from Lolium perenne swards under elevated atmospheric CO_2. *Soil Biology Biochemistry*, *36*, 713-723.

Baker, J. M., Ochsner, T. E., Venterea, R. T., & Griffis, T. J. (2007). Tillage and soil carbon sequestration—What do we really know? *Agriculture, Ecosystems & Environment, 118*(1-4), 1-5.

Bakken, L. R., L. Bergaust, B. Liu, and A. Frostegard. "Regulation of Denitrification at the Cellular Level: A Clue to the Understanding of N2O Emissions from Soils."*Philosophical Transactions of the Royal Society B: Biological Sciences* 367 (2012): 1226-234. Print.

Ball, Bruce C., Scott, Albert, Parker, John P. (1999) "Field N2O, CO2, and CH4 fluxes in relation to tillage compaction and soil quality in Scotland". Soil & Tillage Research, 53, 29-39

Borken, Werner, Kathleen Savage, Eric A. Davidson, and Susan E. Trumbore. "Effects of Experimental Drought on Soil Respiration and Radiocarbon Efflux from a Temperate Forest Soil." *Global Change Biology* 12 (2006): 177-93. Print.

Brady, N. C., & Weil, R. R. (2010). *Elements of the nature and properties of soils* (Third ed.). Upper Saddle River, N.J.: Prentice Hall.

Brown, H. J., R. M. Cruse, and T. S. Colvin. 1989. Tillage System Effects on Crop Growth and Production Costs for a Corn-Soybean Rotation. J. Prod. Agric. 2:273-279

Cavagnaro, T., Jackson, L., Hristova, K., & Scow, K. (2008). Short-term population dynamics of ammonia oxidizing bacteria in an agricultural soil. *Applied Soil Ecology*, *40*, 13-18.

Ciais, P., C. Sabine, G. Bala, L. Bopp, V. Brovkin, J. Canadell, A. Chhabra, R. DeFries, J. Galloway, M. Heimann, C. Jones, C. Le Quéré, R.B. Myneni, S. Piao and P. Thornton (2013). Carbon and Other Biogeochemical Cycles. In: *Climate Change 2013: The Physical Science Basix. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change* [Stocker, T.F, D. Qin, G.K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M Midgley (eds.)]. Cambridge, United Kingdom and New York, NY, USA.

Climate Change 2007: the physical science basis: working group I contribution to the Fourth Assessment Report of the IPCC. (2007). Cambridge: Cambridge University Press.

Coleman, D. C., Crossley, D., Jr., & Hendrix, P. F. (2004). *Fundamentals of Soil Ecology* (2nd ed.). Burlington, MA: Elsevier.

Dawson, Julian J.C., Smith, Pete (2007) “Carbon losses from soil and its consequences for land-use management”. Science of The Total Environment, 382, 2-3, p.165-190.

DeLaune, P. B., J. W. Sij, and L. J. Krutz. "Impact of Soil Aeration on Runoff Characteristics in Dual-purpose No-till Wheat Systems." *Journal of Soil and Water Conservation* 68.4 (2013): 315-24. Print.

DeLaune, P.B., and J.W. Sij. "Impact of Tillage on Runoff in Long Term No-till Wheat Systems." *Soil and Tillage Research* 124 (2012): 32-35. Print.

"Ecosystem Restoration Program Agriculture." *Ecosystem Restoration Program Agriculture*. Web. 7 Mar. 2015. http://www.vtwaterquality.org/erp/htm/agriculture.htm.

Eghball, B., B.J. Weinhold, J.E. Gilley, and R.A. Eigenberg. "Mineralization of Manure Nutrients." *Soil and Water Conservation* 57.6 (2002): 470-73. Print.

Environmental Protection Agency. "DRAFT Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990 – 2013." (2015). Print.

Farm & Farm Products | Shelburne Farms. *Farm & Farm Products | Shelburne Farms*. Web. 27 Jan. 2015. <http://www.shelburnefarms.org/about/farm-products>.

FlashEA NC Soil Analyzer; Thermo Electron Corp., Milan, Italy; CE Elantech, Inc, Lakewood, NJ, USA

Fontaine, Sebastien, Mariotti, Andre, Abbadie, Luc (2003) "The priming effect of organic matter: a question of microbial competition?" Soil Biology & Biochemistry, 35, 837-843.

Fontaine, Sebastien, Barot, Sebastien, Barre, Pierre, Bdioui, Nadia, Mary, Bruno, Rumpel, Cornelia (2007) "Stability of organic carbon in deep soil layers controlled by fresh carbon supply" Nature, 450, 8, 277-281.

Giardina, Christian P., Litton, Creighton M., Crow, Susan E., Asner, Gregory P. (2014) "Warming-related increases in soil CO_2 efflux are explained by increased below-ground carbon flux." *Nature Climate Change*, 4, 822-827.

Granli, T., & Bockman, O. C. (1994). *Nitrous oxide from agriculture*. Ås, Norway: Agricultural University of Norway, Advisory Service.

Greenhouse Gas Working Group, 2010. Agriculture's role in greenhouse gas emissions & capture. Greenhouse Gas Working Group Rep. ASA, CSSA, and SSSA, Madison, WI. htttp://www.crops.org/files/science-policy/ghg report-august-2010.pdf (Sept. 24, 2013).

Hatfield, Jerry L., and Thomas J. Sauer. *Soil Management: Building a Stable Base for Agriculture*. Madison, WI: American Society of Agronomy; 2011. Print.

Haile-Mariam, S., Collins, H. P., & Higgins, S. S. (2008). Greenhouse Gas Fluxes from an Irrigated Sweet Corn (Zea mays L.)--Potato (Solanum tuberosum L.) Rotation. *Journal of Environmental Quality*, *37*, 759-771.

Harrigan, T. M., B. B. Bailey, W. J. Northcott, A. N. Kravchenko, and C. A. M. Laboski. "Field Performance of A Low-Disturbance, Rolling-Tine, Dribble-Bar Manure Applicator." *Applied Engineering in Agriculture* 22.1 (2006): 33-38. Print.

Hogberg P, Nordgren A, Buchmann N et al. (2001) Large-Scale Forest Girdling Shows That Current Photosynthesis drives Soil Respiration. Nature, 411, 789-792.

Inselsbacher, E., Wanek, W., Ripka, K., Hacki, E., Sessitsch, A., Strauss, J., et al. (2011). Greenhouse gas fluxes respond to different N fertilizer types due to altered plant-soil-microbe interactions. *Plant Soil*, *37*, 17-35.

"Intervale Community Farm." Intervale Community Farm. N.p., n.d. Web. 25 June 2014. http://intervalecommunityfarm.com.

IPCC Fourth Assessment Report Climate Change 2007. Geneva: Intergovernmental Panel on Climate Change, 2007. Print.

Johnson, J.M-F. Franzluebbers, A.J., Weyers, S.L., Reicosky, D.C. (2007). Agricultural opportunities to mitigate greenhouse gas emissions. *Environmental Pollution*150 (1):107- 124.

Kaye, Jason P., Ingrid C. Burke, Arvin R. Mosier, and Juan Pablo Guerschman. "Methane and Nitrous Oxide Fluxes from Urban Soils to the Atmosphere." *Ecological Applications* 14.4 (2004): 975-81. Print.

Lachat methods; Flow Injection Analysis, QuikChem 8000, Hach Company, Loveland, CO.

Lal, R. (1999). Soil management and restoration for carbon sequestration to mitigate the greenhouse effect. *Prog Env Sci,* 307-326.

Lloyd J, Taylor JA (1994) On the temperature-dependence of soil respiration. Functional Ecology, 8, 315-323.

Liebig, M., Franzluebbers, A. J., & Follet, R. F. (2012). *Managing Agricultural Greenhouse Gases Coordinated Agricultural Research through GRACEnet to Address our Changing Climate.* Burlington: Elsevier Science.

Lubbers, Ingrid M., Kees Jan Van Groenigen, Steven J. Fonte, Johan Six, Lijbert Brussaard, and Jan Willem Van Groenigen. "Greenhouse-gas Emissions from Soils Increased by Earthworms." *Nature Climate Change* 3 (2013): 187-94. Print.

Matson, P., Lohse, K. A., & Hall, S. J. (2002). The Globalization of Nitrogen Deposition: Consequences for Terrestrial Ecosystems. *AMBIO: A Journal of the Human Environment Ambio, 31*(2), 113-119.

Mer, Jean Le, and Pierre Roger. "Production, Oxidation, Emission and Consumption of Methane by Soils: A Review." *European Journal of Soil Biology* 37 (2001): 25-50. Print.

Millar, Neville, G. Philip Robertson, Peter R. Grace, Ron J. Gehl, and John P. Hoben. "Nitrogen Fertilizer Management for Nitrous Oxide (N2O) Mitigation in Intensive Corn (Maize) Production: An Emissions Reduction Protocol for US Midwest

Agriculture. *"Mitigation and Adaptation Strategies for Global Change* 15 (2010): 185-204. Print.

Montes, F., et al. (2013). "SPECIAL TOPICS-Mitigation of methane and nitrous oxide emissions from animal operations: II. A review of manure management mitigation options." Journal of Animal Science **91**(11): 5070-5094.

"North Williston Cattle Co." workinglands.vermont. N.p., n.d. Web. 25 June 2014. http://workinglands.vermont.gov/sites/ag_wlei/files/NorthWillistonCattleCompany.pdf

O'brien, Sarah L., and Julie D. Jastrow. (2013) "Physical and Chemical Protection in Hierarchical Soil Aggregates Regulates Soil Carbon and Nitrogen Recovery in Restored Perennial Grasslands." *Soil Biology and Biochemistry*: 1-13. Print.

Parkin, Timothy B., and Thomas C. Kaspar. "Temperature Controls on Diurnal Carbon Dioxide Flux." *Soil Science Society of America Journal* 67.6 (2003): 1763. Print.

Parkin, T.B. and Venterea, R.T. 2010. Sampling Protocols. Chapter 3. Chamber-Based Trace Gas Flux Measurements. IN Sampling Protocols. R.F. Follett, editor. p. 3-1 to 3-39.

Post, W.H., Peng, T.H., Emanuel, W.R., King, A. W., Dale, V.H., and DeAngelis, D.L. (1990). The global carbon cycle. *Am. Sci*. 78, 310-326

Post, W.M, Kwon, K.C. (2000). Soil Carbon Sequestration and Land-Use Change: Processes and Potential. *Global Change Biology*, 6, 317-328

Powlson, D.S., Whitmore, A.P., Goulding, K.W.T (2011) "Soil carbon sequestration to mitigate climate change: a critical re-examination to identify the true and the false." European Journal of Soil Science, 62, 42-55.

Powlson, D.S., A. Bhogal, B.J. Chambers, K. Coleman, A.J. Macdonald, K.W.T. Goulding, and A.P. Whitmore. "The Potential to Increase Soil Carbon Stocks through Reduced Tillage or Organic Material Additions in England and Wales: A Case Study." *Agriculture, Ecosystems & Environment* (2012): 23-33. Print.

Pribyl, Douglas W. "A Critical Review of the Conventional SOC to SOM Conversion Factor." *Geoderma* (2010): 75-83. Print.

R Core Team (2013). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL http://www.R-project.org/.

Raich JW, Schlesinger WH (1992) The global carbon-dioxide flux in soil respiration and its relationship to vegetation and climate. Tellus Series B-Chemical and Physical Meteorology, 44, 81-99.

Raich JW, Tufekcioglu A (2000) Vegetation and soil respiration: Correlations and controls. Biogeochemistry, 48, 71-90.

Richardson, David, Heather Felgate, Nick Watmough, Andrew Thomson, and Elizabeth Baggs. "Mitigating Release of the Potent Greenhouse Gas N2O from the Nitrogen Cycle – Could Enzymic Regulation Hold the Key? *"Trends in Biotechnology*: 388-97. Print.

Ruddiman, W. F. (2003). The Anthropogenic Greenhouse Era Began Thousands of Years Ago. *Climatic Change, 61*(3), 261-293.

Ruddiman, W. F. (2005). How Did Humans First Alter Global Climate? *Scientific American, 292*(3), 429-436.

Rustad LE, Huntington TG, Boone RD (2000) Controls on soil respiration: Implications for climate change. Biogeochemistry, 48, 1-6.

Savage, K., R. Phillips, and E. Davidson. "High Temporal Frequency Measurements of Greenhouse Gas Emissions from Soils. *Biogeosciences Discussions* 11 (2014): 2709-2720. Print.

Schattman, Rachel (2013) Climate Change Best Management Practice Tool.

Scott-Denton LE, Rosenstiel TN, Monson RK (2006) Differential controls by climate and substrate over the heterotrophic and rhizospheric components of soil respiration. Global Change Biology, 12, 205-216.

Shrestha, Raj K., Rattan Lal, and Basant Rimal. "Soil Carbon Fluxes and Balances and Soil Properties of Organically Amended No-till Corn Production Systems." *Geoderma* (2013): 177-85. Print.

Singh, B. P., Cowie, A. L., & Chan, K. Y. (2011). *Soil health and climate change*. Heidelberg: Springer.

Tamm, C.O., Lake, J.V., Miller, H.G (1982) “Nitrogen Cycling in Undisturved and Manipulated Boreal Forest”

Thomson, A. J., G. Giannopoulos, J. Pretty, E. M. Baggs, and D. J. Richardson. "Biological Sources and Sinks of Nitrous Oxide and Strategies to Mitigate Emissions. *“Philosophical Transactions of the Royal Society B: Biological Sciences* (2012): 1157-168. Print.

Thornley, J.H.M., and M.G.R. Cannell. "Soil Carbon Storage Response to Temperature: A Hypothesis." *Annals of Botany* 87 (2001): 591-98. Print.

Venterea, R. T. (2014). "Nitrite's Significant Role in Nitrous Oxide Emissions from Soil."

Wallenstein, Matthew D., David D. Myrold, Mary Firestone, and Mary Voytek. "Environmental Controls on Denitrifying Communities and Denitrification Rates: Insights from Molecular Methods." *Ecological Applications* 16.6 (2006): 2143-152. Print.

Wan S, Norby RJ, Ledford J et al. (2007) Responses of soil respiration to elevated CO2, air warming, and changing soil water availability in a model old-field grassland. Global Change Biology, 13, 2411-2424.

Wei, Hui, Bertrand Guenet, Sara Vicca, Naoise Nunan, Hamada Abdelgawad, Valérie Pouteau, Weijun Shen, and Ivan A. Janssens. "Thermal Acclimation of Organic Matter Decomposition in an Artificial Forest Soil Is Related to Shifts in Microbial Community Structure." *Soil Biology and Biochemistry* 71 (2014): 1-12. Print.

Xue, Y., et al. (2013). "Effects of N placement, carbon distribution and temperature on N2O emissions in clay loam and loamy sand soils." Soil Use and Management **29**(2): 240-249.

APPENDIX

Material List for chamber construction:

Item	Source
PVC pipe, 12” diameter, schedule 40	
Straight union fittings, ¼” PP	
Tractor tire tube, 15.5R38	EBay
Thin Plexi-glass for lid (1/4” or thinner)	Lowe’s Hardware Store
Rubber window seal	Lowe’s Hardware Store
Gorilla glue	Lowe’s Hardware Store
Reflective Mylar tape	Lowe’s Hardware Store