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Utilizing Soil Sensors to Assess Soil Health and Investigating Cover Crops Impact on Methane Emission

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UTILIZING SOIL SENSORS TO ASSESS SOIL HEALTH AND INVESTIGATING COVER CROPS IMPACT ON METHANE EMISSION

BY

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THESIS ACCEPTANCE PAGE

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This thesis is approved as a creditable and independent investigation by a candidate for the master's degree and is acceptable for meeting the thesis requirements for this degree. Acceptance of this does not imply that the conclusions reached by the candidate are necessarily the conclusions of the major department.

Advisor Date

Department Head Date

Director, Graduate School Date

This thesis is dedicated to my dear grandfather and grandmother, whose nurturing they gave to an infant born prematurely on stone laden soil during a journey back to the workplace. To a mother, who endured the pain and suffering, her unwavering love, and care, I shall never forget that has been a constant presence in my life.

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ABBREVIATIONS

%Na	sodium percentage
CH_4	methane
CO_2	carbon dioxide
CO_2e	carbon dioxide equivalent
EC	electrical conductivity
$EC_{1:1}$	electrical conductivity of soil solution prepared at 1:1 ratio
EC_a	apparent electrical conductivity
EC_e	electrical conductivity of saturated paste extract
EM	electromagnetic
ESP	exchangeable sodium percent
GDD	growing degree day
GHG	greenhouse gases
GPS	global position satellite
MMO	methane monooxygenase complex enzyme
N_2O	nitrous oxide
IPCC	intergovernmental panel on climate change
SAR	sodium adsorption ratio
SD	South Dakota
SOC	soil organic carbon
V2	corn at V2 stage
V4	corn at V4 stage
VE	corn at emergence stage
WFPS	water filled pore space

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ABSTRACT

UTILIZING SOIL SENSORS TO ASSESS SOIL HEALTH AND INVESTIGATING COVER CROPS IMPACT ON METHANE EMISSION

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Soil health is influenced by climate change, cultural practices, topography, crop rotations, cover crops, soil parent materials, and soil biota. Interactions between these factors can improve or reduce soil health. This thesis investigates two topics, using soil sensors to provide a rapid assessment of soil health, and the impact of cover crops on soil biological activity and greenhouse gas emissions. Chapter 1 explores the use of two types of sensors to measure apparent electrical conductivity (EC_a) and provides examples on the use of these sensors. Even though both sensors measure EC_a using different processes, they provide useful information about temporal and spatial changes in soil health. Chapter 2 explores the use of cover crops to improve environmental health and reduce CH_4 emissions. Prior research shows that cover crops impact both carbon dioxide and nitrous oxide emissions. However, missing from this analysis was their effect on methane emissions. Therefore, chapter 2 assessed the impact of dormant seeded rye on methane emission and the carbon dioxide equivalence prior to termination. These results showed that rye as a cover crop reduced soil methane emission in the range of -1.21 to -9.25 g CH_4-C $(ha \times d)^{-1}$.

STATEMENT OF PROBLEM

Global warming

The burning of fossil fuel accounts for over 75% of global greenhouse emissions, and agriculture and related land use change has been said to generate roughly one-quarter of global GHG emissions. Global warming caused catastrophic weather events are expected to affect the developing countries more than resource affluent countries. For example, because people in a developing country like Nepal depends on glacial meltwater and precipitation for drinking, irrigation, hydroelectric power, food production, clothing, and trade, changes in water available would affect their food, economic, and national security. Hence, there is urgent need to start curtailing greenhouse gas emission to mitigate the impact of climate change.

In addition to global warming and climate change, increasing human population has been putting pressure on the carrying capacity of earth. The human population in the world has reached 8 billion in 2022, adding 2 billion since 1998 and 1 billion since 2010 (UN, 2022). It has been projected to reach 8.5 billion in 2030; 9.7 billion in 2050; and 10.4 billion by 2100. Hence, the demand for food has been expected to rise by 70 to 100 percent by 2050. With limited land resources for production, food insecurity may prevail in various parts of the world. To increase the cultivable area, people may start converting forests and grasslands into agricultural land as well as increase soil manipulation and inputs into soil deteriorating soil health which might ultimately increase agriculture greenhouse gas emissions.

According to IPCC (2023), 195 countries have agreed upon curtailing fossil fuel use and adopting alternative energy sources to achieve the goal toward a 40-70% reduction in greenhouse gas emissions by 2050. To achieve this benchmark, agriculture sector will need to contribute by reducing GHG emissions while accelerating gains in agriculture productivity through better soil health, reduce GHG emissions and feed a growing population while achieving climate goals. Meeting these goals will be difficult and require reducing emissions wherever possible. Two possible areas are the remediation of salt-affected soils and using cover crops to reduce N_2O emissions.

Identifying a point source of high GHG emissions

Over 3.4 and 1.2 million hectares of land in SD are impacted by salinity (high soluble salt in soil) and sodicity (high sodium concentration in soil), respectively. Soil salinity is an accumulation of dissolved minerals and salts in the soil that often includes Na^+, Cl^-, Ca^{2+}, SO_4^{2-}, HCO_3^- K^+, Mg^{2+}, and NO_3^-. The sources of these salts are dissolution of salt-bearing minerals such as marine shales and halite or human management that can accelerate soil genesis or lead to the accumulation of salts in the surface soil. Salinity risks are generally assessed by measuring soil electrical conductivity (EC). However, because the critical EC levels are soil and crop specific, salinity management is closely tied to water, soil, and crop management. Research suggests that these soils can have very high nitrous oxide emissions (Fiedler et al., 2021). Given the potential impact of these soils on air and water quality, it is important to remediate these soils as soon as possible. However, prior to remediation the extent of the problem must be identified. Classically, soil salinity is measured by determining the electrical conductivity (EC) of soil solution. This would require a lot of soil sampling and expensive measurement of EC to ascertain the problem. To reduce these costs alternative techniques are needed.

Effectiveness of a remediation technique

It is generally believed that the adoption of climate smart practices should reduce greenhouse gas emissions. However, many “climate smart” practices have not actually been shown to reduce greenhouse gas emissions. One practice that is widely assumed to reduce GHG emissions is the use of cover crops. Theoretically, this approach should be effective because it reduces soil nutrients and water. However, because testing of this method has been mixed a more detailed analysis is needed (Basche et al., 2014; Reicks et al., 2021).

Rationale

In summary, both problems, soil salinity and greenhouse gas emissions, can be partially minimized by adopting the soil health practices of providing cover, a living root, minimizing disturbance, and increasing biodiversity. Associated with is the need to quantify the underlying problem and improvements with the adopted practices. Electromagnetic sensor (EM) help quantify the extent of the salinity problem through indirect measurement of electrical conductivity, which in turn may be utilized to develop management map onto which remediation strategies may be employed. Different time point measurements can then quantify or detect changes and improvements after remediation. In addition, greenhouse gas analyzers and associated sensors can help quantify the effectiveness of recommended climate smart practices, for example, cover crop, in reducing greenhouse gas emissions.

Hence, our objective is to document how soil proximal sensors could be used for the assessment of salinity problem and quantification of impact of cover crop as a management option for reducing greenhouse gas emissions. The greenhouse gas emissions should be curtailed whenever and wherever possible, hence our goal is to contribute reduce their emission into atmosphere either by aiding reclamation of salinity or quantifying impact of cover crop on emissions.

CHAPTER 1:
USING SOIL SENSORS TO ASSESS SOIL HEALTH
ABSTRACT

Important information to assess soil health is the soil electrical conductivity. Soil electrical conductivity is related to the concentration of dissolved ions in the soil solution. Soils with high concentrations of ions can have low microbial activity and productivity. The model system discussed in this chapter is the use of sensors to assess soil electrical conductivity in salt-affected soil. This chapter discusses the use of electromagnetic (EM) sensors that measure the apparent electrical conductivity (EC_a). In the field, electrical conductivity (EC) can be measured by two primary approaches: physical contact and EM induction. With physical contact, a current is injected into the soil, and the detector measures the resulting voltage. An EM meter does not make direct contact but uses a coil to produce an EM field. A sensor then measures the soil-induced changes in the field. Both types of sensors measure the EC_a, which is different from laboratory-derived EC values. When using an EC_a sensor, it is important to remember that they are sensitive to many factors, including salinity, soil moisture, bulk density, soil texture, and temperature. The purpose of this chapter is to provide an overview on the use of sensors to assess EC_a and to provide examples on the use of EC sensors in the field.

INTRODUCTION

Electrical conductivity (EC) is an intrinsic property of soil and is affected by many soil properties (Clay et al., 2001; Heilig et al., 2011; Logsdon, 2008; McNeill, 1992; Sudduth et al., 2005). One of the properties that affects soil EC is the concentration of Cl^{-}, SO_4^{2-}, HCO_3^{-}, CO_3^{2-}, Na^{+}, K^{+}, Mg^{2+}, Ca^{2+}, and several others in the soil solution. The laboratory measurement of the soil EC involves multiple steps, including collecting, drying, grinding, and mixing the soil with water. The EC of soil water solution is then measured with a meter that quantifies the ability of a solution to transmit an electrical current. The ability of the solution to transmit an electrical current increases with ion concentration. Because different measurement approaches provide different EC values, it is important to consider the measurement approach when interpreting the values.

In general, within a measurement approach, the higher the EC value, the greater the salt concentration in the soil. EC can be measured in the field or laboratory. Field measurements are generally reported at apparent electrical conductivity (EC_a) whereas laboratory measurements are reported as EC_e or $EC_{1:1}$. Field measurements of EC_a complement and do not replace the laboratory measurement of EC.

Laboratory measurements of EC

Electrical conductivity measurements conducted in the laboratory can be combined with the measurement of K^{+}, Na^{+}, Ca^{2+}, and Mg^{2+} to assess the risks of soil dispersion. In general, the higher the Na^{+} concentration, the higher the dispersal risk. However, this risk may be reduced if other soil cations are present in high quantities. The classical approach to assess the risk of soil dispersion from high Na^{+} concentrations is to measure the exchangeable sodium percentage (ESP) (100 x Na^{+} / CEC). However, because the measurement of ESP is expensive, alternative techniques have been developed. The most common approaches are to determine the sodium adsorption ratio (SAR) or %Na. In the laboratory, the measurement of EC is attributed only to differences in the concentrations of the cations and anions in the soil solution, whereas in the field EC_a measurements can be influenced by many factors, including total dissolved salts, soil texture, bulk density, and temperature (Altdorff et al., 2020; He et al., 2018; Doolittle & Brevik, 2014). However, the sensitivity of EC_a readings to these different soil properties also depends on the working mechanisms of sensors. For example, multi-coil based EMI sensors are less sensitive to these soil properties than multi-frequency based EMI sensors. Because in-field EC_a measurements are relatively inexpensive it is well suited for conducting an initial assessment of salinity spatial variability. However, because many factors can influence EC_a and care must be used when interpreting this information.

Field measurements of apparent soil electrical conductivity (EC_a)

There are a variety of sensors designed to measure EC in the field. These sensors may use different technologies. For example, the Veris MSP3 (Veris Technology) system generates a small electrical current that is transferred into the soil through a pair of electrode coulter disks. A second pair of coulters then measures the drop in voltage, which is proportional to the soil's EC. A different method is utilized by an electromagnetic (EM) sensor. This sensor does not come into physical but uses an induction coil to produce an EM field. The sensor then measures the soil-induced changes to the original EM field. Two types of sensors are available, a multi-frequency (MF) or multi-coil (MC). EM sensors range from mobile to stationary, and they can be configured measured multiple ways (Mat Su & Adamchuk, 2023).

In the laboratory the electrical conductivity of a solution is measured, whereas in the field the ability of the soil, including pores, solid, and liquid, to conduct an electrical current is measured. These measurement approaches are generally correlated to each other; however, the relationship is dependent on temporal and spatial changes of many factors including soil temperature, moisture, bulk density, and texture.

THE SEVEN STEPS FOR USING AN EC_a SENSOR TO ASSESS SOIL SALINITY

Soil sensors that measure EC_a have been used to better understand the soil spatial and temporal variability. These sensors can be used to identify soil zones with high bulk densities, soil textural changes, and high salt concentrations. The primary benefit of these sensors is that they provide an inexpensive, quick to conduct, and easy to interpret. By combining sensor information with chemical analysis, locations and elevation information, useful EC_a maps can be created. However, because EC_a sensors are sensitive to many factors, EC_a readings cannot be directly converted to EC_e. To use EC_a to assess salinity issues, a seven-step process should be used.

Step 1: Determine the Likely Problem

Visiting a site and obtaining site histories are important first steps in the creation of a remediation plan that manages excess salts in the soil and improves plant and soil health. This information may include identifying problem, location, conducting a farmer interview to obtain prior histories, making targeted measurements of the soil properties, analyzing historic yield monitor and remote sensing data, and identifying the soil types and chemical characteristics. For example, the image shown in Figure 1-1 highlights areas with different EC_a values. However, prior to assuming that salinity or sodicity are the primary problems, it is important to look for evidence supporting this suspicion. For example, does the soil classification suggest that it is a salt-affected soil or does prior soil sampling indicate that the EC or %Na are relatively high.

In the information collection stage, it is important to consider that multiple problems can have similar symptoms. For example, compaction by itself or when combined with salinity and sodicity can produce similar symptoms. A rapid approach to assess compaction is to determine the soil texture, friability, and penetrometer resistance (Kumar et al., 2016). Soil texture is the relative amount of sand, silt, and clay, whereas friability is the tendency for a soil to crumble into smaller fragments. Penetrometer resistance is the resistance of soil to the insertion of probe. Friability and penetrometer resistance decrease with increasing soil water. Sodium can affect both values by dispersing the soil aggregates.

The measurement of the soil bulk density and water infiltration can provide additional information. Bulk density is the dry weight of soil per unit volume of soil and relative water infiltration can be measured by digging a small hole and measuring how fast the water disappears. Sodium-dispersed soils can have high bulk densities and low water infiltration rates. When inspecting the site, it is important to collect soil samples to determine the soil EC and the amount of exchangeable Na^+ on the exchange sites (ESP, SAR, or %Na).

Figure 1- 1: Inspecting a site for salinity and sodicity problems. Courtesy: Dr. Cheryl Reese, South Dakota State University.

Step 2: Select a Sensor

To develop corrective solutions, the source and extent of the problem must be identified. There are many approaches to assess the extent of a salinity and sodicity problem, including analyzing soil samples for EC_e and ESP collected from a grid design or using an EM sensor to create an EC_a map (Doolittle & Brevik, 2014).

If you choose to collect and analyze soil samples for EC, then a soil sampling protocol must be selected. There are many options to collect soil samples, ranging from composite soil samples from management zones to grid soil sampling (Clay & Carlson, 2016).

When selecting a sensor, consider costs, skill requirements to collect the samples, and desired outcomes. Whenever possible, the sampling protocol should match the problem. For example, if the problem is concentrated in the surface soil, then the sensor should concentrate on the soil surface, whereas if the problem is concentrated at a lower soil depth, then a sensor should be selected that measures deeper in the soil profile (Corwin & Yemoto, 2020). Clues about the location of the salinity and sodicity problem may be provided in the soil name, inspecting the site, or by collecting soil samples from targeted areas for analysis.

Step 3: Calibrate the Sensor

All data collection protocols start with maintenance and calibration. To assess temporal and spatial changes in soil salinity and sodicity, EC_a sensors should be calibrated prior to use. Calibration can be conducted by at least one of two approaches. The first approach is to measure EC_a followed by collecting soil samples from the study area, analyzing for EC_e, followed by comparing EC_a and EC_e values. The second approach is to follow the manufacturer calibration procedures. It is essential to standardize the device to a uniform output to accurately make comparisons from one sampling time to the next. Temperature calibration may also be required.

Step 4: Conduct the Survey

Conducting a survey starts with creating a sampling design. The sampling design should be based on identifying the extent of the problem. Two basic approaches can be used to conduct the survey. In the first approach, a stop and go measurement approach is used. In a stop and go approach, a grid is overlayed on the field. At each grid point, the location and elevation are measured with a global position satellite (GPS) system, and a sensor measures the EC_a. At selected sampling points, a soil sample is collected that will be analyzed for EC using an appropriate protocol.

In the second approach, an EC_a sensor and GPS system is driven across a field. This sensor measures EC_a, elevation, and location/elevation simultaneously. When using this approach, the surveyor selects the distance between the transects. Where possible, the transects should be perpendicular to elevation. The survey should avoid field edges because these areas often are compacted or have high EC resulting from runoff from roads.

Step 5: Analyze the Soil Samples

Soil samples should be analyzed as soon as possible using an appropriate protocol. In most situations, the samples should be analyzed for EC, pH, Cl^-, NO_3^-, SO_4^{2-}, Na^+, K^+, Ca^{2+}, and Mg^{2+} and for the relative amount of Na^+ on the soil's cation exchange sites. The ratio between Na and the CEC is called the exchangeable sodium percent (ESP). However, due to cost, the ESP value is often replaced by the sodium adsorption ration (SAR) or %Na values. Additional information about GPS is available in Shannon et al. (2018).

Step 6: Convert EC_a to EC_e or $EC_{1:1}$

After the survey has been conducted and the associated soil samples are analyzed, the relationship between EC_a and EC_e/EC of a 1:1 soil/water mixture ($EC_{1:1}$) can be determined. For this analysis, it might be necessary to separate the field into management zones. Management zones can be based on a soil survey or elevation map. Across the field or within a zone, determine the relationship and equation relating EC_a to EC_e or $EC_{1:1}$. Based on this relationship, convert the EC_a to EC_e or $EC_{1:1}$.

Step 7: Graphic Display

One of the goals for conducting the survey is to assist in soil sampling and implementing remediation treatments. Prior to displaying the data set, check it for errors that need to be removed. When displaying the map, let the software process the data and create a map. When using this approach, it is important to compare the map with the field and the producer's knowledge. In many situations, overlaying the EC_a map on an elevation map makes sense. Low areas in production fields often have high EC_a values. High values are often linked to areas with high EC_e values or contain compacted soil zones, whereas low values may be associated to coarse textured soil. If the maps are not useful, try adjusting the boundary EC_e values of the zones.

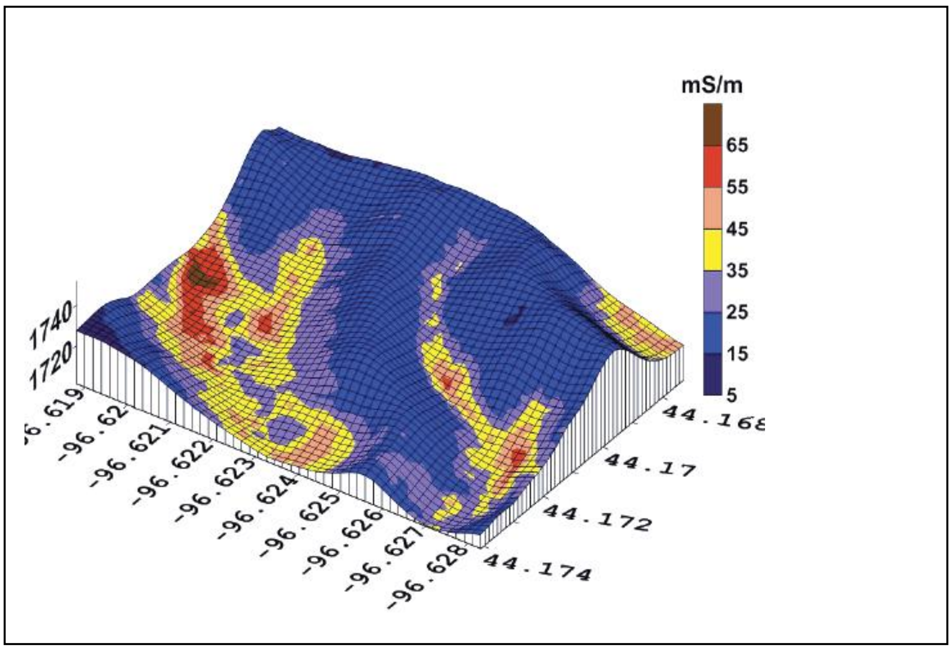

Figure 1- 2: Apparent electrical conductivity (EC_a) overlayed on an elevation map. This map was obtained by pulling an EM sensor behind a vehicle as it drove across a field. This map suggests that EC_a varies across the field and that footslope zone generally had higher values than backslope zones. However, because EM sensors are impacted by many factors, to confirm that the variation was due to salts, soil samples should be collected collected and analyzed for $EC_{1:1}$ of 1:1 soil/water mixture ($EC_{1:1}$). Courtesy of D. Clay, South Dakota State University

In summary, the EM surveys are not designed to replace but to compliment traditional plant and soil health measurements. To create a useful EM map, technical know-how is required about GPS, the EM sensor, and data processing. The EM values are influenced by many factors, such as texture, moisture content, salt concentration, temperature, and bulk density, so when converting EC_a to EC_e, it is important to understand relationships among these parameters. If more than one property is found to be highly correlated to EC_a, a multivariable model may be required. A sensor that measures EC_a is a relatively low-cost approach for creating directed soil maps and when the sensor is maintained, standardized, and calibrated, it can then be used to create management zones.

Case Studies on the Use of EC_a Information

He et al. (2018) collected 1088 soil samples from a 12.2 by 12.2 grid within a 8.1 ha field. Soil samples were analyzed for $EC_{1:1}$, $pH_{1:1}$, soil dispersion, Ca^{2+}, Mg^{2+}, Na^+, and K^+. At the sampling points, EC_a was measured with an EM 38 when the sensor was in the horizontal and vertical dipole modes. The soils at the North Dakota site were Natruaquoll and Calciaquolls. There was a strong correlation between apparent horizontal dipole EC (EC_{ah}) and %Na (r^2 = 0.71), EC_{ah} and EC_e (r^2 = 0.79), and EC_e and %Na (r^2 = 0.77). All geo-referenced data were entered into ArcGIS 10.0, and interpolation maps of %Na, $EC_{1\ 1}$, EC_{ah}, and apparent vertical dipole EC were prepared using the ordinary inverse-distance-weight interpolation. Finally, a management zone map was created in Management Zone Analyst 1.0.1 using the EC_{ah} and %Na datasets, which yielded six zones. After the zones were delineated, soil samples from each zone could be collected to determine the gypsum requirement.

Amezketa (2007) evaluated the ability of using EC_a information for mapping saline/sodic soils in Navarre, Spain. The fields were underlain by saliferous rock strata, and EC_a was measured from an orthogonal grid that varied depending on the field size. Soil water content was approximately field capacity, and EC_a values were corrected to a reference temperature of 25 °C. About 10–30 soil sampling sites that corresponded to a full range of EM 38 measurements were surveyed within each field (site selected with the help of ESAP-RSSD program). Multiple linear regression (preloaded prediction model) included in ESAP-calibrate was then used to estimate the calibration equation by pairing EC_a readings with laboratory-analyzed soil property data. The EC_a and SAR values were strongly correlated ($r > 0.91$). These results were attributed to autocorrelation ($r > 0.93$) between EC_e and SAR in all fields. The calibration models accounted for 87% of the observed variability in salinity and 84% in sodicity. Soil salinity and sodicity raster maps were prepared by IDS interpolation of EM-estimated profile average EC_e and SAR values.

Ganjegunte et al. (2013) used an EM sensor to measure salinity and sodicity in turfgrass soil watered with saline water. Tall fescue (*Festuca arundinacea*) and Kentucky bluegrass (*Poa pratensis*) were irrigated with water that had an EC values of 0.6 and 2.98 dS/m. At the end of the study, EC_a was measured using an EM 38 in the horizontal coil configuration. Location and EC_a was measured every 4.3 m in transects that were separated by 20 m. After completing the survey, 24 locations were selected for soil sampling. At each point, soil samples were collected in 15-cm increments to a depth of 75 cm. Soil samples were analyzed for EC_e, pH, concentration of Na^+, Ca^{2+}, and Mg^{2+} using plasma spectroscopy. Based on these measurements the SAR value was calculated. Based on a strong relationship between EC_a and EC_e, point kriging using a linear model was used to create an EC_e map based on the EC_a data. These findings showed that kriging can be used to create an EC_e map based on EM data.

CONCLUSIONS

Electrical conductivity can be measured in the field continuously using either physical contact or EM induction. With physical contact, an electrical current is injected into the soil, and the detector measures the resulting voltage, whereas an EM meter does not make direct contact but instead uses a coil to induce an EM field into the soil. Both sensors measure EC_a, which is different from laboratory-derived EC values. This chapter discussed the feasibility of using laboratory measurements to convert field-measured EC_a data into EC_e values. A seven-step process for this site-specific conversion was proposed. The primary benefits of the EC sensor approach are the low cost and the increased speed at which useful information can be collected. As a result, the EC sensor approach reduces the cost of conducting a salinity and sodicity survey.

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CHAPTER 2: RYE AS A COVER CROP REDUCES METHANE EMISSION UNTIL ITS TERMINATION.

ABSTRACT

In agriculture, the primary greenhouse gases are CO_2, N_2O, and CH_4. Of these gases, CH_4 is often not considered when determining the carbon dioxide equivalence (CO_2e). The purpose of this paper is to determine if ignoring methane in CO_2e calculations is a valid simplification in fields planted with cover crops. We quantified the effects of a fall-planted living cereal rye (*Secale cereale* L.) cover crop on the following spring's soil temperature, soil moisture, inorganic N, N_2O, and CH_4 emissions and compared these measurements to a no-cover crop treatment. In this replicated study, rye was dormant seeded in 2017, 2018, and 2019, but mostly emerged the following spring. The methane flux was near-continuously measured from early spring through June. Rye reduced soil moisture (cm^3/cm^3) in the surface 5 cm by 23, 2, and 35% in 2018, 2019, and 2020; soil NO_3-N by 21%, 52 and 64% in 2018, 2019 and 2020, respectively; and increased soil CH_4 consumption by 155, 106, and 145% in 2018, 2019, and 2020, respectively. While considering CH_4-C emission alongside N_2O-N emission, the change in carbon-dioxide equivalence figure ranged from 4.08-36.92% in cover crop treatment and 4.39-6.90% in no-cover crop treatment. This study showed that from cover crop germination to termination, CH_4 consumption reduced the CO_2e derived from N_2O by 24.4%, whereas in the no-cover crop treatment soil CH_4 consumption reduced CO_2e derived from N_2O by 6%.

INTRODUCTION

Climate smart practices are site-specific approaches focused on enhancing productivity and efficiency while simultaneously reducing greenhouse gas (GHG) emissions and maintaining ecosystem services. This definition implies that all appropriate gases should be considered when determining the carbon dioxide equivalence (CO2e). In agriculture the important gases are CO_2, N_2O, and CH_4. Carbon dioxide is produced during respiration, whereas nitrous oxide is produced during nitrification and denitrification. The emission of CO_2 and N_2O from soil as impacted by management has been widely reported (Joshi et al., 2022; Joshi et al., 2023; Reicks et al., 2021; Thies et al., 2020). What is often missing from these calculations is the impact of management on CH_4 emissions.

Methane is produced by methanogenic bacteria under anaerobic conditions through the reaction, $C_6H_{12}O_6 \rightarrow 3CO_2 + 3CH_4$. Because this reaction is anaerobic, it is most likely to occur in soil with very high water filled pore space. However, under aerobic condition it can be consumed by methanotrophic bacteria following the equation, $CH_4 + O_2 \rightarrow CO_2 + H_2O$. These biological processes are influenced by many factors including soil temperature, moisture, gas diffusivity, and nitrogen status (Ball et al., 1997; Fest et al., 2017; Grosso et al., 2000; Von fischer & Hedin, 2007). For example, increasing the soil temperature from 10 to 23 °C will increase methanogenic bacteria activity by 660% (Mer & Roger, 2001; Moore & Dalva, 1993). Soil moisture and pore space have a similar impact on methane synthesis and consumption.

When methane synthesis by methanogenic bacteria exceeds methanotrophic bacterial consumption, CH_4 is released into the atmosphere and soil becomes a net methane source (Mer & Roger, 2001; Holmes et al., 1999). However, when the reverse occurs the soil biota can consume methane and become a net methane sink. The switch between source and sink may be related to soil moisture and soil nutrient level (Joshi et al., 2022, 2023; Smeltekop, Clay, & Clay, 2002). Because cover crops often reduce soil moisture and inorganic N, they may contribute to the soil switching from a source to a sink (King & Schnell, 1994; Steudler et al., 1989). Therefore, because few papers have considered the impact of cover crops on CH_4 flux, the objective of this paper was to quantify the influence of dormant seeded rye (*Secale cereale* L.) cover crop on soil temperatures, soil moisture, inorganic N, total CH_4-C emission in a well-drained frigid soil from the start of growth in April/May through cover crop termination in late June.

MATERIALS AND METHODS

Experimental site

The field experiment was conducted at Aurora research farm, Aurora, SD (44°18´20.57´´ N, 96°40´14.04´´ W) which is located on the border between the Bsh (semi-arid) and DFa (continental wet all seasons) Köppen climate groups. The site's soil was a fine-silty, mixed, super active frigid Calcic Hapludoll (Brandt silty clay loam), and the surface soil (15 cm) contained 280 g clay kg^{-1} (28%), 65 g silt kg^{-1} (65%), 7 g sand kg^{-1} (7%), and 36 Mg ha^{-1} (1.8%) of soil organic carbon (SOC). For this soil, the SOC half-life and the no-tillage first order rate constant were 103 years and 0.00675 kg (kgC x year)$^{-1}$, respectively (Clay et al., 2015). The soil $pH_{1:1}$ was 5.8, and the soil parent materials were loess (0-60 cm) over glacial outwash. The surface soil hydraulic conductivity was 0.72 m d^{-1} and the slope was between 0 and 2%. The gravimetric water contents at field capacity and the wilting point were approximately 0.315 and 0.177 g g^{-1}, respectively (Thies et al., 2020). Our study was not irrigated, and it was not cultivated after seeding the rye. Before the study, the long-term rotation was corn (*Zea mays* L.) followed by soybean [*Glycine max* (L.) Merr.] (Reicks et al., 2021).

Experimental treatments

The experimental design was completely randomized with two treatments: cover crop and no-cover crop. Each treatment was replicated 4 times. The dimensions for each experimental unit were 9.1×3.1 m^2. Winter cereal rye (*Secale cereale* L.) was drilled in two rows at a rate of 56 kg ha^{-1}at a depth of 2.5 cm in October in the fall of 2017, 2018, and 2019. The two cover crop rows were separated by 17.5 cm, and they were positioned in the center between 2 corn rows. The cover crop occupied about 25% of the area between the corn rows. All soil was subjected to the prevailing weather conditions, no fertilizer was used, and there was very little residue cover. Seed emergence in late November was 17, 15, and 36% in 2017, 2018, and 2019, respectively (Reicks et al., 2021). These plants did not survive the winter and the remaining plants germinated in the following spring.

Soil sampling

At the start of an experiment in 2018, 2019, and 2020. soil samples from the 0–15 and 15–30 cm soil depths were collected with a 2-cm diameter probe. These samples were collected in an area adjacent to the GHG chambers. At the completion of the study, soil samples from the same depths were collected from the interior area of the chambers. Each sample contained 8 soil cores that were mixed, soil moisture was determined, and frozen until chemical analysis. Gravimetric soil moisture was determined by drying subsamples at 105 °C. Additional soil samples were collected for bulk density analysis, and in 2018, the bulk densities were 1.33 and 1.32 g cm^{-3} for the 0–15 and 15–30 cm depths, respectively. The bulk densities in 2019 were 1.31 and 1.28 g cm^{-3} for the 0–15 and 15–30 cm depths, respectively. The bulk densities in 2020 were 1.29 and 1.33 g cm^{-3} for the 0–15 and 15–30 cm depths, respectively. The % water filled pore space (WFPS) was calculated using the volumetric moisture contents and measured bulk densities. This calculation assumed that the soil particle density was 2.65 g cm^{-3}. Soil samples were air dried at 40 °C, ground (<2 mm) and analyzed for NH_4^+-N and NO_3^--N.

Methane emissions

As soon as it was physically feasible, greenhouse gas emissions measurements were initiated in the spring of each year. Emissions were measured using LI-COR long-term opaque chambers (8100-104 LI-COR, Lincoln NE), which pivoted over and covered the PVC ring to create an enclosed volume. Gas samples were collected for 15-minutes six times daily (between 0000 and 0230 h, 0400 and 0630 h, 0800 and 1030 h, 1200 and 1430 h, 1600 and 1830 h, and 2000 and 2230 h). The chambers were sampled in an ordered sequence at each gas sampling event, and the gas within the chamber was mixed with a pump, a vent equalized the chamber and atmospheric pressures, and the thermistor measured the air temperature.

The gas collected in the chamber was analyzed for CO_2, N_2O, and CH_4, using a Picarro Cavity Ringdown Spectrometer (model G2508: Picarro Inc., Santa Clara CA). The CO_2 and N_2O results were previously reported (Reicks et al., 2021, Joshi et al., 2022). CH_4-C emissions were calculated using 4.01 LI-COR SoilFluxPro software (v. 4.01; LI-COR). Standard gases were used before and after the completion of the experiment to check for accuracy. Soil moisture and temperature for the top 5 cm were monitored using LI-COR soil moisture (LI-8150-205) and soil temperature (LI-8150-203) probes that were pushed into the soil, outside and adjacent to the chambers in a similarly treated region.

From 7 May to 3 July 2018, 26 April to 24 June 2019, and 8 April to 26 June 2020, emissions were measured (Table 2-1). Three times a year, when rye reached 15 cm, they were cut to a height of 3 cm. At each clipping date, rye biomass was dried, weighed, ground, and analyzed for total N and C using a stable isotope C and N analyzer.

Three sampling intervals were chosen to represent the cover crop growth phase (Table 2-1). The first interval was from rye emergence to corn emergence (VE). The second interval was from corn emergence to the V2 growth stage in corn, and the third interval was from the V2 to V4 growth stage in corn. Corn growing degree days (GDD) were calculated using 10^{o} C as the lower limit and 30^{o}C as the upper limit (Nleya et al., 2016).

Statistical analysis

Total CH_4-C emission was determined by integrating the emissions over the 3 study periods. The experiment was repeated in 2018, 2019, and 2020. The t-test was used to determine the cover crop differences ($p \leq 0.05$) on CH_4-C emission, soil temperature, and water filled pore space. Methane and N_2O emissions were converted to carbon dioxide equivalence (CO_2e) multiplying g CH_4 by 27.8 and g of N_2O by 298 following IPCC (2006) guidelines. These two values were then summed up**.**

RESULTS AND DISCUSSION

Weather and Climate

The 30-yr (1989-2020) average annual rainfall was 640 mm, the average season rainfall (May-September) was 452 mm, the average growing degree days (10 ºC base and 30 ºC maximum temperature) from April to October was 1,256, the average annual temperature was 6.3 ºC, and the growing season average temperature was 17.9 ºC (NOAA, 2022). At the study site, the annual and growing season temperatures in 2018 were 6.25 and 19.53 ºC, respectively. Total annual rainfall in 2018 was 682 mm, of which 532 mm occurred during the growing season. The average temperatures from May 7 to July 3 in 2018, from April 26 to June 24 in 2019, and from April 8 to June 26 in 2020, were 20.05, 13.86 and 13.66 ºC, respectively. The cumulative rainfall during these periods is shown in Table 2-1. More information about average annual temperature, growing season temperature (May-September), snow depth and temperature of snow-covered soil for the year 2019 and 2020 can be found in Joshi et al. (2022).

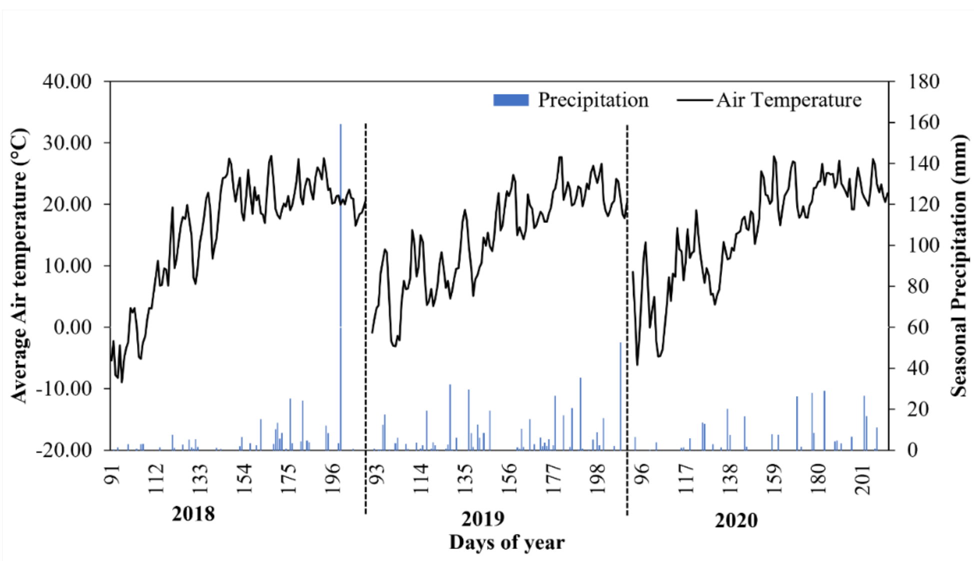

Figure 2- 1: Daily summary of average air temperature, total rainfall during April – July in 2018, 2019, and 2020. Data source: (South Dakota Mesonet, 2024)

Biomass production, inorganic N, precipitation, and N removal from soil

The amount of rye biomass produced was least in 2019 and greatest in 2020 (Table 2-1). The low biomass production in 2019 was attributed to cool and wet conditions. For example, from 26 April to 13 May there were only 25 GDD. The cool conditions also reduce soil microbial activity and root growth.

Table 2- 1: The total precipitation (cm), rye biomass produced (kg ha^{-1}) , carbon and nitrogen content (kg ha^{-1}) in rye biomass and growing degree days (GDD) for each of three growing phases of rye (rye emergence-VE corn, VE-V2 corn, and V2-V4 corn) during the sampling intervals in 2018, 2019, and 2020. The 95% confidence intervals are provided.

Rye emergence to corn's VE growth stage						
Year	**Sampling interval**	**Precipitation**	**Growing degree day**	**Dry rye biomass**	**C in rye biomass**	**N in rye biomass**
		cm	**°C**	**Kg ha^{-1}**	**Kg ha^{-1}**	**Kg ha^{-1}**
2018	7 May-25 May	2.59	132	279±15.76	111.0±6.28	13.39±0.75
2019	26 April-13 May	7.09	25	106±10.77	46.59±4.73	3.48±0.35
2020	8 April- 4 May	0.51	78	951±7.7	403.88±3.27	40.22±0.32
VE to corn's V2 growth stage						
2018	26 May-15 June	2.13	240	392±25.45	157±10.21	16.93±1.0
2019	14 May-29 May	8.05	64	69±11.34	29.75±0.25	1.55±0.26
2020	5 May-29 May	8.46	112	883±24.7	369.84±10.90	27.10±0.75
V2 to corn's V4 growth stage						
2018	16 June-3 July	10.31	225	378±11.76	154±4.82	12.77±0.39
2019	30 May- 24 June	5.44	232	253±19.38	112.38±8.60	4.76±0.36
2020	30 May-26 June	7.80	291	843±71.1	370.16±31.22	20.56±1.73

In 2018, the initial NO_3-N and NH_4-N concentrations in the surface 30 cm were 3.7±0.32 and 6.68±0.57 mg kg^{-1}, respectively, and when rye was terminated on 3 July 2018, the NO_3-N concentrations in the soil and rye treatments were similar but numerically lower in the rye (7.11 ± 0.91 mg kg^{-1}) than the soil (9.03 ± 2.94 mg kg^{-1}). At cover crop termination, the NH_4-N concentrations in the soil and rye treatments were similar, which was 5.41 ± 0.83 mg kg^{-1}. In 2019 when the experiment was initiated the initial NO_3-N concentration (26 April) was 14.3 ± 7.3 and the initial NH_4-N concentration was 20.3 ± 4.75 mg kg^{-1}. When rye was terminated on 24 June 2019, the NO_3-N concentration in the no cover crop and cover crop treatments were 8.66 ± 1.84 and 4.12 ± 0.26 mg $kg^{-1,}$, respectively. However, rye did not influence the NH_4-N concentrations and it was 10.3 ± 3.99 mg kg^{-1} in both treatments. In 2020, the NO_3-N and NH_4-N concentrations in the surface 30 cm prior to the study were 6.25 ± 1.22 and 43.6 ± 21 mg kg^{-1}, respectively. When the experiment was terminated on 26 June 2020, NO_3-N in the surface 30 cm was 7.11 ± 1.95 in the no-cover crop treatment and 2.5 ± 1.56 mg kg^{-1} in the rye cover crop treatment. However, at termination rye did not influence NH_4-N concentration and it was 2.8 ± 1.77 mg kg^{-1} in both treatments.

Vegetative rye impact on soil moisture and temperature.

When assessing GHG emissions, soil moisture and precipitation should be considered. The higher the soil moisture (WFPS), the higher the water filled pore space. When water filled pore space become greater than 60%, oxygen diffusion can become limiting and the soil can switch from aerobic to anaerobic respiration (Linn & Doran, 1984). Under anaerobic conditions, nitrate can be used as the terminal electron acceptor and methane emissions can increase.

Soil moisture varied during the study and was generally greatest following precipitation. Following precipitation, the combined impacts of soil water movement and evapotranspiration reduced soil moisture. However, there were periods when the WFP was higher than 60%, which has been reported to be the tipping point between aerobic and anaerobic conditions (Linn & Doran, 1984). In 2018, the WFPS in no-cover crop treatment frequently exceeded the 60% WFPS between rye germination and corn emergence (VE) and from corn VE to V4 growth stage (Table 2-2, 2-3, 2-4). In the cover crop treatment, rye generally reduced the WFPS during these sampling periods. In 2019, due to high rainfall and limited growth (Table 2-1) rye did not influence WFPS from rye emergence to corn's V4 growth stage (Table 2-2, 2-3, 2-4). In 2020, the WFPS of the cover crop was lower than that of no-cover crop at every period. These results were attributed to rye increasing transpiration. Considering, within year results, rye reduced soil moisture or water filled porosity ($p < .05$) in 2018 and 2020. However, different results were observed in 2019, when rye did not reduce soil moisture.

Table 2- 2: Average daily methane flux (g CH_4-C ha^{-1} day^{-1}) , water-filled porosity space (WFPS,%), and temperature (ºC)of the no-cover crop and the rye cover crop during the interval between Rye emergence to VE corn in 2018, 2019, and 2020. A negative flux indicates that the soil methane oxidation occurred.

Rye emergence to corn's VE growth stage				
Treatment	**Year**	**Methane flux**	**WFPS**	**Soil temperature**
		g ha^{-1} day^{-1}	**%**	**ºC**
Cover crop	2018	-14.05	62	16
No-cover crop	2018	-2.46	74	15
p-value		0.01	<0.01	0.48
Cover crop	2019	-6.56	57	8
No-cover crop	2019	-1.08	51	8
p-value		<0.01	0.11	0.89
Cover crop	2020	1.67	27	8
No-cover crop	2020	8.95	50	9
p-value		<0.01	<0.01	0.46
2018		-8.26	68	15
2019		-3.82	54	8
2020		5.31	38	8
p-value		0.21	0.15	0.001

Cover crop	-2.45	48	10
No-cover crop	3.94	58	10
p-value	0.04	0.37	0.93

Table 2- 3: Average daily methane flux (g CH_4-C ha^{-1} day^{-1}), water-filled porosity space (WFPS,%), and temperature (ºC) of the no-cover crop and the rye cover crop during the interval between VE to V2 corn in 2018, 2019, and 2020. A negative flux indicates that the soil methane oxidation occurred.

VE to corn's V2 growth stage				
Treatment	**Year**	**Methane flux**	**WFPS**	**Soil temperature**
		g ha^{-1} day^{-1}	**%**	**ºC**
Cover crop	2018	-6.37	43	22
No-cover crop	2018	-5.14	53	21
p-value		0.60	<0.01	0.23
Cover crop	2019	-3.87	64	12
No-cover crop	2019	-3.66	59	12
p-value		0.89	0.41	0.86
Cover crop	2020	1.97	45	12
No-cover crop	2020	6.33	64	13
p-value		0.15	<0.01	0.32
2018		-5.76	48	22
2019		-3.77	61	12
2020		4.15	54	12
p-value		0.02	0.43	<0.01
Cover crop		-2.76	50	15
No-cover crop		-0.82	58	15
p-value		0.13	0.37	0.97

Soil temperature was not impacted by the rye cover crop in 2018, 2019, and 2020. These results are different than Blanco-Canqui and Ruis (2020) who conducted a review and reported that cover crops can decrease temperatures during the daytime and increase temperatures at nighttime.

Table 2- 4: Average daily methane flux (g CH_4-C ha^{-1} day^{-1}), water-filled porosity space (WFPS,%), and temperature (ºC) of the no-cover crop and the rye cover crop during the interval between V2 to V4 corn in 2018, 2019, and 2020. A negative flux indicates that the soil methane oxidation occurred.

		V2 to corn's V4 growth stage		
Treatment	**Year**	**Methane flux**	**WFPS**	**Soil temperature**
		g ha^{-1} day^{-1}	**%**	**ºC**
Cover crop	2018	-7.90	50	22
No-cover crop	2018	-3.07	72	21
p-value		0.16	<0.01	0.15
Cover crop	2019	-5.32	44	19
No-cover crop	2019	-4.24	52	16
p-value		0.68	0.05	<0.01
Cover crop	2020	-6.57	30	20
No-cover crop	2020	-6.07	48	21
p-value		0.89	<0.01	<0.01
2018		-5.49	61	21
2019		-4.78	48	18
2020		-6.32	39	20
p-value		0.76	0.32	0.12
Cover crop		-6.60	41	20
No-cover crop		-4.46	57	19
p-value		0.13	0.06	0.65

Vegetative rye impact on methane flux and total emissions.

Methane fluxes studies involving cover crop treatments are very limited (Abdalla, et al., 2014; Sanz-Cobena, et al., 2014; Behnke & Villamil, 2019). For example, from 2014 to 2017, Behnke & Villamil (2019) measured GHG 21 times. Our study used near continuous observations taken 6 times a day and over 7 days, 42 measurements were collected in each plot. From cover crop emergence to corn's VE growth stage there were large differences ($p<0.01$) in CH_4 fluxes in the no-cover crop and cover crop treatments (Table 2-2). It is important to point out that many of the measurements were negative, which means that the soil was a sink. This finding agrees with Sanz-Cobena et al. (2014) who found the greatest difference between treatments after 229 days of sowing cover crop in previous fall, which coincided with rye emergence to corn's VE growth stage in our study. Abdalla, et al. (2014) had slightly different results and reported that in a sandy soil, CH_4 flux ranged from near zero to negative and that the highest flux occurred in summer and that was low or negative during the winter. Sanz-Cobena et al. (2014) reported that in a silty clay loam soil, all cover crop except vetch increased CH_4 removal.

In all three years, the rye cover crop was a stronger methane sink than the no-cover crop treatment. The total methane flux (cumulative) in no-cover crop treatment, considering whole cover crop growing period from rye emergence to corn's V4 growth stage, were -0.21 ± 0.06, -0.16 ± 0.017, and 0.22 ± 0.03 kg CH_4-C ha^{-1} in 2018, 2019, and 2020, respectively while in the cover crop treatment, the fluxes were -0.54 ± 0.01, -0.33 ± 0.01, and -0.10 ± 0.02 kg CH_4-C ha^{-1}, in 2018, 2019, and 2020, respectively. Similarly, the daily methane flux varied across years with a highly negative flux in 2018, followed by 2019 and the least negative in 2020 (Table 2-2, 2-3, 2-4). These temporal differences may be attributed to residual NH_4-N concentration in the soil which was greatest in 2020 and least in 2018. This interpretation is consistent with nitrification and CH_4 competing for the active site on the methane monooxygenase enzyme complex (Bedard & Knowles, 1989; Dalton, 1977; Wang & Ineson, 2003). Hence, cover crop could have assimilated NH_4^+-N in its biomass to decrease the inhibitory effect of this inorganic N to enhance soil methane sink or uptake.

Previous studies have shown that the temperature and soil moisture have confounding effect on methane flux. For example, Bowden et al. (1997), Conrad et al. (1996), Khalil and Baggs (2005), King and Adamsen (1992), and Smith et al. (2003) reported that temperature has a smaller effect on the methane flux than the soil moisture. Schaufler et al. (2010) reported CH_4 to be negatively correlated with soil moisture and only scarcely related (nearly indifferent) to soil temperature. In our experiment, the cover crop biomass in 2019 may not have been sufficient to control the soil moisture, resulting in nonsignificant differences between the no-cover crop and cover crop treatment. Khalil and Baggs (2005) reported that soil moisture content can affect CH_4 fluxes by altering the diffusion of CH_4 and O_2 through the soil profile, and in addition, it can affect methanogens and methanotrophs growth in soil. Shukla et al. (2013) reported that gas diffusion can be restricted when soil moisture content exceeds 56% (WFPS). The moisture content in our study in case of no-cover crop frequently averaged to 50% WFPS, which could have limited O_2 diffusion and oxidation. As reported by Luo et al. (2013), a more aerobic soil condition could inhibit methanogen growth and CH_4 production. This suggested that the cover crop treatment induced decrease in soil moisture and inorganic N (NH_4-N) which might have enhanced the soil methane sink in this study.

Vegetative rye impact on CO_2 equivalence (CO2e).

Carbon dioxide equivalence (CO_2e) is the recommended metric for expressing warming effect of each of the greenhouse gases in reference to CO_2. CH_4, which eventually gets converted to CO_2 in atmosphere, was converted to its carbon dioxide equivalence (CO_2e-CH_4) by multiplying by factor 27.8 and N_2O to CO_2e-N_2O by multiplying by 298 (Heald & Kroll, 2020; IPCC, 2006).

Cover crop reduced methane carbon dioxide equivalence (CO_2e-CH_4) in comparison to no-cover crop treatment. However, this beneficial effect was not constant over years, cover crop increased methane removal from the atmosphere. The CO_2e-CH_4 in no-cover crop treatment was -7.82 ± 2.5, -5.94 ± 0.65, and 7.97 ± 0.12 kg ha^{-1} in 2018, 2018, and 2020, respectively, while in the cover crop treatment it was -19.9 ± 0.63, -12.24 ± 0.49 and -3.62 ± 2.3 kg ha^{-1} in the same years, respectively. The result suggests that cover crop contributed to the reduction of CO_2e or the removal of methane from the atmosphere.

Table 2- 5: Comparison of nitrous oxide carbon equivalence (CO_2e-N_2O, kg ha^{-1}) and nitrous oxide-methane carbon equivalence (CO_2e-N_2O+CO2e-CH_4, kg ha^{-1}) in between cover crop and no-cover crop treatments prior to termination in year 2018, 2019, and 2020. % change is the change in carbon equivalence with and without considering methane in calculation.

Treatment	**CO_2e-N_2O**	**CO_2e-CH_4**	**CO_2e-N_2O+ CO_2e-CH_4**	**Change**
	Kg ha^{-1}	Kg ha^{-1}	Kg ha^{-1}	%
Cover crop	60.11	-11.92	48.19	24.44
No cover crop	126.28	-1.93	124.29	6.00
p-value	0.07	0.03	0.06	

Methane (CH_4) emission might have profound consequences on the atmosphere, which could extend well beyond its warming potential when evaluated in terms of carbon dioxide equivalence. As CH_4 undergoes conversion to CO_2 in the atmosphere, it generates various byproducts that exert broader effects on climate, ecosystem, and human health, transcending the warming effect alone (Smith et al., 2021). For example, chemically very active, CH_4, consumes hydroxyl radicals (OH^{-1}), crucial for oxidizing other trace gases and aerosols. This diminishes the oxidation of sulfate aerosols (SO_2 + OH), which possess a cooling effect. Additionally, it contributes to the production of ozone (O_3) in the troposphere and water vapor (H_2O) in the stratosphere, both of which influence plant growth, human health, and broader climate system (Mar et al., 2022).

Reicks et al. (2021) and Joshi et al (2022) have previously reported that CO_2-C or CO_2e-CO2 was higher in cover crop treatment than no cover crop treatment. However, when considering CO_2 sequestered in cover crop biomass, soil with cover crop was net sink for CO_2. In addition, they reported that N_2O-N or CO_2e-N_2O was significantly lower in cover crop treatment than no cover crop treatment. In our study, we found that the soil with cover crop behaved as sink or removed methane from the atmosphere. Keeping the CO_2 and its sequestration in cover crop as idle, it was deemed worthwhile reporting the changes in CO_2e from N_2O and CH_4 as impacted by cover crop. When methane was considered, the change in equivalence figure ranged from 4.08-36.92% in cover crop treatment and 4.39-6.90% in no-cover crop treatment. Cover crop induced increase in methane sink and decrease in N_2O emission decreased emissions or CO_2e by 61.22% in comparison to no-cover crop prior to termination across the three years.

CONCLUSIONS

In this experiment, the impact of dormant seeded, unfertilized growing cover crop on soil temperatures, soil moisture, inorganic N, total CH_4-C emission and CO_2e in a well-drained frigid soil from the start of growth in April/May through termination in late June were investigated with the use of near continuous measurements of emissions. There are only a few studies which focused on the effect of cover crops on methane emissions. This study is one of the few investigating CH_4-C emission from growing cover crop in Midwest, US.

Our research showed that when compared to no-cover crop, rye reduced soil WFPS in the surface 5 cm by 23, 2, and 35% in 2018, 2019, and 2020, respectively. Rye also reduced the NO_3-N concentration in surface 30 cm of soil by 21, 52 and 64% in 2018, 2019 and 2020, respectively. The change in NH_4-N concentration from the start of experiment till termination was 19, 49, and 93% in 2018, 2019, and 2020, respectively. Our study showed that from cover crop emergence to corn emergence (VE), rye increased the strength of the methane sink in all three years.. The increase in methane sink and decrease in N_2O emission induced by the cover crop resulted in a reduction of emission or CO_2e by 61.22% compared to no-cover crop prior to termination across the three years. The residual soil NH_4-N and soil moisture were the factors impacted by the cover crop, resulting in differences between treatments and across three years. Our study showed that from germination to termination considering CH_4 in the CO_{2e} calculation reduce the CO_2e from N_2O by 24.4% in the cover crop treatment and 6% in the no-cover treatment. Our findings support that CH_4 emission should be considered while reporting CO_2e and that in addition to reducing N2O, cover crops also reduce CH_4 emissions. Ignoring it would underestimate or overestimate total carbon dioxide equivalence. Additional research is needed to confirm these results at other sites.

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FINAL STATEMENT

Use of sensors in assessment of problems will aid in making management decisions. This will add our capacity to intensify experimental observations so that a process can be understood in detail by keeping the informational pieces together. Our research showed that rye cover crop, in comparison to no-cover crop (bare soil), reduced methane emission by 155, 106, and 145%, in year 2018, 2019, and 2020, respectively. The carbon-dioxide equivalents (CO2e) ranged from 7.79 kg ha-1 to -7.82 kg ha-1 in no-cover crop treatment while it ranged from -3.62 kg ha-1 to -19.9 kg ha-1 in cover crop treatment. So, we conclude that CH_4-C emission should also be included while reporting agricultural greenhouse gas emission. These results are attributed to cover crop induced decrease in soil moisture and inorganic N (NH_4^+-N). The finding that the early cover crop growing phase (rye emergence to corn emergence) may be significant contributor to overall effect on methane emission of entire cover crop growing phase is interesting one. This result could be further strengthened by looking into the impact that the cover crop might have on changing the population dynamics of methanogens and methanotrophs using specific markers, qPCR and transcriptome analysis of rye rhizosphere and bulk soil. Further addition of redox potentials measurements into our greenhouse gas emission studies might add more information. So, our study will focus on this direction.

Our immediate goal will be to quantify the impact of decomposing rye cover crop on methane emission, soil moisture and soil temperature. We will then use machine learning based model for prediction of daily methane flux by using variables: soil moisture, rainfall, air temperature, soil temperature, CO_2-C flux, carbon remaining in cover crop biomass as predictors.

Additionally, we will continue to evaluate the impact of various soil management practices such as no-till, cover crop and N-fertilization on greenhouse gas emissions with focus on N_2O-N, CO_2-C and CH_4-C. The biological processes underpinning this emission will be further assessed by the help of soil microbial DNA and RNA analysis. Information from soil analysis, crop biomass analysis, greenhouse gases emission and genetic and transcriptomic analysis will bring a holistic approach to our study.

SUPPLEMENTARY FIGURES AND TABLES

Supplementary table 1: Average daily methane flux (g CH_4-C ha^{-1} day^{-1}), water-filled porosity space (WFPS, %), and temperature (ºC) in the no-cover crop and the rye cover crop treatments during the entire cover crop growing season in 2018, 2019, and 2020. A negative flux indicates that the soil methane oxidation occurred.

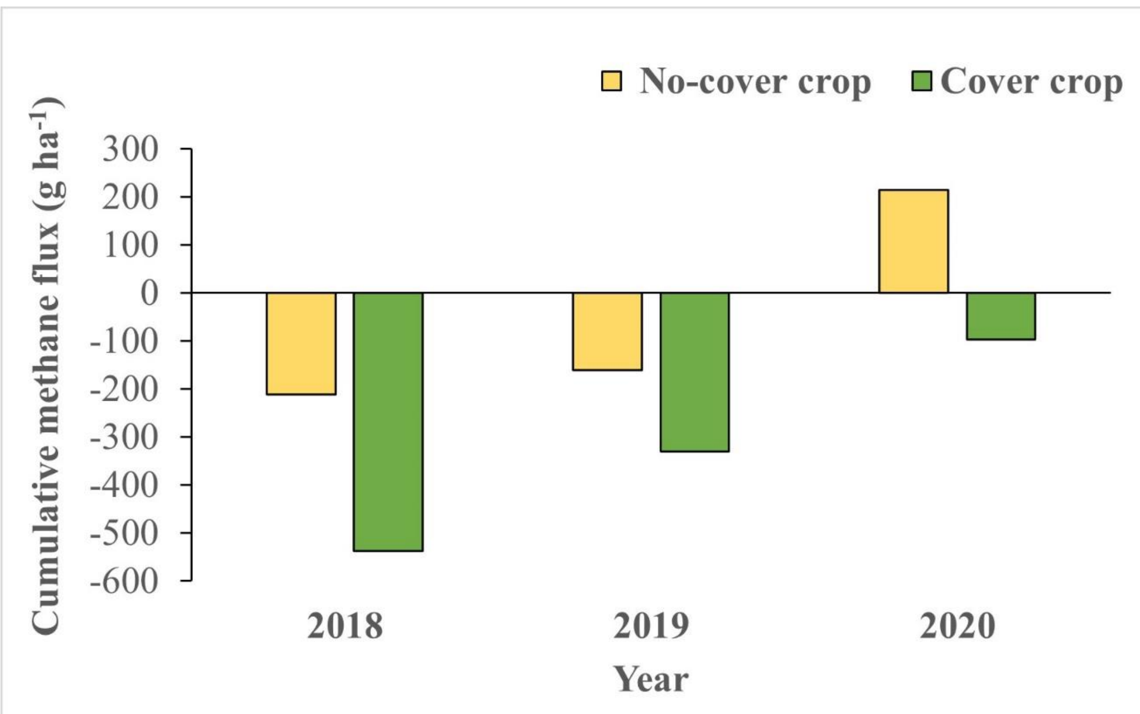
Overall rye growing phase (Rye emergence to Corn V4)				
Treatment	**Year**	**Methane flux**	**WFPS**	**Soil temperature**
		g ha^{-1} day^{-1}	**%**	**ºC**
Cover crop	2018	-9.25	51	20
No-cover crop	2018	-3.63	66	20
p-value		<0.01	<0.01	0.24
Cover crop	2019	-5.49	54	8
No-cover crop	2019	-2.66	53	8
p-value		<0.01	0.82	0.89
Cover crop	2020	-1.21	34	14
No-cover crop	2020	2.68	54	15
p-value		0.03	<0.01	0.16
2018		-5.48	58	22
2019		-4.78	53	18
2020		-6.32	44	21
p-value		0.76	0.45	0.12
Cover crop		-5.31	46	20
No-cover crop		-1.20	57	20
p-value		0.03	0.21	0.65

Supplementary table 2: Cumulative methane flux (g CH_4-C ha^{-1}) from no-cover crop and cover crop treatments during entire cover crop growing season in year 2018, 2019, and 2020.

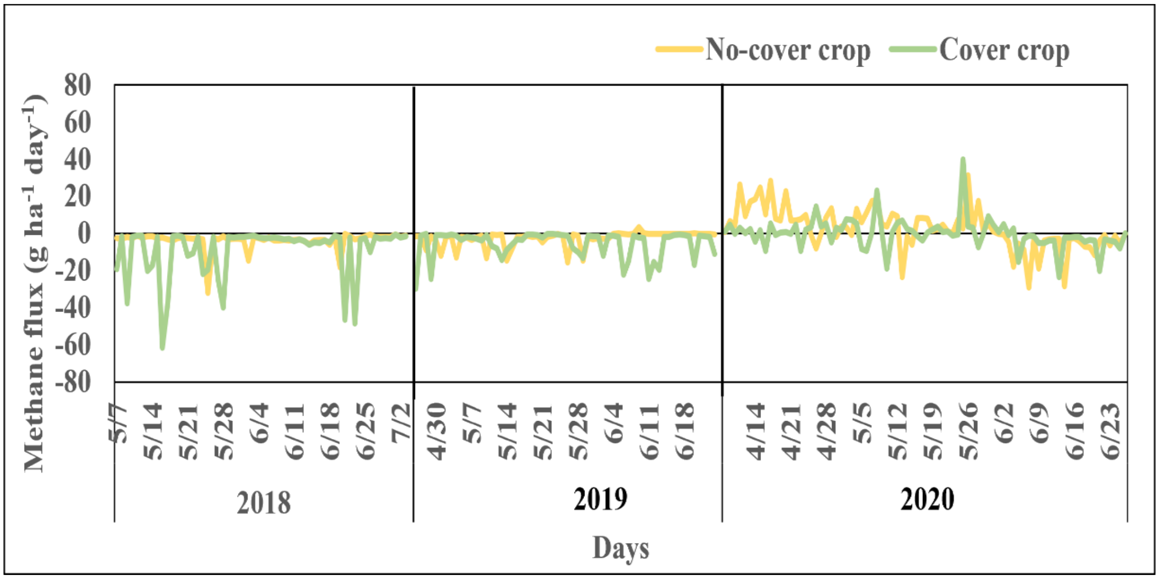
	Cover crop	**No-cover crop**
Year	**Cumulative methane flux**	
	g ha^{-1}	
2018	-536.8	-210.84
2019	-329.93	-160.02
2020	-97.52	214.86
Total	-964.25	-156.00

Supplementary table 3: Comparison of nitrous oxide carbon equivalence (CO_2e-N_2O, kg ha^{-1}) and nitrous oxide-methane carbon equivalence (CO_2e-N_2O+CO_2e-CH_4, kg ha^{-1}) in between cover crop and no-cover crop treatments prior to termination in year 2018, 2019, and 2020. % change is the change in carbon equivalence with and without considering methane in calculation.

Year	CO_2e-N_2O		CO_2e-CH_4		CO_2e-N_2O+CO_2e- CH_4		% Change	
	kg ha^{-1}		kg ha^{-1}		kg ha^{-1}		%	
	Cover crop	No-cover crop	Cover crop	No-cover crop	Cover crop	No-cover crop	Cover crop	No-cover crop
2018	53.89	177.74	-19.90	-7.82	33.99	169.92	36.92	4.39
2019	37.86	88.22	-12.24	-5.94	25.62	82.28	32.32	6.73
2020	88.58	112.88	-3.62	7.97	84.96	120.67	4.08	6.90



Supplementary figure 1: Cumulative methane flux (g CH_4-C ha^{-1}) from no-cover crop and cover crop treatments during entire cover crop growing season in year 2018, 2019, and 2020.



Supplementary figure 2: Daily methane flux (g CH_4-C ha^{-1} day^{-1}) from no-cover crop and cover crop treatments during entire cover crop growing season in year 2018, 2019, and 2020.