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Evaluating the Effects of Organic Amendment Applications on Nitrous Oxide Emissions From Salt-Affected Soils

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Evaluating the Effects of Organic Amendment Applications on Nitrous Oxide Emissions From Salt-Affected Soils

A Dissertation submitted in partial satisfaction
of the requirements for the degree of

Doctor of Philosophy

in

Soil and Water Sciences

by

Namratha Pulla Reddy Gari

December 2013

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DEDICATION

This dissertation is dedicated to my mother and brother, Uma Rani and Nagin Kumar, for their unwavering support and encouragement during my graduate studies. I am indebted to my Uncles, Babu Rao and Chandra Sekhar, for their help in providing me with an opportunity for better education. I would also like to thank my friends at UCR, Vijay Chaganti and Julie Escalera, for their continued support and encouragement during tough times. Lastly, I dedicate my work to my large family who have inspired and motivated me to take interest in science.

ABSTRACT OF THE DISSERTATION

Evaluating the Effects of Organic Amendment Applications on Nitrous Oxide Emissions From Salt-Affected Soils

by

Namratha Pulla Reddy Gari

Doctor of Philosophy, Graduate Program in Soil and Water Sciences
University of California, Riverside, December 2013
Dr. David M. Crohn, Chairperson

Soil salinization and greenhouse gas emissions are major global environmental concerns. The extent of salinization and associated negative effects on soils and crop yields make the reclamation of these soils an international priority. With increasing interest in the use of organic amendments for remediating salt-affected soils, it is important to investigate their interactive effects on soil biogeochemical processes including greenhouse gas emissions, particularly carbon dioxide (CO_2) and nitrous oxide (N_2O). The objectives of this dissertation are to: 1) evaluate the effects of soil salinity, temperature, and carbon availability from organic amendments on CO_2, N_2O, and N_2 emissions; 2) determine the effect of changes in salinity and temperature on soil mineral N concentration from salt-affected soils following organic amendment applications; and 3) determine whether CO_2 and N_2O emissions under field conditions were comparable qualitatively to those observed in the laboratory incubation studies. Organic amendments used in this research included: active greenwaste (AGW), cured greenwaste compost (CGW), active dairy manure (ADM), and cured dairy manure compost (CDM). The methods used in this research included monitoring CO_2 and N_2O emissions, estimating N_2 emissions using acetylene block technique, and analysis of soil mineral N concentrations in the laboratory and field studies.

Results from laboratory incubations showed that increases in soil salinity enhanced cumulative N_2O-N losses but decreased cumulative CO_2-C and N_2 emissions and N_2 to N_2O-N ratios. Increases in soil temperatures greatly enhanced cumulative CO_2-C, N_2O-N, and N_2 emissions and ratios of N_2 to N_2O-N from all treatments. In the field, results validated the laboratory findings that active organic materials, particularly AGW, reduced N_2O emissions compared to cured amendments. In general, dairy manure amendments produced higher N_2O emissions relative to the greenwaste treatments. In both laboratory and field studies, soils amended with greenwaste materials had lower soil nitrate concentrations compared to those treated with dairy manure amendments.

Overall, this work showed that soil salinization resulted in greater N_2O emissions following organic amendment applications. Nitrous oxide emissions are dependent on climatic conditions (*e.g.,* precipitation and temperature), soil properties (*e.g.,* electrical conductivity, microbial respiration, and nitrate concentration), organic amendment properties (*e.g.,* feedstock and processing stage), and their interactions.

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1. LITERATURE REVIEW

Soil degradation is a major widespread environmental constraint with severe negative effects on agricultural productivity. According to Global Assessment of Human-induced Soil Degradation (GLASOD), 1964 Mha of land is degraded globally with 46% under moderate and 16% under strong and extreme levels of degradation (Oldeman, 1994). Soil degradation can occur due to physical (*e.g.,* erosion, desertification, and slaking), chemical (*e.g.,* acidification, alkalinization, salinization, and nutrient depletion), and biological (*e.g.,*depletion of soil organic carbon and decline in biodiversity) processes (Lal et al., 2004).

Soil Salinization

Soil salinization is considered to be the second largest cause of land degradation after soil erosion by the United Nations Environmental Program (UNEP). Saline soils are characterized by excess levels of carbonates, bicarbonates, chlorides, and sulfates of sodium and calcium in the soil solution sufficient to adversely impact plant growth and development. Salt-affected soils occur in more than 100 countries and pose a serious economic problem as they decline agricultural productivity. These soils are more common in arid and semi-arid regions of the world due to the low precipitation and high evaporative demands prevalent in these areas (Pitman and Läuchli, 2002). In addition, anthropogenic activities such as improper land and water management practices particularly irrigation with poor quality water and lack of drainage systems also contribute to soil salinization. The total area of salt-affected soil is about 950 million hectares (Rengasamy, 2006). It is estimated that nearly 20% of the global and 25-30% of the United States irrigated land is salt-affected (Nelson and Maredia, 2001; Wichelns, 1999).

Salt-affected soils are classified into three categories based on the electrical conductivity of saturated soil paste extracts (EC_e) and sodium adsorption ratio (SAR): (i) Saline soils having $EC_e > 4$ dS m^{-1} and SAR < 13; (ii) Sodic soils with $EC_e < 4$ dS m^{-1} and SAR > 13; and (iii) Saline-Sodic soils with $EC_e > 4$ dS m^{-1} and SAR >13 (Richards, 1954). Excessive salts in the soil have detrimental effects on plant growth and yield by limiting plant-available water and or by inducing ion toxicity. Several soil physical, chemical, and biological properties are influenced by soil salinity and or sodicity. These include reduced water and nutrient holding capacity, depressed soil infiltration rates, poor soil aggregation, low cation exchange capacity, and inhibited microbial activity (Lauchli and Grattan, 1990; Tejada et al., 2006).

The seriousness and extent of soil salinization problems around the world necessitates urgent solutions to abate their spread, particularly in arid and semi-arid regions. For many countries, a key to reducing food insecurity will be to effectively ameliorate salt-affected soils to improve their productivity, especially in areas where irrigation infrastructures have been well established (Qadir et al., 2006). Over the past century, many different approaches including chemical amendments, particularly gypsum, tillage operations such as deep plowing, and irrigation strategies have long been used to reclaim salt-affected soils with varying results (Amezketa et al., 2005; Oster et al., 1996). More recently, organic amendments such as manures and composts, varying in their levels of processing and characterization, have been investigated for their reclamation as a low-cost alternative to chemical amendments (Liang et al., 2005; Tejada et al., 2006). It has been demonstrated that the application of organic amendments to saline soils can improve crop yields, as well as soil properties such as fertility, structural stability, bulk density, water infiltration, microbial biomass, and enzymatic activities (Lakhdar et al., 2008; Liang et al., 2003; Liang et al., 2005; Tejada et al., 2006).

While the effects of soil salinization on soil physical, chemical, and biological properties have been well studied, their effects on greenhouse gas emissions such as carbon dioxide (CO_2) and nitrous oxide (N_2O), particularly when being reclaimed using organic amendments remain relatively unexplored.

Nitrous Oxide Emissions

There are several natural and anthropogenic sources of atmospheric N_2O including oceans, biomass burning, industrialization, and soil. Of these, however, soils are identified as a major source of global N_2O emissions with 6.6 Tg yr^{-1} from natural soils and 4.2 Tg yr^{-1} from agricultural soils out of an estimated total of 17.7 Tg yr^{-1} from all sources (IPCC, 2001). In the United States, average N_2O emissions from agricultural soils were approximately 0.79 Tg in 2011 of which cropland accounted for 0.4 Tg, a 21% increase since 1990 (EPA, 2013). In 2005, California Energy Commission (CEC) estimated that about 64% of total N_2O emissions come from agricultural soils. The reliability of this estimate is uncertain, however, due to a lack of sufficient field measurements in California and to the high spatial and temporal variability of N_2O fluxes from soils.

Environmental Impact of Nitrous Oxide

Nitrous oxide can directly contribute to greenhouse effect by absorption of infrared radiation thereby trapping thermal radiation emitted from the earth's surface. The global Warming Potential (GWP) concept originally developed by IPCC is an index that relates the climate impacts of a greenhouse gas emission to that of an equivalent mass of CO_2 produced (IPCC, 2001). In 2013, the GWP of N_2O was estimated at 310 times that of CO_2 and approximately 15 times that of CH_4 over a 100-year period by the United States Environmental Protection Agency (U.S.E.P.A). Its efficiency in trapping light in the infrared region makes it important as a greenhouse gas. Nitrous oxide is a stable and potent greenhouse gas with an atmospheric lifetime of 120 years and plays an important role in stratospheric chemistry (EPA, 2013).

Ozone losses result in higher intensity of UV-B radiation reaching the earth's surface. As a precursor to NO_x gases which include nitric oxide (NO) and nitrogen dioxide (NO_2), nitrous oxide (N_2O) plays a vital role in regulating stratospheric ozone chemistry. Several studies have reported the depletion of stratospheric ozone with increased N_2O concentrations (Bouwman, 1990; Crutzen, 1979; Crutzen and Ehhalt, 1977; Hahn and Junge, 1977). The ozone depletion potential of a chemical (ODP), defined as the amount of ozone depletion caused by a compound relative to that caused by trichlorofluoromethane (CFC-11), was estimated for N_2O to be 0.017 by Ravishankara et al. (2009). This value is comparable to an ODP of less than 0.02 documented for many Class II ozone-depleting substances including hydrochlorofluorocarbons (HCFCs) such as HCFC-123,124, 225ca and 225cb (IPCC, 2005; WMO, 2007) that are currently being included in the Montreal Protocol, an international treaty designed to reduce and or eliminate the production and consumption of ozone depleting substances responsible for the depletion of the ozone layer (UNEP, 2010).

Apart from depleting the ozone layer, N_2O emissions from soils reduce the efficiency of the applied nitrogen (N) fertilizer thereby reducing the soil N pool available for plant uptake. The origin of this gas from agricultural soils is primarily attributed to the anaerobic reduction of nitrate by denitrifying microorganisms; however, aerobic oxidation of ammonium *i.e.* nitrification also contributes to N_2O emissions (Blackmer et al., 1980; Bremner and Blackmer, 1978; Bremner and Blackmer, 1979).

Nitrogen Cycle and N_2O Emissions

Nitrogen is an essential nutrient present in all living organisms including soil microbes such as bacteria and fungi within cells where it is bound in proteins and DNA. Mineralization is a biological process by which free NH_4^+ is formed from the decomposition of organic N compounds by microbes. A two-step process known as nitrification further oxidizes NH_4^+ to NO_2^- and subsequently to NO_3^-. Immobilization of N which involves the incorporation of inorganic N to organic N forms may also occur, thus altering the dynamics of soil N. As a general rule, substrate C:N exceeding a threshold value of 25:1 have been shown to immobilize N whereas those with lower C:N ratios are associated with net N mineralization (Janssen, 1996).

Nitrogen immobilization, mineralization, and nitrification processes occur under aerobic conditions in which heterotrophic microorganisms utilize the available C and N in the substrate to meet their metabolic needs. Under anaerobic conditions, however, NO_3^- can be reduced to N_2 gas during the process of biological denitrification, commonly considered as the last step in soil N cycle. Denitrification is a respiratory pathway performed by a variety of facultative aerobic and heterotrophic microorganisms such as Archaea, bacteria, and fungi with most denitrifiers falling under four major genera: *Pseudomonas, Alcaligenes, Bacillus,* and *Flavobacterium* (Tiedje, 1988). The pathway for complete denitrification involves the reduction of NO_3^- to N_2 gas through a series of intermediate by-products including nitrogen dioxide (NO_2), nitric oxide (NO), and nitrous oxide (N_2O) catalyzed by a range of reductase enzymes (Tiedje, 1988). Denitrifiers thus utilize NO_3^- and N oxides as terminal electron acceptors instead of O_2 in the electron transport phosphorylation to synthesize energy. Denitrifiers mainly depend on the substrate and its availability to produce N_2O or N_2.

Both nitrification and denitrification are known to regulate N_2O production in soils (Davidson et al., 1986). Nitrous oxide is predominantly produced during biological denitrification of NO_3^- to N_2 gas as an intermediate byproduct. Despite the strong role of soil microorganisms, several abiotic factors including soil properties regulate fluxes of N_2O (Bremner and Shaw, 1958).

Influence of Soil Properties on N_2O Emissions

A wide array of soil factors can be identified as potentially significant to the formation of N_2O, including soil texture (Groffman and Tiedje, 1991; Sexstone et al., 1985; Skiba and Ball, 2002), pH (Firestone et al., 1980; Focht, 1974; Goodroad and Keeney, 1984), moisture (Maag and Vinther, 1996; Zheng et al., 2000), temperature (Goodroad and Keeney, 1984; Maag and Vinther, 1996; Smith et al., 1998), and organic matter (Aulakh et al., 1991; Chang et al., 1998; Huang et al., 2004; Meng et al., 2005).

Texture

Soil texture largely influences denitrification by controlling aeration and water holding capacity. In general, coarse textured soils contain more macropores and are therefore well aerated compared to fine textured soils that contain more micropores. Oxygen is a direct and dominant abiotic regulator of denitrification and clay soils can easily become anaerobic than loamy and sandy soils (Papendick and Campbell, 1981) thereby causing greater N_2O production in fine textured soils. Groffman and Tiedje (1991) found higher denitrification rates in clay loam soils even under well-aerated conditions compared to loam soils. Also, increase in clay content has been shown to increase water holding capacity of soils (Gupta and Larson, 1979). Sexstone et al. (1985) documented an increase in denitrification rates in clay loam soil than in a sandy loam soil due to the ability of clay loam soils to retain water for longer duration.

pH

Although denitrification can occur in both acidic and basic soil environments, the optimum pH for denitrification is considered to be 6 to 8 (Bremner and Shaw, 1958). Several studies have demonstrated that denitrification rates are highly correlated with soil pH and increase linearly with increase in pH from 4 to 8 (Struwe and Kjøller, 1994; Wijler and Delwiche, 1954). Bremner and Blackmer (1978) reported that the influence of NO_3^- on N_2O emissions is substantially enhanced by soil acidity. Firestone et al. (1980) likewise observed that acid soils produced greater N_2O emissions than those with neutral or alkaline pH. This is likely due to high sensitivity of the terminal reductase enzymes to low soil pH resulting in the accumulation of denitrification intermediates such as N_2O (Focht, 1974).

Moisture

Soil moisture is a direct and dominant regulator of denitrification in soils. The rate of nitrification and the activity of microbial population is often high in soils following a wetting event (Birch, 1958; Griffiths and Birch, 1961). In general, increases in soil moisture increase denitrification from soils. This can be inferred from several studies. Aulakh et al. (1991) demonstrated that WFPS is a better determinant of denitrification than overall soil water content. Denitrification follows a hysteresis response to soil wetting and drying cycles. Groffman and Tiedje (1988) showed that denitrification rates sharply increased following a wetting cycle and decreased during the drying process. Denitrification rates substantially increased as soil wets from 20-60% WFPS and further wetting of soils to field capacity and saturation did not enhance denitrification to a greater extent. Several authors have demonstrated that incipient denitrification appears to have a threshold value of 60% WFPS and at approximately 80% WFPS, most N gas flux occurs as N_2 (Aulakh et al., 1991; Davidson et al., 1991; Parton et al., 1988). Weier et al. (1993) found that ratios of N_2 to N_2O increased with increasing WFPS from 60-90% in silt loam soils and the conversion of N_2O to N_2 was more profound in treatments receiving greater carbon additions.

Temperature

Temperature both plays an important role in regulating the nitrification process and has a profound effect on N_2O emissions from the denitrification process. Goodroad and Keeney (1984) showed increased rates of nitrification and N_2O production with increasing temperatures from 10 to 30° C. They concluded that under isothermal constant moisture conditions, N_2O production was proportional to nitrate production and that 0.1-0.2% of the nitrified N was evolved as N_2O in dry, well-structured soils. Keeney et al. (1979) reported increased denitrification rates with temperature above 15°C with an optimum of 60°C. They reported that at 40°C, N_2O comprised up to 88% of the total (N_2O+N_2) gas evolved. However, in laboratory studies using a silt loam soil, Lensi and Chalamet (1982) observed that an optimum temperature for denitrification was between 37-45°C, which is a temperature range more typical of soil environments in arid and semi-arid climates, particularly Southern California, where salt-affected soils are commonly found; however, at these temperatures conditions are such that moisture is limiting. Furthermore, the simultaneous effect of soil salinity and temperature on N_2O emissions is still unknown.

Soil Salinity

Typically, microbial biomass and activity are concentrated in the top few centimeters of soil profile (Lavahun et al., 1996; Murphy et al., 1998). Hence, salinization near the soil surface is likely to affect a series of microbiologically mediated processes. Many researchers have reported on the interaction between salt accumulation and soil microbial activity. Some investigations on the effect of soil salinity on soil microbial properties have been shown to suppress soil microbial communities and their biochemical activities (Rietz and Haynes, 2003; Yuan et al., 2007). Nitrification rates have also been shown to decrease with increasing soil salinity due to reduced N mineralization (McClung and Frankenberger, 1985; Pathak and Rao, 1998).

Soil salinity negatively affects nitrification thereby reducing the accumulation of NO_3^-, so it may also influence denitrification by inhibiting microbial activity and associated enzyme activity. Ruiz-Romero et al. (2009) reported that under anaerobic conditions, N_2 production was larger compared to N_2O in a loamy sand soil with EC 12.5 dS m^{-1} than in a sandy clay loam soil with EC 56 dS m^{-1}. This suggests that under moderately saline conditions, more N_2O was reduced to N_2 due to active synthesis of N_2O reductase enzyme by denitrifying bacteria, thereby reducing the N_2O emissions. In case of the highly saline soils (EC 56 dS m^{-1}), production of N_2O was enhanced indicating a decreased N_2O reductase enzyme activity (Ruiz-Romero et al., 2009).

While decreased ammonification and nitrification rates are considered to be the primary effects of soil salinity on N dynamics, information on the effects of saline soils amended with organic materials on nitrification, denitrification rates, and associated N_2O emissions is scant. Because denitrifiers are hetereotrophic microorganisms that obtain energy from available C (Tiedje, 1988), addition of organic C is likely to promote the denitrification process.

Influence of Organic Amendments on N_2O Emissions

Organic amendments directly contribute to the release of N_2O from C and N compounds present in them. They may also produce N_2O indirectly through their effects on various physical, chemical, and biological properties of soils. Organic carbon compounds in the added amendments serve as a source of electron donors for energy and synthesis of cellular constituents for many soil microorganisms including nitrifiers and denitrifers. As most denitrifiers are heterotrophic bacteria, the process of denitrification is strongly correlated to the availability of soil carbon.

Nitrous oxide is produced in soils by nitrification and denitrification. Because these processes require NO_3^--N and organic C, applying organic materials to soil may result in increased N_2O production due to enhanced mineralization. Several studies have reported increased rates of denitrification with the application of organic matter to soils in the form of plant residues, green manures, and farm yard manures compared to unamended or mineral N treated soils (Anderson and Levine, 1986; Ding, 2007; Meng et al., 2005). This is because application of easily decomposable organic matter results in increased demand for nitrate as electron acceptor due to depletion of soil O_2 concentration during decomposition (Anderson and Levine, 1986). Therefore, in soils abundant in mineral N, increased amount of easily decomposable C is more conducive to denitrification.

The production of N_2O from soils treated with organic materials depends on their mineralization potential. Overall, the most important factor to consider while selecting an organic amendment is the C:N ratio because it influences rates of net mineralization, immobilization, and denitrification (Aulakh et al., 1991). Vigil and Kissel (1991) reported that applying residues with a low C:N ratio encouraged mineralization, but applying residues with a higher C:N ratio enhanced N immobilization. Several researchers documented reduced N_2O emissions following incorporation of organic materials with high C:N ratios due to N immobilization (Baggs et al., 2000; Hao et al., 2001; Velthof et al., 2002). McKenney et al. (1993) demonstrated that denitrification rates depend on the C:N ratio of the crop residue. They investigated the denitrification rates from hairy vetch (*Vicia villosa* Roth), red clover (*Trifolium pretense* L.), annual ryegrass (*Lolium temulentum*), and corn which varied in C:N ratio from 11 to 66. In a 5-day aerobic incubation study preceding the denitrifying conditions, they documented 5-17 fold increases in gaseous N losses from soils treated with residues at rates of 5-10 mg kg^{-1} compared to the unamended control with greatest N losses from hairy vetch and lowest from corn amended soils. Furthermore, application of organic materials with high C:N ratios in N limited soils resulted in decreased denitrification losses due to N immobilization (Huang et al., 2004; Yo and Hatano, 2007). Abundant research on the effects of organic matter additions on N_2O emissions have been conducted in non-saline soils. However, the interactive effects of salinity and organic amendments on N_2O emissions are still unknown.

Soil salinity is one of the many factors affecting C and N mineralization rates by directly influencing microbial activity. However, the effect of increased salinity on N mineralization has been a subject of controversy. Reitz and Haynes (2003) documented decreased N mineralization rates at higher soil salinity levels whereas Laura (1974) concluded that soil salinity even up to 5.1% salt addition did not inhibit ammonification. However, Pathak and Rao (1998) demonstrated that addition of *Sesbania Cannabina* to saline soils stimulated N mineralization up to salinity levels of 70 dS m^{-1} and thereafter N mineralization was negatively correlated with salinity. They also demonstrated that soil enzymes such as protease, amidases, and deaminases involved in N mineralization were still active at high salt concentrations. Furthermore, Khoi et al. (2006) found that soil salinity adversely affected N mineralization rates for a short duration and increased at a later stage likely due to the recovery and adaptation of soil microbes to osmotic stress. Since N_2O production is positively correlated to soil NO_3^--N content (Mosier et al., 1983), we may assume that soil salinity and N_2O emissions are also related.

While several studies have investigated the effects of incorporation of organic amendments on greenhouse gases and documented enhanced N_2O emissions from non-saline soils, it is unknown however, if denitrification rates increase in salt-affected soils following application of organic amendments. Additionally, the effect of soluble salts on microbial activity is influenced by soil moisture content and temperature. At low soil moisture levels, dissolved ions are concentrated; at high soil moisture levels, dissolved ions are diluted in the soil solution. This affects the availability of water and substrate to the microbial cells. Microorganisms such as bacteria and fungi adapt to increases in soil salinity either by accumulating solutes (*e.g.,* ammonium and potassium) necessary for their metabolism in their cytoplasm or by selectively excluding salt ions (*e.g.,* sodium and chloride) from their cells to maintain osmotic balance with their surrounding soil solution (Killham, 1994). Furthermore, addition of organic amendments influence soil properties such as water holding capacity, aeration, pH, and substrate availability that are key regulators of denitrification. Therefore, it is also important to understand and relate the impact of applying degradable organic matter such as manures and composts to salt-affected soils on properties that influence N_2O production.

As the United States seeks to develop more low-cost alternative methods to remediate salt-affected soils, vast acreages of these soils are expected to be reclaimed with organic amendments in lieu of inorganic amendments. In order to minimize the ecological impact of reclaiming these arid lands, it is critical to evaluate the contribution of various organic and inorganic amendments to the greenhouse gas emissions such as nitrous oxide and carbon dioxide.

This work examines the effect of incorporating organic amendments varying in their characterization on carbon dioxide, nitrous oxide, and dinitrogen gas emissions from salt-affected soils in Southern California. Chapter 2 quantifies the effects of soil salinity in governing greenhouse gas emissions by regulating the C and N dynamics of the added organic amendments. Specific objectives of chapter 2 are: 1) to evaluate the effects of soil salinity on N_2O and CO_2 emissions and N_2 to N_2O ratios following the addition of organic amendments; and 2) to investigate the impact of labile carbon availability from active and cured organic amendments on the conversion of N_2O to N_2 gas. Chapter 3 examines the impacts of varying temperatures on greenhouse gases, specifically addressing the objectives of: 1) examining the effect of extreme salinity and sodicity on CO_2, N_2O, and N_2 emissions; 2) investigating organic amendment effects on the N dynamics of a saline-sodic soil; and 3) determining how a varying soil temperature altered these processes. Chapters 2 and 3 are laboratory experiments. Lastly, chapter 4 focuses on evaluating N_2O and CO_2 emission rates from an extreme saline-sodic soil under field conditions and the role of active and cured materials in regulating these greenhouse gases.

The different approaches used to study the role of organic amendment amelioration of salt-affected soils allow us to evaluate the impact of active and cured forms of organic amendments on greenhouse gas emissions. These approaches at varying scales from laboratory experiments under controlled conditions to field-scale patterns help advance our understanding of the impact that these organic amendments have on mitigating N_2O emissions. This is essential to make informed decisions prior to further encouraging large scale use of organic amendments to remediate salt-affected soils.

References

Amezketa E., Aragüés R., Gazol R. (2005) Efficiency of sulfuric acid, mined gypsum, and two gypsum by-products in soil crusting prevention and sodic soil reclamation. Agronomy Journal 97:983-989.

Anderson I.C., Levine J.S. (1986) Relative Rates of Nitric Oxide and Nitrous Oxide Production by Nitrifiers, Denitrifiers, and Nitrate Respirers. Applied and Environmental Microbiology 51:938-945.

Aulakh M.S., Doran J.W., Walters D.T., Power J.F. (1991) Legume residue and soil water effects on denitrification in soils of different textures. Soil Biology and Biochemistry 23:1161-1167.

Baggs E., Rees R., Smith K., Vinten A. (2000) Nitrous oxide emission from soils after incorporating crop residues. Soil Use and Management 16:82-87.

Birch H. (1958) The effect of soil drying on humus decomposition and nitrogen availability. Plant and Soil 10:9-31.

Blackmer A., Bremner J., Schmidt E. (1980) Production of nitrous oxide by ammonia-oxidizing chemoautotrophic microorganisms in soil. Applied and Environmental Microbiology 40:1060.

Bouwman A. (1990) Exchange of greenhouse gases between terrestrial ecosystems and the atmosphere. p. 61-127, in: A. Bouwman (Ed.), Soils and the Greenhouse Effect, John Wiley & Sons, New York.

Bremner J., Shaw K. (1958) Denitrification in soil. II. Factors affecting denitrification. The Journal of Agricultural Science 51:40-52.

Bremner J., Blackmer A. (1978) Nitrous oxide: emission from soils during nitrification of fertilizer nitrogen. Science 199:295.

Bremner J., Blackmer A. (1979) Effects of acetylene and soil water content on emission of nitrous oxide from soils. Nature 280:380-381.

Chang C., Janzen H., Cho C. (1998) Nitrous oxide emission from long-term manured soils. Soil Science Society of America Journal 62:677-682.

Crutzen P. (1979) The role of NO and NO_2 in the chemistry of the troposphere and stratosphere. Annual Review of Earth and Planetary Sciences 7:443-472.

Crutzen P., Ehhalt D. (1977) Effects of nitrogen fertilizers and combustion on the stratospheric ozone layer. Ambio 6:112-117.

Davidson E.A., Swank W.T., Perry T.O. (1986) Distinguishing between nitrification and denitrification as sources of gaseous nitrogen production in soil. Applied and Environmental Microbiology 52:1280-1286.

Davidson E.A., Rogers J., Whitman W. (1991) Fluxes of nitrous oxide and nitric oxide from terrestrial ecosystems, in: J. E. Rogers and W. B. Whitman (Eds.), Microbial production and consumption of greenhouse gases: methane, nitrogen oxides, and halomethanes, American Society of Mircobiology, Washington D.C. pp. 219-235.

Ding W. (2007) Effects of long-term amendment of organic manure and nitrogen fertilizer on nitrous oxide emission in a sandy loam soil. Journal of Environmental Sciences (China) 19:185-193.

EPA. (2013) Inventory of U.S. Greenhouse gas emissions and sinks: 1990-2011. U.S. Environemntal Protection Agency, Washington, DC.

Firestone M.K., Firestone R.B., Tiedje J.M. (1980) Nitrous oxide from soil denitrification: factors controlling its biological production. Science 208:749-751.

Focht D. (1974) The effect of temperature, pH, and aeration on the production of nitrous oxide and gaseous nitrogen-a zero-order kinetic model. Soil Science 118:173.

Goodroad L., Keeney D. (1984) Nitrous oxide production in aerobic soils under varying pH, temperature and water content. Soil Biology and Biochemistry 16:39-43.

Griffiths E., Birch H. (1961) Microbiological changes in freshly moistened soil. Nature 178:500-501.

Groffman P.M., Tiedje J.M. (1988) Denitrification hysteresis during wetting and drying cycles in soil. Soil Science Society of America Journal 52:1626-1629.

Groffman P.M., Tiedje J.M. (1991) Relationships between denitrification, CO_2 production and air-filled porosity in soils of different texture and drainage. Soil Biology and Biochemistry 23:299-302.

Gupta S.C., Larson W.E. (1979) Estimating soil water retention characteristics from particle size distribution, organic matter percent, and bulk density. Water Resources Research 15:1633-1635.

Hahn J., Junge C. (1977) Atmospheric nitrous oxide- A critical review. Zeitschrift fuer Naturforschung, Teil a 32:190-214.

Hao X., Chang C., Carefoot J., Janzen H., Ellert B. (2001) Nitrous oxide emissions from an irrigated soil as affected by fertilizer and straw management. Nutrient Cycling in Agroecosystems 60:1-8.

Huang Y., Zou J., Zheng X., Wang Y., Xu X. (2004) Nitrous oxide emissions as influenced by amendment of plant residues with different C: N ratios. Soil Biology and Biochemistry 36:973-981.

IPCC. (2001) Intergovernmental Panel on Climate Change. Climate Change 2001: The Scientific Basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change, in: J. T. Houghton, et al. (Eds.), Cambridge University Press, Cambridge.

IPCC. (2005) Intergovernmental Panel on Climate Change. Special Report on Safeguarding the Ozone Layer and the Global Climate System: Issues Related to Hydrofluorocarbons and Perfluorocarbons, Special Report of the Intergovernmental Panel on Climate Change, Cambridge University Press, Cambridge.

Janssen B. (1996) Nitrogen mineralization in relation to C: N ratio and decomposability of organic materials. Plant and Soil 181:39-45.

Keeney D., Fillery I., Marx G. (1979) Effect of temperature on the gaseous nitrogen products of denitrification in a silt loam soil. Soil Science Society of America Journal 43:1124-1128.

Khoi C.M., Guong V.T., Merckx R. (2006) Predicting the release of mineral nitrogen from hypersaline pond sediments used for brine shrimp *Artemia franciscana* production in the Mekong Delta. Aquaculture 257:221-231.

Killham K. (1994) Soil Ecology Cambridge University Press, United kingdom.

Lakhdar A., Hafsi C., Rabhi M., Debez A., Montemurro F., Abdelly C., Jedidi N., Ouerghi Z. (2008) Application of municipal solid waste compost reduces the negative effects of saline water in *Hordeum maritimum* L. Bioresource Technology 99:7160-7167.

Lal R., Sobecki T.M., Ivari T., Kimble J.M. (2004) Soil Degradation in the United States: Extent, Severity, and Trends CRC Press, Boca Raton, Florida.

Lauchli A., Grattan S.R. (1990) Plant responses to saline and sodic conditions, Agricultural salinity assessment and management, ASCE. pp. 169-205.

Laura R. (1974) Effects of neutral salts on carbon and nitrogen mineralisation of organic matter in soil. Plant and Soil 41:113-127.

Lavahun M., Joergensen R., Meyer B. (1996) Activity and biomass of soil microorganisms at different depths. Biology and Fertility of Soils 23:38-42.

Lensi R., Chalamet A. (1982) Denitrification in waterlogged soils: *In situ* temperature-dependent variations. Soil Biology and Biochemistry 14:51-55.

Liang Y., Yang Y., Yang C., Shen Q., Zhou J., Yang L. (2003) Soil enzymatic activity and growth of rice and barley as influenced by organic manure in an anthropogenic soil. Geoderma 115:149-160.

Liang Y., Si J., Nikolic M., Peng Y., Chen W., Jiang Y. (2005) Organic manure stimulates biological activity and barley growth in soil subject to secondary salinization. Soil Biology and Biochemistry 37:1185-1195.

Maag M., Vinther F. (1996) Nitrous oxide emission by nitrification and denitrification in different soil types and at different soil moisture contents and temperatures. Applied Soil Ecology 4:5-14.

McClung G., Frankenberger W. (1985) Soil nitrogen transformations as affected by salinity. Soil Science 139:405.

McKenney D., Wang S., Drury C., Findlay W. (1993) Dentrification and mineralization in soil amended with legume, grass, and corn residues. Soil Science Society of America Journal 57:1013-1020.

Meng L., Ding W., Cai Z. (2005) Long-term application of organic manure and nitrogen fertilizer on N_2O emissions, soil quality and crop production in a sandy loam soil. Soil Biology and Biochemistry 37:2037-2045.

Mosier A., Parton W., Hutchinson G. (1983) Modelling nitrous oxide evolution from cropped and native soils. Ecological Bulletins:229-241.

Murphy D., Sparling G., Fillery I. (1998) Stratification of microbial biomass C and N and gross N mineralisation with soil depth in two contrasting Western Australian agricultural soils. Australian Journal of Soil Research 36:45-55.

Nelson M., Maredia M. (2001) Environmental impacts of the CGIAR: an assessment. Doc. No. SDR/TAC:IAR/01/11 presented to the Mid-term Meeting, 21-25 May, Durban, South Africa.

Oldeman L. (1994) The global extent of soil degradation, in: D. J. Greenland and I. Szabolos (Eds.), Soil resilience and sustainable land use, Wallingford: CABI. pp. 99-118.

Oster J., Shainberg I., Abrol I. (1996) Reclamation of salt-affected soil. Soil Erosion, Conservation and Rehabilitation, Agassi M (ed.). Marcel Dekker: New York:315-351.

Papendick R.I., Campbell G.S. (1981) Theory and Measurement of Water Potential1, in: J. F. Parr, et al. (Eds.), Water Potential Relations in Soil Microbiology, Soil Science Society of America. pp. 1-22.

Parton W., Mosier A., Schimel D. (1988) Rates and pathways of nitrous oxide production in a shortgrass steppe. Biogeochemistry 6:45-58.

Pathak H., Rao D. (1998) Carbon and nitrogen mineralization from added organic matter in saline and alkali soils. Soil Biology and Biochemistry 30:695-702.

Pitman M.G., Läuchli A. (2002) Global impact of salinity and agricultural ecosystems. Salinity: environment–plants–molecules. Dordrecht: Kluwer:3-20.

Qadir M., Noble A., Schubert S., Thomas R., Arslan A. (2006) Sodicity-induced land degradation and its sustainable management: Problems and prospects. Land Degradation and Development 17:661-676.

Ravishankara A.R., Daniel J.S., Portmann R.W. (2009) Nitrous oxide (N_2O): The dominant ozone-depleting substance emitted in the 21st century. Science 326:123-125.

Rengasamy P. (2006) World salinization with emphasis on Australia. Journal of Experimental Botany 57:1017-1023.

Richards L.A. (1954) Diagnosis and Improvement of Saline and Alkali Soils. Soil Science 78:154.

Rietz D., Haynes R. (2003) Effects of irrigation-induced salinity and sodicity on soil microbial activity. Soil Biology and Biochemistry 35:845-854.

Ruiz-Romero E., Alcántara-Hernández R., Cruz-Mondragon C., Marsch R., Luna-Guido M., Dendooven L. (2009) Denitrification in extreme alkaline saline soils of the former lake Texcoco. Plant and Soil 319:247-257.

Sexstone A.J., Parkin T.B., Tiedje J.M. (1985) Temporal response of soil denitrification rates to rainfall and irrigation. Soil Science Society of America Journal 49:99-103.

Skiba U., Ball B. (2002) The effect of soil texture and soil drainage on emissions of nitric oxide and nitrous oxide. Soil Use and Management 18:56-60.

Smith K., Thomson P., Clayton H., McTaggart I., Conen F. (1998) Effects of temperature, water content and nitrogen fertilisation on emissions of nitrous oxide by soils. Atmospheric Environment 32:3301-3309.

Struwe S., Kjøller A. (1994) Potential for N_2O production from beech (*Fagus silvaticus*) forest soils with varying pH. Soil Biology and Biochemistry 26:1003-1009.

Tejada M., Garcia C., Gonzalez J.L., Hernandez M.T. (2006) Use of organic amendment as a strategy for saline soil remediation: Influence on the physical, chemical and biological properties of soil. Soil Biology and Biochemistry 38:1413-1421.

Tiedje J.M. (1988) Ecology of denitrification and dissimilatory nitrate reduction to ammonium, in: A. J. B. Zehnder (Ed.), Biology of anaerobic microorganisms, John Wiley & Sons, New York. pp. 179-243.

UNEP. (2010) United Nations Environment Programme. The Montreal Protocol on substances that deplete the ozone layer. Available Online at: http://ozone.unep.org/new_site/en/montreal_protocol.php. Accesssed 1 November 2013, Ozone Secretariat. Nairobi, Kenya.

Velthof G.L., Kuikman P.J., Oenema O. (2002) Nitrous oxide emission from soils amended with crop residues. Nutrient Cycling in Agroecosystems 62:249-261.

Vigil M., Kissel D. (1991) Equations for estimating the amount of nitrogen mineralized from crop residues. Soil Science Society of America Journal 55:757-761.

Weier K.L., Doran J.W., Power J.F., Walters D.T. (1993) Denitrification and the dinitrogen/nitrous oxide ratio as affected by soil water, available carbon, and nitrate. Soil Science Society of America Journal 57:66-72.

Wichelns D. (1999) An economic model of waterlogging and salinization in arid regions. Ecological Economics 30:475-491.

Wijler J., Delwiche C. (1954) Investigations on the denitrifying process in soil. Plant and Soil 5:155-169.

WMO. (2007) World Meteorological Organization: Scientific Assessment of Ozone Depletion: 2006, Global Ozone Research and Monitoring Project.Report No. 50, 572 pp., Geneva, Switzerland.

Yo T., Hatano R. (2007) Effect of crop residue C: N ratio on N_2O emissions from Gray Lowland soil in Mikasa, Hokkaido, Japan. Soil Science and Plant Nutrition 53:198-205.

Yuan B.-C., Li Z.-Z., Liu H., Gao M., Zhang Y.-Y. (2007) Microbial biomass and activity in salt affected soils under arid conditions. Applied Soil Ecology 35:319-328.

Zheng X., Wang M., Wang Y., Shen R., Gou J., Li J., Jin J., Li L. (2000) Impacts of soil moisture on nitrous oxide emission from croplands: a case study on the rice-based agro-ecosystem in Southeast China. Chemosphere-Global Change Science 2:207-224.

2. EFFECTS OF SOIL SALINITY AND CARBON AVAILABILITY FROM ORGANIC AMENDMENTS ON NITROUS OXIDE EMISSIONS

Abstract

Soil salinity negatively affects the mineralization and nitrification processes of the N cycle and may also affect the production of nitrous oxide (N_2O) and N_2 to N_2O-N ratios. Application of organic amendments such as manures and composts improve soil physical, chemical, and biological properties of salt-affected soils. However, because these materials both mineralize N and serve as substrates for denitrifiers, they may also increase N_2O emissions. We studied the effect of organic amendments applied to saline soils on N_2O emissions and on N_2 to N_2O-N ratios. Saline soils with EC_e measures of 2.8, 15.2, and 30.6 dS m^{-1} were collected from Coachella Valley, California. Treatments included four organic amendments: active greenwaste compost (AGW), cured greenwaste compost (CGW), active dairy manure compost (ADM), and cured dairy manure compost (CDM). Treatments were incorporated at 50 Mg ha^{-1} and incubated at 65% water-filled pore space (WFPS) for 60 days at 25°C. Evolving CO_2-C and N_2O-N were monitored along with soil ammonium (NH_4^+-N) and nitrate (NO_3^--N) concentrations.

The results showed that increasing soil salinity increased cumulative N_2O-N losses but decreased CO_2-C and N_2 emissions and N_2 to N_2O-N ratios. Of all the amendments, the highest cumulative N_2O-N and N_2 emissions were produced from the ADM treatment at all three salinity levels. In general, incorporation of active compared to cured amendments increased N_2 to N_2O-N ratios at all three salinity levels suggesting that applying active organic materials could be useful in mitigation of N_2O emissions from salt-affected soils under remediation.

Introduction

Global warming and soil salinization are major international concerns. With an ever growing population, there is an increasing need to bring more agricultural land under production to feed people. Unfortunately development, erosion, and other pressures are reducing the earth's arable land. The problem is particularly acute in drylands where overgrazing and deforestation have led to salinization, associated problems of soil fertility, crusting, compaction, and widespread desertification (Dregne, 1983; Singh, 2009). It is estimated that more than 800 Mha of land is salt-affected globally (Martinez-Beltran and Manzur, 2005). Re-desertification of reclaimed lands can also occur if poor quality irrigation water is applied (Banin and Fish, 1996). Reclamation involves leaching after amending soils with organic materials such as manures and composts, or inorganic materials such as gypsum and sulfur. This removes salts and improves soil physical, chemical, and biological properties (Tejada et al., 2006). The extent of salinization and the rate of increase in salt-affected lands make reclamation and maintenance of these soils an international priority.

Soil amendment use also affects the emission of nitrous oxide (N_2O), a potent gas that contributes to both global warming and ozone depletion. Nitrous oxide has been singled out as the most significant anthropogenic ozone depleting compound (Ravishankara et al., 2009). With an estimated 120-year global warming potential 310 times higher than CO_2, N_2O accounts for approximately four percent of United States greenhouse gas emissions (EPA, 2013). This high potency makes even relatively small changes in N_2O emissions potentially significant.

The production of N_2O from agricultural soils is mostly due to the anaerobic reduction of nitrate by denitrifying bacteria. Denitrification, which contributes both N_2 and N_2O, is promoted by anaerobic conditions and the presence of organic substrates for the denitrifiers. Significant N_2O emissions are most likely to occur under reduced oxygen conditions where soil nitrate concentrations are greater than 5 mg NO_3^--N kg^{-1} dry soil, and soil temperatures exceed 5°C (Bremner and Blackmer, 1979; Bremner and Shaw, 1958). The rate of N_2O emission from soils is normally a function of soil moisture, temperature, substrate supply, and structure; but under saline conditions the influence of salts on processes such as nitrification and denitrification should also be considered (Bremner and Shaw, 1958). It is also important to understand and relate the impact of applying N enriched degradable organic matter, such as manures and composts, on soil properties that influence N_2O production including the availability of C sources as substrates for denitrifying microorganisms (Patten et al., 1980).

Many researchers have reported on the interaction between salt accumulation and soil microbial activity. Investigations have shown that saline conditions broadly suppress soil microbial communities and their biochemical activities (Rietz and Haynes, 2003; Yuan et al., 2007). Decreases in mineralization of organic C and N with increasing salt concentrations are well documented (Laura, 1974; McClung and Frankenberger, 1985; Pathak and Rao, 1998) but specific information on the effects of saline soils amended with organic materials on nitrification, denitrification, and associated N_2O emissions is lacking. Typically, organic matter and microbial activity are concentrated in the top few centimeters of soil profile (Lavahun et al., 1996; Murphy et al., 1998). Hence, salinization near the soil surface is likely to affect many microbiologically mediated processes including the microbial production of N_2O in soils.

Given the interest in the use of organic amendments for remediating salt-affected soils, their effects on greenhouse gas emissions particularly N_2O, merits study. The objectives of this study were therefore:

1) to evaluate the effects of soil salinity on N_2O and CO_2 emissions and N_2 to N_2O-N ratios following the addition of organic amendments; and
2) to investigate the impact of labile carbon availability from active and cured organic amendments on the conversion of N_2O to N_2 gas.

Materials and Methods

Site Description and Experimental Treatments

Soils samples from three different salt-affected areas were collected from an abandoned field site in California's Coachella Valley (33° $35^{'}$ $30.5^{''}$ N, 116° $06^{'}$ $20.7^{''}$ W). The soils were characterized as fine-silty, mixed, superactive, calcareous, hyperthermic Aquic Torriorthents. The site had been fallowed for at least 5 years due to salinity issues. Samples from three areas with different EC_e properties (2.8, 15.2, and 30.6 dS m^{-1}) were collected from 0-15 cm depth, air-dried in the greenhouse, and passed through a 2 mm sieve. Particle size analysis using hydrometer method (Gee and Bauder, 1986) for the three soils indicated that the soils were clay loam in texture. Physical and chemical characteristics of the three soils used in this study are given in Table 2.1.

Four organic amendments were considered as treatments in this study: active greenwaste (AGW), cured greenwaste compost (CGW), active dairy manure (ADM), and cured dairy manure compost (CDM). The greenwaste amendments were collected from a commercial greenwaste composting facility located in California. The AGW was comprised of plant material from local roadside cuttings and was collected from an actively managed two-week old turned windrow. The CGW was collected from a screened cured compost storage pile that had also undergone 10 weeks of active windrow composting. Treatments ADM and CDM were collected from actively managed two-week old turned windrow and 12-week old on-farm compost storage piles, respectively, at a dairy farm in California. Samples of all treatments were air-dried in a greenhouse at 20º C until moisture contents stabilized. The air-dried samples were then milled using a Wiley mill to pass through a 4 mm sieve. Chemical characteristics of treatments are presented in Table 2.2.

Incubation Experiment

The three soils used in this study were pre-incubated at 40% WFPS in the dark at 24±1ºC for two weeks prior to adding the organic amendments. The mass of water required to achieve the desired WFPS was calculated using the following equation (Carter, 1993):

$$\theta_g \ at \ \% \ WFPS = \frac{WFPS \times \left(1 - \frac{\rho}{2.65}\right)}{\rho} \quad \text{Equation 1}$$

where, WFPS is the fraction of total pore space filled with water (%), θ_g at % WFPS is the gravimetric soil water content at respective WFPS (g water g^{-1} soil), ρ is the bulk density of the soil (g cm^{-3}), and 2.65 is the soil particle density (g cm^{-3}). During the course of incubation, containers were weighed periodically and water was added on a gravimetric basis to maintain the desired WFPS.

Separate experiments were setup to: 1) determine soil mineral N; and 2) monitor CO_2 and N_2O emissions from saline soils treated with organic amendments. Nitrogen mineralization was studied in 1 L glass jars containing 250 g of air-dried soil while CO_2 and N_2O emissions were studied within 100 mL glass serum bottles containing 25 g of air-dried soil. All organic amendments (AGW, CGW, ADM, and CDM) were incorporated into the soil at an application rate of 50 Mg ha^{-1} while control treatments received no organic amendment. The soil and amendments were mixed thoroughly and packed to achieve a bulk density of 1.25 g cm^{-3}, a value similar to observed field conditions (Table 2.1). The soil water was maintained at 65% WFPS as this has been shown to facilitate N mineralization and also result in denitrification losses (Maag and Vinther, 1996). The 1 L jars were closed using lids with a 1 cm diameter hole in the center to facilitate aerobic conditions while limiting excessive soil moisture loss. Three replicates of each of the five treatments and three soil types were prepared for the 1 L jars while six replicates of the same were prepared for 100 mL serum bottles. Incubations were sustained for 60 days in the dark at 24±1ºC to best represent mean daily temperatures in the arid and semi-arid regions such as California.

Soil and Gas Sampling Procedures

Soil and gas samplings were conducted on Days 0, 4, 7, 11, 15, 20, 25, 32, 37, 45, 53, and 60 days from the commencement of the incubations. Inorganic N (NH_4^+-N and NO_3^--N) was determined by colorimetry on extractions obtained with 2M KCl (ratio 1:5).

For the determination of CO_2 and N_2O emissions, 100 mL incubation serum bottles were aerated for 30 s to remove any traces of CO_2 and N_2O built-up previously and also to facilitate proper oxygenation of the environment. The bottles were then sealed with rubber septa and aluminum crimp seals for a period of 4 hrs. Of the six replicates prepared for each treatment, three replicates were injected with 10% v/v acetylene (C_2H_2) while the other three replicates did not receive C_2H_2. After 4 hrs, the headspace air was mixed twice using a 30 mL gas syringe to remove any internal air stratification. A 30 mL gas sample was collected and injected into pre-evacuated 20 mL headspace vials and then analyzed within 72 hrs of sampling for CO_2 and N_2O.

Carbon dioxide was measured using an infrared gas analyzer (Model EGM-4, PP-Systems, Amesbury, Massachusetts) while N_2O was measured using a HP 5790 A Gas Chromatograph (Hewlett- Packard Co., Fullerton, California) fitted with an electron capture detector with oven and injection temperatures set at 50 ºC and a detector temperature of 300 ºC. Standards were injected after every 10 samples to assure instrument precision.

Statistical Analysis

Cumulative emissions for CO_2-C, N_2O-N, and N_2 were calculated by linear interpolation of daily measurements across consecutive sampling days. Emission rates for each time interval were assumed equal to the average of the emissions measured at the beginning and end of that interval. Two-way analysis of variance (ANOVA) was conducted to determine significant differences ($p < 0.05$) for cumulative CO_2-C, N_2O-N, and N_2 emissions, soil NO_3^--N, and NH_4^+-N concentrations between different soils and treatments. The effect of soil salinity and organic amendments on cumulative emissions for CO_2-C, N_2O-N, and N_2, N_2 to N_2O-N ratios, and soil mineral N over time was tested by performing repeated measures ANOVA. Tukey's multiple comparison tests were used to statistically separate the significant differences between various soil salinity levels and organic amendment treatments. Linear and multiple regression analyses were used to examine the relationships between cumulative CO_2-C, N_2O-N, and N_2 emissions and N_2 to N_2O-N ratios. All statistical analyses were conducted using SPSS 20.0 software.

Results

ANOVA results for the effects of soil salinity and organic amendment application on cumulative CO_2-C, N_2O-N, and N_2, N_2 to N_2O-N ratios, and soil NO_3^--N concentrations are reported in Table 2.3.

Emission of CO_2

Effect of Salinity

Across the incubation period, soil respiration decreased significantly ($P < 0.05$) with increase in soil salinity in all treatments (Fig. 2.1). Elevated daily emissions of CO_2 occurred between weeks 3-8 for all treatments and gradually decreased over the incubation period. Controls had the lowest daily and cumulative CO_2 emissions over the 60 day period which ranged from 3.4-19.0 mg C kg^{-1}. Cumulative CO_2 emissions ranged from 19.0-101.1 mg C kg^{-1} for S3, 10.3-70.7 mg C kg^{-1} for S15, and 3.4-51.0 mg C kg^{-1} for S30 soils (Fig. 2.1). For all organic amendments, increased soil salinity resulted in decreased CO_2 emissions by 28-38% from S3 to S30.

Effect of Organic Amendments

Addition of organic amendments significantly ($P < 0.01$) increased the daily CO_2 emissions regardless of whether the soil was from a high or low salinity level. Over the incubation period, AGW and ADM resulted in consistently higher CO_2 emissions compared to CGW and CDM. Cumulative CO_2 emissions across the three soil salinity levels from AGW were 1.3-1.6 times higher than from CGW while ADM losses were approximately twice that of CDM. Compared to the composted amendments (CGW and CDM), uncomposted amendments (AGW and ADM) significantly ($P < 0.05$) increased CO_2 production (Fig. 2.1). Applying AGW resulted in 25, 30, and 39% increase in CO_2 emissions relative to CGW amendment in S3, S15, and S30 salinity levels, respectively. Similarly, increase in CO_2 production of 49, 53, and 54% was recorded with the application of ADM compared to CDM amendment in S3, S15, and S30 salinity levels, respectively (Fig. 2.1).

Emission of N_2O and N_2

Effect of Salinity

Increases in soil salinity significantly ($P < 0.05$) increased cumulative N_2O-N but decreased N_2 emissions ($P < 0.01$) from all treatments over the incubation period. The highest peak daily loss of N_2O was observed in S3 soils followed by S15 and S30 during weeks 3-5 of the incubation period. For the controls across the three salinity levels, cumulative N_2O emissions ranged from 2.6-4.5 µg N kg^{-1}. Regardless of the treatment, compared to S3 soils, an increase in cumulative N_2O-N production by 18-25 % for S15 soils and 34-87% for S30 soils was observed after the 60 day incubation period (Fig. 2.2).

Increases in soil salinity decreased cumulative N_2 production from all treatments. Peak daily N_2 emissions were observed in weeks 3-5 of the incubation period. Cumulative N_2 emissions for the controls ranged from 4.9-14.3 µg N kg^{-1}. Compared to

S3 soils, percent reduction in cumulative N_2 production ranged from 14-54% for S15 soils and 44-62% for S30 soils (Fig. 2.3). Regardless of the treatment, the ratios of N_2 to N_2O-N were significantly ($P < 0.01$) decreased in S15 and S30 relative to S3 soils. The ratios of N_2 to N_2O-N ranged from 2.2-9.5 for S3, 0.8-5.2 for S15, and 0.6-3.9 for S30 soils (Fig. 2.4).

Effect of Organic Amendments

Regardless of the EC_e of the soil, application of organic amendments significantly ($P < 0.05$) increased N_2O emissions. In all three soils, higher N_2O production was observed from the dairy manure compared to the greenwaste amendments. In case of greenwaste as organic amendment, incorporation of cured material (CGW) ultimately produced higher N_2O emissions compared to the active amendment (AGW). In contrast, for dairy manure as organic amendment, active material (ADM) resulted in higher N_2O emissions relative to the cured material (CDM). Cumulative N_2O-N emissions using CGW compared to AGW amended soils increased by 25, 22, and 46% for S3, S15, and S30 soils, respectively. Soils amended with ADM, however, increased cumulative N_2O emissions relative to CDM by 12, 15, and 21% for S3, S15, and S30 soils, respectively (Fig. 2.2).

At all three soil salinity levels, incorporation of organic amendments resulted in a significant ($P < 0.01$) increase in N_2 production. Incorporation of active materials, AGW and ADM resulted in significantly ($P < 0.01$) greater N_2 production compared to the cured materials, CGW and CDM. Dinitrogen (N_2) gas emissions from active organic amendments ranged from 207.4-552.7 µg N kg^{-1} while those from cured organic amendments ranged from 77.0-222.9 µg N kg^{-1} across the three salinity levels. Application of CGW compared to AGW reduced N_2 emissions by approximately 51% while incorporation of CDM relative to ADM decreased N_2 production by 70, 84, and 62% for S3, S15, and S30 soils, respectively (Fig. 2.3).

Application of organic amendments significantly ($P < 0.05$) decreased the ratios of N_2 to N_2O-N with increasing salinity. Though ratios for CGW exceeded AGW initially in S3 soils, ultimately the active materials, AGW and ADM increased the N_2 to N_2O-N ratios as high as 2.9 and 5.4 times compared to their respective cured materials, CGW and CDM. Ratios of N_2 to N_2O-N for active materials ranged from 1.3-9.5 while those for cured materials ranged from 0.6-3.7 (Fig. 2.4).

Inorganic N

Effect of Salinity

At the commencement of the incubation (Day 0), all three soils had similar amounts of NO_3^--N and NH_4^+-N. After 60 days of incubation, the amount of soil NO_3^--N (Fig 2.5) and mineral N significantly decreased ($P < 0.01$) with increases in soil salinity. No trend was observed for NH_4^+-N alone (data not shown). Across the incubation period, for the controls, the NO_3^- concentrations gradually decreased in S15 and S30 but increased in S3 soils. The mineral N content at the end of the incubation did not significantly differ among the three control soils varying in EC_e.

Effect of Organic Amendments

In all three soils, NO_3^--N, NH_4^+-N, and mineral N concentrations after the 60 day incubation period differed significantly ($P < 0.01$) between the organic amendment treatments. Regardless of soil EC_e, AGW and CGW showed marked decrease in NO_3^--N and mineral N concentrations during the first 28 days while the decrease with ADM and CDM was recorded only during the first 10 days (Fig. 2.5). For all three soils, addition of AGW and CGW significantly decreased the soil NO_3^--N (Fig 2.5) and mineral N (data not shown) content compared with ADM and CDM. Greater NO_3^- -N concentrations were observed for the ADM and CDM treatments compared to the AGW and CGW amendments. At Day 60, compared to S3 soils, NO_3^- -N and mineral N concentrations for S15 and S30 soils amended with greenwaste decreased by 16-33% and reduction with dairy manure by 4-17% were recorded (Fig. 2.5).

Relationship between CO_2, N_2O, N_2, and Soil Inorganic N Concentrations

Soil salinity significantly ($P < 0.05$) and negatively correlated ($r^2 > 0.56$) with soil NO_3^--N concentrations. Cumulative N_2O-N and N_2 emissions and total denitrification losses (N_2O+N_2) were significantly and positively correlated ($0.45 < r^2 \leq 0.88$; $P < 0.05$) with cumulative CO_2-C emissions and soil NO_3^--N concentrations ($0.55 < r^2 \leq 0.81$; $P < 0.01$) at S3, S15, and S30 salinity levels. However, weaker and negative correlations between N_2 to N_2O-N ratios and cumulative CO_2-C production ($0.28< r^2 \leq 0.46$) and soil NO_3^--N concentrations ($0.40 < r^2 \leq 0.68$) were observed at S3, S15, and S30 salinity levels (Table 2.4).

Discussion

Production of CO_2

Soil salinity significantly and negatively affected cumulative CO_2-C emissions suggesting an increased osmotic stress on the soil microbial communities causing a reduction in their activity (Fig. 2.1). Several authors have reported reduced soil respiration rates with increasing soil EC_e (Laura, 1974; McCormick and Wolf, 1980; Pathak and Rao, 1998).

Curves representing cumulative CO_2-C emissions with time at all three salinity levels indicate that soil amendments AGW and ADM had higher cumulative CO_2-C evolution than CGW and CDM (Fig. 2.1). The C substrates in AGW and ADM amendments were decomposed more rapidly than in CGW and CDM which had stabilized during curing. This trend for increased CO_2-C emissions from uncured materials was similar to that reported by Tejada et al. (2006). Cook and Allan (1992) also reported that the labile organic matter fraction is the most degradable and therefore, highly susceptible to mineralization. This may also indicate the presence of greater labile C fraction in AGW and ADM than in CGW and CDM.

Also, our results at each salinity level using regression analysis showed that cumulative CO_2-C emissions significantly and positively correlated with soil NO_3^--N concentrations, cumulative N_2O-N, and N_2 emissions. Stepwise regression used to identify the factors affecting CO_2-C emissions in our study indicated that soil NO_3^--N, cumulative N_2O, and N_2 emissions together accounted for nearly 75% of variability. Microorganisms active in denitrification and the N mineralization rely on organic substrates for energy and generate CO_2 as waste product. The remaining variability is contributed by the degree to which heterotrophs release N at different rates depending on their own needs and substrate characteristics and possibly to the chemotrophic nature of the nitrification process (Robertson and Groffman, 2007).

Inorganic N

Excessive amounts of salts are known to affect microbiological processes such as C and N mineralization which are crucial for the decomposition of organic matter. It has been well established that processes such as N mineralization and nitrification decrease with increasing soil salinity (Laura, 1977; McClung and Frankenberger, 1985). In addition, studies have shown that when soils are amended with high C:N organic materials, soil microbes assimilate the available N into their biomass and utilize it to decompose the added materials (Parton et al., 2007; Taylor et al., 1989). For amendments such as AGW and CGW which are relatively low in N, N was immobilized immediately after the amendments were applied thereby reducing the concentration of NO_3^--N, a substrate for N_2O production during denitrification. In the high N treatments (ADM and CDM) however, more NO_3^--N was available after Day 10 and therefore soils amended with these organic materials resulted in greater N_2O emissions (Fig. 2.5). A strong correlation was observed between mineral N and N_2O and N_2 emissions at 65%

WFPS which was also consistent with a study by Bremner and Blackmer (1981) who found positive correlation between mineral N and N_2O emissions.

Production of N_2O and N_2

Nitrous oxide is produced during N transformations in the soil through two important biological processes: nitrification in the presence of aerobic conditions and denitrification under anaerobic conditions. In this study, aerobic conditions were maintained in the headspace and soil moisture content was adjusted to 65% WFPS to drive the N transformation towards nitrification process (Maag and Vinther, 1996). Although aerobic conditions were maintained throughout our study, denitrification losses can still occur due to the presence of anaerobic microsites in soil aggregates (Müller et al., 2004).

There was also a decrease in N_2 to N_2O-N ratios with increasing soil EC_e. The NO_3^--N in soil is the electron-acceptor for denitrifiers responsible for N_2O and N_2 emissions and elevated N_2O and N_2 emissions are generally associated with greater soil NO_3^--N concentrations. In this incubation study however, we observed higher cumulative N_2O-N emissions from S30 soils which had lower soil NO_3^--N concentrations compared to S3 and S15 soils. In contrast, greater cumulative N_2 emissions were noticed in S3 soils followed by S15 and S30 soils suggesting that the reduction of N_2O to N_2 was decreased with increasing in soil salinity (Fig 2.3). Nearly all N_2O produced was reduced to N_2 in S3 soils. It is likely that osmotic stress at the higher salinity levels may have slowed the denitrifiers responsible for reducing N_2O to N_2 or alternatively salts may have interfered with N_2O reductase activity or its de novo synthesis. A trend of increased cumulative N_2O-N and decreased N_2 emissions with increasing soil salinity was similar to that observed by Ruiz-Romero et al. (2009) who found elevated N_2O emissions in soils with EC_e 56 dS m^{-1} compared to 12 ds m^{-1}.

Substrate availability also affects N cycling. The ratios of N_2 to N_2O-N were higher for AGW and ADM compared to CGW and CDM amendments (Fig. 2.4). Positive correlations for cumulative CO_2-C emissions and N_2 to N_2O-N ratios at all three salinity levels suggest that substrate availability limited the reduction of N_2O to N_2, particularly in the cured materials. The higher N_2 to N_2O-N ratios for active amendments is most likely due to increased support of the microbial biomass due to the bioavailability C from these treatments compared to the cured amendments resulting in enhanced reduction of N_2O to N_2 via denitrification (Firestone and Davidson, 1989; Huang et al., 2004).

Denitrification is an anaerobic process, but Müller et al. (2004) found that even under aerobic conditions NO_3^- reduction is responsible for N_2O production in a grassland soil. Their study demonstrated that with 15-21% O_2 concentration in the entire soil profile, denitrification was still responsible for N_2O emissions. Observations from our incubation study are also consistent with those reported by McKenney et al. (1993) who showed that addition of organic residues resulted in higher denitrification rates in aerobic than in anaerobic conditions. Production of CO_2 may be a factor in the development of anaerobic microsites thereby encouraging denitrification (Huang et al., 2004). Active organic amendments such as AGW and ADM resulted in greater cumulative CO_2-C emissions and total denitrification losses compared to their respective cured amendments, CGW and CDM. Measurements using acetylene inhibition technique which blocks the further reduction of N_2O to N_2 via denitrification also confirms the contribution of denitrification as the primary process responsible for N_2O and N_2 emissions in our study. Although conditions in our incubation study were controlled to favor nitrification, anaerobic microsites likely were present due to enhanced oxygen consumption following the addition of organic amendments and or saturated soil aggregates (Cannavo et al., 2004; Leffelaar, 1986) and should be further investigated.

Organic amendments used for remediating salt-affected soils are often applied repeatedly to the same field. Continual applications could eventually result in more N being mineralized and released into soils than with single application. We did not consider the effect of multiple applications of organic amendments in this study. In addition, N release rates and C decomposition from active and composted organic materials widely vary over time. Therefore, long-term experiments with multiple organic amendment applications would help to evaluate N_2O and N_2 emissions from active and composted greenwaste and dairy manure materials.

Conclusions

Incorporation of organic amendments derived from greenwaste and dairy manure to salt-affected soils significantly affected soil CO_2, N_2O, and N_2 emissions. Cumulative CO_2-C and N_2 emissions decreased with increasing soil salinity. In contrast, enhanced cumulative N_2O-N emissions were recorded as the EC_e of soil increased. The production of N_2 was larger than N_2O-N from active amendments (AGW and ADM) compared to cured amendments (CGW and CDM) resulting in high N_2 to N_2O-N ratios even in soil with EC_e of 30.6 dS m^{-1}. The main factor related to enhanced N_2 emissions was greater cumulative CO_2-C emissions from active organic amendments which promoted the completion of the denitrification process. Therefore, use of active organic amendments to remediate salt-affected soils can prove to be beneficial in mitigating N_2O emissions. However, further field research must be conducted to evaluate management practices such as application rates to determine long-term effects of these added amendments on trace gas emissions. Although uncured amendments were shown to reduce N_2O emissions by promoting the production of N_2, once in the soil these materials will eventually stabilize and their performance may converge to follow the N cycling patterns observed for the cured materials.

Table 2.1: Soil physical and chemical properties of top 0-15 cm soil layer.

Paramter	Non-Saline (S3)	Saline (S15)	Highly Saline (S30)
Sand (%)	25.8	29.4	26.3
Silt (%)	40.0	32.5	37.2
Clay (%)	34.2	38.1	36.5
Bulk Density (g cm^{-3})	1.22	1.25	1.24
EC_e (dS m^{-1})	2.8	15.2	30.6
pH	7.86	7.32	7.91
Total C (%)	1.18	1.38	0.90
Inorganic C (%)	0.72	0.83	0.48
Organic C (%)	0.46	0.55	0.42
Total N (%)	0.12	0.13	0.08
C:N	10.19	10.49	15.48
NO_3^--N (mg kg^{-1})	180.18	175.15	177.39
NH_4^+-N (mg kg^{-1})	2.80	2.47	2.18
Inorganic N (mg kg^{-1})	182.98	177.62	179.57

Table 2.2: Organic amendment properties.

Amendment	Total C (%)	Total N (%)	C:N	NO_3^--N (mg kg^{-1})	NH_4^+-N (mg kg^{-1})
Active Greenwaste (AGW)	33.6	0.76	44.2	3.49	19.1
Cured Greenwaste Compost (CGW)	25.7	0.84	30.6	4.42	53.06
Active Dairy Manure (ADM)	30.9	2.37	13.0	8.46	173.06
Cured Dairy Manure Compost (CDM)	13.1	1.15	11.4	31.93	82.72

Table 2.3: Analysis of variance for different parameters as affected by soil salinity and organic amendments.

Source	Cumulative CO_2-C	Cumulative N_2O-N	Cumulative N_2	N_2:N_2O-N	NO_3^--N	NH_4^+-N
Soil Salinity (S)	*	*	**	**	**	ns
Organic Amendment (A)	**	*	*	*	**	**
Interaction (S×A)	ns	ns	ns	ns	ns	ns

ns, *, and ** represent no significance, significance at $P < 0.05$ and $P < 0.01$,respectively.

Table 2.4: Stepwise regression analysis of cumulative CO_2, N_2O, and N_2 emissions with soil properties.

Treatment	Regression Equation	Combined r^2 (adj)
Cumulative CO_2	$1.262 - 0.307\ EC_e + 0.043\ NO_3^- + 0.099\ N_2 + 0.050\ N_2O$	0.613
Cumulative N_2O	$-71.773 + 2.058\ EC_e + 0.349\ NO_3^- + 0.093\ N_2 + 0.159\ CO_2$	0.653
Cumulative N_2	$106.364 - 2.341\ EC_e - 0.356\ NO_3^- + 3.920\ CO_2 + 1.155\ N_2O$	0.606
Ratio of N_2 to N_2O-N	$18.018 - 0.220\ EC_e - 0.056\ NO_3^- + 0.015\ N_2 - 0.040\ CO_2$	0.418

Note: Units for selected variables are reported as: soil electrical conductivity (dS m^{-1}); NO_3^- concentrations (mg N kg^{-1}), CO_2 emissions (mg C kg^{-1}), and N_2O and N_2 emissions (µg N kg^{-1}).

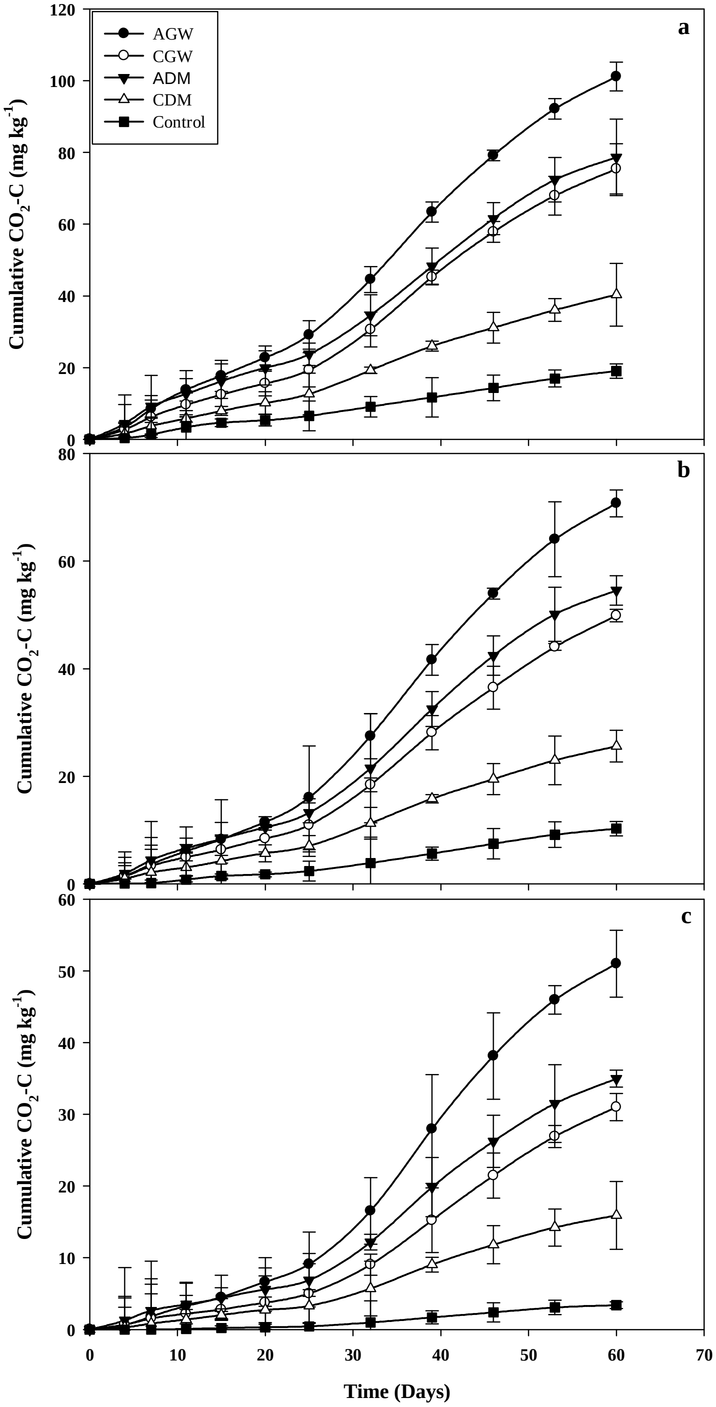

Fig. 2.1. Cumulative CO_2-C emissions for different treatments (mean ± std. error) from (a) S3 soils, (b) S15 soils, and (c) S30 soils.

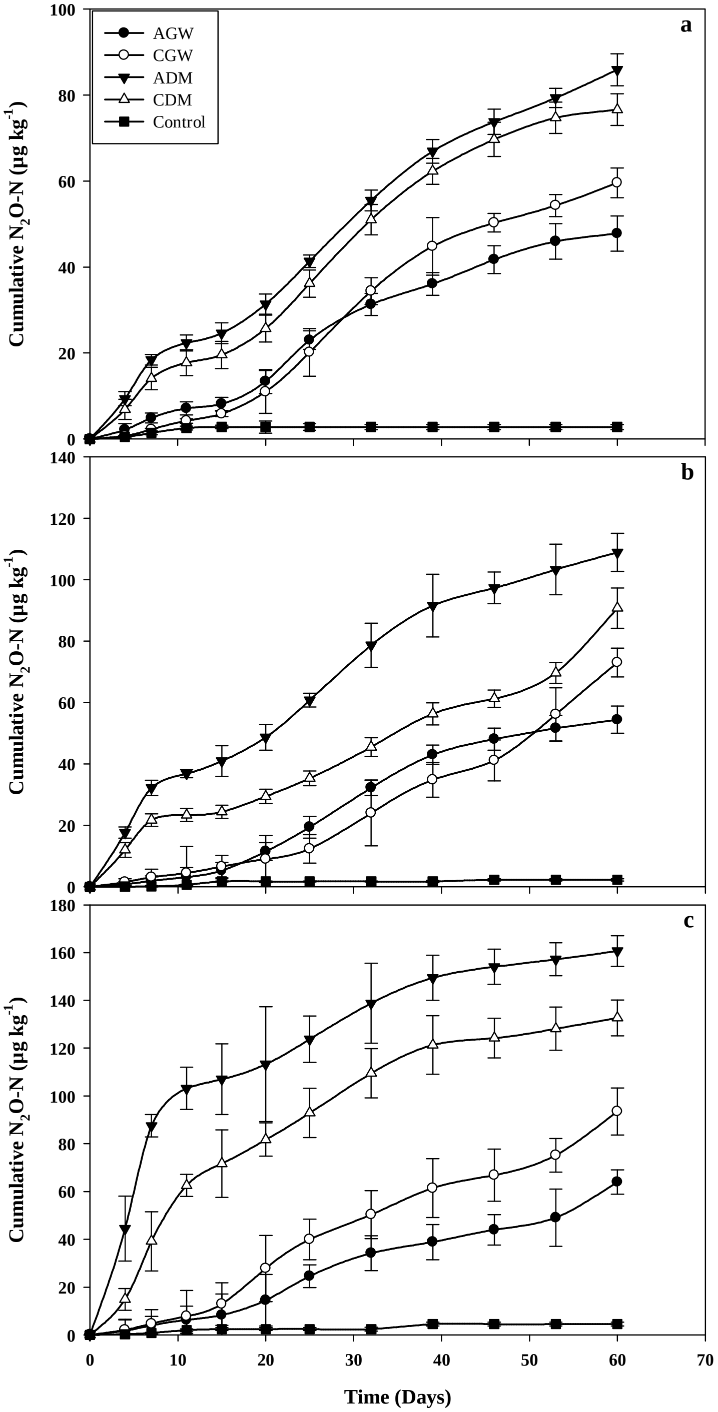

Fig. 2.2. Cumulative N_2O-N emissions for different treatments (mean ± std. error) from (a) S3 soils, (b) S15 soils, and (c) S30 soils.

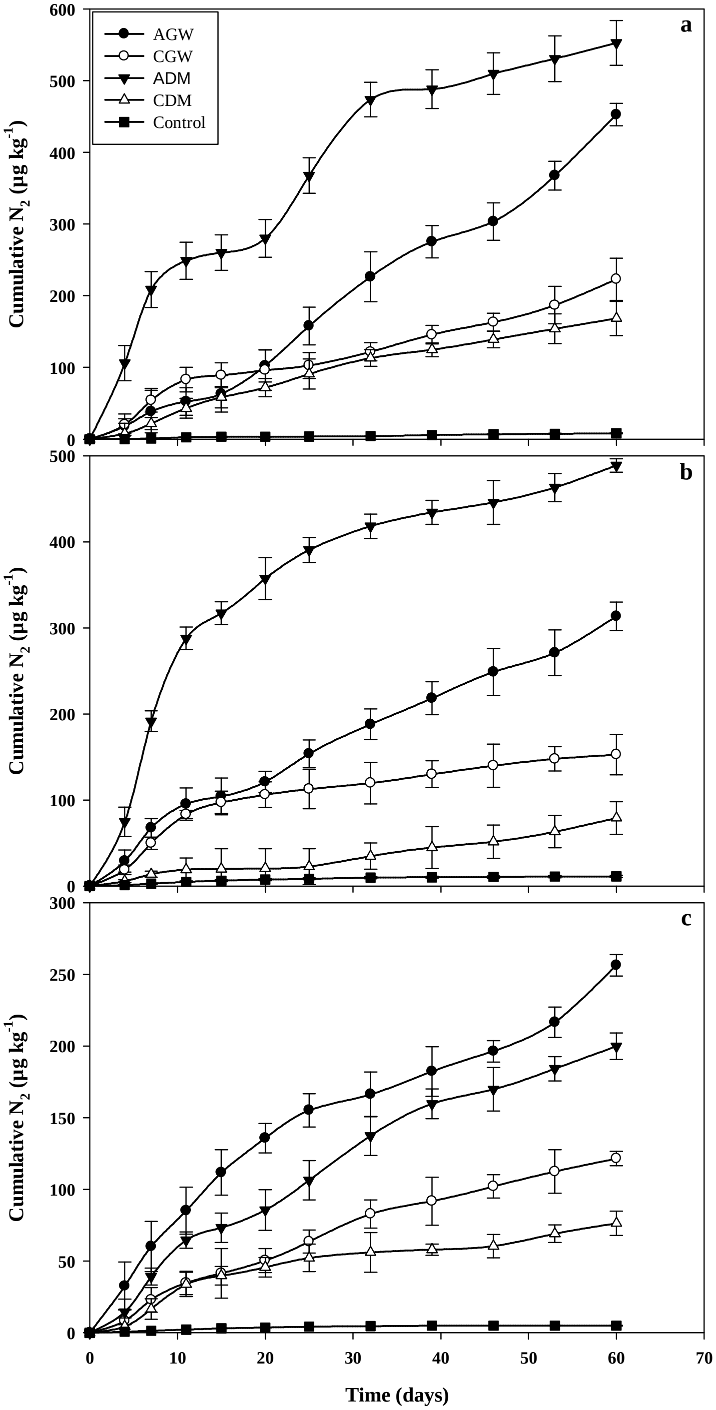

Fig. 2.3. Cumulative N_2 emissions for different treatments (mean ± std. error) from (a) S3 soils, (b) S15 soils, and (c) S30 soils.

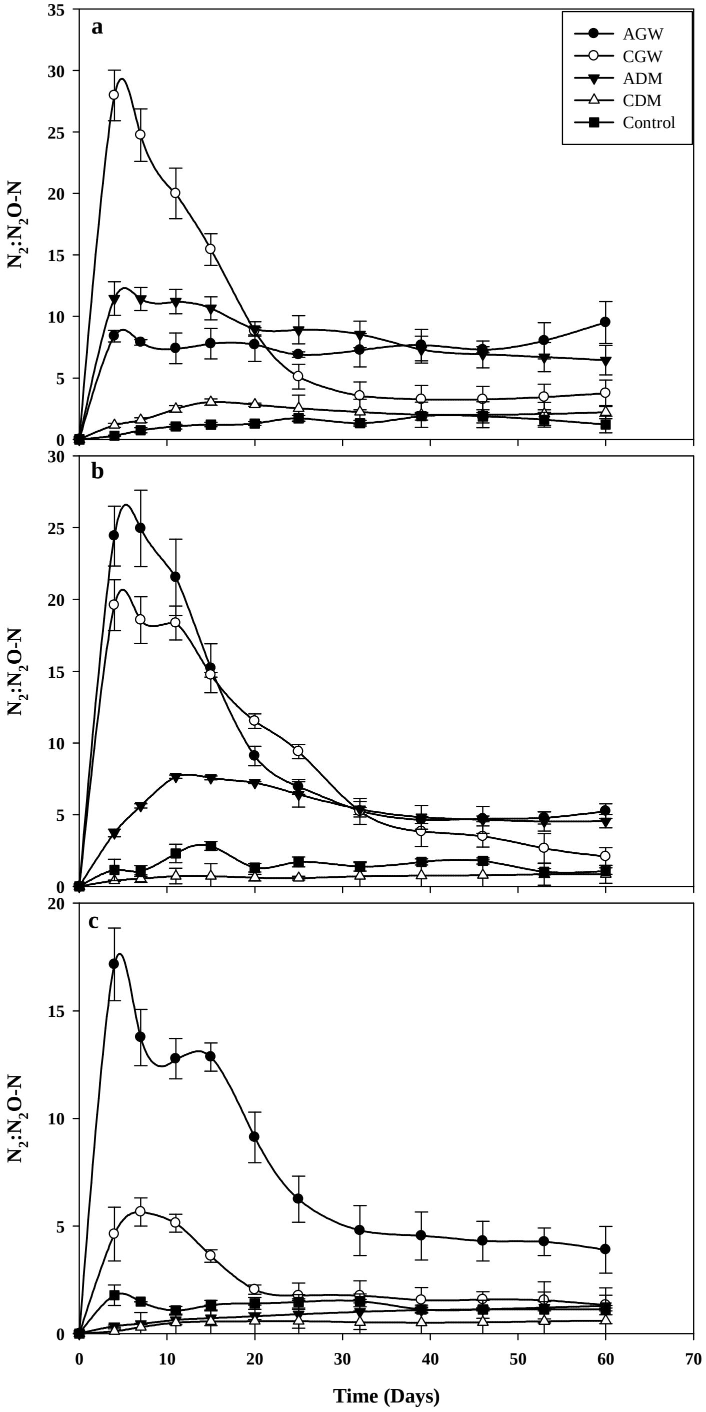

Fig. 2.4. Ratios of N_2 to N_2O-N for different treatments (mean ± std. error) from (a) S3 soils, (b) S15 soils, and (c) S30 soils.

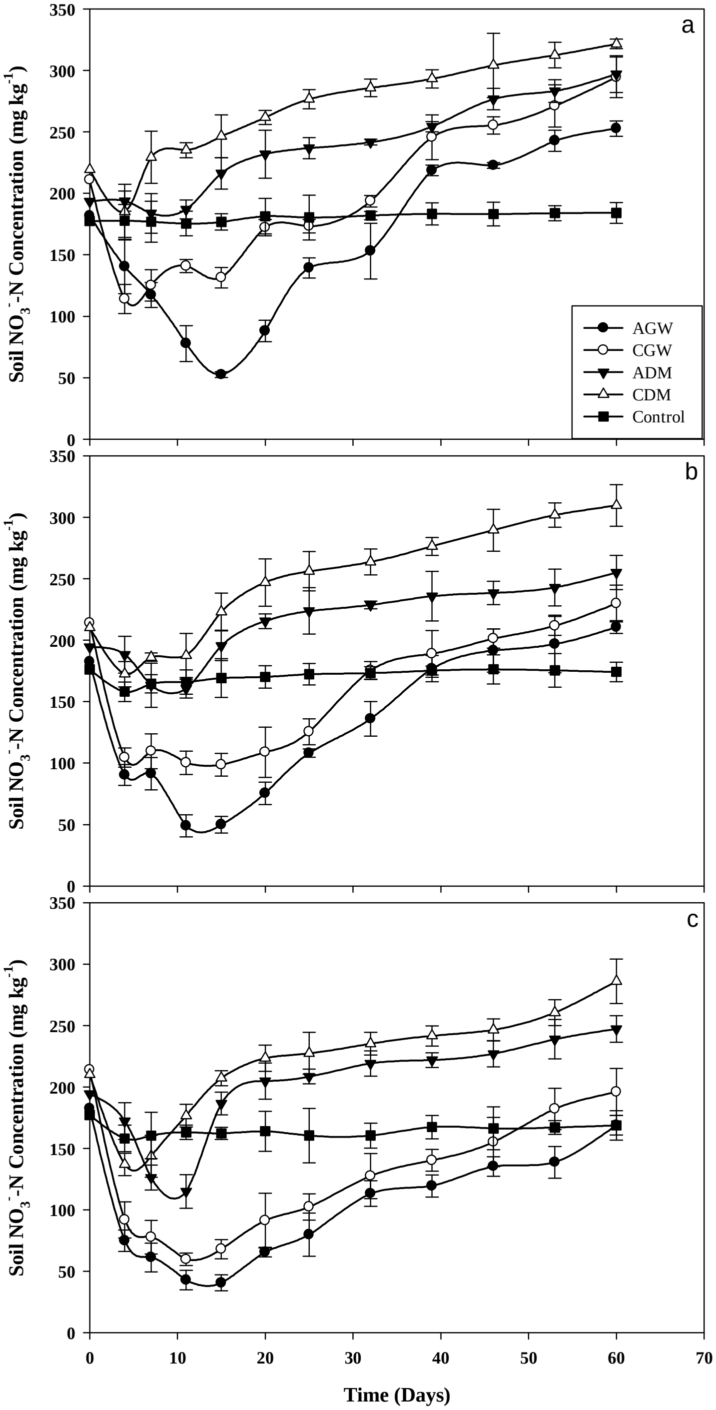

Fig. 2.5. Soil NO$_3^-$-N concentration for different treatments (mean ± std. error) from (a) S3 soils, (b) S15 soils, and (c) S30 soils.

References

Banin A., Fish A. (1996) Secondary desertification due to salinization of intensively irrigated lands: The Israeli experience, Desertification in Developed Countries, Springer. pp. 17-37.

Bremner J., Shaw K. (1958) Denitrification in soil. II. Factors affecting denitrification. The Journal of Agricultural Science 51:40-52.

Bremner J., Blackmer A. (1979) Effects of acetylene and soil water content on emission of nitrous oxide from soils. Nature 280:380-381.

Bremner J.M., Blackmer A.M. (1981) Source of atmospheric nitrous oxide, in: C. C. Delwiche (Ed.), Denitrification, nitrification, and atmospheric nitrous oxide, Wiley, New York. pp. 151-170.

Cannavo P., Richaume A., Lafolie F. (2004) Fate of nitrogen and carbon in the vadose zone: in situ and laboratory measurements of seasonal variations in aerobic respiratory and denitrifying activities. Soil Biology and Biochemistry 36:463-478.

Carter M.R. (1993) Soil sampling and methods of analysis CRC Press, Boca Raton, Florida.

Cook B.D., Allan D.L. (1992) Dissolved organic carbon in old field soils: total amounts as a measure of available resources for soil mineralization. Soil Biology and Biochemistry 24:585-594.

Dregne H.E. (1983) Desertification of arid lands Harwood Academic Publishers, New York.

EPA. (2013) Inventory of U.S. Greenhouse gas emissions and sinks: 1990-2011. U.S. Environemntal Protection Agency, Washington, DC.

Firestone M.K., Davidson E.A. (1989) Microbiological basis of NO and N_2O production and consumption in soil, in: M. O. Andreae and D. S. Schimel (Eds.), Exchange of trace gases between terrestrial ecosystems and the atmosphere., John Wiley & Sons, New York. pp. 7-21.

Gee G.W., Bauder J.W. (1986) Particle-size analysis, in: J. C. Dane and G. C. Topp (Eds.), Methods of soil analysis: Part 4-Physical methods. pp. 278-282.

Huang Y., Zou J., Zheng X., Wang Y., Xu X. (2004) Nitrous oxide emissions as influenced by amendment of plant residues with different C: N ratios. Soil Biology and Biochemistry 36:973-981.

Laura R. (1974) Effects of neutral salts on carbon and nitrogen mineralisation of organic matter in soil. Plant and Soil 41:113-127.

Laura R. (1977) Salinity and nitrogen mineralization in soil. Soil Biology and Biochemistry 9:333-336.

Lavahun M., Joergensen R., Meyer B. (1996) Activity and biomass of soil microorganisms at different depths. Biology and Fertility of Soils 23:38-42.

Leffelaar P. (1986) Dynamics of partial anaerobiosis, denitrification, and water in a soil aggregate: experimental. Soil Science 142:352-366.

Maag M., Vinther F. (1996) Nitrous oxide emission by nitrification and denitrification in different soil types and at different soil moisture contents and temperatures. Applied Soil Ecology 4:5-14.

Martinez-Beltran J., Manzur C.L. (2005) Overview of salinity problems in the world and FAO strategies to address the problem, Proceedings of the international salinity forum, Riverside, California. pp. 311-313.

McClung G., Frankenberger W. (1985) Soil nitrogen transformations as affected by salinity. Soil Science 139:405.

McCormick R.W., Wolf D.C. (1980) Effect of sodium chloride on CO_2 evolution, ammonification, and nitrification in a Sassafras sandy loam. Soil Biology and Biochemistry 12:153-157.

McKenney D., Wang S., Drury C., Findlay W. (1993) Dentrification and mineralization in soil amended with legume, grass, and corn residues. Soil Science Society of America Journal 57:1013-1020.

Müller C., Stevens R., Laughlin R., Jäger H.-J. (2004) Microbial processes and the site of N_2O production in a temperate grassland soil. Soil Biology and Biochemistry 36:453-461.

Murphy D., Sparling G., Fillery I. (1998) Stratification of microbial biomass C and N and gross N mineralisation with soil depth in two contrasting Western Australian agricultural soils. Australian Journal of Soil Research 36:45-55.

Parton W., Silver W.L., Burke I.C., Grassens L., Harmon M.E., Currie W.S., King J.Y., Adair E.C., Brandt L.A., Hart S.C. (2007) Global-scale similarities in nitrogen release patterns during long-term decomposition. Science 315:361-364.

Pathak H., Rao D. (1998) Carbon and nitrogen mineralization from added organic matter in saline and alkali soils. Soil Biology and Biochemistry 30:695-702.

Patten D., Bremner J., Blackmer A. (1980) Effects of drying and air-dry storage of soils on their capacity for denitrification of nitrate. Soil Science Society of America Journal 44:67-70.

Ravishankara A.R., Daniel J.S., Portmann R.W. (2009) Nitrous oxide (N_2O): The dominant ozone-depleting substance emitted in the 21st century. Science 326:123-125.

Rietz D., Haynes R. (2003) Effects of irrigation-induced salinity and sodicity on soil microbial activity. Soil Biology and Biochemistry 35:845-854.

Robertson G.P., Groffman P.M. (2007) Nitrogen Transformations, in: E. A. Paul (Ed.), Soil Microbiology and Biochemistry, Elsevier Academic Press, Burlington, MA. pp. 341-364.

Ruiz-Romero E., Alcántara-Hernández R., Cruz-Mondragon C., Marsch R., Luna-Guido M., Dendooven L. (2009) Denitrification in extreme alkaline saline soils of the former lake Texcoco. Plant and Soil 319:247-257.

Singh G. (2009) Salinity-related desertification and management strategies: Indian experience. Land Degradation and Development 20:367-385.

Taylor B.R., Parkinson D., Parsons W.F. (1989) Nitrogen and lignin content as predictors of litter decay rates: a microcosm test. Ecology 70:97-104.

Tejada M., Garcia C., Gonzalez J.L., Hernandez M.T. (2006) Use of organic amendment as a strategy for saline soil remediation: Influence on the physical, chemical and biological properties of soil. Soil Biology and Biochemistry 38:1413-1421.

Yuan B.-C., Li Z.-Z., Liu H., Gao M., Zhang Y.-Y. (2007) Microbial biomass and activity in salt affected soils under arid conditions. Applied Soil Ecology 35:319-328.

3. INFLUENCE OF TEMPERATURE AND ORGANIC AMENDMENT APPLICATIONS ON CARBON DIOXIDE, NITROUS OXIDE, AND DINITROGEN EMISSIONS FROM A SALINE-SODIC SOIL

Abstract

Incorporation of organic amendments such as manures and composts to remediate saline-sodic soils help improve physical, chemical, and biological properties of these soils. However, little is known how organic matter applications to such soils would affect CO_2, N_2O, and N_2 emissions and soil N dynamics. The effects of soil temperature and organic matter applications to a saline-sodic soil on CO_2, N_2O, and N_2 emissions were investigated. An extreme saline-sodic soil with electrical conductivity (EC_e) 100 dS m^{-1} and pH 7.8 was used in this study. Four organic amendment treatments were selected: active greenwaste (AGW), cured greenwaste compost (CGW), active dairy manure (ADM), and cured dairy manure compost (CDM). All treatments were incorporated at 100 Mg ha^{-1} and incubated at 70% water-filled pore space (WFPS) for 60 days at 15, 25, and 35°C. Increases in soil temperature significantly enhanced cumulative CO_2, N_2O, and N_2 emissions from all treatments. The emissions of CO_2 increased in the AGW and ADM by 1.5 to 2.2 fold compared to CGW and CDM amended soils at all three temperatures. Higher cumulative N_2O-N emissions were released from ADM

compared to AGW and CDM compared to CGW treatments at 25 and 35°C, respectively. At 25 and 35°C, AGW and CDM increased N_2 emissions relative to ADM and CGW, respectively. It was found that while soil temperature had a significant effect on CO_2, N_2O, and N_2 emissions; CO_2 production alone influenced N_2 emissions indicating that active organic materials could be carefully applied to remediate saline-sodic soils while mitigating N_2O emissions.

Introduction

Soil degradation due to salinization is one of the major causes for reduced crop productivity throughout the world (Rengasamy, 2006). In arid and semi-arid regions such as California, low rainfall coupled with intense evaporation enhances salt accumulation in the upper soil profile. Negative effects of excessive salt accumulation on soil physical and chemical properties and microbiological processes have been well documented (Rietz and Haynes, 2003; Wong, et al., 2009). Adverse effects of excessive exchangeable sodium have been shown to decrease structural stability, soil permeability, and infiltration rate (Lauchli and Epstein, 1996). The effects of soil salinity on soil microbial properties include suppression of soil microbial communities and their biochemical activities (Rietz and Haynes, 2003; Yuan, et al., 2007).

In the United States, saline-sodic soils are soils with $EC_e > 4$ dS m^{-1} and high amounts of soluble Na^+ with SAR >15 (Richards, 1954). Soil used in this study is considered "extreme saline-sodic" as $EC_e = 100$ dS m^{-1} and SAR >125. Such soils are commonly degraded with poor drainage and structural stability. With nearly 20% of world's arable land and 50% of permanent cropland affected by soil salinization (Flowers and Yeo, 1995), reclamation of these soils to bring them back to productivity has become an international priority.

Organic amendments such as manures and composts have been used to remediate salt-affected soils (Liang, et al., 2005; Tejada, et al., 2006; Wahid, et al., 1998). Resulting beneficial effects have been attributed to the introduction of organic matter and associated microbial processes (Pathak and Rao, 1998; Wong, et al., 2009). Soil organic matter is known to influence various physical, chemical, and biological properties of soils. Organic carbon compounds found in manures and composts serve as a source of electron donors for energy and synthesis of cellular constituents for many soil microorganisms including nitrifiers and denitrifers which are sensitive to salt toxicity. Several studies have reported decreased soil microbial processes such as organic matter decomposition and nitrification with increase in salt toxicity (Pathak and Rao, 1998; Tejada, et al., 2006). As most denitrifiers are heterotrophic bacteria, the process of denitrification is strongly correlated to the availability of soil carbon. Therefore, soil salinization may also affect nitrous oxide (N_2O) production from denitrification processes.

With a global warming potential 310 times higher than CO_2 over a 100-year period, N_2O is an important greenhouse gas. This high potency makes even relatively small changes in N_2O emissions potentially significant and atmospheric concentrations have been increasing since 1988 at a rate of 0.8 ppb yr^{-1}. In 2006, atmospheric N_2O concentrations reached 321.8 ppb, 3.2 ppb higher than in 2004 (IPCC, 2007). Several studies have also reported the depletion of stratospheric ozone with increased N_2O concentrations (Bouwman, 1990; Crutzen, 1979; Crutzen and Ehhalt, 1977; Hahn and Junge, 1977).

Soils are identified as a major source of N_2O emissions with 6.0 Tg yr^{-1} emitted from natural soils and 4.2 Tg yr^{-1} from agricultural soils, accounting for nearly 70% of N_2O produced from the biosphere (IPCC, 2007). Enhanced denitrification rates with the application of organic matter to soils in the form of plant residues, green manures, and farm yard manures in the laboratory and field studies are well documented (Anderson and Levine, 1986; Cabrera, et al., 1994; Coyne, et al., 1994; Ding, 2007; Meng, et al., 2005; Rolston, et al., 1978). For example, Coyne et al. (1994) observed increased N_2O fluxes accounted for up to 39% of the total N gas flux from manure amended plots compared to plots receiving grass filters. Similarly, Rolston et al. (1978) found that approximately 70% of applied fertilizer was lost via denitrification, primarily as N_2, from a manured soil.

Several soil factors can be identified as potentially significant to the formation of N_2O, including soil texture (Vinther, 1992; Weitz, et al., 2001), moisture (Maag and Vinther, 1996; Zheng, et al., 2000), and pH (Eaton and Patriquin, 1989; Focht, 1974; Goodroad and Keeney, 1984; Martikainen, 1985). In addition to the above mentioned factors, temperature has a profound effect in regulating the C and N mineralization, and denitrification processes. Goodroad and Keeney (1984) showed increased rates of nitrification and N_2O production with increasing temperatures from 10 to 30° C. They concluded that under isothermal conditions, N_2O production was proportional to nitrate production and that 0.1-0.2% of the nitrified N was evolved as N_2O in dry, well-structured soils.

High temperature conditions coupled with high concentrations of salts in soil water are likely to impact the existing microbial community thereby affecting the decomposition of added organic amendments and denitrification. Information on the effect of reclaiming extreme saline-sodic soils using organic amendments at varying temperatures on N_2O and N_2 emissions is lacking however. Heat may affect denitrifiers differently as they decompose available carbon under saline conditions. Because global warming and food production are both critical issues, it is of utmost importance to understand the impact that these organic amendments have on the environment prior to widely using these materials to alleviate soil salinization. We have therefore investigated how the addition of organic amendments varying in their carbon availability to an extreme saline-sodic soil at different temperatures affected the emission of CO_2, N_2O, and N_2 in a laboratory experiment. Specific objectives of this research are:

a) to examine the effect of extreme salinity and sodicity on CO_2, N_2O, and N_2 emissions;
b) to investigate organic amendment effects on the N dynamics of a saline-sodic soil; and
c) to determine how a varying soil temperature altered these processes.

Materials and Methods

Experimental Site

Soil was collected from the upper 10 cm of a fallow field located in San Jacinto, California (33º 50' 44.2"N, 117 º 1' 39.6"W). The field has been abandoned for more than 5 years due to high salt content to a depth of 30 cm, poor soil structure, and impaired drainage. Taxonomic classification for this soil is Grangeville coarse-loamy, mixed, superactive, thermic fluvaquentic haploxeroll. The bulk soil samples were air-dried to 30% water holding capacity and sieved through a 2 mm sieve. Soil physical and chemical properties are illustrated in Table 3.1.

Treatments

Treatments included Control (no-amendment added) and four organic amendments: active greenwaste (AGW), cured greenwaste (CGW), active dairy manure (ADM), and cured dairy manure (CDM). The AGW and GWC were collected from a commercial greenwaste composting facility located in California. The AGW was comprised of plant material from local roadside cuttings and was collected from an actively managed two-week old turned windrow. The CGW was collected from a screened cured compost storage pile that had also undergone 10 weeks of active windrow composting. Organic amendments ADM and CDM were collected from actively managed two-week old turned windrow and 12-week old on-farm compost storage pile, respectively, at a dairy farm in California. Samples of all treatments were air-dried in a greenhouse at 20º C until moisture contents stabilized. The air-dried samples were then milled using a Wiley mill to pass through a 4 mm sieve. Chemical characteristics of treatments are presented in Table 3.2.

Laboratory Incubations

Before treating with amendments, soils were pre-incubated at 40% WFPS in the dark at 25ºC for two weeks. The gravimetric water content required to maintain the soils at the desired WFPS was calculated using Eq. 1.

In this study, all treatments including the controls were subjected to three temperature regimes of 15, 25, and 35ºC. Soil inorganic N content and CO_2 and N_2O emissions at each temperature condition were monitored using two different experimental setups. Inorganic N was measured in 1 L jars containing 250 g dry soil while CO_2 and N_2O production was measured in 100 mL glass serum bottles containing 25 g dry soil. All treatments were uniformly incorporated into the soil at an application rate of 100 Mg ha^{-1} and packed to a bulk density of 1.3 g cm^{-3}. De-ionized water was used to maintain soil moisture content at 70% WFPS as this has been shown to maximize nitrification and still result in denitrification losses (Maag and Vinther, 1996). Aerobic conditions were maintained for all treatments in each of the experimental setups throughout this study. Three replicates of each treatment were setup for the 1 L jars while six replicates were prepared for the 100 mL serum bottles. Incubations were conducted at 15, 25, and 35ºC for 60 days in the dark.

Analytical Methods

Soil and gas samplings were conducted on Days 0, 1, 3, 5, 9, 14, 20, 26, 33, 40, 50, and 60 days from the commencement of the incubations. Soil inorganic N (NH_4^+-N and NO_3^--N) concentration was determined by colorimetry on 2M KCl (ratio 1:5) extractions.

All containers used to collect gas samples were flushed with air for 30 s to remove previously accumulated trace gases and sealed with rubber septa and aluminum crimps. Acetylene (C_2H_2) inhibition technique was used in all treatments including controls to determine the conversion of N_2O to N_2. Of the six replicates prepared for measuring CO_2 and N_2O, three received 99.999% pure C_2H_2 that was injected to replace 10% of the headspace volume. To promote uniform distribution of the added C_2H_2, headspace gas was mixed thoroughly using a large syringe. Following a 4 hr sealed incubation, 30 mL gas samples were collected from the headspace using a gas-tight syringe and immediately transferred to 20 mL pre-evacuated gas vials. Samples were analyzed for CO_2 using infrared gas analyzer (Model EGM-4, PP- Systems, Amesbury, Massachusetts) while N_2O was measured using HP 5790 A Gas Chromatograph (Hewlett- Packard Co., Fullerton, California) fitted with 80/100 mesh Porapak Q column and ^{63}Ni electron capture detector. Oven and injection temperatures were set at 50 ºC and a detector temperature of 300 ºC was maintained. Standards were injected after every 10 samples to assure instrument precision.

Statistical Analysis

The statistical analysis was performed using the SPSS version 20.0 software. Linear interpolation of daily measurements across consecutive sampling days was used to assess cumulative CO_2-C, N_2O-N, and N_2 emissions. Two-way factorial analysis of variance was used to assess organic amendment and soil temperature effects on cumulative CO_2-C, N_2O-N, and N_2 emissions, ratios of N_2 to N_2O-N, and soil mineral N concentrations. The impact of treatments and soil temperature on the aforementioned parameters over time was tested using repeated measures ANOVA. Significant differences, if any, between various soil temperature levels and organic amendment treatments were separated using Tukey's multiple comparison tests. Correlation matrices and stepwise regression analyses were conducted to examine relationships between soil temperature, CO_2-C, N_2O-N and N_2 emissions, ratios of N_2 to N_2O-N, and soil mineral N concentrations.

Results

Emission of CO_2

Influence of Temperature

Soil respiration from all organic amendments was significantly increased ($P < 0.05$) by elevating temperatures over the incubation period (Fig. 3.1). Daily and cumulative CO_2 emissions increased between 15 and 25°C and 25 and 35°C. Peak daily

CO_2 emission rates were observed from Days 5-14 and gradually decreased thereafter. Cumulative CO_2 production ranged from 4.0-44.8 mg C kg^{-1} at 15°C, 14.5-122.2 mg C kg^{-1} at 25°C, and 19.7-320.7 mg C kg^{-1} at 35°C soils. For all organic amendments, apparent values of Q_{10} for cumulative CO_2-C ranged from 2.6-3.2.

Influence of Organic Amendments

At all three temperatures, organic amendments added to the soil significantly increased ($P < 0.05$) CO_2 production. Emissions from active materials (AGW, ADM) exceeded those from cured materials (CGW, CDM) at all three temperatures. Cumulative CO_2 emissions across the three temperatures from ADM were 1.8-2.3 times higher than from CDM while AGW losses were approximately 1.6 times that of CGW. Compared to CGW, addition of AGW increased cumulative CO_2-C emissions by 42, 38, and 35% at 15, 25, and 35°C, respectively. Similarly a 55, 49, and 45% increase in CO_2 production was observed in 15, 25, and 35°C soils respectively, with the application of ADM compared to CDM treatment.

Emission of N_2O and N_2

Influence of Temperature

Cumulative N_2O-N and N_2 emissions were significantly enhanced ($P < 0.05$) with increasing temperature (Fig. 3.2 and 3.3). Greater N_2O and N_2 production occurred from all treatments at 35°C while the lowest emissions were observed from soils at 15°C temperature. Apparent Q_{10} values for cumulative N_2O-N emissions ranged from 0.9-4.0

between 15 and 25°C and 1.7-7.3 between 25 and 35°C soils. In case of cumulative N_2 production, Q_{10} values ranged from 2.4-9.2 between 15 and 25°C and 1.1-3.2 between 25 and 35°C.

Cumulative N_2O-N production ranged from 1.1-49.1 µg N kg^{-1} at 15°C, 1.0-194.7 µg N kg^{-1} at 25°C, and 7.5-351.7 µg N kg^{-1} at 35°C soil temperature (Fig. 3.2). Cumulative N_2 production ranged from 0.6-51.1 µg N kg^{-1} at 15°C, 4.0-288.9 µg N kg^{-1} at 25°C, and 4.7-728.1 µg N kg^{-1} at 35°C temperature (Fig. 3.3). Regardless of the treatment, the ratios of N_2 to N_2O-N were significantly ($P < 0.05$) higher at 25°C and 35°C compared to 15°C. The ratios of N_2 to N_2O-N ranged from 0.5-1.5 at 15°C, 1.2-4.8 at 25°C, and 0.6-4.1 at 35°C (Fig. 3.4).

Effect of Organic Amendments

Regardless of the temperature, application of organic amendments significantly ($P < 0.05$) increased N_2O emissions. At all three temperatures, CGW and ADM resulted in greater cumulative N_2O-N emissions relative to their counterparts AGW and CDM respectively (Fig. 3.2). Compared to AGW, application of CGW increased cumulative N_2O-N emissions by 35, 73, and 24% at 15, 25, and 35°C temperatures, respectively. In contrast, use of CDM as soil amendment decreased cumulative N_2O-N production by 15, 25, and 30% compared to ADM amended soils at 15, 25, and 35°C, respectively (Fig. 3.2).

At all three temperature levels, incorporation of organic amendments resulted in a significant ($P < 0.05$) increase in N_2 production. Active amendments (AGW and ADM) produced significantly greater N_2 emissions compared to the cured materials (CGW and CDM) at 25 and 35°C while CGW and ADM produced highest N_2 emissions at 15°C. Across the three temperatures, cumulative N_2 emissions from active amendments ranged from 31.4-728.1 µg N kg^{-1} while those from cured treatments ranged from 22.4-467.7 µg N kg^{-1}. Of the two active amendments used in this study, AGW produced greater N_2 emissions compared to ADM at 25 and 35°C temperatures. However, at 15°C temperature, an opposite trend was observed with ADM producing more N_2 emissions than AGW. Compared to AGW, incorporation of CGW reduced N_2 emissions by 63, 58, and 47% in soils maintained at 15, 25, and 35°C, respectively. Similarly, application of CDM relative to ADM decreased N_2 production by 48, 21, and 18% at 15, 25, and 35°C, respectively (Fig. 3.3).

Application of organic amendments significantly ($P < 0.05$) increased the ratios of N_2 to N_2O-N at all temperatures. Ratios of N_2 to N_2O-N from active materials ranged from 0.9-4.8 while that for cured materials ranged from 0.5-1.8. Increases in N_2 to N_2O-N ratios from active materials, AGW and ADM, ranged from 1.3-4.1 times while those from cured materials, CGW and CDM, ranged from 0.8-3.3 fold at 25°C and 35°C relative to 15°C (Fig. 3.4).

Inorganic N

Effect of Temperature

Soil mineral N concentration ($P < 0.05$) was significantly affected by temperature. After 60 days of incubation, the amount of soil NO_3^--N and total inorganic N significantly ($P < 0.05$) increased with increasing temperature (Fig. 3.5 and 3.6). Regardless of the treatment, gradual increase in mineral N was recorded at all temperatures after 26 days into the incubation period. This increase was more pronounced at 35°C followed by 25°C compared to 15 °C (Fig. 3.6). The amount of soil mineral N present after the 60 days incubation in soils amended with organic materials increased by 19-26% between 15 and 25°C and 27-41% between 15 and 35°C.

Effect of Organic Amendments

Incorporation of organic amendments significantly ($P < 0.05$) increased soil NO_3^--N and NH_4^+-N concentration at all temperatures. Across the incubation period, regardless of soil temperature, lower mineral N concentrations were recorded for AGW and CGW treatments compared to the ADM and CDM amendments (Fig 3.6). Immediately after organic amendment applications, soil NO_3^--N concentrations were drastically decreased in all treatments. This decreasing trend in AGW and CGW amended soils lasted for 28, 28, and 10 days at 15, 25, and 35°C, respectively. Similar decrease in soil NO_3^--N concentrations were recorded for ADM and CDM treated soils as well but the decrease only lasted up to 2 weeks at all three temperatures (Fig. 3.5).

Relationship between CO_2, N_2O, N_2, and Soil Inorganic N Concentrations

Soil temperature significantly and positively correlated with soil NO_3^--N concentrations ($r^2 = 0.67$; $P < 0.05$) and with cumulative CO_2-C, N_2O-N and N_2 emissions ($0.85 < r^2 \leq 0.92$; $P < 0.01$). Soil respiration significantly and positively correlated with N_2O ($r^2 > 0.71$; $P < 0.05$) and N_2 ($r^2 > 0.88$; $P < 0.05$) fluxes at each of the sampling times. However, ratios of N_2 to N_2O-N across all treatments were significantly but weakly explained by temperature and soil respiration ($0.42 < r^2 \leq 0.55$; $P < 0.05$). A combination of soil temperature, soil NO_3^-, and CO_2 production could explain up to 89% variability while soil NO_3^--N concentrations alone accounted for 67% of variation in N_2O-N emissions (Table 3.3). For N_2 emissions however, a combination of soil temperature and CO_2 evolution accounted for nearly 95% variation of which 93% was explained by CO_2 alone. This suggests CO_2 flux was a better predictor of cumulative N_2 emissions from this salt-affected soil.

Discussion

Production of CO_2

Soil respiration and microbial activity are known to be decreased in moderately saline soils (EC_e < 15 dS m^{-1}) due to increased osmotic stress on soil microbial community (Laura, 1974; McCormick and Wolf, 1980; Pathak and Rao, 1998). However, the production of CO_2 throughout the 60-day incubation period in this study suggests that the heterotrophic microbes were still active even under extremely saline conditions. Evolution of CO_2 was lowest at 15°C and highest at 35°C suggesting that the microbial activity was enhanced at increased temperature (Fig. 3.1). Similar findings were documented by several authors in non-saline soils (Moore and Dalva, 1993, Schaufler, et al., 2010; Schlentner and Cleve, 1985) and our results have demonstrated that this effect holds true even under extremely saline soils. Organic amendments increased soil CO_2 fluxes at all temperatures with significantly higher emissions from AGW and ADM compared to CGW and CDM suggesting that active treatments decomposed more rapidly than the cured counterparts. These findings are consistent with those reported by Tejada et al. (2006) who reported greater cumulative CO_2-C emissions from saline soils amended with poultry manure compared to cotton gin compost. These results were likely related to the presence of large amounts of labile C in active materials than in cured amendments. The soil used in this study had a pH of 7.8 which is above the threshold carbonate dissociation value of 7.2. This eliminates the possibility of chemical dissociation of soil carbonates and subsequent evolution of CO_2.

Production of N_2O and N_2

Findings from this study show that organic amendment application had a major influence on N_2O and N_2 emissions. These results are consistent with other studies that have shown greater N_2O emissions following organic matter applications (McKenney, et al., 1993; Meng, et al., 2005; Yao, et al., 2010). Positive correlations between CO_2 and N_2O and CO_2 and N_2 imply that denitrification may have occurred when CO_2 emissions peaked and created temporary anaerobic microsites (Cannavo, et al., 2004; Huang, et al., 2004). In addition, greater availability of C from active treatments than from cured amendments resulted in increased N_2 emissions from AGW and ADM, thereby completing the denitrification process in its entirety.

The availability of N following application of organic amendments is another important factor driving the production of N_2O. Losses were higher from the dairy manure than the greenwaste amendments which had more readily available N for heterotrophic denitrifiers involved in N_2O production. At all temperatures considered in this study, consistently greater losses of N_2O were observed from ADM followed by CDM and CGW while the lowest emissions were from AGW amendment suggesting that ADM and CDM were more prone to mineralization than AGW and CGW treatments. Presumably this difference would be reduced over time as labile compounds in the active materials decompose.

Both N_2O and N_2 emissions were increased dramatically in all treatments by increasing soil temperature from 15 to 35°C. These results are consistent with other studies that have shown increased denitrification rates with increased soil temperatures (Goodroad and Keeney, 1984; Maag and Vinther, 1996). The apparent Q_{10} values for N_2O emissions from various treatments used in this study between 15 and 35°C are in accordance with those reported in the literature which varied from 1.5-3.0 between 10 and 35°C (Jacobson and Alexander, 1980; Knowles, 1982).

Elevated N_2O and N_2 emissions are associated with greater soil NO_3^--N concentrations (Weier, et al., 1993). In this study, greater cumulative N_2O-N emissions were obtained from ADM and CDM compared to AGW and CGW treatments at all three temperature levels. However, corresponding increases in N_2 emissions were not observed for the same treatments. Instead, at 25 and 35°C, greater cumulative N_2 production was observed from AGW and ADM followed by CDM and CGW amendments. There may be two reasonable explanations for this result. First, bioavailability of C may have been higher in active treatments compared to cured treatments as evident from cumulative CO_2-C emissions (Fig. 3.1). This might have further supported the heterotrophic denitrifies responsible for reduction of N_2O to N_2 (Huang, et al., 2004). Second, more rapid mineralization of the dairy manure treatments resulted in greater NO_3^- availability compared to the greenwaste amendments (Fig 3.5). This created a more favorable condition for NO_3^- reduction to N_2O in ADM and CDM.

Of the various organic amendments used in this study, at all three temperatures, greater N_2 to N_2O-N ratios were recorded only for the AGW treatment. This may be explained by the increased availability of C and lower availability of N from AGW than from other treatments. Although higher N_2 emissions were also recorded for ADM, the ratio of N_2 to N_2O-N did not increase significantly with increasing temperature. This may be attributed to the increased availability of inorganic N from ADM that enhanced N_2O emissions initially in the incubation period (Table 3.2).

Inorganic N

The effect of higher concentrations of salts in the soil profile have been shown to negatively affect C and N mineralization by inhibiting microbial activity (McClung and Frankenberger, 1985; McCormick and Wolf, 1980). Studies have also shown that when soils are amended with a high C:N organic material containing <1% N, soil microbes immobilize the available N (Parton, et al., 2007; Robertson and Groffman, 2007). It was expected that AGW and CGW amendments both of which had high C:N and < 1% total N, would immobilize available N to a greater extent than ADM and CDM which had lower C:N ratios and > 1% N . The fact that there was significantly higher levels of NH_4^+-N in ADM compared to CDM and to a lesser extent in CGW than in AGW indicates that ADM and CGW were more biochemically active than CDM and AGW, respectively (data not shown). Although the percentage increase in soil mineral N differed among various organic amendments, there was a rapid increase in NH_4^+-N in order ADM > CDM > CGW > AGW. This suggests that mineralization of organic matter was greater in dairy manure compared to greenwaste amendments and this drastically increased with increasing temperature (Fig. 3.5).

In this study, all treatments were maintained at 70% WFPS and no leaching occurred throughout the incubation. Therefore, the amount of mineral N measured in organic amended soils during the incubation represents the amount of mineralized N minus the gaseous N losses as N_2O and N_2 emissions. Both organic amendments AGW and CGW in this study had lower soil mineral N concentrations than the control by Day

28 at 15 and 25°C and Day 7 at 35°C indicating that increasing soil temperature reduced the duration of the immobilization phase in both these treatments (Fig. 3.6). This could have supplied lower amounts of NO_3^-, a substrate for denitirification, resulting in lower N_2O emission from AGW and CGW (Fig. 3.2). Similarly, greater availability of soil NO_3^- from ADM and CDM may have resulted in greater N_2O emissions as evident by significantly positive correlation between NO_3^- and N_2O production. These results are consistent with findings in the literature which indicate positive correlation between soil NO_3^- levels and denitrification (Bremner and Blackmer, 1981; Weier, et al., 1993).

Remediation of salt-affected soils using organic amendments involves repeated flooding in order to leach excess salts to deeper layers of soil profile (Hanay, et al., 2004). This is usually followed by an extended period of non-irrigated conditions and inversion of top soil by tillage. This tends to widely vary decomposition of the added amendments and dynamics of C and N over time. In this study, the effect of alternate wetting and drying conditions on N_2O and N_2 emissions was not considered. Therefore, further studies should focus on evaluating the effects of varied moisture conditions on trace gas emissions from the added organic amendments in salt-affected soils.

Conclusions

Organic amendments can be significant sources of CO_2, N_2O, and N_2 emissions. Cumulative CO_2-C, N_2O-N, and N_2 emissions were significantly increased at elevated temperatures. In addition, emission of N_2 was greater in active treatments compared to cured amendments. The main factors related to the N_2O emissions were soil temperature, CO_2 emissions, and soil NO_3^- concentrations. Manure amendments with their greater N contents, tended to increase soil NO_3^- concentrations and associated N_2O emissions compared to greenwaste treatments at all three temperatures considered in this study. Use of active organic amendments initially reduced N_2O emissions compared to the application of cured materials, but presumably this benefit would be reduced as the amendments stabilize in the soil and deplete available C over time. Measured N_2 to N_2O-N ratios were highest at 35°C suggesting that spring applications of organic amendments may impact greenhouse gas emissions by taking advantage of elevated summer temperatures more efficiently than applications in the fall. Labile compounds would be rapidly decomposed to more stable forms resulting in more favorable N_2 to N_2O-N ratios. Long-term effects of incorporating organic amendments in efforts to remediate salt-affected soils under varying environmental conditions prior to their adoption on a large scale also merits additional study.

Table 3.1: Physical and chemical properties of top 0-15 cm soil layer.

Parameter	Value
Sand (%)	43.12
Silt (%)	37.43
Clay (%)	19.45
Bulk Density (g cm^{-3})	1.26
EC$_e$ (dS m^{-1})	101.46
pH	8.24
SAR	149.75
Total C (%)	1.27
Total N (%)	0.08
C:N	15.53
NO_3^--N (mg kg^{-1})	205.57
NH_4^+-N (mg kg^{-1})	2.37

Table 3.2: Characteristics of organic amendments

Organic Amendment	Total C (%)	Total N (%)	C:N	NO_3^--N (mg kg^{-1})	NH_4^+-N (mg kg^{-1})
Active Greenwaste (AGW)	33.60	0.75	44.80	3.82	19.11
Cured Greenwaste Compost (CGW)	31.86	1.08	29.50	5.69	31.39
Active Dairy Manure (ADM)	21.39	1.76	12.15	10.34	123.06
Cured Dairy Manure Compost (CDM)	14.61	1.30	11.24	30.42	82.72

Table 3.3: Stepwise regression analysis of cumulative CO_2, N_2O, and N_2 emissions with soil properties.

Treatment	Regression Equation	Combined r^2 (adj)
Cumulative CO_2	$152.37 - 0.621\ NO_3^- + 0.361\ N_2 + 0.359\ N_2O$	0.959
Cumulative N_2O	$-305.74 + 1.350\ NO_3^- - 0.225\ N_2 + 1.007\ CO_2$	0.895
Cumulative N_2	$-308.47 + 1.247\ NO_3^- + 2.405\ CO_2 - 0.533\ N_2O$	0.949
Ratio of N_2 to N_2O-N	$1.564 + 0.008\ N_2 - 0.012\ N_2O$	0.752

Note: Units for selected variables are reported as: soil NO_3^- concentrations (mg N kg^{-1}), CO_2 emissions (mg C kg^{-1}) and N_2O and N_2 emissions (µg N kg^{-1}).

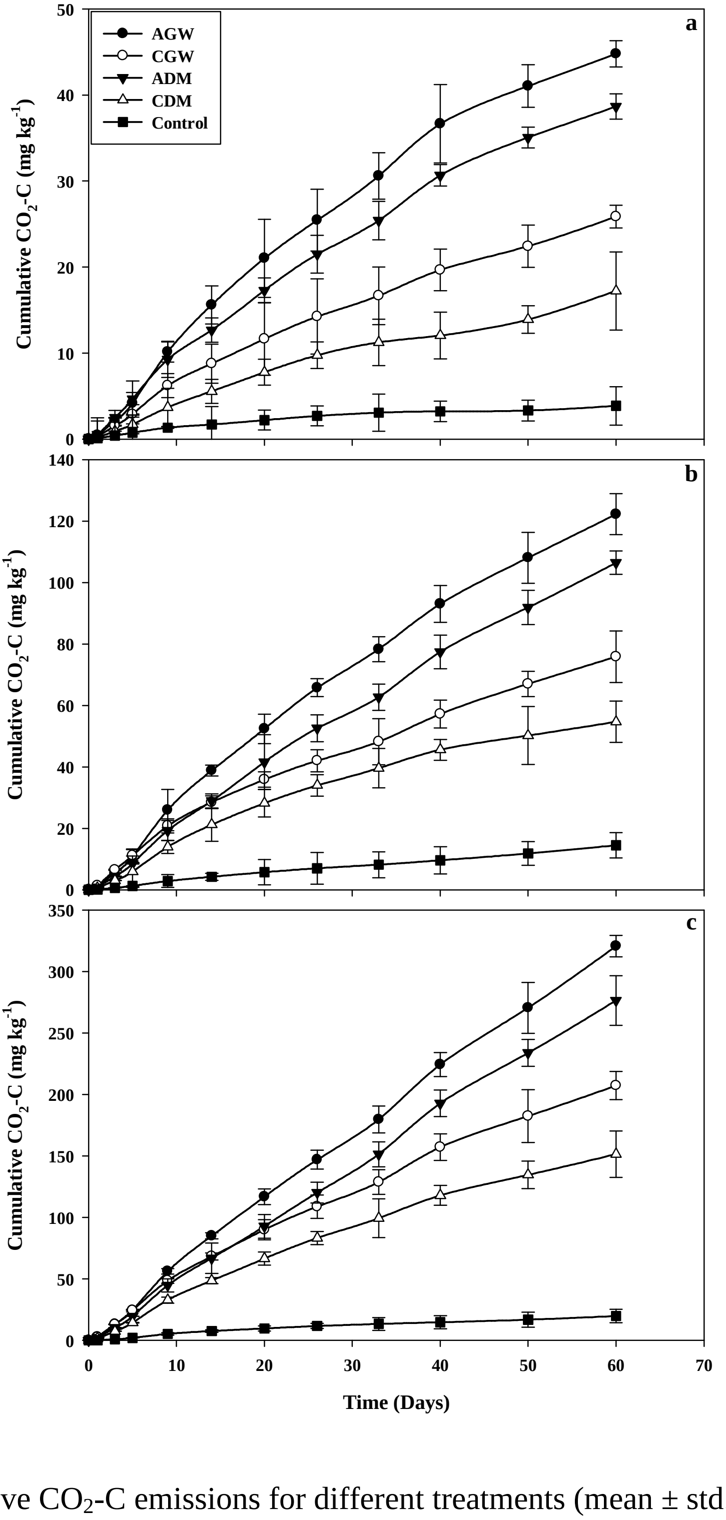

Fig. 3.1. Cumulative CO_2-C emissions for different treatments (mean ± std. error) in soils incubated at (a) 15°C, (b) 25°C, and (c) 35°C.

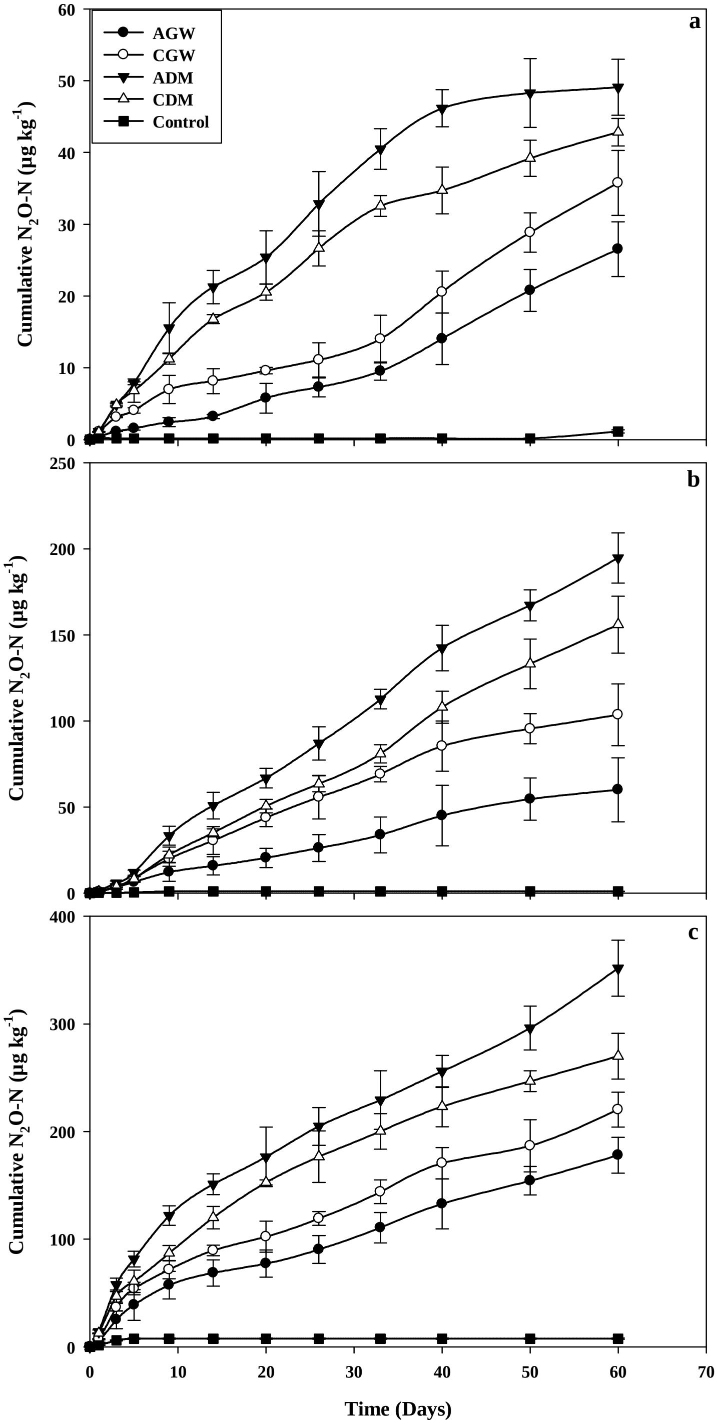

Fig. 3.2. Cumulative N_2O-N emissions for different treatments (mean ± std. error) in soils incubated at (a) 15°C, (b) 25°C, and (c) 35°C.

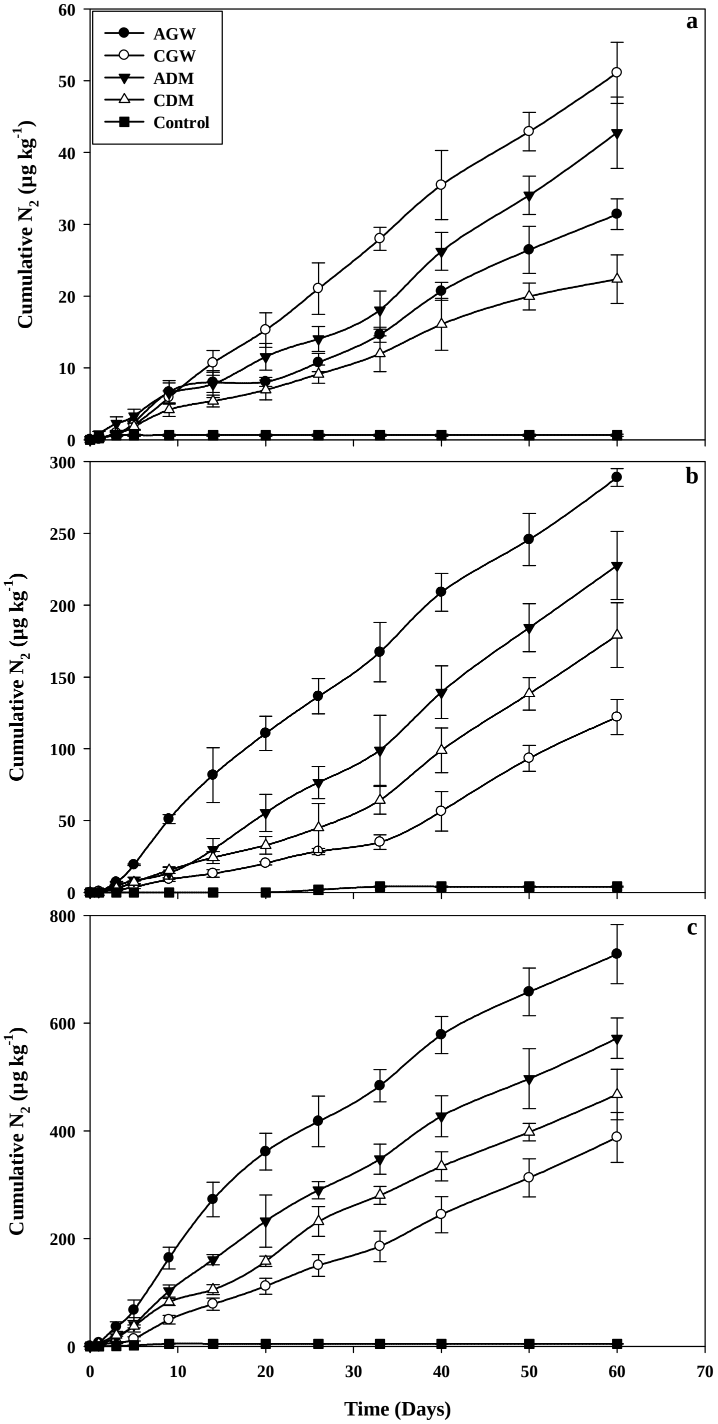

Fig. 3.3. Cumulative N_2 emissions for different treatments (mean ± std. error) in soils incubated at (a) 15°C, (b) 25°C, and (c) 35°C.

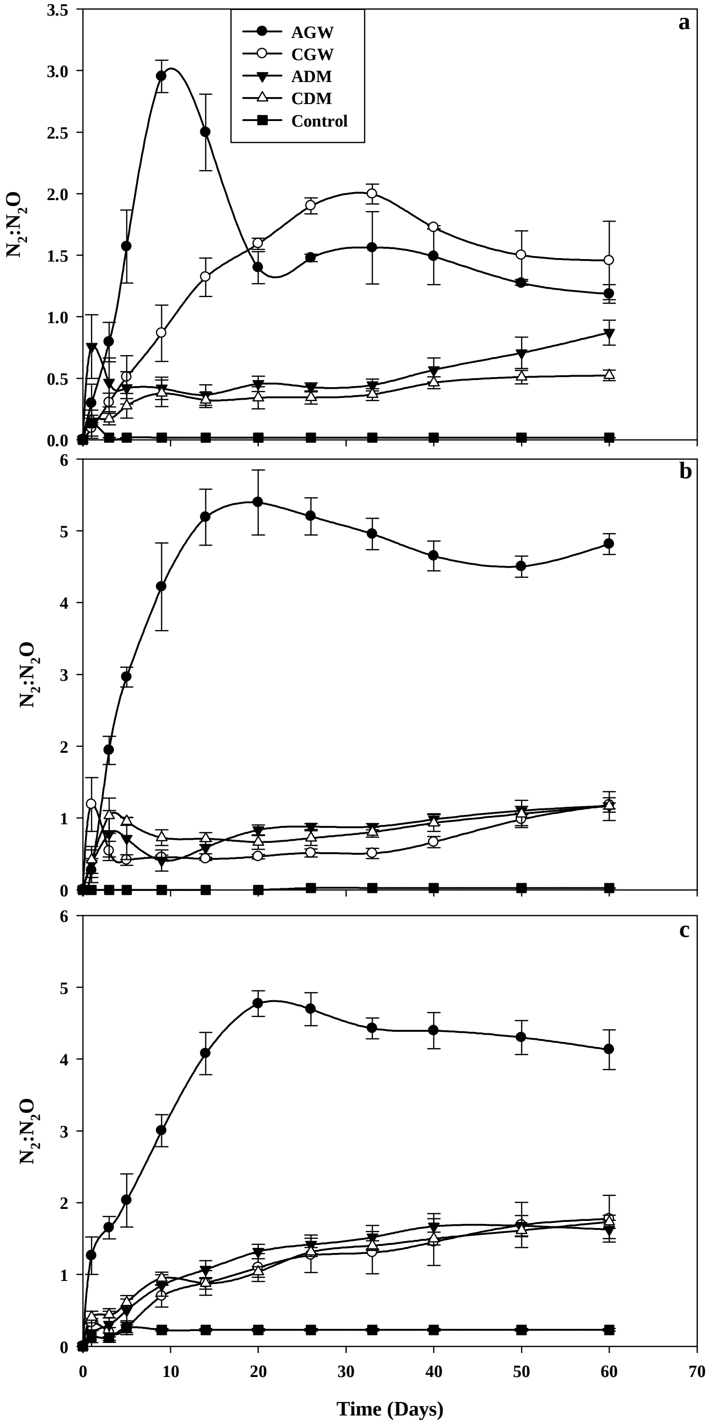

Fig. 3.4. Ratios of N_2 to N_2O-N for different treatments (mean ± std. error) in soils incubated at (a) 15°C, (b) 25°C, and (c) 35°C.

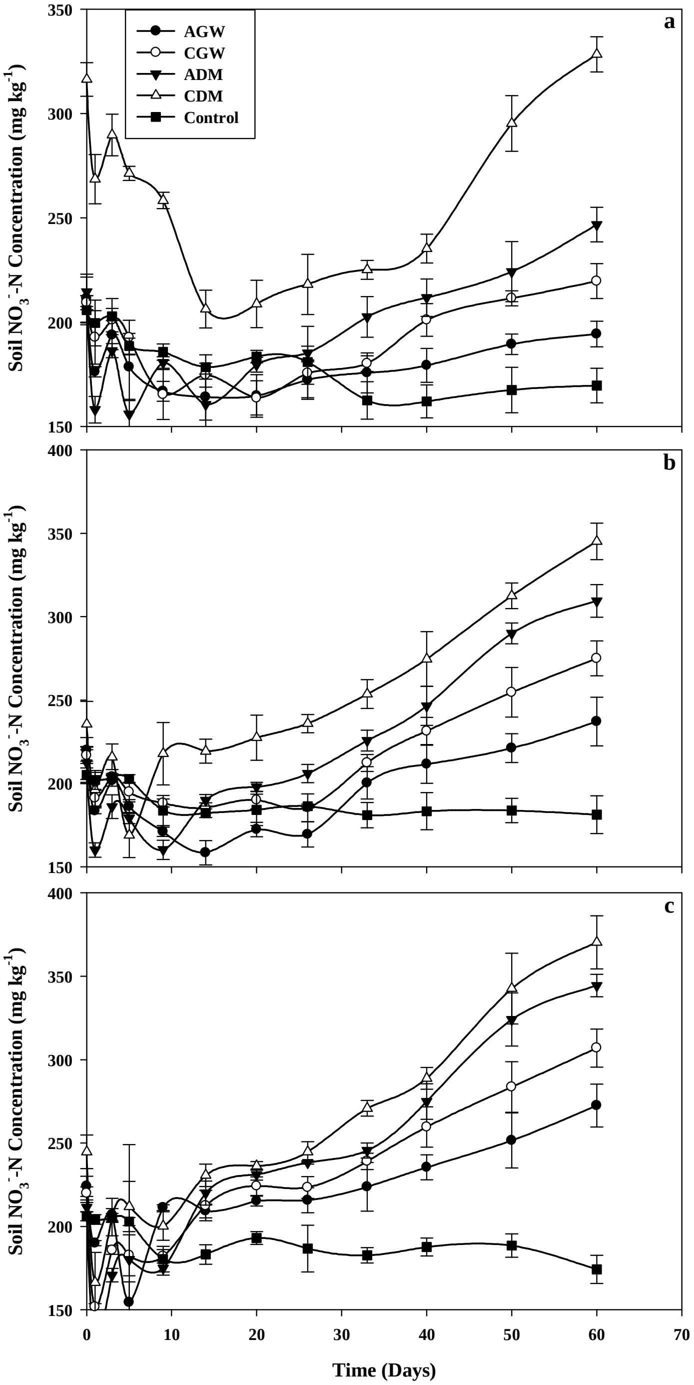

Fig. 3.5. Soil nitrate-N concentrations for different treatments (mean ± std. error) in soils incubated at (a) 15°C, (b) 25°C, and (c) 35°C.

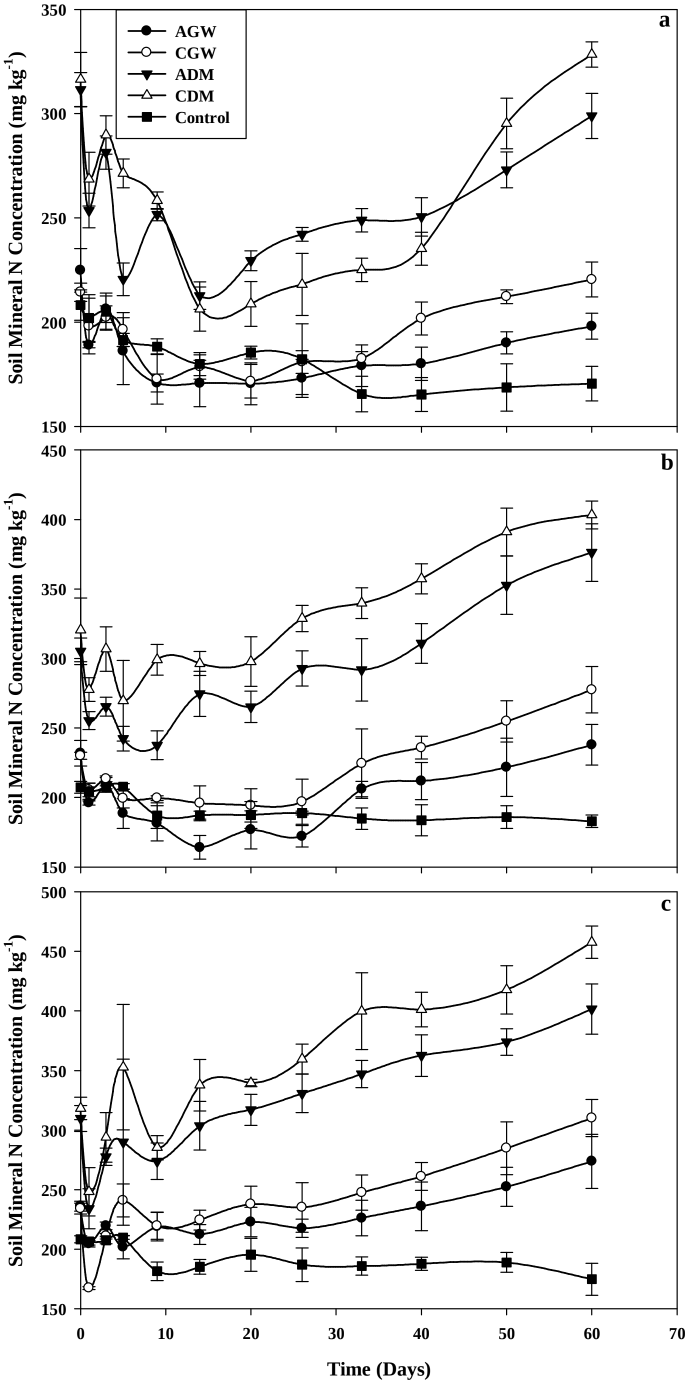

Fig. 3.6. Soil mineral-N concentrations for different treatments (mean ± std. error) in soils incubated at (a) 15°C, (b) 25°C, and (c) 35°C.

References

Anderson I.C., Levine J.S. (1986) Relative Rates of Nitric Oxide and Nitrous Oxide Production by Nitrifiers, Denitrifiers, and Nitrate Respirers. Applied and Environmental Microbiology 51:938-945.

Bouwman A. (1990) Exchange of greenhouse gases between terrestrial ecosystems and the atmosphere. p. 61-127, in: A. Bouwman (Ed.), Soils and the Greenhouse Effect, John Wiley & Sons, New York.

Bremner J.M., Blackmer A.M. (1981) Source of atmospheric nitrous oxide, in: C. C. Delwiche (Ed.), Denitrification, nitrification, and atmospheric nitrous oxide, Wiley, New York. pp. 151-170.

Cabrera M., Chiang S., Merka W., Pancorbo O., Thompson S. (1994) Nitrous oxide and carbon dioxide emissions from pelletized and nonpelletized poultry litter incorporated into soil. Plant and Soil 163:189-195.

Cannavo P., Richaume A., Lafolie F. (2004) Fate of nitrogen and carbon in the vadose zone: in situ and laboratory measurements of seasonal variations in aerobic respiratory and denitrifying activities. Soil Biology and Biochemistry 36:463-478.

Coyne M.S., Gilfillen R., Blevins R. (1994) Nitrous oxide flux from poultry-manured erosion plots and grass filters after simulated rain. Journal of environmental Quality 23:831-834.

Crutzen P. (1979) The role of NO and NO_2 in the chemistry of the troposphere and stratosphere. Annual Review of Earth and Planetary Sciences 7:443-472.

Crutzen P., Ehhalt D. (1977) Effects of nitrogen fertilizers and combustion on the stratospheric ozone layer. Ambio 6:112-117.

Ding W. (2007) Effects of long-term amendment of organic manure and nitrogen fertilizer on nitrous oxide emission in a sandy loam soil. Journal of Environmental Sciences (China) 19:185-193.

Eaton L., Patriquin D. (1989) Denitrification in lowbush blueberry soils. Canadian Journal of Soil Science 69:303-312.

Flowers T., Yeo A. (1995) Breeding for salinity resistance in crop plants: where next? Functional Plant Biology 22:875-884.

Focht D. (1974) The effect of temperature, pH, and aeration on the production of nitrous oxide and gaseous nitrogen-a zero-order kinetic model. Soil Science 118:173.

Goodroad L., Keeney D. (1984) Nitrous oxide production in aerobic soils under varying pH, temperature and water content. Soil Biology and Biochemistry 16:39-43.

Hahn J., Junge C. (1977) Atmospheric nitrous oxide- A critical review. Zeitschrift fuer Naturforschung, Teil a 32:190-214.

Hanay A., Bueyueksoenmez F., Kiziloglu F., Canbolat M. (2004) Reclamation of saline-sodic soils with gypsum and MSW compost. Compost Science & Utilization 12:175-179.

Huang Y., Zou J., Zheng X., Wang Y., Xu X. (2004) Nitrous oxide emissions as influenced by amendment of plant residues with different C: N ratios. Soil Biology and Biochemistry 36:973-981.

IPCC. (2007) Agriculture, in: B. Metz, et al. (Eds.), Mitigation, Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, Cambridge University Press, Cambridge, United Kingdom/New York, NY, USA.

Jacobson S.N., Alexander M. (1980) Nitrate loss from soil in relation to temperature, carbon source and denitrifier populations. Soil Biology and Biochemistry 12:501-505.

Knowles R. (1982) Denitrification. Microbiological Reviews 46:43-60.

Lauchli A., Grattan S.R. (1990) Plant responses to saline and sodic conditions, Agricultural salinity assessment and management, ASCE. pp. 169-205.

Laura R. (1974) Effects of neutral salts on carbon and nitrogen mineralisation of organic matter in soil. Plant and Soil 41:113-127.

Liang Y., Si J., Nikolic M., Peng Y., Chen W., Jiang Y. (2005) Organic manure stimulates biological activity and barley growth in soil subject to secondary salinization. Soil Biology and Biochemistry 37:1185-1195.

Maag M., Vinther F. (1996) Nitrous oxide emission by nitrification and denitrification in different soil types and at different soil moisture contents and temperatures. Applied Soil Ecology 4:5-14.

Martikainen P. (1985) Nitrous oxide emission associated with autotrophic ammonium oxidation in acid coniferous forest soil. Applied and Environmental Microbiology 50:1519.

McClung G., Frankenberger W. (1985) Soil nitrogen transformations as affected by salinity. Soil Science 139:405.

McCormick R.W., Wolf D.C. (1980) Effect of sodium chloride on CO_2 evolution, ammonification, and nitrification in a Sassafras sandy loam. Soil Biology and Biochemistry 12:153-157.

McKenney D., Wang S., Drury C., Findlay W. (1993) Dentrification and mineralization in soil amended with legume, grass, and corn residues. Soil Science Society of America Journal 57:1013-1020.

Meng L., Ding W., Cai Z. (2005) Long-term application of organic manure and nitrogen fertilizer on N_2O emissions, soil quality and crop production in a sandy loam soil. Soil Biology and Biochemistry 37:2037-2045.

Moore T., Dalva M. (1993) The influence of temperature and water table position on carbon dioxide and methane emissions from laboratory columns of peatland soils. Journal of Soil Science 44:651-664.

Parton W., Silver W.L., Burke I.C., Grassens L., Harmon M.E., Currie W.S., King J.Y., Adair E.C., Brandt L.A., Hart S.C. (2007) Global-scale similarities in nitrogen release patterns during long-term decomposition. Science 315:361-364.

Pathak H., Rao D. (1998) Carbon and nitrogen mineralization from added organic matter in saline and alkali soils. Soil Biology and Biochemistry 30:695-702.

Rengasamy P. (2006) World salinization with emphasis on Australia. Journal of Experimental Botany 57:1017-1023.

Richards L.A. (1954) Diagnosis and Improvement of Saline and Alkali Soils. Soil Science 78:154.

Rietz D., Haynes R. (2003) Effects of irrigation-induced salinity and sodicity on soil microbial activity. Soil Biology and Biochemistry 35:845-854.

Robertson G.P., Groffman P.M. (2007) Nitrogen Transformations, in: E. A. Paul (Ed.), Soil Microbiology and Biochemistry, Elsevier Academic Press, Burlington, MA. pp. 341-364.

Rolston D.E., Hoffman D., Toy D. (1978) Field measurement of denitrification: I. Flux of N_2 and N_2O. Soil Science Society of America Journal 42:863-869.

Schaufler G., Kitzler B., Schindlbacher A., Skiba U., Sutton M.A., Zechmeister B.S. (2010) Greenhouse gas emissions from European soils under different land use: effects of soil moisture and temperature. European Journal of Soil Science 61:683-696.

Schlentner R.E., Cleve K.V. (1985) Relationships between CO_2 evolution from soil, substrate temperature, and substrate moisture in four mature forest types in interior Alaska. Canadian Journal of Forest Research 15:97-106.

Tejada M., Garcia C., Gonzalez J.L., Hernandez M.T. (2006) Use of organic amendment as a strategy for saline soil remediation: Influence on the physical, chemical and biological properties of soil. Soil Biology and Biochemistry 38:1413-1421.

Vinther F. (1992) Measured and simulated denitrification activity in a cropped sandy and loamy soil. Biology and Fertility of Soils 14:43-48.

Wahid A., Akhtar S., Ali I., Rasul E. (1998) Amelioration of saline-sodic soils with organic matter and their use for wheat growth. Communications in Soil Science and Plant Analysis 29:2307 - 2318.

Weier K.L., Doran J.W., Power J.F., Walters D.T. (1993) Denitrification and the dinitrogen/nitrous oxide ratio as affected by soil water, available carbon, and nitrate. Soil Science Society of America Journal 57:66-72.

Weitz A., Linder E., Frolking S., Crill P., Keller M. (2001) N_2O emissions from humid tropical agricultural soils: effects of soil moisture, texture and nitrogen availability. Soil Biology and Biochemistry 33:1077-1093.

Wong V., Dalal R., Greene R. (2009) Carbon dynamics of sodic and saline soils following gypsum and organic material additions: a laboratory incubation. Applied Soil Ecology 41:29-40.

Yao Z., Zhou Z., Zheng X., Xie B., Mei B., Wang R., Butterbach-Bahl K., Zhu J. (2010) Effects of organic matter incorporation on nitrous oxide emissions from rice-wheat rotation ecosystems in China. Plant and Soil 327:315-330.

Yuan B.-C., Li Z.-Z., Liu H., Gao M., Zhang Y.-Y. (2007) Microbial biomass and activity in salt affected soils under arid conditions. Applied Soil Ecology 35:319-328.

Zheng X., Wang M., Wang Y., Shen R., Gou J., Li J., Jin J., Li L. (2000) Impacts of soil moisture on nitrous oxide emission from croplands: a case study on the rice-based agro-ecosystem in Southeast China. Chemosphere-Global Change Science 2:207-224.

4. EVALUATING THE EFFECTS OF ORGANIC AMENDMENTS ON NITROUS OXIDE AND CARBON DIOXIDE EMISSIONS FROM A SALINE-SODIC SOIL - A FIELD STUDY

Abstract

Incorporation of organic amendments to remediate salt-affected soils has been a common practice to improve the physical, chemical, and biological properties of soil. The high amounts of C and N added with their use may pose a threat to the environment in terms of their potential for enhanced greenhouse gas emissions. A field experiment was conducted to evaluate the influence of different organic amendments varying in composting stages on carbon dioxide (CO_2) and nitrous oxide (N_2O) emission rates from an extreme saline-sodic soil. Four organic waste amendments were selected: active greenwaste (AGW), cured greenwaste compost (CGW), active dairy manure (ADM), and cured dairy manure compost (CDM). All treatments were incorporated into the soil as a single application at 100 Mg ha^{-1}. Gaseous fluxes were measured using a closed chamber method. Organic amendment applications significantly affected CO_2 and N_2O emission rates compared to the control. Carbon dioxide fluxes were significantly increased with the application of AGW and ADM compared to CGW and CDM, respectively. Daily N_2O emissions were decreased with the use of AGW compared to CGW, ADM, and CDM treatments. Average daily emission fractions of N_2O were increased by 66% and 84% in soils incorporated with cured forms compared to active forms of dairy manure

and greenwaste amendments, respectively. Compared to the cured treatments, incorporation of active organic amendments reduced average daily N_2O emission fractions by 42% most likely by reducing the availability of mineral N as evident by significant correlation matrices between N_2O fluxes and soil NO_3^- concentrations. With a global warming potential of 310 times that of CO_2, even a small reduction in N_2O emissions with the use of active organic amendments can be considered as a positive impact on the environment.

Introduction

Soil degradation and salinization are widely recognized as mounting terrestrial sustainability concerns. According to Global Assessment of Human-induced Soil Degradation (GLASOD), nearly 1964 M ha of land is degraded globally. Of these lands, 46% have been identified as moderately impacted and 19% have been greatly degraded (FAO, 2003). The most widespread form of soil degradation is salinization which affects nearly 50% of cropland worldwide (FAO, 2003) and 25-30% of irrigated land in the United States (Wichelns, 1999). Salt-affected soils occur in more than 100 countries, mainly in the arid and semi-arid regions of the world due to the low precipitation rates and elevated evapotranspiration potentials that prevail in these areas.

Salt toxicity not only adversely effects plant growth but also negatively influences the physical, chemical, and biological processes of soil (Rietz and Haynes, 2003; Wahid et al., 1998; Walker and Bernal, 2008; Wong et al., 2009). Increasing soil electrical conductivity (EC) decreases soil enzymatic activities, structural stability, bulk density, soil permeability, and water infiltration (Tejada and Gonzalez, 2006). Because the economic, ecological, and social consequences of soil salinization can be severe, there are widespread efforts to reclaim salt-affected soils that have gone out of production (Lakhdar et al., 2009). A wide array of organic amendments such as manures and composts varying in their characteristics and processing have been evaluated for their effectiveness in remediating salt-affected soils (Lakhdar et al., 2008; Tejada et al., 2006). It is widely known that increases in soil organic C content associated with the application of organic amendments helps to improve soil properties. Both the amount and composition of the incorporated organic amendments influence the extent to which soil physical, chemical, and biological properties are improved. In soils, organic amendments act as substrates that can stimulate microbial activity even under saline conditions (Pathak and Rao, 1998). Several studies have documented increased structural stability and infiltration rates, reduction in exchangeable sodium percentage and bulk density, and improved soil microbial biomass and enzyme activity with the incorporation of organic amendments (Calderón et al., 2004; Liang et al., 2003; Liang et al., 2005; Tejada and Gonzalez, 2006; Tejada et al., 2006). It has been demonstrated that the improvement in the biological properties of soil, particularly microbial biomass and enzyme activity, are responsible for changes in biochemical processes such as N mineralization and nitrification following organic matter additions in saline soils (Lakhdar et al., 2008; Tejada, et al., 2006).

Agricultural soil is a significant source for two of the most important greenhouse gases: carbon dioxide (CO_2) and nitrous oxide (N_2O). Nitrous oxide is a potent greenhouse gas with a 120 year long atmospheric lifetime and a global warming potential (GWP) of 310 (EPA, 2013). In 2011, N_2O emissions accounted for an estimated 3.7% of total greenhouse gas emissions on a mass basis in the United States; of which 69.3% was attributed to agriculture alone, a 8.5% increase compared to that in 1990 (EPA, 2013). However, the accuracy of these estimates is uncertain due to insufficient field measurements and also the high spatial and temporal variability of N_2O fluxes from soils. Globally, N_2O emissions are escalating at an annual rate of 0.2-0.3% and agricultural

N_2O emissions are further expected to increase by 35-60% by 2030 due to increased manure and chemical fertilizer applications essential to feed the ever-growing global population (FAO, 2003).

Increased N_2O emissions from agricultural soils can be largely attributed to the use of inorganic and organic fertilizers resulting in enhanced soil mineral N. Nitrification and denitrification are the two major processes responsible for N_2O emissions in soils. Under anaerobic conditions, N_2O is produced as an intermediate byproduct during the reduction of NO_3^- and or NO_2^- to N_2 by heterotrophic denitrifying bacteria in soils (Robertson and Tiedje, 1987). Furthermore, changes in soil properties such as soil moisture, organic C, and N may also result in production of N_2O via nitrification during oxidation of NH_4^+ to NO_2^- and further to NO_3^- (Castaldi and Smith, 1998).

The effects of incorporating organic waste materials into non-saline soils on N_2O can vary. Studies have shown both increases and decreases in N_2O emissions following incorporation of organic waste materials such as straw, farmyard manure, greenwastes, and composts (Chang et al., 1998; Dalal et al., 2009; Ding, 2007; Huang et al., 2004). In general, organic amendments with low C:N ratios tend to produce greater N_2O emissions due to increased N mineralization than those with higher C:N ratios that cause N immobilization, thus controlling NO_3^- availability for denitrifiers (Huang et al., 2004; Velthof et al., 2003). By contrast, the simultaneous effects of soil salinity and incorporation of organic amendments for the purpose of remediation on N_2O emissions has not been studied previously.

Salt-affected soils often require leaching with large quantity of water to displace excess salts from soil surface to deeper layers. Poor soil structure and low saturated hydraulic conductivity, commonly associated with saline-sodic soils, results in ponding which in turn creates anaerobic conditions suitable for denitrification. Addition of organic amendments to remediate these soils can further promote denitrification, particularly N_2O emissions by supplying organic C, a source of food for most denitrifiers which are heterotrophic.

This field study was intended to validate certain conclusions that have been drawn from laboratory incubation experiments. Laboratory incubation studies (Chapters 2 and 3) found that adding organic amendments to salt-affected soils maintained at 65-70% WFPS under controlled temperature conditions increased N_2O emissions. Results from these 60-day laboratory incubations also showed that application of active materials compared to the cured amendments increased N_2 relative to N_2O emissions due to the presence of high amounts of labile C. Also, low soil mineral N was observed for high C:N materials compared to the low C:N amendments. This trend was consistent for incubations involving both non-saline and saline soils and at varying temperatures suggesting that the availability of labile C and soil NO_3^--N are important factors affecting N_2O production during denitrification. However, it is important to validate these laboratory results with a field study to more fully characterize the benefits of applying active amendments relative to cured amendments. Under field conditions, soils are open to the environment and soil temperatures, wetting patterns, and other factors vary in ways that may not be fully captured using laboratory studies alone. Pulses of high N_2O fluxes observed under field conditions are often attributed to changes in soil physical and chemical properties, crop type, climate, and management practices such as irrigation, N fertilization, and incorporation of crop residues (Bouwman et al., 1993).

This study compares how wet and dry soil moisture conditions influence CO_2 and N_2O fluxes from a saline-sodic soil following organic amendment applications. In the field, soil temperature and moisture conditions change with consequent effects on the C and N dynamics of the added amendments. Wet conditions can increase N_2O fluxes but the magnitude to which this occurs is controlled by many factors including the availability of C within the feedstock, a property that varies depending on the processing stage of applied organic amendments. Artificial wetting patterns were used in this study to simulate irrigation or precipitation events. However, continuous leaching was not conducted during this study as the goal was to monitor and validate the results found in the laboratory and not to remediate these soils.

The primary objective of this research was therefore to evaluate CO_2 and N_2O emissions from naturally developed saline-sodic soils treated with organic amendments such as manures and composts applied for the purpose of soil remediation. Specific objectives of this study were:

a) to assess whether active amendments decrease N_2O emissions from an extremely saline-sodic soil under varying environmental conditions; and
b) to investigate how CO_2 and N_2O fluxes respond to changes in soil moisture and temperature conditions.

Based on my previous laboratory findings, I hypothesize that incorporation of organic amendments to saline-sodic soils in the form of active manures as alternative to cured composts will reduce N_2O emissions and increase CO_2 fluxes by supplying more readily available labile C. I also hypothesize that low gaseous emission rates will be detected during hot and dry periods compared to the cool and wet periods.

Materials and methods

Site Description and Experimental Treatments

The field experiment was conducted from April, 2012 to August, 2013 at a fallow site in San Jacinto, California (33º 50' 44.2"N, 117 º 1' 39.6"W). The site is located in Southern California and this soil is characterized as a Grangeville coarse-loamy, mixed, superactive, thermic fluvaquentic haploxeroll (NRCS, 2012). The climate is semi-arid with an annual average maximum temperature of 27.2 °C, annual average minimum temperature of 9.1 °C, and average annual precipitation of 30.9 cm (WRCC, 2013). The general properties of this soil (0-25 cm) are shown in Table 4.1. The total C and N in the treatments were analyzed using a Thermo Electron Corporation NC soil analyzer, organic matter was determined by loss on ignition method, EC and pH were determined from saturation paste extracts, and mineral N was analyzed in 1:5 extracts using 2M KCl solution. All soil analyses were performed as per the guidelines of standard soil sampling and methods of analysis (Carter, 1993).

The organic amendments considered in this study were: active greenwaste (AGW), cured greenwaste compost (CGW), active dairy manure (ADM), and cured dairy manure compost (CDM). Table 4.2 reports key properties of organic materials used in this study which were analyzed using standard procedures described in the Test Methods for the Examination of Composting and Compost (TMECC) (Thompson et al., 2001). The field experiment consisted of 5 treatments: four organic amendments and an unamended control, arranged in a randomized complete block design with six replications. Plots were 2x2 m^2 in size. All organic amendments were applied at 100 Mg ha^{-1} (dry weight) except for the control which was unamended. Organic amendments were spread manually on 8 April, 2012 and immediately incorporated into the soil to a depth of 15 cm with a rotary tiller.

Gaseous Flux Measurements

Carbon dioxide and N_2O fluxes were monitored monthly from May 2012 through August 2013. Some measures were conducted under as-found moisture conditions while for others soils were first wetted. In 2012, gaseous fluxes were measured under unwetted conditions on: May 17, June 25, July 27, August 8, September 22, October 4, October 30, November 16, and December 12. During 2013, unwetted condition measurements for N_2O and CO_2 fluxes were conducted on: January 24, March 27, April 18, June 25, and July 23. Prior to sampling on seven additional dates, wetted conditions for sampling air were created by filling the chambers with 0.75 L of de-ionized water simulating a 1.5 cm irrigation event one day before sampling to better understand the influence of irrigation or abrupt soil moisture changes on gaseous fluxes. Measurements under artificially wetted conditions were conducted in 2012 on: August 29, September 24, October 5, October 19, November 1, November 9, and November 20. It is a common practice to flood salt-affected soils to a depth of 1-2 cm to leach the salts in the cool season during the reclamation process (Bruce Scott, Farmer, 15 May, 2012).

Closed chambers built as per the guidelines provided by Parkin and Venterea (2010) were used to measure trace gas emissions. Briefly, chamber anchor and tops were made using polyvinyl chloride (PVC) pipes. Anchors were made by cutting a PVC pipe section measuring 240 mm i.d x 6.3 mm wall thickness x 152.4 mm deep. One anchor was inserted 10 cm deep into the soil in the center of each plot using a back-hoe 30 days after incorporating the organic amendments and remained undisturbed throughout the study. Chamber tops were made from an identical cylindrical PVC pipe (240 mm i.d., 183 mm height, 6.3 mm wall thickness) and covered with 6.5 mm thick PVC sheet. A 20 mm i.d. sampling port was installed in the center of each chamber top. During sampling, the chamber top was sealed to each anchor with a 10 cm rubber ring which formed a gas-tight seal and butyl rubber septa were used to seal the sampling port.

Before collecting air samples, all chambers were closed for 1 hr. Gas samples were taken immediately after enclosure and again after 1 hr of sealing. Samples were taken using airtight 60 mL syringes and transferred to 20 mL pre-evacuated borosilicate glass vials. After collecting the second sample, the chamber was immediately opened to establish ambient air condition within the chamber (Parkin and Venterea, 2010).

All air samples were analyzed for CO_2 and N_2O within 48 hrs of collection. Samples were analyzed for CO_2 using infrared gas analyzer (Model EGM-4, PP-Systems, Amesbury, Massachusetts) while N_2O was measured using HP 5790 A Gas Chromatograph (Hewlett- Packard Co., Fullerton, California) fitted with 80/100 mesh Porapak Q column and ^{63}Ni electron capture detector. Oven and injection temperatures were set at 50 ºC and a detector temperature of 300 ºC was maintained (Poth and Focht, 1985). Standards were injected after every 10 samples to assure instrument precision.

Two gas measurements were made in the headspace of each chamber at the start and end of 1 hr of closing and fluxes of CO_2 and N_2O were estimated as the difference in gas concentration over time. Chamber heights were measured at each sampling and volume of each chamber was calculated. Average daily CO_2-C and N_2O-N emissions were calculated as the average of daily measurements across the duration of this study.

Average Daily Nitrous Oxide Emission Fractions

Average daily nitrous oxide emission fractions (AEF) for treatments used in this study were calculated as the fraction of applied N that was lost as observed average daily N_2O emissions from organic amendments using the following equation:

$$AEF = \frac{(E_t - E_c)}{N_a} \times 100 \qquad \text{Equation 2}$$

where, *AEF* is the average daily N_2O-N emission fraction (g kg^{-1}), E_t is the average daily N_2O-N emission (g N ha^{-1}) from organic waste treated plot, E_c is the average daily N_2O-N emission (g N ha^{-1}) from untreated control plot, and N_a is the N applied as organic amendments expressed in kg N ha^{-1}.

Soil Sampling and Analyses

Soil samples were collected from 0-10 cm depth from outside each chamber on the day of each gas sampling in plastic bags and stored at 4°C until further analysis. At the beginning of the irrigation events, soil samples were collected from a 25 x 25 cm^2 area 0.5 m north of the anchors which was also wetted to create soil conditions similar to that inside the anchors.

Soil mineral N was extracted by shaking 10 g of field-moist soils with 50 mL of 2M KCl for 1 hr. Subsequently, the extracts were obtained by filtering the soil slurry through a Whatmann no. 42 paper and analyzed for NO_3^--N and NH_4^+-N concentrations by colorimetry (Carter, 1993).

Soil temperature (0-10 cm) was monitored in five randomly selected plots at the time of gas sampling. Gravimetric water content was determined by drying a sub-sample at 105 ºC for 24 hr. Soil bulk density was determined from four undisturbed soil core samples. Soil water content represented as water-filled pore space (WFPS) was determined using Eq. 1 as described by Carter (1993) to understand the effect of varying soil moisture conditions on N_2O emissions.

Data Analysis

One-way analysis of variance (ANOVA) was conducted to test significant differences in average daily CO_2 and N_2O emissions, and average soil NH_4^+-N and NO_3^--N concentrations between various treatments. Significant differences, if any, between various organic amendment treatments were separated using Tukey’s multiple comparison tests. Linear and multiple regression analysis were performed to identify parameters that significantly regulated gas fluxes. Log-transformation of N_2O and NO_3^- data was performed to improve the distribution of residuals. All statistical analysis was performed using SPSS 20.0 and graphs were prepared using SigmaPlot 10.0 graphing software.

Results

Average annual air temperature, precipitation, and irrigation information at the field site from April 2012 through July 2013 are presented in Fig. 4.1. Soil temperature and soil moisture contents during the sampling events are presented in Fig. 4.2. Soil temperatures ranged from 16-27°C from May through September 2012 and June-July 2013 and gradually decreased from October 2012 through April 2013. Under unwetted conditions, soil moistures ranged from 11-35% WFPS. After wetting, soil temperatures were low (8.1-15.3 °C) and soil moisture contents were 68-77% WFPS.

Gaseous Emissions

Measured Daily Fluxes

Daily CO_2 and N_2O fluxes exhibited strong spatial and temporal variations (Figs. 4.3 and 4.4) throughout this study. Application of organic amendments showed significant positive effect ($P < 0.05$) on CO_2 and N_2O fluxes. Across both unwetted and wetted conditions, the daily N_2O fluxes of active materials ranged from 0.02-68.6 g N ha^{-1} day^{-1} while those from cured amendments ranged from 0.06-88.7 g N ha^{-1} day^{-1}. The daily CO_2 fluxes of active materials ranged from 0.05-24.9 kg C ha^{-1} day^{-1} while those from cured amendments ranged from 0.01-23.6 kg C ha^{-1} day^{-1}.

The highest fluxes were observed soon after artificial wetting periods in dairy manure treatments followed by the greenwaste amendments and the peak heights decreased over the course of the wetting period (Figs. 4.3 and 4.4). In general, daily N_2O emission rates were lowest in AGW and highest in CDM treatments. The order of treatments for most daily N_2O emission rates was: CDM > ADM > CGW > AGW while those for CO_2 fluxes were: AGW > ADM > CDM > CGW.

Under unwetted conditions, N_2O fluxes were detected only from May-September 2012 and June-July 2013. During these periods, addition of organic amendments significantly increased ($P < 0.01$) N_2O emissions compared to the controls. However, at each of the sampling days during this study, no significant differences in N_2O fluxes between treatments were observed with the exception of AGW, the smallest contributor.

Highest N_2O fluxes were recorded on August 8 and September 22, 2012 (Fig. 4.4). Increases in CO_2 and N_2O fluxes for the enclosure period (1 hr) were not detected on October 4, October 30, November 16, and December 12, 2012 or on January 24, March 27, and April 18, 2013 and were considered as outliers.

After wetting, CO_2 and N_2O fluxes were significantly increased by organic amendments compared to the controls on all sampling days. Carbon dioxide fluxes were significantly increased in the AGW and ADM treatments compared to the CGW and CDM amendments. Fluxes of CO_2 ranged from 9.3- 24.9 kg C ha^{-1} day^{-1} for active materials and 4.0-18.9 kg C ha^{-1} day^{-1} for cured treatments. Significant ($P < 0.05$) differences between treatments on N_2O emission rates occurred only on October 5, November 1, and November 20, 2012. On these sampling occasions, active treatments significantly lowered N_2O emissions compared to the cured treatments.

Average Daily Emissions and Average Daily Emission Fractions

Average daily N_2O and CO_2 emissions are presented in Table 4.3. Average daily CO_2 emissions were significantly greater from the active treatments compared to the cured amendments. With the active amendments, use of AGW increased average daily CO_2 emissions by 42% relative to CGW while this increase was 62% with the application of ADM compared to CDM. Feedstock types also mattered as incorporation of AGW and CGW increased the average daily CO_2 emissions by 29% and 47% relative to ADM and CDM amendments, respectively.

The lowest average daily N_2O emissions were detected from the AGW treatment and CDM produced the highest average daily N_2O emissions during this study. Average daily N_2O emissions were decreased from active treatments compared to their respective cured amendments (Table 4.3). Incorporating AGW and CGW significantly decreased average daily N_2O emissions by 64% and 34% compared to ADM and CDM, respectively. Between the dairy manure amendments, ADM decreased average daily N_2O emissions by 22%. Similar reduction in N_2O emissions were observed with the AGW treatment, with a decrease of 58% relative to the CGW treatment.

Average daily N_2O emission fractions (AEF), the proportion of applied N released as average daily N_2O from treatments alone, are also listed in Table 4.3. The mean annual AEF’s of N_2O were on an average increased 1.7 fold from cured treatments compared to the active amendments. Dairy manure treatments contributed significantly ($P = 0.03$) more N_2O than the greenwaste amendments. Treatments ADM and CDM increased AEF’s by 42% and 27% compared to AGW and CGW amendments, respectively. Between the greenwaste amendments, CGW increased AEF’s by 84% relative to AGW. Similarly, CDM enhanced AEF by 66% compared to ADM treatments, among the dairy manure amendments.

Soil Mineral N

Mean soil NH_4^+ and NO_3^- concentrations (0-10 cm) are shown in Table 4.3. Application of organic amendments significantly affected ($P < 0.05$) soil NO_3^--N and NH_4^+-N concentrations compared to the control. No significant ($P < 0.05$) differences between treatments were observed for average soil NH_4^+ levels. Soil NH_4^+ concentrations for the controls were in the range of 0.07-0.82 kg N ha^{-1} while those for active amendments ranged from 0.19-13.58 kg N ha^{-1} and for cured treatments from 0.73-16.88 kg N ha^{-1}. Compared to treatments AGW and ADM, average soil NH_4^+ levels with the incorporation of CGW and CDM amendments increased by 15% and 53%, respectively.

Treatment effects on NO_3^--N concentrations were significant ($P < 0.05$) throughout this study. The soil NO_3^--N concentrations were significantly higher after the addition of amendments, with 2.3 fold and 1.8 fold increases observed in the dairy manure and greenwaste amended plots over those of the control, respectively. The NO_3^--N concentrations for CDM and CGW plots were on an average 16% and 12% higher than for the ADM and AGW treatments respectively, over the course of this study (Table 4.3). Lower NO_3^--N concentrations were recorded for greenwaste treatments than for the dairy manure amendments. Soil NO_3^--N concentrations for ADM and CDM treatments were on an average 33% higher than those in AGW and CGW treatments.

Relationship between Daily N_2O Emission Rates and Soil Properties

In general, daily N_2O fluxes (log-transformed) were significantly and positively correlated with soil moisture, soil NO_3^--N concentrations, and daily CO_2 emissions (Table 4.4). A combination of soil moisture and NO_3^- could explain 66% of variability in N_2O fluxes. Stepwise regression analysis performed separately for active and cured treatments identified a combination of soil NO_3^--N and CO_2 emissions responsible for 48% and 32% of N_2O flux variations from active and cured treatments, respectively (Table 4.5).

Some relationships applied only to wetted or unwetted conditions. No significant ($P < 0.05$) relationship between N_2O flux and soil temperature was observed when the data for unwetted and wetted conditions were considered together but the rate of N_2O emissions (log-transformed) were significantly correlated with soil temperature ($P < 0.05$) when unwetted conditions were considered alone. No significant relationship between soil temperature and N_2O emission rates (log-transformed) existed under artificially wetted conditions. Within the observed WFPS range (15-95%), emission rates of N_2O (log-transformed) increased with increasing WFPS with a significant correlation.

Under unwetted conditions, rates of N_2O emission (log-transformed) were significantly ($P < 0.01$, $n = 168$) but weakly ($0.37 < r < 0.42$) correlated with soil moisture and NO_3^--N concentrations (log-transformed). In case of the wetted conditions, fluxes of N_2O (log-transformed) were significantly and positively ($P < 0.01$, $n = 168$) correlated with soil moisture and NO_3^--N concentrations but also weakly and negatively with CO_2 emissions.

Discussion

Effect of Organic Amendments on Gaseous Emissions

Incorporating organic amendments increased soil CO_2 fluxes. Significantly higher emissions were observed from active treatments than cured amendments, an observation consistent with the material stability measures (Table 4.2). These results are consistent with those observed in the incubation experiment in Chapter 3 at 15, 25, and 35°C temperatures where in AGW and ADM significantly enhanced CO_2 production compared to the CGW and CDM amendments. Although these incubated and field soils had higher carbonate content, chemical release of CO_2 caused by the dissociation of soil carbonates can be excluded from this study. This is because the soil pH (8.1) of this soil was well above the threshold value of 7.2 for carbonate dissociation.

Daily fluxes of N_2O under both unwetted and wetted conditions did not differ significantly between CGW, ADM, and CDM treatments (Fig. 4.4). Lower N_2O fluxes were recorded from AGW throughout the study suggesting that soil NO_3^- levels were too low and or that the N_2O produced was converted to N_2 due to the availability of highly labile C. Both these possibilities cannot be excluded as I found similar results in Chapter 3 from AGW amended soils that had low soil NO_3^- concentrations which resulted in decreased N_2O production but enhanced N_2 emissions at 25 and 35°C. In this field study, CGW and ADM resulted in similar N_2O fluxes. Contrastingly, ADM resulted in greater N_2O emissions than CGW at higher temperatures of 25 and 35°C under laboratory conditions. Daily N_2O emissions under different treatments in this study were in the order of CDM ≥ ADM ≥ CGW > AGW. In contrast, the general trend in the 60-day incubation study was of the order ADM > CDM > CGW > AGW which suggests that the availability of NO_3^--N for denitrification from the CDM was greater than ADM under field conditions. Losses were highest from the dairy manure treatments as N was in readily available NO_3^- form for denitrifying bacteria as against the greenwaste amendments. Although, soil NO_3^- concentrations did not differ significantly from October through December 2012 and January through May 2013, the gas fluxes that were considered outliers (zero flux measurements) during this study were likely due to the combined effect of very low soil moisture (11-18 % WFPS) and temperature conditions (8.1-11.6°C). Gas fluxes were increased again in June and July 2013 corresponding to 5 and 0.4 mm rainfall that occurred on June 22 and July 18, 2013, respectively (Fig. 4.1).

Average daily N_2O emissions from ADM was 30.28 g N ha^{-1}, which is significantly lower than the mean daily N_2O loss of approximately 195 g N ha^{-1} reported by Chang et al. (1998) from a non-saline soil incorporated with feedlot manure applied at 104 Mg ha^{-1} on a dry weight basis. Average daily N_2O emissions from cured organic amendments were significantly higher compared to the active treatments suggesting that the availability of NO_3^--N, a substrate for denitrification, to soil microorganisms was a driving factor. Compared to CDM, ADM decreased average daily N_2O emissions by only 22% as opposed to a 58% reduction by AGW relative to CGW (Table 4.3). This suggests that the dairy manure treatments were more prone to mineralization than the greenwaste amendments which likely have high lignin content.

Nitrous oxide in soils is primarily produced by microbial mediated soil processes of nitrification and denitrification which may be altered by organic amendment applications (Ma et al., 2012). Khalil et al. (2002) reported negative correlation between C:N ratios of organic amendments and N_2O emissions. Application of organic amendments with higher C:N ratios promote N immobilization thereby resulting in lower N_2O emissions (Baggs et al., 2000; Huang et al., 2004). Soil NO_3^--N concentrations in the AGW and ADM (higher C:N) treatments were comparatively lower than the cured CGW and CDM (lower C:N) amendments (Table 4.3). As expected, ADM and CDM with much lower C:N ratios compared to the AGW treatment promoted N_2O emissions (Table 4.2). Also, at 43, the AGW C:N ratio was slightly above the threshold value of 40 for stimulating N mineralization (Vigil and Kissel, 1991). Despite this, AGW generated NO_3^- and produced N_2O emissions. This is likely due to the decay of labile C fraction with a higher C:N ratio which would release NO_3^- and supply the denitrifiers with energy (Flessa and Beese, 1995).

Results from Chapter 2 using soils varying in salinity suggest that increasing soil salinity increased N_2O emissions and decreased N_2 production from the added organic amendments. The soil considered in Chapter 3 and this field study represents an extreme saline-sodic condition with an average EC_e 104 dSm^{-1} and SAR 148. When remediated with organic amendments, EC and SAR values are expected to decrease over time (Tejada et al. 2006; Tejada and Gonzales, 2006). Therefore, N_2O emissions from these soils are likely to decrease promoting gaseous N losses in the form of N_2 from active organic amendments containing high amounts of labile C. However, N_2O emissions from salt-affected soils undergoing remediation with organic amendments merits further study as the comprehensive effect of temporal variations in soil properties on greenhouse gas emissions must be evaluated.

Average Daily Emission Fractions of Nitrous Oxide

The applied N sources increased N_2O emissions. The average daily N_2O emission fractions for all organic amendments applied at 100 Mg ha^{-1} in this study fall in the range of 0.01-0.03 g kg^{-1}. These values were also lower than those calculated using Eq.2 from the results reported by Chang et al. (1998), which indicate an estimated AEF of N_2O up to 1.8 g kg^{-1} in non-saline soils from feedlot manure applied at 104 Mg ha^{-1}. The AEF values of AGW and CGW treatments observed in this study were also significantly lower than 0.08-0.11 g kg^{-1} straw-induced range calculated using Eq.2 using the results reported by Li et al. (2013). Although 28% more N was applied with ADM compared to CDM, the AEF of N_2O from ADM was 40% lower than CDM. In contrast, similar reduction in the AEF of N_2O from AGW was observed even though N applied from CGW was 41% higher (Table 4.3). Two considerations arise when evaluating daily N_2O emission rates from organic amendments. First, the added N is not readily available and is immobilized by soil microbes, thus reducing N_2O emissions. Second, the labile C present in organic materials can induce ideal conditions for denitrification and produce greater N_2 emissions. Both these possibilities were demonstrated in Chapter 3 using the same saline-sodic soil and organic amendments applications exposed to temperatures similar to those observed in this study. Despite the use of average daily N_2O emissions from observations instead of annual N_2O emissions to calculate emission fractions, these values can be used as indicators to predict the potential of the tested organic amendments on N_2O emissions. However, the quantitative effects of the added amendments on long-term N_2O emissions and associated emission factors needs further evaluation.

Effect of Soil Properties on Carbon Dioxide and Nitrous Oxide Emissions

Daily N_2O fluxes increased with increasing soil temperatures from 16-28 °C. The low correlation coefficient between N_2O fluxes and soil temperature (Table 4.4) under unwetted conditions is likely due to the association of heat with dry conditions (Table 4.4). This finding is in contrast to that observed under laboratory conditions with the same soil maintained at 70% WFPS wherein soil temperature strongly and positively correlated ($r^2 = 0.76$; $P < 0.01$) with N_2O fluxes suggesting that other environmental factors besides temperature may have influenced N_2O emission rates in the field.

N_2O production occurs via nitrification and denitrification processes in soil. That there was no significant relationship between the amount of NH_4^+-N in the soil and the rate of N_2O emissions, suggests that the role of nitrification in N_2O production can be deemphasized under both wetted and unwetted conditions. Soil NO_3^--N, a substrate for denitrification, had a significant positive correlation with N_2O emissions, suggesting that denitrification is the dominant process for N_2O production in these saline-sodic soils (Table 4.4). Denitrification is an anaerobic process but it can still occur under aerobic soil conditions when there are anaerobic microsites (Renault and Stengel, 1994). This soil had relatively little structure through which to transmit O_2 and anaerobic microsites would likely develop in the vicinity of decaying amendment particles. Dairy manure amendments with high initial N contents and low C:N ratios released greater soil NO_3^- concentrations compared to greenwaste amendments, particularly AGW which tended to immobilize N. ADM and CDM consistently released N but AGW and CGW immobilized N from May through November, 2012 of this study.

Besides N availability, the artificial wetting of collars enhanced CO_2 and N_2O emissions from all organic amendment treated plots (Fig. 4.3 and 4.4). Nitrous oxide fluxes strongly correlated with WFPS for active and cured amendments and weakly correlated with moisture under wetted and unwetted conditions. This is because the range of WFPS measures was much narrower under unwetted (11-37%) and wetted (55-92%) conditions compared to active (11-82%) and cured (11-92%) amendments.

Correlations observed between N_2O fluxes and CO_2 emissions were weaker than between N_2O emission rates with soil moisture. Carbon dioxide fluxes positively correlated with N_2O emission rates for active and cured amendments. However, CO_2 emissions correlated weakly and negatively with N_2O fluxes under unwetted and wetted conditions. Because respiration is required for both the creation of N_2O and its further reduction to N_2, respiration rates alone would be expected to correlate less strongly with overall N_2O losses (Table 4.4). Although weaker, these positive correlations are consistent with findings from other authors who reported positive correlations between CO_2 and N_2O emissions both under field and laboratory studies (Ding, 2007; Groffman and Tiedje, 1991; Huang et al., 2004). Positive correlation between N_2O fluxes with both CO_2 emission rates and soil moisture likely reflects correlation between CO_2 and WFPS.

Soil moisture significantly correlated with CO_2 fluxes for all treatments under both wetted and unwetted conditions. Significant ($P < 0.01$) correlations between WFPS and CO_2 emissions for active ($r^2 = 0.64$) and cured ($r^2 = 0.37$) organic amendments (data not shown) suggests that increases in soil moisture content likely enhanced greater C decomposition from AGW and ADM compared to CGW and CDM treatments, respectively. Decomposition of organic materials can further promote denitrification “hot spots” by depleting oxygen (Velthof et al., 2003). Furthermore, the possibility that lower N_2O fluxes from active treatments (particularly AGW) caused by a higher N_2 to N_2O ratio resulting from the availability of labile C to heterotrophic denitrifiers, cannot be excluded (Granli and Boeckman, 1994). This possibility had been demonstrated from the results in the laboratory experiments using the same saline-sodic field soil (Chapter 3) which showed that N_2 emissions were significantly greater from active compared to cured amendments and greater N_2 to N_2O ratios were recorded for AGW alone at all three temperatures throughout the incubation period.

Under unwetted and wetted conditions, WFPS weakly correlated with N_2O emission rates due to less variability in WFPS measured within each group (Table 4.4). Nitrous oxide was negatively correlated with CO_2 fluxes under unwetted and wetted conditions as increased CO_2 emissions may have driven the conversion of N_2O to N_2, thus completing denitrification. This observation is consistent with the findings from laboratory incubations (Chapters 2 and 3) wherein treatments producing greater CO_2 emissions produced higher N_2 emissions and N_2 to N_2O ratios with varying salinity levels from 2.8-30.6 dS m^{-1} at 25°C and with EC_e 100 dS m^{-1} at 15, 25, and 35°C temperature levels.

To examine the factors governing N_2O fluxes separately under unwetted and wetted conditions, multiple regression analysis was performed with soil properties such as temperature, moisture, NO_3^--N, and CO_2 emission rates. In this study, biophysical factors such as soil temperature, WFPS, NO_3^--N, and CO_2 emissions could explain only 66% of the variability in N_2O emission rates (Table 4.5) suggesting that a combination of other biophysical factors including enzyme activities likely altered by high application rate of organic amendments may be responsible for N_2O fluxes from these saline-sodic soils.

The high spatial and temporal variability observed in field N_2O emissions in this study is consistent with findings from other authors (Dalal et al., 2003; Granli and Boeckman, 1994). Parton et al. (2001) attributed this to the complex multiplicative rather than the additive effects of the biophysical factors on N_2O emissions from soils. Monitoring other additional soil properties such as dissolved organic C, microbial populations, and their enzyme activity may provide a better understanding of the factors that govern N_2O emissions in highly saline-sodic conditions as observed in this study. The extent of N_2O dissolution in the soil solution and its rate of diffusion to the soil surface are additional parameters that may explain some of the variability of N_2O

production rates unaccounted for under field conditions (Jury et al., 1982). Thus, although N_2O emission rates are regulated by various biophysical properties in soil, close relationship between soil N_2O production and N_2O emission rates may rarely be achieved.

Conclusions

Incorporation of different organic amendments to saline-sodic soils significantly affected soil N_2O and CO_2 emissions. The amount and quality of organic C they supplied controlled the production of N_2O and CO_2 by regulating the soil C and N dynamics. Dairy manure applications to soil in both active and cured forms released significantly higher amounts of mineral N than the greenwaste treatments which was either accumulated as NH_4^+-N or converted to NO_3^--N in the soil causing ADM and CDM to increase N_2O emissions compared to the AGW and CGW treatments. Compared to the dairy manure treatments, approximately 34-64% reduction in average daily N_2O emissions was achieved with greenwaste amendments even at an application rate of 100 Mg $ha^{-1}$and their average daily N_2O emission fractions ranged from 0.01-0.03 g kg^{-1}. Additionally, soil moisture, NO_3^--N, and CO_2 emissions explained 66% of the variations in daily N_2O fluxes.

I conclude that active amendments, particularly AGW, proved to be efficient in reducing N_2O emissions compared to the cured treatments. Results from this study confirm the findings of the 60-day laboratory incubations at varying temperatures suggesting that remediation with greenwaste amendments seems to be a valuable substitute for dairy manure treatments because of a low average daily emission fraction as well as low soil NO_3^- concentrations. Based on the findings from Chapter 2, I also contend that organic amendment induced N_2O emissions will be lower from salt-affected soils with low electrical conductivity values than those observed in this study representing extreme saline-sodic conditions. However, a more comprehensive long-term evaluation on the effect of these amendments on remediating salt-affected soils needs to be considered. Further site-specific research is also needed to accurately compare the tested treatments under varying salinity conditions with extended flux measurements to include long-term impacts on associated greenhouse gas emissions.

Table 4.1: Physical and chemical properties of soil (0-25 cm)

Parameter	Value
Sand (%)	43.12
Silt (%)	37.43
Clay (%)	19.45
Bulk Density (g cm^{-3})	1.33
EC_e (dS m^{-1})	107.61
pH	8.14
Total C (%)	1.27
Organic C (%)	0.92
Total N (%)	0.08
NO_3^--N (mg kg^{-1})	198.28
NH_4^+-N (mg kg^{-1})	1.46
Sodium Adsorption Ratio	148.36

Table 4.2: Composition of organic amendments expressed on a dry weight basis.

Parameter	AGW	CGW	ADM	CDM
Total C (%)	33.41	30.54	26.63	18.10
Total N (%)	0.78	1.10	1.72	1.34
C:N	42.91	27.81	15.59	13.62
NO_3^--N (mg kg^{-1})	2.36	6.03	12.56	33.11
NH_4^+-N (mg kg^{-1})	15.02	28.61	132.81	85.67
Organic Matter (%)	73.64	65.05	62.24	37.98
Stability Index (mg CO_2-C g OM^{-1} day^{-1})	5.33	2.15	4.92	1.87
Particle Size Distribution		% by Weight		
> 15.8 mm	18.40			
9.5-15.8 mm	19.24			
6.3-9.5 mm	12.31		10.78	
4.0-6.3 mm	15.33	100	89.22	100
2.0-4.0 mm	10.83			
< 2.0 mm	23.34			

Table 4.3. Nitrogen applied, average soil NH_4^+-N and NO_3^--N (0-10 cm) concentration, average daily gaseous fluxes, and average daily N_2O emission fractions (mean ± std. error) for different organic amendments applied at 100 Mg ha^{-1} to an extreme saline-sodic soil.

Treatment	N Applied (kg ha^{-1})	Average NH_4^+-N (kg ha^{-1})	Average NO_3^--N (kg ha^{-1})	Average Daily CO_2-C (kg ha^{-1})	Average Daily N_2O-N (g ha^{-1})	Average Daily N_2O Emission Fraction (g kg^{-1})
Control	0	0.36 ± 0.23 a	138.92 ± 25.39 a	1.69 ± 0.14 a	1.37 ± 0.21 a	-
AGW	780	1.64 ± 0.61 b	236.87 ± 38.66 b	12.81 ± 1.02 d	10.77 ± 0.94 b	0.012 ± 0.02 a
CGW	1100	1.88 ± 0.97 b	267.33 ± 31.54 c	9.04 ± 0.83 c	25.73 ± 5.03 c	0.022 ± 0.06 b
ADM	1720	5.26 ± 1.11 c	306.51 ± 30.92 d	9.95 ± 0.95 c	30.28 ± 2.67 c	0.017 ± 0.04 ab
CDM	1340	8.08 ± 1.53 c	336.39 ± 42.86 e	6.13 ± 1.08 b	38.82 ± 6.24 d	0.028 ± 0.07 c

Note: Different letters within the same column denote significant ($P < 0.05$) differences between treatments.

Table 4.4: Pearson correlation coefficients between daily N_2O emission rates and soil properties.

Treatment	n	CO_2-C	NO_3^--N	WFPS	Soil Temperature
Active Amendments	168	0.59^{**}	0.37^{**}	0.79^{**}	-0.43^{*}
Cured Amendments	168	0.44^{**}	0.35^{**}	0.85^{**}	-0.31^{*}
Unwetted Conditions	168	-0.22^{*}	0.37^{*}	0.42^{*}	-0.28^{*}
Wetted Conditions	168	-0.23^{*}	0.49^{*}	0.47^{*}	ns
Combined Data	336	0.42^{**}	0.39^{*}	0.75^{**}	ns

Note: ns, *, ** represent non-significant and significant correlation at $P < 0.01$ and $P < 0.05$, respectively.

Table 4.5: Stepwise regression analysis of daily N_2O emission rates with soil properties.

Treatment	Regression Equation	Combined r^2 (adj)	P-Value
Active Amendments	$- 7.764 + 2.853 \text{ Ln } NO_3^- + 0.158 \text{ } CO_2$	0.480	< 0.01
Cured Amendments	$1.484 + 1.465 \text{ Ln } NO_3^- + 0.086 \text{ } CO_2$	0.319	< 0.05
Unwetted Conditions	$-1.226 + 1.712 \text{ Ln } NO_3^- - 0.207 \text{ } CO_2 + 7.637 \text{ WFPS}$	0.317	< 0.01
Wetted Conditions	$5.957 + 0.701 \text{ Ln } NO_3^- + 1.34 \text{ WFPS}$	0.301	< 0.01
Combined Data	$-2.478 + 1.706 \text{ Ln } NO_3^- + 5.738 \text{ WFPS}$	0.665	< 0.05

Note: Units for selected variables are reported as: Soil NO_3^- concentrations (mg N kg^{-1}), daily CO_2 emission rates (kg C ha^{-1} day^{-1}) and water-filled pore space (%).

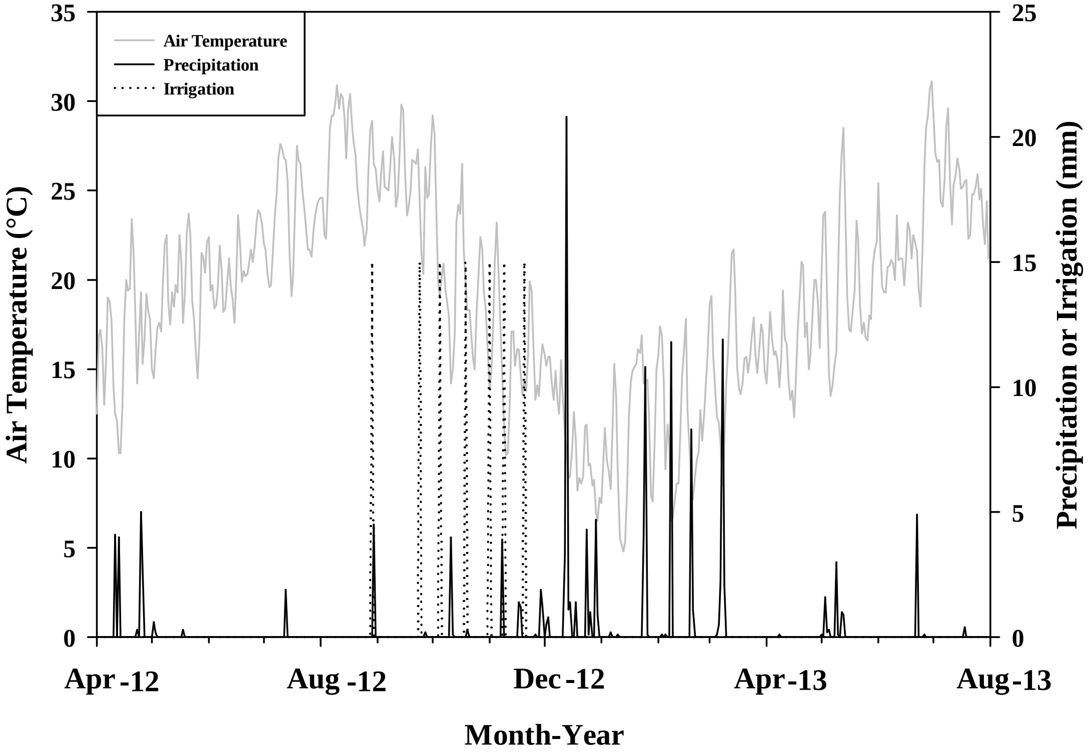

Fig. 4.1. Average Air Temperature, Precipitation and Irrigation at the field site from April 2012 through July 2013.

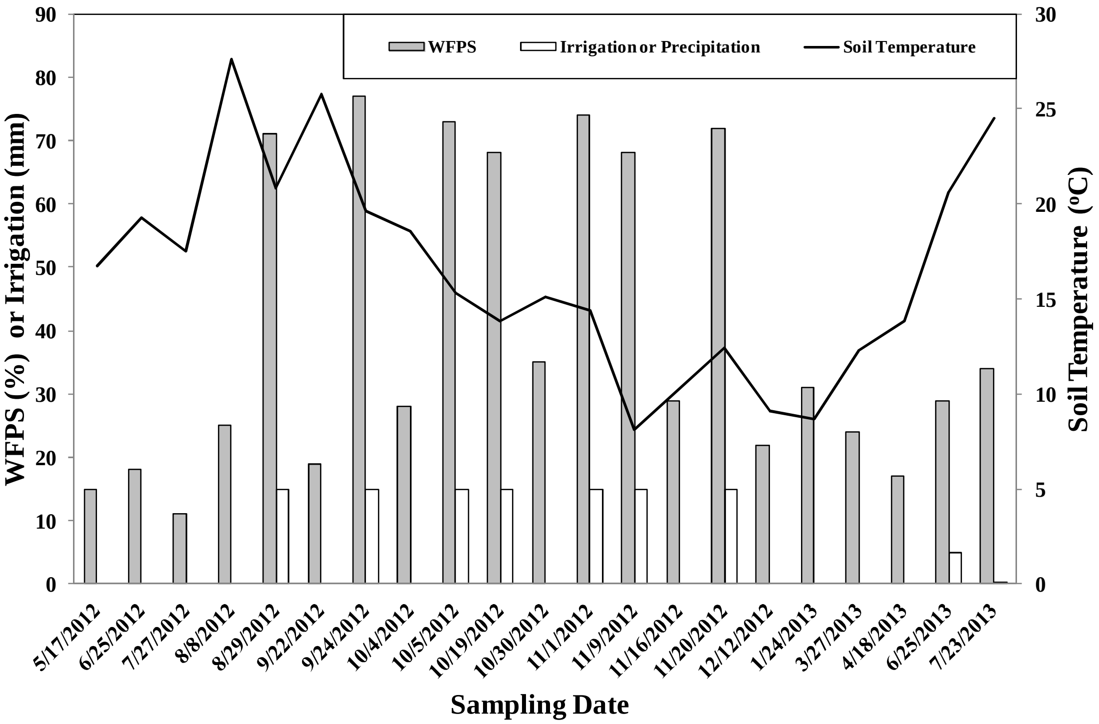

Fig. 4.2. Soil Temperature, Water-Filled Pore Space, and Irrigation or Precipitation corresponding to gas sampling events.

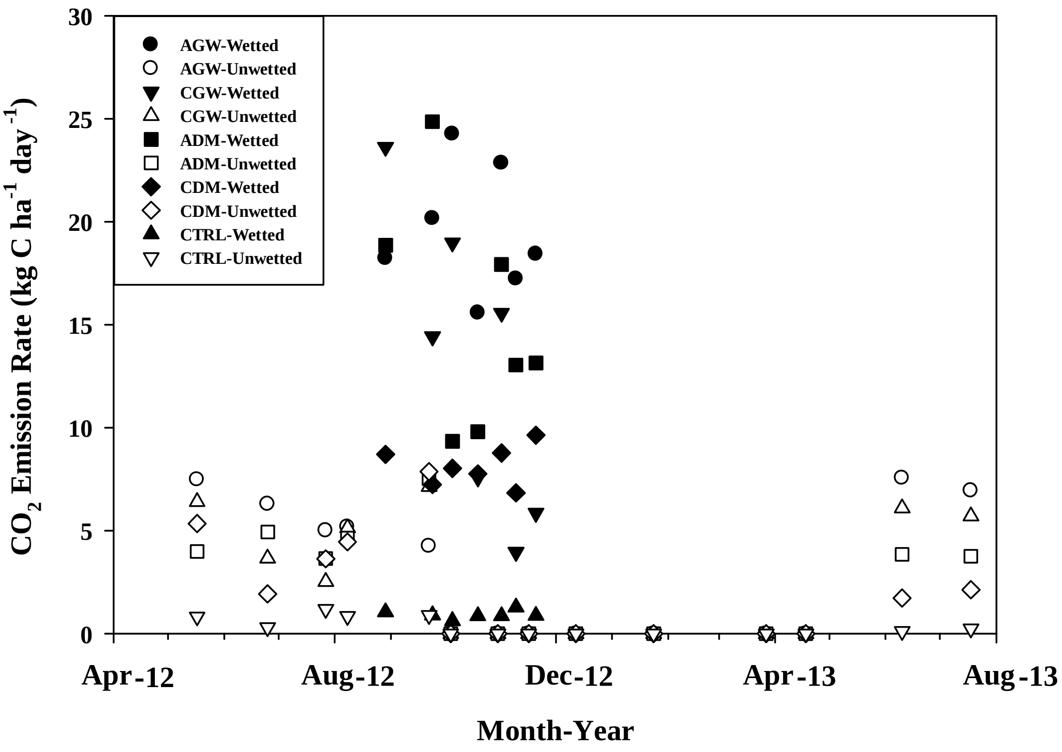

Fig. 4.3. Daily CO_2 fluxes from organic amendments incorporated into a saline-sodic soil.

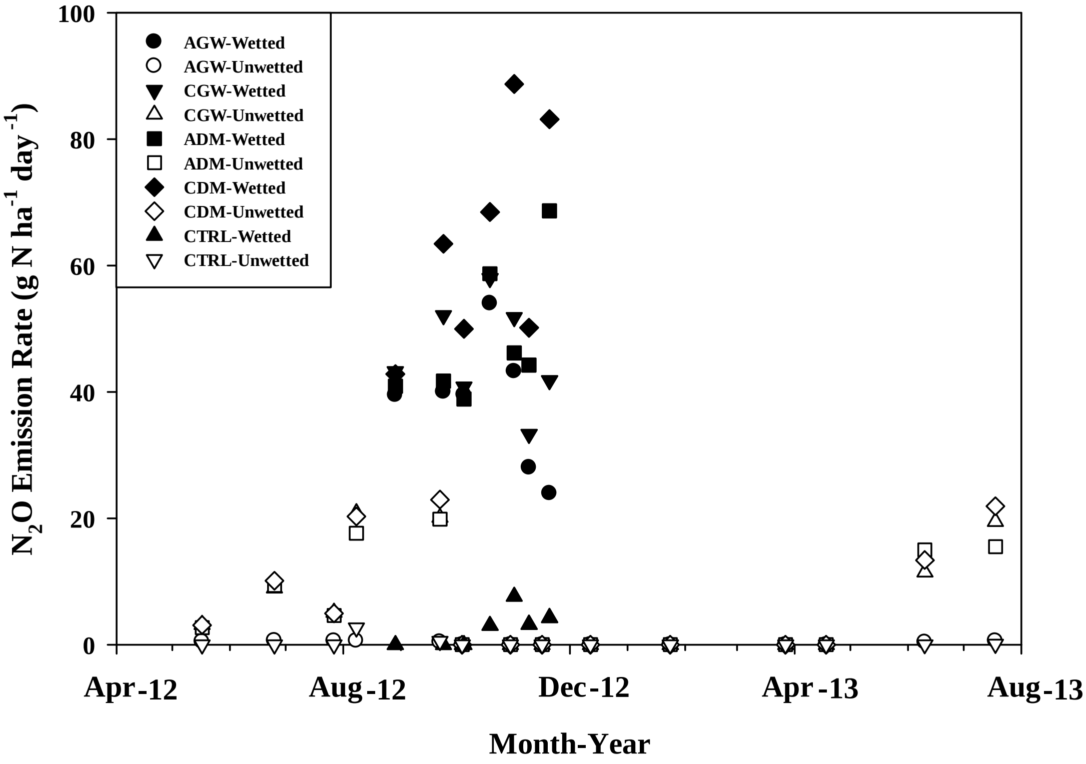

Fig.4.4. Daily N_2O fluxes from organic amendments incorporated into a saline-sodic soil.

References

Baggs E., Rees R., Smith K., Vinten A. (2000) Nitrous oxide emission from soils after incorporating crop residues. Soil Use and Management 16:82-87.

Bouwman A., Fung I., Matthews E., John J. (1993) Global analysis of the potential for N_2O production in natural soils. Global Biogeochemical Cycles 7:557-597.

Calderón F.J., McCarty G.W., Van Kessel J.A.S., Reeves J.B. (2004) Carbon and nitrogen dynamics during incubation of manured soil. Soil Science Society of America Journal 68:1592-1599.

Carter M.R. (1993) Soil sampling and methods of analysis CRC Press, Boca Raton, Florida.

Castaldi S., Smith K.A. (1998) The effect of different N substrates on biological N_2O production from forest and agricultural light textured soils. Plant and Soil 199:229-238.

Chang C., Janzen H., Cho C. (1998) Nitrous oxide emission from long-term manured soils. Soil Science Society of America Journal 62:677-682.

Dalal R., Wang W., Robertson G., Parton W. (2003) Nitrous oxide emission from Australian agricultural lands and mitigation options: a review. Australian Journal of Soil Research 41:165-195.

Dalal R.C., Gibson I.R., Menzies N.W. (2009) Nitrous oxide emission from feedlot manure and green waste compost applied to Vertisols. Biology and Fertility of Soils 45:809-819.

Ding W. (2007) Effects of long-term amendment of organic manure and nitrogen fertilizer on nitrous oxide emission in a sandy loam soil. Journal of Environmental Sciences (China) 19:185-193.

EPA. (2013) Inventory of U.S. Greenhouse gas emissions and sinks: 1990-2011. U.S. Environemntal Protection Agency, Washington, DC.

FAO. (2003) World agriculture: towards 2015/2030. An FAO Perspective. USA.

Flessa H., Beese F. (1995) Effects of sugarbeet residues on soil redox potential and nitrous oxide emission. Soil Science Society of America Journal 59:1044-1051.

Granli T., Boeckman O.C. (1994) Nitrous oxide from agriculture. Norwegian Journal of Agricultural Sciences.

Groffman P.M., Tiedje J.M. (1991) Relationships between denitrification, CO_2 production and air-filled porosity in soils of different texture and drainage. Soil Biology and Biochemistry 23:299-302.

Huang Y., Zou J., Zheng X., Wang Y., Xu X. (2004) Nitrous oxide emissions as influenced by amendment of plant residues with different C: N ratios. Soil Biology and Biochemistry 36:973-981.

Jury W., Letey J., Collins T. (1982) Analysis of chamber methods used for measuring nitrous oxide production in the field. Soil Science Society of America Journal 46:250-256.

Khalil M., Rosenani A., Van Cleemput O., Boeckx P., Shamshuddin J., Fauziah C. (2002) Nitrous oxide production from an ultisol of the humid tropics treated with different nitrogen sources and moisture regimes. Biology and Fertility of Soils 36:59-65.

Lakhdar A., Rabhi M., Ghnaya T., Montemurro F., Jedidi N., Abdelly C. (2009) Effectiveness of compost use in salt-affected soil. Journal of hazardous materials 171:29-37.

Lakhdar A., Hafsi C., Rabhi M., Debez A., Montemurro F., Abdelly C., Jedidi N., Ouerghi Z. (2008) Application of municipal solid waste compost reduces the negative effects of saline water in *Hordeum maritimum* L. Bioresource technology 99:7160-7167.

Li L.-J., Han X.-Z., You M.-Y., Horwath W.R. (2013) Nitrous oxide emissions from Mollisols as affected by long-term applications of organic amendments and chemical fertilizers. Science of The Total Environment 452:302-308.

Liang Y., Yang Y., Yang C., Shen Q., Zhou J., Yang L. (2003) Soil enzymatic activity and growth of rice and barley as influenced by organic manure in an anthropogenic soil. Geoderma 115:149-160.

Liang Y., Si J., Nikolic M., Peng Y., Chen W., Jiang Y. (2005) Organic manure stimulates biological activity and barley growth in soil subject to secondary salinization. Soil Biology and Biochemistry 37:1185-1195.

Ma Y., Wang J., Zhou W., Yan X., Xiong Z. (2012) Greenhouse gas emissions during the seedling stage of rice agriculture as affected by cultivar type and crop density. Biology and Fertility of Soils 48:589-595.

NRCS. (2012) Soil Survey Staff, Natural Resources Conservation Service, United States Department of Agriculture. Official Soil Series Descriptions.Available online at http://soils.usda.gov/technical/classification/osd/index.html. Accessed 13 January 2012, USDA-NRCS, Lincoln, NE.

Parkin T.B., Venterea R.T. (2010) Chamber-based trace gas flux measurements, in: R. F. Follett (Ed.), USDA-ARS GRACEnet Sampling Protocols. pp. 1-39.

Parton W., Holland E., Del Grosso S., Hartman M., Martin R., Mosier A., Ojima D., Schimel D. (2001) Generalized model for NOx and N_2O emissions from soils. Journal of Geophysical Research: Atmospheres (1984–2012) 106:17403-17419.

Pathak H., Rao D. (1998) Carbon and nitrogen mineralization from added organic matter in saline and alkali soils. Soil Biology and Biochemistry 30:695-702.

Poth M., Focht D.D. (1985) ^{15}N kinetic analysis of N_2O production by Nitrosomonas europaea: an examination of nitrifier denitrification. Applied and Environmental Microbiology 49:1134-1141.

Renault P., Stengel P. (1994) Modeling oxygen diffusion in aggregated soils: I. Anaerobiosis inside the aggregates. Soil Science Society of America Journal 58:1017-1023.

Rietz D., Haynes R. (2003) Effects of irrigation-induced salinity and sodicity on soil microbial activity. Soil Biology and Biochemistry 35:845-854.

Robertson G., Tiedje J. (1987) Nitrous oxide sources in aerobic soils: nitrification, denitrification and other biological processes. Soil Biology and Biochemistry 19:187-193.

Tejada M., Gonzalez J. (2006) Crushed cotton gin compost on soil biological properties and rice yield. European journal of agronomy 25:22-29.

Tejada M., Garcia C., Gonzalez J.L., Hernandez M.T. (2006) Use of organic amendment as a strategy for saline soil remediation: Influence on the physical, chemical and biological properties of soil. Soil Biology and Biochemistry 38:1413-1421.

Thompson W., Leege P., Milner P., Watson M. (2001) Test methods for examination of composting and compost, in: J. Smith (Ed.), The US Composting Council, US Government Priting Office.

Velthof G.L., Kuikman P.J., Oenema O. (2003) Nitrous oxide emission from animal manures applied to soil under controlled conditions. Biology and Fertility of Soils 37:221-230.

Vigil M., Kissel D. (1991) Equations for estimating the amount of nitrogen mineralized from crop residues. Soil Science Society of America Journal 55:757-761.

Wahid A., Akhtar S., Ali I., Rasul E. (1998) Amelioration of saline-sodic soils with organic matter and their use for wheat growth. Communications in Soil Science and Plant Analysis 29:2307 - 2318.

Walker D., Bernal M. (2008) The effects of olive mill waste compost and poultry manure on the availability and plant uptake of nutrients in a highly saline soil. Bioresource technology 99:396-403.

Wichelns D. (1999) An economic model of waterlogging and salinization in arid regions. Ecological Economics 30:475-491.

Wong V., Dalal R., Greene R. (2009) Carbon dynamics of sodic and saline soils following gypsum and organic material additions: a laboratory incubation. Applied Soil Ecology 41:29-40.

WRCC. (2013) Western Regional Climate Center. Cooperative Climatological Data Summaries. Available Online at http://www.wrcc.dri.edu/cgi-bin/cliMAIN.pl?casjac+sca. Accessed 24 June 2013.

5. CONCLUSIONS

The purpose of this dissertation was to examine the influence of soil salinization and properties of organic amendments on greenhouse gas emissions. In summary, CO_2, N_2O, and N_2 emissions were significantly affected by soil salinity, temperature, type of feedstock, and C availability from organic amendments. Soil respiration and mineralization were decreased with increases in salinity from 2.8 to 30.6 dS m^{-1}. The general decrease in CO_2 production and soil NO_3^--N content with increasing soil EC may have been due to direct influence of osmotic stress on microbially driven biogeochemical processes. Increases in soil EC_e enhanced N_2O emissions but decreased N_2 emissions suggesting the possibility of deactivation of N_2O reductase enzyme responsible for the conversion of N_2O to N_2, the final product of denitrification. Also, increases in temperature significantly increased greenhouse gas emissions and soil NO_3^--N concentrations. Production of N_2O, N_2, and N_2 to N_2O-N ratios increased at elevated temperatures of 25 and 35°C compared to 15°C as did the CO_2 production associated with improved microbial activity at higher temperatures.

As expected cumulative CO_2-C emissions were highest from active organic amendments (AGW and ADM) due to the presence of high amounts of labile C that is more easily decomposed. In addition, cumulative N_2O emissions increased from soils treated with dairy manure amendments compared to greenwaste materials. This was most likely associated with lower C:N ratios of dairy manure amendments that resulted in faster mineralization and nitrification resulting in availability of NO_3^-as substrate for denitrification. Soil NO_3^- concentrations in soils treated with greenwaste amendments, particularly AGW, were significantly lower than other treatments including unamended controls indicating that N immobilization was responsible for reduced N_2O emissions. However, regardless of soil EC_e and temperature levels, AGW also resulted in greater CO_2 emissions and N_2 to N_2O-N ratios. This is likely due to the presence of large amounts of labile C in AGW which may be responsible for the conversion of N_2O to N_2. This was most evident with a strong correlation between CO_2 and N_2 emissions.

In the field, patterns from the laboratory experiments were verified under unwetted and artificially wetted conditions. Under unwetted conditions, CO_2 and N_2O fluxes were generally low compared to artificially wetted scenarios. Regardless of soil water content, lower daily N_2O fluxes were consistently observed from AGW alone. In contrast, significantly higher CO_2 fluxes were recorded for both AGW and ADM treatments. Within the 14-month study period, mineral N content from greenwaste treatments applied in both active and cured forms was significantly lower than from the dairy manure amendments. Active materials incorporated in the form of greenwaste reduced average daily N_2O emissions by 64% compared to dairy manure amended plots. Average daily emission fractions (AEF) of N_2O ranged from 0.01-0.03 g kg^{-1} from all organic amendments used in this study. In addition, AEF's from active treatments were on an average 42% lower than cured amendments.

The laboratory and field studies examined the influence of soil and environmental factors that regulate N_2O emissions following the incorporation of organic amendments. These studies found evidence to support: 1) higher soil salinity levels enhanced N_2O emissions regardless of the type and stage of organic amendment incorporated; 2) increased CO_2 and N_2 emissions in response to elevated temperatures and addition of labile C source in the form of active organic amendments; and 3) reduced cumulative N_2O emissions from greenwaste treatments relative to those that were treated with dairy manure amendments. These findings suggest that greenwaste materials, due to their effectiveness in reducing N_2O emissions, can be used to substitute dairy manure amendments for remediation of salt-affected soils. Furthermore, results from this work also demonstrate the existence of complex interactions between abiotic and biotic factors in the soil (*e.g.,* electrical conductivity, temperature, soil moisture, and microbial respiration) that serve as important regulators for biologically driven denitrification process and associated N_2O and N_2 emissions.