Geoderma 307 (2017) 267–276

Contents lists available at ScienceDirect

Geoderma

journal homepage: www.elsevier.com/locate/geoderma

Short-term effects of biochar and salinity on soil greenhouse gas emissions from a semi-arid Australian soil after re-wetting

C. Maucieri[a,b,*], Y. Zhang[a,c], M.D. McDaniel[a,d], M. Borin[b], M.A. Adams[a]

[a] Centre for Carbon Water and Food, Sydney Institute of Agriculture, University of Sydney, Camden, New South Wales 2570, Australia
[b] Department of Agronomy, Food, Natural Resources, Animals and Environment – DAFNAE, University of Padua, Agripolis Campus, Viale dell'Università 16, 35020 Legnaro, PD, Italy
[c] Jiangsu Key Laboratory of Low Carbon Agriculture and GHGs Mitigation, Nanjing Agricultural University, Nanjing 210095, China
[d] Department of Agronomy, Iowa State University, 2517 Agronomy Hall, Ames, IA, United States

A R T I C L E I N F O

Keywords:
Carbon dioxide
Methane
Nitrous oxide
NaCl
Biochar
Water holding capacity

A B S T R A C T

Arid and semi-arid soils often show a pulse of soil greenhouse gas (GHG) emissions upon re-wetting – whether from irrigation water or rainfall. We used a laboratory incubation to elucidate interactions of salinity, biochar amendment, and simulated wetting intensity in emissions of carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) in a semi-arid Australian soil. A factorial experimental design was used with three main factors: irrigation water salinity (using NaCl, control or ~0.9 dS m^{-1}, 5 dS m^{-1} and 10 dS m^{-1}), biochar amendment (0% and 5% by mass of *Eucalyputs polybractea* biochar) and soil moisture (25%, 50%, 75% and 100% of water-holding capacity, WHC – a proxy for wetting intensity after irrigation or rainfall). The strongest single regulating variable of rates of soil CO_2 emission was WHC (+171% increase between 25% and 100% WHC). Salinity reduced CO_2 emissions (relative to controls) by −19% at 5 dS m^{-1} and −28% at 10 dS m^{-1}. Soils amended with biochar produced less (−10%) CO_2 emissions. All treatments showed negative CH_4 emissions (or CH_4 oxidation) that were only influenced by WHC. Soil N_2O emissions increased with salinity (+60%), while biochar additions reduced them slightly (−12%). N_2O emissions were not influenced by WHC. Overall, results showed that biochar additions can mitigate some of the "pulse" effects of rainfall on emissions (~10% in term of global warming potential across all treatments).

1. Introduction

Agriculture plays an important role in global anthropogenic greenhouse gas (GHG) emissions. Understanding management practices that mitigate agriculture's impact on GHG emissions is crucial for all nations, but particularly for those countries where agriculture comprises large portions of total national emissions. In Australia, for example, agriculture contributes 23% of total national emissions (Hatfield-Dodds et al., 2007; Maraseni and Cockfield, 2011). Over 70% of Australian land is classified as either semi-arid or arid (Wolfe, 2009), and soil moisture limits crop growth. Irrigation is sparingly used with grain crops but may be more widely used in future. Both natural rainfall and irrigation can be significant regulators of overall soil emissions, by causing disproportionately large "pulses" of emissions relative to normal conditions (Davidson, 1992; Kessavalou et al., 1998; Sponseller, 2007; McDaniel et al., 2014a).

Globally, 75 countries have been recognized as having large areas of salt-affected lands for a total of approximately 831 million hectares (Amini et al., 2016). Different types of salinization, with a prevalence of sodium salts, affect about 30% of Australia, and >60% of soils in agricultural zones in Australia are at least partially sodic (Rengasamy, 2006). Salinization can also affect GHG emissions, but has received less attention than other management effects and its impact remains unclear. Studies have shown increases (N_2O, Low et al., 1997), decreases (CH_4, Marton et al., 2012), or no effect (CO_2, CH_4, N_2O, Kontopoulou et al., 2015) in GHG emissions with increased salinity.

Water shortages around the world emphasize the significance of irrigation to global change and expanding populations (Cheng et al., 2009; Sietz and Van Dijk, 2015), and with this, an increased likelihood of having to turn to low-quality water for irrigation. Low quality water is frequently characterized by high salinity, and in some cases sodicity, which can influence soil microbial communities with flow-on consequences for carbon (C) and nitrogen (N) transformations (Ndour et al., 2008) and crop production (Filippini et al., 2011; Okorogbona et al., 2015). Salinity has had variable consequences for emissions and warrants further research. In particular, water and sodicity are likely to

* Corresponding author.
E-mail address: carmelo.maucieri@unipd.it (C. Maucieri).

http://dx.doi.org/10.1016/j.geoderma.2017.07.028
Received 10 February 2017; Received in revised form 20 July 2017; Accepted 24 July 2017
Available online 30 August 2017
0016-7061/

produce contrasting effects on soil GHG emissions. For example, using open-field irrigation, Kontopoulou et al. (2015) found no effect of irrigation water salinity on soil CO_2, CH_4 and N_2O emissions within a crop of common bean. In contrast, Zhang et al. (2016a) reported that salinity significantly increased N_2O emissions within a cotton crop. Short-term exposure to salinity has also been reported to increase anaerobic C mineralization, and reduce CH_4 emissions (Marton et al., 2012). On the other hand, soil moisture has well-described roles in regulating GHG emissions (Zou et al., 2005; Muhr et al., 2008; Case et al., 2012). For example, in a laboratory experiment using intact soil cores from 13 European sites, Schaufler et al. (2010) noted positive correlations of soil moisture with N_2O emission, negative correlations with CH_4 oxidation, and maximum CO_2 emissions at intermediate soil moistures. It is well established that water can physically limit soil aeration (Mentges et al., 2016; Maucieri et al., 2017), whereas in very dry soils, osmotic stress can limit activity of microbial communities (Stark and Firestone, 1995; Smith et al., 2003).

Biochar – a by-product of thermal degradation of organic materials in the absence of oxygen (pyrolysis) – has now been widely studied and promoted as a soil amendment that can increase soil C sequestration, improve soil fertility, and thus help mitigate climate change (Lehmann, 2007; Zhang et al., 2010; Jeffery et al., 2011; Nelissen et al., 2014; Partey et al., 2016; Wang et al., 2016; Subedi et al., 2016; Zhang et al., 2016b; Subedi et al., 2017). Biochar may also reduce emissions by influencing soil microbial community size (Zhang et al., 2014) and composition (Lehmann et al., 2011), as well as the availability of substrates to microbes (Spokas and Reicosky, 2009; Singh et al., 2010) and water and/or O_2. Biochar has also been reported to change soil redox environments (Cayuela et al., 2013, 2014a). Overall, most studies report that biochar amendments reduce emissions (Liu et al., 2011; Feng et al., 2012; Case et al., 2015; Ameloot et al., 2016), notwithstanding some reports to the contrary (Clough et al., 2010; Cheng et al., 2012; Wang et al., 2014).

Here we examine interacting effects on GHG emissions of two possible interventions for Australian grain production – irrigation with saline water and additions of biochar. The role of soil moisture content was also examined, inter alia, as a proxy for wetting intensity – whether that be from irrigation or natural rainfall. Our aim was to evaluate the single and interactive effects of these treatments on CO_2, CH_4 and N_2O emissions from an Australian vertisol soil. In situ flux measurements previously suggested these soils are particularly prone to producing pulses of GHG after rain events, especially after long periods of dryness (McDaniel et al., 2017). We tested three main hypotheses: 1) increasing the wetting intensity (or soil moisture) will increase the pulse of soil GHGs, 2) the biochar will reduce the pulse effect of re-wetting, and 3) salinity will increase the pulse of soil GHG emissions. We used a short-term laboratory incubation to test these hypotheses.

2. Materials and methods

2.1. Soil sampling and biochar

Soil samples (0–10 cm depth cores) were collected from a clay-loam vertisol soil (USDA, 1999) managed with chickpea (*Cicer arietinum* L.) and winter wheat (*Triticum aestivum* L.) crop rotation at the experimental station of the University of Sydney, Narrabri, Australia (30°19′ S, 149°46′ E). Sixty-six soil cores were collected on May 28th, 2015 across a 16 ha field and after sampling they were pooled to obtain a representative sample. Visible plant detritus and any fragments were removed after air-drying at room temperature for 20 days, and soils were then passed through a 2-mm stainless steel sieve. We used air-dried soils for two reasons: 1) ease of manipulation of soil moisture, beginning with dry soils, and 2) simulation of 'pulse' events over a gradient of irrigation or rainfall intensities – which were simulated by establishing four different soil moistures. Soil physicochemical properties are shown in Table 1. The biochar used as a soil amendment was produced from woody remains of blue mallee (*Eucalyptus polybractea*), via low-temperature pyrolysis (< 500 °C) by Biochar Energy Systems Pty. Ltd., Bendigo, Australia. Biochar was characterized by a pH of 9.6, total carbon content of 54.9% and a carbon to nitrogen ratio of 39.5, more information about the chemical characteristics of biochar are reported in Keith et al. (2015). Biochar used in this study was also air-dried and ground to pass through a 2-mm sieve.

Table 1
Soil physicochemical properties (mean ± standard deviation) before the incubation.

Property	Value
Clay (%)	28.2 ± 1.7
Silt (%)	45.8 ± 1.1
Sand (%)	26.0 ± 1.2
Volumetric water content ($m^3\ m^{-3}$)	0.238 ± 0.043
Total C (%)	1.35 ± 0.24
Total N (%)	0.11 ± 0.03
Soil C-to-N ratio	13.4 ± 4.5
Dissolved organic C ($mg\ kg^{-1}$)	106 ± 39
Dissolved organic N ($mg\ kg^{-1}$)	28 ± 13
NH_4^+-N ($mg\ kg^{-1}$)	10.63 ± 1.55
NO_3^--N ($mg\ kg^{-1}$)	17.91 ± 0.81
pH	7.25 ± 0.03
Electrical conductivity ($\mu S\ cm^{-1}$)	185.87 ± 2.85
Water holding capacity ($g\ g^{-1}$)	0.597 ± 0.034

2.2. Experimental design and set-up

We used a 3 × 2 × 4 factorial design with the following main factors: 1) irrigation water salinity as the main factor (control or 0 $dS\ m^{-1}$, 5 $dS\ m^{-1}$ and 10 $dS\ m^{-1}$); 2) biochar amendment (5%) as a secondary factor (control and with biochar addition); 3) soil moisture as a proxy for wetting intensity (25%, 50%, 75% and 100% water-holding capacity, WHC). In total, there were twenty-four treatment combinations in our experiment, each with three replicates. For each treatment, 30 g of soil (on air-dry weight basis) was added to a glass mason jar (1-L) and sealed tightly with a screw-cap lid fitted with a stopcock for subsequent gas sampling.

Soil water salinity was adjusted using deionized water and NaCl. Background salinity of the deionized water was 0.09 $dS\ m^{-1}$, but we refer to the treatment as '0 $dS\ m^{-1}$'. The jars receiving biochar had 1.5 g of air-dried biochar added to each soil (or 5% w/w), and soil and biochar were thoroughly mixed. Water content was adjusted using the deionized or salinized water. Due to sieving and drying, the natural porosity of the soil was changed. However, we estimated that the water contents were roughly equivalent to 24%, 48%, 72%, and 96% WFPS. The incubation began when water was added to all soils to bring them to the four soil moisture contents. The jars were monitored and re-adjusted for any soil moisture loss. Jars were kept at 20 ± 1 °C during the entire 30 d incubation period.

2.3. Greenhouse gas measurements

Using a protocol similar to that described by McDaniel et al. (2014b), gas emissions were measured after 1, 2, 3, 4, 5, 7, 8, 9, 10, 11, 12, 14, 15, 16, 18, 21, 24, 28 and 30 days of incubation. To begin each gas-sampling event, jars were thoroughly flushed with ambient air for 5 min by wafting ambient air into all jars. Gas samples from the headspace were collected by syringe (35 ml) just after jars closing (one sample per jar, to provide initial concentrations) and after 24 h to calculate an emission. For each sampling 15 ml of sampled headspace air were injected into a 12-ml evacuated 'Exetainer' vial (i.e. a slight over-pressure) sealed with butyl-rubber septa. Vials were analyzed for greenhouse gases content within 6–8 days of gas sampling. N_2O and CH_4 concentrations were measured by LGR N_2O/CH_4 analyzer (*Los Gatos Research*, *Mountain View*, *CA*, *USA*) and a LI-820 (LI-COR, Lincoln,