Science of the Total Environment 465 (2013) 279–287

Contents lists available at ScienceDirect

Science of the Total Environment

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

Pyrolysing poultry litter reduces N_2O and CO_2 fluxes

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HIGHLIGHTS

- ► Raw poultry litter (PL) had higher N_2O flux cf. PL-biochar or urea at similar N rates.
- ► PL increased labile C in soil and had higher emissions of CO_2 cf. PL-biochar.
- ► The majority of N_2O flux was from non-urea sources and was not influenced by PL-biochar.
- ► PL-biochar improved plant available P which was linked to greater crop N uptake.

ARTICLE INFO

Article history:
Received 26 September 2012
Received in revised form 11 February 2013
Accepted 18 February 2013
Available online 15 March 2013

Keywords:
Poultry litter
Biochar
Nitrous oxide
C mineralisation
Crop productivity
Ferrosol

ABSTRACT

Application of poultry litter (PL) to soil can lead to substantial nitrous oxide (N_2O) emissions due to the co-application of labile carbon (C) and nitrogen (N). Slow pyrolysis of PL to produce biochar may mitigate N_2O emissions from this source, whilst still providing agronomic benefits. In a corn crop on ferrosol with similarly matched available N inputs of ca. 116 kg N/ha, PL-biochar plus urea emitted significantly less N_2O (1.5 kg N_2O–N/ha) compared to raw PL at 4.9 kg N_2O–N/ha. Urea amendment without the PL-biochar emitted 1.2 kg N_2O–N/ha, and the PL-biochar alone emitted only 0.35 kg N_2O–N/ha. Both PL and PL-biochar resulted in similar corn yields and total N uptake which was significantly greater than for urea alone. Using stable isotope methodology, the majority (~80%) of N_2O emissions were shown to be from non-urea sources. Amendment with raw PL significantly increased C mineralisation and the quantity of permanganate oxidisable organic C. The low molar H/C (0.49) and O/C (0.16) ratios of the PL-biochar suggest its higher stability in soil than raw PL. The PL-biochar also had higher P and K fertiliser value than raw PL. This study suggests that PL-biochar is a valuable soil amendment with the potential to significantly reduce emissions of soil greenhouse gases compared to the raw product. Contrary to other studies, PL-biochar incorporated to 100 mm did not reduce N_2O emissions from surface applied urea, which suggests that further field evaluation of biochar impacts, and methods of application of both biochar and fertiliser, are needed.

1. Introduction

The poultry industry is one of the fastest growing agricultural industries in the world and significant issues exist with the disposal and management of litter to minimise its environmental impacts (Bolan et al., 2010). Runge et al. (2007) estimate that 1.6 million tonnes of PL is produced annually in Australia, whilst MacDonald et al. (2009) estimate that over 44 million tonnes is produced annually in the United States. PL is a mixture of the bedding material (sawdust, rice hulls) used for the initial bedding placed on the floor of a clean shed, and the excreta, feathers and other detritus from the chickens plus wasted feed and water. Whilst PL is increasingly being used as a soil amendment, only limited information is available about its impacts on nutrient use efficiency and greenhouse gas emissions from soil. Application of PL to soil provides a rich source of nutrients (Sharpe et al., 2004) and readily available organic C to support the activities of microbial populations including ammonifiers, nitrifiers and denitrifiers (Bolan et al., 2010). However, the N released via mineralization of the litter can therefore be lost via ammonia volatilization, leaching, and nitrification and denitrification processes in soil (Moore et al., 1996; Akiyama et al., 2004). Ammonia fluxes from field-applied PL have been reported in the

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Abbreviations: PL, poultry litter.

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http://dx.doi.org/10.1016/j.scitotenv.2013.02.054

range of 3.3% of applied N in winter to 24% in summer (Sharpe et al., 2004), thus reducing N use efficiency. The direct emission of the greenhouse gas (GHG) nitrous oxide (N_2O) from soil, and indirect emissions of N_2O through volatilization and leaching (De Klein et al., 2006) following PL application are a matter of concern because of their high global warming potential i.e. 298 over a 100 year period (Forster et al., 2007).

Although the IPCC (De Klein et al., 2006) assigned a value of 1% conversion of applied N to N_2O for N additions from mineral fertilisers and organic amendments, it is well established that N losses from applied manure can exceed this value. For example, Sistani et al. (2011) showed that between 3.2 and 5.8% of the N in PL applied to soil during a corn crop was converted to N_2O, compared to a measured conversion factor for urea of 0.40–1.2% of applied N. Cabrera et al. (1994) showed that up to 6.8% of N in raw PL applied to a loamy sand was converted to N_2O. Toyoda et al. (2011) suggested that nitrification was a more significant source of N_2O for synthetic fertiliser applied to Andisols, whilst denitrification was a dominant mechanism for N_2O production when the same soil was amended with PL. This is likely due to co-application of relatively labile organic C, which enhances microbial activity and creates conditions favourable for denitrification.

In recent years, biochar production via pyrolysis of organic waste resources and its application to soil has been proposed as a way to stabilise organics C and N in slow cycling forms (Cross and Sohi, 2011; Singh et al., 2012; Wang et al., 2012), stabilise native and added soil C (Keith et al., 2011; Slavich et al., 2012), and reduce greenhouse gas emissions from soil (Singh et al., 2010; Van Zwieten et al., 2009, 2010). Most evidences presented on the potential of biochar to reduce soil N_2O emissions are based on laboratory studies. Several mechanisms have been proposed to explain biochar's role in reducing greenhouse gas emissions from soil (Van Zwieten et al., 2009) including; a) reduced bioavailability of labile organics C and N due to their sorption on biochar surfaces and their subsequent stabilisation through interaction with soil clay minerals (Keith et al., 2011; Lin et al., 2012), (b) increased soil pH through addition of alkalinity in biochars (Kookana et al., 2011), and/or (c) increased soil porosity due to reduction in soil bulk density (Yanai et al., 2007). However, field measurements have shown inconsistent results, with reductions in some situations, but neutral effects or small increases in others (Scheer et al., 2011; Zhang et al., 2010; Taghizadeh-Toosi et al., 2011). Furthermore, there has not been a quantitative comparison in emissions between applications of PL and pyrolysed PL to soil with respect to impacts on soil C, soil fertility, crop yield, N uptake and GHG flux.

This study examines an alternative means to manage organic C and nutrient resources in PL through the production of a value-added amendment (PL-biochar) for use in agricultural soils. The manuscript describes the use of pyrolysed PL in a field trial with regard to its impacts on key soil properties, N-uptake and N-use efficiency and emissions of N_2O, relative to direct PL or urea only treatments, or as a co-amendment.

2. Methods

2.1. Biochar production and characterisation

PL (freshly removed from a commercial poultry shed) with sawdust bedding was sourced by Pacific Pyrolysis (Somersby, NSW) and 1000 kg was pyrolysed using a continuous 300 kg/h pilot slow-pyrolysis unit. The PL-biochar was produced at a highest treatment temperature of 550 °C, at a heating rate of 5–10 °C/min. The plant used for the production of the biochar operates as a continuous process with a mean residence time in the reactor of approximately 30 min. Biochar was sieved to <2 mm before analysis and application to soils. The feedstock and resulting biochar were tested for calorific value, ultimate and proximate analysis using the Australian Standard methods AS 1038.5, AS 1038.6.1 and AS 1038.3.

2.2. Study site and experimental design

This field study was conducted on a highly permeable red ferrosol derived from basalt at the Wollongbar Primary Industries Institute (28.49″S, 153.23″E; elevation 140 m), Wollongbar, New South Wales, Australia commencing in November 2011. The climate is sub-tropical and rainfall is summer dominated. The field site was prepared by mowing existing pasture and removing most of the above ground biomass. The site was sprayed with glyphosate and after 2 weeks, was rotary hoed to 100 mm. The site was surveyed and 24 plots of 10 m × 2.5 m pegged in an array 2 plots deep and 12 plots wide.

Treatments standardised C input from PL and PL-biochar amendments, each to supply 4.5 Mg C/ha. A total of 6 treatments were applied according to a randomised complete block design. Treatments comprised; 1) nil amendment; 2) urea at 116 kg N/ha; 3) raw PL at 12 Mg dry wt/ha; 4) PL-biochar at 10 Mg dry wt/ha; 5) PL-biochar at 10 Mg dry wt/ha plus 116 kg N/ha as urea; and 6) raw PL at 12 Mg dry wt/ha in combination with biochar at 10 Mg dry wt/ha. Treatment 6 supplied 9 Mg C/ha. The rationale behind the selection and standardisation of N application is given in the Discussion.

The site was rotary hoed to incorporate the C amendments into the 0–100 mm soil layer, whilst the urea was surface applied following seeding. The nil amendment plots were also rotary hoed. To minimise limitations in P or K nutrition, single super 300 kg/ha and muriate of potash 140 kg/ha; equivalent to 26.4 kg P and 70 kg K were applied prior to sowing the summer corn crop (*Zea mays* var H5). Soil bulk densities across the 24 plots in the 0–100 mm profile ranged between 1.01 and 1.03 40 d post-sowing with no trends between treatments.

Each plot was mechanically planted with six rows of corn in November 2011 and harvested manually in late February 2012. Above ground biomass (close to 100%) was manually removed shortly after harvest of cobs. A representative subsample of whole plant (dried and ground material) was used for analysis of N uptake into the above ground component of the crop.

2.3. Soil sampling and analysis

Soil samples were taken at days 41 and 113 following amendment. Three soil cores (50 mm diameter) from the 0–100 mm soil layers were taken from each plot, air-dried, composited and passed through a 1 mm sieve prior to analysis. Soil and biochar chemical analyses were undertaken in a NATA (National Association of Testing Authorities, Australia) facility accredited to ISO17025.

Total C and N were measured by Dumas combustion using an Elementar vario MAX CN analyser with combustion chamber set at 900 °C and an O_2 flow rate of 125 mL/min. Permanganate-oxidisable organic C (using 333 mM $KMnO_4$) was assayed according to methods described in Blair et al. (1995). The pH was measured in 0.01 M $CaCl_2$ (1:5) according to method 4B2 (Rayment and Higginson, 1992). Exchangeable cations were assessed using 1 M NH_4OAc described in method 15E1 (Rayment and Higginson, 1992). The liming value of PL-biochar was measured as the carbonate equivalent using method 19A1. The acid extractable elements were determined according to USEPA 6010 using a Varian 720-EC ICP-OES, Inductively Coupled Plasma, Optical Emission Spectrometer (ICP-OES). Plant available P was tested using Bray #1 extraction as described in method 9E2 (Rayment and Higginson, 1992).

Biochar was tested for total P, water-soluble P, citrate-insoluble P, and available P (citrate-soluble) according to AOAC Official Methods 977.01 and 963.03 (AOAC International, 2000).

Nitrate analyses were done on days 9, 16, 23, 35, 42, 55, and 66 days following amendment whilst a limited number of ammonium analyses (days 35, 42, 55, 66) were undertaken. Additional soil sampling and analyses occurred prior to amendment and at harvest. Soils from each plot were sampled (0–50 mm profile) using a corer (40 mm diameter), homogenised in the field and extracted immediately with KCl before