



The effect of lignite on nitrogen mobility in a low-fertility soil amended with biosolids and urea



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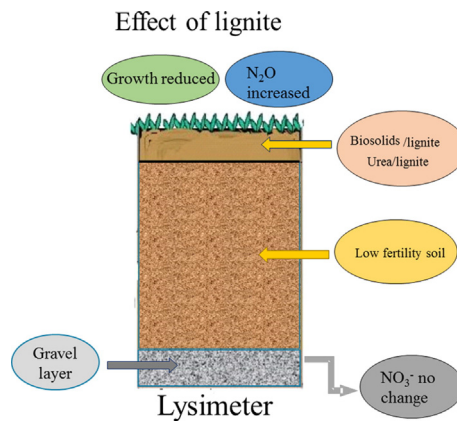
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HIGHLIGHTS

- Lignite reduces the beneficial effects of urea and biosolids on plant growth.
- Lignite has minimal effect on nitrate leaching and plant metal uptake.
- Lignite exacerbates nitrous oxide emissions from soil.

GRAPHICAL ABSTRACT



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ABSTRACT

Lignite has been proposed as a soil amendment that reduces nitrate (NO_3^-) leaching from soil. Our objective was to determine the effect of lignite on nitrogen (N) fluxes from soil amended with biosolids or urea. The effect of lignite on plant yield and elemental composition was also determined. Batch sorption and column leaching experiments were followed by a lysimeter trial where a low fertility soil was amended with biosolids (400 kg N/ha equivalent) and urea (200 kg N/ha equivalent). Treatments were replicated three times, with and without lignite addition (20 t/ha equivalent). Lignite did not reduce NO_3^- leaching from soils amended with either biosolids or urea. While lignite decreased NO_3^- leaching from an unamended soil, the magnitude of this effect was not significant in an agricultural context. Furthermore, lignite increased cumulative N_2O production from soils receiving urea by 90%. Lignite lessened the beneficial growth effects of adding biosolids or urea to soil. Further work could investigate whether coating urea granules with lignite may produce meaningful environmental benefits.

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1. Introduction

Countries with sewage treatment plants produce approximately 27 kg (dry mass) of biosolids (sewage sludge) per person per year (Hue, 2014). Biosolids can be used to rebuild low-fertility or degraded soils (Rigby and Smith, 2013; Robinson et al., 2011), but may have detrimental effects through contributing to nitrate (NO_3^-) leaching (Correa et al., 2006b) and due to accumulation of heavy metals in both soils and plants (Civeira and Lavado, 2008). Most nitrogen (N) in biosolids is present as organic N, which only oxidizes slowly, but there are often high concentrations of ammonium (NH_4^+) and NO_3^- (Correa et al., 2006a). This can result in a large flush of NO_3^- leaching if sufficient rainfall occurs shortly after application. High concentrations of NO_3^- in soil can also result in large emissions of N_2O (Di and Cameron, 2002), a greenhouse gas that has a global warming potential 298 times that of CO_2 (Ravishankara et al., 2009).

Knowles et al. (2011) reported that NO_3^- leaching from biosolids applied to soil could be mitigated by mixing the biosolids with biochar, produced by pyrolyzing wood-waste. However, biochar is not available in sufficient quantities for large-scale commercial operations and thus it is currently expensive (US\$200–US\$500) per ton (Kulyk, 2012). Lignite, a low grade coal (Kabe et al., 2004) may be an alternative option for reducing NO_3^- leaching from biosolids. Lignite is globally abundant with ca. 195×10^9 t of proven and recoverable lignite resources, (WEC, 2010) including 333×10^6 t located in New Zealand. Lignite has a similar structure to many biochars (Kwiatkowska et al., 2008). Typically, biochar comprises 54.9% carbon (C), 2.14% hydrogen (H), 4.97% N and 20.04% oxygen (O) (Ozcimen and Karaosmanoglu, 2004), whereas lignite has 65% C, 5% H, 1.2% N and 27.8% O (Janos et al., 2011). Lignite has a greater number of organic functional groups than biochar, with a resultant higher Cation Exchange Capacity (CEC) 20–70 cmol_c/kg (Wong et al., 1996), compared to 20–35 cmol_c/kg for a typical biochar (Gundale and DeLuca, 2007). The higher CEC of lignite may provide greater retention of NH_4^+ and thus the lower the potentially for NO_3^- leaching loss.

There is a paucity of scientific research examining the effects of unmodified lignite on NO_3^- leaching. Commercially, however, lignite has been combined with urea $\text{CO}(\text{NH}_2)_2$ and sold as “black urea” (Ferguson, 2002). In South Africa, van Vuuren and Claassens (2009), have reported that in pot trials and field trials the yield of maize (*Zea mays* L.) increased by 20–46% when “black urea” was applied and compared with standard urea. They also showed that 10–20% and 30–50% less “black urea” was required than normal urea was required when applied as topdressing on an alkaline and acidic soil respectively. ARTH (2009) reported that the fertilizer requirements of field grown maize were reduced by 15–35% when “black urea” and “black DAP (lignite coated di-ammonium phosphate fertilizer) were applied, compared to the uncoated fertilizers. There is little scientific validation of these claims. In contrast, there are many studies demonstrating heavy metal sorption by lignite (Domańska and Smolinska, 2012; Daskocil and Pekar, 2012; Jezierski et al., 2000). Qi et al. (2011) showed that lignite mixed with dewatered biosolids sludge could reduce heavy metal leaching. Simmler et al. (2013) also demonstrated that lignite could reduce plant cadmium (Cd) uptake from biosolids-amended soils. Lignite may also offset potentially toxic effects of high copper (Cu) and zinc (Zn) concentrations (Lafferty and Hobday, 1990) that are typically found in biosolids and which can accumulate in a soil. We aimed to evaluate the combined effect of lignite amendments on both N and trace metal mobility in a soil.

We hypothesised that incorporating lignite into soils amended with biosolids would (i) reduce N leaching as a result of increased CEC and (ii) reduce the plant uptake of heavy metals. These hypotheses were tested using batch studies, column-leaching studies and a lysimeter experiment.

2. Materials and methods

2.1. Soil

Lismore Stony Silt Loam (NZ classification: Pallic Orthic Brown Soil) was collected (0–30 cm depth) from the Ashley Dene sheep farm at Lincoln University ($43^\circ 39' 05.82''\text{S}$ $172^\circ 19' 41.47''\text{E}$), New Zealand. The low-fertility Lismore soil (Table 1) was formed from gravelly glacial outwash gravels and has a variable depth of silty loess deposited at the surface. The soil was air-dried to a gravimetric moisture content (θ_g) of 11.9% and sieved to <2 mm. Table 1 gives the chemical properties of the soil.

2.2. Biosolids

Biosolids were obtained from the Kaikōura Regional treatment works ($42^\circ 21' 47.78''\text{S}$ $173^\circ 41' 20.32''\text{E}$), New Zealand. Some 160 kg of stockpiled and weathered biosolids were collected and homogenized using a concrete mixer and passed initially through a 20 mm sieve. A 2 kg sub-sample was then passed through a 2 mm nylon sieve. Biosolids θ_g equalled 53%. Table 1 gives the properties of the biosolids.

2.3. Lignite

Three types of powdered lignites were provided by Solid Energy New Zealand Ltd. The New Vale lignite (NV), a low calorific value lignite, was sourced from an open cast mine in Southland, New Zealand, ($46^\circ 08' 23.80''\text{S}$ $168^\circ 45' 11.62''\text{E}$), the Charleston lignite, was sourced from Charleston open cast mine near Charleston, West Coast, New Zealand ($41^\circ 54' 51.77''\text{S}$ $171^\circ 25' 58.62''\text{E}$), and the Millerton lignite, was sourced from the Millerton section of the Stockton open cast mine on the West Coast, New Zealand ($41^\circ 39' 47.67''\text{S}$ $171^\circ 51' 52.17''\text{E}$). The lignites were sorted on a crusher with an air swept classifier, which separated the lignite particles based on size (Simmler et al., 2013). Table 1 shows the chemical properties of the respective lignites.

The pH of the soil, biosolids and lignites was determined in water using a sample: water ratio (w/w) of 1:2.5 following the method of Blakemore et al. (1987), using a Mettler Toledo, GmbH, Switzerland. Carbon (C) and N concentrations were determined using an Elementar Vario MAX CN analyser (Elementar GmbH Germany). The CEC was measured with the silverthiourea method described by Blakemore et al. (1987). Extractable NH_4^+ and NO_3^- concentrations in the soil and biosolids were determined with a 2 M KCl extract following the method of Blakemore et al. (1987) and Clough et al. (2001).

Water soluble C (WSC) was determined using cold (20 °C) and hot (80 °C) water extracts (Ghani et al., 2003). To measure WSC, 3 g of oven dried material and 30 mL of cold distilled water were placed in polypropylene centrifuge tubes on an end-over-end shaker for 30 min and then centrifuged for 20 min at 2253 g. The extracts were then decanted off and filtered through 0.45 μm cellulose nitrate membrane filters. The sample remaining in the centrifuge tube had a further 30 mL of distilled water added before it was then placed in a hot water bath at 80 °C for 16 h, where after it was centrifuged and filtered as before. Total carbon (TC), inorganic carbon (IC) and total organic carbon (TOC) concentrations of the WSC samples were measured using a TOC-5000A analyser (Shimadzu Oceania Pty Ltd., NSW, Australia).

Element concentrations were measured in acid digests by the ICP-OES (Varian 720-ES, Melbourne, Australia). Samples (0.5 g dry weight) of soil, lignites, biosolids or pasture (harvested from the lysimeters) were digested in 5 mL $\text{HNO}_3/1$ mL H_2O_2 (Merck hydrogen peroxide 30%) at 175 °C for 20 min. The digest was diluted with deionized water (Milli Q, Barnstead, EASYpure RF, 18.2 M Ω -cm) to a volume of 25 mL and filtered through a Whatman 52 filter paper (Simmler et al., 2013). A Wageningen reference soil (ISE 989) and plant (IPE 100) material were analysed for quality assurance (van Dijk and Houba, 1998).

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