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Short communication

Grassland plants affect dissolved organic carbon and nitrogen dynamics in soil

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Abstract

Dissolved organic carbon (DOC) and nitrogen (DON) are central in many nutrient cycles within soil and they play an important role in many pedogenic processes. Plants provide a primary input of DOC and DON into soil via root turnover and exudation. Under controlled conditions we investigated the influence of 11 grass species alongside an unplanted control on the amount and nature of DOC and DON in soil. Our results showed that while the presence of plants significantly increases the size of a number of dissolved nutrient pools in comparison to the unplanted soil (e.g. DOC, total phenolics in solution) it has little affect on other pools (e.g. free amino acids). Grass species, however, had little effect on the composition of the DOC, DON or inorganic N pools. While the concentration of free amino acids was the same in the planted and unplanted soil, the flux through this pool was significantly faster in the presence of plants. The presence of plants also affected the biodegradability of the DOC pool. We conclude that while the presence of plants significantly affects the quantity and cycling of DOC and DON in soil, comparatively, individual grass species exerts less influence.

Keywords: Carbon cycling; Nitrogen cycling; Mineralization; Phenolics; Rhizosphere; Rhizodeposition

While inorganic nutrient cycling has been intensively studied in soils, there is comparatively less information available on the dynamics and functional significance of dissolved organic nutrients in soil (Kalbitz et al., 2000). In many natural ecosystems, the primary input of nutrients occurs in an organic form due to the addition of plant and animal residues (Shand et al., 2002; Stockdale et al., 2001). While there is some plant input of dissolved organic C (DOC) and N (DON) to soil from above-ground litter and throughfall (Schwendenmann and Veldkamp, 2005), the main inputs arise from below-ground root/mycorrhizal turnover and rhizodeposition (Nguyen, 2003). Rhizodeposition is predicted to lead to an increase in the concentration of the soil's low molecular weight DOC pool (Miller et al., 2005). Evidence from mathematical modelling suggests that the concentration of C in the rhizosphere will be considerably higher than in the bulk soil, however, this is predicted to be highly temporally and spatially variable and dependent upon the rate of microbial C removal (Toal et al., 2000). It is also likely to be plant species specific (Nguyen, 2003). The aim of this study was to evaluate the influence of plant species on the concentration, quality and turnover of DOC and DON in soil in comparison to an unplanted soil.

Soil (Eutric cambisol) was obtained from an agricultural grassland located in Abergwyngregyn, UK (53°14′ N, 4°01′ W). Further details of the soil and site can be found in Table 1 and Jones et al. (2004). Three independent samples from the surface Ah horizon (0–20 cm) were used in the experiments. Field moist soil from each replicate sample was placed into opaque nylon pots to give a bulk density of $1.0\,g\,cm^{-3}$. Rhizon in situ soil water samplers (Rhizosphere Research Products, Wageningen, Netherlands) were then inserted into the pots to recover soil solution. Seeds of 11 grass species were then sown in monoculture within the pots: *Lolium perenne* L., *Poa annua* L., *Dactylis glomerata* L., *Agropyron repens* L., *Holcus lanatus* L., *Phleum pratensis* L., *Alopecurus pratensis* L., *Agrostis tenuis* Sibth., *Festuca rubra* L., *Lolium multiflorum*

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Table 1
Chemical and physical characteristics of the grassland soils used in the study

	Parameter
$EC_{1:1}$, $\mu S\,cm^{-1}$	80±4
pH ($_{1:1}$, H_2O)	5.90±0.03
$CaCO_3$, $g\,kg^{-1}$	0.11±0.02
Water holding capacity, $g\,kg^{-1}$	520±20
Moisture content, $g\,kg^{-1}$	160±10
Organic C, $g\,kg^{-1}$	2.1±0.1
Total N, $g\,kg^{-1}$	0.16±0.01
C-to-N ratio	13.3±0.6
Exchangeable cations	
Na, $mmol\,kg^{-1}$	1.2±0.1
K, $mmol\,kg^{-1}$	2.4±0.3
Ca, $mmol\,kg^{-1}$	12.5±0.4
Mg, $mmol\,kg^{-1}$	3.0±0.6
Al, $mmol\,kg^{-1}$	0.8±0.1
Root biomass, $g\,m^{-3}$	0.39±0.01
Soil respiration, $g\ CO_2\ m^{-2}\,h^{-1}$	0.60±0.02

All values represent means±SEM ($n = 3$).

Lam., and *Cynosurus cristatus* L.. Three pots were also left unplanted (control). The pots were then placed in a randomized design in a climate-controlled growth room with day/night rhythm of 18/22 °C, 70% relative humidity, photoperiod of 16 h and light intensity of 500 μmol photons $m^{-2}\,s^{-1}$ PAR. Pots were watered to field capacity three times weekly using artificial rainwater (NaCl, 96 μM; K_2SO_4, 20 μM; $CaCl_2$, 5 μM; $MgCl_2$, 6 μM; NH_4NO_3, 15 μM; KH_2PO_4, 0.1 μM). When the grass sward was >15 cm in height, the above ground biomass was removed to 2.5 cm above the soil surface. Approximately, 3 months after sward establishment, and after the first sward cut, soil solution was recovered biweekly over a 3-month period. Soil solution was collected 24 h after the soils had been watered to field capacity and at least 14 d after sward cutting. Soil solutions were analysed for DOC, DON, total free amino acids NO_3^- and NH_4^+ as described in Jones et al. (2004). Phenol containing substances were assessed according to Swain and Hillis (1959) while DOC biodegradability was assessed according to McDowell et al. (2006).

To determine the rate of free amino acid mineralization in each of the soils, 500 μl of soil solution was spiked with a mixture of 16 uniformly ^{14}C-labelled amino acids (50 μl; <10 nM; 37 kBq ml^{-1}; Jones et al., 2005). The ^{14}C-labelled amino acid mixture was then injected back into the soil between the grass plants in each pot. An opaque plastic cylinder was then placed over the labelled area (approximately 10 cm^2) and pushed into the soil. A 1 M NaOH trap was then placed inside the cylinder and the cylinder hermetically sealed at the top. Over a 14 d period the NaOH trap was periodically replaced and the $^{14}CO_2$ in the NaOH traps determined by liquid scintillation counting. Amino acid mineralization was fitted to a double first-order exponential decay model:

$$S = [a_1 \times \exp(-b_1 t)] + [a_2 \times \exp(-b_2 t)], \quad (1)$$

where S is the ^{14}C-label remaining in the soil, b_1 is the rate constant describing the primary mineralization phase (i.e. depletion from soil solution; Jones et al., 2004, 2005), b_2 is the rate constant describing the secondary mineralization of the microbial biomass, a_1 and a_2 describe the size of pools b_1 and b_2 and t is time.

All experimental treatments were performed in triplicate. Statistical analysis (*t*-tests, linear regression and ANOVA followed by Tukey pair-wise comparison) was performed with the computer programs Excel 12.0 (Microsoft Corp., CA) and Minitab 14.0 (Minitab Inc., State College, PA). The fitting of the double first-order kinetic model to the experimental amino acid mineralization data was performed with Sigmaplot 8.0 (SPSS Inc., Chicago, IL).

Generally, there was a significant difference in soil solution chemistry in the unplanted soil in comparison to those containing plants over the 6 sampling events (Fig. 1). The unplanted soil had significantly higher NO_3^- and DON concentrations and lower DOC and phenolic concentrations in comparison to the planted soil ($P<0.001$), however, there was no significant difference in total free amino acid concentration ($P>0.05$). The levels of NH_4^+ in all soil solution samples were below detection limits at all sampling dates (<0.05 mg $N\,l^{-1}$). Overall, plant species had relatively little effect on the chemistry of the soil solution (Fig. 1).

On average, 35±2% of the DOC was biodegraded during the 7 d incubation period in accordance with previous studies (Don and Kalbitz, 2005; Fig. 2). The biodegradability of the DOC in soil solution was greatest in the unplanted soil in comparison to the planted soil when expressed in the conventional way ($P<0.01$; McDowell et al., 2006; Fig. 2). At the end of the assay the DOC in the unplanted soil solution (7.5±0.4 mg $C\,l^{-1}$) remained significantly lower than in the planted soil (22.3±1.4 mg $C\,l^{-1}$). There was no significant influence, however, of plant species on the biodegradability of DOC in soil solution ($P>0.05$; Fig. 2).

In agreement with previous studies, the rate of ^{14}C loss after the addition of amino acids to the soil was biphasic with an initial fast phase of $^{14}CO_2$ loss (0–0.5 d) followed by a significantly slower phase of $^{14}CO_2$ evolution (0.5–14 d). Overall, the experimentally measured rates of $^{14}CO_2$ loss conformed extremely well to a double first-order kinetic model ($r^2 = 0.98 \pm 0.01$, $n = 12$; Eq. (1)). The calculated rate of amino acid turnover in the unplanted soil was about 2-fold slower than observed in the planted soil (Fig. 3). This difference reflected the slower turnover in the initial mineralization phase (0–1 h). Although, there were small differences in amino acid half-life between some of the grass species, these were not statistically significant ($P>0.05$).

The results presented here clearly show that plants are involved in regulating DOC and DON concentrations in soil. This supports previous studies in forest ecosystems and flooded rice paddy soils which have shown increases in DOC in the presence of plants (Lu et al., 2000; Yano et al., 2004). One explanation for the plant-induced increases in DOC is due to direct inputs into the soil solution from root exudates which are known to be C rich (Toal et al., 2000).