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Decomposition kinetics of soil carbon of different age from a forest exposed to 8 years of elevated atmospheric CO_2 concentration

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ABSTRACT

Ecosystem exposure to elevated atmospheric CO_2 concentration can often leads to increased ecosystem carbon (C) fluxes, as well as greater net primary production. Changes in the soil C pool with elevated [CO_2] are more difficult to measure and therefore remain poorly understood. In this study, we carried out a series of laboratory soil incubations, in order to determine whether 8 years of ecosystem exposure to elevated [CO_2] altered decomposition dynamics of two age classes of soil C in a temperate coniferous forest. Our objectives were to determine whether there were differences in the decomposition kinetics of soil C up to 8 years old ($C_{post\text{-}tr}$) and soil C older than 8 years ($C_{pre\text{-}tr}$), in the absence of concurrent plant activity. We collected soil from the Duke Forest Free Air CO_2 Enrichment site in North Carolina and incubated whole and crushed (all macroaggregates dispersed) soil from two depth increments (0–5 cm and 5–15 cm) for 102–127 days. We found that mineral soil from the treatment plots had higher respiration rates in the absence of concurrent plant activity than mineral soil from plots under ambient CO_2 conditions. These differences in respiration rate were only significant in 0–5 cm soil and could be largely explained by higher initial respiration rates of soil collected from the CO_2-treated plots. Disruption of soil macroaggregates did not result in a difference in efflux rate in soil from this forest under ambient or elevated CO_2 conditions at either depth. The specific respiration rate of $C_{post\text{-}tr}$ was higher than that of $C_{pre\text{-}tr}$ in the top 5 cm of soil, while the opposite was true for 5–15 cm of soil. Even though $C_{post\text{-}tr}$ was assimilated by the ecosystem more recently than $C_{pre\text{-}tr}$, their decay constants were similar at both depths. These results suggest that, in the absence of plant activity, the mineralization of soil C of different ages in this forest may be under similar biological and/or biochemical control. Therefore, if the higher initial rates of decomposition of $C_{post\text{-}tr}$ seen in these experiments are sustained in the field, greater labile pool size of recently added C, and potentially faster cycling of this pool, may in part explain higher soil respiration rates and limited soil C accumulation under elevated [CO_2] in this forest.

1. Introduction

Rising atmospheric CO_2 concentration ([CO_2]) and its potential to alter global climate have amplified research attention to the carbon (C) storage capacity of the terrestrial biosphere, in both plant biomass and soil pools. Increasing atmospheric [CO_2] can alter the cycling of C in terrestrial ecosystems by increasing the magnitude of ecosystem C fluxes (King et al., 2004; Bernhardt et al., 2006; Ainsworth and Rogers, 2007), as well as the pool size of above- and belowground plant biomass (DeLucia et al., 1999; Matamala and Schlesinger, 2000; Norby et al., 2005; Moore et al., 2006). Even though greater plant biomass production under elevated [CO_2] may result in increased soil C inputs, most ecosystem-scale CO_2 enrichment experiments report no significant increases in soil C content (Leavitt et al., 2001; Schlesinger and Lichter, 2001; Van Groenigen et al., 2003; Hoosbeek et al., 2004; Lichter et al., 2005; but see Jastrow et al., 2005). The large spatial heterogeneity of SOM may prevent the detection of small increases in soil C content with elevated [CO_2] (Hungate et al., 1996; Jastrow et al., 2005). Another potential explanation for this inconsistency is that enhanced C inputs under elevated [CO_2] may be allocated to labile C pools that turn over rapidly and are not incorporated into stable SOM pools with long residence times (Lichter et al., 2005; Taneva et al., 2006), but may lead to observed increases in soil respiration rates (King et al., 2004). The mechanisms involved in the storage, transformation, and turnover of the additional C entering soils when ecosystems are exposed to elevated atmospheric [CO_2] are not well understood, but are of critical

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importance to the accurate evaluation of the long-term storage capacity of soils for anthropogenic C.

Soil C balance is a function of plant C inputs and C outputs through heterotrophic respiration. Increased C inputs to soils with elevated [CO_2] have been hypothesized to lead to both soil C and nutrient accumulation (Diaz et al., 1993) and, in contrast, to greater soil C and N cycling rates and increased decomposition (Zak et al., 1993). Furthermore, there is increasing evidence that enhanced plant activity and belowground C inputs under elevated [CO_2] can, directly or indirectly, lead to increases in SOM decomposition dynamics (Hoosbeek et al., 2004; Subke et al., 2004; Trueman and Gonzalez-Meler, 2005; Allard et al., 2006). Ecosystem exposure to elevated [CO_2] may therefore indirectly lead to interactions between labile and more recalcitrant soil C pools, but it remains unclear whether such changes in soil C cycling are mainly brought about by the presence of plant activity or are the result of further and lasting changes in soil C pool dynamics.

Recently, the importance of the relationship between soil physical protection and SOM dynamics has been highlighted (Jastrow et al., 1996; Six et al., 2000) and SOM sequestration in afforested soils and restored grasslands has been shown to occur mainly within soil aggregates (Jastrow et al., 1996; DeGryze et al., 2004). Labile organic C can become physically protected within soil aggregates, which can render it unavailable to microorganisms, and thus relatively new C can accumulate in soil pools with long residence times (Oades, 1984; Elliott and Coleman, 1988; Jastrow et al., 1996). Soil physical protection mechanisms depend in part on the quantity and quality of belowground C inputs (Six et al., 2001), as well as on soil microbial activity (Six et al., 1999), which have been shown to be affected by elevated atmospheric [CO_2] in some ecosystems (Matamala and Schlesinger, 2000; Montealegre et al., 2002). Anthropogenic changes in environmental conditions that may alter mechanisms of soil C protection or aggregate turnover can therefore potentially accelerate the loss of soil C by exposing protected C to microbial degradation.

In this study, we conducted a series of laboratory incubation experiments to evaluate the dynamics of soil C oxidation as a function of C age and macroaggregate protection in soils collected from the Free Air CO_2 Enrichment (FACE) experiment in the Duke Forest (Chapel Hill, NC, USA). Our objectives were: (1) to determine whether differences in soil C decomposition dynamics between soil from the elevated CO_2-treated and control plots exist in the absence of concurrent plant activity; (2) to determine the extent to which soil macroaggregate protection affects soil C mineralization; and (3) to evaluate the mineralization kinetics of soil C of two different age groups: C assimilated by the ecosystem before fumigation began 8 years previously and soil C assimilated over the previous 8 years in both soil with and without intact macroaggregates.

2. Materials and methods

2.1. Site description

The Forest Atmosphere Carbon Transfer and Storage 1 (FACTS-1) research site is located in the Blackwood Division of the Duke Forest, near Chapel Hill, NC, USA (35°58′N 79°05′W). The Free Air CO_2 Enrichment (FACE) experiment at FACTS-1 is composed of six 30-m diameter plots in an intact loblolly pine (*Pinus taeda*) plantation. Three of the experimental plots are fumigated with CO_2 to maintain an atmospheric [CO_2] that is approximately 200 μl/l above ambient, or about 567 ± 4 μl/l (averaged from 1996 to 2004; K. Lewin and R. Nettles, personal communication); the three control plots are fumigated with ambient air only (Hendrey et al., 1999). Continuous fumigation of all plots began on 27 August 1996, when the trees were 15 years old. Fumigation is switched off when temperatures are below 5 °C and when sustained wind speed exceeds 5 m/s. Starting 16 December 2002, fumigation was reduced to daytime only.

The loblolly pine plantation was established in 1983, with 3-year-old seedlings planted at 2 × 2.4 m spacing. Through natural regeneration, a number of hardwood species have become established in the understory, the most abundant of which are *Acer rubrum*, *Liquidambar styraciflua*, *Liriodendron tulipifera*, *Ulmus alata*, and *Cercis canadensis*. Soils at the site are clay-rich, low fertility Ultic Alfisols, derived from igneous rock, with a pH of ~5. Fine roots are found mostly in the upper 20 cm of the soil profile (Matamala and Schlesinger, 2000). Mean annual temperature is 15.5 °C and mean annual precipitation is 1140 mm.

2.2. Ecosystem ^{13}C tracer

The CO_2 used for fumigation at FACTS-1 is strongly depleted in ^{13}C vs. PDB ($\delta^{13}C \approx -43.1 \pm 0.6‰$ SE, where $\delta^{13}C = [(R_{sample} - R_{reference})/R_{reference}] \times 1000$ and $R = {}^{13}C/{}^{12}C$). By increasing atmospheric [CO_2] by 200 μl/l in the treatment plots, the $\delta^{13}C$ of atmospheric CO_2 is changed from about −8 to −20 ± 0.4‰. Consequently, new needles and fine roots produced under FACE have a $\delta^{13}C$ of −41.8 ± 0.3 ‰ and −39.2 ± 0.8‰ compared to $\delta^{13}C$ of −29.9 ± 0.2‰ and −27.6 ± 0.2‰ at ambient conditions, respectively (Matamala et al., 2003). The fumigated forest plots have been exposed to a continuous ecosystem ^{13}C label since the beginning of the CO_2 treatment in 1996 and, through its incorporation into plant biomass, the ^{13}C label has been incorporated into soil organic matter and can be detected in soil-respired CO_2 (Andrews et al., 1999; Taneva et al., 2006).

2.3. Soil sampling and preparation

Soils were sampled to a depth of 30 cm in March 2005 at four randomly chosen locations within each plot at FACTS-1, using a 5.4 cm diameter slide hammer soil corer. The soil was frozen to −20 °C and transported on dry ice to the laboratory. Soil cores were divided into 0–5 cm, 5–15 cm, and 15–30 cm depth increments prior to processing; the 15–30 cm increment was not part of the experiments described in this study. Roots, plant fragments, debris and rocks were removed by hand from moist soil. Root-free soil was sieved to pass a 2-mm screen and then dried at 65 °C for 5–7 days. In order to disperse all physical protection of SOM by soil aggregates, a subsample of soil was ground manually with a mortar and pestle to pass through a 180 μm sieve. Thus, there were two soil treatments for the soil incubation experiments—whole soil (WS), with all micro- and macro-aggregates intact, and crushed soil (CS), with all soil macroaggregates disrupted. The $\delta^{13}C$ values of the soil were determined with a Finnegan Delta Plus XL isotope ratio mass spectrometer (Bremen, Germany).

2.4. Laboratory soil incubations

Mineralizable soil C and the degree of protection from decomposition provided by soil macroaggregates were evaluated during extended laboratory incubations of root-free moist samples of either whole soil (WS) or crushed soil (CS) at the 0–5 cm and 5–15 cm depth increments. Moisture content of the dried soil samples was adjusted to and maintained at 80% field capacity with deionized water. Duplicate, 30 g subsamples from each FACE plot (ambient and elevated [CO_2]), each depth (0–5 cm and 5–15 cm), and each soil treatment (WS and CS) were placed into a 0.9 l Mason jar and incubated in the dark at 25 °C. A layer of glass wool was placed on the bottom of the jar to facilitate aeration and to prevent drying of the soil sample. The soil samples were mixed with 30 g of sterilized sand to avoid changes in soil aggregation during the incubation period. Jars were capped with a perforated lid to avoid anoxic conditions. Two control jars contained no soil. Soil samples