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Sensitivity of soil organic carbon stocks and fractions to different land-use changes across Europe

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ABSTRACT

Land-use changes (LUC) influence the balance of soil organic carbon (SOC) and hence may cause CO_2 emissions or sequestration. In Europe there is a side by side of LUC types that lead to SOC loss or SOC accumulation. However, there is a lack of studies covering all major LUC types to investigate qualitative and quantitative LUC effects on SOC. In this study we sampled 24 paired sites in Europe to a depth of 80 cm, covering a wide range of pedo-climatic conditions and comprising the major European LUC types cropland to grassland, grassland to cropland, cropland to forest and grassland to forest. To assess qualitative changes and the sensitivity of different functional SOC pools with distinct turnover times, we conducted a fractionation to isolate five different fractions of SOC. The mean SOC stock changes after LUC were 18 ± 11 Mg ha^{-1} (cropland to grassland), 21 ± 13 Mg ha^{-1} (cropland to forest), -19 ± 7 Mg ha^{-1} (grassland to cropland) and -10 ± 7 Mg ha^{-1} (grassland to forest) with the main changes occurring in the topsoil (0–30 cm depth). However, subsoil carbon stocks (>30 cm depth) were also affected by LUC, at 19 out of 24 sites in the same direction as the topsoil. LUC promoting subsoil SOC accumulation might be a sustainable C sink. Particulate organic matter (POM) was found to be most sensitive to LUC. After cropland afforestation, POM accounted for 50% (9.1 ± 2.3 Mg ha^{-1}) of the sequestered carbon in 0–30 cm: after grassland afforestation POM increased on average by 5 ± 2.3 Mg ha^{-1}, while all other fractions depleted. Thus, afforestations shift SOC from stable to labile pools. The resistant fraction comprising the so-called inert carbon was found to be only slightly less sensitive than the total SOC pool, suggesting that an inert carbon pool was not chemically extracted with NaOCl oxidation, if there is any inert carbon.

1. Introduction

Land-use changes (LUC) affect soil organic carbon (SOC) stocks and can either lead to sequestration or emission of CO_2 (Houghton, 2003). The worldwide SOC pool is estimated at 1400 Pg, which equals the sum of the atmospheric and biotic carbon pools (Hiederer and Köchy, 2011). Therefore any manipulation of this pool can significantly influence the concentration of atmospheric CO_2. Historically, when croplands have been established on land previously used for native vegetation, the soil C pool has been a major source of atmospheric CO_2, contributing about 180–200 Pg C over the last two centuries (DeFries et al., 1999) which is about 40% of the total anthropogenic CO_2 emissions (Marland et al., 2000). The conversion from natural vegetation to cropland often leads to a depletion of the SOC stock due to reduced input of biomass and enhanced decomposition after physical disturbance (Poeplau et al., 2011). Approximately 10% of the total land surface has been converted into croplands (DeFries et al., 1999). Consequently, world soils now contain a lower C pool than their potential capacity. Thus, LUC are acknowledged as a measure to mitigate climate change if the depleted cropland pools are refilled with carbon (The Terrestrial Carbon Group, 2010). However, carbon loss from soils due to LUC is still going on. Worldwide, LUC is responsible for estimated net emissions of 1.1 ± 0.7 Pg C per year of the first decade of 2000s. This is mainly due to deforestation and the conversion of natural vegetation into cropland in the tropics (Don et al., 2011; Houghton, 2003). While afforestation is acknowledged as a measure to regain SOC, several studies have shown that the afforestation of grassland can also lead to a depletion of the SOC stock (Alfredsson et al., 1998; Davis and Condron, 2002; Thuille and Schulze, 2006). This is explained by the lower incorporation of above ground biomass and lower input of below-ground biomass (Pérez-Cruzado et al., 2011). LUC in Europe is estimated to currently and in the near future be a net sink for CO_2 owing to extensification with land abandonment and afforestation (mainly in the North and East) (Schulze et al., 2010; Zaehle et al., 2007). However, due to increased production of bioenergy crops on croplands, a new pressure on land resources is arising with more LUC to croplands. Thus, in contrast to other regions such as the tropics where mainly natural vegetation is converted to agricultural land, in Europe there is a side by side of different LUC types that either lead to SOC losses or SOC accumulations. Surprisingly, there are hardly any studies investigating LUC effects on SOC that covers all major European LUC types allowing direct comparisons.

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SOC consists of several components, which differ in their physico-chemical properties and hence their degree of stabilization and turnover time. To estimate the effect of LUC on the total SOC stocks, it is crucial to quantify and understand the sensitivity of the different functional SOC pools to disruptions such as LUC (Martin et al., 1990). These functional pools are mostly defined by their distinct turnover time ranging from years to millennia (Bol et al., 2009). Dead plant material or particulate organic matter (POM) is easily available to microorganisms and hence rapidly decomposable (Zimmermann et al., 2007b). The longer dead plant material stays in the soil system, the more it experiences physico-chemically transformation and is stabilized in aggregates and bonded onto the surface of minerals such as clay and silt (Six et al., 2002). Along with this stabilization process, a small SOC fraction might even become biochemically recalcitrant with low bioavailability and long turnover times (Krull and Skjemstad, 2003). From this conceptual point of view, fractions with a shorter turnover time should respond more rapidly to LUC than more stabilized fractions. While several studies found POM to be more sensitive to LUC than other SOC fractions (Conant et al., 2004; Franzluebbers and Stuedemann, 2002; Six et al., 1998), other studies found no effect of management or land-use change on this labile pool (Conant et al., 2003; Jastrow, 1996; Leifeld and Kögel-Knabner, 2005). Jastrow (1996) reported for an Australian prairie restoration, that more than 80% of the total accumulation occurred in the mineral-associated fraction. A wide range of different physical and chemical fractionation methods have been developed to identify SOM pools with distinct properties, functions and turnover times (Amelung and Zech, 1999; Christensen, 2001; Hassink et al., 1997; Six et al., 2001). A major goal of most fractionation methods is to isolate fractions that correspond to functionally defined SOC pools, to e.g., initialize and parameterize SOC turnover models (Scharnagl et al., 2010). This is a challenging task, since measured fractions are operationally defined and do not necessarily correspond to functional pools (Cambardella, 1998). Zimmermann et al. (2007c) described a method to measure different fractions that are quantitatively highly related to modeled pools in the Rothamstad Carbon model (RothC) (Coleman and Jenkinson, 1996). The obtained fractions are free particulate organic matter (POM); particulate organic matter which is occluded in stable aggregates or attached to the sand fraction (S + A); dissolved organic carbon (DOC), SOC that is bonded onto silt and clay particles without being biochemically recalcitrant (S + C-rSOC) and a resistant fraction (rSOC). So far, not many studies have used this approach (Dondini et al., 2009; Xu et al., 2011) although it might have the potential to become a new standard fractionation method since it can be used to parameterize a widely used turnover model.

The concentration and turnover of SOC are usually highest in the surface soil (Conant et al., 2001). Therefore, most studies investigating LUC effects on SOC are restricted to a depth of 30 cm (Lorenz and Lal, 2005). This involves a lack of knowledge about the sensitivity of subsoil SOC to LUC (Chabbi et al., 2009). However, there is rising evidence that LUC affects SOC dynamics also in the subsoil (Poeplau et al., 2011; Wright et al., 2007), while we do not sufficiently understand the processes leading to the stabilization of SOC in the subsoil (Don et al., 2007; Lützow et al., 2006). Since the subsoil C pool accounts for up to 30–75% of the total SOC (Batjes, 1996; Rumpel and Kogel-Knabner, 2011; Rumpel et al., 2002), an investigation of only the surface layer may lead to biased conclusions (Baker et al., 2007; Wiesmeier et al., 2012). Don et al. (2009) even found a depletion of SOC in the subsoil after the conversion from cropland to grassland. In our study we assessed SOC dynamics down to a depth of 80 cm or down to the bedrock on shallower soils.

LUC in Europe are estimated to be a net sink for CO_2 (Schulze et al., 2010; Zaehle et al., 2007). However, SOC dynamics are often underrepresented in these estimates or based on worldwide means due to insufficient European data sources (Schulze et al., 2010). In the present study we sampled 24 sites in Europe comprising the four major LUC types to assess the sensitivity of SOC stocks to LUC both qualitatively and quantitatively. We aim to test the following three hypotheses: i) converting cropland to grassland or forest leads to a substantial gain in SOC, while the afforestation of grassland has a slightly negative effect; ii) Subsoil SOC stocks are also affected by LUC, but with less intensity than topsoil SOC stocks; iii) there is a strong sensitivity gradient within the different fractions with the strongest change occurring in the POM fraction, while the resistant SOC fraction remains unaffected.

2. Materials and methods

2.1. Study sites and soil sampling

We selected 24 study sites across Europe. These sites comprise the following four LUC types with six replicate sites each: cropland to grassland and vice versa, cropland to forest and grassland to forest, which are the most relevant LUC types in Europe. We tried to find sites located in regions, where the particular LUC is of actual relevance, such as grassland afforestation in the Alps or in Great Britain and Ireland; cropland to grassland and forest conversion in Lithuania and grassland to cropland conversion in Germany (Table 1). To measure changes in SOC we used the paired plot approach, where a reference plot represents the conditions before land-use change (Time 0) and a conversion plot represents the conditions at Time *x* after LUC. Two thirds of all sites were established long-term research sites, where SOC stocks have not necessarily been investigated yet, and one third was found by talking to farmers or land owners. The sites were selected by accounting for the following preconditions: i) the reference plot has not undergone any major land-use change in the past ~100 years, ii) the conversion plot exists in the present land-use for at least 20 years to ensure that the major effect of SOC dynamic has already occurred, iii) the two plots of a pair are situated directly adjacent or close to each other to ensure comparable pedological conditions. The only site where the time since conversion was below 20 years was Vilnius (16 years). 18 out of 24 paired plots were directly adjacent to each other, 6 pairs had a distance ranging from a few hundred meters to around 6 km, but were still located on very similar soils. Plot selection in the field was done after scanning the soil with a hand-auger in order to minimize the variability within a plot and between the reference plot and the conversion plot. At six sites a triple plot could be sampled to obtain two different LUC types. At three sites (Kungsängen, Hordorf and Trenthorst) the triple plot was permanent grassland, permanent cropland and grassland, which had been cropland before. Besides two different land-use changes (grassland to cropland and cropland to grassland), we could also assess the potential SOC dynamic as a function of time (chronosequence approach). The two plots of a pair were sampled in a 14 m × 21 m grid (Fig. 1) with 6 replicates each. This relatively small plot size ensures that comparable paired plots can be found, even at sites with a high topographical and pedological heterogeneity. Samples were taken with a petrol driven auger (87 mm diameter) down to a depth of 80 cm. Because of shallow bedrock, at 5 sites the soil could only be sampled down to a depth of 50 cm. Compactions of the soil core were estimated by the difference of the core length and the borehole depth. Compaction correction was performed for the compacted area, which could be visually detected on the core and was often at a depth of 40–80 cm. Compaction increased with soil moisture and sand content and ranged from no compaction in fine textured soils to 8 cm in wet sandy soils. The soil cores were segregated into 0–10, 10–20, 20–30, 30–50 and 50–80 cm increments and packed in air-tight plastic bags. At forest sites the forest floor was sampled using a 20 × 20 cm metal frame at the same position before the soil core was extracted.

2.2. Sample preparation and analysis

All soil samples were dried at 40 °C, all forest floor samples at 70 °C and weighed to determine the total dry mass. Coarse roots