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# Large changes in carbon storage under different land-use regimes in subtropical seasonally dry forests of southern South America

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ABSTRACT

Net emissions of $CO_2$ from land-use conversion represent a significant driver of global climatic change. This is especially true for subtropical seasonally dry Chaco forests from southern South America, now experiencing one of the highest loss rates globally. However, direct quantifications of the effect of accelerated deforestation on carbon (C) pools of these systems are rare. Considering five dominant ecosystem types within the dry Chaco forest of Argentina, derived by land-use change from the same original vegetation, substrate, and climate, we quantified the magnitude and change of total C pools including trees and shrubs, non-woody plants, coarse and fine debris, and soil organic (SOC) and inorganic (SIC) pools up to 2 m depth. Soil C pools represented the largest C stocks (>74%). Shrubs also represented a large C pool (at least 28% of the aboveground standing biomass), which we quantified in detail for the first time. The conversion of forests to open shrublands and croplands was associated to large losses of organic C both in aboveground biomass and in soils down to 30 cm depth (from 43 to 64%). Although SIC is usually considered as a relatively stable compartment, the forest to crop transition presented here involved carbonate losses of c. 68% at soil depths between 1 and 2 m. Our results indicate that the landscape transformations expected in the region under business-as-usual socioeconomic scenarios will probably lead to a marked reduction of the C stored, with a consequent net C emission and a decline in other C storage-related ecosystem services provided by these ecosystems.


## 1. Introduction

Unequivocal evidence of global climatic change as a result of rising levels of $CO_2$ in the atmosphere (IPCC, 2013) has stimulated interest in the conservation and enhancement of natural biological reservoirs of carbon (C), particularly forests. Land-use changes, including forest cover replacement by crops and grazing lands, are one of the main proximate causes of the average annual net $CO_2$ emissions of 0.9 Gt of C to the atmosphere in the last decades (IPCC, 2013; Peters et al., 2013; van der Werf et al., 2009).

In spite of having one of the highest rates of annual cover loss globally (Hansen et al., 2013), the subtropical seasonally dry forests of southern South America have been comparatively less studied than temperate and tropical forests, where the effect of land-use on the carbon pools and associated emissions are fairly well studied (Baccini et al., 2012; Don et al., 2011; Harris et al., 2012; Saatchi et al., 2011). The Chaco forest – the geographically most extended seasonally dry forest in South America – is now undergoing massive clearing for agricultural expansion, including annual crops and to a lesser extent grazing lands. In particular, the annual deforestation rates reached by the Chaco forests of Argentina during the last decade are among the highest in the world (Hansen et al., 2013). Emissions from deforestation in the semi-arid Chaco forests of the same region between 1996 and 2005 have been estimated at 15.65 Gg C $y^{-1}$, representing the largest source from land-cover change in the subtropical forests of southern South America (Gasparri et al., 2008).

The process of C loss from vegetation removal due to forest management and deforestation is often accompanied by direct C losses from surface soils as a result of a reduction in its physical protection and a rise in soil temperature in the short term (Martínez-Mena et al., 2002; Post and Kwon, 2000), as well as a reduction in the amount of litter fall in the long term (Don et al.,

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2011; Jandl et al., 2007; Yanai et al., 2003). In the Chaco forest, previous works have found a significant decline in soil organic C (SOC) from the top 0–20 cm (ranging from 33 to 78% of C loss) due to overgrazing and forest degradation (Abril and Bucher, 2001; Bonino, 2006). Deeper soil is assumed to be largely insensitive to the effects of management activities (Dungait et al., 2012; Jandl et al., 2007). However, land-use practices could re-distribute SOC within the soil profile (e.g., by tillage) leading to a priming effect (i.e., a stimulation of C mineralization in deeper layers due to the supply of fresh plant-derived C from surface layers) (Chaopricha and Marín-Spiotta, 2014; Fontaine et al., 2007); or simply by breaking the soil structure and making C more available to decomposers (Salomé et al., 2010; Schmidt et al., 2011). There is also increasing evidence that root-derived inputs drive SOC storage and stability along the soil profile in different ecosystems (Kätterer et al., 2011; Rasse et al., 2005; Tefs and Gleixner, 2012). Since dominant woody species from water-limited ecosystems are expected to have deep root distributions (Schenk and Jackson, 2005; Schenk, 2005) and high allocation to belowground biomass (Poorter et al., 2012), a change in the vegetation cover (e.g., the replacement of forest by non-woody vegetation or crops) could affect SOC vertical distribution and reduce the amount of belowground carbon input, especially in deeper layers (Jobbagy and Jackson, 2000). To date, there is no published empirical record of the SOC compartment below 30 cm for the Chaco forest, or its response to land-use change.

Generally the quantification of soil C has been focused on the organic compartment (SOC) because it is more ubiquitous, while soil inorganic C (SIC, primarily as calcium carbonate) is generally restricted to arid and semi-arid regions, and becomes more important with soil depth (Lal and Kimble, 2000; Schlesinger, 1982). In addition, SIC has been considered to be much less sensitive to land-use changes than SOC (Sanderman, 2012). However, irrigation, the use of fertilizers and other land management practices that alter soil $CO_2$ levels, pH and dissolved salt concentration, could enhance either carbonate precipitation or its dissolution, thus changing the size of the SIC pool (Lal, 2004; Mi et al., 2008; Wu et al., 2009). Recent studies in semi-arid woody ecosystems of central Argentina have shown that forest disturbance produce significant changes in the water content and salinity of soils (Jayawickreme et al., 2011; Marchesini et al., 2013), which could have significant consequences for the C dynamics of a long-term reservoir like SIC. Although SIC can be the main form of C in the dry and deep soils of semi-arid ecosystems (Sanderman, 2012) like Chaco, to our knowledge no empirical work quantifying this pool have been carried out in this ecosystems.

Methods for C estimation are well established and standardized for tropical and temperate forest across the world (Brown, 1997; Chave et al., 2014, 2005; Don et al., 2011; Gibbs et al., 2007; Keith et al., 2009; Marín-Spiotta and Sharma, 2013). However, several compartments that become important in semi-arid ecosystems (like multi-stemmed trees and shrubs, succulents, and soils with high carbonate content) require an adaptation of these standard procedures. The most important challenges that has to be considered for a complete and effective C pools quantification of these forest includes: (i) an adaptation of the globally standardized methods to effectively quantify plant biomass from different species showing different allometric relationships; (ii) deep soil sampling to include most of the SOC distribution across soil profiles of these systems; and (iii) quantification of SIC which could be an important and dynamic component of the total C of semi-arid systems. The research presented in this paper aimed to quantify in detail the aboveground and soil C pools of ecosystems under different land-uses in the semi-arid Chaco of Argentina, including trees, shrubs, non-woody vegetation, fine and coarse woody debris, and superficial and deep soil organic and inorganic C.

## 2. Materials and methods

### 2.1. Study area and experimental design

The study was carried out in the southern extreme of the Gran Chaco, in Córdoba Province, Argentina (c. 31°17′–31°50′S and 65°16′–65°32′W). The climate is subtropical with a mean annual precipitation of 600 mm distributed in spring–summer (October–March) and a mean annual temperature of 18 °C. Soils are mainly sandy-loam aridisols of alluvial origin (Gorgas and Tassile, 2003). The dominant vegetation is a xerophytic forest with *Aspidosperma quebracho-blanco* and *Prosopis flexuosa* as canopy and subcanopy dominants, respectively. The shrub layer is often dense and dominated by *Mimozyganthus carinatus*, *Acacia gilliesii*, and *Larrea divaricata* (Cabido et al., 1992).

During the December 2007–February 2008 summer, we sampled five different ecosystem types in the area, originally developed from the same vegetation, under the same climate and on highly similar substrate and topography. The selected ecosystem types corresponded to the most common combinations and intensities of livestock grazing and logging over the original Chaco forest. Sampled ecosystem types were: Primary forest, with no significant logging or livestock grazing in the past seven and five decades, respectively; Secondary forest, which has currently light selective logging and low cattle and goat stocking rates; Closed species-rich shrubland, with a high shrub species diversity and under current moderate to heavy logging and moderate-high cattle and goat stocking rate; Open shrubland strongly dominated by *L. divaricata*, with heavy logging and high cattle and goat stocking rate during the past decades, now decreasing due to declining productivity; and irrigated crop (potato) traditionally cultivated in the area using a gravitational irrigation system. Potato production systems typically include two potato crops per year (in summer and winter) and an average water requirement for the area of 500 mm per crop (Tapella, 2012). We identified four plots (replicates) of each ecosystem type at a distance of at least 1 km from each other, except in the case of the potato culture, where the plots where closer. Each sampling plot had 50 m × 50 m and was as homogeneous as possible in terms of vegetation, soil, and topography. At each site we sampled basic soil parameters including total N (%), pH and texture to confirm similarities among plots. All these procedures were carried out following Sparks (1996).

### 2.2. C Pools quantification

In order to estimate the vegetation C pools, at each 50 m × 50 m plot, we quantified biomass in trees, shrubs, non-woody vegetation, fallen leaf, and fine woody material (fine debris), as well as coarse fallen woody material and standing dead trees (coarse woody debris). The sum of C in trees, shrubs, and non-woody vegetation represented the C pool in the aboveground plant standing biomass compartment (ASB). The sum of C in fine debris and that in coarse woody debris (CWD) represented the C pool in the aboveground plant dead biomass compartment (ADB).

In order to estimate soil C pools, we took a compound soil sample (3 cores) at each plot and at four depths (0–10, 10–30, 30–100, and 100–200 cm) with a soil corer of 10 cm diameter. From each soil sample, we determined soil organic and inorganic C (SOC and SIC, respectively). We expressed all plant and soil C pools in Mg C ha$^{-1}$.

#### 2.2.1. Aboveground plant standing biomass (ASB)

We surveyed all trees greater than 5 cm diameter at breast height in each plot and estimated their dry biomass (ASB, kg) by the allometric model proposed by Chave et al. (2014) using