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# Deforestation impacts on soil organic carbon stocks in the Semiarid Chaco Region, Argentina

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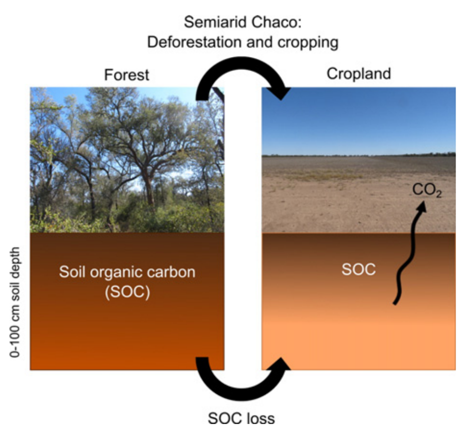
## HIGHLIGHTS

- In the Semiarid Chaco, SOC stock decreased due to cropping after deforestation.
- SOC loss was positively associated with the proportion of soybean in the rotation.
- Forest to cropland conversion modified SOC vertical distributions.
- SOC loss in deeper soil layers was high due to cropping after deforestation.

## GRAPHICAL ABSTRACT

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## ABSTRACT

Land use change affects soil organic carbon (SOC) and generates $CO_2$ emissions. Moreover, SOC depletion entails degradation of soil functions that support ecosystem services. Large areas covered by dry forests have been cleared in the Semiarid Chaco Region of Argentina for cropping expansion. However, deforestation impacts on the SOC stock and its distribution in the soil profile have been scarcely reported. We assessed these impacts based on the analysis of field data along a time-since-deforestation-for-cropping chronosequence, and remote sensing indices. Soil organic C was determined up to 100 cm depth and physically fractionated into mineral associated organic carbon (MAOC) and particulate organic C (POC). Models describing vertical distribution of SOC were fitted. Total SOC, POC and MAOC stocks decreased markedly with increasing cropping age. Particulate organic C was the most sensitive fraction to cultivation. After 10 yr of cropping SOC loss was around 30%, with greater POC loss (near 60%) and smaller MAOC loss (near 15%), at 0–30 cm depth. Similar relative SOC losses were observed in deeper soil layers (30–60 and 60–100 cm). Deforestation and subsequent cropping also modified SOC vertical distribution. Soil organic C loss was negatively associated with the proportion of maize in the rotation and total crop biomass inputs, but positively associated with the proportion of soybean in the rotation. Without effective land use polices, deforestation and agricultural expansion can lead to rapid soil degradation and reductions in the provision of important ecosystem services.

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## 1. Introduction

Soil degradation, food insecurity and climate change are among the major concerns of the last 50 years. These aspects that jeopardize humankind sustainability are strongly interlinked. Land degradation promotes climate change and food insecurity. Therefore, enhancing soil quality is a way to address these global issues (Lal, 2011). Soil organic carbon (SOC) affects nearly all soil properties related to ecosystem functioning (Powlson et al., 2011). Soil organic carbon content is then considered crucial for maintaining soil quality and health (Weil and Magdoff, 2004). Soil organic carbon depletion entails degradation of soil functions that support relevant ecosystem services to societies (Palm et al., 2007; Banwart et al., 2015). Moreover, soils play a key role in global climate change because SOC is the main terrestrial carbon (C) reservoir (Janzen, 2004). Soil organic carbon loss by mineralization leads to C dioxide ($CO_2$) emission under aerobic conditions and methane ($CH_4$) emission under anaerobic condition (Conrad, 1996). Thus, maintenance and restoration of the SOC pool is an important issue to address the global environmental crisis (Lal, 2011).

Deforestation has globally dominated land use changes (Smith et al., 2016) causing severe environmental impacts such as biodiversity loss, climate change, and land degradation (Lal, 2001; Foley et al., 2007). Replacement of forest by crops is the land use change that has the greatest impact on SOC causing losses between 24 and 52%, depending mainly on climate and site conditions (Smith et al., 2016). Tropical dry forests are among the most threatened ecosystems (Hoekstra et al., 2005). Dry forests in the Chaco region of South America represent the second largest forested ecosystem of the continent after the Amazon, and had the highest deforestation rate in the world between 2000 and 2012 (Hansen et al., 2013). The Semiarid Chaco Region of Argentina occupies approximately 29 Mha and deforestation rates have increased exponentially since 1976, reaching the maximum between 2006 and 2012 (2,5%, Vallejos et al., 2014). The main driver of Chaco's deforestation rates has been the agricultural expansion and in particular soybean crops (*Glycine* max (L.) Merr) (Grau et al., 2005; Gasparri et al., 2013).

Land use changes are the second source of $CO_2$ emissions to the atmosphere, only after fossil fuel burning (IPCC, 2013). Since SOC plays a key role in these emissions, the Intergovernmental Panel on Climate Change (IPCC) has developed a simple tool to estimate $CO_2$ emission/removal associated to SOC variation in the 0–30 cm depth (IPCC, 2006). Unfortunately, this tool has not demonstrated a good performance when compared against observed data in the Argentine Pampas (Berhongaray and Álvarez, 2013; Villarino et al., 2014b). Local simulation models have indicated that deforestation in the Chaco region (Fig. 1) accounts for significant $CO_2$ emissions to the atmosphere at a global scale (Gasparri et al., 2008). However, only few studies have estimated the impacts of deforestation on SOC stock changes and their contribution to $CO_2$ emissions in this region. Osinaga et al. (2016) studied soil profile up to 100 cm depth and observed less SOC stock up to 0–80 cm depth after more than 20 years of cropping compared to the native forest. The highest SOC loss (45%) was found in the top 20 cm of the soil. This SOC loss was similar to the reported by Rojas et al. (2016), who only evaluated the top 20 cm of soil. On the other hand, Conti et al. (2014) studied soil profile up to 200 cm depth under different ecosystem types (native and secondary forest, two different shrublands and cropping) in the southern Semiarid Chaco and observed less SOC stock under cropping than under native forest only in the 0–10 cm depth. However, these last two studies (Conti et al., 2014; Rojas et al., 2016)

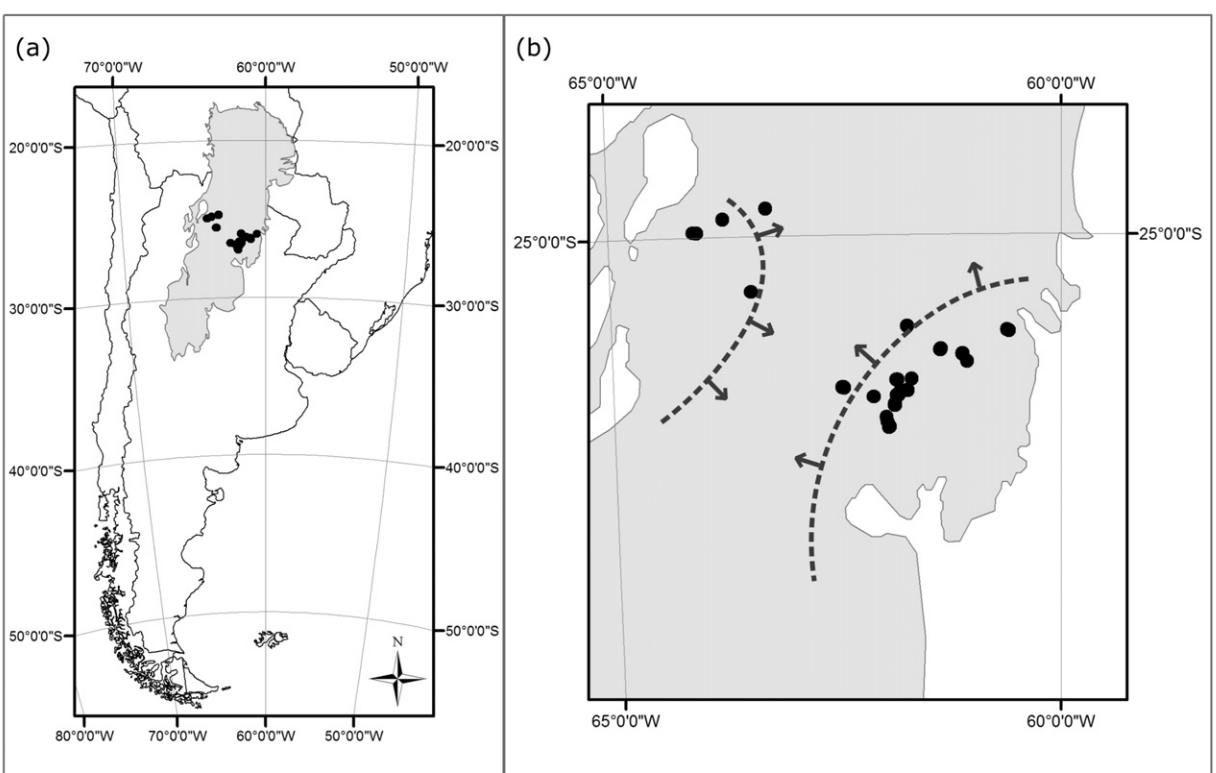


**Fig. 1.** Sampling sites location in the Semiarid Chaco (gray area). Black circles are the sampled sites. Dashed lines with arrows indicate the position of the deforestation frontier and its direction of advance (adapted from Vallejos et al., 2014). Panel (a) shows South American south cone with its national boundaries and panel (b) shows north-central Argentina.

did not report the elapsed time since deforestation and, therefore, their contribution to the understanding of the dynamics of SOC is limited (Dean et al., 2012). Hence, magnitude and soil depth at which SOC stocks changes after deforestation and its relation with cropping time, still remains unclear in the Semiarid Chaco.

Soil organic C stocks result from the balance between C inputs and outputs. Therefore, it is usually positively correlate with the amount of above and below ground plant litter (Powlson et al., 2011; Janzen, 2006). In the soil, organic C is transformed and stabilized as a result of complex interactions between the biotic and abiotic environment, as well as by the dynamics of C input. Therefore, SOC stock is considered an ecosystem property resulting of those complex and scarcely known interactions (Schmidt et al., 2011). The dry forest of the Semiarid Chaco contains high vegetation diversity at different canopy layers (forest, shrub and herbaceous) (Biani et al., 2006). This defines C inputs along the year in terms of the amount, moment and chemical composition. On the contrary, croplands show a homogenous herbaceous canopy layer, with seasonal and less aboveground net primary production (ANPP) (Volante et al., 2012). On the other hand, vertical distribution of SOC is strongly affected by the vegetation type that dominates an ecosystem (Jobbágy and Jackson, 2000). Soil organic C stocks are expected to be more stable in deep soil layers, where C inputs are lower than in the surface (Fontaine et al., 2007), but highly affected by root system growth and structure (Rumpel and Kögel-Knabner, 2011). Therefore, we hypothesize that agricultural expansion over forests may affect not only SOC stocks but also its vertical distribution in the soil profile.

Soil organic C comprises a wide variety of organic substances with different dynamics within the soil (Stevenson and Cole, 1999). Physical fractionation methods have been used in order to separate SOC into pools of functional relevance (Gregorich et al., 2006; Cambardella and Elliott, 1992). Particulate organic C (POC), a fraction with particle size between 53 and 2000 μm, is a SOC pool that has proven to be highly sensitive and useful for the early detection of soil changes due to management. These changes are generally reflected much later in the mineral associated organic carbon fraction (MAOC, fraction with particle size smaller than 53 μm) and total SOC (Wander, 2004; Haynes, 2005). Also, POC contributes to many soil functions such as the maintenance of physical properties (Six et al., 2004), the provision of cellular C and energy for the living microorganisms (Haynes, 2005), and nutrient supply to plants (Willson et al., 2001). Therefore, POC has been proposed as an indicator of labile SOC and soil quality (Wander, 2004; Haynes, 2005).

Large dry forest areas have been cleared in the Semiarid Chaco Region because of cropping expansion and it is likely that this change will continue in the future (Gasparri et al., 2013; Vallejos et al., 2014). Since SOC plays a key role in soil functions and $CO_2$ emissions (Powlson et al., 2011; Stockmann et al., 2013), the main objective of this study was to estimate the impact of forest conversion to cropland on SOC stocks in order to provide tools for decision-making and recommendations in agroecosystems management. The specific objectives were: i) to estimate agricultural impacts on SOC, POC and MAOC stocks along a time-since-deforestation-for-cropping chronosequence, ii) to evaluate the effects of crop type and biomass inputs as drivers of these changes, and iii) to determine possible changes in the vertical distribution of SOC between forest and croplands.

## 2. Materials and methods

### *2.1. Study area and soil sampling*

The Semiarid Chaco Region is a vast plain located in the north-central part of Argentina (Fig. 1). The main soils of the study area (Fig. 1b) are Mollisols and Entisols, with loam-silty and loam textures (Table 1). Natural vegetation of this region is composed by a matrix of xerophytic forest with grassland patches (Morello et al., 2005; Torrella and Adámoli, 2006). Forest plant species are adapted to semiarid climate (Table 1) and usually have small and deciduous leaves and thorns. The most representative species belong to the genera *Schinopsis*, *Prosopis*, *Acacia*, *Aspidosperma,* and *Bulnesia* (Biani et al., 2006).

Changes in SOC stocks due to deforestation were evaluated based on 21 sampling sites distributed across the region. Sixteen of these sites were located in the south-eastern deforestation frontier (Fig. 1) and were sampled in 2012 (five sites), 2013 (eight sites) and 2014 (three sites). Five more sites were located in the north-western deforestation frontier, and were sampled in 2010. In each site, composite samples were taken from soils under remnant forest and a paired adjacent cropping plot. Therefore a pair sampling design was used. Silt + clay and sand contents of paired samples under remnant forest and cropped plots were similar (differences were always less than 5%), hence we assumed soil was the same between land uses in each site. Sampling sites were selected in order to include cropped plots with different elapsed time since deforestation, that is, to include sites with different "cropping ages". Cropping ages ranged between 2 and 40 years, and therefore we were able to construct a chronosequence of years after deforestation and cropping initiation. We found only two plots with a very long cropping history (40 years), but without adjacent remnant forests. In order to overcome this limitation, paired samples to compare with these cropping plots were taken from similar soils under remnant forest, located 55 and 75 km away. Soil samples were collected using a 2 cm diameter soil corer, taking separate composite samples per plot which contained 10 to 24 subsamples. Samples were collected in discrete intervals up to one-meter depth, except for samples taken in 2012 that were collected only up to 30 cm depth.

All cropping plots had been managed under no-till during the last 15 years (and previously with plowing), and species planted were soybean, maize (*Zea mays* L.), and wheat (*Triticum aestivum* L.). Cropping history and crop rotations for each site were estimated based on farmer's records when available and from enhanced vegetation index

**Table 1**
Soil and climate information of the study area (Fig. 1).

| Soil information | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| Soil order[a] | Soil order area[a] (%) | Main soil subgroup[a] | Soil properties[b] | | | | | |
| | | | Clay ($g\ kg^{-1}$) | Lime ($g\ kg^{-1}$) | Sand ($g\ kg^{-1}$) | SOC ($g\ kg^{-1}$) | pH | CEC ($meq\ 100\ g^{-1}$) |
| Mollisols | 50 | Typic Haplustol | 15 | 50 | 36 | 17.6 | 6.3 | 20 |
| Entisols | 26 | Typic Ustifluvent | 14 | 40 | 47 | 11.5 | 7.0 | 16 |
| Alfisols | 24 | Typic Natracualf | 29 | 48 | 23 | 22.6 | 6.9 | 21 |

| Climate information[c] | | |
|---|---|---|
| Annual precipitation (mm) | Mean annual temperature (°C) | Annual potential evapotranspiration (mm) |
| 653 | 22 | 1110 |

SOC: soil organic carbon; CEC: cation exchange capacity.
[a] INTA (1990).
[b] Angueira et al. (2007).
[c] Bianchi and Cravero (2010).

(EVI) time series of each plot derived from MODIS images (see 2.3 section). The proportion of soybean, maize and wheat in the rotation ranged between 53 and 80%, 18 to 30% and 0 to 29%, respectively. In plots with >15 years under continuous cropping, cotton (*Gossypium hirsutum* L.) monocultures were sown under conventional tillage originally and then replaced by soybean, maize and wheat under no-till during the last 15 years before sampling.

### 2.2. Laboratory analysis

Total wet weight of each sample was recorded. After homogenization, an aliquot was taken from each fresh soil sample and then oven dried at 105 °C to determine soil moisture content. The rest of the sample was oven dried at 30 °C and sieved through 2 mm mesh, identifiable plant material was eliminated manually, and soil was stored until fractionation and analysis. Total wet weight of each sample was corrected with soil moisture to calculate total dry weight. To estimate soil bulk density (BD), the total dry weight was divided by the total volume (volume of each soil core * number of subsamples in the composite sample).

Dry soil samples from 0 to 30 cm depth were re-grounded and physically fractionated according to Cambardella and Elliott (1992) into the POC and MAOC. Soil organic C contents were determined in the total soil mass (i.e. SOC) and in the MAOC by wet combustion, maintaining the reaction temperature at 120 °C for 90 min (Schlichting et al., 1995). Particulate organic C concentration was obtained by the difference between SOC and MAOC (Cambardella and Elliott, 1992), except for the samples taken in 2010. In this case the organic C content was determined in the mineral associated fraction (MAOC) and in the particulate fraction (POC) by dry combustion with an automatic elemental analyzer (Carlo Erba NA 1500 Elemental Analyzer) in the Stable Isotope Laboratory, Duke University, Durham, NC, USA. Therefore, the SOC was calculated as the sum of MAOC and POC. To compare both methods, we selected 52 soil samples originally analyzed with the wet combustion method, covering a wide range of SOC concentration (0.59 to 6.92%), and were also analyzed by the dry combustion method (LECO C/N Analyzer). Non-significant differences were found between methods ($p = 0.58$) and organic C concentrations among methods were strongly associated ($R^2 = 0.98$). These results were similar to those reported by Eyherabide et al. (2014). An aliquot from each dry sieved sample (~2 g) was acidified with diluted HCl solution in order to test carbonate presence. Only a few samples from deep soil layers (30–60 and 60–100 cm depths) showed reaction. Since inorganic carbon does not interfere with SOC determination by wet combustion (Kalra and Maynard, 1991), this method was used in the samples that contained carbonates.

### 2.3. Comparison of SOC, POC and MAOC stocks between forest and croplands

Because soil BD of croplands were always higher than those of paired native forests, SOC comparisons were expressed in an equivalent soil mass (Davidson and Ackerman, 1993). Given that soil compaction under agriculture increased with cropping age (Fig. 2), we used this relationship to estimate soil BD of samples taken in 2012, when BD was not measured. To report SOC, POC and MAOC contents in an equivalent soil mass, soil sampling depths under croplands were corrected using the following equation (Solomon et al., 2002; Eclesia et al., 2012):

$$CD = (BD_F/BD_C)\ D \tag{1}$$

where CD is the corrected soil depth (cm), $BD_F$ is the BD of forest ($g\ cm^{-3}$), $BD_C$ is the BD of cropping situation ($g\ cm^{-3}$) and D is the sampled soil depth (cm). Soil organic C, POC and MOAC stocks were estimated with Eq. 2:

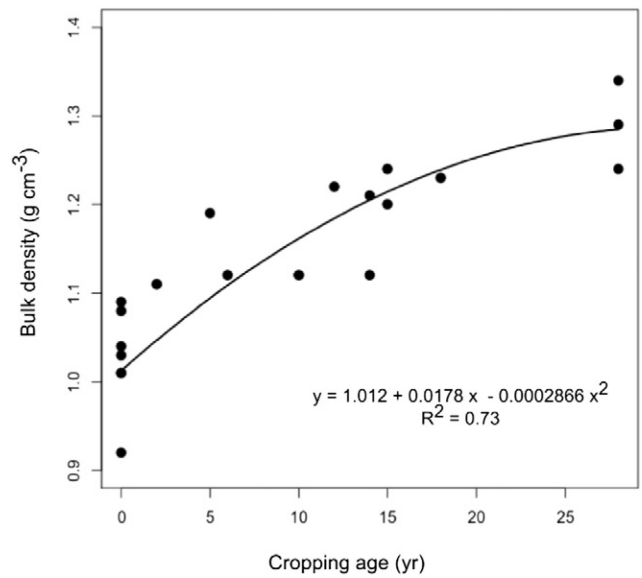


**Fig. 2.** Bulk density and cropping age relation at 0–30 soil depth.

$$C_F = (W_F/W_T)\%C_F\ BD\ (CD\ or\ D) \tag{2}$$

where $C_F$ is C stock of the fraction ($Mg\ ha^{-1}$), $W_F$ is soil fraction weight (g), $W_T$ is total soil weight (g), $\%C_F$ is C concentration of the fraction. Corrected D (CD) was used for cropping situations and D, for native forests.

Soil organic C changes across time after deforestation were described with a logarithmic linear model (Eq. 3):

$$y_s = \alpha + \beta \ln(x) + \varepsilon \tag{3}$$

where $y_s$ is organic C stock change under cropping respect to the corresponding paired native forest (%), $\alpha$ is the mean C stock change at the first year of cropping, $\beta$ is a model parameter, ln is natural logarithm, x is cropping age (yr) and $\varepsilon$ is the experimental error. Crop proportions in the rotation and average C inputs were later added to this model (Eq. 3) as new variables, in order to evaluate their role for explaining changes on SOC, POC and MAOC. Interaction effects were tested and they were incorporated when significant ($p < 0.05$). The adjusted $R^2$ was used to define whether the simpler model (Eq. 3) was enhanced by adding new variables (i.e. crop residue input, proportion of maize in the rotation, etc.). Statistical analyses were performed with the R software (R Core Team, 2013).

### 2.4. Satellite images analysis

We used the model proposed by Monteith (1972) to estimate crop ANPP, expresed as dry matter per hectare. This model assumes that the ANPP is proportional to the fraction of photosynthetically active radiation absorbed by green vegetation (FAPAR), the incoming photosynthetically active radiation (PAR) and the radiation use efficiency (RUE) (Eq. 4).

$$ANPP = FAPAR * PAR\left(MJ\ m^{-2}\ d^{-1}\right) * RUE\left(kg\ dry\ matter\ MJ^{-1}\right) \tag{4}$$

The FAPAR can be estimated through several spectral indices provided by remote sensors aboard satellites, such as the EVI. In this study we used an empirical approach to estimate FAPAR from EVI using the linear relationship proposed by Grigera and Oesterheld (2006). We used the MOD13Q1 MODIS product (scene h12v11) that provides EVI data for the period 2000–2015 every 16 days (23 images per year) with a spatial resolution of 250 m. We also used the QA band to exclude pixels with cloud presence, shadows, and/or aerosols in the atmosphere at the time that the sensor registered earth's surface radiance. Enhanced vegetation index values of such low-quality pixels (with cloud presence, shadows, and/or aerosols in the atmosphere) were deleted and replaced by the average EVI value from the immediately preceding and subsequent dates (Ma et al., 2013). For each sampling site, we selected the MODIS pixels that fell completely within the crop or forest plots. Incomig solar radiation was obtained from a South America radiation solar data base (DSA/CPTEC/INPE), an used to estimate PAR (as 48% of the incomig radiation (McCree, 1972; Blackburn and Proctor, 1983)). Aboveground biomass inputs were estimated based on ANPP estimates and using published harvest indexes (HI) and radiation use efficiencies (RUE), while belowground biomass inputs were estimated based on published root:shoot ratios (Table 2).

### *2.5. Evaluation of SOC vertical changes in the soil profile*

Cumulative SOC proportion from surface up to each sampling depth was calculated and expressed as the proportion respect to total SOC at 0–100 cm. Vertical distribution of cumulative SOC proportion was described with six mathematical models (Table 3). Data from different depths of the same soil profile are not assumed to be independent from each other (Jobbágy and Jackson, 2000). Hence, autoregressive structure of first order between observation errors from the same soil profile was modeled and random effects of soil profile were incorporated in some of the model parameters. Therefore, correlation structures modeled had two components: one assuming constant correlation for all observations from the same profile and another one assuming that correlation between errors decreased with the increasing depth distance. These mixed effects models were fitted with the *nlme* function from *nlme* package, version 3.1–121 (Pinheiro et al., 2015). Best models were selected through graphical analysis of the residuals and restricted maximum likelihood ratio test.

Best fitted models were validated by comparing their estimates against independent data obtained from the Geographic Information System of Santiago del Estero (SigSE) (Angueira et al., 2007). Almost the whole area of Santiago del Estero province belongs to the Semiarid Chaco and therefore, soils from this database are similar to the sampled ones (Fig. 1). Texture and SOC concentration in soil profiles under different land uses are described in this database, although BD values are not provided. Hence, in order to calculate SOC stocks, BD was estimated with pedotransfer functions based on soil texture (Rawls, 1983; Villarino et al., 2014a). Selected soil profiles corresponded to 19 croplands and 11 remnant forests. This observed data was regressed over the model predictions and 95% confidence intervals (CI) of the parameters were calculated. Statistical analyses were performed with the R software (R Core Team, 2013).

## 3. Results

Total SOC, POC and MAOC stocks showed variable changes during the first years after deforestation, but decreased markedly at all sites with increasing cropping age (Table 4 and Fig. 3). Estimated losses after 40 years of continuous cropping compared to the native forest at the top 5 cm of soil were 61, 80 and 34% for SOC, POC and MAOC, respectively (Table 4 and Fig. 3). Particulate organic C was the most sensitive fraction. After 10 years of continuous cropping, POC loss at the top 5 cm of soil was 54%, while MAOC loss for the same period was 9% (Fig. 3). Between 10 and 40 years after deforestation, SOC loss for the first 30 cm of the soil was around 30%, again with grater POC loss (near 60%) and smaller MAOC loss (near 15%). These C losses account for nearly 19 Mg C $ha^{-1}$ of SOC, composed by 12 Mg C $ha^{-1}$ of POC and 7 Mg C $ha^{-1}$ of MAOC (Fig. 4). Soil organic C losses in deep soil layers (30–60 and 60–100 cm) were similar to the losses in the surface soil (0–30 cm) (Fig. 3). However, SOC stock losses at 30–100 cm depth account for 30 Mg C $ha^{-1}$, and this value is around 11 Mg C $ha^{-1}$ higher than the SOC stock loss at 0–30 cm (Fig. 4).

Soil organic C losses after deforestation at 0–5 and 0–30 cm depths were associated with time after deforestation, but also with the percentage of maize and soybean in crop rotations and with the average crop biomass inputs (Table 5). Equation coefficients were positive on all fitted models between SOC and maize-proportion in the rotation or total crop biomass inputs (Table 5). This indicates that increases of maize percentage and biomass inputs reduces SOC losses. On the contrary, equation coefficients for soybean-percentage in the rotation were always negative (Table 5). Therefore, increases in soybean frequency leads to SOC (at 0–30 depth), POC (at 0–5 depth) and MAOC (at 0–5 depth) losses.

Beta and Potential models (Table 3) were the only models that showed good performances in describing SOC vertical distributions for croplands and native forest, respectively (Fig. 5). Because SOC vertical distributions were described with different models for forest and croplands, it is likely that land use change modified SOC vertical distributions. The major difference in SOC vertical distribution between native forest and croplands was around 40 cm of soil depth (Fig. 5). At this depth, cumulative SOC proportions were 0.57 and 0.64 in native forest and cropping situations, respectively (Fig. 5). Therefore, below 40 cm soil depth, native forests accumulated a larger proportion of SOC (0.43) than cropping situations (0.36).

**Table 2**
Average values of radiation use efficiency (RUE), harvest indexes (HI) and root:shoot ratio of wheat, soybean and maize, used to estimate above and belowground crop biomass.

| Crop | RUE | HI | Root:shoot |
|---|---|---|---|
| Wheat | 0.73 | 0.35 | 0.88 |
| Soybean | 0.86 | 0.45 | 0.88 |
| Maize | 1.64 | 0.46 | 0.85 |
| References | Sinclair and Muchow, 1999 | Calderini et al., 1995, Baigorri et al., 1995, Andrade et al., 1996 | Buyanovsky and Wagner, 1986 |

**Table 3**
Models fitted to describe cumulative proportion of soil organic carbon (SOC) (y) as a function of soil depth (x).

| Model | Equation[a] | Reference |
|---|---|---|
| Beta | $y = 1 - \beta^x$ | Jobbágy and Jackson (2000) |
| Gompertz | $y = \exp.(-\alpha \exp(-\beta x))$ | Nelder (1961) |
| Log | $\ln(y) = \alpha + \beta x$ | Jobbágy and Jackson (2000) |
| Log-log | $\ln(y) = \alpha + \beta \ln(x)$ | Jobbágy and Jackson (2000) |
| Logistic | $y = 1/(1 + \exp(-\alpha (x - \beta)))$ | Franses (1994) |
| Potential | $y = \alpha x^\beta$ | Berhongaray et al. (2013) |

[a] ln is natural logarithm, exp. is exponential function, α y β are model parameters.

**Table 4**
Summary of statistics of logarithmic linear models fitted for soil organic carbon (SOC), particulate organic carbon (POC) and mineral associated organic carbon (MAOC) changes respect to the corresponding paired native forest as a function of cropping age.

| | | α parameter | | β parameter | | | |
|---|---|---|---|---|---|---|---|
| SOC fraction | Soil layer | Estimated | SE[a] | Estimated | SE[a] | $R^2$ | p-Value |
| SOC | 0-5 cm | 12.78 | 7.82 | −20.07 | 2.98 | 0.71 | < 0.0001 |
| SOC | 0–30 cm | 33.03 | 6.76 | −20.75 | 2.58 | 0.77 | < 0.0001 |
| SOC | 30–60 cm | −10.35 | 7.74 | −9.06 | 2.94 | 0.51 | 0.013 |
| SOC | 60–100 cm | −4.56 | 11.64 | −13.98 | 4.41 | 0.53 | 0.011 |
| POC | 0–5 cm | −10.54 | 9.9 | −18.89 | 3.77 | 0.57 | < 0.0001 |
| POC | 0–30 cm | 38.28 | 12.9 | −32.91 | 5.04 | 0.7 | < 0.0001 |
| MAOC | 0–5 cm | 32.91 | 8.61 | −18.22 | 3.28 | 0.62 | < 0.0001 |
| MAOC | 0–30 cm | 29.59 | 7.4 | −14.08 | 2.89 | 0.57 | 0.0001 |

[a] Standard error.

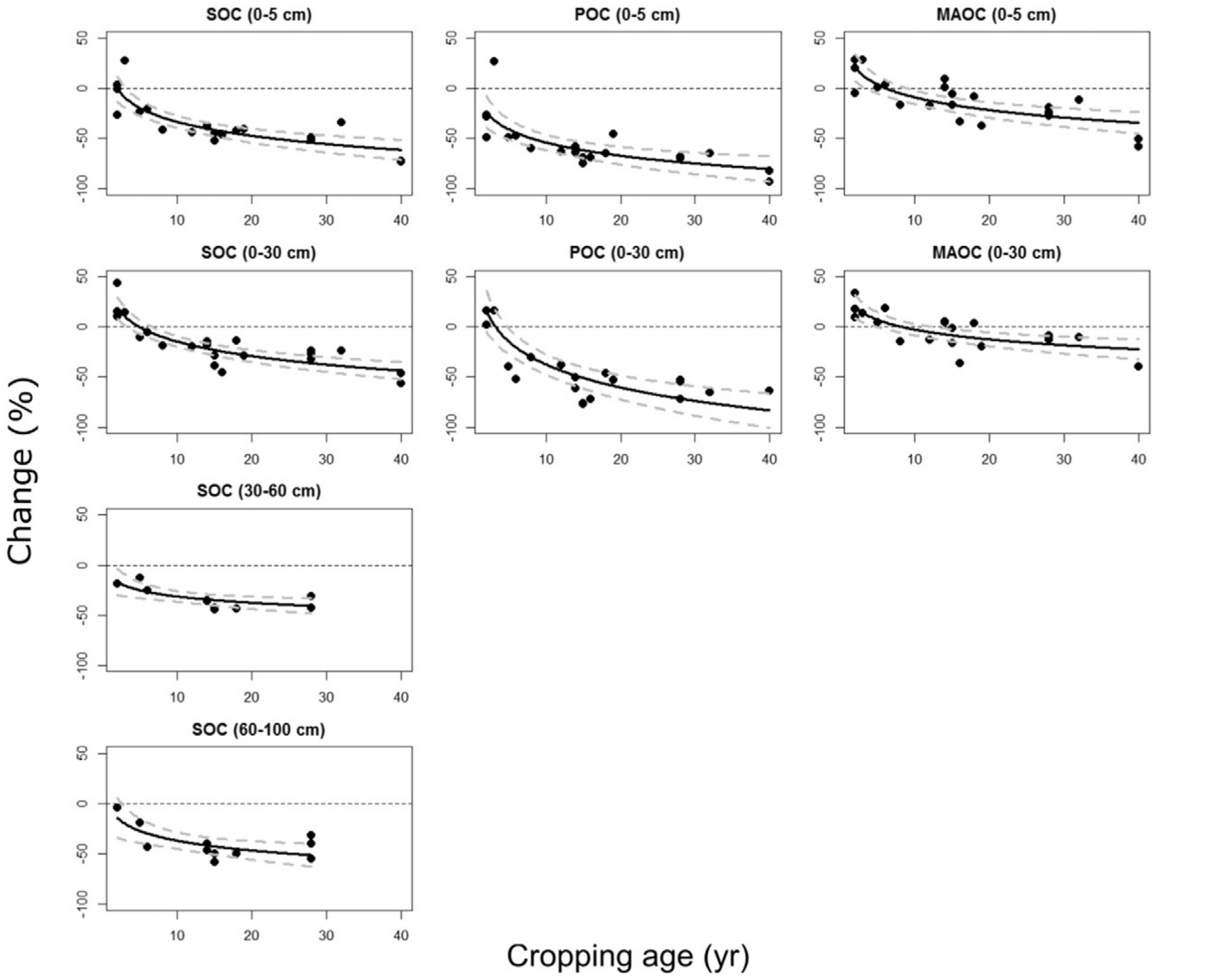


**Fig. 3.** Relative changes in soil organic carbon (SOC), particulate organic carbon (POC) and mineral associated organic carbon (MAOC) after deforestation, as a function of cropping age at different soil depths (showed within parentheses). Black lines are the fitted models (Table 4) and gray dashed lines are the 95% confidence intervals of the mean.

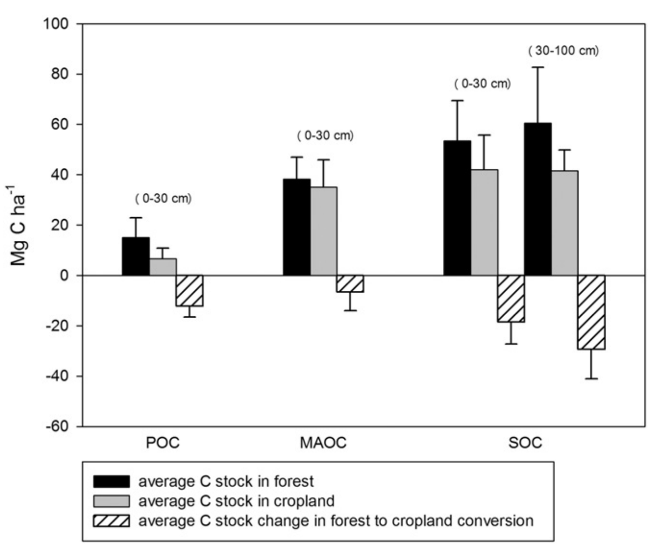


**Fig. 4.** Average stocks of total soil organic C (SOC), particulate organic C (POC) and mineral associated organic C (MAOC) in forest and in croplands, and average C stock changes in forest to cropland conversion. The croplands sites had cropping ages between 12 and 40 years old for 0–30 cm soil depth and between 14 and 32 years old for 30–100 cm soil depth. Soil depths are showed within parentheses above the bars.

Models selected to describe SOC vertical distributions had a good performance when their estimates were compared with independent data ($R^2 = 0.93$, Fig. 6). In the forest model validation, the intercept was significantly different from zero (CI: 0.01 to 0.14), while the slope was not different from 1 (CI: 0.88 to 1.09) (Fig. 6). Therefore, forest models had a slight tendency to underestimate SOC accumulated proportions. On the other hand, cropland model was unbiased, since the intercept and slope were not significantly different from 0 and 1, respectively (Fig. 6).

## 4. Discussion and conclusion

### *4.1. Changes in SOC stocks and its vertical distribution after deforestation*

Stored C in total biomass (above and belowground biomass) of Semiarid Chaco forest has been estimated as 59 Mg C ha$^{-1}$ (Gasparri et al., 2008). In this study, we estimated that the average stored C in total biomass under cropping was 6 Mg C ha$^{-1}$, and the remaining potential C input after harvest was around 3 Mg C ha$^{-1}$ (data not shown). In the forest, the observed SOC stock in the first meter of soil was 114 Mg C ha$^{-1}$ (Fig. 4), almost twice of the total C biomass. The average estimated SOC losses after >10 years of continuous cropping (48 Mg C ha$^{-1}$, Fig. 4) was slightly lower than C stored in biomass (59 Mg C ha$^{-1}$ under forest, Gasparri et al., 2008). Therefore, deforestation led

**Table 5**
Equation parameters of models fitted between SOC, POC and MAOC (%) changes after cropping as a function of cropping age (x, yr), average crop biomass input (BM, Mg $ha^{-1}$ $yr^{-1}$), maize percentage in crop rotation (M, %), wheat percentage in crop rotation (W, %) and soybean percentage in crop rotation (S, %).

| Fraction | Soil depth (cm) | Fitted model | Parameters estimated A | $\beta_1$ | $\beta_2$ | $\beta_3$ | Adjusted $R^2$ of the fitted model | Adjusted $R^2$ of the original model (Eq. 3) |
|---|---|---|---|---|---|---|---|---|
| SOC | 0–5 | SOC = α + β1 ln(x) + β2 M + ε | −17.75 | −17.88 | 1.13 | | 0.75 | 0.69 |
| | 0–30 | SOC = α + β1 ln(x) + β2 BM + ε | 6.43 | −20.13 | 3.03 | | 0.78 | 0.76 |
| | | SOC = α + β1 ln(x) + β2 S + ε | 70.23 | −21.34 | −0.55 | | 0.78 | |
| POC | 0–5 | POC = α + β1 ln(x) + β2 M + ε | −47.84 | −16.21 | 1.38 | | 0.63 | 0.55 |
| | | POC = α + β1 ln(x) + β2 S + ε | 29.86 | −19.52 | −0.59 | | 0.56 | |
| | 0–30 | POC = α + β1 ln(x) + β2 DM + β3 xBM + ε | −118.4 | 28.41 | 19.05 | −7.5 | 0.82 | 0.69 |
| | | POC = α + β1 ln(x) + β2 S + β3 xS + ε | 350.17 | −135.2 | −4.56 | 1.49 | 0.80 | |
| MAOC | 0–5 | MAOC = α + β1 ln(x) + β2 C + ε | 6.12 | −17.59 | 3.05 | | 0.61 | 0.6 |
| | | MAOC = α + β1 ln(x) + β2 S + ε | 74.09 | −18.87 | −0.6 | | 0.62 | |

* α, β1, β2 and β3 are model parameters; ε is the experimental error; xBM and xS are the interactions between x and BM, and x and S, respectively.

to a loss of approximately 104 Mg C $ha^{-1}$ (56 Mg $ha^{-1}$ in total biomass plus 48 Mg $ha^{-1}$ in SOC stock).

Plant root systems play a major role in SOC allocation (Jobbágy and Jackson, 2000, Rumpel and Kögel-Knabner, 2011). Therefore, it is likely that native vegetation type could define soil depths where SOC contents are affected by cropping. In regions of Argentina where long-lasting croplands (>25 year old) expanded over grassland, SOC losses occurred up to 0–50 cm depth, with major changes taking place at 0–25 cm depth (Berhongaray et al., 2013). In the Semiarid Chaco, croplands replaced native forest and this change led to high SOC losses at 0–30, 30–60 and 60–100 cm depths (Fig. 3).

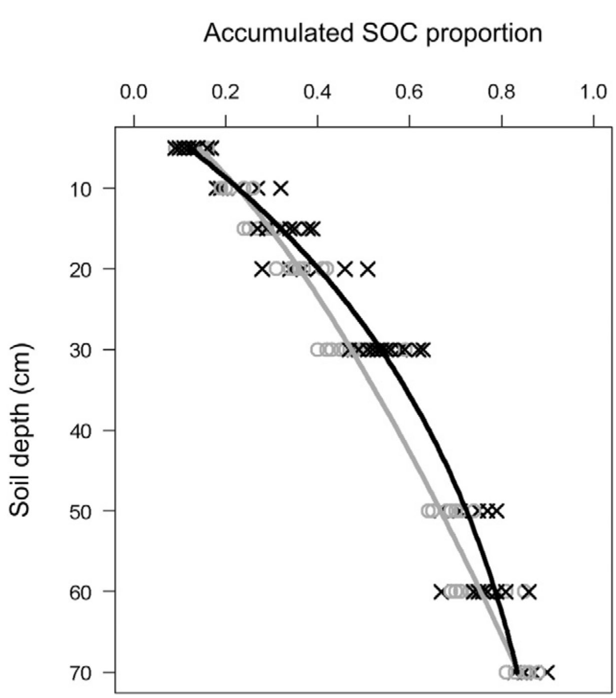


**Fig. 5.** Models selected to describe soil organic carbon (SOC) vertical distribution under forest and cropland. The black line is the beta model fitted for soil under croplands ($y = 1 - 0.975^{x}$) and the gray line is the potential model fitted for soil under forest ($y = 0.048\ x^{0.67}$). Crosses correspond to data of soil under cropland and circles correspond to data of soil under forest. The independent variable is the soil depth.

The most widespread deforestation method in the study area consists in land clearing with heavy bulldozers, burning the remaining vegetation and then plowing down the residues (Boletta et al., 2006). This practice produces strong changes in soil, such as surface exposure to precipitation, wind and solar radiation, aeration and temperature increases, root removal, and aboveground biomass and charcoal incorporation. These changes severely affect SOC dynamics in different ways. Surface exposure to climatic conditions, aeration and temperature increases, and root removal could deplete SOC stocks, either by mineralization increases or by water and wind erosion. However, aboveground biomass and charcoal incorporation could have the opposite effect and this could explain the higher SOC stocks during the first 2–3 years of cropping (Fig. 3). Soil organic C, POC, and MOAC (0–5 and 0–30 depths) losses under longer periods of cropping are probably due to decreases in the ANPP in croplands (Volante et al., 2012), increases in the mineralization rate due to higher temperature and aeration, and wind erosion (Rojas et al., 2013). In a global review performed by Murty et al. (2002), it was shown that relative SOC changes after deforestation for cropping ranged between 0 and −60% and mean SOC changes tended to stabilize at −30% after 10 years of cropping. Thus, SOC changes observed in our study are in agreement with these global estimates (see Fig. 3).

Crop rotation and crop BM inputs are major drivers of SOC dynamic (Studdert and Echeverría, 2000). At 0–5 cm soil depth, SOC losses decreased as maize percentage increased (Table 5). Generally, maize contributed with high amounts of aboveground residues that protects the soil from erosion and promotes intense decomposition and humification processes near the soil surface, especially under no-till cropping (Mazzilli et al., 2014). In agreement with other authors, crop BM inputs were negatively associated with SOC losses (Álvarez and Lavado, 1998; Studdert and Echeverría, 2000; Lal, 2011) and POC was the most sensitive fraction to cropping (Buyanovsky et al., 1994; Dominguez et al., 2009; Ferrary Laguzzi et al., 2014), suggesting that this fraction could be a good estimator of labile SOC (Wander, 2004). Since POC is considered a key soil quality indicator (Haynes, 2005), it is very likely that high losses under cropping are indicating strong soil degradation in the region. In addition, at 0–5 depth, POC depletion was negatively associated to maize percentage in the rotation but increased with soybean percentage (Table 5). Similar results but for total SOC stocks were reported by Studdert and Echeverría (2000) in the Pampas region of Agrentina. Soybean had less ANPP and, therefore, lower amount of crop residues leftover than maize. This, together with the low C/N ratio of its residues (Studdert and Echeverría, 2000; Mazzilli et al., 2014), could lead to a decrease in POC stock.

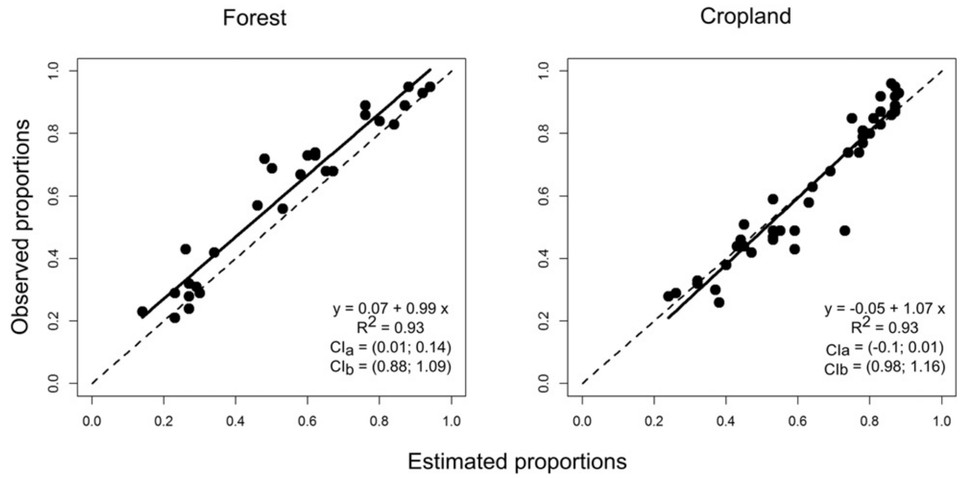


**Fig. 6.** Comparison of estimated proportions with Beta model for croplands and Potential model for forests against independent observed data, obtained from the Geographic Information System of Santiago del Estero (SigSE) (Angueira et al., 2007). Regression equation of observed vs. estimated values, $R^2$ and confidence intervals at 95% for intercept ($CI_a$) and slope ($CI_b$) are shown in the bottom right of each panel. Full lines are the regressions lines and dashed lines indicate the y = x lines.

### *4.2. Vertical changes in SOC and its implications for $CO_2$ emission estimates*

Land use changes from forests to cropland affected SOC stocks not only in the surface soil but also in deeper soil layers, with implications on the estimation of the SOC stocks and $CO_2$ emissions using standard IPCC protocols (IPCC, 2006). $CO_2$ emissions are likely to be underestimated with this method in the Chaco region because the IPCC tool considers only the 0–30 cm soil depth (IPCC, 2006). For example, based on the IPCC (2006) tool, our region is estimated to have SOC stocks near 38 Mg $ha^{-1}$ under native forest and 30.7 Mg $ha^{-1}$ under no-till cropping systems with medium C inputs (IPCC, 2006). This results in an estimated $CO_2$ emission of 26.8 Mg $CO_2$ $ha^{-1}$ (7.3 Mg C). Our vertical distribution models (equations in Fig. 5) estimate that the cumulative SOC proportion at 0–30 cm depth is 0.47 under forest and 0.53 under cropland (Fig. 5). Therefore, SOC stock estimation with the IPCC tool corrected to 0–100 cm depth with our models would be around 81.1 and 57.7 Mg $ha^{-1}$ under forest and cropland, respectively. When considering 0–100 cm depth, the estimated $CO_2$ emissions are 85.7 Mg $CO_2$ $ha^{-1}$ (23.4 Mg C), 320% larger than the emissions estimated from 0 to 30 cm depth. The IPCC tool assigns SOC stocks according to soil and climate criteria but does not differentiate between vegetation types. Therefore, we suggest to explore the possibility of adding vegetation transitions in the IPCC tool in order to define SOC stocks under native vegetation and the soil depth at which SOC stocks are affected by cropping.

### *4.3. Implications for soil management decisions and regulations*

The cropping systems that are expanding into the Semiarid Chaco are the same as those cultivated in temperate regions of Argentina (mainly the Pampas Region), where soil conditions are very suitable for crop and livestock production (Hall et al., 1992). The incorporation of these cropping systems in the Semiarid Chaco has led to a rapid and high SOC loss, particularly in the POC fraction, indicating high soil vulnerability. Moreover, cropping expansion was accompanied by increases in soybean monoculture (Gasparri et al., 2013), and our results suggest an increase in soil vulnerability when there is a higher soybean percentage in the rotation.

In summary, our analysis showed that the conversion of forests to cropland and the proportion of different crops in the rotation have a strong effect on SOC, a critical component of environmental sustainability. Therefore, deforestation and agricultural expansion in the Semiarid Chaco without effective land use polices can lead to rapid soil degradation and reductions in the provision of important ecosystem services (Powlson et al., 2011). In 2007, Argentina passed the Law No. 26,331, Native Forest Protection Act, which requires provinces to design and implement land-use plans to restrict deforestation in areas of high and medium conservation value (García Collazo et al., 2013). In addition to a weak implementation, an important drawback in the design of provincial land use plans lies in a poor consideration of the below-ground processes that strongly affect environmental sustainability. To overcome this limitation, future land-use plans should complement existing requirements of the Forest Law in two directions. First, land allocation to categories of conservation value (low, medium, high) should incorporate indicators of soil quality and health as criteria for land zoning. Soil organic C can be a suitable indicator to incorporate soil quality as criteria for land zoning (Weil and Magdoff, 2004; Powlson et al., 2011; Rojas et al., 2016). However, to correctly incorporate this indicator in land use decisions we need to know the threshold value of SOC contents beyond which soil functionality becomes impaired. Unfortunately, there is low consensus about this value (Loveland and Webb, 2003), although there is some agreement that it is strongly influenced by soil texture (Stockmann et al., 2013). Therefore, future research should be oriented towards finding SOC thresholds under different soil texture conditions, in order to obtain an easy-to-measure indicator on which to base decisions for land-use planning.

Second, land-use plans should not only regulate the conversion of forests to cropland, but also the management factors involved in cropping systems. Here we showed that crop rotations have a significant effect on SOC, POC and MAOC changes and therefore should be incorporated in land-use plans (e.g. as in the Uruguayan legislation regarding crop rotation plans). Due to the fragility of Semiarid Chaco environment, cropping in this region should include management practices at the landscape scale and beyond crop rotations to avoid soil degradation, such as the conservation of remnant forest fragments to protect soil from wind erosion, the restoration of degraded forest fragments to reestablish connectivity and regeneration capacity, and the incorporation of cover crops in fallow periods. Very little is known about these management practices at landscape-scale and, therefore, future research in the Semiarid Chaco should also address this knowledge gap.

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