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Soil volume and carbon storage shifts in drained and afforested wetlands of the Paraná River Delta

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Abstract Wetland ecosystems have a high carbon storage potential as a result of high primary productivity and low decomposition rates dictated by water saturation. In the herbaceous wetlands of the Paraná River Delta, drainage and afforestation with poplars represents one of the dominant land uses. We explored the effects of these interventions on the volume and carbon storage of the young sedimentary soils of the region. At three sites we identified paired stands occupying similar landscape positions and soil types but subject to natural flooding and covered by natural herbaceous communities or drainage and flood control by dikes and covered by poplar plantations established 12, 17 and 19 years ago. Soil sampling at these sites revealed a reduction of the litter compartment (−86 %) and decreasing volume and porosity of its underlying mineral layer (0–10 cm in the wetland reduced to 0–4 cm in the plantation). Our comparisons of carbon storage accounted for these volumetric shifts by using accumulated mineral mass rather than depth as a reference, showing that tree plantations gained in the mineral soil (22 Mg C ha^{-1}) almost as much as what they lost in the litter. These gains were particularly large at intermediate depths (4–43 cm in the plantations) were soil porosity remained unaffected and C was raised by 64 % explained by (1) the pulse of inputs from overlaying litter and organic layers subject to rapid decomposition and mobilization after drainage and (2) root colonization, since tree plantations had 75 % of their fine root biomass at these intermediate soil depths, whereas roots in the wetlands did not explore the mineral soil profile and were completely confined to the organic layer. A neutral C balance following wetland drainage and afforestation resulted from the opposing effects of aeration, favoring decomposition in the organic layer, root colonization and organic matter stabilization, favoring its accumulation in the mineral soil.

Keywords Carbon sequestration · Soil compaction · Flood control · *Populus deltoides* · Argentina

Introduction

In spite of the relatively small fraction of the continental area that they cover, wetlands are among the most valuable ecosystems in terms of their services

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to society (Costanza et al. 1997). As interfaces between aquatic and terrestrial systems, wetlands play key roles for water provision, hydrological regulation, carbon storage and biodiversity conservation. While the productive use of wetland territories has involved their drainage and replacement by non-inundated systems, including crops and pastures found in neighboring "dry" areas; the realization of their value as "wet" ecosystems is questioning this widespread trend (Mitsch and Gosselink 2007). Here we explore the effects of the conversion of natural deltaic wetlands into "dry" commercial forestry systems on soil carbon storage.

The development of wetland soils, in close interaction with vegetation succession, is influenced by sedimentation/erosion rates and water saturation conditions (Mausbach and Parker 2001). Both forces can be strong and variable on river deltas where high sediment transport and water level shifts take place. Hydrological fluctuations dictate the alternation of aerobic and anaerobic conditions (Stepanauskas et al. 1996; Davidsson and Leonardson 1997) regulating biogeochemical transformations and cycling (Gutknecht et al. 2006). In addition to these controls, human activities influence (intentionally or not) the hydrology of river deltas through the development of local infrastructure and the replacement of vegetation (Kercher and Zedler 2004).

The conversion of river deltas from a "wet" to a "dry" condition is often implemented through the simultaneous development of water evacuation infrastructure, which includes drains, sometimes coupled with pumps, at the plot level and channels at the landscape to regional level; and flood control infrastructure, that includes barriers such as dikes capable of preventing surface water inputs (Vörösmarty et al. 2009). These interventions depress water table levels favoring soil aeration and making land suitable to grow crops with low waterlogging tolerance and to support the transit of machinery. From a biogeochemical perspective, the transition from anaerobic to aerobic conditions favors the oxidation of soil materials, including faster respiration and release of carbon to the atmosphere (Megonigal et al. 2005), and the loss of organic-bound nutrients from the ecosystems towards the hydrological network (Sigua et al. 2006, 2009; Dunne et al. 2007).

Soil changes accompanying wetland drainage involve physical effects, such as compaction and subsidence (Hargreaves et al. 2003; Kool et al. 2006; Gebhardt et al. 2009; Hamza and Anderson 2005), and biogeochemical effects, which include changes in decomposition rates and oxidation states and mobility of nutrient species (Olila et al. 1997; Hargreaves et al. 2003; Lal 2007). In wetlands, organic materials are an important component of the soil matrix and their transformation can play a significant role in soil fertility and biogeochemical cycles from the local to the global level. While drainage in these systems favors carbon release to the atmosphere, it can also promote the incorporation of part of surface organic carbon stocks into the mineral soil, where it can become stabilized by soil minerals (Oades 1988; Chan and Heenan 1999; Evrendilek et al. 2004; Cheng 2009; Gartzia-Bengoetxea et al. 2009).

Beyond drainage and flood control interventions, vegetation changes can trigger soil changes, including shifts in carbon storage associated with changes in primary productivity and above/belowground allocation. The replacement of natural herbaceous ecosystems by tree plantations can be accompanied by soil carbon storage reductions that are particularly important under humid climates. These carbon losses, which are on average 10 %, are likely resulting from lower belowground biomass production (Berthrong et al. 2012). On the other hand, some studies suggest soil organic matter increases where tree plantations gain a better access to resources (water and nutrients) than grasslands, achieving higher primary productivity rates (Jobbágy and Jackson 2004; Nosetto et al. 2006). No herbaceous wetland have been included in these studies of grassland to tree plantations afforestation as far as we are aware. The combination of drainage and tree establishment could favor the penetration of roots into a mineral soil that lost its saturated condition, favoring the incorporation of organic matter into deep soil layers (Brimhall et al. 1991, 1992).

When important soil volumetric changes take place, as is commonly observed following drainage and, occasionally, vegetation changes, it becomes indispensable to quantify them to achieve correct carbon or nutrient stock balances (Davidson and Ackerman 1993; Ellert and Bettany 1995; Wuest 2009; Toriyama et al. 2011). In these cases, comparisons at similar depth intervals are avoided (a given depth range represents different segments of the initial soil column if it suffered volumetric changes) and the cumulative content of a resistant mineral is used to adjust depths for comparisons (Brimhall et al. 1991, 1992; Anderson et al. 2002). The mineral mass of soils is appropriate for this adjustment when dissolution/precipitation or erosion/deposition processes are unimportant and volume changes are due to porosity changes caused by organic matter losses/gains (Ellert and Bettany 1995; Ellert et al. 2001; Vandenbygaart 2006; Vandenbygaart and Angers 2006; Lee et al. 2009; Toriyama et al. 2011).

In this paper we characterize organic carbon content (litter and soil) and soil volume shifts in response to drainage and tree planting on herbaceous wetland ecosystems in the lower Paraná River Delta of Argentina. Two sets of alternative hypotheses on possible volumetric and organic processes guide our observations. As a result of porosity loss and compaction following the transition from saturated to unsaturated conditions, we expect a volumetric reduction of the mineral soil column (Hypothesis 1A). However, the aeration that accompanies drainage combined with the establishment of deep rooted trees may enhance the porosity and cause a volumetric expansion of the mineral soil column as roots colonization and senescence cycles start to take place (Hypothesis 1B). Regarding organic carbon stocks in mineral soil, we expect reductions stemming from the enhancement of aeration and increasing decomposition rates brought by drainage (Hypothesis 2A). On the other hand, we can anticipate a transient organic carbon stock rise associated with the pulse of inputs from overlaying litter and organic soil layers subject to rapid decomposition and mobilization after drainage (Hypothesis 2B). To explore these hypotheses we characterize soil profiles in paired plots that host similar wetland environments but offer stands that remain under natural conditions covered by native herbaceous vegetation or have been subject to drainage, dike construction, and poplar planting 12–19 years ago (tree plantation). This approach assumes that herbaceous wetlands represent the original conditions of tree plantation stands, allowing us to infer changes through time based on spatial contrast (Richter and Markewitz 2001).

Materials and methods

Study system

The Paraná River Delta hosts a mosaic of wetlands of 17,000 km^2 (Bonfils 1962; Malvárez 1997; Bó 2006), formed by its own sediments and those brought by the Uruguay River in the area where they discharge their waters into the flat bedded and slow flowing Río de la Plata. This study was conducted in one of the dominant wetland ecosystems of the Paraná River Delta, corresponding to herbaceous communities locally known as "pajonales" ("pajonales y bosques del Bajo Delta" according to Malvárez 1997) (Fig. 1). These environments are regularly flooded or have their water table levels very close (<30 cm) to the surface, propitiating anaerobic conditions that slow down biomass decomposition. Under wetland conditions a 30 cm-thick organic layer that includes biomass in different stages of decomposition is found above the mineral soil. In the Río Carabelas area, this vegetation type has been replaced by poplar plantations after the construction of channels and dikes that alter the hydrological regime and favor the establishment and growth of these trees. Under drained and afforested conditions these environments have deeper water table levels. Throughout our study we found water table levels below 2 m of depth during the growing season (September to March) and levels between 1 and 2 m during the rest of the year. In tree plantations a thinner organic horizon is found over the mineral soil, rarely exceeding 3 cm of thickness. This area hosts the largest afforested spot of the Paraná Delta and had originally 80 % of its territory covered by wetlands dominated by *Scirpus giganteus* ("grass covered") and *Schoenoplectus californicus* ("sedge covered") (Kandus 1997).

The region has a temperate humid climate (Malvárez 1997) with a mean annual temperature of 16.5 °C and narrow seasonal variation (mean temperature of the coldest and warmest month being 11.5 and 22.5 °C, respectively) and mean annual precipitation of 1,100 mm distributed evenly throughout the year (1960–2010; INTA Delta agrometeorological station). Soils in the region are young, with most of surface sediments deposited during the Holocene (Bonfils 1962; Wermter et al. 1977; Gómez and Ferrao 1986). The low regional slope and fluvial influence results in poorly drained soils that have most of their profile saturated, displaying signs of chemical reduction such as the presence of a gley horizon and red/yellow mottles above it (Pereyra et al. 2004). The dominant soils are classified as Entisols (Soil Survey Staff 2010) subject to a Fluvic and Aquic environmental regime.

Under natural conditions this area is periodically flooded by the Paraná River in its high water periods

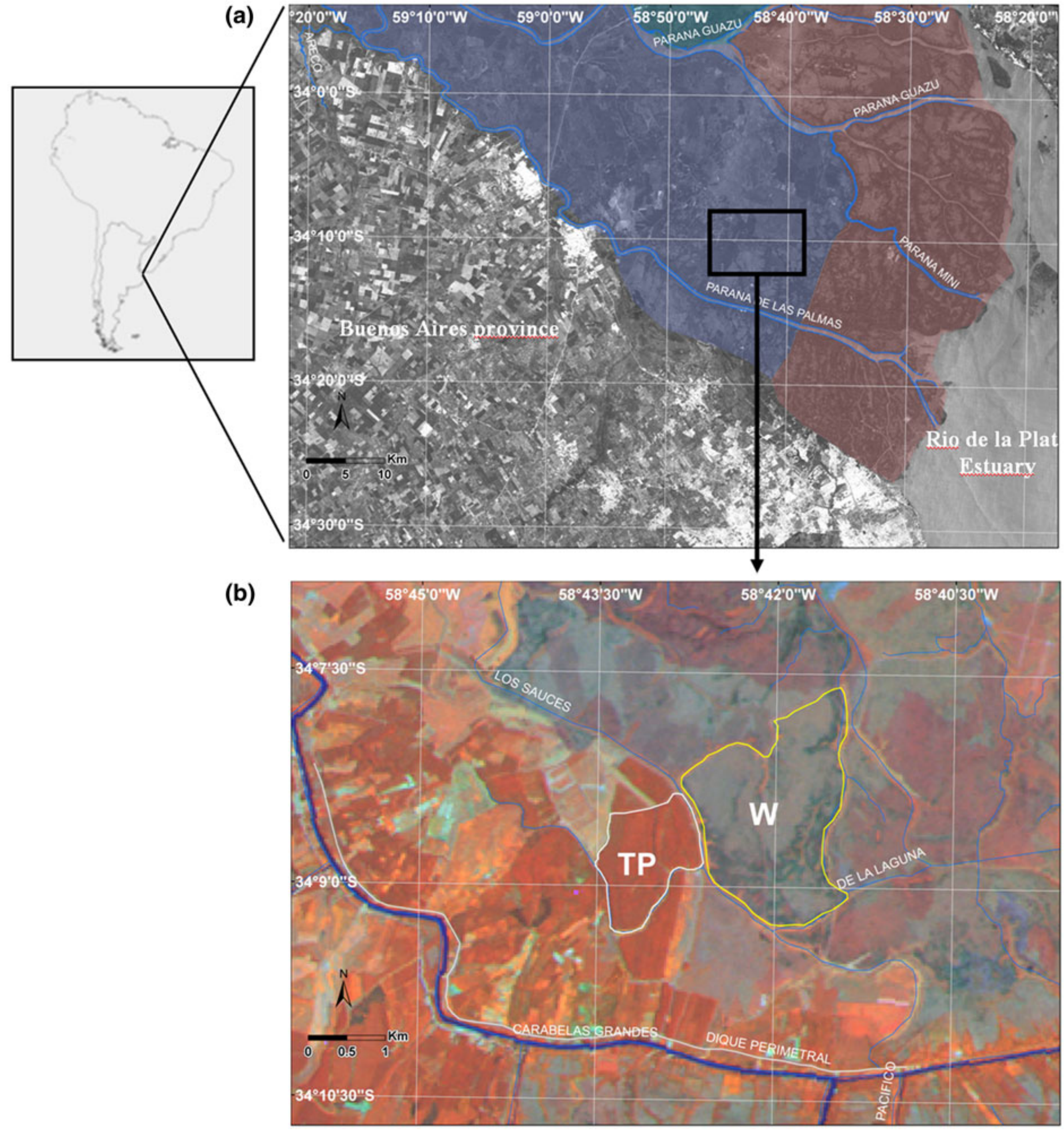


Fig. 1 Geographic location of the Paraná River Delta showing **a** the upstream zone dominated by wetlands and native forests of the lower delta (*blue*) and the downstream zone that corresponds to the advancing sedimentary front (*red*) (Malvárez 1997). A detail of site 1 on a false coloured LANDSAT image **b** shows the pair of stands of natural wetland (W) and tree plantation (TP). (Color figure online)

and, to a lesser extent, by rises in the Río de la Plata level that result from the combined effect of tides and strong winds opposing its flow direction (Kandus 1997). Forestry is the most important land use of the area with 83,370 ha covered by willow and poplar plantations (DPF MINAGRI 2011). While willow plantations are established on wetland soils with little hydrological interventions, poplars, more sensitive to waterlogging than willows, are planted after flood control infrastructure is developed. Flood control is based on the depression of water tables through drainage and channel construction (enhanced water output) and the control of river water inflow through the establishment of dikes along the perimeter of the areas to be planted (reduced water input).

Site description and experimental design

We selected three sites hosting paired stands of natural wetland and poplar plantation that represented the most typical environments of the region. This paired stand approach sought to reconstruct temporary soil changes based on the assumption that natural stands represent the conditions found in their afforested counterpart prior to hydraulic interventions and planting (Richter and Markewitz 2001). Stands of *S. giganteus* with no hydraulic alterations where located <500 m away from poplar plantation stands that where drained and protected by dikes. Soil similarity between stand pairs was evaluated through the description of the sequence and texture of soil horizons, which were analyzed using the Hydrometer method (Bouyoucos 1927, 1962; Elliot et al. 1999; Pansu and Gautheyrou 2006). All afforested stands were planted with *Populus deltoides* of the 'Australia 129/60' clone on *S. giganteus* wetlands and were on their first growth cycle, being never subject to harvesting (Table 1). Both in wetlands and plantation stands, mineral soil sampling points were randomly located within an area of 0.25 ha.

Soil and plant analyses

To avoid potential edge effects, sampling areas were >50 m away from fences or borders. Within each wetland and plantation stand, we randomly obtained seven samples of the organic layer, which included both litter (dead plant structures with different degrees of fragmentation and decomposition) and live or recently dead roots and stems (in wetlands), and just litter (in plantations), using a 1 × 0.5 m frame. This material was sorted and dried at 60 °C during 72 h. In three randomly distributed sampling locations at each wetland and plantation stand we sampled mineral soil at depth intervals of 0–5, 5–10, 10–20, 20–35, 35–50, 50–75, and 75–100 cm. We used a hand auger in wetlands. In the forested stands, we excavated 1 × 1 m wide soil pits, in the center among four trees, from which four walls we obtained subsamples for a composite sample per horizon. Mineral soil samples were air-dried and aggregates were broken to pass a 2-mm sieve. The mineral mass proportion of all soil samples was determined after they were ashed at 700 °C for 8 h in a muffle oven.

We obtained bulk density (BD) values in five soil pits per stand (Elliot et al. 1999) through the cylinder method (Campbell D and Henshall 1991), using 8.2 cm diameter cores in the plantations and a "Mc Cauley" auger of 4 cm diameter in the wetlands. Samples of known volume were weighted after drying at 105 °C during 72 h. We estimated soil porosity based on BD measurements, the relative mass proportion of mineral and organic materials, and their respective particle density values. For the mineral fraction we used a typical particle density value for silicate minerals of 2.65 g cm^{-3}, confirmed for sediments in this region (Pereyra et al. 2004), for the organic fraction we assumed a value of 0.24 g cm^{-3} (Paul et al. 2002).

We characterized the root biomass in all stands; in wetland, where roots were located in the organic layer and did not penetrate mineral soil, we obtained seven 1 × 0.5 m samples of the organic layer. Samples were separated into litter and live and recently dead plant structures, oven-dried at 65 °C, and weighted. In tree plantations, where roots explored the mineral soil, we

Table 1 Location and characteristics of three poplar plantation stands used in this study

Site	Latitude (S)	Longitude (W)	Age (years)	Size of stand (ha)	Density (stems ha^{-1})		DBH (cm)	H (m)	Drainage age (years)
					Initial	Current			
1	34°8′34.76″	58°43′22.97″	12	22	667	407	27.6	24.5	12
2	34°7′49.88″	58°47′12.32″	17	10	833	283	32.1	29.6	>20
3	34°8′9.04″	58°47′43.20″	19	11	833	340	39.0	32.0	>20

The geographic location, age, size, initial and final densities, average diameter at breast height (DBH), average height (H) and history of use are indicated

Table 2 Regression coefficients of the models used to estimate fine roots at each soil depth interval according to the equation y = a + b * x, where x corresponds to the height (m) of individual trees and y to total root dry mass (kg)

Depth (cm)	Coef *a*	Coef *b*	r^2	*P*
0–5	−0.002	0.005	0.72	0.0074
5–25	−0.226	0.129	0.73	0.0065
25–50	−0.039	0.028	0.73	0.0069
50–75	−0.012	0.006	0.72	0.0078
75–100	−0.005	0.003	0.72	0.0081

estimated fine root (<5 mm) distribution at the same depth intervals described above but at four horizontal locations that ranged from 10 cm away from the tree base to the intermediate point between two trees in the same row. Samples were sieved and roots separated, oven-dried at 65 °C, and weighted. We adjusted allometric models that based on tree height allowed us to obtain total root biomass at each depth interval (Table 2). In all samples (litter, roots and soil), total organic C was measured through dry combustion using a CHN autoanalyzer system (LECO CR12) (Gill et al. 2002).

Data analysis

Estimates of soil volumetric changes were based on the assumption that mineral soil mass was conservative (no gains or losses) for the period of time (less than two decades) and for the land use interventions that took place (drainage and afforestation), and were due to changes in the organic matter content and/or porosity (Fig. 2). We adapted the method proposed by Anderson et al. (2002) to estimate soil strain, an indicator of compression/expansion. Soil strain quantification across depths was performed using the following equation:

$$E_i = \frac{BD_W \times MM_W}{BD_{TP} \times MM_{TP}} - 1$$

where the strain (E_i) of a given soil depth i, is a function of the ratio of its BD (Mg m^{-3}) multiplied by its mineral mass proportion (MM) under wetland condition (W) divided by the same values under tree plantation conditions (TP).

In order to account for volumetric changes in C stock comparisons we used cumulative mineral soil mass as our reference (Davidson and Ackerman 1993; Ellert and Bettany 1995; Ellert et al. 2001; Wuest 2009; Toriyama et al. 2011). We adapted the method proposed by Ellert and Bettany (1995) and Ellert et al. (2001) to adjust depth intervals in the tree plantation profiles in order to encompass the same mineral mass stored in the fixed depth sampling intervals of the wetland profiles. We initiated our cumulative mineral mass calculations from the surface of wetland profiles down to each predefined depth (e.g. 0–5 cm in

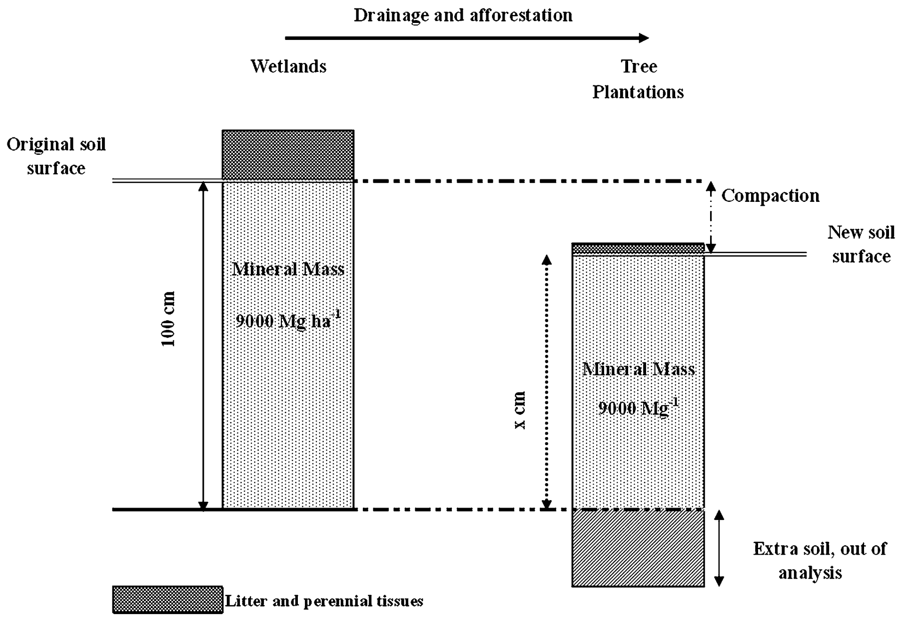


Fig. 2 Scheme of volumetric changes and readjustment of sampling depths for comparison between wetland and tree plantation stands. Mineral mass has been used as a conservative tracer of the initial soil volume. Because all samples were obtained down to 1 m depth, some of them, corresponding to the bottom of the profile of the tree plantation stands, have been discarded or partially considered in the analyses

wetlands, mineral mass = X) seeking to achieve the same accumulated mineral mass in the tree plantation profile through linear interpolation (e.g. 0–5 cm in tree plantation mineral mass = 2X; readjusted layer for C comparisons and balance is 0–2.5 cm with mineral mass = X). The remaining depth of cut intervals (e.g. 2.5–5 cm in tree plantation) was transferred to the following layers and the matching calculation was repeated. Since compaction prevailed at all sites the mineral mass found in the top meter of wetlands was captured in 85–96 cm of depth in the tree plantations; mass sampled below this depth was not considered in our analyses.

True replication in our study was derived from the comparisons of effects across sites, which were evaluated using paired *t*-tests (Zar 1984). Statistical comparisons between stands within each site were also performed with a *t*-test in order to constrain error estimates and using Statgraphics Plus 5.1.

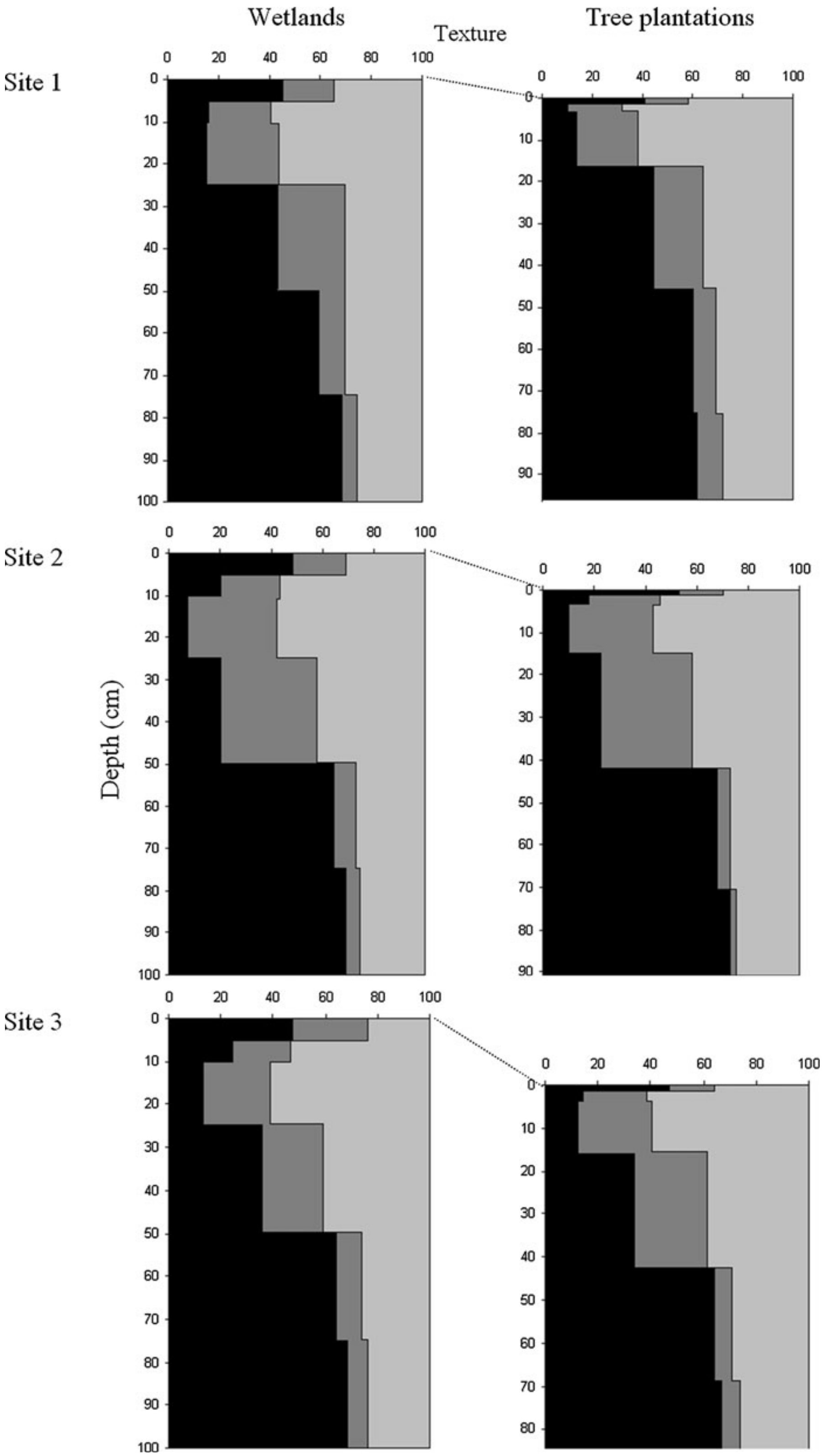
Results

The combination of drainage and afforestation caused a volumetric reduction of soils at all study sites. This compaction was evidenced by the reduction of soil horizon thickness within the similar stratigraphic sequence displayed by tree plantation and wetland profiles (Fig. 3). BD and mineral mass measurements confirmed soil compaction patterns and provided a quantitative estimate of its magnitude at different depths (Fig. 4). While similar soil strata sequences across paired stands supported our use of spatial differences to explore temporal effects (i.e. wetlands soils represented correctly the initial condition of tree plantation soils), the magnitude of volumetric changes justified the adjustment of soil depths in our comparisons of carbon storage based on a conservative soil component, which in our case was total mineral mass.

Bulk density increased with depth under both vegetation types, with largest shifts in the top 25 cm of the profiles. At all study sites drainage and afforestation generated a strain throughout the top meter of mineral soil of −10 ± 3 cm. This whole-profile strain, however, resulted from contrasting strains at different soil depths. While compaction prevailed in the top 10 cm of profiles, volumetric expansion was evident between 25 and 50 cm (Fig. 5). These volumetric shifts can be linked to porosity (physical component) and soil organic matter content (biogeochemical component) shifts. To isolate the contribution of these two components to net volumetric shifts we focused on two depth intervals of the mineral soil column (0–10 and 10–50 cm under original wetland conditions). For the surface layer (0–10 cm in the wetland mineral soil) we estimated a volumetric reduction of 6.2 ± 0.2 cm (this layer was reduced to a 3.8 ± 0.2 cm thickness of mineral soil). This reduction was explained by the physical component, after a loss of 82 % of the original porosity (3.3 ± 0.1 vs. 0.6 ± 0.1 $\text{cm}^3\ \text{g}^{-1}$, for wetland and tree plantation, respectively; $P < 0.01$). The biogeochemical component did not contribute to the compaction of this soil layer since its organic matter stock remained unchanged (34.3 ± 2.0 vs. 37.0 ± 7.0 Mg ha^{-1}, $P = 0.21$). The second layer (10–50 cm in the wetland soil) displayed a volumetric expansion in two of the three study sites (6 ± 3 %, Fig. 5). While the physical component remained unaffected in this layer, with porosity differences being non significant (0.8 ± 0.1 vs. 0.7 ± 0.1 $\text{cm}^3\ \text{g}^{-1}$, $P = 0.3$), the biogeochemical component explained the volumetric expansion, with an increment of 83 % in the organic matter stock (53.3 ± 11.0 vs. 89.4 ± 9.6 Mg ha^{-1}, $P = 0.03$). While the top 10 cm of the wetland soil experienced compaction due to porosity losses, the next 40 cm of the profile expanded as a result of organic matter gains.

Carbon concentration in mineral soils decreased with depth under both vegetation types (Fig. 6), with largest concentrations in the surface soil (0–10 cm) of wetlands. Tree plantations lost the organic layer present in wetlands but gained carbon in the mineral soil, compared to wetlands. The litter compartment that hosted 25.5 ± 1.9 Mg ha^{-1} of C in the wetlands (Table 3) had only 14 % of that amount under tree plantations. This loss was compensated by gains in the mineral soil accounting for 21.6 ± 2.0 Mg ha^{-1}, therefore did not show significant changes in total C stock (litter + mineral soil column), −0.8 ± 0.6 Mg C ha^{-1} was accounted. Drained and afforestated stands, hosted 35.1 ± 8.4 % ($P = 0.1$) more organic carbon than wetlands in mineral soil, gaining 1.3 ± 0.1 Mg C ha^{-1} year^{-1}. The largest increase took place in the 10–50 cm depth range of the original wetland soil (equivalent to 4–43 cm of depth in tree plantations), with significant differences ($P = 0.02$)

Fig. 3 Soil texture variation across sites, stands and depths. Depths in tree plantation stands were adjusted to account for soil volumetric changes. Values correspond to the percentage of soil mass corresponding to sand (*black*), silt (*dark grey*) and clay (*light gray*)



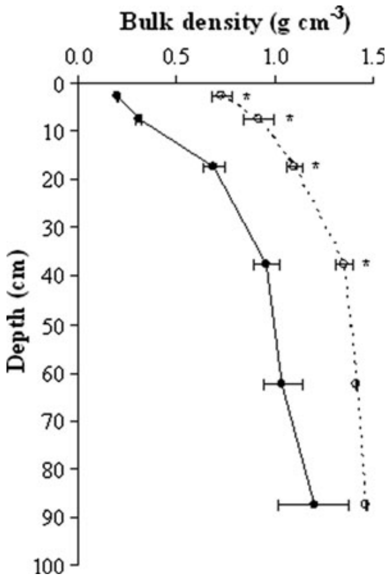


Fig. 4 Bulk density at different depths of soil for wetlands (*solid line*) and tree plantations (*dotted line*) (mean ± SE; based on site-to-site variability, N = 3). *Indicates significant differences between pairs ($P < 0.05$). Note that no volumetric adjustment has been applied in this figure and depth intervals in the tree plantation do not correspond to the same original layers in the wetland

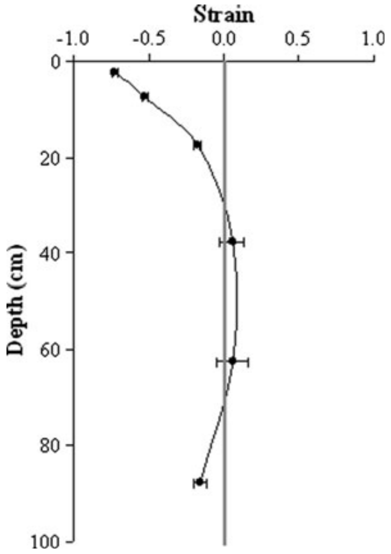


Fig. 5 Soil strain of wetland soils after drainage and afforestation. Values range from −1 to 1, respectively showing volumetric reduction or expansion (mean + SE; based on site-to-site variability, N = 3) for depth intervals as found in the original wetland soil are shown

between 10 and 25 cm, where C storage increased 154 % (Fig. 7).

Vertical root distributions were very contrasting between wetlands and tree plantations mirroring soil organic carbon distribution patterns. While marshes roots in the wetland were located over the mineral soil, within an organic layer of 29 ± 2 cm of thickness; tree roots in the drained plantations were located in the mineral soil, with 75 % of the their total biomass found between 10 and 50 cm (Fig. 8). Noticeably, this depth range showed the highest C gains as well. At all study sites the observed profiles suggested that drainage and/or vegetation replacement favored the proliferation of roots into mineral soil layers that remained untapped under natural wetland conditions.

Discussion

In Spite of we cannot eradicate all uncertainties regarding our assumption of equal conditions at both plots (wetland and tree plantation) in pre-afforestation times we have some evidence supporting it which includes (a) sites located in the same landscape positions as corroborated by satellite images and aerial photography and interviews with local land managers, (b) similar soil texture supporting no soil type bias. Our selection of stands was carefully guided by (a) and our first soil analysis confirmed (b). The fact that the three pairs of sites (real replicates) displayed a similar behavior encourages as to sustain our conclusions.

Based on the results, we infer that the surface mineral soil at our study sites subsided at a rate of 0.6 cm per year as a result of porosity losses in the top 10 cm of the profile. This rate is lower than those described for more organic wetlands such as peatlands after they are subject to drainage and cultivation, where organic matter rather than porosity losses are the main cause of volumetric changes (Wösten et al. 1997, 2006; Grønlund et al. 2008). This finding confirmed that wetland soil compaction is an important and intense process in our study system, suggesting that profile de-saturation (Hypothesis 1A) has been a key driver (Fredlund and Rahardjo 1993; Aust et al. 1995; Engelaar et al. 2000; Schanz 2005; Hadas 2006). In other mineral wetland soils subject to drainage, Rovdan et al. (2002) have found volumetric losses of 7–31 % in the top 10 cm of the profile resulting from

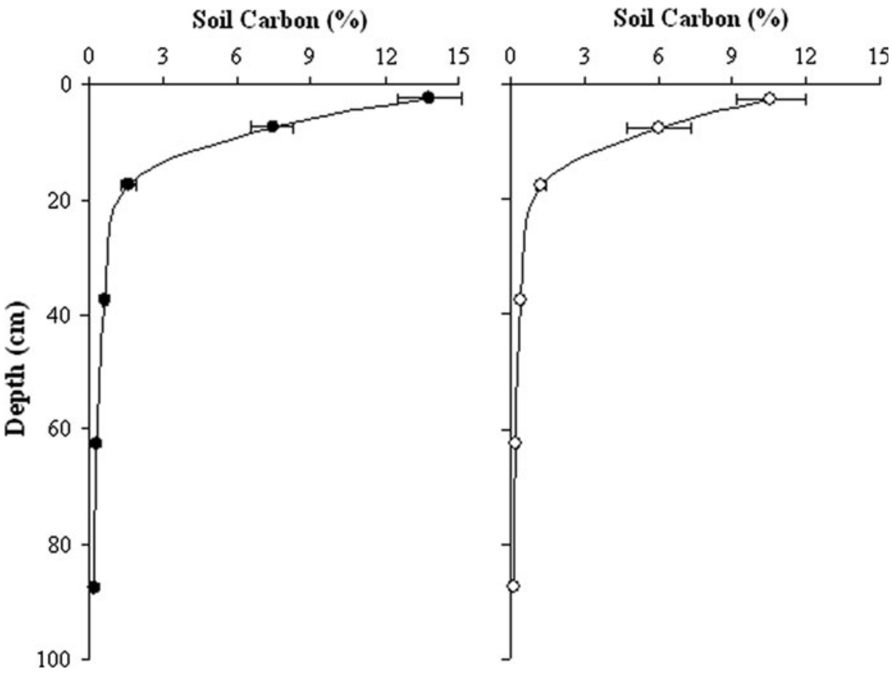


Fig. 6 Soil carbon concentration at different depths in wetland (*left*) and tree plantation (*right*) stands (mean ± SE; based on site-to-site variability, N = 3). Note that no volumetric adjustment has been applied in this figure and depth intervals in the tree plantation do not correspond to the same original layers in the wetland

Table 3 Soil carbon storage (wetlands and tree plantations) and balance (conversion of wetlands to tree plantations) for the litter and the first meter of mineral soil (SE; based on site-to-site variability, N = 3)

Site	Vegetation type	Equivalent depth (cm)		Carbon (Mg ha^{-1})					
				Content		Total balance		Mineral balance	
		Mean	SE	Mean	SE	Mean	SE	Mean	SE
1	Wetlands (litter)			19.3	2.2				
	Wetlands (mineral)	100		64.5	3.7				
	Tree plantation (litter)			2.3	0.5				
	Tree plantation (mineral)	95.7	0.6	79.1	3.8	−2.4	6.8	14.6	7.5
2	Wetlands (litter)			31.1	2.4				
	Wetlands (mineral)	100		50.9	2.9				
	Tree plantation (litter)			4.4	0.4				
	Tree plantation (mineral)	90.8	2.6	76.8	2.7	−0.9	3.4	25.8	1.4
3	Wetlands (litter)			26.1	3.1				
	Wetlands (mineral)	100		75.9	2.3				
	Tree plantation (litter)			4.0	0.7				
	Tree plantation (mineral)	84.5	0.3	100.1	6.0	2.1	6.2	24.2	3.9

In tree plantations depth has been adjusted based on cumulative mineral mass to account for volumetric change

porosity reduction, values that are low compared with figures of more than 80 % described here. The combination of wetland soils that are initially very porous with the load of heavy trees individuals to which they are subject may explain the intense compaction that we described here. Compaction may affect vegetation growth in multiple ways including interference in water, nutrient and oxygen availability (Hamza and Anderson 2005), whether these problems affect the tree plantations that we studied still needs to be known.

In contrast to what we found in surface soil, at intermediate depths (10–50 cm), our study sites showed a little volumetric expansion that could have

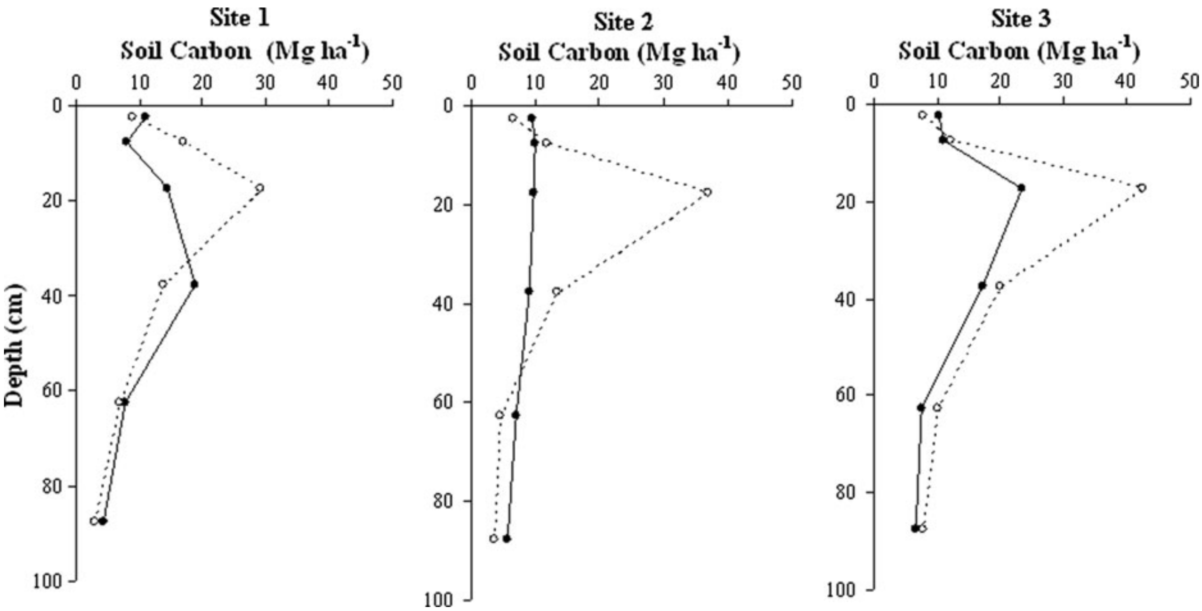


Fig. 7 Soil carbon storage in wetland (*filled line*) and tree plantation (*dotted line*) stands. Values indicate the contribution of each soil depth interval to storage per unit of area. In tree plantations soil carbon values have been recalculated in order to compensate for volumetric changes in the profiles

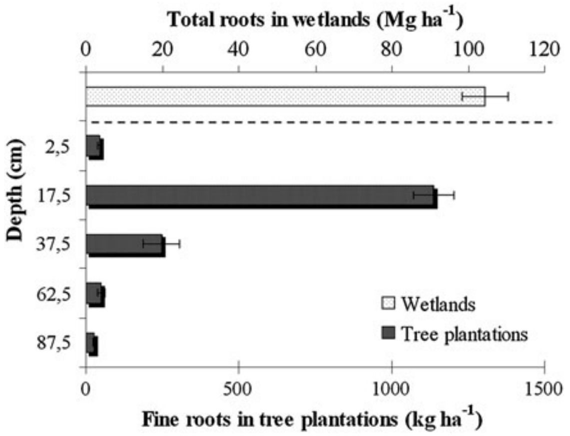


Fig. 8 Vertical distribution of perennial tissues biomass in wetlands and fine roots (<5 mm) tree plantation (mean ± SE; based on site-to-site variability, N = 3). The *dotted line* indicates the surface of the mineral soil. Wetlands have no roots in the mineral soil

been caused by the penetration of roots and the enhancement of biological activity in a soil region that was free of roots under natural conditions and most likely subject to highly anoxic conditions (Hypothesis 1B) (Cerisola et al. 2005). In European wetlands, deeper and denser root systems followed drainage (Wills et al. 1999; Braekke 2005). Roots could have caused a deep volumetric expansion through the physical generation of macropores, the biological stabilization of soil structure, and the addition of soil organic matter (Neary et al. 2009; Rong et al. 2010; Cerisola et al. 2005; Lal 2007; Deurer et al. 2009; Gill et al. 2009; Ruehlmann and Körschens 2009). Complementing root effects, soil drainage and flood control in afforested stands may have propitiated desiccation and wetting cycles that could favor soil cracking and expansion in the most clay-rich subsurface horizons (Pires et al. 2009; Pillai and McGarry 1999). Our observations suggest that compaction and expansion mechanisms following drainage coexisted (Hypotheses 1A and 1B) but took place at different soil depths.

Twelve to nineteen years after drainage and tree planting, study sites gained organic carbon in the mineral soil, particularly in subsurface horizons. Simultaneously, surface litter in tree plantations hosted only 14 % of the mass found in the same comportment in wetlands. This finding support our initial expectation of C losses through enhanced decomposition and respiration (Hypothesis 2A) for the surface organic layer but not for the mineral zone. While C gains prevailed in the mineral soil, they cannot be completely attributed to the redistribution from the organic layer (Hypothesis 2B), given their vertical distribution. Instead, the observed match between tree fine root distributions and C gains points to root colonization as the most likely mechanism. Although we found no reports of soil C shifts following afforestation of herbaceous wetlands, many studies report the effects of poplar plantations on other ecosystem types, indicating losses (Hansen 1993; Ross et al. 1999), gains (Hansen 1993; Sartori et al. 2007; Arevalo et al. 2009; Rong et al. 2010) or lack of changes (Grigal and Berguson 1998; Coleman et al. 2004; Dowell et al. 2009). Arevalo et al. (2009) showed that, after 9 years of grassland replacement by poplar plantations in the Canadian prairie, soils gained carbon at rates of 1.7 Mg ha^{-1} year^{-1}, similar to what is reported here, mostly as a result of root contributions. A similar rate of C gains and role of roots has been suggested for poplar plantations in Wisconsin (Hansen 1993), and linked to the high root inputs of these trees and their enhancement of soil biological activity that make them more effective raising soil C than other species (Lodhyal et al. 1995a, b; Paul et al. 2002; Coleman et al. 2004; Vesterdal et al. 2002; Laganiere et al. 2010; Rong et al. 2010).

After drainage and afforestation, the dominant wetland ecosystems of the Paraná River Delta experienced a strong compaction of their surface mineral soil but an expansion at greater depth and neutral C balance when considering litter and mineral soil. These changes were accompanied by organic matter losses above the mineral soil surface but gains below it, suggesting that these young sedimentary wetland soils may have less negative carbon balances than those hosting more developed organic soils (e.g. peatlands). While these changes are consistent across sites, they emerge after just one decade of ecosystem interventions, soils will likely sustain their trajectory of change for longer periods, making their monitoring in the long-term critical to understand the net effects of drainage on C balance and ecosystem productivity.

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