

## Latrubesse Special Issue

# Carbon budget alteration due to landcover–landuse change in wetlands: the case of afforestation in the Lower Delta of the Paraná River marshes (Argentina)

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### Keywords

carbon budget; climate change; drainage; emission; forestry; wetlands.

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### Abstract

The objective of this work was to analyse the effects on carbon budget caused by the replacement of freshwater marshes by salicaceae plantations at ecosystem and regional scales, in the Lower Delta of the Paraná River in Argentina. Biomass and net aboveground primary productivity were estimated for one of the main species planted in the delta, *Salix babylonica* var. *sacramenta*, and for the dominant species of the marshes that are most widespread in the study area, *Scirpus giganteus* and *Schoenoplectus californicus*. Soil organic carbon was also estimated. Afforestation and marshes were mapped based on the digital classification of Landsat imagery. Aboveground biomass stored by afforestation is higher than by marshes (113.4 Mg/ha vs. around 10 Mg/ha). Nevertheless, soil organic matter accumulation is 10-fold higher in the marshes. A huge amount of carbon is released into the atmosphere when marshes are converted to forest due to change from a wetland to a terrestrial condition.

## Introduction

Forest carbon sinks were included in the Kyoto Protocol as one of the mechanisms for mitigating climate change, because these sinks are known to play an important role in the global green house gas (GHG) balance. Globally, annual carbon sequestration by terrestrial ecosystems was estimated to be 2.3 GtC in the 1990s, while emissions from landuse change were 1.6 GtC/year (IPCC 2000). Wetlands contain a disproportionately large share of the earth's carbon stocks (Schlesinger 1984). Although they occupy only 3–5% of the earth's total land area, they contain nearly 26% of the total C stocks in plant matter and soils (Mitra *et al.* 2005). The reason for the high levels of carbon in many wetlands is their relatively high net primary productivity (NPP) in terms of fixing carbon from the atmosphere coupled with a relatively low rate of decomposition of accumulated organic matter. The latter is due to anaerobic soil conditions in wetlands, which can increase organic matter half-lives by nearly an order of magnitude over aerobic decomposition in upland ecosystems (Reddy *et al.* 1986).

On the other hand, anaerobic conditions favour the formation of methane, which might offset some of the GHG sequestration benefits associated with wetlands. Wetlands are estimated to contribute from 20 to 40% of the annual global atmospheric CH<sub>4</sub> flux (Bartlett & Harriss 1993). However, there are important differences among wetland types. Reviews consistently report the highest rates of CH<sub>4</sub> emission from tropical wetlands (51% of the total wetland flux) and the lowest emissions from temperate wetlands (10%) (Aselmann & Crutzen 1989). Nevertheless, in temperate regions, re- and afforestation rates tend to be high (IPCC 1996), and it is important to note that most plantations need strong hydrologic modifications of the wetlands to be successful. Ditching, draining and levee building are specifically designed to dry wetlands out and to control floods. As a result, wetlands are modified in a number of ecological ways by changing normal hydrologic patterns. Straight deep channels and ditches are the most common methods to achieve wetlands' drying, and they allow more rapid runoff of freshwater than the normal shallow and sinuous channels do; water levels fluctuate more rapidly and the water table

declines, thus drying the marshes. In other cases, artificial levees may block overbank sheet flow, reducing the sediment supply to the marsh (Mitch & Gosselink 2000).

Given these alterations, and their potential for changing carbon dynamics, there is a need to better understand the net sequestration potential of the wide variety of wetlands, and how they might be affected by management practices and changes in landcover and hydrological conditions. The aim of this study was to analyse the changes in carbon budget due to temperate tidal freshwater marsh substitution by terrestrial *Salix* spp. (willow) afforestation.

Aboveground biomass (AB), annual net aboveground primary production (NAPP) and soil organic matter for the dominant species of the two most important marsh communities (*Scirpus giganteus* and *Schoenoplectus californicus*) and for a common willow plantations (*Salix babylonica* var. *sacramenta*) were estimated for the Buenos Aires section of the Lower Delta of the Paraná River. A major hypothesis was that primary carbon allocation differs significantly from that revealed by standing biomass between marshes and terrestrial forest. The consequences were analysed at ecosystem and regional scales.

## Main features of the study area

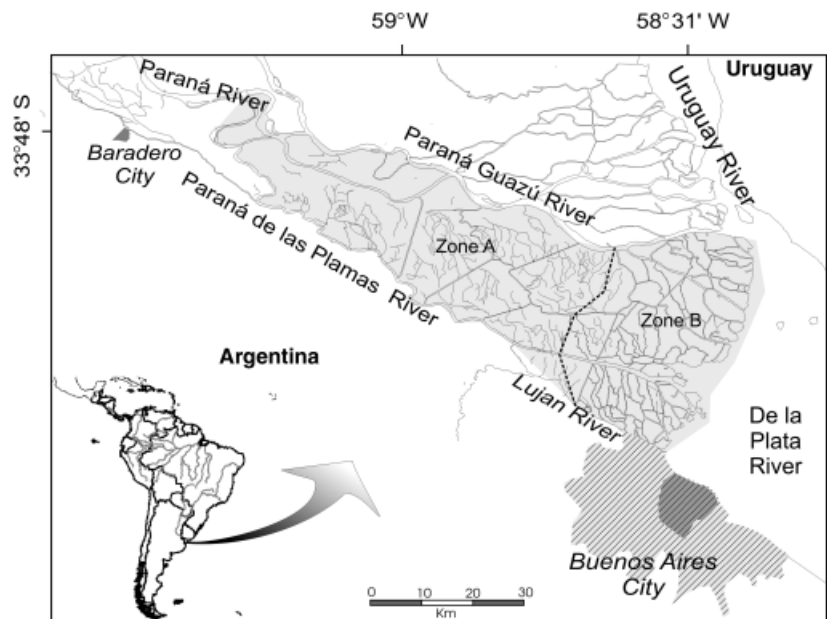
The region has temperate–subtropical climate conditions, with a mean annual temperature around 16.7 °C and an average annual rainfall close to 1073 mm, without a particular rainy season.

The islands of the Buenos Aires section of the Lower Delta of the Paraná River cover 2700 km<sup>2</sup>, and are located at the end of this river, just in the upper portion of the De

La Plata estuary (Fig. 1). They belong to a prograding delta with an estimated rate of growth calculated at 70 m/year over a recent 160-year interval (Iriondo & Scotta 1979). Most of the area is formed by mature islands with a saucer shape, with levees surrounding the perimeter and a depressed central portion temporarily or permanently flooded. According to Bonfils (1962), levees and central depressions occupy an average of 20 and 80% of an island's area, respectively, and their elevation is around 3–0.5 m above the sea level. Natural forests develop on levees and relatively high elevation sites. In the lower portion of the topographic gradient, marshes dominated by *S. californicus* or *S. giganteus* are found instead. These marshes grow, forming almost monoespecific stands under a tidal freshwater environment. As with many emergent macrophytes commonly found in saturated soils, *S. giganteus* and *S. californicus* are perennial rhizomatous species that grow through the repeated production of ramets and emergent structures develop at nodes located in rhizomes below the ground surface.

Located very close to the Buenos Aires City, which has over 11.5 million inhabitants, the delta islands show a long history of use, which is nowadays mainly represented by salicaceae (*Salix* and *Populus* species) afforestation. In fact, landscape patterns are defined by a complex mosaic of natural and man-made landcover types (Kandus *et al.* 1999). Salicaceae plantations replace the natural vegetation of almost all levees and they extend towards lowlands of the central portion of the islands.

According to Kandus *et al.* (2003), two main ecological units are identified in the region: Zones A and B (Fig. 1). In the upstream portion of the delta (Zone A),



**Fig. 1.** The Buenos Aires portion of the lower Delta of the Paraná River (grey area) and their location in South America. Zones A and B correspond to the ecological units defined by Kandus *et al.* (2003).

*S. californicus* dominates marsh communities (with *Azolla filiculoides*, *Pontederia lanceolata* and *Echinodorus grandiflorus* as co-occurring species) and plantations are protected from flooding by man-made dikes up to 3 m in height. The dikes are built principally to prevent flooding from the seasonal discharge of the Paraná River, which has peak discharge in March, rather than lesser flood stages from tides originating in the Rio de la Plata estuary. Flooding from the Paraná is triggered by the El Niño phenomenon, which can exert a significant effect on hydroperiod, tree mortality and sediment accumulation on the islands (Karszenbaum et al. 1999). In contrast, in Zone B, the inner portion of the islands is occupied by *S. giganteus* marshes with the following co-occurring species: *S. californicus* under prolonged waterlogging conditions and *Baccharis penningtonii*, *Eupatorium tremulum* and *Aeschynomene montevidensis* under intermittent waterlogging conditions.

This zone is affected mainly by lunar tides and wind regimes of the Rio de la Plata estuary. The semidiurnal tidal amplitude is normally about 1 m. Floods that primarily affect this portion of the region are due to south-southeastern winds that can increase water levels up to 2.5 m over the average, producing floods that may range between several hours and a few days (Iriondo & Scotta 1979). Afforestation activity in Zone B is facilitated more by the constructions of ditches and channels (rather than dikes) to allow the more rapid runoff of freshwater.

## Methodology

To accomplish the proposed objective, biomass, NAPP and soil organic matter were measured in representative stands of marsh and plantation. These cover types were mapped using a digital classification over a set of Landsat Thematic Mapper images and the area of each landcover type was calculated.

## Biomass and NAPP measurements

### Marshes

The AB has been studied through an annual cycle, and the NAPP in the two most important marshes of the study area dominated by *S. giganteus* and *S. californicus* was

estimated, respectively (Kandus 1997; Kandus et al. 2003). The estimate of NAPP has already been presented and described in Pratolongo (2005) and Pratolongo et al. (2005, 2007, 2008). Ten plots (50 × 50 cm) were established in two stands belonging to the different kind of marshes from September 2002 to September 2003. To minimize the disturbance caused to the vegetation and soil surface by walking on the stand, sampling began along a 100 m line, with plots randomly located 2 m from the line. On each subsequent sampling date, plots were established 2 m from the previous ones, towards an undisturbed area. All live ramets were tagged in each plot by piercing the entire mat at the ground level with a steel needle and a polyester thread. Nodes remain below the surface, and so as leaves grow, the pierced points move up above the substrate. On subsequent sampling dates, all ramets present in previously marked plots were cut at the ground level, 10 new plots were established and new ramets were tagged. In the laboratory, the collected material was rinsed, and each leaf was cut at the level of the thread. The portion below the thread as well as new ramets that appeared inside the quadrat were classified as leaf tissue grown during the sampling interval, and the portion above the thread as remaining leaf tissues.

### *Salix* afforestation

AB and NAPP were estimated in *S. babylonica* var. *sacramento* plantations based on allometric relationships calculated from 20 individuals representative of different diameters for this study area (Whittaker & Woodwell 1968; Whittaker et al. 1974; Williams et al. 2003). Table 1 presents the parameters estimated by least squares to fit the model  $\ln(Y) = a + b \ln(X)$  for individuals of *S. babylonica* var. *sacramento* in the Lower Delta of the Paraná River plantation according to Burghi (2002). Afterwards, 20 plantations were selected, of different basal areas and ages, and 50 trees were randomly chosen for diameter breast height (DBH) measurements. DBH values were sorted into 2 cm classes. In each one of the 20 plantations, AB (Mg/ha) was calculated applying the coefficients of the DBH-biomass regressions for each diameter class. Finally, the total AB was calculated by multiplying each class biomass by the

**Table 1** Parameters estimated by least squares to fit the model  $\ln(Y) = a + b \ln(X)$  for individuals of *Salix babylonica* var. *sacramento* in the Lower Delta of the Paraná River plantation

Dependent variable (Y)	Independent variable (X)	a coefficient	b coefficient	R <sup>2</sup>	F	P-level
Total aboveground biomass	DBH	− 0.598	2.089	0.957	178.374	<0.001
NAPP	DBH	− 0.583	1.218	0.739	22.707	<0.001

Source: Burghi (2002).

DBH, diameter breast height; NAPP, net aboveground primary production.

density of trees in the diameter classes and then we added the biomass for all the classes.

### Soil organic matter

Soil organic matter content was estimated in marshes in the Paraná River Delta by Pratolongo *et al.* (2007). Field sampling was conducted during spring–summer 2002 and five samples were collected in each marsh type, using a McCauley peat sampler. Each sample comprised of a cylindrical piece of 25 cm length and 4 cm diameter, collected to a mean depth of 12.5, 37.5, 62.5 and 87.5 cm. Soil cores were transported to the laboratory in air-tight plastic bags and refrigerated until sample processing. Samples were oven dried (80 °C) and weighed to determine the dry mass of ground and bulk density was calculated as a relation between dry mass and sample volume (dry g/cm<sup>3</sup>). Organic matter content was determined using the loss-on-ignition method (LOI%) and estimated as the relation between the weight lost after heating (500 °C, during 3 h) and dried weight. For data analysis, total organic matter only up to 75 cm depth was considered. LOI% was converted to weight (Mg/ha) by transforming bulk density into real density by subtracting porous space.

### Landcover–landuse map

A landcover–landuse map was constructed based on a satellite image classification according to the procedure already described in Kandus *et al.* (1999). Three Landsat Thematic Mapper images were acquired from the Instituto Nacional de Pesquisas Espaciais (INPE), Brazil. The acquisition dates of the images were 3 August 1993 (winter), 6 October 1996 (spring) and January 1994 (summer). The spring image was rectified to a Gauss Kruger coordinate system with a pixel size of 28.5 m. The other images were registered with an RMS error <0.25 pixel for the fit between images. Row data were converting into reflectance values at the top of the atmosphere (Kandus 1997).

The landcover classification strategy was based on the use of the multitemporal layer classification procedure. Because of the complexity and fragmentation of the landscape, an iterative clustering technique was used to further improve separation between classes. The final landcover–landuse map was the result of combining a set of intermediate layers that were generated after each individual image was classified, labelled and recoded. For the classification procedure, the unsupervised isodata algorithm available in the software package ERDAS 8.2 was used (Jensen 2000). Although 15 landcover–landuse classes were originally mapped, only *S. giganteus* marsh, *S. californicus* marsh and salicaceae plantations landcover types were particularly highlighted in the map presented

in this work. The remaining landcover types were labelled here as ‘other landcover types’.

### Regional projection

AB, soil organic matter and NAPP at regional scales were estimated assuming a simplistic general model that considers all *Salix* spp. plantations as being *S. babylonica* var. *Sacramento*. It is also assumed that willow afforestation in Zone A replaced only *S. californicus*, although in this zone, there are also *S. giganteus* marshes. In Zone B, only *S. giganteus* marshes were replaced. Finally, it was assumed that the age distributions of the willow plantations were homogeneous with a harvest rotation of 14 years.

### Results and discussion

AB and NAPP results for *S. babylonica* afforestation and *S. giganteus* and *S. californicus* marshes are shown in Table 2. The values on this table indicate that *S. babylonica* afforestation has a 10-fold higher AB than *S. giganteus* and *S. californicus* marshes. In contrast, NAPP for all the landcover types falls within a similar range, resulting in a much higher production to biomass ratio (NAPP/AB), which would be an estimator of productivity efficiency. In addition to the AB annual turnover, approximately 65% of the *S. giganteus* litterfall is decomposed after a year and likely is incorporated into the soil profile (Pratolongo 2005).

Table 3 shows soil organic matter content between 0 and 75 cm depth and total organic matter including only

**Table 2** Mean aboveground biomass (AB) (Mg/ha), net aboveground primary production (NAPP) (Mg/ha/year)  $\pm$  1 standard deviation (SD) and production to biomass ratio (NAPP/AB) for *Salix babylonica* plantation and *Scirpus giganteus* and *Schoenoplectus californicus* marshes in the Lower Delta of the Paraná River

Landcover–landuse	AB (mean $\pm$ SD)	NAPP (mean $\pm$ 1 SD)	NAPP/AB
<i>S. babylonica</i> plantation <sup>a</sup>	113.4	15.1	0.13
<i>S. giganteus</i> marsh	10.3 $\pm$ 2.4	14.9 $\pm$ 3.8	1.5
<i>S. californicus</i> marsh	8.8 $\pm$ 0.6	16.5 $\pm$ 3.5	1.9

<sup>a</sup>As estimated from a regression, SD does not correspond here.

**Table 3** Soil organic matter (0–75 cm depth) and total organic matter (OM) (including only aboveground and soil organic matter) for *Salix babylonica* afforestation and *Scirpus giganteus* and *Schoenoplectus californicus* marshes in the Lower Delta of the Paraná River

Landcover type	Soil OM (Mg/ha) Mean $\pm$ SD	Total OM (Mg/ha) mean $\pm$ SD
<i>S. babylonica</i> plantation	125 $\pm$ 50	239 $\pm$ 26
<i>S. giganteus</i> marsh	1513.44 $\pm$ 488	1523.7 $\pm$ 490
<i>S. californicus</i> marsh	225.85 $\pm$ 294	234.6 $\pm$ 295

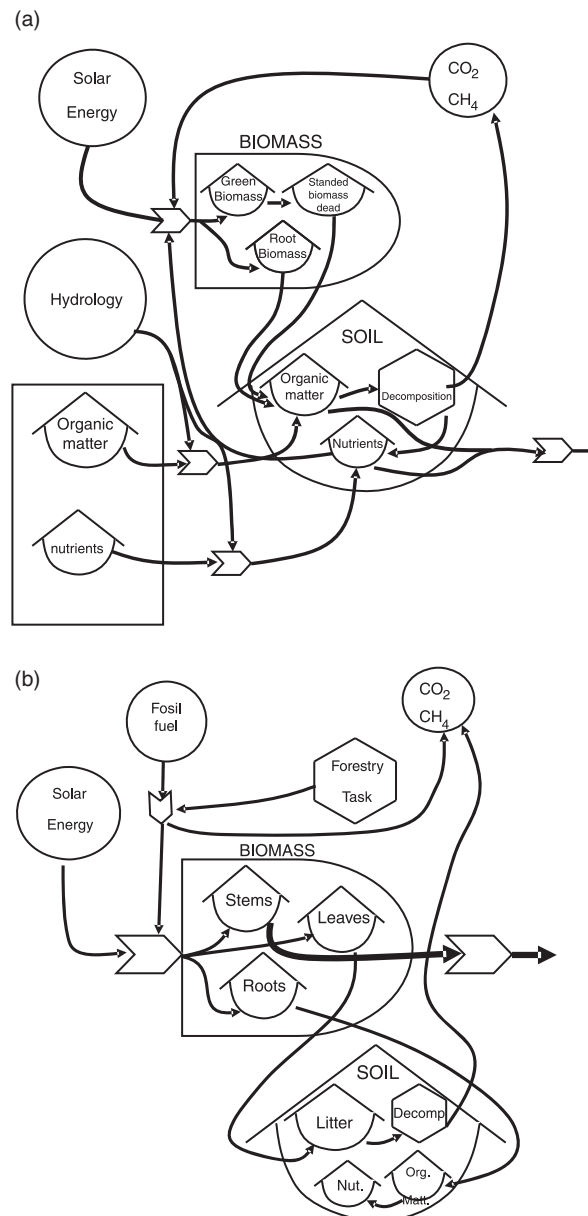
aboveground and soil organic matter, for *S. babylonica* afforestation and *S. giganteus* and *S. californicus* marshes. In this way, it is noticeable that the amount of soil organic matter stored in the first 75 cm depth in afforestation was half of that of *S. californicus* marsh soil and it was around 10-fold less than the value of *S. giganteus* marsh soil. These values are coincident with the results of Schlesinger (1984), who found that the average organic carbon of a soil profile of 1 m depth of wetland areas is around 720 Mg/ha. As a consequence, from an ecosystem point of view, organic matter accumulation in marsh soils, plus aboveground organic matter in plant tissues, is enormous compared with that in the afforestation environments.

It is important to note that in order to gain a complete view of the net ecosystem exchange (NEE), it will be necessary to estimate belowground biomass in both marshes and afforestation of the delta region in a forthcoming work. According to the literature, there are considerable practical problems in sampling underground biomass (Kvet & Westlake 1998). As a consequence, conversions of aboveground standing crop into total biomass, aboveground production and underground production are often made by assuming standard ratios such as the root/shoot ratio ( $B_u/B_a$ ) (Whigham 1978). In this way, Kvet & Husák (1978) found a  $B_u/B_a$  of 2.3–3.9 for *Schoenoplectus lacustris* and 0.4–0.6 for *Typha latifolia* in different sites. For *Cyperus papyrus*, Thompson *et al.* (1979) estimated a relation of 0.45, and in temperate marshes, Fiala (1976) reported 1.8–9.9 for *Phragmites australis* marsh.

In the case of trees, some species such as *Acer rubrum* develop very shallow root systems in response to flooding. In aerated soils, the same species will develop deep roots (Mitch & Gosselink 2000). Rytter (2001) estimated above and belowground biomass of *Salix viminalis* under experimental conditions. According to her results, the  $B_u/B_a$  ratio in this species was around 0.1 during the 3 years after planting.

Based on the data of Table 2 and choosing the most conservative  $B_u/B_a$  ratio values in marshes (0.4 for *S. lacustris*), the estimated underground biomass is around 4 Mg/ha for *S. giganteus* and 3.5 Mg/ha for *S. californicus*, being 11 Mg/ha for *Salix* using the rate derived from the results of Rytter (2001). Even considering 43 Mg/ha as an average value for trees belowground biomass in freshwater wetlands (Brinson 1990), these values, along with our results, suggest that the inclusion of root biomass would not change significantly the trends for total organic matter shown in Table 3.

The energy flow models in Fig. 2 show some of the main changes occurring at the ecosystem level when marshes are replaced by afforestation. First of all, the alteration of the hydrologic regime leads to changes in



**Fig. 2.** Energy flow model describing biomass accumulation (bird-house symbols), productivity processes, respiration and cycling; input and output of nutrients and organic matter; and the factors (both internal and external) believed to control rates. (a) Model for *Scirpus giganteus* and *Schoenoplectus californicus* marshes. (b) *Salix babylonica* var. *sacramento* afforestation model. Symbols are from Odum (1983).

the soil aeration condition, moving soils from a carbon sink to a carbon source (Lugo *et al.* 1990). Apart from in situ NAPP, significant quantities of carbon from both wetland and nonwetland sources may also be trapped and stored in wetland sediments. Thus, soil carbon reservoirs may supply large amounts of carbon to the atmosphere if water levels are reduced or land management practices result in the oxidation of soils (Kusler 1999).

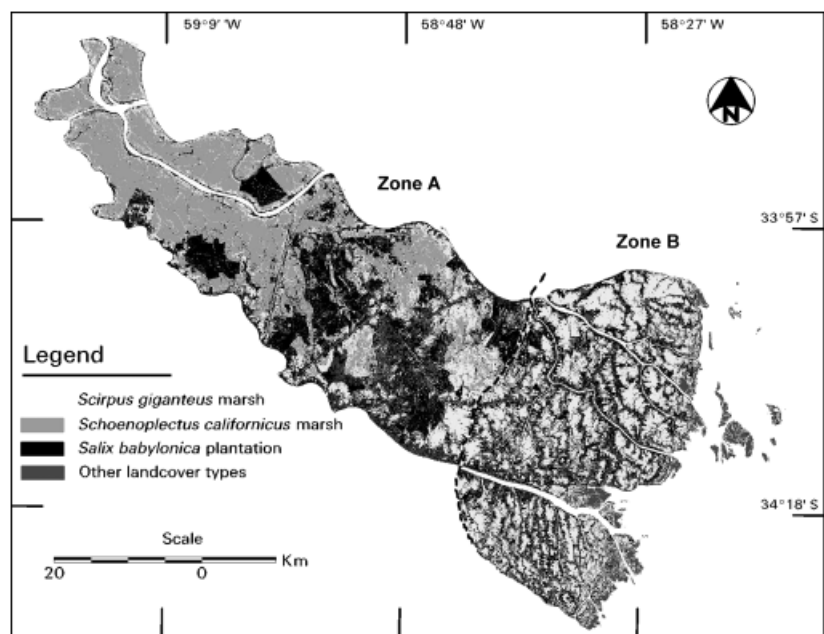
Freshwater wetlands are a natural source of  $\text{CH}_4$ , estimated to release 100–200 Tg  $\text{CH}_4$ /year due to anaerobic decomposition of organic material in the wetland soils (Cicerone & Oremland 1988). Destruction of freshwater wetlands, through drainage or filling, would result in a reduction of  $\text{CH}_4$  emissions and an increase in  $\text{CO}_2$  emissions due to increased oxidation of soil organic material (Moore & Knowles 1989). The magnitude of these effects is largely a function of soil temperature and the extent of drainage (i.e. the water content of the soil). Also, because dryland soils are a sink of  $\text{CH}_4$ , drainage and drying of a wetland could eventually result in the wetland area changing from a source to a sink of  $\text{CH}_4$  (IPCC/UNEP/OECD/IEA 1997). Nevertheless, in the NEE balance, it should be taken into account that the afforestation involves an additional source of energy in fossil fuel form that contributes to an additional emission of carbon to the atmosphere.

On the other hand, changes are very different considering plantations inside polders (dikes) or drained by ditches and channels. In the first case, the practice leads to the isolation from the surrounding environments and prevents overbank flooding, thus reducing or impeding the inflow of water, nutrient and sediments as well as the outflow to the estuary. This is the most common practice in Zone A (Fig. 1). In the second one, ditches and channels are designed in Zone B to carry floodwaters off as rapidly as possible, increasing the export rate of nutrient and particulate organic matter to the main streams and, finally, into the Río de la Plata estuary.

The landcover–landuse map shows the area occupied by plantations and both types of marshes (Fig. 3). As mentioned before, marshes of the inner portion of the islands of Zone A are mostly represented by *S. californicus* marshes and those of Zone B by *S. giganteus* marshes. In relation to *Salix* spp. afforestation in Zone A, plantations cover large portions of the islands, while in Zone B, plots are smaller and are distributed always close to the streams.

The area of ‘other landcover types’ class is also illustrated to represent areas not considered in the analysis and consists mostly of levee landcover–landuse types and includes *Populus* spp. (poplar) plantations. Poplar is planted on the levees in Zone B, but in Zone A, they are restricted to areas inside the dikes at lowest elevations. Although this kind of plantation is of increasing importance, it was not analysed in this work. Abandoned afforestation in lowlands and *Erythrina crista-galli* forest were included in *S. giganteus* marsh class due to their understorey being completely dominated by *S. giganteus*. Table 4 presents the area occupied by marshes and afforestation by 1994. In this table, the summation of the three landcover types analysed corresponds to the 80% of the surface cited by Bonfils (1962) as lowlands. It is noted that both zones were similarly affected by willow (*Salix* spp.) afforestation, although the ecological consequences in each one might be different considering the different practices that were discussed previously.

Table 5 illustrates the trend at the regional scale in AB, soil organic matter accumulation, NAPP and total organic matter already shown at the ecosystem level. As may be



**Fig. 3.** Landcover–landuse map for the Lower Delta of the Paraná River at Buenos Aires Province. Zones A and B correspond to ecological units (Kandus et al. 2003) and are limited by dashed line.

expected, in the entire region, aboveground biomass was around ninefold higher in afforestation than in marshes but soil organic matter accumulated showed the opposite pattern (10.4:1).

These results indicate that the total organic matter accumulated in marshes (*S. giganteus* and *S. californicus* together) is huge (107 070 200 Mg) compared with that in the afforestation and represents 85% of the total organic matter accumulated at the regional scale. It is important to note that 90% of this magnitude corresponds to *S. giganteus* marsh located principally in Zone B.

Assuming that afforestation in Zone A replaced *S. californicus* marshes and in Zone B replaced *S. giganteus* marshes, the NEE of organic carbon by the entire wetland system due to afforestation practices was around 64 475 400 Mg by 1994. In this way, it is interesting to note that the carbon sink in Argentina due to changes in forest and other woody biomass stocks and abandonment of managed lands in 1994 was only 29 610 850 Mg (Fundación Bariloche 2005). Furthermore, this report indicates that it was 31 430 300 Mg in 1997 and 41 118 250 Mg in 2000. Nowadays, the afforestation area in the Buenos Aires section of the Lower Delta of the Paraná River has not only been maintained but has increased.

Our data show that the loss of soil carbon generated by changes in land use in wetlands in the lower delta is

higher than the carbon sink generated by afforestation in the Argentine.

Others research efforts should be made in order to quantify CH<sub>4</sub> wetland emissions, belowground biomass and biomass and NPP of other forest species.

According to Brinson & Malvárez (2002), in South America, 'most conservation actions rely exclusively on creating and maintaining protected areas such as National Parks, Ramsar sites and Man and Biosphere reserves without a general policy framework for proper management. Inventories of wetland areas and change due to human activity are lacking. Economic pressures to convert wetlands outside of protected areas to agriculture will continue, particularly for countries with the burden of high external debt'. In this way, the authors would like to point out through the results presented that wetland conservation policies should take into account not only structural biodiversity parameters but also the functional properties of the ecosystems. In addition, it would be interesting for national forest policies to consider the type of ecosystem that would be replaced in their planning and their consequences in the short and long term.

Finally, in those cases where afforestation is a traditional activity, the question is whether it is possible to maintain soil organic stocks (i.e. drain less severely) and grow plantations at the same time. Future researches should seek management practices that would allow this practice.

**Table 4** Area in square kilometres and percentage occupied by *Scirpus giganteus* and *Schoenoplectus californicus* marshes and *Salix babylonica* var. *Sacramenta* plantations in the Lower Delta of the Paraná River by 1994

Landcover type	Zone A		Zone B		Total area	
	km <sup>2</sup>	%	km <sup>2</sup>	%	km <sup>2</sup>	%
<i>Salix</i> spp. plantation (willow)	459	35	355	34	813	34
<i>S. giganteus</i> marsh	130	10	504	48	634	27
<i>S. californicus</i> marsh	446	34	0	0	446	19
Other landcover types	287	21	194	18	486	20

Zones according to Kandus et al. (2003).

## Conclusions

(1) From an ecosystem point of view, organic matter accumulation in marsh soils, plus aboveground organic matter in plant tissues in the Lower Delta of the Paraná River marshes, is enormous compared with that in the afforestation environments.

(2) The total organic matter accumulated in marshes (*S. giganteus* and *S. californicus* together) is huge compared

**Table 5** Estimated aboveground biomass, soil organic matter, productivity and total organic matter for each of the landcover types for *Scirpus giganteus* and *Schoenoplectus californicus* marshes and *Salix* spp. plantations in the Lower Delta of the Paraná River region

Landcover type	Aboveground biomass (Mg)			Soil organic matter (Mg) <sup>a</sup>		
	Zone A	Zone B	Total area	Zone A	Zone B	Total area
<i>Salix</i> spp. plantation	5 205 060	4 025 700	9 230 760	5 737 500	4 437 500	10 175 000
<i>S. giganteus</i> marsh	133 900	519 120	653 020	19 674 200	76 275 360	95 949 560
<i>S. californicus</i> marsh	392 480	0	392 480	10 075 140	0	10 075 140
<i>Salix</i> spp. plantation	693 090	536 050	927 140	10 942 560	8 463 200	19 405 760
<i>S. giganteus</i> marsh	193 700	750 960	1 013 349	19 808 100	76 794 480	96 602 580
<i>S. californicus</i> marsh	735 900	0	735 900	10 467 620	0	10 467 620

<sup>a</sup>Includes only aboveground and soil organic matter.

Zones follow Fig. 1.

with that in the afforestation and represents 85% of the total organic matter accumulated at the regional scale. It is important to note that 90% of this magnitude corresponds to *S. giganteus* marsh located principally in Zone B. (3) The loss of soil carbon generated by changes in land use, mainly marshes of *S. giganteus* and *S. californicus* by afforestation of *Salicaceae*, in wetlands in the lower delta is higher than the carbon sink generated by all afforestation in the Argentine.

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