Research Article

Post-Fire Effects in Wetland Environments: Landscape Assessment of Plant Coverage and Soil Recovery in the Paraná River Delta Marshes, Argentina

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ABSTRACT

During 2008, under a region-wide drought, there were a large number of simultaneous fires in the Paraná River Delta region: the most affected vegetation was in marshes dominated by *Schoenoplectus californicus* (C.A.Mey.) Soják or *Cyperus giganteus* Vahl. The objective of this paper was to study fire severity in terms of fire effect on vegetation cover and soil properties, and the recovery of those properties after one growing season, using optical remote sensing techniques and fieldwork data. To this aim, we performed unsupervised classification of Landsat TM imagery and conducted vegetation censuses and soil sampling in November 2008 and May 2009. Our results show that we could identify three fire severity classes: low severity, medium severity, and high severity. These classes are characterized by a remnant vegetation cover of approximately 75%, 25%, and 5%, respectively, and a diminution of soil organic carbon and nitrogen of 66% and 59% in the case of medium severity and high severity. Fire had almost no effect over pH and a slight effect on electrical conductivity. After one growing season, vegetation recovery is dependent on fire severity and hydrological condition, while soil properties did not show signs of recovery. This is one of the first studies of fire effects and recovery on fluvial herbaceous wetlands.

Keywords: *Cyperus giganteus*, fire severity, fluvial wetlands, optical remote sensing, Paraná River Delta, post-fire recovery, *Schoenoplectus californicus*, soil properties, vegetation cover

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INTRODUCTION

In South America, wetlands cover extensive areas associated with the floodplains of large rivers: Amazonas, Orinoco, and Paraná (Neiff *et al.* 1994, Iriondo 2004). Flooding regime is widely recognized as a main factor that determines the presence, extension, and ecological features of wetlands in alluvial plains (Junk 1989, Middleton 2002, Mitsch and Gosselink 2007). However, besides floods, fire and grazing are two other main factors highly related to evolutionary history, organization, and dynamics of the wetlands (Batzer and Sharitz 2006, Keddy 2010).

In the Paraná River Basin, fire has been used historically as a management practice for wildlife hunting and for the elimination of species with little or no forage value (Brinson and Malvárez 2002). This practice is used in foraging lands, especially at the end of winter, to favor growth of tender and more palatable grasses, and the area affected by these fires usually is around a few hectares in size, favoring habitat patchiness. Fire is also used, with much less frequency, after summer season and over the autumn. During the last decade, soybean agricultural expansion resulted in the conversion of many cattle ranches into soybean fields (Aznar *et al.* 2011). This created the need for new grassland fields for cattle ranches, mainly in what were previously considered marginal lands, like wetlands. As a consequence, the encroachment of cattle ranch activities promotes an increase in fire events in the Paraná floodplain, threatening ecosystem health as well as goods and services provided by these wetlands.

The Paraná River wetlands usually store large amounts of carbon in aboveground plant biomass, but also in soil layers, given the high productivity and low decomposition rates that occur when there is a sustained presence of a water table over the mineral soil surface (Mitra *et al.* 2005, Ma and Lu 2008). Sequestration of carbon above the soil surface creates fuel for fires, especially dangerous during dry periods, affecting fire occurrence and dispersion across the landscape (Thompson and Shay1988, Laubhan 1995, Keddy 2010). In wetlands, the presence of an aboveground water table or saturated soil restricts downward penetration of heat into the soil, which affects fire dispersion and severity (Are *et al.* 2009).

Vegetation response to fire is mainly conditioned by life history and functional traits of the species present at any site, as well as disturbance characteristics like frequency, intensity, and fire extent (Hobbs and Huennekke 1992). Fire involves biomass loss, altering the structure and functioning of populations and communities, as well as general ecosystem features (Grime 1979, Keddy 2010). Plant community response to fire also depends on the occurrence of fire in relation to life cycles of those species able to invade burned areas, and on the season of the burn, which affects biomass available as fuel (Laubhan 1995, Turner *et al.* 1998). Fire becomes important during prolonged periods of drought. Low intensity fires can simply remove aboveground biomass, shift composition from woody to herbaceous wetlands, and increase plant diversity (Thompson and Shay 1988).

Direct effects of fire and the overall changes to the ecosystem can both lead to short-, medium-, and long-term changes in the soil (Doerr and Cerda 2005). Physical, chemical, and biological soil properties can be altered by fires (Neary *et al*. 2009, Úbeda and Outeiro 2009). The magnitude and duration of these changes depend on factors like fire severity, peak temperature, and soil type. During severe fires, the most typical effects are loss of organic matter and nutrients through volatilization, ash entrapment in smoke columns, leaching and erosion, alterations of microbial communities, and deterioration of soil structure by affecting aggregate stability (Mataiz-Solera and Cérda 2009). Hydrological and biogeochemical cycles in the post-fire period are dependent on the status of the soil surface. Reduction in vegetation cover will also lead to changes in soil thermal regime and, in general, higher soil and water losses (Certini 2005).

Besides affecting ecosystem structure, fire also impacts ecosystem function and consequently the goods and services provided by wetlands. Carbon storage, flood dampening, and habitat for wildlife are among key wetland ecosystem services (Järvellä 2003, Baigun *et al*. 2008, Kandus *et al*. 2010, Vicari *et al*. 2011) that are affected by varying fire severity (Lal 2007, Salvia *et al.* 2009*a*).

Fire severity describes the magnitude of the disturbance and reflects the degree of change in ecosystem components (Neary *et al.* 1999). From a landscape perspective, post-fire assessment involves the estimation of the affected area, but also the spatial distribution of burned patches and patch severity damage in terms of fuel consumption in fires (Lentile *et al.* 2006, Keeley 2009). Achieving a detailed and rapid knowledge of the degree of environmental change caused by fire in terms of the level of damage and its spatial distribution is essential in order to quantify the impact of fires (van Wagtendonk *et al.* 2004), select and prioritize treatments applied on any site (Patterson and Yool 1998, Bobbe *et al.* 2001), plan and monitor activities of restoration and recovery, and finally, provide baseline information for future monitoring (Brewer *et al.* 2005).

In this way, maps derived from remote sensing data are widely recognized as powerful tools in fire ecology studies at landscape scales, and indeed in most fire-related land management programs, because they can describe spatial patterns of severity of fire events (Kremens *et al.* 2010). The range of remote sensing methods dealing with mapping fire severity includes, among others: differencing of pre- and post-fire original bands or indices (Key and Benson 1999, Loboda *et al.* 2007), thresholding of original bands or indices (Hall *et al.* 1980, Huesca *et al.* 2008, Gonzalez-Alonso and Merino-de-Miguel 2009), unsupervised or supervised classification of original bands or indices (Milne 1986, Miller and Yool 2002), using active fire detections (Justice *et al.* 2002, Giglio *et al.* 2003), spectral mixing analysis (Cochrane and Souza 1998, Gonzalez-Alonso *et al.* 2007), and time series analysis (Milne 1986, Roy *et al.* 2002, Roy *et al.* 2005). At regional to global scales, the detection of burned areas using satellite data has been traditionally carried out by the Advanced Very High Resolution Radiometer (AVHRR) because of its high temporal resolution (Kaufman *et al.* 1990, Chuvieco *et al.* 2008). However, the focus of AVHRR-fire studies to date have been on terrestrial ecosystems like forests (Briz *et al.* 2003, Falkowski *et al.* 2005), grasslands (Kauffman *et al.* 1994, 1998; Mohler and Goodin 2010), or shrublands (Lozano *et al.* 2007). Few works are available on wetlands (Ramsey *et al.* 2002, Cassidy 2007), probably due to their reduced area, natural fragmentation, and difficulty of access.

During 2008, under an extended drought, there were a large number of simultaneous fires in the lower Paraná floodplain, named the Paraná River Delta (Figure 1). Given the fact that fire is a common practice on South American floodplains, which is a potential risk to wetland health, the objective of this paper was to analyze the effect of fires that occurred dur-

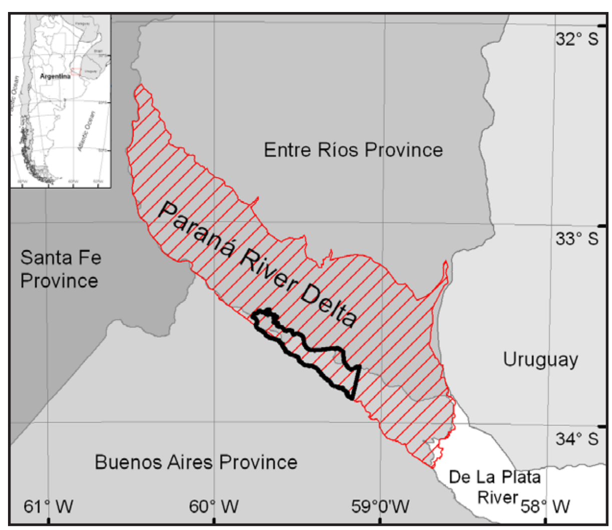

Figure 1. Paraná River Delta Region.

ing 2008 in wetlands of the Paraná River Delta and the recovery of its vegetation and soils after one growth season.

MATERIAL AND METHODS

The Paraná River Delta

The Paraná River Delta region is a mosaic of wetlands extending 300 km along the lower Paraná Basin in Argentina (Malvárez 1999). It comprises approximately 17 500 km^2, between 32º05'S 58º30'W, south of Diamante, and 34º29'S 60º48'W, close to Buenos Aires (Figure 1). The Paraná River drains a 2 310 000 km^2 area and, given its length, basin size, and water discharge, is considered the second most important river in South America. Moreover, among the great rivers throughout the world, it is the only one that flows from tropical to temperate latitudes, where it converges with the Uruguay River into De La Plata River and its estuary. Therefore, the Paraná Delta region is a complex floodplain having unique biogeographic and ecological characteristics in Argentina (Malvarez 1997).

According to Enrique (2009), only 6 % of the delta area is occupied by native forests. Most of the delta is comprised of herbaceous plant communities such as grasslands, marshes, and aquatic plant prairies (Salvia *et al.* 2009*b*). Although these communities are usually dominated by a few species, the landscape heterogeneity of the region entails a mosaic of different environmental conditions that result in a high ecological plant diversity (Kandus 1997, Malvarez 1997, Kandus *et al.* 2006).

One of the most extensive vegetation communities are marshes dominated by equisetoid plants: *Schoenoplectus californicus* (C.A.Mey.) Soják, commonly known in the region as junco, and *Cyperus giganteus* Vahl, regionally known as pirí (Kandus *et al.* 2003, Salvia *et al.* 2009*b*). These plant associations are present in lowlands of the inner portion of islands, covering 17 % of the total delta area (Salvia 2010). Peak standing biomass of these marshes was reported as 1009.91 ±265.64 g m^{-2} (Pratolongo *et al*. 2008), and Net Primary Production (NPP) was around 1299 ±179 g m^{-2} yr^{-1} (Pratolongo 2004). Soils are flooded or saturated during most of the year, giving a low oxygen environment that, along with high plant productivity, promotes a considerable organic matter accumulation rate (752 g m^{-2} yr^{-1} of dry biomass; Pratolongo 2004).

Junco and pirí plant communities provide valuable regional ecosystem services such as flood dampening and sediment retention (Järvelä 2003), as well as wildlife habitat (Quintana *et al.* 2002), and carbon uptake (Pratolongo 2004, Vicari *et al.* 2011). These plant communities have a long history of wild and man-induced fire disturbance to favor growth of forage grasses, resulting in a fire-induced mosaic of stands with different junco and pirí plant development.

During 2008, under an extended drought, there were an unusually large number of simultaneous fires in the Paraná River Delta, some of them of high intensity and persistence. This burning process started early in summer (February) and extended throughout the year, with a main peak of fires in April and May, and a second peak from August to November.

Fires affected almost 15 % of the Paraná River Delta area, but did not equally affect all communities. Junco and pirí marshes were the most affected, having nearly 40 % of their area burned to some degree (Stamati *et al*. 2009, Salvia 2010). Due to their ecological importance and the magnitude of the area affected, we focused the analysis in these marshes, setting our study area on the islands located northeast of the cities of San Pedro, Baradero, and Zárate, between 59º51'W 33º31'S and 59º5'W 34º2'S, which covers 114 290.82 ha (Figure 1).

Methodology

We analyzed Landsat 5 TM optical imagery for the period 2008 to 2009, and we classified two scenes (Path/Row: 226/83). The first one (11 November 2008) belongs to a post-fire situation, and the second one (2 May 2009) corresponds to a recovery state after one growing season. We used unsupervised procedures of classification based on ISODATA (Iterative Self-Organizing Data Analysis Technique) algorithm, obtaining a land cover map for each date. Land cover classes from the 11 November 2008 image classification were recoded based on fire severity; otherwise, land cover classes from 2 May 2009 were related to marsh recovery.

Normalized Burn Ratio (NBR; Key and Benson 1999) was calculated only for the post-fire situation, and results were compared with the classification results. The NBR was defined for Landsat TM images as:

$$NBR = (TM4 - TM7)/(TM4 + TM7), \quad (1)$$

where TM4 is near infrared band and TM7 is mid infrared band from Landsat 5 TM.

All Landsat 5 TM bands used were calibrated and partially corrected for atmosphere effects, considering Rayleigh scattering (Stumpf 1992), and geometrically corrected using ERDAS IMAGINE 8.4 software (Intergraph Corporation, Huntsville, Alabama, USA).

In order to analyze the recovery of the areas under different burning severity, we performed a cross tabulation procedure between land cover classes of November 2008 and May 2009 classifications. Thus, we obtained the amount of hectares for each combination of land cover classes (2008 and 2009) and its corresponding percentage.

Field work was conducted at the end of November 2008, a few weeks after the fires had been extinguished, and at the beginning of May 2009, after one growing season. Satellite imagery acquisition was temporally consistent with the field work periods. We conducted vegetation censuses and soil sampling at landscape scale in order to cover a wide spectrum of burn severity classes and unburned portions of junco and pirí marshes present on the study area. In order to compare fire damage over different sites across the study area, severity classes were defined in terms of remaining standing biomass and litter after fire consumption. In non-flooded sites, and sites saturated but not covered by water, litter consumption was estimated by visual inspection and also by measurement of litter depth from soil samples. In flooded sites, only measurements from soil samples were made. Recovery classes after one growth season were defined in terms of total plant cover.

During field work, we set five transects through the interior of the islands, across the topographical gradient, including areas with different plant communities dominated by junco and pirí. Sampling sites were placed along the transects, as near as possible to the center of the land cover types identified in the ISODATA 23 November 2008 image classification: five replicates in unburned areas, seven in medium severity burned areas and 11 in high severity burned areas. In each site (replicate), we estimated the total plant cover, and we made five censuses in 1 m^2 plots. Plots were randomly distributed along a circle of 5 m radius using the scientific calculator random function and a compass. The number obtained from the calculator was multiplied by 360, ensuring the representation of any possible direction of the compass. In each plot, we recorded total plant cover and cover by species using Braun-Blanquet scale (Muller-Dombois and Ellenberg 1974).

At the same time, in each site, we took samples of soil at the following depths: 0 cm to 10 cm, 10 cm to 20 cm, 20 cm to 40 cm, and 40 cm to 60 cm for the November 2008 field work. Due to the fact that the third depth (40 cm to 60 cm) did not show significant differences between burned and unburned sites, only the 0 cm to 10 cm, and the 10 cm to 30 cm depths were sampled during the May 2009 field work. The collected samples were dried at 60 °C for 48 hr and submitted to the laboratory, where they were sieved with 2 mm mesh. Color was determined in wet and dry condition using the Munsell Soil Color Chart (ColorAccuracy.com, North Brunswick, New Jersey, USA). Soil organic carbon (SOC) and total nitrogen (N) were measured using dry combustion and a CHN autoanalyzer system LECO CR12 (LECO Corporation, St. Joseph, Michigan, USA; Gill *et al.* 2002), and soil pH and electrical conductivity (EC) were measured on the supernatant of a 1:1 soil water extract (Thomas 1996).

To establish the effects of fire on soil variables, we performed analyses of variance (ANOVA) and Tukey test ($\alpha = 0.05$) between the severity classes. To analyze soil recovery, we performed a *t*-test between the data from the two field work dates for each variable only in the first range of depths (0 cm to 10 cm) using the program Statgraphics Plus 5.1 (StatPoint Technologies, Inc., Warrenton, Virginia, USA).

RESULTS

The land cover map obtained by classification of a Landsat 5 TM image from 23 November 2008 shows unburned or low severity areas, medium severity, and high severity areas (Figure 2). From the total analyzed area (109 401 ha), 32 % was not affected by fires, corresponding mostly to levees and polders, while unburned marshy lowlands were rare. Among the medium severity burned areas, some were flooded or had saturated soil at the moment of image acquisition, which was also

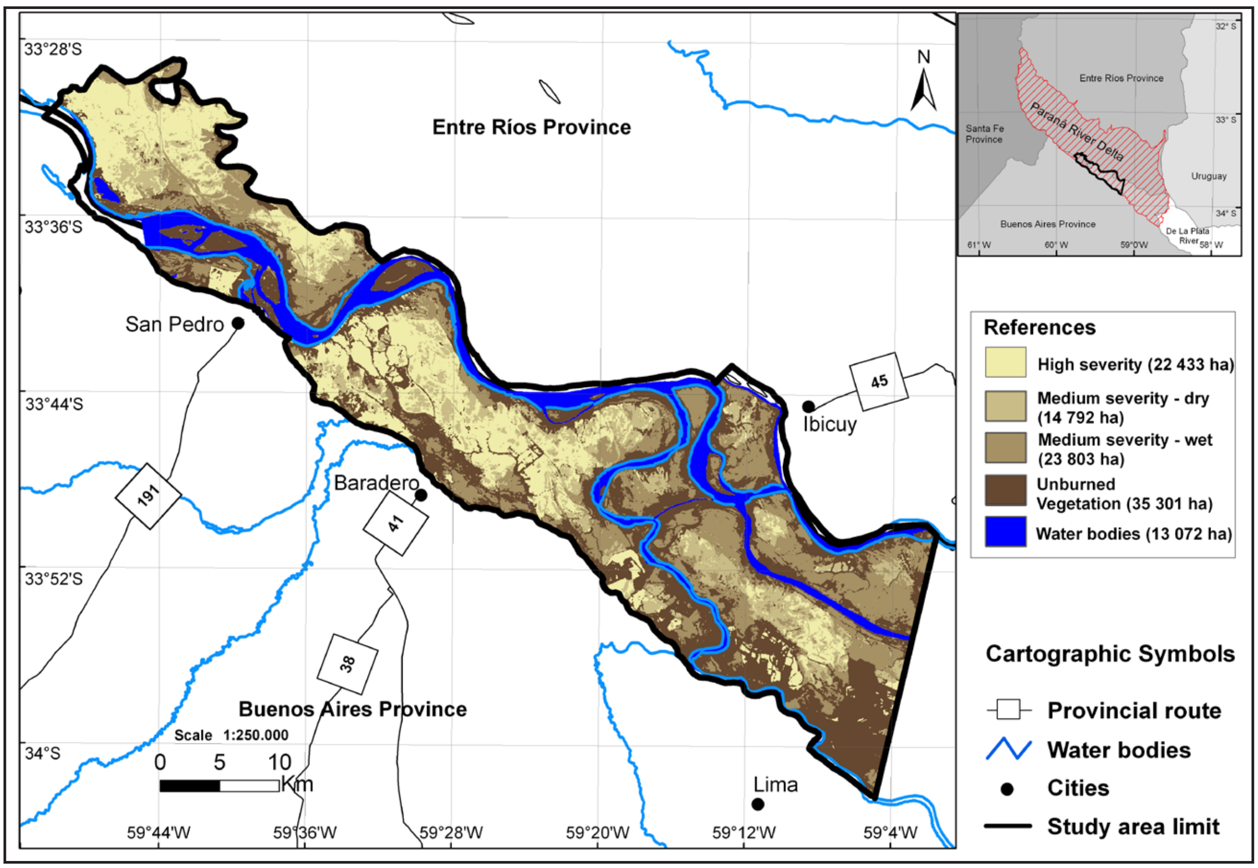

Figure 2. Burned area and severity map, showing the environmental situation as of 23 November 2008.

verified during fieldwork. Landsat spectral signatures can distinguish between severity classes; however, NBR is more sensitive to background material (dry soil or wet soil) than to total plant coverage (Figure 3); that is, there is a better separation between moderate severities with different soil moistures than between moderate severity with dry soil and high severity (dry soil).

The November 2008 field work allowed us to evaluate classes on the map belonging to burned areas, as well as to characterize fire severity (Figure 2). High severity areas show aerial biomass, roots and rhizomes, as well as the organic matter in the top soil layer and litter almost completely incinerated. Only some early and isolated plant regrowth was recorded, particularly under wet conditions. The surface of soil affected by the fire showed Hue 2.5 YR 7/6 according to standard Munsell Soil Color Charts, and a layer of ashes, and a laminar structure, the product of clay particles exposed to high temperatures.

Medium severity areas consistently included two situations within image classification results: 1) sites with dry soil where aerial biomass was burned at the base of plants, causing them to collapse but not be incinerated. In some cases, given the evidence of ash over the top mineral soil layer, part of the litter may have been burned (dry sites). 2) sites with soil saturation where plant aerial biomass was mostly consumed but where the fire didn't affect soil or underground biomass (wet sites). In both medium severity situations, there was some plant regrowth.

Fire severity classes varied in terms of mean cover by species, total plant cover, and species richness recorded during field work in November 2008 (Table 1). Unburned sites tended to be dominated by *S. californicus* or by *C. giganteus*. These species form a continuous top layer with coverage of about 21% to 28% and 2 m to 2.5 m high. The rest of the species recorded were climbers or creeping herbs and grasses that had less coverage.

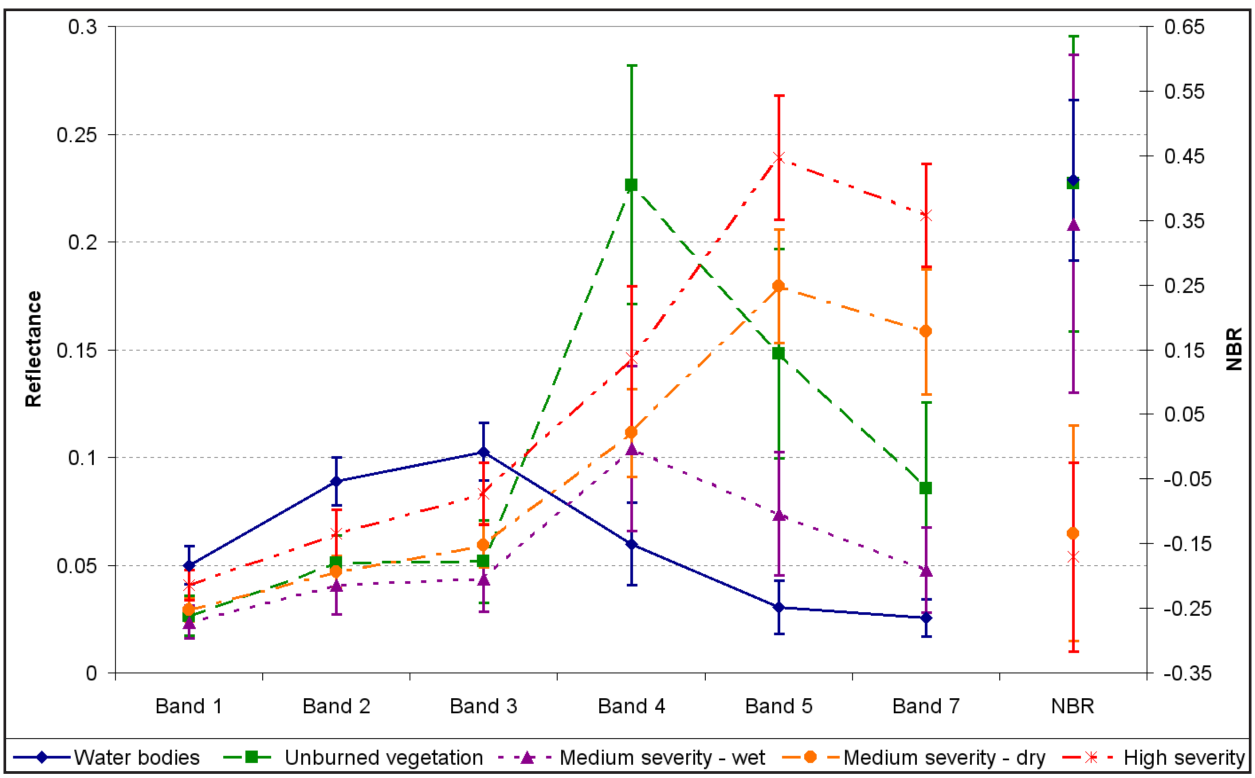

Figure 3. Spectral signatures and NBR index of severity classes as of 23 November 2008.

Table 1. Species list, abundance (in percent cover), and richness in each of the burn and recovery categories found on the study area.

	Post-fire severity (28 Nov 2008)				Marsh coverage (2 May 2009)	
Species	**Unburned**	**Medium wet**	**Medium dry**	**High**	**High**	**Medium**
Cyperus giganteus Vahl	28.10	10.38		1.98	20.04	1.00
Schoenoplectus californicus (C.A.Mey.) Soják	21.20	3.11	1.50	0.73	28.36	12.87
Polygonum hispidum Kunth	7.85	1.29		0.28	0.64	1.50
Alternanthera spp.[1] Forssk.	5.30	5.94	24.00	2.48	0.42	0.03
Echinochloa helodes (Hackel) Parodi	0.25	3.81		0.15		
Cissus palmata Poir.	0.10	0.05			0.04	
Polygonum meissnerianum Cham. & Schltdl.	0.05	0.08				
Mikania micrantha Kunth	0.10	0.06			0.00*	0.30
Polygonum hydropiperoides Michx.		0.01				
Vigna luteola (Jacq.) Benth.	5.90	0.16				
Pontederia cordata L.	0.10				0.01	
Senecio bonariensis Hook. & Arn.	0.00*					
Polygonum stelligerum Cham.	0.10	0.04			0.06	
Zizaniopsis bonariensis (Balansa & Poitr.) Speg.		0.05	0.20	0.03		
Echinodorus grandiflorus (Cham.& Schltdl.) Micheli		0.03				
Carex spp. L.		0.08			0.00*	0.00*
Hymenachne grumosa (Nees) Zuloaga		0.25			7.83	
Mimosa tweedieana Barneby ex Glazier & Mackinder		0.01				
Hydrocotyle bonariensis Lam.		0.03				
Tripogandra elongata (G.F.W. Mey.) Woods.	0.10					
Tradescantia anagallidea Seub.	3.75	0.94				
Jaborosa integrifolia Lam.				0.08		
Aeschynomene montevidensis Vogel					0.00*	
Polygonum punctatum Elliott					0.10	7.67
Typha spp. L.					0.76	
Aspilia silphioides (Hook. & Arn.) Benth. & Hook.f					0.07	
Sesbania punicea (Cav.) Benth.					0.00*	
Scirpus giganteus Kunth						1.83
Azolla filiculoides Lam.						0.03
Ranunculus apiifolius Pers.						0.001
Low bearing grasses					24.81	1.9
Total coverage	76.25	31.13	26.75	5.50	76.37	31.9
Specific richness[2]	14	18	3	7	30	19
Median height of junco or pirí (m)	1.8				1.8	1.5

[1] Includes *A. philoxeroides* (Mart.) Griseb. and *A. reineckii* Briq.
[2] Includes species that were found in a very low cover.
* Species with cover <0.001.

Medium severity or wet sites showed high species richness despite total cover being low; on the other hand, sites with dry soils showed very low species richness and coverage was dominated by *Alternanthera* spp., which show creeping behavior, covering large surfaces. High severity areas showed the lowest total cover.

Both unburned areas and the two severity classes showed SOC and N concentrations diminishing with depth. However, for the upper 10 cm, there was a significant decrease of 66% and 59% in SOC and N concentration, respectively, in high severity areas when compared with unburned sites (Figure 4a and 4b). In deeper layers, there was no significant difference, indicating that fire only affected the top layer of the mineral soil. In addition, the top soil layer showed a significant increase of EC in high severity burned sites when compared to unburned sites. On the other hand, pH showed no significant differences between either sites or depths (Figure 4c and 4d).

In the May 2009 Landsat scene classification, after one growing season, we identified five classes: almost 25% of the study area was marsh with high plant cover, nearly 20% was marsh with medium plant cover, about 8 % was marsh with low plant cover, only 3% of the study area had bare soil, and water bodies covered nearly 12% of the total area, similar to that of November 2008 (Figure 5). Water bodies and bare soil classes show distinguishable spectral signatures while those of the three plant cover classes have the same shape, but mostly differ in the infrared bands (Figure 6).

In general, after one growing season, the junco and pirí marshes with high plant cover reached values similar to those encountered at unburned sites in November 2008 (Table 1). However, in comparing with undisturbed marshes (14 species recorded), high cover marshes have a significant increase in species richness, registering an average of 30 species. Medium cover marshes had lower plant cover and a richness of 20 species. Unfortunately, given its scarce land cover (8%), we could not sample low coverage marshes. After one growth season, the concentration of SOC and N remained at the same levels as in the post-fire sample (Figure 7), so the loss of C and N was not recovered.

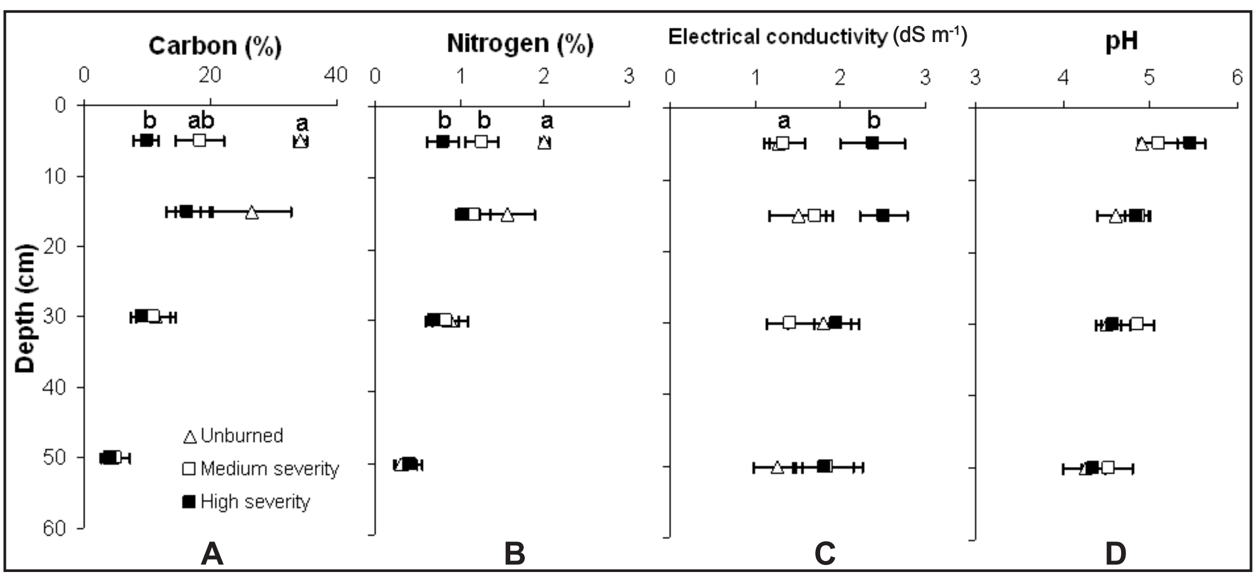

Figure 4. Soil organic carbon (%), total nitrogen (%) concentrations, electric conductivity (dS m^{-1}), and pH for the three different fire severity classes recorded in four soil depths (0 cm to 10 cm, 10 cm to 20 cm, 20 cm to 40 cm, 40 cm to 60 cm). Different letters indicate significant differences between the unburned class and the two fire severity classes ($P < 0.05$).

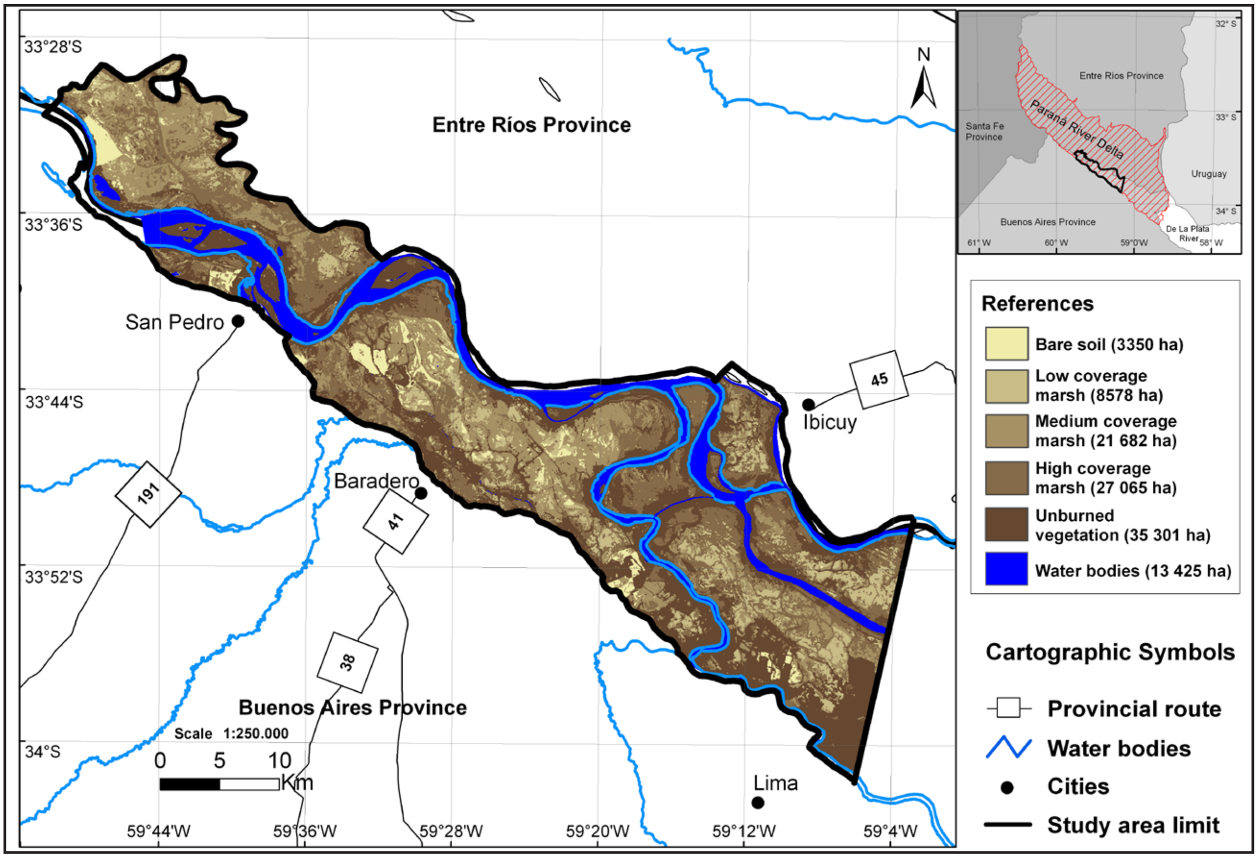

Figure 5. Map of post-fire vegetation recovery up to 2 May 2009.

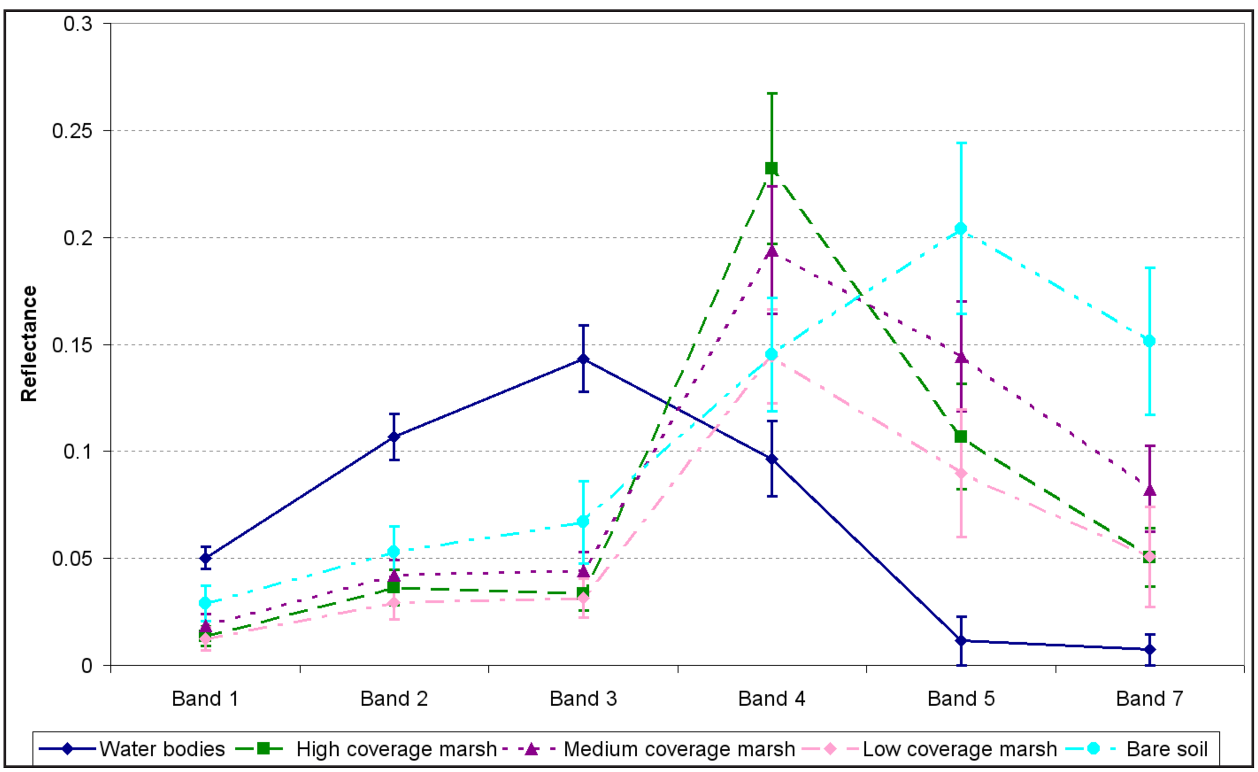

Figure 6. Spectral signatures of vegetation recovery classes.

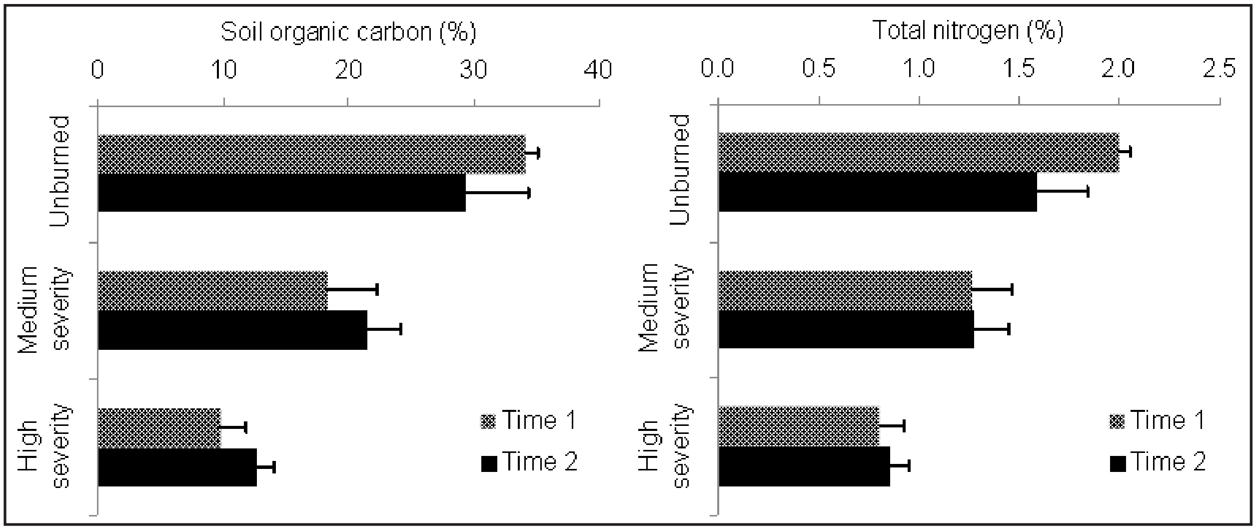

Figure 7. Soil organic carbon (%) and nitrogen (%) concentrations for unburned sites and two burning severity classes recorded in the first soil depth (0 cm to 10 cm) in November 2008 and May 2009 after a period of growth. No significant differences were found ($P < 0.05$) between dates.

The cross tabulation of the November 2008 and May 2009 map classes shows that the recovery of the marshes on the May 2009 image classification is closely related to the burning severity found in November 2008 (Table 2). It can be seen that, in most of the area that had fires of high severity, junco marsh developed only partially (medium cover marshes), while where fires were less severe, junco marshes had a greater recovery (a mixture of high cover marshes and medium cover marshes). Those areas that had dry soils and medium fire severity had a variable recovery by May 2009: 41% of its surface had medium cover junco marshes, and 41% had high cover junco marshes. On the other hand, most of the areas with medium fire severity and wet or saturated soils turned into high cover junco marshes (66%).

DISCUSSION

The November 2008 map shows that almost all the junco marsh of the studied area was burned. This is consistent with the results presented at regional scale by Stamati *et al.* (2009) and also by Salvia (2010): the former authors used on-screen visual interpretation of Landsat and CBERS (China-Brazil Earth Re-

Table 2. Recovery categories percent cover for each burn severity class. Bolded numbers indicate the highest percentage values.

		June 2009 coverage					
		Water bodies	**Bare soil**	**Low marsh**	**Medium marsh**	**High marsh**	**Total**
November 2008 severity	Medium-wet	2.48	0.20	16.45	14.63	**66.24**	100
	Medium-dry	0.02	1.67	16.72	**41.08**	**40.51**	100
	High	0.02	13.61	9.52	**53.94**	22.91	100

sources Satellite program) images, and the latter used a segmentation procedure over the near infrared band of a SACC-MMRS (Satélite de Aplicaciones Científicas C-Multispectral Medium Resolution Scanner) image (pixel size: 175 m). Both results estimate that, in this study area, almost 90 % of junco marshes were burned.

The map obtained in this work by means of an unsupervised classification of Landsat 5 TM reflective bands distinguished between three fire severity classes: unburned, medium burn severity, and high burn severity. In addition, it also differentiated soil conditions (dry and wet) in the medium burn severity class. The spectral signatures associated with this map show that all Landsat bands, especially those of infrared wavelengths, contribute to spectrally differentiate the severity classes. The unburned class has a typical signature of vegetation cover, and high severity class has a spectral signature resembling the one for bare soil. The class "medium severity-wet" has lower reflectance than unburned vegetation class in all Landsat bands, but still holds the same shape. This is because medium severity wet pixels are composed of junco marsh vegetation and wet or saturated soil, so its spectral signature is a mixture of that of water and plant cover. On the other hand, the "medium severity-dry" signature has the same shape as "high severity" class, but with lower reflectance on all bands.

In contrast, the NBR index performed better in distinguishing between different backgrounds (i.e., soil flooding conditions) than between fire severities. It showed a better separation between dry and wet soil, than between medium and high fire severities when the soil was dry.

This finding is consistent with the literature about this index. Even when NBR and dNBR indices are used widely for fire detection and assessment, they were originally designed for forest fires (Key and Benson 1999), and were mostly used on terrestrial ecosystems. Nevertheless, Epting *et al.* (2005) found that the NBR index did not perform well on unforested areas, like the marshes evaluated in this paper. Furthermore, Kremens *et al.* (2010) state that the relationships between the NBR family of indices and commonly observed post-fire effects are non-linear and exhibit dependencies on scale, ecosystem, and soil type.

Fire severity classes obtained through spectral classification were consistent with plant cover recorded during field work. Since there are few papers that address and quantify fire effects on herbaceous wetlands, we believe that our results on vegetation consumption due to fire severity could be compared with those on grassland fires. Kauffman *et al.* (1994) found, in grasslands and savannas of the Brazilian Cerrado, that over 97 % of the aerial plant biomass was consumed under controlled fires. Under high severity fires, we found that plant coverage was reduced, on average, to 5.5 %, which represents a decrease of 70 %, though in some sites there was almost no remaining plant cover. On the other hand, Kauffman *et al.* (1998) found that the proportion of consumed plant biomass in pastures of the Brazilian Amazonia varied between 21 % and 84 %, which is similar to what we found in medium severity burned areas.

After one growth season, during field work performed in May 2009, we found that most of the area showed some degree of recovery, which is what is also seen in the map obtained from the May 2009 image classification. Most of the area that was burned in November 2008 corresponds to one of the three vegetation classes for which spectral signatures are shown in Figure 6. The bare soil class has low cover (approximately 13 %) and water bodies had no change in their areas.

Laterra (2003), in the case of *Paspalum quadrifarium* Lam. marshes in the Pampa Deprimida (Argentina), found that after fire and in absence of grazing, vegetative cover is restored to values similar to undisturbed stands within the first growing season. However, compared with undisturbed marshes we found a significant increase in species richness, and half the cover difference is made up for by the presence of low-bearing grasses and herbs (i.e., *Paspalum quadrifarium*, *Ranunculus apiifolius* Pers., and others), which are not common in these lowlands that tend to be permanently saturated. Suding (2001) and Laterra (2003), despite the finding that there is a recovery in the plant cover, remark that fire events modify the relationship between species, producing changes in species dominance as a result of species-specific responses. In general terms, it has been observed that in early stages of post-fire succession, there is an increase in diversity and biomass, while in later stages, diversity diminishes. In early stages, low competition for resources is expected, and as it increases in the following stages, diversity declines (Kunst *et al.* 2003, Laterra 2003, Norton and De Lange 2003).

Furthermore, we found that the degree of vegetation recovery is linked to both fire severity and soil hydrologic condition. Fire severity and vegetation recovery are negatively related (i.e., less severity led to higher recovery). This could be due to the fact that exposure of dry soils to intensive fires leads to burning of the rhizomes, which are responsible for junco and pirí marshes vegetative expansion and growth, and guarantee their fast recovery after disturbances like fires. During the field work, it was seen that these organs were exposed, and in many occasions charred, leaving large patches without vegetal elements that could start the recovery process. Within the areas that were burned with medium severity, rhizomes would have maintained at least part of their structural and functional integrity, promoting community recovery.

We also found that areas that had a wet or saturated soil had higher recovery than those with dry soils. Being that a few days before November 2008 field work there had been an eolic tide of large magnitude, it is fair to assume that those areas that, during the mentioned fieldwork (and image acquisition), were saturated or remained wet, are the ones that, in general terms, have better water supply from streams, which, in a drought period such as the years 2008 and 2009, is the main water source.

The color (Hue 2.5 YR 7/6) determined for the top mineral soil layer of sites with high fire severity is consistent with the exposure of clay particles to high temperatures. Hepper *et al.* (2008) show that high temperatures (>500 °C) in Entisols with loam texture change the texture of the soil, reducing the proportion of clay and increasing sand proportion. Ulery and Graham (1993) indicate that, from thermal decomposition, the clay releases compounds of amorphous silicon and aluminum that act as cement for the particles, increasing the fraction of sand. When plant cover and litter layers are consumed, there is a reduction of organic matter that potentially could enter the mineral soil, resulting in changes of the soil's physical properties and a decrease of the soil's physical resilience (Neary *et al.* 1999, Certini 2005, Hubbert *et al.* 2006).

The soil parameters of junco and pirí marshes showed that changes in carbon, nitrogen, and electric conductivity are restricted to the top soil layer, and with significant differences between burned and unburned sites in 0 cm to 10 cm depth. There was no effect recorded below 10 cm depth. This is consistent with literature that states that, owing to the low thermal conductivity of mineral soils, the depth affected by fire is limited to the top few centimeters (De Bano *et al.* 1998, Mataiz-Solera *et al*. 2011), and that in moist soils the high specific heat and the evaporation of water will not allow soil temperatures to exceed the water boiling point (Campbell *et al.* 1995). Furthermore, Frandsen and Ryan (1986) and Clement *et al*. (2011) state that most of the energy released by combustion of aboveground fuels is not transmitted downward. Moreover, Packham and Pompe (1971) found that only about 5 % of the heat released by a surface fire was transmitted into the ground.

Severe fires can often cause changes in successional rates, alter above- and belowground species composition, generate volatilization of nutrients and ash entrainment in smoke columns, produce rapid or decreased mineralization rates, alter C:N ratios, and result in subsequent nutrient losses through accelerated erosion, leaching, or denitrification. In addition, changes in soil hydrologic functioning, degradation of soil physical properties, decreases in micro- and macrofauna, and alterations in microbial populations and associated processes can occur (Neary *et al.* 1999).

The greatest potential loss of carbon in these ecosystems happens through the burning of the organic layer formed by litter and roots, because 83 % of the total site carbon (61 Mg C ha^{-1}) in flooded marshes of *Scirpus giganteus*, in the lower delta of the Parana, is in the organic layer (Ceballos *et al.* 2012). This loss of C and N was not recovered after one growth season. This coincides with the findings of Vergnoux *et al.* (2011), who reported that fires in Mediterranean forest led to significant decreases of total organic carbon that had not returned to levels in unburned forest 16 yr after the last fire. Where fire is an important agent of disturbance, it reinforces the likelihood of N limitation; fire in particular volatilizes much more N than it does phosphorus (Vitousek *et al.* 2010).

Sites with high fire severity increased EC in the top soil layer, probably due to an increase in exchangeable cations in the soil (Pettit and Naiman 2007; Notario del Pino *et al.* 2008).

Based on the scale of conductivity (United States Salinity Laboratory Staff 1954), those soils not affected by fire, and those in sites with medium severity fires (1.5 dS m^{-1}) are classified as soils without saline effects. Sites with high severity fires increased their EC (mean EC 2.5 dS m^{-1}), constraining the growth of species that are highly sensitive to soil saline content. The pH showed no significant difference, probably because the ash was exported by precipitation or flooding and not incorporated into the particularly acid soil in this region of the Paraná Delta.

CONCLUSIONS

In this work, we mapped fire severity and vegetation recovery at landscape scale over the Parana River floodplain by means of unsupervised classification of Landsat 5 TM multispectral data. The use of NBR index, which was designed originally for forest fires, did not prove to be useful in the herbaceous wetland in this case. However, its usefulness in other wetlands and herbaceous landscapes remains to be tested.

We documented vegetation recovery in this wetland composed primarily of herbaceous communities of clone perennial species after one growth season. However, the degree of recovery is negatively linked to fire severity (i.e. less severe fires led to higher vegetation recovery).

The low oxygen availability and high productivities of Parana River Delta's communities allow them to capture high amounts of carbon, especially on the accumulated organic layer over the surface of mineral soil. By the combination of long drought periods and the effect of fire, this organic layer was totally removed, releasing the accumulated carbon to the atmosphere and the water courses, even diminishing carbon concentration over the first 10 cm of mineral soil.

We assert that fire also affects nutrient availability, modifying the dynamics of vegetation communities as well as loss of elements to water courses, given the frequent flooding of these wetlands, and modifying the chemical composition of these water bodies.

In addition, the effect of fire will not be the same if it happens at the end of summer or at the end of winter and, on wetlands, apart from season the effect of fires also depends on the hydrological condition, particularly the presence of an aboveground water table or soil moisture condition. This feature is critical for downward penetration of heat into the soil, or to upward heat transfer to aboveground biomass and the atmosphere. when soil moisture is low, fires will be of higher intensity, given the dry fuel material. However, when the soil is saturated fire affects principally the dry aerial biomass so that rhizomes and carbon, nitrogen, and soil nutrients storage is less affected.

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