EFFECTS OF INTENSIVE AQUIFERS EXPLOITATION ON GROUNDWATER SALINITY IN COASTAL WETLANDS

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Abstract

The coastal plain of the Río de la Plata constitutes a large wetland which develops on the right margin of the river estuary. Anthropic activities such as intensive exploitation of groundwater carried out in the vicinity of the wetland can modify the natural hydrological regime. The aim of this work is to asses the effects of intensive aquifer exploitation in coastal wetlands using hydrogeological models. Such models allow to evaluate changes in the environmental conditions of wetland at regional level. The hydrogeological model exposed in this work shows how the intensive groundwater exploitation affects the wetland area, generating important variations both in the groundwater flows and in the salinity of the groundwater. Identification of these modifications to the environment is important to generate guidelines leading to minimize these affectations.

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1. Introduction

Coastal wetlands are among the most fluctuating and productive ecosystems of the world, and perform a wide range of ecosystem services of socio-economic value (Odum, 1978). These values include shoreline stabilization, sediment and nutrient retention, high primary and secondary production, habitat and food resources for terrestrial, aquatic fauna, coastal water quality buffering, biomass and biodiversity reservation, and recreation and tourism amenities (Mitsch and Gosselink, 1993; Mitsch and Gosselink, 2000; Gedan et al., 2011). These ecosystems play an important role in wildlife conservation and can also act as sinks or sources of a wide range of substances, such as nutrients, organic matter, pollutants, etc. (Boorman, 1999). Ecological systems, however, are usually affected by human activities, which may cause a loss and degradation of their natural status, a decline of their biodiversity, an alteration of their ecological functioning, and a limitation of their ecosystem services (Brinson and Malvárez, 2002; Yamashiki et al., 2006; Moiwo et al., 2010a).

In Argentina, many coastland sectors are characterized by the presence of wetlands. Most of them are natural reserves sites, and some of them are included in the world heritage network of the RAMSAR sites (Isacch et al., 2010). Despite these valuable ecosystems are protected, they undergo high ecological and physical pressures due to anthropogenic activity (Brinson and Malvárez, 2002; Carol et al., 2016; Cellone et al., 2016; Carol et al., 2017; Idaszkin et al., 2017).

The coastal plain of the Río de la Plata constitutes a large wetland which develops in the coastal zone on the right margin of the Río de la Plata estuary. Although numerous natural reserve areas are present in the wetland, there are also large urban sites, ports and industries along the middle estuary. The wetland sector, located in the vicinity of La Plata city, presents a hydrological functioning which depends on the contribution of water coming from rains, tidal flows and groundwater discharge. Such groundwater discharge constitutes one of the main hydrological components that sustain the wetland and it derives both from the local flows in the unconfined aquifer and from the regional flows in the underlying semi-confined aquifer (Carol et al., 2013). Although in this wetland sector natural reserve areas are present, urban and industrial development in other areas has affected surface and groundwater dynamics and chemistry (Vecchioli, 1998, Santucci et al., 2017a, Santucci et al., 2018). Likewise, anthropic activities such as intensive exploitation of groundwater carried out in the vicinity of the wetland (Kruse et al., 2013) can modify water flows affecting its natural hydrological regime (Carol et al., 2013). The objective of this work is to asses the continental aquifer intensive exploitation effects on groundwater salinity of adjacent coastal wetlands, through the use of hydrogeological models. This will enable us to evaluate changes in the environmental conditions of wetland at regional level, and to generate bases for the water resource management in the region.

2. Study Area

The study area comprises the wetland sector of the coastal plain of the Río de la Plata adjacent to La Plata city (Fig. 1). This wetland sector presents anthropized sectors and also reserve areas where ecological characteristics are preserved. In the most continental sector, the wetland is bordered by a loess plain environment which is topographically higher than coastal plain, and consists of eolian sediments with thicknesses close to 40 m, which have been reworked by water in some sectors (Teruggi, 1957).

The area is characterized by a humid temperate climate with a mean annual precipitation of 1061 mm (period: 1901-2002). The mean annual temperature is 16 °C and, according to the soil water balance (Thornthwaite and Matter, 1957), actual evapotranspiration is 783 mm/year, infiltration 225 mm/year and runoff 53 mm/year (Kruse et al., 2004).

This sector of the coastal plain is part of the lower basin that comprises a set of streams which drain the water from the adjacent loess plain to the Río de la Plata. The wetland receives the local groundwater discharge from the unconfined aquifer and also the regional discharge from the underlying semi-confined aquifer. Precipitation is the source of recharge of the unconfined aquifer, whose regional groundwater flow is towards the Río de la Plata (Carol et al., 2013). The recharge of the underlying semi-confined aquifer is local and indirect from the unconfined aquifer mostly occurring in the high basin sector, and its groundwater flow is also towards the Río de la Plata. The semi-confined aquifer presents high salt contents in the adjacent area to the middle Río de la Plata estuary due to a paleo-seawater intrusion that is preserved at present (Santucci et al., 2016; Santucci et al., 2017b).

In the loess plain sector, the unconfined aquifer is composed of silty to silty-clayed sediments with calcium carbonate concretions. The mean aquifer thickness is 32 m and it generally wedges out towards the Río de la Plata. In the coastal plain sector the unconfined aquifer corresponds to silty to clayed sediments that alternate with lenses of fine-grained sands and marine shells (Fucks et al., 2017). The uppermost sediments in the coastal plain consist of fine-grained sands corresponding to alluvial deposits of the levee in the vicinity of the Río de la Plata. Together, both units have an average thickness of 30 m. The unconfined aquifer overlays a 4 m average thick silty-clayed aquitard which hinders the hydraulic transmission between the unconfined aquifer and the underlying semi-confined aquifer. The semi-confined aquifer is composed of fine- to medium-grained quartz sands of fluvial origin with a mean aquifer thickness of 21 m and overlie green clays.

In the loess plain, where La Plata city is located, the semi-confined aquifer is intensively exploited due to the fact that it constitutes the main source of water supply in the region. The urban population has increased progressively during the last years having registered 694613 inhabitants for 2010 (INDEC 2010). This urban sprawl took place from the inner area towards the environs of the city, especially in the south region. In a more restricted south sector, distant from the urban area and in the loess plain uplands, intensive horticultural activity is developed. This activity is performed in small pieces of land. This generates an important production, in which supplementary flood irrigation was introduced in the 1980s and drip irrigation in the 1990s. As consequence, during the last years, land use changed from open field to greenhouses, decreasing the unconfined aquifer recharge due to the plastic coverage of the greenhouses structures and increasing the need for artificial irrigation (Delgado et al., 2018). This intensive water exploitation generated both in the urban area for human supply and in the agricultural area for irrigation, changes the water balance at the regional level (Auge, 2005).

3. Methodology

A numerical model of groundwater flows within the unconfined and semi-confined aquifers was developed using a three-dimensional (3-D) system modelling with the MODFLOW software package (Harbaugh et al., 2000), and the generic computer program SEAWAT (Langevin et al., 2008) was applied to simulate the salinity changes of water in the wetland area. It allowed us to simulate the temporal variations in groundwater salinity in the wetland area at regional level (loess plain and coastal plain). In previous research, MODFLOW was applied for steady state groundwater flow simulation (the natural conditions was used as the initial condition for transient state) and transient simulations for 1940 (a prior 10-year simulation was run to achieve model calibration), 1988 and 2008 (Kruse et. al., 2013). In the modelled area, topographic curves, streams, aquifer levels and exploitation wells distribution were digitized. Groundwater recharge occurs due to the infiltration of rainfall excess and was considered as constant in all models. Thus, considering the average values of the rainfall records (period: 1909-2015), it can be observed that in the years analyzed in such models, the mean precipitation values are relatively similar (1024 mm/year in 1940, 1020 mm/year in 1988 and 1069 mm/year in 2008). In addition, the variations in the mean annual temperatures are slight, with values of 16.1 °C for the decade that includes 1940, 15.9 °C for the one including 1988 and 16.0 °C for the one including 2008. Based on those data, it was considered a constant water balance at a regional scale. Groundwater flow and potentiometric map of the natural condition were made according to data from primitive wells in the region and to the configuration of groundwater flow maps of neighboring basins unaltered by exploitation. The resulting head value distribution was the initial condition for transient state simulation. The calibration was performed in transient state, to simulate groundwater levels in 1940. Hydraulic parameters from 45 pumping wells and a potentiometric map were considered for calibration purpose. The sensitivity analysis for the 1940 numerical model was carried out by systematically varying hydraulic conductivity (k_h and K_v) or the storage coefficient (S). Head withdrawal for 2008 were simulated based on data from 154 pumping wells (flow rate and a potentiometric map), and a head distribution from 1988 as an intermediate control point (with flow rate from 88 pumping wells), where the flow model was calibrated by Kruse et al., 2013.

For this purpose, different hydrological situations were modeled considering a natural condition and, two other situations where changes by pumping and waterproofing in recharge areas were documented for the years 2008 and 2015. Based on the temporal analysis of satellite images (Google Earth software), urbanized areas and greenhouses were delimited for the years 2008 and 2015. All data were processed in a Geographic Information System (ArcGIS).

For the year 2008, 154 wells with an exploitation volume of 1.44 10^3 m^3/day per well were considered. Those wells were installed in the semi-confined aquifer to urban needs supply. An actual well distribution, depth of extraction (−40 m asl) and exploitation rate were used. In addition, greenhouses for horticulture use a volume of 3.2 10^3 m^3/day for irrigation purpose, and their location comprises 36 km^2 in the loess plain uplands (Fig. 1). The pumping system for irrigation was simulated with an uniform well distribution between the cells of semi-confined aquifer (−40 m asl), below the agricultural area.

A constant recharge of 6.1 10^{-4} m/day given by the rainfall infiltration from surface to the unconfined aquifer was considered for the whole area (Kruse et al., 2004), except for the greenhouses and urban areas. The infiltration rate was set as constant because the years considered for the simulation show similar hydrological situations regarding the average conditions of the water balance (Kruse et al., 2013). In greenhouses areas, it was considered a withdrawal of 3.8 10^{-4} m/day, taking into consideration the impermeable greenhouses roofs, and 1.3 10^4 m^3/day of groundwater extraction from the unconfined aquifer for irrigation purposes. For the urban areas it was considered a hypothetical recharge of 0 m/day. Potentiometric head for 2015 were simulated from 2008 numerical model results. For 2015, it was considered an increase of 66 km^2 in agricultural land use, and also an increase in water needs for irrigation (6 10^4 m^3/day from semi-confined aquifer and 2.4 10^4 m^3/day from unconfined aquifer). The increase of horticultural areas was defined through satellite images analysis.

The regional model was developed for a 915 km^2 basin, which is drained by five main streams. The modeled area was discretized into 95 x 97 cells with 440 m × 315 m in x and y directions. In the z direction, a three-layered system with a thickness ranging from 15 m to 48 m in Layer 1 (unconfined aquifer), 3 - 18 m in Layer 2 (aquitard) and 8 - 47 m in Layer 3 (semi-confined aquifer) was defined. The thicknesses of the modeled aquifer units and the aquitard were adjusted according to the records of perforations and isopach maps of the area (García et al., 2017). Regionally, groundwater flows from the basin limit (the west limit) to Río de la Plata (to northeast). For boundary condition, a specified-head boundary of 0 m asl was considered at the northeastern limit of the basin, where the Río de la Plata is, which is a natural discharge boundary. For the unconfined aquifer, the southwestern limit was considered as no-flow boundary, because is a basin perimeter. For the semi-confined aquifer and the confined layer, the western limit was considered as specified head boundary, to account by the regional inflow from semi-confined aquifer who discharge into Río de la Plata. The hydraulic head was specify from 14 m asl to 25 m asl (from southwest to northwest), based on the potentiometric maps of the semi-confined aquifer (Kruse et al., 2013). The northwest and southeast boundaries are hydraulic boundary (no-flow boundaries), for modeling purpose. There were defined from the potentiometric maps of the unconfined and the semi-confined aquifer (Kruse et al., 2013) as groundwater streamlines, perpendicular to Río de la Plata and parallel to the equipotential lines. It was assumed no-flow through the streamline. Constant head was considered for Río de la Plata and drains for the streamlines, existing groundwater discharge zones when the water table exceeds the terrain level.

The hydraulic parameters used in each of the modeled layers are based on the values estimated in field tests through several pumping tests (Kruse et al., 2004; Auge, 2005). As a result of these tests, in the unconfined aquifer, the hydraulic conductivities vary between 3 and 10 m/d, the average transmissivity is 310 m^2/d and the effective porosity is 0.08. This layer was divided into two main zones: one zone coincides with the high plain area (loess sediment), and the other zone coincides with the coastal plain area (wetland). In the aquitard, the horizontal hydraulic conductivity (Kh) range from 0.4 to 0.04 m/day and a vertical anisotropy of 10 (kh/kv, where Kv is the vertical hydraulic conductivity) it was estimated for modeling purposes. In the semi-confined aquifer, the hydraulic conductivity is 20 m/day, the effective porosity is 0.20, and the transmissivity is 500 m^2/day. For modelling purposes, an underlying impermeable layer of clay was considered between -40 and -60 m asl. Recharge from precipitation, drainage system and withdrawal from pumping (discharge) at semi-confined aquifer, only in agricultural land use, were indicated on the first layer (unconfined aquifer). The pumping wells for urban water supply and for irrigation installed at the semi-confined aquifer were located on the third layer. To simulate agricultural impact in 2008 groundwater head, a simulation for a 10-year periods was carried out considering land use, water extraction for urban and agricultural needs and groundwater levels from 2008 model (from Kruse et al., 2013). With the adjusted 2008 model a transient simulation for 2015 (7 years onwards) was carried out.

Based on the hydrodynamic numerical model, the generic computer program SEAWAT (Langevin et al., 2008) was applied to simulate the salinity changes of groundwater for steady state (natural conditions) and transient state (2015). SEAWAT is a coupled version of MODFLOW (Harbaugh et al., 2000) and MT3DMS (Zheng and Wang, 1999; Zheng, 2006) and it was used to simulate three-dimensional, variable-density groundwater flow. The variable-density groundwater flow equation was solved by the VDF Process of SEAWAT. The VDF Process uses MODFLOW methodology to solve the variable-density ground-water flow equation (which include parameters as fluid density and salt concentration) (Langevin et al., 2003). SEAWAT integrates MT3DMS Transport Process to solve the solute transport equation (Langevin et al., 2008).

Boundary condition for salinity in the unconfined aquifer was defined as: a) constant concentration of 0.5 g/L at Río de la Plata (Santucci et al., 2016) (northeast); b) watershed boundaries with constant concentration from 0.44 g/L (northwest) to 0.86 g/L (southwest) (reference data from observation wells P21 and P23) (Table 1). Boundary condition for salinity in the semi-confined aquifer and in the aquitard was defined as: a) constant concentration of 19 g/L for the paleo-seawater intrusion, at Río de la Plata boundary (Santucci et al., 2016) (northeast); b) West boundaries with constant concentration from 0.31 g/L (northwest) to 0.51 g/L (southwest) (reference data from observation wells P22 and P17) (Table 1). Recharge with variable salinity from 0.44 to 0.8 g/L of TDS (Total Dissolved Solids), was defined based on the lowest salinity observed in the unconfined aquifer (Table 1). Initial salt concentration for the transport simulation resulted from the interpolation of the boundaries condition for salinities, to the grid. It was run a simulation with a total elapse time of 18250 days, with 50 time steps, one for each year. The model results were compare with the TDS observation data from the wells located outside the urban area and close to the boundaries, selected as representatives for the regional unmodified salinity condition. For natural condition and 2008 the initial salt concentration for the transport simulation resulted from the interpolation of the boundaries salinities to the grid. Resulting salt concentration from 2008 transport model (at 1095 days simulation time) was considered as initial condition (starting concentration) for 2015 transport simulation.

Variable-density, saturated ground-water flow simulation for natural condition was carried out using 10-year stress period (3650 days elapsed time) with 10 time steps and 365 length each (base on steady state flow model result as flow condition). The transport transient simulation for 2008, was run for 10-year stress period (3650 days elapsed time) with 10 time steps and 365 length each (the same discretization than the flow model). Groundwater flow simulation for 2015 was run for a total elapse time of 2555 days (7 year stress period) with 7 time steps, one for each year (the same discretization than the flow model).

For the transport simulation in the aquifer (MT3DMS/SEAWAT software), the longitudinal and transverse dispersivity (αL, αT) were set to 3.6 m and 0.36 m lengths, respectively (Perera et al., 2008; Shoemaker, 2004), and the molecular diffusion to 1.0 10^{-10} m^2/d. Fresh water density was set to 1000 g/L and salt water density to 1024.5 g/L. The fluid density at the reference concentration (DENSEREF) and the slope of the linear equation of state that relates fluid density to solute concentration (DRHODC or DENSESLP, from Langevin et al., 2003) are parameters needed to compute fluid density by the variable-density groundwater flow equation (VDF Process in SEAWAT routine). The fluid density in the simulation was calculated as a function of the specie TDS, in this case. A DENSEREF value of 1000 indicates that the reference fluid density (freshwater in this case) at 25ºC is 1000 g/L. A DRHODC value of 1.3 indicates that the density will linearly vary between 1000 g/L and 1024.5 g/L for freshwater and for saltwater respectively. DRHODC can be estimated by dividing the density difference by the concentration difference. In this case, it will be as follows:

(1024.5 g/L) - (1000 g/L) / (19.93 g/L) - (0.306 g/L) = 1.3

The calibration for salt concentration in steady state was assessed on the basis of observation data of salt concentration from 11 piezometers located at the modeling area boundary, of unconfined aquifer (4 wells) and semi-confined aquifer (7 wells) (Table 1). A reasonable match between the salt concentration results of data measured in field and in the model was achieved after a time simulation of 10 years. The result from 2008 transport model for TDS was assess based on observed data from 20 piezometers located on the unconfined aquifer (8 wells) and on the semi-confined aquifer (12 wells) (Table 1). The residual error between calculated and observed TDS was estimated. After the elapse time of 3650 days, the results from the transport model indicated a mean absolute residual error of 0.38 g/L, with standard deviation of 0.54 g/L and a root mean square residual (RMS) of 0.52 g/L. The correlation coefficient between observed and calculated concentrations is 0.99 (Fig. 2a). For 2015, the results from the transport model indicated a mean absolute residual error of 0.35 g/L, with standard deviation of 0.47 g/L and a root mean square residual (RMS) of 0.50 g/L. The correlation coefficient between observed and calculated concentrations is 0.99 (Fig. 2b).

In accordance with modeling results, an assessment of changes in groundwater salinity and in the variation of water flows within the wetland was made. The salt concentration resulting from natural condition (considered as relative zero salt concentration) was compare with the salt concentration resulting from 2008 simulation (with over exploitation and waterproofing) and from 2015 simulation (with over exploitation and increase in waterproofing). This approach allowed to calculate spatial TDS changes, due to variables as pumping and waterproofing related to the development of urban and agricultural areas, on a relative analysis.

4. Results

At regional scale, rain water infiltration is the recharge source of the unconfined aquifer throughout the study area, while the semi-confined is indirectly recharged by water infiltration from the unconfined aquifer. Groundwater flows and the hydrodynamic relation between the unconfined and semi-confined aquifers vary spatially as observed in the hydrogeological numerical models (Fig. 3 and 4).

Under natural conditions, regional groundwater flows from the basin header areas located in the southwest towards the coastal wetland and the Río de la Plata, in both semi-confined and unconfined aquifers (Fig. 3 a and b). Analysis of the hydraulic load differences between both aquifers (Fig. 3 c) shows that, in the watershed divide areas, the water level of the unconfined aquifer is higher than that of the semi-confined. In this way, the watershed divide sectors constitute preferential areas for semi-confined aquifer recharge. On the other hand, the water courses and the wetland constitute zones of discharge of the unconfined aquifer and in some of these sectors the semi-confined aquifer water levels are higher than in the unconfined aquifer. This evidences that in these sectors a flow of water from the semi-confined to the unconfined aquifer exists.

In the loess plain area, the groundwater has low salinity both in the unconfined and semi-confined aquifers, while the salinity of the groundwater in the wetland area is quite variable (Fig. 3 d and e). In the wetland, the unconfined aquifer presents areas with low water salinity (areas with higher rainwater recharge) and sectors with saline water (transit zones and groundwater discharge). On the other hand, in the semi-confined aquifer the water is saline, being present as a wedge-shape (Fig. 3 e and f).

The settlement of La Plata city and the urban growth generated around the city, produced modifications in the potential infiltration of the substrate as waterproofing due to buildings and pavement of the streets, that tend to decrease the recharge of aquifers in the sector of the urban area. Likewise, the intensive exploitation of the semi-confined aquifer causes a cone of depression that modifies the groundwater flows of both the semi-confined and unconfined aquifers, and also modifies the hydraulic relation between the aquifers. Besides that, the development of greenhouses, which generate waterproof areas, and the increase of pumping in preferential recharge zones of aquifers constitutes another modification to the natural hydrogeological functioning of the system.

The models generated in this work show how the intensive exploitation of the semi-confined aquifer generates a cone of depression that also affects the water levels of the unconfined aquifer (Fig. 4 a, b and c). Within the influence area of the depression cone, an inversion in the groundwater flow takes place causing the water to flow from the wetland towards the loess plain in the sector adjacent to the wetland. Groundwater flow outside the radius of the depression cone remains the same as in natural conditions, being the water from the unconfined aquifer a contribution to the wetland through groundwater discharge, to which the semi-confined aquifer discharges in some sectors. The development of greenhouses in the more continental sectors of the study area, decreases the recharge of the unconfined aquifer and when it is pumped, together with the semi-confined aquifer, depresses its water level, affecting the natural recharge of the semi-confined aquifer. Changes in the hydrodynamics in the intensive exploitation zone, cause an advance of the semi-confined aquifer salt wedge towards the more continental sectors (Fig. 4 d, e and f).

Salinity variation analysis between natural conditions and those modified by intensive exploitation shows the important changes generated in the groundwater salinity in the two aquifers (Fig. 5 a and b). In sectors affected predominantly by anthropic activities (central - northwest sector), the salinity both in the unconfined and semi-confined aquifers tends to increase, which is most evident in the semi-confined aquifer, in the sector close to the limit coastal plain - loess plain. Conversely, in mainly natural sectors (southeast), the increase in salinity is not noticeable.

5. Discussion

Coastal wetlands have been suffering from serious degradation, alteration or loss of ecosystem services due to intense anthropogenic activities (Lemly et al., 2000; Newton et al., 2012; Zhao et al., 2016), and thus, coastal wetlands are listed amongst the most heavily damaged of natural ecosystems worldwide (Barbier, 2011; Jones et al., 2018). Wetlands degradation associated with modifications in hydrological functioning is a problem that affects many coastal regions worldwide and it has accelerated in recent years not only owing to anthropogenic activity but also because of climate change (Perillo et al., 2009).

Hydrological changes and the effects they produce in wetlands can be easily studied with field detail studies at local level. But at the regional level, it is necessary to acquire models that allow not only to identify the problems but also to analyze their evolution over time. The results of such models should allow to obtain the prediction in changes of coastal wetlands to human impacts, so that the adaptation strategies can be put in place (Reyes, 2009).

Groundwater flows constitute one of the main water sources that sustain coastal wetlands, being the discharge of local and regional groundwater flows a significant hydrological component in coastal areas (Custodio, 2000). The results obtained with the regional hydrogeological modelling show how the intensive exploitation of aquifers in continental areas adjacent to the coastal wetland affect the groundwater flows toward the wetland, which is a feature that had already been evidenced in previous studies (Carol et al., 2013). However, the novelty of the model is that it shows that these modifications in the groundwater flows lead to changes in the groundwater salinity of both the unconfined and semi-confined aquifers in the wetland area. If salinity changes of the wetland sectors, which are affected and not affected by the intensive exploitation of groundwater, are compared, a contrasting evolution from the natural state is shown.

In the wetland sector, without intensive exploitation affectation, the salinity in both the unconfined and semi-confined aquifers tends not to increase or slightly increase (southeast sector in Fig. 5). This behaviour is what would be expected the wetland to expose in natural conditions over time. The groundwater flow of fresh water from the more continental sectors tends to displace the saline paleo-wedge towards the estuary of the Río de la Plata. This means that the salinity of groundwater in the semi-confined aquifer in the wetland area tends to decrease over time.

Conversely, in the sector where the groundwater flow is affected by intensive exploitation and the recharge decrease as consequence of waterproofing by greenhouses, the salinity varies. In this sector, the salinity does not show important variations in the unconfined aquifer, while in the semi-confined aquifer the salinity increases (central - northwest sector in Fig. 5). This salinization, although mainly affects the sector close to the limit coastal plain - loess plain, also causes a salinization of the groundwater in the wetland itself. This is due to the fact that the intensive pumping of the semi-confined aquifer from the continental sector reverses the groundwater flow, causing the intrusion of the saline paleo-wedge towards the continent. This causes a groundwater salinity increase in the semi-confined aquifer in the wetland sector adjacent to the exploitation zone.

The models generated show, as it has been exposed in other regions of the world, that intensive exploitation of groundwater is one of the main causes of wetland deterioration (Winter, 1988; Bernaldez et al., 1993; Suso and Llamas, 1993; Rochow, 1994; Serrano and Serrano, 1996; Wise et al., 2000; Cooper et al., 2003; Moiwo et al., 2010; Johansen et al., 2011; Carol et al., 2013; Cooper et al., 2015; Park et al., 2019). The modification in the underground flows, that support the wetlands, leads to a decrease in the areas of wetlands and / or to ecological effects in them (Bernaldez et al., 1993). This problematic may further be worsened in coastal wetlands due to the saline intrusion processes that characterize littoral areas. In these, intensive pumping in sectors adjacent to wetlands or within them, not only leads to modifications in water flows but also to their salinization. If we consider that salinity of groundwater is one of the main factors of the physical environment determining the environmental characteristics of the ecosystems (Watson and Byrne, 2009), it is expected that changes in wetland environments slowly begin to take place. Given that no significant alterations have been recorded in the wetland ecosystems today, the diagnosis generated from the modelling allows to alert the managers of the water resource. Until now, they have only evaluated the problems of intensive exploitation associated with water urban supply, without considering agricultural impacts on recharge and on groundwater exploitation, and the consequences of what this produces in adjacent wetland environments.

6. Conclusions

The regional-level hydrogeological model shows that the intensive exploitation of aquifers in areas adjacent to the wetland, produces variations both in the salinity and groundwater flows of the wetland. A decrease of water inputs from the regional groundwater discharge of the unconfined and semi-confined aquifers occurs as a result of the investment of the flow caused by intensive exploitation. These flow modifications lead to changes in the salinity of the groundwater within the wetland area. Flow variations showed that in the area affected by intensive exploitation, the unconfined aquifer salinity will tend to slightly vary, while in the semi-confined aquifer it will tend to increase, which is most evident in the sector close to the limit coastal plain - loess plain. These changes will modify the long-term environmental characteristics of the wetland, and hence their early identification is important for the generation of guidelines that tend to minimize these affectations.

Human activities and climate change have turned coastal wetlands into highly vulnerable environments. Wetlands monitoring and diagnosis used to mitigate these changes generally tends to evaluate the changes at the local level within the wetland area. The hydrogeological model at a regional level evidences how anthropic affectations taking place outside the wetland area can also affect its natural environment.

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8. Data Availability Statement

The data that support the findings of this study are available from the corresponding author upon reasonable request.

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Figure captions

Figure 1. Location of the study area and delimitation of wetland and loess plain sectors. Distribution of the pumping and observation wells, greenhouses and urban area.

Figure 2. Observed concentration vs. calculated concentrations in a) 2008 and b) 2015.

Figure 3. Hydrogeological model for natural conditions. Piezometric levels modelled for a) unconfined aquifer, b) semi-confined aquifer and c) hydraulic load differences profile between both aquifers. Salt concentration modelled for d) unconfined aquifer, e) semi-confined aquifer and f) profile showing the salinity distribution for both aquifers. The black line indicates the profile trace.

Figure 4. Hydrogeological model for intensive exploitation conditions. Piezometric levels modelled for a) unconfined aquifer, b) semi-confined aquifer and c) hydraulic load differences profile between both aquifers. Salt concentration modelled for d)

unconfined aquifer, e) semi-confined aquifer and f) profile showing the salinity distribution for both aquifers. The black line indicates the profile trace. In b and e, the pattern of yellow dots indicate greenhouses y pumping wells.

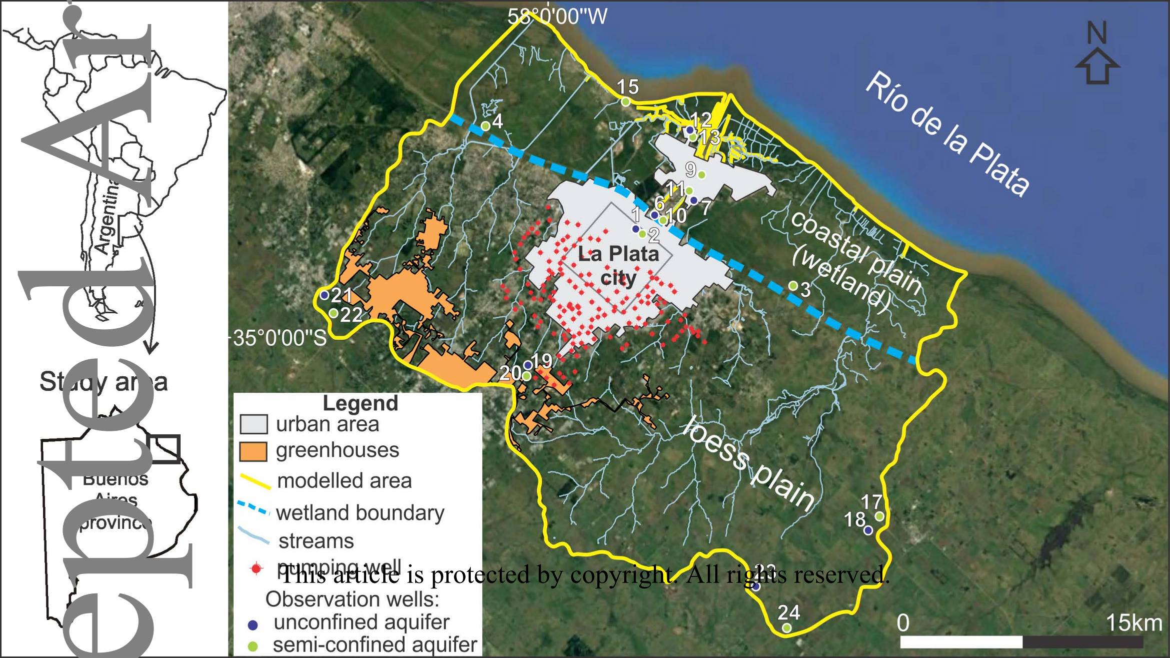
Figure 5. Salinity variation between natural condition and 2015 for a) unconfined aquifer and b) semi-confined aquifer.

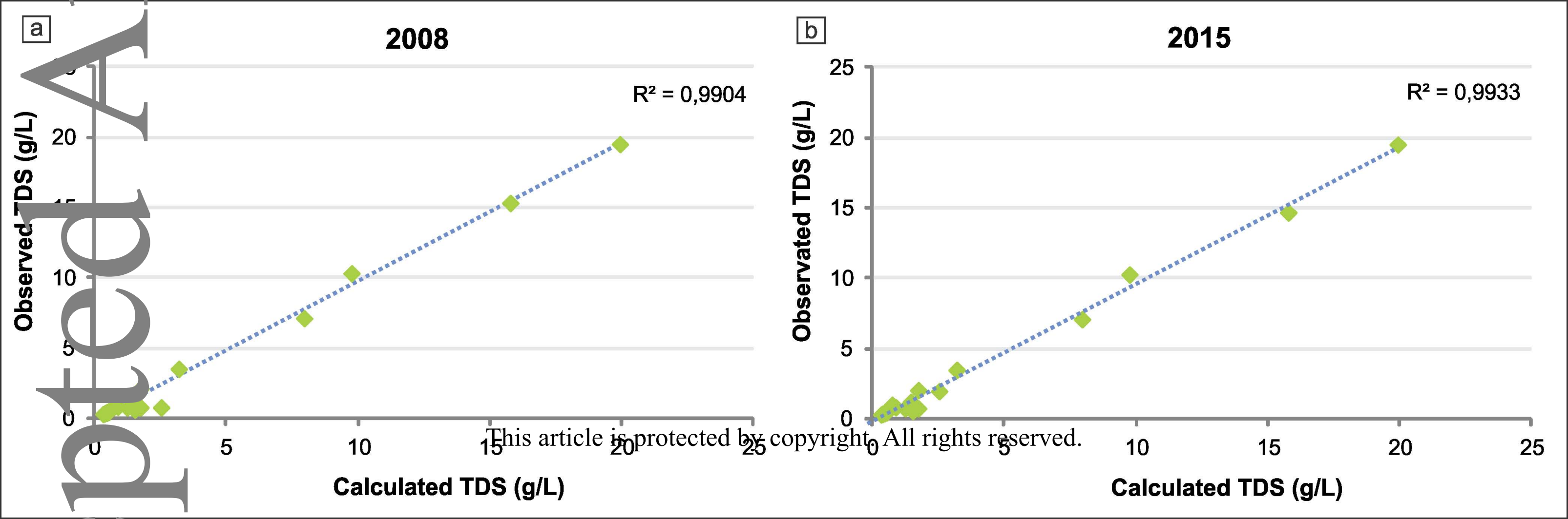
Table caption

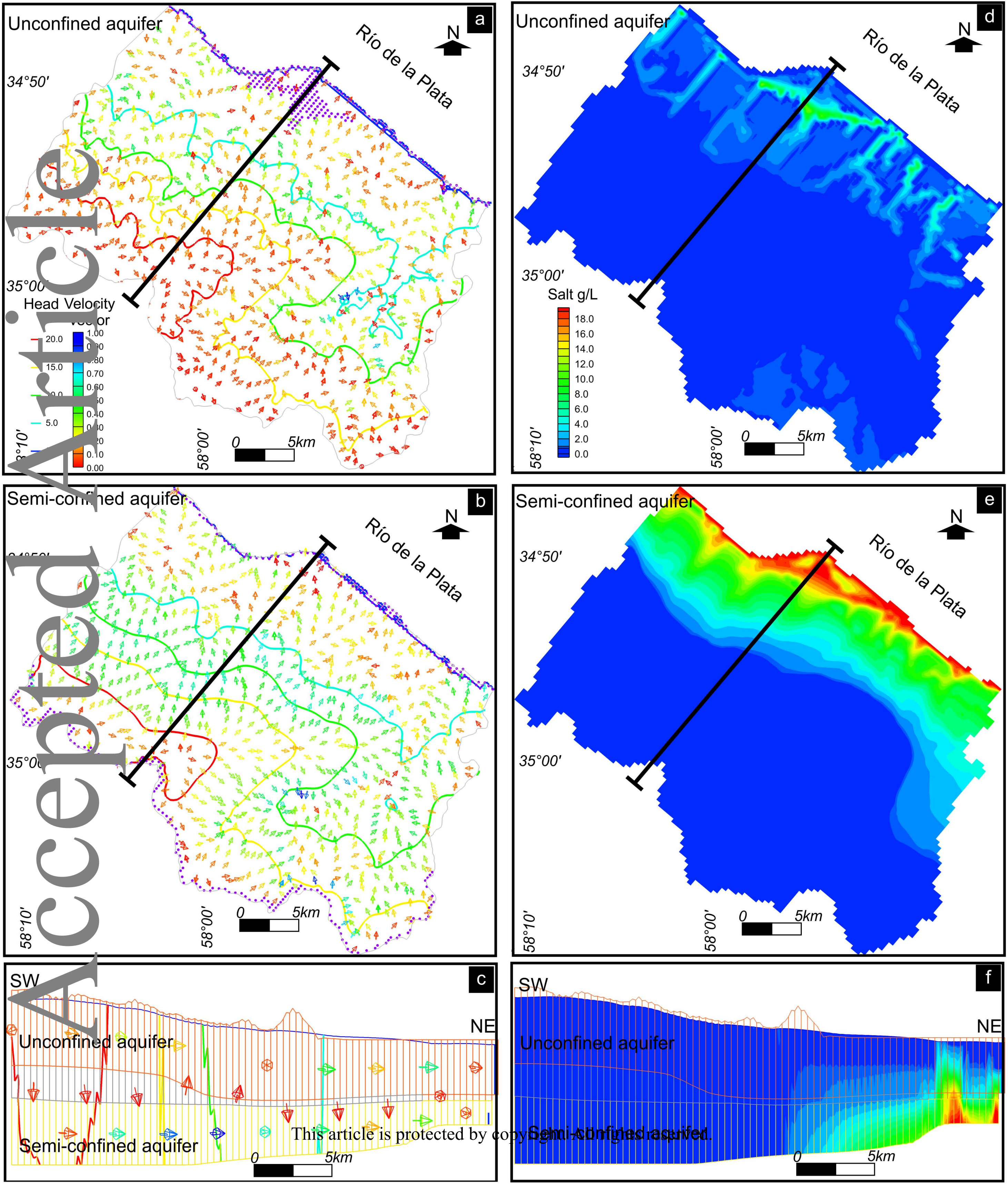
Table 1. Salt concentration in observation wells (g/L). UC: Unconfined aquifer. SC: Semiconfined aquifer.

Table 1. Salt concentration in observation wells (g/L). UC: Unconfined aquifer. SC: Semiconfined aquifer.

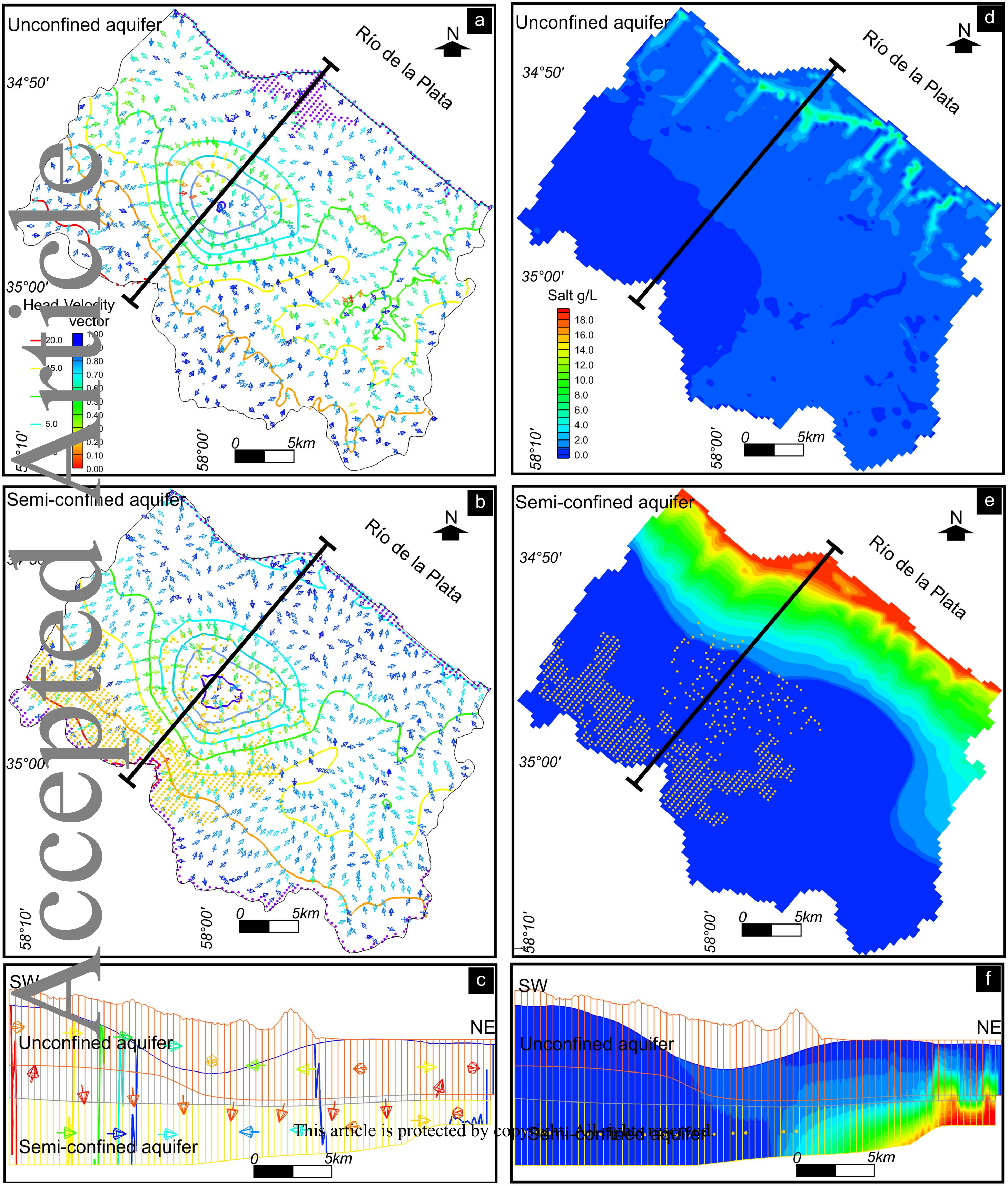
Well	Aquifer	Latitude S	Longitude W	Depht (m)	Observation wells Salt Concentration (g/L)
P1	UC	34°54'25.20"	57°55'57.60"	20	1.72
P6	UC	34°53'47.90"	57°55'44.90"	4.75	1.22
P7	UC	34°53'11.40"	57°54'22.70"	4.75	2.51
P12	UC	34°51'10.5"	57°53'59.80"	6	0.73
P18	UC	32°11'35.95"	61°3'25.53"	20-30	0.52
P19	UC	32°27'4.97"	60°58'22.86"	20-30	0.40
P21	UC	32°32'37.65"	60°54'47.79"	20-30	0.44
P23	UC	32°17'41.48"	61°6'24.07"	20-30	0.86
P2	SC	34°54'25.20"	57°55'57.60"	65	1.72
P3	SC	34°56'37.00"	57°50'21.00"	62	1.51
P4	SC	34°50'47.70"	58°03'2.90"	56	1.44
P9	SC	34°52'29.76"	57°53'52.08"	28	9.74
P10	SC	34°53'57.67"	57°55'27.35"	60	3.18
P11	SC	34°53'10.08"	57°54'24.67"	40	7.94
P13	SC	34°51'10.50"	57°53'59.80"	40	15.77
P15	SC	34°49'49.80"	57°56'55.10"	51	19.93
P17	SC	35°4'53.36"	57°44'48.20"	50	0.51
P20	SC	35°0'33.45"	58°3'39.69"	50	0.36
P22	SC	34°58'47.57"	58°10'18.47"	50	0.31
P24	SC	35°8'59.82"	57°49'21.07"	50	0.46


58°0'00"W
N
Río de la Plata
coastal plain (wetland)
loess plain
La Plata city
35°0'00"S
Argentina
Study area
Buenos Aires province
Legend
urban area
greenhouses
modelled area
wetland boundary
streams
pumping well
Observation wells:
unconfined aquifer
semi-confined aquifer
0
15km
1 2 3 4 6 7 9 10 11 12 13 15 17 18 19 20 21 22 23 24

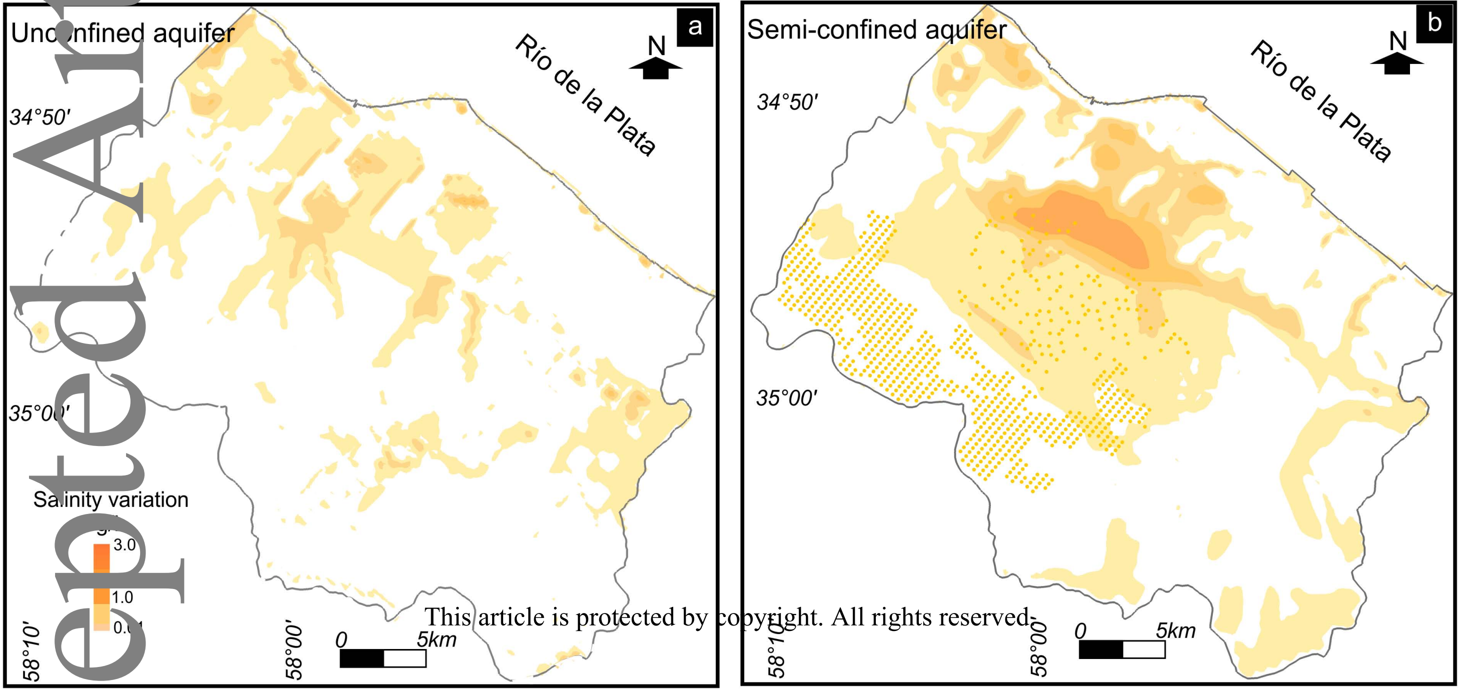

a
2008
R² = 0,9904
Observed TDS (g/L)
Calculated TDS (g/L)
b
2015
R² = 0,9933
Observated TDS (g/L)
Calculated TDS (g/L)



Unconfined aquifer
a
Río de la Plata
N
34°50'
35°00'
Head
Velocity vector
20.0
15.0
5.0
1.00
0.80
0.70
0.60
0.10
0.00
58°00'
0
5km
Unconfined aquifer
d
Río de la Plata
34°50'
35°00'
Salt g/L
18.0
16.0
14.0
12.0
10.0
8.0
6.0
4.0
2.0
0.0
58°10'
58°00'
Semi-confined aquifer
b
e
35°00'
SW
NE
c
f
Semi-confined aquifer



a
Unconfined aquifer
Río de la Plata
N
34°50'
35°00'
Head
Velocity vector
20.0
15.0
5.0
1.00
0.80
0.70
0.40
0.10
0.00
58°00'
0
5km
b
Semi-confined aquifer
Río de la Plata
N
35°00'
58°10'
58°00'
0
5km
c
SW
NE
Unconfined aquifer
Semi-confined aquifer
0
5km
d
Unconfined aquifer
Río de la Plata
N
34°50'
35°00'
Salt g/L
18.0
16.0
14.0
12.0
10.0
8.0
6.0
4.0
2.0
0.0
58°10'
58°00'
0
5km
e
Semi-confined aquifer
Río de la Plata
N
34°50'
35°00'
58°10'
58°00'
0
5km
f
SW
NE
Unconfined aquifer
Semi-confined aquifer
0
5km



Unconfined aquifer
a
Río de la Plata
N
34°50′
35°00′
Salinity variation
3.0
1.0
58°10′
58°00′
0
5km
Semi-confined aquifer
b
Río de la Plata
N
34°50′
35°00′
58°10′
58°00′
0
5km