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Long term monitoring of aquifer salinization processes in a physical analog model

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Abstract: A long-term experiment on an unconfined shallow aquifer physical analog model was performed to study the salinization processes in finely graded sandy-silty sediments. The unconfined shallow aquifer was reconstructed within a laboratory large tank (4x8x1.4 m) equipped with 26 standard piezometers. The sediments (~35 m^3) pertaining to a crevasse splay deposit of the paleo-Po river system were sourced in March 2004 from an unconfined alluvial aquifer near Ferrara (Northern Italy). In July 2013, 600 l of NaCl saline solution (35 g/l) was injected in the aquifer physical analog model and the fate and transport of the saline solution was monitored since nowadays. The system was subjected to high evaporation rates, which caused the upper soil salinization in presence of saline groundwater. The movement of the saline solution was monitored in time and space with both fully screened piezometers and high resolution multi-level coring (every 5 cm). The results highlighted how the saline front migrated within the aquifer physical model in response to different boundary conditions: lateral input of water from irrigation canals, sprinkler irrigation and evaporation, all in controlled conditions. This study aims to elucidate the processes of soil salinization in reclaimed agricultural lands characterized by highly saline groundwater.

Key words: soil salinization; saline groundwater; evaporation; low permeability

1. INTRODUCTION

In the following decades, the increase of surface water resources losses as a result of widespread pollution and of the increasing water demand for economic development, will drive a progressive stress on shallow groundwater bodies, especially in coastal areas (Kundzewicz and Döll 2009). Like various low-lying coastal areas around the Europe (Feseker 2007; Oude Essink et al. 2010), the Po river Delta shallow coastal aquifer was found to be affected by autonomous salinization caused by the upward flux of saline and hypersaline groundwater (Colombani et al. 2016a). Thus, soil salinization mechanisms have to be rigorously quantified using controlled conditions, by using long term laboratory apparatus. Large tank experiments can be extremely useful to infer long term changes in groundwater and aquifers matrix (Werner et al. 2016). This can be achieved accelerating the groundwater flow conditions as typically can be done in column experiments, but using standard monitoring techniques like standard monitoring wells and accounting for soil heterogeneities (Lee et al. 2008; Mastrocicco et al. 2011). In this study, a three years long experiment was performed in a physical analog model to quantify the foreseeable impacts of climate change on the unconfined aquifer of the Po Delta. This was achieved modifying the boundary conditions and monitoring piezometric heads and salinity using multi-level sampling (MLS) devices.

2. MATERIALS AND METHODS

The experiment was performed in a physical analog model (4 m wide by 8 m long and 1.4 m deep) located in the Hydrogeology Laboratory at the Scientific and Technological Pole of the University of Ferrara. The physical analog model was filled with 42 m^3 of unconsolidated material

(35 m^3 of natural sediments and 7 m^3 of gravel), by means of a bulldozer equipped with a 2-ton tilting shovel mounted on a 8-m-long telescopic crane.

An external reservoir (constant head) was connected to the tank via three inflow pipes. Twenty-six piezometers were installed using a hand driven auger, on the base of a semi-regular monitoring grid. A complete description of the tank set-up can be found in Mastrocicco et al. (2011). Groundwater levels were measured using an electric contact phreatimeter and salinity was measured converting electrical conductivity and temperature collected data. Core samples were collected several times along the maximum salinity gradient of the physical analog model. Core samples were taken every 5 cm by mean of a micro auger equipment. Soil water content was measured gravimetrically and pore water salinity was measured extracting with ultrapure water the water-soluble fraction from the sediment samples, using a sediment to water weight ratio of 1:5.

The experiment started in July 2013, here an input of 600 L of saline water (35 g/l) was provided from the constant head at the end of the model. Then, the boundary conditions were left unaltered for 8 months (until February 2014) to monitor the salinization of the physical analog model. From February 2014 to February 2016 the physical analog model was no more supplied with water and left under bare evaporation conditions. From February 2016 a freshwater (0.35 g/l) constant head was provided. Finally, from May 2016 to June 2016 three different irrigation rates were applied (1, 5 and 10 mm/h for 24 hours each) on the physical analog model ground surface and then monitored until October 2016.

3. RESULTS AND DISCUSSION

The long term experiment here described tried to resemble the possible scenario of a portion of the unconfined aquifer, which could occasionally be salinized by irrigation canals affected by seawater ingression as reported by different authors as a mean of aquifer salinization (Feseker 2007; Oude Essink et al. 2010; Colombani et al. 2016a). The movement of the saline wedge was extremely slow due to the low hydraulic conductivity of the sediments. In fact, after 6 months from the injection, the saline wedge had just travelled a maximum distance of 2 m from the injection boundary (Figure 1). The contour maps of Figure 1 show that the saline wedge moved preferentially in zone characterized by a higher hydraulic conductivity values. In addition, saline front did not behave as sharp interface, but was rather diluted, indicating a relatively large dispersion coefficient both in the longitudinal and transversal direction. While the vertical dispersivity seems less pronounced. This behaviour was due to the intra-well artificial mixing that led to a biased saline distribution within the monitoring wells (Britt 2005; Colombani et al. 2016b). In fact, Figure 1 shows the vertical salinity distribution observed in the monitoring wells and it can be noticed that within the saltwater wedge the vertical salinity distribution was quite different between the open borehole logs and the pore water.

Another aspect that is appreciable only employing MLS, is the top soil salinization. In the right chart of Figure 2, is apparent an accumulation of salts in the top soil even in absence of saline groundwater, this was due to continuous evaporation since 2006, when the physical analog model was assembled and continuously flushed with freshwater (Mastrocicco et al. 2011).

Moreover, that changing the boundary conditions increased the top soil salinity in time due to evaporation processes combined with high capillary rise of the sediments. A high top soil salinity was found at the end of the desiccation period (February 2016) in the zones reached by saline groundwater. Salinity higher than 15 g/L was found in surficial samples collected below Y coordinate of 1.5 m (see Figure 1). Consequently, when irrigation was supplied a sudden salinity increase was observed in all the monitoring wells up to Y coordinate of 2.0 m. However, given that the physical analog model was almost completely desiccated, the first two irrigation events did not produce any groundwater head increase. On the contrary, the third irrigation event produced a sharp rise in the water table, but since the irrigation rate was excessive, surface run-off and ponding occurred, spreading the saline water in an area larger than the one affected by the saltwater wedge.

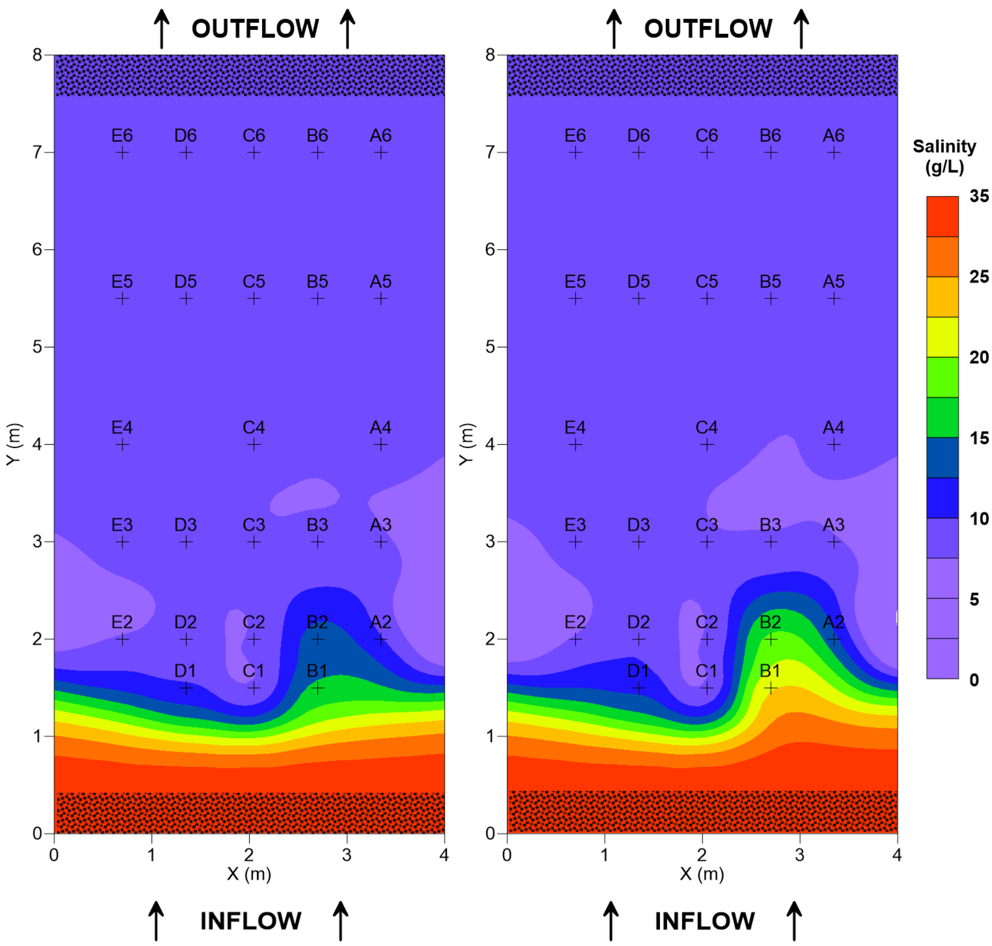

Figure 1. Upper left panel: groundwater salinity contour map at the water table in February 2014. Upper right panel: groundwater salinity contour map at the model bottom in February 2014.

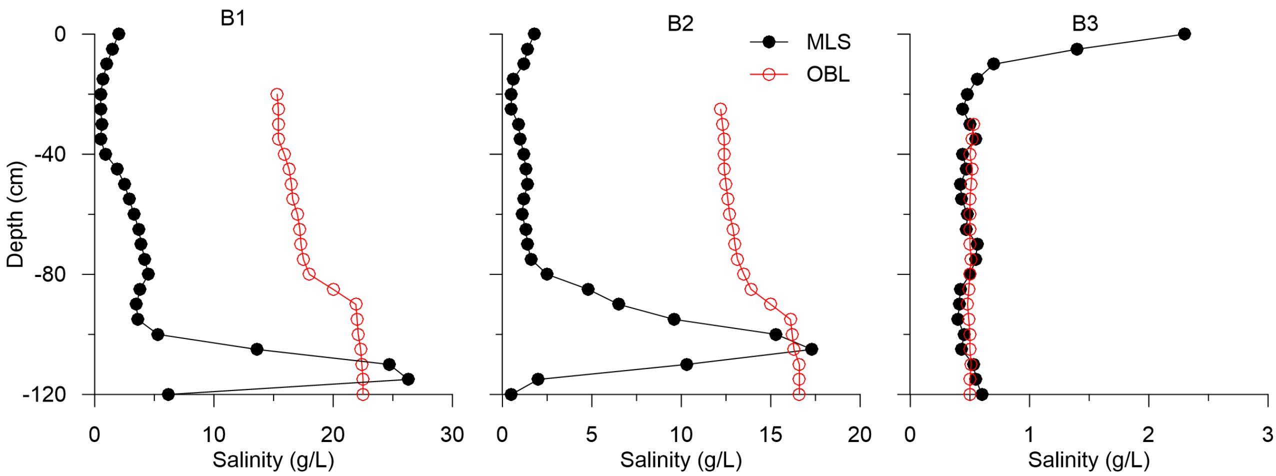

Figure 2. Vertical pore water salinity distribution (MLS) and open borehole logs (OBL) of three selected points in February 2014.

More information is given in Table 1, where the salinity of monitoring wells is reported for each irrigation event. As can be seen from table 1, the last irrigation event produced a salinity increase respect to the background value of 0.3 g/L. In fact, a salinity increase of 1-2 g/L in many observation wells even beyond the Y coordinate of 4.0 m. On the contrary, since the inflow boundary was set up with freshwater from February 2016, the salinity in the monitoring wells near to the inflow decreased nearly to background values (Table 1).

Table 1. Salinity in monitoring wells at the end of the monitoring period (10/10/2016).

i.d.	**Salinity (g/L)**	i.d.	**Salinity (g/L)**
A2	0.76	C4	1.02
A3	1.32	C5	3.64
A4	7.00	C6	2.36
A5	3.19	D1	2.80
A6	1.43	D2	1.17
B1	0.59	D3	0.85
B2	0.97	D5	1.19
B3	1.11	D6	1.71
B5	2.97	E2	3.74
B6	3.82	E3	5.36
C1	0.81	E4	0.92
C2	1.02	E5	3.71
C3	2.76	E6	2.16

Finally, given that sprinkler irrigation often produce surface run-off and ponding (Gillies and Smith 2005), this mechanism could be a mean to spread salt contamination in large areas at the border of the saltwater wedge or near salinized water canals and rivers (Da Lio et al. 2015).

4. CONCLUSIONS

A three years long experiment was performed in an aquifer physical analog model to study the lateral and vertical migration of saline groundwater. Vertical and lateral saltwater migration was affected by heterogeneities of sediment grain size, creating preferential pathways and stagnant zones. The monitoring wells did not captured the vertical distribution of salinity due to artificial mixing within the monitoring wells and this can be a source of bias when attempting to build up a robust conceptual model of saltwater intrusion. The top soil salinization increased in time due to evaporation processes combined with high capillary rise of the sediments. The irrigation practices like sprinklers can eventually increase the areas of salt contamination if the top soil salinity is carried along the agricultural fields via surface run-off.

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