Inferring the interconnections between surface water bodies, tile-drains and an unconfined aquifer–aquitard system: A case study

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Inferring the interconnections between surface water bodies, tile-drains and an unconfined aquifer-aquitard system via flow and transport modelling

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ABSTRACT

Shallow lenses in reclaimed coastal areas are precious sources of freshwater for crop development, but their seasonal behaviour is seldom known in tile-drained fields. In this study, field monitoring and numerical modelling provide a robust conceptual model of these complex environments. Crop and meteorological data are used to implement an unsaturated flow model to reconstruct daily recharge. Groundwater fluxes and salinity, water table elevation, tile-drains' discharge and salinity are used to calibrate a 2D density-dependent numerical model to quantify non-reactive solute transport within the aquifer-aquitard system. Results suggest that lateral fluxes in low hydraulic conductivity sediments are limited, while water table fluctuation is significant. The use of depth-integrated monitoring to calibrate the model results in poor efficiency, while multi-level soil profiles are crucial to define the mixing zone between fresh and brackish groundwater. Measured fluxes and chloride concentrations from tile-drains not fully compare with calculated ones due to preferential flow through cracks.

Keywords: modelling, groundwater, salinization, monitoring, lowland landscape, drainage.

1. INTRODUCTION

In reclaimed coastal agricultural land, the presence of saline and brackish groundwater is the rule rather than the exception. Many examples are found in The Netherlands (Raats, 2015), Belgium (Vandenbohede et al., 2010), Italy (Da Lio et al., 2015), Australia (Kobryn et al., 2015), USA (McMahon et al., 2015) and China (Sun et al., 2012), just to cite the most recent. Tile-drains in reclaimed agricultural soils are often used as a mean to limit soil salinity in the root zone (Ali et al., 2000; Ayars et al., 2006), leading to complex groundwater flow patterns that can add significant uncertainty to the prediction of local-scale flow and transport (Gooday et al., 2008). In this framework, the numerical modelling of flow and transport processes has undergone some significant advances in the last years, with the general recognition of tile-drains as an important feature linking surface waters to groundwater (De Schepper et al., 2015; Warsta et al., 2013). Despite of these general advances, there are many aspects that still need to be clearly understood both at the site's scale and at the watershed scale. In general, there is little agreement on how to deal with the input of a specific yield parameter to simulate water table fluctuation (Acharya et al., 2012), how to model the flow resistance toward tile-drains (Hansen et al., 2013; Kohler et al., 2001), or how to model evapotranspiration fluxes from the saturated zone (Adeloye et al., 2012; Camporese et al., 2015). The literature paid more attention on modelling nutrient leaching through tile-drains, since they usually

provide a preferential pathway for eutrophication of surface freshwater resources and heavy metal export (Rozemeijer et al., 2010); while only recently field based numerical models have been implemented including variable density flow and transport features, or drains and recharge fluxes from the vadose zone (Colombani et al., 2015; De Louw et al., 2013). Undoubtedly, without a clear and robust conceptual model of the water fluxes in complex environments it is meaningless to build up a mass balance of reactive species like nutrients or heavy metals.
Thus, the aim of this study is to quantify the variable density flow and transport patterns within an unconfined aquifer-aquitard system affected by brackish groundwater and located beneath a tile-drained agricultural field. The data consist of precipitation and evapotranspiration, groundwater levels, groundwater and soil water salinities and chloride concentrations, tile-drains' discharge and salinity. A set of nested numerical flow and transport models are used to support the development of a robust conceptual model concerning the seasonal behaviour of tile-drains and to give some insight on the capabilities/limitations of the actual monitoring techniques.

2. MATERIALS AND METHODS

2.1. The study area

The experimental site is located in the Po River lowland, Northern Italy (44°50'33'' N and 12°05'40'' E), on the boundary of two different soil facies: the Thionic Fluvisols and the Humic Thionic Fluvisols, following the W.R.B. classification from F.A.O. (2014).
The area is characterized by an interplay of deltaic and littoral deposits (Stefani and Vincenzi, 2005), with a diachronous splay of Po distributary channels bounded to the North by Alps sourced paleo-channels of the Adige River and to the South by Apennines derived paleo-rivers, associated with inland delta deposits reclaimed from 1860 to the present day (Di Giuseppe et al., 2014). The crop type is a rotation of rain fed winter cereals and maize. Tile-drains (fig. 1) are employed to prevent water-logging conditions during winter time and for sub-irrigation purposes during prolonged dry periods. This field has been selected and studied within the ZeoLIFE project - Water pollution reduction and water saving using a natural zeolitite cycle (LIFE+10 ENV/IT/00321).

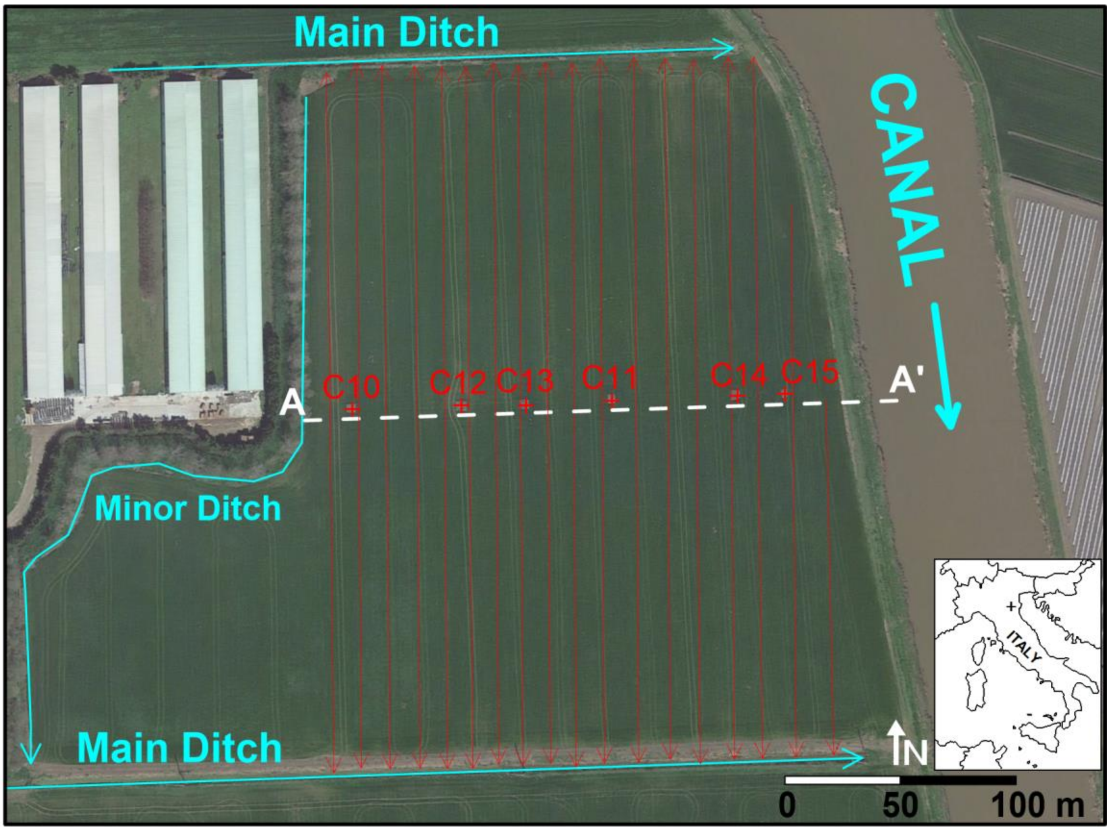

Figure 1: Field site location with the main hydraulic features highlighted: tile-drains (red lines), ditches (cyan lines), monitoring wells (red crosses) and the model transect A-A' (dashed white line).

2.2. Field site characterization and analytical procedure

The soil, consisting of recent interfluvial silty-clay deposits (Mastrocicco et al., 2013), was investigated from the top layer to -4 m below ground level, whose elevation varies from -2.75 to -3.30 m above sea level (a.s.l.). Six 1" PVC monitoring wells, 4 m long and screened in the last meter, were installed along a transect (fig. 1). Core samples were collected several times from October 2011 to September 2014 in proximity of the six monitoring wells; samples were taken every 30-50 cm by mean of an auger equipment, stored in a cool box at 4 °C and immediately transported in the laboratory for sedimentological and chemical analysis. Particle size curves of 50 soil samples were gained using a sedimentation balance for the coarse fraction and an X-ray diffraction sedigraph for the fine fraction; the two particle size curves were linked using a spreadsheet. Samples were grouped into two different types: an upper tilled horizon and a lower undisturbed horizon (Table 1).
The dry bulk density and the water content were determined gravimetrically. The gravimetric water content was measured at saturated condition. The residual water content was measured gravimetrically in triplicates on air dried sediments after heating for 24 hours at 105 °C. The organic matter content was determined by dry combustion (Tiessen and Moir, 1993).
To estimate the hydraulic conductivity (k_s) distribution along the vadose zone profile both pedotransfer functions and field measurements via a Guelph Permeameter were performed (Mastrocicco et al., 2013). To obtain k_s in the saturated zone, slug tests were performed in some monitoring wells (fig. 1). From the k_s characterization a clear decreasing trend is found in the first 3 m of soil, here designated as unconfined aquifer; while a nearly constant k_s value is found from approximately -5 m a.s.l., here designated as aquitard. The k_s decreasing trend in the unconfined aquifer is most probably due to sediments compaction rather than changes in sediments' grain size distribution.

Table 1. Sediment characteristics and their standard deviation.

Parameter	Upper Horizon (-3 to -3.4 m a.s.l.)	Lower Horizon (-3.4 to -4.0 m a.s.l.)
Grain size (%)		
Coarse sand (630-2000 μm)	0.00±0.0	0.00±0.0
Medium Sand (200-630 μm)	0.6±0.1	0.0±0.0
Fine Sand (63-200 μm)	7.4±0.3	1.5±1.4
Silt (2-63 μm)	49.2±3.1	62.4±4.2
Clay (< 2 μm)	42.0±3.4	36.1±2. 9
Organic matter (%)	8.1±1.5	8.9±2.4
Hydraulic conductivity (m/day)	1.7±2.4	1.1±0.3
Bulk density (Kg/m^3)	1.15±0.05	1.45±0.1
Saturated water content (%)	58.5±0.6	39.3±0.4
Residual water content (%)	13.0±0.2	14.8±0. 3

Chloride (Cl^-) was determined both in pore-water and groundwater. Ultrapure water was used to extract water-soluble Cl^- from the sediment samples, using a sediment to water weight ratio of 1:5. The sediment and the water were mixed and sealed in bakers, shaken for 1 h, and centrifuged for 1 h at 25 °C to separate the sediment from the solution. Pore-water samples were filtered through 0.22 μm Dionex polypropylene filters, prior to be analysed for anions. Groundwater samples were collected in each monitoring well via a low flow purging and sampling technique. The collected groundwater samples were filtered through 0.22 μm Dionex polypropylene filters, stored in a cool box at 4 °C and analysed in the laboratory. Major anions in groundwater and soil were analysed using an isocratic dual pump ion chromatography. Quality Control (QC) samples were run every 10 samples and the standard deviation for all QC samples run was better than 4%.

2.3. Numerical modelling and tracer selection

Along the selected transect A-A', perpendicular to the canal (fig. 1), a non-reactive transport model is used to investigate the freshening and salinization processes taking place within the unconfined aquifer-aquitard system. To compute the recharge from the unsaturated zone, the finite element model Hydrus-1D (Šimunek et al., 2008) is employed. The groundwater flow simulation is carried out using SEAWAT 4.0 (Langevin et al., 2007) to simulate the interaction between the brackish groundwater and the freshwater canal, taking into account the effect of variable-density transport processes. For the solute transport, no chemical reactions are considered since this study was planned to investigate physical control on conservative chemical transport.

Cl^- is selected as environmental tracer, since in this aquifer-aquitard system the correlation between the electrical conductivity (EC) and the total dissolved solids (TDS) in groundwater is very weak (fig. 2), thus excluding the possibility of using classical relationships between EC and TDS. Also Cl^- and EC compare rather poorly, since the aquifer-aquitard system is dominated by seasonal pyrite oxidation into sulphate (SO_4^{2-}) within the water table fluctuation zone. This mechanism produces an inverse relationship between Cl^- and SO_4^{2-} (fig. 2). In fact, when oxic recharge water (poor in Cl^-) percolates down through the aquifer-aquitard system it triggers pyrite oxidation with the consequent release of large amount of SO_4^{2-}. Besides, fertilizers input can contribute to enrich the recharge water in both Cl^- and SO_4^{2-}, complicating the observed trend (Giambastiani et al., 2015). Groundwater Cl^- concentrations match well with TDS, thus Cl^- is used as a proxy of groundwater salinity and is simulated to track the seasonal vertical variation of the fresh/brackish water interface. Groundwater heads and Cl^- in six monitoring wells (fig. 1) are used for model calibration. Cl^- profiles in soil pore-water and Cl^- concentrations within the tile-drains are used for model validation.

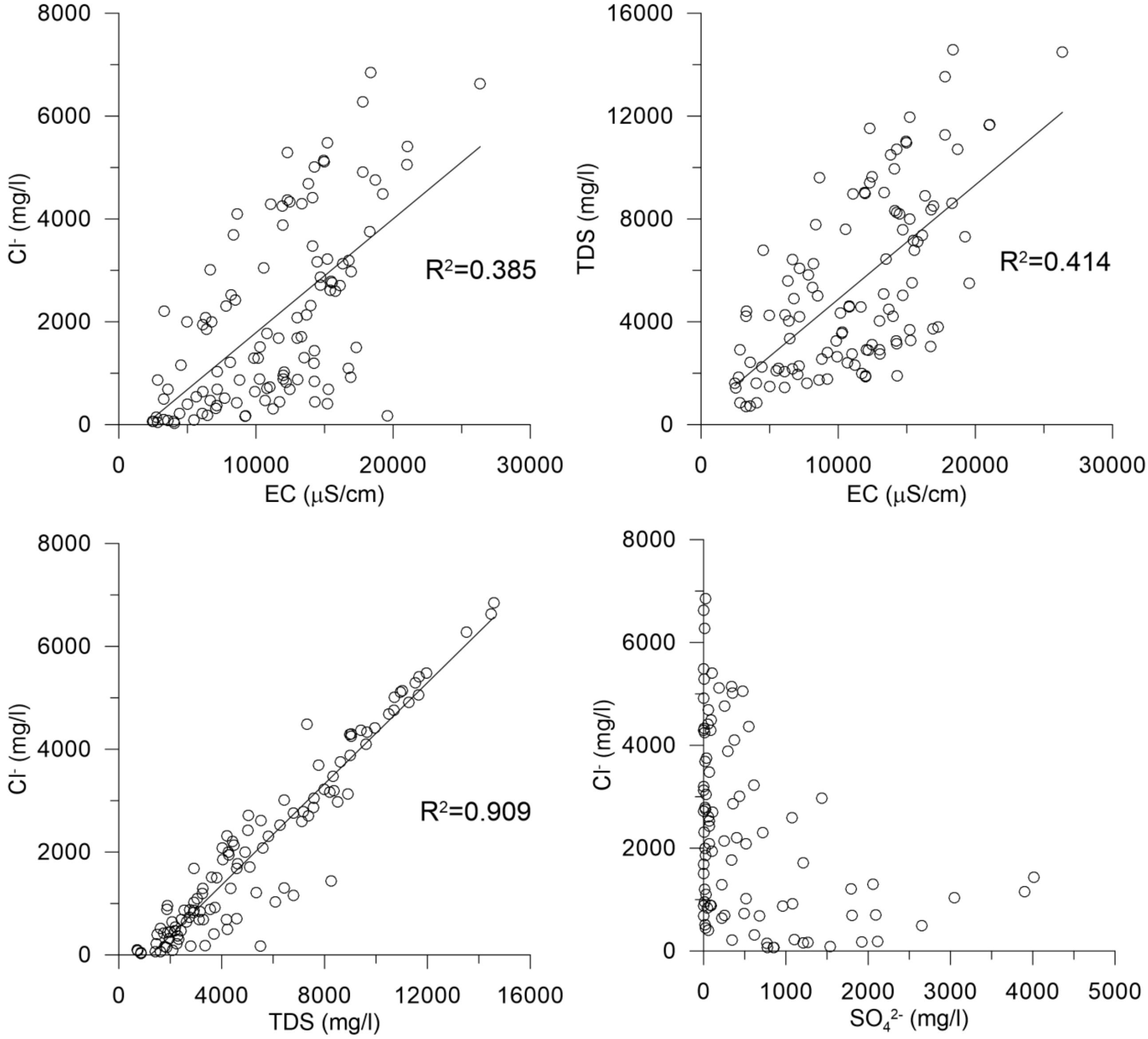

Figure 2: Scatter plots of Cl^- versus EC (upper left), TDS versus EC (upper right), Cl^- versus TDS (lower left) and Cl^- versus SO_4^{2-} (lower right). The first three plots are provided with linear regression lines.

2.3.1. Unsaturated flow model setup

The daily recharge is calculated using the finite element model Hydrus-1D, the model domain is discretized in 101 nodes of 1 cm each, to form a regular vertical grid of 1 m. The soil hydraulic parameters are assigned using the values found by Colombani et al. (2014) (Table 2). The initial water content is assumed to be constant throughout the model profile. At the soil surface, variable flux and head conditions are selected to represent the atmospheric boundary. Runoff is assumed to be negligible due to the flat topography and based on other studies in this area (Mastrocicco et al., 2013). Daily reference evapotranspiration (ET_0) is calculated with the FAO-56 recommended Penman-Monteith equation (Allen et al., 1998). A meteorological station located 0.5 km from the site, recording daily rainfall, wind speed, temperature, solar radiation and humidity, is used to derive input data for the Penman-Monteith equation.

The calculated ET_0 is then multiplied for the appropriate crop coefficient for maize and sorghum as defined by Allen et al. (1998). Potential transpiration and evaporation are split using an estimated surface cover fraction of 0.8 (Ritchie, 1972).
The actual root water uptake is simulated using the maize and sorghum dimensionless water stress response function available in the HYDRUS-1D database (Feddes et al., 1978), a variable height depending on the phenological state and a maximum rooting depth of 0.6 m, defined by visual inspection during core sampling in pre-harvest campaigns, are employed in the calculation.

Table 2: Hydraulic parameters used in the HYDRUS-1D model for the two soil horizons.

Parameter	Upper Horizon (-3 to -3.4 m a.s.l.)	Lower Horizon (-3.4 to -4.0 m a.s.l.)
K_s (cm/d)	1.7	1.1
θ_s (cm^3/cm^3)	0.59	0.39
θ_r (cm^3/cm^3)	0.13	0.15
α (1/m)	2.5*	2.5*
n (-)	1.2*	1.2*
ω (1/d)	0.47*	0.49*
α_2 (1/m)	2.11*	2.05*
n_2 (-)	1.58*	1.49*

* from Colombani et al. (2014).

The dimensionless constant for the radiation extinction by the canopy is set to 0.463, the interception constant is set to 0.25 mm and the leaf area index is derived from crop height. Free drainage is selected as lower boundary condition and the cumulative bottom flux is expressed as negative flux (recharge towards the aquifer-aquitard system). The daily recharge is subsequently discretized in the 25 stress periods used in the SEAWAT model (fig. 3).

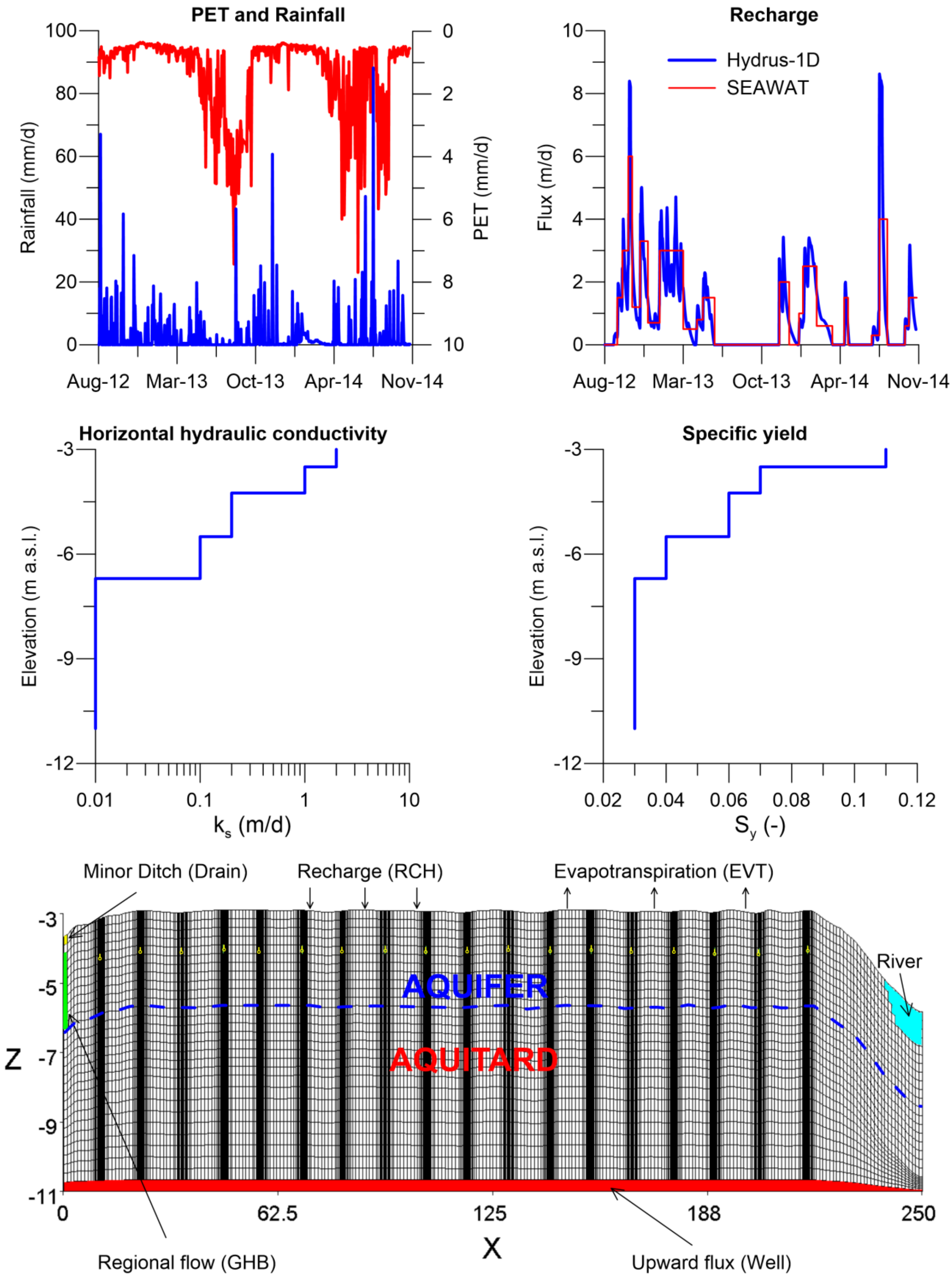

Figure 3: Temporal trends of precipitation and potential evapotranspiration (upper left plot) used to calculate the recharge in HYDRUS-1D and SEAWAT (upper right plot). Horizontal hydraulic conductivity (middle left plot) and specific yield (middle right plot) discretization with depth. Grid discretization of the model domain and boundary conditions applied (lower panel); the tile-drain cells (yellow symbols) are activated as Drain or GHB depending on the stress period.

2.3.2. Saturated flow and transport model setup

The domain of the cross-sectional model consists of 555 columns and 20 layers with variable width and thickness to allow the exact reconstruction of the tile-drains (fig. 3).
The vertical k_s is assumed to be 5 times lower than the horizontal k_s for all layers, following De Louw et al. (2013), while the effective porosity is taken as 0.18 for the entire model, based on the column experiments on the same soil discussed in Colombani et al. (2014). The specific yield (S_y) for silty-clay sediments is estimated using the approach proposed by Acharya et al. (2012). Specific storage is taken from literature values (Fetter, 2001). For a conservative solute as Cl^-, the molecular diffusion coefficient in porous media is taken equal to $1x10^{-4}$ m^2/d.
Initially a steady state simulation is run to achieve a representative distribution of heads and Cl^- concentrations within the aquifer-aquitard system; the Cl^- profiles collected in summer 2012 near the monitoring wells are used as initial concentrations. In the transient simulation, time is subdivided into 25 stress periods to simulate the variation of the recharge, the regional groundwater flow and the canal stage. The irrigation canal, on the right side of the model, is modelled by the River Package (fig. 3). The Drain Package is used to simulate the effect of the Minor Ditch (left side of the model profile), which removes groundwater from the aquifer-aquitard system at a rate proportional to the head difference between the aquifer-aquitard system and the ditch. When the hydraulic head in the aquifer-aquitard system is greater than the ditch elevation, groundwater flows into the ditch and is removed from the model domain. Recharge from the Minor Ditch to the underlying aquifer-aquitard system is always zero, regardless of the hydraulic head in the aquifer-aquitard system. The ditch elevation is set at -3.9 m a.s.l. and a conductance of 10 m/d is assigned. The Drain Package is also used to simulate the tile-drains when the surrounding ditches have a stage lower than the tile-drains elevation (fig. 1); tile-drains elevation is set at -4.1 m a.s.l. and a conductance of 1 m/d is assigned. To account for freshwater input in the aquifer-aquitard system from the surrounding ditches, the drain cells are substituted with General Head Boundary (GHB) cells when the stage in the ditches is higher than the tile-drains elevation. The Well Package is used to simulate the upward flux induced by the reclamation system (Giambastiani et al., 2013) and the specific discharge rate is calculated using the approach of de Louw et al. (2013). The Recharge Package is used to simulate the effective precipitation averaging the daily recharge values calculated by Hydrus-1D in the 25 stress periods used in SEAWAT 4.0 (upper right panel of fig. 3). The average Cl^- concentration in the pore-water of the top soil samples is used as Cl^- concentration of the recharge water for each stress period (Mastrocicco et al., 2015). This procedure allows to reproduce the real Cl^- concentration input from recharge, given that during the two years of simulation each plot has been treated with different amounts of pig slurry manure and urea, which have released pulses of Cl^- within the aquifer-aquitard system.
The evapotranspiration is simulated with the Evapotranspiration Package, the maximum evapotranspiration rate during the growing seasons is set to 5 mm with an extinction depth of -6 m a.s.l. for maize, and to 5 mm with an extinction depth of -4 m a.s.l. for sorghum. The maximum Cl^- concentration transpirable is set to 2000 mg/l.

2.3.3 Inverse modelling and model performance evaluation

Inverse modelling via the numerical code PEST (Doherty, 2010) is performed to estimate the following parameters: horizontal k_s, drain conductance, S_y, longitudinal and vertical dispersivities. The specific storage for the transient simulation was judged insensitive in a preliminary sensitivity analysis and thus not included in the model calibration. Prior information on the ratio between horizontal and vertical k_s values and on the measured horizontal k_s values along the profile, are used in order to constrain parameters estimation. Upper and lower limits of ±200%, with respect to the measured horizontal k_s values, are set as optimization constraints. Upper and lower limits of one order of magnitude, with respect to the initial estimate of longitudinal and vertical dispersivity values, are set as optimization constraints. The objective function consists of piezometric heads, tile-drains fluxes and Cl^- concentrations in both groundwater and tile-drains. Each screen interval is subdivided according to the layer proportion to ensure that calculated heads are related to the observed heads. The horizontal k_s values are log-transformed, while the other parameters are left linear for the inverse modelling.

The calibration of the transient state model is performed with the objective of matching modelled with measured hydraulic heads and Cl^- concentrations, recorded in the monitoring wells and in the tile-drains. To accurately verify the model's parameters consistency and robustness, the validation step is performed using an independent dataset (Oreskes et al., 1994). The independent dataset consists of Cl^- pore-water profiles recorded in May 2013, which are inherently different from the groundwater Cl^- concentrations since they are collected using different sampling techniques and different points in space. In fact, groundwater Cl^- concentrations are collected using a submersible pump in each monitoring well, while Cl^- pore-water profiles are collected using an auger sampler near to each monitoring well location. The model calibration and validation are assessed by calculating and comparing the value of the mean absolute error (MAE), percent of bias (PBIAS) and Wilcox index of agreement (d) for groundwater heads, fluxes in tile-drains, Cl^- concentrations in groundwater and Cl^- pore-water profiles. PBIAS values are calculated as (Gupta et al., 1999):

$$PBIAS = \frac{\sum_{j=1}^{n}\left(Obs_j - Calc_j\right)\times 100}{\sum_{j=1}^{n}\left(Obs_j\right)} \qquad (1)$$

where $Calc_j$ is the calculated value corresponding to the observed Obs_j. The optimal value of PBIAS is 0.0, with low-magnitude values indicating accurate model simulation. Positive values indicate model underestimation bias and negative values indicate model overestimation bias (Gupta et al., 1999). d values are calculated as (Willmott, 1981):

$$d = 1 - \frac{\sum_{j=1}^{n}\left(Calc_j - Obs_j\right)^2}{\sum_{j=1}^{n}\left(\left|Calc_j - Obs\right| + \left|Obs_j - Obs\right|\right)^2} \qquad (2)$$

where Obs is the mean value of the observed data. This statistical indicator can vary from 0 to 1, a computed value of 1 indicates a perfect agreement between the measured and predicted values, and 0 indicates no agreement at all (Willmott, 1981).

3. RESULTS AND DISCUSSION

3.1. Flow model results

Figure 4 shows the calibration results of measured versus computed hydraulic heads in all the monitoring wells. A good calibration is achieved for heads throughout the simulated period, showing a large water table fluctuation within the modelled period, induced by both evapotranspiration and surface water level variations in the main ditches connected with the tile-drains.

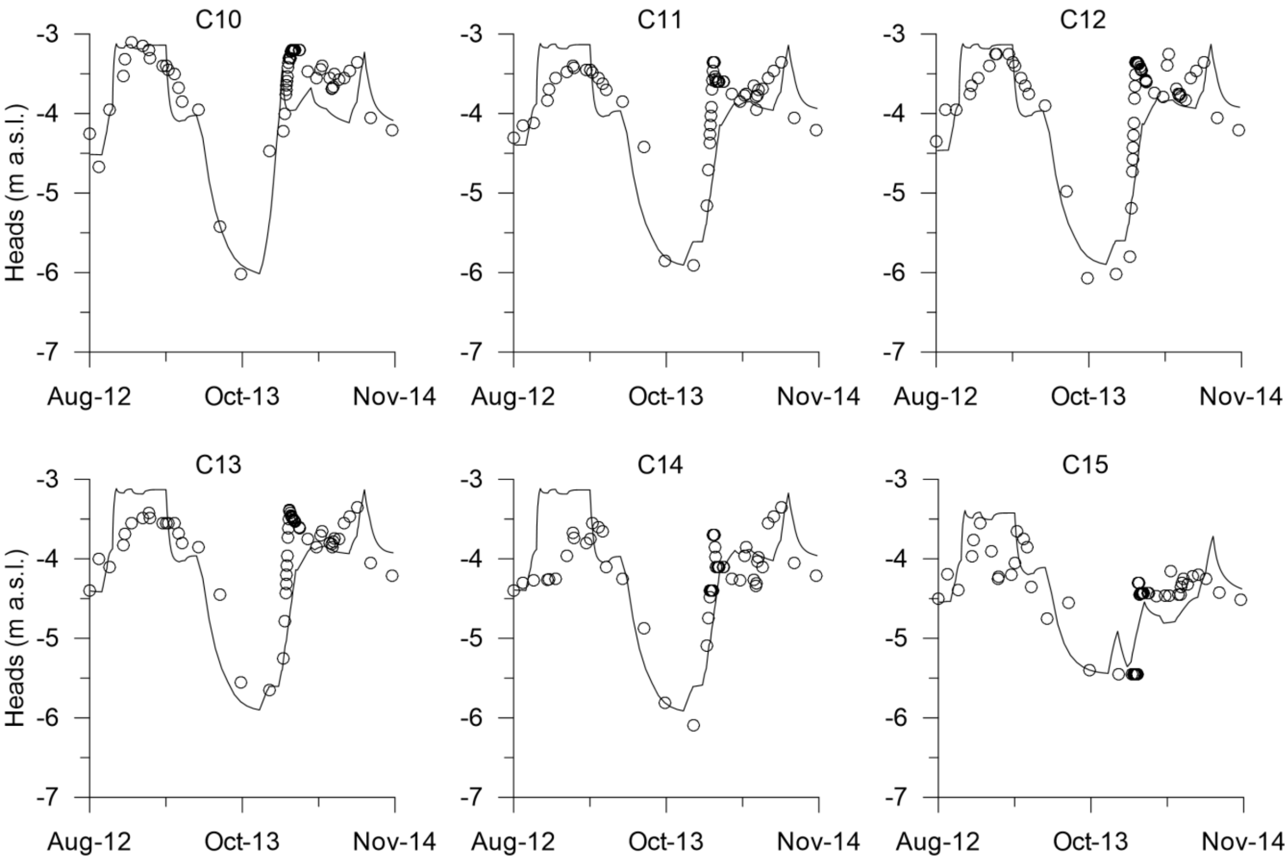

Figure 4: Modeled groundwater heads (black line) compared with observed heads (open circles) in different monitoring wells.

The latter was the main driver for the extremely fast aquifer-aquitard system re-wetting during winter 2013. In fact, when the water level in the main ditches is above the tile-drains elevation, water is injected in the aquifer-aquitard system at a depth of -4.35 m a.s.l. bypassing the top soil. This behaviour is simulated switching from Drain cells to GHB cells in different stress periods. Model scenarios not accounting for this behaviour are not able to capture the observed water table trends, even using different values for k_s and S_y. The monitoring well that is less affected by water table fluctuations is C15 located close to the canal (see fig. 1 for location), although even this monitoring well exhibited a pronounced drawdown during summer 2013. The inverse parameters estimation procedure with PEST resulted in an AME, PBIAS and *d* between simulated and observed heads of 0.19 m, 7.0% and 0.79, respectively. These errors correspond to less than 5.0% of the observed piezometric range during the monitored period so, following the model performance rating of Moriasi et al. (2007), the overall model statistics can be considered good. Being the daily variability of recharge and evapotranspiration very high (fig. 3), to further improve the accuracy of the simulation, a much finer temporal discretization would be required. Nevertheless, the model satisfyingly reproduces the piezometric head trend induced by the complex interplay of recharge events, evapotranspiration, tile-drains recharge/drainage and canal water level changes.

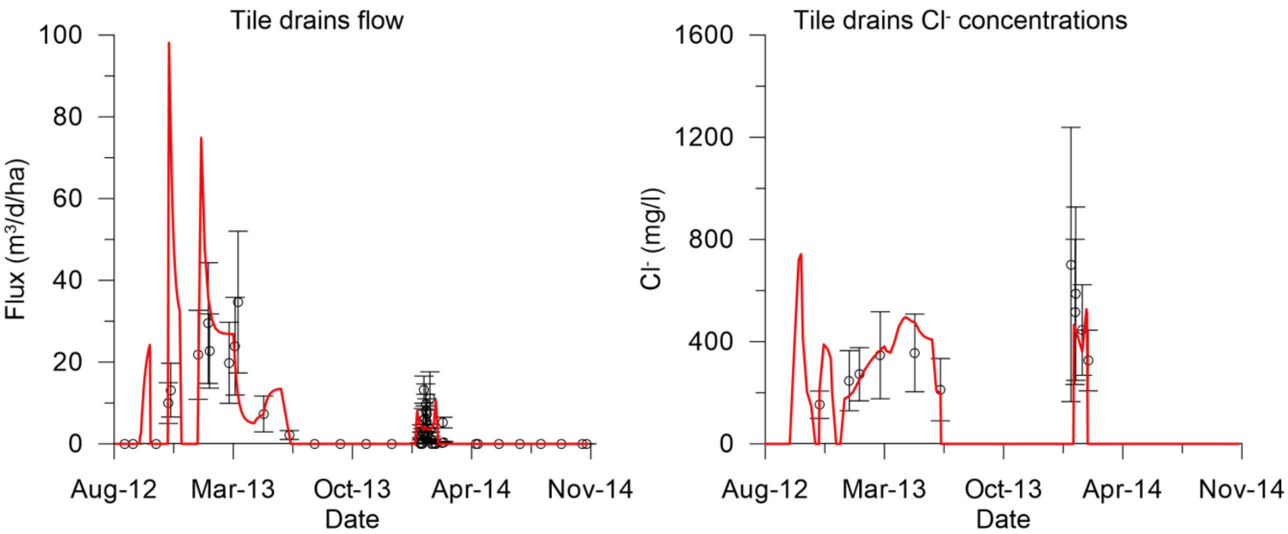

Figure 5: In the left panel, the modeled bulk groundwater flux towards the tile-drains (red line) is compared with the sum of the observed fluxes (open circles) in all the tile-drains. The error bars represent the standard deviation of the observed fluxes in each sampling round. In the right panel, the modeled bulk Cl^- concentration in the tile-drains (red line) is compared with the sum of the observed Cl^- concentrations (open circles) in all the tile-drains. The error bars represent the standard deviation of the observed Cl^- concentrations in each sampling round.

Figure 5 shows the results of the water budget sub-routine, where the total flux towards the tile-drains is extracted and compared with the total flux spilled from the tile-drains during different sampling campaigns. Initially, an attempt to calibrate the observed flux from each tile-drain versus the modelled one was done, but given the extremely large variance of the observed fluxes (influenced by local factors, like small changes in k_s) this approach resulted to be too time consuming; then, the overall modelled flux versus the bulk flux observed in all the tile-drains in each campaign is optimized. This approach gives appreciable results, with a good agreement between simulated and observed fluxes: the MAE, PBIAS and *d* are 1.43 m^3/d/ha, 4.3% and 0.91, respectively. During winter 2013 the observations were less frequent than in winter 2014, and in this period some model underestimation is visible, most probably due to fluxes from the unsaturated zone, not directly accounted for in the model. The error bars give an indication of the extremely large variability of the observed fluxes from the tile-drains.

The model performance statistics for the non-reactive solute transport show a good agreement between simulated and observed Cl^- concentrations in the tile-drains: the MAE, PBIAS and *d* are 33.3 mg/l, 6.8% and 0.85, respectively. Once again, the variability of the observed Cl^- concentrations confirm the extreme heterogeneity of salt discharge from tile-drains. To minimize the bias induced by the extreme variability of observations, a large number of spatially and temporally distributed observations (tile-drains fluxes and Cl^- concentrations) is advisable if the salt export has to be estimated from tile-drains observations alone.

Finally, a simple model scenario is implemented to evaluate the role of tile-drains in linking surface water and groundwater at the field site. By switching off both the Drain and GHB cells pertaining to the tile-drains (see fig. 3 for location), a numerical scenario is created from the previously calibrated model. Examining the resulting water budget, the cumulative groundwater flux towards the surface water is 1652 m^3 versus 2252 m^3 calculated by the calibrated simulation. When compared to the calibrated model, the scenario without the tile-drains results affected by water-logging conditions after prolonged recharge events, since the only outflows are the minor ditch and the canal located at the model boundaries (see fig. 3 for location). For this scenario, the cumulative surface water flux towards the aquifer-aquitard system is 203 m^3 versus 782 m^3 calculated by the calibrated simulation. The scenario without the tile-drains displays an inflow from surface water nearly four time lower than the calibrated model, highly limiting the freshening of the aquifer-aquitard system. Moreover, the activation of the tile-drains in the recharge mode (GHB), once the stage in the ditch exceeded the tile-drains elevation, results not to be linked with local rainfall events but is rather governed by human management of the ditches employed in the local reclamation works. As a consequence, given the unpredictability of human intervention, it is not possible to use the calibrated model to predict tile-drains behaviour.

This comparison highlights the role of the tile-drains in augmenting the interaction between surface water and groundwater, with respect to an undrained field. This enhanced communication complicates the flow pattern and the water budget and, to be accurately simulated, it requests more field data with a much finer spatial and temporal discretization.

3.2. Transport model results

Figure 6 shows the modelled and observed Cl^- concentrations in groundwater. In comparison to the simulated heads, the dissolved Cl^- concentrations are poorly captured by the model, although the general trend is correctly represented. The recorded Cl^- concentrations are inversely correlated to the water table drawdown suggesting that dilution/accumulation processes play an important role in governing dissolved Cl^- concentrations.

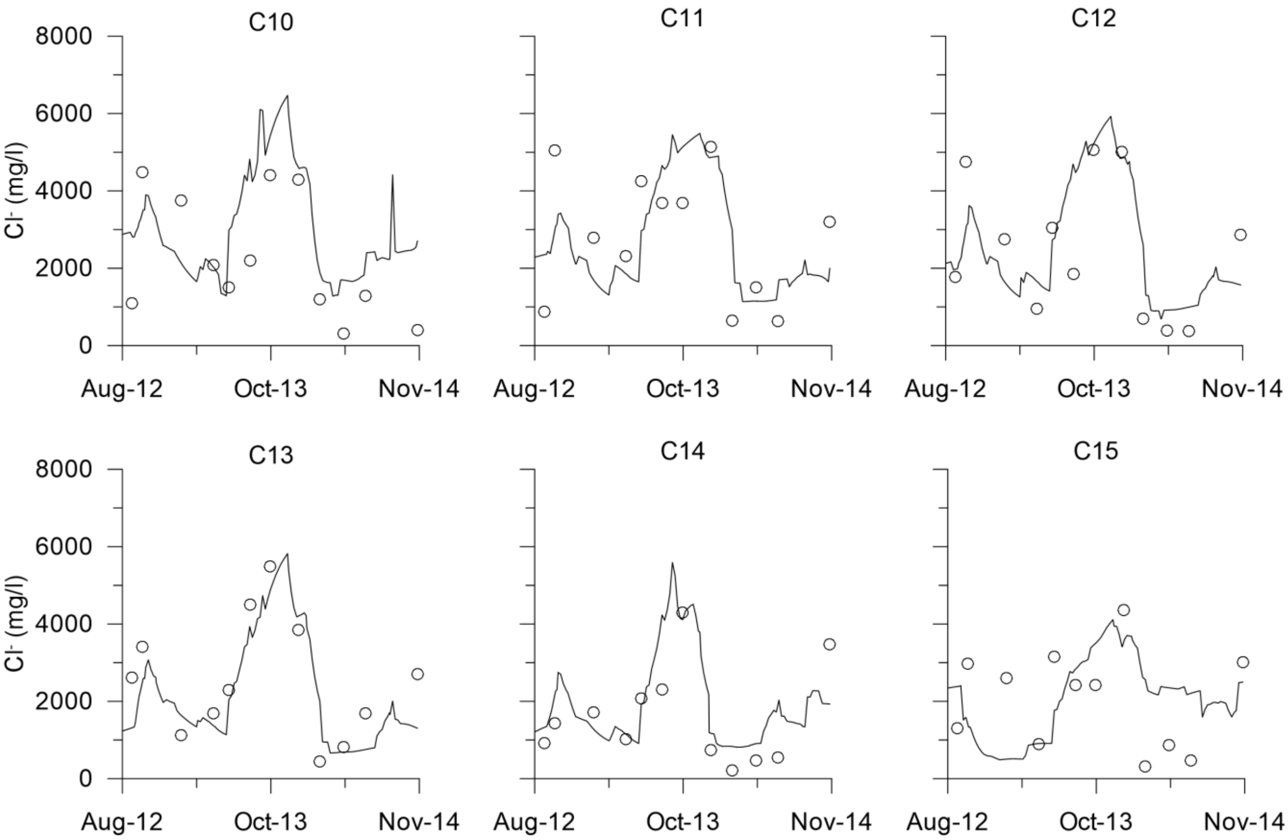

Figure 6: Modeled groundwater Cl^- concentrations (black line) compared with observed Cl^- concentrations (open circles) in different monitoring wells.

The inverse parameters estimation procedure with PEST resulted in an AME, PBIAS and *d* between simulated and observed Cl^- concentrations of 337.6 mg/l, -2.4% and 0.69, respectively. The model performance statistical indicators clearly deteriorate with respect to the heads and fluxes ones; this is probably imputable to the relatively long screen of the monitoring wells that induces an artificial mixing, which conceals the hydrogeochemical stratification of the aquifer-aquitard system (fresh, oxic, Ca-Cl-SO_4 type in the upper part and brackish, reducing, Na-Cl type in the lower part). The simulated Cl^- concentrations do not accurately reproduce the various Cl^- sub-peaks observed in the monitoring campaigns mainly because the lateral Cl^- heterogeneities are significant within the aquifer-aquitard system, since preferential pathways may develop because of cracks formation in the clay soil during the summer season. The seasonal changes in k_s is not accounted for in the simulation, given that both HYDRUS and SEAWAT consider k_s as a constant parameter and not as a variable. The Cl^- concentration in the recharge water is initially assumed homogeneous over the model domain during each stress period, while in reality appreciable Cl^- variations have been detected during seasonal soil sampling. Including the observed spatial variation in the Cl^- concentration of the recharge improves the calibration process. Then, the addition of a layer proportion of the modelled Cl^- concentrations, directly related to the layer k_s, positively influences the model performance. This assumption is linked to the ability of the more permeable layers to release more water during well sampling, thus Cl^- concentrations are not an arithmetic mean of each intercepted layer but rather a weighted average based on the k_s of each intercepted layer.

3.3. Model validation

Figure 7 shows the pore-water Cl^- profiles collected close to four monitoring wells used to validate the numerical model. The model well reproduces the Cl^- profiles, with an AME, PBIAS and d between simulated and observed Cl^- concentrations of 142.2 mg/l, -2.1% and 0.83, respectively. The model performance statistical indicators largely improve with respect to the monitoring wells, demonstrating that most of the uncertainty is essentially due to artificial mixing within the long screen monitoring wells.

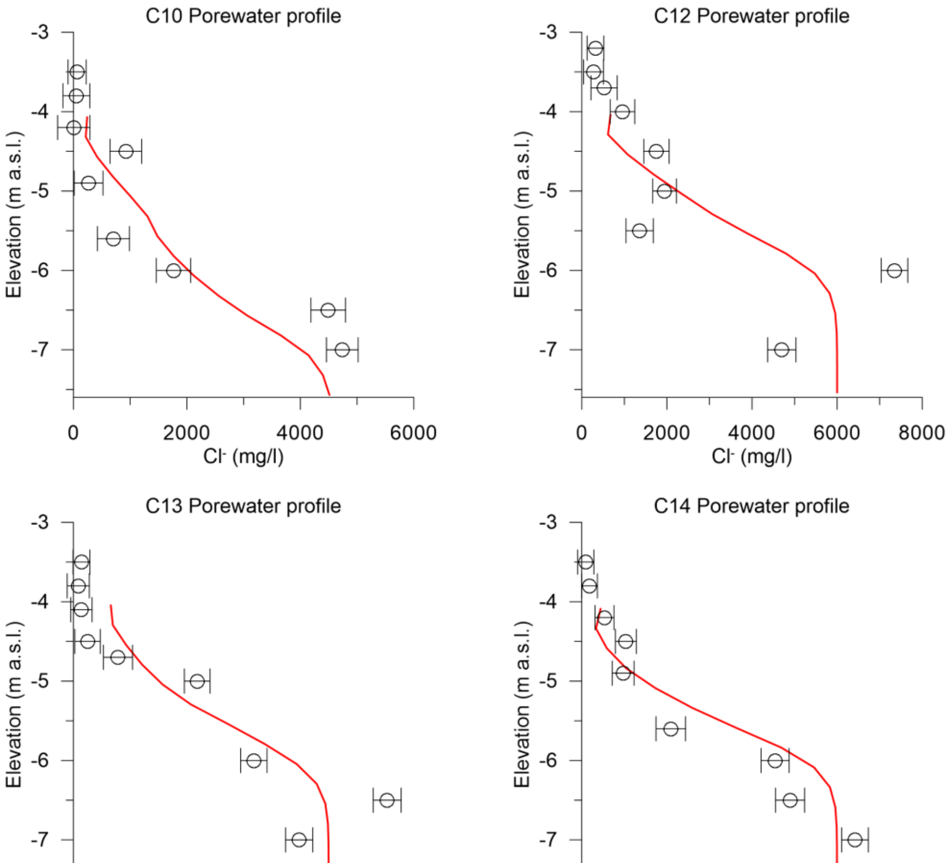

Figure 7: Modeled pore-water Cl^- concentrations in the saturated zone (red line) compared with observed pore-water Cl^- concentrations (open circles) for both the unsaturated and saturated zones, in different soil profiles; error bars represent the analytical errors.

From figure 7, it could be noted that the mixing zone between brackish and freshwater is not sharp, and the freshwater is mainly restricted to the unsaturated zone. The smooth gradient between saline and freshwater could be justified by the relatively low value of vertical k_s that impede fast mass fluxes. Cl^- concentrations in the saturated zone increase gradually downward, reaching an average value of 6000 mg/l, except for C10, located at the border of the agricultural field, which is affected by the diluting effect of the recharge coming from the regional groundwater flow provided by the GHB (fig. 3). Results of the validated model confirm the interconnection between surface water bodies, tile-drains and the aquifer-aquitard system; the transport model clearly elucidates that the lateral movement of solutes is very limited compared to the vertical movement (see Video).

3.4. Sensitivity analysis

The model sensitivity to perturbation of the input parameters is addressed with PEST. Figure 8 shows that the most influencing parameters are the k_s values of the layers located right below and above the tile-drains, followed by the S_y of the layers above the tile-drains and by the evapotranspiration rate.

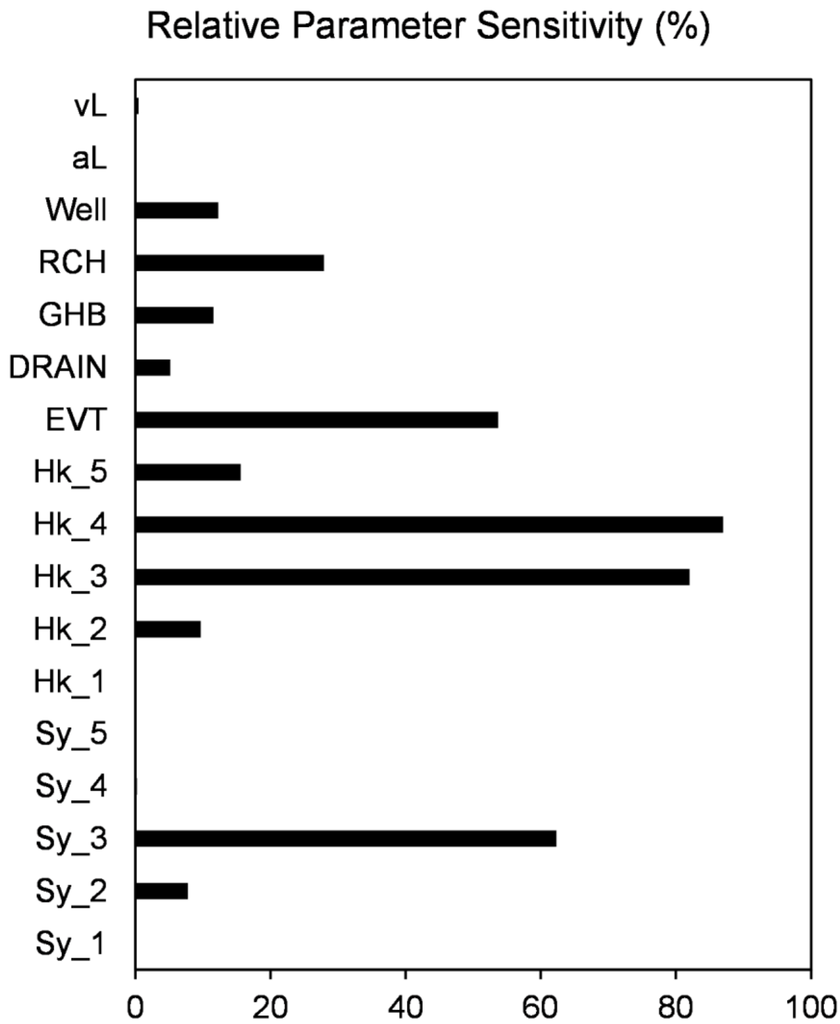

Figure 8: Results of the sensitivity analysis with PEST.

The high k_s and S_y sensitivities are not surprisingly, since those parameters drive the surface water-groundwater exchanges, besides being usually the most uncertain parameters in small scale models (Colombani et al., 2015; De Louw et al., 2013; Hansen et al., 2013). Moreover, S_y drives the vertical fluctuation of the water table in time, so this parameter is even more important than k_s in modelling

silty-clay units. The evapotranspiration rate is extremely important since it provokes the complete desiccation of the aquifer-aquitard system during summer 2013. The conductance values for both Drain and GHB are not influencing the objective function as much as S_y, since the number of head observations are 3 times higher than the flow observations. In addition, for values of the drain conductance below 10 m^2/d, flooding conditions are calculated in winter time, but the agricultural field was never submerged by ponding water; thus, low conductance values are a priori excluded. The upward flux simulated by the Well Package influences the Cl^- concentrations and not the hydraulic heads and fluxes toward the tile-drains, so model sensitivity is quite low. Finally, the transport related parameters (longitudinal and vertical dispersivities) barely influence the model outputs, given that the solute transport in this aquifer-aquitard system is mainly diffusion driven.
In table 3 the Cl^- cumulative mass balance from SEAWAT, for the first year and at the end of the simulation, are reported. Most of the Cl^- mass exiting from the tile-drains (Drains+GHB applied to tile-drains) is calculated for 2013 with 5594 kg-Cl^-/ha/y, while only 237 kg-Cl^-/ha/y are exported in 2014. This is due to salt accumulation during summer 2013 induced by limited Cl^- transpiration from plants. Summing all the surface water bodies outflows (Drains+Rivers+GHB applied to tile-drains), the cumulated Cl^- mass export at the end of the simulation is 7373 kg-Cl^-/ha, that gives an average value of 3385 kg-Cl^-/ha/y. Despite being quite elevated, this mean value is comparable with other literature values (Vandenbohede et al., 2011).

Table 3: Cumulative mass balance of Cl^- during the first year and at the end of the simulation period.

	1st year		**Final**	
	IN (kg/ha)	**OUT (kg/ha)**	**IN (kg/ha)**	**OUT (kg/ha)**
WELLS	1818.117	0	3971.27	0
DRAINS	0.0	-5534.5	0.0	-5771.7
RIVERS	0.0	-523.8	0.0	-664.2
GHB	0.0	-937.4	7.0	-937.4
RECHARGE	4640.8	0.0	6511.1	0.0
EVT	0.0	-130.4	0.0	-1584.5
STORAGE	6316.8	-5633.5	8197.6	-9734.0
(TOTAL)	12775.7	-12759.7	18687.0	-18691.7
NET (IN-OUT)	16.0		-4.8	
DISCREPANCY (%)	0.1		0.0	

The large mass of dissolved salts leaving the aquifer-aquitard system via the tile-drains is extremely important, since it can carry in solution not only Cl^- but also other chemical species potentially hazardous for aquatic life, like heavy metals (Mastrocicco et al., 2015) or reactive nitrogen species (Di Giuseppe et al., 2014). For this reason, an accurate mass balance can be obtained only using calibrated and validated numerical models accounting for all the specific features of reclaimed environments.

4. CONCLUSIONS

In reclaimed coastal areas saline groundwater is rather common, thus shallow freshwater lenses, generated by vertical recharge from rainfall of lateral recharge from surface water bodies, are essential for crop development. Often the seasonal behaviour of these freshwater lenses is not fully understood, especially in tile-drained agricultural fields. To untangle the issue of quantifying the interaction between surface water bodies, tile-drains and an unconfined aquifer-aquitard system, in this study a set of nested numerical flow and transport models are used to develop a robust conceptual model and

to give some insight on the capabilities/limitations of the monitoring techniques employed to gain the model parameters.
This study demonstrates that tile-drains augment the interaction among groundwater and surface water, guaranteeing not only the avoidance of water-logging condition in the field but also contributing to the freshening of the upper portion of the aquifer-aquitard system, thus ultimately ensuring the cultivability of reclaimed soils. The inclusion of the effect of tile-drains on the aquifer-aquitard system leads to: (i) a much complicated flow pattern and water balance, (ii) the need for a greater number of filed data with a much finer temporal and spatial resolution, especially to account for the high variability of evapotranspiration, (iii) the obligation to employ high resolution soil profiles to avoid artificial mixing within monitoring wells, that do not allowed a good match among the observed and calculated concentrations of dissolved species, (iv) and the need to use a complex and time-demanding set of numerical models (e.g. HYDRUS and SEAWAT), that anyway cannot reproduce seasonal changes in k_s to simulate the seasonal variability of preferential flow through cracks in the soil.
Even though some difficulties have occurred in the simulation of the tile-drains and in the spatial and temporal discretization of some input parameters, still field monitoring and numerical modelling allow to obtain an advanced and robust conceptual model of this complex environment that would be an appropriate basis to build up a mass balance of reactive species like nutrients or heavy metals. Nevertheless, even taking into account all the factor complicating the conceptual model within the numerical simulation, it is impossible to use the numerical model as a tool to predict the activation of the tile-drains both in the drainage mode and in the recharge mode, since their activation is not ruled by natural phenomenon (e.g. rainfall) but it is rather due to human intervention.

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