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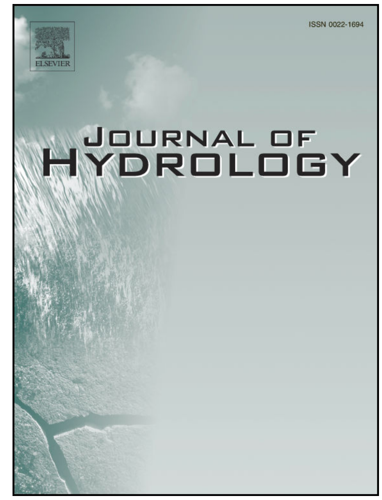

Research papers

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Managed Phreatic Zone Recharge for Irrigation and Wastewater Treatment

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Abstract

Managed phreatic zone recharge with marginal water, using (existing) drainage systems, raises the water table and increases water availability for crops. This is a newly developed method of freshwater conservation and marginal water treatment and disposal, but risks crop and environmental contamination. The fate of contaminants of emerging concern (CECs) within the irrigated water is addressed. We introduce numerical and analytical models, inspired loosely by a field site where treated domestic wastewater is used for subsurface irrigation. The treated wastewater would otherwise have been discharged into rivers, thereby spreading downstream. Model results show that minimal amounts of CECs are transported to deeper aquifers. Crops are not contaminated, except during dry years where small amounts of mobile CECs rise to the root zone, but then only directly above each irrigation drain. Under an annual precipitation surplus, less-mobile solutes are thus unlikely to ever enter the root zone. The primary mechanism of solute transport is lateral advection within the phreatic aquifer. Despite spatio-temporal heterogeneity in water flux magnitudes and directions, contaminant retardation does not significantly alter mass balance outcomes, only how fast it gets there. Therefore, persistent CECs pose the greatest risks, though overall environmental and crop contamination risks appear low. To maximize complementarity with subsurface irrigation systems, future advances in water treatment technologies should focus on removing persistent CECs. However, the system may be unsuitable for climates with annual precipitation shortages, as CECs may accumulate in the root zone and crops.

Keywords:

Groundwater aquifer; Irrigation and drainage; Contaminant transport; Wastewater reuse; Bioremediation

Highlights:

New subsurface irrigation and drainage method that raises the phreatic water table.

New method synergistic with marginal water irrigation: crops not directly exposed.

Tool for irrigation, freshwater conservation, and wastewater treatment and disposal.

Crop contamination risk is low, except by mobile contaminants in very dry years.

Crop and wider environmental contamination risk largest for persistent contaminants.

1. Introduction

Even in temperate humid regions, periodic shortages of fresh water resources may constrain irrigation. Use of treated wastewater for irrigation may be an effective strategy to balance regional fresh water supply and agricultural water demand (Dingemans et al., 2021, Pronk et al., 2021, Narain-Ford et. al 2021). Water stress caused by climate change exacerbates the need to re-use treated wastewater (Elliott et al., 2014). Moreover, the treated wastewater may contain nutrients, which enriches the soil (Lubello et al., 2004). However, wastewater treatment does not remove all impurities from the water, such as pharmaceuticals, pathogens, and other contaminants of emerging concern (CECs) (Yang et al., 2017; Richardson and Ternes, 2018). Surface irrigation with treated wastewater may cause adverse health effects amongst farm workers and the general public through direct exposure or crop contamination (Qadir et al., 2010; Devaux et al., 2001). It may also adversely affect soil fertility because it induces nutrient leaching (Dole et al., 1994) and alters the pH and composition of root zone water (Mohammad and Mazahreh, 2003). Meanwhile, treated industrial and domestic wastewater is often discharged via surface water channels towards the sea, spreading contaminants along these discharge paths (Beard et al., 2019). Accordingly, utilizing treated wastewater in subsurface irrigation may reduce these risks (Narain-Ford et al., 2020). An estimate of the potential of subirrigation for the Netherlands shows that it is logistically plausible for 78% to 84% of all treated wastewater to be used in subirrigation, which would reduce freshwater demand in agriculture by 8% to 12% (Narain-Ford et al., 2021).

Subsurface irrigation and drainage of phreatic aquifers using marginal water as water supply, is a new irrigation and drainage technology that allows for treated wastewater to undergo further in-situ filtration and remediation in the soil, while simultaneously ameliorating crop water stress and freshwater scarcity (Narain-Ford et al., 2022). Subirrigation is applied using controlled drainage systems, which can drain excess water (prevent waterlogging) and recharge the phreatic aquifer (subirrigation) when necessary. In this system, subirrigation pipes are buried in the phreatic zone some distance below the root zone. Therefore, the irrigation raises the water table. The CEC-free soil moisture located between the pipes and the root zone then rise towards the root zone due to capillary rise, and the effluent does not enter the root zone directly (Bern et al., 2013). Controlled drainage may also reduce nutrient losses from the root zone caused by high groundwater levels (Bonaiti and Borin, 2010; Peng et al., 2015; Borin et al., 2001; Borin et al., 1998), and reduce the spreading of the CECs contained in the effluent. Buried pipes that are already widely present in agricultural fields for the purpose of drainage (De Wit et al., 2022) may be used for subirrigation, making this a potentially low-cost and accessible option.

Irrigation from the subsurface rather than the surface aims to minimize direct crop exposure to CECs. This nature-based solution uses the soil, groundwater, and rainwater between the drains and root zone as a buffer zone for biodegradation and adsorption, similar to river bank filtration and constructed wetlands (Narain-Ford et al., 2020). Due to biodegradation and adsorption, the soil functions as a bioreactor. Due to these processes and dilution, water recovered by the drains during wet periods may contain lower concentrations of CECs than the injected effluent, and thereby be of higher quality. This improved water may be used instead of freshwater for some appropriate purpose, such as industrial processes, thus further conserving freshwater. Many CECs, being organic compounds, are biodegradable to some extent by soil microbes, both in the unsaturated (Bekele et al., 2011) and saturated (Rauch-Williams et al., 2010) zones. However, there is still a large uncertainty in the biodegradation rate and extent (Greskowiak et al., 2017). Persistent CECs may enter the root zone through dispersion and capillary rise, or be advected further into the environment by groundwater flow. With subsurface irrigation, the exposure of crops to CECs is indirect, and the groundwater is directly exposed, which is opposite to traditional surface irrigation systems. Therefore, it is important to investigate the risks of both crop and environmental contamination specific to subirrigation systems.

In this paper we construct numerical and analytical models to investigate effluent transport in the subsurface and the risks of crop and environmental contamination with CECs in subirrigation systems. We build upon a controlled drainage and subsurface irrigation system fed with treated domestic wastewater, that has been implemented in the Netherlands and monitored since 2015 (Narain-Ford et al., 2022). Solute fate is modelled for tracers and a reactive solute sampled during the experiment. These results may be translated to other hydrogeological properties and CEC types by varying the parameters in the model. We emphasize that the models constructed here are not meant to recreate the field site, but rather, meant to illustrate the fate of CECs in a generic controlled drainage-subirrigation system that represents a simplified concept of the field experiment, in climatic conditions typical of the Netherlands. The simple and generic nature of the models would better elucidate physical processes governing contaminant fate, and allow it to be more easily adapted to other implementations of the system. Some data from the field experiment is presented in this study as it serves as proof-of-concept, to show that the model created is able to generally account for the important physical processes that govern solute transport and fate, particularly mass balances, in a generic subsurface irrigation and drainage system.

With the models, the processes that affect the fate of solutes contained within the effluent (tracers and geochemically reactive CECs that undergo adsorption and biodegradation) are characterized. Solute mass balances, breakthrough curves, and plume transport behavior are analyzed. Partitioning of solute fate is between crop solute uptake, advection in the phreatic groundwater, leaching to deeper groundwater, biodegradation, soil water, and drainage by the drainage system. The partitioning of the fate of irrigated solute mass (i.e. the relative solute mass balance) ultimately determines the extent of crop and environmental contamination. Furthermore, we also use the model to investigate the potential of the subirrigation system to lead to adverse environmental effects in the long-term, such as long-term accumulation of CECs in the soil. Hence, the model reveals aspects of the system that cannot be obtained from available experimental data, such as a more extensive spatio-temporal characterization of CEC transport processes, suitability of the system under different climates and hydrological conditions, the potential for CEC transport to deeper aquifers and surface water, and the risk of crop uptake of CECs.

2 Methods

2.1 Field site

The subsurface irrigation and drainage system we use as an example is implemented in a 58500 m^2 field site in Haaksbergen, the Netherlands, and is the only such system in operation. Maize grain destined to be livestock fodder is grown. Treated wastewater from a domestic wastewater treatment plant is fed into the subsurface irrigation and drainage system, which has been in operation since May 2015. Further details of the field site are available in detail in Narain-Ford et al (2022), and will not be reproduced here.

2.2 Hydrogeological model

The subsurface irrigation/drainage system was modeled with HYDRUS-2D (Šimůnek et al., 2016). Many studies in the literature have shown that HYDRUS-2D is suitable for simulating subirrigation (Cai et al., 2019; Saefuddin et al., 2019; Siyal et al., 2013) and drainage (Kacimov and Obnosov, 2021) systems. For brevity, a description of HYDRUS-2D will not be reproduced here. An overview of the parameter values and the sources they were obtained from can be found in Table 1. Some hydrogeological parameters (Table 1) were calibrated by comparing groundwater levels simulated without irrigation and crops, against field data obtained from a grassfield 1km from the experimental plot, with a post-calibration Nash-Sutcliffe efficiency of 0.61 (Figure 1), which is considered good (Moriasi et al., 2007). The subsurface soil and phreatic aquifer is simulated to a depth of 4m, as the two sampled tracer types were not detected beneath that level in six years of experimental data (Figure 2). Furthermore, borehole soil profiling has revealed that a poorly conductive loam layer is present at 4m depth at the middle of the experimental site.

We construct a 13m by 4m model domain (Figure 3a) to represent the subsurface to a depth of 4m from the soil surface, with a root zone of the maize crop that reaches a depth of 0.6m, and two drainage/irrigation pipes buried 1.2m beneath the surface spaced 6m apart. We insert breakthrough curve observation points at the same approximate locations as in the field experiment (0.2m, 0.6m, 1.0m beneath the soil surface, and 0.25m, 0.8m, 1.3m beneath drain level), with one column of observation points directly above and below the downstream irrigation drain, and one column midway between the two drains, for a total of twelve points.

The top boundary for water flow is an atmospheric boundary that represents precipitation; excess precipitation beyond the soil's infiltration capacity is lost to runoff (Šimůnek et al., 2016). Daily precipitation and reference evapotranspiration data (calculated according to Makkink, 1957) were obtained from the Dutch meteorological institute (KNMI). The daily potential evapotranspiration of the root zone during the crop season was calculated by multiplying the reference evapotranspiration with a weekly crop factor for Dutch grain maize crops (LAGO, 1984). Outside of the crop season, the reference evapotranspiration was used as the potential evapotranspiration. These atmospheric fluxes are shown in Figure S1 in the supplementary material. The root distribution, which controls the spatial distribution of root water uptake, is modelled with a maximum intensity at depth $z = 0.05m$ (the first root zone layer in Figure 3b), and thereafter a decreasing intensity in the following three root zone layers, with relative intensities 8,4,2,1. The potential evapotranspiration rate is partitioned across each node in the root zone proportionally to the root density at each node (Šimůnek and Hopmans, 2009). The actual root water uptake flux is then determined with the pressure head reduction function of Feddes et al (1978), with the parameters for corn (Wesseling et al, 1991) that are pre-programmed into Hydrus 2D.

Each irrigation drain is modelled as a circular opening with a 4cm internal diameter and 8cm external diameter. The crop season in the model lasts for 150 days every year starting from the first of May, while the drainage season occurs the rest of the time. The irrigation drains have imposed pressure head boundaries during the crop season, and 'seepage face' boundary conditions (Liu et al., 2021) outside the crop season. The bottom boundary is a deep drainage boundary condition: an imposed flux boundary whose magnitude depends on the groundwater level (Hopmans and Stricker, 1989) according to the formula $q = A\exp(B\,|h - Z|)$ m/day, where h is the hydraulic head at the boundary, and A,B,Z are empirical constants. The left (upstream) boundary for water flow is an imposed head gradient boundary representing the regional head gradient. The right (downstream) boundary is a Cauchy boundary condition comprised of a prescribed head and prescribed conductivity different from the rest of the aquifer.

The initial conditions for water flow were set to hydrostatic equilibrium relative to a water table depth of 1m, corresponding to measurement data from the unirrigated field on the day before the crop season of 2016 began. Initial conditions for solutes were set as the rainwater concentration at the top boundary, the groundwater concentration at the bottom boundary, and a linear distribution with depth within the model domain. The simulation period of the model is four years, starting from the crop season of 2016.

The regional groundwater head gradient has a large and direct effect on solute breakthrough curves, especially at the observation points located at a larger distance from the drains. This is caused by the direct impact of regional groundwater head on the lateral advection velocity, and because it is responsible for a much larger proportion of the flux passing through the simulated domain than any other flux source or sink. Since detailed information on the regional head gradient over the measurement period is unavailable, the regional head gradient also has a significant contribution to uncertainty in the model parameterization. The combination of relatively large uncertainty in a relatively impactful parameter might manifest as uncertainty in the simulation results. To identify the extent to which uncertainty in the regional head gradient would affect the accuracy of the simulation results, we briefly investigate the effect of varying the regional head gradient on the simulated breakthrough curves. In the base model, a single constant value of the regional head gradient (0.0014m/m) was used. We repeated the numerical simulations with two other regional flow scenarios, namely zero head gradient (0 m/m) and a high estimate of the regional head gradient (0.0022 m/m). Unless otherwise mentioned, subsequent results and discussions refer to the base model.

2.3 Tracer and contaminant transport model

Three indicators of effluent spreading are analyzed: Cl:Br, solution EC, and concentrations of the antiepileptic pharmaceutical carbamazepine. We also simulate a generic tracer with identical effluent concentration timeseries as carbamazepine to illustrate the effect of adsorption and biodegradation on solute transport. We assume that Cl:Br (Davis et al., 1998) and EC (Chaali et al., 2013; Scott et al., 2020) behave as tracers of the effluent, and carbamazepine as a reactive solute that undergoes instantaneous equilibrium adsorption and first-order biodegradation (Durán-Álvarez et al., 2012; Williams et al., 2014; Williams et al., 2006). Although adsorption isotherms (e.g. Langmuir isotherm) and biodegradation rates (e.g. Michaelis-Menten kinetics) generally follow a non-linear functional response with respect to the concentration of the solute being transported, the trace concentrations of CECs in the effluent would be small enough for the isotherms and rates to be effectively first-order.

For all substances except EC, we assume that all solutes in soil water are taken up by crop roots along with the water regardless of concentration; this is reasonable for CECs, which are present in trace amounts (Christou et al., 2019). As Cl:Br is a ratio of trace elements, not a concentration, it is reasonable that root uptake does not affect levels in the soil water. Since root salt uptake rates increase linearly with soil salinity up to some physiological threshold (Moya et al., 1999), we assume for simplicity that the maximum root uptake concentration of the ions that contribute to EC is the rainwater concentration. The rainwater ionic content is small compared to the groundwater and effluent, thus our implementation is in principle similar but more realistic than other studies in the literature (e.g. Siyal et al., 2013) that assume no ions are taken up by roots. In the simulations, excess ionic content in the soil water beyond the rainwater concentration is left behind in the soil water, where it accumulates and increases the EC.

The greatest determinant and the largest source of uncertainty regarding solute fate is the adsorption coefficient and the biodegradation rate, as they span multiple orders of magnitude, are highly uncertain in the field, and vary not only with CEC identity but also environmental parameters. In this study we focus on tracers and a relatively mobile and persistent CEC (carbamazepine), and investigate whether they may contaminate crops and the wider environment. The opposite extreme of immobile CECs represent a rather trivial case (with respect to a contaminant transport model), where the CECs are mostly adsorbed or biodegraded within the immediate vicinity of the irrigation pipes (Narain-Ford et al., 2022). Adsorption coefficients and biodegradation rates of carbamazepine available in the literature vary over orders of magnitude, depending on environmental factors and in-situ physical and biochemical conditions (Durán-Álvarez et al., 2012; Williams et al., 2014; Williams et al., 2006). Therefore, we calibrated these biochemical parameters so that the simulated breakthrough curves match the observed carbamazepine concentrations in the field. This resulted in a half-life of 125 days (biodegradation rate coefficient of 0.008 day^{-1}) and an adsorption coefficient of 1 L/kg, implying a retardation factor of 2.5, falling well within the range provided in the literature.

Daily effluent Cl:Br ratio, EC, and carbamazepine concentrations from the field site were interpolated from the two to four effluent samples measured per year and used as realistic input values for the model analysis. Daily effluent EC values became available after June 2018, and were incorporated into the simulations. The Cl:Br ratio of rainwater is estimated at 100 from samples taken from the shallow soil (20cm depth) in 2016, and agrees roughly with the literature (e.g. Davis et al, 1998). The EC of rainwater is estimated at 100 μS/cm from early shallow soil samples, and also agrees in general with the literature (e.g. Zdeb et al (2018)). Groundwater Cl:Br (300) and EC (800 μS/cm) were estimated from field samples obtained from 4m beneath the soil surface or deeper, as they both are uniformly and constantly measured at these respective values over the entire subsurface profile deeper than 4m (Figure 2), during the measurement period. These values are also in approximate agreement with various studies in the literature on environmental Cl:Br and EC values (e.g. Alcalá and Custodio, 2008; Van den Brink et al., 2007). Groundwater and rainwater carbamazepine concentrations are assumed to be zero. Concentration flux boundary conditions were applied for solute transport at all boundaries. Daily effluent Cl:Br ratio, EC, and carbamazepine concentrations from the field site were interpolated from the two to four effluent samples measured per year and used as realistic input values for the model analysis.

3. Results

3.1 Hydrology

Model results show that the system is able to stably maintain the groundwater level within a narrow band during the crop season. Most of the variations in groundwater levels occur during the drainage season, because then the drains control the fluxes in only one direction. In other words, during the drainage season, the drains remove water when the groundwater level is high, but do not add water when the groundwater level is low, unlike during the crop season. As the maize crops grow throughout the crop season that spans from May to September, their water requirements increase. Hence, evapotranspiration rates gradually increased between the start and the end of the crop season. However, the precipitation volume was low in May and June, and significantly higher in July, August and September, in accordance with historical averages. Overall, these trends cause the irrigation flux to be at a maximum midway through the crop season (Figure 4a). Therefore, such a subsurface irrigation system must be designed to meet the large water demands of July and August.

The evapotranspiration rates were at a minimum in January, while precipitation rates in January were around the annual average. Hence, the surplus precipitation (P-ET) is at a maximum in January, which causes most drainage to occur in January (Figure 4a). Since there is a period of three months between the end of the crop season and January, most solutes intercepted and recovered by the drains will be from the outer edge of the plume, which was injected at the start of the crop season and transported to the root zone by capillary flux during the crop season, and which is moving back into the saturated zone in January (Figure 5a). Note that lateral advection is minimal above the water table, hence the solutes migrating downwards at this time from the unsaturated zone to the water table can be intercepted by the drains. On the other hand, 'newer' solutes that were injected near the end of the crop season would have left the vicinity of the drains by January through lateral and downwards advection, and either be discharged into the environment, or extracted by a downstream drain if the plume does not sink too quickly (Figure 5b). Since the biodegradation of CECs require the solutes to reside in the soil for a period of time, and since (first-order) biodegradation has diminishing returns with increased residence times, it is beneficial that 'older' effluent is recovered by drainage instead of 'newer' effluent. The 'older' effluent that is recovered would comprise a larger fraction of non-biodegradable contaminants that pose a larger risk of long-term accumulation in the soil, while the 'newer' effluent left in the soil would be able to biodegrade further. Simulations (not shown) with a non-retarding but biodegradable solute indeed result in lower drained fractions of solute mass than full tracers.

Table 2 shows the water balances of the simulations and field experiment. The incoming regional groundwater flow (Figure 4b), outgoing downwards flux (Figure 4c), and outgoing regional flow (Figure 4d), are positively related to the height of the groundwater level. Of these three boundary fluxes, the outgoing regional flow is the most sensitive, while the incoming regional flow is the least sensitive to the groundwater level. The outgoing fluxes vary greatly over time because of irrigation, drainage, and crop evapotranspiration, which greatly affect the water balances and groundwater levels of the domain. The downwards flux varies less over time than the outgoing regional flow, which reflects that the resistance of the bottom boundary (implicit in the deep drainage bottom boundary condition) is larger than that of the resistance to horizontal flow. Most of the variations in these fluxes occur during the drainage season, because as previously discussed, most of the fluctuations in groundwater levels occur then. By reducing the groundwater levels and hence the outwards fluxes of water and solute, drainage outside of the crop season helps arrest the spreading of effluent.

Table 2 shows that more irrigation is required and less drainage of water occurs, as the regional flow head gradient decreases. This occurs because the natural (non-irrigated) average groundwater level decreases as the regional head gradient decreases. For this same reason, the vertical outflow is smaller when the regional head gradient is smaller. Altogether, the saturated zone flux out of the simulated domain (lateral + vertical) is lowest when there is no regional flow, even though the irrigation flux is higher and drainage flux is lower. This means that the rate of transport of effluent in the saturated zone increases with the rate of regional flow: the average outwards flux in the high regional flow scenario is about 1.6 times that of the zero regional flow scenario. The absence of regional flow does not necessarily imply that no lateral discharge of water from the simulated domain occurs. When hydraulic gradients arise between the simulated domain and the boundaries due to precipitation or irrigation in excess of evapotranspiration, not all the excess water will be discharged downwards through the aquifer bed, as some water will be discharged laterally to the downstream boundary. Therefore, the lateral discharge is significant even when the regional head gradient is zero.

3.2 Tracer transport

The simulated total irrigation and drainage volumes show good agreement with the field observations (Table 2). The simulated breakthrough curves also show good agreement in general with the field data (Supplementary material Figure S2.1 – S2.6). In this section (Section 3), key points in the solute mass balances (Table 3) and breakthrough curves (Supplementary material Figure S2.1 – S2.6) are elaborated. The fit between simulation data and field measurements appears slightly better for Cl:Br than for the EC, especially at the measuring points midway between drains. This is consistent with Cl:Br being likely a more reliable tracer than EC, because Cl:Br is more conservative (Davis et al., 1998) than EC (Pellerin et al., 2008).

Results of the base model show that the solute concentrations in the root zone (0.6m depth and above) reset to nearly background levels before the start of every crop season, with no evidence of long-term accumulation, due to the large excess precipitation that occurs outside the crop season. This is in agreement with field observations for a normal hydrological year (Narain-Ford et al., 2022). The tracer reaches the root zone within the first crop season, and achieves an annual periodic steady-state at the end of the drainage season in terms of effluent plume shape, location, size, solute concentrations, and solute mass balances within four years. Therefore, the size of the model domain and a simulation period of four years is sufficient for long-term analyses of the system.

During the crop season, the effluent plume moves upwards due to capillary rise when crop water requirements are not fulfilled by precipitation. Therefore, the upper part of the effluent plume rises in the summer to above drain level, then sinks back to drain level during the drainage season, and may be partially drained away. For EC, where we assumed that the maximum root uptake concentration is the rainwater level, the root zone concentration may significantly exceed the effluent concentration during the crop season. This is because the ions that contribute to EC are left behind in the soil when water is absorbed by crops, increasing the EC of the remaining soil water, as was also previously observed by Siyal et al (2013) and Fujimaki et al (2006). In our simulations, this is observed only in the soil directly above a drain, and not in the root zone soil midway between drains, because little effluent reaches the root zone midway between drains. Such large EC levels will decrease to ambient levels by the start of the following crop season, and thus should not result in long-term salt accumulation. Hence, crop contamination that might occur during any one irrigation period would likely not carry over to the following years, regardless of whether they accumulate in the root zone (EC) or not (Cl:Br). However, if the effluent is very saline, there is a possibility that the irrigation system causes crops directly above drains to experience salinity stress during dry years (Heidarpour et al., 2007).

3.3 Reactive solute transport

The mass balances of carbamazepine and the generic tracer are presented in Table 3. No downwards vertical discharge of carbamazepine from the model domain occurs, due to the effects of adsorption and biodegradation. For the generic tracer, the amount of downwards vertical solute discharge from the domain is finite but essentially negligible. Furthermore, less than 10% of the generic tracer and 1% of carbamazepine are taken up by the crop (Table 3). Hence, the model shows that most of the subirrigated solutes, whether tracers or carbamazepine, is advected laterally out of the domain along with regional groundwater flow.

Small amounts of carbamazepine spread to the sampling points directly above and below drains within four years. Comparatively much smaller amounts of carbamazepine spread to the soil and aquifer midway between drains. The mass influx of the generic tracer and carbamazepine at the drains varies between 0mg to 10^{-2}mg per day during subirrigation. The generic tracer mass flux at the downstream lateral boundary reaches 10^{-3}mg per day by the first crop season, and remains above that level for most of the rest of the simulation. However, carbamazepine mass fluxes at the downstream lateral boundary never exceeds 10^{-3}mg per day, and only reaches 10^{-4}mg per day during the second crop season. No significant levels of the generic tracer nor carbamazepine reaches the bottom boundary even after four years. In the phreatic groundwater, carbamazepine spreads less than 3m from the drains after four years. Hence, the transport of carbamazepine is highly limited compared to that of the tracer.

When drainage occurs during the crop season, the concentrations of tracer and carbamazepine in the drained water are similar. When drainage occurs outside the crop season, the concentration of carbamazepine drained is larger than the tracer, despite carbamazepine undergoing biodegradation, because adsorption retains the carbamazepine plume close to the drains. This explains why more carbamazepine than tracer mass is drained in total (Table 3), even though carbamazepine biodegrades in the soil but not the tracer. This means that immobile contaminants are more likely to be recovered during drainage than mobile contaminants. Hence, highly immobile and persistent contaminants can potentially be prevented from accumulating in the agricultural soil by draining it during wet periods, both during and outside the crop season.

Differences in the mobility of the tracer and carbamazepine result in different spatial distributions of root solute uptake. The root zone directly above drains receives the most solutes due to capillary rise. Table 3 shows that the relative amounts of total root solute uptake differ with the horizontal position and depth of the roots. For the tracer, the roots at the nodes at 0.2m depth take up roughly four times as much tracer as the roots at 0.6m depth, even though solute concentrations are 10% higher at 0.6m depth, because the root density is four times larger at 0.2m depth. The roots directly above drains take up roughly four times as much tracer as the roots midway between drains, because solute concentrations directly above drains are two to four times larger. These patterns are also observed for carbamazepine. Carbamazepine uptake directly above drains is over ten times that midway between drains, because adsorption arrests the spreading of carbamazepine. Therefore, most root uptake of tracer and carbamazepine occurs directly above drains, and this spatial heterogeneity in root solute uptake is stronger for less mobile and less persistent solutes.

Unlike the tracer, for which appreciable levels of root solute uptake, horizontal discharge and vertical discharge were observed, very little carbamazepine had been taken up by crops or discharged from the domain by the end of the simulation (Table 3). For carbamazepine too, concentrations in the root zone essentially reset on an annual basis due to the annual precipitation surplus, which means that the crop solute uptake of carbamazepine is not expected to increase with the number of years of operation of the subirrigation system. Since less than 1% of the irrigated carbamazepine is taken up by crops, and since most crop solute uptake of carbamazepine is concentrated in roots directly above the drains, carbamazepine levels in crops is likely negligible everywhere except directly above drains, where it is present in very small concentrations. This also implies that the system studied here might not be suitable for irrigation in a climate with an annual precipitation shortage.

Here we highlight the key findings of the experimental data analyses (Narain-Ford et al, 2022) and how they relate to the results of the model introduced in this study. Of the 55 CECs found in the wastewater effluent but not in the control field beside the experimental plot, the fraction of CECs classified as mobile and persistent (MP CECs) is 19/55. Next to the surface water stream located 5m beside the agricultural plot, and in deep groundwater under the agricultural plot, the average detected concentrations of MP CECs as a fraction of the effluent concentrations was smaller than 0.01 in deep groundwater and in the surface water stream on all sampling occasions. An exception was that in the middle of the crop season during the drought of 2018, somewhat elevated levels of MP CECs were detected next to the surface water stream, but not in deeper groundwater. Less than 1% of MP CEC mass reaches deeper groundwater in the field site. In addition, from the field data we observe that the concentrations of all CECs in the root zone, regardless of mobility or persistence, reset to background levels by the start of the following year's crop season, except at very close distances from the drains, or if there is a period of severe drought such as that of 2018. In the field experiment, carbamazepine levels in crop samples obtained in September 2019 (a hydrologically typical year with no drought) based on solid-phase extraction of 30g plant material were everywhere below the detection limit (6ng/L), including for crops directly above drains. All of the above experimental results agree with the introduced model.

3.4 Effects of regional groundwater fluxes on solute transport

Despite the large effect of the regional head gradient on the breakthrough curves, varying the regional head gradient had little effect on the relative solute mass balance (Table 3). Most importantly, in the simulations with high and zero regional head gradients, the main findings of the base model continue to apply: Crop solute uptake is around 10% for tracers and 1% for carbamazepine and occurs primarily directly above irrigation drains, little tracer mass seeps to deeper aquifers, no carbamazepine seeps to deeper aquifers, little solute mass is drained by the drainage system, and the rest of the irrigated solute mass is discharged horizontally out of the simulated domain, in agreement with field observations (Narain-Ford et al., 2022). Therefore, uncertainty in the regional head gradient does not undermine the findings of the study. In fact, we have shown that while the regional head gradient may affect the rate at which CECs are laterally advected, it does not significantly change our conclusions relating to crop contamination risk and the overall mass balance of the CECs. Since there is an annual precipitation surplus, the average annual transport direction of the CECs can only be laterally downstream and/or vertically downwards, regardless of the regional groundwater head gradient. Since the calibrated lateral flow resistance is effectively much lower than the downwards flow resistance, the transport of CECs is primarily lateral even in the absence of regional flow. This explains why the relative solute mass balance is not sensitive to the regional head gradient.

When the regional head gradient is low, the natural groundwater level is deeper, therefore requiring a higher irrigation flux to maintain target water table levels. In the base model (0.0014m/m head gradient), 16.3mg of carbamazepine was irrigated over the four year simulation period (Table 3). The corresponding values in the model with no regional flow (0 m/m) and high regional flow (0.0022m/m) were 25.4mg and 10.9mg respectively, which implies that even such a large uncertainty in regional head gradient would translate only to a 50% difference in the irrigated solute mass. Nevertheless, in practice it is easy to monitor the absolute volume of water used by the irrigation system. Therefore, knowledge of the relative solute mass balance, which is not sensitive to the regional head gradient, is sufficient to evaluate the fate of the irrigated CECs.

4 Discussion

4.1 Limitations of the base numerical model

The numerical model is unable to describe the rapid rise in CEC concentrations at two observation points directly beneath an irrigation pipe. The measurement points located 0.8m and 1.3m beneath drain level, directly below the drains, are among the deepest measurement points where effluent has been detected in the field site. For these two measurement points, the base model consistently and significantly underestimates concentrations of Cl:Br, EC, and carbamazepine (Supplementary material Figure S2.1, S2.3, S2.5 e and f). The fact that this is observed experimentally for all three modelled compounds, which have vastly different adsorption, biodegradation, and accumulation behavior in the soil, suggests a physical rather than chemical reason for the discrepancy. A likely explanation is that in the field experiment some solutes are rapidly transported vertically downwards from the pipes by preferential flow due to soil heterogeneity, a mechanism not considered in the model.

Further evidence that vertical preferential flow is likely the cause of the discrepancy is that measured concentrations of carbamazepine at these outlier points are almost identical to measured effluent concentrations, which implies that carbamazepine did not experience significant adsorption and biodegradation while it was transported from the irrigation drains to those points, despite exhibiting retardation across the rest of the domain. The fast downwards transport of solutes described above is consistent with solute transport in large macropores or fractures, which have small pore surface-area-to-volume ratios and hence less adsorption sites, thereby causing effluent to travel vertically downwards much faster than simulated in the model.

In general, the simulations with no regional head gradient result in breakthrough curves that deviate more significantly from field data than the base model, especially at observation points between drains. However, at the two aforementioned observation points where the base model tends to fit poorly with the experimental data (directly beneath drains, 0.8m and 1.3m below drain level), the simulations with no regional head gradient have better fits with the experimental data than the base model. Since the base model has better fits with the data when considering all other observation points, it is unlikely that overestimation of the regional head gradient is responsible for the base model's poor fitting at 0.8m and 1.3m directly beneath drains. The fact that the zero head gradient case exaggerates the extent of downwards transport of solutes also reinforces the hypothesis that vertical preferential flow is the cause of the rapid increase in effluent concentrations 0.8m and 1.3m directly beneath drains in the field.

In this section we have argued that the disagreement between model and field data at these two points is very likely caused by vertical preferential flow. Since the disagreement occurs only for two observation points out of twelve, and since these two points are located at the same direction from the drains suggesting a common unknown cause, we conclude that the model approximates the field situation well in general. Hence, the model successfully captures the general but simplified flow and solute transport patterns observed in the field. This also implies that although most solutes are laterally advected out of the subsurface of the agricultural field, the presence of vertical preferential flow may lead to some downwards leaching, although this adverse possibility was not observed in the experiment (Figure 2)

4.2 Effects of soil heterogeneity on solute fate

Section 4.1 shows that soil heterogeneity may cause solutes to be transported within the soil more rapidly and at higher concentrations. Furthermore, possible clogging of the soil around irrigation drains due to the growth of biofilms, precipitation of minerals, and deposition of particulate matter was observed in the experiment. This may eventually alter the hydrological and biogeochemical properties of the soil, in a spatially heterogeneous manner. Therefore, we further investigate the effects of soil heterogeneity on solute fate, using a simple model of soil heterogeneity. To the generic tracer base model, we add soil heterogeneity with Miller-Miller similitude (Miller and Miller, 1956), where the pressure head and hydraulic conductivity at a certain soil water content are scaled according to a scaling factor m, which is randomly distributed across space in the soil (see Roth (1995) for more information). Roth (1995) found that the spatial structures of soil hydraulic properties (water retention, pressure head, and flow velocity) becomes highly sensitive to random heterogeneity in the scaling factor m if the standard deviation of log10(m), σ_m, is larger than 0.7. Therefore, we simulate the generic tracer base model with 30 random fields of the scaling factor m, with horizontal and vertical exponential autocorrelation lengths of 2m and 0.5m respectively, and with $\sigma_m = 0.25$ and $\sigma_m = 0.75$, to simulate weak and strong heterogeneity respectively.

Summary statistics boxplots of the simulations with weak heterogeneity (Figure 6a) show that the medians and means of the outcomes are very close to the homogeneous base model. The simulated heterogeneity is responsible for less than a half-order of magnitude variation in outcomes. Consider that the 30 random fields simulated can be considered as 30 subplots of a single larger agricultural plot. The statistical outcomes therefore imply that in an agricultural plot with weakly heterogeneous soils, crop solute uptake and saturated zone solute discharge should be spatially heterogeneous but on average be similar to that of a homogeneous soil. However, spatially uneven distribution of crop solute uptake is already a feature of the subsurface irrigation system even in homogeneous soils (Section 3.3), as most crop solute uptake occurs directly above the individual drains, whereas crops located midway between drains take up little solute.

Strong heterogeneity, on the other hand, led to significantly worse environmental outcomes compared to the base model (Figure 6b). On average, about twice as much effluent had to be irrigated into the system to maintain the target groundwater levels. More saturated zone solute discharge occurred, while less solute was drained away. Therefore, most of the additional irrigated effluent caused by soil heterogeneity is discharged into the saturated zone through high conductivity channels. Consequently, solutes and CECs in the saturated zone has a farther reach and larger mass, which may render subirrigation unfeasible in strongly heterogeneous soils. However, the root solute uptake was on average similar as in homogeneous soils, because the crop solute uptake is limited by the water requirements of the crop. It is also noteworthy that of all the key simulation outcomes, root solute uptake had the lowest variance across realizations, for both mild and strong heterogeneity. Therefore, even though spatial heterogeneity was present both in the root zone and beneath the root zone, most of its effects were concentrated in the saturated zone.

Altogether, this analysis shows that crop contamination is on average similar in homogeneous and heterogeneous soils, even under strong heterogeneity. In contrast, as we have shown, groundwater contamination in the saturated zone is more sensitive to soil heterogeneity. This is consistent with our observation in Section 4.1 that field sampled solute concentrations in the saturated zone displayed more behavior that was not explained by the homogeneous model, than sampled concentrations in the unsaturated zone.

Although weak soil heterogeneity had little impact on solute mass balance on average, it had a large effect on solute breakthrough curves at observation points, due to the possibility of streamlines bypassing or concentrating around observation points. Even under weak heterogeneity, peak tracer concentrations at observation points (both in the saturated zone and root zone) varied by over a factor of 10 across the individual realizations of heterogeneous simulations (not shown). This agrees with heterogeneity being an explanation for the discrepancies observed in Section 4.1.

Whether subirrigation with treated wastewater in strongly heterogeneous soils poses a significant environmental problem, depends on the ability of contaminants to undergo further in-situ bioremediation in the groundwater, and whether the groundwater will be abstracted at downstream locations for other uses. Nevertheless, Figure 6b shows that in the strong heterogeneity scenario, AbsContIn, AbsContOutRight, and log10AbsContOutDown (see Figure 6 caption for variable name explanations), all approximately double. Therefore, these outcomes all appear to be proportionally related. Indeed, across the 30 simulations with strong heterogeneity, AbsContIn and AbsContOutRight were related with an R-squared of 0.99. Similarly, AbsContIn and log10AbsContOutDown were related by an R-squared of 0.46. Hence, even in strongly heterogeneous soils, it is possible to limit lateral contaminant discharge to the groundwater by simply lowering the irrigation flux, though water availability for crops may also decrease accordingly. Future studies could investigate the effects of soil heterogeneity on geochemically reactive CECs. In physically heterogeneous soils, the spatial distribution of adsorption sites and microbes responsible for biodegradation will also be heterogeneous, leading to a highly complex problem that is beyond the scope of this paper. The potential for short or long term soil structure changes due to biomass growth and clogging, and methods to prevent or remediate them, should also be investigated in future studies.

4.3 Environmental impact beyond the agricultural plot

We have shown that most of the effluent leaves the crop field by lateral advection. In heterogeneous soils, where more water has to be irrigated to maintain target groundwater levels, most of the additional solute mass introduced into the subsurface leaves the plot through lateral advection. Beyond the agricultural field, the downwards transport rate of the CECs would decrease further, as the absence of irrigation reduces the downwards water fluxes. Therefore, most of the CECs that leave the crop field likely remain in the phreatic zone until they are discharged into a surface water body, such as a stream or river.

In Ternes et al's (2007) study of 54 CECs present in treated domestic wastewater in Germany, carbamazepine was found to be one of the most persistent and mobile CECs in the soil and phreatic zone. Therefore, the spreading of many other CECs in the environment is likely to be even more limited than what we have observed for carbamazepine. Any CEC that is more mobile and persistent than carbamazepine would exhibit behavior that is intermediate between carbamazepine and tracers, both of which have been discussed in this study. Since the effluent would have likely been directly discharged into surface water in the absence of the subirrigation system, the subirrigation system therefore possibly reduces the adverse environmental impacts associated with treated wastewater discharge, thereby leading to better surface water quality in the vicinity of the treatment plant. Still, some attention should be paid to possible ecotoxic effects of transformation products (Reemtsma et al., 2016).

4.4 Potential for further adoption

Around 35% of agricultural land in the Netherlands (NL) is currently drained using subsurface drainage pipes. In agricultural land, drainage pipes are mostly 0.9m to 1.2m deep, but can vary slightly depending on factors such as soil type, or due to processes such as soil settling (Maasop & Schuiling, 2016). The placement of the drains (depth and inter-drain distance) beneath agricultural lands is designed according to national standards in which the local soil type and topography are factored (Cultuurtechnische Vereniging, 1988). Since the drainage response of the drainage systems are nationally similar, the wetting patterns in the soil when subirrigation is performed with similar fluxes should not be too different. Under similar hydrological and hydrogeological conditions, this leaves the crop type as likely the key determining factor in crop contamination risk, through several aspects of crop physiology that could be studied in further detail in future research: the spatial distribution of its roots, its disposition towards solute uptake, and its water requirements.

Drought is another factor that should be considered in the adoption of the proposed subsurface irrigation approach with treated wastewater. Our study area in the Netherlands is vulnerable to occasional drought, which occurred in 2018–2019 during our study period. In our model, of the total tracer (and carbamazepine) solute mass taken up by crops over the four year simulation period (Table 3), 21% (21%) occurred in 2016-2017, 35% (43%) occurred in 2018, and 44% (36%) occurred in 2019. Simulated crop solute uptake in 2019 was similarly high as in 2018 because some of the solutes irrigated in 2018 remained in the topsoil even by the start of the crop season in 2019, whereas they would have been flushed away by precipitation in a normal hydrological year. Therefore, soil water concentrations of contaminants should be monitored more closely during the occasional drought year, and the following year. If such precautions are taken, then crop contamination risks are manageable in regions that experience occasional drought. In regions with perpetual drought, the proposed subsurface irrigation approach is not recommended altogether, as large amounts of water (and CECs) will have to be introduced into the subsurface, and there is insufficient precipitation to flush out CECs from the root zone during the non-crop season, possibly leading to accumulation of persistent CECs in the root zone. A relative precipitation shortage caused by farming crops with large water requirements, instead of by drought, may pose similar risks. In this regard, note that this study was conducted on maize agriculture, which is one of the most water-intensive common crops in the Netherlands, aside from tree fruits (LAGO, 1984).

A new EU regulatory framework intends to stimulate and regulate direct reuse of treated domestic wastewater for irrigation purposes (European Commission, 2020). A risk management plan (Maffetone & Gawlik, 2022) and irrigation water quality requirements (Alcade Sanz & Gawlik, 2017) is part of the EU regulation and includes an analysis of the effect of water reuse on farmers, soil, groundwater and ecosystems. As water reuse through subirrigation is a special form of irrigation, process-based knowledge and modeling tools as presented in this study are required to identify potential risks and take appropriate precaution measures.

5 Conclusion and outlook

The results and analyses of the model have provided additional understanding of the physical processes of CEC transport under subsurface irrigation using marginal water in phreatic aquifers, a novel method of managed aquifer recharge in regions with annual precipitation surpluses. In the long term, effluent contamination in both the root zone and the phreatic zone within the agricultural plot becomes periodically steady-state, with larger concentrations during the crop season and nearly background concentrations during the drainage season. No long-term accumulation of CECs in the root zone occurs due to the annual precipitation surplus. Despite the possibility of minor crop contamination by mobile contaminants, most of the crop solute uptake occurred for crops located directly above drains, whereas crops located midway between drains are barely exposed even to tracers. Combined application with intercropping, where non-food crops are placed above drains, could be the next step in the development and adaptation of subsurface irrigation with treated wastewater. Transport of contaminants from the phreatic zone to deeper groundwater accounts for a negligible portion of the solute mass balance for the simulated scenarios. Around 90% of the effluent tracer leaves the simulated domain along with lateral advection by regional groundwater flow, so most tracers in the effluent would end up discharged to surface water.

Using the treated wastewater in subsurface irrigation therefore likely leads to better surface water quality, compared to the alternative of direct discharge of the treated wastewater into rivers and canals. Furthermore, such a system of subsurface irrigation with treated wastewater may be implemented using existing subsurface drainage pipes. Using such alternative water resources for agricultural water supply reduces the use of groundwater resources and other sources of freshwater, which are increasingly under pressure (Pronk et al., 2021). Hence, this system has the potential to reduce anthropogenic environmental damage associated with discharge of sewage treatment plant effluent at a low technical difficulty and initial investment cost.

Despite the front-loaded initial investment costs, the reduction in groundwater usage, improvements in the quality of the surface water, and increased crop yields due to improved crop access to water, may lead to economic returns and wider adoption in the long term. However, this should not lead to a reduced effort to minimize the contaminant load of wastewater discharged into the environment, as the capacity of the soil and soil microbes to biodegrade CECs remains uncertain and difficult to quantify exactly. New EU rules on treating urban wastewater will lead to improved quality of treated wastewater, thus further reducing the potential environmental impact of using it for irrigation. Of all possible adverse solute fates, the most sensitive to CEC mobility is crop contamination, as the CEC must rise to the root zone within a single crop season for this to occur. This did not occur in this study even with tracers, implying that mobility causes less of an environmental contamination risk than persistence. We therefore recommend that future improvements to water treatment technologies focus on removing contaminants that persist in the soil, to prevent their proliferation in the wider environment. Immobile but persistent CECs spread less easily and will be removed from the soil to a larger degree during drainage, whereas less persistent CECs will more likely be remediated within the soil. Focusing on persistent contaminants during primary treatment would thus maximize the degree of complementarity between wastewater treatment technology and nature-based secondary treatment solutions such as the irrigation and drainage system introduced in this paper.

The models in this study were constructed loosely within the context of an experimental field in the Netherlands, and are meant to apply to similar subsurface irrigation and drainage systems in general. A sensitivity analysis of the model will reveal the environmental and crop safety consequences of adapting the system to other regions with various hydrogeological and climatic properties. Another avenue for further research may be to investigate the ecotoxicology and spreading of biodegradation metabolites, which may not be present in the original CEC cocktail.

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Table 1: Parameters used in the simulations. Soil 1 is present from field level to 0.2m depth. Soil 2 is present from 0.2m to 0.6m depth. Soil 3 makes up the bulk of the aquifer. Sources of parameter values: 1) Heinen et al (2020); 2) Dutch geological survey (TNO); 3) Ernst and Feddes (1979); 4) Calibrated to experimental data; 5) Estimates

Parameter	**Value**	**Source**
θr (soil 1,2,3) Residual saturation	0.01	1
θs (soil 1,2,3) Maximum saturation	0.42	1
α (soil 1,2,3) [1/m] van Genuchten soil water retention parameter	2	1
n (soil 1,2,3) van Genuchten soil water retention parameter	1.5	1
Ks (soil 1) [m/day] Saturated conductivity	0.5	1
Ks (soil 2) [m/day] Saturated conductivity	2	1
Ks (soil 3) [m/day] Saturated conductivity	5	1
L (all soils) Tortuosity parameter	0.5	1
Regional head gradient	0.0014	2
Reference depth Z [m] Deep drainage boundary parameter	0	3
A [m/day] Deep drainage boundary parameter	0.0025	3

B [1/m] Deep drainage boundary parameter	-1.250	3
Water table depth at downstream boundary [m]	1.6	4
Conductivity of downstream boundary [m/day]	0.02	4
Irrigation drain conductivity [m/day]	0.025	4
Irrigation pressure [m]	0.3	4
Drainage backpressure [m]	0.3	4
Longitudinal dispersivity D_l [m]	0.2	5
Transverse dispersivity D_t [m]	0.02	5
Soil bulk density ρ [kg/L]	1.5	5

Table 2: Water mass balances over the entire duration of the modelled scenarios. Water balances are expressed in millimeters for comparison with rainfall volumes. Field data for entries with a dash were not measured.

Water balances	**Irrigated water (mm)**	**Drained water (mm)**	**Crop water uptake (mm)**	**Rainfall (mm)**	**Lateral flow in (mm)**	**Lateral flow out (mm)**	**Vertical flow out (mm)**
Field data	1828	280	–	–	–	–	–
Base model	1895	269	1916	3152	2785	4041	1004
No regional flow model	2997	39	1916	3152	0	2870	831
High regional flow model	1218	728	1916	3152	4339	4722	1127

Table 3: Solute mass balances over the entire duration of the modelled scenarios. Since each observation node is a zero-dimensional object with no volume, the crop solute uptake at each node is dimensionless. Therefore, the crop solute uptake values per observation node are normalized (separately each for carbamazepine and the generic tracer) to the highlighted values.

Solute mass balances	**Irrigated solute (mg)**	**Drained solute (mg)**	**Crop solute uptake (mg)**	**Crop solute uptake fraction**	**Horizontal solute discharge (mg)**	**Vertical solute discharge (mg)**	**Biodegraded solute fraction**
Base model (generic tracer)	16.3	0.520	1.75	0.107	12.7	0.0330	0
No regional flow model (generic tracer)	25.4	0.176	2.40	0.094	14.1	2.82	0
High regional flow model (generic tracer)	11.1	0.909	1.23	0.111	9.29	0.0037	0
Base model (Carbamazepine)	16.3	0.689	0.136	0.00833	1.46	0	0.666
No regional flow model (Carbamazepine)	25.4	0.177	0.214	0.00840	1.07	0.00116	0.707
High regional flow model (Carbamazepine)	10.9	1.03	0.0941	0.00860	1.41	0	0.672

Total root solute uptake at observation node	Directly above drain		Midway between drains	
	0.2m depth	0.6m depth	0.2m depth	0.6m depth
Base model (generic tracer)	1	0.291	0.281	0.0818
No regional flow model (generic tracer)	1.29	0.355	0.306	0.0879
High regional flow model (generic tracer)	0.678	0.211	0.188	0.0529
Base model (Carbamazepine)	1	0.663	0.0566	0.0487
No regional flow model (Carbamazepine)	1.76	1.10	0.0052	0.0056
High regional flow model (Carbamazepine)	0.558	0.349	0.0896	0.0734

859

860

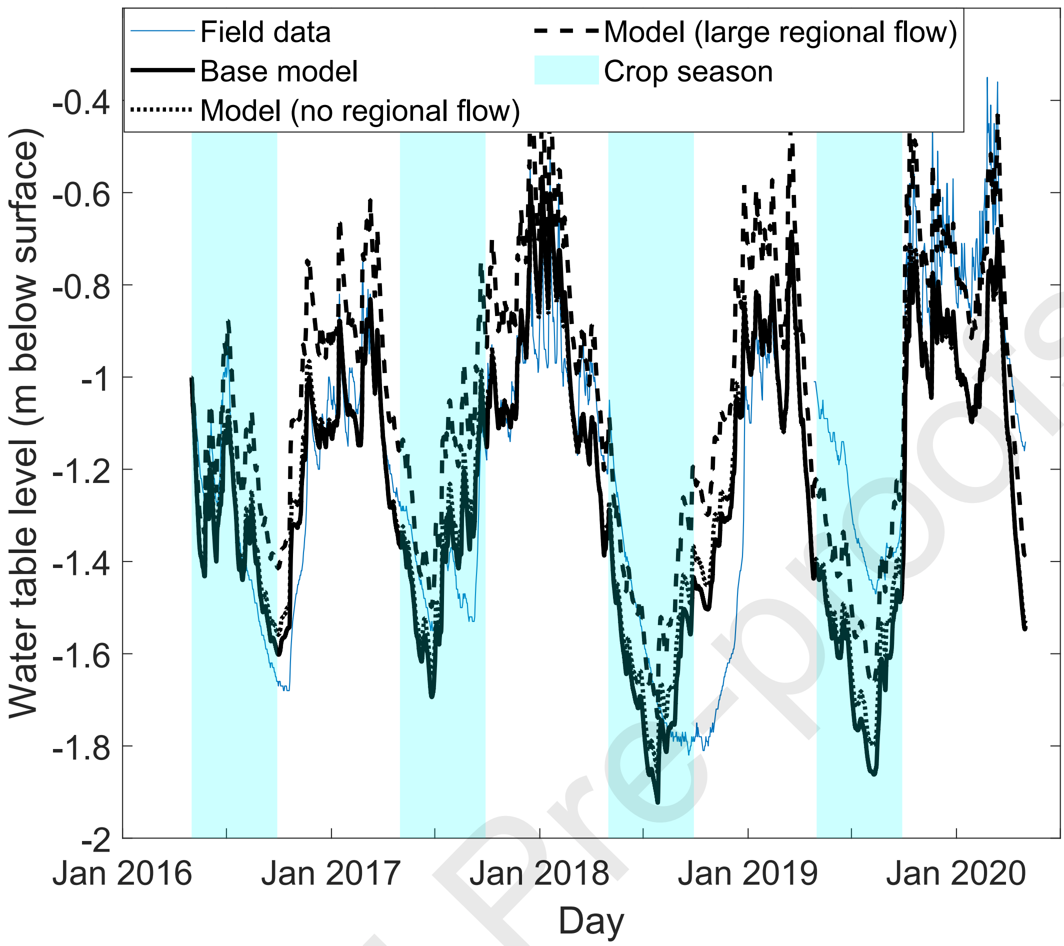
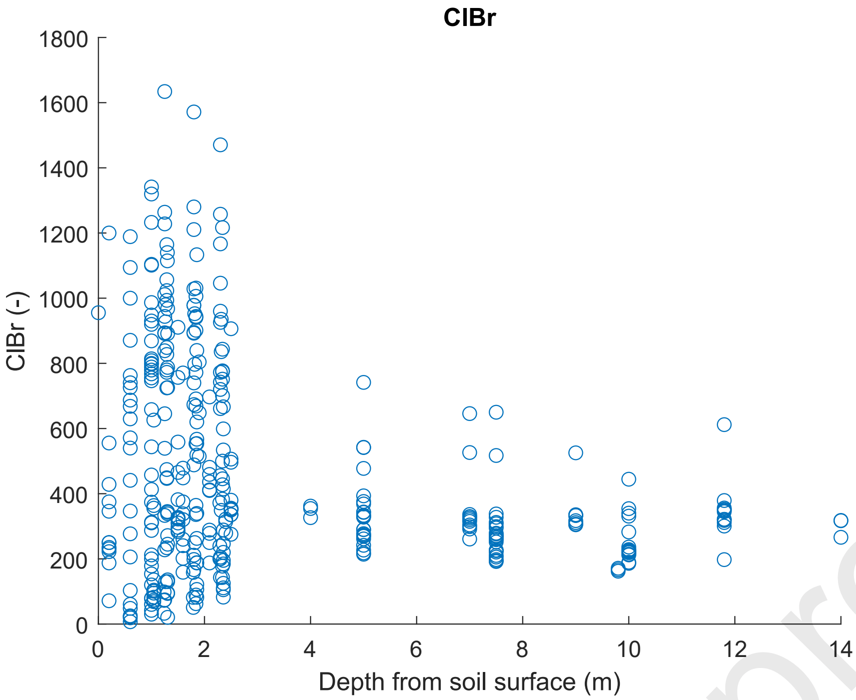
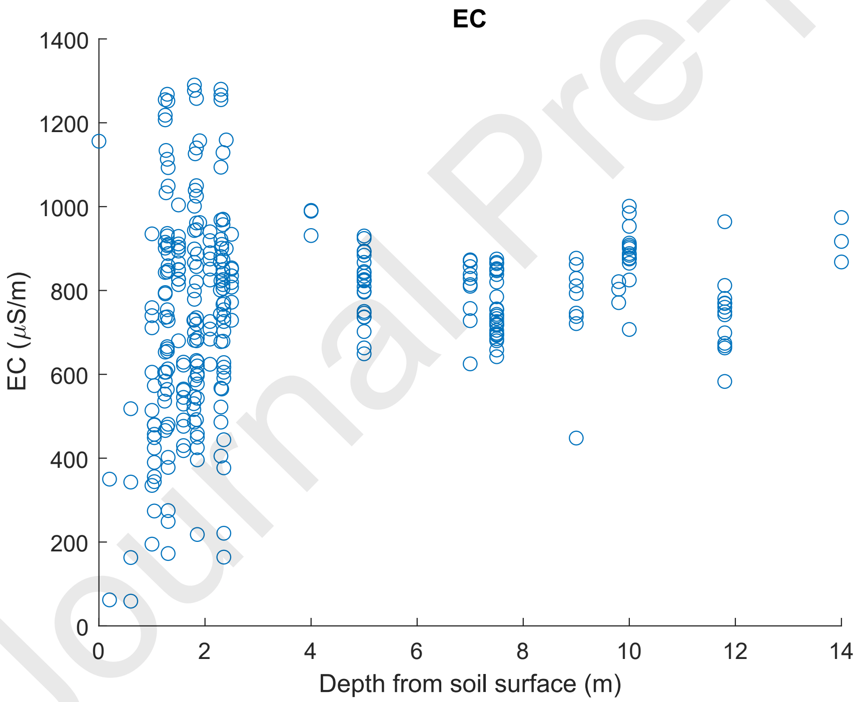


Figure 1: Groundwater levels from reference field data and the model without irrigation and drainage.



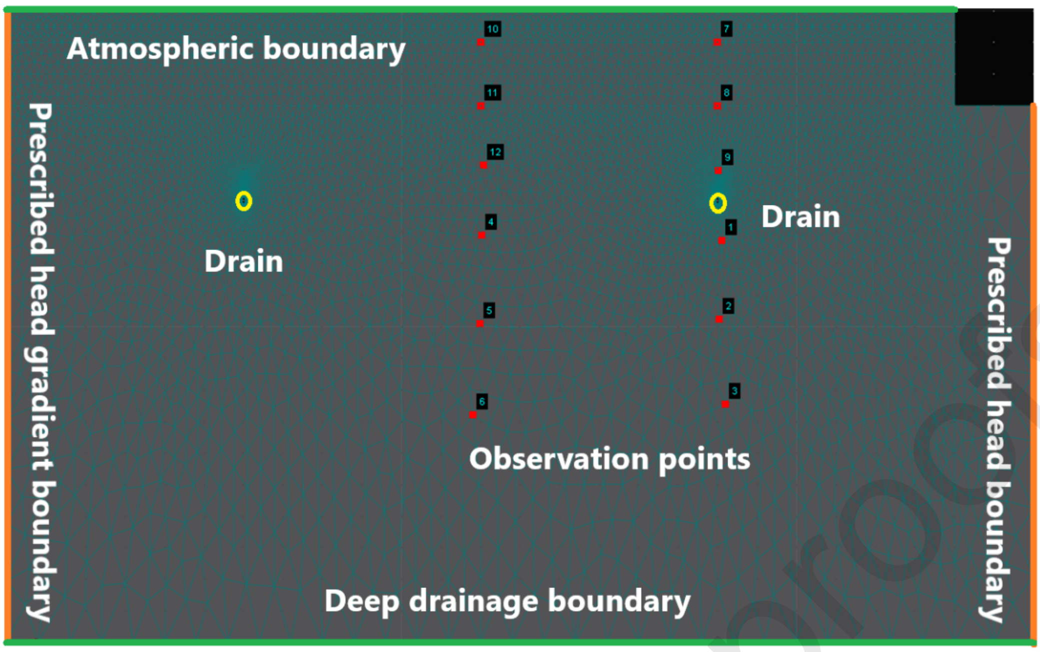
a



b

Figure 2: Scatterplots of (a) Cl:Br and (b) EC of subsurface water obtained from field measurements taken at various locations and times over the experimental period.

a



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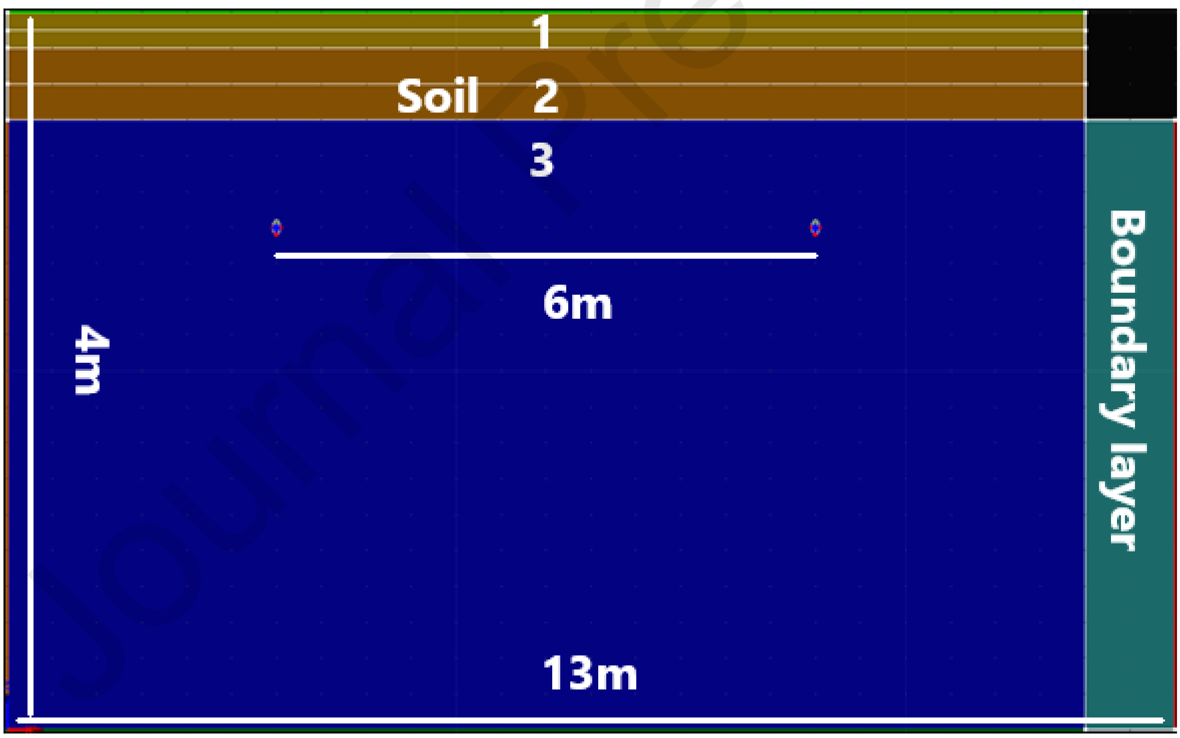
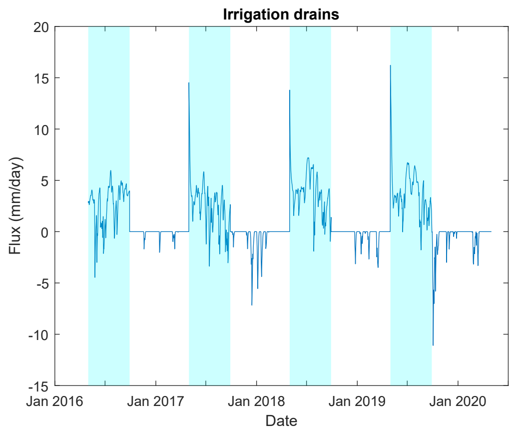
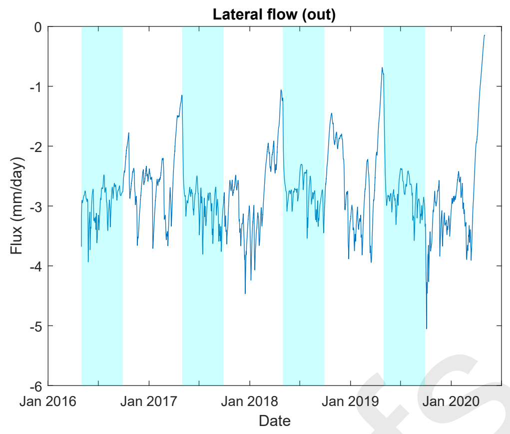


Figure 3: (a) Cross-section of the numerical model domain showing the locations of observation points, the drains, and the four boundary conditions. Regional groundwater flow flows from left to right. (b) Domain of the numerical model showing soil types and root zones. The four brown layers are the root zone layers (Section 2.2).



a.



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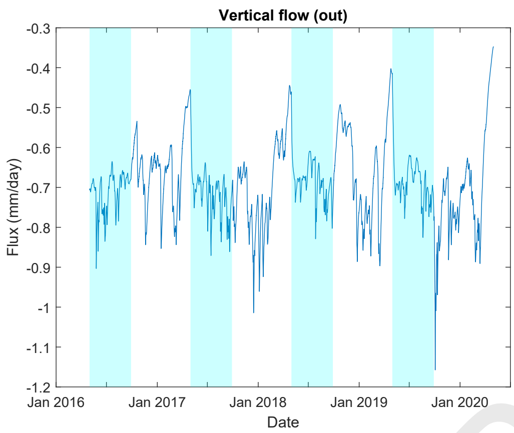

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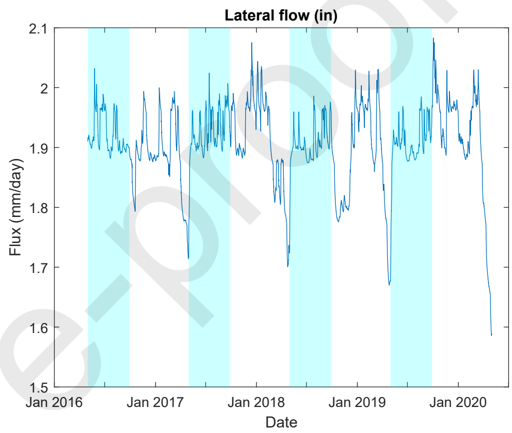
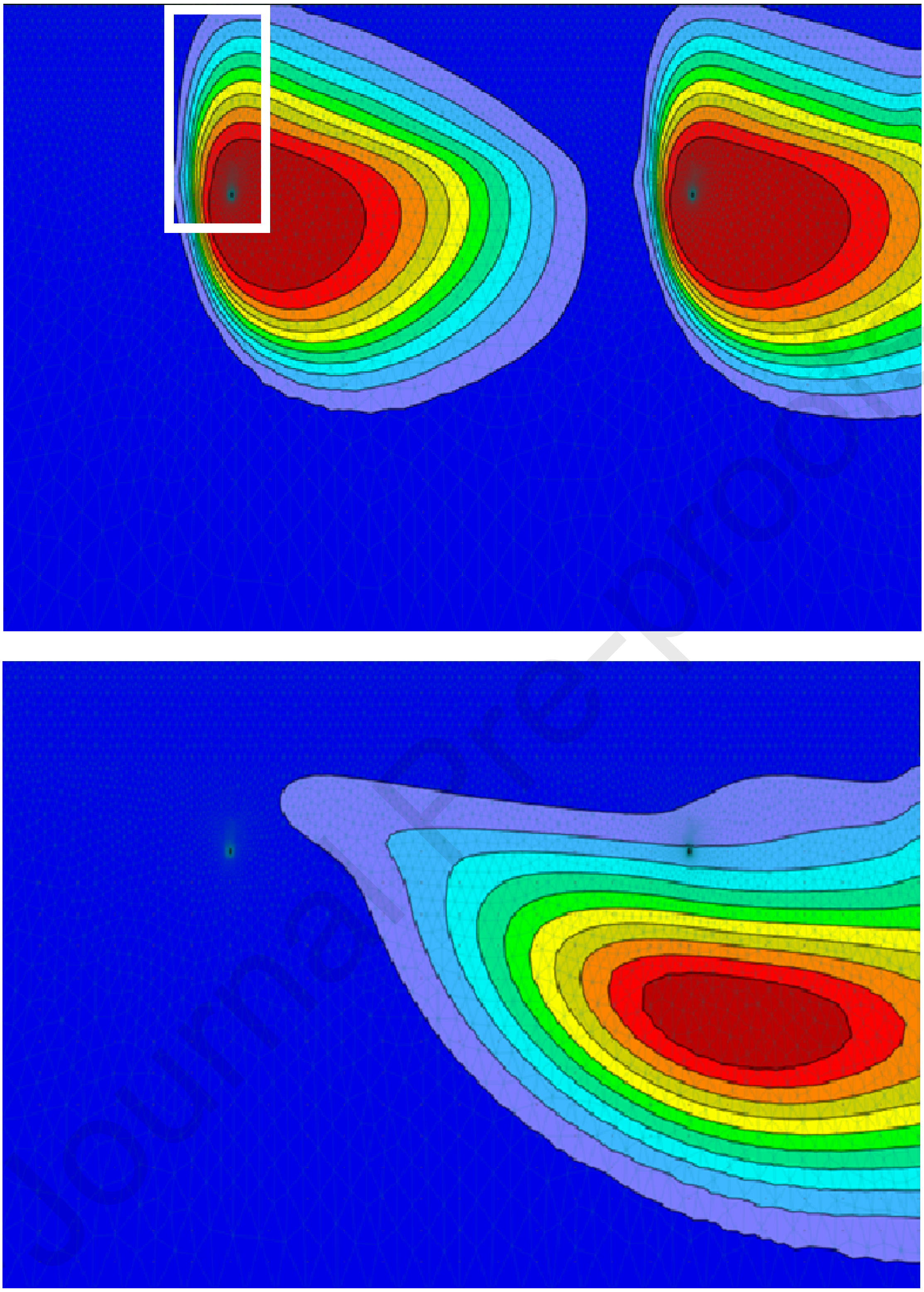


d

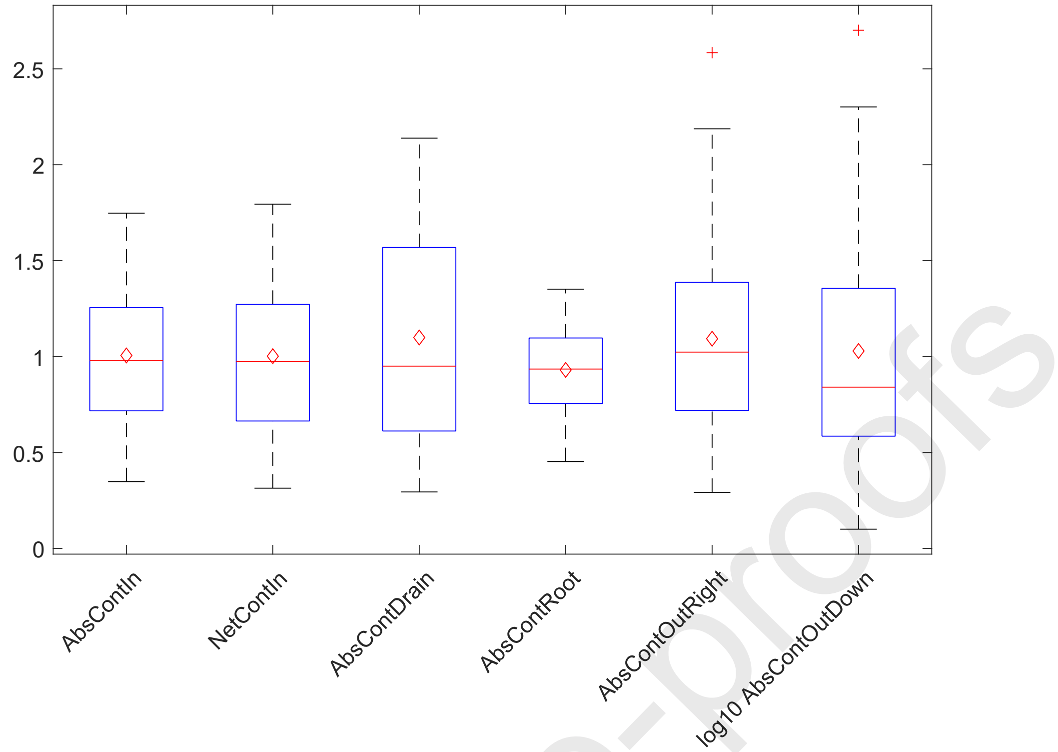
Figure 4: Water fluxes for the base model. Positive fluxes are fluxes into the domain, and negative fluxes are fluxes out of the domain.



A

b

Figure 5: (a) Tracer plume in the base model at the end of the first crop season. The white box illustrates approximately the area of the plume that may potentially be recovered by the drains after the crop season. (b) Tracer plume in the base model at the end of the fourth drainage season. The plume encompassing the drain on the right was originally injected by the drain on the left during the fourth crop season. The continuum of colors indicate tracer concentrations, with a scale normalized to the irrigated (red) and background (blue) concentration.



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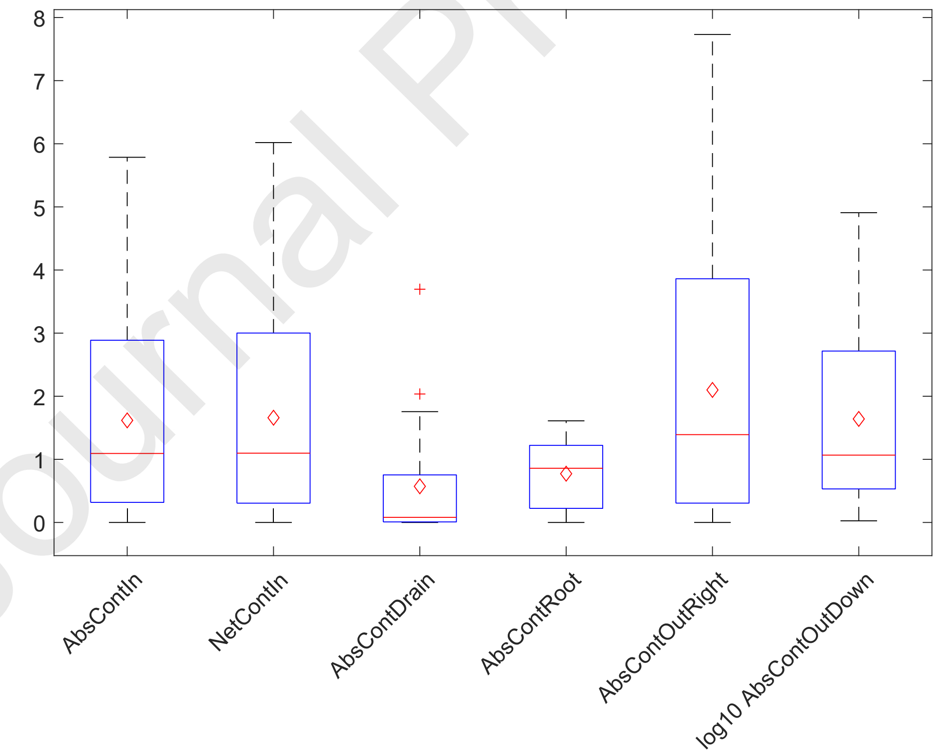

b

Figure 6: Summary statistics boxplots of the 30 simulations with (a) weak and (b) strong spatial heterogeneity, normalized against the values of the homogeneous base model. The boxes represent the interquartile range, whiskers represent 1.5x interquartile range, the plusses are outliers, the red lines are the medians, and the diamonds are means. Variable names are AbsContIn = absolute injected solute mass; NetContIn = net injected solute mass (injected – drained), ContRoot = total root solute uptake, AbsContDrain = absolute drained solute mass, AbsContOutRight = absolute horizontal solute discharge,

AbsContOutDown = absolute vertical solute discharge. Statistics of log10(AbsContOutDown) are displayed because the raw AbsContOutDown data is observed to be exponentially distributed.

Highlights:

New subsurface irrigation and drainage method that raises the phreatic water table.

New method synergistic with marginal water irrigation: crops not directly exposed.

Tool for irrigation, freshwater conservation, and wastewater treatment and disposal.

Crop contamination risk is low, except by mobile contaminants in very dry years.

Crop and wider environmental contamination risk largest for persistent contaminants.

CRediT authorship contribution statement

Darrell Tang: Conceptualization, Formal analysis, Investigation, Methodology, Software, Writing - original draft

Sjoerd van der Zee: Conceptualization, Funding acquisition, Supervision, Writing - review & editing

Dominique Narain-Ford: Investigation

Ge van den Eertwegh: Investigation

Ruud Bartholomeus: Conceptualization, Funding acquisition, Investigation, Supervision, Writing - review & editing

Abstract

Managed phreatic zone recharge with marginal water, using (existing) drainage systems, raises the water table and increases water availability for crops. This is a newly developed method of freshwater conservation and marginal water treatment and disposal, but risks crop and environmental contamination. The fate of contaminants of emerging concern (CECs) within the irrigated water is addressed. We introduce numerical and analytical models, inspired loosely by a field site where treated domestic wastewater is used for subsurface irrigation. The treated wastewater would otherwise have been discharged into rivers, thereby spreading downstream. Model results show that minimal amounts of CECs are transported to deeper aquifers. Crops are not contaminated, except during dry years where small amounts of mobile CECs rise to the root zone, but then only directly above each irrigation drain. Under an annual precipitation surplus, less-mobile solutes are thus unlikely to ever enter the root zone. The primary mechanism of solute transport is lateral advection within the phreatic aquifer. Despite spatio-temporal heterogeneity in water flux magnitudes and directions, contaminant retardation does not significantly alter mass balance outcomes, only how fast it gets there. Therefore, persistent CECs pose the greatest risks, though overall environmental and crop contamination risks appear low. To maximize complementarity with subsurface irrigation systems, future advances in water treatment technologies should focus on removing persistent CECs. However, the system may be unsuitable for climates with annual precipitation shortages, as CECs may accumulate in the root zone and crops.