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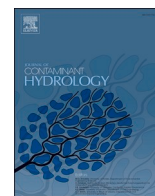
Publication Date

2022-04-01

DOI

10.1016/j.jconhyd.2022.103964

Peer reviewed



Potential effects on groundwater quality associated with infiltrating stormwater through dry wells for aquifer recharge

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ARTICLE INFO

Keywords:

Stormwater infiltration
Dry wells
Groundwater quality
Vadose zone modeling
Aquifer recharge
Urban stormwater quality

ABSTRACT

Dry wells (gravity-fed infiltration wells) have frequently been used to recharge aquifers with stormwater, especially in urban areas, as well as manage flood risk and reduce surface water body contamination from stormwater pollutants. However, only limited assessment of their potential adverse impacts on groundwater quality exists. Dry well recharge can bypass significant portions of the filtering-capacity of the vadose zone. Stormwater and groundwater monitoring data and analysis of transport of a wide range of historic and current-use stormwater chemicals of concern is lacking. To address these gaps, two dry wells were constructed with vegetated and structural pretreatment features to assess the likelihood of stormwater contaminants reaching the aquifer. We monitored, assessed, and compared the presence of contaminants in stormwater to water quality in the vadose zone and shallow groundwater after it passed through the dry well. The dry wells were installed at a suburban residential and at a suburban commercial site. The selected sites were overlying a regional, unconsolidated, and highly heterogeneous alluvial aquifer system. Stormwater, vadose zone, and groundwater samples were collected during five storms and analyzed for over 200 contaminants of concern. Relatively few contaminants were detected in stormwater, generally at low concentrations. Prior to stormwater entering the dry well, 50–65% of contaminants were removed by vegetated pretreatment. In groundwater, metals such as aluminum and iron were detected at similar concentrations in both upgradient and downgradient wells, suggesting the source of these metals was not dry well effluent. Naturally occurring metals such as chromium and arsenic were not detected in stormwater but were found at elevated concentrations in groundwater. A modeling assessment suggests that the travel time of metals and hydrophobic organic contaminants to the water table at these sites ranges from years to centuries, whereas water soluble pesticides would likely reach the water table within days to months. The modeling assessment also showed that more vulnerable sites with higher fraction of alluvial sands would have much shorter contaminant travel times. However, none of the contaminants assessed reached concentrations that pose a risk to human health across the scenarios considered. No evidence was found, either through direct measurements or vadose zone modeling, that contaminants present in suburban stormwater degraded or would degrade groundwater quality at the studied sites and site conditions. Future work is needed to address emerging contaminants of concern.

1. Introduction

The necessity of water resources stewardship has been brought into sharper focus by the impacts of climate change over the past decade. In

many parts of the world, more extreme weather events, such as flooding and drought, have motivated water managers as well as local and state government agencies to re-examine water management policies and practices (Chauhan et al., 2021; Dillon et al., 2019). In the United States

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<https://doi.org/10.1016/j.jconhyd.2022.103964>

Received 24 October 2020; Received in revised form 8 January 2022; Accepted 27 January 2022

Available online 2 February 2022

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of America for example, the impacts of drought and climate change on the availability of water resources have led to the adoption of the Sustainable Groundwater Management Act in California, legislation enacted to minimize overdraft and work toward sustainable groundwater use (<http://groundwater.ucdavis.edu/SGMA/>).

Urban stormwater is one source of water for recharge, but there are a number of challenges to optimizing its use. In some areas, a low-permeable subsurface (e.g., clay), may prevent or limit recharge. Dry wells, large diameter vertical perforated pipes, can penetrate these barriers and release water into more permeable units within the vadose zone closer to the aquifer. However, concerns about groundwater contamination have prevented dry wells from being widely used in some areas (NASEM, 2015). Metals, pesticides, and organics may be entrained in runoff as it flows over the land surface (Masoner et al., 2019; Stein et al., 2006; Tiefenthaler et al., 2008). Infiltration of this runoff through dry wells releases the pollutants closer to the aquifer, diminishing the attenuation role of the unsaturated zone.

Apart from managing recharge, dry wells are also used to reduce runoff to and pollution of urban creeks (Clark and Pitt, 2007; NRC, 2009). A variety of low impact development practices, including bio-retention cells, swales, and infiltration basins, have been adopted to prevent contaminated stormwater from adversely affecting waterways and aquatic life (Askarizadeh et al., 2015). However, due to the presence of low permeability surface soils and near surface sediments in many arid locations, these practices can be ineffective at infiltrating stormwater. Dry wells are an alternative method to enhance infiltration by allowing water to bypass low permeability layers and facilitating the movement of stormwater into more permeable units of the subsurface. Given the general lack of data and analysis of potential contamination from dry wells, regulators and water managers are sometimes hesitant to support their use.

There are relatively few scientific studies that have evaluated the risk of groundwater contamination associated with dry well use, most of them having been performed over 20 years ago (Edwards et al., 2016). The few longer term (> 5 years) studies that have been performed have indicated that dry wells did not result in appreciable aquifer contamination (Dallman and Spongberg, 2012; Jurgens et al., 2008). In some cases, existing groundwater contamination was diluted by the infiltrating stormwater (LASGRWC, 2010). Pretreatment chambers that capture sediment and associated metals and organics have been reported to be effective at significantly reducing the concentration of contaminants prior to runoff entering the dry well (Wilson et al., 1989; Clark and Pitt, 2007). Additionally, the composition of the vadose zone can have a significant impact on attenuation of metals and organics; some contaminants will sorb strongly in clay layers, while unconsolidated sand and gravel layers have limited attenuation capacity (Bandeem, 1987; Wogsland, 1988; Wilson et al., 1989).

Much of our current knowledge is based on site-specific reports that address specific regulatory requirements, while rigorous, comprehensive scientific experiments are lacking. Studies have analyzed stormwater or groundwater quality but not both; some did not include upgradient and downgradient groundwater monitoring wells (Lindemann, 1999; Adolfson Associates, 1995; Olson, 1987; Barraud et al., 1999; Wogsland, 1988). While some long-term stormwater and groundwater analysis exist for a broad range of contaminants (Wilson et al., 1989; Dallman and Spongberg, 2012), most are limited in the scope of contaminants investigated or in the duration of the period of analysis. Several recent studies have employed vadose zone modeling to quantify dry well performance in terms of stormwater infiltration rates and volumes, but these studies did not model stormwater contaminant transport (Liang et al., 2018; Sasidharan et al., 2018; Sasidharan et al., 2019; Qi et al., 2020). Only one previous study used modeling to assess risks associated with dry well use, but it did not include any field monitoring (Bandeem, 1987).

The goal of this study was to perform an integrated monitoring and assessment study that simultaneously considers performance of

stormwater pretreatment on water quality and potential impacts on the vadose zone and shallow groundwater, with respect to over 200 contaminants of interest, within two representative urban settings. We installed dry wells (with pretreatment) and an associated monitoring networks at an urban residential site and at an urban commercial site. Monitoring included observation of stormwater prior to pretreatment, prior to entering the dry well, in the vadose zone adjacent to the dry well, and both, upgradient and downgradient of the dry well in groundwater. Observations occurred over a two-year period. We analyzed for several contaminant groups including metals, volatile and semi-volatile organics, and also for current-use pesticides (pyrethroids). A vadose flow and transport assessment using HYDRUS was performed to assess potential long-term impacts of the monitoring data collected, and their implication for other sites overlying unconsolidated alluvial sediments. To our knowledge, this is the first study that includes pyrethroids, a widely used class of pesticides commonly detected in urban stormwater sampling in California (Amweg et al., 2006), and the first study to consider fipronil and imidacloprid, stormwater contaminants of emerging concern. This is also the first comprehensive modeling assessment to expand on necessarily intermediate-term, site-specific experimental results, across the pretreatment-dry well-vadose zone-groundwater continuum.

2. Material and methods

2.1. Dry well system and setting

Two dry wells and associated monitoring well networks were constructed at two locations in Elk Grove, California, in the United States. Elk Grove is in the Sacramento Valley, which is the northernmost watershed in California's Central Valley, and has a Mediterranean climate with distinct annual wet and dry seasons. Precipitation normally occurs between the months of October and May, with an average annual rainfall of 47.04 cm (NOAA, 2010). The drought that began in California during 2012 persisted through 2014, 2015, and 2016, the years during which the drywell field study occurred. Elk Grove received approximately 17.8 cm of precipitation during the 2014 water year, 39.4 cm during the 2015 water year, and 37.6 cm during the 2016 water year (Weather Underground, 2016).

Elk Grove lies between the Cosumnes and Sacramento Rivers, and the geology of the area is characterized by fluvial sediment deposition that creates unconsolidated, interbedded layers, consisting of lenses of gravel, sand, silt, clay, and various mixtures of the three (Meirovitz et al., 2017). The groundwater system underlying the Elk Grove area is consequently semi-confined, with the aquifers occurring in sand and gravel layers, and the finer grained layers serving as confining beds, or aquitards. County-wide shallow aquifer water table depths have historically ranged from 6.1 m to 21.3 m below land surface (Department of Water Resources, 1997).

Each dry well was 1 m (m) wide and 14 m deep. One dry well was constructed at the Strawberry Detention Basin (SDB), located within a 70.0 ha (ha) residential neighborhood. Stormwater at this site was collected from a single-family residential community which included a local high school with large areas of turf. The site of the second dry well was the City's Corporation Yard (CY), a 0.26 ha drainage shed that serves as a bus fleet servicing, maintenance and parking facility. Runoff that drained to this dry well system was collected from the rooftop of an office/maintenance building and the hardscape of the expansive bus parking lot. Each dry well consisted of a drilled borehole 1.1 m in diameter containing a 0.76 m diameter cylindrical polypropylene casing approximately 13.7 m deep, with 2.5 cm (cm) perforations along its length. A 10 cm diameter PVC stilling pipe capped at the top and open at the dry well bottom, perforated with 1.3 cm diameter holes spaced at 13 cm intervals on two sides of the pipe, was placed within the casing. The void space between the polypropylene casing and the stilling pipe was filled with 6 mm (mm) diameter pea gravel, as was the annular space

between the borehole wall and the casing. Both dry wells were completed at least 2.5 m above the seasonal-high water table (Fig. 1).

The monitoring wells, located upgradient and downgradient of the dry wells, were completed at the water table. Monitoring wells were 6 cm in diameter and were completed at 76 m bgs at the CY and 58 m bgs at SDB. At SDB, the upgradient well was MW3 while at the CY, it was MW1. Fig. 2 shows the locations of the monitoring wells in comparison to the dry wells at each site. These wells were fitted with pressure transducers to determine the depth to the water table and estimate the local groundwater gradients. A vadose zone well (MW2) was also constructed 4.6 and 7.6 m downgradient of each dry well. During the rainy season, the screen on MW2 at SDB was actually below the water table, so it functioned as a fourth water table well.

The dry well system was composed of three parts: a vegetated pretreatment feature designed to remove suspended solids and associated pollutants; a structural pretreatment feature (a sedimentation chamber) that provided additional treatment; and the dry well itself (Fig. 1). At SDB, the vegetated feature was an existing water quality basin containing small trees, shrubs, and grass, while at the CY a deep grassy swale was constructed to detain water for up to 10 min prior to entering the sedimentation well. The sedimentation chambers were approximately 1.5 m deep and 0.76 m in diameter. The pipes connecting the sedimentation chambers to the dry wells were placed approximately 0.5 m from the bottom of the chambers. Due to insufficient depth and pipe inlet height, stormwater did not have sufficient residence time in the sedimentation chambers to permit most sediment and associated pollutants to settle, therefore pretreatment was performed primarily by the vegetated features.

Each dry well was approximately 13.5 m deep and was completed 4.5 (SDB) and 9 (CY) meter above the water table. At both sites, clay layers composed significant portions of the lithology; 27% at the CY and 45% at SDB. At the CY there was a meter of clayey material between the bottom of the dry well and the water table.

2.2. Stormwater monitoring

Stormwater samples were collected five times over a two-year period when storm precipitation totals were 6.4 mm or greater and when there was less than 2.54 mm of rain during the preceding 72 h. The water year at our locations begins in October, with the onset of winter rains. Most

precipitation stops in March, but can extend into April or May. The study was conducted during two water years with below average rainfall: 26.7 cm in the first year and 34.3 cm in the second. Normal rainfall for the region is 46 cm. During each rain event, runoff was collected at two locations at each site, as indicated by the red arrows in Fig. 1. The stormwater samples were collected in pre-cleaned bottles every 15–60 min, depending on the intensity and duration of the rain event, and placed on ice. Total volume of stormwater runoff was measured at each site, then an algorithm was used to select samples for preparing the flow-weighted composites. Appropriate volumes from each collection bottle were added to a large carboy (reflecting 80% of the runoff volume), mixed, then aliquoted into appropriate vials for analytical chemistry. Total storm runoff volumes and representative flow rates were calculated using the areas of orifices or holes in a stand pipe that controlled the volume of water entering the sedimentation chamber. Each orifice could be plugged so that stormwater could enter the sedimentation chamber only through the open hole. Different sized orifices were associated with different flow rate, based on lab calibration testing. In addition to these samples, during two rain events, composite samples were collected at the stormwater outfall or curb cut (green arrow, Fig. 1). Additional grab samples were collected at curb cut/storm drain outfall during all other rain events. During the second wet season of the study, monitoring occurred during the first storm event of the water year in order to capture the “first flush” of contaminants.

Water samples were analyzed for over 200 contaminants at a local EPA and ELAP-certified (Environmental Laboratory Accreditation Program) laboratory employing standard methods for sample bottle preparation, chain of custody, and reporting of both detection and reporting limits for all analytes. Contaminants for analysis were chosen based on the frequency of their detection in local stormwater samples as well as recommendations from local and state stormwater managers as well as reports of emerging contaminants. They included total suspended solids (US EPA 160.2), pyrethroid pesticides (California Dept. Fish Wildlife, Water Pollution Control Laboratory Method 53), chlorophenoxy herbicides (US EPA 8151A), total petroleum hydrocarbons and motor oil (US EPA 8015-diesel and gas), pyrogenic polycyclic aromatic hydrocarbons (US EPA 8310), semi-volatile organics (US EPA 625), volatile organics (US EPA 8260B), 25 metals and trace elements (measured as dissolved metals; US EPA 200 series), and hexavalent chromium when indicated (US EPA 200 series), general physical constituents such as turbidity and

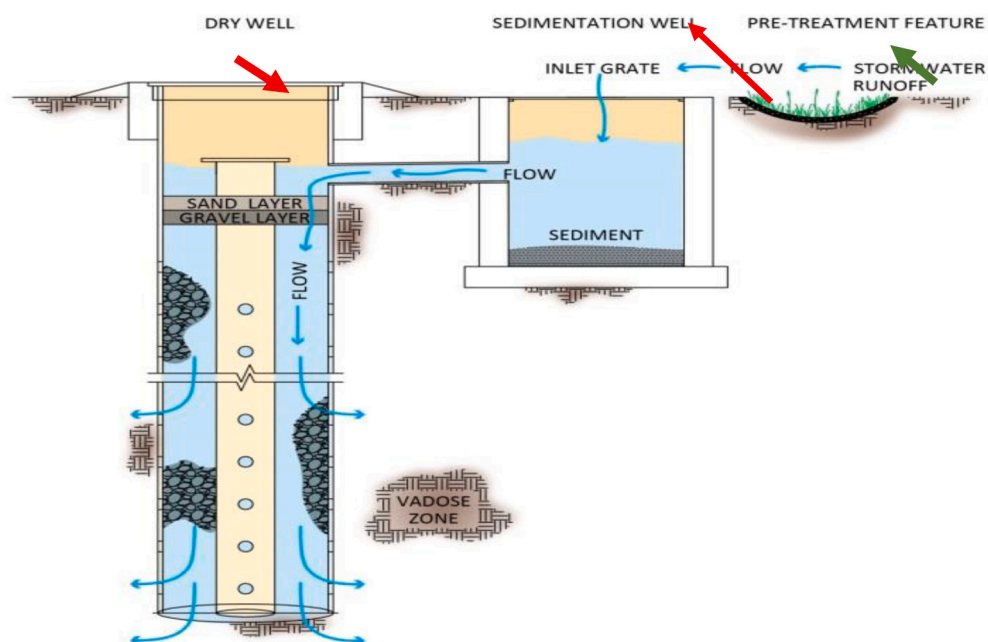


Fig. 1. The dry well system, composed of a vegetated feature, a 122 cm deep swale at one site (CY) or a water quality basin at the other site (SDB); a structural feature, a sedimentation chamber at both sites; and a dry well. Due to its lack of depth, the sedimentation chamber was unable to effectively sequester larger particles; fine particles would likely escape capture regardless of depth. Arrows indicate locations where water samples were collected; circles areas within the dry well indicate rocks used to fill the dry well.

a.

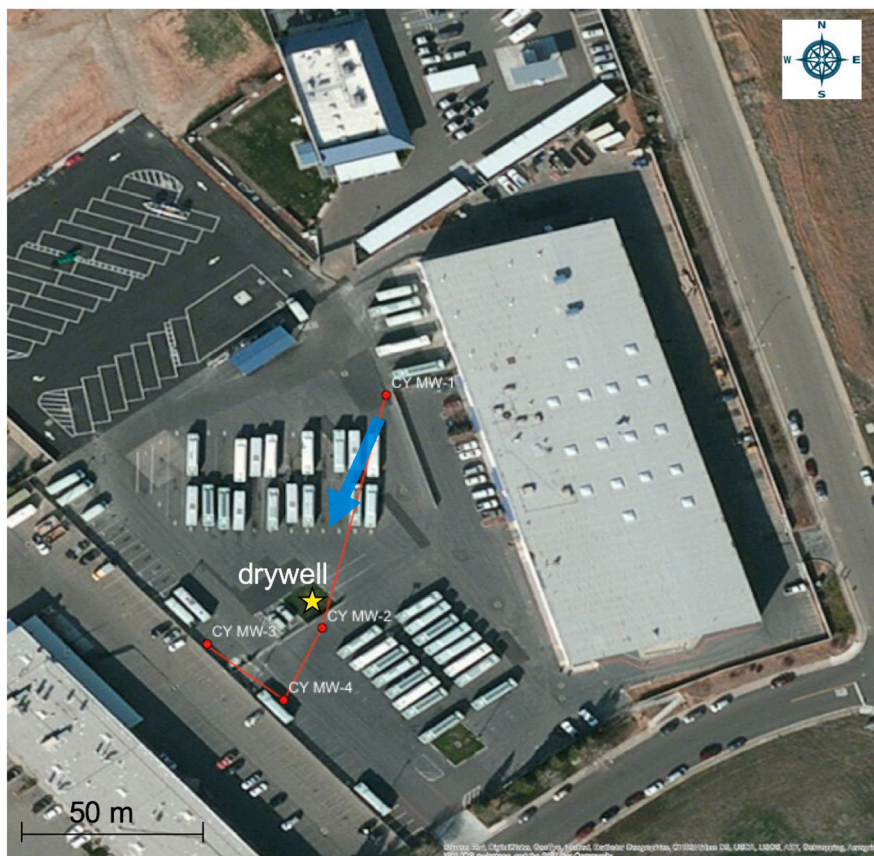
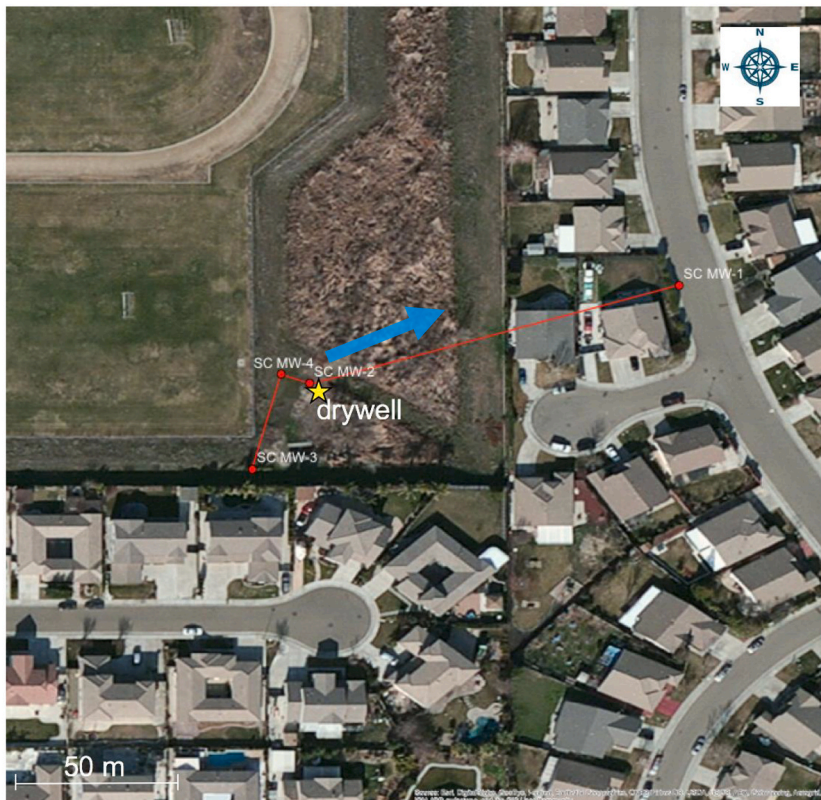


Fig. 2. Aerial views of the two dry well field sites showing the locations of the dry wells and monitoring wells (MW1, MW2, MW3, and MW4) for the CY site (a), and the SDB site (b). The line marks the profile of each site's lithologic cross-section, and the arrow marks the predominant direction of groundwater flow during the study period. The yellow star indicates the location of the dry wells. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

b.



colour (US EPA STDM), general mineral constituents including chloride, nitrate, and sulfate (US EPA STDM), total coliform (US EPA SM 9221), and glyphosate (final monitoring event only). Field blanks, field blank duplicates, and field duplicates were also collected to control for field contamination and reproducibility of collection methods. Laboratory quality assurance included analysis of matrix spike and matrix spike duplicates as well as laboratory control samples. Data was reported for the detection and reporting limits but only results above the reporting limits were used for statistical analysis.

2.3. Groundwater and vadose zone monitoring

Groundwater samples were collected from groundwater and vadose zone monitoring wells prior to the construction of the dry wells to characterize ambient groundwater quality, one to two days after the five storm sampling events from the vadose zone well and two to seven days after the rain event from the groundwater wells. Additionally, groundwater samples were collected after the completion of each wet season. The timing of wet season collections was informed by water level data from pressure transducers installed in some of the wells. Samples were analyzed for the same set of contaminants measured in stormwater. All groundwater samples that were used for analysis of metals were filtered in the field, thus the analysis reflects dissolved concentrations. This was not the case for other stormwater samples, which were not filtered. Outside of this difference, all other samples were collected in the same manner.

2.4. Statistical methods

Non-parametric statistics were used to analyze contaminant data because the data were not normally distributed (Helsel, 2005). Contaminant concentrations at the sites of sample collections were compared to assess the relationships between concentrations in stormwater collected as it entered the dry well system after pretreatment, in the vadose zone, and in the upgradient and downgradient water table well samples. Censored data was recoded by subtracting non-detects from a constant higher than the maximum concentration for each contaminant prior to statistical analysis (Helsel, 2005). The Kruskal-Wallis test was used for those contaminants with a single analytical reporting limit, a statistically derived number that was between two and ten fold greater than the minimum detection limit. If significance was identified ($p \leq 0.05$), the concentrations of contaminants at each study site were ranked, then Tukey's test of honestly significant differences was used to identify differences between groups. By analyzing differences in ranks, not the actual values, Tukey's test substituted for Bonferroni's correction for Type I errors (limits the false discovery rate; Helsel, 2005). For those contaminants with two or greater reporting limits, Generalized Wilcoxon tests were performed. Gehan test, a variation of the Generalized Wilcoxon test used to determine whether two group percentiles are the same or if one is higher than the other, was used to identify differences between groups (Helsel, 2005). Correlation analysis was performed on data used to investigate redox couples and ion exchange reactions in the subsurface. All statistical analyses were performed using Statistica (Tibco Software Inc., Palo Alto, CA, USA).

2.5. Vadose zone contaminant transport modeling

Given the relatively short two-year period of this study, the long-term risks to groundwater quality associated with dry well use could not be assessed from groundwater monitoring alone. Numerical modeling analysis was used to estimate potential long-term effects. We sought to determine the breakthrough time of 1% of the contaminant's input concentration, the time at which the contaminant would reach any existing regulatory levels, and the peak concentration at the water table after 500 years of infiltration for each contaminant at each dry well site for a range of subsurface conditions, chemical transport parameters, and

contaminant input concentrations.

The one-dimensional vadose zone water flow and solute transport modeling software HYDRUS 1D was used to model the vertical transport of select contaminants in infiltrating stormwater through the subsurface domain representing the length between the bottom of the dry well and the site's seasonal-high water table (known as the separation distance), at each site (Šimůnek et al., 2013). 1D rather than 2D or 3D modeling was deemed appropriate because of the laterally extensive layered structures of the vadose zone sediments at these sites, which are also typical of the California Central Valley and many other sedimentary basins. Specifically, the lateral extent of the sediment layers representing coarse and fine-grained hydrofacies (e.g., sands, gravels, silts, clays) in the study area have been estimated to be about 100 m to 400 m (Meirovitz et al., 2017; Maples et al., 2019), making more complex 2D or 3D flow patterns due to laterally discontinuous layering unlikely at these vadose zone sites. This strong horizontal stratification is also reflected in the stratigraphic cross sections shown in Fig. 3. In other words, although the subsurface in the region is highly heterogeneous in 3D at the kilometers scale, at the scale of these vadose zone sites and due to the strong tendency for vertical flow to dominate in the vadose zone, we concluded a 1D modeling would be both appropriate and most likely to represent a direct pathway to the water table (i.e., a worst-case scenario). HYDRUS 1D was used to obtain travel times and contaminant concentrations at the water table. To gain insight into the case where the silt and clay layers that impede flow are missing, two hypothetical, coarse-grained sites in Los Angeles were also modeled.

Stormwater contaminants detected during field testing were selected for the modeling based on their frequency of detection and toxicity. Input contaminant concentrations were calculated from the results of the stormwater quality sampling analyses. Further, fipronil and imidacloprid, two pesticides that were not analyzed for in site stormwater but whose presence has been detected in urban stormwater with increased frequency, were also included in the model. Their input concentrations were based on data obtained from the California Department of Pesticide Regulation. The amount of water coming into the domain was determined from the results of groundwater stage analyses. Water and solutes were input at the top of the model domain and allowed to drain freely from the bottom, which represented the elevation of the seasonal-high water table.

The HYDRUS 1D domains were created using lithologic cross-sections created for each study site (Fig. 3). These cross-sections were created using lithologic logs obtained during the drilling of monitoring wells. The sediment layers between the dry well bottom and the seasonal-high water table were estimated from the cross-sections, and the textural classifications and layer thicknesses were duplicated in the HYDRUS 1D domain sediment profiles. The CY domain was 9.75 m in length and composed of seven layers of sediment (from top to bottom: 70 cm of clay, 165 cm of silty sand, 60 cm of sandy silty clay, 60 cm of silty clay, 285 cm of sand, 85 cm of silty clay, and 250 cm of sandy silt). The SDB domain was 2.78 m in length and composed of two layers (from top to bottom: 150 cm of sandy silty clay, and 128 cm of sand). The time-variable boundary conditions were determined based on the results of the stormwater sampling, the results of the dissolved concentration calculations, and the stage records for each dry well site.

The water flow boundary condition (BC) at the top of both model domains is variable pressure head and flux, and the solute transport BC at the top of both model domains is concentration flux. The bottom water flow BCs are free drainage, i.e. zero-gradient boundary conditions. Here, we chose the free drainage boundary condition to represent worst case (most downward flux) flow and transport conditions, typical of conditions when the water table is lower than the seasonal high position chosen to represent the bottom of the domain. The bottom solute transport BC for both domains is zero concentration gradient, which in effect accomplishes a specified solute flux that is a function of the water velocity and the computed concentration at the boundary, but does not account for the effect of dispersion across the boundary. The latter can

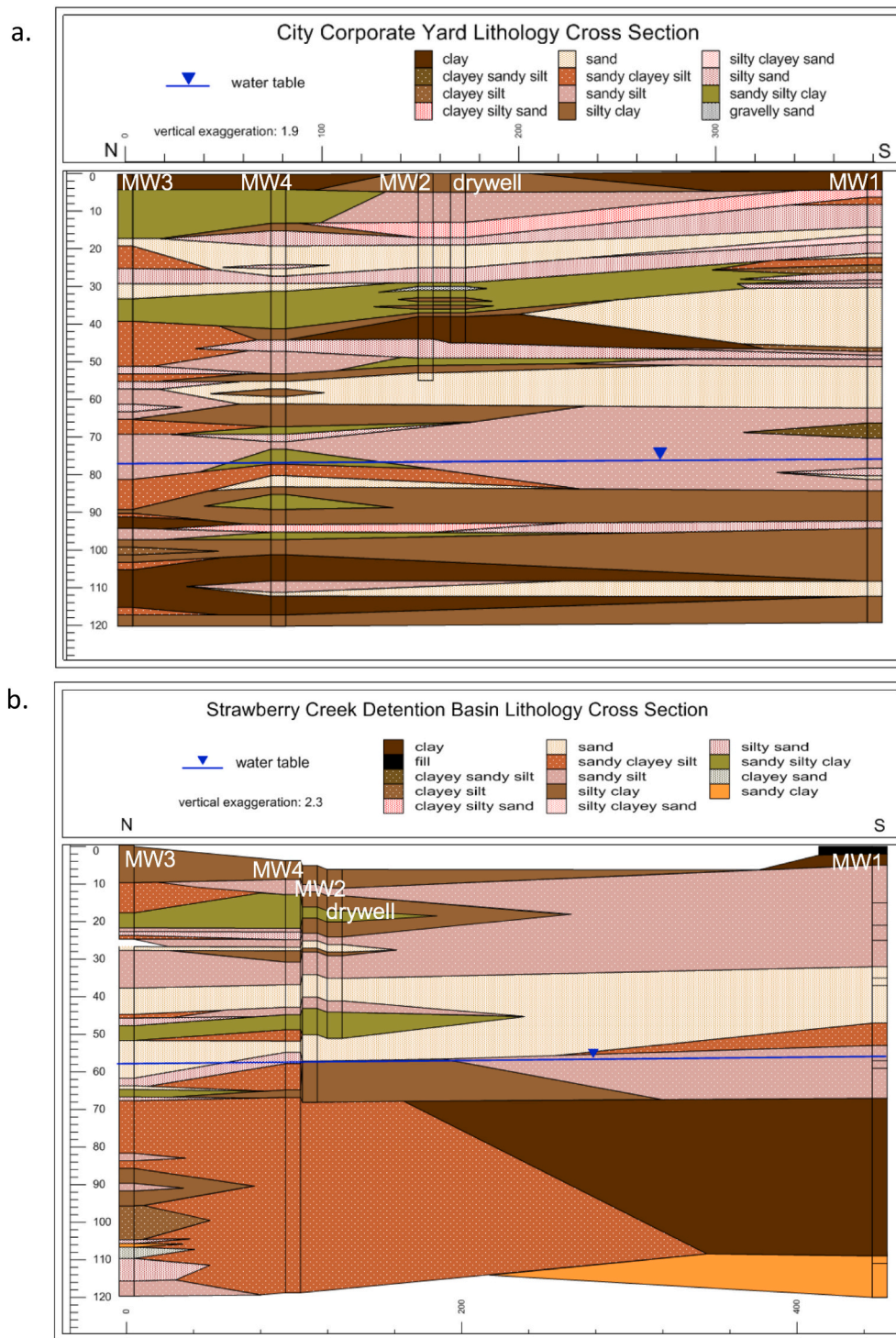


Fig. 3. Lithologic cross-sections created for the CY dry well site (a) and the SDB dry well site (b) using textural classifications obtained from driller’s logs. Depths are reported in feet (one foot is 0.31 meters).

be shown to contribute negligibly to the overall solute flux under the simulated conditions.

The top water flow and solute transport BCs are time-variable. A 365-day record of daily pressure head or flux and contaminant concentration in the influent water was created for each drywell site. The CY model time-variable BC is a constant pressure head of 400 cm for 230 days, and 0 flux for the remaining 135 days, repeated each year. The SDB model domain’s upper water flow boundary condition is a constant pressure head of 300 cm for 215 days, and 0 flux for the remaining 150 days,

repeated each year. The time-variable BCs were determined based on the results of the stormwater sampling, the results of the dissolved concentration calculations, and the stage records for each drywell site. The heads of 400 cm and 300 cm represent the average pressure head (stage) during the wet season for the 2016 water year. Contaminant concentration remains constant over time in influent water for both sites. The annual boundary condition time-series is repeated as needed for each model run to capture the full breakthrough curve of each contaminant under each scenario (model run time periods range from one year to

many years depending on the transport parameters of the modeled contaminant).

The group of contaminants selected for modeling were those that were regularly reported in stormwater monitoring at either of the two study sites or were newer generation pesticides regularly detected in urban stormwater monitoring in by the California Department of Pesticide Regulation (2016). Input variables specified for the modeling included sediment hydraulic properties and contaminant chemical properties that affect transport. The saturated hydraulic conductivity (K_s) of the lowest hydraulic conductivity layer was estimated using field data, and the Rosetta Dynamically Linked Library neural network, accessed through HYDRUS 1D, was used to estimate the K_s of the other sediments based on their textural classification (Schaap et al., 2001). Literature sources were used to determine values for subsurface chemical parameters and for all contaminant transport parameters. Because of uncertainty in both the subsurface hydraulic and chemical conditions, and the transport characteristics of the selected contaminants, eight sensitivity scenarios were run for each of the chosen contaminants at each site. The variables that were changed to create the eight scenarios, selected due to site variability, were contaminant input concentration, K_s and fraction organic carbon (f_{oc}) values for organic contaminants or soil-water partitioning coefficient (K_d) values for metals. The K_s values were calculated from actual site conditions. F_{oc} and K_d values were obtained from relevant literature (Allison and Allison, 2005; Baes et al., 1984; Interstate Technology and Regulatory Council, 2015; Office of Air and Radiation USEPA, 1999; Sheppard and Sohlenius, 2009; Streng and Peterson, 1989). The worst-case-scenario conditions were modeled using total contaminant concentrations in stormwater, high K_s values, and low f_{oc} or K_d values, maximizing the input concentration of contaminants and the movement of those contaminants through the vadose zone. The most-likely-case was modeled using the mobile fraction of each contaminant, the dissolved concentration which was calculated from total concentration using the relevant partition coefficient. See Table 1 and Table 2 for selected values of soil hydraulic properties and contaminant transport parameters for the CY and SDB sites, respectively. The K_d value for an organic contaminant is dependent on the fraction organic carbon (f_{oc}) of the sediment the contaminant is moving through. To capture this variability in the model, the f_{oc} values used for clay and silt sediments were varied between 0.01 or 0.001, and 0.001 or 0.0001 for sands (Interstate Technology and Regulatory Council, 2015).

In addition to the two Elk Grove sites, modeling was performed for two theoretical sites in Los Angeles, California, United States of America. The Sacramento region within which the City of Elk Grove is located, as well as the greater California Central Valley, are known for the presence of clay and silt layers interbedded with the aquifer sediments. Given the capacity of clay to bind many contaminants, there was concern that Elk Grove modeling results could suggest a greater protection of groundwater than might be found in other parts of the state or country, where sand can compose significant portions of the unsaturated zone. Further, the diversity and concentration of contaminants could be larger than what might be seen in a suburban area such as Elk Grove. Modeled sites in the Los Angeles metropolitan area were used to fill this gap. Stormwater contaminants and concentrations for the Los Angeles area were determined from stormwater quality literature, and sediment hydraulic and chemical properties were sourced from relevant literature (Allison and Allison, 2005; Baes et al., 1984; Interstate Technology and Regulatory Council, 2015; Office of Air and Radiation USEPA, 1999; Sheppard and Sohlenius, 2009; Stein et al., 2006; Streng and Peterson, 1989; Tiefenthaler et al., 2008). See Table 3 for selected values of soil hydraulic properties and contaminant transport parameters for the theoretical dry well sites in Los Angeles. The f_{oc} values used for the LA domain were 0.005 for clay, 0.001 for sandy loam and loam, and 0.0001 for sand. Parameter values were not modified between model runs, as the two sediment profiles created for the Los Angeles numerical analysis already represent theoretical average and worst-case subsurface scenarios.

Table 1

HYDRUS 1D solute transport parameters for the CY dry well site, where D_w is the contaminant's diffusivity in water, D_G is the contaminant's diffusivity in air, K_d is the contaminant's soil-water partitioning coefficient, H is the contaminant's Henry's Law coefficient, and k is the contaminant's first order degradation rate constant.

Contaminant	D_w (cm ² /day)	D_G (cm ² /day)	K_d (mL/g)	H (–)	k (day ⁻¹)
Aluminum	0	0	1500 ^a	0	–
Di(2-ethylhexyl)-phthalate	0.3629 ^a	1468.8 ^a	1.2E5 x $f_{oc}^{a,b}$	1.10E-5 ^b	clay: 3.47E-3 ^{b,c,d} sand: 6.93E-3 ^{b,c,d}
Iron	0	0	high: 100 ^e low: 25 ^f	0	–
Manganese	0	0	high: 90 ^{h,i} low: 30 ^{i,f}	0	–
Permethrin	0.4147 ^a	1641.6 ^a	1.2E5 x f_{oc}^g	7.70E-5 ^{h,g}	clay: 9.24E-3 ^{g,h} sand: 2.77E-2 ^{g,h}
Tert-butyl alcohol	0.9850 ⁱ	8510.4 ⁱ	2.0 x f_{oc}^i	3.70E-4 ^j	clay: 3.47E-3 ^j sand: 3.47E-3 ^j
Fipronil	0.4061 ^k	4138.6 ^k	6000 x $f_{oc}^{l,m}$	3.47E-8 ⁿ	clay: 3.15E-2 ^{l,m} sand: 3.85E-2 ^{l,m}
Imidacloprid	0.4916 ^k	4821.1 ^k	300 x f_{oc}^o	2.66E-9 ^p	clay: 3.03E-3 ^q sand: 3.65E-3 ^q

^a US EPA, 2016.

^b Institute for Health and Consumer Protection, 2008.

^c Carlisle et al., 2009.

^d Liang et al., 2018.

^e Allison and Allison, 2005.

^f Baes et al., 1984.

^g Imgrund, 2003.

^h Kaufman et al., 1977.

ⁱ New Jersey Department of Environmental Protection, 2015.

^j National Center for Environmental Assessment, 2014.

^k Tucker and Nelken, 1982.

^l Bower and Tjeerdema, 2017.

^m Gunasekara and Troung, 2007.

ⁿ Sander, 2015.

^o Fossen, 2006.

^p Roberts and Hutson, 1999.

^q Rouchard, Gustin, and Wauters, 1994.

2.6. Analysis of desorption of arsenic and chromium

Arsenic and chromium are naturally occurring toxic metals present in geologic formations in the Sacramento region (GAMA, 2021). To investigate the possibility that constituents such as salts or metals in stormwater could cause desorption and solubilization of either of these toxic metals, a preliminary analysis of the relationship between ion and redox couples in groundwater was performed. Concentrations of arsenic, chromium, and numerous ions were assessed as part of stormwater and groundwater quality analysis. Comparisons between upgradient and downgradient concentration of arsenic and chromium at the water table were made, and correlation analysis of redox couples was performed.

Table 2

HYDRUS 1D solute transport parameters for the CY dry well site, where D_w is the contaminant's diffusivity in water, D_G is the contaminant's diffusivity in air, K_d is the contaminant's soil-water partitioning coefficient, H is the contaminant's Henry's Law coefficient, and k is the contaminant's first order degradation rate constant.

Contaminant	D_w (cm ² /day)	D_G (cm ² /day)	K_d (mL/g)	H (-)	k (day ⁻¹)
Aluminum	0	0	1500 ^a	0	-
Bifenthrin	0.3888 ^a	1555.2 ^a	2.37E5 x foc ^{a,b}	4.10E-4 ^{a,b}	clay: 2.31E-3 ^b sand: 3.47E-3 ^b
Iron	0	0	high: 100 ^c low: 25 ^d	0	-
Manganese	0	0	high: 90 ^{a,d} low: 30 ^{a,d}	0	-
Tert-butyl alcohol	0.9850 ^e	8510.4 ^e	2.0 x foc ^e	3.70E-4 ^{e,f}	clay: 3.47E-3 ^f sand: 3.47E-3 ^f
Fipronil	0.4061 ^g	4138.6 ^g	6000 x foc ^{h,i}	3.47E-8 ^j	clay: 3.15E-2 ^{h,i} sand: 3.85E-2 ^{h,i}
Imidacloprid	0.4916 ^g	4821.1 ^g	300 x foc ^k	2.66E-9 ^{k,l}	clay: 3.03E-3 ^m sand: 3.65E-3 ^m

^a USEPA, 2016.

^b Fecko, 1999.

^c Allison and Allison, 2005.

^d Baes et al., 1984.

^e New Jersey Department of Environmental Protection, 2015.

^f National Center for Environmental Assessment, 2014.

^g Tucker and Nelken, 1982.

^h Bower and Tjeerdema, 2017.

ⁱ Gunasekara and Troung, 2007.

^j Sander, 2015.

^k Fossen, 2006.

^l Roberts and Hutson, 1999.

^m Rouchard et al., 1994.

3. Results

3.1. Stormwater infiltration rates and volumes

Estimates of cumulative annual stormwater volumes infiltrated through each dry well were obtained using the results of orifice flow measurements taken during sampled storm events as well as dry well stage data gathered from pressure transducers. The total volumes of stormwater infiltrated through the CY and SDB dry wells during the 2016 water year were estimated to be approximately 246.7 cubic meters (m³) and 863.4 m³, respectively. The SDB dry well received approximately 3.5 times greater volumes of influent stormwater per storm event than the CY dry well did. This is because the SDB dry well receives stormwater from an area of approximately 70 ha, compared to the CY dry well's paved 0.26 ha surface.

The infiltration rates at SDB were on average 12.7 cubic meters/h (m³/h), considerably higher than the 7.2 m³/h measured at the CY. Factors accounting for this disparity include differences in the volumes of water entering the dry well, causing differences in head, as well as differences in vadose zone composition (see Fig. 3).

3.2. Stormwater and groundwater quality.

Contaminants that were detected in stormwater samples followed similar patterns, based on their origin. Anthropogenic contaminants such as the pyrethroid bifenthrin, aluminum, and iron, typically had elevated concentrations in stormwater, which declined to varying

Table 3

HYDRUS 1D solute transport parameters for the theoretical Los Angeles dry well locations, where D_w is the contaminant's diffusivity in water, D_G is the contaminant's diffusivity in air, K_d is the contaminant's soil-water partitioning coefficient, H is the contaminant's Henry's Law coefficient, and k is the contaminant's first order degradation rate constant.

Contaminant	D_w (cm ² /day)	D_G (cm ² /day)	K_d (mL/g)	H (-)	k (day ⁻¹)
Aluminum	0	0	1500 ^a	0	-
Benzo[a]pyrene	0.4838 ^a	4147.2 ^a	5.9E5 x foc ^b	1.90E-5 ^a	clay: 9.49E-4 ^c sand: 2.31E-3 ^c
Bifenthrin	0.3888 ^a	1555.2 ^a	2.37E5 x foc ^{a,d}	4.10E-4 ^{a,d}	clay: 2.31E-3 ^d sand: 3.47E-3 ^d
Copper	0	0	35 ^a	0	-
Fipronil	0.4061 ^e	4138.6 ^e	6000 x foc ^{f,g}	3.47E-8 ^h	clay: 3.15E-2 ^{f,g} sand: 3.85E-2 ^{f,g}
Imidacloprid	0.4916 ^e	4821.1 ^e	300 x foc ⁱ	2.66E-9 ^{j,k}	clay: 3.03E-3 ^k sand: 3.65E-3 ^k
Iron	0	0	25 ^{l,m}	0	-
Lead	0	0	900 ⁿ	0	-
Naphthalene	0.7258 ^a	5184.0 ^a	1.50E3 x foc ⁿ	1.8E-2 ^a	clay: 3.47E-3 ^{n,o} sand: 7.70E-3 ^{n,o}
Permethrin	0.4147 ^a	1641.6 ^a	1.2E5 x foc ^p	7.70E-5 ^{a,p}	clay: 9.24E-3 ^{p,q} sand: 2.77E-2 ^{p,q}
Zinc	0	0	62 ^a	0	-

^a USEPA, 2016.

^b Irwin, 1997.

^c USEPA, 1984.

^d Fecko, 1999

^e Tucker and Nelken, 1982.

^f Bower and Tjeerdema, 2017.

^g Gunasekara and Troung, 2007.

^h Sander, 2015.

ⁱ Fossen, 2006.

^j Roberts and Hutson, 1999.

^k Rouchard et al., 1994.

^l Allison and Allison, 2005

^m Baes et al., 1984

ⁿ Park et al., 1990.

^o Cerniglia, 1992.

^p Imgrund, 2003.

^q Kaufman et al., 1977.

degree as runoff passed through the vegetated pretreatment feature (Fig. 4, Panels A, B, and C). None were detected above the reporting limit in groundwater. Among the organic compounds analyzed, pyrethroid pesticides and motor oil were detected most frequently. Of the nine pyrethroids measured, bifenthrin was detected at the highest concentrations, primarily at the SDB (Fig. 4A). However, bifenthrin was not detected in any of the groundwater samples. The average concentration of bifenthrin declined by about 50% as stormwater passed through the pretreatment features while the median concentration in groundwater was below the reporting limit. Of the other pesticides measured, a single chlorophenoxy herbicide, dalaphon (in groundwater only), and glyphosate (in stormwater only) were each detected once; in both cases, the concentrations were below the relevant regulatory standard. Most volatile organics were not detected in stormwater. Some volatiles including toluene, acetone, and tert-butyl alcohol (TBA) were identified, near the reporting limit and below the regulatory standard. Acetone was detected once at 300 µg/L in stormwater entering the CY swale, but the

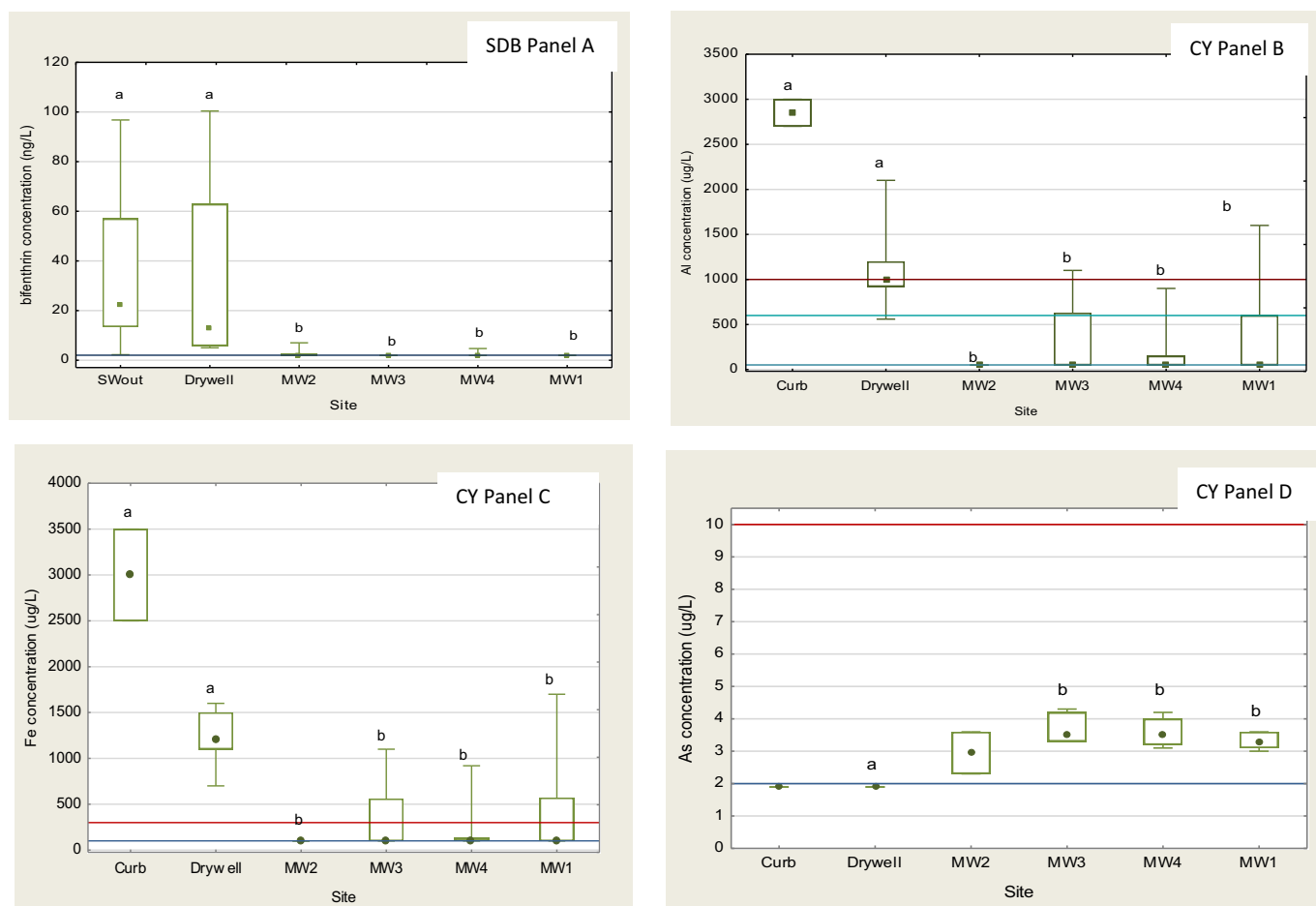


Fig. 4. Concentrations of anthropogenic and naturally-occurring toxicants in stormwater and groundwater at the two study sites. Examples of the pattern of contaminant detection at various stages of passage through the dry well system. Each point represents the median concentration at each of six sampling sites; the box reflects the 25th and 75th percentile values, and the whiskers represent the minimum and maximum values measured. The blue line indicates the reporting limit. $n = 5$ except for the curb-cut sample, $n = 2$. Different superscripts indicate significant differences in concentration. Sampling groups are as follows: SWout = location where stormwater entered vegetated pretreatment; Drywell = concentration as runoff entered the drywell; MW2 = vadose zone well; MW1 and 4 = downgradient water table wells; MW3 = upgradient water table well. During the winter, MW2 at SDB functioned as a water table well due to a reduce depth to groundwater associated with increased precipitation. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

concentration fell by 50% after pretreatment. Acetone was not quantifiable in groundwater because its concentration was below the reporting limit. At SDB, acetone was occasionally noted in stormwater at low levels and once just above the reporting limit in a downgradient groundwater monitoring well (data not shown). Not surprisingly, motor oil was detected in stormwater at both sites, but at a ten-fold higher concentration at the CY than at the SDB (median values: 1.2 vs. 0.11 mg/L, respectively). Pretreatment resulted in a three-fold or greater reduction in concentration. None was found in groundwater samples at either site. Metals, measured as dissolved in water, were frequently detected in stormwater and groundwater at both sites. For example, the concentration of aluminum in influent stormwater was 3 mg/L at the CY, three times the MCL, but fell to 1 mg/L after pretreatment (Fig. 4B). The second most commonly detected metal was iron (Fig. 4C). Exceedances of the secondary regulatory standard for taste and odor were found at both sites for influent stormwater and stormwater measured after pretreatment, although the concentrations were considerably reduced after pretreatment.

In contrast to these pollutants, a completely different pattern of detections was observed for naturally occurring metals such as arsenic (Fig. 4D) and chromium (data not shown); concentrations in stormwater were not detectable while concentrations in groundwater were elevated in both upgradient and downgradient wells. These differences were statistically significant in the case of arsenic. Over the two year study

period, concentrations of both metals were elevated in the summer (dry season) and declined during the rainy season, likely due to dilution caused by the infiltration of stormwater.

No significant increase of arsenic or chromium concentrations were observed downgradient of the dry well compared to the upgradient well, suggesting mobilization of metals during the two-year study period did not occur. When data from all groundwater samples at both sites was pooled, no significant correlations were found between arsenic and sulfate, bicarbonate, or manganese. The single exception to this pattern was a weak positive correlation between arsenic and iron ($p = 0.056$). Mobilization due to competing effects could not be excluded, and therefore further monitoring of common competing anions such as phosphate, silicate or vanadate is needed. No significant correlation was found between chromium and manganese (data not shown), despite the fact that manganese oxides are recognized as the major viable oxidants which oxidize insoluble trivalent chromium to soluble hexavalent chromium under a range of environmental conditions (Eary and Rai, 1987; Guha et al., 2001).

Nitrate is the single water soluble constituent that can be toxic at high concentrations (>10 mg/L NO_3^- -N; US EPA regulatory standard). Nitrate concentrations were detectable in stormwater and groundwater, exceeding the regulatory standard in groundwater at both sites (Fig. 5). It was notable that at the CY, samples from the upgradient water table monitoring well had a significantly higher concentration than samples

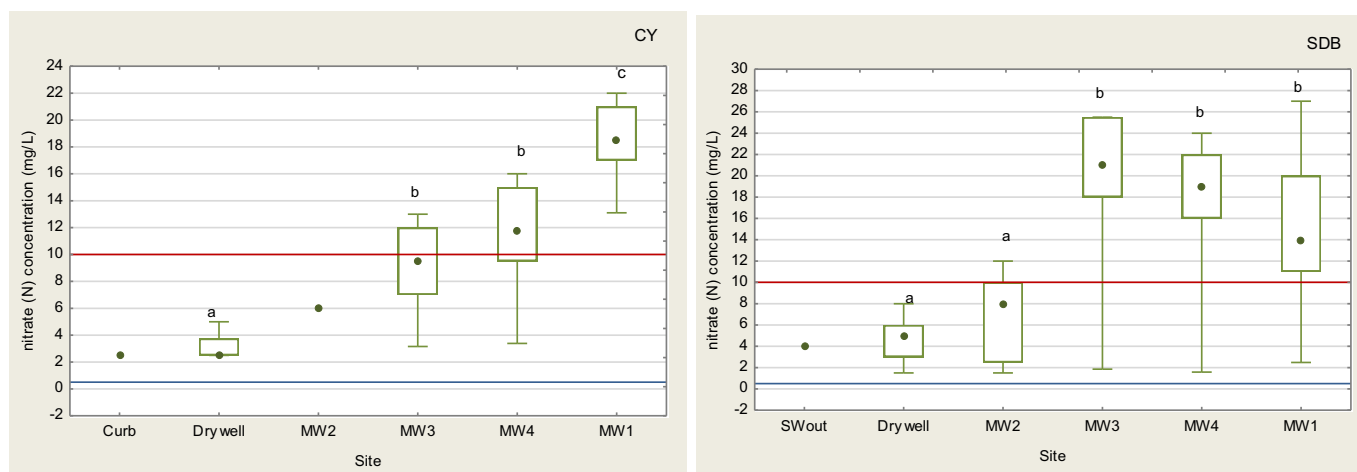


Fig. 5. Nitrate concentrations in stormwater and groundwater at the CY (left) and SDB (right). Notations are the same as for other figures.

from the two downgradient wells, suggesting that stormwater might have actually diluted nitrate concentrations.

3.3. Estimate of the effectiveness of pretreatment

The concentrations of a subset of contaminants for which there were detectable quantities and that have varied chemical properties were compared before and after pretreatment (Table 4). The change in concentration was used as an estimate of the effectiveness of the grassy swale or water quality basin at removing pollutants. The different types of vegetated pretreatment (a grassy swale at the CY vs. a water quality basin with varied vegetation at SDB) were reflected in the different degrees of contaminant reduction. The change in total suspended sediment, which adsorbs many organics and metals, was 63% at the CY and 50% at SDB. The estimate of removal efficiency is likely most valid for total suspended solids due to the larger sample size.

3.4. Vadose zone modeling: Corporation Yard

The contaminants selected for modeling at the CY and the SDB were a combination of those detected in stormwater at each location and two pesticides of emerging concern, fipronil and imidacloprid. In the most likely scenario modeled for the CY site (Table 4), the input concentration for half of the contaminants modeled was so low that the groundwater concentration would not exceed the reporting limit within the modeling timeframe of 500 years. Contaminants that fell into this group included aluminum, di-ethyl hexyl phthalate (DEHP), and permethrin, a commonly used pyrethroid. For example, aluminum was predicted to arrive at the water table at a concentration 1% of its input concentration after 265 years, but would be unlikely to reach the analytical reporting limit due to its low input concentration. In contrast, fipronil and iron were predicted to arrive at the water table at measurable concentrations

Table 4

Percent change in concentration of contaminants before and after pretreatment. Statistical analysis was not performed on this data due to the small sample size ($n = 2$) with the exception of TSS ($n = 5$).

Contaminant	% reduction	
	CY	SDB
Aluminum	65	50
Bifenthrin	100 ^a	42
Manganese	53	-44
NO ₃ -N	0	-25
Motor oil	67	55
TSS	63	50

^a Below reporting limit of 0.002 $\mu\text{g/L}$.

after 6 and 18 years respectively. A third group of contaminants, those that are water soluble, reached the water table over the course of days to weeks. For example, imidacloprid and tert-butyl alcohol (TBA) were shown to reach quantifiable levels after 85 and 129 days respectively.

The worst-case scenario (high hydraulic conductivity and high solubility of the contaminant) results for the CY site modeling showed most contaminants reach reporting limits at the water table after a period of days to years (Table 4). The rate of transport of aluminum was slow; it would likely take hundreds of years to reach the water table. The other modeled metals and pyrethroids moved more rapidly through the vadose zone, reaching the water table in between 5 and 17 years. The water soluble contaminants would reach reporting limits at the water table between 10 and 133 days. Included in Table 5, for comparison, are the results of running a conservative tracer through the model domains. All chemical parameters for the conservative tracer were set to 0 to represent transport through the domain without any retardation, dispersion, or degradation.

3.5. Vadose zone modeling: Strawberry Detention Basin

A slightly different group of contaminants were modeled at the SDB than at the CY, reflecting differences in pollutants detected (Table 6). The ubiquitous pyrethroid pesticide, bifenthrin, was regularly measured at elevated concentrations in stormwater at SDB. Given its hydrophobicity, it was the least mobile of all those modeled at SDB, predicted to take about 470 years to reach a reportable concentration at the water table. Under the average case scenario, the concentration would likely reach about 3 ng/L after 500 years while using worst case scenario assumptions, the concentration would reach its influent concentration of 99 ng/L in 5 years.

At SDB, under the average case scenario, the water soluble contaminants are predicted to have greatly reduced transport times. Imidacloprid and fipronil were shown to reach the water table in about three months and seven years, respectively. TBA was shown to take fewer than 20 days of travel time. All the other monitored contaminants are never shown to reach sufficiently high concentrations to be reportable. The worst-case scenario results at the SDB site differed from the CY in that all contaminants would likely reach reportable concentrations in less than 50 years (Table 6). Most notably, imidacloprid and TBA were shown to reach the aquifer in 3 days, and fipronil in 18 days.

3.6. Vadose zone modeling: Los Angeles

A different group of contaminants, drawn from regional stormwater data, were modeled for the hypothetical Los Angeles, California, USA, scenarios. The pollutant list included contaminants such as benzo[a]

Table 5

Modeled travel time and concentrations based on the average and worst-case scenario model outputs for the CY site. Parameters used in this scenario included the use of dissolved (mobile) concentrations of contaminants in stormwater and domain input parameters representing site conditions. NR – not reportable; concentration below the reporting limit, NA – not applicable due to the lack of a regulatory value. The notification level, a health advisory value that is not regulatory, is the reference value for tert-butyl alcohol (TBA). Criteria values refer to a regulatory or advisory standard. The criteria value for Al (aluminum) is the maximum contaminant level (MCL); iron is a secondary standard, based on taste and odor; for TBA, a notification level. Imidacloprid and fipronil were not measured during the study. The calculated dissolved concentrations of TBA, fipronil, and imidacloprid were within 10% of their total concentrations, and so only total concentrations were modeled average values found in stormwater were provided by the California Department of Pesticide Regulation. These results do not consider dilution of contaminants in groundwater, only the concentration when the contaminant reaches the water table. Results for a conservative tracer are included for comparison, with the reporting limit in this case being 1% of the input concentration, and the criteria value being the input concentration. The input concentration is similar to the chloride concentrations observed in stormwater at the dry well sites.

CY contaminant (input concentration for average worst-case scenarios)	Travel time to reach reporting limit at water table		Time to reach criteria value		Peak concentration at water table in 500 years	
	Average	Worst-case	Average	Worst-case	Average	Worst
aluminum (0.042 2.1 mg/L)	NR	274 years	NR	351 years	0.041 mg/L	2.1 mg/L
DEHP (0.062 3.01 µg/L)	NR	NR	NR	NR	0.056 µg/L	2.7 µg/L
iron (0.16 1.6 µg/L)	25 years	5 years	NR	6 years	0.16 µg/L	1.6 µg/L
manganese (10 31 µg/L)	NR	8 years	NR	NR	10 µg/L	31 µg/L
permethrin (2.4 12.2 ng/L)	NR	17 years	NA	NA	1.7 ng/L	8.8 µg/L
TBA (19 19 µg/L)	12 days	10 days	13 days	12 days	18 µg/L	18 µg/L
fipronil (0.5 0.5 µg/L)	6 years	133 days	NA	NA	0.47 µg/L	0.47 µg/L
imidacloprid (0.9 0.9 µg/L)	85 days	16 days	NA	NA	0.85 µg/L	0.86 µg/L
conservative tracer (5.0 mg/L)	9 days	7.5 days	20 days	19.5 days	5.0 mg/L	5.0 mg/L

pyrene, naphthalene, and zinc that were frequently reported in the stormwater literature from the Los Angeles region (Tiefenthaler et al., 2008; Stein et al., 2006). The average scenario modeling results indicated that all but one of the contaminants would not reach concentrations above criteria values (Table 7). Four contaminants were predicted to reach the water table at detectable concentrations within 15 years: copper, fipronil, imidacloprid and naphthalene after 15 years, 3 years, 34 days, and 188 days, respectively.

In contrast, under the worst-case scenario, modeling showed that contaminants reach the water table much more rapidly and achieved higher concentrations than estimated for the average case (Table 7). For example, benzo[a]pyrene (B[a]P) traveled through the unsaturated zone, which was 80% sand, in three months compared to 250+ years in the average case scenario, close to a 1000 fold difference. Similar comparisons can be made for copper, where a greater than 200 fold difference was calculated; a 90 fold difference for lead, and bifenthrin, the most hydrophobic of the modeled contaminants, was predicted to never reach a reportable value compared to a 48 day travel time in the worst-case scenario. It appeared that the greater the hydrophobicity of the contaminant, the larger the contrast between the influence of the two subsurface profiles on transport time. However, with the two water

Table 6

Modeled travel time and concentrations based on average and worst-case scenario model outputs for SDB. The calculated dissolved concentrations of TBA, fipronil, and imidacloprid were within 10% of their total concentrations, so total concentrations were used in the modeling effort. Notations are as for Table 5. These results do not consider dilution of contaminants in groundwater, only the concentration in percolating water when the contaminant reaches the water table. Results for a conservative tracer are included for comparison, with the reporting limit in this case being 1% of the input concentration, and the criteria value being the input concentration. The input concentration is similar to the chloride concentrations observed in stormwater at the dry well sites.

SDB contaminant and input concentration for average worst-case scenarios	Travel time to reporting limit at water table		Time to reach criteria value at water table		Peak concentration at water table in 500 years	
	Average	Worst-case	Average	Worst-case	Average	Worst-case
Al (0.006 0.3 mg/L)	NR	50 years	NR	NR	0.0022 mg/L	0.30 mg/L
bifenthrin (11 100 ng/L)	470 years	5 years	NA	NA	3.0 ng/L	99 ng/L
iron (0.042 0.42 µg/L)	NR	190 days	NR	2 years	0.042 µg/L	0.42 µg/L
manganese (14 41 µg/L)	NR	2 years	NR	NA	14 µg/L	41 µg/L
TBA (20 20 µg/L)	19 days	3 days	23 days	4 days	19 µg/L	20 µg/L
fipronil (0.5 0.5 µg/L)	7 years	18 days	NA	NA	0.24 µg/L	0.47 µg/L
imidacloprid (0.9 0.9 µg/L)	103 days	3 days	NA	NA	0.86 µg/L	0.90 µg/L
conservative tracer (5.0 mg/L)	9 days	1.5 days	21.5 days	3 days	5.0 mg/L	5.0 mg/L

soluble pesticides the differences were considerably less. It is worth noting that with the exception of B[a]P, none of the concentrations of contaminants reached their respective criteria values (e.g., MCL, PHG, or other benchmark).

4. Discussion

4.1. Contaminants in stormwater

The presence of contaminants in stormwater in the United States has been widely documented for over 50 years. The concern regarding stormwater has historically been focused on the adverse impacts on aquatic life. A large body of literature exists on the classes, sources, and effects of contaminants on aquatic life (Brown and Peake, 2006; Kuivila et al., 2012; Weston and Lydy, 2012; Budd et al., 2015; Pitt et al., 1999; Maestre and Pitt, 2006). However, more recently, additional concerns regarding the harmful effects of stormwater contaminants on groundwater has been the focus of considerable interest due to efforts to infiltrate stormwater to reduce the effects of hydromodifications on urban waterways as well as use it as a source of groundwater replenishment (NRC, 2009). Given the history of groundwater contamination associated with industrial and commercial site and on military installations and the lack of nationwide information, the United States Geologic Survey conducted a comprehensive study of 50 runoff events at over 20 sites in the United States. Their data showed that stormwater entrains pesticides, pharmaceuticals, PAHs, as well as a variety of industrial and household chemicals (Masoner et al., 2019). Of the 438 organic chemicals analyzed, their average detection rate was greater than 70%.

In contrast to findings of Masoner and others, we found surprisingly few organic contaminants detected in stormwater samples at the two study sites in Elk Grove. Fewer than 10% of the contaminants analyzed at any one sampling event were detected above their reporting limit.

Table 7

Modeled travel time and concentrations based on average and worst-case scenario model outputs for two theoretical sites in Los Angeles. The calculated dissolved concentrations of TBA, fipronil, and imidacloprid were within 10% of their total concentrations, so total concentrations were used in the modeling effort. Notations are as for Table 5. These results do not consider dilution of contaminants in groundwater, only the concentration in percolating water when the contaminant reaches the water table.

LA contaminant and input concentration for average worst-case scenarios)	Travel time to reach reporting limit at water table (years unless otherwise noted)		Time to reach criteria value at water table (years unless otherwise noted)		Peak concentration at water table in 500 years	
	Average	Worst-case	Average	Worst-case	Average	Worst-case
benzo[a]pyrene (29 100 ng/L)	253	120 days	338	175 days	25 ng/L	100 ng/L
bifenthrin (9 100 ng/L)	NR	48 days	NA	NA	0.0000035 pg/L	9.0 ng/L
copper (18 18 µg/L)	15	24 days	NR	NR	18 µg/L	18 µg/L
fipronil (0.5 0.5 µg/L)	3	2 days	NA	NA	0.21 µg/L	0.49 µg/L
imidacloprid (0.9 0.9 µg/L)	34 days	1 day	NA	NA	0.85 µg/L	0.90 µg/L
lead (1.6 8 µg/L)	364	4	NR	NR	0.90 µg/L	8.0 µg/L
naphthalene (35 35 ng/L)	188 days	1 day	NR	NR	31 ng/L	35 ng/L
zinc (77 µg/L)	26	43 days	NR	NR	77 µg/L	77 µg/L

This appears to be due to a few factors. One is the fact that most of the contaminant analysis was performed on samples collected after pretreatment. Due to the high cost of analytical chemistry, on only two occasions were untreated stormwater samples collected. Most were collected at the dry well after pretreatment had occurred, so it was expected that the concentrations of contaminants would be lower than untreated runoff. However, even the two untreated stormwater samples contained fewer than 10 contaminants above their reporting limits. The two study sites (a residential neighborhood and at a bus parking/servicing area) are common urban land uses that have been associated to varying degrees with contaminated stormwater (SSQP, 2017; Maestre and Pitt, 2006; Stein et al., 2006). Influent stormwater at these sites contained few polycyclic aromatic hydrocarbons (e.g., pyrene, fluoranthene, phthalates, and benzo[a]pyrene) and volatile organics (e.g., carbon tetrachloride, chloroform, and toluene). In contrast, pyrethroid, total petroleum hydrocarbons, and metals were regularly detected. The low frequency of detections could under report the actual frequency because of the small sample size. Given that the first year of the study occurred during a major drought in California, one might expect contaminants to build up on the landscape in the absence of rain and be released into stormwater at higher than average concentrations during subsequent average or wet years. However, this was not reflected in the monitoring results, possibly due to the hydrophobicity of PAHs.

Combustion by-products are a major source of urban organic contaminants (summarized in Edwards et al., 2016; Stein et al., 2006) so we expected elevated concentrations at the City's Corporation Yard. However, unlike many bus fleets, Elk Grove's buses are fueled with natural gas, which is known to produce lower emissions than diesel-fueled buses (Nylund et al., 2004). This could account for the low number of detections of PAHs and other urban organics associated with vehicle exhaust at the CY. On the other hand, pyrethroids and motor oil were consistently measured at the drywells. Among the pyrethroids, bifenthrin was reported most frequently. Other pyrethroids, such as cyfluthrin and cyhalothrin, were detected less frequently, consistent with previous reports (Weston et al., 2005). Stormwater collected at the SDB drywell showed more frequent detections of pyrethroids than samples from the CY. Given that bifenthrin is one of the most widely used pyrethroids for the control of ants and other insects in residential and commercial settings (Weston et al., 2005), these findings are not surprising. Other studies in the region have shown median concentrations of bifenthrin similar to what we observed at SDB (Weston and Lydy, 2012; SSQP, 2017). Motor oil and, to a lesser degree, a handful of volatile organics such as acetone and tert-butyl alcohol were also detected at elevated concentrations at the CY in influent stormwater, but this group of organics were seen only at very low levels at SDB. None were detected in groundwater at either site.

Metals have been commonly reported in urban stormwater runoff (Tiefenthaler et al., 2008; Sacramento Stormwater Quality Partnership,

2013). While numerous metals were detected at both study sites, most were below their reporting limits. One exception was aluminum, which was detected at up to a three-fold greater concentration than the 1 ppm regulatory standard at the CY. Given the fact that aluminum is present in the highest concentration of any metal in the earth's crust, and that aluminum is a common contaminant at locations where vehicles are housed (Federal Register, 1996), it is possible that the source of these elevated concentration was dirt that had been rinsed from the undercarriage and tires of buses.

4.2. Contaminants in groundwater

In general, there was a lack of a relationship between the contaminants in stormwater and those found in groundwater. For example, small concentrations of cyfluthrin, a pyrethroid, were detected in stormwater. With the exception of a low level detection of cyfluthrin in groundwater at SDB prior to the installation of the dry well, none was detected in groundwater. Numerous other organic contaminants, including methylphenol, dichlorobenzidine, benzofluoranthene, benzo[a]pyrene, and multiple phthalates (data not shown) were detected in stormwater below the reporting limit and above the detection limit, so were not quantifiable, but none were detected in either vadose zone or water table wells.

Similarly, there was little to suggest that dry wells facilitated the movement of anthropogenic metals into groundwater. While aluminum exceeded the regulatory standard on two occasions and exceeded the Public Health Goal of 600 µg/L on multiple occasions in both an upgradient and downgradient monitoring well, the fact that none was detected in the vadose zone well and that elevated concentrations were seen in both upgradient and downgradient wells suggests the source of aluminum in groundwater was unlikely to be linked to effluent from the dry wells. Furthermore, modeling results suggest that at the CY, even in scenarios with higher domain hydraulic conductivity values, the travel time for this contaminant to reach the Public Health Goals, health based advisory standards, at the water table would be over 330 years.

At SDB, the median concentration of iron in the vadose zone or upgradient wells was at or below the reporting limit, but iron was detected on two occasions in both downgradient wells. Modeling results suggest travel time to the water table would be five years, so it is unlikely these detections were related to dry well effluent. External sources, legacy metals in the surrounding soils, or desorption of naturally occurring iron could account for these findings. The fact that elevated concentrations of iron are commonly reported in drinking water in the region supports this interpretation (Elk Grove Water Service, 2016). Similarly, elevated concentrations of aluminum (110–640 µg/L) have been reported multiple times in monitoring and water supply well samples in the area (California State Water Resources Control Board, 2018). Other metals including boron, manganese, antimony, lead,

copper, and zinc, were also detected at low levels in stormwater but not in groundwater.

4.3. Role of vadose zone attenuation

Vadose zone modeling suggests a reason few of the 200 plus contaminants analyzed were detected in groundwater samples. There is an abundant amount of clay in the unsaturated zones surrounding the dry wells, 28% at the CY and 52% at SDB, which appeared to have limited the movement of organic and metal contaminants. Both dry wells were completed above or in a clay unit, forcing runoff to leave the dry well through holes in the sides and releasing the water into clay. Compared to sand or silt, clay has a high adsorptive capacity due its large surface areas ($10 \text{ m}^2/\text{g}$) and reactive surface functional groups (Essington, 2015). Clay appeared to attenuate the movement of pollutants via sorption, as reflected by the consistent difference between contaminant concentrations in water entering the dry well and concentrations collected from vadose zone and water table monitoring wells. When Lindemann (1999) sampled the subsurface material below the dry well, he found consistently elevated concentrations of PAHs, in some cases, many times greater concentrations than in the influent stormwater, suggesting sequestration of pollutants in the vadose zone. Taken together, these findings point to the important role the vadose zone plays in pollutant attenuation. This importance is reflected in the policy of some states in the United States, where the composition of the vadose zone is used as one of two key factors (the other being stormwater contaminant concentration) used to determine the type of pretreatment required at a dry well installation (Washington State Department of Ecology, 2006).

The key role the vadose zone plays in attenuation can be most easily seen when comparing the worst case with the average-case scenario in Los Angeles (Table 7). The vadose zone profile of the average-case Los Angeles conditions was very similar to the two sites in Elk Grove, all three containing less than 50% sand. The distance to the water table at SDB is about the same as the worst case scenario profile in Los Angeles, both less than 3 m. The key difference between the two is the sand content: 46% at SDB and 80% for worst-case conditions in Los Angeles. The effect of the high amount of sand in LA resulted in a decrease in the travel time. The difference in the peak concentration at the water table after 500 years was much less prominent, with the biggest differences seen for the contaminants with the highest K_{oc} values, such as B[a]P and bifenthrin. The greater percentage of sediment with higher f_{oc} value increased the contaminant sorption occurring within the model domain, thereby decreasing the amount of the contaminant that reached the water table. The model results for contaminants with high K_d or K_{oc} values can be compared to the results obtained for contaminants that do not readily sorb in the subsurface. TBA has a K_{oc} value orders of magnitude lower than the other modeled contaminants, and when multiplied by the f_{oc} values of the model domain sediment its K_d was close to zero, making its travel times comparable to that of the modeled conservative tracer.

The results showing greater contamination in higher permeability cases is not surprising, and is consistent with reports of others. Bandeen (1987) used UNSAT 2 to estimate the migration of stormwater through an unsaturated zone composed primarily of highly permeable material (e.g., gravel; Case 1), one composed of gravel underlain by sandy loam (Case 2), and a third that was a mix of sand and gravel underlain by sandy/clay loam (Case 3). The migration time through the 100 ft deep vadose zone was over 90-fold faster in Case 1 compared to Case 3. The authors concluded that a multi-layered subsurface profile with a significant amounts of clay would provide the maximum attenuation. Barraud et al. (1999) found that coarse material in the subsurface was ineffective at attenuating metals in stormwater. Similarly, Wogslund (1988) in a field study of two dry wells concluded that clay and amorphous oxides were very effective at attenuating metals, oil, and grease. Additional research efforts, summarized by Edwards et al. (2016) have

drawn similar conclusions. Knowledge of the vadose zone lithology is a key factor in estimating risk to groundwater quality. Further, vadose zone profiles that contain clay are effective at attenuating contaminant migration and reducing the risk of groundwater contamination.

The important role of clay layers shown by the model assumes that these layers are laterally extensive and not just small lenses or inclined toward the water table, allowing infiltrating water to find alternative preferential pathways through connected sands and gravels. In these heterogeneous alluvial sediments, it has been shown that preferential layer dipping can generate lateral gradients that allow water to recharge the water table faster than in the case of large lateral, essentially flat stratigraphic layers, especially clay layers (Kung, 1990; McCord et al., 1991). Laterally less extensive, flat-lying low permeability inclusions lead to lateral spreading (e.g., Russo et al., 1994; Harter and Yeh, 1996). For a dry well with an annual discharge rate $Q_{\text{dry-well}}$, the minimum extent, r , from a well of an essentially flat-lying local clay inclusion large enough to absorb the full dry well discharge can be computed by postulating that:

$$Q_{\text{dry-well}} < Q_{\text{clay}} \quad (1)$$

where Q_{clay} is the vertical discharge rate through the clay lens under saturated, unit-gradient gravity discharge conditions. Q_{clay} can be computed from Darcy's law:

$$Q_{\text{clay}} = A \cdot K_{\text{clay}} \cdot l = \pi r^2 K_{\text{clay}} \quad (2)$$

where K_{clay} is the vertical saturated hydraulic conductivity of clay, A is a minimum circular clay lens area with radius r from the well. It then follows that:

$$r > \text{sqrt} (Q_{\text{dry-well}} / [\pi K_{\text{clay}}]) \quad (3)$$

For $Q_{\text{dry-well}} = 1000 \text{ m}^3 \text{ a}^{-1}$ and $K_{\text{clay}} = 10^{-1} \text{ m a}^{-1}$, the minimum distance from the well to which a sufficiently protective clay lens must extend is on the order of 10^2 m or more. As stated previously, estimated mean lengths of the hydrofacies in the study area are 100 to 400 m (Meirovitz et al., 2017; Maples et al., 2019). The laterally extensive layering conditions assumed in the one-dimensional unsaturated zone flow and transport model are therefore not unrealistic in the examples simulated.

It is worth noting that even for the worst-case-scenario conditions in Los Angeles, where bifenthrin concentration at the water table was estimated to reach 100 ng/L after 500 years (data not shown), this contaminant's concentration is unlikely to pose a risk to human health. Risk to health was assessed by calculating the health protective concentration (HPC) of bifenthrin, using the methodology employed to develop Public Health Goals (PHG) in California, was estimated to be 314 $\mu\text{g/L}$. PHGs are drinking water standards that are based solely on risk to human health (OEHHA, 2021). Health protective concentrations were calculated since regulatory standards or PHGs have not been established in the US. In Australia, where drinking water guidelines for bifenthrin currently exists, the standard is 0.2 mg/L (NHRMC and NRRMC, 2011). Both of these criteria values are orders of magnitude greater than the modeled concentration. Thus, the health risk posed by bifenthrin is quite small.

Taken together, few contaminants were found in samples collected at the water table. Often, detections were observed but they were mostly below the reporting limits. For the majority of the contaminants that were analyzed by modeling, their concentration would never reach a level approaching a health advisory standard, such as the California Public Health Goal or a US Notification Level (NL), under the most-likely case scenarios. These include metals and DEHP. Under worst-case scenario conditions, however, iron would reach the secondary regulatory standard for taste after as few as 5 years at CY, or 190 days at SDB. Tert-butyl alcohol would reach the NL within days. Other contaminants, such as fipronil and imidacloprid, would not reach their respective health

protective concentrations.

It is difficult to validate modeling results for the two dry well sites given the short duration of water quality monitoring and the fact that any contaminants derived from dry well infiltration that arrive at the water table mixed with the groundwater, diluting the concentrations of whatever pollutants were present. As noted earlier, the majority of the contaminants that were modeled were not predicted to reach the water table within the time period of the field study. Of the four contaminants that were predicted to reach detectable levels at the water table within the two-year period under worst-case scenario conditions, two (fipronil and imidacloprid) were not measured in groundwater or stormwater samples. The third, TBA, was predicted to reach detectable levels at both sites within 12 days; in the case of iron, after 190 days at the SDB site. TBA wasn't detected in groundwater during the monitoring period. This may be because mixing of dry well effluent with groundwater diluted the influent TBA concentration to below reporting limits. The fourth contaminant was iron. It was frequently detected in stormwater and groundwater, and was detected at significant concentrations downgradient of the SDB dry well. Iron was detected at low concentration, but in a groundwater sample taken from a downgradient well in February, 2015. It was also detected above its secondary MCL in upgradient and downgradient groundwater in September, 2015 (higher concentration downgradient), after which it was not detected in SDB groundwater again. The detection of iron is consistent with the results of the vadose zone modeling.

4.4. Role of pretreatment

On average, concentrations of a representative group of five organic and metal contaminants were reduced by 50% as a result of passage through the water quality basin at SDB and by 67% as they passed through the grassy swale at the CY. Although the water quality basin was much larger than the swale, vegetation was more irregular and preferential flow pathways permitted water to reach the dry well with less-than-optimal pretreatment. In contrast, the grassy swale consisted of smooth, long grass within a deep swale that retained water for 5–15 min prior to entering the sedimentation well. This pretreatment was effective for most contaminants. For example, the median concentration of aluminum declined about three-fold as it passed through the grassy swale. Many other contaminants followed this pattern, including bifenthrin, motor oil, acetone, and aluminum (not all data shown). Our findings are consistent with reports in the literature. [Adolfson \(1995\)](#) compared the pretreatment features of three different dry well systems: with no pretreatment, with a structural pretreatment only (a sedimentation well), and with both a grassy swale and a sedimentation well. It was found that the greatest pollutant removal occurred with the third design where the removal efficiency was greater than 80% in some cases. Reports in the [International Stormwater BMP database \(2016\)](#) suggest that a grassy swale typically removes about 30% of total suspended solids, on which contaminants are frequently adsorbed. The swale at the CY removed about double that amount, approximately 63% of suspended solids. These results highlight the importance of including a pretreatment feature in a dry well system.

Unfortunately, the other portion of the pretreatment train, the 1.5 m deep sedimentation wells, were insufficiently deep to optimize deposition of sediment particles. Little sediment was ever observed at the bottom of either sed well. Further, commercial and municipal sedimentation wells are typically 3 to 4.5 m deep in order to provide sufficient depth for the settling of suspended material ([Torrent Resources, Phoenix, AZ, USA; Bureau of Environmental Services, City of Portland, OR, USA, 2006](#)). Given that most of the contaminants of concern in this study are particle bound, this can be an efficient way to remove not only particles, which can clog a dry well over time, but also contaminants ([Clark and Pitt, 2007](#)). [Wilson et al. \(1989\)](#), [Olson \(1987\)](#), [Adolfson \(1995\)](#), and other studies reviewed by [Edwards et al. \(2016\)](#) found efficient removal of suspended sediment and pollutants such as

polycyclic aromatic hydrocarbons, oil and grease, multiple phthalates and most metals, in these chambers. There does not appear to be any literature that examines the attenuation potential of grassy swales on water soluble pesticides, but it is doubtful if structural pretreatment would be efficient for water solubles. Nonetheless, for the majority of contaminants, structural pretreatment has been found reduce pollutant load.

4.5. Contaminants of special concern

Nitrate is one example of a pollutant that is notoriously difficult to manage. It was one of the few other contaminants that were detected at elevated concentrations in groundwater. Nitrate concentration in stormwater was consistently below the concentration found in groundwater at both study sites. At SDB, where groundwater concentrations of nitrate exceeded the MCL, there were no significant differences between the upgradient and downgradient nitrate concentrations, suggesting that infiltration through the dry wells was not a major source, although it could have been a contributor. Given the historical use of this area for agriculture, it is likely that nitrate accumulated in the unsaturated zone and has been leaching to groundwater for decades (e.g., [Harter et al., 2017](#)).

Nitrate is difficult to manage with vegetated features. Constructed wetlands are a frequently used option for removal of nitrogen in urban stormwater and agricultural runoff ([Vyazal, 2007](#)). However, processes in the dry well's pretreatment basin, which remain wet only during rain events, are likely different from wetland conditions. Due to its water solubility, ([Intl. BMP database, 2016](#)), neither bioretention cells nor grassy swales have been shown to be effective at attenuating nitrate. In fact, the concentration of nitrate is often greater leaving a bioretention cell than when it entered, primarily due to nitrification ([Li and Davis, 2014](#)). In the Sacramento region, nitrate + nitrite in stormwater has been consistently less than 1 mg/L over the past 15 years, considerably below the regulatory standard of 10 mg/L for nitrate-N ([SSQP, 2017](#)). Similar concentrations have been reported in other parts of the U.S. ([Li and Davis, 2014](#)). However, it appears that nitrogen concentrations in runoff are location-specific. For example, a Southern California study reported a median nitrate-N concentration from lawns of 5.42 mg/L ([Toor et al., 2017](#)). Due to the water solubility of nitrate, the varying concentrations possible in stormwater, its existing ubiquitousness in groundwater in many locations, and its well documented toxicity to infants ([Fewtrell, 2004](#)), special consideration is warranted in the selection of pretreatment and dry well siting to minimize its migration.

Lastly, our modeling results suggest that two water soluble pesticides, neonicotinoids and fipronil, moved relatively rapidly through the unsaturated zone. These pesticides have been detected in urban runoff with increasing frequency ([Teerlink, 2017; California Department of Pesticide Regulation, 2016; Masoner et al., 2019](#)). The modeling suggested they could move through the vadose zone in as little as 24 h ([Los Angeles scenario parameters included 2 m vertical separation in sandy soil](#)) up to as much as six to seven years at the Elk Grove sites. Research examining the mobility of these pesticides in soil has shown that the neonicotinoids are more mobile than fipronil primarily due to their high water solubility, 610 mg/L for imidacloprid vs. 3.8 mg/L for fipronil ([Bonmatin et al., 2015](#)). Their mobility is altered by the composition of soil; organic carbon and clays act to attenuate these pesticides to varying degrees, with little retention evident in sandy soils ([Fossen, 2006; Teerlink, 2017](#)). Because of their solubility, the California Department of Pesticide Regulation has placed imidacloprid on the Groundwater Protection List. To date, field studies have not been performed examining attenuation of these pesticides by various forms of pretreatment. This is an area of research that needs further attention.

While the mobility of both neonicotinoids and fipronil pesticides raises concern, their concentrations in stormwater are relatively low, reducing the potential risk to drinking water quality. [Masoner et al. \(2019\)](#) found the median concentration of imidacloprid in samples

collected from 50 storm events was approximately 25 ng/L; the fipronil concentration was approximately 20 ng/L. These concentrations were a magnitude lower than the ones used for modeling in this study, 0.9 and 0.5 µg/L respectively. The modeled concentrations reflected the higher end of values collected from sampling of stormwater in California, conducted by the Department of Pesticide Regulation in 2016. Estimates of the concentration of these two pesticides at the water table after 500 years were compared to the health protective concentrations. The HPC of imidacloprid is 358 µg/L, a value about 400-fold greater than the estimated concentration of 0.9 µg/L at the water table. This difference has a greater margin of safety compared to fipronil, where the concentration for the average case scenario after 500 years at both sites was slightly more than half of the HPC of 1.19 µg/L. This calculation assumed that the concentration of the pesticide entering the dry well was the same as what would be found in drinking water, assuming no dilution in the aquifer. Both are conservative assumptions. Considering the mobility and concentration in stormwater of these two pesticides, it does not appear that either would pose a significant risk to the quality of drinking water or human health. This could change in the future if the use of these classes of pesticides were to increase.

There clearly is a need to develop pretreatment facilities that could capture water soluble pesticides and nitrate. Methods to attenuate nitrate using nitrogen fixing plants such as alfalfa in vegetated features has been investigated (Kim et al., 2003). These investigators found greater than 90% efficiency of removal of nitrate with alfalfa use in bio-retention. Others are exploring the use of anion/cation pillows that could be added to dry well systems to attenuate water soluble pesticides (J. Mayer, 2014, pers. comm.). While the risk to human health will vary with specific conditions, the need to better understand the behavior of these types of contaminants and identify ways to efficiently manage them would be worthwhile.

4.6. Risk of mobilizing naturally occurring toxic metals

Constituents in stormwater such as ions and certain metals have the potential to mobilize naturally-occurring arsenic and chromium. These two metals were regularly detected in groundwater but not in stormwater. For example, at both sites, arsenic concentrations in stormwater were too low to quantify; however, it was detected in all groundwater wells at concentrations of 2 to 4 µg/L. In many locations in California, and the southwestern USA in general, arsenic is endemic in drinking water. In the study region, the median arsenic concentration in groundwater, based on samples collected over the last 20 years, was about 5 µg/L, half of the regulatory standard of 10 µg/L, however, one public supply well less than a half mile from the CY site had a concentration of arsenic as high as 64 µg/L in 2008 (California State Water Resources Control Board, 2018). Arsenic is a minor component of granite, a major geologic feature of the Sierra Nevada mountain range in eastern California, and it is also present in metamorphic rocks, often bound to iron complexes, on the western slopes of the Sierras. Under oxidizing conditions in the presence of iron, arsenic is primarily retained in the solid phase. A shift to reducing conditions, however, could cause arsenic to be released through a variety of mechanisms (Bose and Sharma, 2002). Both ions such as sulfate or bicarbonate and metals contained in stormwater could alter the redox state, thus mobilizing arsenic.

A weak positive correlation between iron and arsenic suggested that iron in stormwater effluent could cause a reduction of arsenic-bearing iron oxides, resulting in the mobilization of arsenic (Bose and Sharma, 2002; Pierce and Moore, 1982). Modeling results of the worst case scenario suggested that iron would reach the water table at the CY in about five years. Therefore, in this two-year study, it was unlikely that stormwater effluent influenced the redox conditions. The case was different at the SDB, where iron was estimated to reach the water table within a few months. Yet, no correlation was found between iron and arsenic at this site (data not shown). This could be due to the small

sample size or the lack of a meaningful relationship. While this short study produced no evidence of stormwater constituents mobilizing toxic metals, other research suggests that this could occur. Jurgens et al. (2008) found weak positive correlations between iron and dissolved arsenic in a study of the aquifer in Modesto, CA, USA. They also observed a positive correlation between select infiltrated metals and uranium concentration in groundwater. Further research would help clarify these relationships.

5. Conclusion

This study presents the most comprehensive data and analysis to date on water quality phenomena associated with dry wells for over 200 potential contaminants at actual field sites. The data include water quality sampling of not only the storm (influent) and dry well waters, but also the surrounding vadose zone and underlying groundwater, allowing a total-system evaluation of possible water quality threats from these structures that are increasingly being considered for enhancement of groundwater recharge to address critical groundwater overdraft problems. Results suggest that using dry wells to infiltrate stormwater would pose minimal risk to groundwater quality when proper pretreatment is employed and source water does not contain potentially mobile groundwater contaminants. Most contaminants commonly found in urban runoff would either be attenuated in a pretreatment system or in the vadose zone for varying degrees of time (ranging from decades to centuries). The exceptions would be for highly water soluble, non-degradable, and/or possibly mobile contaminants such as nitrate, usually found in low concentration in most urban runoff, and water soluble pesticides such as the neonicotinoids. This also includes the group of less mobile *per-* and polyfluoroalkyl substances (PFAS), which are an emerging contaminant of concern due to their long-term persistence (Xiao et al., 2015). Research is needed to find new methods of pretreatment that assist in minimizing the risk of mobile or persistent water soluble pollutants leaching from dry wells. Additional research to quantify the capacity of the unsaturated zone to adsorb contaminants would also help clarify potential long term risks. Our field and modeling work suggests that given appropriate consideration for construction (including pre-treatment) and siting, dry wells can serve as a valuable tool to advance the effective use of water resources.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

This work would not have been possible without the support and efforts of numerous people. At OEHHA, the following staff and interns made important contributions: Ari Ashoor, Nelson Pi, Kathleen Dorn, David Katz, and Hamad Hamad, Carmen Milanes, Elaine Khan, Chris Banks, and David Siegel. At cbec ecoengineering: Chris Campbell, Ben Taber, and Rafael Rodriguez. At Ludhorff and Scalmanini Consulting Engineers: Reid Bryson and Casey Meirovitz.

There were no conflicts of interests for any authors involved in this study.

Support for the study was provided by the California State Water Quality Control Board, Proposition 84 California State Water Bond funds, with additional funding from the City of Elk Grove, and the Office of Environmental Health Hazard Assessment, California Environmental Protection Agency.

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