Review papers

# Assessing the effectiveness of drywells as tools for stormwater management and aquifer recharge and their groundwater contamination potential

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## ARTICLE INFO

## SUMMARY

Drywells are gravity-fed, excavated pits with perforated casings used to facilitate stormwater infiltration and groundwater recharge in areas where drainage and diversion of storm flows is problematic. Historically, drywells have predominantly been used as a form of stormwater management in locations that receive high volumes of precipitation; however the use of drywells is increasingly being evaluated as a method to supplement groundwater recharge, especially in areas facing severe drought. Studies have shown that drywells can be an effective means to increase recharge to aquifers; however, the potential for groundwater contamination caused by polluted stormwater runoff bypassing transport through surface soil and near surface sediment has prevented more widespread use of drywells as a recharge mechanism. Numerous studies have shown that groundwater and drinking water contamination from drywells can be avoided if drywells are used in appropriate locations and properly maintained. The effectiveness of drywells for aquifer recharge depends on the hydrogeologic setting and land use surrounding a site, as well as influent stormwater quantity and quality. These parameters may be informed for a specific drywell site through geologic and hydrologic characterization and adequate monitoring of stormwater and groundwater quality.

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# 1. Introduction

## 1.1. Background

The natural hydrologic cycle has been altered in much of the world due to climate change and human land development (Maloney et al., 2014; Rusu et al., 2012). Urban development limits the permeability of ground surfaces; precipitation that would normally reach natural land surface and infiltrate into the underlying aquifer instead runs off, traveling over paved areas or areas with low surface soil permeability until it evaporates, or enters surface water bodies or stormwater management facilities (Clark and Pitt, 2007; Rusu et al., 2012). Rapid human population growth is further stressing the allocation of water resources, and groundwater usage in some areas is occurring at potentially unsustainable rates (Gorelick and Zheng, 2015; Rusu et al., 2012). One of the ways to address the challenges of managing stormwater runoff and replenishing depleted groundwater resources is through the use of deep infiltration practices such as drywells. Drywells are vadose zone infiltration wells that end before the water table and are used extensively throughout the United States and other parts of the world to dispose of stormwater in areas with low ground surface permeability. However, more recently their potential to provide additional aquifer recharge has been recognized (Natural Resources Defense Council, 2014). There is some concern that drywells allow stormwater pollutants more direct passage to the water table without undergoing surface soil and near surface sediment attenuation processes. In some cases, drywells have been linked to groundwater and drinking water contamination events (Environmental Protection Agency, 1999a,b). This has created controversy over their use and made some regulatory agencies reluctant to support further installations (EPA, 1999a,b).

## 1.2. Rationale

While drywells are a prevalent form of stormwater infiltration device in some parts of the world, relatively few studies have been performed to quantify either the quantity of recharge entering aquifers from drywell infiltration, or the potential for this infiltration to contaminate groundwater and drinking water. Groundwater contamination events associated with the use of drywells have been reported, however in many cases these events are the result of mismanagement of the facilities and can be traced to surface pollutant spills or illicit dumpings (Adolfson, 1995; EPA, 1999a,b; Jurgens et al., 2008). Some regulations pertaining to drywell installation, design, and usage were instated before many comprehensive drywell studies had been performed, and therefore can be lacking quantifiable basis and may mandate separation distances that are based on those set for other forms of stormwater or wastewater management (EPA, 1999a,b; Minnesota Department of Transportation, 2009). In some locations where successful full or pilot scale drywell studies have been performed, preexisting regulations or permitting processes have been reformed based on the studies' conclusions (Brody-Heine et al., 2011; City of Portland Bureau of Environmental Services, 2014; Wilson et al., 1992). It has been shown that drywells can offer an effective solution for both stormwater management and aquifer recharge; however, little has been done to synthesize these findings. The purpose of this paper is to review the available literature pertaining to drywell performance in terms of both stormwater management and groundwater quality control. General stormwater quality will be summarized along with the findings of studies focused on the impact of drywells on groundwater recharge quantity and quality and their performance compared to other forms of stormwater infiltration devices. We use the information reviewed to discuss the factors that affect the potential for drywells to cause groundwater contamination, and the possible means by which to predict the timescale and magnitude of contamination.

## 1.3. Drywell design and usage

A drywell by simple definition is a well that is deeper than its widest surface dimension and is used to transmit surface water to the subsurface (EPA, 1999a,b). An important distinction must be made between drywells and soakaways. Soakaway is a term commonly seen in European stormwater management literature, and refers to an infiltration system that transmits stormwater to the subsurface; however, soakaways are not necessarily deeper than they are wide, and so while some soakaways may be classified as drywells, some are too shallow and wide to qualify. In this paper, soakaway will be used as a broad term, and the specification will be made whether or not a described soakaway is also a drywell. A typical drywell design consists of a perforated precast casing, usually made of concrete but in some cases PVC, with an average diameter of approximately 1.2 meters (m), and a depth of anywhere from 0.6 to 26 m, usually backfilled with gravel and/or sand (Adolfson and Clark, 1991; Adolfson, 1995; Bandeen, 1984, 1987; Barraud et al., 1999; Chen et al., 2007; City of Portland, 2008; Clark and Pitt, 2007; Dallman and Spongberg, 2012; Izuka, 2011; Jurgens et al., 2008; Lindemann, 1999; Pitt et al., 2012; Wilson et al., 1990; Wogsland, 1988). Fig. 1 depicts the design of a typical drywell. Drywells are also referred to as underground injection control wells (UICs), and are classified by the United States Environmental Protection Agency (USEPA) as class V wells, which are defined as shallow wells used to place fluids directly below the land surface (EPA, 1999a,b). They are further categorized as stormwater drainage wells (SWDWs), which are bored and dug wells and improved sinkholes designed to manage stormwater runoff (EPA, 1999a,b). In 1999, there were an estimated 247, 522 SWDWs in the United States. A more current national estimate has not been made, and a worldwide estimate is not available.

As drywells have become more prevalent, their design has increased in complexity, and more modern drywells usually include some form of Best Management Practice (BMP) or pretreatment in their design. Sedimentation can be a major problem for drywells, and so sedimentation traps, manholes, filters, or settling chambers are often constructed to receive influent stormwater
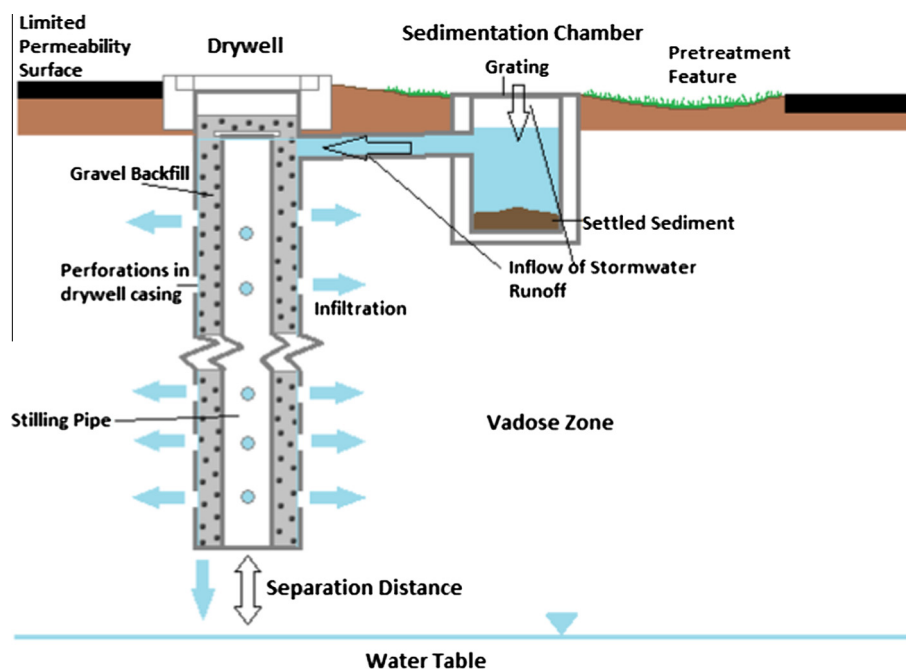
**Fig. 1.** The design of a typical drywell, including a grass swale and sedimentation chamber pretreatment. Drywells are typically backfilled with gravel or cobbles, and extend to a completion anywhere from 61 to zero meters above the water table.

before it reaches the drywell (Adolfson and Clark, 1991; Adolfson, 1995; Bandeen, 1987; City of Portland, 2008; Dallman and Spongberg, 2012; Olson, 1987; Wilson et al., 1990). See Fig. 1 for example pretreatment design. A drywell's separation distance is defined as the vertical distance between a drywell's bottom perforation and the local seasonal high water table (see Fig. 1) (City of Portland, 2008). Many drywell regulatory guidelines stipulate a mandatory separation distance for drywell installation (City of Portland, 2008, 2014; EPA, 1999a,b; Washington State Department of Ecology, 2006).

Although commonly built to control stormwater runoff, drywells have also been characterized as artificial recharge systems, defined in Bouwer (2002) as "engineered systems where surface water is put on or in the ground for infiltration and subsequent movement to aquifers to augment groundwater resources" (p. 122). There are many different forms of artificial recharge systems. The broader category can be split into three subcategories based on the depth of penetration of the recharge structure: surface infiltration devices, vadose zone infiltration devices, and injection wells. Surface infiltration devices include infiltration basins, detention basins, vegetated swales, managed aquifer recharge ponds, and any other system where water is put on the ground surface for infiltration into underlying groundwater (Bouwer, 2002). Vadose zone infiltration devices are open to the atmosphere; however most of their surface area extends either vertically or horizontally under the ground's surface. These devices include recharge trenches and drywells. Injection wells are wells that are used to recharge water directly into the aquifer, and are not to be confused with drywells (Bouwer, 2002).

Each type of artificial recharge system has its own set of benefits and potential drawbacks. Surface infiltration practices may lose potential recharge to evaporation, as they store and dispose of water at the surface. Their installations typically requires highly permeable sediments and a large surface area—both of which makes them suited to agricultural or low development areas, but somewhat limits their use in urbanized areas. Surface infiltration devices, however, provide the most opportunity for contaminants to be attenuated due to reactive transport through surface and subsurface soils and sediments, and vegetated facilities provide opportunities for bioretention (Bouwer, 2002). When recharge is an objective, however, the surface systems have the disadvantage of evaporative water losses. In contrast, injection well systems have no or minimal evaporative losses, but provide little opportunity for natural attenuation processes to remove pollutants that typically occur in urban runoff, and so stormwater contaminant levels must be below regulatory standards before injection. Recharge trenches and drywells completed in the vadose zone also have minimal evaporative losses, while allowing some contamination to be attenuated in unsaturated subsurface sediments prior to entering groundwater. Recharge trenches and drywells do not require large installation areas, which can make them especially suitable for urban settings.

## 2. Review of the literature

### 2.1. Stormwater quality

Several stormwater quality surveys have been conducted within the last twenty years, and the pollutants commonly found in these studies are similar to those detected in water entering drywells (EPA, 1999a,b; Hamilton et al., 2004; Pitt et al., 2005). In 2005, the United States' National Stormwater Quality Database released information regarding the pollutants found in stormwater across the United States. During the course of the study, 3765 storm events were sampled in 200 municipalities in 17 states in residential, commercial, industrial, institutional, freeway, open space, and mixed land use areas (Pitt et al., 2005). The contaminants detected include oil and grease; fecal coliform; fecal streptococcus; total coliform; E. Coli; nitrogen as ammonia, nitrite, nitrate, and total Kjeldahl nitrogen; phosphorus; antimony; arsenic; beryllium; cadmium; chromium; copper; lead; mercury; and zinc (Pitt et al., 2005).

Data summarized in the USEPA's 1999 Class V well report indicates that antimony, arsenic, beryllium, cadmium, chromium, cyanide, lead, mercury, nickel, nitrate, selenium, and organics such as

benzene, benzo(a)pyrene, bis(2-ethylhexyl)phthalate, chlordane, dichloromethane, pentachlorophenol, tetrachloroethylene and trichloroethylene concentrations have exceeded primary regulatory levels in stormwater, and that aluminum, chloride, copper, iron, manganese, total dissolved solids (TDS), zinc, and methyltertbutyl ether have exceeded secondary regulatory levels and United States health advisory levels (EPA, 1999a,b).

The National Water-Quality Assessment (NAWQA) program conducted by the United States Geological Survey (USGS) is an ongoing program that seeks to assess both surface and groundwater quality across the United States (Hamilton et al., 2004). Included in their study units are streams fed by stormwater runoff; this streamwater quality data can be used to ascertain land use and locational conditions that contribute to the presence of certain stormwater contaminants, as well as seasonal and temporal trends in the contaminants and contaminant concentrations found in stormwater (Hamilton et al., 2004). Ninety-four percent of agricultural runoff samples collected between 1991 and 2001 across the United States contained pesticides (Hamilton et al., 2004). Nitrogen was detected in most surface water samples, and 20% of shallow groundwater samples contained nitrogen concentrations above the MCL (Hamilton et al., 2004). Volatile organic compounds (VOCs) including trichloroethylene (TCE), trichloroethane (TCA), perchloroethylene (PCE), and methyltertiary butyl ether (MTBE) were commonly found in urban runoff, as well as phosphorous and insecticides, which are linked to homeowner use and also found in suburban and commercial areas (Hamilton et al., 2004).

Differences between contaminants found in areas of similar land use across the United States can be attributed to differences between local climate, geology, hydrology, and soil types. For example, the analyzed NAWQA data showed groundwater nitrate concentrations to be low in agricultural areas in the southeastern U.S. due to the high organic content of the soils, whereas streams in California's Central Valley, the Northwest, and the Great Plains experience relatively high concentrations of nitrate because of low organic content in soil (Hamilton et al., 2004). Local geology plays a part in the presence of naturally occurring elements such as phosphorous, radon, arsenic and uranium that can be mobilized from soils due to runoff. An example of the effects of the hydrologic setting and seasonal variations on surface water contamination can be seen in the pattern of atrazine concentrations in streams. Concentrations of the herbicide peaked in the Mississippi River basin at Baton Rouge during May and June; this location receives runoff from large areas across the Midwest (Hamilton et al., 2004). In an area that is more hydrologically isolated, atrazine concentrations peaked earlier and were higher due to the high concentration of atrazine in urban and agricultural runoff inputs and the lack of dilution from mixing with cleaner inputs (Hamilton et al., 2004). Water temperature can also have an effect on contaminant concentrations. VOCs were detected in a stream in Pittsburgh, PA, during winter months when the water was colder, but not during summer months due to the volatilization of VOCs in warmer temperatures (Hamilton et al., 2004). Seasonal trends found in pesticide contamination can be seen in diazinon concentration patterns. Concentrations of diazinon increased after summer applications in Ohio, Michigan, Indiana, Pennsylvania, and New York, whereas diazinon increased during winter months in the San Joaquin-Tulare Basins in California after dormant orchard sprayings (Hamilton et al., 2004).

Certain surface water contaminants are also associated with various land uses; if an analyte is typically applied or deposited due to activities performed in a certain anthropogenic land use category, it may end up in surface water runoff. Table 1 presents contaminant categories that are typically associated with agricultural land use, and commercial, industrial, and residential urban land use. Many pesticides are often associated with agricultural use, while heavy metals and petroleum by-products are more likely to be found in urban runoff. Stormwater quality data for Elk Grove, California, can provide an example of contaminants associated with predominantly residential areas, and shows the presence of eight herbicides and pesticides, including bifenthrin, diazinon and simazine; six polycyclic aromatic hydrocarbons (PAHs), including naphthalene and pyrene; as well as copper; E. Coli; fecal coliform; iron; lead; methylmercury; nitrate and nitrite; phosphorus; and zinc (Sacramento Stormwater Quality Partnership (SSQP), 2009). E. coli, fecal coliform, iron, and lead reached, exceeded, or met their drinking water regulatory maximums. The other listed contaminants tested for were well below regulatory levels (SSQP, 2009).

The physicochemical properties of stormwater contaminants also affect their presence in stormwater. Some contaminants, such as nitrate and chloride, are highly soluble in water, and are therefore more mobile and likely to be present in stormwater than contaminants such as 1,3-dichlorobenzene and chlordane, which have very low water solubilities (Clark and Pitt, 2007). Contaminants that have low degradation rates are also more likely to be present in stormwater than contaminants that degrade quickly. Contaminants that are susceptible to hydrolysis and photolysis are also less likely to be present in stormwater.

Many contaminants are commonly found in surface water runoff at varying concentrations; however ultimately the impact of drywell facilitated stormwater infiltration is dependent on the ability of the local subsurface material to attenuate pollutants, as discussed in Section 3.1, and the drywell's functionality, as discussed in Section 2.2.

### 2.2. Runoff control performance

Drywells can be an effective means of diverting stormwater runoff to the subsurface; however it is well known that stormwater

**Table 1**
Categories of contaminants commonly found in runoff from various land use types.

| Land Use Type | Commonly associated stormwater contaminant categories | Specific frequently detected contaminants | Sources of contaminants |
|---|---|---|---|
| Agricultural | Herbicides, insecticides, nutrients, VOCs | Atrazine, cyanazine, diazinon, fipronil, nitrogen, phosphorous | Crop applications, fertilizers, livestock, dairy, and poultry production |
| Urban: commercial | Bacteria, metals, nutrients, petroleum by-products | Arsenic, chromium, copper, lead, nickel, zinc, nitrogen, phosphorus | Pavement runoff |
| Urban: industrial | Bacteria, metals, nutrients, PAHs, petroleum by-products | Fluoranthene, pyrene, antimony, beryllium, boron, cadmium, chromium, copper, lead, mercury, nickel, zinc | Pavement runoff, industrial processes |
| Urban: residential | Bacteria, dissolved minerals, herbicides, metals, nutrients, petroleum by-products | Chloride, sulfate, *E. Coli*, fecal coliform, fecal streptococcus, phosphorus, oil and grease, beryllium, copper, lead, mercury, zinc | Yard applications, pavement runoff, septic systems, household usage |

Hamilton et al. (2004), Pitt et al. (1999) and Pitt et al. (2005).

infiltration devices (basins, trenches, infiltration wells) used for extended periods of time are subject to clogging and other physical malfunctions (Bouwer, 2002; Chen et al., 2007; Gonzales-Merchan et al., 2012; Le Coustumer and Barraud, 2007; Lindsey et al., 1992). Clogging in drywells can cause ponding of stormwater, reduction in infiltration capacity, and reduction in pre-treatment functionality (Gonzales-Merchan et al., 2012; Lindsey et al., 1992). It is possible to estimate the likelihood of an infiltration device becoming clogged based on local stormwater runoff quality and quantity parameters. A study done in Chassieu, France correlated the clogging rate of an infiltration basin to the volume, total suspended solids (TSS) load and chemical oxygen demand (COD) of influent stormwater as well as areal climatic factors (Gonzales-Merchan et al., 2012). The hydraulic resistance of the basin's clogging layer was used as an indicator of the degree to which the basin had been clogged; hydraulic resistance is a material property calculated from the thickness of the layer divided by its hydraulic conductivity, and was calculated for the study's basin in order to characterize the clogging layer that developed along the basin's bottom (Bouwer, 2002; Gonzales-Merchan et al., 2012). The study monitored the progression of hydraulic resistance in the basin over a period of seven years (2004–2010) while taking yearly measurements of TSS and COD (Gonzales-Merchan et al., 2012; Le Coustumer and Barraud, 2007). The results showed that the hydraulic resistance through the bottom of the basin increases as a function of increasing TSS load and COD in the influent water (Gonzales-Merchan et al., 2012). Data also indicates that the presence of vegetation can reduce the rate of clogging, as the basin's hydraulic resistance increased at a lower rate with the presence of spontaneous vegetation than without (Gonzales-Merchan et al., 2012). The results have application to drywells, as drywells can experience similar clogging layers, and pre-treatment for both infiltration device types is similar.

A study done in the United Kingdom in 2007 evaluated the functionality of four soakaways that had been in use in a residential area since the 1930s (Chen et al., 2007). It is common for soakaways in Great Britain to lose infiltration capacity due to clogging by silt and floating material (Chen et al., 2007). Field infiltration tests were performed at each soakaway, all of which qualify as drywells due to their dimensions, to determine the time needed for the soakaway to drain to a depth halfway below the stormwater inlet pipe (Chen et al., 2007). The drywells were filled to a depth just below the location of the stormwater inlet pipe, and if the time needed to drain half of the filled volume was less than 24 h, the soakaway was determined to be functioning acceptably (Chen et al., 2007). All four of the tested drywells met this hydraulic performance criteria after 70 plus years of use (Chen et al., 2007).

In a study performed to compare the effectiveness of different Best Management Practices (BMPs) for stormwater management in Maryland, drywells had a higher success rate than many other forms of artificial recharge devices (Lindsey et al., 1992). Of the 258 BMP facilities inspected, 22 were drywell sites. Of the 22 drywells, 17 were functioning as designed (77%), 17 were performing water quantity control as designed, and 20 were providing water quality benefits (91%), which was the highest percentage out of the eight reviewed categories of facility: dry basins, wet detention basins, infiltration basins, infiltration trenches, drywells, underground storage, vegetated swales, and an "other" category (Lindsey et al., 1992). Only six of the drywells needed maintenance (27%), the lowest percentage in all of the categories. In terms of performance criteria compliance, none of the drywells experienced structural failures, only three (14%) experienced slow infiltration, clogging of the facility occurred at four wells (18%), and four experienced excessive sediment or debris, again the lowest percentage of the eight categories (Lindsey et al., 1992). In terms of maintenance criteria, drywells performed best in five of the six vegetative conditions requirements, and three of the seven sediment conditions requirements. Drywells had the smallest amount of sediment entering any of the facilities (six wells, or 27%) (Lindsey et al., 1992).

### 2.3. Groundwater recharge quantity

There have been few studies performed that have attempted to quantify the volume of runoff that a drywell is able to infiltrate into the subsurface to provide groundwater recharge. A study was done of the recharge potential of drywells and septic tanks in the Portland Basin in Oregon, US (Snyder et al., 1994). A representative urban area that housed 5700 drywells at the time was chosen for analysis, and a simple mass balance was performed for which it was assumed that all of the runoff entering drywells would infiltrate into the underlying aquifer (Snyder et al., 1994). The basin receives approximately 1.07 m of rain annually; the runoff into drywells was set to equal the volume of precipitation that fell on impervious surfaces minus the volume of precipitation that was detained on the surface or evaporated before infiltrating (Snyder et al., 1994). The results of the analysis indicated that approximately 75% of precipitation falling on impervious surfaces in the Portland Basin enters drywells (Snyder et al., 1994). This translates to 53 cm per year (cm/year), or 38% of total groundwater recharge in urban areas (Snyder et al., 1994).

A similar analysis was performed in Los Angeles, California, which found that 48% of precipitation in the LA Basin runs off impermeable land surfaces (the Los Angeles and San Gabriel Watershed Council, 2010). Theoretically, assuming that 1.9 cm (cm) of rainfall on every land parcel from each storm event could be turned into infiltration through the use of drywells, as much as 473.7 gigaliters/year (GL/year) could be added to aquifers (the Los Angeles and San Gabriel Watershed Council, 2010). Currently, however, the Los Angeles Basin does not have enough drywells or other stormwater infiltration devices installed to offer that scale of groundwater recharge benefit.

A more recent study taking place in Bend, Oregon, calculated that the upwards of 5000 drywells within City boundaries only contributed 1.8% of yearly recharge and inflow into the aquifer underlying the city (11.9 out of a total 648.1 GL/year) (Brody-Heine et al., 2011). Lateral subsurface flow into the groundwater basin was reported as providing approximately 61% of recharge and inflow, the remaining 39% being provided by surface water sources such as stream losses and precipitation (Brody-Heine et al., 2011). Of this 39%, drywell infiltration was 5% and surface infiltration of precipitation was 3% (Brody-Heine et al., 2011). As estimates of drywell recharge volume were based on precipitation runoff analyses (meaning that drywell infiltration and surface infiltration account for all of the city's precipitation infiltration, 8% of the total groundwater recharge and inflow), drywells therefor were estimated to capture and infiltrate more than half of the city's total precipitation (Brody-Heine et al., 2011).

### 2.4. Groundwater recharge quality

#### 2.4.1. Drywell infiltration of surface spills

There have been reports of underground sources of drinking water being contaminated as a result of local drywells; however, most of these events occur either because of contaminant spills in the vicinity of a drywell, or because of abuse and undesignated use of the drywell (Adolfson, 1995; EPA, 1999a,b). In Fairborn, Ohio, a commercial petroleum distributing facility released 79,500 liters (L) of fuel oil from an aboveground storage tank (EPA, 1999a,b). The fuel overflowed the emergency dikes and entered two stormwater wells. In Los Gatos, California, groundwater was contaminated with gasoline and other chemicals

originating from a commercial site where surface spills of fuel washed into drywells (EPA, 1999a,b). In Morgan Hill, California, industrial wastewater containing TCE, TCA, and cis-1,2-dichloroethane (DCE) from an auto-glass company was discharged into a stormwater retention pond with three drywells at its bottom (EPA, 1999a,b). At the McChord Air Force Base in Washington, organic waste solvents and sludge that had been disposed of in leach pits and storm drains was found in drinking water sources (EPA, 1999a,b). Motor oil was found in five wells in Modesto, California, and 70 gallons of used crankcase oil were found in a drywell in Pierce County, Washington (Adolfson, 1995; EPA, 1999a,b). The sources for both are theorized to be illicit dumpings.

### 2.4.2. Drywell field studies

*2.4.2.1. Description of drywell field studies.* Even when spills do not take place and drywells are used in their intended manner, groundwater contamination can occur owing to contaminants in stormwater runoff. Many studies have been implemented to test the efficacy of drywells as a safe form of stormwater infiltration. Table 2 summarizes the design, duration, and scope of the studies reviewed for this paper. The studies differ from one another in many aspects, including the design of the drywells, the setup and extent of monitoring, the contaminants monitored, the duration of the study, the methods used to evaluate the results, and the geologic, hydrologic, and land use characteristics of the study sites (see Table 2). The studies were performed at sites in France, the United Kingdom, and the United States.

*2.4.2.2. Study hydrogeologic settings.* Because of the vast differences in subsurface geology around the world, not all studies of drywells are comparable. A study was done on the Island of Hawai'i, Hawai'i, where the subsurface is predominantly basalt from the five basaltic, mid-plate, hotspot shield volcanoes that formed the island (Izuka, 2011). The lack of weathering and high porosity of the island's geologic material result in aquifers with high permeability, especially in areas of flank lava flows. These areas can have hydraulic conductivities of up to 1000 m per day (Izuka, 2011). Hawai'i has a unique hydrogeology compared to most continental locations, and so the infiltration volume, contaminant concentrations, and plume dynamics determined from the performance of a model study for conditions on the island would not be easily applicable to most other locations.

Drywell studies performed in Arizona, US, dealt with layered stratigraphy, which can be found in many parts of the world. Arizona houses a high percentage of the United States' drywells and uses them extensively to manage stormwater runoff (Arizona Department of Water Quality, 2015). The studies examined here dealt specifically with the Tucson Basin, which has variable subsurface permeability: perching low permeability strata exist amidst more permeable materials. These low permeability layers are composed of either clay or caliche (Bandeen, 1984, 1987; Olson, 1987; Wilson, 1983). The vadose zone at the Arizona field site contained alternating beds of sand, silt, clay, and gravel (Bandeen, 1984, 1987). The transition between relatively permeable sand and gravel to less permeable sand, silt, and clay mixtures occurs at a depth of about 9.1 m, and the water table is approximately 33.5 m below ground surface (Bandeen, 1984, 1987). The layers are laterally discontinuous, extending over an area of no more than 8000 square meters ($m^2$). Their depths range from less than 1.5 m to 50 m below ground surface in the Tucson Basin (Bandeen, 1984, 1987). These impermeable and low permeability layers can cause a large amount of lateral flow, as was shown with vadose zone modeling. Lateral flow can potentially lead to the contamination of drinking water due to rapid travel to a highly permeable well pack and direct flow to the underlying aquifer (Bandeen, 1984).

A multi-year study conducted at a site in the Eastern San Joaquin Valley, California, found that varying permeabilities and layering in the vadose zone, among other things, leads to the creation of preferential flow paths that extend to the water table separated by areas of more stagnant flow (Harter et al., 2005; Botros et al., 2011). While the study was conducted specifically in reference to nitrate and other agrochemicals, preferential flow paths and rapid lateral flow occurring adjacent to low permeability materials has the potential to both shorten the arrival times of peak contaminant concentrations while also prolonging transport of the total mass for many groundwater contaminants owing to sequestration in the slower zones (Harter et al., 2005).

While heterogeneity in the vadose zone can cause potential contamination hazards, homogeneity can in some cases have greater detrimental impacts due to a loss of contaminant attenuation. Washington, US, has more drywells than any state in the country. The study done in Pierce County, Washington, where open bottom drywells had been in use for 30 years preceding the study, had site soil composition of mostly coarse gravel and sand (Adolfson, 1995). It was found that contaminants were not sufficiently attenuated between the bottom of the drywell and the seasonal high water table (Adolfson, 1995). Further vadose zone modeling performed for sites in Arizona used domains with different layerings of gravelly sand, sandy loam, and loamy clay, and showed that pollutant attenuation was related to soil particle size and plume exposure degree (see Section 2.4.6 for further details) (Bandeen, 1987). Recommendations were made for drywell installation in locations that contained clay layers in the subsurface to increase contaminant adsorption.

*2.4.2.3. Study monitoring methods.* Each of the reviewed case studies that were based on pilot scale or full scale drywell installations had some form of monitoring system in place for the duration of the study. The complexity of monitoring implementation varied from four drywells sampled one time each as was the case for the study conducted in Pima County, Arizona, to a single drywell sampled for 17 storm events over the course of five years as was the case for the study done in Pierce County, Washington, to 30 drywells sampled for five storm events each year for ten years as was the case for the study conducted in Portland, Oregon (Adolfson, 1995; City of Portland, 2008; Wilson et al., 1990). During the Pierce County study, stormwater quality monitoring samples were taken at the point where the water entered the drywell, and also beneath the drywell, after the water had infiltrated. A groundwater monitoring well was not installed due to site constraints (Adolfson, 1995). The influent stormwater samples were taken by lowering a bailer to the bottom of the well and withdrawing it; this was done in order to ensure that the collected sample represented the entire water column. Samples labeled as "treated" were taken from a T pipe directly beneath the drywell (Adolfson, 1995). This pipe was very difficult to purge, and it is possible that samples taken from this point were affected by the resuspension of concentrated particulates (Adolfson, 1995). Samples were taken within three hours of the onset of precipitation for 17 storm events over the course of five years. Samples were frequently unavailable from the drywell during the beginning of the study due to the rapid draining of the system (Adolfson, 1995). As the system experienced siltation and infiltration proceeded at a slower rate, this problem subsided. Sediment samples were also taken from the bottom of the sedimentation chamber (Fig. 1) three times throughout the five years of monitoring (Adolfson, 1995). Analytes and characteristics tested for in stormwater included pH, temperature, conductivity, fecal coliform, nitrate-nitrogen, total Kjeldahl nitrogen, total and dissolved arsenic, copper, lead and zinc, chemical oxygen demand, total suspended solids and total petroleum hydrocarbons (Adolfson, 1995). Analytes and characteristics tested for in soil

**Table 2**

Summary of drywell studies examined for this review.

| Study location, years of study, and report associated with study | Number of drywells used | Drywell design | Subsurface conditions | Land use of study sites | Contaminants/parameters in analytical suite | Monitoring implementation |
|---|---|---|---|---|---|---|
| Valence, France 1996 Barraud et al. (1999) | 1 | 1 m diameter, 3 m deep perforated chamber | Subsurface is rough alluvia, water table is less than 1 m below drywell bottom | Residential | Metals, other organics, (total petroleum hydrocarbons) TPH | 9 rain events sampled with continuous samples taken inside drywell and composite sediment sample taken at end of monitoring period, grab groundwater samples taken |
| United Kingdom 2007 Chen et al. (2007) | 10 | Precast, perforated concrete rings of various diameters and depths | Varied | Commercial, industrial, and residential | Metals, VOCs, TPH | Grab samples taken of sediment and water in or below drywell |
| Pima County, Arizona, USA 1986–1989 Wilson et al. (1990) | 4 | 7.6–10.6 m deep gravel filled drywells connected to concrete sedimentation chamber | Subsurface is predominantly coarse sand and gravel with low percentages of silt and clay. Water table depth range between 130 and 76.2 m bls | Commercial, industrial, and residential | Metals, VOCs, PAHs, pesticides, organics | One stormwater grab samples taken at entrance to sed. chamber, one sediment sample taken from bottom of sedimentation chambers. Vadose zone water samples taken from boreholes, and groundwater samples taken from three sites |
| Tucson, Arizona, USA 1987 Olson (1987) | 149 | 1.2 m diameter, 4.6–7.6 m deep concrete settling chamber ending in filter fabric above 1.2 m diameter borehole ending with injection screen in granular material | Subsurface is caliche layers with sand and clay interbeddings | Commercial, industrial, and residential areas | Sediment samples: metals, VOCs, semivolatile organic carbons (SVOCs), pesticides, other organics. Water samples: VOCs, SVOCs, PAHs, pesticides, other organics, TPH | Sediment samples taken from top 10 cm of sediment in settling chamber; one stormwater sample taken for each site; vadose zone samples collected at industrial sites by taking samples every 1.5 m bls for length of drywell |
| Los Angeles, California, USA 2001–2007 Dallman and Spongberg (2012) | 2 | Drywells designed to accommodate 19 mm/h and 51 mm/h rainfall events, specifics of design not indicated | | Commercial and residential | Metals, SVOCs, VOCs, other organics, minerals, E. coli, pathogens, TPH | Stormwater, vadose zone, and groundwater samples taken for 12 storm events from 6 years at the commercial site, and for 6 storm events from 3 years at the residential site and groundwater samples taken in fall and spring |
| Modesto, California, USA 2003–2005 Jurgens et al. (2008) | More than 11,000 drywells in Modesto | 1 m diameter drilled holes between 15 and 25 m deep, backfilled with rock aggregate with perforated casing 15 cm diameter and 6 m long | Unconfined aquifer composed of sand and gravel layers with discontinuous clay layers. perched, shallow, intermediate, and deep aquifers at 8.5 m, 29 m, 51 m, and 100 m bls | Agricultural and urban | Metals, VOCs, other organics, pesticides, organics, nutrients, minerals | 23 monitoring wells ranging from 4.9 to more than 91.4 m in depth. Urban stormwater samples taken near two drywells once during winter, and sprinkler irrigation water samples taken once during summer. All monitoring wells sampled 1–5 times over the course of one year, and depth-dependent samples taken from public supply well |
| Missoula, Montana, USA 1987 Wogsland (1988) | 2700 drywells in Missoula | 1.2 m long small diameter pipes placed in the bottom of stormwater catch basins | Unconfined aquifer extending to 30.5–61 m bls composed of layers of boulders, cobbles, gravel, sand, silt, and clay | Commercial and residential | VOCs, PAHs, nutrients, TPH, salts, and other USEPA organic priority pollutants | Composite samples collected at two sites, grab samples collected from two stormwater outfalls and four parking lots, vadose zone water samples collected from lysimeters installed at two drywell sites, groundwater samples taken from two sites from downgradient monitoring wells |
| Millburn, New Jersey, USA 2011 Pitt et al. (2012) | Approximately 1500 drywells in Millburn | Perforated, open-bottomed concrete casings with 1.8 m diameters and depths that end in 0.6 m layer of crushed stone | Low permeability surface soils above high permeability subsurface layers | Residential | Metals, pesticides, nutrients, organics, bacteria | Grab samples collected from drywell underdrains during and immediately after storm events for ten events per year. samples also taken from roof runoff cistern |
| Bend, Oregon, USA 2006–2011 Brody-Heine et al. (2011) | More than 4580 drywells in Bend | Stormwater inlet leading to 1.2 m diameter dug well with perforated casing ranging from 1.2 to 5.8 m deep with closed bottom | Subsurface composed of thick basaltic lava flows with approximately 3 m thick sedimentary interbeds | Commercial, industrial, residential, and urban | Metals, SVOCs, VOCs, PAHs, pesticides, other organics | Stormwater samples taken at 8 UICs and 2 stormwater outfalls between 2006 and 2011, 10 samples taken at each of the 8 UICs during spring 2011 |
| Portland, Oregon, USA 2006–2014 | 19,000 UICs in the City of | Solid concrete sedimentation manhole 3 m deep and 1.2 m in diameter, | Subsurface is coarse and fine grained sedimentary deposits, | Commercial, industrial, | Metals, SVOCs, VOCs, PAHs, pesticides, other | Monitoring performed for 10 years: during year 1, thirty wells were monitored, 15 of |

**Table 2** (continued)

| Study location, years of study, and report associated with study | Number of drywells used | Drywell design | Subsurface conditions | Land use of study sites | Contaminants/parameters in analytical suite | Monitoring implementation |
|---|---|---|---|---|---|---|
| City of Portland (2008, 2014) | Portland at the time of the study | connected to 1.2 m diameter dug drywell ranging in depth from 0.2 to 12.2 m with perforated casing | cemented gravel mixed with sand and silt | residential, and urban | organics | which were rotating locations, all sampled for 5 storm events per year. 41 UICs sampled during year 2 in the same manner with the addition of 10 UICs near drinking water sources. Years 3–10 monitoring was done for UICs on a case-by-case basis. Stormwater samples were taken at entrance to sed. chamber |
| Pierce County, Washington, USA 1990–1995 Adolfson and Clark (1991) and Adolfson (1995) | 1 | 1.2 m diameter, 1.7 m deep sedimentation manhole connected to 1.2 m diameter drywell encased in filter fabric and perforated concrete casing to 2.7 m depth filled with drain rock with open bottom | Low permeability till in the north of the county, highly permeable coarse sand and gravel in the south. Deeper subsurface is alternating glacial and non-glacial strata | Commercial | Sediment: metals, TPH, stormwater; in addition to analytes sampled for in sediment, nutrients, fecal coliform, TSS | Sediment samples taken once,16 stormwater sampling events during which stormwater composite grab sample was taken, and further samples taken through tube that extended down directly below drywell |
| Park Ridge, Wisconsin, USA 1996–1998 Lindemann (1999) | 14 | 1.5–2.4 m deep perforated pipe with gravel inlet and open bottom draining into gravel | Shallow subsurface consists of gravel and sand deposits overlying bedrock | Residential | Water samples analyzed for metals, VOCs, PAHs, nutrients, minerals | Three grab stormwater samples taken underneath the drywell inlet grates from five drywells and three roadside ditches. Sediment cores taken from beneath drywell. Groundwater samples taken from downgradient and upgradient monitoring wells ten times over two years |

include total arsenic, copper, lead, zinc, total petroleum hydrocarbons, and total solids.

In Portland the permit issued by the Department of Environmental Quality (DEQ) under which the UIC program operated required that stormwater monitoring occur for 10 years after the start of the study. Thirty of the 9000 wells involved in the study were randomly chosen through statistical analysis to be monitored during Year 1 (City of Portland, 2008). Of these, 15 were rotating locations sampled for 5 storm events per year, and the remainder were fixed locations also sampled for 5 storm events during the wet season of the water year (City of Portland, 2008). Water samples were taken from the point where stormwater enters the top of the sedimentation manhole, and were analyzed for common pollutants, such as metals, VOCs, SVOCs, PAHs, pesticides, and a Priority Pollutant Screen (PPS), composed of a list of common contaminants that includes benzene, PCP, chromium, toluene, xylenes, phthalates, arsenic, copper, and nitrogen (City of Portland, 2008). Details concerning the temporal nature of the storm event samplings were not included in the study reports. Forty-one UICs were sampled during Year 2. Once again, 15 rotating locations and 15 fixed locations were monitored as well as one UIC that had not met Permit compliance in Year 1, and 10 UICs located near drinking water wells (City of Portland, 2008). Five sampling events were again completed. Year 2 monitoring well samples were tested for common pollutants. Groundwater monitoring wells were installed and results of monitoring had been analyzed before the start of the study to establish background conditions (City of Portland, 2008). Sediment analyses had also taken place before the start of the study. Further monitoring was performed on a case-by-case basis when deemed necessary. Any drywells identified as non-compliant with the DEQ permit due to the quality of influent stormwater underwent Groundwater Protectiveness Demonstrations (GWPD), which includes the analysis of results obtained with a spreadsheet model, further discussed in Section 2.4.6 (City of Portland, 2008). After the completion of GWPDs, the drywells were marked as compliant, backfilled to increase separation distance, outfitted with a pretreatment feature, or decommissioned depending on results (City of Portland, 2008).

*2.4.2.4. Study results for stormwater quality.* The contaminants most commonly found in stormwater runoff in the referenced drywell studies fall under the categories of heavy metals, nutrients, organics, pathogens, pesticides, and salts. Metals detected include arsenic, cadmium, chromium, copper, iron, lead, magnesium, mercury, nickel, thallium, uranium, and zinc (Adolfson, 1995; Barraud et al., 1999; Brody-Heine et al., 2011; Chen et al., 2007; City of Portland, 2008; Dallman and Spongberg, 2012; Izuka, 2011; Jurgens et al., 2008; Olson, 1987; Pitt et al., 2012; Wogsland, 1988). The metals most commonly detected were copper, lead, and zinc. Out of all of the contaminant categories, metals were most often detected above advisory levels. Six different metals were detected above regulatory levels in stormwater at six sites as shown in Table 3: lead in Portland and Bend, Oregon; lead, copper, and zinc in Pierce County, Washington; lead and cadmium in Tucson, Arizona, arsenic in Los Angeles, California; and uranium and arsenic in Modesto, California (both detected in groundwater but linked to stormwater infiltration) (Adolfson, 1995; Brody-Heine et al., 2011; City of Portland, 2008; Dallman and Spongberg, 2012; Jurgens et al., 2008; Olson, 1987).

Pesticides including 2,4-D, methoxyvhlor, atrazine, and simazine were detected during two studies as seen in the table, however they were not detected at levels above health advisory limits (City of Portland, 2008; Jurgens et al., 2008). Nutrients were detected at five sites, and nitrogen was found above its MCL at the drywell site in Park Ridge, Wisconsin (Lindemann, 1999). Organics detected include VOCs, semivolatile organic compounds (SVOCs),

**Table 3**

Contaminants detected in stormwater samples, vadose zone water samples, or groundwater monitoring well samples taken during drywell studies.

| Study | Stormwater contaminants detected | | | | | | |
|---|---|---|---|---|---|---|---|
| | Metals | Semi-volatile and volatile organic carbons | Polycyclic aromatic hydrocarbons | Pesticides | Other organics | Nutrients/minerals | Biological contaminants |
| Barraud et al. (1999) | Cadmium, lead, zinc | – | – | – | Mineral oil | Nitrate | – |
| Chen et al. (2007) | Lead, zinc | – | Unspecified | – | TPH | – | – |
| Wilson et al. (1990) | Silver, arsenic, cadmium, chromium, copper, mercury, nickel, lead, zinc | Chloro-methane[a], 1,3-dichlorobenzene[a], ethylbenzene, methylene chloride, toluene[a], xylenes | Acenaphthene[a], acenaphthylene, anthracene, benzo(a)anthracene, benzo(a)pyrene[b], benzo(b)fluor-anthene, bis(s-ethylhexyl)phthalate, chrysene, di-n-butyl- phthalate[a], di-n-octyl-phthalate[a], dibenzo(a,h)- anthracene[a], fluoranthene, fluorene, indeno(1,2,3-c,d) pyrene, naphthalene, phenanthrene, pyrene[a] | Chlordane, chlorpyrifos, 2,4-DB, 4–4′-DDT, dioxathion | – | – | – |
| Olson (1987) | Arsenic, cadmium[b], chromium, lead[b], mercury, nickel, thallium | – | Anthracene, benzo(A)pyrene[b], pyrene | – | – | – | – |
| Dallman and Spongberg (2012) | Various heavy metals, arsenic[b], copper, lead, zinc | – | – | – | Petroleum hydrocarbons | Chlorine[a], nitrogen, salts perchlorate | Bacteria |
| Jurgens et al. (2008) | Arsenic[b], uranium[b] | Benzene, chloroform, ethylbenzene, tetrachloro-ethylene[a], toluene | – | 10 total including atrazine and simazine | – | Nitrogen[b] | – |
| Wogsland (1988) | Cadmium, chromium, copper, iron, lead, manganese, nickel, zinc | – | – | – | Oil, grease | Calcium, chlorine, manganese, nitrogen[b], potassium, sodium[b] | – |
| Pitt et al. (2012) | Lead | – | – | Chlordane | Endosulfan-1, heptachlor epoxide | – | Bacteria |
| Brody-Heine et al. (2011) | Antimony, arsenic, cadmium, copper, lead[b], zinc, barium, cyanide | Toluene | – | – | – | Nitrogen as nitrate | – |
| City of Portland (2008) | Copper, lead[b] | Di(2-ethylhex-yl)phthalate, pentachloro-phenol[b], toluene | Benzo(A)pyrene, naphthalene | 2,4-D, methoxychlor | – | – | – |
| Adolfson (1995) | Arsenic, copper[b], lead[b], zinc[b] | – | – | – | TPH | Nitrogen | Fecal coliform |
| Lindemann (1999) | – | Benzene[b], carbon tetrachloride[b], naphthalene, trichlorofluor-omethane, toluene, xylene | Benzo(a)anthra-cene, benzo(A)-pyrenee[b], benzo(b)fluor-anthene[b], chrysene[b] | – | – | Chlorine, nitrogen[b], sodium[b] | – |

[a] Detected in trace amounts.
[b] Detected above regulatory levels.

PAHs, and benzene byproducts; organic contaminants were detected at seven sites. Pentachlorophenol (PCP) exceeded regulatory levels in Portland, Oregon, and several VOCs and SVOCs were detected above regulatory levels in Park Ridge, Wisconsin (City of Portland, 2008; Lindemann, 1999). Benzo(a)pyrene exceeded its health advisory levels in Tucson, Arizona (Olson, 1987).

Salts were found at three sites, and exceeded MCLs or health advisory levels at two: Park Ridge, and Missoula, Montana (Dallman and Spongberg, 2012; Lindemann, 1999; Wogsland, 1988). Pathogens (namely bacteria) were found at three sites (Adolfson, 1995; Dallman and Spongberg, 2012; Pitt et al., 2012). Refer to Table 3 for further details.

The potential contaminants a drywell may introduce into groundwater can in some cases be predicted by the land use type surrounding the facility as well as by certain seasonal trends and the composition of the local subsurface, as discussed in Section 2.1. For instance, in regards to the seasonality of certain pollutants, between the months of February and March chloride and sodium levels were commonly found to be elevated in groundwater in the vicinity of drywells that received runoff from major roads that experienced snow in the winter (Lindemann, 1999; Wogsland, 1988). As the snow thawed and snowmelt runoff flowed across the road and surrounding area into the drywell, it picked up dissolved deicing salts that had been deposited on the road during winter months. After a few months the chloride plume dissipated and levels returned to background (Lindemann, 1999).

A study conducted in Pierce County, Washington, provides an example of land use being useful for predicting potential contaminants (Adolfson and Clark, 1991; Adolfson, 1995). These studies compared three different forms of stormwater management at different sites, and the drywell used in the study was located in a largely commercial area, in an asphalt parking lot immediately adjacent to an auto repair shop (Adolfson, 1995). The well was fed by runoff from a 0.4047 hectare (ha) commercial area (Adolfson and Clark, 1991; Adolfson, 1995). The drywell's proximity to the auto repair shop made it susceptible to contamination from chemicals and compounds commonly used for automobile maintenance. Early on in the monitoring period, 265 L of used crankcase oil were allegedly dumped into the drywell (Adolfson, 1995). This resulted in very high levels of TPH in the stormwater samples taken from below the drywell (Adolfson, 1995). The oil was removed after a few months. TPH was not detected in subsequent samples taken from below the drywell. The drywell site also had the highest levels of lead and zinc in runoff out of the three sites used in the study (Adolfson, 1995). Used batteries and crushed oil containers were observed on the ground at the site; this can also be attributed to the proximity of the automobile repair shop and is a likely source of metals (Adolfson, 1995).

An extensive study conducted by the USGS in Modesto, California, detected ten different pesticides, primarily atrazine degradation products and simazine, in shallow groundwater, all in concentrations below regulatory levels (Jurgens et al., 2008). Drywells of a simple design lacking pre-treatment have been in use in Modesto since the 1950s; the city currently has over 11,000 rock aggregate filled drywells between 15 and 25 m deep (Jurgens et al., 2008). Nitrate, analyzed as nitrite plus nitrate, was detected in all 23 groundwater samples with a median concentration of 4.0 mg/L. Nitrate concentrations exceeded regulatory levels (10 mg/L) in six out of 23 monitoring wells: in three shallow groundwater monitoring wells in concentrations ranging from 1.0 to 12.9 mg/L, in a water table monitoring well located near an unsewered city subdivision at a concentration of 11.1 mg/L, and in two water table monitoring wells located below farmland in concentrations of 12.6 mg/L and 17.4 mg/L (Jurgens et al., 2008). These detections were linked to agricultural land use in the area surrounding the city (Jurgens et al., 2008). Inorganic fertilizer was found to be one of the sources.

Uranium was also detected in groundwater, and in this case was most likely the result of salts in agricultural runoff infiltration increasing the alkalinity and oxicity of shallow groundwater (above 50 m below land surface). Desorption of naturally occurring uranium bound to sediments is favored in highly alkaline and oxic groundwater (Jurgens et al., 2008). No contaminants were found in the deepest aquifer (greater than 199 m bls). The authors of the study concluded that agricultural chemicals not commonly used in urban areas were the primary sources of contamination in deeper aquifers (Jurgens et al., 2008).

Although contaminants were detected in stormwater samples in all of the examined drywell studies, the conclusion of the majority of studies is that when conducted properly and allowing for a sufficient separation distance and subsurface pollutant attenuation, drywell infiltration of stormwater does not pose a threat to groundwater and drinking water sources (Barraud et al., 1999; Brody-Heine et al., 2011; Dallman and Spongberg, 2012; Jurgens et al., 2008; Olson, 1987). Although some contaminants were detected in stormwater samples above regulatory levels, these contaminants were rarely detected in groundwater at similar levels (Jurgens et al., 2008; Olson, 1987; Wilson et al., 1990). Even in Modesto, California, where drywells have been used for more than half a century, no contaminants detected in downgradient groundwater monitoring wells exceeded regulatory values (Jurgens et al., 2008). Studies that showed insufficient pollutant attenuation in the drywell or underlying vadose zone sediment concluded that incorporating pretreatment into the drywell design would reduce influent contaminant concentrations (Adolfson, 1995; Wogsland, 1988).

*2.4.2.5. Drywell infiltration modeling studies.* Contaminant attenuation in the vadose zone was indicated in many studies, and contaminant fate and transport modeling performed as part of some studies further supports this conclusion (Bandeen, 1987; Brody-Heine et al., 2011; City of Portland, 2008; Izuka, 2011; Olson, 1987; Wilson et al., 1990; Wogsland, 1988). Four out of the 13 drywell studies performed predictive contaminant fate and transport modeling as part of their determination of groundwater protectiveness (Bandeen, 1984, 1987; Brody-Heine et al., 2011; City of Portland, 2008; Izuka, 2011).

The goals of the two modeling efforts performed for studies in Arizona were to estimate the subsurface distributions of stormwater infiltrating through a drywell under various subsurface conditions (Bandeen, 1984, 1987). A model study relating to the composition of the subsurface beneath a drywell was conducted using UNSAT 2 (Bandeen, 1987). Three simulations were created for this study, each run with two five year, one hour storm events separated by 24 h. Case 1 simulated infiltration through a homogeneous layer of gravelly sand and sandy gravel to the water table at a 30.5 m depth. Case 2 simulated infiltration through the same gravel/sand layer for 9.1 m and then through a uniform clay/silt loam type soil to the water table at 30.5 m depth. Case 3 ran the same simulation as Case 2, however with a sandy loam type soil instead of the clay/silt loam layer. All three soil types are common in the Tucson Basin. Attenuation of pollutants was found to be related to the size of soil particles and the degree of exposure the drainage plume had to soil. Another model study using UNSAT 2 produced results delineating the geometry of the infiltrated stormwater plume, the infiltration rate, and the hydraulic head distribution at various time steps (Bandeen, 1984). These results were used to make recommendations about the subsurface conditions suitable for drywell installation sites (Bandeen, 1984, 1987).

Numerical contaminant fate and transport modeling was done for the drywell study conducted for the Island of Hawaii, the goal of which was to provide results for the effects of drywell infiltration relevant to the entire island. Unsaturated zone models were

coupled with groundwater models in order to estimate stormwater contaminant attenuation in the vadose zone as well as groundwater contaminant plume migration (Izuka, 2011). Three-dimensional domains were created to represent various subsurface conditions on the island; the domains were 1.0 km (km) in width, 7.6 km in length, and ranged in depth from 43 to 339 m depending on the thickness of the unsaturated zone and aquifer being represented (Izuka, 2011). A single drywell was modeled 1.0 km from the downgradient boundary, and a pulse of 141.6 L/s was applied to the unsaturated zone models for one hour to simulate drywell stormwater infiltration (Izuka, 2011). MODFLOW-2005 and UZF1 add on package were used to simulate unsaturated flow of stormwater contaminants through the vadose zone to groundwater (Harbaugh, 2005; Izuka, 2011; Niswonger et al., 2006). SEAWAT Version 4 (MODFLOW-2000 coupled with MT3DMS) was used to simulate saturated zone contaminant plume flow (Langevin et al., 2008; Zheng and Wang, 1999). The unsaturated zone models produced time-variable infiltration rates that were input into the saturated zone models, which then produced three-dimensional renderings of the migration of groundwater contaminant plumes over the modeled time period (Izuka, 2011). The results were used to make general recommendations about drywell usage on the island (Izuka, 2011).

The modeling done for the Portland, Oregon, study was implemented as a solute transport spreadsheet model designated the Groundwater Protection Demonstration (GWPD) tool, and was developed to evaluate the results of stormwater monitoring, determine necessary vertical separation distances between the bottom of a UIC and the water table, and provide generic conditions for which to describe groundwater protectiveness (City of Portland, 2008). The GWPD Tool predicts how much a pollutant's concentration in infiltrating stormwater will decrease as stormwater flows out of the UIC and through the unsaturated zone before it reaches groundwater, and is based on the one-dimensional constant source Advection Dispersion Equation (ADE) (City of Portland, 2008). Stormwater contaminant data obtained from analyzing stormwater samples collected during monitoring events was used to choose the contaminants used in fate and transport modeling and the input concentrations of those contaminants (City of Portland, 2008). Toluene was selected to represent VOCs; PCP and di(2-ethylhexyl)phthalate (DEHP) to represent SVOCs; benzo(a) pyrene and naphthalene for PAHs; 2,4-D and Methoxychlor for pesticides and herbicides; and copper and lead to represent metals (City of Portland, 2008). Scenarios depicting average and worst-case conditions were created and models were run for both five and seven foot separation distances through each of the chosen representative soil facies: gravel with silt and sand, coarse sand and silt, and cemented gravel (City of Portland, 2008). The model's results show that even with only a 1.5 m separation distance and traveling through gravel with silt and sand, the most permeable geologic material, all of the selected non-metal pollutants are reduced by more than 99% before they reach the water table (City of Portland, 2008). It was estimated that it would take copper and lead 1600 and 2150 years respectively to reach the water table (City of Portland, 2008).

The drywell study performed for the City of Bend, Oregon, also used a one-dimensional fate and transport model to determine groundwater protectiveness (Brody-Heine et al., 2011). The representative contaminants chosen were copper, lead, benzo(a)pyrene, naphthalene, PCP, DEHP, 2,4-D, and toluene (Brody-Heine et al., 2011). Flow from UICs into subsurface basaltic fracture network was simulated; representative pollutants were chosen, and site-specific geologic and hydrogeologic conditions were used. The Fate and Transport Tool (FTT) used a one-dimensional ADE to estimate pollutant attenuation during vadose zone transport (Brody-Heine et al., 2011). The FTT simulation results for the average scenario

indicate that all of the evaluated pollutants were attenuated to below detection limits with 1.5 m of transport (Brody-Heine et al., 2011). Copper, lead, benzo(a)pyrene, PCP, and DEHP were also shown to attenuate to below detection limits with 1.5 m of transport under worst case scenario conditions; 2,4-D and toluene were attenuated to below detection limits after 11.3 and 8.8 m respectively, and were not predicted to reach the depth of the City's seasonal high water table (Brody-Heine et al., 2011).

## 3. Discussion

### 3.1. Evaluation of performed drywell studies

#### 3.1.1. Rigor of studies' review

Five of the thirteen discussed drywell studies were published in peer-reviewed journals. The reports created for the field studies were predominantly produced by consulting firms to address local regulatory concerns, and were not meant to provide insight into the functionality of drywells on a broader scale. The peer-reviewed studies tended to focus on more regional scales, or longer time periods. The study performed for drywell use in Hawaii, USA, used generalized regional geology, and a non-specific contaminant to provide results that could be applied to the entire state (Izuka, 2011). The study performed at unspecified locations in the UK focused on the types of contaminants entering the drywells in different land use areas, and not the fate of the contaminants in the local subsurface (Chen et al., 2007). The study performed to examine long-term use of drywells in Modesto, California, observed trends in groundwater contamination linked to drywell use in agricultural areas after over 50 years of use (Jurgens et al., 2008). The comparative study performed in Valence, France, used the results of groundwater and stormwater quality sampling for a newly installed drywell and an older soakaway to predict the effects of long-term stormwater infiltration (Barraud et al., 1999).

Soakaway use in the UK and other parts of Europe is regulated, and guidelines exist for design parameters and infiltration testing (Chen et al., 2007). There are no unifying drywell regulations in the United States, however, and permitting and usage vary from state to state (EPA, 1999a,b). The reports for studies performed in the United States mostly answer specific questions to exhibit the protectiveness of drywells for groundwater in specific, localized areas. This could be the reason that the majority of reports for drywell studies performed outside of the US are peer-reviewed, and the majority of reports produced for studies done in the US are not. Existing guidelines and regulations can provide a foundation for the performance of more comprehensive and focused investigations.

#### 3.1.2. Monitoring methods

Monitoring complexity and rigor varied between the drywell studies. The majority of studies used upgradient and downgradient groundwater monitoring wells, and took stormwater runoff samples at the point where water entered the drywell (Dallman and Spongberg, 2012; Jurgens et al., 2008; Lindemann, 1999; Olson, 1987; Wilson et al., 1990; Wogsland, 1988). Some studies did not perform any sort of groundwater monitoring, focusing instead solely on the contaminants entering the wells. This may be an appropriate method if groundwater protectiveness is to be determined by the quality of water entering the wells; however, these studies do not capture any observable data regarding subsurface contaminant attenuation and fate, which can play a major role in groundwater protectivenss.

Studies that included regular monitoring, taking stormwater and vadose zone water samples for multiple storm events every year, seemed to obtain more representative results that were

better able to capture seasonal trends. One of the challenges for studies implemented on a large scale, such as the City of Portland study, was how to accurately monitor hundreds, if not thousands, of drywells. It is possible that a rotational sampling method would leave some contamination events undetected.

None of the studies were able to test for all possible stormwater contaminants, and so the success of their stormwater and groundwater sample testing depended on the appropriateness of the contaminants chosen for analysis. Choosing to test for select high priority contaminants along with a range of common and expected contaminants appears to be the standard method, however this could lead to certain lower-profile contaminants being overlooked, and causing groundwater contamination. Ideally, a wide range of contaminants would be analyzed in the initial sampling events, after which the scope of the analysis could be narrowed based on initial results.

The method of sampling conducted during stormwater events may also play a role in the success of a study's monitoring; grab sampling may miss first flush stormwater contaminants (due to the small number of samples taken and the limited volume of stormwater runoff captured by grab samples) that automated systems would catch (Maestre and Pitt, 2006). Manual/grab sampling was used most often in small studies with less than five drywells, while automated sampling was used for large studies with the funds to equip wells with automated samplers (Adolfson, 1995; City of Portland, 2008; Wilson et al., 1990). Ideally, automated samplers would be used, and water samples would be taken from influent stormwater, water infiltrating out of the drywell, and groundwater in order to determine analyte concentrations in each of these zones, and allow conclusions to be reached concerning the occurrence of contaminant attenuation in or below the drywell.

### 3.1.3. Numerical methods

There have been few numerical models created to simulate drywell infiltration, and those that were discussed in this review accomplish specific goals, but fall short of comprehensive. The modeling performed to propose potential drywell effects on the Island of Hawaii did not use sampled stormwater contaminant concentration data in the fate and transport modeling, nor did it model any real contaminants, only a generic solute (Izuka, 2011). The coupling of different vadose zone domains with various aquifer conditions and groundwater flow regimes does provide varied results that may accurately describe contaminant flow in Hawaii's groundwater, however, without observed stormwater contaminant concentration inputs, the models may not be used to predict actual groundwater quality effects.

The modeling performed for the Tucson, AZ, sites also did not use site specific stormwater contaminant concentration inputs, however effort was made to simulate the stratigraphy underlying drywell sites in the area (Bandeen, 1984, 1987). The more detailed and specific domain used in the Tucson models may provide insight into flow patterns and plume dimensions below infiltrating drywells, however the shortcoming of these models may be the time period modeled: each model only simulated two hour-long storm events in a 26 h period (Bandeen, 1987). This allows for short-term, storm-event-specific flow and transport to be predicted, but says little about the potential effects of drywell aided stormwater infiltration ten years or even one year in the future.

The modeling performed for the Oregon drywell studies was used to evaluate the protectiveness of the mandated 1.5 m separation distance between the bottom of the drywell and the seasonal high water table (Brody-Heine et al., 2011; City of Portland, 2008). Many assumptions were made for the modeling, but most were justified by the fact that all assumptions were skewed toward representing the worst-case scenario results of stormwater infiltration

on groundwater quality (Brody-Heine et al., 2011; City of Portland, 2008). The spreadsheet models described in section 2.4.6 only allow for contaminant transport through a single material (Brody-Heine et al., 2011; City of Portland, 2008). This may be representative of some physical drywell sites, however in cities that employ drywells in the tens of thousands as Portland and Tucson do, only a fraction of the drywell sites may be accurately represented (Arizona Department of Water Quality, 2015; City of Portland, 2008).

This raises the question of the value of a highly detailed, site-specific model. A model simulation that accurately describes the underlying stratigraphy, pollutant concentrations and attenuation, and water flux of a single drywell site may not be applicable to any other drywell site. Drywell success, as has been discussed previously in this paper, can only be assessed on a case-by-case basis. However, communities that employ the use of drywells on a large scale may not be able to realistically perform modeling efforts for each and every drywell. Therefore, it seems that a successful model of drywell infiltration would err on the conservative side, focus on the specific stormwater contaminants found in the modeled area, and attempt to use long-term simulation results to identify areas that are unsuitable for drywell installation due to the probability of either insufficient subsurface pollutant attenuation, or elevated contaminant concentrations in influent stormwater.

### 3.2. Factors that affect the groundwater pollution potential of drywells

#### 3.2.1. Land use

Land use has been shown by the literature to correlate with contaminants present in stormwater. Stormwater quality studies show that certain categories of contaminants are likely to be found in agricultural, commercial, industrial, and residential areas, and the results of the drywell study samplings also show a correlation between contaminants found in stormwater runoff, vadose zone water samples, sediment samples taken from inside drywells, and groundwater samples, and drywell site land use. The data collected from stormwater surveys, summarized in Table 1, shows that stormwater runoff from agricultural areas is likely to contain VOCs, pesticides, and nutrients. The results of the drywell field study samplings showed that VOCs, pesticides, and nutrients were found in agricultural runoff, and that metals were present in groundwater due to mobilization from subsurface sediment by salts in infiltrating runoff (Jurgens et al., 2008). Land use proved to be an accurate predictor of infiltration contaminants for agricultural areas.

Stormwater survey data indicated that runoff from commercial areas is likely to contain metals, organics, nutrients, and biological contaminants; runoff from industrial areas is likely to contain metals, PAHs, organics, nutrients, and biological contaminants; and runoff from residential areas is likely to contain metals, pesticides, organics, nutrients, and biological contaminants. Land use predictions were also somewhat accurate for urban land uses: the contaminant types listed in Table 1 were all found at least one study site for each of the three urban land uses (excluding biological contaminants from industrial sites). However, VOCs, PAHs, pesticides, organics, and nutrients were also found at all urban land use sites, which was not predicted. Some contaminants were linked to specific functions of the land surrounding the drywell site, for example, metals at the drywell site located next to an auto body repair shop, and chloride at the drywell sites located next to roads that received de-icing salts in the winter (Adolfson, 1995; Lindemann, 1999). The more specific information is known about the land surrounding a drywell site, the more possible it is to make an accurate prediction of the presence of specific stormwater contaminants and thus the potential effect on the local groundwater system.

### 3.2.2. Drywell design and pretreatment

Only five of the field study reports describe some form of pretreatment in their drywell design. All of the pretreatments described include a sedimentation chamber that holds influent stormwater before it enters the drywell, and one described pretreatment includes the use of filter fabric inside of the drywell (Adolfson, 1995). Few of the studies placed emphasis on the importance of pretreatment incorporation into drywell design; however, those that did highly recommended its use not only for decreasing contaminant concentrations in influent stormwater before water enters a drywell, but also to reduce the amount of fine sediment entering the drywell and potentially leading to clogging. Sedimentation chambers and vegetative pretreatment systems were shown to reduce the clogging rate of stormwater infiltration systems, thus maintaining their functionality for longer periods and allowing them to continually provide beneficial stormwater runoff management and aquifer recharge. It should be noted, however, that the simply designed rock wells employed in Modesto, California, had functioned for more than fifty years before an analysis of their effects was performed, and although they did not employ any form of pretreatment, they did not appear to contribute to groundwater contamination.

### 3.2.3. Hydrogeologic setting

A major part of the success of a drywell is the local subsurface composition of its installation site. Drywells by design are meant to transfer water from poorly drained or low permeability areas to subsurface areas where infiltration and storage of water is possible. However, once influent stormwater has passed the low-permeability surface layer and entered the vadose zone, its transport and fate are dependent on the subsurface hydrogeology. Many of the drywell study reports did not provide detail regarding the hydrogeology present at installation sites. Even when hydrogeology was described, it was not often considered in terms of possible contaminant transport and attenuation.

The key characteristics of a drywell site's underlying hydrogeology include the hydraulic conductivity of its material, the layering structure and composition of layers, and factors that indicate attenuation potential (such as the presence of clays and organic matter). If the effective hydraulic conductivity is too high, contaminants will be transported quickly without much attenuation; if the effective hydraulic conductivity is too low, stormwater infiltration will not occur quickly and the drywell will not serve either a stormwater runoff management of aquifer recharge function. What values quantify an effective hydraulic conductivity as either too high or too low cannot be broadly defined: these values must be determined on a site-by-site basis depending on the quality of stormwater runoff and the amount of runoff a drywell is receiving.

Some studies concluded that layered stratigraphies with some clay layers are suited for receiving drywell infiltration. A composition of sand, loam, and clay layers could provide both a sufficiently high effective hydraulic conductivity as well as the potential for contaminant attenuation. However, the possibility of preferential flow paths and lateral migration of contaminants has also been discussed, and the risk of preferential lateral flow pathways increases in the presence of sand lenses on top of clays. It is potentially possible to describe a subsurface composition ideal to drywell installation, but what is more important is an awareness of subsurface conditions at real sites being considered for drywell usage: if groundwater quality is going to be protected, then the subsurface hydrogeologic conditions must be known, and should be able to accept stormwater infiltration quickly enough as well as provide the necessary contaminant attenuation.

## 4. Conclusions and recommendations

### 4.1. Conclusions from the literature

This review has presented relevant findings from the existing literature pertaining to drywell design, stormwater quality, drywell maintenance, and 13 drywell field studies conducted in the past 40 years. In summary, studies suggest that drywells are an effective and practical means by which to manage stormwater runoff and recharge groundwater aquifers. Their success, however, is dependent on local subsurface conditions, drywell facility maintenance, and the quantity and quality of influent water. A review of stormwater quality literature has shown that temporal and spatial trends exist in the contaminants found in stormwater. Certain contaminants are associated with specific land uses, geographical areas, and times of the year. Drywell field studies support these associations; many of the stormwater contaminants predicted to be associated with a specific land use were found during stormwater sampling performed for drywell studies sited in areas of that land use. Various stormwater and groundwater monitoring methods were described, and it was concluded that an ideal monitoring system would provide multi-year sampling data gathered with an automated sampler for multiple storm events per year, and taken from surface water, vadose zone water, and groundwater to capture any short-term stormwater infiltration effects on groundwater.

Eight of the thirteen drywell studies detected contaminants above their regulatory levels in influent stormwater; however, only a few of the studies determined stormwater infiltration to be a possible source of groundwater contamination. This is because of predicted, observed, or modeled contaminant attenuation in the subsurface below drywells. Although heavy metals were frequently detected in monitoring samples from all land use areas and were the most common contaminant detected above advisory levels, metals have a low contamination potential due to their high adsorptivity in soils with some clay content. This was shown by results of the modeling performed for the drywell study in Portland, Oregon. The studies that performed groundwater monitoring consistently show decreased concentrations of contaminants between stormwater samples taken at the land surface and samples taken from groundwater, which could indicate contaminant attenuation in the vadose zone. However, it is possible that the monitoring period of the studies was not sufficient enough to capture longer-term stormwater infiltration effects on groundwater; few of the studies were performed over a multi-year time period, and few attempts were made to predict possible long-term effects of drywell use at study sites. The exception to this is the study performed in Modesto, which indicated that even after 50 years of use drywells had not detrimentally affected local groundwater resources. In order to fill the knowledge gap, some studies performed vadose zone or groundwater contaminant fate and transport modeling to predict when specific stormwater contaminants would enter groundwater, and how subsurface hydrogeology might affect contaminant transport.

### 4.2. Recommendations for drywell usage

Because the effect of drywell-aided stormwater infiltration is so dependent on local stormwater quality, land use, subsurface hydrogeology, and drywell design, it is difficult to make broadly applicable recommendations for drywell usage. It is recommended that stormwater quality monitoring be performed at any location being considered for drywell installation, ideally over a time period sufficient enough to capture seasonal trends, and that surface water quality monitoring be accompanied by groundwater quality

monitoring in order to observe any short-term detrimental effects on groundwater and potentially drinking water sources.

Studies have shown that the subsurface conditions suited for drywell infiltration are stratified, contain some layers with high clay content, and have a sufficient distance between the seasonal high water table and the lowest perforation of the drywell in order to allow for vadose zone pollutant attenuation. However, the danger does exist of extensive lateral migration of contaminants along preferential flow paths (typically movement of water through sand or gravel layers on top of clay layers), which further constrains the vadose zone compositions appropriate for drywell installation. Homogenous subsurface geologies with high hydraulic conductivities are not recommended because they may allow influent stormwater runoff to enter groundwater without sufficient pollutant attenuation or residence time. It is therefor critical that the subsurface composition and hydrology of an area be known before drywells are installed.

Drywells, like many forms of stormwater infiltration device, are prone to clogging after prolonged use, and so proper maintenance of a drywell system as well as periodic performance testing in order to ensure that the drywell's functionality is not compromised is advisable. One study showed that drywells could be used functionally for more than 70 consecutive years (Chen et al., 2007). The inclusion of pretreatment features in drywell design can reduce clogging due to siltation, and can also reduce contaminant levels in water infiltrating through drywells. Drywell installation for management of stormwater runoff and aquifer recharge is recommended if stormwater monitoring results indicate low levels of stormwater contaminants, groundwater monitoring results indicate little effect of stormwater infiltration on groundwater, and the subsurface hydrogeology of a site appears to satisfactorily attenuate contaminants.

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