Bioretention and Green Roof Systems in Semi-arid and Arid Climates: Evaluating and Optimizing the Retention and Pollutant Removal Utility

Charlotte Marie Devine
Charlottesville, VA

Bachelor of Arts, Middlebury College, 2017

A Thesis presented to the Graduate Faculty
of the University of Virginia in Candidacy for the Degree of
Master of Arts

Department of Environmental Sciences
University of Virginia
December 2018

Abstract

Implementation of low impact development (LID) techniques is becoming increasingly important as a means of reducing stormwater runoff volumes and treating deteriorated water quality resulting from climate change and urbanization. Although stormwater management is difficult under all climate conditions, in semi-arid and arid climates increased erosion, flooding, and pollutant buildup occur due to long dry periods broken by monsoonal events. Furthermore, long periods of drought reduce vegetation survival in these climates. Through an examination of existing literature on the performances of green roofs and bioretention systems in semi-arid and arid climates, this review has identified both the utility of these technologies and potential limitations. With some modifications involving strategic design decisions and vegetation selection, bioretention and green roof technologies can be utilized as effective LID technologies to reduce runoff volume and remove potentially hazardous contaminants from these environments. Ultimately, semi-arid and arid bioretention basins have utility in both retaining runoff and reducing pollutants in effluent. However, green roofs fail to consistently improve stormwater quality. Both technologies have limited utility individually in large or high-intensity storm events. Therefore, combined systems referred to as “treatment trains” should be utilized as a means to better mitigate increased runoff volumes while simultaneously treating stormwater for potential reuse in irrigation.

Table of Contents

List of Figures

List of Tables

List of Abbreviations

A_{bio}	Bioretention surface area
ADD	Antecedent dry days
ADWPs	Antecedent dry weather periods
AMC	Antecedent moisture content
AN	Ammioniacal nitrogen
BMP	Best Management Practices
CAM	Crassulacean acid metabolism
COD	Chemical oxygen demand
DD	Discharge duration
EC	Electrical conductivity
EMC	Event mean concentration
EPA	Environmental Protection Agency
ET	Evapotranspiration
g_L	Leaf stomatal conductance
h_{max}	Maximum water on surface
IWS	Internal water storage
K	Hydraulic conductivity
K_{sat}	Saturated hydraulic conductivity
LAI	Leaf area index
LID	Low Impact Development
MeHg	methylmercury
mPa	Megapascal
MUSIC	Model of Urban Stormwater Improvement Conceptualization
$NO_{2,3}$-N	Nitrate and nitrite
OD	Onset delay
Ortho-P	orthophosphates
PAHs	Polycyclic aromatic hydrocarbons
PCBs	Polychlorinated biphenols

PD	Peak delay
PE	Potential evaporation
PR	Peak reduction
P_{th}	Dryness threshold
Q_e	Peak flow rate
RCPs	Representative Concentration Pathways
RD	Rainfall duration
SAR	Sodium adsorption ratio
SPI	Site Pollution Index
SSPs	Shared Socioeconomic Pathways
SWC	Soil water content
SWMM	Stormwater Management Model (EPA)
TC	Total carbon
T_c	Time of concentration
TDS	Total dissolved solids
TN	Total nitrogen
TOC	Total organic carbon
TP	Total phosphorous
TSS	Total suspended solids
U	Consumptive use
V_e	Effluent volume
WHC	Water holding capacity
WUE	Water use efficiency

Acknowledgements

I would like to thank my advisor, Stephen Macko, and the members of my committee, Karen Firehock and Michael Pace, for their support and guidance as I completed this thesis.

I would also like to thank the rest of the faculty, staff, and graduate students in the Department of Environmental Sciences for creating a wonderful learning community this past year.

I would like to thank the gardeners at Monticello for their encouragement and insight as I researched and wrote this thesis.

I would like to thank my family for their unending support, patience, and positive energy as I completed this degree.

I would like to thank John Dukes for keeping me sane and always offering a sunny outlook on things.

1. Introduction

Stormwater management studies have only increased in importance over the years as urbanization increases, new contaminants are released into the biosphere, and both natural and human processes change the characteristics and fate of stormwater runoff. In particular, low impact development (LID) has emerged as a popular sustainable stormwater management technique. This approach emphasizes creating systems that imitate the natural hydrology of the drainage basin to return runoff characteristics to a predevelopment state (Ahiablame *et al.* 2012). In doing so, the effects of development, which can be extensive, are minimized. In particular, development frequently affects the volume of runoff, as well as degrading the overall quality of the runoff (Liu *et al.* 2014). These changes in runoff characteristics often include increased rates and quantities that are accompanied by decreased baseflow, groundwater recharge, and infiltration volumes (Figure 1) (Liu *et al.* 2014).

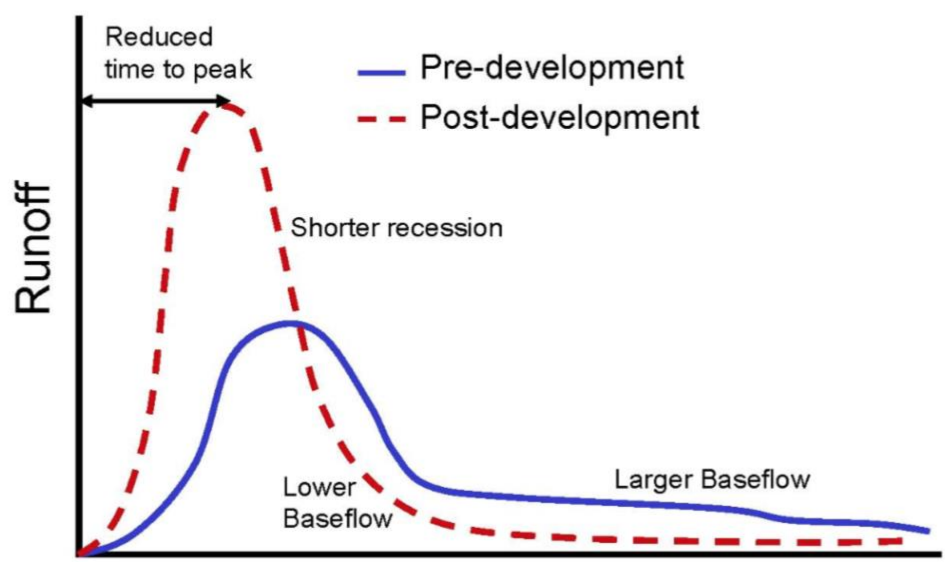


Figure 1: Effects of development on hydrological variables (Liu *et al.* 2014).

Studies show that development causes flooding in response to increased impermeable surface cover and, in some cases, may bring about extensive erosion owing to increased runoff intensity (Foley *et al.* 2005; Liu *et al.* 2014). Furthermore, the concentrations of pollutants in stormwater often increase alongside development due to the lack of permeable surface to absorb and filter contaminants and the increased production of pollutants alongside human activity (Gobel *et al.* 2007). For example, a recent study shows that runoff volume increased by 85% over a 30-year period in the Erbil City subbasin in Iraq due to urban growth and the accompanying increase in impervious land cover by more than 50 km^2 (Figure 2) (Hameed *et al.* 2017). Concentrations of harmful pollutants and excess nutrients, such as phosphorous and nitrogen, also accumulate at increased rates and contaminate waterbodies as urbanization occurs (Qin *et al.* 2014). Low impact development technologies involve an array of techniques such as infiltration, filtration, sedimentation, plant uptake, and storage, amongst others, to reduce stormwater quantity and improve quality (EPA 2007).

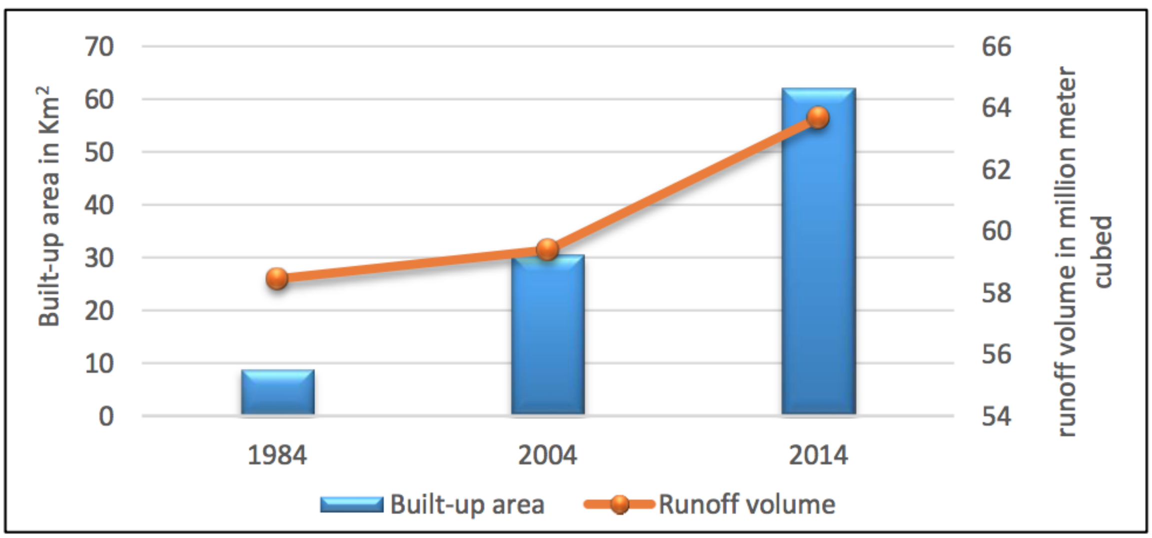


Figure 2: Increase in runoff volume in relation to impervious surface expansion in the Erbil City, Iraq subbasin (Hameed *et al.* 2017).

This paper seeks to address whether bioretention and green roof technologies, two relatively common LID technologies, have potential utility in semi-arid and arid climates. It does so by investigating the potential and shortcomings of each technology to manage stormwater in these climates, identifying the mechanisms by which performance might be optimized, and suggesting ways in which the technologies might be improved to better mitigate stormwater runoff quantity and improve quality going forth. Based on the mechanisms of each technology, I hypothesized that both technologies would function to improve stormwater quality and quantity to some extent, albeit at a decreased rate compared to systems in wet and humid climates. I also predicted that bioretention systems would achieve higher pollutant removal rates than green roofs, while green roofs would better reduce stormwater runoff volumes. The following sections of this introduction provide background material regarding basic bioretention and green roof functioning and components, as well as explaining the specific needs and challenges of semi-arid and arid climates that justify the focus of this report on these climate types. This literature review builds upon a 2016 guide published by the EPA that cites studies evaluating stormwater management of different LID technologies in semi-arid and arid climates and identifies areas for future research (EPA 2016). This paper expands upon the findings discussed in the EPA report and introduces new research that has emerged in the past few years (EPA 2016). It also concludes with a review of "treatment trains," which are combined systems including several LID technologies with different mechanisms to manage both stormwater quality and quantity to a greater extent than individual systems. My review analyzes treatment train performance in a variety of climates relative to individual LID systems as a means of identifying a potential solution and new area of study for future improvement of semi-arid and arid stormwater management practices.

1.1 Bioretention function and components

Bioretention and green roofs are two LID technologies that utilize vegetation to counter the negative effects of development on the natural watershed. Bioretention systems, also called rain gardens and bioswales, are vegetated basins typically filled with components including strategic porous soil mixes, underdrains, and location-specific plant species (Figure 3) (EPA 2007; SEMCOG 2008). Collectively, these various parts work to enhance infiltration and reduce runoff volume, particularly peak flow (EPA 2007). The vegetation included in bioretention systems prompts enhanced uptake of water and excess nutrients contained in runoff for plant growth, and it also structures the soil to reduce erosion (Jurries 2003; Manganka *et al.* 2015; Batalini de Macedo *et al.* 2017). Typically, the "ponding area," a shallow depression at the top of the system, permits temporary storage and evaporation of water from the system (EPA 1999). The relatively deep planting soil layer, often about 24 inches in depth, lies below the ponding area and allows for runoff to infiltrate the ground surface and limits pooling within the system (France 2002). High sand content is often present to increase soil porosity, thus increasing infiltration and limiting excessive pooling (Sileshi *et al.* 2015). However, a certain amount of clay in the soil mixture allows for pollutants to absorb to soil particles, which reduces the contaminant concentrations in runoff (Sileshi *et al.* 2015; EPA 1999). In some cases, inlet and outlet controls can be used to slow the velocities of runoff and carry overflow offsite in areas of intense or high-volume rains (Prince George's County 1999). Underdrains and filter layers serve similar purposes by increasing drainage within bioretention systems and returning excess water to storm drains (Schueler *et al.* 2007).

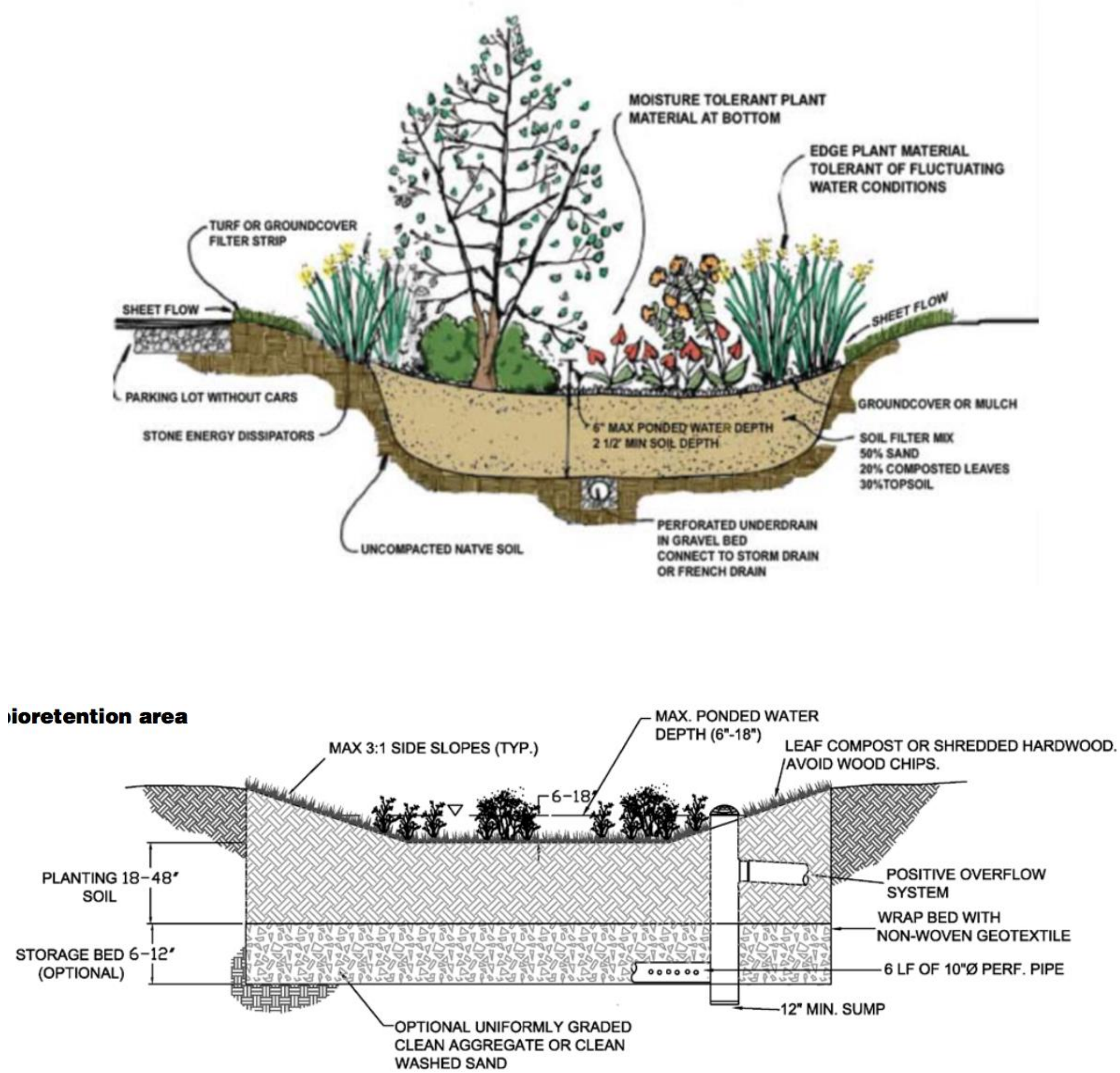


Figure 3: Illustrative and technical diagrams of typical bioretention basins with common features and dimensions labeled (SEMCOG 2008).

Despite sharing a common technology and purpose, bioretention systems vary greatly in design based on location and site-specific features. As a result, basins also vary greatly in their function based on design decisions, including the size of the surface bowl, the soil pore volume, the moisture content of the soil, and the chosen drainage system (Davis *et al.* 2012). In a study by Manganka *et al.* (2015), researchers looked at the effects of different hydrologic and hydraulic factors on the pollutant treatment performance of different bioretention basins. They concluded that antecedent dry days, rainfall depth, and pollutant leaching all affect the performance of a basin (Manganka *et al.* 2015). Furthermore, land use, soil type, and the specific site at which a design is implemented also affects the overall performance of a system (Eckhart *et al.* 2017). Based on a recent study conducted in the warm temperate monsoon climate of Xi'an, bioretention systems have high pollutant removal capacities for total suspended solids (TSS), total phosphorus (TP), and total nitrogen (TN), showing up to a 91% reduction in all pollutants annually (Jiang *et al*. 2017). Perhaps just as importantly, the study showed that one system also reduced runoff volumes by up to 98.0% (Jiang *et al.* 2017). However, other basins in this study only reduced pollutant loads by about 54.3% and reduced runoff volume by 54.1%, which the researchers attributed to differences in inflow volume and design variations (Jiang *et al.* 2017). Despite the general success of bioretention basins in runoff reduction and pollutant removal, these systems may also contribute to pollutant buildup and fail to address large rain events effectively. Hunt *et al.* (2012) warns that these systems might leach N and P if excessive organic matter is present within the system, and intense rainfall can result in overflow that causes erosion and flooding.

1.2 Green roof function and components

In contrast to bioretention systems, green roofs function primarily to retain rainfall and thus reduce runoff that would otherwise contribute to flooding, erosion, and associated hazards. Roofs may be either extensive or intensive; the former has a shallow substrate and typically has smaller vegetation while the latter has deeper substrate and can sustain shrubs and even small trees (Oberndorfer *et al.* 2007). Typical components of a green roof system often include vegetation, a substrate layer, a filter membrane, a drainage layer, and a waterproofing layer, although these layers frequently vary in design or may be modified (Figure 4) (Oberndorfer *et al.* 2007; Vijayaraghvan 2016). In relatively recent studies, the average amount of rainfall retained by green roofs was cited as between 20% and 100% (Ahiablame *et al.* 2012). The efficacy of any particular system, as with bioretention, varies according to green roof design characteristics and characteristics of rainfall events. Berndtsson (2010) reported that soil thickness, slope of roof, type of vegetation, and soil type are amongst the variables affecting water retention performance. Weather conditions affecting retention might include the antecedent dry periods, the climate, and the intensity or duration of any specific rain event (Berndtsson 2010). In terms of nutrient retention, a consensus is lacking as to whether green roofs are overall beneficial or detrimental to reducing pollutant concentrations in stormwater. Whereas some studies find that certain metals are reduced in green roof runoff, many find that P and N concentrations increase significantly in runoff; this suggests leaching from the soil medium that actually may further deteriorate water quality (Dietz 2007; Berndtsson 2010; Ahiablame *et al.* 2012).

Figure 4: Typical components included in a green roof system (Vijayaraghvan 2016).

1.3 Bioretention and green roof research gaps in semi-arid and arid climates

Research into the use of bioretention and green roof systems in semi-arid and arid climates is limited. “Arid” climates are either “cold steppe/desert” or “hot steppe/desert” depending on whether their annual temperature is below or above 18°C (Kottek *et al.* 2016). The “dryness threshold,” P_{th}, is based on the average annual temperature in C° and on the “annual cycle of precipitation,” and P_{th} is also used in the Köppen climate scheme to classify steppes versus deserts (Kottek *et al.* 2006).

In this study, locations classified as “semi-arid” or “arid” by the researchers are included for evaluation. Furthermore, some studies focused on the bioretention and green roof performances in Mediterranean areas with the Köppen classification “Cs,” meaning they have “hot-dry summers,” are also included (Kottek *et al.* 2016). Aschmann (1973) describes the “Mediterranean shrub or chaparral climate” as areas where more than 65% of annual rainfall is consolidated within the winter portion of the year, and plants in the summer months are drought-stressed. Other studies designate the annual rainfall depth as the only distinguishing feature of Mediterranean regions relative to their semi-arid or arid neighbors (Lavee *et al.* 1998). Furthermore, these areas have been identified as particularly susceptible to desertification and increasingly arid conditions resulting from climate change (Gao and Giorgi 2008). These trends are already apparent in plant phenology changes due to increased temperature and decreased precipitation in the Mediterranean basin (Gordo and Sanz 2010). Therefore, these areas are included in this analysis given their increasingly similar characteristics to those of semi-arid regions.

In an early review of bioretention system research and design by Roy-Poirier *et al.* (2010), the authors remark that there is a shortage of research into use and performance of bioretention basins in climates that differ much from that of the Eastern U.S., particularly semi-arid and arid climates. Houdeshel *et al.* (2013) called for increased research in the area, as well as providing design recommendations for future study based on “ecological principles” (Houdeshel *et al.* 2013). Two years later, Jiang *et al.* (2015) reviewed the efficacy of systems in arid and semi-arid climates. The review considered very few studies, but they found that data showed an average runoff reduction of only 53% in these climates (Jiang *et al.* 2015). Although the concentration of TSS in semi-arid and arid climates were similar to those of humid climates (91%), pollutant loads for TP and TN actually increased by 133% to 350%, likely due to soil leaching (Jiang *et al.* 2015). This study also cited the need for additional work to improve bioretention performance in arid climates (Jiang *et al.* 2015). Similarly, green roof studies conducted in arid and semi-arid climates are relatively scarce. Furthermore, many green roof studies investigate the survival and feasibility of different planting and maintenance schemes in arid climates and not stormwater retention or pollution reduction. For example, the EPA conducted a study in Denver, Colorado that put emphasis primarily on determining location-specific plant species survival in varying environmental and design conditions rather than each system’s stormwater management efficiency (EPA 2012).

1.4 Stormwater management challenges in semi-arid and arid climates

Stormwater management is a critical concern in all climates, but semi-arid and arid regions, because of the disproportionate effects of climate change and urbanization exacerbated by extreme weather conditions, are a research priority. Guatam *et al.* (2010) acknowledged that the hydrometeorology of the arid western United States varies drastically compared to locations with humid climates. Rainfall depth and intensity per storm event are often larger than in humid regions, but dry periods between storms can be extensive (Guatam *et al.* 2010). Table 1 shows the differences in rainfall characteristics based on climate type for different western U.S. cities; while cities such as Seattle receive high cumulative precipitation and relatively few days with rainfall depth greater than 25.4 mm, semi-arid Los Angeles receives a similar number of high depth rainfall days but only about a third as much rain annually (Guatam *et al.* 2010). These rainfall characteristics, typical of semi-arid and arid climates, result in heavy rainfall being confined to only a small number of storms with intense drought between storm events, as seen in the precipitation patterns of several semi-arid and arid cities illustrated in Figure 5 (Houdeshel *et al.* 2013). Furthermore, high temperatures result in higher evapotranspiration, which can further exacerbate plant stress in drought periods (Guatam *et al.* 2010). For example, Guatam *et al.* (2010) describes how rainfall in the U.S. southwest desert region is the lowest annually, but this area also experiences the highest evaporation rates (Figure 6). Due to the extreme nature of precipitation events in semi-arid and arid climates, urbanization can have a disproportionately significant effect on runoff behavior. Urbanization results in increased impermeable surface, leading to decreased infiltration and more flooding (Guatam *et al.* 2010).

	Daily precipitation(mm) >=				Total annual precipitation (mm)
	0.25	2.54	12.7	25.4	
	No. of days				
Albuquerque	63.9	27.3	3.9	0.5	240.5
Austin	87.8	51.2	21.2	9.3	854.7
Denver	91	39.5	7.1	2.4	401.6
Boise	89.7	40.9	3.3	0.1	309.6
Las Vegas	28.6	12.1	2.3	0.3	114.0
Los Angeles	35.5	22.6	9	3.6	334.0
Phoenix	37.7	19.3	4.9	1	210.6
Reno	53.6	21.8	2.7	0.4	190.0
Seattle	150	92.1	21.2	4.5	941.6

Table 1: Differences in rainfall amounts per day and per year for western U.S. cities with different climate classifications (Guatam *et al.* 2010; National Climatic Data Center 2004).

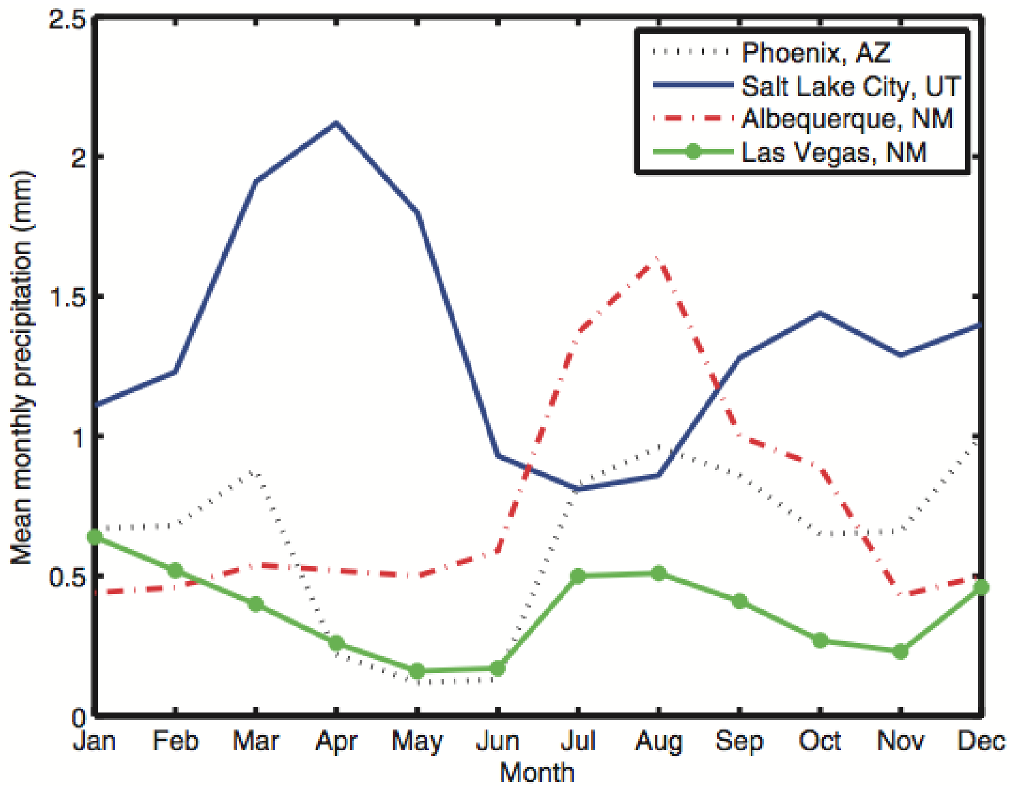


Figure 5: Seasonal rainfall patterns featuring monsoonal periods and intermittent dry periods for several semi-arid and arid cities (Houdeshel *et al.* 2012).

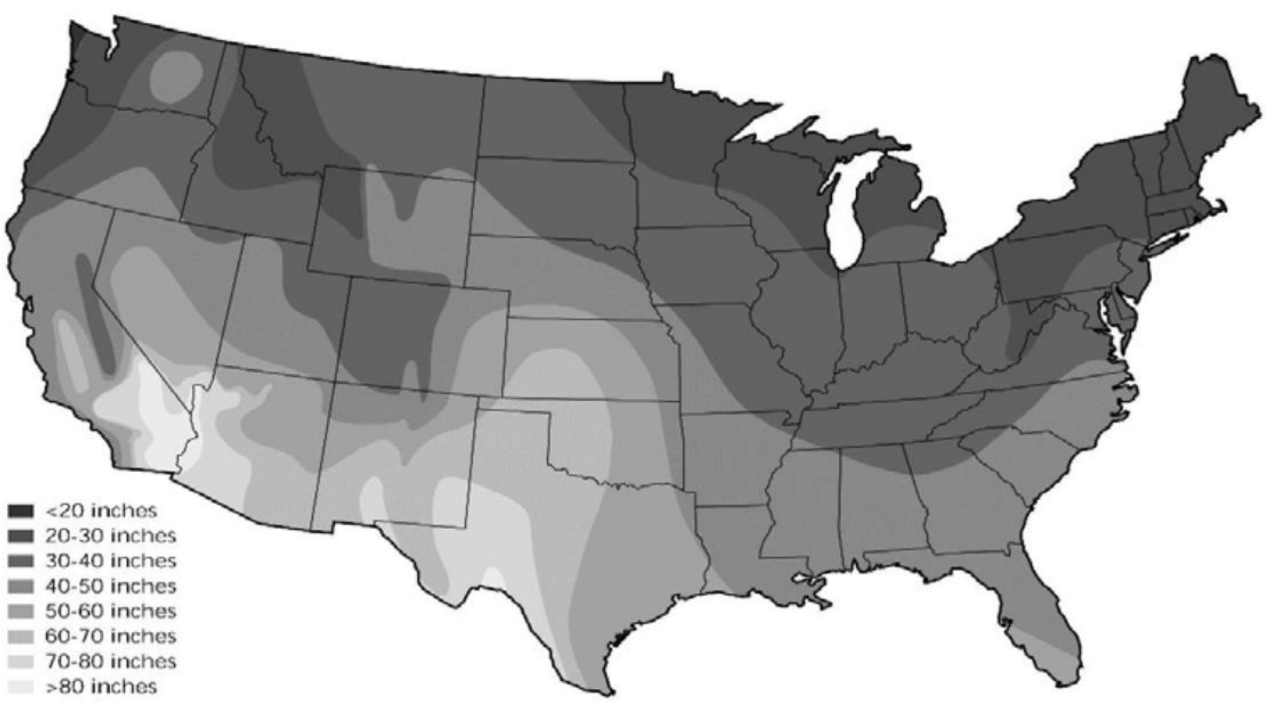


Figure 6: Average annual lake evaporation from 1946-1955 across the United States (Guatam *et al.* 2010; FISRWG 1998).

In terms of water quality, dry-region runoff often has higher pollutant levels due to the greater buildup of pollutants occurs between storm events (Guatam *et al.* 2010). The lack of substantial vegetation in arid and semi-arid climates poses another substantial threat to increased runoff. Soils lack the thick organic layer more common to humid climates, and the “maximum flood peaks, flash-flood potentials, and runoff potentials” are much higher in semi-arid regions compared to humid areas due to lower soil infiltration and easy erosion of barely-vegetated surfaces (Guatam *et al.* 2010; Osterkamp and Friedman 2000). Vegetation for bioremediation is hard to sustain in rain-deprived environments, and additional irrigation is often necessary (EPA 2010). The EPA states that almost “one-third of all freshwater withdrawals in the United States are used for irrigation,” and climate change likely will impact semi-arid and arid parts of the world further in terms of water shortages (EPA 2016). As a result, utilizing and maintaining stormwater management technologies in sustainable and responsible ways is paramount to protecting the resources of arid and semi-arid locations (EPA 2016).

The impetus for investigating semi-arid and arid climates stems from the large area of semi-arid, arid, and Mediterranean climates, as well as the disproportionately detrimental effects that climate change and urbanization will likely have on rainfall and stormwater runoff characteristics in these locations. Semi-arid and arid climates, as well as chaparral climates, comprise a large part of the western and southwestern United States, much of Australia, eastern Europe, and parts of the Mediterranean (Figure 7) (Kottek *et al.* 2006). Beck *et al.* (2005) determined land area change of each climate classification over the 20th century in response to climate change. Comparing the five main Köppen climate types, land categorized as “arid” (climate type B) increased the most from 1955 to 1995, which is a trend that other studies confirm (Figure 8) (Beck *et al.* 2005; Huang *et al.* 2016).

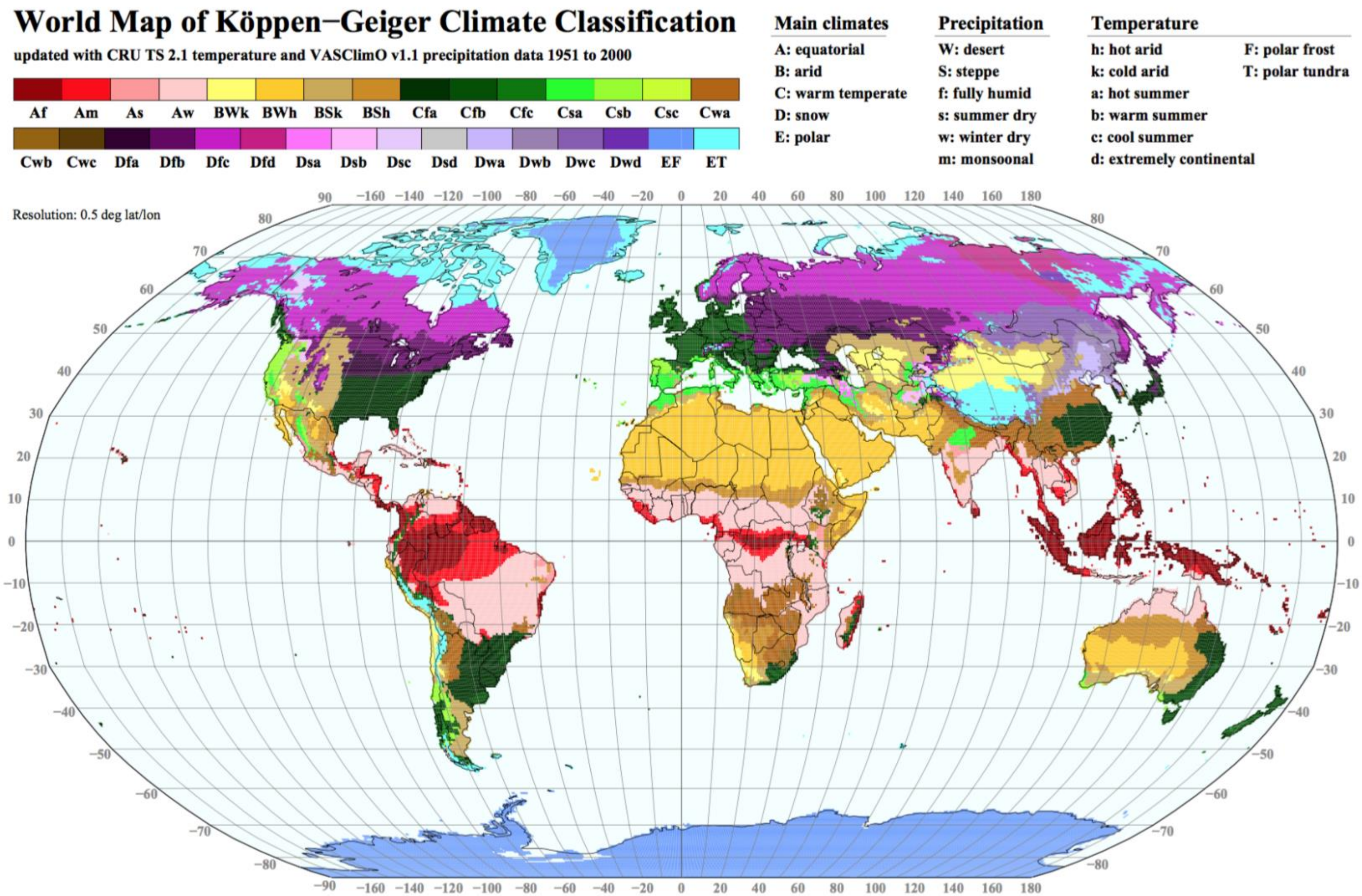


Figure 7: Distribution of climate types globally based on the Köppen-Geiger climate classification system with semi-arid and arid climate locations colored shades of yellow (Kottek *et al.* 2006).

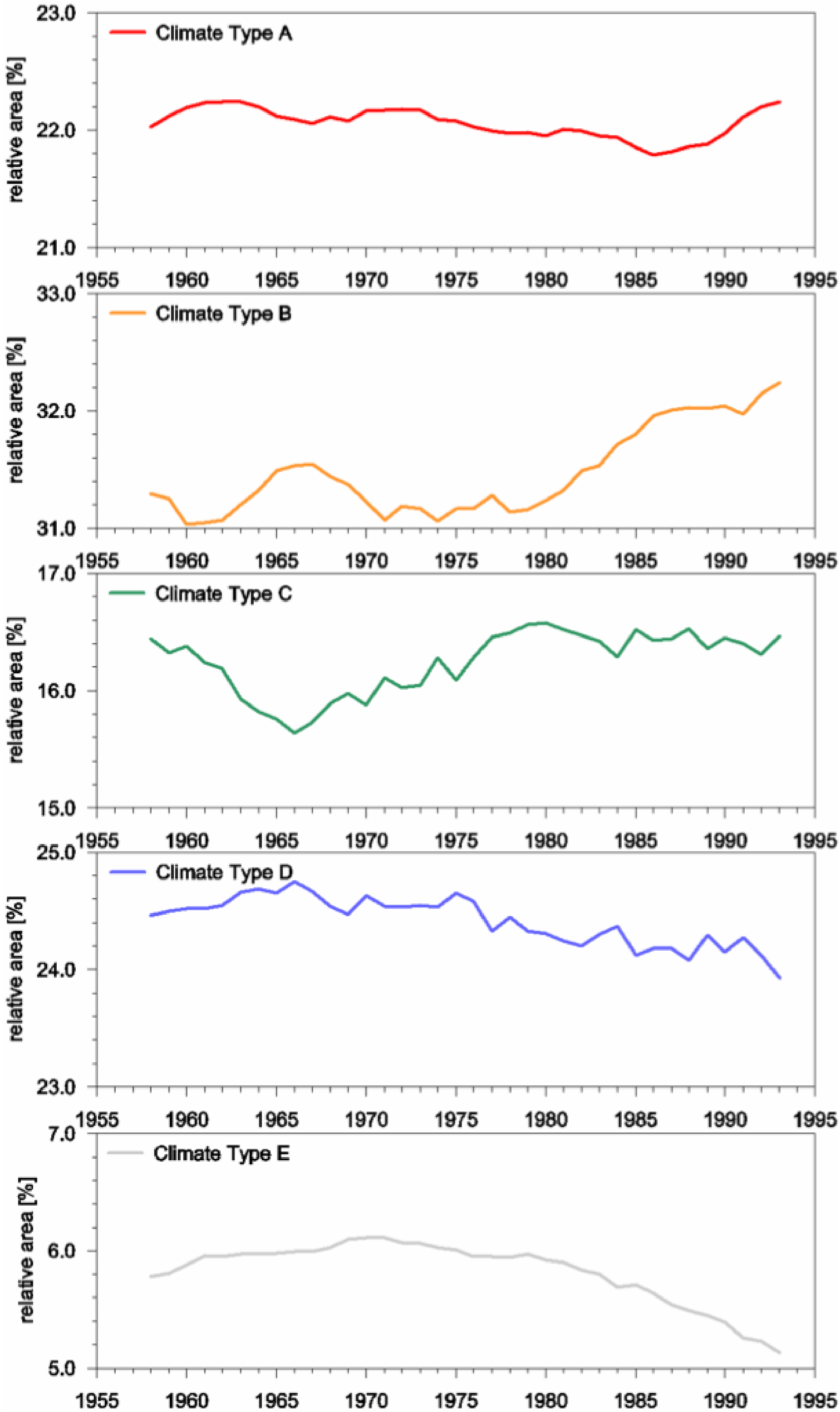


Figure 8: Change in percentage of global land area falling under tropical (Type A), arid (Type B), warm temperate (Type C), snow and boreal forest (Type D), and polar (Type E) Köppen climate types from 1955 through 1995 (Beck *et al.* 2005).

1.5 Utility of runoff quality and quantity controls in semi-arid and arid climates

As mentioned previously, through this paper, I seek to address what utility green roofs and bioretention systems have in semi-arid and arid climates. I also strive to determine what characteristics optimize the function of each and identify shortcomings that warrant future study. Chapter 2 of this thesis provides an overview of bioretention and green roof development, applications, and performances in temperate, cold, and tropical climate types in order to demonstrate the potential for use in a variety of conditions. Chapters 3 and 4 summarize research evaluating the retention capacity and pollutant removal of bioretention systems in semi-arid and arid climates; in these sections, I identify the mechanisms and design variations by which stormwater management might be optimized under climatic conditions. Chapters 5 and 6 do the same for green roof studies in semi-arid and arid climates. Chapter 7 discusses the design variations that limit and enhance the performance of green roof systems, particularly in terms of identifying characteristics that will best preserve the survivorship of vegetation. Chapter 8 introduces the possibility of using treatment trains to better improve management functions. It includes a review of literature on treatment train performance in a variety of climates in order to demonstrate the improved functionality of combined systems compared to individual techniques under different conditions. This includes the possibility of harvesting and reusing water within the system in order to irrigate and further treat stormwater runoff on-site. The final section, Chapter 9, summarizes the ideal stormwater management techniques and design for use in semi-arid and arid climates, ascertains the best options for particular management goals, and argues for the use of each technology in the treatment train format to attain more comprehensive management in the future.

Ultimately, I argue that bioretention systems improve the quality of runoff and help retain excess quantities. Green roofs reduce runoff volumes significantly in semi-arid and arid climates, but they often contribute additional pollutants to the system rather than remove significant amounts. Given the limited performance of each system individually in terms of managing both components runoff, as well as the unique and extreme conditions of semi-arid and arid regions, I suggest that combined systems in the form of treatment trains be utilized in the future in order to more effectively manage runoff. I also recommend that water harvesting and recycling be implemented within each system in order to improve quality and reduce quantity further, as well as to responsibly source water for irrigation in a manner that continues to improve overall functionality.

2. Existing studies of bioretention and green roof systems in various climates

2.1 History of bioretention technology

Despite the growing integration of LID technologies into urban design in a variety of climates and sites, bioretention and green roof technologies were developed in response to the environmental conditions around which they originated. Bioretention technology was first developed and examined in the humid subtropical climate of Prince George's County in eastern Maryland (Coffman *et al.* 1994). The performance of bioretention in various urban retrofit and residential settings around Prince George's was studied to determine the versatility and efficacy of each design. The ponding area, root zone, sand bed, and organic layer (Figure 9), as the four mandatory components of the system, were considered, and Coffman *et al.* (1994) chose the dimensions and plants to resemble a deciduous forest ecosystem, which would better capture the natural hydrology of the sites. Coffman *et al.* (1994) also recommended a "stratified" planting scheme to create a microclimate and buffer the system from ephemeral phenomena, such as wind and direct sunlight. The study determined that the bioretention area should comprise about 7% of the entire site area in order to reduce runoff sufficiently in a 0.7-inch rainfall scenario (Coffman *et al.* 1994). Ideal soil qualities were sandy loam textures with a 10-25% clay content and pH of between 5.5. and 6.5, as these values allow for ideal infiltration and adsorption of nutrients within the soil (Coffman *et al.* 1994).

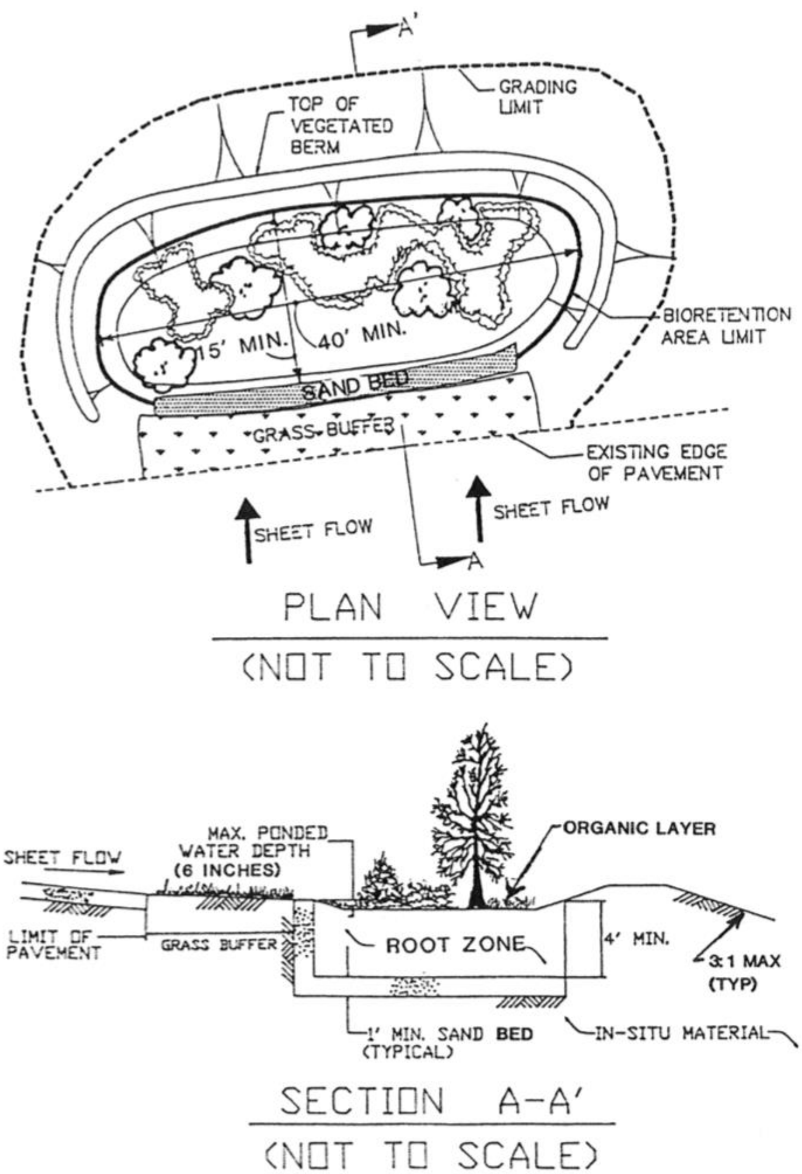


Figure 9: Plan and section of an early bioretention design created for Prince George's County, MD (Coffman *et al.* 1994).

While Coffman *et al.* (1994) set the precedent for bioretention design, later studies tested and modified the characteristics of bioretention units to improve both stormwater volume reduction and nutrient removal capacity. Davis *et al.* (2008) found that for 49 rainfall events at the University of Maryland campus, the bioretention systems produced no overflow for 18% of the events, and mean peak flow was reduced by 49-58%. This study and others introduced an "anoxic zone," or saturated area for denitrification, into the bioretention template; some studies also favored a clay content of less than 10% and added a mulch layer to the top of the cell to provide enhanced infiltration and adsorption, respectively (Kim *et al.* 2003; Davis *et al.* 2008; SEMCOG 2008; EPA 1999). Underdrain additions, located below the planting soil layer (Figure 10), also quickly became typical of bioretention basins. Underdrains originally consisted of a perforated pipe or series of pipes placed within the gravel layer of the system to help prevent flooding of the system as a whole, especially in high-intensity storm events, by diverting excess water to regular storm drains (Davis *et al.* 2001; Schueler *et al.* 2007). Later studies found that inclusion of an "internal water storage layer" (IWS), synonymous with the anoxic zone mentioned previously, not only improved water quality, but it also increased the hydraulic retention time of water within the system so that the final outflow volume was much less (Brown and Hunt 2011).

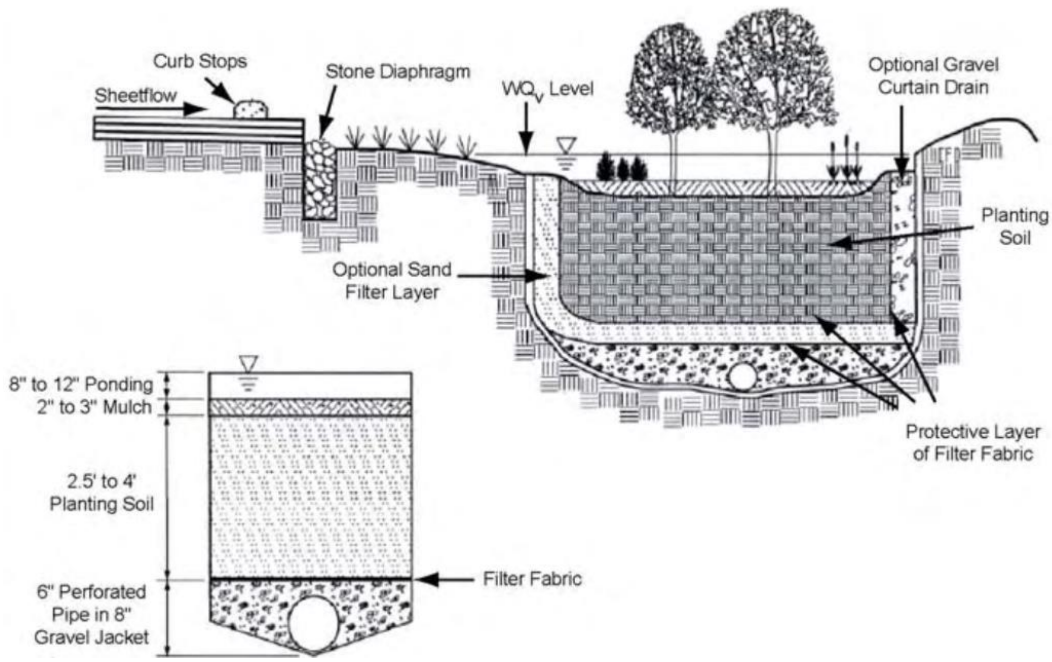
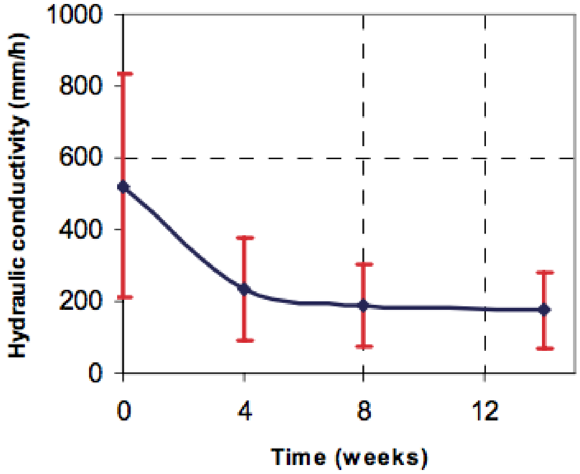


Figure 10: Plan and section of bioretention system with underdrain included (Schueler *et al.* 2007).

2.2 Performance of bioretention in temperate climates

The vast majority of bioretention applications and modelling studies evaluate the technology's performance in locations with humid, temperate climate regimes. Given the higher quantity of rainfall, bioretention efficacy largely depends upon the soil's hydraulic conductivity, a means of measuring the flow of water through the soil based on soil characteristics like pore space (Westholm 2006). Focused on the sub-tropical and temperate climate zones of Australia, Le Coustumer *et al.* (2007) conducted field and laboratory tests examining the maintenance of hydraulic conductivity in bioretention basins over time. They determined that in "normal" rainfall scenarios, hydraulic conductivity decreased in the first four weeks before stabilizing at the relatively constant value (Figure 11) (Le Coustumer *et al.* 2007). In heavy rainfall scenarios, however, hydraulic conductivity (K) decreased almost twice as quickly due to greater loading rates; they argue that this shows the importance of both the design storm size and media type chosen for each system (Le Coustumer *et al.* 2007).



Time	K mm/h	Std. Dev. (mm/h)	Cv (%)
Initial	521	325	62
4 weeks	235	143	61
8 weeks	189	117	62
14 weeks	176	103	58

*Number of points used =124

Figure 11: Initial reduction and gradual stabilizing of hydraulic conductivity (mm/h) over time in a "normal" rainfall scenario in a temperate Australian bioretention system (Le Coustumer *et al.* 2007).

Given the array of variable design parameters and differences in the characteristics of each location, studies cite a wide range of retention and pollutant removal efficiencies for temperate climate systems. Hunt *et al.* (2006) found that bioretention cells installed in North Carolina, USA significantly reduced the overflow and amount of water funneled into the storm drain system by an annual average of 78%. However, they determined that seasonality had great effect in this climate. For example, winter had a significantly higher outflow:runoff ratio compared to other seasons (Table 2), which was likely due to lower ET rates and a higher water table during the cold weather months (Hunt *et al.* 2006). Volume reduction generally decreases as rainfall depth and duration of rainfall increases (Li *et al.* 2009), and studies have suggested increasing the bowl depth and increasing the surface area of each basin as means to address larger storms (Hunt *et al.* 2012). Other studies cite runoff volume retention ranging from around 33% in Australia for an undersized treatment train of bioretention cells with various filter media

(Hatt *et al.* 2009) to 100% in Rocky Mount, NC, USA for a sand bioretention cell with IWS layer included (Brown and Hunt 2011).

	OF:RO		ANOVA p-value			
Season	Mean	Standard deviation	Spring	Summer	Fall	Winter
Spring	0.14	0.18	—	0.25	0.85	*0.002*
Summer	0.07	0.07	—	—	0.14	*<0.001*
Fall	0.13	0.10	—	—	—	*0.002*
Winter	0.54	0.21	—	—	—	—

Note: As measured by outflow:runoff ratio, any p-value less than 0.05 (significance) is italicized.

Table 2: Differences between outflow:runoff ratios for each season in a North Carolina bioretention cell (Hunt *et al.* 2006).

In terms of peak flow reduction, bioretention systems are quite effective. Hunt *et al.* (2008) reported an average 99% peak flow reduction for small to medium storms in Charlotte, North Carolina. Davis (2008) reported a lower peak flow reduction of only 44% in his College Park, MD study, but noted that bioretention cells also contributed a delay in peak flow by a factor of about 2. Figure 12 shows what the study refers to as "typical hydrographs," with peaks in effluent flow delayed by about 2 hours owing to the influence of bioretention cells (Davis 2008). As with overall runoff reduction performance, the studies collectively show that variations in design parameters and restraints have an effect on the performance of each system even in temperate climates.

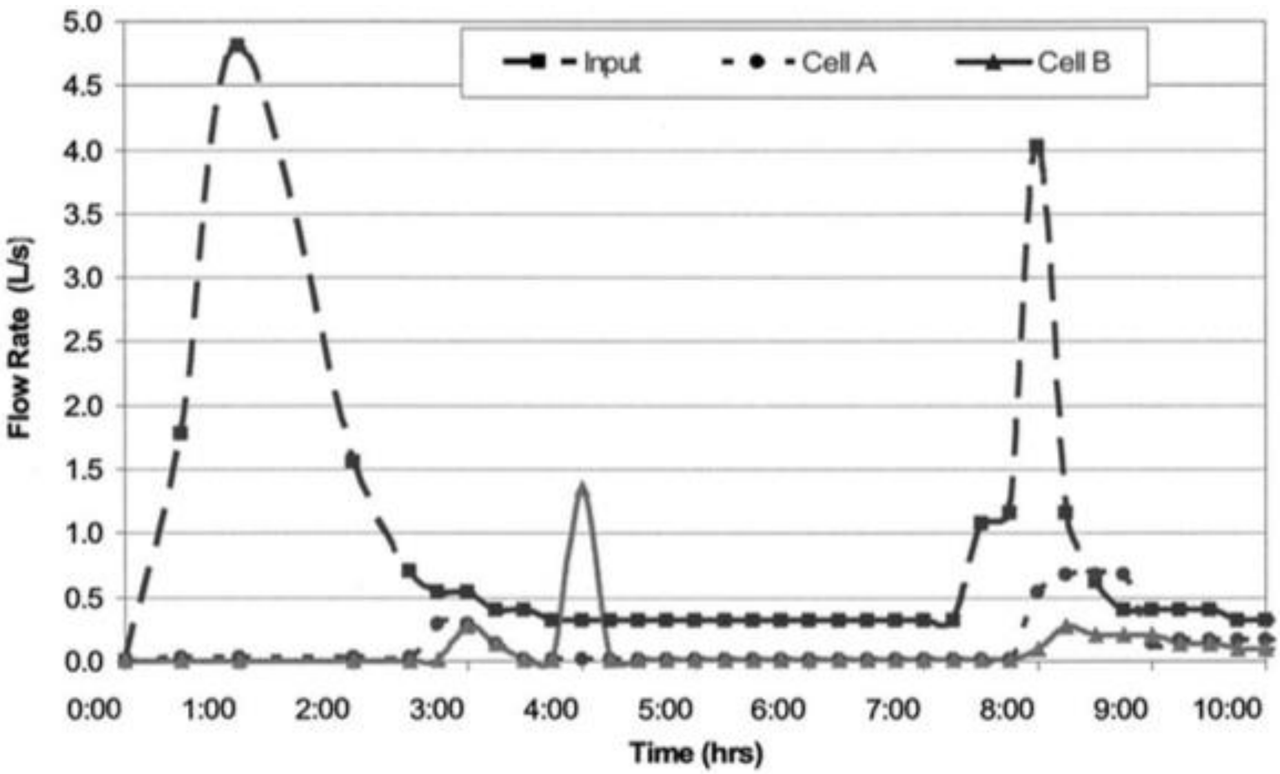


Figure 12: Hydrographs illustrating the delay in peak flow achieved by a shallow (Cell B) and deep (Cell A) bioretention cell installed at the University of Maryland in response to initial runoff inflow (Input) (Davis. 2008).

Bioretention systems also have one of the highest pollutant removal capacities of LID technologies. TSS is reduced up to 99% in temperate climates; this occurs via settlement of particles as runoff ponds and passes through the filter media (Ahiablame *et al.* 2012). Initial TSS removal might be less significant as the system stabilizes, and clogging may limit the efficacy of the system if proper maintenance, such as filter media replacement and weeding, are not sustained (Roy-Poirier *et al.* 2010). Dissolved nutrients including phosphorous and nitrogen, bacteria, and metals are also removed at high rates in temperate climates. This often occurs via adsorption and plant uptake within the system (EPA 1999). Soil media with low organic matter and phosphorus is important in avoiding the possibility of phosphorous leaching out of the system (LeFevre *et al.* 2015). Furthermore, phosphates can bind to amendments added to media, such as fly ash or aluminum hydroxides; this improves the overall P removal of the system (LeFevre *et al.* 2015). Nitrogen removal can be more difficult to achieve given the soluble nature of nitrate and nitrite which prevents easy adsorption (LeFevre *et al.* 2015). However, with the addition of an IWS layer (Figure 13), a permanent saturated zone provides for denitrification and enhances the removal of nitrate (Kim *et al.* 2003). Average metal reductions vary between 30% and 99% for temperate studies, and fly ash amendment is one design intervention that can improve the retention capacity of inefficient systems (Ahiablame *et al.* 2012). Soil media accounts for much more metal removal in bioretention systems than plant uptake (88%-97% and 0.5%-3.3%, respectively), and systems should be cleaned occasionally via harvesting of contaminated plant materials and replacement of soil media to ensure continued metal adsorption (Roy-Poirier *et al.* 2010). Retention of bacteria is also high (70% to 99%), and the efficacy of removal increases as a system ages (Ahiablame *et al.* 2012).

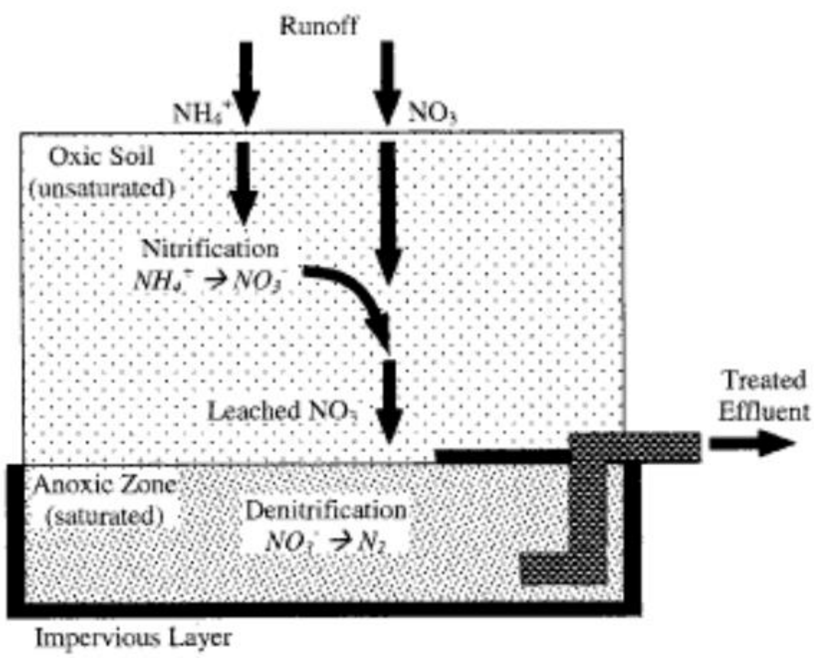


Figure 13: Diagram of a bioretention cell with an IWS layer (anoxic zone) included included and the transformation of nitrogen throughout the system (Kim *et al.* 2003).

2.3 Application of bioretention technology in alternative climates: cold and tropical

Later literature reviews note that the emergence of design specifications for different locales prompts much of the variation prevalent in bioretention design specifications. At this point, more overt deviations in design emerge for different climate types, primarily. Bioretention cells function well in cold climates despite periods of surface layer freezing. Khan *et al.* (2012) studied the hydrologic performance of bioretention systems in Calgary, Alberta during cold periods of around 0°C compared to normal warm periods in the same location. They found that the basins could reduce runoff volume by an average 91.5% and reduce the peak flow rate by 95.3%; there was no significant difference observed between change in volume for cold and warm periods when considering all rain events (Khan *et al.* 2012). However, when researchers did not consider small events (depth < 32 mm), they found a significant difference in change in volume between seasons (Khan *et al.* 2012). Figure 14 shows what the researchers identify as an increase in effluent volume (V_e), peak flow rate (Q_e), and peak delays caused by the frozen surface layer deviating and slowing flow until the flow encounters an open path downwards (Khan *et al.* 2012). Therefore, large or high-intensity events may reduce bioretention performance slightly in cold climates compared to warm climates.

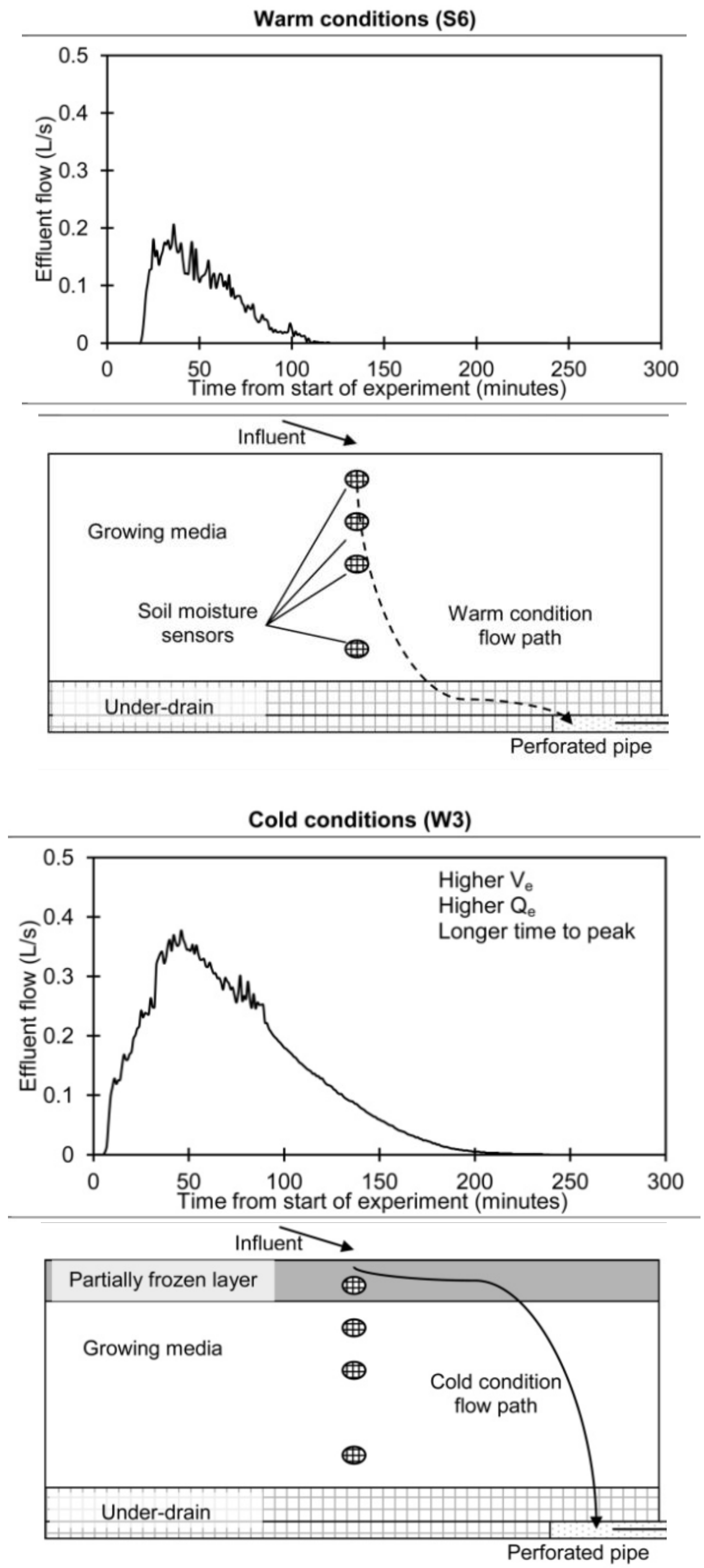


Figure 14: Differing effects of warm and cold conditions on the effluent flow characteristics and infiltration movement of water in a bioretention system in Calgary, Alberta (Khan *et al.* 2012).

Other studies confirm that although cold weather causes frozen topsoil, the hydraulic functioning of the overall system is not significantly affected, and frost does not necessarily affect the permeability of the filter media (Roseen *et al.* 2009). Paus *et al.* (2016) determined that in a bioretention study conducted in Finland, a minimum saturated hydraulic conductivity (K_{sat}) of 10 cm/h increased the water infiltrated by the bioretention system. The percentage of runoff infiltrated by the system increased almost exponentially as the saturated hydraulic conductivity increased (Figure 15) (Paus *et al.* 2016). The study also found that increasing hydraulic conductivity has a much greater effect on increasing infiltration than either increasing the surface area (A_{bio}) of the basin or increasing the maximum level of water on the surface (h_{max}) (Figure 15) (Paus *et al.* 2016). Unsurprisingly, the choice of a well-draining soil is of utmost importance in cold weather climates, as unsaturated soils exposed to freezing temperatures maintain infiltration, while saturated soils under the same conditions experience “concrete frost” and low permeability (LeFevre *et al.* 2009).

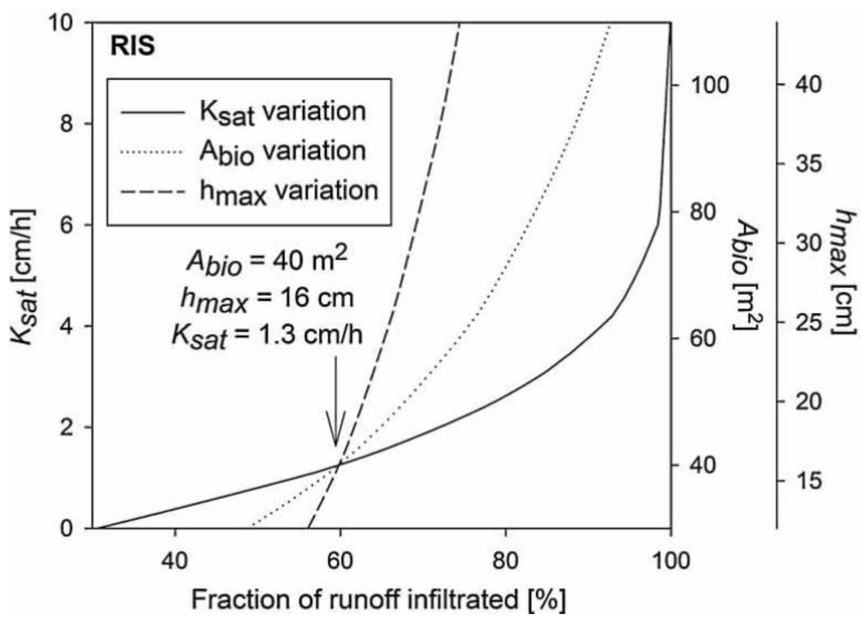


Figure 15: Relationship between fraction of runoff infiltrated and saturated hydraulic conductivity (K_{sat}), bioretention surface area (A_{bio}), and maximum height of ponded water (h_{max}) of a bioretention basin (Paus *et al.* 2016).

Bioretention systems retain most nutrients, regardless of the season (Muthanna *et al.* 2007; Blecken *et al.* 2007). The presence of vegetation within the system is necessary in order to allow for infiltration of water and additional pollutant removal even in colder months (Valtanen *et al.* 2017). Deicing salt, commonly used in cold weather climates, resulted in lower copper and aluminum retention in the Valtanen *et al.* (2017) study conducted in Finland. They attributed this to the mobilization of metals alongside salt due to processes such as cation exchange. However, other pollutants were retained to similar extents in both warm and cool seasons despite the cold climate of Finland (Valtanen *et al.* 2017).

In particularly wet, tropical climates, research is somewhat limited compared to that conducted in cold climates. Research is overwhelmingly focused on the potential removal of runoff contaminants by bioretention, although studies note that, as in temperate climates, overflow occurs when heavy rainfall events take place (Wang *et al.* 2017). Furthermore, climate change modeling conducted in Singapore (Figure 16) shows that with increasingly extreme scenarios of urbanization (SSPs) and climate change (RCPs), peak runoff increased substantially in response (Wang *et al.* 2016). Therefore, future studies need to focus specifically on the potential for bioretention to act as quantity control for excess runoff in tropical climates.

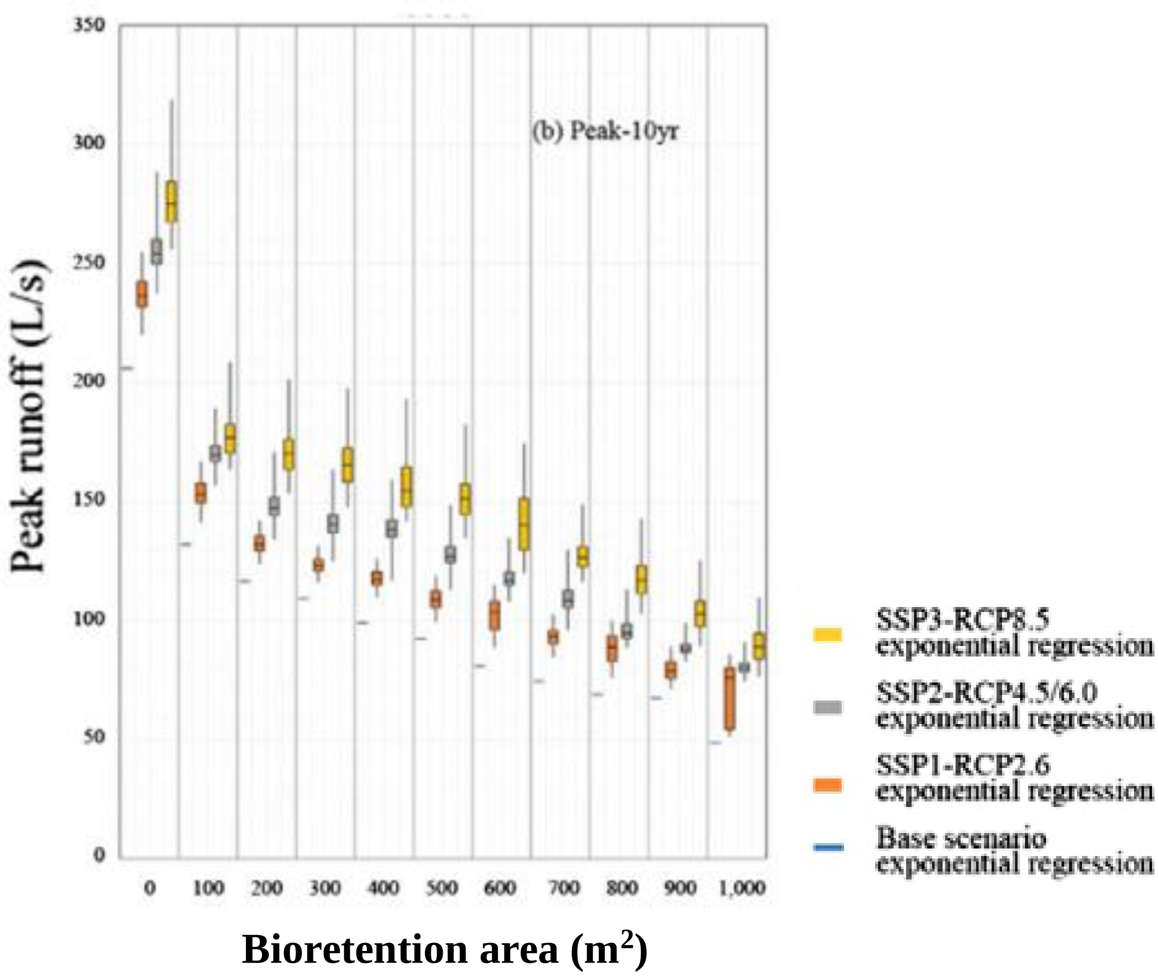


Figure 16: Simulated peak runoff values for different climate change (RCP) and urbanization(SSP) scenarios with different bioretention areas (m^2) (Wang *et al.* 2016).

Study of pollutant removal performance shows limitations to bioretention systems in tropical climates that are based on temperate climate designs due to higher rainfall depth and more frequent days of rainfall. Wang *et al.* (2017) concluded that based on 96 storms monitored in a bioretention system in Singapore, average pollutant removal reductions were only 46% for TP, 25% for TN, and 53% for TSS, respectively. Nutrient and suspended sediment retention varied greatly based on individual storm characteristics (Table 3). The researchers attribute this to overflow that occurs when rainfall depth is greater than 10-30 mm and limits the potential for pollutant removal (Wang *et al.* 2017). Table 3 shows storm events in increasing order from left to right in terms of rainfall depth, and a relationship can be noted between increased rainfall and decreased pollutant removal. In addition to the role that heavy rain can have in decreasing pollutant removal, tropical climates also generally have less pollutant buildup due to frequent and heavy rainfall (Wang *et al.* 2017). As a result, storm events often have lower event mean concentrations (EMCs), which is obtained by dividing the loading mass of pollutants by runoff volume, than in temperate climates where pollutants have a greater opportunity to build up; thus the “first flush” function of bioretention systems often fails in tropical locations (Wang *et al.* 2017). Interestingly, Muha *et al.* (2016) observed that for a small scale bioretention system in tropical Malaysia modeled using MUSIC, a common stormwater management modeling platform, and also monitored through observation, pollutant removal percentages were high and comparable to temperate values. The system attained about 92-98%, 82-86%, and 61-77% removal for TSS, TP, and TN, respectively (Muha *et al.* 2016). Tropical studies, as with cold climate studies, highlight the importance of a vegetated presence within each system, as plants allow for a proper hydraulic conductivity of between 100-200 mm/hour to be achieved by taking up water that would otherwise contribute to the flow volume (Goh *et al.* 2015). This, in turn, allows for increased pollutant removal by limiting overflow and clogging of the system as the system ages (Goh *et al.* 2015).

	Removal rate (%)							
	#3	#2	#1	#5	#6	#4	Mean	SD
TN	68	58	35	10	−24	4	25	35
TKN	72	63	61	58	−46	12	37	46
ON	75	80	74	69	6	12	53	34
NH_3–N	61	−23	36	96	−120	12	10	76
NO_3–N	96	82	−55	−14	−31	−22	9	63
NO_2–N	98	69	−118	−234	−129	−25	−56	128
TP	73	75	69	33	−7	32	46	32
TDP	55	58	59	4	4	66	41	29
PO_4–P	66	54	60	10	4	75	45	30
TOP	74	74	64	−13	−15	2	31	44
TDOP	35	73	53	−65	−22	56	22	54
TPIP	73	83	77	51	−25	−7	42	47
TPOP	82	74	67	5	2	−70	27	59
TSS	92	91	86	31	−12	29	53	43
COD	55	60	56	22	−94	−8	15	60

Table 3: Removal rates (%) of a tropical bioretention system for 6 different storm events which increase in rainfall depth (mm) from left to right (3.2mm, 8.0mm, 10.2mm, 29.4mm, 33.2mm, 40.2mm, respectively) (Wang *et al.* 2017).

2.4 History of green roof technology

While bioretention is a relatively new concept, green roofs have a much longer usage and history of development. Initial "intensive" gardens were largely for aesthetic purposes and contained a variety of plants requiring high maintenance, such as the "hanging gardens" of Semiramis (Kohler *et al.* 2002). However, "extensive" roofs were utilized as thermal mechanisms to insulate houses in extreme weather and thus always had a functional component (Kohler *et al.* 2002). At the beginning of the 20th century, Germany modified the green roof to act as a fire-protective measure, and later in the 1970s, it became adopted widely throughout the country and beyond due to its many environmental benefits (Oberndorfer *et al.* 2007). Today, green roofs are integrated into urban planning, with cities such as Germany constructing an estimated 14% of its new rooftops as green roofs (Oberndorfer *et al.* 2007). Studies continue to explore potential vegetation to utilize in green roofs, as well as investigating the possible benefits to air quality and other ecosystem services that might be provided by green roofs within a variety of climates (Oberndorfer *et al.* 2007).

2.5 Performance of green roof systems in temperate climates

While bioretention systems have the ability to impact both the quantity and quality of stormwater, green roof systems function as storage components for stormwater, thus managing volume and peak flow more than improving the overall quantity of runoff. In terms of reducing runoff volume, studies show that typical retention ranges between 20% and 100% (Ahiablame *et al.* 2012; Dietz 2007). A wide range of retention values exists largely due to the dynamic interactions and impacts of different design components and site conditions. As with bioretention systems, high rainfall depth and high intensity events both decrease the efficacy of the system's volume retention (Ahiablame *et al.* 2012; Berndtsson 2010). One study determined that as rainfall depth increases, the percentage of retention decreases; for example, green roofs retained 88% of the runoff from <25.4 mm storms compared to only 48% of that from >76.2 mm storms (Berndtsson 2010). Furthermore, green roof systems help delay runoff by about 10 minutes compared to traditional roofs (Berndtsson 2010). Studies show that in terms of design characteristics, water holding capacity of soil media, depth of substrate, slope, and vegetation composition all affect the performance of different systems (Berndtsson 2010). Typically, drier and deeper soil layers result in a higher rate of retention, and the relative contribution of substrate is higher than that of vegetation, particularly during the winter months (Berndtsson 2010). The effects of vegetation vary according to studies, with some noting more than double the retention capacity of vegetated systems compared to non-vegetated systems due to their ability to hold water for gradual release; alternatively, others observe significantly higher retention in bare soil, which they speculate is due to the vegetation lowering both temperatures and evapotranspiration (Berndtsson 2010). The age and slope of a green roof are two additional factors that affect the retention capacity of each system. Typically, lower slopes result in lower runoff, particularly in cases of low intensity rainfall (Berndtsson 2010). The overall performance also increases as the roof ages because the pore space and organic matter increase over time, increasing the storage capacity of the system (Oberndorfer *et al.* 2007; Berndtsson 2010). However, other studies show that the relationship between slope and retention performance is weak (Dietz 2007).

While both of the evaluated LID technologies have potentially effective runoff retention performances, green roofs stand in stark contrast with bioretention systems in temperate climates in terms of pollutant removal capacity. Overall, green roofs are not consistently effective at removing most pollutants, and in most cases, they contribute to high concentrations (Rowe *et al.* 2010). Studies vary greatly in terms of whether effluents from green roofs contain higher concentrations of phosphorous and nitrogen than control roofs (Berndtsson 2010). Factors affecting TP and TN concentrations are the age of the roof, fertilization and maintenance practices, the amount of water volume reduction, and plant and soil types (Berndtsson 2010; Rowe *et al.* 2010; Ahiablame *et al.* 2012). For example, pollutants likely decrease over time as a green roof ages and organic matter decomposes and leaches out of substrate material, and green roofs maintained with "conventional fertilizers" have higher nutrients in runoff than those using "controlled release fertilizers" (Rowe *et al.* 2010).

High metal concentrations in green roof effluent is less of a concern than nutrient pollution according to most studies. Ahiablame *et al.* (2012) reports that certain substrate amendments such as coal bottom ash and lava rock likely add metals to green roof runoff, and studies show the potential for both reduction and addition of Pb and Zn (Ahiablame *et al.* 2012).

Despite variable reports of nutrient and metal contamination in green roofs, green roofs appear to increase the pH of runoff, which Rowe *et al.* (2010) attributes to carbonate in the substrate. This has significance in terms of its potential to mitigate the effects of acid rain, and therefore, it also helps to prevent the leaching of metals that accompanies decreased pH (Rowe *et al.* 2010). Ultimately, Berndtsson (2010) attributes the overall deteriorated water quality of green roof effluent primarily to the substrate properties and the addition of fertilizers to the soil media. Therefore, limited use of fertilizer and careful selection of soil and plants might help to mitigate potential quality issues.

2.6 Application of green roof technology in alternative climates: cold and tropical

While relatively few studies have been conducted in tropical and cold climates compared to temperate locations, those that exist show great promise in terms of stormwater volume retention. As in temperate climates, retention performance in tropical regions is limited for long, intense, and high-depth rainfall events (Figure 17) (Simmons *et al.* 2008; Fang 2010; Wong and Jim 2014; Ferrans *et al.* 2018). Simmons *et al.* (2008) and Fang (2010) observed that retention capacity is “inversely proportional” to the intensity of the rainfall (Fang 2010). Light rain in Taiwan resulted in 87-100% retention, while for a heavy storm the study green roof only retained 26-33% of rainfall (Fang et al. 2010). A green roof in Singapore had average retention between 57-68% (Lim and Lu 2016), while average retention in a 3-year study in Columbia was 85% (Ferrans et al. 2018). Most studies also find that green roofs are of limited utility in monsoon season due to saturation of substrate (Wong and Jim 2014; Lim and Lu 2016). However, they still manage to be very effective at peak reduction and peak delay. For instance, Kok *et al.* (2016) found that up to 47.3% peak flow reduction could be achieved for a 60-minute storm by a Malaysian green roof, but the average peak discharge reduction for all storms was only 23.6%.

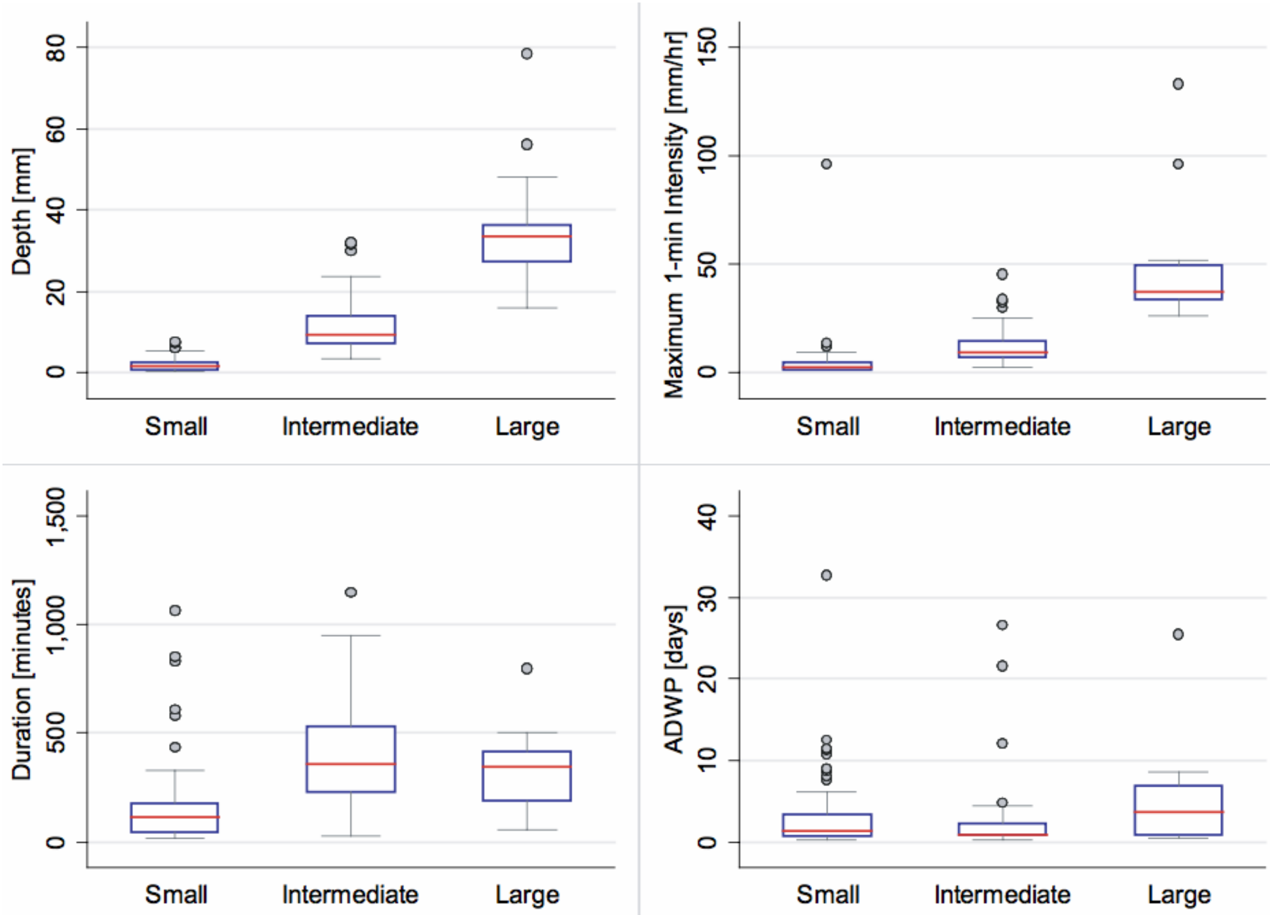


Figure 17: Range of rainfall event variables including duration, depth, intensity, and antecedent dry weather period (ADWP) observed and categorized for small, intermediate, and large rain events in the tropical climate of Colombia (Ferrans *et al.* 2018).

Wong and Jim (2014) evaluated the hydrological performance of green roof modules with four different substrate depths over a 10-month period in Hong Kong for parameters including peak delay, peak runoff reduction, and others (Table 4). They found that even though there were more heavy rain events (>10 mm) than small or medium events, the mean peak delay was about 25-35 minutes while the median was about 10 minutes (Table 5) (Wong and Jim 2014). At times, this resulted in a peak delay of up to 5 hours (Wong and Jim 2014). In terms of peak reduction, the mean peak reduction ranged between about 41-58% for all treatments because green roof layers lengthened the time it took precipitation to pass through the system prior to discharge (Wong and Jim 2014). The discharge onset delay was about one hour long, and the discharge duration was longer than the rainfall duration (Wong and Jim 2014). The longer discharge duration is beneficial for preventing floods and reducing erosion, both of which are particularly prevalent in tropical regions.

Term	Symbol	Definition
Peak delay	PD	The time difference between the peak of rainfall and the peak of discharge
Peak reduction	PR	The percentage of reduced peak discharge intensity relative to the respective peak rainfall intensity
Onset delay	OD	The time difference between the start of rainfall and the start of discharge
Discharge duration	DD	The total duration of discharge
Rainfall duration	RD	The total duration of rainfall
Time of concentration	T_c	The duration required for precipitation fallen at the farthest point of the basin to reach the drainage basin outlet

Table 4: List and definitions of parameters typically measured in stormwater management performance evaluations (Wong and Jim 2014).

Performance indicator	Treatment	Mean	Median	Max	Min
Retention, R (%)	40	38.9	24.3	100.0	0.4
	40RW	40.0	31.3	100.0	1.2
	80	45.3	36.3	100.0	0.9
	80RW	44.3	32.6	100.0	0.9
Peak delay, PD (min)[a]	40	24.9	4.0	264.0	0.0
	40RW	26.9	6.5	222.0	0.0
	80	35.4	18.0	269.0	1.0
	80RW	33.3	9.0	306.0	0.0
Peak reduction, PR (%)[a]	40	40.6	40.9	88.6	−21.3
	40RW	47.4	52.4	90.7	−4.1
	80	58.3	57.7	92.1	19.5
	80RW	53.2	59.0	84.2	0.4
Discharge onset delay, OD (h)	40	0.8	0.1	7.7	−0.1
	40RW	1.1	0.4	7.6	0.0
	80	1.2	0.5	10.0	−0.1
	80RW	1.2	0.4	8.3	−0.1
Discharge duration, DD (h)	40	11.2	7.3	89.7	1.1
	40RW	11.2	6.5	87.8	1.6
	80	12.1	8.1	90.1	0.2
	80RW	11.5	8.0	90.5	0.7
Time of concentration, T_c (h)	40	3.2	2.7	13.5	−3.5
	40RW	3.3	2.5	17.7	−3.7
	80	4.2	3.8	16.8	−2.7
	80RW	3.6	2.8	15.3	−3.9

[a] >10 mm events only ($n = 26$).

Table 5: Stormwater management performance of 4 green roofs with different substrate depths and composition (40=40mm sandy loam, 40RW=40 mm layered sandy loam and rockwool, 80=80mm sandy loam, 80RW=80mm layered sandy loam and rockwool) (Wong and Jim 2014).

Vegetation type, substrate type, and substrate depth are other factors affecting green roof volume management in tropical areas. Ayub *et al.* (2015) observed the performance of test beds with 3 different plant species in tropical Malaysia. They found that *A. compressus* achieved the highest peak flow reduction within the range of 51-67%, which they attributed to the grass' high-density coverage (Ayub *et al.* 2015). *K. pinnata* followed with lower peak flow attenuation between 29-47% due to wider and larger leaves (Ayub *et al.* 2015). Tan and Sia (2005) conducted a green roof study in Singapore and concluded that even in humid tropical locations, “irregular periods of depleted water in the root zone” requires use of drought-resistant plants. Furthermore, almost 50% of the *Sedum* varieties used in the study performed poorly, which supports the claim that extended wet periods make succulents less appropriate to this climate (Tan and Sia 2005).

Despite the effect that vegetation has on the system, most studies agree that substrate plays a greater role than vegetation in terms of managing overall stormwater volume (Simmons *et al.* 2008; Fang 2010). Fang (2010) argues that substrate accounts for 77-98% of retention, while vegetation accounts for only 2-23% because plants retain water more slowly than substrate. The study also suggests that substrate be greater than 15 cm in depth and notes that a lightweight substrate with a ratio of 20:20:20:40 of perlite, vermiculite, peat moss, and sandy soil, respectively, performed better than substrates with lower sandy soil ratios (Figure 18) (Fang 2010). Wong and Jim (2014) found no significant difference between different depths in terms of the performance of green roofs in Hong Kong. They argued that the successful peak mitigation of even the shallow 40 mm substrate shows that even shallow green roofs are effective at reducing peak flow in tropical locations (Wong and Jim 2014). Simmons *et al.* (2008) found large differences between green roof types, which they attributed to a combination of design decisions including substrate type and additional drainage layer characteristics, such as the sizing of the drainage cups and drainage holes (Figure 19).

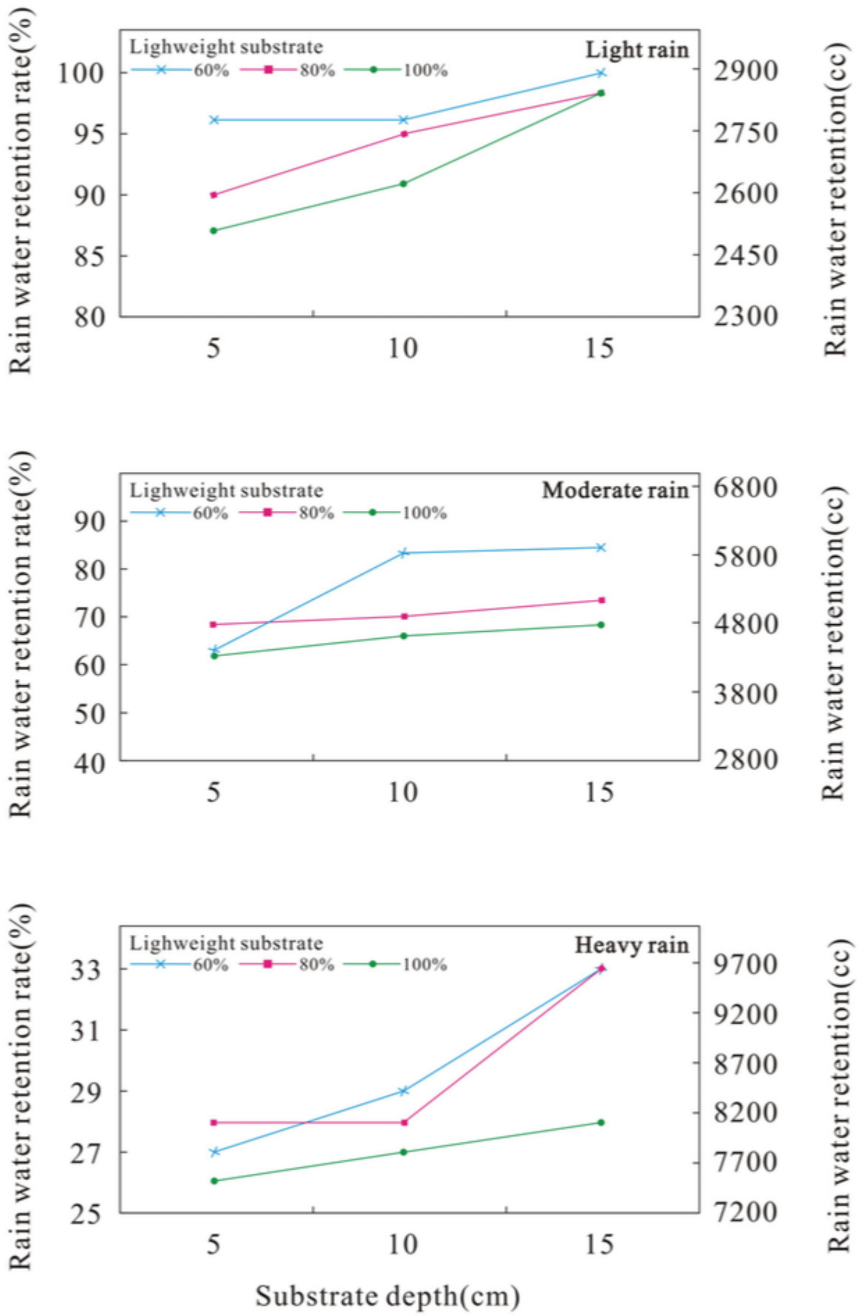


Figure 18: Stormwater retention performance of green roofs with different substrate types under light (10mm), medium (30mm), and heavy (100mm) rain events (Fang 2010).

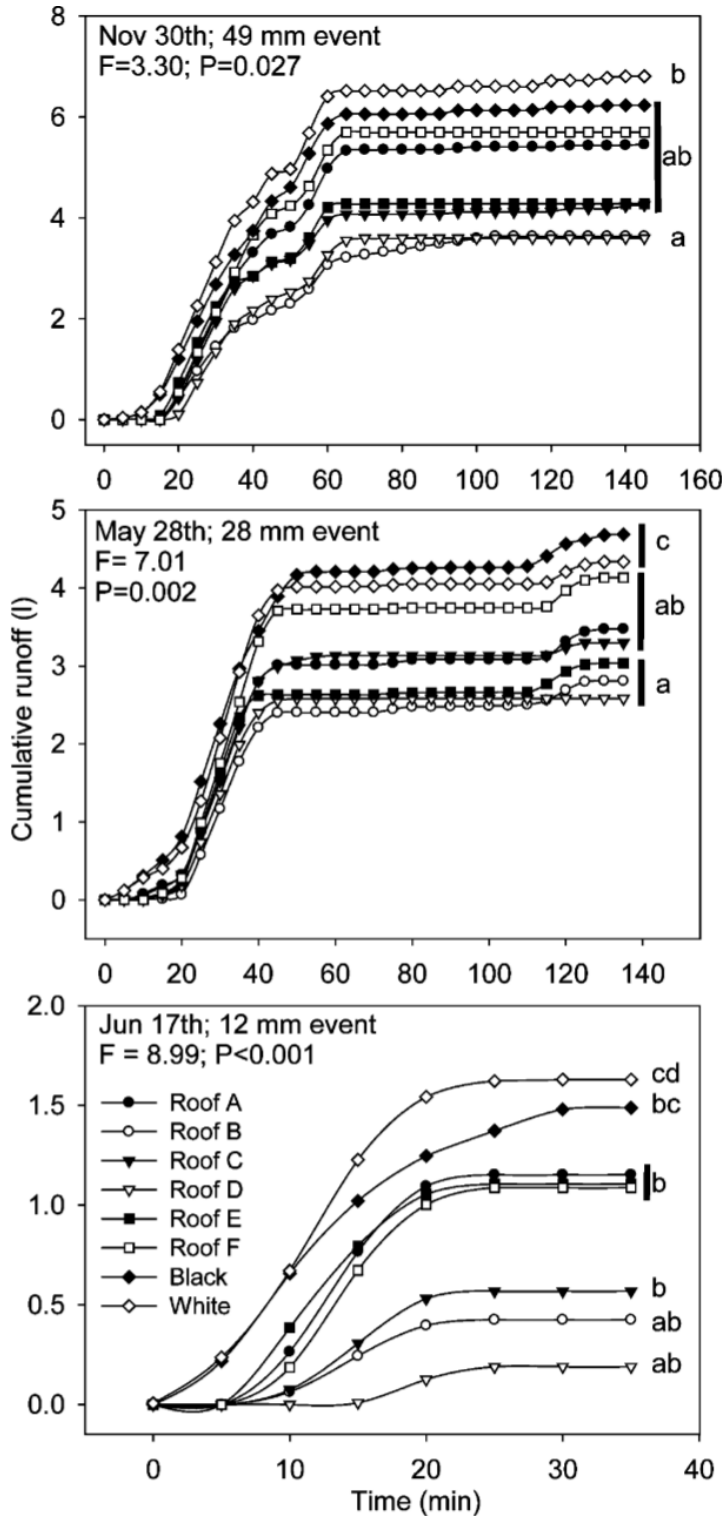


Figure 19: Variation in green roof runoff reduction performance caused by differences in roof type, substrate material, and structural components for light, medium, and heavy rain events in subtropical Austin, TX. Different letters show significant differences (repeated measures ANOVA, p<0.05) (Simmons *et al.* 2008).

As in temperate climates, stormwater quality improvement is relatively limited and varies significantly from one green roof to another in tropical climates. Fertilizer and substrate composition are frequently faulted as sources of contaminants in green roof effluent, particularly in the case of phosphates, potassium, most metals, and nitrogen (Chen 2013; Kok *et al.* 2016; Lim and Lu 2016; Ferrans *et al.* 2018). Chen (2013) reported that sediment and nutrient levels were 10 times higher for green roof runoff than conventional roof runoff in Taiwanese cities, which they attributed partly to poor maintenance, cleaning, and weeding efforts. Furthermore, concentrations of pollutants might decrease over time as plants grow and take up more nutrients (Kok *et al.* 2016).

Despite low pollutant removal efficiencies for tropical green roofs in most studies, some papers report that vegetation removes large quantities. In Ayub *et al.* (2015), measurement of effluent revealed variation between a maximum 80% and 89% reduction for TP and between a 93% and 95% reduction for ammioniacal nitrogen (AN) depending on which species was present within the system (Figure 25). However, TP and K still leached in great amounts from the system throughout most of the study period, which the researchers attributed to fertilizer application (Ayub *et al.* 2015). Van *et al.* (2015) arrived at a similar conclusion and found that in a wetland green roof system, TP and TN removal varied by plant. Vo *et al.* (2018) examined the growth and treatment by 9 plant species in a shallow wetland green roof system and found that the amount of biomass produced correlated with the amount of TP and TN removed from runoff. While the pollutant removal capacity of tropical green roofs is highly variable, they appear to have a positive effect on increasing the pH of runoff (Kok *et al.* 2015; Ferrans *et al.* 2018), although Sultana *et al.* (2015) found the opposite to be true in humid tropical Malaysia. This has significance because it shows the potential of green roofs to mitigate acid rain effects in tropical countries by acting as a buffer for contaminated stormwater (Kok *et al.* 2015).

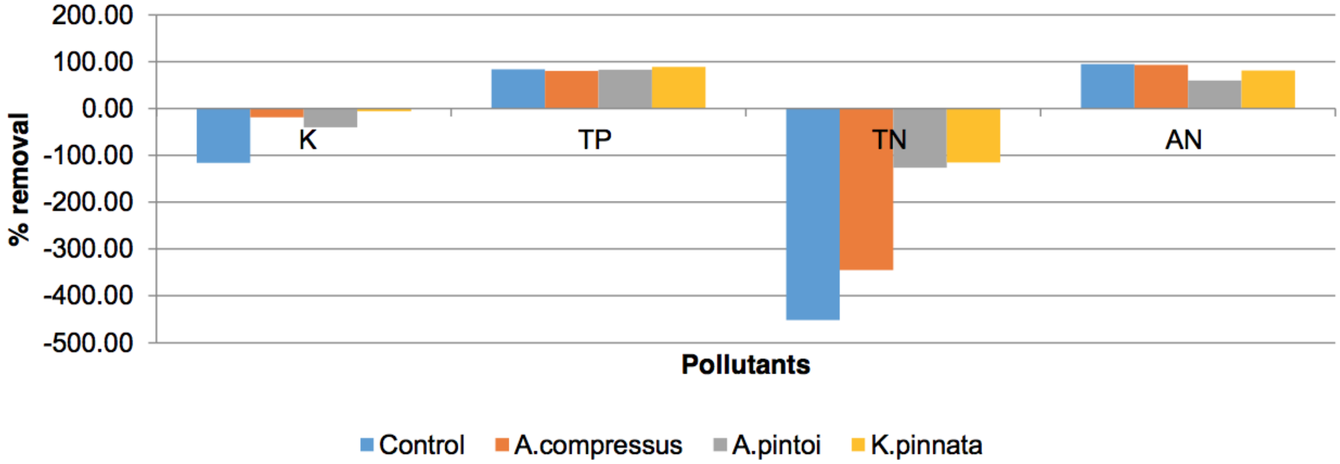


Figure 20: Removal and addition of different pollutants to green roof systems by different plant species including *Kalanchoe pinnata*, *Axonopus compressus*, and *Arachis pintoi* in tropical Malaysia (Ayub *et al.* 2015).

Few studies evaluate the performance of green roofs in cold weather climates, particularly in terms of pollutant removal potential. However, a couple studies have performed some basic analysis of retention capacity and vegetation survival in cold weather green roof systems, and they find that retention capacity generally decreases in colder weather. Elliott *et al.* (2016) looked at the effect of seasonality on retention performance using 4 years of data collected from a New York City green roof. Retention was highest in warm months, as green roofs retained more rain due to longer antecedent dry weather periods (ADWPs) and the accompanying increase in ET that allows for higher storage prior to the next storm (Elliott *et al.* 2016). Additionally, seasonal effects were more noticeable for thinner green roof substrates, which suggests that locations experiencing colder weather should pursue deeper substrate implementation (Elliott *et al.* 2016).

Johannessen *et al.* (2017) evaluated the effects of storage capacity and ET on retention across different climates in Northern Europe. The highest retention "in absolute values" was in the wettest climates, while highest retention in terms of the "percentage of annual precipitation" was in the climates that were the warmest and driest overall (Johannessen *et al.* 2017). This study's results agreed with the findings of Elliott *et al.* (2016) that highest retention occurs in the summer, but other characteristics varied according to each system's climate (Johannessen *et al.* 2017). Cold and dry climate green roofs had storage capacities of only about 25 mm compared to the 40-50 mm capacities of warm and dry climate alternatives (Johannessen *et al.* 2017). Furthermore, cold and wet locations would not necessarily achieve better volume reduction by increasing storage capacity because the ET would have to increase significantly first to have any effect (Johannessen *et al.* 2017). Therefore, the lowest absolute retention was found in the most northern and coldest locations; maximum retention was about 17% for the coldest location (Tromsø) compared to 58% for the driest (Malmö) (Johannessen *et al.* 2017). Furthermore, although almost no seasonal variation in retention was observed for different locations, the highest "magnitude of seasonal variation" was observed for cold locations (Johannessen *et al.* 2017). Ultimately, this study's conclusions suggest that green roofs are less effective in cold weather locations compared to places with alternate climate types (Johannessen *et al.* 2017). However, local site factors such as variations in direction of orientation, slope, altitude, and wind exposure might also alter the performance of cold weather systems, and thus should be taken into account when conducting future studies (Johannessen *et al.* 2018).

In addition to surveying the retention capacity of green roofs in cold climates, several studies analyze the health and potential establishment of different plant species in cold climate regions. Whittinghill and Rowe (2011) studied the varying levels of salt tolerance of different green roofs species, including 5 *Sedum* species, 2 *Allium* species, and a mixture of turf grasses, in response to 6 different concentrations of salinity. They measured survival, growth, and health, and they found that 3 plants were "highly salt tolerant" (*Allium cernuum*, *Allium senscens*, and *Sedum ellecombianum*), while 2 were intolerant (*Sedum reflexum* and turf grass) (Whittinghill and Rowe 2011). Typically, volume indexes decreased as salt concentrations increased, and effects of soil inundation and saline spray on the plants included chlorosis, reduced health, and smaller leaves. Therefore, salt tolerant species should be chosen and planted in areas with salt exposure, which is a potential danger to plant growth in areas utilizing deicing materials (Whittinghill and Rowe 2011).

Price *et al.* (2011) evaluated the effect of irrigation on seasonal health for different species in Alabama and determined that pre-winter irrigation had no effect on overwintering success. Lower temperature also reduced survival of some plants, particularly drought-tolerant ice plants, as the "shallow soil depth and the high exposure of the green roof environment" make it difficult to rely upon hardiness zone classifications to choose suitable plants (Price *et al.* 2011). In addition to irrigation having no effect on overwinter survival, Clark and Zheng (2012) determined that fertilizer has no effect on *Sedum* survival over the winter, as well. Therefore, *Sedum* species are resilient and suitable for fall green roof installation in northern climates irrespective of which fertilizer type is used.

3. Bioretention runoff volume reduction utility in semi-arid and arid climates

In arid and semi-arid climates, the success of bioretention systems at reducing stormwater runoff volume depends largely upon the storm characteristics, as well as specific design and maintenance features. This explains the frequently large variation in efficacy on a per-system basis. The primary function of bioretention systems in these climates is to reduce peak flow of stormwater runoff (Liu *et al.* 2014), but some limitations exist to their performance based on several key parameters. Amongst the key parameters, the characteristics of rainfall events, such as rainfall depth, intensity, and frequency, most often affect performance. Design variations including vegetation type, soil media qualities, and extra layers also affect performance.

3.1 General retention performance of bioretention in semi-arid and arid climates

In a study conducted in Tianjin, China, Huang *et al.* (2014) compared the performances of 5 different LID technologies in reducing runoff. Tianjin City has an average annual precipitation of 550 mm, but 58.6% of this rain falls within a period of one month, making flood mitigation of paramount concern (Huang *et al.* 2014). Compared to other LID technologies in Tianjin City, bioretention provides the best peak flow reduction, at a rate of 41.7% compared to 28.7% of porous pavement, as well as the best peak delay (Huang *et al.* 2014). However, bioretention reduced the average runoff the least of all, with only a 9.1% average reduction overall compared to the other LID technologies shown in Table 6 (Huang *et al.* 2014). Jiang *et al.* (2015) evaluated runoff volume reduction performance of a bioretention basin in Lakewood, CO over the course of 3 years and found that almost 53% reduction could be achieved. Such a range of values shows the importance of conducting multi-year studies, as well as the impact that study-specific factors such as vegetation palette, rainfall characteristics, and dimensions of the system can have on overall runoff volume attenuation.

Hydrological variables	Bio-retention	Grass swale	Infiltration trench	Porous pavement	Cistern system
Average runoff (mm)	3.08	3.07	2.35	2.19	2.96
Change of total runoff (%)	9.10	9.60	30.80	35.58	12.85
Peak flow (0.01 $m^3/(s \cdot ha)$)	0.19	0.25	0.26	0.23	0.31
Change of peak flow (%)	41.65	23.56	19.44	28.66	13.00
Change of lag time (min)	21	18	12	7	10
Ratio of runoff to precipitation	0.33	0.33	0.25	0.23	0.31
Rainfall captured by LID on site (mm)	0.31	0.33	1.05	1.21	0.44
Iw	0.017	0.020	0.027	0.052	0.221

Table 6: Comparison of bioretention system impact on hydrological variables in semi-arid Tianjin City, China compared to other LID technologies for events <25.4mm (Huang *et al.* 2014).

3.2 Performance and survival of bioretention vegetation

While several studies investigate the survivorship of specific plant species in bioretention systems of arid and semi-arid climates, few observe the impact of vegetation type on runoff reduction performance. One study that notes significant differences on the basis of plant species is the 2011 Li *et al.* study, which investigated the performance of 5 different bioretention plant species in reducing runoff in semi-arid Bryan, Texas. With each bioretention test box containing either shrub species, common highway grass species, native grasses, Bermuda grass, or bare soil (Table 7), they found that peak flow reduction could be achieved within a range of 14.4% to 74.8%, which they credited to both soil media and vegetation selection within the system (Li *et al.* 2011). Significant variation can be observed based on the presence and type of vegetation within each cell. Boxes with vegetation generally reduced peak flows temporarily, but control boxes without vegetation actually had the highest peak flow reduction rate. This resulted in less outflow for the control cell, particularly in the first hour, compared to vegetated bioretention cells. Furthermore, ponding occurred immediately in the control box, while preferential pore paths created by plant roots enhanced the infiltration of rain in the vegetated boxes and delayed the ponding response until the second hour of rainfall (Li *et al.* 2011). Figure 21 shows the increase in outflow caused by the inclusion of various plant species in the system compared to the control, which remained non-vegetated (Li *et al.* 2011). The ponding space of a bioretention system is an important feature because it stores water temporarily to decrease runoff, allows opportunity for water to evaporate, and allows for particulates to settle (EPA 1999). As a result, rainfall with ponding area limited or deviated out of the bioretention cell by vegetation and underdrain reduces the runoff reduction performance of the system as a whole, particularly in terms of peak flow reduction.

Box	Species: Botanical Name	Species: Common Name	Planting/Seeding Rates (kg/ha)
Shrub	*Ilex vomitoria*	Stroke dwarf yaupon holly	3 counts
	Morella cerifera	Wax myrtle	3 counts
	Leucophyllum frutescens	Texas sage (Cenizo)	3 counts
Texas DOT seed mix for sandy soil (Bryan District)	*Cynodon dactylon*	Bermuda grass	1.7
	Eragrostis curvula	Weeping lovegrass (Ermello)	0.7
	Eragrostis trichodes	Sand lovegrass	0.7
	Leptochloa dubia	Green sprangletop	0.3
	Paspalum notatum	Bahiagrass (Pensacola)	8.4
	Coreopsis lanceolata	Lance leaf Coreopsis	1.1
Native grass seed mix	*Bouteloua curtipendula*	Sideoats grama	11.2
	Leptochloa dubia	Green sprangletop	5.6
	Schizachyrium scoparium	Little bluestem	5.6
	Eragrostis trichodes	Sand lovegrass	5.6
	Desmanthus illinoensis	Illinois bundleflower	7.9
	Chamaecrista fasciculata	Partridge pea	5.6
Bermuda grass	*Cynodon dactylon*	Bermuda grass	18.6
Control	No vegetation		—

Table 7: List of plant species planted within each bioretention box in the Li *et al.* 2011 study modelling the effects on bioretention in treating and reducing highway runoff in semi-arid Bryan, TX (Li *et al.* 2011).

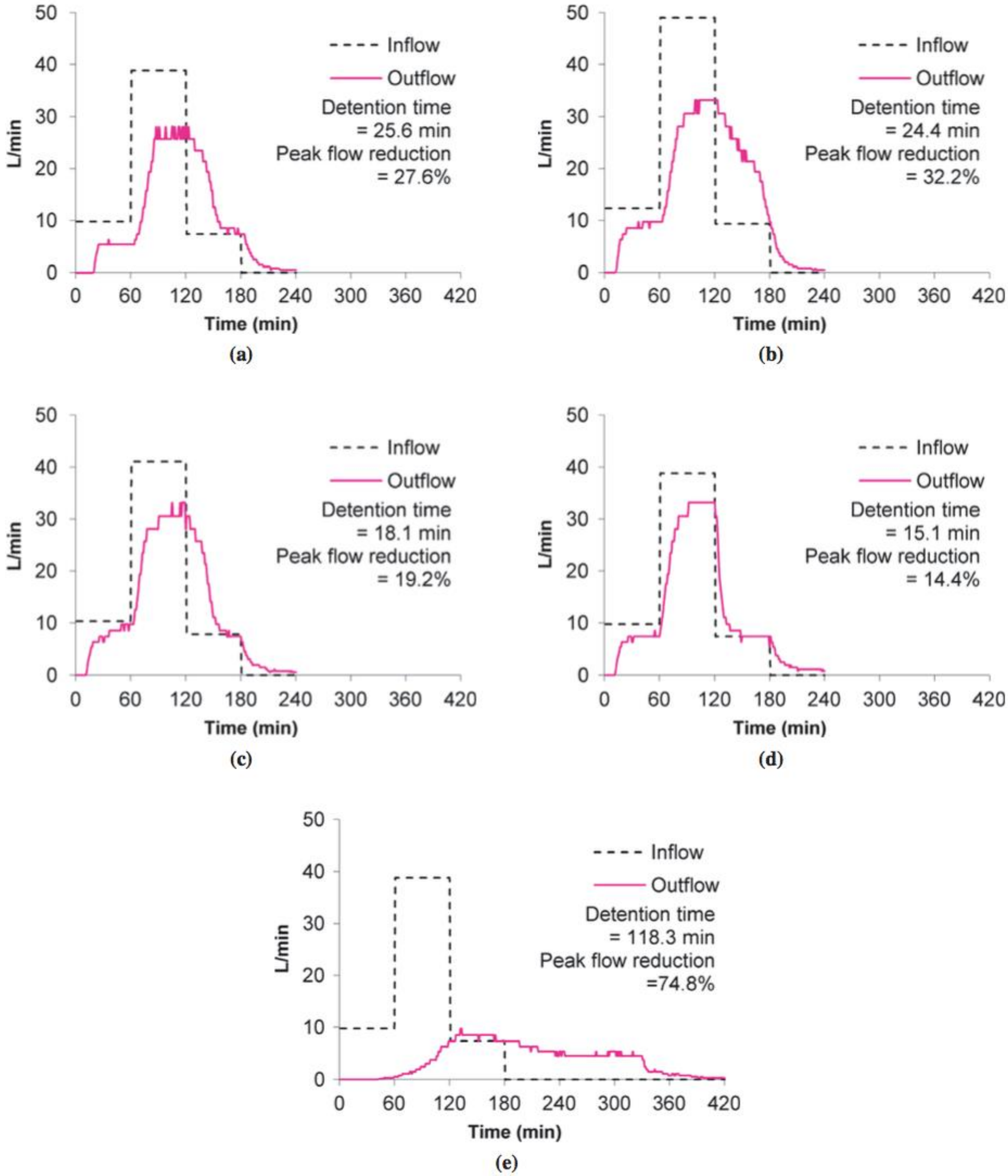


Figure 21: Effects of different vegetation palettes on the runoff characteristics of bioretention boxes compared to systems without vegetation in semi-arid Bryan, TX. Each box contains one vegetation type: a) shrub, b) Texas DOT sandy soil seed mix, c) native grass seed mix, d) Bermuda grass, and e) control (no vegetation) (Li *et al.* 2011).

Most studies examine the potential of different plant species to survive in the frequently extreme conditions of semi-arid and arid regions. Houdeshel (2012) highlights the importance of plant selection in water-limited regions because plants must be capable of standing both stress from both the soil and an atmosphere lacking much water. Guatam *et al.* (2010) calls for revision of design guidelines, including plant species and soil characteristics, for semi-arid conditions, as few field-based studies exist thus far to determine the extent for needed modifications. Houdeshel and Pomeroy (2013) sought to identify the most efficient decentralized irrigation system in the semi-arid climates of the western U.S. In order to do so, they monitored soil water potential and the survivorship of different species in order to determine the need and efficacy of irrigation efforts. Ironically, the minimum soil water potential fell only to -2.56 mPa after the longest dry period of 45 days (Figure 22) (Houdeshel and Pomeroy 2013). As this value is above the level causing water stress to these species (-5 mPa), almost all plants survived this dry period without any need for additional irrigation (Houdeshel and Pomeroy 2013). These results are promising in terms of signifying that the plants likely have the ability to survive long droughts. Many of the chosen species had adaptive physical mechanisms, such as deep roots, "seasonal dormancy," and stomatal manipulation to prevent water loss, making them particularly suitable to these climates (Houdeshel and Pomeroy 2013). The study concluded that bioretention systems with drainage areas 16 times the area of the bioretention basin area itself should be able to harvest and store water appropriate to support plants through an entire summer without additional irrigation (Houdeshel and Pomeroy 2013).

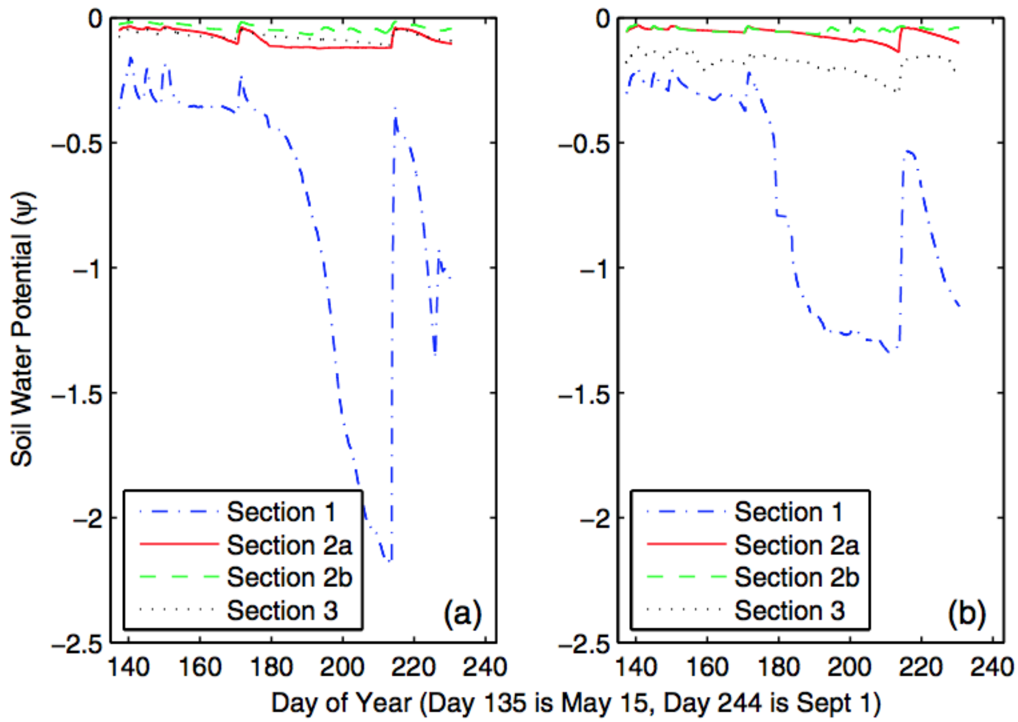


Figure 22: Soil water potential of plants at 25 cm (a) depth and 50 cm (b) depth over a 4-month period including a 45-day dry period in Salt Lake City, UT (Houdeshel and Pomeroy 2013).

Ambrose and Winfrey (2015) compared bioretention systems in California to those in Australia with emphasis on key differences. Southern California has a Mediterranean climate, while Melbourne, Australia has a temperate climate; as a result, southern California receives much longer dry periods and more sporadic distribution of rainfall from one year to the next. Comparing 13 bioretention basins in each location, they noted that the presence of a "submerged zone" could increase the survival of plants by allowing roots to harvest water for longer (Ambrose and Winfrey 2015). However, they also warned that additional irrigation would probably be required to support this layer throughout the drought periods. The report also recommended that future research investigate the potential use and suitability of the more than 800 endemic species in California bioretention systems (Ambrose and Winfrey 2015).

Despite the success of some studies in maintaining vegetation cover, finding plants capable of surviving both extensive dry and inundated periods can be a challenge. Although they used frequent irrigation, Li *et al.* (2011) found that only one specific desert species, Texas sage, flourished, and they emphasized that plants must be able to withstand both standing water and dry conditions. Wetland species in particular were incapable of surviving in bioretention cells situated in Bryan, TX due to low soil moisture and long droughts (Li *et al.* 2011). Houdeshel *et al.* (2012) recommends that bunchgrasses be planted along with native evergreens and shrubs in xeric, or "very dry," climates. Grasses provide soil stability and enhance infiltration of water into the soil, while native shrubs feature long roots that can penetrate deeper water reserves in the soil and facilitate hydraulic lift to make deep water accessible to shallower grass roots (Houdeshel *et al.* 2012). Native plants are encouraged due to lower water requirements and the ability to use annual water budget planning, rather than the monthly monitoring suggested for non-native plantings (EPA 2010). Application of xeriscaping principles, or landscaping principles that require little added water, for bioretention systems in arid and semi-arid locations should be investigated further, particularly due to their low irrigation requirements. For example, Sovocool *et al.* (2006) found that xeriscaping in arid Las Vegas Valley, Nevada rather than using turfgrass in yards could lower the winter to summer "peak water demand ratios" by 48% (Sovocool *et al.* 2006). Although native plants should be capable of surviving in the long-term without additional irrigation, Houdeshel *et al.* (2012) recommends irrigation each week until roots become well-developed.

3.3 Impact of rainfall event characteristics

Bioretention performance varies immediately according to the quantity of rain falling within a certain span of time. Both diurnal and monthly rainfall characteristics influence the performance of bioretention systems, as well as air temperature. Lizárraga-Mendiola *et al.* (2017) compared the monthly runoff reduction efficacy of a combined infiltration trench and bioretention cell in one dry year (1982) and one wet year (2010) in semi-arid Central Mexico. They found that the design infiltrated 5.37% of runoff volume maximum for the wettest month in 1982, but during the wettest month in 2010, the maximum runoff reduction dropped to 2.25% with the rest of the rainfall moving out of the system as outlet runoff (97.75%) (Lizárraga-Mendiola *et al.* 2017). Infiltration rates for the two years were not significantly different, but consumptive use by plants and evaporation were significantly higher in 2010 (Table 8). Therefore, vegetation and evaporation have the potential to affect the outflow volume of semi-arid bioretention systems variably in response to weather conditions. Regardless of rainy or dry spells, they conclude that semi-arid Mexico, as temperature increases and rainfall decreases, consumption by plants increases and potential evaporation (PE), the measure of water that would evaporate given ideal available water, stays the same (Liázarraga-Mendiola *et al.* 2017). In conclusion, multiple components of the system contribute to the degree of runoff reduction achieved throughout any given year. In terms of peak flow reduction, they observed that each bioretention cell intercepted water at a positive rate for the first 5-10 minutes of a storm, but that after 20 minutes, the rate of water infiltrated (m/s^2) became negative (Figure 23); at this point, the cell was saturated, and any remaining runoff was excess overflowing the system (Lizárraga-Mendiola *et al.* 2017). Therefore, fluctuations in rainfall on both annual and diurnal scales can significantly impact the runoff reduction capacity of an individual system, and design should be based on high rainfall years for a certain location. Lizárraga-Mendiola *et al.* (2017) conclude from their study that plants often require extra irrigation during periods without rain, but they can recover from drought stress when dry conditions are followed by rainy bouts.

Variable	1982				2010			
	Min.	Max.	Mean	Standard Deviation	Min.	Max.	Mean	Standard Deviation
Temperature (T) [°C]	2.33	21.32	14.00 *	7.11	18.85	24.16	21.74 *	1.57
Rainfall (R) [mm]	0.00	53.00	15.09	15.05	0.00	249.40	48.80	69.00
Evaporation (Ev) [mm]	25.60	143.70	85.54 *	34.29	76.75	174.93	116.41 *	30.16
Infiltration (Inf) [mm]	0.00	42.40	12.07	12.04	0.00	199.52	39.04	55.20
Consumptive use (U) [mm]	0.36	89.15	38.94 *	27.74	54.93	117.86	85.40 *	20.31
Potential evapotranspiration (PE_m) [mm]	2.76	4.51	3.64 *	0.63	25.80	41.56	33.47 *	5.28

Note: * Means significantly different in a t-Student test ($p < 0.05$).

Table 8: Differences in hydrological variables between years based on differences in temperature and rainfall for a dry year (1982) and a wet year (2010) in semi-arid Central Mexico (Lizárraga-Mendiola *et al.* 2017).

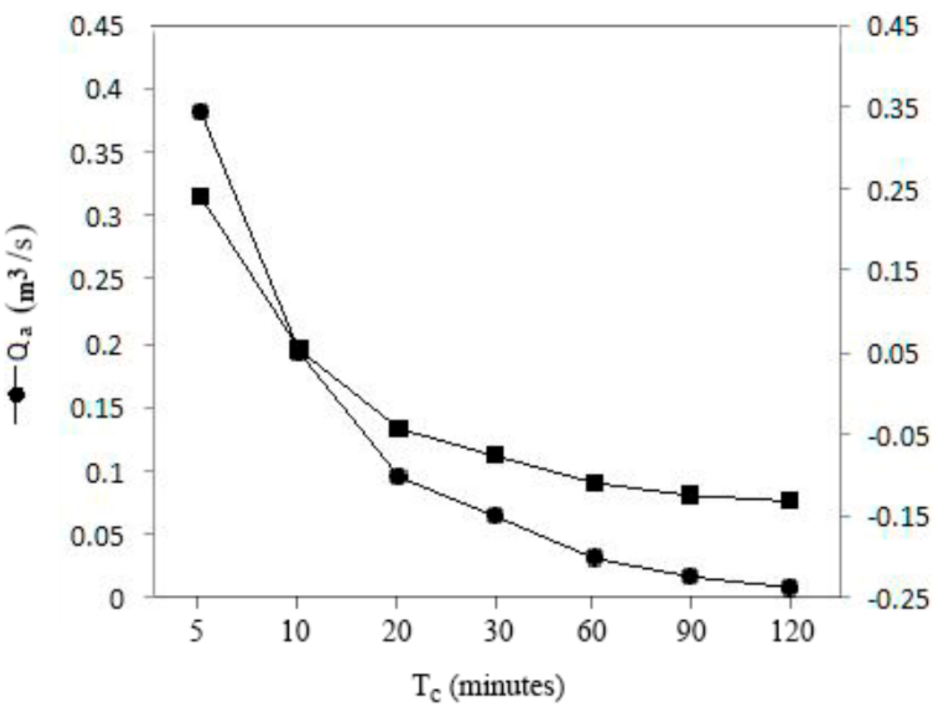


Figure 23: Reduction in runoff retention and subsequent overflow of system after first 20 minutes of storm event in semi-arid Central Mexico. Q_a is input runoff, Q_{out} is effluent runoff, and T_c is time of concentration (Lizárraga-Mendiola *et al.* 2017).

Interestingly, Feng *et al.* (2016) arrived at a different conclusion in their study of wet, dry, and normal years in the semi-arid urban catchment of Salt Lake City, UT. The study used the EPA's stormwater management model (SWMM) to compare the combined effect of bioretention and green roof systems in Salt Lake City to that of baseline and natural hydrology scenarios specific to the city. Figure 24 breaks down the water budgets of baseline, green infrastructure, and natural hydrology scenarios for each of the three years (Feng *et al.* 2016). The researchers noted that the annual surface runoff reduction for a wet year with green infrastructure was 43%, while that for a dry year was only 35% (Feng *et al.* 2016). The model results showed that surface discharge was 30% for the dry year, while it remained only 13% for the wet year (Figure 24) (Feng *et al.* 2016). Furthermore, evapotranspiration (ET), percolation, surface storage, and soil moisture were higher in the wet year than in the dry year, and the study notes that ET was an important part of the water balance equation for all scenarios modeled (Figure 24) (Feng *et al.* 2016). Feng *et al.* (2016) also mention that greater quantities of excess water stored in soil and in the storage space of green infrastructure can be used by plants in ET during the months of drought that inevitably follow; this explains the slightly higher ET observed in wet years, which contributes somewhat to higher runoff reduction compared to the dry year.

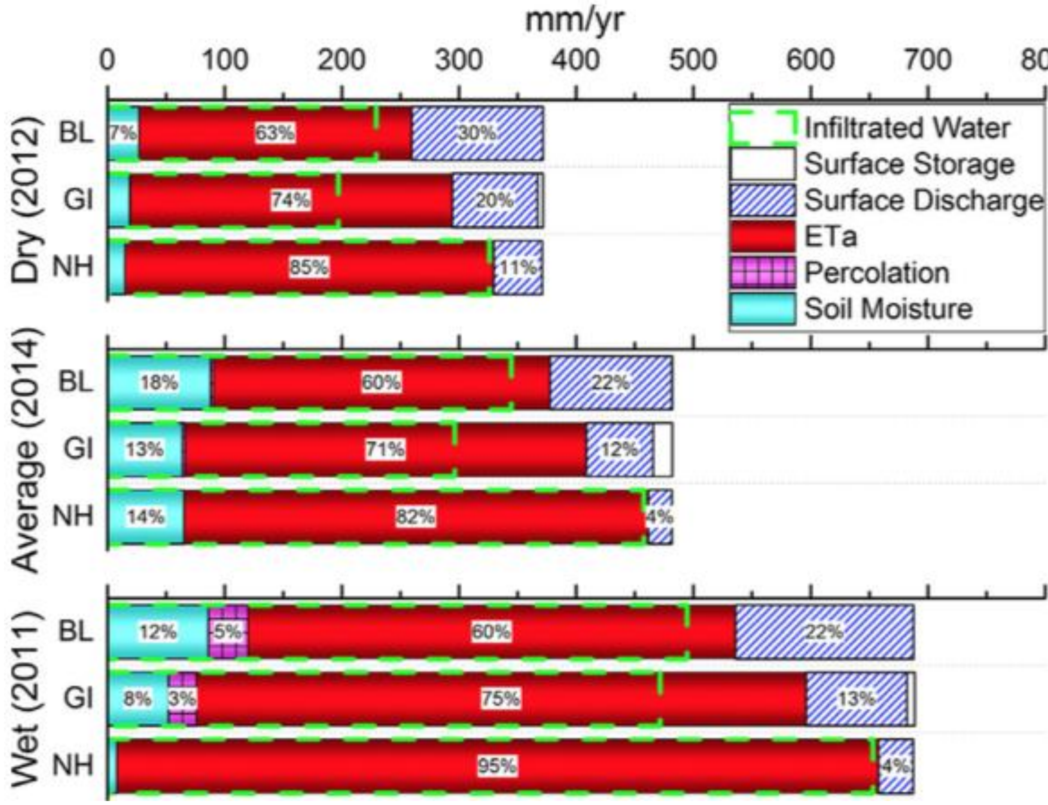


Figure 24: Breakdown of water budgets for baseline (BL), green infrastructure (GI), and natural hydrology (NH) scenarios modeled using SWMM for dry, average, and wet years in semi-arid Salt Lake City, UT (Feng *et al.* 2016).

3.4 Effect of additional design parameters and maintenance

Despite the general efficacy of bioretention systems with only the features typical to humid climates, several suggestions have been made to modify systems to function better in terms of runoff reduction in semi-arid and arid climates. Dussaillant *et al.* (2005) modeled water flow using another stormwater management model, RECARGA, over 3 bioretention layers in different world climates. Using hourly rainfall data for each city over multiple years, they found that in both humid and semi-arid climates, the ideal ratio of impervious groundcover to garden cover was 10-20%; however, for arid climates, this ratio dropped to 5% (Dussaillant *et al.* 2005).

This might be the case because precipitation is minimal, and there is almost no natural recharge, resulting in less runoff overall (Dussaillant *et al.* 2005).

The addition of an internal water storage (IWS) layer has also been suggested as a method to improve both runoff reduction and nutrient removal performance in semi-arid and arid bioretention systems. Also called a “submerged zone,” the IWS layer creates an anaerobic environment that promotes denitrification and also can reduce outflow events by performing as a “storage sump,” or basin to temporarily hold contaminated water to treat (North Carolina Cooperative Extension Service). Li *et al.* (2013) compared systems with and without IWS layers in Bryan, Texas and observed that while both designs caused a lower peak discharge and longer detention time of runoff, the IWS layer design did so to a greater extent. The synthetic tests showed similar peak discharge reduction for both non-IWS and IWS design, but hydraulic performance determined using natural rainfall showed consistently higher peak flow reduction and detention times for the IWS design (Li *et al.* 2013). Ambrose and Winfrey (2015) also introduced the benefit of using an IWS layer in southern Californian and Australian systems. They observed that this addition can both enhance runoff reduction and increase the survival of the plants by providing an additional available water resource. From a growth standpoint, the efficacy of this layer might be somewhat dependent on location, as the antecedent dry period per location might require extra irrigation water through drought periods to ensure proper IWS layer performance (Ambrose and Winfrey 2015).

3.5 Limitations to runoff volume reduction

Just as some conditions and features are optimal for bioretention function, there are some characteristics that studies show might limit bioretention performance, as well. Dry months demand additional irrigation, and thus the efficiency of the system is less in these months, even in years of above-average rainfall (Lizárraga-Mendiola *et al.* 2017). Furthermore, in dry years, additional irrigation might be necessary even throughout the rainy season when very dry conditions are periodically present due to the reduced retention of runoff overall during the year (Lizárraga-Mendiola *et al.* 2017). Another way to consider the negative effect of extended dry periods on bioretention runoff reduction performance is to consider the number of antecedent dry days (ADD) for bioretention cells in a specific location. Ambrose and Winfrey (2015) observe that bioretention in Los Angeles, California compared to that in Melbourne, Australia would be less effective because long periods of ADD in Los Angeles are 17 times more common than in Melbourne. Therefore, more regular rainfall in Melbourne will result in better bioretention performance than in Los Angeles (Ambrose and Winfrey 2015). A final factor limiting bioretention performance can be long or particularly heavy singular rainfall events. As mentioned previously, Lizárraga-Mendiola *et al.* (2017) found that after 20 minutes, the runoff reduction value became negative and the remaining water bypassed the system altogether.

4. Bioretention pollutant removal utility in semi-arid and arid climates

While many LID technologies reduce runoff volume, few achieve the same degree of pollutant removal as bioretention systems. Despite the strong performance of bioretention in other climates, lesser amounts of rainfall in semi-arid and arid locations may cause higher pollutant concentrations to build up over time, resulting in higher than average concentrations in the “first flush” of stormwater runoff (Guatam *et al.* 2010). Therefore, bioretention must be evaluated separately from bioretention systems in humid temperate areas due to higher levels of contaminants present in runoff.

4.1 Nutrient removal

Investigations of nutrient removal in semi-arid and arid bioretention systems primarily evaluate the effects of variations in vegetation treatment on removal efficiencies. Great variation exists in the pollutant removal capacities of semi-arid and arid bioretention systems, but overall, all systems appear capable of removing a substantial amount of pollutants from stormwater runoff. As with runoff volume reduction, two obvious designs to compare are non-vegetated control cell and vegetated test cells. Li *et al.* (2011) observed that although all bioretention cells in their Bryan, Texas study retained some pollutants, the non-vegetated control cell removed almost double the amount of total suspended solids (TSS) that the vegetated cells retained on average (84% and 26%, respectively). In terms of TN and TP, however, negative percentages indicate that the systems added more contaminants to the system than they removed (Figure 25) (Li *et al.* 2011). The control cell contributed a greater percentage of TN (434% increase in TN) than each of the vegetated cells (431% to 20% increase), which the study attributes to the vegetation's ability to take up TN to a greater extent and lower the amount leaving the system through leaching (Li *et al.* 2011). However, vegetation also caused higher P concentrations in runoff outflow due to leaching and reduced runoff detention time within the vegetated boxes relative to the non-vegetated control (Li *et al.* 2011). Jiang *et al.* (2015) arrived at similar conclusions and determined that a bioretention cell in Lakewood, CO plants including dry weather vegetation and a seed mixture removed 91% of TSS, but it failed to significantly improve stormwater quality in terms of TN and TP. They explained this shortcoming as resulting from nutrients in the soil media leaching in runoff, thus augmenting rather than reducing the pollutant concentrations in effluent (Jiang *et al.* 2015).

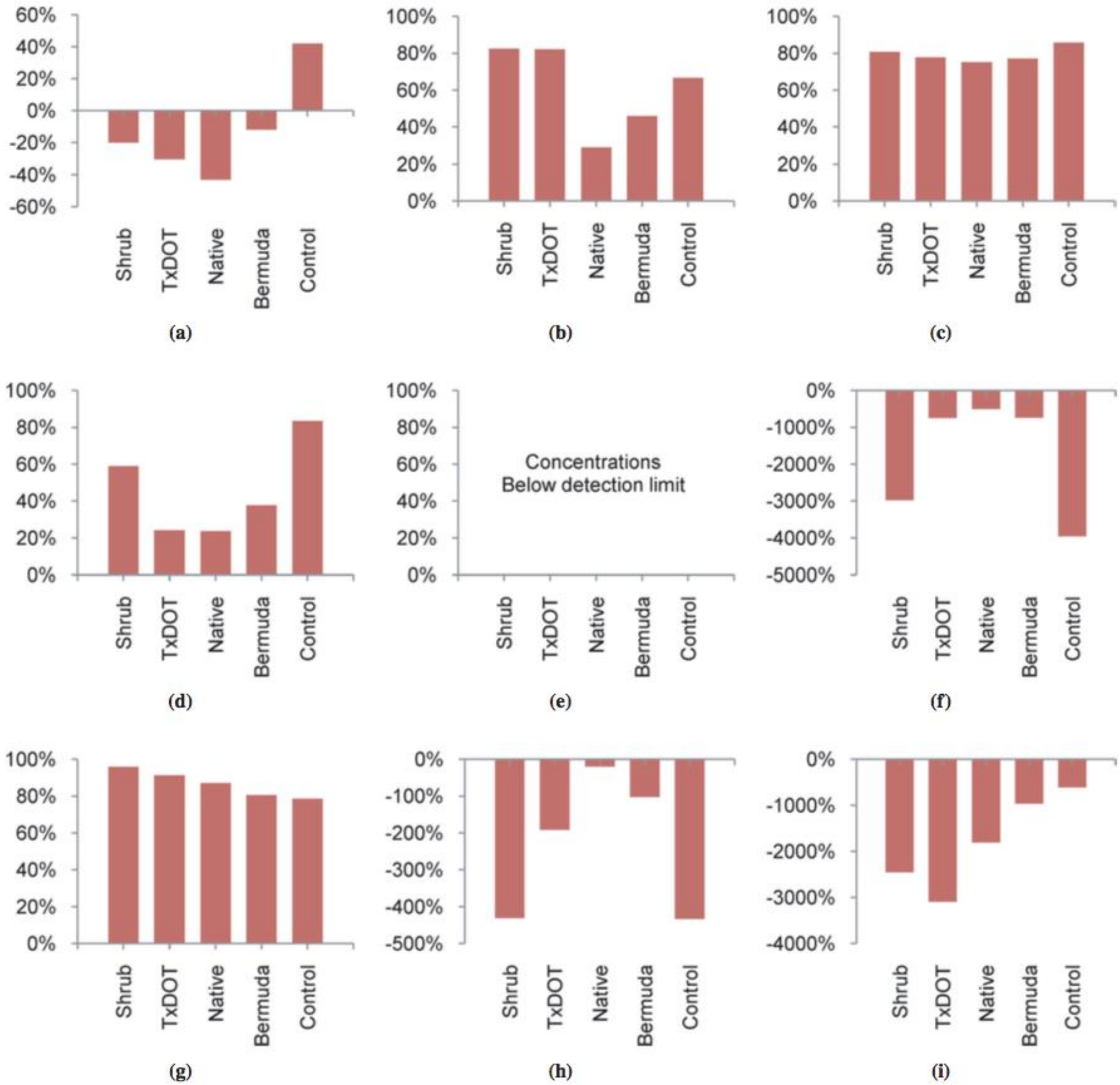


Figure 25: Pollutant removal capabilities of bioretention boxes with different planting schemes for the following pollutants: a) Cu, b) Zn, c) Pb, d) TSS e) $N0_2$-N, f) NO_3-N, g) NH_3-N, h) TN, and i) TP (Li *et al.* 2011).

Houdeshel *et al.* (2015) arrived at different conclusions and found that although a non-vegetated cell allowed for greater runoff reduction in the Salt Lake City study, it also resulted in the least nutrient removal capacity compared to irrigated wetland plant and unirrigated upland plant cells. The study featured 3 bioretention cells planted with either no vegetation, an upland native plant community, or a wetland community that needed additional watering, all of which were exposed to an average of less than 400 mm of rain annually. Table 9 lists the species chosen for both cell types (Houdeshel *et al.* 2015). Water was applied synthetically with timing and quantity simulating a typical year of storm patterns in Salt Lake City, and effluent was analyzed to determine nutrient contents and retention over time. Although phosphate removal rates were more than 50% for both vegetated and non-vegetated cells, the vegetated cells retained slightly more than non-vegetated, and only the vegetated cells retained TN (Houdeshel *et al.* 2015). The type of vegetation chosen for a particular system is also important for nutrient retention. As depicted in Figure 26, cells vegetated with wetland plants were the only cells shown to reduce nitrate concentrations, as both unirrigated cells had no means by which denitrification could occur to reduce the NO_3 produced by nitrification (Houdeshel *et al.* 2015). Therefore, although the wetland vegetation cell required additional water supplies, it might be more suitable for areas with high nitrate pollution (Houdeshel *et al.* 2015).

	Species name	Common Name	Form
Upland cell	*Schizachyrium scoparium*	Little blue stem	Bunch grass
	Bouteloua gracilis	Buffalo grass	Bunch grass
	Sorghastrum nutans	Indian grass	Bunch grass
	Amelanchier utahensis	Utah serviceberry	Shrub
	Cercocarpus ledifolius	Curl-leaf mahogany	Shrub, evergreen
	Cercocarpus montanus	Mountain mahogany	Large shrub
	Artemisia cana	Silver sage	Shrub
Wetland cell	*Juncus effusus*	Common rush	Rush
	Dactylis glomerata	Oarchard grass	Bunch grass
	Typha sp.	Cattail	Bunch grass
	Phragmites sp.	Phragmites	Bunch grass
	Salix exigua	Coyote willow	Shrub, tree
	Medicago sativa	Alfalfa	Forb

Table 9: List of plant species installed within upland and wetland cell types in Salt Lake City, UT study (Houdeshel *et al.* 2015).

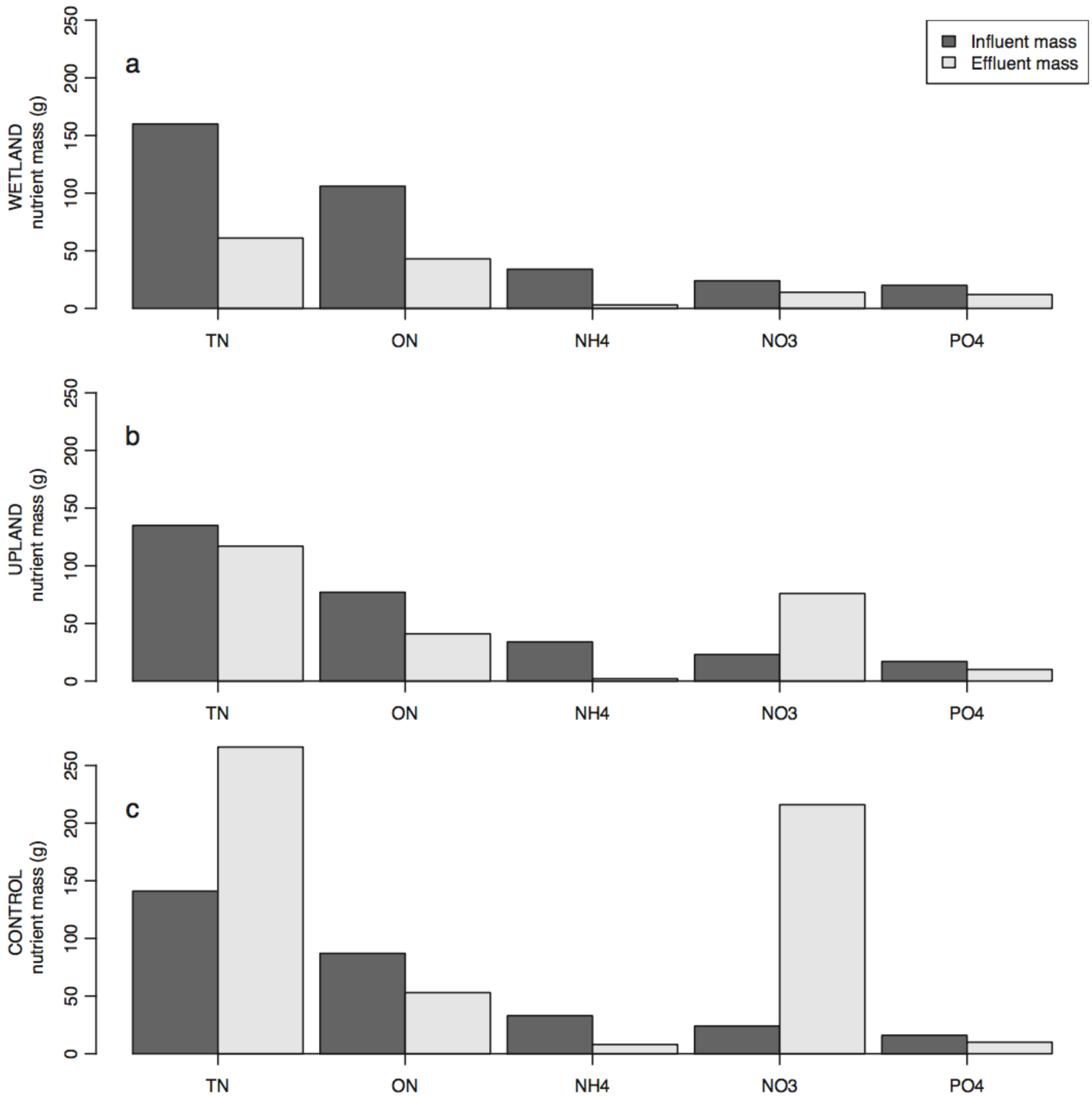


Figure 26: Comparison of nutrient masses in influent and effluent of wetland, upland, and control bioretention cells in Salt Lake City, UT. Note that NO_3 mass decreased only in the wetland-vegetated cell because this was the only cell with a mechanism for denitrification to occur (Houdeshel *et al.* 2015).

4.2 Bacteria removal

Bioretention cells have potential to remove bacteria and pathogens carried in runoff. Kim *et al.* (2012) studied the efficacy of *E. coli* removal for 5 different vegetated bioretention systems in semi-arid Texas and found that the highest removal occurred in the control cell (97%), followed by the shrub cell (88%). Although they admitted that the removal mechanisms for *E. coli* require more research before arriving at a definitive answer, they noted a correlation between longer hydraulic retention time and higher *E. coli* removal. This supports their conclusion that straining and adsorption are the “two major removal mechanisms” for *E. coli* (Kim *et al.* 2012). Furthermore, the type of vegetation included within each system also has an effect, as porosity of substrate, root paths, and “rhizosphere environments” all vary with the type of plants used (Kim *et al.* 2012).

4.3 Metal removal

Generally, bioretention systems in semi-arid and arid climates have high metal removal efficiencies. Both Li *et al.* (2011) and Li *et al.* (2013) determined that bioretention systems removed Zn and Pb at high rates, and in the case of the non-vegetated control, Cu was removed, as well. Li *et al.* (2011) also remarked that Cu removal might be low in this study due to lower concentrations in runoff than usual compared to Zn and Pb concentrations. However, Jiang *et al.* (2015) determined that although bioretention in Lakewood, CO removed some metals, including Pb, at a comparable rate to other studies, other metals shown in Table 10, such as Cu, actually increased as measured by mean effluent.

Davis *et al.* (2003) observed that copper, lead, and zinc removal are generally very high in bioretention systems regardless of rainfall duration, runoff acidity, rainfall intensity, and overall removal. However, shallow bioretention systems of under 30 cm in depth frequently showed decreased metal removal rates, and a mulch layer was identified as a key component of bioretention metal removal performance (Davis *et al.* 2003).

Constituents	Mean Influent	Mean Effluent	Event-Based Concentration Reduction Rate (%)				
			5th *	Median	Mean	95th *	5th *
TSS (mg/L)	264.3	51.3	−5%	91%	63%	98%	−183%
$NO_3 + NO_2$ (mg/L)	0.7	2.1	−1327%	−128%	−559%	−5%	−752%
TKN (mg/L)	3.1	2.6	−363%	18%	−31%	70%	−225%
NH_3-N (mg/L)	0.7	0.0	−378%	96%	−28%	100%	−31%
TP (mg/L)	0.4	0.7	−1947%	−133%	−494%	76%	−1960%
Ortho-P (mg/L)	0.2	0.4	−844%	−252%	−269%	−99%	−443%
Diss. P (mg/L)	0.5	1.1	−1357%	−196%	−358%	49%	−842%
Tot. sol. P (mg/L)	0.1	0.4	−560%	−350%	−317%	−104%	−655%
Tot. Cu (μg/L)	16.6	23.4	−393%	−12%	−73%	63%	−338%
Tot. Pb (μg/L)	8.1	5.0	−503%	98%	−30%	100%	−360%
Tot. As (μg/L)	3.3	4.4	−139%	−100%	−112%	100%	−755%
Tot. Be (μg/L)	0.0	0.1	−100%	−100%	−33%	80%	−265%
Tot. Cd (μg/L)	0.1	0.2	−407%	−100%	−107%	100%	−177%
Tot. Cr (μg/L)	2.9	1.4	−170%	100%	32%	100%	100%
Tot. Sb (μg/L)	0.4	0.5	−100%	32%	26%	100%	45%
Tot. Se (μg/L)	0.1	0.1	−100%	−100%	−68%	11%	−170%

Table 10: Nutrient and metal removal capacity of a bioretention cell in Lakewood, CO (Jiang *et al.* 2015).

5. Green roof runoff volume reduction utility in semi-arid and arid climates

Green roofs in semi-arid and arid climates successfully reduce stormwater runoff, specifically through the absorption of rainwater into soil media and use by plants. In these climate types, green roofs may have high stormwater runoff retention, but as with bioretention systems, the performance of each depends on vegetation type, rainfall characteristics, and substrate type, among other factors.

5.1 General retention performance of green roofs in semi-arid and arid climates

The efficacy of a semi-arid or arid green roof system at reducing stormwater runoff volume is actually greater than that of a humid green roof in many studies. Sims *et al.* (2016) set up duplicate green roofs in locations with three different climates, including semi-arid continental Calgary, Alberta, humid continental London, Ontario, and humid maritime Halifax, Nova Scotia. They then measured the cumulative retention, volumetric retention, and single-event retention over the course of about 2 years. They determined that compared to humid continental and humid maritime climates, semi-arid continental Calgary had the highest percentage of cumulative stormwater retention (67% compared to 48% for London and 24% for Halifax) (Sims *et al.* 2016). Individual events classified as “small” and “large” for each location did not show significant differences in retention (Table 11) (Sims *et al.* 2016). Researchers attributed the similarity between small events to ET rates high enough to restore storage space, and the similarity between large events to the inevitable drainage resulting from restricted storage space for high precipitation quantity (Sims *et al.* 2016). On the other hand, Table 11 also shows that different climates had significantly different retention for medium-sized events, as a low antecedent moisture content (AMC) determined retention ability and thus was lower in climates with higher ET and dry periods (Sims *et al.* 2016). However, the researchers also noted that because Calgary is drier than the other locations, the volumetric retention performance is actually lower; while ET rates were high and pore space was therefore clear, low precipitation levels meant that potential storage space was often underutilized. They also concluded that the differences in AMC had a greater distinguishing effect overall between locations than rainfall patterns (Sims *et al.* 2016). Therefore, manipulating the AMC to be lower before an event via careful substrate selection, vegetation selection, and roof orientation can improve the performance of the green roof, regardless of climate (Sims *et al.* 2016).

Events	Event size	Retention % (σ)
London, Ontario		
51	Small (<3 mm)	93.8 (14.8)
81	Medium (3–15 mm)	77.2 (29.9)
28	Large (>15 mm)	42.8 (35.2)
160	All events	76.5 (32.1)
Calgary, Alberta		
38	Small (<3 mm)	94.5 (11.2)
39	Medium (3–15 mm)	91.7 (16.7)
9	Large (>15 mm)	58.5 (37.0)
86	All events	89.6 (21.0)
Halifax, Nova Scotia		
32	Small (<3 mm)	89.6 (18.6)
36	Medium (3–15 mm)	52.2 (36.0)
30	Large (>15 mm)	36.4 (24.8)
98	All events	59. 6 (35.4)

Table 11: Average annual retention (%) for small, medium, and large storm events in London, Calgary, and Halifax. Only medium-sized events showed a significant difference in retention percentages (Sims *et al.* 2016).

Feng *et al.* (2016) studied the effect of combined green roof and bioretention implementation across semi-arid Salt Lake City using SWMM modeling to compare results to the baseline (urbanized) and natural hydrology scenarios over the course of 3 years. They found that green infrastructure had a greater impact in reducing runoff volume compared to humid climates and that partial restoration of the area's evapotranspiration (E_a) component was responsible for the bulk of infiltrated water retained by the system (Feng *et al.* 2016). Furthermore, the green infrastructure mimicked natural hydrology both temporally and in terms of the amount if retention achieved (Figure 27) (Feng *et al.* 2016). Alternatively, the baseline scenario consistently had the lowest retention volume and highest discharge due to less places for storage on the surface and in the soil and lower rates of ET_a, respectively (Figure 27). A shortcoming of this study might be that it assumed that underdrains for both green roofs and bioretention cells drained at an identical rate, which means the performance of each is presumed to be the same, when in reality this is not the case (Feng *et al.* 2016).

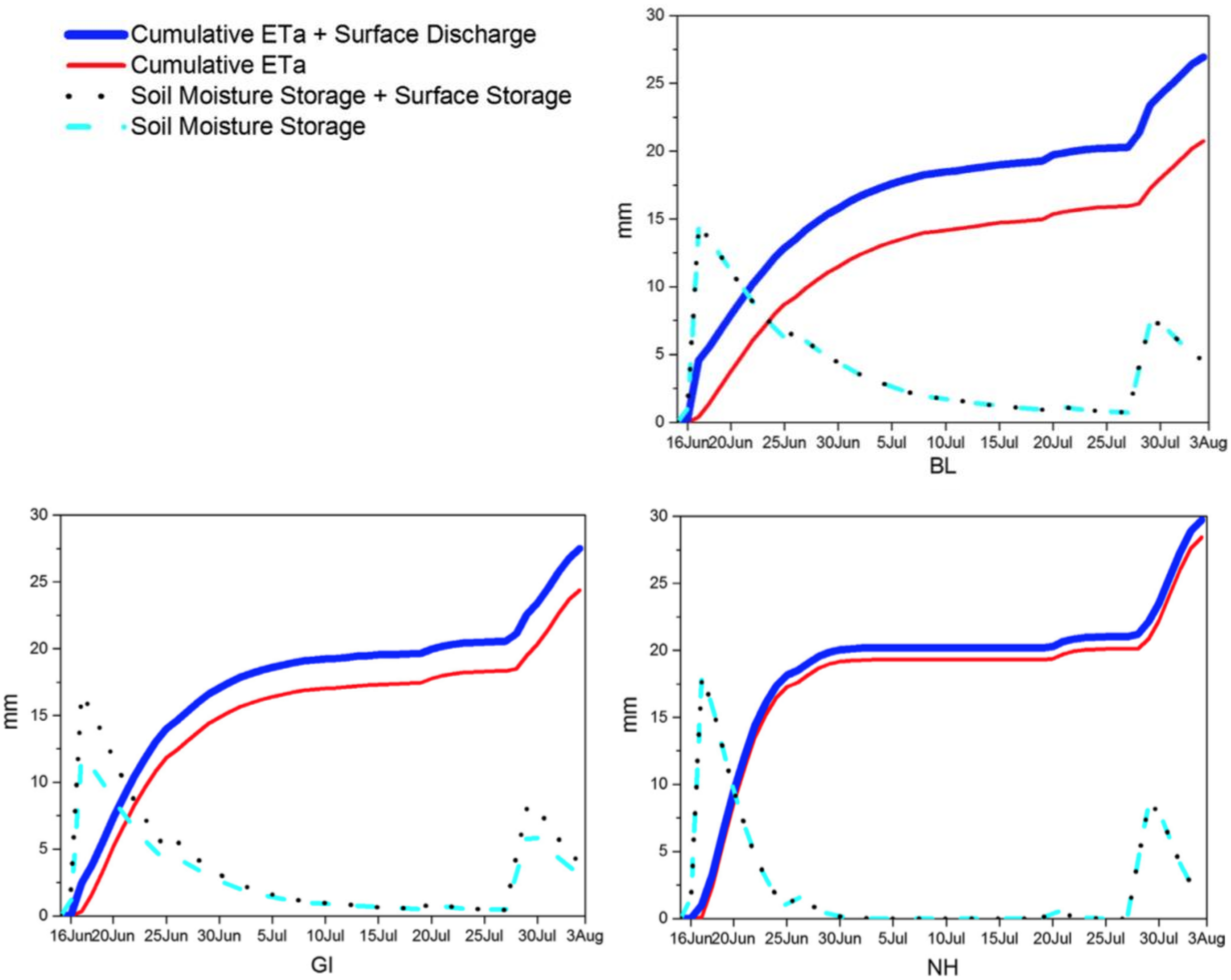


Figure 27: Differences in ET_a, soil moisture storage, and surface storage for baseline (BL), green infrastructure (GI), and natural hydrology (NH) scenarios for two typical storm events in Salt Lake City, UT (Feng *et al.* 2016).

Soulis *et al.* (2017) conducted a study observing the effect of varying rainfall depth, initial moisture content of the soil, and the design of each green roof system on the runoff reduction attained by each in Athens, Greece for one year. They made the observation that as rainfall depth and soil moisture increases, the runoff reduction capacity of the green roof system decreases (Figure 28) (Soulis *et al.* 2017). Given the sporadic nature of rain events in semi-arid and arid climates, rainfall depths and intensities per storm event can vary massively. Soulis *et al.* (2017) observed that depths varied between 0.6 mm and 45.4 mm during the length of their study, while intensities varied from 0.01 mm/min to 1.40 mm/min; this helps to explain the massive runoff retention capacity of the green roof systems ranging from 2% to 100%. The 2% runoff retention value was observed during the second largest rain event while 100% reduction was observed in multiple small rain events (Soulis *et al.* 2017). Overall, however, retention percentages could be achieved up to an average 81.1% for a green roof planted with *Origanum onites* and a 16-cm substrate depth, which shows the positive effect on retention that strategic design choices can have even in semi-arid locations (Table 12) (Soulis *et al.* 2017).

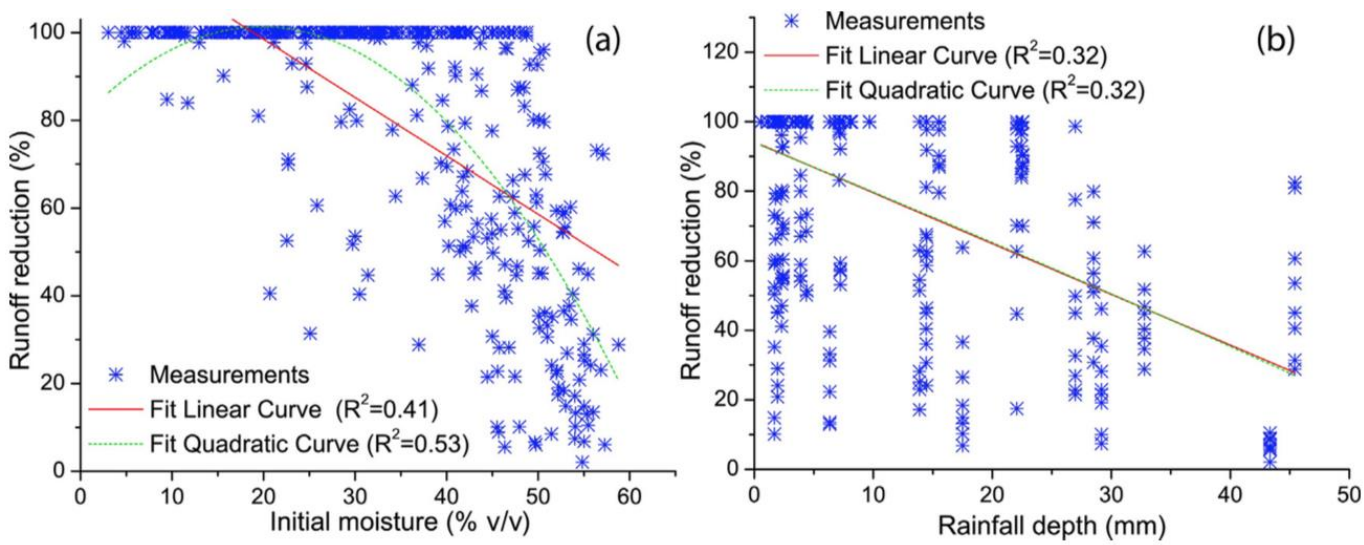


Figure 28: Fitted curves showing strong correlation between a) runoff reduction (%) and initial substrate moisture (% v/v) and b) runoff reduction (%) and rainfall depth (mm) over a one-year study in Athens, Greece (Soulis *et al.* 2017).

Parameters		Rainfall	Runoff							
			No veget. 16 cm	No veget. 8 cm	Origan. 16 cm	Origan. 8 cm	Sedum 16 cm	Sedum 8 cm	Festuca 16 cm	Festuca 8 cm
Total rainfall/runoff depth (mm) for the entire study period										
Max		45.4	42.4	40.7	38.8	39.5	40.7	40.9	39.7	40.4
Min		0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sum		465.6	210.6	230.1	88.2	169.6	185.1	228.9	145.1	210.0
Average		10.3	4.7	5.1	2.0	3.8	4.1	5.1	3.2	4.7
Total rainfall/runoff depth (mm) for each season										
Winter	Total depth	106.2	43.6	52.4	6.3	27.4	22.7	39.8	29.0	48.1
Spring		138.3	44.1	50.9	18.6	39.1	27.5	50.5	41.0	59.4
Summer		14.6	0.0	0.0	0.0	0.0	3.3	4.7	0.4	0.8
Autumn		206.5	122.8	126.8	63.3	103.0	131.7	133.8	74.7	101.7
Total runoff reduction (%) for the entire study period										
Min		–	6.6	10.4	14.6	13.1	10.3	9.9	12.7	11.0
Max		–	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0
Average		–	54.8	50.6	81.1	63.6	60.3	50.8	68.8	54.9
Total runoff reduction (%) for each season										
Winter		–	58.9	50.7	94.0	74.2	78.7	62.5	72.7	54.7
Spring		–	68.1	63.2	86.6	71.7	80.1	63.5	70.3	57.1
Summer		–	100.0	100.0	100.0	100.0	77.7	67.6	97.5	94.6
Autumn		–	40.5	38.6	69.3	50.1	36.2	35.2	63.8	50.7

Table 12: Rainfall depth (mm) and runoff reduction (%) varying in response to different vegetation and substrate depth combinations. Substrate depths were either 8 cm or 16 cm, and each bed consisted of one of the following: *Origanum onites* (Origan.), *Sedum sediforme* (Sedum), *Festuca arundinacea* (Festuca), or No veget. (non-vegetated) (Soulis *et al.* 2017).

5.2 Effect of substrate and plant selection on retention

A few studies have analyzed the effect of substrate type on green roof runoff reduction performance. Conclusions that might be drawn from these studies are that deeper substrates and the inclusion of plants, particularly those with xerophytic traits, enhance the runoff volume reduction of a semi-arid or arid green roof. Beecham and Razzaghmanesh (2015) first compared the effects of intensive substrate designs with extensive substrate designs for four green roofs in Adelaide, Australia. They found that intensive green roofs (depth=300 mm) had higher peak retention and peak runoff delay values compared to extensive roofs (depth=100 mm); intensive vegetated roofs attained between 60% and 95% runoff retention and the extensive beds retained between 55% and 86%, approximately (Beecham and Razzaghmanesh 2015). Holding substrate type, vegetation presence, and slope constant, intensive green roofs consistently perform better than extensive green roofs in terms of mean retention (Figure 29) (Beecham and Razzaghmanesh 2015). However, only intensive roofs with vegetation showed statistically significant performance over extensive roofs with vegetation; differences in mean annual retention values between non-vegetated intensive and extensive roofs were not statistically significant (Figure 29) (Beecham and Razzaghmanesh 2015). The researchers theorized that the greater depth of intensive systems allowed for higher water holding capacity, particularly in cases of prolonged storms where outflow otherwise would have occurred. Furthermore, the presence of vegetation enhanced the retention performance of the system, as evapotranspiration by the plants plays an important role in reducing stormwater quantity more rapidly (Beecham and Razzaghmanesh 2015).

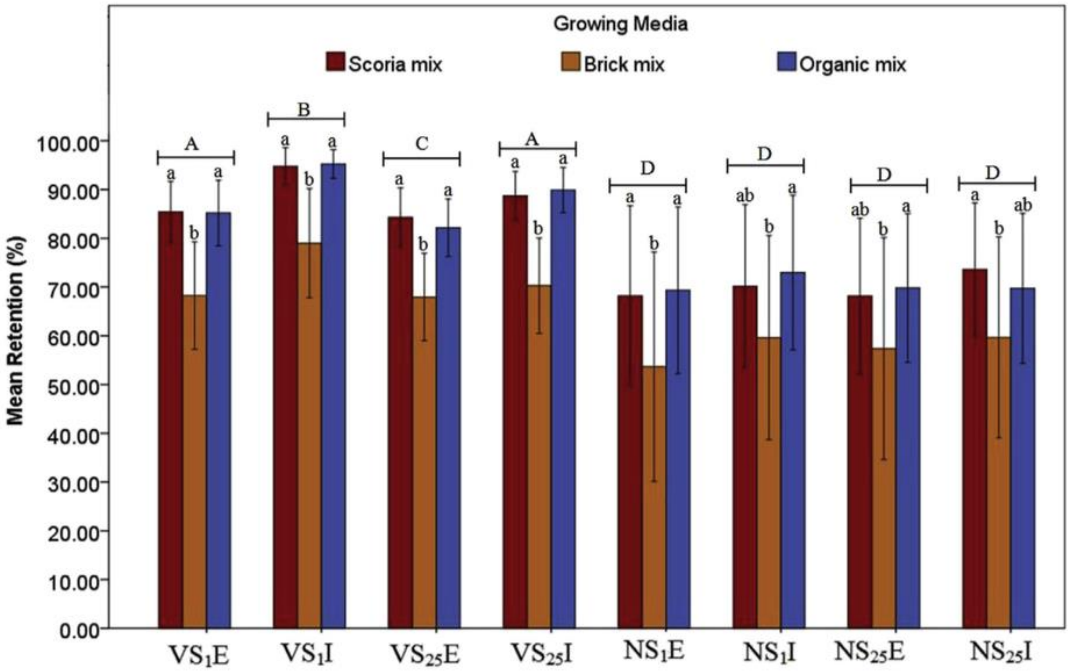


Figure 29: Differences in annual mean retention (%) for vegetated (VS) and non-vegetated (NS) green roofs with varying substrate depths (E=100mm, I=300mm), substrate types (Scoria mix, Brick mix, and Organic mix), and slopes ($_{1}$=1º slope, $_{25}$= 25º slope) in the dry climate of Adelaide, Australia. Different lowercase letters show significant differences (two-way ANOVA, $p<0.05$), and different capital letters show significant differences (one-way ANOVA, $p<0.05$) (Beecham and Razzaghmanesh 2015).

Variations in both plant type and substrate depth can also affect the water retention properties of a semi-arid green roof. In another study, Razzaghmanesh *et al.* (2014A) observed that in Adelaide, South Australia, vegetation planted in both intensive and extensive green roofs actually featured roots that grew beyond the substrate layer and broke through the drainage layer to access water. They concluded that this successful root growth into the storage layer can further reduce peak flow entering the system and might be beneficial in locations where runoff quantity is a primary concern (Razzaghmanesh *et al.* 2014A). Additionally, substrates composed of scoria (media B) and a mixture of scoria and organic material (media C) had different water use efficiencies (Figure 30); while media C has the higher water use efficiency (WUE) that might allow for longer plant survival in drought periods, media B likely facilitates increased evapotranspiration from the system, thus reducing runoff volumes to a greater extent and more rapidly (Razzaghmanesh *et al.* 2014A). These characteristics have particular utility in large storm events.

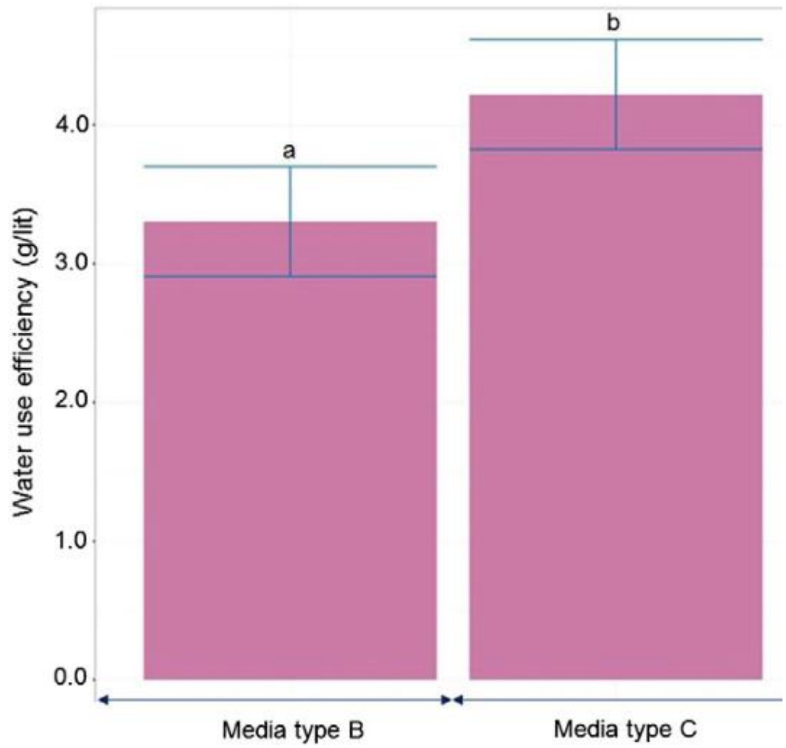


Figure 30: Significant differences in water use efficiencies (WUE) of two green roof media types installed in Adelaide, Australia. Media evaluated in the study includes either scoria (media B) or both scoria and organic matter (media C) (Razzaghmanesh *et al.* 2014A).

Soulis *et al.* (2017) also looked at the runoff reduction performances of different substrate depth and plant cover combinations based on data from 30 lysimeters in Athens, Greece. They arrived at a similar conclusion as Beecham and Razzaghmanesh (2015) and argued that deeper substrates always showed greater retention performance (Soulis *et al.* 2017). However, they also examined the effect of plant type in conjunction with each substrate type. Green roofs with deeper substrate supporting xerophytic plants resulted in the highest runoff reduction rate compared to deep substrate planted with either turfgrass or succulents (Soulis *et al.* 2017). *Origanum onites* and *Festuca arundinacea* showed faster reduction in soil moisture after rain events than *Sedum sediform*, which researchers attributed to higher evapotranspiration requirements of the former two species (Figure 31) (Soulis *et al.* 2017). They speculated that the "leaf seasonal dimorphism" of xerophytes allowed the plants to adjust evapotranspiration needs according to whether more or less water was available at a given time (Soulis *et al.* 2017). The heightened evapotranspiration of xerophytes in times of high substrate moisture content and their drought tolerance under dry soil conditions allowed the plants to reduce runoff in rainy periods and continue to thrive in times of drought. In contrast, succulents failed to utilize water rapidly enough to efficiently reduce runoff when most necessary after large storms (Soulis *et al.* 2017). Interestingly, in contrast to the higher runoff volume reduction capability of non-vegetated controls relative to vegetated bioretention systems, the non-vegetated control green roofs of both 16 cm and 8 cm depths had the lowest peak runoff reduction rates (Soulis *et al.* 2017). This might be because higher leaf area index typically correlates with higher evapotranspiration and subsequently greater water loss from the system, as determined by other studies referenced in Soulis *et al.* (2017).

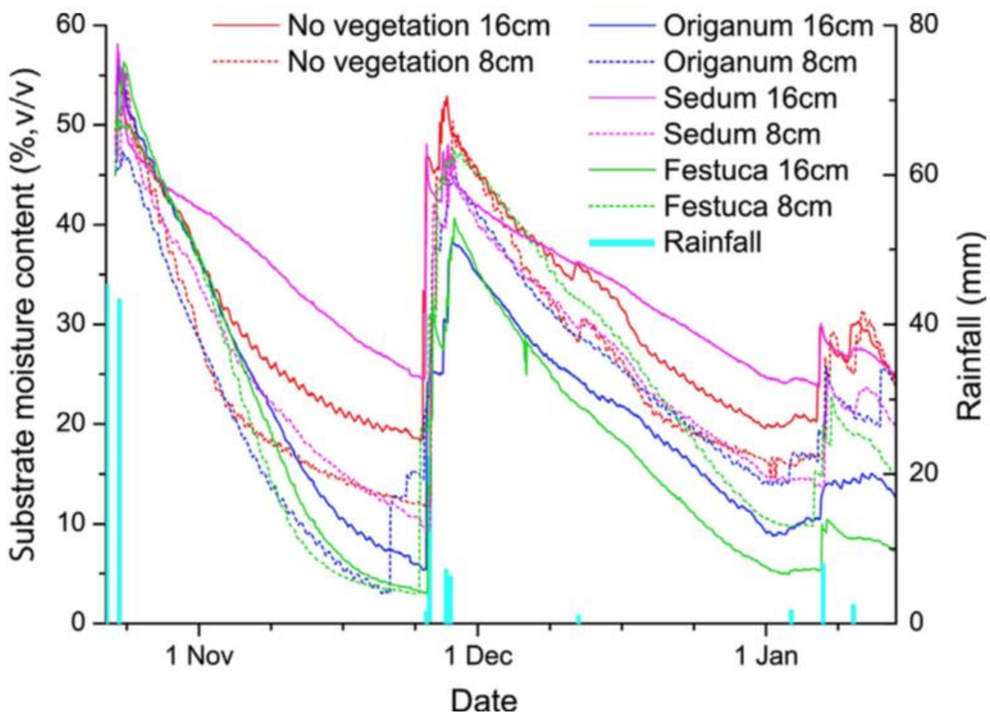


Figure 31: Effect of vegetation type in reducing the substrate moisture content of green roofs following three storm events in Athens, Greece. Substrate depths were either 8 cm or 16 cm, and observed plants include *Origanum onites* (Origanum), *Sedum sediforme* (Sedum), *Festuca arundinacea* (Festuca), or non-vegetated (No vegetation) (Soulis *et al.* 2017).

Studies of dry periods in temperate climates demonstrates the impact that vegetation can have on water storage and substrate moisture characteristics in terms of both total moisture content and distribution through the soil profile (Berretta *et al.* 2014). Moisture content has been shown to increase with depth under vegetated beds (TB1, TB2, and TB3, Figure 32), but no such vertical gradient exists in non-vegetated beds (TB4, Figure 32) (Berretta *et al.* 2014). Therefore, plants and their roots might affect the overall field capacity of the green roof, and through transpiration, enhance stormwater retention compared to non-vegetated roofs (Beretta *et al.* 2014). Despite the validity of Beretta *et al.* (2014) in observing the performance of green roofs in dry weather, this study might be of only tentative usefulness given it being conducted in the otherwise temperate climate of Sheffield, England, with dry periods of only about 10 days.

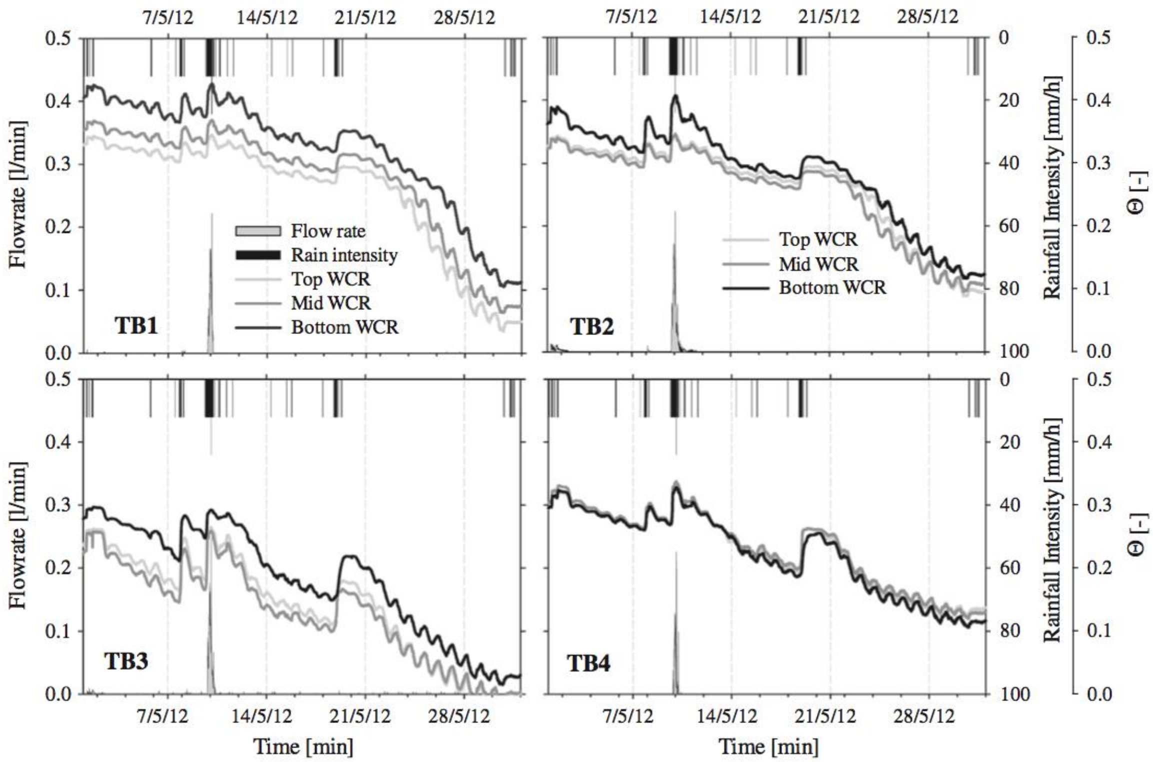


Figure 32: Distribution of soil moisture throughout the vertical profile of green roof test beds with vegetation (TB1, TB2, TB3) and without vegetation (TB4) over the course of a dry period in Sheffield, England. Soil profiles are considered at three depths below surface: top (20 mm), mid (40 mm), and bottom (60 mm) (Berretta *et al.* 2014).

6. Green roof pollutant removal utility in semi-arid and arid climates

Few studies examine the performance of green roofs in reducing pollutant levels in stormwater runoff. However, the few that do so show that green roofs remove few pollutants from stormwater and end up exporting many more. It should be noted that in all climates, retention of different pollutants varies based on vegetation type, media type, and maintenance practices. For example, studied vegetated roofs have retained more P and N than non-vegetated roofs with the same substrate type, and significant differences have been found between retention properties of green roofs planted with *Sedum* and herbaceous perennials (Rowe *et al.* 2010). Generally, as green roofs age, they can retain higher concentrations of pollutants, which Rowe *et al.* (2010) attributes to the reduction of the decomposed organic material that the substrate contains at installation and replacement by fully-developed plants with greater potential for nutrient uptake. In addition to investigating the removal of common nutrients from runoff, research has also focused on the potential for green roof systems to remove metals. As with nutrient retention, studies report varying findings, although frequently substrate and substrate amendments contribute additional metals to the runoff, as discussed below. In order to improve the efficacy of stormwater quality improvement, Oberndorfer *et al.* (2007) recommends taking care in choosing maintenance regimes and utilizing plants that require high levels of nutrients in order to prevent contributing to runoff pollutants. Ahiablame *et al.* (2012) suggests combining green roofs with other LID practices in the form of treatment trains in order to improve quality. On this basis, green roofs should not be considered a primary management tool for improving the quality of runoff. Leaching of substrate material in particular has been identified as a cause of many heightened chemical levels in green roof effluent (Rowe *et al.* 2010).

6.1 Effects of design variations on runoff quality

Design variations in green roofs can negatively affect the quality of runoff to varying degrees. Razzaghmanesh *et al.* (2014B) compared extensive roof and intensive roof designs with two different media types in semi-arid Australia to determine the impact of each on stormwater quality. Substrates were either composed of crushed brick, scoria, coir fibre, and composted organics (substrate A) or of scoria, composted pine bark, and hydro-cell flakes (substrate B). They found that generally for all green roofs scenarios, pollutant levels improved from the beginning of the 9-month study period to the end as more materials leached out early on and the system stabilized (Razzaghmanesh *et al.* 2014B). However, intensive roofs with substrate A showed the highest values for pH, turbidity, nitrate, phosphate, and potassium, which the researchers attribute to potentially either the material of substrate A or the addition of fertilizer to the system, both of which might have contributed nutrients such as phosphate through the addition and type of organic matter added (Razzaghmanesh *et al.* 2014B). They attributed the lesser contamination of extensive green roofs to a thinner soil media and subsequently less material to leach out of the system. The values of green roofs at the end of the study for turbidity, nitrate, chloride, phosphate, TSS, and potassium were higher for green roofs relative to the control roofs; thus, green roofs in almost all chemical parameters contaminated stormwater more than improved its quality compared to aluminum and asphalt control roofs (Figure 33) (Razzaghmanesh *et al.* 2014B).

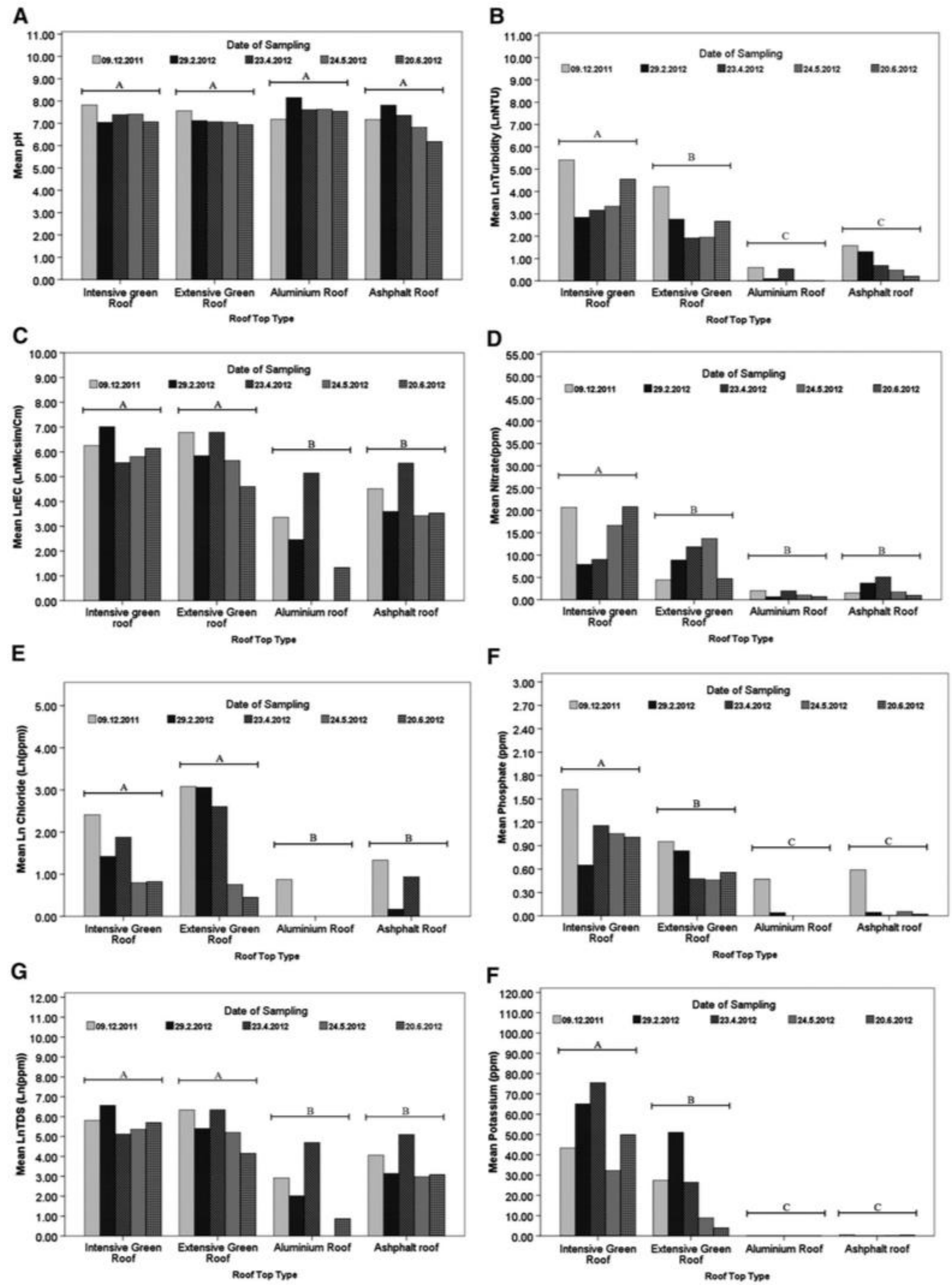


Figure 33: Water quality parameters measured for effluent off an intensive green roof, extensive green roof, aluminum roof, and asphalt roof in Adelaide, Australia for 5 sample days. Measured parameters include: mean pH (A), mean turbidity (B), mean electrical conductivity (C), mean nitrate (D), mean chloride (E), mean phosphate (F), mean total dissolved solids (G), and mean potassium (F) (Razzaghmanesh *et al.* 2014B).

Gnecco *et al.* (2013) investigated green roofs as sinks and sources of pollutants (Table 13) by evaluating outflow data from each storm event for a 3-year period in the Mediterranean climate of Genoa, Italy. They found that compared to bulk deposition input, green roof effluent caused higher TDS concentrations, as well as increased calcium, iron, and potassium (Table 14). However, green roofs reduced concentrations of zinc and copper through enhanced retention of the pollutants in the green roof layers, and they also featured lower concentrations of both solid particulates and metals compared to conventional roofs considered in this study (Gnecco *et al.* 2013). In most cases, in large rainfall events (e.g. October 4-5, Table 14), green roofs acted as sources of most pollutants, while for smaller events, including November 30-December 1, they became sinks (Gnecco *et al.* 2013). Although this study concluded that green roofs had an overall positive effect on pollutant loads compared to conventional rooftops, the pollutant retention is much less significant than that attained by bioretention systems in the same climate, and in any significant amount of rainfall, they generally contribute to overall contamination (Gnecco *et al.* 2013).

	EMC_{in}		EMC_{out}	
Parameter	**Mean**	**Std. Dev.**	**Mean**	**Std. Dev.**
pH [–]	5.4	0.7	7.1	0.4
EC [µS/cm]	20.7	8.6	64.6	16.2
TSS [mg/l]	2.9	4.2	2.8	3.2
TDS [mg/l]	20.7	9.5	99.0	17.1
COD [mg/l]	2.64	6.06	18.1	10.4
Cu [µg/l]	44	26	38	32
Fe [µg/l]	100	151	121	26
Mn [µg/l]	3.28	1.13	1.73	0.68
Zn [µg/l]	77	77	32	30
Ca [mg/l]	1.437	0.888	4.421	0.755
K [mg/l]	0.478	0.215	3.189	0.533

Table 13: Comparison of influent water quality influenced by bulk deposition (EMC_{in}) and effluent water quality (EMC_{out}) from a green roof in Genoa, Italy (Gnecco *et al.* 2013).

a)

Parameter	Rainfall–runoff events: 4–5 Oct.	Rainfall–runoff events: 30 Nov. –1 Dec.
TDS [mg/l]	○	○
COD [mg/l]	○	○
Cu [μg/l]	●●	●●
Fe [μg/l]	●●	○
Zn [μg/l]	○	●●
Ca [mg/l]	○	○
K [mg/l]	○	○

b)

Parameter	Rainfall–runoff events: 4–5 Oct.	Rainfall–runoff events: 30 Nov. –1 Dec.
TDS [mg/l]	○	●●
COD [mg/l]	○	●●
Cu [μg/l]	●●	●●
Fe [μg/l]	●●	●●
Zn [μg/l]	○	●●
Ca [mg/l]	○	●●
K [mg/l]	○	●●

Table 14: Variation in green roof pollutant removal performance as a source (white circle) or sink (double black circles) based on storm size and intensity in terms of a) concentration and b) mass in Genoa, Italy. October 4-5 was a large rain event with 111.6 mm depth and 114.4 mm/h maximum intensity, while November 30-December 1 was a small rain event with 15.6 mm depth and 4.8 mm/h maximum intensity (Gnecco *et al.* 2013).

Agra *et al.* (2018) compared the effects of using coal ash and perlite on runoff quality of green roofs in Ma'ale Tzva, Israel. Both substrate types had an effect on TP, pH, and EC over time. Figure 34 shows the levels of each water quality variable at different points throughout the year (Agra *et al.* 2018). The researchers determined that for both substrates, TP increased over time as vegetation required lower levels of nutrients later in the growing season (Figure 34a). Furthermore, both grey water and compost in the substrate added phosphorous to the system (Agra *et al.* 2018). Peak EC of runoff occurred following summer, which the researchers attributed to “first flush” of the system following prolonged buildup of salts in the substrate (Figure 34b) (Agra *et al.* 2018). The pH of runoff was comparable for each substrate type (pH_{ash}=8.0, $pH_{perlite}$=8.2), but both were slightly higher than average when compared to that of irrigation water (pH_{irr}=6.5 to 8.0) (Figure 34c) (Agra *et al.* 2018). Ultimately, both substrates were deemed suitable for use in green roofs, although high EC levels should be addressed by diluting salts within the system with irrigation water to prevent potential injury to plants (Agra *et al.* 2018).

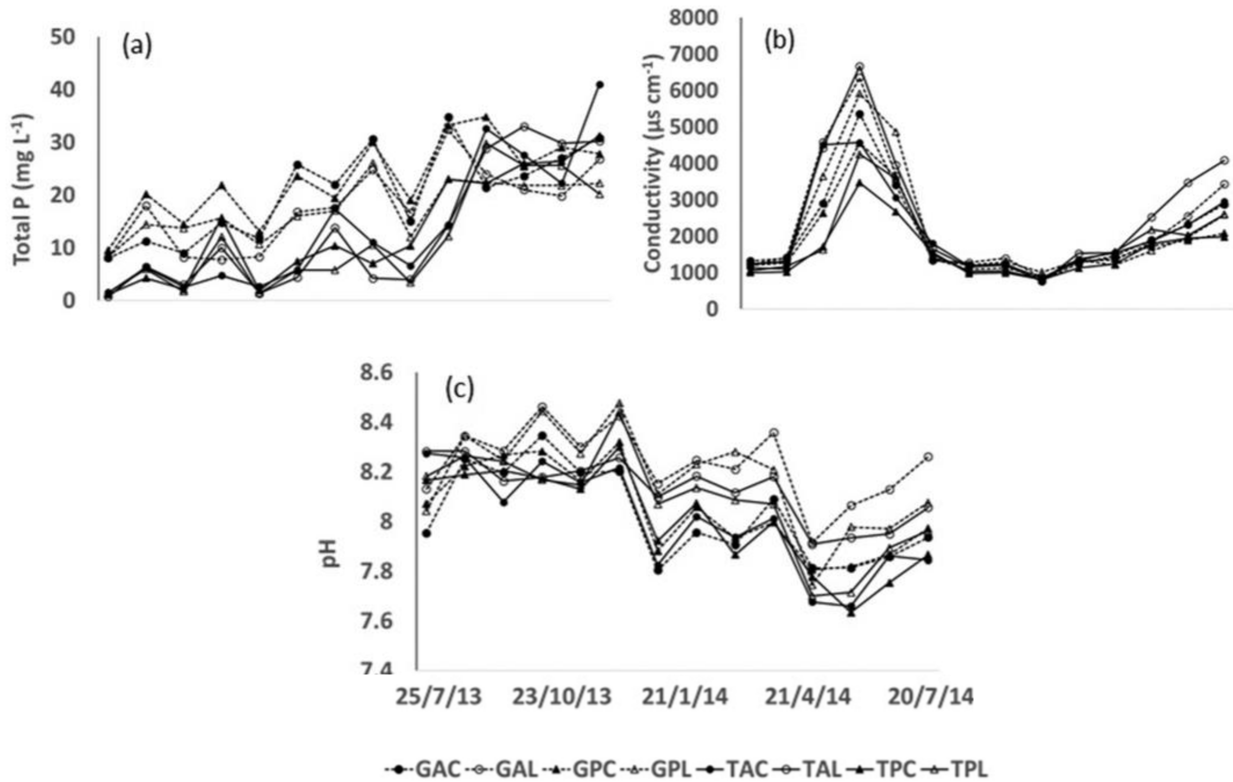


Figure 34: Change in TP, conductivity, and pH of green roof runoff in Ma'ale Tzva, Israel over the course of a one-year study conducted by Agra *et al.* 2018. Dashed lines show tap water (T), solid lines show grey water (G), circles show green roof area planted with ash (A) substrate, and triangles show area planted with perlite substrates (P) (Agra *et al.* 2018).

6.2 Additional ecological benefits of green roofs in semi-arid and arid climates

Despite the pollutant loading tendencies of green roofs in semi-arid and arid climates, runoff from these systems might still be utilized and reused. Monteiro *et al.* (2016) determined that runoff flowing from a semi-arid green roof over 12 months had pollutant values low enough to be recycled for non-potable uses like irrigation. Although not focused exclusively on the runoff quality of green roofs, Ondoño *et al.* (2016) concluded that green roofs in Santomera, Spain, particularly those with compost, soil, and brick-based substrate, successfully sequestered high levels of both carbon and nitrogen due to enhanced microbial activity, which specifically was highest under irrigated plots. Therefore, although pollutant removal potential within stormwater runoff might be limited in semi-arid green roofs, other benefits to ecological functioning and responsible uses of stormwater should be explored further.

7. Vegetation survival and suitability for semi-arid and arid green roofs

The majority of studies investigating green roofs in semi-arid or arid climates focus on the survivorship of plants installed within each system. Arid and semi-arid regions face the multi-faceted problem of lower rainfall depth, higher evaporation rates, and relatively thin vegetation cover that often requires additional irrigation where it does exist (Caraco 2000). As a result, many studies experiment with different vegetation types to determine whether certain plants suitable for green roofs in humid climates can be utilized in drier climates, as well. Furthermore, they also investigate the effect of different substrate characteristics on plant survival for different species.

7.1 Effects of substrate water-holding capacity on plant survivorship

Unsurprisingly, studies generally agree that substrate with higher water holding capacity usually accompanies enhanced survivorship in plants. Farrell *et al.* (2012) compared the performance of 5 succulents and 3 substrates under drought and well-watered conditions in Melbourne, Australia. Substrates were either a scoria mix, a crushed roof tile mix, or a bottom-ash mix, and each had distinguishing properties (Table 15). Plant biomass was higher when plants were adequately watered, and they determined that plants survived 12 days longer in bottom-ash substrate due to its higher water holding capacity, which allowed for higher evapotranspiration overall even in drought conditions because plants had extended access to water (Farrell *et al.* 2012). Alternatively, scoria had lower water holding capacity and plants survival was shorter in this substrate, perhaps due to greater “air-filled porosity” in this substrate

(Figure 35) (Farrell *et al.* 2012). The water-holding capacity of each substrate should also be considered in relation to the type of vegetation within the system, as some species drain water from the system at a much faster rate than others and therefore might limit the efficacy of even substrates with high water-holding capacity.

Substrate	pH	$EC_{1:5}$ dSm^{-1}	WHC %	AFP %	Bulk density gcm^{-3}
Scoria mix 60% scoria 8 mm minus, 20% 7 mm scoria aggregate and 20% coir	6.9[c] (± 0.03)	0.14[a] (± 0.003)	45.9[b] (± 0.3)	13.8[b] (± 0.5)	1.26[b] (± 0.01)
Crushed roof tile mix 80% 8 mm minus crushed roof tile and 20% coir	6.4[a] (± 0.01)	0.15[a] (± 0.004)	44.0[a] (± 0.8)	7.14[a] (± 0.3)	1.56[c] (± 0.01)
Bottom ash mix 60% bayswater sand (less than 2 mm diameter), 20% eraring filter (less than 10 mm diameter) and 20% coir	6.7[b] (± 0.03)	0.48[b] (± 0.004)	51.7[c] (± 0.4)	7.65[a] (± 0.6)	1.11[a] (± 0.01)

Table 15: Differences in characteristics of three common substrate mixes for use in green roof plots examined in Farrell *et al.* 2012. Recorded properties include pH, electrical conductivity (EC), water-holding capacity (WHC), air-filled porosity (AFP), and bulk density (Farrell *et al.* 2012).

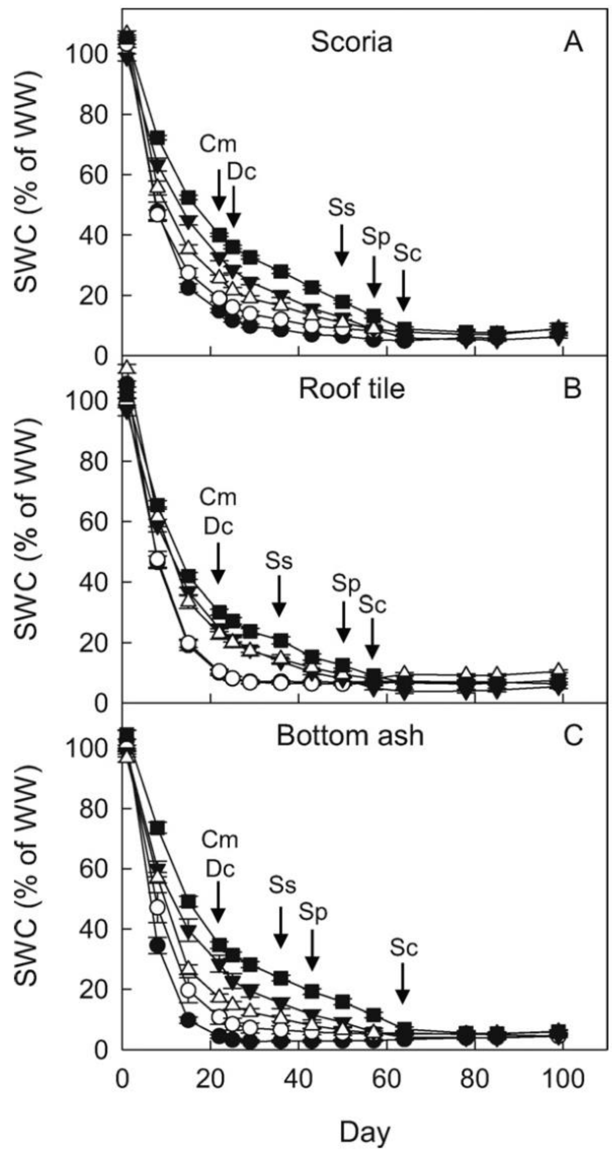


Figure 35: Change in soil water content over time for green roof plots in Melbourne, Australia planted with different vegetation in scoria (A), roof tile (B), and bottom ash (C) substrates. Substrate properties are listed in Table 12. Vegetation consisted of the following species: *Carpobrotus modestus* (Cm), *Disphyma crassifolium* (Dc), *Sedum pachyphyllum* (Sp), *Sedum spurium* (Ss) and *Sedum clavatum* (Sc) (Farrell *et al.* 2012).

Raimondo *et al.* (2015) explored the retention characteristics of two substrates in a green roof at the University of Messina in Italy. They found that the amount of available water for substrate A, which had smaller grain size, higher electrical conductivity, lower organic matter, higher pH, and higher porosity, was about 12% greater than that of substrate B, resulting in higher soil water potential values in plants after 48 hours (Raimondo *et al.* 2015). They concluded that small differences in substrate properties can have large effects on the performance of vegetation in semi-arid green roofs (Raimondo *et al.* 2015). While highly porous soils allow for greater volumes of water to be stored, substrates with lower water holding capacities achieve lower initial moisture content much earlier, which is beneficial for restoring the retention capabilities of a green roof following a large storm event (Raimondo *et al.* 2015).

Berretta *et al.* (2014) observed that for dry periods in the temperate climate of Sheffield, comparing 3 types of substrate, those with higher field capacity ("Heather with Lavender Substrate (HLS)" and "Sedum Carpet Substrate (SCS)") also had higher initial moisture content than the substrate with low field capacity ("Lightweight Expanded Clay Aggregate (LECA)"). They found that substrate moisture loss correlated with temperature, and therefore, in dry and warm periods, lower evapotranspiration accompanied lower soil moisture. For example, they note that the LECA-based substrate, which had the lowest field capacity and lowest percentage of fine particles, resulted in an immediate decrease in moisture content as a dry period began (Figure 26). Generally, this restores the retention capacity for future storms more quickly than other substrates, but this might also prompt plant water stress more rapidly (Berretta *et al.* 2014). Therefore, in semi-arid and arid locations where evapotranspiration and temperatures are likely high, substrates with low field capacity should likely be avoided, or plants should be irrigated or chosen selectively to prevent excessive drying.

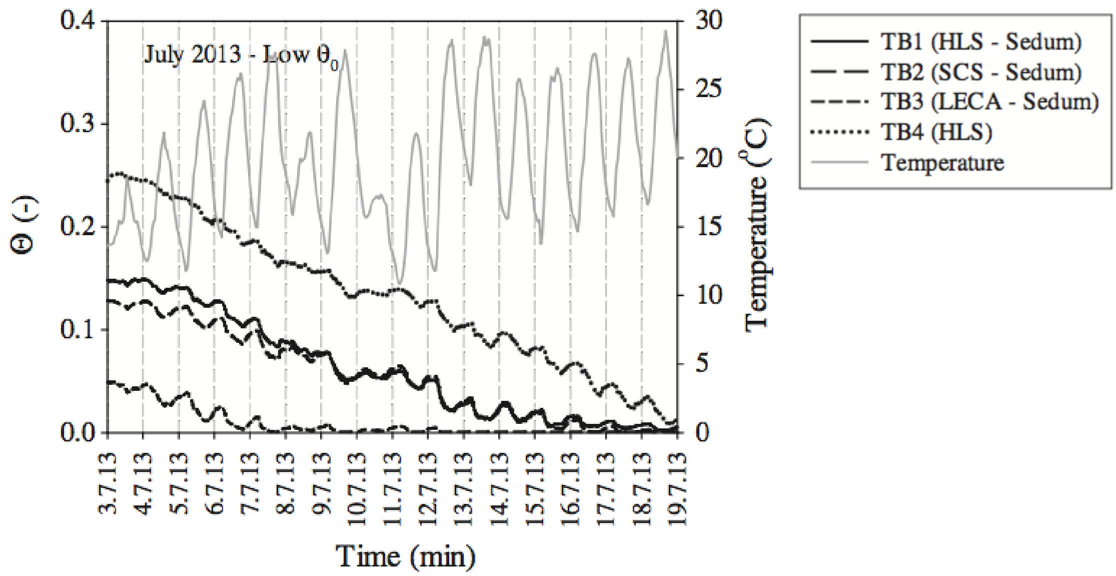


Figure 36: Response of green roof substrates with different soil moisture contents to a dry period in Sheffield, England. In order of decreasing field capacity and initial soil moisture content, substrates in the Berretta *et al.* 2014 study include heather with lavender (HLS), *Sedum* carpet (SCS), and lightweight expanded clay aggregate (LECA). Test beds 1 through 3 (TB1-TB3) are vegetated with *Sedum* while test bed 4 (TB4) is non-vegetated (Berretta *et al.* 2014).

7.2 Effects of substrate amendments on plant survivorship

Traditionally, substrate should have several key characteristics to be functional, such as a small amount of organic matter, high hydraulic conductivity, strong binding properties to roots, and good water retention capacity (Petrović *et al.* 2017). As a result, traditionally a few substrate types dominate in green roof designs; these include perlite, vermiculite, pumice, or sand augmented by organic matter (Ampim *et al.* 2010). As green roof technology advances however, new substrates have been introduced. Agra *et al.* (2018) suggests recycling coal ash by using it as a growing substrate in semi-arid regions. The study compared the effects of combinations of substrates (coal ash or perlite), different water types (grey or tap), and plant species (*Phyla nodiflora*, *Concolculus mauritanicus,* or no vegetation) on the growth of green roofs plants in semi-arid Ma’ale Tzva, Israel. The performance of plants grown in coal ash rivaled that of the perlite traditionally used commonly in green roofs, and in the case of the bed planted with *Convulvulus mauritanicus*, the coal ash actually surpassed perlite in its capacity to promote plant growth in a semi-arid location (Agra *et al.* 2018). The pH of substrate composed of coal ash did not differ significantly from that of perlite by the end of the 12–month study period, and no negative effects occurred when grey water was used in conjunction with coal ash substrate; this allows for two potential environmental contaminants to be utilized in green roof design in a sustainable way (Agra *et al.* 2018). Despite the general success of both species in coal ash, *Phyla* plants still performed slightly better in perlite than in coal ash while the inverse was true for *Convolvulus* plants (Agra *et al.* 2018). Therefore, variations in performance by plant species should be considered when introducing new recycled materials such as coal ash as substrate amendment.

Papafotiou *et al.* (2013) found that using substrate amended with grape marc compost (grape marc compost:soil:perlite at 2:3:5) produced greater xerophyte growth compared to the control peat-amended substrate (peat:soil:perlite at 2:3:5) in a semi-arid Mediterranean climate. Of particular importance is the fact that the addition of grape marc compost to substrate enhanced plant growth even in shallow and unirrigated green roof cells compared to deep, irrigated peat-amended cells (Figure 37) (Papafotiou *et al.* 2013). This allows for less water use and thinner substrate depth in green roof design, which is an important implication for rooftops which are structurally unable to bear the weight of an extensive green roof. The researchers attributed this higher plant growth to greater percentage of nutrients in the compost compared to the peat, although they warn that “the K/Mg rate in the compost substrate was high (6.6) and this could have led to a suppressive effect of K on Mg plant uptake” and caused chlorosis in older leaves (Papafotiou *et al.* 2013). Although it discusses the chemical content of each substrate amendment, this study fails to comment on the possible increase in effluent pollutants resulting from leaching of each substrate. (Papafotiou *et al.* 2013).

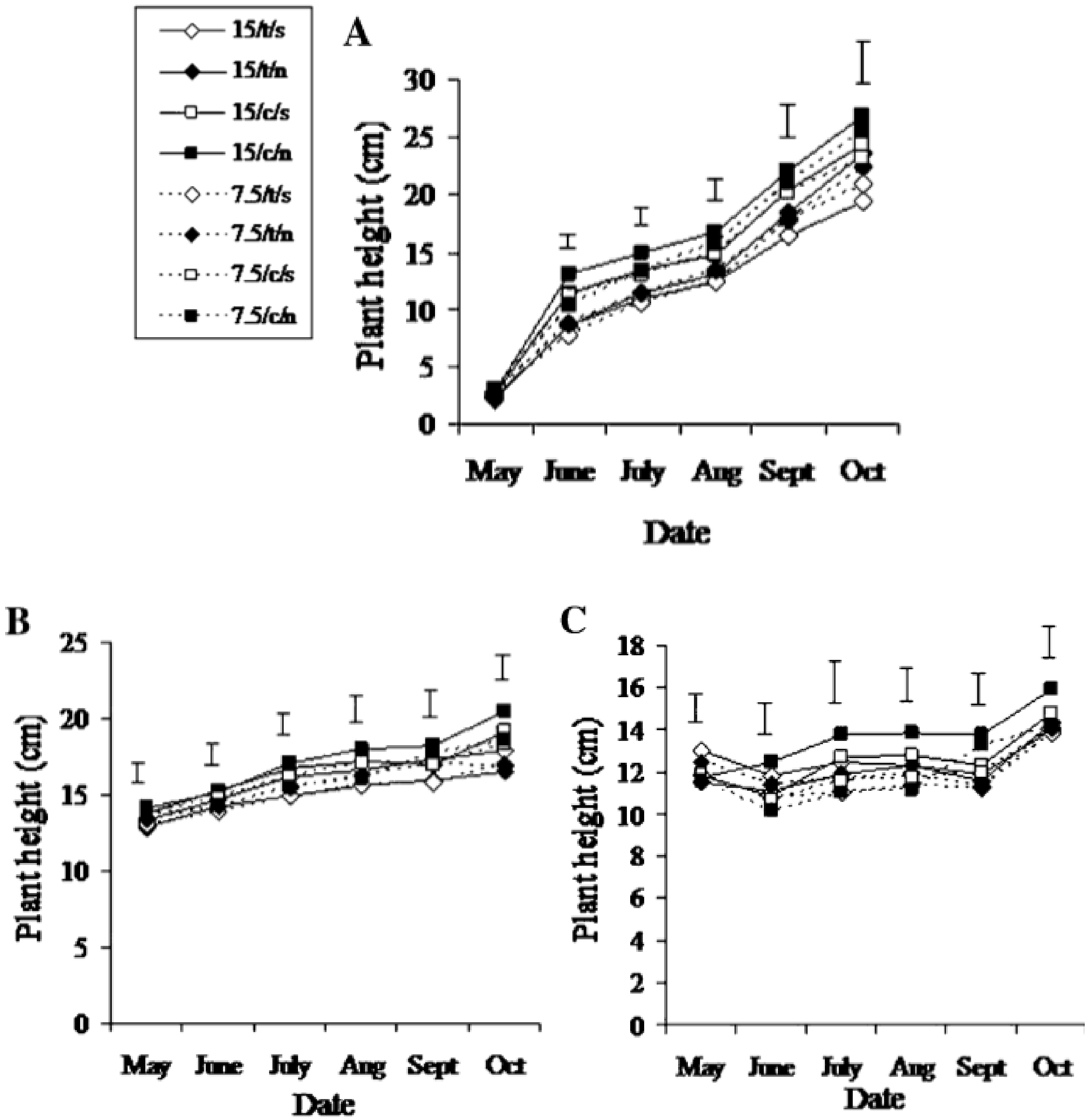


Figure 37: Performance of peat-amended substrate (diamonds) and grape mark compost-amended substrate (squares) on the growth of three different plant species (*Artemisia absinthium* (A), *Helichrysum italicum* (B), and *Helichrysum orientale* (C)) under different irrigation regimes (n=normal, s= sparse) and substrate depths (15=15cm, 7.5=7.5cm) in semi-arid Athens, Greece (Papafotiou *et al.* 2013).

Savi *et al.* (2014) looked at the effect of including hydrogels in substrate on plant available water and associated drought tolerance of *Salvia officinalis* at the University of Trieste's green roof in Italy. A hydrogel is a polymer that allows for enhanced absorption of water by several hundreds of times its own weight, increasing the water-holding capacity of substrate overall (Savi *et al.* 2014). Test planting beds of different depths were designed with different concentrations of hydrogels added (0.6% and 0.8%), and the researchers measured the water available to vegetation and associated plant growth. Hydrogels in this study absorbed up to 115 times their weight and increased the plant available water by up to 131% (Savi *et al* 2014). The study found that 0.6% hydrogel had the best ability to increase available water under drought stress, and its "functional advantage" was optimized with shallower substrate depths, as this allowed for less plant biomass and subsequently the highest performance in stomatal conductance (Figure 38) (Savi *et al.* 2014). Therefore, hydrogels are recommended for use in semi-arid climates in order to alleviate water stress in substrates with lower water holding capacities (Savi *et al.* 2014).

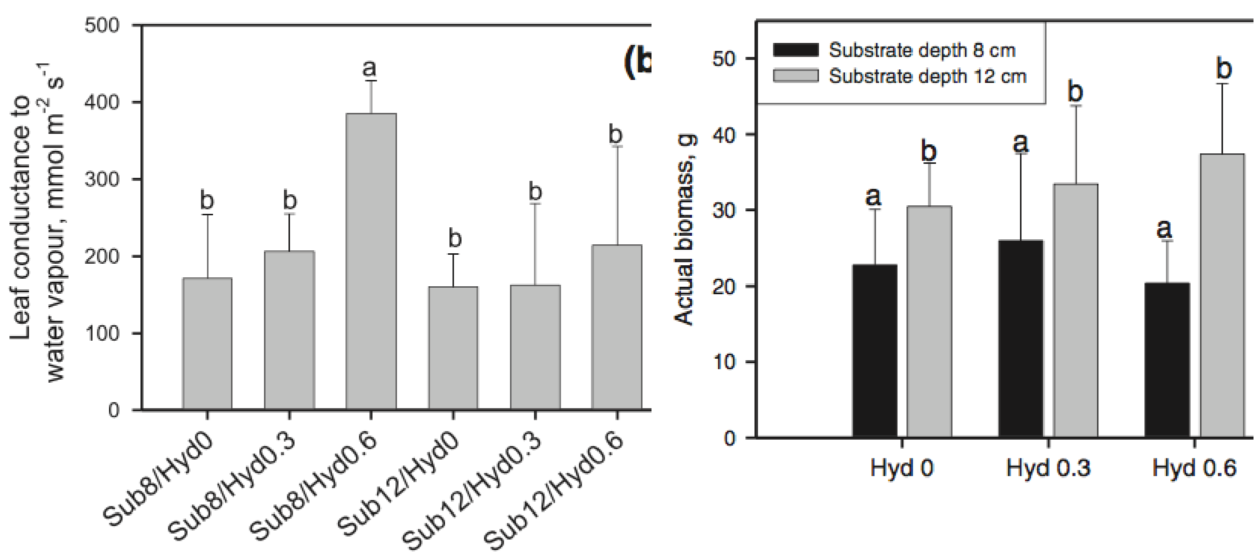


Figure 38: Differences in stomatal conductance and biomass of *Salvia officinalis* in response to substrate depth (8 cm and 12 cm) and hydrogel concentration (Hyd 0= 0% hydrogel, Hyd 0.3=0.3% hydrogel, Hyd 0.6%=0.6% hydrogel) in Treiste, Italy (Savi *et al.* 2014).

The EPA evaluated the potential for materials such as zeolite to be used as a green roof media in the semi-arid climate of Denver, Colorado (EPA 2012). They tested the effect of different percentages (33%, 66%, and 100%) of zeolite combined within a generic GreenGrid substrate on the growth of four different native *Sedum* varieties (*S. acre, S. album, S. spurium* 'Dragons Blood', and *S. spurium* 'John Creech') (EPA 2012). Although some species survived in the 100% zeolite mixture, others died off over the winter months, which the study speculates might have been the result of low precipitation combined with the water holding capacity of the zeolite (EPA 2012). This decrease in plant available water likely resulted in the desiccation of the plant root zones during overwintering (EPA 2012). Furthermore, the researchers hypothesize that environmental conditions such as minimal snow cover and higher albedo in the 100% zeolite test bed might have reduced plant survivorship, as well (EPA 2012). Therefore, the efficacy of specific amendments and substrate mixtures likely varies according to plant species and location-specific features, and thus universal application of a specific amendment or media composition should be discouraged.

Other studies adopt a multi-factorial approach and look at the success of green roof cells on the basis of an ideal combination of different distinctive parameters. Razzaghmanesh *et al.* (2014A) examined the effect of the interaction between roof slope, substrate type, substrate depth, and plant species on green roof plant growth in semi-arid South Australia. They determined that substrate A (crushed brick, scoria, coir fibre, and composted organics) was unconducive to plant growth, but that substrate B (scoria, composted pine bark, and hydrogel) and substrate C (substrate B with additional 50% organic matter) could both support plant growth with an average 90% rate of survival (Razzaghmanesh *et al.* 2014A). However, as is the case with other substrates, substrate C had significantly higher water use efficiency due to higher organic matter content and thus promoted slightly better plant performance relative to substrate media B (Razzaghmanesh *et al.* 2014A).

Ondoño *et al.* (2016) tested two substrates similar to those described in Razzaghmanesh *et al.* (2014B), but they compared the performance of a substrate containing compost, soil, and brick (CSB) and one containing only compost and brick (CB) in Murcia, Spain. They determined that the CB substrate had "higher porosity and lower density," but that CSB had "greater organic matter content and higher water-holding capacity" (Ondoño *et al.* 2016). Therefore, as in other studies, the substrate with higher organic matter content and water-holding capacity usually proved more capable of sustaining plants relative to even a more porous alternative. Furthermore, deeper substrate enhanced the plant available water content further, increasing survival even more (Ondoño *et al.* 2015). The researchers also expanded upon the higher organic matter content and described that in particular, the amount of humic material was higher in CSB, as were TC, TOC, and TN levels (Ondoño *et al.* 2016). Given that humic substances are credited by the researchers as stimulating root development and positively affecting plant growth in this study, the greater success of native plants growing in substrate of higher humic content might be attributed to this tendency and to the greater amount of available plant nutrition in CSB (Ondoño *et al.* 2016).

7.3 Effects of substrate depth on plant survivorship

The effect of substrate depth on the water retention capacity of each green roof was discussed previously, but substrate depth has also been studied in the context of plant survival. Generally, plant growth positively correlates with deeper substrate depth, although some exceptions have been noted. On a fundamental level, substrate depth controls the temperature of the soil, which has the potential to be too warm for adequate growing conditions in semi-arid and arid locations (Reyes *et al.* 2016). Reyes *et al.* (2016) analyzed 3 different substrate depths (5 cm, 10 cm, and 20 cm) in Santiago, Chile to see the effects of each on temperature and water requirements. They discovered that the shallowest green roof substrate (5 cm) actually responded in conjunction with ambient temperature extremes, with water content following the same trend. Deeper substrates of 10 cm and 20 cm had temperatures reasonably below air temperature, as well as stable and suitable volumetric water contents (Reyes *et al.* 2016). Owing to thermal amplitudes that were too high for plant growth in the 5-cm roofs, shown in Figure 39a, as well as rapid fluctuations in the volumetric water content, Reyes *et al.* (2016) warns against the use of extensive green roofs with depths of 5 cm or less in semi-arid climates.

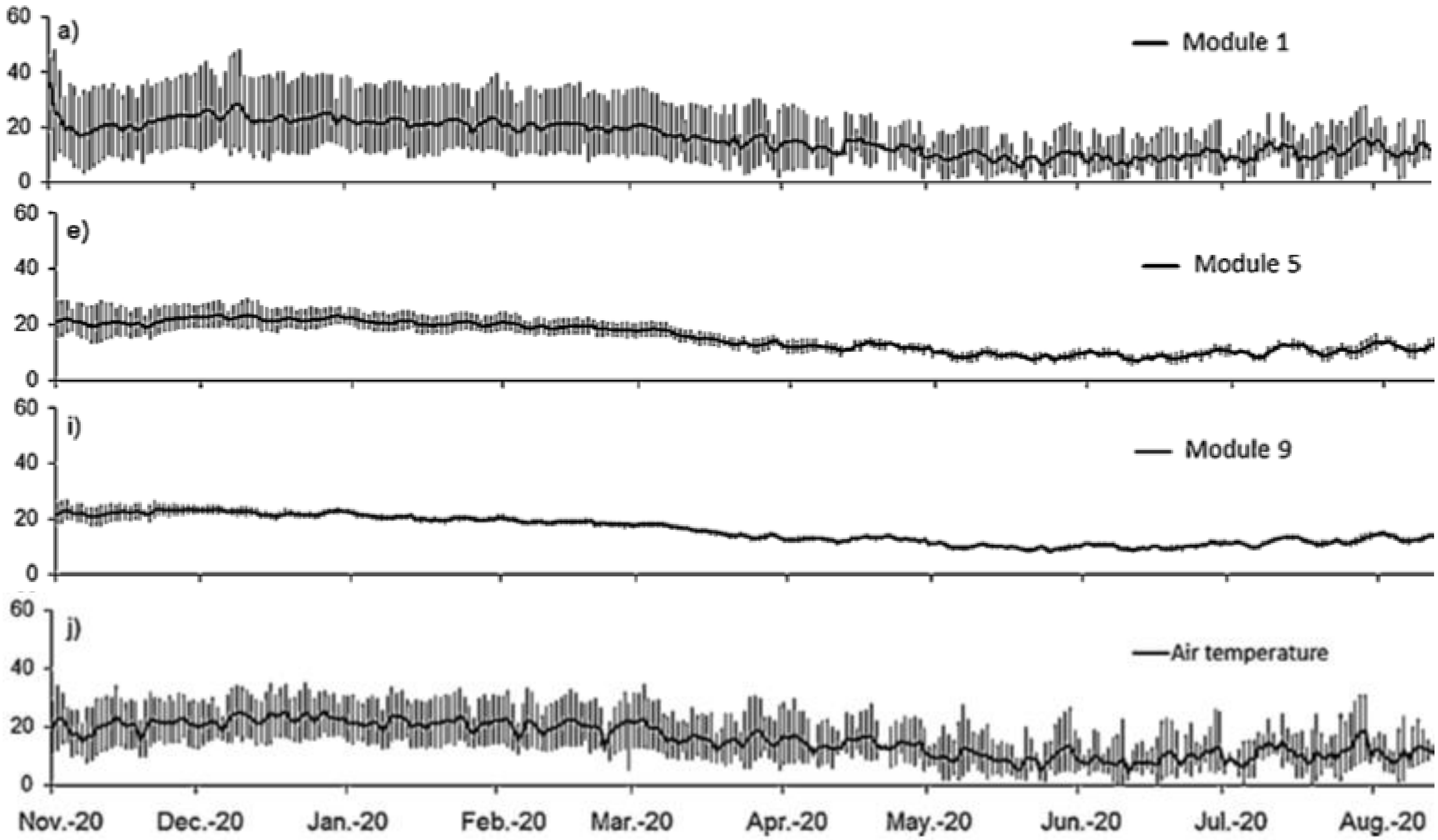


Figure 39: Soil temperature trends for samples of different depth substrates in Santiago, Chile. Cells with substrate depths of 5 cm (a) showed greater oscillation and closer correlation with air temperature (j) trends than those with either 10 cm (e) or 20 cm (i) depth (Reyes *et al.* 2016).

Soulis *et al.* (2017) compared plant performance in a green roof in Athens, Greece for three different plant covers in substrate depths of 8 cm and 16 cm. They found that the highest vegetation cover, regardless of whether plants were xerophytic, succulent, or turfgrass, occurred in the deeper substrate and that the lowest vegetation cover took place when substrate was shallow. Greater leaf area index (LAI) associated with higher plant cover enabled increased capacity for retention and higher evapotranspiration in the green roofs, and therefore deeper substrate might be better in green roof design (Soulis *et al.* 2017). Deep substrates have higher initial moisture content, as found in this study, that likely prompts greater plant cover and provides plants with access to water for longer periods of time during droughts (Soulis *et al.* 2017). Savi *et al.* (2014) arrived at the same conclusion and found that substrate of 12 cm depth on a green roof in Treiste, Italy increased aboveground biomass by almost 50% compared to 8 cm beds, which they also credited to the enhanced water holding capacity of deeper substrates throughout dry periods. They elaborated to attribute this heightened growth to the correlation between available soil volume and developed root mass. However, in terms of overall suitability, the researchers concluded that plants with lower biomass actually might be more appropriate for semi-arid green roofs (Savi *et al.* 2014). They recognized that plants grown in shallower substrate also experienced less growth and subsequently more careful water use, which strengthened plant water status in the "establishment phase" (Savi *et al.* 2014).

Razzaghmanesh *et al.* (2014A) also used root depth as a means of determining plant growth in Adelaide, Australia. They found that roots were deeper in the intensive system with 30 cm substrate depth compared to the 10 cm-depth extensive system, and they admitted that root depth was often deeper than both systems' substrate depths, and that some plants actually grew into the drainage layer below the system for both shallow and deep substrates (Razzaghmanesh *et al.* 2014A). Significant differences in root development varied according to species in the deep substrate whereas in the shallow substrate, no significant difference was observed (Figure 40) (Razzaghmanesh *et al.* 2014A). Therefore, deeper substrate might be more conducive to green roofs where a variety of plant species with differing root depths are present, as they enable more growth than intensive roofs with a variety of vegetation (Razzaghmanesh *et al.* 2014A).

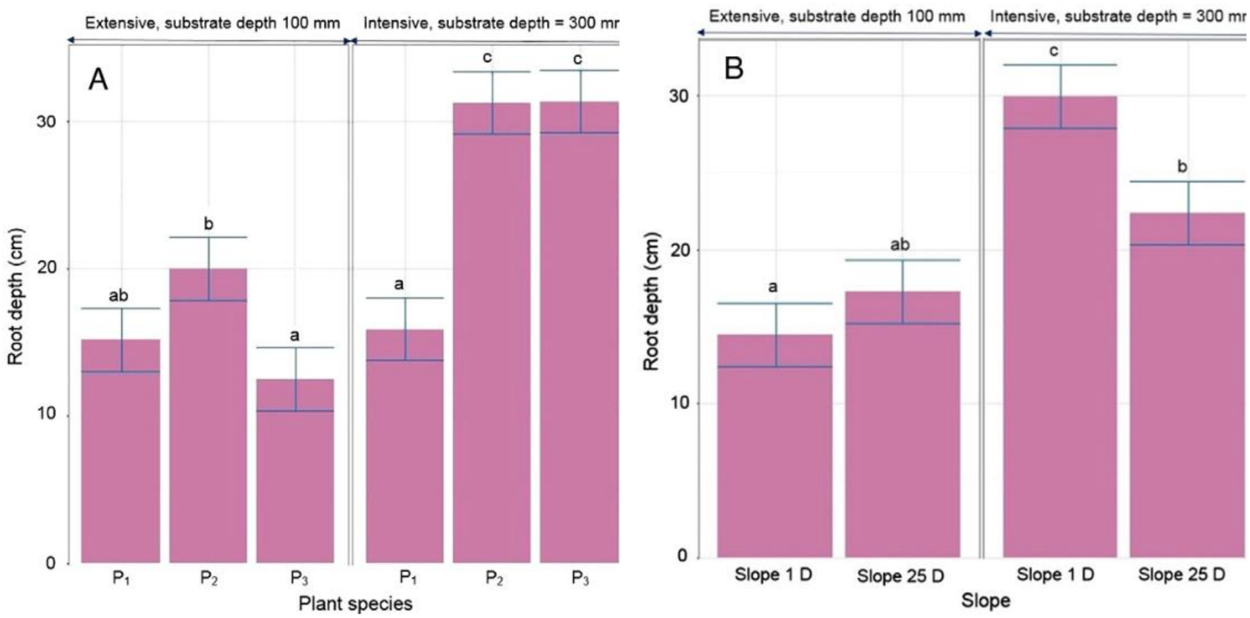


Figure 40: Effects of plant species and slope on the average root depth of plants grown in a green roof in Adelaide, Australia. Slopes were either 1º or 25º, and plant species were *Brachyscome multifida* (P1), *Chrysocephalum apiculatum* (P2), and *Disphyma crassifolium* (P3). Significant differences are marked by individual letters above columns (Razzaghmanesh *et al.* 2014A).

Despite many studies professing that greater substrate depth increases water content, reduces temperature extremes, and promotes biomass development, Papafotiou *et al.* (2013) found that depth had little effect on plant growth in the semi-arid climate of Athens, Greece. However, they admitted that other studies reported higher growth in deeper substrate associated with higher water-holding capacity during dry periods. Rather, in their study, plant performance varied based on how much compost was added to the substrate, with grape marc compost-amended soils performing better than those amended with peat owing to higher plant nutrient content (Papafotiou *et al.* 2013). This finding, as well as the general success of plants in many of the preceding studies in even shallow substrate system, supports the idea that many design parameters affect the performance of plants in green roofs, and frequently these factors act in conjunction with each other.

7.4 Survivorship of different plant species

Perhaps the greatest challenge to green roof design in extreme climates is the selection of plant species that can survive conditions of both extreme dryness and occasional inundation. Traditionally, *Sedum* varieties dominate extensive green roof design given their tolerance of drought, ability to thrive in shallow soils, and ability to withstand extreme weather conditions, as well as their ability to significantly reduce stormwater runoff in wet and humid climates (Villarreal and Bengtsson 2005; Dvorak and Volder 2010). However, additional research into plants suitable for semi-arid green roofs should be conducted given the risks associated with low genetic diversity within a garden (Bousselot *et al.* 2009). Furthermore, studies have found that the choice to use *Sedum sediforme* cover on green roofs could have a negative environmental impact. Use of *Sedum* in the Mediterranean actually increased net CO_2 emissions for the system compared to control rooftops, resulting in a net negative CO_2 balance throughout the dry season in this climate type (Agra *et al.* 2017).

Existing study looks at plant performance on a species-specific basis, as well as based on characteristics of certain plants. Commonly, studies argue that reliance on northwestern European species in green roof design is limiting, as many cannot handle the intense characteristics of a Mediterranean climate (Van Mechelen *et al.* 2014). They thus encourage the use of native plants or those of a Mediterranean variety instead based on the general success of these species relative to more common non-native alternatives. Van Mechelen *et al.* (2014) used 10 plant traits to screen 372 potential native green roof species and found that 28 Mediterranean plants within the list were not currently included but had good potential within dry-summer Mediterranean green roof systems (Van Mechelen *et al.* 2014). Species listed in Table 16 without an "x" are those previously unidentified explicitly as green roof plants that show promise in terms of performance and survival. *Sedum* varieties in particular scored the highest. Many species were kept off the list because of deep rooting systems, the inability to handle stress, or a general lack of knowledge on evaluated traits (Van Mechelen *et al.* 2014).

Species name	Also in GRTOT	Species list	Adjusted score (%)	Raunkiaer life form
Aethionema saxatile		MEDVEG	67	H
Ajuga genevensis	x	MEDVEG	67	H
Alyssum montanum	x	MEDVEG	71	H
Anacamptis pyramidalis		MEDLIT	63	H
Arabis alpina		MEDLIT	63	H
Arabis collina		MEDLIT	67	H
Armeria arenaria		MEDVEG	67	H
Athamanta cretensis	x	MEDLIT	71	H
Brachypodium distachyon		MEDVEG	67	T
Briza media	x	MEDVEG	63	H
Bromus rubens		MEDVEG	83	T
Campanula glomerata	x	MEDLIT	63	H
Campanula persicifolia	x	MEDLIT	63	H
Campanula rotundifolia	x	MEDVEG	63	H
Carex alba		MEDLIT	63	H
Carex mucronata		MEDLIT	63	H
Ceterach officinarum		MEDLIT	71	H
Crassula tillaea		MEDVEG	71	T
Crepis capillaris		MEDLIT	71	T
Dianthus sylvestris		MEDVEG	63	H
Dittrichia viscosa		MEDLIT	67	H
Filago pyramidata		MEDVEG	63	T
Galium boreale		MEDLIT	60	H
Helianthemum nummularium	x	MEDVEG	71	H
Helleborus foetidus		MEDLIT	63	H
Hieracium murorum	x	MEDVEG	71	H
Hieracium pilosella	x	MEDVEG	63	H
Hippocrepis comosa	x	MEDVEG	63	H
Linum tenuifolium		MEDVEG	71	H
Minuartia hybrida		MEDVEG	71	T
Poa badensis	x	MEDVEG	63	H
Polygala vulgaris		MEDVEG	63	H
Polypodium vulgare	x	MEDLIT	71	H
Potentilla pusilla		MEDVEG	67	H
Primula vulgaris	x	MEDLIT	63	H
Rubia peregrina		MEDLIT	67	H
Rumex scutatus		MEDLIT	63	H
Saponaria ocymoides	x	MEDVEG	71	H
Scabiosa columbaria	x	MEDVEG	63	H
Scrophularia canina		MEDLIT	67	H
Sedum acre	x	MEDVEG	100	H
Sedum album	x	MEDVEG	100	H
Sedum dasyphyllum	x	MEDLIT	86	H
Sedum ochroleucum	x	MEDVEG	100	H
Silene vulgaris	x	MEDLIT	63	H
Succisa pratensis		MEDLIT	63	H
Tanacetum corymbosum		MEDLIT	63	H
Taraxacum sect. ruderalia	x	MEDVEG	71	H
Teucrium montanum	x	MEDVEG	83	H
Thymus praecox	x	MEDVEG	71	H
Thymus serpyllum	x	MEDVEG	71	H
Veronica chamaedrys		MEDLIT	63	H
Viola reichenbachiana		MEDLIT	63	H

Table 16: List of Mediterranean plant species identified as having "good green roof potential" in the Mediterranean area by Van Mechelen *et al.* 2014. Species without an "x" are newly identified plants for green roof use in the Mediterranean region. Also shown for each species is the species list in which a plant was originally listed (MEDVEG and MEDLIT), adjusted score for green roof suitability, and the Raunkiaer life form (H=Hemicryptophyte, G=Geophyte, T=Therophyte). Adjusted score is determined by dividing a plant's total score for 10 traits by the number of traits with existing information (Van Mechelen *et al.* 2014).

An early study by Bousselot *et al.* (2009) compared plant data for the Rocky Mountain area to see the compatibility of native plants to a green roof system in Denver, CO. They found that all six native plants (Figure 41) performed well within the semi-arid system. However, plants exhibited either early season or late season growth patterns that affected the time at which the majority of their growth occurred (Figure 41) (Bousselot *et al.* 2009). Several of the plant groups investigated in this study and in others on a comparative basis are often succulents, forbs, xerophytes, and turfgrasses (Bousselot *et al.* 2009).

Gioannini *et al.* (2018) studied the performance of native plants compared to non-native plants in semi-arid New Mexico over a span of two years for three plant groups: groundcover, grasses, forbs, and succulents. Given that greater plant cover allows for cooler substrate temperatures and greater water retention, Gioannini *et al.* 2018 argued that plants with greater surface area should be prioritized to meet the 60% plant cover standard implemented by the green roof industry. Native plants performed better or comparably with non-native plants in most cases, with only the non-native succulent (*Delosperma nubigenum*) achieving greater groundcover than the native succulent (*Chrysactinia mexicana*) (Gioannini *et al.* 2018). The researchers concluded that only the grasses (non-native *Sedum kamtschiaticum* and native *Festuca glauca*) attained 63 and 62% cover, respectively, by the end of the second year, and therefore might be more appropriate to this climate (Gioannini *et al.* 2018).

While native species had overwhelming success relative to non-native species in Gioannini *et al.* (2018), other studies note that often some non-native species perform better than natives. Rayner *et al.* (2016) evaluated the survivorship of 32 different plant species of both native and non-native origin on an Australian green roof and discovered that although three species of non-native succulents survived in a green roof setting at a rate of 100%, only two native species of succulents survived at a rate of <30%, and none of these survived past the 18th week. They attributed this to succulents using more water during drought compared to *Sedum* species, and the native succulents also faced more extreme temperatures on the rooftop compared with their normal environment (Rayner *et al.* 2016).

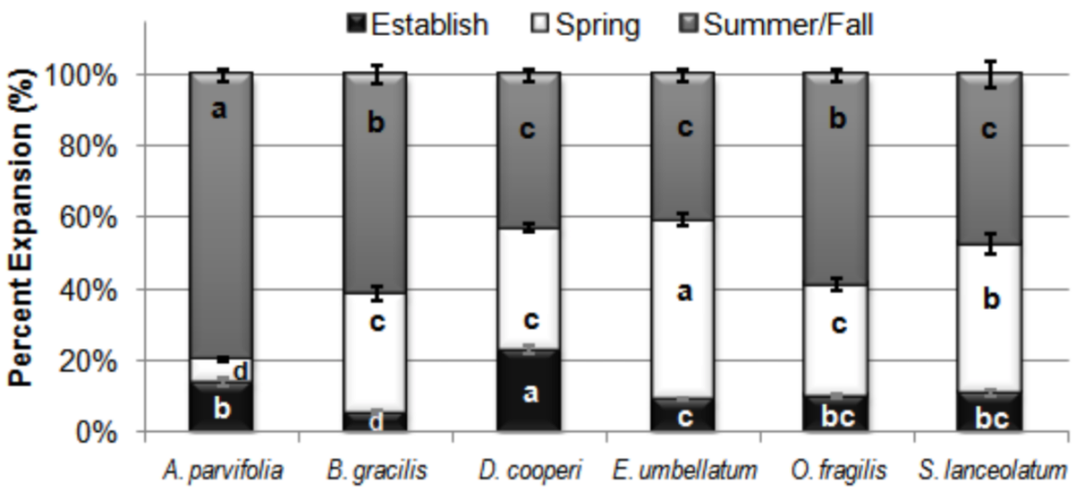


Figure 41: Percent expansion rate showing seasonal growth patterns for each species in a semi-arid green roof in Denver, CO. Seasons include Establishment (growth to March 22), Spring (March 22 to June 25), and Summer/Fall (June 25 to October 14). Species include the following: *Antennaria parvifolia*, *Bouteloua gracilis*, *Delosperma cooperi*, *Eriogonum umbellatum*, *Opuntia fragilis*, and *Sedum lanceolatum*. Lowercase letters show significant differences (Bousselot *et al.* 2009).

While native plants show decent potential, most studies investigate plant performance on the basis of specific plant characteristics. Specific traits that designate plant species suitable for semi-arid and arid green roofs are succulence, CAM pathways, and water use tendencies. Rayner *et al.* (2016) observed that succulents proved capable of survival after 25 to 34 weeks of low water content in a dry period in Melbourne, Australia, in contrast to the <40% survival of forbs and monocots after 10 to 12 weeks in the same semi-arid conditions. The percentage of plant survival was positively correlated with degree of leaf succulence when considering succulents collectively (Figure 42); higher succulence determines the water accessible to plants after substrate water is depleted (Rayner *et al.* 2016). The study stressed the overt failure of forbs and monocots chosen using the "habitat template approach" to emphasize the importance of choosing plants based on "local climatic extremes" rather than previous performance at a different location with vaguely similar attributes (Rayner *et al.* 2016).

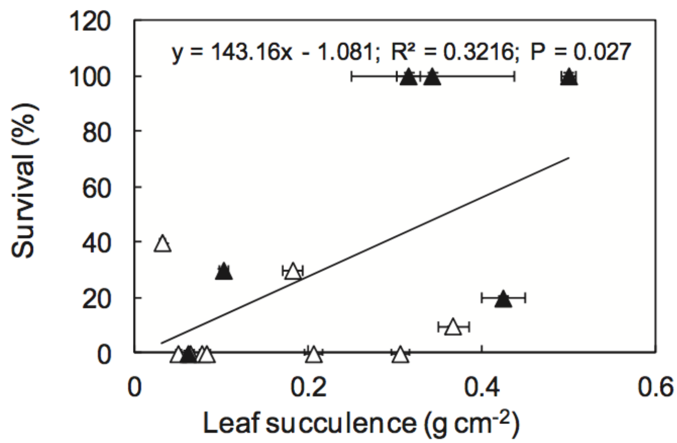


Figure 42: Relationship between degree of leaf succulence and succulent survival in a dry period in Melbourne, Australia. Although a significant correlation existed between variables for all succulents, considering prostrate succulents (white triangles) and upright succulents (black triangles) separately was not significant (Rayner *et al.* 2015).

The EPA's "Moisture Deficit Study," conducted in semi-arid Denver, Colorado, found that the foliage of succulent species can survive more than five times longer than that of the herbaceous species grown under identical conditions of increasingly long dry periods (EPA 2012). The study evaluated plant growth, dieback, and revival in greenhouse conditions and outdoor conditions over the course of 151 days. Tables 17 and 18 show the overall performance of herbaceous and succulent species in terms of water use, days to dieback, and revival percentage for outdoor and greenhouse trials, respectively. Succulents showed an average 111.8 days to dieback and 41.8% revival rate, while herbaceous species took about 19.8 days to dieback and recovered after drought stress at an average of only 22.6% in the greenhouse studies (EPA 2012). In the outdoor studies, only one succulent showed dieback at 31.2 days with 20.8% revival, while all herbaceous species showed dieback at an average of 27.6 days and an average revival of 41.7% (EPA 2012). The researchers attributed these differences to different ET rates, differences in root zones, and different exposure to radiation and wind (EPA 2012). As a result, herbaceous green roof plants in semi-arid and elevated areas, such as those in this study, should be irrigated with supplemental water supplies every 14 days even though succulents can withstand only monthly irrigation (EPA 2012).

Species	Plant type	Mean relative water use (SE)	Days to dieback (SE)	Revival[x]
Antennaria parvifolia	Herbaceous	-0.71% (0.21) b[1]	31.4 (0.24) c	54.17%
Artemisia frigida	Herbaceous	-3.21% (0.31) de	20.0 (0.31) a	50.00%
Buchloe dactyloides	Herbaceous	-1.57% (0.15) bc	27.7 (0.25) b	41.67%
Herbaceous mean		-1.83%	27.58	41.67%
Delosperma cooperi	succulent	-2.62% (0.25) d	NA[2]	NA
Penstemon pinifolius	succulent	-1.82% (0.08) c	31.2 (0.42) c	20.83%
Sedum album	succulent	-3.35% (0.36) e	NA	NA
Sedum lanceolatum	succulent	-0.94% (0.23) bc	NA	NA
Sedum spurium 'John Creech'	succulent	-2.57% (0.20) d	NA	NA
Sempervivum 'Royal Ruby'	succulent	+0.23% (0.15) a	NA	NA
Succulent mean		-1.85%	NA	NA

[1] Lower case letters show significant differences at the $p \leq 0.05$ level
[2] NA = Not applicable (due to truncation of study from freezing temperatures.)

Table 17: Survivorship and resilience of herbaceous and succulent species evaluated by mean water use, days to dieback, and revival rates in a semi-arid green roof in Denver, CO (EPA 2012).

Species	Plant type	Mean relative water use (SE)	Days to dieback (SE)	Revival
Antennaria parvifolia	herbaceous	-1.53% (0.19) c[1]	22.79 (0.65) d	31.25%
Artemisia frigida	herbaceous	-6.23% (0.74) k	16.08 (0.32) a	8.33%
Bouteloua gracilis	herbaceous	-4.63% (0.46) hi	18.23 (0.71) ab	22.92%
Buchloe dactyloides	herbaceous	-3.56% (0.35) f	20.19 (0.90) bc	37.50%
Carex flacca	herbaceous	-4.77% (0.50) hij	20.13 (0.90) bc	27.08%
Penstemon pinifolius	herbaceous	-3.63% (0.32) fg	20.09 (0.67) bc	0.00%
Thymus pseudolanuginosus	herbaceous	-2.26% (0.20) de	20.75 (0.87) c	31.25%
Herbaceous mean		-3.80%	19.75	22.62%
Allium cernuum	succulent	-1.13% (0.25) b	59.25 (1.77) f	91.67%
Delosperma cooperi	succulent	-5.24% (0.60) ij	52.25 (1.44) e	0.00%
Delosperma nubigenum	succulent	-5.56% (0.69) ijk	107.06 (3.46) g	2.08%
Sedum acre	succulent	-2.72% (0.31) e	107.67 (6.46) g	2.08%
Sedum album	succulent	-4.48% (0.60) gh	151.00 (0.00) j	58.33%
Sedum lanceolatum	succulent	-1.22% (0.27) bc	138.71 (2.53) i	54.17%
Sedum spurium 'John Creech'	succulent	-2.00% (0.22) cd	127.87 (3.72) h	56.25%
Sempervivum 'Royal Ruby'	succulent	+2.04% (0.36) a	151.00 (0.00) j	69.44%
Succulent mean		-2.54%	111.75	41.75%

Lower case letters show significant differences at the p≤ 0.05 level

Table 18: Survivorship and resilience of herbaceous and succulent species evaluated by mean water use, days to dieback, and revival rates in a greenhouse trial in semi-arid Denver, CO (EPA 2012).

As discussed previously, substrate moisture content is of critical concern when considering plant sustainability. Plants with lower water use are therefore preferred before those requiring regular irrigation, and many studies evaluate plant suitability as related to water usage and drought tolerance. Farrell *et al.* (2012) concluded that plant species should have conservative water use and high succulence to enjoy success in semi-arid green roofs. The study looked at the effects of drought on the health and growth of five succulents in three substrate materials in a hot and dry Australian climate. Given that Australian species in the study reduced the water in substrate faster than the *Sedum* species, *Sedum* species showed increased survival compared to their native alternatives; Australian succulent species died earlier because they failed to reduce biomass under drought stresses, while *Sedum* successfully reduced its growth in response to water limitations (Farrell *et al.* 2012).

Azeñas *et al.* (2018) also evaluated the growth and survival of succulents, but they compared the success of succulent CAM plants relative to other C4 and C3 native Mediterranean species to determine which drought adaptations are most effective in semi-arid Mediterranean environments. While *Sedum sediforme*, a CAM-facultative species, achieved the greatest plant cover, the other four species (*Brachypodium phoenicoides, Crithmum maritimum*, *Limonium virgatum,* and *Sporobolus pungens*) also maintained greater than 70% cover during the dry season, showing great tolerance to water-limited conditions (Azeñas *et al.* 2018). An important discovery is the difficulty of some plant species to survive the cooler temperatures of winter months, even in semi-arid climates. *Sporobolus pungens,* a perennial native to coastal sandy soil with an underground storage organ, showed decreased soil cover in the winter due to a C4 metabolism that limited growth (Azeñas *et al.* 2018). Furthermore, CAM-facultative species have the critical limitation of being less efficient in stormwater volume reduction capacity, making them of little usefulness in a stormwater management capacity (Azeñas *et al.* 2018).

In addition to sometimes having succulent characteristics, often plants display different survival strategies, such as the isohydric and anisohydric characteristics. Raimondo *et al.* (2015) compared the performance of two different shrub species (anisohydric *Salvia officinalis* and isohydric *Arbutus unedo*) in the Mediterranean within two substrates of differing water retention capacities. Anisohydric plants tolerate intense water stress while maintaining transpiration and photosynthesis, although plants in this study reduced their leaves and showed lower hydraulic conductance (k_{plant}) as dry periods lengthened compared to the well-irrigated samples (Queensland 2009; Raimondo *et al.* 2015). Alternatively, isohydric plants reduce leaf stomatal conductance (g_L) and show much more consistent hydraulic conductance compared to anisohydric alternatives in order to minimize transpiration in times of low water potential (Figure 43) (Raimondo *et al.* 2015; Queensland 2009; Sade *et al.* 2012). Despite different strategies, Raimondo *et al.* (2015) recommend that isohydric plants, such as *A. unedo,* be used in semi-arid areas with little to no irrigation options. Anisohydric species can also be used, as they often reduce runoff quantities to a greater degree than isohydric options, but these might not be as optimal for green roof designs, as they require frequent and regular irrigation to prevent senescence (Raimondo *et al.* 2015).

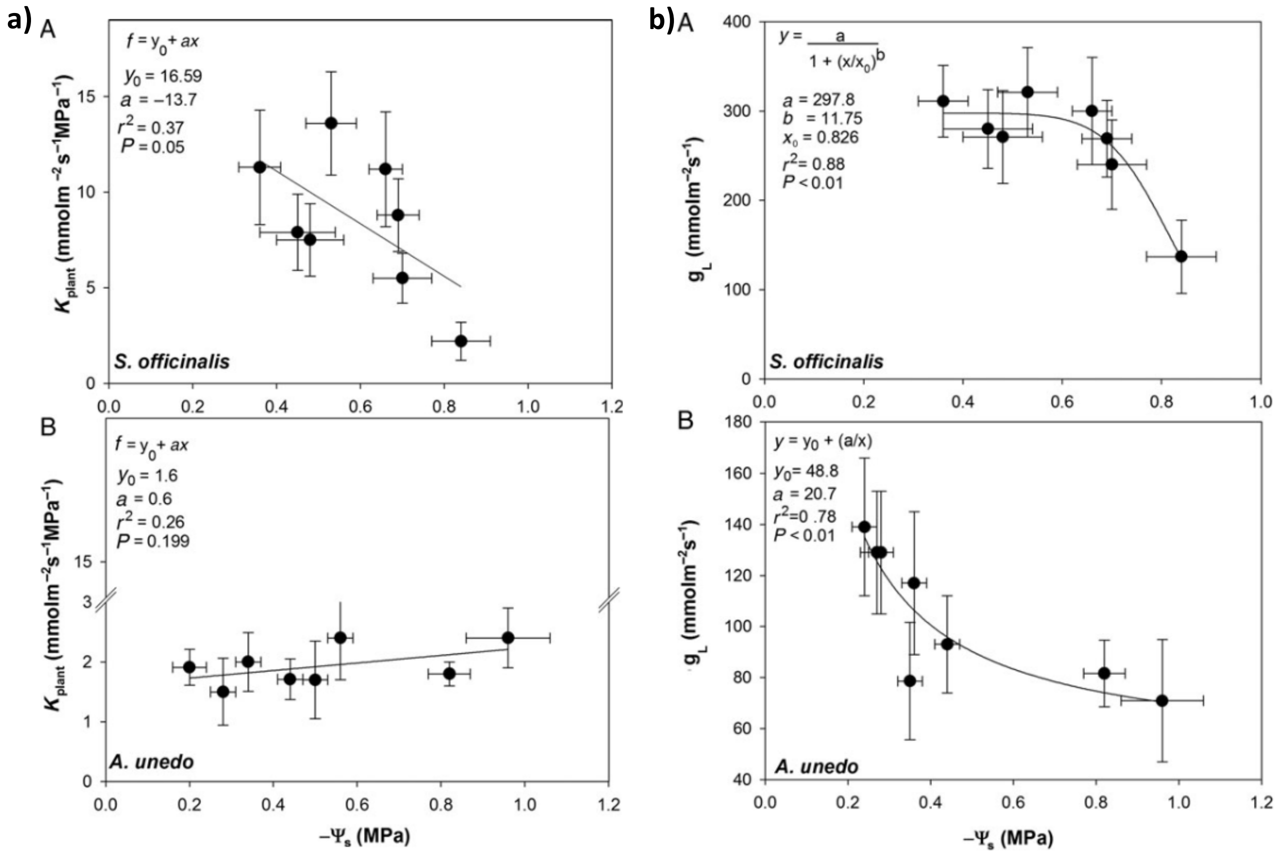


Figure 43: Comparison of the relationship between a) hydraulic conductance and substrate water content and b) between maximum leaf stomatal conductance and substrate water content for anisohydric *Salvia officinalis* and isohydric *Arbutus unedo* in a green roof system in Trieste, Italy (Raimondo *et al.* 2015).

Whereas most studies promote the use of succulents in semi-arid green roof designs, many acknowledge the benefits of pursuing a community-based approach to planting schemes. Often pests and blights will decimate a monoculture-based system, while communities more frequently avoid such devastation, although they may experience the added element of increased plant competition (Gioannini *et al.* 2018). Often there is no difference in the performance of community-based systems relative to monocultures in semi-arid climates. Gioannini *et al.* (2018) designed green roof cells in Las Cruces, NM that included vegetation randomly assigned to either a community- or monoculture-based design in a desert location. Plants were either groundcover, forb, succulent, or grass, and both native and non-native plants were used. After two years, they saw that statistically, plants performed equally well in either a community or monoculture setting in semi-arid climates, but they did not necessarily surpass them in performance, as other studies have found (Gioannini *et al.* 2018). Furthermore, for the succulent group and grass group, native communities and monoculture performed better than non-native alternatives in terms of relative growth (Figure 44). Groundcover and forb groups failed to show this same trend (Gioannini *et al.* 2018). Therefore, native plants might be considered as potentially superior options to non-natives when choosing community members.

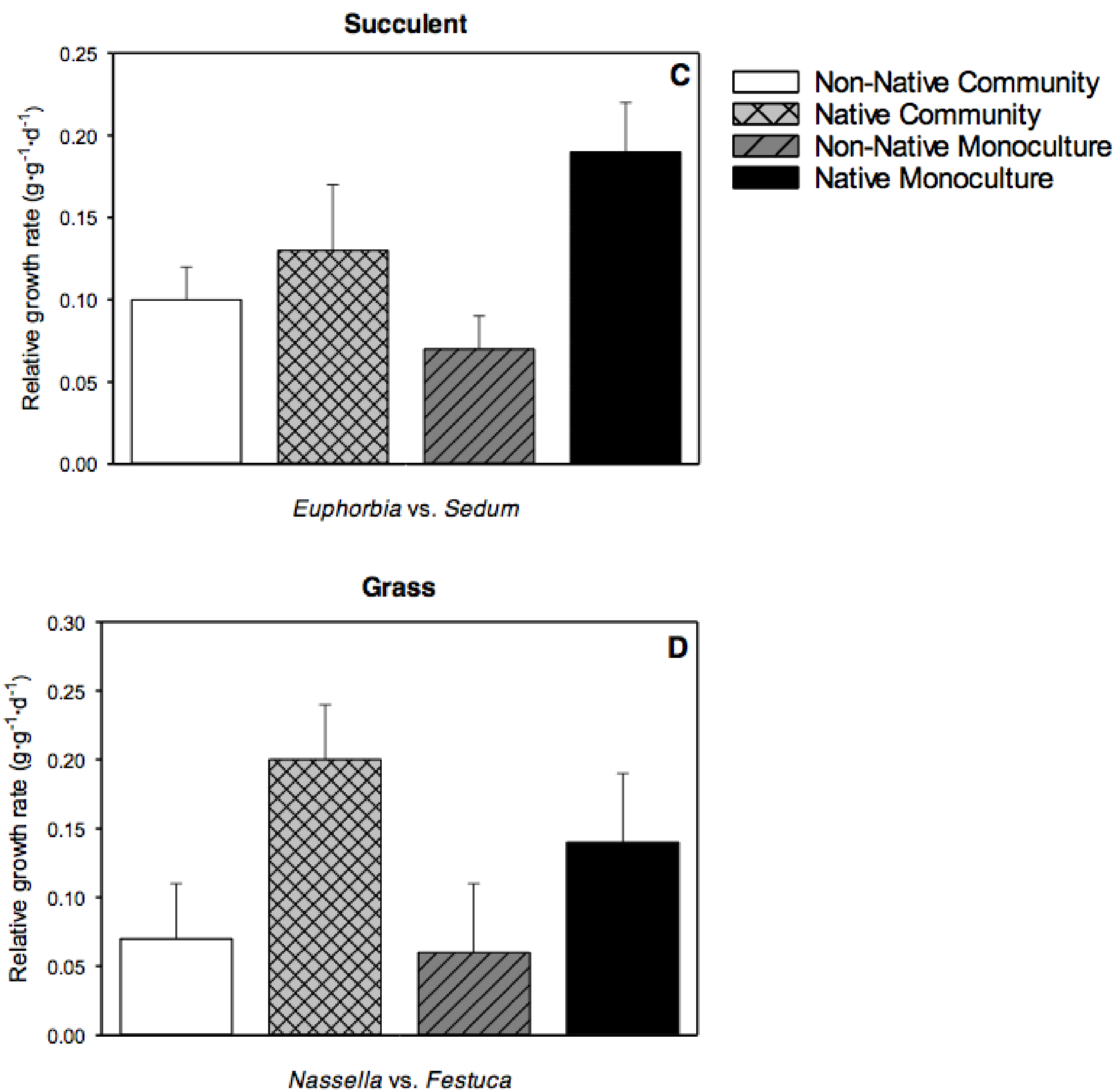


Figure 44: Comparison of growth rates of native and non-native species of succulents (*Euphorbia antisyphilitica* and *Sedum kamtschiaticum, respectively*) and grasses (*Nassella tenuissima* and *Festuca glauca*) in green roof community and monoculture plantings in Las Cruces, NM (Gioannini *et al.* 2018).

In 2012, the EPA conducted a "Mixed Species Study" that combined eight different species into ten planting boxes filled with either GreenGrid growing media or a mix of 50% GreenGrid and 50% zeolite in semi-arid Denver, Colorado (EPA 2012). The EPA found that in the mixed species study, of the eight species tested, only *Eriogonum umbelatum aureum* increased more in plant cover when grown in a community (64%) compared to its growth as a monoculture (12.5%) (EPA 2012). Although the remaining plants all achieved some degree of plant cover within a community setting, they failed to reach the cover they achieved in a comparable study of each species grown individually. The addition of zeolite to the growth media increased the peak cover by an average 26% for herbaceous plants in community beds and 36% for succulent plants in community beds, but many of the species experienced less overwintering performance and decreased percent plant cover when zeolite was added compared to substrate with no amendment (Table 19, Table 20) (EPA 2012). The study concluded that zeolite was generally not advantageous to the growth of herbaceous species within a community setting, and while succulent plant cover might have increased, overwintering success generally remained the same or less (Table 19) (EPA 2012). The presence of multiple species within a green roof system may thus affect the success of one or more particular species on the basis of competition and water use dynamics, and therefore, studies should always examine community interactions in specific semi-arid locations prior to widespread planting (EPA 2012). Furthermore, the effects of different substrate amendments on plant species cover might vary depending on whether a plant is inserted into a community or monoculture environment (EPA 2012).

Species	No amendment	50% zeolite amendment
Allium cernuum	100%	75%
Antennaria parvifolia	40%	40%
Bouteloua gracilis	60%	60%
Eriogonum umbellatum aureum	60%	40%
Herbaceous Mean	65%	54%
Delosperma cooperi	100%	60%
Opuntia fragilis	100%	100%
Sedum lanceolatum	100%	100%
Sempervivum 'Royal Ruby'	100%	100%
Succulent Mean	100%	90%

Table 19: Percentage of herbaceous and succulent species with successful winter survival for substrate with no amendment and substrate with the addition of 50% zeolite amendment in a mixed-species green roof in Denver, CO (EPA 2012).

Species	No amendment	50% zeolite amendment
Allium cernuum	25%	- 43%
Antennaria parvifolia	10%	- 63%
Bouteloua gracilis	- 66%	- 75%
Eriogonum umbellatum aureum	64%	0%
Herbaceous Mean	8%	- 45%
Delosperma cooperi	22%	0%
Opuntia fragilis	50%	64%
Sedum lanceolatum	74%	37%
Sempervivum 'Royal Ruby'	32%	46%
Succulent Mean	44%	36%

Table 20: Change in plant cover for different herbaceous and succulent species in substrate with no amendment and substrate with the addition of 50% zeolite amendment from September 19, 2008 to May 13, 2009 in a mixed-species green roof in Denver, CO (EPA 2012).

Another benefit of community-based planting schemes is that combined use of different plants can provide more effective groundcover, even throughout changing seasons; this, in turn, may allow for enhanced stormwater management (Ondoño *et al.* 2016). In their study of the effects of substrate depth on the performance of the perennial *Silene vulgaris* and annual *Lagurus ovatus* in Santomera, Spain, Ondoño *et al.* (2016) recommend the integrated use of both annual and perennial species within one system in order to maintain continuous plant cover.

Furthermore, annual grasses provide immediate cover and therefore rapid retention performance, as well as higher C and N sequestration compared to perennial options (Ondoño *et al.* 2016). The addition of perennials then compensates for the early wilting of annual grasses in early spring (Ondoño *et al.* 2016). In their study, *L. ovatus* contributed more detritus to the soil media than the evergreen *S. vulgaris*, which likely resulted in variable levels of nutrients in green roof runoff compared to roofs planted with a monoculture (Ondoño *et al.* 2016).

A final consideration that should be taken into account when choosing vegetation for a particular site is the demands of the particular location. For example, some sites may require more ground cover, while others might face more intense sunlight exposure and long periods of heat. Razzaghmanesh *et al.* (2014A) draws attention to the possible vegetation choices that should be chosen based on site-specific features. For instance, although an average 90% of plants survived in the study, different interactions between slope, depth, species, and growing media determined that *Disphyma crassifolium*, or “round-leaved pigface,” is better suited for areas with a need for groudcover (Razzaghmanesh *et al.* 2014A). This is due to the fact that it responds well to a design parameters in terms of maintaining high leaf succulence, great groundcover, and high relative growth compared to the two alternative plants (Razzaghmanesh *et al.* 2014A). On the other hand, the researchers recommend *Brachyscome multifidi*, or “cut-leaved daisy,” as a more suitable choice for areas that required tall plants and plants that have the best survival rates, as these achieve better height and higher overall rates of survival (Razzaghmanesh *et al.* 2014A). Van Mechelen *et al.* (2014) identifies four overarching “clusters” of native Mediterranean plants with “distinct climatic, geographic and soil-related properties” (Table 21) (Van Mechelen *et al.* 2014). Based on the location of any particular green roof, they note that characteristics such as average annual precipitation, average annual temperature, growing degree days, and elevation affect which cluster or type of vegetation should be chosen from within the list (Van Mechelen *et al.* 2014). For instance, green roofs in locations with high average temperatures and low precipitation are more likely to support plants within the most drought-resistant clusters (Van Mechelen *et al.* 2014).

	Cluster 1 △	Cluster 2 ○	Cluster 3 +	Cluster 4 ●
Plots from locations	1, 3, 6–9	2, 4, 5	10–13	14–20
No of plots	84	36	46	87
Characteristic plant traits				
Low values	A, Sand, Aridity, BL	A, Aridity, OM, AAP, BS	Silt, PET, AAT, Dry, BS, BL	Silt, PET, AAT, Dry, BL, C, GDD
High values	C, BS, Silt, Clay, AAT, PET, Dry	Silt, AAT, Dry, C, BL, PET, S	A, C, BS, Sand, Aridity, GDD	A, BS, Sand, EC, OM, AAP, Aridity
ISA results				
#Species	76	112	53	55
#Species significantly belonging to cluster	46	98	40	25
Indicator species	*Brachypodium retusum, Convolvulus cantabricus, Linum strictum, Euphorbia exigua, Dactylis glomerata, Phlomis lychnitis* and *Sedum sediforme*	*Poa bulbosa, Vulpia ciliata, Cerastium pumilum, Saxifraga tridactylites, Minuartia hybrida, Bromus madritensis, Helianthemum nummularium, Erophila verna and Brachypodium distachyon*	*Thymus praecox, Trinia glauca, Koeleria vallesiana, Festuca marginata, Galium pusillum, Potentilla neumanniana, Ononis striata* and *Anthyllis montana*	*Bromus erectus, Thymus serpyllum, Linum suffruticosum, Coris monspeliensis, Galium lucidum* and *Helianthemum italicum*
Phytosociology (Appendix B)	Class *Thero-Brachypodietea*; Association *Brachypodietum ramosi*	Class *Thero-Brachypodietea; (la Crau: association Asphodeletum fistulosi)*	Class *Ononido-Rosmarinetea*; Association *Ononidion striatae*	Class *Ononido-Rosmarinetea*
Vegetative description	*garrigue* vegetation, typical for thermophilic limestone pavements, with a high amount of therophytes and a mosaic of different biological types (Loisel, 1976)	Very species rich, dry and basophilic with many therophytes, mosses and lichens (Molinier and Tallon, 1949)	Typical for mesophilic, open calcareous grasslands and prairies with a low abundance of thero- and geophytes (Van den Berghen, 1963)	(meso)xerophilous *garrigue* and dry grasslands, with a very low abundance of therophytes (Loisel, 1976; Van den Berghen, 1963)

Table 21: Characteristics of different "clusters" of vegetation types defined in Van Mechelen *et al.* 2014. Clusters allow for species to be most suitably chosen for specific green roof locations to ensure highest survival rates (Van Mechelen *et al.* 2014).

The maintenance regime possible for a particular green roof also varies by location and can determine the plants species that can be adequately sustained given a certain degree of oversight. As discussed earlier, low water use is a prerequisite of the plants that can survive in the semi-arid green roof landscape, but a distribution of plants along the spectrum of irrigation needs is often still present. For example, Raimondo *et al.* (2015) concluded that *Arbutus unedo* might be used in situations where very little supplemental irrigation is possible, while *Salvia officinalis* is more appropriate in scenarios where irrigation can be consistently applied and rapid water reduction is a crucial design demand (Raimondo *et al.* 2015).

8. Potential for treatment train application in semi-arid and arid locations

As discussed previously, semi-arid and arid climates feature some unique characteristics that limit the utility and overall performance of distinct bioretention and green roof systems in terms of both runoff volume reduction and quality improvement. This literature review has shown that in terms of runoff reduction, both systems can achieve a relatively high degree of retention. However, both systems rapidly decrease in efficacy in terms of managing high depth or high intensity rainfall events. This is of particular concern in semi-arid and arid climates, as long periods of drought and low annual average rainfall characterize the area for the majority of the year, but “the extreme value of rainfall depth and intensity can be significant” and urbanization can have a significant effect (Guatam *et al.* 2010). Figure 45 shows that for cities like Denver with low annual rainfall (401.6 mm), the hourly rainfall intensity might be substantially higher than that observed in cities with high annual rainfall, such as Seattle (941.6mm) (Guatam *et al.* 2010). For example, Soulis *et al.* (2017) observed only a 2% reduction in stormwater runoff for the second largest event (43.3mm) observed in their year-long study of green roofs in Athens, Greece; this was partly attributed to a relatively high initial moisture content percentage in the substrate at the time (54.8%). Alternatively, during smaller events, green roof designs could achieve up to 100% retention (Soulis *et al.* 2017). Although enlarging systems and modifying substrate and plant characteristics might help maximize storage space, modification of potential storage often means that potential storage space is underutilized during low precipitation events and throughout most of the year (Sims *et al.* 2016). Bioretention systems share the problem of limited retention utility during the large storm events typical of semi-arid and arid climates. Lizárraga-Mendiola *et al.* (2017) found that after the first 10 minutes of a rain event in semi-arid Central Mexico, the rate of infiltration became negative (Figure 24) implying overflow from the then-saturated system (Lizárraga-Mendiola *et al.* 2017).

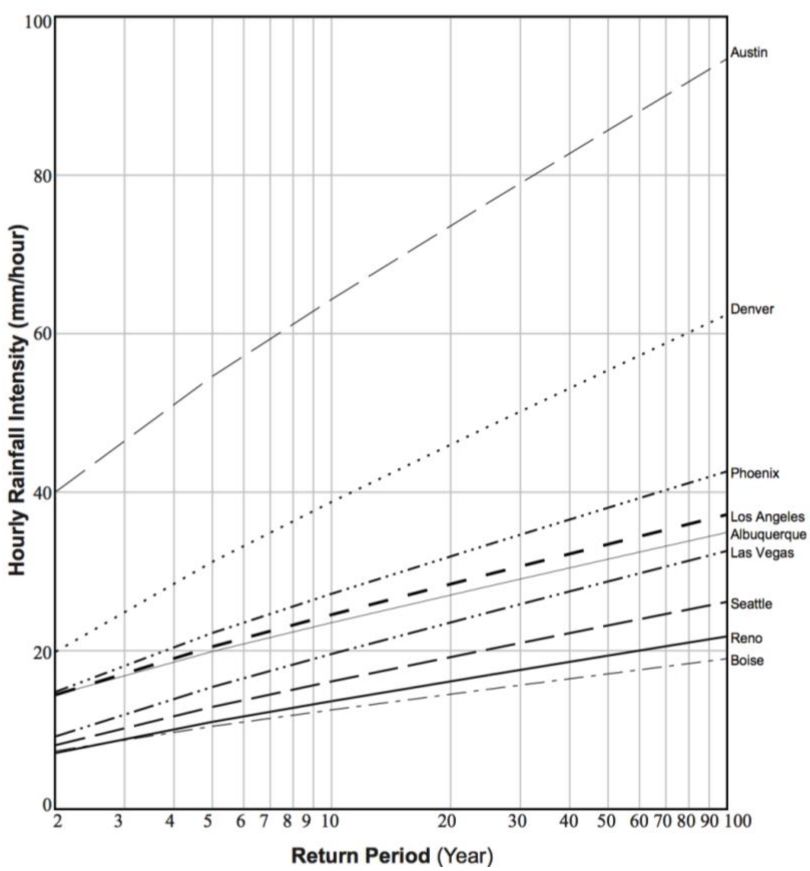


Figure 45: Return periods of rainfall intensities for western U.S. cities with different climates and annual rainfall depths (Guatam *et al.* 2010).

In addition to difficult and extreme environmental characteristics common to semi-arid and arid locations, the underlying mechanisms of each system differ to the extent that they do not achieve both the same degree of quantity and quality control on their own. Although both might be considered "infiltration practices" to some extent in that they both contain vegetation that helps to reduce runoff quantity and uses excess nutrients, the EPA classifies bioretention systems more as a "filtration practice" because it simultaneously reduces runoff volume through infiltration, has the potential to recharge groundwater, and also provides high pollutant removal benefits (EPA 2007). Green roofs might be better considered a "runoff storage practice" in that they are oriented on top of impermeable surface in order to collect water to reduce peak flow and overall volume that might cause erosion or flooding to the surrounding landscape (EPA 2007). As a result of different mechanisms, the relative pollutant removal and runoff reduction capacities of each vary generally and in response to different environmental conditions.

The analysis of existing studies shows that individually, bioretention and green roof systems have only limited capacity to remove contaminants from runoff in semi-arid and arid climates. Bioretention systems have at least general success in improving semi-arid stormwater runoff quality. Bioretention removes TSS in high quantities (Li *et al.* 2011; Jiang *et al.* 2015) and the few studies examining pathogen and metal removal note success in the removal of most of these contaminants, as well (Kim *et al.* 2012; Davis *et al.* 2003). However, limitations exist, and TN and TP are frequently cited as being released in greater quantities in effluent from bioretention systems than they entered in influent (Li *et al.* 2011; Jiang *et al.* 2015), although Houdeshel *et al.* (2015) reported that phosphate removal could be around 50% for both vegetated and non-vegetated cells. Studies show that green roof substrate can leach higher concentrations of some pollutants in effluent compared to control roof runoff (Razzaghmanesh *et al.* 2014A), although some studies disagree and find that TSS and metals can be retained to some extent relative to aluminum and asphalt roofs (Gnecco *et al.* 2013). Regardless, green roofs are less effective as pollutant removal devices relative to bioretention systems.

Given the aforementioned reasons, in this section I propose that bioretention and green roof systems be combined with each other or with other LID technologies to form treatment trains to manage stormwater runoff. Treatment trains combine distinctive chemical, biological, and physical mechanisms of LID technologies to retain water on-site to reduce volume and improve quality to a greater extent than singular LID systems (Revitt *et al.* 2014). I also propose that additional subterranean storage or other rain harvesting techniques be implemented in order to store excess runoff from large rain events on-site for further quality improvement and for use in irrigating the vegetated elements of the systems. This section first defines a treatment train and summarizes existing literature that compares the performance of treatment trains relative to singular LID systems or conventional stormwater management techniques. It also discusses the potential for other LID technologies to be combined with bioretention and green roofs to further manage stormwater. It briefly summarizes existing treatment train studies carried out in semi-arid and arid climates. Reviewed studies have primarily been conducted in temperate or humid climates with few studies taking place explicitly in semi-arid and arid climates. Therefore, this chapter concludes with a call for future research and pilot studies investigating treatment train performance coupled with water harvesting practices in semi-arid and arid climate locations.

8.1 Definition of treatment train and utility of application

A treatment train may be defined as "two or more treatment systems used in combination to maximize the availability of different pollutant removal processes," as well as to reduce the quantity of runoff (Revitt *et al.* 2014). The *Minnesota Stormwater Manual* distinguishes between two categories of treatment trains (Figure 46): the “LID configuration” and the “traditional development configuration” (Minnesota Pollution Control Agency). First used as a phrase in the 1980s, “treatment train” initially referred to any stormwater quality and quantity management technique using a series of either natural or artificial landscaping features (Perini and Sabbion). Today, multiple LID techniques incorporated into a continuous system to treat the same impervious runoff on-site is acknowledged as the most effective method to improve stormwater quality and to reduce quantity through a variety of different mechanisms (Rushton 2001). Treatment trains are frequently placed next to parking lots and other areas with high impervious surface area and heavy contamination, as many singular systems fail to adequately remove pollutants and reduce volumes adequately (Minnesota Pollution Control Agency).

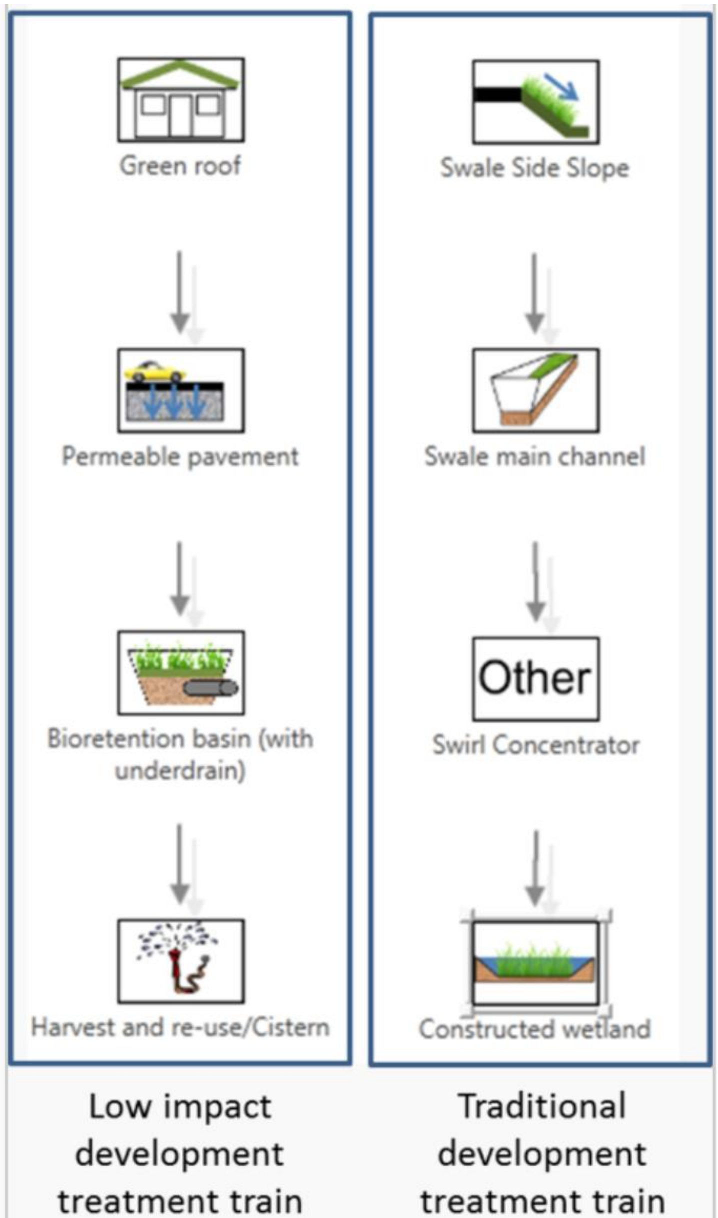


Figure 46: Diagram showing examples of a LID treatment train and a "traditional development" treatment train (Minnesota Stormwater Manual).

8.2 Runoff volume reduction performance of bioretention treatment trains

Although exact values vary according to each study, treatment train design, and location, all reviewed studies that compared treatment train performance to singular LID system performance observed an increase in runoff retention when multiple systems were combined. In 2001, Rushton *et al.* compared the performance of an early LID parking lot treatment train design in Florida with several control alternatives. The treatment train consisted of permeable pavement and a bioswale, and this was compared to variations of asphalt and concrete with and without swales. They found that the treatment train performed best in terms of both hydrology and quality parameters relative to traditional paving techniques, with a 32% reduction in runoff compared to other pavements coupled with swales (Rushton *et al.* 2001). They noted that the performance of this treatment train was reduced slightly for larger rain events (Figure 47), which they attributed to soils being more saturated during the rainy season's large, high-intensity storms (Rushton *et al.* 2001).

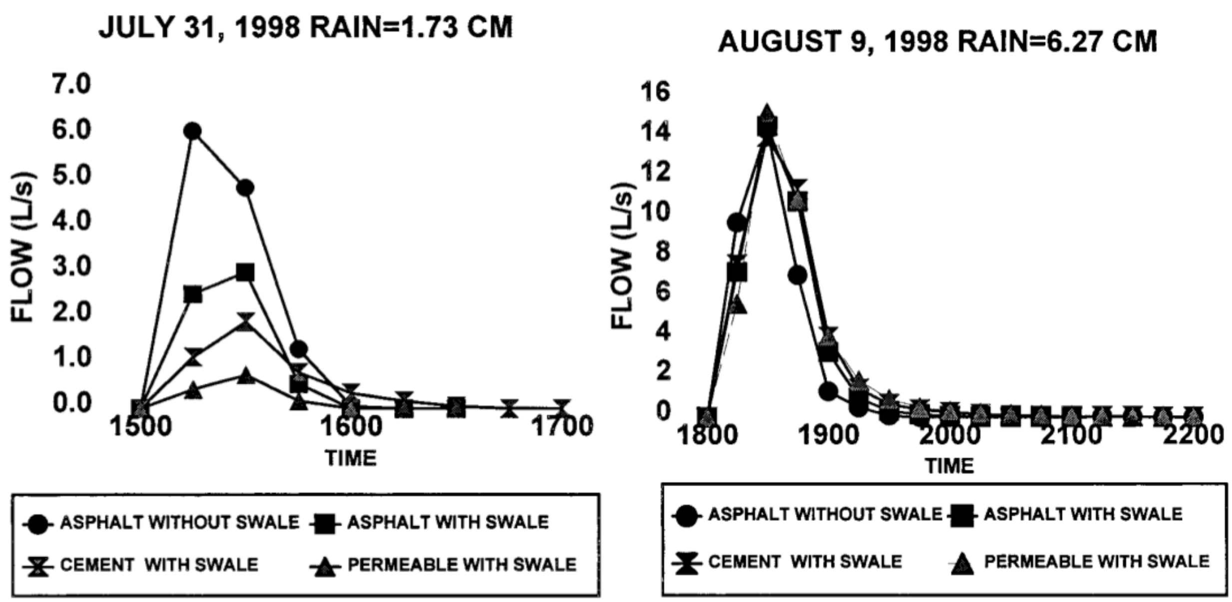


Figure 47: Comparison of different treatment train performances in small (depth=1.73cm) and large (depth=6.27cm) rain events. Pavement types include asphalt, cement, and pervious paving, and systems were either coupled with a swale or were not coupled (Rushton *et al.* 2001).

Brown *et al.* (2012) looked at a similar train utilizing permeable concrete and a bioretention cell in Nashville, NC for 17 months at a location with a high water table. Because the site featured a high water table, the cell was only 1.6 feet deep, but volume reduction for the system was still remarkably high. Outflow occurred for 33 out of 80 storm events, and the average volume reduction for the entire train was 69% (Brown *et al.* 2012). Compared to individual LID systems, the treatment train had significantly lower rates of peak outflow and discharged almost half as much annual outflow as an individual bioretention system (Brown *et al.* 2012). As expected, the authors observed that with good storage volume, which was common as antecedent dry days increased in number, runoff volume was almost completely reduced. Figure 48 reflects the impact of longer antecedent dry days in reducing outflow volumes, although it shows that large rainfall depth still produces relatively high outflow volume (Brown *et al.* 2012).

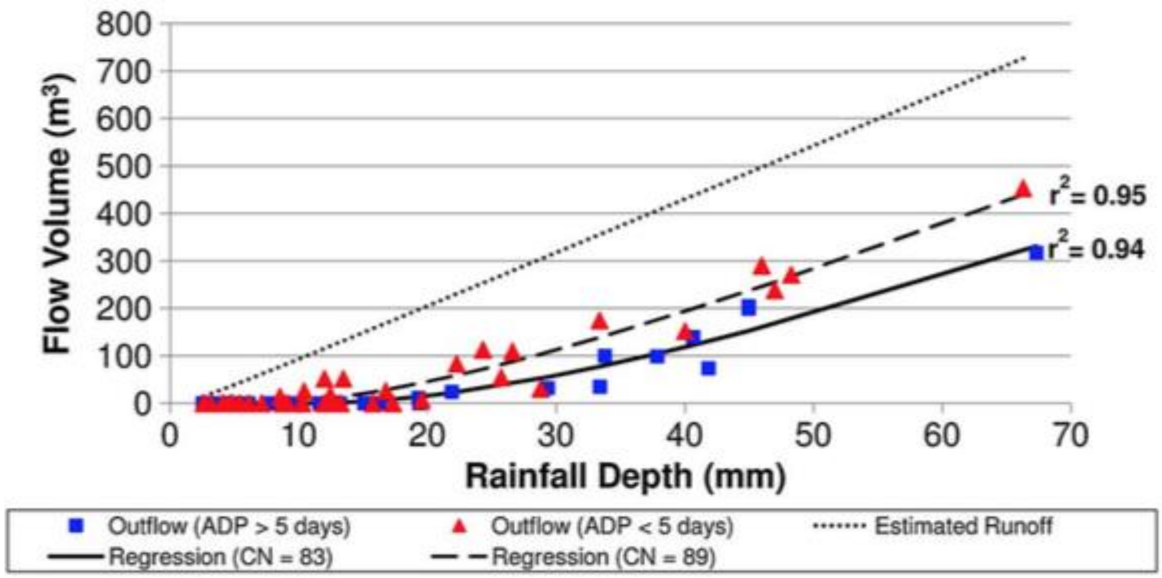


Figure 48: Relationship between flow volume and rainfall depth for two outflow events of different antecedent dry day periods (ADP) (Brown *et al.* 2012).

Braswell *et al.* (2018) also looked at the hydrological and quality effects of treatment trains incorporating permeable pavement, but they examined the interaction between the pavement and a Filterra system in a North Carolina parking lot for 22 months. Filterra systems are filter boxes including a tree or shrub, planting soil, an underdrain, and other features (Figure 49), frequently placed next to curb cuts to filter runoff from parking lots or other urban surfaces (Imbrium Systems Inc.). While Filterra systems are not synonymous with bioretention basins, both utilize similar mechanisms of biofiltration prior to expulsion via underdrain, and Filterra systems are particularly versatile given their small overall dimensions. While the permeable pavement reduced the runoff volume by 56%, 38% of the rain at the site failed to receive treatment and continued on as surface runoff because of clogging and underestimation of the system's treatment area (Braswell *et al.* 2018). Although the Filterra system played a minimal role in runoff reduction generally (6% volume reduction), in cases where underdrain flow from the permeable pavement took place, the Filterra system provided extra storage space for runoff; this would be of particular utility in storm events with high rainfall depth (Braswell *et al.* 2018).

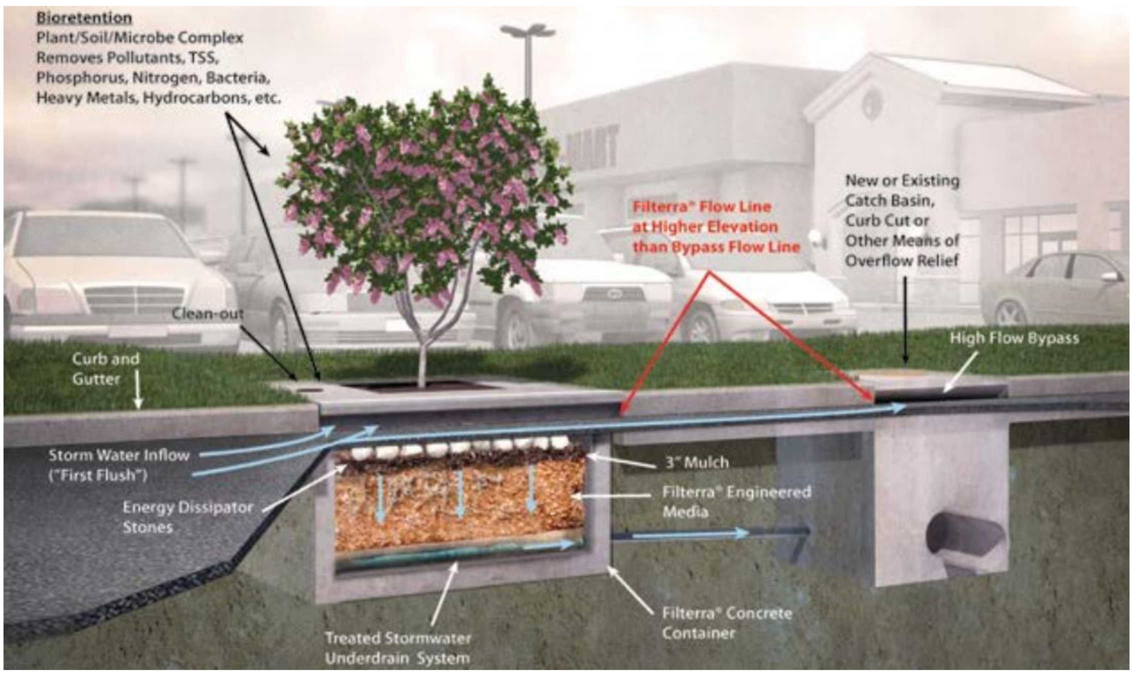


Figure 49: Diagram of a standard Filterra system (Imbrium Systems Inc.).

Flanagan *et al.* (2016) observed the pairing of a bioretention swale and filter strip for stormwater management modelling using SWMM in France. They determined that the filter strip and vegetated hill were responsible for the bulk of the runoff retention (89-90%), but the bioretention swale still helped keep the maximum runoff coefficient (C_R) at 35% overall (Flanagan *et al.* 2016). As seen in Rushton *et al.* (2001), with increasing rainfall intensity, C_R

increased, and the researchers also found that variable hydraulic conductivities of different soil types affected whether or not water was infiltrated by the filter strip or continued on to be treated and intercepted by the bioretention system (Flanagan *et al.* 2016). For example, the lower hydraulic conductivity of clay (k= 0.254 mm hr^{-1}) results in a higher runoff coefficient than sandy loam (k=23.8 mm hr^{-1}); subsequently, less water is infiltrated by the filter strip when the soil type of the system is heavy in clay content (Figure 50) (Flanagan *et al.* 2016).

The 2015 Jia *et al.* study also evaluated the paired use of a filter strip and bioretention cell, but they expanded the LID practices involved to include a bioretention basin, three grass swales, two infiltration pits, and a constructed wetland. The study took place in the humid subtropical climate of southern China and evaluated the retention and treatment performance of both individual components and the train as a whole. They found that the bioretention cell and swales both reduced both runoff peak flow (66% and 44.3%, respectively) and runoff volume (62.2% and 36.2%, respectively) significantly, but the bioretention cell performed much better than the swale, particularly during large storm events (Figure 51) (Jia *et al.* 2015).

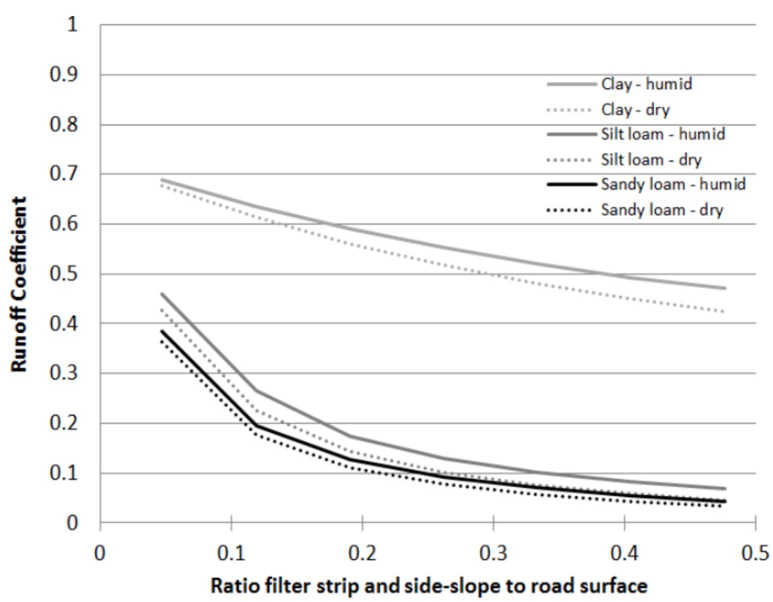


Figure 50: Relationship between runoff coefficient and ratio of pervious LID to impervious surface area for different soil types under different degrees of initial humidity (Flanagan *et al.* 2016).

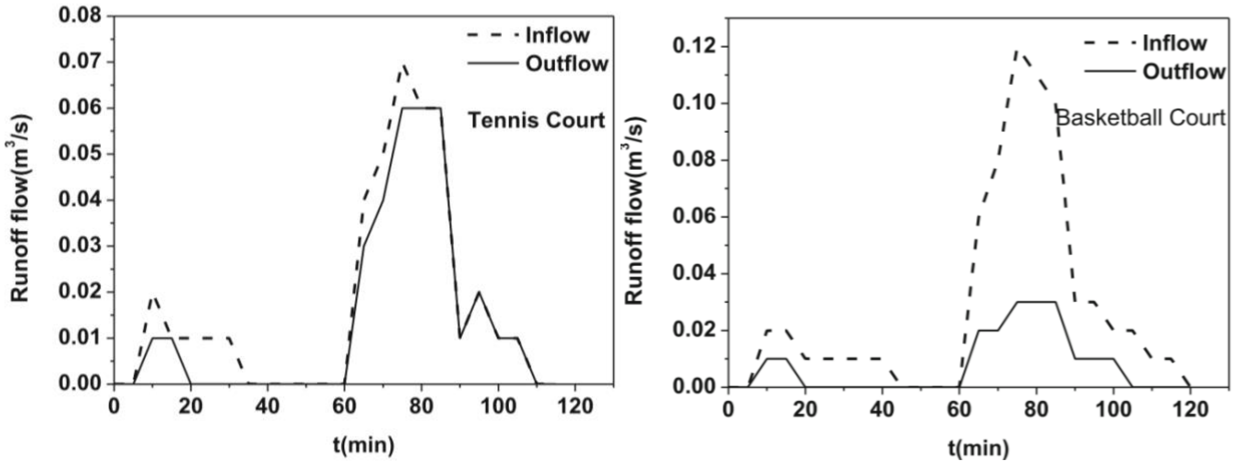


Figure 51: Runoff hydrographs for a large storm event on May 27, 2012 in southern China for treatment train components including a) two grassed swales and b) a bioretention cell (Jia *et al.* 2015).

Wilson *et al.* (2014) compared a conventional train with swales and a detention pond to a LID treatment train with swales, bioretention, infiltration pits, and cisterns in North Carolina. The study revealed that for large events of 79.5mm, 89% volume reduction was achieved and that for smaller 16.5mm events, 100% reduction was possible with the LID train (Wilson *et al.* 2014). Wilson *et al.* (2014) acknowledged that both conventional and LID trains were over 98% effective at reducing peak flow, but the cumulative runoff depth reductions differed between the two trains significantly; the LID treatment train reduced runoff by an average 97.0% while the conventional train only achieved 49% reduction (Figure 52) (Wilson *et al.* 2014). Furthermore, the study speculated that much of the runoff retained by the LID train helped to recharge groundwater supply (Wilson *et al.* 2014). Almost 55% of the water resources of the arid and semi-arid desert United States Southwest comes from groundwater reserves; therefore, finding new means to resupply uncontaminated groundwater is of both economic and environmental importance and makes LID treatment trains particularly attractive alternative to traditional best management practices (BMP) trains (EPA 2010).

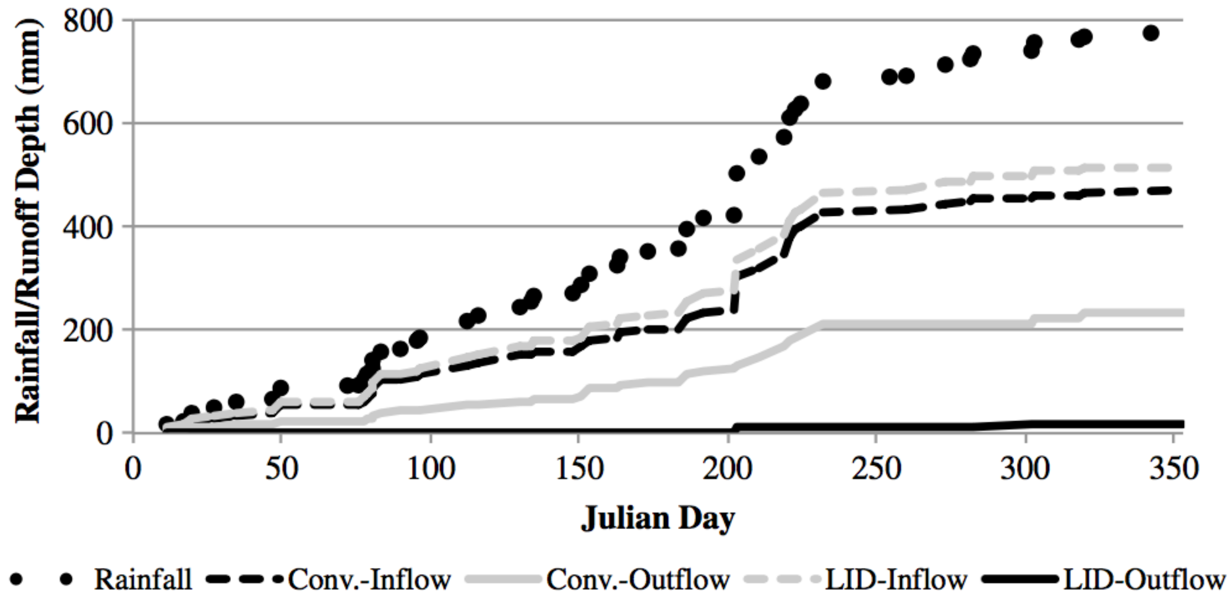


Figure 52: Comparison of runoff reduction performance of a conventional treatment train and LID treatment train over the course of a one-year study (Wilson *et al.* 2014).

Another approach to reducing stormwater runoff is to incorporate storage components into treatment train design, often through either cistern use or subterranean storage. Doan *et al.* (2017) designed a train at the University of Maryland (Figure 53) that channeled runoff into a 3-cell, terraced bioretention system prior to potentially entering a cistern installed proximal to the cells. The water could then be pumped back into the bioretention system or used as irrigation water (Doan *et al*. 2017). The study determined that this particular design would produce overflow for events more than 0.75 cm in rainfall depth in 2 hours, and for the observation period from March 10-November 10, the overflow was 89%, infiltration in the bioretention system was 8%, and cistern storage was 3% (Doan *et al.* 2017). This system had limited volume reduction because it was undersized due to “site restraints” and failed to treat 40% of the drainage area, which is a relatively common reason for bioretention failure (Doan *et al.* 2017). The researchers also agreed with other studies that antecedent moisture content, cistern size, storm characteristics, and soil characteristics affected the performance of the train in reducing runoff volume, and a summer of regular rainfall events likely also impacted volume reduction negatively (Doan *et al.* 2017).

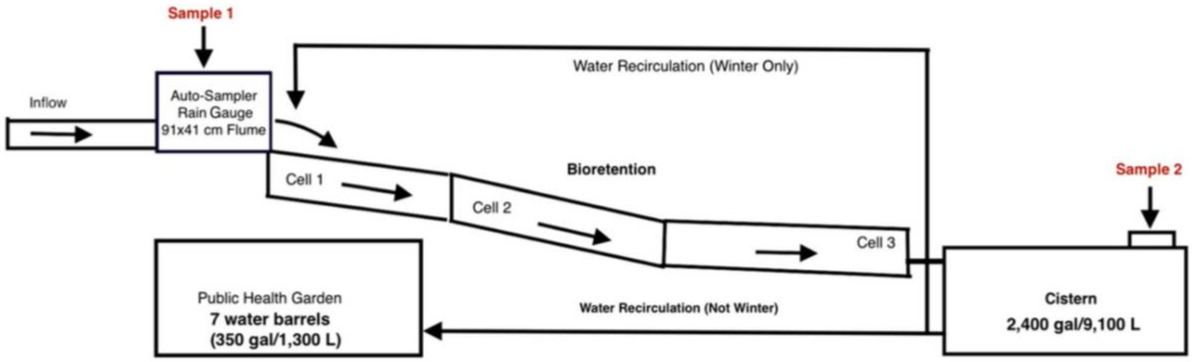


Figure 53: Bioretention and cistern storage treatment train designed and evaluated in the Doan *et al.* 2017 University of Maryland study (Doan *et al.* 2017).

8.3 Runoff quality improvement performance of bioretention treatment trains

The value of treatment trains incorporating bioretention cells extends to runoff quality improvement perhaps to an even greater extent than volume reduction. Overall, treatment trains incorporating bioretention cells produce runoff with very low EMCs, and frequently different parts of any observed treatment train remove different pollutants from runoff to collectively improve overall quality. In their analysis of a train with a buffer strip, a bioretention cell, grass swales, two infiltration pits, and a constructed wetland, Jia *et al.* (2015) determined that the bioretention cell had generally positive removal, but TSS and TP were negative for the first year before the system stabilized. Collectively, however, the treatment train achieved great removal rates (Figure 54); the authors attributed this to the fact that even though the bioretention cell actually added TP and TSS to the system, the flat and long swales and infiltration pit did not contain any high-nutrient planting soil, and this allowed for the TP to be well-absorbed prior to effluent leaving the system (Jia *et al.* 2015).

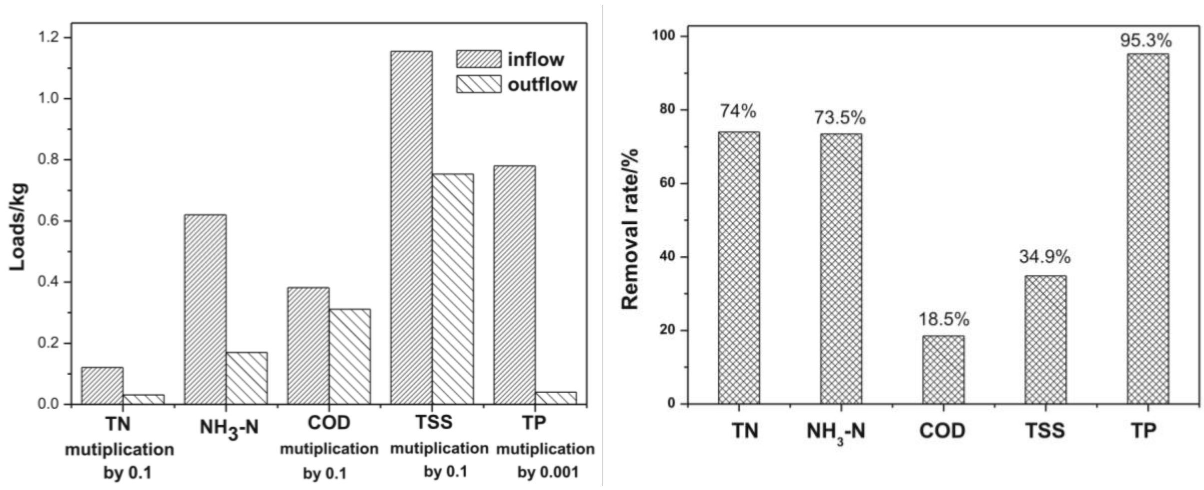


Figure 54: Pollutant removal rates of the Jia *et al.* 2015 treatment train for different contaminants (Jia *et al.* 2015).

Wilson *et al.* (2014) noted that compared to conventional treatment trains, a LID treatment train including swales, a bioretention cell, infiltration pits, and a cistern in Raleigh, North Carolina significantly lowered pollutant loads in discharge by 23 to 85 times compared to a conventional development train (Table 22). The researchers attributed this to the LID train reducing runoff volume much more significantly than the study's conventional train, resulting in a much lower quantity of pollutants exiting the system within runoff (Wilson *et al.* 2014). They concluded that these results reinforced "the importance of both pollutant and hydrologic mitigation to maximize function" (Wilson *et al.* 2014). Additionally, the "overdesign" of this particular system resulted in EMCs lower than other studies using similar LID approaches within treatment trains (Wilson *et al.* 2014).

Pollutant	Conventional development (n = 20) (kg/ha/year)	LID (n = 20) (kg/ha/year)	Ratio: conventional/LID	Test statistic[a]	p
TKN	3.43	0.09	37	$M = 9$	<0.0001
TAN	0.46	0.02	24	$M = 9$	<0.0001
NO_X	1.10	0.05	23	$M = 9$	<0.0001
Ortho-P	0.16	0.002	85	$t = 7.75$	<0.0001
TSS	96.9	1.56	62	$M = 9$	<0.0001
TN	4.52	0.14	32	$M = 9$	<0.0001
TP	0.53	0.01	52	$S = 104$	<0.0001

[a]Test statistics *M* (Sign test), *t* (Student's *t* test), and *S* (Wilcoxon signed rank test) depended on the normality of the data.

Table 22: Average pollutant loadings in outflow runoff of the conventional development train and LID treatment train evaluated in Wilson *et al.* 2014 (Wilson *et al.* 2014).

Despite the minimal volume reduction of the Doan *et al.* (2017) multi-celled bioretention and cistern treatment train at the University of Maryland, water quality improvement similar to that observed in other studies was achieved. TSS removal varied from 78% to 99%, and other water quality parameters including TN, TP, and various metals all improved significantly compared to incoming water, as well (Doan *et al.* 2017). One shortcoming of the system was its inability to lower the electrical conductivity of runoff after winter, as snowmelt often contains deicing agents that impact water quality severely. However, a suggested solution is recycling water through the bioretention cells for further volume reduction rather than harvesting it for irrigation use (Doan *et al.* 2017).

Studies pairing bioretention basins with permeable pavement in treatment trains also appear to be effective in improving the overall quality of stormwater runoff. Rushton *et al.* (2001) determined that the inclusion of a swale alongside a permeable pavement system in Florida resulted in 50% less pollution emitted from the system. The swales assisted in reducing pollutant loads by 75% for the metals and TSS alone compared to the baseline scenario of an asphalt parking lot design with no swales, which the researchers credited partly to the ponding area and opportunity for gradual infiltration that the LID treatment train provided (Rushton *et al.* 2001).

Brown *et al.* (2012) observed that positive pollutant reductions in outflow varied from 49% for TN to 89% for TSS using a pervious concrete-bioretention basin train in Nashville, NC. However, the researchers credited these reductions to the decrease in runoff volume by 69% (Brown *et al.* 2012). Furthermore, orthophosphate (Ortho-P) and nitrate and nitrite ($NO_{2,3}$-N) actually increased overall in effluent concentrations (Brown *et al.* 2012). Compared to individual LID systems, the treatment train treated an extra 10% of the runoff volume, but base flow influx from the groundwater actually increased TN in output from the system (Brown *et al.* 2012). The researchers mention that the lack of IWS layer prevented a denitrification mechanism for the system, and this coupled with an early lack of vegetation likely explains the high TN output (Brown *et al.* 2012). High water tables, such as the one present in this study, might be a slight challenge to implementation of treatment trains in semi-arid and arid climates. Due to the failure of the impermeable liner used within the system, groundwater moved into the bioretention basin and potentially contributed to the increased movement of TN to surface water (Brown *et al.* 2012). However, compared to single systems under the same site conditions, most water quality variables improved when the treatment train components were evaluated collectively (Table 23) (Brown *et al.* 2012).

Sampling location	TAN	TKN	$NO_{2,3}$-N	TN	Ortho-P	TP	TSS	Volume (m^3)
	Pollutant load							
Asphalt parking lot runoff (kg/ha · year)	1.99	5.93	1.77	7.70	0.10	0.58	278	15,353
Storm outflow (kg/ha · year)	0.15	1.80	2.14	3.94	0.12	0.28	30.7	4,697
Reduction (%)[a]	93	70	−21	49	−14	51	89	69
Reduction (%)[b]	90[c]	48[c]	−35	39[c]	−51	39[c]	86[c]	67[c]
Base flow (kg/ha · year)	0.09	0.73	7.96	8.69	0.05	0.12	5.6	11,338
Total outflow (kg/ha · year)	0.24	2.53	10.10	12.64	0.16	0.40	36.3	16,035
Reduction (%)[a]	88	57	−471	−64	−60	30	87	−4

[a]Arithmetic reduction computed from differences in annual export.
[b]Reduction in storm event loads computed statistically by using the least squares means analysis.
[c]Indicates significance at the 0.05 level.

Table 23: Pollutant loads and reductions over the course of a year in the Brown *et al.* 2012 pervious concrete-bioretention treatment train evaluated in Nashville, NC (Brown *et al.* 2012).

8.4 Volume reduction and treatment performance of green roof treatment trains

Very few studies evaluate the performance of green roofs within treatment trains, but a couple suggest that they might supplement other LID technologies well in both volume and quantity control. Bastien *et al.* (2010) discussed the use of different BMPs and LID techniques within treatment trains to compare their performance against each other and against “end of pipe” solutions. They determined that, based on other studies’ findings, green roofs are the most cost-effective method to store runoff within a treatment train (Bastien *et al.* 2010). A study by the University of Florida investigated the performance of a treatment train including a green roof system paired with a cistern and bioretention cell for the four months of monitoring. They discovered that neither the cistern nor the green roof experienced overflow in the amount of time that they were in use, which confirms that the treatment train was effective at reducing runoff volume (Kelly *et al.* 2007). Future research needs to study the impact of including green roofs in treatment trains to aid in stormwater volume retention, as a significant research gap exists in this area.

Despite the fact that the primary function of a green roof is storage within treatment trains, they might also be paired with filtration technologies, such as bioretention, to effectively reduce some contaminant concentrations. Revitt *et al.* (2014) examined the ability of individual BMPs to reduce pollutant levels in parking lot runoff in the United Kingdom. They considered three treatment train scenarios: A) a permeable paving system collects runoff before channeling it to the bioretention cell for further treatment, B) the runoff from a green roof is moved to a bioretention cell for treatment, and C) both stormwater management techniques are implemented (Revitt *et al*. 2014). Comparing the treatment performance of each scenario on the basis of the site pollution index (SPI), they found that although scenario B reduced TSS and TP to acceptable levels, it failed to lower Zn appropriately (Figure 55) (Revitt *et al.* 2014). Scenario A had higher rates of removal overall, but scenario C actually achieved the lowest SPI overall (Revitt *et al.* 2014). Interestingly, the researchers concluded that despite the somewhat superior performances of scenarios A and C, scenario B was actually the best-suited to this particular type of site; given that heavy vehicles frequented the area, LID trains involving permeable pavement might be less conducive to a car park’s stormwater management approach over time due to compaction and clogging limiting treatment performance (Revitt *et al.* 2014). Such an observation speaks to the sensitivity of stormwater management techniques to site constraints when choosing the best technologies for a specific location. Finally, the study also mentioned that green roofs are effective for frequent storms, but with long-term scenarios of accumulated surface contaminants, there is likely a need to add more BMPs to the treatment train (Revitt *et al.* 2014).

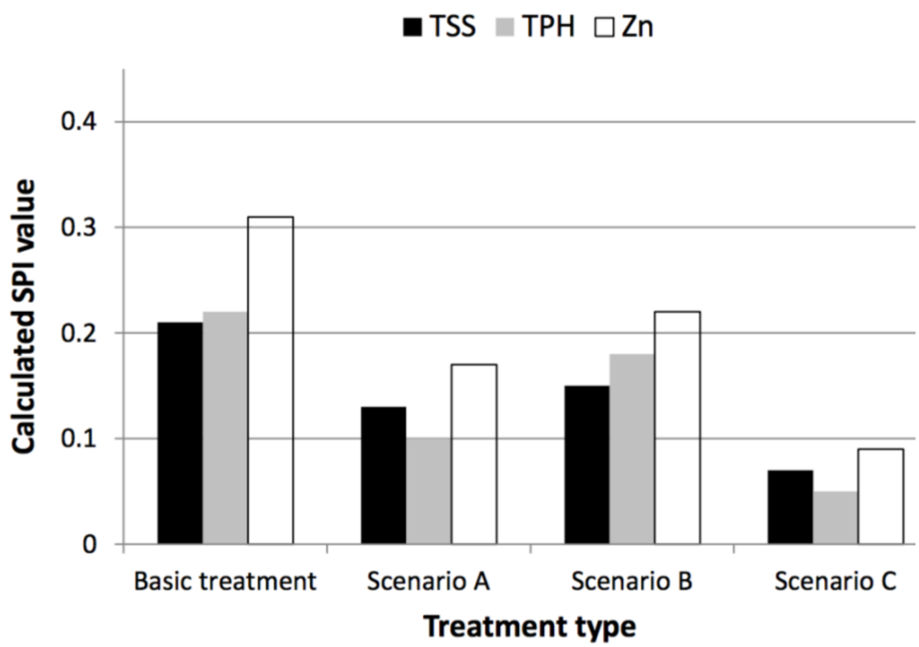


Figure 55: Site Pollution Index (SPI) values determined for 4 treatment train scenarios treating parking lot runoff in the United Kingdom. Evaluated water quality indicators included TSS, TPH, and Zn (Revitt *et al.* 2014).

8.5 Research in the application of treatment trains in semi-arid and arid climates

Studies examining the performance of treatment trains are still relatively few, and the huge majority of these are conducted in temperate and humid climates. Despite this topic being relatively new in garnering interest, a few recent publications have acknowledged the potential for treatment train implementation in semi-arid and arid climates, and some have even begun to investigate their performance. Hunt *et al.* (2012) recommends that the bowl depth of bioretention systems be increased and the basins be augmented with other BMP techniques to form a treatment train in order to manage peak storm runoff, regardless of climate. David *et al.* (2015) evaluated a train of four bioretention cells and one swale in reducing both common (suspended sediment concentration (SSC), Cd, Cu, Ni, Pb, Zn, and polycyclic aromatic hydrocarbons (PCBs)) and unusual pollutants (total Hg and polychlorinated biphenols (PCBs)) in semi-arid Daly City, CA. Hydrologically, the flow volume could only be reduced by about 10%

for the system at the time of the study (David *et al.* 2015). Limitations to flow reduction in this trial were likely the higher than normal annual precipitation of the study period and the presence of clay soils that reduced infiltration (David *et al.* 2015). However, the researchers speculated that this volume reduction would likely increase substantially in future years as plants matured, and this shows potential utility of this treatment train to contribute to quantity control of runoff in semi-arid regions (David *et al.* 2015). Similar systems likely would be more effective in areas that feature soils with higher hydraulic conductivity and in years with drier winters and longer periods between rain events, which are typical of semi-arid regions.

David *et al.* (2015) also evaluated water quality improvement in their bioretention-swale treatment train, the volume reduction performance of which was discussed previously. Effluent quality improved overall for all pollutants monitored with the exception of MeHg, which might have increased due to poor aeration (Table 24) (David *et al.* 2015). TSS loading rates were reduced by 74%, and mean metal loading rate reductions ranged from 59% to 90%, which is comparable with well-functioning systems in temperate climates (David *et al.* 2015). Two pollutants less commonly assessed in stormwater are PAH and PCBs, the loading rates of which this treatment train reduced by 97% and 82%, respectively (David *et al.* 2015).

Parameters	Concentration (%)	Flow weighted mean concentration (%)	Loading rate (%)
SSC	29	82	84
MeHg	–152	–87	–56
HgT	18	78	59
HgD	50	55	64
Copper	83	94	96
Zinc	93	97	98
Nickel	20	74	79
Lead	51	86	87
Cadmium	84	96	96
PCBs	44	78	82
PAHs	90	97	97
OCDD	97	98	97

Table 24: Water quality values for various pollutants following treatment by a bioretention-swale treatment train in Daly City, CA (David *et al.* 2015).

In addition to reducing water quantities to prevent flooding and erosion and improving quality to protect waterways and health, semi-arid and arid climates would benefit from the additional water resources that might be provided through runoff conditioning by different treatment trains. In an early study of stormwater management practices, Lazarova *et al.* (2001) observed the disparity between available water resources and the need for water in Israel and other semi-arid Mediterranean countries. They identified the interest and investment in wastewater reuse in Israel, which composed 20% of Israel's total water supplies in 1994, as stemming from this need (Lazarova *et al.* 2001). Chowdhury (2015) examined whether bioretention systems in arid areas could remove pollutants from grey water sources. Using small bioretention test units in arid Al Ain, UAE, he found that this technology was successful for most pollutants but concluded that bacteria, pH, and K ions would be better treated by the use of a treatment train (Chowdhury 2015).

Most recently, Kazemi *et al.* (2018) examined a bioretention-permeable pavement paired system in the semi-arid climate of Adelaide, Australia. They evaluated its efficacy in reducing two concentrations of salinity (500 mg/L and 1500 mg/L) for the sake of making the runoff suitable for irrigation use and storage for future use. They found that both individual systems and a combined system reduced the sodium absorption ratio (SAR), an index "representing the risk of soil damage due to excessive sodium in irrigation waters," to within a reasonable range (Figure 56) (Kazemi *et al.* 2018). However, when considering individual parameters, often the combined system worked more effectively to improve water quality. For example, the limestone material of the permeable pavement added Ca and Mg to the runoff prior to filtration through the bioretention system, which effectively buffered the highly saline water added to the system (Kazemi *et al.* 2018). Furthermore, pH increased as it passed through the permeable pavement layers (Figure 56), and here the bioretention basin acted as a buffer by lowering pH, regardless of whether the degree of salinity was low or high (Kazemi *et al.* 2018). Two measures of water quality that deteriorated in this study were turbidity and electrical conductivity. Turbidity increased significantly after permeable pavement was run through the bioretention basin, but the researchers attributed this to the new state of the system and the fine particles washing out from the basin prior to the system stabilizing (Kazemi *et al.* 2018). Despite increased electrical conductivity, levels were still within the range of "good" and "permissible" for low and high concentrations of salinity, respectively (Kazemi *et al.* 2018). Therefore, both systems individually or combined can still be used as effective storage without compromising quality (Kazemi *et al.* 2018).

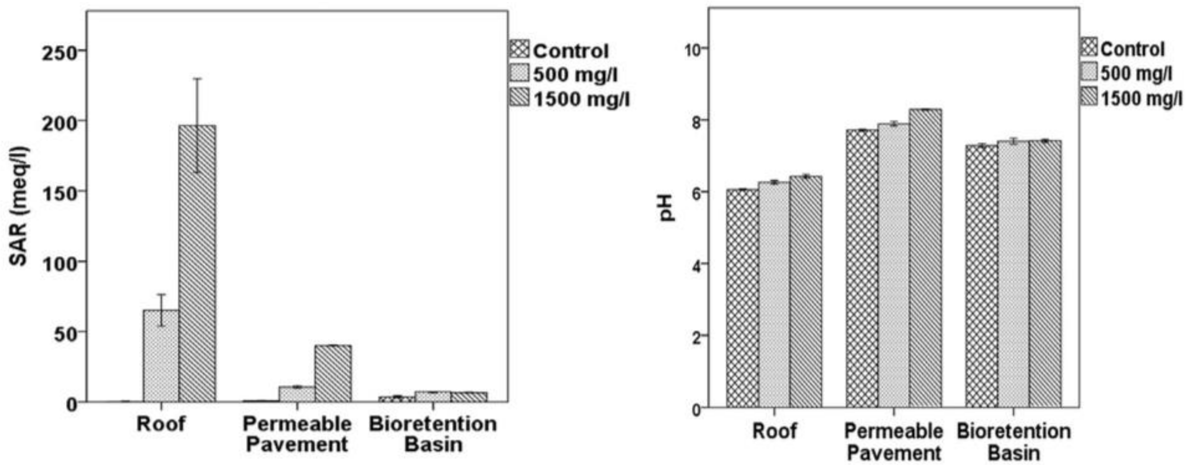


Figure 56: Effect of treatment train permeable pavement and bioretention on a) the SAR and b) the pH of roof runoff after adding concentrations of 0 mg/l (Control), 500 mg/l, and 1500 mg/l of salinity to the systems in semi-arid Adelaide, Australia (Kazemi *et al.* 2018).

9. Conclusions and future research

Based on this literature review, bioretention and green roofs have significant potential as LID stormwater management techniques in semi-arid and arid climates. Bioretention systems provide both substantial stormwater volume retention and pollutant removal. Bioretention systems show great potential to reduce and delay stormwater peak flow, although heavy rainfall, long rainfall, or deep rainfall, as well as undersized systems, have the potential to produce overflow quickly in these climates. Pollutant removal is also high for bacteria, nutrients, and metals, although TP and TN removal is highly variable amongst systems. Variations in design including the addition of an IWS layer, strategic selection of vegetation, and maintenance practices can significantly affect the utility of these systems in semi-arid and arid climates.

Green roofs provide an alternative service to bioretention in that they primarily act to store water for gradual volume reduction. Like bioretention systems, the retention reduction percentage decreases for heavy or large storm events, but overall, semi-arid and arid green roofs provide greater cumulative stormwater retention than their counterparts in humid climates.

Deeper substrate and the inclusion of plants enhance the retention performance of these systems, but plant debris often contributes to extra nutrients leaching from the system. Although green roofs in these climates have been shown to remove some metals, generally substrate materials leach many nutrients and contaminate stormwater further. Therefore, green roofs should be avoided when striving to improve stormwater quality in semi-arid and arid climates. Many studies have also tested the survival of different plant species in different green roof systems, and overall, deeper substrates and certain plant characteristics enhance survival significantly. A synopsis of significant findings determined through this review is provided below.

Bioretention utility in semi-arid and arid climates:

- Bioretention systems reduce peak flow and average annual runoff volume by up to 74.8% and 53%, respectively in semi-arid and arid climates (Li *et al. 2011*; Jiang *et al.* 2015). However, individual systems remain sensitive to design variations including vegetation, substrate characteristics, and dimensions, as well as rainfall characteristics, that can limit retention. For instance, other systems have shown volume reduction as low as 9.1% annually (Huang *et al.* 2014)
- Bioretention performance quickly deteriorates with high-depth and intense rainfall events. Overflow may occur within 20 minutes of the start of a rain event (Lizárraga-Mendiola *et al.* 2017).
- Evapotranspiration accounts for the largest portion of the water budget in semi-arid and arid climates (Feng *et al.* 2016; Lizárraga-Mendiola *et al.* 2017). Plant consumptive use increases with drought and decreases immediately following rainfall events (Lizárraga-Mendiola *et al.* 2017).
- Peak flow and volume reduction vary greatly based on the species of vegetation within each system. More studies should examine the performance of vegetated versus non-vegetated bioretention systems in terms of stormwater mitigation, as non-vegetated cells have achieved higher peak flow reduction than vegetated counterparts (Li *et al.* 2011).
- With strategic selection, plants are capable of surviving without additional irrigation in semi-arid and arid climates for long periods of drought (Houdeshel and Pomeroy 2013).
- Proper plant selection is paramount to plant survival in the extreme conditions of these climates. Texas sage and other species with both drought tolerance and the ability to tolerate standing water are successful; further research should investigate the ability of wetland species to survive with additional irrigation (Li *et al.* 2011). Species with different functional characteristics, such as bunchgrass and native shrubs, might be planted together to maximize water access (Houdeshel *et al.* 2012).

- Xeriscaping efforts show potential in reducing water demands significantly in semi-arid and arid climates (Sovocool *et al.* 2006). Future research needs to investigate the potential for rain harvesting for use in irrigation and other non-potable reuses.
- The addition of an IWS layer can improve both runoff reduction and nutrient removal in semi-arid and arid climates (Li *et al.* 2013; Ambrose and Winfrey 2015). Furthermore, in order to optimize performance, the ratio of impermeable surface area to bioretention area should be calculated specifically for each site, as the ideal ratio has been shown to vary between locations with arid and semi-arid climates (Dussaillant *et al.* 2005).

- In terms of pollutant removal, bioretention basins show high capacity to remove TSS by up to 91% (Li *et al.* 2011). Alternatively, the ability of each system to remove TP and TN from runoff varies drastically, and in some cases, bioretention basins contribute additional TN and TP due to leaching of soil media and decomposition of vegetation (Li *et al.* 2011).
- Vegetated cells often show a greater ability to remove some contaminants than non-vegetated cells, and removal rates vary according to species (Li *et al.* 2011; Jiang *et al.* 2015; Houdeshel *et al.* 2015).
- Bioretention systems can remove high quantities of bacteria and select metals from stormwater runoff in semi-arid and arid climates (Davis *et al.* 2003; Li *et al.* 2011; Kim *et al.* 2012; Li *et al.* 2013).

Green roof utility in semi-arid and arid climates:

- Green roof systems in semi-arid and arid climates can retain higher percentages of stormwater annually than identical systems in humid climates due to higher ET, longer dry periods, and lower AMC (Sims *et al.* 2016). Significant differences were observed between medium-sized events (3-15 mm) in each climate and not for large or small events (Sims *et al.* 2016).
- The water budget is affected more by green roof implementation in semi-arid climates than in humid climates. Widespread implementation of green roofs shows the potential to restore the water budget to more closely resemble that of a predevelopment scenario (Feng *et al.* 2016).
- Green roof retention is low for large rainfall events and high for small events, and retention decreases with increased duration and soil moisture content (Soulis *et al.* 2017).
- Deeper substrates increase volume and peak flow reduction. The presence of vegetation significantly increases retention relative to non-vegetated roofs due to evapotranspiration by plants (Beretta *et al.* 2014; Beecham and Razzaghmanesh 2015; Soulis *et al.* 2017).
- Substrates with lower water use efficiencies have superior volume retention capacities (Razzaghmanesh *et al.* 2014A).
- Xerophytic plants and species with higher evapotranspiration demands allow for a more rapid decrease in soil moisture compared to succulents, which are inefficient in reducing runoff after large storms (Soulis *et al.* 2017).

- Semi-arid and arid green roofs consistently show poor pollutant removal performance; rather, they more frequently contribute additional contaminants to runoff due to substrate media leaching and fertilizer use (Gnecco *et al.* 2013; Razzaghmanesh *et al.* 2014B; Agra *et al.* 2018).

- Use of amendments, such as coal ash, as well as green roof substrates high in organic matter and compost result in higher leaching and greater contamination of runoff (Razzaghmanesh *et al.* 2014B; Agra *et al.* 2018).
- Despite heightened pollutant loading, runoff from green roofs has shown low enough pollutant levels for potential non-potable reuse (Monteiro *et al.* 2016).
- Green roofs can successfully sequester carbon and nitrogen as a result of heightened microbial activity (Ondoño *et al.* 2016).

Vegetation survival and suitability in semi-arid and arid green roof systems:

- Porous substrates with greater water holding capacity and more plant available water promote greater plant growth in semi-arid and arid green roof systems (Farrell *et al.* 2012; Beretta *et al.* 2014; Razzaghmanesh *et al.* 2014B; Raimondo *et al.* 2015).
- Soil amendments including coal ash (Agra *et al.* 2018), grape marc compost (Papfotiou *et al.* 2013), and hydrogels (Savi *et al.* 2014) can enhance plant growth relative to more traditional substrate types (e.g. perlite, vermiculite, etc.). Different species may respond differently to amendments, however; therefore, more studies should examine the response of specific species individually and within a community setting (EPA 2012).
- Substrates with higher organic content support plant growth to a greater extent than substrates without soil or those with low organic matter content (Razzaghmanesh *et al.* 2014B; Ondoño *et al.* 2016).
- Plant growth generally correlates with deeper substrate depth. Shallow substrates have lower initial moisture contents, reducing the biomass (Savi *et al.* 2014; Soulis *et al.* 2017). Shallow substrates also reduce the extent of root development downwards (Razzaghmanesh *et al.* 2014A).
- Shallow substrates often respond too closely to ambient air temperature and as a result may have soil temperature values too high for plants to tolerate in semi-arid climates (Reyes *et al.* 2016).
- Native plant species show great potential in green roof environments (Bousselot *et al.* 2009; Gioannini *et al.* 2018). Research should continue to strive to identify potential new species to introduce to green roof environments, and those that have already been identified should be evaluated in pilot green roof studies (Van Mechelen *et al.* 2014).
- *Sedum* species and some non-native species survive well in green roof settings despite extreme or atypical conditions (Van Mechelen *et al.* 2014; Rayner *et al.* 2016).
- High succulence (Rayner *et al.* 2016; EPA 2012), low water use (Farrell *et al.* 2012), and CAM metabolism (Azeñas *et al.* 2018) are traits that can improve plant survival in semi-arid and arid climates.
- Community-based planting schemes can perform as well as monocultures in terms of vegetation survival on semi-arid and arid green roofs (Gioannini *et al.* 2018). However, in other cases, competition and water use dynamics can result in poor survival by certain species in community settings (EPA 2012). More research should be conducted regarding the combinations of different plant species within a community prior to widespread implementation of a community-based approach (EPA 2012).
- Ultimately, plant species should be chosen based on the specific needs, both aesthetic and practical, and conditions of a site (Razzaghmanesh *et al.* 2014A; Van Mechelen *et al.* 2014). This involves irrigation and maintenance needs, in particular (Raimondo *et al.* 2015).

General treatment train performance:

- LID treatment trains show higher volume retention than individual LID systems and conventional development trains in various climates types, and permeable pavement-bioretention trains have been shown to be particularly effective in this regard (Rushton *et al.* 2001; Wilson *et al.* 2014; Jia *et al.* 2015; Flanagan *et al.* 2016).
- Undersized trains can significantly reduce the retention performance of the overall system. However, pollutant removal can still be high even when retention is low (Doan *et al.* 2017).
- In terms of pollutant removal, treatment trains incorporating bioretention can reduce pollutant loads by up to 85 times those of conventional development trains (Wilson *et al.* 2014). Bioretention-paired trains have achieved high TSS, TP, TN, and metal removal in a variety of climates (Rushton *et al.* 2001; Brown *et al.* 2012; Jia *et al.* 2015). Pollutant levels might be slightly higher initially before the system stabilizes (Jia *et al.* 2015).
- Green roofs are the most cost-effective means of storing runoff within a treatment train (Bastien *et al.* 2010).
- Green roof treatment trains are few and should be investigated further. The few that exist show that no overflow results (Kelly *et al.* 2007), and green roof trains can reduce TSS and TP to within an acceptable limit (Revitt *et al.* 2014).

Semi-arid and arid treatment train potential:

- Few studies investigate treatment trains in semi-arid and arid climates. Those that have noted up to a 74% reduction in TSS loading and reductions in mean metal loading between 59% and 90% (David *et al.* 2015). Methylmercury can increase in runoff if proper drainage is not provided (David *et al.* 2015).
- Components of treatment trains in semi-arid and arid climates have been shown to work better in tandem than individually. For example, increased runoff pH raised upon passing through permeable pavement was buffered by its passage through the bioretention basin (Kazemi *et al.* 2018).
- Combined systems can also lower salinity in stormwater to within a range reasonable for reuse in irrigation and storage (Kazemi *et al.* 2018).
- Future research should investigate the potential for treatment trains to harvest or store water for future use in irrigation. Water might also be pumped and recycled through the system for further treatment.

Given the pattern of long drought periods interspersed with high rainfall days, individual LID systems appear to be of limited efficacy in semi-arid and arid climates when monsoonal weather occurs. Therefore, application of the “LID treatment train” might be a more suitable means of managing stormwater runoff for events that cause the greatest damage. Treatment train studies from a variety of climate types evaluated in this review show that combined systems consistently improve stormwater management relative to individual systems. By combining storage and filtration treatment technologies, such as bioretention cells and green roofs, treatment trains can handle higher quantities of water in terms of both retention and pollutant removal. Furthermore, bioretention systems in particular can be used to address the problem of increased contaminants contained in green roof effluent. Future studies should investigate pairing multiple LID technologies with different mechanisms to evaluate their efficacy while continuing to vary design parameters. Methods to recycle water from high-rainfall events to further improve quality and responsibly irrigate plants in dry periods should also be investigated. Should these technologies be improved and appropriate water harvesting techniques identified, the palette of appropriate species might be broadened and the performance of each system enhanced to better counter the effects of climate change and urbanization as they manifest themselves.

References

Agra, H., T. Klein, A. Vasl, H. Shalom, G. Kadas, and L. Blaustein. 2017. "Sedum-dominated green-roofs in a semi-arid region increase CO_2 concentrations during the dry season." *Science of the Total Environment* 584-585: 1147-1151, doi: 10.1016/j.scitotenv.2017.01.176.

Agra, H., A. Solodar, A. Bawab, S. Levy, G.J. Kadas, L. Blaustein, and N. Greenbaum. 2018. "Comparing grey water versus tap water and coal ash versus perlite on growth of two plant species on green roofs." *Science of the Total Environment* 633: 1272-1279, doi: 10.1016/j.scitotenv.2018.03.291.

Ahiablame, L., B.A. Engel, and I. Chaubey. 2012. "Effectiveness of low impact development practices: Literature review and suggestions for future research." *Water Air Soil Pollution* 223: 4253-4273, doi: 10.1007/s11270-012-1189-2.

Ambrose, R.F. and B.K. Winfrey. 2015. "Comparison of stormwater bioinfiltration systems in Southeast Australia and Southern California." *WIREs Water* 2: 131-146, doi: 10.1002/wat2.1064.

Ampim, P.A.Y., J.J. Sloan, R.I. Cabrera, D.A. Harp, and F.H. Jaber. 2010. "Green Roof Growing Substrates: Types, Ingredients, Composition and Properties." *Journal of Environmental Horticulture* 28 (4): 244-252, doi: 10.24266/0738-2898-28.4.244.

Aschmann, H. 2012. "Distribution and Peculiarity of Mediterranean Ecosystems." *Mediterranean Type Ecosystems: Origin and Structure*, *7*, 11.

Ayub, K.R., A.A. Ghani, N.A. Zakaria, and S. Shaharud. 2015. "Efficiency of intensive green roof in high intensity rainfall for stormwater treatment: selection of vegetations." E-proceedings of the 36th IAHR World Congress, 36(1), 6730-6735.

Azeñas, V., I. Janner, H. Medrano, and J. Gulías. 2018. "Performance evaluation of five Mediterranean species to optimize ecosystem services of green roofs under water-limited conditions." *Journal of Environmental Management* 212: 236-247, doi: 10.1016/j.jenvman.2018.02.021.

Bastien, N., S. Arthur, S. Wallis, and M. Scholz. 2010. "The best management of SuDS treatment trains: a holistic approach." *Water Science and Technology* 61(1): 263-272, doi: 10.2166/wst.2010.806.

Batalini de Macedo, M., A. Rosa, C.A. Ferreira do Lago, E.M. Mendiondo, and V.C. Borges de Souza. 2017. "Learning from the operation, pathology and maintenance of a bioretention system to optimize urban drainage practices." *Journal of Environmental Management* 204: 454-466, doi: 10.1016/j.jenvman.2017.08.023.

Beck, C., J. Grieser, M. Kottek, F. Rubel, and B. Rudolf. 2005. "Characterizing global climate change by means of Köppen climate classification." *Klimastatusbericht*.

Beecham, S. and M. Razzaghmanesh. 2015. "Water quality and quantity investigation of green roofs in a dry climate." *Water Research* 70: 370-384, doi: 10.1016/j.watres.2014.12.015.

Berndtsson, J.C. 2010. "Green roof performance towards management of runoff water quantity and quality: A review." *Ecological Engineer*ing 36: 351-360, doi: 10.1016/j.ecoleng.2009.12.014.

Berretta C., S. Poe, and V. Stovin. 2014. "Moisture content behaviour in extensive green roofs during dry periods: The influence of vegetation and substrate characteristics." *Journal of Hydrology* 511: 374-386, doi: 10.1016/j.jhydrol.2014.01.036.

Blecken, G.T., Y. Zinger, T.M. Muthanna, A. Deletic, T.D. Fletcher, and M. Viklander. 2007. "The influence of temperature on nutrient treatment efficiency in stormwater biofilter systems." *Water Science and Technology* 56(10): 83-91, doi: 10.2166/wst.2007.749.

Bousselot, J.M., J.E. Klett, and R.D. Koski. 2009. "High elevation semi-arid taxa: evaluations on an extensive green roof." Paper Presented at Seventh Annual Greening Rooftops for Sustainable Communities Conference, Atlanta, GA.

Braswell, A.S., A.R. Anderson, and W.F. Hunt III. 2018. "Hydrologic and Water Quality Evaluation of a Permeable Pavement and Biofiltration Device in Series." *Water* 10(1), 1-21, doi: 10.3390/w10010033.

Brown, R.A. and W.F. Hunt. 2011. "Underdrain configuration to enhance bioretention exfiltration to reduce pollutant loads." *Journal of Environmental Engineering* 137(11): 1082-1091, doi: 10.1061/(ASCE)EE.1943-7870.0000437.

Brown, R.A., D.E. Line, and W.F. Hunt. 2012. "LID treatment train: pervious concrete with subsurface storage in series with bioretention and care with seasonal high water tables." *Journal of Environmental Engineering* 138(6): 689-697, doi: 10.1061/(ASCE)EE.1943-7870.0000506.

Caraco, D. 2000. "Stormwater strategies for arid and semi-arid watersheds." *Watershed Protection Techniques 3* (3): 695-706.

Chen, C. 2013. "Performance evaluation and development strategies for green roofs in Taiwan: A Review." *Ecological Engineering* 52: 51-58, doi: 10.1016/j.ecoleng.2012.12.083.

Chowdhury, R.K. 2015. "Greywater reuse through a bioretention system prototype in the arid region." *Water Science and Technology* 72(12): 2201-2211, doi: 10.2166/wst.2015.442.

Clark, M.J. and Y. Zheng. 2012. "Evaluating fertilizer influence on overwintering survival and growth of Sedum species in a fall-installed green roof." *Horticultural Science* 47(12): 1775-1781, doi: ISSN: 2327-9834.

Coffman, L., R. Green, M. Clar, & S. Bitter. 1994. Development of Bioretention Practices for Stormwater Management. In W. James, (Ed.), *Advances of Modeling the Management of Stormwater Impacts*, (23-42). Chelsea, MI: Ann Arbor Press.

David, N., J.E. Leatherbarrow, D. Yee, and L.J. McKee. 2015. "Removal efficiencies of a bioretention system for trace metals, PCBs, PAHs, and dioxins in a semiarid environment." *Journal of Environmental Engineering* 141(6): 1-8, doi: 10.1061/(ASCE)EE.1943-7870.0000921.

Davis, A.P., M. Shokouhian, H. Sharma, and C. Minami. 2001. "Laboratory study of biological retention for urban stormwater management." *Water Environment Research* 73(1): 5-14.

Davis, A.P., M. Shokouhian, H. Sharma, C. Minami, and D. Winogradoff. 2003. "Water quality improvement through bioretention: lead, copper, and zinc removal." *Water Environment Research* 75: 73-82, doi: 10.2175/106143003X140854.

Davis, A.P. 2008. "Field performance of bioretention: Hydrology impacts." *Journal of Hydrological Engineering* 13(2), 90-95, doi: 10.1061/ ASCE 1084-0699 2008 13:2 90.

Davis, A.P., ASCE, F., Traver, R.G., Hunt, W.F., ASCE, M., Lee, R., Brown, R., ASCE, A.M., & Olszewski, J.M. 2012. Hydrologic Performance of Bioretention Storm-Water Control Measures. *Journal of Hydrologic Engineering*, 17(5): 604-614.

Dietz, M.E. 2007. "Low impact development practices: A review of current research and recommendations for future directions." *Water Air Soil Pollution* 186: 351-363, doi: 10.1007/s11270-007-9484-z.

Doan, L.N. and A.P. Davis. 2017. "Bioretention-cistern-irrigation treatment train to minimize stormwater runoff." *Journal of Sustainable Water in the Built Environment* 3(2): 1-10, doi: 10.1061/ JSWBAY.0000820.

Dussaillant, A.R., A. Cuevas, and K.W. Potter. 2005. "Raingardens for stormwater infiltration and focused groundwater recharge: simulations for different world climates." *Water Science and Technology Water Supply* 5 (3): 173-179, doi: 10.2166/ws.2005.0097.

Dvorak, B. and A. Volder. 2010. "Green roof vegetation for North American ecoregions: A literature review." *Landscape and Urban Planning* 96: 197-213, doi: 10.1016/j.landurbplan.2010.04.009.

Eckhart, K., McPhee, Z., & Bolisetti, T. 2017. Performance and implementation of low impact development; A Review. *Science of the Total Environment*, 607-608, 413-432. doi: 10.1016/j.scitotenv.2017.06.254.

Elliott, R.M., R.A. Gibson, T.B. Carson, D.E. Marasco, P.J. Culligan, and W.R. McGillis. 2016. "Green roof seasonal variation: comparison of the hydrologic behavior of a thick and a thin extensive system in New York City." *Environmental Research Letters* 11: 1-15, doi: 10.1088/1748-9326/11/7/074020.

EPA (Environmental Protection Agency). 1999. Storm Water Technology Fact Sheet: Bioretention. Washington, D.C.

EPA (Environmental Protection Agency). 2007. Reducing Stormwater Costs through Low Impact Development (LID) Strategies and Practices. Washington, D.C.

EPA (Environmental Protection Agency). 2010. "Green infrastructure in arid and semi-arid climates." 1-9. EPA-833-B-10-002.

EPA (Environmental Protection Agency). 2012. "Evaluation of green roof plants and materials for semi-arid climates." Office of Research and Development. EPA/600/R-12/592.

EPA (Environmental Protection Agency). 2016. "Arid green infrastructure for water control and conservation." U.S. Environmental Protection Agency Office of Research and Development. EPA-C-15-001.

Fang, C. 2010. "Rainwater retention capacity of green roofs in substropical monsoonal climatic regions: A case study in Taiwan." *Design and Nature* 138: 239-249, doi: 10.2495/DN100211.

Farrell, C., R.E. Mitchell, C. Szota, J.P. Rayner, and N.S.G. Williams. 2012. "Green roofs for hot and dry climates: Interacting effects of plant water use, succulence and substrate." *Ecological Engineering* 49: 270-276, doi: 10.1016/j.ecoleng.2012.08.036.

Feng, Y., S. Burian, and C. Pomeroy. 2016. "Potential of green infrastructure to restore predevelopment water budget of a semi-arid urban catchment." *Journal of Hydrology* 542: 744-755, doi: 10.1016/j.jhydrol.2016.09.044.

Ferrans, P., C.V. Rey, G. Pérez, J.P. Rodríguez, and M. Díaz-Granados. 2018. "Effect of green roof configuration and hydrological variables on runoff water quantity and quality." *Water* 10(7), 1-18, doi: 10.3390/w10070960.

FISRWG. 1998. *Stream corridor restoration: Principles, processes and practices.* GPO Item No. 0120-A; SuDocs No. A 57.6/2:EN 3/PT.653.

Flanagan, K., B. Phillippe, R. David, and G. Marie-Christine. 2016. "Evaluation of the relative roles of a vegetative filter strip and a biofiltration swale in a treatment train for road runoff." *Water Science Technology* 75(3-4): 987-997, doi: 10.2166/wst.2016.578.

Foley, J.A., R. DeFries, G.P. Asner, C. Barford, G. Bonan, S.R. Carpenter, F.S. Chapin, M.T. Coe, G.C. Daily, H.K. Gibbs, J.H. Helkowski, T. Holloway, E.A. Howard, C.J. Kucharik, C. Monfreda, J.A. Patz, I.C. Prentice, N. Ramankutty, and P.K. Snyder. 2005. "Global consequences of land use." Science 309: 570-574, doi: 10.1126/science.1111772.

France, R., & P. Craul. 2002. "Retaining water: Technical support for capturing parking lot runoff (Ithaca, New York)." In France, R.L., (Ed.), *Handbook of Water Sensitive Planning and Design*, Boca Raton, FL: Lewis Publishers.

Gao, X. and F. Giorgi. 2008. "Increased aridity in the Mediterranean region under greenhouse gas forcing estimated from high resolution simulations with a regional climate model." *Global* *and Planetary Change* 62(3-4): 195-209, doi: 10.1016/j.gloplacha.2008.02.002.

Gautam, M.R., K. Acharya, and M. Stone. 2010. "Best management practices for stormwater management in the desert southwest." *Journal of Contemporary Water Research and Education* 146: 39-49, doi: 10.1111/j.1936-704X.2010.00390.x.

Gioannini, R., M. Al-Ajlouni, R. Kile, D. VanLeeuwen, and R. St. Hilaire. 2018. "Plant communities suitable for green roofs in arid regions." *Sustainability* 10(6): 1755, doi: 10.3390/su10061755.

Gnecco, I., A. Palla, L.G. Lanza, and P. La Barbera. 2013. "A green roof experimental site in the Mediterranean climate: the storm water quality issue." *Water Science and Technology* 68 (6): 1419-1424, doi: 10.2166/wst.2013.196.

Gobel, P., C. Dierkes, and W.G. Coldewey. 2007. "Storm water runoff concentration matrix for urban areas." *Journal of Contaminant Hydrology* 91: 26-42, doi: 10.1016/j.jconhyd.2006.08.008.

Goh, H.W., C.K. Chang, T.Z. Lau, K.Y. Foo, and N.A. Zakaria. 2015. "The role of tropical shrub with enhanced bioretention media in nutrient rich runoff treatment." E-proceedings of the 26th IAHR World Congress, Netherlands.

Gordo, O. and J.J. Sanz. 2010. "Impact of climate change on plant phenology in Mediterranean ecosystems." *Global Change Biology* 16(3): 1082-1106, doi: https://doi.org/10.1111/j.1365-2486.2009.02084.x.

Guatam, M.R., K. Acharya, and M. Stone. 2010. "Best management practices for stormwater management in the Desert Southwest." *Journal of Contemporary Water Research and Education* 146(1): 39-49, doi: *10.1111/j.1936-704X.2010.00390.x.*

Hameed, H.M. 2017. "Estimating the effect of urban growth on annual runoff volume using GIS in the Erbil sub-basin of the Kurdistan region of Iraq." *Hydrology* 4 (12): 1-13, doi: 10.3390/hydrology4010012.

Hatt, B.E., T.D. Fletcher, and A. Deletic. 2009. "Hydrologic and pollutant removal performance of stormwater biofiltration systems at the field scale." Journal of Hydrology 365: 310-321, doi: 10.1016/j.jhydrol.2008.12.001.

Houdeshel, D. and C. Pomeroy. 2013. "Storm-water bioinfiltration as no-irrigation landscaping alternative in semiarid climates." *Journal of Irrigation and Drainage Engineering* 140(2): 1-6, doi: 10.1061/(ASCE)IR.1943-4774.0000663.

Houdeshel, C.D., C.A. Pomeroy, and K.R. Hultine. 2012. "Bioretention design for xeric climates based on ecological principles." Journal of the American Water Resources Association 48(6): 1178-1190, doi: 10.1111/j.1752-1688.2012.00678.x.

Houdeshel, C.D., K.R. Hultine, N.C. Johnson, and C.A. Pomeroy. 2015. "Evaluation of three vegetation treatments in bioretention gardens in a semi-arid climate." *Landscape and Urban*

Planning 135: 62-72, doi: 10.1016/j.landurbplan.2014.11.008.

Huang, J.J., Y. Li, S. Niu, and S.H. Zhou. 2014. “Assessing the performances of low impact development alternatives by long-term simulation for a semi-arid area in Tianjin, northern China.” *Water Science and Technology* 70(11): 1740-1745, doi: 10.2166/wst.2014.228.

Huang, J., M. Ji, Y. Xie, S. Wang, Y. He, and J. Ran. 2016. “Global semi-arid climate change over last 60 years.” *Climate Dynamics* 46: 1131-1150, doi: 10.1007/s00382-015-2636-8.

Hunt, W.F., A.R. Jarrett, J.T. Smith, and L.J. Sharkey. 2006. “Evaluating bioretention hydrology and nutrient removal at three field sites in North Carolina.” *Journal of Irrigation and Drainage Engineering* 132(6): 600-608, doi: 10.1061/(ASCE)0733-9437(2006)132:6(600).

Hunt, W.F., J.T. Smith, S.J. Jadlocki, J.M. Hathaway, and P.R. Eubanks. 2008. “Pollutant removal and peak flow mitigation by a bioretention cell in urban Charlotte, N.C.” *Journal of Environmental Engineering* 134(5): 403-408, doi: 10.1061/ ASCE 0733-9372 2008 134:5 403.

Hunt, W.F., Davis, A.P., Traver, R.G. 2012. “Meeting Hydrologic and Water Quality Goals through Targeted Bioretention Design.” *Journal of Environmental Engineering*, 138(6), 698-707. doi.org/10.1061/(ASCE)EE.1943-7870.0000504.

Imbrium Systems Inc. 2017. Filterra Bioretention System. Imbrium Systems Inc. IB-Filterra Bro 3/16. http://www.imbriumsystems.com/Portals/0/documents/fl/brochures/Filterra%20Brochure.pdf.

Jia, H., X. Wang, C. Ti, Y. Zhai, R. Field, A.N. Tafuri, H. Cai, and S.L. Yu. 2015. “Field monitoring of a LID-BMP treatment train system in China.” *Environmental Monitoring and Assessment* 187: 1-18, doi: 10.1007/s10661-015-4595-2.

Jiang, C., Li, J., Li, H, Li, Yajiao, & Chen, L. 2017. Field Performance of Bioretention Systems for Runoff Quanitity Regulation and Pollutant Removal. *Water Air Soil Pollution*, 228, 468, doi: 10.1007/s11270-017-3636-6.

Jiang, Y., Yuan, Y., and H. Piza. 2015. “A review of applicability and effectiveness of low impact development/green infrastructure practices in arid/semi-arid United States.” *Environments* 2: 221-249, doi: 10.3390/environments2020221.

Johannessen, B.G., H.M. Hanslin, and T.M. Muthanna. 2017. “Green roof performance potential in cold and wet regions.” *Ecological Engineering* 106: 426-447, doi: 10.1016/j.ecoleng.2017.06.011.

Johannessen, B.G., T.M. Muthanna, and B.C. Braskerud. 2018. “Detention and retention behavior of four extensive green roofs in three Nordic climate zones.” *Water* 10: 1-23, doi: 10.3390/w10060671.

Jurries, D. & Oregon DEQ. 2003. Biofilters (Bioswales, Vegetative Buffers, & Constructed Wetlands) For Storm Water Discharge Pollution Removal. Salem, OR.

Kazemi, F., Golzarian, M.R., and Myers, B. 2018. “Potential of combined water sensitive urban design systems for salinity treatment in urban environments.” *Journal of Environmental Management* 209: 169-175, doi: 10.1016/j.jenvman.2017.12.046.

Kelly, M., M. Hardin, and M. Wanielista. 2007. “A Stormwater Management Evaluation of the 2007 New American Home Stormwater Treatment System and the Stormwater Academy Greenroof.” FDEP Project Number: WM 864 Task 4. September, 2007.

Khan, U.T., C. Valeo, A. Chu, and B. van Duin. 2012. “Bioretention cell efficact in cold climates: Part 1- hydrologic performance.” *Canadian Journal of Civic Engineering* 39L 1210-1221, doi: 10.1139/l2012-110.

Kim, H., E.A. Seagren, and A.P. Davis. 2003. “Engineered bioretention for removal of nitrate from stormwater runoff.” *Water Environmental Research* 75(4): 355-367.

Kim, M.H., C.Y. Sung, M. Li, and K. Chu. 2012. “Bioretention for stormwater quality improvement in Texas: Removal effective of Escherichia coli.” *Separation and Purification Technology* 84: 120-124, doi: 10.1016/j.seppur.2011.04.025.

Kohler, M., M. Schmidt, F.W. Grimme, M. Laar, V. Paiva, and S. Tavares. 2002. “Green roofs in temperate climates and in the hot-humid tropics- far beyond aesthetics.” *Environmental Management and Health* 13(4): 382-391, doi: 10.1108/09566160210439297.

Kok, K.H. L.M. Sidek, M.F. Chow, M.R.Z. Abidin, H.Basri, and G. Hayder. 2016. “Evaluation of green roof performances for urban stormwater quantity and quality controls.*” International Journal of River Basin Management* 14(1): 1-7, doi: 10.1080/15715124.2015.1048456.

Kottek, M., J. Grieser, C. Beck, B. Rudolf, and F. Rubel. 2006. “World map of the Köppen-Geiger climate classification updated.” *Meteorologische Zeitschrift* 15(3): 259-263, doi: 10.1127/0941-2948/2006/0130.

Lavee, H., A.C. Imeson, and P. Sarah. 1998. “The impact of climate change on geomorphology and desertification along a Mediterranean-arid transect.” *Land Degradation and Development* 9(5): 407-422, doi: 10.1002/(SICI)1099-145X(199809/10)9:5<407::AID-LDR302>3.0.CO;2-6.

Lazarova, V., B. Levine, J. Sack, G. Cirelli,, P. Jeffrey, H. Muntau, M. Salgot, and F. Brissaud. 2001. “Role of water reuse for enhancing integrated water management in Europe and Mediterranean countries.” *Water Science and Technology* 43(10): 25-33.

Le Coustumer, S., T.D. Fletcher, A. Deletic, and S. Barraud. 2007. “Hydraulic performance of biofilter for stormwater management: first lessons from both laboratory and field studies.” *Water Science and Technology* 56(10): 93-100, doi: 10.2166/wst.2007.735.

LeFevre, N.J., J.D. Davidson, and G.L. Oberts. 2009. “Bioretention of simulated snowmelt: cold climate performance and design criteria.” 14th Conference on Cold Regions Engineering, Duluth, Minnesota, USA: 145-154.

Li, H., L.J. Sharkey, W.F. Hunt, A.P. Davis. 2009. "Mitigation of impervious surface hydrology using bioretention in North Carolina and Maryland." *Journal of Hydrologic Engineering* 14(4): 407-415, doi: 10.1061/ ASCE 1084-0699 2009 14:4 407.

Li, M., C.Y. Sung, M.H., Kim, and K. Chu. 2011. "Assessing performance of bioretention boxes in hot and semiarid regions." *Journal of the Transportation Research Board* 2262: 155-163, doi: 10.3141/2262-15.

Li, M., M. Swapp, M.H. Kim, K. Chu, and C.Y. Sung. 2013. "Comparing bioretention designs with and without an internal water storage layer for treating highway runoff." *Water Environment Research* 86: 1-11, doi: 10.2175/106143013X13789303501920.

Lim, H.S. and X.X. Lu. 2016. "Sustainable urban stormwater management in the tropics: An evaluation of Singapore's ABC Waters Program." *Journal of Hydrology* 538: 842-862, doi: 10.1016/j.jhydrol.2016.04.063.

Liu, J., D.J. Sample, C. Bell, and Y. Guan. 2014. "Review and research needs of bioretention used for the treatment of urban stormwater." *Water* 6: 1069-1099, doi: 10.3390/w6041069.

Lizárraga-Mendiola, L., G.A. Vazquez-Rodriguez, C.A. Lucho-Constantino, C.. bigurra-Alzati, R.I. Beltran-Hernandez, J.E. Ortiz-Hernandez, and L.D. Lopez-Leon. 2017. "Hydrological design of two low-impact development techniques in a semi-arid climate zone of central Mexico." *Water* 9 (8): 1-18, doi: 10.3390/w9080561.

Manganka, I., Liu, A., Egodawatta, P., & Goonetilleke, A. 2015. Performance of a stormwater treatment bioretention basin. *Journal of Environmental Management, 150*, 173-178. doi.org/10.1016/j.jenvman.2014.11.007.

Minnesota Pollution Control Agency. "Using the treatment train approach to BMP selection" in *Minnesota Stormwater Manual,* https://stormwater.pca.state.mn.us/index.php?title=Using_the_treatment_train_approach_to_BMP_selection&oldid=22962 (accessed September 28, 2018).

Mobilia, M., A. Longbardi, and J.F. Sartor. 2015. "Green roofs hydrological performance under different climate conditions." *WSEAS Transactions on Environment and Development* 11: 264-271, E-ISSN: 2224-3496.

Monteiro, C.M., C.S.C. Calheiros, C. Pimentel-Rodrigues, A. Silva-Afonso, and P.M.L. Castro. 2016. "Contributions to the design of rainwater harvesting systems in buildings with green roofs in Meditteranean climate." *Water Science and Technology* 73 (8): 1842-1847, doi: 10.2166/wst.2016.034.

Muha, N.E., L.M. Sidek, and M. Jajarmizadeh. 2016. "Water quality improvement through reductions of pollutant loads on small scale of bioretention system." *IOP Conference Series: Earth and Environmental Science* 32: 1-4, doi: 10.1088/1755-1315/32/1/012010.

Muthanna, T.M., M. Viklander, N. Gjesdahl, S.T. Thorolfsson. 2007. “Heavy metal removal in cold climate bioretention.” *Water Air Soil Pollution* 183: 391-402, doi: 10.1007/s11270-007-9387-z.

National Climatic Data Center (NCDC). 2004. *Climatography of the United States No. 20, 1971-2000.*

North Carolina Cooperative Extension Service. 2012. “Urban waterways: Designing bioretention with an internal water storage (IWS) layer.” Ed. Hunt, W.F. and W.G. Lord.

Oberndorfer, E., and Coauthors, 2007. Green Roofs as Urban Ecosystems: Ecological Structures, Functions, and Services. *BioScience* 57(10): 823-833, doi: 10.1641/B571005.

Ondoño, S., J.J. Martinez-Sanchez, and J.L. Moreno. 2016. “The composition and depth of green roof substrates affect the growth of Silene vulgaris and Lagurus ovatus species and the C and N sequestration under two irrigation conditions.” *Journal of Environmental Management* 166: 330-340, doi: 10.1016/j.jenvman.2015.08.045.

Osterkamp, W.R. and J.M. Friedman. 2000. “The disparity between extreme rainfall events and rare floods- with emphasis on the semi-arid American West.” *Hydrological Processes* 14(16): 2817-2829, doi: 10.1002/1099-1085(200011/12)14:16/17<2817::AID-HYP121>3.0.CO;2-B.

Papafotiou, M., N. Pergialioti, and L. Tassoula. 2013. “Growth of Native Aromatic Xerophytes in an Extensive Mediterranean Green Roof as Affected by Substrate Type and Depth and Irrigation Frequency.” *Horticultural Science* 48 (10): 1327-1333.

Paus, K.H., T.M. Muthanna, and B.C. Braskerud. 2016. “The hydrological performance of bioretention cells in regions with cold climates: seasonal craiation and implications for design.” *Hydrology Research* 47(2): 291-304, doi: 10.2166/nh.2015.084.

Perini, K. and P. Sabbion. 2017. *Urban Sustainability and River Restoration*. Wiley-Blackwell: Oxford, p. 70.

Petrovic, N., J. Radosavljevic, N. Tosic, and A. Vukadinovic. 2017. “Types and features of green roof substrates.” 10th International Scientific Conference: SED, 29-35.

Prince George’s County, Maryland Department of Environmental Resource Programs and Planning Division. 1999. Low-Impact Development Design Strategies: An Integrated Design Approach. Largo, MD.

Price, J.G., S.A. Watts, A.N. Wright, R.W. Peters, and J.T. Kirby. 2011. “Irrigation lowers substrate temperature and enhances survival of plants on green roofs in the Southeastern United States.” *HorTechnology* 21(5): 586-592.

Qin, H., Q. Su, S. Khu, and N. Tang. 2014. “Water quality changes during rapid urbanization in the Shenzhen river catchment: An integrated view of socio-economic and infrastructure development.” *Sustainability* 6: 7433-7451, doi: 10.3390/su6107433.

Queensland Department of Primary Industries and Fisheries. 2009. "Isohydric and anisohydric characterisation of vegetable crops." PR09–4248.

Raimondo, F., P. Trifilo, M.A. La Gullo, S. Andri, T. Savi, and A. Nardini. 2015. "Plant performance on Mediterranean green roofs: interaction of species-specific hydraulic strategies and substrate water relations." *AoB Plants* 7: 1-12, doi: doi/10.1093/aobpla/plv007/199388.

Rayner, J.P., C. Farrell, K.J. Ranor, S.M. Murphy, and N.S.G. Williams. 2016. "Plant establishment on a green roof under extree hot and dry conditions: the importance of leaf succulence in plant selection." *Urban Forestry and Urban Greening* 15: 6-14, doi: 10.1016/j.ufug.2015.11.004.

Razzaghmanesh, M., S. Beecham, and C.J. Brien. 2014A. "Developing resilient green roofs in a dry climate." *Science of the Total Environment* 490: 579-589, doi: 10.1016/j.scitotenv.2014.05.040.

Razzaghmanesh, M., S. Beecham, and F. Kazemi. 2014B. "Impact of green roofs on stormwater quality in a South Australian urban environment." *Science of the Total Environment* 470-471: 651-659, doi: 10.1016/j.scitotenv.2013.10.047.

Revitt, D.M., L. Lundy, F. Coulon, and M. Fairley. 2014. "The sources, impact and management of car park runoff pollution: a review." *Journal of Environmental Management* 146: 552-567, doi: 10.1016/j.jenvman.2014.05.041.

Reyes, R., W. Bustamante, J. Gironás, P.A. Pastén, V. Rojas, F. Suárez, S. Vera, F. Victorero and C.A. Bonilla. 2016. "Effect of substrate depth and roof layers on green roof temperature and water requirements in a semi-arid climate." *Ecological Engineering* 97: 624-632, doi: 10.1016/j.ecoleng.2016.10.025.

Roseen, R.M., T.P. Ballestero, J.J. Houle, and P. Avellaneda. 2009. "Seasonal performance variaitons for storm-water management systems in cold climate conditions." *Journal of Environmental Engineering* 135(3): 128-137, doi: 10.1061/ ASCE 0733-9372 2009 135:3 128.

Rowe, D.B. 2010. "Green roofs as a means of pollution abatement." *Environmental Pollution* 159(8-9): 2100-2110, doi: 10.1016/j.envpol.2010.10.029.

Roy-Poirier, A., Champagne, P., ASCE, A.M., & Filion, Y. 2010. Review of Bioretention System Research and Design: Past, Present, and Future. *Journal of Environmental Engineering* 136(9), 878-889. doi: 10.1061/(ASCE)EE.1943-7870.0000227.

Rushton, B.T. 2001. "Low-impact parking lot design reduces runoff and pollutant loads." *Journal of Water Resources Planning and Management* 127(3): 172-179, doi: 10.1061/(ASCE)0733-9496(2001)127:3(172).

Sade, N., A. Gebremedhin, and M. Moshelion. 2012. "Risk-taking plants: anisohydric behavior as a stress-resistance trait." *Plant Signal Behavior* 7(7): 767-770, doi: 10.4161/psb.20505.

Savi, T., M. Marin, D. Boldrin, G. Incerti, S. Andri, and A. Nardini. 2014. "Green roofs for a drier world: Effects of hydrogel amendment on substrate and plant water status." *Science of the Total Environment* 490: 467-476, doi: 10.1016/j.scitotenv.2014.05.020.

Schueler, T., Hirschman, D., Novotney, M., & Zielinski, J. 2007. Manual 3: Urban Stormwater Retrofit Practices. In Center for Watershed Protection, Urban Subwatershed Restoration Manual Series. Ellicott City, MD.

SEMCOG. 2008. *Low Impact Development Manual for Michigan.* Detroit, MI.

Sileshi, R., Pitt, R., and Clark, S. 2015. "Impacts of Soil Texture, Structure, and Compaction on Bioinfiltration Device Performance: Results of Lab and Field Investigations." In Clar, M.L., Traver, R.G., Clark. S.E., Lucas, S., Lichten, K., & Ports, M.A. (Eds.), *Low Impact Development Technology: Design Methods and Case Studies*, 4-15.

Simmons, M.T., B. Gardiner, S. Windhager, and J. Tinsley. 2008. "Green roofs are not created equal: the hydrologic and thermal performance of six different extensive green roofs and reflective and non-reflective roofs in a sub-tropical climate." *Urban Ecosystems* 11: 339-348, doi: 10.1007/s11252-008-0069-4.

Sims, A.W., C.E. Robinson, C.C. Smart, J.A> Voogt, G.J. Hay, J.T. Lundholm, B. Powers, and D.M. O'Carroll. 2016. "Retention performance of green roofs in three different climate regions." *Journal of Hydrology* 542: 115-124, doi: 10.1016/j.jhydrol.2016.08.055.

Soulis, K.X., N. Ntoulas, P.A. Nektarios, and G. Kargas. 2017. "Runoff reduction from extensive green roofs having different substrate depth and plant cover." *Ecological Engineering* 102: 80-89, doi: 10.1016/j.ecoleng.2017.01.031.

Sovocool, K.A., M. Morgan, and D. Bennett. 2006. "An in-depth investigation of Xeriscape as a water conservation measure." *Journal American Water Works Association* 98 (2): 82-93, doi: 10.1002/j.1551-8833.2006.tb07590.x.

Sultana, N., S. Akib, M.A. Ashraf, M.R.Z. Abidin. 2015. "Quality assessment of harvested rainwater from green roofs under tropical climate." *Desalination and Water Treatment* 57: 75-82, doi: 10.1080/19443994.2015.1015307.

Tan, PY, and A. Sia. 2005, *A pilot green roof research project in Singapore*, paper presented at the Proceedings of Third Annual Greening Rooftops for Sustainable Communities Conference, Awards and Trade Show, Washington, DC, USA.

Valtanen, M., N. Sillanpää, and H. Setälä. 2017. "A large-scale lysimeter study of stormwater bioinfiltration under cold climatic conditions." *Ecological Engineering* 100: 89-98, doi: 10.1016/j.ecoleng.2016.12.018.

Van, P.T.H., et al. 2015. "Nutrient removal by different plants in wetland roof systems treating domestic wastewater." *Desalination and Water Treatment* 54: 1344-1352, doi: 10.1080/19443994.2014.915767.

Van Mechelen, C., T. Dutoit, and M. Hermy. 2014. “Mediterranean open habitat vegetation offers great potential for extensive green roof design.” *Landscape and Urban Planning* 121: 81-91, doi: 10.1016/j.landurbplan.2013.09.010.

Vijayaraghavan, K. 2016. “Green roofs: A critical review on the role of components, benefits, limitations and trends.” *Renewable and Sustainable Energy Review* 57: 740-752, doi: 10.1016/j.rser.2015.12.119.

Villarreal, E.L. and L. Bengtsson. 2005. “Response of Sedum green-roof to individual rain events.” *Ecological Engineering* 25: 1-7, doi: 10.1016/j.ecoleng.2004.11.008.

Vo, T-D-H., X-T Bu, D-D. Nguyen, V-T. Nguyen, H-H. Ngo, W. Guo, P-D. Nguyen, C-N. Nguyen, and C. Lin. 2018. “Wastewater treatment and biomass growth of eight plants for shallow bed wetland roofs.” *Bioresource Technology* 247: 992-998, doi: 10.1016/j.biortech.2017.09.194.

Wang, M., D. Zhang, A. Adhityan, W.J. Ng, J. Dong, and S.K. Tan. 2016. “Assessing cost-effectiveness of bioretention on stormwater in response to climate change and urbanization for future scenarios.” *Journal of Hydrology* 543: 423-432, doi: 10.1016/j.jhydrol.2016.10.019.

Wang, J., L.H.C. Chua, and P. Shanahan. 2017. “Evaluation of pollutant removal efficiency of bioretention basin and implications for stormwater management in tropical cities.” *Environmental Science: Water Research and Technology* 3: 78-91, doi: 10.1039/c6ew00285d.

Westholm, L.J. 2006. “Substrates for phosphorous removal- potential benefits for on-site wastewater treatment?” *Water Research* 40(1): 23-36, doi: 10.1016/j.watres.2005.11.006.

Whittinghill, L.J. and D.B. Rowe. 2011. “Salt tolerance of common green roof and green wall plants.” *Urban Ecosystems* 14: 783-794, doi: 10.1007/s11252-011-0169-4.

Wilson, C. E., W.F. Hunt, R.J. Winston, and P.K. Smith. 2014. “A Comparison of Runoff Quality and Quantity from a Urban Commercial Infill Low Impact Development and a Conventional Development.” In *World Environmental and Water Resources Congress 2013: Showcasing the Future*, pp. 2910-2923.

Wong, G.K.L. and C.Y. Jim. 2014. “Quantitative hydrologic performance of extensive green roof under humid-tropical rainfall regime.” *Ecological Engineering* 70: 366-378, doi: 10.1016/j.ecoleng.2014.06.025.