Hydrologic processes that govern stormwater infrastructure behavior

Journal:	*Hydrological Processes*
Manuscript ID	HYP-17-0322.R1
Wiley - Manuscript type:	SI Urban Hydrology
Date Submitted by the Author:	06-Sep-2017
Complete List of Authors:	Eger, Caitlin; Syracuse University, Civil and Environmental Engineering Chandler, Dave; Syracuse University, Civil and Environmental Eng. Driscoll, Charles; Syracuse University, Civil and Environmental Engineering
Keywords:	Low Impact Development (LID), Best Management Practices (BMPs), Green Infrastructure (GI), Sustainable Drainage Systems (SuDS), Water budget, Urban stormwater management

Hydrologic processes that govern stormwater infrastructure behavior

Caitlin G. Eger[a1], David G. Chandler[a] and Charles T. Driscoll[a]

[a]Department of Civil and Environmental Engineering, 151 Link Hall, Syracuse University, Syracuse, NY 13244, USA

[1]Corresponding author: (CGE) email: cgeger@syr.edu, phone: 1-(315) 443-4443

Contact information for DGC: dgchandl@syr.edu

Contact information for CTD: ctdrisco@syr.edu

Abstract

Using water budget data from published literature, we demonstrate how hydrologic processes govern the function of various stormwater infrastructure technologies. Hydrologic observations are displayed on a Water Budget Triangle, a ternary plot tool developed to visualize simplified water budgets, enabling side-by-side comparison of green and grey approaches to stormwater management. The tool indicates ranges of hydrologic function for green roofs, constructed wetlands, cisterns, bioretention and other stormwater control management structures. Water budgets are plotted for several example systems to provide insight on structural and environmental design factors, and seasonal variation in hydrologic processes of stormwater management systems. Previously published water budgets and models are used to suggest appropriate operational standards for several green and grey stormwater control structures and compare between conventional and low-impact development approaches. We compare models, characterize and quantify water budgets and expected ranges for green and grey infrastructure systems, and demonstrate how the Water Budget Triangle tool may help users to develop a data-driven approach for understanding design and retrofit of green stormwater infrastructure.

Key Words
Low Impact Development (LID), Best Management Practices (BMPs), Green Infrastructure (GI), Sustainable drainage systems (SuDS), Urban stormwater management, Water budget

1 Introduction

International attention to the problem of urban stormwater has resulted in the innovation of many new civil infrastructure solutions. These new structures and design principles have many names and acronyms (Fletcher et al. 2015) such as: low impact development (LID; Low Impact Development Center 2014), green infrastructure (GI; US EPA Office of Water 2016), best management practices (BMPs; WERF et al. 2014), stormwater control measures (SCMs; Fassman-Beck et al. 2016), sustainable (urban) drainage systems (SUDS), water-sensitive urban design (WSUD; Melbourne Water 2016), blue-green infrastructure (BGI; Thorne et al. 2015) and 'soft-path' water infrastructure (Brown 2014). The breadth of names for various stormwater management structures does not lead to a clear understanding of functional differences in performance by practitioners and stakeholders. We aim to characterize and quantify physical differences in hydrologic function associated with various stormwater technologies using simplified, representative water budgets. Ultimately, our goal is to inform both technical and non-technical users about hydrologic processes occurring in stormwater systems.

Stormwater BMPs are used to capture urban runoff to reduce or eliminate wastewater inputs, combined sewer overflows (CSOs), and meet total maximum daily pollutant load (TMDL) goals; however, there are many co-benefits of GI (CNT and American Rivers 2011, Zahmatkesh et al. 2015). US regulators have attempted to credit BMPs in state stormwater design standards by incorporating a 'Runoff Reduction' method into design manuals (Hirschman et al. 2008, NYSDEC and CWP 2015, NERR 2016). This method helps practitioners select appropriate BMP options from a suite of choices based on projected hydrologic function.

Application of this method stems from recognition that water quality benefits from stormwater control structures are largely volume-driven (Hirschman et al. 2008, Ballestero and UNHSC 2012, Eger 2012). Although the ‘Runoff Reduction’ method credits ‘green’ BMPs, it neither credits nor discredits the selection of conventional stormwater (gray) structures over GI (e.g. NYSDEC 2013). This approach presumes conventional ‘gray’ technologies are either environmentally benign or hydrologically superior to green engineering strategies. Often, neither is the case. Therefore, these two broad stormwater management approaches are implemented on unequal terms, rather than as complementary technologies that should be evaluated on the same assessment scale. This perspective unfortunately limits the breadth of information available to urban hydrologists, engineers, planners and civic decision makers when choosing among GI and conventional structures for stormwater management.

The advantages of GI over conventional grey infrastructure for stormwater abatement are widely reported (De Sousa et al. 2012, Lucas and Sample 2015). However, runoff reduction values for GI often have wider operating ranges than conventional grey systems (Driscoll et al. 2015). As a result, implementation of GI has met resistance and regulatory barriers, which mandate inflexible standards or prescribe specific performance metrics, and from communities with fractured or complex stormwater regulation (Worstell 2013, EPA Green Infrastructure Technical Assistance Program 2013, Green Infrastructure for New Hampshire Coastal Communities 2014). In some cases, civil infrastructure professionals and permitting organizations have expressed concern over the uncertainty of adopting green infrastructure BMPs for runoff mitigation (Matthews et al. 2015, Thorne et al. 2015). Technical concerns from the engineering community about performance uncertainty and undefined operational ranges can be interpreted by non-technical decision makers as increased risk for implementation of BMPs relative to conventional stormwater infrastructure. In contrast, risk of implementing grey infrastructure is less commonly addressed, despite established social, economic and ecological impacts (Walsh et al. 2005, Vineyard et al. 2015). Concerns about inconsistent performance stem from an absence of clear metrics to compare and contrast green and grey systems in straightforward, meaningful ways. The proliferation of field studies on GI systems has been accompanied by greater availability of performance data and range of metrics in the literature, including volumetric reduction, peak flow reduction and delayed time-to-peak, among others (Stovin et al. 2015). However, some of these metrics are not well suited for comparison of GI to grey systems. For example, recent reports indicate that GI often outperforms grey infrastructure on a percent volumetric or mass reduction basis, however, this metric is not commonly used for conventional stormwater infrastructure monitoring (Bhaskar and Welty 2012, Driscoll et al. 2015). Peak flow reduction metrics are routinely used in stormwater reporting, but this measure is generally inappropriate for monitoring subterranean sewer systems, green roofs and porous pavement installations. The range of function among BMP technologies and designs has impeded efforts to gather consensus on the benefits associated with these practices. Moreover, the lack of traditional descriptive metrics acts as a barrier to decision-makers whose options may be restricted by regulatory code. For example, retention and detention ponds generally exhibit favorable time-to-peak delay but poor overall volumetric reduction, which caps the benefits realized for downstream water quality (Driscoll et al. 2015). Further, comparison among similar designs is complicated by different climatic conditions and scales (Driscoll et al. 2015). Common reporting methods are necessary to synthesize datasets and identify which physical factors have the greatest influence over hydrologic and water quality variables. In this assessment, we conduct a quantitative comparison among BMP types, and evaluate the hydrologic processes occurring within green systems (primarily infiltrative and evaporative) alongside grey systems (primarily conveyance) using the common water budget metric and a ternary plot tool, the Water Budget Triangle.

2 Methods

2.1 Water Budget Triangle, a tool for graphical analysis

The Water Budget Triangle was developed to address the fundamental question: "*How does stormwater leave the unit volume of a control structure?*" The tool facilitates comparative assessment of dissimilar systems by providing graphical depiction of simplified hydrologic budgets exiting the control volume of an engineered or natural stormwater structure. It is intended to visually represent the fractional distribution of volumetric (or mass) outflow among discharge (Q), percolation (I) and evaporation (ET) on a ternary diagram (Eger et al. 2014). The tool assumes that after influent stormwater enters a system there are three potential pathways for water loss:

1. Discharge to a pipe or surface water (Q, right axis);
2. Evaporation or transpiration into the atmosphere (ET, top axis); or
3. Drainage into soil pores/groundwater (I, left axis).

Our analysis and visualization approach is similar to the Piper plot diagrams developed by Lent et al. (1997) to characterize and visualize hydrologic indices for wetlands (See Appendix S1). However, the Water Budget Triangle is inverted from the Piper plot to emphasize the importance of prioritizing ET and percolation in GI design, and does not account for influent sources of water. Other simplified water budget visualization tools also depict water budgets for both individual structures and whole watersheds (see Askarizadeh et al. 2015, Stroud Water Research

Center 2016). The method was developed to lower communication barriers that limit GI implementation, including: 1) conveying technical information about various stormwater devices to technical and lay stakeholders; 2) providing a systematic visualization tool to compare performance of dissimilar systems; and 3) eliminating ambiguity in the description of BMPs for non-technical stakeholders.

The methodology uses a water balance approach to account for fractional fluxes of water leaving the boundaries of the stormwater control device along each pathway (Q, ET, and I). The mass water flux of any given system may be represented by the water balance equation:

$$R + P = Q + ET + I + \Delta S \qquad (1)$$

R represents influent water or run-on, P represents direct precipitation input and Q, ET and I (defined above) are calculated for the time step of interest. This approach may include stored water released after the event hydrograph. Note water stored (ΔS) in the system control volume can be depleted after a stormwater event, however, the time-step represented on the triangle must be uniform for all axes. The change in water storage (ΔS) within the system is not explicitly represented in the ternary diagram, since it is not a flux. The tool makes a distinction between changes in stores and fluxes because it is used to quantify the relative importance of different loss processes occurring over the time-step of interest. Following increases in inputs from a storm event there is an increase in water storage for a period,

$$P + R > 0 \text{ and } \Delta S \geq 0. \qquad (2,3)$$

After the event, losses will eventually exceed inputs (resulting in decreasing storage),

$$P + R = 0 \text{ and } \Delta S < 0, \qquad (3,4)$$

$$\text{therefore } -\Delta S = ET + I + Q. \tag{5}$$

It is convenient to choose the time scale for analysis as the period over which ΔS = 0, when storage returns to the initial condition prior to a runoff event and the mass balance is fully described by the loss terms in the diagram (i.e., steady-state). However, steady-state condition is not a requirement for application of the tool, as long as the time-step remains constant across all loss pathways.

3 Results and Discussion

It is essential that stormwater infrastructure designers understand how physical design, drainage media preparation, long-term maintenance, and plant species affect the water budget of a built system. To explore this, we present a series of case studies from stormwater management technologies found in the literature, including: runoff reduction calculations; hypothetical assessment of dynamic behavior; comparison of modeled and measured behavior for both constructed and natural BMPs; and comparison of multiple GI and conventional technologies. This synthesis supports reasonable expectations that modifying contributing catchment area, basin area, hydraulic retention time, media depth or soil particle characteristics, rooting depth and other ecohydrologic characteristics will change the water budgets of engineered stormwater systems. Equipped with a quantitative understanding of the hydrologic function of stormwater technologies, the application of watershed models should allow for projections of the stormwater management actions needed to achieve water resource objectives. Using the datasets collected from the literature, we calculate acceptable operational ranges for these structures and understand design factors that influence hydrologic processes (Tables 1 and 2). This data-driven approach supports the development of a "sliding scale" performance credit system (Brown et al.

2011). Supporting information is available for additional stormwater technologies not presented here (Appendix S2).

3.1 Natural and Constructed Wetlands

The hydrologic function of natural wetlands has been explored for more than 50 years (Crisp 1966), and offers a good reference for comparison with constructed systems. Wetlands exhibit a wide range of hydrologic behavior, due to a) varying hydrogeomorphic controls for natural wetlands and b) diverse design objectives for constructed wetlands (Lent et al. 1997, Nungesser and Chimney 2006). Dominant hydrologic fluxes in wetlands may change over relatively short time scales throughout the period of surface runoff, and may reverse during baseflow, flood levels or tidal extremes (Hughes et al. 1998, Choi and Harvey 2000). For example, the major hydrologic fluxes from kettle wetlands are evapotranspiration and infiltration, which may exhibit diel or seasonal fluctuation, depending on temperature and precipitation (Hollands 1989). Natural wetlands typically have more complex water budgets than small constructed wetlands. It may not be possible to depict water budgets for estuaries and other wetland systems with tidal forcing using the simple Water Budget Triangle (see Hughes et al. 1998).

Lent et al. (1997) presented a Piper plot water-budget method to classify natural wetlands and lakes, summarized in Figure 1 alongside data from constructed wetlands and wetland models. Natural wetlands show greater evapotranspiration fractions than constructed wetlands (46% vs 8% in Table I), but similar proportions of flux to groundwater (5 to 10%). Models for natural and constructed wetlands also reflect this difference. The compiled data (Figure 1, Table I) also show constructed wetlands produce 22 to 26% more runoff on average than natural wetlands. This observation makes a good case for preservation of naturally occurring wetlands during landscape development rather than building “replacement” constructed wetlands, an intervention pertinent to developers and land managers in watersheds struggling to control downstream flooding. The difference may arise from: 1) reduced ET in constructed wetlands associated with lower vegetation density; 2) lower ET related to soil carbon content (or different humic material structure), which affect relative infiltration and evaporative fluxes from the system; or 3) seasonally high groundwater surfaces that are shorter in duration in constructed than natural wetlands. In comparison, lakes (Figure 1) exhibit a slightly greater fraction of infiltration than natural wetlands.

3.2 Retention Basins and 'Wet' Ponds

Like many constructed wetlands, retention basins are designed to maintain permanent standing water. As with wetlands, low seepage and limited groundwater exhange in retention ponds is controlled by subsurface hydrology: natural groundwater table, impermeable soils, compaction, and/or presence of a liner (PA DEP 2006, Hartigan and Kelly 2009). Water budget data were obtained for measured values for seven retention ponds in Florida and two years of modeled wet pond water budgets from the City of Austin Stormwater Treatment Section (Harper et al. 2003, Teague and Rushton 2005, Hartigan and Kelly 2009, Harper 2010a, 2010b, 2010c, 2011). Cumulative monthly water budgets, sized by the monthly precipitation (Figure S1) show that retention ponds may behave as zero-discharge systems during seasonally dry conditions, but typically discharge 85-95% of influent. The remaining water is lost to ET (8-13 %) or groundwater (<5%). Hydraulic retention time (average length of time a unit of water spends in the basin storage volume) is a significant predictor of the fraction of inflow occurring as runoff ($R^2 = 0.81$, $p = 0.0006$) and ET ($R^2 = 0.98$, $p = 1.32 \times 10^{-7}$) (Figure 2), but not infiltration in retention basins. Increased hydraulic retention time increases evaporation, but not infiltration. Infiltration losses are explained more by site location than other variables, indicating this pathway a groundwater control (Figure S1). These observations are in line with the design assumption that percolation is not an important sink for retention ponds, although the Poppleton, Palm Bay and Tampa sites show seasonal groundwater connectivity (Figure S1).

3.3 Detention Basins and 'Dry' Ponds

Detention ponds are commonly engineered for 6 to 72 hours of transient storage to attenuate peak flows. Unlike retention ponds, they are not designed to maintain permanent standing water, so are typically dry except for periods of wet weather. Despite their widespread use, detention structures are rarely studied from an ecohydrological perspective; consequently, retention and detention ponds are rarely considered to be types of constructed wetlands. However, this is a somewhat artificial classification, since hydrologic behavior of retention and detention systems places them alongside constructed wetlands on the same continuum (compare Figures 1 and 3).

Much of the literature reporting hydrologic performance of detention and retention basins has focused on the event-scale (Geosyntec and Wright Water Engineers 2011), which overlooks the longer-term roles of ET or groundwater recharge (I) from detention ponds (WEF and ASCE 1998). Use of the triangle tool to study detention ponds requires defining an appropriate time scale to partition percolated drainage (I) and "event runoff" (Q). Event water detained during the period of surface runoff is considered beneficial to watershed function if released gradually during baseflow, and is comparable to percolated drainage. The water balance of detention basins is more variable than retention ponds (compare Figures 2 and 3). Analysis of five detention basins from California and Nevada, and 11 grass-lined detention basins from the International Stormwater BMP Database estimates volumetric reduction to be between 8 – 33% (that is, $Q \approx 67 - 92$) (2^{nd} Nature 2006, Geosyntec and Wright Water Engineers 2011), suggesting these systems may behave similarly to retention basins, or may produce substantially less runoff. Unlike wet ponds, dry detention basins are thought to have good hydrologic connectivity with groundwater, depending upon the infiltration area, soil hydraulic conductivity and unsaturated depth (2nd Nature 2006). Figure 3 indicates this assumption overlooks ET as a loss pathway for detention basins, and calls hydrologic connectivity into question. The presence of mowed vegetation in detention basins also contributes to greater long-term evaporative losses.

Further study of detention basin water budgets may indicate specific design criteria (area to depth ratio) or management techniques (mowing, aeration, planting strategies) may improve long-term stormwater retention by increasing I or ET losses. Analysis of detention and retention pond design characteristics from a water budget standpoint may lead to improved hydrologic or water quality performance of constructed wetlands.

3.4 Bioretention Cells, Infiltration Trenches, Swales and Rain Gardens

Analysis by Traver and DeBarry (2003) of precipitation and runoff in southern Pennsylvania indicated that 80 percent of total annual precipitation volume can be captured by retaining the first 25 mm of each rainfall event. Previous studies have found rain gardens and bioretention cells reduce runoff volume by 30-99% (Schlea 2011, Strauch et al. 2016). Newcomer (2014) modeled volumetric reduction of 58 – 79% for infiltration trenches, compared with 8-33% reduction on irrigated grass lawn. The wide range in performance arises from a large number of design factors that significantly affect the long-term water budgets of swale systems. These design factors include: presence of an underdrain, liner or internal water storage zone; contributing catchment area ratio; direct connection of impervious surfaces; ponding depth; media depth, composition and particle size distribution; and plant density and species composition (Bratieres et al. 2008, Li et al. 2009, Roy-Poirier et al. 2010). Non-design factors that affect water budget include: native sub-base drainage and water table height; event depth and intensity; season and temperature; age and maturity of the planted system; and particle clogging. To our knowledge there are no known studies or reviews that examine all of these factors in controlled experiments and prioritize their relative importance to hydrologic performance. However, there are published case studies, multiple models, and a general intuitive understanding about how these factors affect performance at individual sites (Wardynski and Hunt 2012).

Winston et al. (2016) observed that the underlying sub-base conductivity is a key factor for volume reduction in bioretention cells, reporting that even poorly drained soils can be effective for events smaller than ~6 mm if design allows for an internal water storage zone. Thus far, discussion of internal water storage zones has mostly assumed that better hydrologic performance arises from increases in exfiltration (I increases). However, lysimetry studies indicate that bioretention ET can become water-limited under dry conditions (Wadzuk et al. 2015). ET from bioretention cells decreases as a function of decreasing soil moisture below field capacity: drier soils evaporate less water than wet soils (Buckingham 1907). ET is greatest following a rain event and decreases over subsequent days, resulting in water limitation of bioretention cells. Therefore, internal water storage zones also may maintain system capacity for ET by limiting plant water stress and maintaining sufficient capillary conductance and connectivity to the soil surface. Wadzuk et al. (2015) demonstrate ET limitation by water availability using weighing lysimeters with and without an internal water storage zone. Lysimetry data from Hess (2014) clearly implicate ET as an important loss pathway for bioretention (Figure 4, in green).

Several models of bioretention estimate ET at or below 5% of the water budget, which is much less than estimates from lined bioretention cells (19%) and weighing lysimeters (40-78%) and also less than estimates for unplanted pavement (~10-20%). The discrepancy could be due to the time step used in model calculations, which is narrowed to the event scale + 24 hours, over which little ET occurs. However, 50-year climate simulations using DRAINMOD also largely underestimate the evaporative fraction of long-term water budgets (Wardynski (2011); Figure 4, orange box, lower left). This pattern is corroborated by Hess et al. (2017) and Wadzuk et al. (2015), who report that using Penman-Monteith tends to underestimate ET while using Hargreaves tends to overestimate ET. The yellow circles in Figure 4 represent two cells where ET was estimated using Penman-Monteith. Additional work is needed to more closely constrain annual ET estimates for swale systems before making long-term performance projections under varying climate conditions. The most comprehensive work on ET in bioretention thus far has come from three weighing lysimeters with differing soil types (Hickman 2011, Hickman et al. 2011, Hess 2014, Hess et al. 2015, 2017, Wadzuk et al. 2015). Trials from Hess et al. (2015, 2017) indicate that soil composition controls whether percolation or ET is a more important loss pathway over the long-term (i.e., right to left I-ET axis on the triangle). Using lysimetry, Denich and Bradford (2010) report summer ET rates of 4.2 to 7.7 mm/d in Ontario; Hess et al. (2017) report average ET rates of 2.9 to 4.3 mm/d during the growing season, 1.5 mm/d in winter, with an annual ET of approximately 600 mm in Pennsylvania.

Peak flow reduction is a major goal in mitigating downstream flooding, and is used as a common assessment metric for bioretention cells, retention and detention ponds. However, for infiltrative systems like bioretention cells and porous pavement, volumetric reductions (ET and I) drive peak flow reduction, whereas temporary storage (ΔS) accounts for peak flow mitigation in retention/detention systems. This distinction is significant for understanding both site-level and watershed-scale impacts of engineered stormwater systems. Also, unlike peak flow reduction, volumetric reduction is not related to event intensity. Modest increases in volumetric reduction seem to drive large peak flow attenuation in bioretention and porous pavement systems, but less so for grassed swales, detention and retention ponds. For instance, researchers at NC State and Ohio Department of Natural Resources reported runoff reduction of 36 to 60%, but median peak flow reduction of 97-100%, with maximum flow rates decreasing by at least 29% (NERR 2016). Strauch (2016) reported that only 39 out of 255 events produced measurable runoff at a bioretention facility in Nebraska; volumetric reduction was 33-100% on an event scale and mean peak flow reduction was 63%. Additional research is needed at the event scale to determine if there is a predictable relationship between peak flow reduction and volumetric reduction for different stormwater technologies (under uniform climate conditions).

3.5 Pervious Pavement

Porous or pervious pavement includes permeable asphalt or concrete, interlocking pavers, grassed paver surfaces, and many proprietary mixes for walking, driving and parking surfaces. Infiltration rates of engineered porous surfaces can vary widely, ranging from 2.4 to 4.0 mm/min, greater than double the infiltration rates of natural soils or grassed surfaces (Valinski and Chandler 2015). Most designs and models attribute runoff reduction volumes to infiltration (I = ~90%, Q = 10), but ignore evaporation (Drake et al. 2014). However, Pratt et al (1995) reported lined porous pavement systems equipped with underdrains reduced runoff by 20 to 50% due to increased evaporation (I = 0, Q = 50-80%) at rates between 0.2 and 5.5 mm/day.

Evaporation loss estimates for unplanted porous pavement range from 3-44% and are heavily influenced by the time step of the monitoring period. In general, surface runoff from porous pavements is more sensitive to rainfall intensity than rainfall depth, so results from event-scale studies are more common than long-term cumulative water balances. Event scale studies frequently assume evaporative losses are negligible, since values are less than 0.5 mm/day between March and November (Göbel et al. 2013). This assumption is likely reasonable at the event scale, because porous pavement can have a runoff threshold of up to 7 mm, and because low-intensity, small precipitation events (less than 2 mm) are sometimes excluded from observations. However, evaporative losses over longer timescales are substantial, with annual values reaching 150 mm, easily 10-20% or more of an annual water budget in North America or Europe (Figure 5) (Hein et al. 2013, Göbel et al. 2013). Martin and Kaye (2014) indicate ~1 mm/day is a conservative ET estimate for porous pavements without underdrains. Göbel et al. (2013) estimate cold-weather ET rates from porous pavement during December through February around 0.24 mm/day. Similarly, winter evaporation can be substantial during cold, dry weather: (Drake et al. 2014) reported >20 mm cumulative ET over a winter, accounting for ~9-13% of a winter water budget in Ontario, Canada. Ignoring winter measurements further contributes to the underestimation of ET on an annual basis, especially at sites with intermittent snow cover. A two-year water balance study on three types of lined (I = 0) porous pavement measured 95 mm of evaporation and estimated ET losses to be between 2.4 and 7.6% of annual precipitation (Brown and Borst 2015); however, the authors conclude that design could be modified to enhance evaporation to between 7 and 12%. This is a conservative range for long-term model estimates.

Surface color is a key factor affecting ET losses for porous pavements, because the energy for evaporation is conducted to pore water through the thermal conductivity of the paver; dark-colored pavements may increase ET by up to ~20% (Starke et al. 2010, Göbel et al. 2013). Seam area is also an important factor influencing both infiltration and evaporation rates (Starke et al. 2010). Grassed pavers or pairing unplanted porous pavements with street trees increases transpiration and rainfall interception (Vico et al. 2014). Vegetated grass pavers may evaporate

~1.5 mm/day, accounting for more than 50% of annual precipitation (Göbel et al. 2013). Like green-roofs, evaporative losses from both grassed pavers and un-vegetated pavement may become water-limited during dry periods (ET may decrease in periods of low pore moisture content) (Pratt et al. 1995, Brown and Borst 2015). Thus, eliminating underdrains or including an upturned elbow for internal water storage can increase exfiltration time (I increases) and prevent water-limiting conditions from occurring between storms (ET increases).

3.6 Green Roofs

The primary water sinks for green roof systems are evapotranspiration and discharge, with minimal permanent storage and no infiltration occurring acros the impermeable membrane below the growth media (Wadzuk et al. 2013). Since green roofs are disconnected from ground infiltration, below a minimum event threshold they can operate as zero-discharge systems (Figure 6; top-right corner of triangle), with larger events plotting progressively closer to the lower vertex. Every green roof has a maximum water retention limit; progressively larger and more intense events retain and evaporate proportionally less water. The hydrologic function of a green roof is greatly affected by media depth. Deeper media cells capture incrementally more water; Fioretti et al (2010) show study roofs with greater than 15 cm of substrate retain more water than shallower systems (2-15 cm). However, they also show that a green roof with a modest 2 cm media depth retained more than 400% more precipitation than a conventional roof. Soil media characteristics also play an important role, since particle size distribution determines water holding capacity and retention (Graceson et al. 2013). The chemical properties of green roof soil media of are less well studied, but agronomic and soil science studies have demonstrated that some soil media characteristics enhance water retention capacity (Bleam 2016). Vegetation increases retention by enhancing transpiration losses; the blue (unplanted) roofs presented in Figure 6 show lower ET than planted roofs. Morgan et al (2013) report that a minimum of 20-25% vegetated roof coverage is needed to increase stormwater retention beyond the capacity of the growth media alone. Lab-scale roofs do not seem to capture the full range of performance shown in full-scale green roofs (Figure 6). This discrepancy is possibly due to shorter monitoring periods, lower vegetation density, or greater soil moisture range and sensitivity in test roofs. The roof media storage volume is also dependent upon ambient temperature, and to a lesser extent, on the carbon content and root biomass. Green roofs presented in Figure 6 represent performance across a range of seasons and climates; green roofs in warmer seasons and climates generally capture and evaporate more water than cold-climate green roofs. However, cold-climate green roofs still perform well in comparison with conventional roofs. The mean and median values of the 59 green roof water budgets from the literature are very similar (Figure 6), Q is approximately 35-38% and ET around 62-64%.

3.7 Modeling and Diagnostic Estimates

Basic statistics from results presented in the previous sections are summarized in Table I. This data summary can be used for modeling efforts, and as a diagnostic benchmark to identify underperforming sites. Mean and median estimates of runoff are within 5% for many systems, apart from bioretention, porous pavement, natural lakes, retention ponds and sewer pipe sections.

Given the variability of event-scale measurements and site-to-site comparisons, operational ranges are likely more useful for both modeling and diagnostic applications than individual summary statistics. Estimates of short-term operational range and long-term performance range for each engineered system type are displayed in Table II. These water budget ranges are intended as a benchmark for comparing hydrologic function among unlike technologies. The discharge numbers boxed on Table II represent a suggested worst-case value for performance for each technology. Correspondingly, it would be prudent for designers to aim for designs that meet or exceed the underlined values. For example, bioretention cells that discharge more than 59% on an annual basis should be examined for retrofit or design changes that can improve performance (Table II). Similarly, sewershed networks that discharge less than 81% of the water conveyed should be examined for leakage (Table II). These diagnostic ranges provide a baseline that should help to optimize designs in the future. However, a larger and more representative water budget dataset would provide more robust certainty about reasonable expectations for retention ponds (n = 7 ponds and 2 models), constructed wetlands (n = 8), detention ponds (n = 6 ponds and 1 model) and bioretention cells (n = 3 lysimeters, 5 undersized retrofits, 1 lined cell with underdrain, and 1 unlined cell with underdrain). Additionally, there may be future innovations that improve volumetric capture; performance ranges may change over time to reflect this improvement.

3.8 Data Limitations

The water balances reported here are not uniform among sites – specifically, the various time-steps stem from differences in study questions and approaches. Hydrologic monitoring at some sites was conducted over multiple water years, while most studies of GI collected six to nine months of data. These differences are notated to the extent possible to provide the greatest context for comparisons. The data reported also represent diverse field techniques due to a wide range of approaches used in measurement and reporting among institutions and study cases. While these differences are not unimportant, the datasets represented in the literature show clear patterns for summary and interpretation, despite varied methods. Runoff (Q) is typically measured directly, using a tipping bucket gage or structure calibrated with a stage-discharge relationship. Infiltration measurements are estimated using a linear scale or pressure transducers to quantify the height of the water table and/or soil moisture sensors to estimate pore water saturation. Evaporation estimates are generally modeled by energy balance techniques typical of micrometeorology studies (Wadzuk et al. 2015). A few studies use pan evaporation or other direct measurement techniques, but this approach is less common. As a result, the estimated proportional importance of ET within the water balance is likely incorrect, and probably underestimated over long time and large spatial scales. Underestimation of ET is likely because many models project by upscaling short-term/small-scale measurements during periods when ET is assumed to be minimal.

The relative simplicity of a green roof water budget (as compared with systems with complex groundwater hydrology) highlights the effect that divergent measurement and analysis techniques have on conclusive outcomes. Several of the green roof case studies presented demonstrate both time- and scale-dependent results (Berndtsson 2010, Fioretti et al. 2010, Voyde et al. 2010, Stovin et al. 2012) that contribute to the broad array of performance (Figure 6). For example, lab-scale studies generally result in more runoff than full-scale green roofs, and event-scale monitoring results in more variable ranges than cumulative studies. Event-scale analysis tends to focus on storms above a specific threshold, creating a cumulative water budget that is less representative of overall annual performance, usually reflecting less ET and more runoff. However, using event-scale analysis water budgets can help elucidate the role of antecedent soil moisture and event precipitation depth, and may be appropriate for investigating other design variables, such as particle size distribution or roof slope.

Studies collected over a single season are less valuable than those collected for longer durations, and water budgets for a single season are rarely indicative of annual performance. Monthly observations vary widely due to seasonal changes in hydrologic and plant function, temperature, and variability in the nature of precipitation and runoff events. Therefore, we recommend collecting at least a full water year of daily data to monitor hydrologic functional. Green roofs, bioretention and constructed wetlands are dynamic living systems; there is evidence that water retention and evaporation increase as the vegetation extent and density increase following installation, which may take more than one or two growing seasons to develop (Figure S2). There are few studies of GI function in the literature with more than 5 years of data. It is likely inappropriate to use values collected in the first year to represent of long-term performance of dynamic systems, as they require time for plant maturation, soil settling, and particle loss/accumulation to equilibrate (Figure S2). For long-term (decadal) modeling applications, we recommend collecting continuous data for at least five years for water budget analyses, including the winter season – shorter duration datasets are recommended to be analyzed at the event scale, where median performance is likely more representative than mean.

Fortunately, the ternary diagram is not particularly sensitive to small values changes in the dataset, which means that measurements with the correct order of magnitude provide an adequate level of precision. As a result, users can obtain an accurate “sense” of hydrologic performance by collecting “ballpark” measurements, despite the field challenges of collecting complete water budgets for a site.

3.9 Factors Affecting Hydrologic Performance

Using the triangle to compare dissimilar systems allow study of how individual design factors affect water budget partitioning. Although some design changes may affect all three water budget variables simultaneously, there are several design factors that primarily influence the tradeoff between two water budget variables while remaining isometric in proportion to the third variable. This feature is represented visually on the Water Budget Triangle by points tracking parallel to one side of the triangle; compare Figures 5, S3, and S4, which show observations tracking parallel to the left side of the triangle, indicating a tradeoff between infiltration and runoff (I-Q axis). For example, the most simplified systems showing infiltration-runoff (I-Q) axis tradeoffs are sewers and cisterns (Figures S3 and S4). Understandably, increasing the tank volume of a cistern will increase the fraction of water harvested (I for cisterns), resulting in less runoff. In the same way, increasing pipe volume in a sewershed network will increase the total capture volume, generally leading to more infiltration and decreased runoff, while ET remains negligible. By synthesizing the data presented above and examining engineered systems as a suite of green and grey tools, it is possible to parse which design factors affect these tradeoffs between site water budget fractions when other design variables are held constant (Figure 7).

3.9.1 Factors primarily affecting I-Q tradeoff

The structural design factors that show I-Q tradeoff behavior are: volume, contributing area, and the presence of an underdrain or liner. Larger tank or basin volumes increase the initial amount of water captured by a system, usually leading to higher infiltration rates and lower runoff; this is especially apparent at the event scale. Structural analogs that control capture volume include: above ground empty basin volume (constructed wetlands, retention and detention ponds); "ponding" basin volume or depth (bioretention and swales); tank volume (cisterns), sub-surface collection boxes, pipes or tanks (porous pavement, bioretention, infiltration trenches, tree boxes, sewers, etc.). Increasing the volume of subsurface storage increases infiltration losses, but also may affect ET if the system is water-limited by inducing either standing water or an internal water storage zone.

Nearly all engineered systems have design variations that include options for a subsurface drainage outlet ("underdrain") or an impermeable liner; systems may have one of these features, both of them, or neither. Designs that may include underdrains or subsurface outlets exist for constructed wetlands, bioretention, porous pavement, green roofs, and retention and detention ponds. Liners may be present in wetlands, retention and detention ponds, porous pavement and bioretention. Sealed pipe joints between sewer sections act as a liner analog and affect the proportion of I-Q fractions. The presence of a drain increases runoff (through the drain), decreases stored capture and reduces infiltration; ET from drained bioretention, constructed wetlands and porous pavement systems remains relatively low (< 30%). The presence of an impermeable liner, geotextile or compaction layer impedes infiltration and constrains hydrologic performance to the right-hand axis (I = 0; see Figure 4, lined bioretention cell; lined porous pavement in Figure 5 and green roofs in Figure 6).

Holding other design variables constant, changing the contributing area to a system will tend to affect the balance between runoff and infiltration. Auxiliary (but related) factors such as the land use of the contributing area or direct connection of impervious surfaces will also affect system performance along the I-Q axis. Wetlands, retention, detention, bioretention, porous pavement and cisterns show performance changes along the I-Q axis when contributing area changes. External environmental factors affecting infiltration fraction include the hydraulic conductivity of sub-base, plant root establishment, particle clogging, event depth and intensity.

3.9.2 Factors primarily affecting Q-ET tradeoff

The most simplified systems showing runoff-ET (Q-ET) axis tradeoffs are green roofs and other lined systems. The structural design factors that affect Q-ET tradeoff behavior are: presence of an internal water storage zone or standing water, amended media depth, and particle size distribution and surface chemistry. Much of the soil media research that has previously been published for green roofs is relevant for ground-based infiltration systems, although weight load limitations are not. The amended media depth and particle characteristics strongly affect the volume of moisture retained at field capacity, once all ponded water has drained through soil media (~24 hours after the end of the rain event). Soil moisture retained within the media matrix after the system has drained to field capacity likely leaves the system through evaporative losses, not through infiltration into the sub-base. Shallower amended media depths retain less soil moisture, allow systems to become water-limited and decrease the importance of ET as a loss pathway. Likewise, designs that use an up-turned elbow drain or raised outlet elevation to promote internal water storage prevent systems from becoming water-limited, and increase ET. Preventing internal water-limitation is a design consideration that is relevant for nearly all types of engineered stormwater systems to promote ET. Maintaining a small amount of residual soil moisture between events also allows for higher hydraulic conductivity at the soil surface, reduces hydrophobicity, and promotes infiltration rates at the onset of subsequent rain events. However, sites that do not effectively evaporate soil moisture between storm events remain relatively saturated and result in increased runoff. External environmental factors affecting soil moisture and evaporation losses include season, temperature, and natural depth to groundwater onsite.

3.9.3 Factors primarily affecting ET-I tradeoff

The addition of plants to an unplanted system increases loss by ET; there is a shift from left to right visible in Figure 5 for grassed pavers, as compared with unplanted permeable pavement. Site planting density, species, and root density also likely affect systems along the ET-I axis; this is a factor for constructed wetlands, bioretention cells, green roof and vegetated pavers. Site management of emergent vegetation also plays a role in limiting or encouraging ET from retention ponds and constructed wetlands. Mowing frequency, surface aeration and site management likely influence relative losses of ET vs I in detention ponds, and grassed swales. Environmental factors affecting the ET-I tradeoff include the surface roughness of contributing areas or initial abstraction of rainfall.

4 Conclusions

The benefits of using the Water Budget Triangle are four-fold:

1) Provides a visual aid to compare green and grey stormwater tools as an integrated suite of management options;
2) Eliminates non-technical uncertainty in language in favor of comparisons based on observable hydrologic behavior;
3) Facilitates communication of detailed, technical information to both scientific and lay audiences; and
4) Illuminates how environmental factors and site design affect hydrological performance, and allows simplified (two-pathway) systems to act as proxies for analysis of more complex systems.

The results of this study indicate:

1) Event-scale understanding of stormwater systems is not representative of long-term performance for GI or conventional systems; short-term monitoring underestimates ET during dry and cold periods. Green infrastructure case studies should collect measurements appropriate for the spatial and temporal scales of interest, and long-term modeling should not simply upscale event-scale measurements;

2) Cursory estimation of water budget variables may be adequate to provide an understanding of constructed system water budgets. However, more accurate estimation of ET is necessary for both living and non-living technologies to account for discrepancies between current model estimates and weighing lysimetry studies.

3) Constructed green infrastructure ecosystems may not adequately replace natural ones, as indicated by differences in natural and constructed wetlands;

4) Modest increases in volumetric reduction (30%) can achieve large peak flow attenuation (90%). Employing permanent reduction pathways (ET, I) instead of temporary storage (ΔS) is a more effective management strategy for mitigation of stormwater.

Future study recommendations:

1) Additional measurements of water budgets are necessary to better predict hydrologic performance of green infrastructure, especially retention and detention ponds.
2) A full water year of daily data is a good starting dataset for this method. Shorter duration or intermittently collected datasets are recommended to be analyzed at the event scale.

Abbreviations

BMP (best management practice); ET (evapotranspiration); GI (green infrastructure); I (percolated or infiltrated drainage); LID (low impact development); Q (discharged runoff); ΔS (change in system storage).

Acknowledgements

This material is based upon work supported by the National Science Foundation under Grant No. DGE-1449617, the Surdna Foundation under Grant No. 20140225, and a Syracuse University Fellowship awarded to CE. Gratitude to Cliff Davidson, Babak Kasaee Roodsari, Kathy Fallon Lambert, Peter Groffman, Neil Bettez and Christa Kelleher for their feedback, as well as Thomas Evans, Aditi Padhye and Megan Daley for editing and literature review. Special thanks to Nicholas Hamilton for development of the ggtern package for producing ternary plots in R. Lastly, this publication relies heavily on the hard work of many other academics, professionals and students over the past 50 years, we are grateful for their contributions of water budget data to the literature.

References

1. 2nd Nature, LLC. (2006). *Lake Tahoe BMP monitoring evaluation process: Synthesis of existing research* (Final report). Santa Cruz, CA: USFS Lake Tahoe Basin Management Unit. 82 pages.
2. Ahiablame, L. M., B. A. Engel, and I. Chaubey. (2012). Effectiveness of low impact development practices: Literature review and suggestions for future research. *Water, Air and Soil Pollution* **223**, 4253–4273.
3. Amick, R. S., and E. H. Burgess. (2003). *Exfiltration in sewer systems* (Project summary). Cincinnati, OH: EPA National Risk Management Research Laboratory. 8 pages.
4. Amick, R. S., E. H. Burgess, and A. Selvakumar. (2000). *Exfiltration in sewer systems*. Cincinnati, OH: EPA National Risk Management Research Laboratory. 41 pages.
5. Askarizadeh, A., M. A. Rippy, T. D. Fletcher, D. L. Feldman, J. Peng, P. Bowler, A. S. Mehring, B. K. Winfrey, J. A. Vrugt, A. Agha Kouchak, S. C. Jiang, B. F. Sanders, L. A. Levin, S. Taylor, and S. B. Grant. (2015). From rain tanks to catchments: Use of low-impact development to address hydrologic symptoms of the urban stream syndrome. *Environmental Science & Technology,* **49**, 11264–11280.
6. Ayub, K. R., N. A. Zakaria, R. Abdullah, and R. Ramli. (2010). Water balance: Case study of a constructed wetland as part of the bio-ecological drainage system (BIOECODS). *Water Science and Technology,* **62**,1931–1936.
7. Ballestero, T. (2012). *University of New Hampshire Stormwater Center 2012 Biennial Report*. Durham, NH: UNH Stormwater Center. 36 pages.

8. Berghage, R., D. Beattie, A. Jarrett, C. Thuring, F. Razaei, and T. O'Connor. (2009). *Green roofs for stormwater runoff control*. Cincinnati, OH: EPA National Risk Management Research Laboratory. 81 pages.

9. Berghage, R., A. Jarrett, D. Beattie, K. Kelley, S. Husain, F. Rezai, B. Long, A. Negassi, and R. Cameron. (2007). *Quantifying evaporation and transpirational water losses from green roofs and green roof media capacity for neutralizing acid rain*. State College, PA: Pennsylvania State University and North Carolina State University. 94 pages.

10. Berndtsson, J. 2010. Green roof performance towards management of runoff water quantity and quality: A review. *Ecological Engineering,* **36**, 351–360.

11. Bhaskar, A. S., and C. Welty. (2012). Water balances along an urban-to-rural gradient of metropolitan Baltimore, 2001-2009. *Environmental & Engineering Geoscience,* **18**, 37–50.

12. Bleam, W. F. (2016). *Soil and Environmental Chemistry* (2nd ed.). London: Academic Press.

13. Bratieres, K., T. Fletcher, A. Deletic, and Y. Zinger. (2008). Nutrient and sediment removal by stormwater biofilters: A large-scale design optimisation study. *Water Research,* **42**, 3930–3940.

14. Brown, H. (2014). *Next Generation Infrastructure: Principles for Post-Industrial Public Works*. Washington, DC: Island Press.

15. Brown, R. A., and M. Borst. (2015). Quantifying evaporation in a permeable pavement system. *Hydrological Processes,* **29**, 2100–2111.

16. Brown, R., W. Hunt, and R. Skaggs. (2011). *Long-term modeling of bioretention hydrology with DRAINMOD*. Raleigh, NC: Water Resources Research Institute of the University of North Carolina. 115 pages.

17. Buccola, N., and G. Spolek. (2011). A pilot-scale evaluation of greenroof runoff retention, detention, and quality. *Water, Air, and Soil Pollution,* **216**, 83–92.

18. Buckingham, E. (1907). *Studies on the movement of soil moisture*. Washington, DC: US Department of Agriculture, Bureau of Soils. 61 pages.

19. Caldwell, P. V., M. J. Vepraskas, R. W. Skaggs, and J. D. Gregory. (2007). Simulating the water budgets of natural Carolina bay wetlands. *Wetlands,* **27**, 1112–1123.

20. Carpenter, D., and P. Kaluvakolanu. (2011). Effect of roof surface type on storm-water runoff from full-scale roofs in a temperate climate. *Journal of Irrigation and Drainage Engineering,* **137**, 161–169.

21. Carson, T. B., D. E. Marasco, P. J. Culligan, and W. R. McGillis. (2013). Hydrological performance of extensive green roofs in New York City: Observations and multi-year modeling of three full-scale systems. *Environmental Research Letters,* **8**, 024036.

22. Carter, T. L., and T. C. Rasmussen. (2005). Use of green roofs for ultra-urban stream restoration in the Georgia Piedmont (USA). *Proceedings of the 2005 Georgia Water Resources Conference*. Athens, GA: SMARTech Georgia Tech Library.

23. Carter, T. L., and T. C. Rasmussen. (2006). Hydrologic behavior of vegetated roofs. *Journal of the American Water Resources Association,* **42**, 1261–1274.

24. Choi, J., and J. W. Harvey. (2000). Quantifying time-varying ground-water discharge and recharge in wetlands of the northern Florida everglades. *Wetlands,* **20**, 500–511.

25. City of Detroit Water and Sewerage Department and Michigan Department of Environmental Quality. (2001). *Long term CSO control plan update: Detroit and Rouge rivers*. Detroit, MI: MDEQ. 199 pages.

26. CNT and American Rivers. (2011). *The value of green infrastructure: A guide to recognizing its economic, environmental and social benefits.* (Technical Guide for Stakeholders). Chicago, IL: Center for Neighborhood Technology & American Rivers. 80 pages.

27. Crisp, D. T. (1966). Input and output of minerals for an area of pennine moorland: the importance of precipitation, drainage, peat erosion and animals. *Journal of Applied Ecology,* **3**, 327–348.

28. Daniels, W. L., A. Cummings, M. Schmidt, N. Fomchenko, G. Speiran, M. Focazio, and G. M. Fitch. (2000). *Evaluation of methods to calculate a wetlands water balance*. (Final Contract Report). Charlottesville, VA: Virginia Transportation Research Council. 70 pages.

29. De Sousa, M., F. A. Montalto, and S. Spatari. (2012). Using life cycle assessment to evaluate green and grey combined sewer overflow control strategies. *Journal of Industrial Ecology,* **16**, 901–913.

30. Denich, C., and A. Bradford. (2010). Estimation of evapotranspiration from bioretention areas using weighing lysimeters. *Journal of Hydrologic Engineering,* **15**, 522–530.

31. Drake, J., A. Bradford, and T. Van Seters. (2014). Hydrologic performance of three partial-infiltration permeable pavements in a cold climate over low permeability soil. *Journal of Hydrologic Engineering,* **19**, 04014016-1–9.

32. Driscoll, C. T., C. G. Eger, D. G. Chandler, C. I. Davidson, B. K. Roodsari, C. D. Flynn, K. F. Lambert, N. D. Bettez, and P. M. Groffman. (2015). *Green infrastructure: Lessons from science and practice*. Syracuse, NY: Science Policy Exchange. 32 pages.

33. Eger, C. G. (2012). *Nutrient retention in roadside retrofit rain gardens* (Master's thesis). Columbus, OH: The Ohio State University. Retrieved from OhioLINK Electronic Theses

and Dissertations Center database http://rave.ohiolink.edu/etdc/view?acc_num=osu1343837490

34. Eger, C. G., D. G. Chandler, B. K. Roodsari, C. I. Davidson, and C. T. Driscoll. (2014). Water Budget Triangle: A new conceptual framework for comparison of green and gray infrastructure. *Proceedings of the American Society of Civil Engineers International Conference on Sustainable Infrastructure 2014* (pp. 1010–1017). Long Beach, CA: ASCE.

35. Ellis, J. B., D. M. Revitt, P. Lister, C. Willgress, and A. Buckley. (2003). Experimental studies of sewer exfiltration. *Water Science and Technology,* **47**, 61–67.

36. Emerson, C. H. (2003). *Evaluation of the additive effects of stormwater detention basins at the watershed scale* (Master's thesis). Philadelphia, PA: Drexel University. Retrieved from https://pdfs.semanticscholar.org/a089/5a3e908036b21f5e26a4c5c5845a8721e672.pdf

37. EPA Green Infrastructure Technical Assistance Program. (2013). *Green infrastructure opportunities and barriers in the greater Los Angeles region*. Los Angeles, CA: US Environmental Protection Agency. 54 pages.

38. Fassman-Beck, E., W. Hunt, R. Berghage, D. Carpenter, T. Kurtz, V. Stovin, and B. Wadzuk. (2016). Curve number and runoff coefficients for extensive living roofs. *Journal of Hydrologic Engineering,* **21**, 04015073.

39. Fassman-Beck, E., E. Voyde, R. Simcock, and Y. S. Hong. (2013). 4 Living roofs in 3 locations: Does configuration affect runoff mitigation? *Journal of Hydrology,* **490**, 11–20.

40. Fioretti, R., A. Palla, L. G. Lanza, and P. Principi. (2010). Green roof energy and water related performance in the Mediterranean climate. *Building and Environment,* **45**, 1890–1904.

41. Fletcher, T., W. Shuster, W. Hunt, R. Ashley, D. Butler, S. Arthur, S. Trowsdale, S. Barraud, A. Semadeni-Davies, J.-L. Bertrand-Krajewski, P. Steen Mikkelsen, G. Rivard, M. Uhl, D. Dagenais and M. Viklander. (2015). SUDS, LID, BMPs, WSUD and more – The evolution and application of terminology surrounding urban drainage. *Urban Water Journal*, **12**, 525-542.

42. Geosyntec, and Wright Water Engineers. (2011). *Technical Summary: Volume Reduction*. International Stormwater BMP Database. 35 pages.

43. Getter, K., D. Rowe, and J. Andresen. (2007). Quantifying the effect of slope on extensive green roof stormwater retention. *Ecological Engineering,* **31**, 225–231.

44. Göbel, P., P. Starke, A. Voss, and W. G. Coldewey. (2013). Field measurements of evapotranspiration rates on seven pervious concrete pavement systems. *Novatech 2013 Conference Proceedings.* Lyon, France: GRAIE.

45. Graceson, A., M. Hare, J. Monaghan, and N. Hall. (2013). The water retention capabilities of growing media for green roofs. *Ecological Engineering,* **61**, 328–334.

46. Green Infrastructure for New Hampshire Coastal Communities. (2014). *Overcoming barriers to green infrastructure factsheet*. Durham, NH: UNH Stormwater Center.

47. Gregoire, B., and J. Clausen. (2011). Effect of a modular extensive green roof on stormwater runoff and water quality. *Ecological Engineering,* **37**, 963–969.

48. Guizani, M. (2016). Storm water harvesting in Saudi Arabia: a multipurpose water management alternative. *Water Resources Management, 2016*, 1819–1833.

49. Hamilton, N. (2016*). ggtern: An extension to "ggplot2", for the creation of ternary diagrams*. R package version 2.2.1. www.ggtern.com

50. Harper, H. H. (2010a). *Navy Canal stormwater facility performance efficiency evaluation*. (Final Report). Seminole County, FL: Environmental Research & Design, Inc. 102 pages.

51. Harper, H. H. (2010b). *Evaluation of the performance efficiency of the Poppleton Creek wet detention pond*. (Final Report). Stuart, FL: Environmental Research & Design, Inc. 84 pages.

52. Harper, H. H. (2010c). *Elder Creek stormwater facility performance efficiency evaluation*. (Final Report). Seminole County, FL: Environmental Research & Design, Inc. 126 pages.

53. Harper, H. H. (2011). *Performance efficiency evaluation of the Club II regional stormwater facility (RSF)*. (Final Report). Seminole County, FL: Environmental Research & Design, Inc. 233 pages.

54. Harper, H. H., D. M. Baker, and C. Harper. (2002). *Effectiveness of stormwater treatment systems in the Florida Keys*. (Final Report). Orlando, FL: Environmental Research & Design, Inc. for Florida DEP. 140 pages.

55. Harper, H. H., D. M. Baker, and C. Harper. (2003). *Performance evaluation of the Palm Bay Basin 7 wet detention pond*. (Final Report) Orlando, FL: Environmental Research & Design, Inc. 61 pages.

56. Harper, H. H., J. L. Herr, D. M. Baker, and E. H. Livingston. (1999). Performance evaluation of dry detention stormwater management systems. *Proceedings of the 6th Biennial Stormwater Research and Watershed Management Conference* (pp. 162–178). Tampa, FL: Southwest Florida Water Management District.

57. Hartigan, P., and M. Kelly. (2009). *Daily wet pond water balance spreadsheet*. (Excel sheet). Austin, TX: Watershed Protection Department Stormwater Treatment Section.

58. Hathaway, A., W. Hunt, and G. Jennings. (2008). A field study of green roof hydrologic and water quality performance. *Transactions of the ASABE,* **51**, 37–44.

59. Hein, D. K., E. Strecker, A. Poresky, R. Roseen, and M. Venner. (2013). *Permeable shoulders with stone reservoirs*. (Final Report). Applied Research Associates, Inc. and Geosyntec Consultants for the National Cooperative Highway Research Program, Transportation Research Board, Project 25-25, Task 82.

60. Hemond, H. F. (1980). Biogeochemistry of Thoreau's Bog, Concord, Massachusetts. *Ecological Monographs,* **50**, 507–526.

61. Hess, A. (2014). *Monitoring of evapotranspiration and infiltration in rain garden designs* (Master's thesis). Villanova, PA: Villanova University. Retrieved from https://www1.villanova.edu/content/dam/villanova/engineering/vcase/vusp/AHess_Thesis.pdf

62. Hess, A., B. Wadzuk, and A. Welker. (2015). evapotranspiration and infiltration in rain garden systems. Pages 261–270 *in Floods, Droughts and Ecosystems.* ASCE.

63. Hess, A., B. Wadzuk, and A. Welker. (2017). Evapotranspiration in rain gardens using weighing lysimeters. *Journal of Irrigation and Drainage Engineering,* **143**, 6.

64. Hey, D. L., K. R. Barrett, and C. Biegen. (1994). The hydrology of four experimental constructed marshes. *Ecological Engineering,* **3**, 319–343.

65. Hickman, J. M. (2011). *Evaluating the role of evapotranspiration in the hydrology of bioinfiltration and bioretention basins using weighing lysimeters* (Master's thesis). Villanova, PA: Villanova University. Retrieved from https://www1.villanova.edu/content/dam/villanova/engineering/vcase/vusp/Hickmen-Thesis%202011.pdf

66. Hickman, J. M., B. M. Wadzuk, and R. G. Traver. (2011). Evaluating the role of evapotranspiration in the hydrology of a bioinfiltration basin using a weighing lysimeter. Pages 3601–3609 *in Bearing Knowledge for Sustainability*. ASCE.

67. Hirschman, D., K. A. Collins, and T. Schueler. (2008). *The runoff reduction method*. (Technical Memorandum). Ellicott City, MD: Center for Watershed Protection & Chesapeake Stormwater Network. 25 pages.

68. Hoffman, L., G. Loosvelt, and R. Berghage. (2010). *Green roof thermal and stormwater management performance: The Gratz Building case study, New York City*. New York, NY: Pratt Center for Community Development and New York State Energy Research and Development Authority. 140 pages.

69. Hollands, G. G. (1989). Regional analysis of the creation and restoration of kettle and pothole wetlands. Pages 287–304 *in* J. A. Kusler and M. E. Kentula, editors. *Wetland Creation and Restoration: The Status of the Science, Vol 1.* U.S. Environmental Protection Agency, Environmental Research Laboratory.

70. Hughes, C. E., P. Binning, and G. R. Willgoose. (1998). Characterization of the hydrology of an estuarine wetland. *Journal of Hydrology,* **1998**, 34–49.

71. Hutchinson, D., P. Abrams, R. Retzlaff, and T. Liptan. (2003). Stormwater monitoring two ecoroofs in Portland, Oregon, USA. *Proceedings for Sustainable Communities Conference, 2003* (18 pp). Chicago, IL: Chicago Greening Rooftops.

72. Jones, J. A., I. F. Creed, K. L. Hatcher, R. J. Warren, M. B. Adams, M. H. Benson, E. Boose, W. A. Brown, J. L. Campbell, A. Covich, D. W. Clow, C. N. Dahm, K. Elder, C. R. Ford, N. B. Grimm, D. L. Henshaw, K. L. Larson, E. S. Miles, K. M. Miles, S. D. Sebestyen, A. T. Spargo, A. B. Stone, J. M. Vose, and M. W. Williams. (2012). Ecosystem

processes and human influences regulate streamflow response to climate change at long-term ecological research sites. *BioScience,* **62**, 390–404.

73. Kosmerl, P. (2012). *Water balance of retrofit, right-of-way rain gardens* (Master's thesis). Columbus, OH: The Ohio State University. Retrieved from OhioLINK Electronic Theses and Dissertations Center database https://etd.ohiolink.edu/rws_etd/document/get/osu1337347745/inline

74. Lent, R. M., P. K. Weiskel, F. P. Lyford, and D. S. Armstrong. (1997). Hydrologic indices for nontidal wetlands. *Wetlands,* **17**, 19–30.

75. Li, H., L. Sharkey, W. Hunt, and A. Davis. (2009). Mitigation of impervious surface hydrology using bioretention in North Carolina and Maryland. *Journal of Hydrologic Engineering,* **14**, 407–415.

76. Liu, K., J. Minor, and City of Toronto. (2005). *Performance evaluation of an extensive green roof*. Toronto: Institute for Research in Construction and National Research Council Canada.

77. Low Impact Development Center. (2014). LID Center Projects website. http://www.lowimpactdevelopment.org/design.htm

78. Lucas, W. C., and D. J. Sample. (2015). Reducing combined sewer overflows by using outlet controls for green stormwater infrastructure: Case study in Richmond, Virginia. *Journal of Hydrology,* **520**, 473–488.

79. Martin, W. D., and N. B. Kaye. (2014). Hydrologic characterization of undrained porous pavements. *Journal of Hydrologic Engineering,* **19**, 1069–1079.

80. Matthews, T., A. Y. Lo, and J. A. Byrne. (2015). Reconceptualizing green infrastructure for climate change adaptation: Barriers to adoption and drivers for uptake by spatial planners. *Landscape and Urban Planning,* **138**, 155–163.

81. Melbourne Water. (2016). Water sensitive urban design (WSUD) website. http://www.melbournewater.com.au/wsud

82. Mentens, J., D. Raes, and M. Hermy. (2006). Green roofs as a tool for solving the rainwater runoff problem in the urbanized 21st century? *Landscape and Urban Planning,* **77**, 217–226.

83. Millar, G., B. Yu, and T. Gardner. (2003). Rainfall catch efficiency for domestic water supply. *Proceedings of the 28th International Hydrology and Water Resources Symposium*. Wollongong, New South Wales: Institution of Engineers.

84. Mitsch, W. J., L. Zhang, E. Waletzko, and B. Bernal. (2014). Validation of the ecosystem services of created wetlands: Two decades of plant succession, nutrient retention, and carbon sequestration in experimental riverine marshes. *Ecological Engineering,* **72**, 11–24.

85. Moran, A., W. Hunt, and J. Smith. (2004). *Hydrological and water quality performance from greenroofs in Goldsboro and Raleigh, North Carolina*. Raleigh, NC: North Carolina State University Biological and Agricultural Engineering.

86. Morgan, S., S. Celik, and W. Retzlaff. (2013). Green roof storm-water runoff quantity and quality. *Journal of Environmental Engineering,* **139**, 4.

87. Nawaz, R., A. McDonald, and S. Postoyko. (2015). Hydrological performance of a full-scale extensive green roof located in a temperate climate. *Ecological Engineering,* **82**, 66–80.

88. NERR. (2016). *Implementing credits and incentives for innovative stormwater management.* (Final Report). North Carolina State University and Ohio Department of Natural Resources for the National Estuarine Research Reserve System Science Collaborative, NOAA Grant Number NA09NOS4190153. 600 pages.

89. Newcomer, M. E., J. J. Gurdak, L. S. Sklar, and L. Nanus. (2014). Urban recharge beneath low impact development and effects of climate variability and change. *Water Resources Research,* **50**, 1716–1734.

90. Nungesser, M. K., and M. J. Chimney. (2006). A hydrologic assessment of the Everglades Nutrient Removal Project, a subtropical constructed wetland in South Florida (USA). *Ecological Engineering,* **2006**, 331–344.

91. NY State Department of Environmental Conservation. (2013). Runoff Reduction Worksheets. http://www.dec.ny.gov/chemical/8694.html

92. NY State Department of Environmental Conservation and Center for Watershed Protection. (2015). *New York state stormwater management design manual*. (Design Manual). Albany, NY: NYSDEC. 578 pages.

93. PA Department of Environmental Protection. (2006). *Pennsylvania stormwater best management practices manual*. (Design Manual). Harrisburg, PA: PA DEP Bureau of Water Management.

94. Palla, A., J. J. Sansalone, I. Gnecco, and L. G. Lanza. (2011). Storm water infiltration in a monitored green roof for hydrologic restoration. *Water Science and Technology,* **64**, 766–773.

838 95. Pitt, R., S. Clark, and J. Voorhees. (2007). Water removal in bioretention devices by evapotranspiration processes and related issues affecting performance. *Intelligent Modeling to Improve Stormwater Management.* Arcata, CA: ASCE.

841 96. Poff, N. L., and J. K. Zimmerman. (2010). Ecological responses to altered flow regimes: A literature review to inform the science and management of environmental flows. *Freshwater Biology,* **55**, 194–205.

844 97. Pratt, C. J., J. D. G. Mantle, and P. A. Schofield. (1995). UK research into the performance of permeable pavement, reservoir structures in controlling stormwater discharge quantity and quality. *Water Science and Technology,* **32**, 63–69.

847 98. Rieckermann, J., M. Borsuk, P. Reichert, and W. Gujer. (2005). A novel tracer method for estimating sewer exfiltration. *Water Resources Research,* **41**, W05013.

849 99. Rim, Y.-N. (2011). *Analyzing runoff dynamics of paved soil surface using weighable lysimeters* (Doctoral dissertation). Berlin, Germany: Technischen Universität Berlin. Retrieved from http://dx.doi.org/10.14279/depositonce-2826

852 100. Rodell, M., H. K. Beaudoing, T. S. L'Ecuyer, W. S. Olson, J. S. Famiglietti, P. R. Houser, R. Adler, M. G. Bosilovich, C. A. Clayson, D. Chambers, E. Clark, E. J. Fetzer, X. Gao, G. Gu, K. Hilburn, G. J. Huffman, D. P. Lettenmaier, W. T. Liu, F. R. Robertson, C. A. Schlosser, J. Sheffield, and E. F. Wood. (2015). The observed state of the water cycle in the early twenty-first century. *Journal of Climate,* **28**, 8289–8318.

857 101. Roy-Poirier, A., P. Champagne, and Y. Filion. (2010). Review of bioretention system research and design: past, present, and future. *Journal of Environmental Engineering,* **136**, 878.

102. Rutsch, M. (2006). *Assessment of sewer leakage by means of exfiltration measurements and modelling tests* (Doctoral dissertation). Dresden, Germany: Technischen Universität Dresden. Retrieved from http://www.qucosa.de/recherche/frontdoor/?tx_slubopus4frontend%5bid%5d=urn:nbn:de:swb:14-1175685350577-01644

103. Rutsch, M., J. Rieckermann, and P. Krebs. (2005). *Quantification of sewer leakage - a review*. Copenhagen, Denmark: Technical University of Denmark Institute of Environment & Resources. 9 pages.

104. Schlea, D. (2011). *Retention and management of stormwater runoff with rain gardens and rainwater harvesting systems* (Master's thesis). Columbus, OH: The Ohio State University. Retrieved from OhioLINK Electronic Theses and Dissertations Center database https://etd.ohiolink.edu/rws_etd/document/get/osu1306853271/inline

105. Shukla, A., S. Shukla, and M. D. Annable. (2015). Using nocturnal water level fluctuations for estimating seepage from stormwater detention systems. *Hydrological Processes,* **29**, 5465–5476.

106. Starke, P., P. Göbel, and W. G. Coldewey. (2010). Urban evaporation rates for water-permeable pavements. *Water Science and Technology,* **6**, 1161.

107. Steffen, J., M. Jensen, C. A. Pomeroy, and S. J. Burian. (2013). Water supply and stormwater management benefits of residential rainwater harvesting in U.S. cities. *Journal of the American Water Resources Association,* **49**, 810–824.

108. Stovin, V. (2010). The potential of green roofs to manage urban stormwater. *Water and Environment Journal,* **24**, 192–199.

109. Stovin, V., S. Poe, and C. Berretta. (2013). A modelling study of long term green roof retention performance. *Journal of Environmental Management,* **131**, 206–215.

110. Stovin, V., G. Vesuviano, and S. De-Ville. (2015). Defining green roof detention performance. *Urban Water Journal,* **0**, 1–15.

111. Stovin, V., G. Vesuviano, and H. Kasmin. (2012). The hydrological performance of a green roof test bed under UK climatic conditions. *Journal of Hydrology,* **414–415**, 148–161.

112. Strauch, K., D. Rus, and K. Holm. (2016). *Water balance monitoring for two bioretention gardens in Omaha, Nebraska, 2011-14*. (Scientific Investigation Report). Reston, VA: US Geological Survey. 19 pages.

113. Strosnider, W. H., Hitchcock, D. R, Burke, M. K, Lewitus, and A. J. (2007). Predicting hydrology in wetlands designed for coastal stormwater management (Paper Number 077084). *Proceedings of the American Society of Agricultural and Biological Engineers 2007* (17 pp.). St Joseph, MI: ASABE.

114. Stroud Water Research Center. (2016). Model My Watershed: Simple Water Balance Simulator. http://www.stroudcenter.org/mmw/mini/

115. Teague, K., and B. Rushton. (2005). *Stormwater runoff treatment by a filtration system and wet pond in Tampa, Florida*. (Final Report). Brooksville, FL: Southwest Florida Water Management District. 237 pages.

116. Teemusk, A., and Ü. Mander. (2007). Rainwater runoff quantity and quality performance from a greenroof: The effects of short-term events. *Ecological Engineering,* **30**, 271–277.

117. Thorne, C., E. Lawson, C. Ozawa, S. Hamlin, and L. Smith. (2015). Overcoming uncertainty and barriers to adoption of Blue-Green Infrastructure for urban flood risk management. *Journal of Flood Risk Management*.

118. Traver, R., and P. DeBarry. (2003). *Designing for infiltration — A perspective*. (World Water & Environmental Resources Congress 2003). Philadelphia, PA: ASCE. Pages 1–7.

119. TRCA. (2006). *Evaluation of an extensive green roof York University, Toronto, Ontario*. Toronto: Sustainable Technologies Evaluation Program, Toronto and Region Conservation Authority. 167 pages.

120. US EPA Office of Water. (2016). Green Infrastructure webpage. http://www.epa.gov/green-infrastructure. Accessed January 27, 2016.

121. Valinski, N. A., and D. G. Chandler. (2015). Infiltration performance of engineered surfaces commonly used for distributed stormwater management. *Journal of Environmental Management,* **160**, 297–305.

122. Van Seters, T., L. Rocha, D. Smith, and G. MacMillan. (2009). Evaluation of green roofs for runoff retention, runoff quality and leachability. *Water Quality Research Journal of Canada,* **44**, 33–47.

123. Vanuytrecht, E. and C. Van Mechelen. (2014). Runoff and vegetation stress of green roofs under different climate change scenarios. *Landscape and Urban Planning,* **122,** 68–77.

124. VanWoert, N. D., D. B. Rowe, J. A. Andresen, C. L. Rugh, R. T. Fernandez, and L. Xiao. (2005). Green roof stormwater retention: Effects of roof surface, slope, and media depth. *Journal of Environmental Quality,* **34**, 1036–1044.

125. Vico, G., R. Revelli, and A. Porporato. (2014). Ecohydrology of street trees: design and irrigation requirements for sustainable water use. *Ecohydrology,* **7**, 508–523.

126. Villarreal, E. L., and L. Bengtsson. (2005). Response of a sedum green-roof to individual rain events. *Ecological Engineering,* **25**, 1–7.

127. Vineyard, D., W. W. Ingwersen, T. R. Hawkins, X. Xue, B. Demeke, and W. Shuster. (2015). Comparing green and grey infrastructure using life cycle cost and environmental impact: A rain garden case study in Cincinnati, OH. *Journal of the American Water Resources Association,* **51**, 1342–1360.

128. Voyde, E., E. Fassman, and R. Simcock. (2010). Hydrology of an extensive living roof under sub-tropical climate conditions in Auckland, New Zealand. *Journal of Hydrology,* **394**, 384–395.

129. Wadzuk, B. M., J. M. Hickman, and R. G. Traver. (2015). Understanding the role of evapotranspiration in bioretention: Mesocosm study. *Journal of Sustainable Water in the Built Environment,* **1**, 04014002.

130. Wadzuk, B., D. Schneider, M. Feller, and R. Traver. (2013). Evapotranspiration from a green-roof storm-water control measure. *Journal of Irrigation and Drainage Engineering,* **139**, 9.

131. Walsh, C. J., A. H. Roy, J. W. Feminella, P. D. Cottingham, P. M. Groffman, and R. P. Morgan. (2005). The urban stream syndrome: current knowledge and the search for a cure. *Journal of the North American Benthological Society,* **24**, 706–723.

132. Wardynski, B., W. Hunt, and R. Brown. (November 30, 2011). Ohio EPA Municipal Storm Water Training Series, Low Impact Development (LID) Stormwater Educational Workshop, Part 6: “DRAINMOD: Introduction to the Ideal Bioretention Model.” Columbus, OH.

133. Wardynski, B. J., and W. F. Hunt. (2012). Are bioretention cells being installed per design standards in North Carolina? A field study. *Journal of Environmental Engineering,* **138**, 1210–1217.

134. Water Environment Federation and American Society of Civil Engineers. (1998). *Urban Runoff Quality Management.* Alexandria and Reston, VA: ASCE.

135. Water, Environment and Research Foundation, Environmental Water Research Institute, US Department of Transportation, American Public Works Association, Wright Water Engineers, Geosyntec, and US Environmental Protection Agency. (2014). International Storwmater BMP Database. http://www.bmpdatabase.org/index.htm

136. Winston, R., J. Dorsey, and W. Hunt. (2016). Quantifying volume reduction and peak flow mitigation for three bioretention cells in clay soils in northeast Ohio. *Science of the Total Environment,* **553**, 83–95.

137. Worstell, C. (2013). *Green infrastructure in the state of New Jersey: Statutory and regulatory barriers to green infrastructure implementation*. Trenton, NJ: New Jersey Future. 27 pages.

138. Xu, Z., H. Yin, and H. Li. (2014). Quantification of non-stormwater flow entries into storm drains using a water balance approach. *Science of the Total Environment,* **487**, 381–388.

139. Zahmatkesh, Z., S. J. Burian, M. Karamouz, H. Tavakol-Davani, and E. Goharian. (2015). Low-impact development practices to mitigate climate change effects on urban stormwater runoff: Case study of New York City. *Journal of Irrigation and Drainage Engineering,* **141**, 04014043.

140. Zhang, X., and M. Hu. (2014). Effectiveness of rainwater harvesting in runoff volume reduction in a planned industrial park, China. *Water Resources Management,* **2014**, 671–682.

Tables

Table I Comparative summary of hydrologic function of stormwater technologies and natural systems. *Mean and median values from the compiled datasets for each system type, calculated for each fractional variable of the water budget (Q, I and ET). Note that the values represent the mean or median of each fractional loss pathway calculated independently from the other fractions; totals may not sum to exactly 100.*

		Mean			Median		
System	N	Q	I	ET	Q	I	ET
Porous pavement	23	34	54	11	20	64	10
Continent*	6	37	7	55	38	7	54
Natural lake	13	38	15	47	45	12	43
Green roof	59	39	0.30	61	36	0	64
Bioretention cell*	10	46	16	37	52	17	28
Cistern*	36	43	55	3	41	58	1.5
Detention pond*	7	43	37	21	48	28	24
Natural wetland	19	46	11	42	45	5	46
Constructed wetland	8	68	19	13	71	10	8
Retention pond	9	73	5.3	21	85	0.20	8.4
Sewer pipe section*	13	76	24	0	83	17	0
Sewershed network*	12	88	12	0	91	9	0

*calculation includes at least one model estimate

Table II Diagnostic operational ranges for engineered stormwater systems. *Estimates of short-term operational range for each engineered system, calculated as the 5th and 95th quantile of the dataset, rounded to the nearest 5th percentile. Estimates of long-term performance, which is the 5th and 95th percentile confidence interval range about the calculated mean, estimated by resampling the distribution of means 1000 times and rounding to the nearest percentile. Confidence intervals and quantiles were calculated for each loss variable separately and may not sum to 100.* **Bold values** *represent worst-case performance scenarios, underlined values represent good performance design benchmarks.*

		5th - 95th%ile Short-term Operating Range			5th - 95th CI about the Mean Long-term Performance		
System	N	Q	I	ET	Q	I	ET
Sewershed network*	12	65 - 100	0 - 35	0	**81** - 93	7 - 19	0
Retention pond*	9	30 - 95	0 - 25	0 - 70	*58* - **87**	0 - 13	8 - 37
Sewer pipe section*	13	50 - 90	10 - 50	0	**68** - 83	17 - 32	0
Constructed wetland	8	25 - 95	0 - 60	0 - 40	*51* - **81**	6 - 35	6 - 22
Bioretention cell	10	10 - 75	0 - 30	20 - 65	*33* - **59**	11 - 22	29 - 47
Detention pond*	7	10 - 70	15 - 70	10 - 35	*29* - **57**	24 - 50	15 - 27
Cistern*	36	20 - 70	30 - 80	0 - 10	*39* - **48**	50 - 59	2 - 4
Green roof	59	20 - 70	0	30 - 80	*36* - **43**	0	57 - 64
Porous pavement	23	0 - 80	5 - 90	5 - 20	*26* - **44**	45 - 63	10 - 13

*calculation includes at least one model estimate

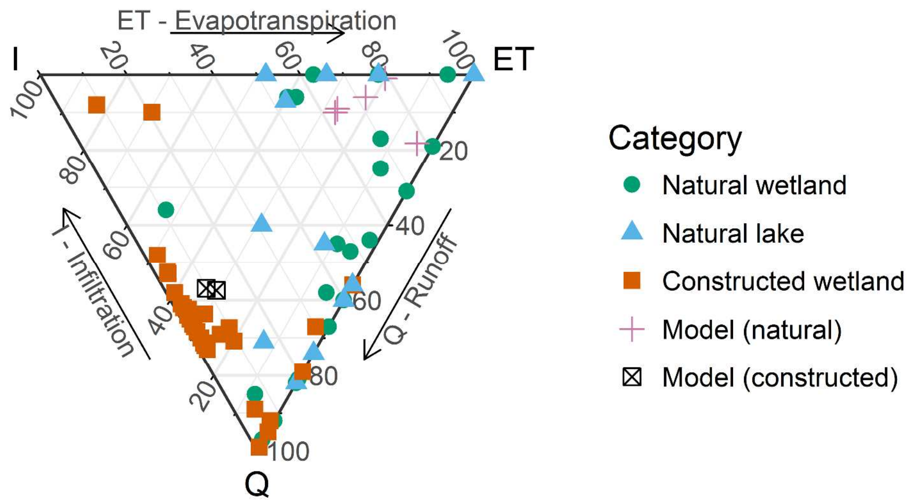


Figure 1 Comparative variability in hydrologic performance of natural wetlands and lakes (n = 21 wetlands and 13 lakes) with data reported for eight constructed wetlands (n = 34 annual or greater measurements). Results from two modeling efforts were used to estimate modeled water budgets in natural wetlands (n = 5 model estimates) and constructed wetlands (n = 2 model estimates). Data compiled from Crisp 1966, Hemond 1980, Hey et al. 1994, Lent et al. 1997, Daniels et al. 2000, Choi and Harvey 2000, Nungesser and Chimney 2006, Strosnider et al. 2007, Caldwell et al. 2007, Ayub et al. 2010, Mitsch et al. 2014.

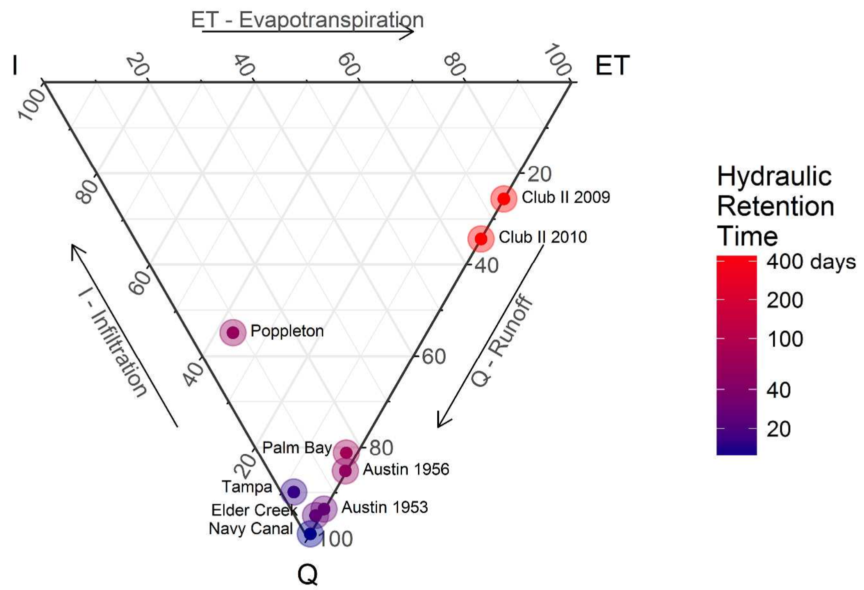


Figure 2 Cumulative water budgets for seven retention ponds and modeled wet pond performance in Austin, Texas for 1953 and 1956 (n = 9). Symbol color represents measured or estimated hydraulic retention times calculated from basin volume and average pond influx per day. Data compiled from Harper et al. 2003, Teague and Rushton 2005, Hartigan and Kelly 2009, Harper 2010a, 2010b, 2010c, 2011.

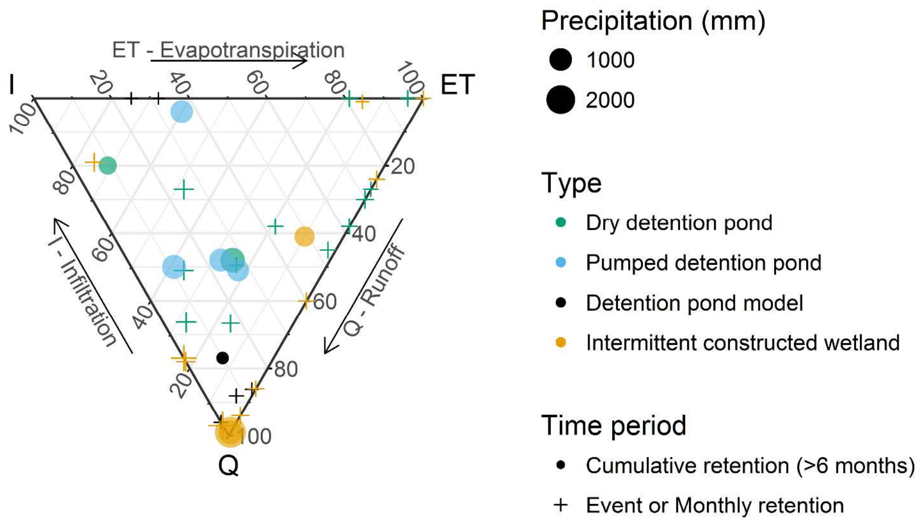


Figure 3 Circles display cumulative water budgets from gravity-fed detention ponds (n=2) and pump-fed detention ponds (n=4, two annual measurements for two ponds), a Green-Ampt detention infiltration model (n=1) and constructed wetland systems that report having detention facilities (n = 3). Crosses show the single-event or monthly retention variation for the same detention systems (n = 48). Data complied from Harper et al. 1999, 2002, Daniels et al. 2000, Emerson 2003, Ayub et al. 2010, Shukla et al. 2015.

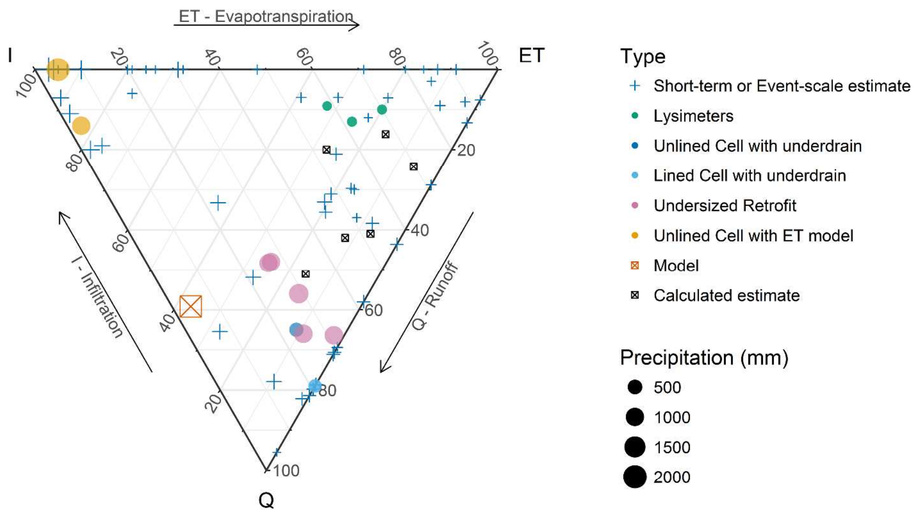


Figure 4 Water budgets for three lysimeters in Pennsylvania (green), a pair of lined and unlined cells with underdrains from North Carolina (light and dark blue), measurements from two sets of undersized, unlined retrofits with underdrains in Ohio (pink), and two unlined cells from Nebraska with ET fraction estimated using the Penman-Monteith method (yellow). Several shorter-term estimates from the same locations are also presented (blue crosses, n = 59), along with calculated estimates using volumetric moisture content constraints (black boxes, n = 6) and a DRAINMOD estimate with very low ET (orange box, n=1). Data from Pitt et al. 2007, Li et al. 2009, Wardynski et al. 2011, Kosmerl 2012, Hess 2014, Strauch et al. 2016.

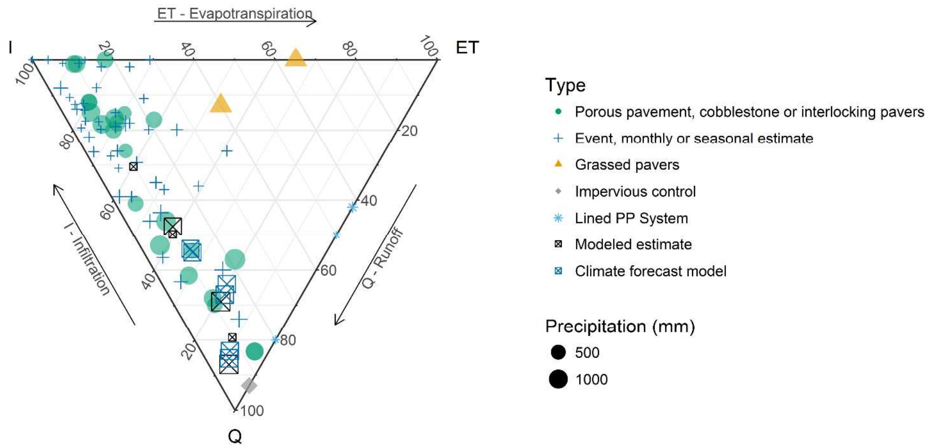


Figure 5 Reported for water budgets from unplanted porous aphalt, permeable concrete, cobblestone and interlocking or tongue-and-groove pavers (n = 15, some estimated more than once) alongside models (n = 12) and short-term measurements (n = 43) for the same locations. For reference, estimates for grassed pavers (n = 2), an impervious surface (n = 1) and lined porous pavement systems (n = 3) are also presented. Data compiled from Pratt et al. 1995, Rim 2011, Göbel et al. 2013, Drake et al. 2014, Brown and Borst 2015, NERR 2016.

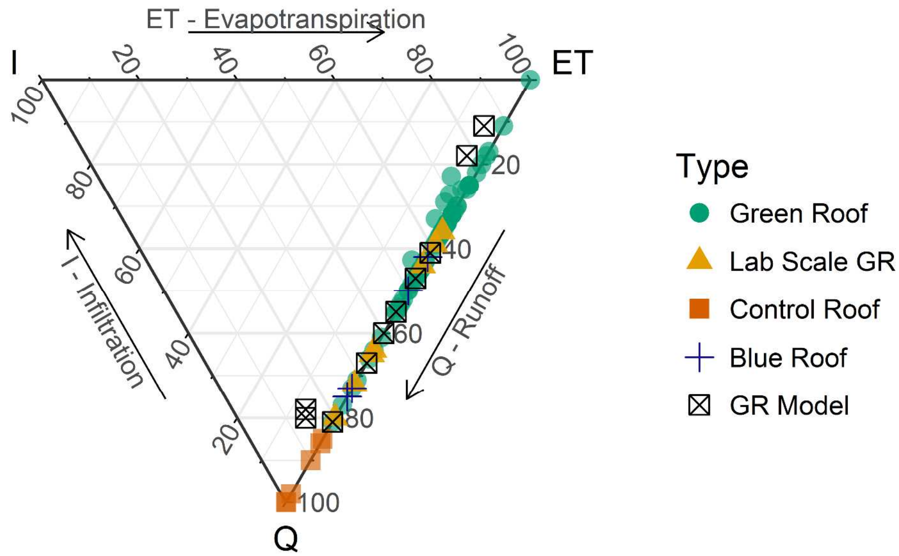


Figure 6 Eighty-seven water budgets representing installed green roofs (n = 59, in green), lab-scale study roofs (n = 7, in yellow), measured control roofs (n = 6, in orange), blue roofs (consisting of of gravel or other unplanted substrate, n = 4, in blue), and green roof models (n = 10, in black). In some cases, the initial abstraction value was reported; this value was used to estimate I for green roofs and ET for control roofs, otherwise the I is assumed to be zero because it is minimal over long timescales. Data compiled from Hutchinson et al. 2003, Liu et al. 2005, Moran et al. 2005, Carter and Rasmussen 2005, 2006, VanWoert et al. 2005, Villarreal and Bengtsson 2005, TRCA 2006, Mentens et al. 2006, Berghage et al. 2007, 2009, Teemusk and Mander 2007, Getter et al. 2007, Hathaway et al. 2008, Van Seters et al. 2009, Berndtsson 2010, Fioretti et al. 2010, Stovin 2010, Voyde et al. 2010, Hoffman et al. 2010, Palla et al. 2011, Carpenter and Kaluvakolanu 2011, Buccola and Spolek 2011, Gregoire and Clausen 2011, Stovin et al. 2012, 2013, Ahiablame et al. 2012, Carson et al. 2013, Wadzuk et al. 2013, Fassman-Beck et al. 2013, Vanuytrecht et al. 2014, Nawaz et al. 2015.

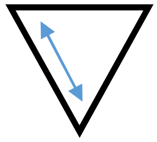
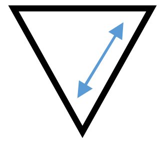
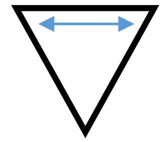
Factors affecting tradeoff between two water budget variables	I - Q axis 	Q - ET axis 	ET - I axis 
Physical, Structural & Internal Design Factors	• System capture volume or ponding depth • Contributing catchment area • Direct connection of impervious surfaces • Presence of drain • Presence of liner	• Presence of internal water storage zone or standing water • Particle size distribution • Particle surface chemistry • Media depth	• Planting density & species composition • Site management practices
External, Site & Environmental Design Factors	• Hydraulic conductivity of sub-base • Plant Establishment • Particle clogging • Event depth & intensity	• Season & temperature • Groundwater table height	• Surface roughness or Initial abstraction

Figure 7. Design factors affecting hydrologic performance. *Design factors that primarily drive a tradeoff between two water budget variables while remaining isometric in proportion to the third variable (holding all other design variables constant). Arrows represent visual direction of influence when data is plotted on a water budget triangle.*

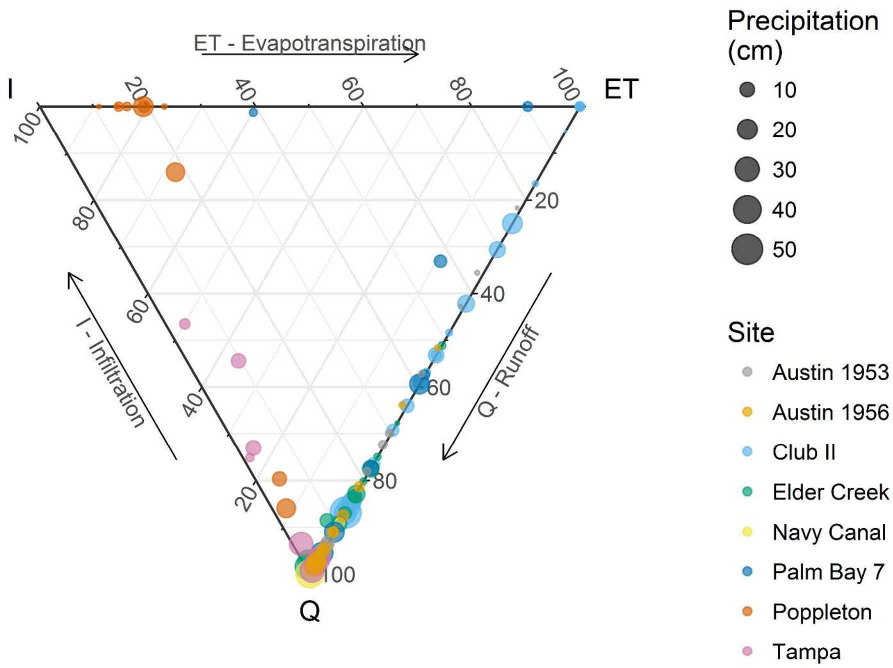


Figure S1 106 monthly water budgets from monitoring reports of seven retention ponds in Florida (Club II, Elder Creek, Navy Canal, Palm Bay 7, Poppleton and Tampa) and two years of modeled wet pond performance in Austin, Texas (Austin 1953, Austin 1956). Symbols are sized by monthly precipitation depth. Data compiled from Harper et al. 2003, Teague and Rushton 2005, Hartigan and Kelly 2009, Harper 2010a, 2010b, 2010c, 2011.

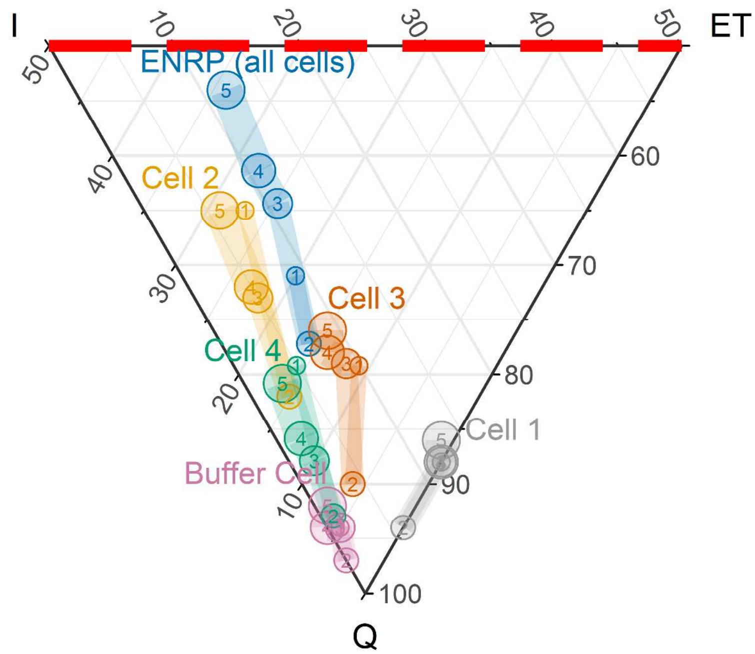


Figure S2 Temporal patterns of water budgets from the first five years of monitoring the Everglades Nutrient Removal Project (ENRP), a constructed wetland site in Florida, USA. Circled numbers represent the year of operation. Each colored line series represents data from one of the five cells in the constructed wetland. Note that the scale is expanded to show only the bottom 50% of the ternary diagram. The second year shows a decrease for all cells, likely due to a combined ecosystem establishment period and higher total influx in year 2. Overall,the ENRP's volumetric reduction of stormwater improves approximately 25% over time (in blue); ~20% is attributable to increased groundwater infiltration and ~5% attributable to greater ET. Data from Nungesser and Chimney (2006).

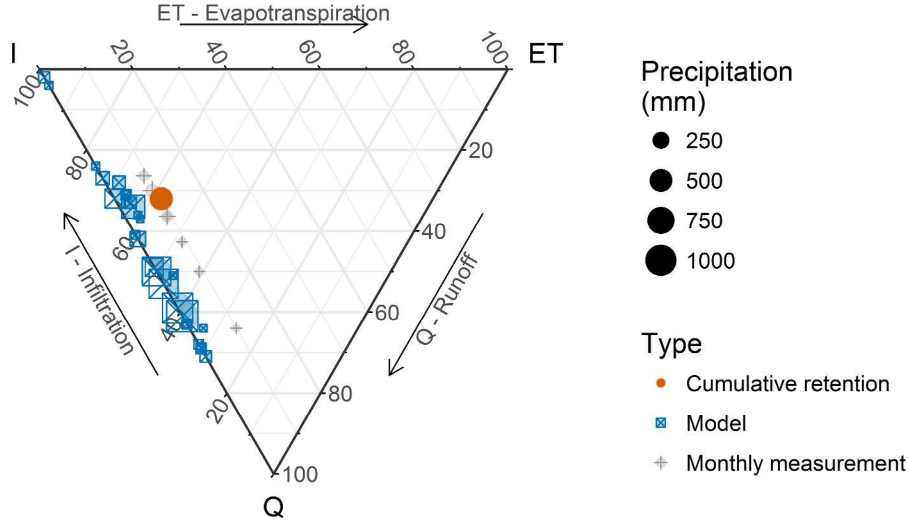


Figure S3 Reported cumulative and monthly water budgets from a cistern in Queensland, Australia (n = 1 long-term measurement and 8 monthly measurements), and model estimates for 17 locations in the US, China, Saudi Arabia and Australia (n = 27 model estimates with varied climates). Data from Millar et al. 2003, Steffen et al. 2013, Zhang and Hu 2014, Guizani 2016.

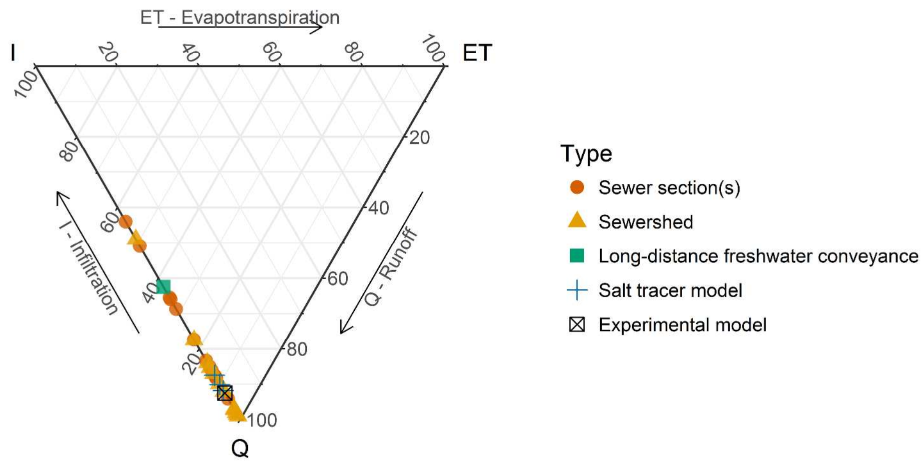


Figure S4 Thirty sewer exfiltration water budgets measured in pipe sections (n = 13), whole sewersheds (n = 12), a long-distance water supply pipeline in Saudi Arabia (n = 1), estimates from salt tracer models (n = 3) and an experimental laboratory model (n = 1). Evaporation from pipe sections and sewershed networks is assumed to be 0. Data compiled from Amick et al. 2000, City of Detroit Water and Sewerage Department and Michigan Department of Environmental Quality 2001, Ellis et al. 2003, Amick and Burgess 2003, Rieckermann et al. 2005, Rutsch et al. 2005, Rutsch 2006, Xu et al. 2014, Guizani 2016.

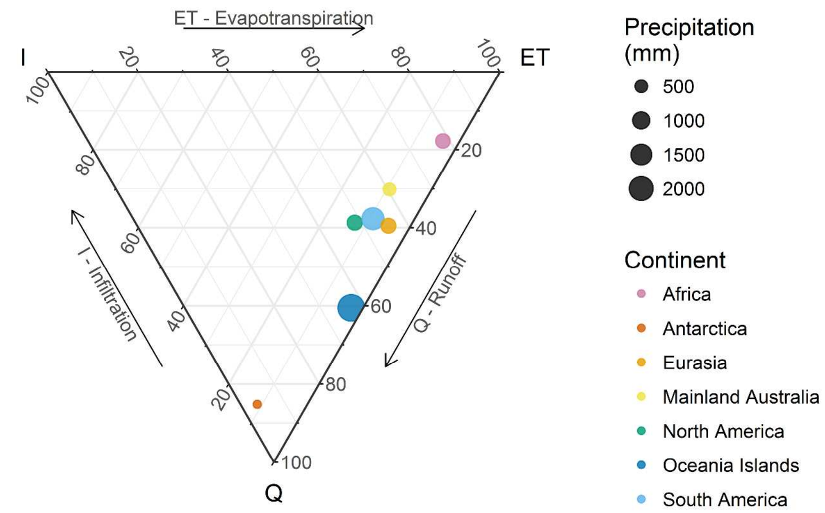


Figure S5 Continental water budget estimates from Rodell et al. (2015), where infiltration is estimated as the magnitude of vertical groundwater flux (n = 7).