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# Economic and Environmental Assessment of Office Building Rainwater Harvesting Systems in Various U.S. Cities

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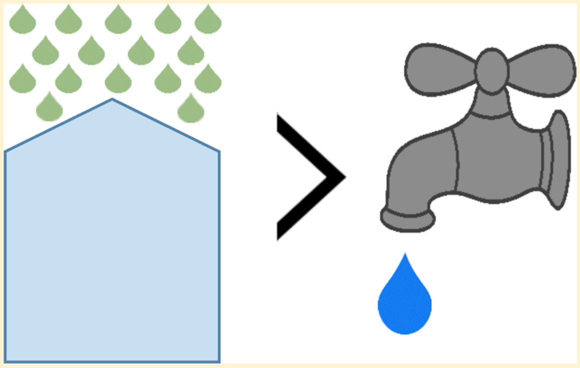
Ⓢ *Supporting Information*

**ABSTRACT:** Rainwater harvesting (RWH) systems implemented in office buildings under heterogeneous urban settings in the United States, including combined and separated storm sewer systems, will result in varying environmental and economic costs and benefits across multiple water sectors. The potable water saving and stormwater abatement potentials were found to strongly correlate with the local annual precipitation totals and patterns, specifically the long-period antecedent dry weather period. Given the current water rates and stormwater fees in large U.S. cities, RWH systems implemented in office buildings may not be cost-effective compared to the municipal supplies over their lifetime, except in Seattle, which has the highest stormwater fees in the country (\$77.50/1000 sf impervious surface/month). The minimum net life cycle costs range from −\$1.60 (Seattle) to \$11.9 (Phoenix) per m$^3$ of rainwater yield, resulting in a potential economic gain of over \$520 (Seattle) to a net loss of \$800 (Phoenix) per building annually. By preventing the rooftop runoff from entering the wastewater treatment plant, between 3 and 9 kg N eq per year could be reduced in combined sewer systems depending on local conditions. This N reduction comes at the expense 0.7−4.6 kg CO$_2$ eq per m$^3$ rainwater yield. In separate sewer systems, eutrophication reduction benefits result from reducing N loading associated with stormwater runoff. The overall sustainability of implementing RWH depends on the site-specific functional, economic, and environmental benefits, impacts, and trade-offs.



## 1. INTRODUCTION

Rainwater harvesting (RWH) is a low-impact, decentralized technique for capturing, storing, and delivering rainwater for potable or nonpotable purposes.[1,2] During the past decade, this centuries-old practice has received increasing attention for urban water management in regions experiencing increasing population and water demand, decreasing freshwater availability, and aging water infrastructures.[3−6] The implementation of RWH systems can synergistically provide water and stormwater functions while achieving a variety of socio-ecological benefits[6−8] (Figure 1) enabling total water management[9] and advancing efficient, resilient, and service-oriented water infrastructure.[10−12]

The most prevalent urban application of RWH has been collecting rainwater from rooftops for nonpotable purposes.[4,13] A wide range of water saving potential by RWH systems have been reported, ranging from 2% to 100%, e.g. refs 3,7,14, and 15, depending on annual precipitation,[3,4,7,16] dedicated water usage,[3,17,18] building type and occupancy,[4,15,19] cistern size,[3,7,20] and contributing roof area.[4,16] A few studies investigated both water saving and runoff reduction potential of RWH systems identifying a trade-off; regions with higher precipitation have higher water-saving potential but require larger cisterns for stormwater management.[3,20]

There is ongoing uncertainty about RWH cost-effectiveness of substituting for the public water supply. A whole life cycle cost assessment of household RWH systems in the U.K.[21] and partial life cycle cost assessment of building-scale RWH systems in the U.S.,[22,23] among other modeling efforts,[24,25] concluded that RWH systems have a negative return on investment (ROI) at the end of their useful life. These modeled results are supported by an empirical study in the U.S.[17] Other studies, based on partial life cycle cost (e.g., omitting maintenance) or qualitative analyses suggest RWH systems have both water and cost saving potential[6,15] due to the limited treatment required for the nonpotable water substitution.[6] The required maintenance costs are significant contributors to the unfavorable ROI of RWH systems.[21] Capital costs of a RWH system are dominated by the costs of the cistern with diminishing ROI of added cistern capacity,[20] highlighting the importance of appropriate sizing.[15] The relatively high life-cycle costs can be offset by incorporating the benefits of stormwater management, which were often neglected or understated in previous

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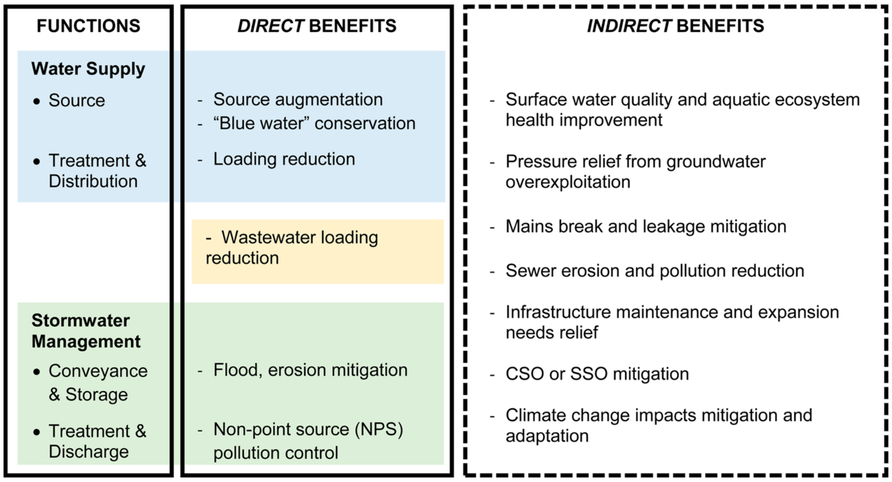


**Figure 1.** Synergistic water, wastewater, and stormwater management functions and (in)direct benefits provided by rainwater harvesting systems.

**Table 1. Relevant Physical and Socioeconomic Characteristics of the Selected Cities**

| EPA Region | City, State (Abbrev.) | Annual Precipitation | | Utility Rates | | | Sewer Structure | | Electricity | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | Total* (cm/ yr) | Per Capita** ($m^3$/person/y) | Potable Water*** (\$/$m^3$) | Sewage*** (\$/$m^3$) | Stormwater**** (\$/$10^3$ sf/month) | Mixed | Separated | Price***** (cents/kWh) | Grid Carbon Intensity****** ($CO_2$ eq./MWh) |
| 1 | Boston, MA (BOS) | 108 ◆ | 20.95 | **2.06** | **2.58** | - | √ | | 15.44 | 330.0 |
| 2 | New York, NY (NYC) | 111 ◆ | 10.33 | 1.26 | 2.01 | - | √ | | 16.86 | 282.9 |
| 3 | Philadelphia, PA (PHIL) | 107 ◆ | 24.12 | **0.74** | **0.74** | 8.99 | √ | | 10.09 | 456.8 |
| 4 | Atlanta, GA (ATL) | 127 ◆ | 96.43 | **2.18** | **5.53** | - | √ | | 10.36 | 617.4 |
| | Tampa, FL (TPA) | 117 ◆ | 95.91 | **2.22** | 2.07 | - | | √ | 10.12 | 545.1 |
| 5 | Chicago, IL (CHI) | 91 ◆ | 19.76 | 0.88 | 0.84 | - | √ | | 8.92 | 685.6 |
| 6 | Dallas, TX (DAL) | 87 ◆ | 61.55 | **0.67** | **0.85** | 1.59 | | √ | 8.33 | 554.7 |
| | Albuquerque, NM (ABQ) | 23 ◆ | 19.07 | 0.48 | 0.61 | - | | √ | 10 | 536.6 |
| 7 | Wichita, KS (ICT) | 80 ◆ | 72.43 | **2.41** | 0.79 | 0.94 | | √ | 9.95 | 820.4 |
| 8 | Salt Lake City, UT (SLC) | 40 ◆ | 59.79 | **0.79** | 0.71 | 1.80 | | √ | 8.54 | 384.2 |
| 9 | San Francisco, CA (SF) | 51 ◆ | 7.31 | 1.91 | 2.33 | - | √ | | 13.56 | 278.2 |
| | San Diego, CA (SD) | 25 ◆ | 15.48 | 1.47 | 1.33 | *0.02* | | √ | 13.56 | 278.2 |
| | Phoenix, AZ (PHX) | 19 ◆ | 15.42 | **1.33** | 0.66 | - | | √ | 9.35 | 536.6 |
| 10 | Seattle, WA (SEA) | 96 ◆ | 31.91 | **2.24** | 4.14 | 77.48 | √ | | 8.11 | 384.2 |

[a]* -- Average annual precipitation calculated from daily precipitation data from Jan 4th, 1960 to Jan 3, 2014 (NOAA, 2014);[33] the cities are clustered into three groups according to the annual precipitation (i.e., >105 cm (blue), 75–100 cm (green), and <55 cm (gold)) as illustrated by the color coded diamonds. [b]** -- Based on the 2013 population estimate and city land area information from the U.S. Census.[34] [c]*** -- 2014 potable water and sewer rates of the public supply systems (sources of the information were included in Table S1 in the SI); the highest inside-city marginal prices or peak usage prices are used (in bold); average prices are used instead if marginal prices are unavailable. [d]**** -- Stormwater fee in San Diego (in italic) is in \$/$m^3$ of water use; San Diego charges users other than single-family residences according to their monthly water usage. [e]***** -- Electricity prices in March, 2014 (US EIA, 2014).[35] [f]****** -- From eGRID 2010.[32]

studies.[17] RWH systems implemented in larger-scale commercial buildings are suggested as more financially viable than those in smaller residences[26]

Comparing the energy consumption and carbon footprint of the public water supply and RWH systems has also received recent attention. Pumping was found to dominate the energy consumption, and thus the carbon footprints, of RWH system operation.[4,8,27] Pumping energy estimates for RWH systems, both empirical and modeled, range widely from 0 to 4.9 kWh/m,[3,5,14,28] with empirical estimates tending to be higher due to inappropriate model assumptions (i.e., no standby and start-up energy consumption, no frictional losses).[8] According to empirical data, the average energy and carbon intensity of RWH systems is likely to be higher than the public water supply in the U.S. and many other countries[4,5,8] and depends on the emission factors of the regional energy grid.[4] Pumping needs, and subsequently energy and carbon costs, can be considerably minimized by placing cisterns at a higher elevation relative to the end uses,[14] using a gravity-fed system,[28] using an automatic mains switch,[5] and improving pumping efficiency.[5,28] The life-cycle energy and carbon impacts of RWH systems could be lower than those of public water supply systems when 1) the avoided wastewater treatment and distribution needs are considered[14,27,29] and 2) the service population density is reasonably high.[4]

The "credits" of water saving, the better matching of water quality to the intended demand, and the descriptions of "low-impact" and "green" for RWH tend to invite the assumption that RWH is a sustainable initiative;[8,15] however, the potential environmental and socioeconomic trade-offs associated with the RWH systems are still largely unknown.[30] Accordingly, this research has two objectives: 1) to systematically assess and reveal the multiple direct and indirect functional, economic, and environmental benefits and costs of implementing RWH systems and 2) to investigate RWH systems under heterogeneous urban settings (i.e., precipitation level and pattern, combined/separate sewer systems, water and electricity rates) in the U.S. for informed decision.

## 2. RESEARCH SCOPE AND METHOD

**2.1. Site Description.** Fourteen representative cities were selected from across the U.S. to consider the role of physical-socioeconomic heterogeneity throughout the country (Table 1, Figure S1 in the SI). The analysis considered the local differences in precipitation characteristics; utility rates for potable water, sewage, and stormwater services; sewer system configuration (i.e., combined and separated sewers); and the carbon intensity of the regional electricity grid. Average annual precipitation in the 14 cities ranges from <20 cm (Phoenix) to >120 cm (Atlanta) per year or 7 $m^3$ (San Francisco) to over 70 $m^3$ (Wichita) rainfall per capita annually. Current water and sewer rates also varied significantly, partly due to the different rate structures adopted—about half of the 14 cities do not have marginal rates. Note the weak correlations between per capita precipitation and water rates indicating that scarcity is largely not reflected in water pricing ($R^2 = 0.137$, see Figure S2 in the SI). Stormwater fees are assessed in limited U.S. cities despite that <20% of stormwater utilities nationwide report adequate funding.[31] Similarly, carbon intensity of the regional energy grids varies from 278 kg $CO_2$ eq/MWh (California) to 820 kg $CO_2$ eq/MWh (Kansas).[32]

**2.2. RWH System Description.** This study modeled a typical three-story U.S. office building with a rooftop catchment area ($A$) of 458 $m^2$ and occupied by 100 employees;[36,37] 60 of the employees are male and 40 of them are female.[38,23] Water use benchmarks for toilet and urinal flushing (1.6 gpf and 1.0 gpf, respectively) were assumed based on U.S. standards[39] resulting in nonpotable water demand of 2 $m^3$/day for 260 work days annually (520 $m^3$/year).

The comparative conventional building relies solely on the public water supply and has no active stormwater management facilities. The RWH system-enabled building relies on a direct feed system, using a submersible pump to lift the rainwater from a fiberglass cistern underground to end uses at different floors of the building (Figure S3 in the SI). An automatic supply switch placed after the pump ensures a reliable backup from the public water supply.[5] The emerging, gravity-fed RWH systems only support small storage[4] and thus would not be appropriate for the larger scale systems investigated by this study. Detailed information and further assumptions of the building configuration and water usage calculations are included in the SI.

**2.3. Water Saving and Runoff Reduction Simulation.** Water balance analysis of the cistern is critical to simulating water saving and runoff volume reduction potentials. The water balance was modeled using the yield after spillage (YAS) operating rule, recognized for accuracy in representing actual RWH system performance.[7,15,26] Primary inputs were daily precipitation of the 14 cities from 1960 to 2014 (detailed information in the SI) and workday demand. Additional model inputs included the roof area (458 $m^2$), roof runoff coefficient ($C = 0.9$),[40,26,41] and system filtration efficiency ($E_f = 90\%$)[5,40,42] with varying cistern sizes varying 2 to 30 $m^3$ (1 to 15 days of nonpotable water demand). Runoff losses from filtration and first-flush diversion, which are considered sufficient as treatment for nonpotable uses,[15,40,42] were accounted by $E_f$ in the model.

The water saving efficiency represents the average percent of water demand substituted by rainwater yield over time period $T$ determined by

$$W_T = \sum_t^T Y_t / \sum_t^T D_t \tag{1}$$

where $Y_t$ and $D_t$ represent the rainwater yield and water demand during time step $t$ = 1 day, respectively.

The runoff volume reduction potentials illustrates the average percent of rooftop runoff captured relative to that generated calculated by

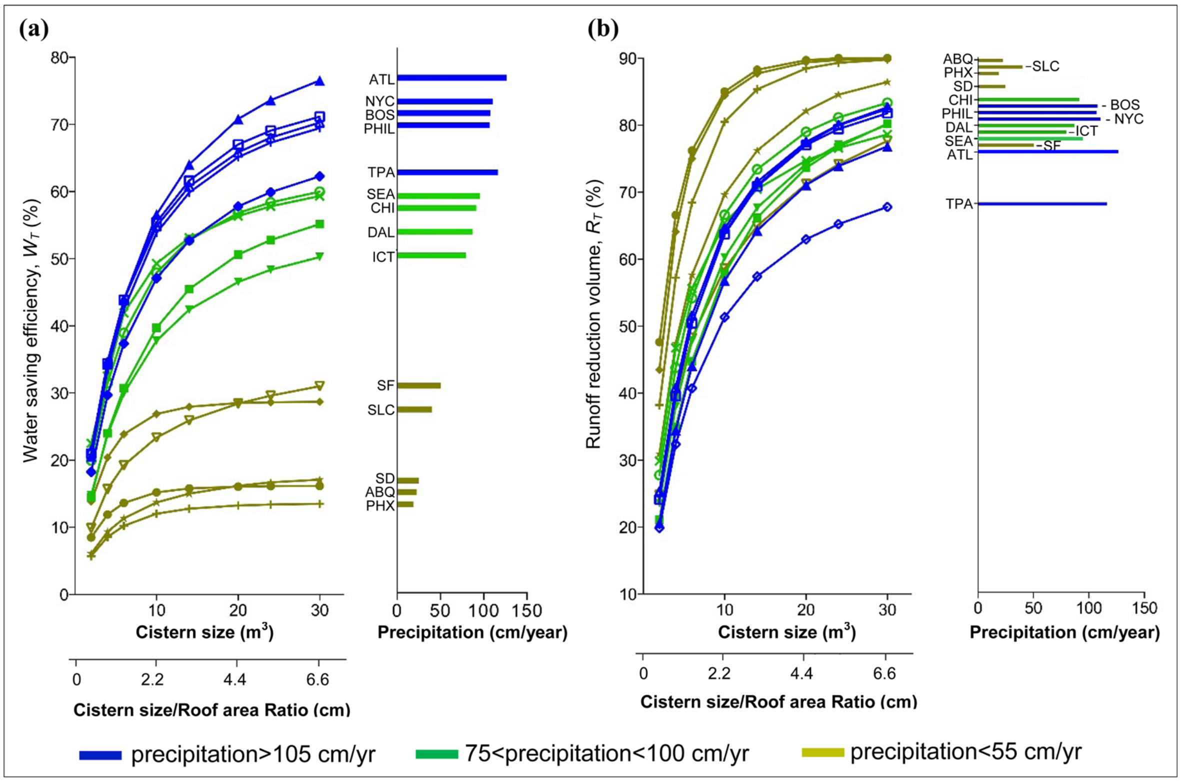
$$R_T = \sum_t^T Y_t / \sum_t^T RR_t \tag{2}$$

where $RR_t$ is the rooftop runoff generated during time step $t$.

**2.4. Life Cycle Inventory (LCI) of RWH Systems.** Existing life cycle inventories of RWH systems were either based on the more aggregated EIO-LCA framework[43,22] or focused on major components and processes,[14,29] ignoring the infrastructure maintenance or installation. To address some of these limitations, this analysis took the hybrid approach and inventoried the RWH system over six life-cycle stages: *Material extraction and manufacturing*, *Material transportation*, *RWH system installation*, *RWH system maintenance*, *RWH system operation*, and "*Avoided products*". Detailed LCI of the RWH system is reported in the SI. Briefly, the LCI data were retrieved from the databases of EcoInvent v3, US Input Output, and US LCI accessed through SimaPro 8.0,[44] supplemented with other literature sources as detailed in the SI, and assembled in SimaPro 8.0. The component and material requirements, operation, and transportation were modeled using available RWH system design guidelines[40,42,45,46] and inventoried using a process-based approach. The installation and maintenance stages were inventoried through economic input–output (I–O) LCA using the cost information on rain cistern installation and maintenance obtained from the WERF BMP and LID Whole Life Cost Tools[47] and estimates from previous studies.[23,48] Note that the I–O data were adjusted using the eGRID information[32] to reflect the grid-specific carbon intensity of the appropriate electricity grid for each installation site. The operation stage consisted primarily of the electricity consumed for pumping. This was simulated accounting for stand-by and start-up energy consumption and pump efficiency (50%)[28] as well as considering pump parameters (SI) and harvested water volumes.[28] All of the construction materials were assumed to be locally available within a 16 km (10 miles).

Cities running combined versus separate sewer systems were modeled differently (Table 1). For cities with a combined sewer system, the RWH system was modeled with credits for both substituted public water supply and avoided treatment of the runoff to WWTPs. For a separated storm sewer system, the credits were modeled as the substituted public water supply and the avoided discharge of NPS pollutants to receiving waters by reducing the rooftop runoff with assumed mean concentrations of the common NPS pollutants from a pitched asphalt/composite shingle roof,[49,50] the most common roofing material for RWH systems in the U.S.[13] All credits were inventoried using the process-based approach (detailed in the SI).

For San Diego and Phoenix, where public supply relies substantially on long-haul water transfers, the energy intensities of the public water supply were reported as 2.93 kWh/$m^3$ and



**(c)**

| Cistern Sizes (m$^3$) | $W_T$ $R^2$ Annual rainfall | $W_T$ $R^2$ # ADWP≥21 days | $R_T$ $R^2$ Annual rainfall | $R_T$ $R^2$ Rainfall variation* | $R_T$ $R^2$ Rainfall intensity* |
|---|---|---|---|---|---|
| 2 | 0.85 | 0.85 | 0.62 | 0.87 | 0.95 |
| 6 | 0.92 | 0.86 | 0.60 | 0.87 | 0.93 |
| 10 | 0.95 | 0.85 | 0.57 | 0.83 | 0.86 |
| 20 | 0.97 | 0.83 | 0.55 | 0.73 | 0.70 |
| 30 | 0.98 | 0.82 | 0.52 | 0.63 | 0.55 |

*Rainfall variation was computed as $\sqrt{\sum_t^T (P_t' - \bar{P_t'})^2/19723}$ , Rainfall intensity was computed as $\sum_t^T P_t''/\sum_t^T P_t$, where $P_t$ is rainfall depth (in tenth mm per day) during time step $t$ =1 day, from Jan 4th, 1960 to Jan3, 2014 (i.e. $T$=54 years or 19724 days), $P_t' = P_t$ when $P_t > 0$, $P_t'' = P_t$ when $P_t > 1.1$ *cm/day*

**Figure 2.** (a) Average water saving efficiency ($W_T$), (b) average runoff volume reduction ($R_T$), and (c) regression analysis results with respect to different cistern sizes: precipitation variation parameters critical for $W_T$ and $R_T$, respectively; cities are color coded into three groups according to the annual precipitation.

0.86 kWh/m$^3$, respectively.[51,52] For the remaining cities, the national average estimate (0.396 kWh/m$^3$) was adopted as the energy intensity of public water supply.[51] The energy intensity estimates do not account for the energy consumed during the water's use phase.

**2.5. Environmental Life Cycle Assessment.** A consequential LCA was performed to investigate the potential environmental implications associated with the addition of a RWH system to the model office building. According to previous research, adding new water infrastructure can lead to a trade-off of achieving the water management goals (e.g., eutrophication reduction) at the expense of energy use and GHG emissions.[53] This research, therefore, selected four out of the 18 indicators available from ReCiPe 2008 midpoint (hierarchist) method[54] as exemplars: climate change (kg $CO_2$ eq), fossil fuel depletion (kg oil eq), freshwater and marine eutrophication (kg P eq and kg N eq, respectively). It is important to note that although ReCiPe assigns phosphorous and nitrogen indicators of freshwater and marine eutrophication impacts, respectively, reductions of both nutrients is essential for water quality improvement in many freshwater and marine ecosystems.[55] The functional unit of this study is 1 m$^3$ rainwater harvested.

Equation 3 provides the generic formula for calculating the annual impact for a given environmental midpoint $i$, where the annual impact $E_i$ is a sum of the impacts for the environmental midpoint at each life cycle stage

$$E_i = \left( \frac{E_{material,i}}{T_L} + \frac{E_{transportation,i}}{T_L} + \frac{E_{installation,i}}{T_L} + E_{maintenance,i} + E_{operation,i} - E_{avoidedproducts} \right) \quad (3)$$

and the environmental impacts caused by the first three life cycle stages are amortized over the expected lifespan ($T_L$ = *50 years*) of the RWH system. $E_{maintenance,i}$ is based on the annual average maintenance accounting for routine and intermittent activities as well as capital replacements. Impacts of RWH operation and the credits depend on the carbon intensity of the electric grid, energy intensity of the public water supply systems, the average $W_T$ and $R_T$, thus varying from city to city.

**2.6. Life Cycle Cost Assessment.** Using a 5.5% discount rate,[47] a life cycle cost (LCC) assessment was performed to investigate the economic viability of implementing RWH systems in the 14 cities. Neglecting "*material transportation*", LCC is computed as

$$LCC = \sum_{T=1}^{50} (CAPEX + O\&M - WP - SF) \tag{4}$$

where *CAPEX* is the present value (PV) of capital expenditure, including capital investment costs and installation costs, *O&M* represents PV operation costs (i.e., electricity costs), routine and intermittent maintenance costs, and asset replacement costs, *WP* represents PV cost savings from avoided public supply water purchases, and *SF* represents PV cost savings from avoided stormwater fees. While stormwater fees are often charged based on percentage of impervious surface, runoff collection and reduction from the roof by RWH is equivalent to reducing the impervious surface of the roof. Therefore, $R_T$ is treated as the rate of impervious area reduction, leading to stormwater fee savings. The avoided wastewater treatment in combined sewer systems does not generate a direct cost saving to property owners and therefore was not considered.

*CAPEX* and maintenance costs were retrieved from the WERF BMP and LID Whole Life Cost Tools[47] and other literature sources as detailed in the SI. The operation costs were computed using the most recent state-level electricity prices.[35] *WP* and *SF* were based on the current potable water rates and stormwater fees charged by the cities (Table 1).

**2.7. Sensitivity Analysis.** The model was based on typical office building configuration and RWH design and existing utility rates of the 14 cities in the U.S. Model sensitivity to potential changes of the system setting was tested by altering the initial values of each constant in the model by ±10%. The effects on the functional, economic, and environmental performances of RWH were analyzed, and insights for future RWH design were discussed.

## 3. RESULTS AND DISCUSSION

**3.1. Functional Performance.** Throughout the 14 cities studied, $W_T$ (eq 1) increased with cistern capacity and annual precipitation levels (Figure 2a). When a 30 $m^3$ cistern was adopted, $W_T$ varied from 60 to 80%, 50%, and 10−30% depending on high (>105 cm), medium (75−100 cm), or low (<55 cm) annual precipitation. Tampa, which has the second-highest annual precipitation level, was the only exception. Compared to other wet cities (i.e., Atlanta, Boston, and New York City), Tampa experienced more incidences of long antecedent dry weather period (ADWP) since 1960, resulting in more frequent low-level cistern storage and subsequent shortage of the rainwater supply (Table 1). The relatively lower $W_T$ observed for Tampa and San Francisco when smaller cisterns were adopted can also be explained by the more incidences of long ADWP than those with less annual rainfall (i.e., the high-tier and low-tier cities, respectively). In addition to annual precipitation level,[3,7] previous studies have suggested *average* ADWP[56] and rainfall frequency[57] as the main hydrologic characteristics affecting $W_T$ of RWH systems. Here, we performed regression analysis to further investigate the impact of *incidences of ADWP of various time lengths.* The results demonstrate that the *annual precipitation level* and *incidences of ADWP longer than 21 days* are the most important variables for the resulting $W_T$ of RWH systems under various precipitation schemes across the United States, with $R^2$ ranging from 0.85 to 0.98 and 0.82−0.86, respectively (Figure 2c).

Sensitivity analysis (±10%) of cistern size = 30 $m^3$ demonstrated that the water saving efficiencies were most sensitive to the variations of daily water demand in all of the 14 cities, especially in the arid ones (e.g., Salt Lake City and Albuquerque). The ±10% variations of supply side factors (i.e., roof area, runoff coefficient, and filtration efficiency) also had considerable effects (±5−±10% changes) on $W_T$, especially in more arid cities. It is also worth noting that tank size had a relatively small impact (0−2%) on $W_T$ beyond the 30 $m^3$ capacity, when precipitation volume becomes the limiting factor. Therefore, simultaneous water conservation and efficiency measures coupled with RWH implementation can enhance the effectiveness and reliability of the RWH system with more significant implications for rainfall-limited cities.

Unlike $W_T$, the annual precipitation level plays a moderate role in $R_T$ ($R^2$ = 0.52−0.62), which decreases as the cistern size increases; however, the effects of rainfall variability and rainfall intensity were more significant (Figure 2b, c). Contrary to $W_T$, RWH systems were more effective for runoff volume reduction in arid cities than in the wet ones. When a 30 $m^3$ cistern was modeled, 80−90% of rooftop runoff was captured in the arid cities, while ∼68−83% rooftop runoff was captured in the wet cities. Different from $W_T$, the most critical hydrologic parameters for $R_T$, as shown by regression analysis, are variation of daily rainfall depths $P_t$ when $P_t > 0$ ($R^2$ = 0.63−0.87) (Figure 2c) and the rain intensity determined by the percentage of rain events larger than 1.1 cm/day contributing to total precipitation. The uneven temporal distribution of precipitation and bigger contribution of more intense rainfall events for total precipitation results in more frequent spills. The effects were best illustrated in Tampa and San Francisco, which experienced more various, intense rainfalls in the 55 years simulated and achieved much lower $R_T$ than those receiving comparable annual precipitation that distributed more evenly.

An increase in filtration efficiency and daily demand or a decrease in runoff coefficient could help improve runoff management, as shown by the sensitivity analysis (SI). It is important to note that varying runoff coefficient, roof area, and daily demand generated opposite effects on $W_T$ and $R_T$. For example, increasing roof catchment area was recommended by previous research as a cost-effective way to collect more rainwater and enhance water savings.[25] However, this neglects the need for greater cistern capacity and the additional challenges on stormwater management. It is also important to clarify the two metrics commonly used to assess RWH systems' water saving benefits: $W_T$ measures the system's water supply ability relative to the desired water demand, while water saving or conservation potential measures the absolute volumes of potable water substituted by rainwater. This explains why reducing water demand can enhance $W_T$, while expanding rainwater consumption can drive total rainwater yield for greater potentials of water conservation and stormwater mitigation.

As illustrated by Figure 2, the precipitation regimes cause a trade-off between $W_T$ and $R_T$: regions with higher precipitation have higher water-saving potentials but require larger cisterns for stormwater management; the trade-off is further complicated by the different and less clear effects of precipitation

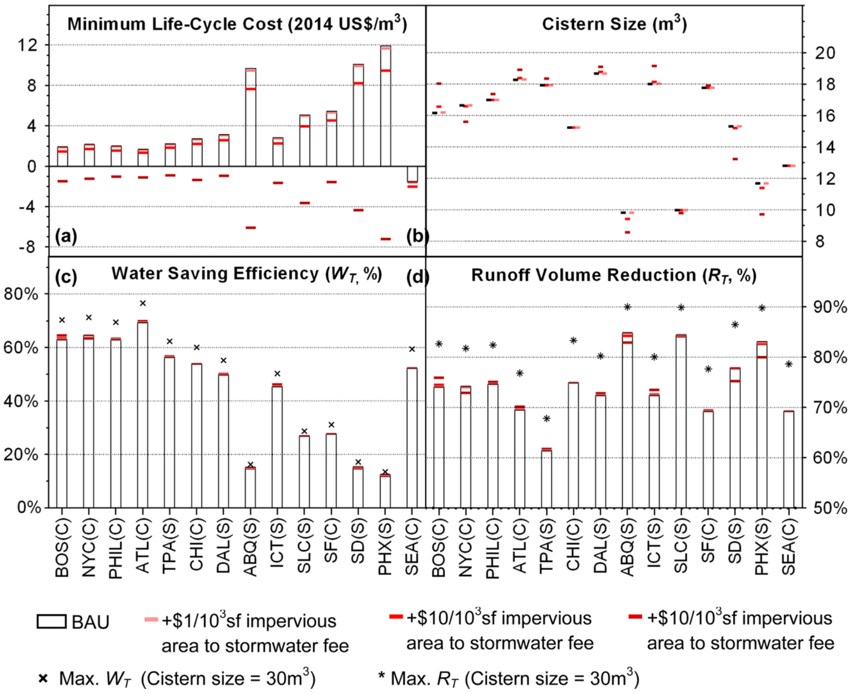


**Figure 3.** RWH economic costs and water yield performances (a) minimal life-cycle costs, (b) cistern size, (c) water saving efficiency, and (d) runoff volume reduction; "(c)" and "(s)" in the *x*-axis labels indicate combined or separated sewer systems, respectively.

variation on $W_T$ and $R_T$. Besides supplementing water supply and reducing runoff, RWH systems would reduce the influent loadings to wastewater treatment plants or mitigate the discharges of NPS pollutants, for combined and separated sewer systems, respectively. The volumes of wastewater or stormwater loadings reduced are equal to the volumes of municipal water supply substituted. A recent study has found that due to the changed rainfall patterns in the UK, larger rain tanks are needed by 2080 in order to meet the same nonpotable water demand as in the present.[58] Given the future of more intense droughts and increased precipitation variability in North America in the next decades,[59] this analysis suggests that the effects of climate variability on RWH across multiple water sectors may become more significant and require further investigation.

**3.2. Cost Effectiveness of Rainwater Harvesting Systems.** Although more benefits of water supply and stormwater abatement can be achieved by using a larger cistern (Figure 2), the incremental advantage of increasing cistern size has to be balanced with the associated incremental economic costs and diminishing returns. For per m³ of rainwater yield (i.e., the functional unit), the minimal LCC, including RWH costs and water system credits, for the 14 cities are shown in Figure 3(a). Compared to operating a public supply only system, RWH systems resulted in a potential nominal net economic loss at end of life ($T$ = *50 years*) in 13 cities except Seattle, which has the country's highest unit stormwater fee.[31] In Seattle, the RWH system could save up to $1.60 per m³ rainwater yield, achieving an economic gain of approximately $520 per year per building. In other cities, the minimum net costs range from $1.70 (Atlanta) to $11.90 per m³ of rainwater yield (Phoenix). The minimum unit life cycle costs of rainwater supply were achieved at cistern sizes ranging from 9.8 m³ (Albuquerque) to 18.7 m³ (Dallas), much lower than the maximum size (30 m³) simulated (Figure 3b).

According to a recent RWH survey in the U.S., cost-effectiveness was identified by household rainwater harvesters as the most common motivation for RWH installation.[13] This economic assessment suggests that the office-building RWH systems, which are generally considered more cost-effective than the household systems, may be unfavorable over the systems' lifetime, given the current water rates and stormwater fees in large cities. It is important to note that this economic assessment was based on the property owners' perspective. Therefore, in a combined sewer system where there is no stormwater fee (i.e., Boston, New York, Atlanta, Chicago, and San Francisco), the economic costs avoided by not treating as much runoff at a WWTP were not credited to the RWH system because the property owners do not pay the stormwater management fees. From the "society's perspective", if those cost savings were accounted, RWH practices would become more cost-effective than the public supply systems in both Atlanta and Boston but still not in the other cities.

The lack of financial incentives has been increasingly recognized as a critical barrier for promoting RWH systems specifically[60] and decentralized water infrastructures generally.[61,62] Higher utility rates and accounting for the stormwater benefits were suggested by previous studies to offset the high life cycle expenses[17,63] and provide significant incentives for adopting RWH practices.[18,62] Compared to other countries, the U.S. is known for its low potable water rates[64] with EPA proposing "full-cost" pricing nested in a block water rate systems since 2007.[65] Recall that nearly half of the 14 study cities are still charging water use based on an average unit rate and that water availability is weakly correlated with the water rate. As a result, current potable water rates in many U.S. cities likely do not reflect the real physical or financial conditions of the public supply systems, compromising the fairness of comparison between the RWH systems and the existing public water supply systems.

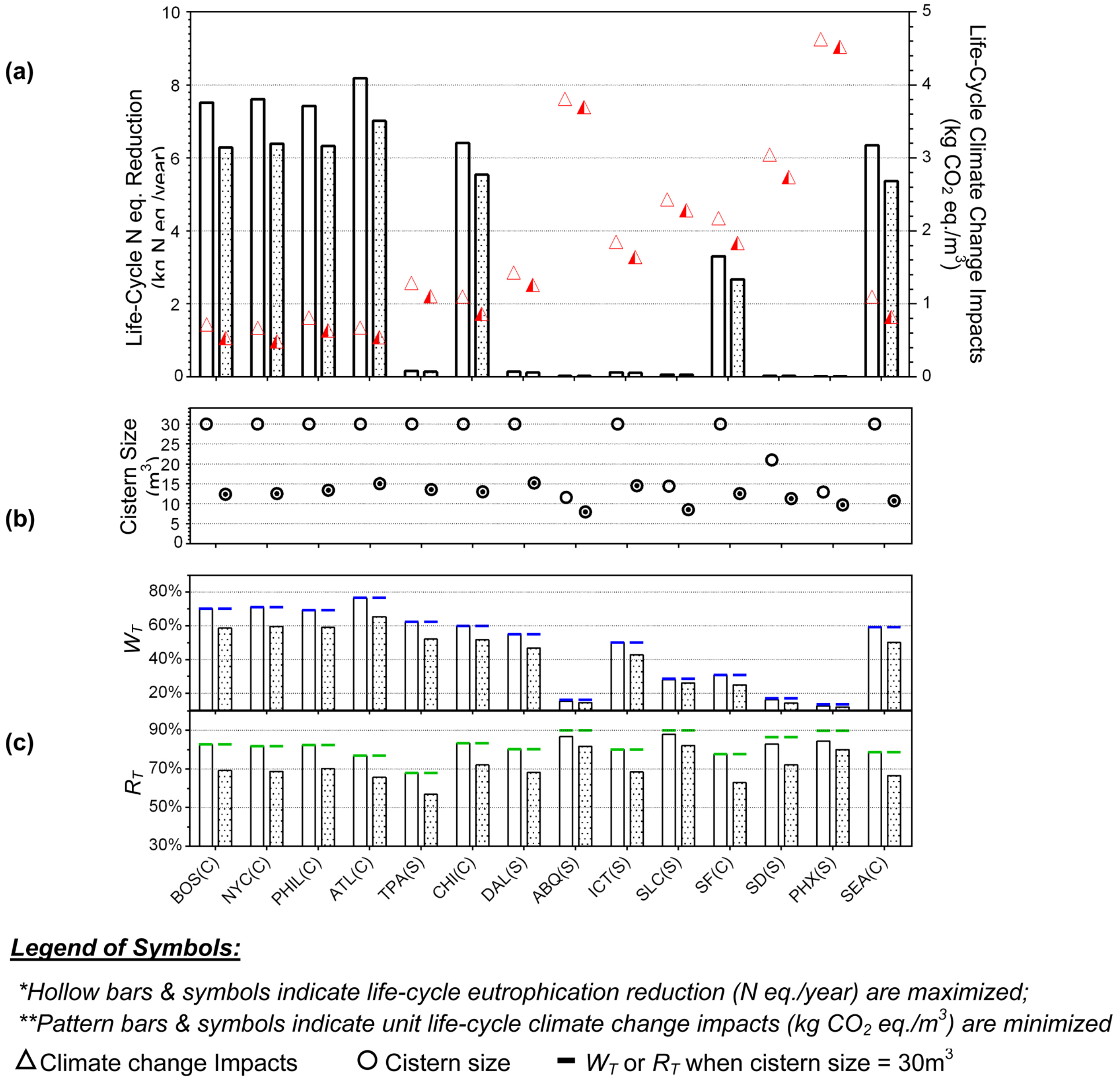


**Figure 4.** (a) Environmental benefits (bars) and drawbacks (triangles), (b) cistern sizes, and (c) $W_T$ and $R_T$; "(c)" and "(s)" in the $x$-axis labels indicate combined or separated sewer systems, respectively.

As such, this study explored the policy implications of increasing stormwater fees. While an additional \$1 stormwater fee per $10^3$ sf of impervious area had minimal effects, increasing to \$10/$10^3$ sf would reduce the RWH systems' LCCs more substantially (Figure 3a). If every city adopted Seattle's stormwater fee of \$77.50/$10^3$ sf of impervious area, then the cost of RWH systems would be economically favorable in all 14 cities. At this \$77.50/$10^3$ sf, the maximum net savings range from \$0.90/$m^3$ (Tampa) to \$7.20/$m^3$ (Phoenix) resulting in average annual savings of \$230 (San Francisco) − \$510 (Salt Lake City). Further, the arid cities achieved much more savings in every cubic meter of rainwater harvested reflecting RWH's more effective stormwater management in the arid areas (Figure 3b). To incentivize RWH, a number of U.S. states and cities are offering tax credits, rebates, or fee reductions through stormwater utilities. RWH may also be eligible for funding by a few federal and state grant programs.[61,65] With a ROI ≥ 1, the substantial upfront costs of RWH projects can be financed by models that were originally designed for energy efficiency financing to further enhance the financial feasibility.[62]

The water cost on a per unit basis is an important metric and helpful for decision makers to compare among various water supply alternatives. Results in this analysis suggest that the optimum unit cost-effectiveness (i.e., minimum unit cost or maximum unit saving) of RWH systems does not guarantee an optimum solution and thus should not be used as the only measure to guide the sizing of the system. As shown in the case when every city adopted Seattle's stormwater fee, offsetting stormwater fees became the dominant factor in LCC in arid cities and their optimum unit cost-effectiveness were achieved at increasingly smaller cistern sizes (Figure 3b), due to the faster diminishing functional gains of RWH systems with increasing cistern size (Figure 2). While the smaller cisterns yield the minimum unit life cycle costs for all 14 cities, with or without enhanced stormwater fee incentives, this leads to ∼5% less of potable water substitution in wet cities and 5−10% less

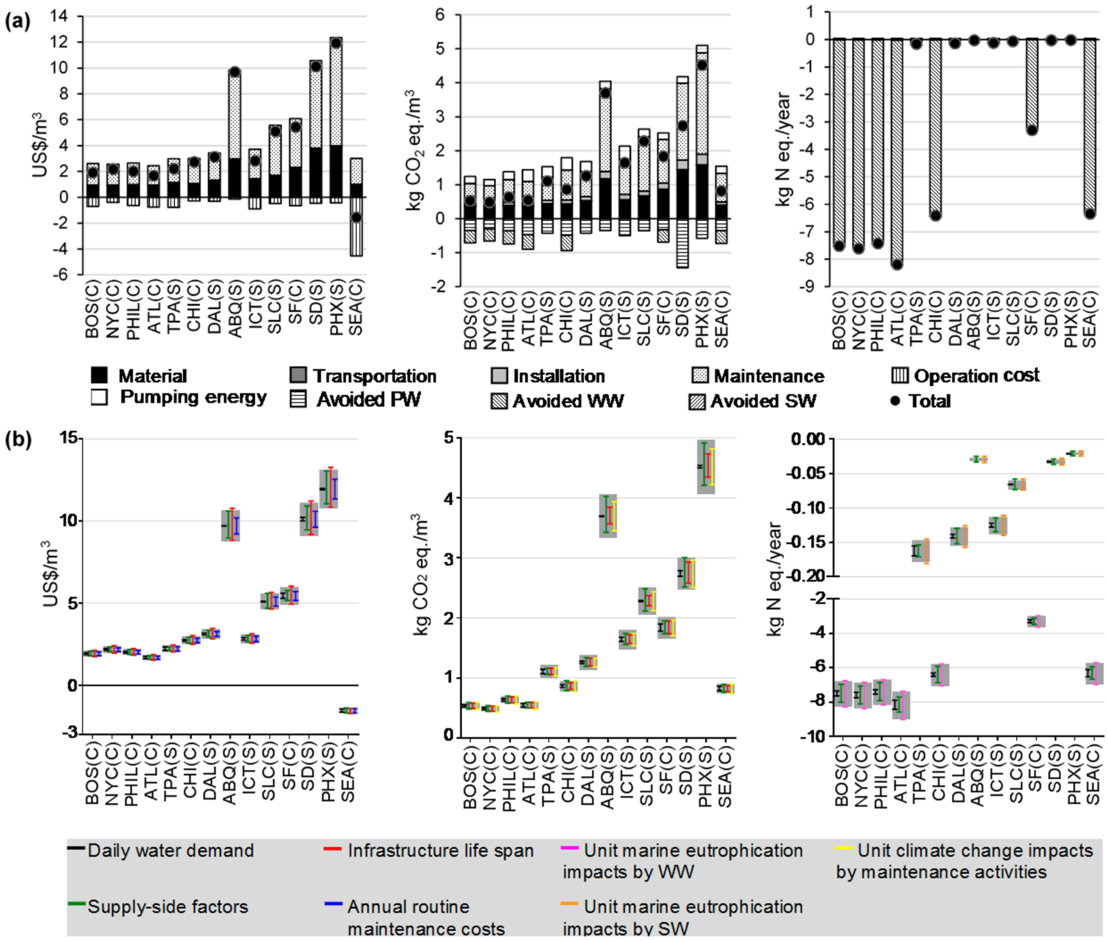


**Figure 5.** (a) Economic and environmental impacts by life-cycle stage and (b) selected SA results. Note: PW, WW, and SW represent the potable water supply, wastewater management, and stormwater management services; gray shadings indicate ±10% variations of the result.

rooftop runoff capture compared to the rates when the optimum cistern size for these functions was adopted (Figure 3c,d).

### 3.3. Environmental Viability of Rainwater Harvesting Systems.

LCA results demonstrate both environmental benefits (e.g., water quality improvement by reducing eutrophication) and drawbacks (e.g., climate change impacts) (Figure 4). Eutrophication benefits by the RWH systems arise directly from the avoided potable water treatment, NPS pollution, and wastewater treatment, especially the latter (detailed in the SI). The seven combined sewer systems could reduce up to 0.2–0.4 kg P eq and 3–9 kg N eq annually when cisterns were sized to the maximum capacity of 30 m$^3$. In separate sewer systems, eutrophication reductions are less significant. It is important to note that RWH system implementation comes with the costs of additional energy consumption and greenhouse gas emissions, caused by building, operating, and maintaining the systems as compared to the public water supply. The minimum life cycle climate change impacts were 0.5–4.5 kg $CO_2$ eq per m$^3$ rainwater yield (i.e., the functional unit) among the cities. Due to the relatively low rainwater yield potentials, rainwater harvested in arid cities were more carbon-intensive than those by the wet cities.

Most previous estimates on energy and carbon footprints of RWH systems only consider the direct energy consumption of pumping and those embedded in major capital components.[14,27,29] Results in this analysis demonstrate that the direct energy (i.e., pumping) was 0.44 kWh/m$^3$ and accounted for 5–52% of total energy and carbon footprints or 4–22% neglecting the credits. Maintenance activities and material manufacturing made the largest contributions to the indirect emissions. For example, from Atlanta to Phoenix, these two stages generated 2 to 20× more $CO_2$ eq emissions than the operation stage, respectively (SI). This analysis showed that maintenance, often omitted in previous studies, was a critical factor in the life cycle environmental assessment of RWH systems, but the current analysis is similarly limited by the scarce empirical information on maintenance requirements and the implications to system reliability and life span and aggregated EIO-LCA data available.

### 3.4. Economic and Environmental Viability: Influences and Leverages along the Life Cycle.

The sensitivity analysis demonstrated that results of LCA and LCC are overall robust to ±10% changes of the conditioning parameters of the model: higher utility fee offsets, $W_T$, and $R_T$, led to lower economic and carbon costs per cubic meter rainwater; reductions of marine eutrophication impacts were higher for RWH systems implemented in a combined sewer system (full SA results are detailed in the SI). Depending on the RWH systems' life-cycle impact profiles that varied geographically, the economic and environmental performances responded differently to the changing conditions (Figure 5).

RWH system life cycle economic and climate costs were driven largely by the upfront capital investment (i.e., material) and recurring maintenance activities, which includes annual routines and intermittent activities as well as capital replacements (Figure 5B), making infrastructure life span a critical variable. If the maintenance parameters are 10% more than those suggested by the best estimates available, the economic and climate change impacts of RWH systems could increase by 6−9% and 6−11%, respectively. Maintenance efforts are closely linked to system reliability and will affect the quantities of operational water savings, energy use, and related carbon emissions.[4,15] Assuming sufficient maintenance activities to advance longer system life span and lower maintenance costs, economic and carbon costs of RWH systems can be reduced, highlighting the importance of efficient and effective maintenance plans.

Unlike in other cities, the favorable economic performance of RWH in Seattle relies on stormwater fees, e.g. a 10% reduction in the current fee would reduce economic gains by 24%. In the remaining cities, the stormwater fees are either nonexistent or too low to yield an impact in economic performances. The SA results support policies for increased stormwater fees and on-site retention standards to effectively incentivize RWH.

The eutrophication reduction potentials were largely attributable to the avoided wastewater and stormwater processes in CSS and MS4, respectively, and therefore would reply on wastewater treatment standards and processes adopted by the utilities and the quality of rooftop runoff. Although RWH developers and owners have limited leverages over those variables, results here suggest that urban water managers should account for the water supply and wastewater and stormwater management services provided by the RWH systems while making decisions for, e.g. upgrading, expanding, and retiring the centralized facilities and the decisions' implications on the overall services achieved by both the centralized and decentralized facilities. In urban areas, the quality of local rooftop runoff quality is affected by from both wet and dry depositions of inorganic and organic contaminants derived from heavy traffic, industry, incinerators and smelters, animal feces, etc.[49,66] Among the 14 cities studied, Chicago, Phoenix, San Francisco, and Philadelphia are having the most critical air quality issues, particularly with high concentrations of particulate matters,[67] indicating some threat to the rooftop runoff quality. However, the detailed and complex modeling of the NPS pollutants and local air pollutants are beyond the scope of this analysis.

*Reflections on Assessing Multifunctional Infrastructures.* Current urban water management practices are often tailored to an individual goal for water supply, wastewater management, or stormwater management. This was reflected in existing studies where RWH systems were primarily evaluated to address scarcity concerns.[4] Previous assessments that attribute the expenses of implementing a multifunctional infrastructure to a specific goal of target, such as the water supply service of RWH, and neglect the "unintended" but equally valuable benefits realized simultaneously may have resulted in an inequitable evaluation of the infrastructure's cost-effectiveness. Further, the fragmented paradigm of managing water flows of various quality and service potential using single-purpose projects has been criticized for causing a repetitive, wasteful use of resources and significant impacts on the environment,[9,68] impeding a sustainable development of the water infrastructure systems. Employing a life-cycle perspective, this analysis demonstrated that RWH systems may promote environmental and economic benefits or drawbacks in multiple water sectors that vary with location and context.[69−71]

## ASSOCIATED CONTENT

### Supporting Information

Figures S1−S11, Tables S1−S18, and text. This material is available free of charge via the Internet at http://pubs.acs.org.

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### Notes

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