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Onsite Non-potable Reuse for Large Buildings: Environmental and Economic Suitability as a Function of Building Characteristics and Location

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ABSTRACT

Onsite non-potable reuse (NPR) is a way for buildings to conserve water using onsite sources for uses like toilet flushing, laundry and irrigation. Although early case study results are promising, aspects like system suitability, cost and environmental performance remain difficult to quantify and compare across broad geographic contexts and variable system configurations. In this study, we evaluate four NPR system types – rainwater harvesting (RWH), air-conditioning condensate harvesting (ACH), and source-separated graywater and mixed wastewater membrane bioreactors (GWMBR, WWMBR) – in terms of their ability to satisfy onsite non-potable demand, their environmental impacts and their economic cost. As part of the analysis, we developed the Non-potable Environmental and Economic Water Reuse Calculator (NEWR), a publicly available U.S. EPA web application that allows users to generate planning-level estimates of system cost and environmental performance using location and basic building characteristics as inputs. By running NEWR for a range of scenarios, we find that, across the U.S., rainfall and air-conditioner condensate are only able to satisfy a fraction of the non-potable demand typical of large buildings even under favorable climate conditions. Environmental impacts of RWH and ACH systems depend on local climate and were comparable to the ones of MBR systems where annual rainfall exceeds approximately 10 in/yr or annual condensate potential exceeds approximately 3 gal/cfm. MBR systems can meet all non-potable demands but their environmental impacts depend more on the composition of the local energy grid, owing to their greater reliance on electricity inputs. Incorporation of thermal recovery to offset building hot water heating requirements amplifies the influence of the local grid mix on environmental impacts, with mixed results depending on grid composition and whether thermal recovery offsets natural gas or electricity consumption. Additional environmental benefits are realized when NPR systems are implemented in water scarce regions with diverse topography and regions relying on groundwater sources, which increases the benefits of reducing reliance on centralized drinking water services. In terms of cost, WWMBRs were found to have the lowest cost under the largest range of building characteristics and locations, achieving cost parity with local drinking water rates when those rates were more than $7 per 1000 gallons, which occurred in 19% of surveyed cities.

Introduction

Urban water reuse is receiving more attention across the United States, as demonstrated by the recent, cross-agency National Water Reuse Action Plan (U.S. EPA, 2020). Traditional, single use approaches are increasingly scrutinized for their inefficient use of material and energy resources as well as their exacerbation of long term water scarcity (CNRA, 2020; Daigger, 2009; Lahnsteiner et al., 2018). The use and reuse of locally available water sources such as wastewater, graywater, rainwater and air conditioner (AC) condensate provide an opportunity to reduce this strain on surface and groundwater resources.

As the water reuse field progresses, research agendas are being implemented simultaneously with action at all levels of governance. Guidance for treatment that is protective of human health has evolved from presumptive criteria to risk-based approaches (Schoen et al., 2017; Sharvelle et al., 2017) while cities like San Francisco have pioneered ordinances requiring onsite non-potable reuse (NPR) systems in large construction projects (SFWPS, 2015).

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Projects to reuse local water resources are becoming more common across the country, including centralized wastewater reuse in Virginia, Florida and Texas (Crook, 2004) and decentralized reuse projects in Oregon, Colorado and Minnesota (U.S. EPA, 2018a). Globally, the United Nations and partner organizations are motivating and coordinating sustainability efforts through frameworks such as the Sustainable Development Goals (SDG), which aims to “substantially increase water use efficiency” (SDG 6.4) by 2030 (United Nations, 2018). Still, considerable uncertainty exists regarding how best to implement these practices. To become widespread, water reuse practices must be economically viable. To be beneficial, practices must not simply shift environmental impacts, favoring, for example, water conservation goals at the expense of greater energy use and carbon emissions.

In an urban setting, water can be reused in a number of ways. Centralized wastewater reuse represents one end of the spectrum, where sewer and treatment plant layouts closely resemble existing centralized facilities but allow for redistribution of treated wastewater. Recently, decentralized water reuse for non-potable end uses has gained attention owing to its potential to reduce burdens associated with complex collection and distribution networks. Early results have shown that economies of scale in treatment often outweigh diseconomies of scale in distribution and collection, but that net benefits can be realized in areas with increasing topographical relief and settlement dispersion (Eggimann et al., 2015; Kavvada et al., 2016; Kavvada et al., 2018; Newman et al., 2014). Onsite NPR occupies one end of the decentralization spectrum, where water sources that are generated onsite – rainwater, AC condensate, graywater or wastewater – are treated and redistributed within single dwelling or building footprints. Although onsite systems lack economies of scale, they are easier to implement in the near-term as they can be applied to single buildings rather than entire communities, requiring far less capital investment, financial risk and management. It is partly for this reason that early water reuse ordinances (e.g., SFWPS, 2015) focus on onsite NPR as an impetus for innovation in the broader water reuse field. Similarly, we focus here on onsite NPR not because it is necessarily superior to other forms of water reuse, but because it is an approach that is within technical and regulatory reach for many municipalities and geographically comprehensive evaluations that compare-and-contrast feasible onsite NPR options are still limited. Numerous studies have been conducted on the reliability, environmental impact and cost effectiveness of rainwater harvesting (RWH) systems in various regions (Allison et al., 2017; Amos et al., 2018; Campos Cardoso et al., 2020; Cook et al., 2014; Faragò et al., 2019; Ghimire et al., 2017; Lani et al., 2018). Of the studies identified, each consisted of a case study approach where results were contingent upon unique geographies and system configurations. Most concluded that RWH systems could be economically viable or environmentally beneficial if certain factors – annual rainfall, water utility rates, electricity grid emissions, demand, etc. – were optimal. For example, Campos Cardoso et al. (2020) found that RWH systems in a particular Brazilian city can be viable if “demand is low and climatic conditions are favorable”.

Similarly, a number of case studies have looked at hybrid systems, combining facets of RWH with graywater recycling (Faragò et al., 2019; Hasik et al., 2017; Jeong et al., 2018; Leong et al., 2019; Marinoski & Ghisi, 2019; Stephan & Stephan, 2017; Zanni et al., 2019). Though graywater was found to be a more reliable water source, system size, electricity consumption and electricity grid emissions were found to heavily influence environmental and cost performance of each system. As expected, larger systems serving multi-family buildings had lower relative cost and environmental impacts than single dwelling units (Faragò et al., 2019; Jeong et al., 2018; Zanni et al., 2019) though differences in climate and system design make it difficult to draw other generalizable conclusions.

Much of the research on NPR of graywater or wastewater refers to low energy systems – wetlands, lagoons, sand filters, etc. – that require large footprints, tend to be uneconomical relative to centralized systems, and provide variable treatment reliability (Arden & Ma, 2018; Hasik et al., 2017; Hendrickson et al., 2015; Zanni et al., 2019). Recently, advancements in membrane bioreactor (MBR) technology have shown that onsite recycle of wastewater flows can be accomplished with a small land requirement and can be environmentally preferable when accounting for offset potable water consumption (Cashman et al., 2018; Morelli et al., 2019) and can be sufficiently protective of human health (Schoen et al., 2018). The environmental and economic performance of aerobic MBRs relative to recirculating vertical flow wetlands and anaerobic MBRs (Morelli et al., 2019) guided our focus on aerobic MBRs in the present research.

Research on the cost and environmental performance of AC condensate harvesting (ACH) systems is perhaps the most limited. Early work focused on measuring or estimating the condensate generating potential of air handler units (AHU), with most studies evaluating potential in individual cities (Lawrence et al., 2010a; Painter, 2009). Exceptions include a global generation potential analysis (Loveless et al., 2013) and an assessment of the economic viability of using condensate for cooling tower water in U.S. cities (Lawrence et al., 2012). Comparing ACH to RWH systems, Ghimire et al. (2019) found the preferred option to differ in two cities with different climate conditions.

The relative nature of these study conclusions makes it difficult to translate results to areas with different climates, utility rates and building characteristics, let alone different system designs. Critically, systems designed to provide recycled water for indoor use must be protective of human health, which requires the use of robust and redundant disinfection processes (Sharvelle et al., 2017; U.S. EPA, 2012). Disinfection processes require energy, chemicals and infrastructure and, although some of the above case study systems included disinfection steps (e.g., Faragò et al., 2019; Ghimire et al., 2017; Hasik et al., 2017; Leong et al., 2019), only the systems of Ghimire et al. (2019), Morelli et al. (2019) and Schoen et al. (2018) were designed according to the most current human health protection guidance (Sharvelle et al., 2017).

The goal of this research is to assess the economic and environmental performance of several common onsite NPR options for all U.S. ZIP Codes as a function of building characteristics and location.. In conjunction with this research, a publicly available, online tool – U.S. EPA’s Non-potable Environmental and Economic Water Reuse Calculator (NEWR) – was built that allows users to estimate the cost and environmental performance of these NPR systems as a function of ZIP Code and building characteristics. Using NEWR, we first evaluate the availability of alternative water sources relative to the non-potable demands of a large building. We then evaluate the life cycle costs and potential environmental impacts of these onsite NPR systems across a range of building sizes and occupancy rates for the entire U.S.

Methods

Scope

To evaluate the environmental impacts and economic cost of onsite NPR options, models were created for NPR systems designed to collect, treat and distribute building-generated source waters at any ZIP Code in the U.S. (Fig. S1). General building characteristics and a range of geographic attributes were used to parameterize the models so that suitability could be evaluated as a function of building type, size, occupancy, end-use characteristics and location. An overview of the modeling approach is provided in this methods section and supported with additional detail in Section 1 of the SI.

System models were created for RWH, ACH, and treatment of source-separated graywater or mixed wastewater using an aerobic membrane bioreactor (GWMBR and WWMBR, respectively). All treatment systems were designed to satisfy general guidelines for unrestricted, indoor NPR including maintenance of a free chlorine residual of 1 mg/L and effluent BOD5 and suspended solids concentrations of less than 10 and 5 mg/L, respectively (U.S. EPA, 2012). All systems were also designed to meet recommended log reduction targets (LRTs) for microbial pathogens (Sharvelle et al., 2017), as documented in Morelli et al. (2019). Model development, including background data sources, is further described below and throughout Section 1 of the SI.

Life cycle assessment (LCA) was used to estimate potential environmental impacts following guidelines specified by the International Organization for Standardization (ISO) (ISO, 2006b, 2006a). The developed LCA estimates integrated environmental impacts during operation of a water reclamation system and throughout upstream supply-chains as resources are extracted, processed, distributed and consumed. Benefits (or avoided impacts) from shifting away from centralized water supply were also considered. Cost estimates were developed using life cycle cost analysis (LCCA), a similarly comprehensive approach to approximating economic costs over the lifetime of a system (Fuller & Petersen, 1996). LCA and LCCA results were calculated and presented per 1 gallon of NPR water provided to the building, termed the functional unit. LCA and LCCA results were generated for Global Warming Potential (GWP), Cumulative Energy Demand (CED), Fossil Fuel Depletion Potential (FDP), Water Consumption (WC), Water Scarcity (WS) and Net Present Value (NPV). More detail on the methods used for each can be found in Section S1.11.

Source Water Availability

For each system, a water balance was defined as a function of source water availability and non-potable demand. Source water availability for RWH and ACH systems depends on local climate and certain building characteristics, while availability for GWMBR and WWMBR systems depends on building occupancy and water fixture efficiency. Non-potable water demand includes toilet flushing, laundry, outdoor irrigation and other miscellaneous uses. Per capita design flows for water use categories, mixed wastewater generation and graywater generation were adapted from DeOreo et al. (2016), Mayer et al. (2011) and Morelli et al. (2019) and are listed in Tables S1 and S2.

Rainwater availability depends on the building area available for collection, rainfall rate and storage tank size. To determine rainwater availability, roof area was assumed to be equivalent to the building's footprint. A 75% collection efficiency (Ghimire et al., 2019) was applied to monthly rainfall totals for a given ZIP code. Monthly precipitation data were primarily obtained from the North America Climate Dataset (NACD) (Museum of Vertebrate Zoology, 2011) and supplemented with Modern-Era Retrospective analysis for Research and Applications, Version 2 (MERRA-2) data (Gelaro et al., 2017) where necessary. Precipitation that falls as snow and ice was excluded by assuming zero precipitation for months in which a hard freeze occurs, which is defined as 4 or more consecutive hours with a temperature of 28°F (2.2°C) or less. Hard freeze months were determined using spatial interpolation of Typical Meteorological Year 3 (TMY3) data (Wilcox & Marion, 2008). The rainwater collection tank was sized by taking the smaller of either maximum monthly demand or average monthly rainwater collection following the method developed by (Ghimire et al., 2019). Additional detail, including an illustration of annual rainfall totals across the U.S. (Fig. S2), is provided in Section S1.3.

AC condensate production is based on the difference between the relative humidity of outdoor air and indoor air and the amount of outdoor air introduced into the building. Condensate generation potential was calculated following the methods of Lawrence et al. (2012, 2010b). TMY3 data were used to define hourly outdoor air relative humidity for a full year at 1020 individual stations (Fig. S3), while indoor air (leaving the AHU) relative humidity was assumed to be 55°F (13°C) at 90% humidity (Glawe, 2013; Lawrence et al., 2010b). The amount of outdoor air introduced to the building was calculated as a function of building occupancy and total floor area assuming the HVAC system was designed according ANSI/ASHRAE Standard 62.1 (ANSI/ASHRAE, 2010), which requires a minimum ventilation rate of 5 cfm/person and 0.06 cfm/ft^2. Additional detail, including an illustration of annual condensate potential across the U.S. (Fig. S4), is provided in Section S1.4.

Outdoor irrigation demand is calculated using an approach based on California's Water Budget Workbook (CDWR, 2010), where demand is a function of monthly reference evapotranspiration, irrigated area and a plant water use factor (a measure of relative transpiration). Average monthly reference evapotranspiration for the U.S. was obtained from the MERRA-2 dataset with an averaging period of 1989-2018, while plant water use factors were categorized as high (0.75), medium (0.5) or low (0.25) based on general ranges provided in the Water Use Classification of Landscape Database (Costello & Jones, 2014). Additional detail is provided in Section S1.5.

Source water quality influences the selection of treatment process and the level of treatment required to meet NPR guidelines. Influent characteristics of mixed wastewater, graywater and effluent quality criteria were used to design the treatment systems as described in Morelli et al., (2019) and in Section S1.6. Water quality was not explicitly defined for rainwater and AC condensate systems, and it was assumed to be suitable for direct disinfection and reuse.

Life Cycle Inventory Development

Material and energy inventories for rainwater harvesting and AC condensate production were adapted from previous LCAs for NPR in multi-story buildings (Ghimire et al., 2017, 2019). The systems were designed according to American Rainwater Catchment Systems Association specifications for supply of toilet and urinal flush water in a four-story building serving 1000 people. Both systems include components for the collection, storage, disinfection and distribution of treated water (Fig. S5). The size of individual system components was scaled based on system size and the scaling methods described in Section S1.7. UV and chlorine disinfection processes were revised to meet roof runoff LRT guidelines – log reduction of 3.5 – for enteric bacteria (Sharvelle et al., 2017) and chlorine residuals.

The aerobic MBR life cycle inventory (LCI) was adapted from an LCA of NPR of mixed wastewater and graywater for large buildings and districts in San Francisco (Morelli et al., 2019). Original LCI values have been scaled to maintain the original design specifications across system size ranges utilized in this study as described in Section S1.8. The systems include screening, an aerated equalization basin, membrane tanks and disinfection processes in the form of ultra-violet (UV) radiation and chlorination, as well as a separate distribution system for treated water (Fig. S5). Treatment systems were designed to meet LRTs for NPR of mixed wastewater and graywater using log reduction values (LRVs) for individual treatment processes and disinfection doses. Details of LRV assignments and their relation to LRTs can be found in Morelli et al. (2019).

For scenarios examining the effect of thermal energy recovery using a water-to-water heat pump, recovered thermal energy is assumed to offset the consumption of electricity or natural gas to provide hot water to the building. The available thermal energy in mixed wastewater and graywater varies due to differences in flowrate and influent temperature. Section S1.10 describes the calculations used to estimate the quantity of avoided natural gas or electricity. Inventory items for the thermal recovery unit are assumed constant per unit flow and are described in greater detail in Morelli et al. (2019).

Avoided burdens associated with reduced potable water demand were estimated using an adapted LCI model of drinking water treatment and distribution for the Greater Cincinnati Water Works Richard Miller Treatment Plant (Xue et al., 2019). Although variation of material components of the LCI is outside the scope of this project, electricity requirements are varied according to geography and general system characteristics based on influencing factors identified in a literature review of electricity demand for water acquisition, treatment and distribution. Results of the literature review showed that, generally, electricity demand for water acquisition from groundwater resources (median of 0.16 kWh/m^3) is greater than acquisition from surface water resources (median of 0.06 kWh/m^3). County level USGS data is used to identify the source water mix for individual ZIP Codes (Dieter et al., 2018) and a composite demand is calculated based on the weighted average of each source water type. Electricity demand for water treatment is defined as the median of identified values, or 0.09 kWh/m^3. Electricity demand of water distribution is estimated relative to other regions based on the relative slope of individual U.S. ZIP Codes. Fig. S9 illustrates the modeled total electricity demand of drinking water provision for the U.S. Additional discussion on the model for displaced drinking water treatment and distribution is provided in Section S1.9.

Emission Factors and Utility Rates

In addition to climate-based time series, several geographic datasets were used to identify how geography influences the environmental impacts and cost of NPR options. Fig. S10 depicts water scarcity factors for watersheds across the U.S., with red regions corresponding to areas with high water stress (Boulay et al., 2018). U.S. EPA's eGRID dataset was used to estimate the environmental impact of energy consumption across U.S. regions (U.S. EPA, 2018b). Fig. S11 illustrates one of the five factors, GWP, used to characterize the impacts of electricity production within eGRID sub-regions. Regional electricity, natural gas and drinking water rates were obtained for LCCA calculations (AWWA, 2019; EIA, 2019; NREL, 2017) (Section S1.11).

Life Cycle Cost Analysis

The LCCA was performed by estimating system NPV (Fuller & Petersen, 1996) over an assumed 30-year period. The NPV method allows one-time, periodic and annual costs to be assessed on a consistent basis that considers the time-value of money using a 5% real discount rate. Additional detail is provided in Section S1.12.

Model Simulation Sets

To explore the effects of geography and building characteristics on system performance, NEWR was used to generate result sets using three different approaches, corresponding to unique model inputs over specific geographic coverages (Table 1).

The first simulation set was used to explore the effects of geography on system performance in a large building, holding building characteristics constant. Building characteristics were defined following Morelli et al. (2019) and Ghimire et al. (2019) – mixed use, 1,100 occupants, 19 floors, a footprint of 20,000 ft^2 (1,860 m^2), high-efficiency fixtures and end-uses of toilet flushing and laundry. Simulations were run for all 40,873 ZIP Codes (Fig. S1).

The second simulation set was used to compare the cost of a large building system relative to potable water supply rates, holding building characteristics constant. For this set, we used the same large building characteristics as Set 1 but ran simulations only for those ZIP Codes located within one of the 234 major cities included within the AWWA Rate Survey (AWWA, 2019).

The third simulation set was used to explore system performance as a function of all possible system variables, including location and building characteristics. To do so, we created an equal-area grid of ZIP Code points and, at each point, randomly generated a single set of building characteristics and modeled each system type for those building characteristics. The equal area grid (Fig. S12) was created to obtain more uniform geographic representation than the full ZIP Code dataset (Fig. S1) and reduce processing time (1,276 simulations vs. 40,873). Each grid cell is 60 miles square and was assigned the ZIP Code point from Fig. S1 that was nearest to its center. The range of building characteristics used for the random scenario generator is based on plausible ranges for onsite NPR implementation. A secondary variable – building footprint/occupant – was used to constrain building footprint to values that were reasonable based on randomly generated occupancies.

Results and Discussion

Source Water Availability

To isolate the effects of geography on system performance, RWH and ACH systems were modeled for a large building across the U.S. ("Large Building" Set, Table 1). Fig. 1 shows the fraction of this demand that can be met by RWH and ACH systems. Overall, we find that RWH systems have the potential to satisfy 0.1 to 42% of demand. Although this is in line with some studies (5-35%, Cook et al., 2014; Ghimire et al., 2019; Stephan & Stephan, 2017), it is much less than two studies conducted in Malaysia (>90%, Lani et al., 2018; Leong et al., 2019) where demand was lower and annual rainfall exceeded 70 inches (1800 mm). ACH systems can potentially satisfy between 0% and 26% of building demand. By comparison, graywater (13.1 gpcd or 49.6 lpcd) and wastewater (29 gpcd or 110 lpcd) satisfy 100% of demand.

Fig. 1 shows that large areas across the West, Midwest, Northeast, and Mid-Atlantic region have relatively low rainfall and condensate generating potential. For RWH systems, low availability results from either low total precipitation or a high fraction of precipitation falling as snow or ice. Base generation results (Fig. S4) align well with those of Lawrence et al. (2012), with their model predicting, for example, 1.2 gal/cfm in San Francisco (we predict ~1.3 gal/cfm), 9.8 gal/cfm in Washington D.C. (we predict ~8.6) and 31.4 in Miami (we predict ~29.1). The Gulf Coast Region, which is humid and warm, has the highest condensate generating potential, while the Gulf Coast and Pacific Northwest have the highest rainfall availabilities. Both Loveless et al. (2013) and Lawrence et al. (2012) showed high AC generating potential in the Southeast U.S.

Geographic Suitability – Background Factors

In addition to source water availability for RWH and ACH systems, there are several background factors that influence the impacts and benefits of all NPR systems. Fig. 2 provides a composite illustration of those factors, which include drinking water energy requirements (Fig. S9), water scarcity (Fig. S10) and eGRID GWP (Fig. S11). Generally speaking, regions with higher estimated energy demand for potable water production, higher water scarcity

Table 1
Coverage and inputs for model simulation sets.

Simulation Parameter	Simulation Set 1 – "Large Building"	Simulation Set 2 – "Large Building – AWWA"	Simulation Set 3 – "Random Generator"	Note (Units):
			Geographic Coverage	
Geographic Coverage	Entire U.S.	AWWA Cities[a]	Entire U.S.	see Fig. S1 for Simulation Set 1, Fig. S12 for Simulation Set 3
# of ZIP Codes	40,873	3,382	1,276	
			NEWR Inputs	
Building Type	Mixed Use	Mixed Use	Mixed Use	70% residential, 30% commercial
Building Occupants	1,100	1,100	min = 50 max = 1,100	count (persons)
Building Floors	19	19	min = 2 max = 20	count (floors)
Building Footprint/Occ.	18.2	18.2	min = 10 max = 20	Used to constrain area/occupant ratio (ft^2/person)
Building Footprint	20,000	20,000	min = 500 max = 22,000	Calculated as building occupants x area/occupant (ft^2)
Irrigated Area	0	0	min = 0% max = 100%	High water use area as a percentage of total building footprint (ft^2)
			Resulting Water Balance[b]	
SWA[c] – RWH	17 – 4,703	163 – 610	159 – 1227	Variable (gpd)
SWA – ACH	0 – 2,956	12 – 356	0.8 – 173	Variable (gpd)
SWA – GWMBR	14,445	14,445	959 – 13,172	Generation of 13.1 gpcd (gpd)
SWA – WWMBR	31,925	31,925	2,119 – 29,110	Generation of 29.0 gpcd (gpd)
Non-potable Demand	11,166	11,166	909 – 10,463	Per-capita demand of 10.2 gpcd multiplied by occupancy (gpd)

a – each of the 234 cities included within AWWA's 2019 rate survey (AWWA, 2019).
b – for Simulation Set 3, water balance results represent simulated ranges, not maximum ranges based on NEWR inputs.
c – SWA = Source Water Availability.

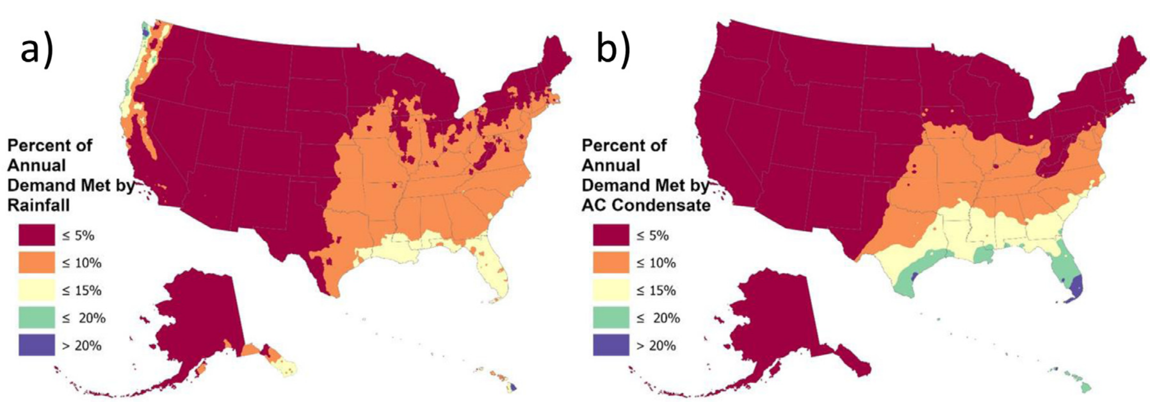

Fig. 1. Percent of annual non-potable demand met by RWH (a) and ACH (b) for a typical large building (Simulation Set 1, Table 1) having a total demand of 10.2 gpcd (38.6 lpcd).

and higher environmental impacts associated with the electricity grid (i.e., higher composite metric in Fig. 2) will generally result in lower net impacts associated with water reuse projects. Fig. 2 shows that, prior to consideration of any specific NPR system, geographic suitability is generally highest in the Southwest and Midwest and lowest in the Northeast, Mid-Atlantic Coast and Alaska.

Life Cycle Assessment

Results were generated for five environmental metrics for each simulation set in Table 1. All results represent net impacts, which account for system impacts less any avoided burdens such as those associated with displaced drinking water and thermal recovery. To allow for a more concise presentation of environmental results we performed regressions of overlapping result metrics to identify correlations. Figs. S13 and S14 illustrate the close correlation between GWP, CED and FDP. In the analyses, trends in GWP are therefore assumed to be indicative of trends in CED and FDP. WC results primarily depend on the volume of displaced potable water, which does not vary on a per gallon basis across the assessed source waters. WC results for all source waters evaluated result in savings of 4-5 liters H_2O per gallon of water provided (Fig. S15), which is more than displaced demand owing to network leakage. WS, which is calculated as a location's Water Scarcity Factor multiplied by the system WC, is directly dependent on trends illustrated in Fig. S10.

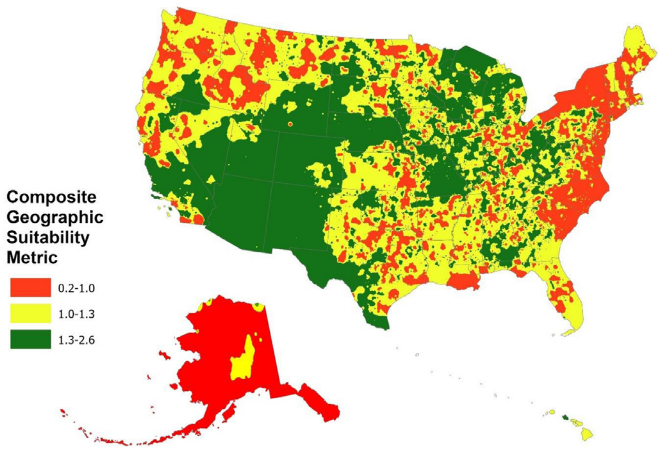

Fig. 2. Composite Geographic Suitability Metric calculated as the sum of linearly normalized (minimum = 0, maximum = 1) values for displaced drinking water electricity demand (kWh/m^3, scale of 0.30-0.59, Figure S9) AWARE Water Scarcity Factor (unitless, scale of 0-100, Figure S10), and eGRID subregion global warming potential (kg CO_2 eq./kWh, scale of 0.21-1.1, Figure S11). Low, medium and high categories represent equal frequency of occurrence.

Fig. 3 illustrates the range of net GWP results for different system types across the U.S. and across plausible building characteristic mixes ("Random Generator" Set, Table 1). Fig. 3a shows how system impact varies with annual treatment volume (system size). For RWH and ACH systems, GWP per gallon sharply increases in areas of low source water availability (e.g., Fig. 1). The difference in impacts between RWH and ACH systems, which become more pronounced at smaller system sizes, is due to inclusion of a vortex filter (to filter large roof debris such as leaves) for RWH systems but not ACH systems. Fig. S16 shows the same data plotted instead against annual rainfall (Fig. S16a) and annual condensate potential (Fig. S16b). As shown on those figures, it generally takes at least 10 in/yr (250 mm/yr) of annual rainfall or 3 gal/cfm of annual condensate potential for GWP of RWH or ACH systems, respectively, to be comparable to MBR systems. RWH and ACH systems in locations with greater than 15 inches (380 mm) of annual rainfall (Fig. S16a) or greater than 5 gal/cfm of annual condensate potential (Fig. S16b) generally outperform MBRs (without thermal recovery) in terms of GWP. Across the range of simulated building characteristics and locations, RWH is able to provide 0.1-27% of total demand, while ACH systems are able to provide 0-23% of total demand (Fig. 3 simulation results).

The variability in environmental performance of MBR systems is more affected by the environmental performance of the electricity grid than by system size. Fig. 3a shows that GWMBRs have slightly lower GWP than WWMBRs due mostly to lower energy requirements for treatment of lower strength graywater (Morelli et al., 2019). For the sizes of systems modeled (~1,000-11,000 gpd or ~3,800-42,000 lpcd), GWMBR systems require 1.1-0.75 kWh/m^3 and WWMBR systems require 1.28-0.94 kWh/m^3, which includes all pumping and disinfection processes associated with each treatment system but excludes distribution energy requirements (Table S6). By comparison, the hybrid rainwater/graywater MBRs modeled by Jeong et al. (2018) were smaller (160-480 gpd or 610-1800 lpd) but treated lower strength wastewater and used 0.62-0.45 kWh/m^3. Conversely, the MBR systems modeled by Kavvada et al. (2016) were larger (~5,000-500,000 gpd or ~19,000-1,900,000 lpd) but used 3.87-0.97 kWh/m^3. Their electricity input was based on a regression of much smaller, early versions of the technology, which may be the reason for the much higher energy demands of the smaller systems.

Fig. 3b shows GWP of GWMBR system variants plotted against the GWP of the underlying electricity grid. Results for GWMBR without thermal recovery (GWMBR_NoTR) are the same as those in Fig. 3a, while additional results are shown for GWMBR systems with thermal recovery units to offset hot water heating natural gas requirements (GWMBR_NGTR) or electricity requirements (GWMBR_ElecTR). Results show that displacing natural gas can lead to either GWP benefits or impacts depending on grid characteristics. Electricity is required to run the thermal recovery heat pump, and in instances where the grid has a larger carbon footprint than the natural gas combustion being displaced, GWP impacts will increase. By comparison, displacing electric hot water heaters always leads to environmental benefits (negative GWP), especially for locations with higher grid GWP (Fig. S11).

Fig. 4 shows the cumulative effect of geographic variables (climate, eGRID GWP, displaced drinking water) on GWP of large building systems ("Large Building" Set, Table 1). Results illustrate the geographic uniformity of MBR systems relative to RWH and ACH systems as performances of the latter are very climate-dependent. Areas of the West, particularly those outside the hydrologic influence of the Pacific Ocean, are not suitable for RWH or ACH systems. Moreover, MBRs are marginally less impactful in these areas and in New York state due to lower grid GWP (Fig. S11). High water scarcity in the West (Fig. S10) reinforces MBR suitability for locations in the West and away from the coast. For NPR-suitable areas east of the continental divide (except New York State), RWH and ACH systems may be more suitable for large buildings, though would only satisfy a fraction of demand.

Incorporation of thermal recovery adds another layer of geographic complexity to base system results. As seen in Fig. 3b, thermal recovery units have the potential to considerably increase or decrease system GWP. Fig. S17 shows results for the same "Large Building" set as was used in Fig. 4, but with incorporation of both types of thermal recovery units for GWMBR and WWMBR systems. As expected from Fig. 3b, results correspond very closely to eGRID subregions. For systems that offset natural gas consumption (Fig. S17, top two tiles), areas of the West and New York State are the most suitable owing to their clean grids. For systems that offset electricity consumption (Fig. S17, bottom two tiles), the largest benefits will be realized in areas with high-carbon grids.

Life Cycle Cost Analysis

Fig. 5 presents NPV results for each system type. Fig. 5a displays results for randomly generated scenarios ("Random Generator" Set, Table 1) as a function of system size to show how economies of scale influence system cost. Fig. 5b uses AWWA cities ("Large Building – AWWA" Set, Table 1) to illustrate how the cost of large building systems compare to the local cost of potable water; values greater than one indicate that onsite NPR is more expensive than local potable supply.

Fig. 5a shows that all systems exhibit strong economies of scale, with RWH and ACH systems being comparably cost-competitive at medium system sizes (~50,000 to 500,000 gpy or ~190-1,900 m^3/yr) and MBR systems being more cost-competitive for larger system sizes (>500,000 gpy or 1,900 m^3/yr). In addition, Fig. 5a shows that for equivalent system sizes, ACH systems are less expensive than RWH systems (though the difference is small) and WWMBR systems are less expensive than GWMBRs. The difference between ACH and RWH systems is due to the cost of a vortex filter (see Section S1.7) while differences in MBR systems are more complex. For example, although WWMBRs require slightly more electricity than GWMBRs (e.g., Fig. 3a), GWMBR systems require a separate water collection system. Even in areas with high electricity costs (possible range of 0.03 to 0.45 $/kWh (NREL, 2017)), GWMBRs are more expensive. Adding thermal recovery units de-

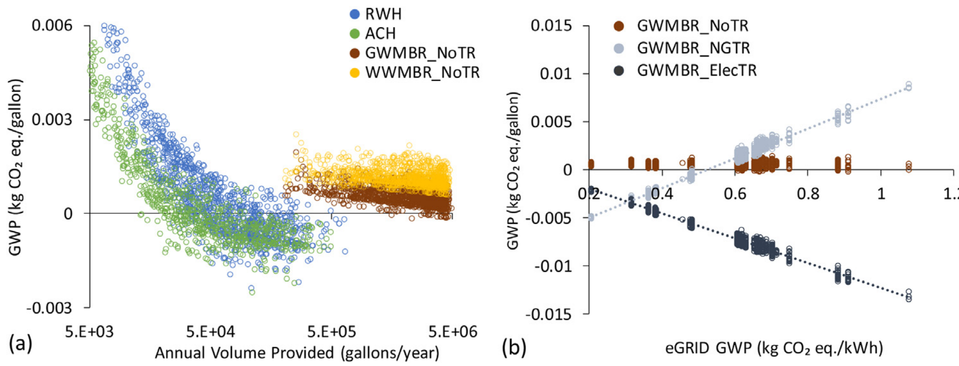

Fig. 3. Global warming potential (GWP) per gallon of recycled water delivered for Simulation Set 3 (Table 1). Tile (a) shows results for base systems – rainwater harvesting (RWH), air conditioner condensate harvesting (ACH), graywater membrane bioreactor (GWMBR) and wastewater membrane bioreactor (WWMBR) – as a function of annual non-potable water delivered by each system type. Tile (b) shows results for GWMBR with no thermal recovery (GWMBR_NoTR), GWMBR incorporating thermal recovery to offset natural gas consumption (GWMBR_NGTR) and GWMBR incorporating thermal recovery to offset electricity consumption (GWMBR_ElecTR) as a function of eGRID GWP (Figure S11).

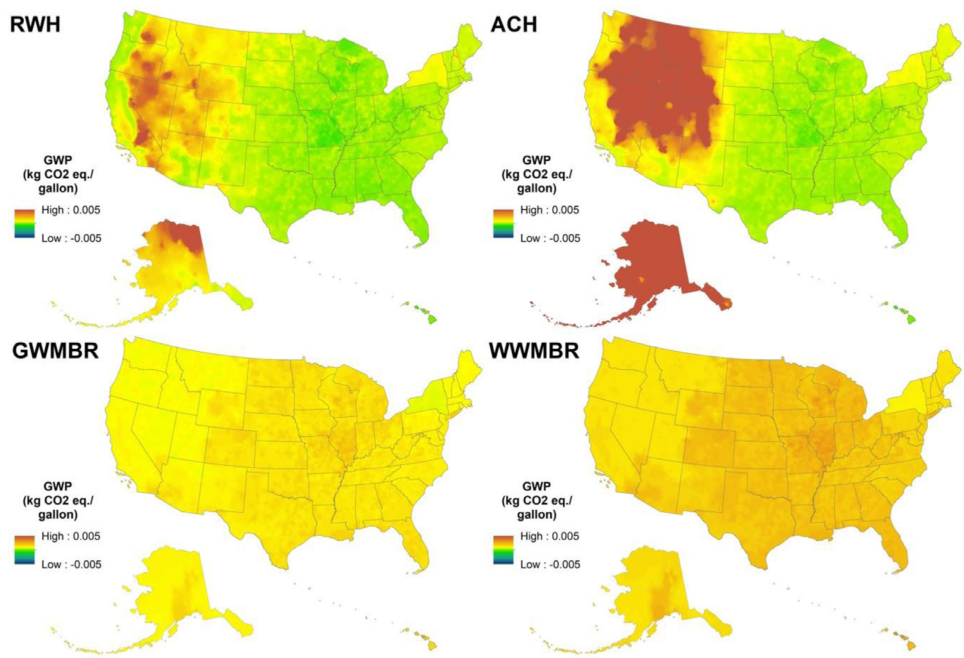

Fig. 4. Map of GWP impact of NPR system types for a large building (Simulation Set 1, Table 1).

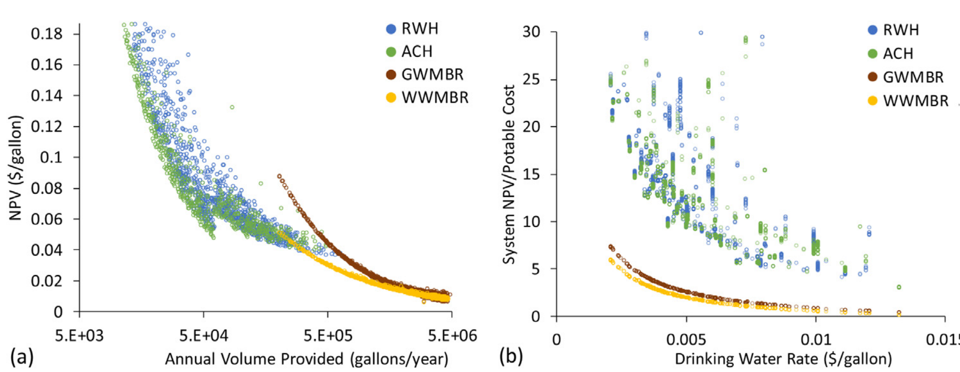

Fig. 5. a) Results of Simulation Set 3 showing net present value (NPV) per gallon of recycled water delivered as a function of system size and b) Results of Simulation Set 2 showing NPV per gallon of recycled water delivered divided by the cost of local potable supply as a function of the cost of local potable supply. System types include rainwater harvesting (RWH), air-conditioning condensate harvesting (ACH), graywater membrane bioreactors (GWMBR) and wastewater membrane bioreactors (WWMBR).

creases the NPV of MBR systems through offset of natural gas or electric utility costs. On average, incorporation of thermal recovery to offset natural gas decreases GWMBR NPV by 8% and WWMBR by 10%, while incorporation of thermal recovery to offset electricity decreases both system type NPVs by approximately 9%.

Fig. 5b illustrates the cost-competitiveness of NPR systems with local drinking water costs. Distinctive clusters visible in the figure correspond to individual cities. For RWH and ACH systems, less than 2% of systems have an NPV that is less than or equal to the local cost of potable water. For GWMBR and WWMBR systems, this figure is slightly better at 9% and 19%, respectively. Rates for locations that achieved cost parity were at least $9 per 1,000 gallons (3,785 liters) for GWMBR systems and $7 per 1,000 gallons (3,785 liters) for WWMBR systems. The majority of RWH and ACH systems are more than 5 times more expensive than equivalent potable water on a per gallon basis, and in most cases many times more. This is consistent with most previous studies, which found that higher drinking water costs (Allison et al., 2017), higher rainfall rates, subsidies (Leong et al., 2019; Stephan & Stephan, 2017), or some combination of each (Amos et al., 2018; Lani et al., 2018; Zanni et al., 2019) were required to make RWH system costs comparable to local drinking water costs.

Study Limitations and Future Research

While the list of reuse projects is growing, widespread adoption of each of these practices is limited. As such, we do not claim to be comprehensive in our evaluation of available technologies, system designs reviewed here may not be fully optimized, and there may be other system types that prove more effective upon further study (e.g., Gassie & Englehardt, 2017; Hasik et al., 2017; Leong et al., 2019). Moreover, there are additional, unconsidered factors that have the potential to affect our study results.

For RWH and ACH systems, storage tank size has a large effect on system performance. Numerous researchers have pointed to larger tanks as being critical to consistently meeting onsite demand (Lani et al., 2018; Roebuck et al., 2011; Stephan & Stephan, 2017), yet storage tanks can represent one of the largest contributors to system cost and impact (Ghimire et al., 2017, 2019). In our study, we used a constant tank sizing algorithm intended to find a balance between storage volume and cost/impact, however a full sensitivity analysis could lead to a more optimal design for a given climate and building configuration (e.g., Stephan & Stephan, 2017).

Systems that combine redundant infrastructure or multiple source waters can also result in reduced costs or impacts. Ghimire et al. (2019) found that a combined RWH/ACH system had lower impacts than individual systems and yielded a greater and more constant water supply. Similarly, several researchers evaluated systems that combined graywater recycle with RWH (Hasik et al., 2017; Leong et al., 2019; Marinoski & Ghisi, 2019; Stephan & Stephan, 2017). Future research should explore these potential synergies using a similarly comprehensive framework as the one developed here.

The microbial risk profiles of RWH and ACH systems still entail considerable uncertainty. While treatment systems were designed in accordance with risk-based guidelines (Sharvelle et al., 2017), specific LRTs have not been defined for condensate and remain uncertain for rainwater due to lack of available pathogen data (Schoen et al., 2017). Risk-based specifications may therefore change with ongoing development of risk models for these water sources, impacting treatment design (U.S. EPA, 2020).

MBRs have yet to reach technological maturity (Parker, 2011), which has implications for system costs and impacts. The energy consumption of MBR systems (0.75-1.3 kWh/m^3), one of the major contributors to system impacts, was estimated based on aeration models adapted from conventional activated sludge plants. Comparisons to similar systems showed wide variability in operational characteristics (0.45-3.9 kWh/m^3) (Jeong et al., 2018; Kavvada et al., 2016), suggesting that further system optimizations are likely. Operational and design refinements that utilize new materials and incorporate energy recovery also show promise in reducing system impacts (Harclerode et al., 2020; Smith et al., 2014).

Onsite recycle of wastewater also has several potentially beneficial consequences that were not considered here. First, reduced flows can result in wastewater cost savings depending on the sewer rate structure. Reduced flows also lessen wastewater treatment plant loadings, theoretically reducing treatment impacts. Last, onsite graywater recycle results in the remaining wastewater stream becoming more concentrated, which is more conducive to energy recovery at centralized treatment works (McCarty et al., 2011).

The focus of this work is utilization of onsite, alternative source waters for building-scale NPR. The current research is intended to support building projects or municipalities that are considering implementation of the discussed options for NPR, which can lead to further development of reuse technologies such as direct potable reuse (DPR) or district-scale NPR. The authors recommend continued research and discussion on the environmental and economic performance of both NPR and DPR projects, particularly their comparative performance as it relates to suitability for onsite reuse.

Conclusions

Results of this analysis highlight the importance of both building characteristics and location on source water availability, environmental performance and system cost.

RWH and ACH suitability depends largely on source water availability. Large portions of the country do not have suitable climates to allow RWH or ACH systems to meet even 10% of a typical large building's non-potable demand. In areas that are suitable, RWH and ACH systems can be environmentally preferable options though costs remain considerably higher than local potable water rates in most locations.

Mixed wastewater and graywater systems, on the contrary, can meet 100% of non-potable demand. GWMBR systems have slightly lower environmental impacts than WWMBR systems due to lower energy requirements, though impacts of both system types track closely with the environmental performance of the local electricity grid. Thermal recovery is a promising option to reduce environmental impacts of MBR systems, especially in areas with carbon intensive electricity grids. GWMBR systems are slightly more expensive than WWMBR systems due to the need for a separate collection system, though both system types can be cost-competitive where local drinking water prices are high—above $7 per 1000 gallons and $9 per 1000 gallons for GWMBR and WWMBRs, respectively.

Onsite NPR systems have the potential to provide substantial water savings and increase supply resiliency, especially in areas already experiencing water scarcity. Still, systems must be assessed holistically to minimize burden shifting and achieve long-term environmental and economic goals. This study, along with the companion calculator NEWR, provides a framework that communities across the U.S. can use to aid in the planning and design of onsite NPR systems that fit their specific needs.

Declaration of Competing Interest

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

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Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.watres.2020.116635.

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