

Assessing the Potential of Nontraditional Water Sources for Landscape Irrigation

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ADDITIONAL INDEX WORDS. environmental impact, graywater, ornamental plants, reclaimed water, salinity, specific ion toxicity, water quality

SUMMARY. Scarcity and competition for good quality and potable water resources are limiting their use for urban landscape irrigation, with several nontraditional sources being potentially available for these activities. Some of these alternative sources include rainwater, stormwater, brackish aquifer water, municipal reclaimed water (MRW), air-conditioning (A/C) condensates, and residential graywater. Knowledge on their inherent chemical profile and properties, and associated regional and temporal variability, is needed to assess their irrigation quality and potential short- and long-term effects on landscape plants and soils and to implement best management practices that successfully deal with their quality issues. The primary challenges with the use of these sources are largely associated with high concentrations of total salts and undesirable specific ions [sodium (Na), chloride (Cl), boron (B), and bicarbonate (HCO_3^-) alkalinity]. Although the impact of these alternative water sources has been largely devoted to human health, plant growth and aesthetic quality, and soil physicochemical properties, there is emergent interest in evaluating their effects on soil biological properties and in natural ecosystems neighboring the urban areas where they are applied.

Municipal tap water, potable and often of the highest quality for direct human consumption and uses (cooking, bathing, etc.), has been the traditional source for urban landscape irrigation. In some regions, such as Texas, as much as one-half of the residential usage of municipal tap water has been devoted to outdoor use/landscape irrigation during the growing season (Cabrera et al., 2013). Accelerated population growth, climate

change (in particular severe and prolonged droughts), and fierce competition for water resources among agricultural, urban, and industrial sectors are effectively calling for a significant reduction, suspension, or both of the use of municipal tap water resources for irrigation of landscapes and lawns [Duncan et al., 2009; U.S. Environmental Protection Agency (USEPA), 2012]. Recent examples include those implemented in Texas, California, and Massachusetts during the severe droughts experienced during 2010–14, 2011–15, and 2015–17, respectively [Hanak et al., 2016; Rosen, 2017; Texas Water Development Board (TWDB), 2017]. Therefore, aside from the logical premise of radically changing the design of urban landscape plantings, based on the careful selection of suitable plant materials

that thrive and aesthetically perform well within the limits of the expected local precipitation (Cabrera et al., 2013; Tanji et al., 2007; USEPA, 2013b), it is contended that any supplemental landscape irrigation will—predictably and mandatorily—have to be provided by nonpotable, alternative water sources (Duncan et al., 2009; Tanji et al., 2007).

Irrigation water quality considerations

Compared with most other agronomic and horticultural species, the aesthetic appearance, visual quality, or both of ornamental and landscape plants are generally considered more important than growth rate (Bernstein et al., 1972; Cassaniti et al., 2012; Niu and Cabrera, 2010). Therefore, considerable attention is often paid to environmental factors and cultural/management practices that impact the aesthetic appearance of the above-ground components of landscape plants and turfgrass species. The chemical quality of the water sources used to irrigate managed landscape sites, thus, takes prominence, particularly when applied through conventional overhead irrigation systems (Farnham et al., 1985; Miyamoto and White, 2002; Tanji et al., 2007). The chemical constituents of water with the greatest impacts on the growth and aesthetic qualities of traditional landscape plants are total soluble salts [expressed as electrical conductivity (EC)], pH, Na and its related sodium adsorption ratio (SAR), Cl, B, and HCO_3^- alkalinity [or alternatively calcium carbonate alkalinity], leading to the development of specific recommendations regarding their most suitable (or less problematic) levels or concentrations (Table 1).

This article results from the workshop “Maintaining Healthy Landscapes Under Drought and/or Permanent Water Restrictions” held on 20 Sept. 2017, at the ASHS Annual Conference, Waikoloa, HI, and sponsored by the Ornamentals/Landscape and Turf (O/LT) Professional Interest Group.

We would like to acknowledge the grant funding, for projects related to this study, by the Floriculture Nursery Research Initiative of the USDA-NIFA, the Horticultural Research Institute (Washington, DC), the New Jersey Agricultural Experiment Station, Texas Water Resources Institute, and Texas A&M AgriLife Research.

The use of trade names in this publication does not imply endorsement by Rutgers, The State University of New Jersey, U.S. Department of Agriculture, and Texas A&M AgriLife Research of products named nor criticism of similar ones not mentioned.

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https://doi.org/10.21273/HORTTECH04023-18

Units

To convert U.S. to SI, multiply by	U.S. unit	SI unit	To convert SI to U.S., multiply by
0.4047	acre(s)	ha	2.4711
1233.4819	acre-ft	m ³	0.0008
29.5735	fl oz	mL	0.0338
0.0929	ft ²	m ²	10.7639
3.7854	gal	L	0.2642
2.54	inch(es)	cm	0.3937
1.6093	mile(s)	km	0.6214
1	mmho/cm	dS·m ⁻¹	1
1	ppb	ng·g ⁻¹	1
1	ppm	mg·L ⁻¹	1
0.9072	ton(s)	Mg	1.1023

Table 1. Selected chemical quality parameters in water suitable for (sprinkler) irrigation of ornamental and landscape plants, and found in a mildly brackish aquifer, municipal reclaimed waters from different ecogeographic regions of the United States, and residential graywater.

Chemical parameter ^z	Suitable water ^y	Mildly brackish aquifer ^x	Municipal reclaimed water					Residential graywater ^r
			Santa Rosa, CA ^w	El Paso, TX ^v	San Antonio, TX ^u	Lake Buena Vista, FL ^t	Evesham, NJ ^s	
pH	6.0–8.0	7.3–8.3	7.3	7.2–7.8	7.2–7.5	7.1–7.9		6.4–8.7
EC (dS·m ⁻¹)	<1.0	1.6–4.7	0.7	1.5–1.9	0.9–1.2	0.6–0.7	0.5–0.6	0.3–1.4
Na (mg·L ⁻¹)	<70	50–560	82	160–280	90–102	74–100	54–58	30–480
Cl (mg·L ⁻¹)	<110	30–510	—	200–340	135–190	99–140	56–61	3–140
B (mg·L ⁻¹)	<1.0	0.1–0.4	0.4	0.2–0.3	0.2–0.3	—	0.2–0.3	0.4–1.5
HCO ₃ ⁻ (mg·L ⁻¹)	<110	105–250	—	194–220	259–305	—	142–153	70–470
SAR	<6.0	2.8–8.7	—	5.1–9.0	2.4–2.6	—	2.3–2.4	4.2–28.1

^zEC = electrical conductivity; Na = sodium; Cl = chloride; B = boron; HCO₃⁻ = bicarbonate; SAR = sodium adsorption ratio; 1 dS·m⁻¹ = 1 mmho/cm; 1 mg·L⁻¹ = 1 ppm.

^yCompiled from Farnham et al. (1985), Duncan et al. (2009), and Tanji et al. (2007).

^xHuaco Bolson aquifer in El Paso, TX (Anderholm and Heywood, 2003; George et al., 2011).

^wAverages for 2000–11, City of Santa Rosa Water (2018). Provides irrigation for 1280 and 5120 acres of urban and agricultural land, respectively.

^vEl Paso Utilities (Buszka et al., 1994; Ornelas, 2005). Providing municipal reclaimed water for urban landscape irrigation in El Paso, TX, since 1963.

^uSan Antonio Water System (2013). Among the largest U.S. municipal reclaimed water delivery systems [130 miles (209.2 km) of purple-coded pipeline].

^tReedy Creek Improvement District (2012–16). Provides municipal reclaimed water to irrigate 1515 acres (613.1 ha) at the Walt Disney World Resort.

^sSoil and Water Consulting Brookside Laboratories (2014); Mattson (2006). The Indian Springs Country Club in Evesham, NJ, is a 150-acre (60.5 ha) golf course irrigated with municipal reclaimed water since 1999.

^rCompiled from Eriksson et al. (2002), Gross et al. (2005), Holgate et al. (2011), Negahban-Azar et al. (2012), and Sheikh (2010).

The effects of total salt, ionic composition, or both of irrigation waters on plant growth and aesthetic quality are commonly and broadly described in three categories: osmotic effects, specific-ion toxicity effects, and nutrient imbalances (Bernstein et al., 1972; Cassaniti et al., 2012; Grattan and Grieve, 1999; Grieve et al., 2012; Niu and Cabrera, 2010). Total salt concentration of the soil solution dictates its osmotic strength, whose increases can significantly decrease soil water potential with concomitant reductions in plant water uptake, growth parameters, and overall productivity (biomass accumulation). A long-standing recommendation for the maximum desirable concentration of total soluble salts expressed in EC units for most ornamental and landscape plants is below 1.0 dS·m⁻¹ when irrigated through sprinklers (Table 1). Dominance of the anion and cation balance in irrigation waters and soil solutions, by specific ions (Farnham et al., 1985; Grattan and Grieve, 1999), in particular Na, Cl, and B (Table 1), leads to their significant accumulation in plant tissues and the development of characteristic ion injuries (toxicities) that can negatively affect plant aesthetic quality (Cassaniti et al., 2012; Niu and Cabrera, 2010). In the case of trees, vines, and other woody plants, the detrimental effects (foliar injury) caused by specific-ion toxicities are exacerbated when the tissues are wetted by sprinkler irrigation, whereas the

response is lessened when saline waters are applied through drip or flood irrigation (Grieve et al., 2012). The total salt concentration and dominance of specific ions in irrigation water or soil solutions also affect or disrupt the availability, uptake, transport, and accumulation/partitioning within the plant of other essential nutrients (Grattan and Grieve, 1999), leading to nutrient imbalances that are often difficult to diagnose by conventional visual and analytical tissue testing procedures (Costello et al., 2003).

The availability of nontraditional irrigation water sources varies across ecogeographic regions, subjected largely to the regional/temporal nature and accessibility of the primary water resources and uses in each location (Cabrera et al., 2015; Duncan et al., 2009; Niu and Cabrera, 2010). Some of the potentially available nontraditional water sources for landscape/urban irrigation include rainwater/stormwater, brackish (naturally saline) aquifer water, MRW, A/C condensates, and residential graywater. A common trait shared by several of these alternative water sources is their relatively high content of salts, specific ions, or both (Table 1). These pose a major challenge to most common ornamental plants and landscape plantings. The literature indicates that woody ornamentals, along with woody fruit crops, are relatively more sensitive to salinity stresses in comparison with agronomic crops and amenity turfgrasses (Boland, 2008; Duncan et al., 2009; Farnham

et al., 1985; Grieve et al., 2012; Niu and Cabrera, 2010). Furthermore, intensive and extensive use of these alternative waters not only presents challenges to the plants irrigated with them, but also impacts the soils and environments within and surrounding the vicinity of the landscapes where they are applied.

Nontraditional water sources

RAINWATER AND STORMWATER. There is a resurgent interest for rooftop rainwater harvesting (RWH) systems for urban/suburban irrigation and water conservation (Thomas et al., 2014; TWDB, 2005; USEPA, 2013a), particularly in the drier, arid locations of the southwestern United States and in other regions stricken by recent severe droughts (Hanak et al., 2016; Rosen, 2017; TWDB, 2017). Depending on the design and characteristics of the RWH systems, this source offers the potential to have the best irrigation water quality; i.e., low salt content, compared with most other sources. Nevertheless, rainwater collected from rooftops could have substantial concentrations of contaminants such as nutrients, heavy metals, particulate matter, and microbes derived from precipitation and atmosphere (wet and dry depositions, respectively), wildlife (feces), and the rooftop materials (Campisano et al., 2017). However, the intensity and frequency of rainfall events and limitations in RWH storage capacity are major restrictive

conditions for consistent irrigation of entire average-sized residential landscape areas. To put it in perspective, a single 1/2-inch irrigation event to a typical 5000-ft² residential mixed (lawn plus shrubs, trees, and flower beds) landscape (Cabrera et al., 2013) requires a water volume of 1558 gal. Filling a tank or cistern of such capacity will require 1.57 inches of rainfall captured with 80% efficiency (TWDB, 2005) on an average-sized house rooftop of 2000 ft² area. Thus, passive, small-scale residential RWH systems (USEPA, 2013a) would in most instances (due to functional and financial constraints) provide limited volumes, sufficient only for irrigation of very small areas such as beds, planters, and containerized plants. Conversely, the use of active RWH systems for irrigation of public and commercial sites with larger ratios of rooftop area to landscape areas, and large storage capacities, has a higher practical and financial potential for implementation. Rainwater harvesting systems, in particular the larger capacity, active type, require careful consideration and adherence to municipal ordinances, state guidance manuals, or both before their implementation/installation (USEPA, 2013a).

Reuse of stormwater (runoff from impervious urban surfaces) has been considered a significant and important alternative source for urban irrigation, particularly in extensive lawn/turfgrass areas (Duncan et al., 2009; Hatt et al., 2006). However, stormwater runoff commonly accumulates pollutants such as sediments, oil and grease, chemicals, and heavy metals—in addition to deicing salts in some regions—as it moves across urban impervious surfaces. Traditionally, stormwater has been almost exclusively addressed for the purpose of drainage and flood control, with a management emphasis focused on protecting the water quality of the waterways where it is discharged (Duncan et al., 2009; USEPA, 2013a). Stormwater recycling could be significant in urban water budgets but its reuse in applications with high potential for human contact, such as pressurized sprinkler irrigation, still needs the development of innovative techniques to collect, treat, and store these runoff effluents (Hatt et al., 2006; Shuttleworth et al., 2017).

There are design propositions and engineered applications, such as rain gardens and bioswales, that already harvest or retain stormwater in urban vegetated areas, contributing to the reduction of heat island effects through enhanced evapotranspiration and surface cooling (Coutts et al., 2013; Shuttleworth et al., 2017).

There are increased local and regional (i.e., state) interests and initiatives in the United States and abroad to integrate RWH systems, both small-passive and large-active, as a significant and incentivized/creditable best management practice for stormwater management (Campisano et al., 2017; Shuttleworth et al., 2017; USEPA, 2013a). This augurs a widespread and official promotion of rainwater as a leading source for supplemental urban irrigation, with the eventual prohibition, partial or complete, of potable water use for this purpose.

AIR-CONDITIONING CONDENSATES.

In some regions, condensate water from A/C systems is a potential source for landscape irrigation (Guz, 2005), particularly in sites with a relatively large indoor footprint vs. landscape footprint, such as public and commercial buildings. Because a condensate is formed from moisture in the air, its chemical quality can be good, requiring minimal to no treatment for storage, immediate irrigation use, or both. San Antonio, TX, was the first city in the United States, in 2006, to have an ordinance requiring all new commercial buildings (≥ 10 tons of total cooling capacity) to capture condensate from A/C systems, with the intention of using it for applications such as irrigation (Glawe, 2013). There are still design, engineering, and maintenance issues being addressed for onsite condensate collection and use systems. In the case of landscape irrigation applications, these include storage and treatment of algal and microbial (e.g., *Legionella*) growth in pipes and components, and hookup to irrigation systems (Glawe, 2013; Guz, 2005). From an economical standpoint, A/C condensate collection and reuse in applications such as landscape irrigation is more viable for large buildings ($> 100,000$ ft²) located in relatively humid climates (i.e., San Antonio, TX, and Dallas, TX, as opposed to El Paso, TX, and Phoenix, AZ), where more condensate is produced (Belzer Adams,

2017). However, the condensate output from small residential heating, ventilation, and A/C systems is rather small to be a significant alternative water source for outdoor irrigation.

BRACKISH GROUNDWATER. Some inland/continental regions of the United States have vast water resources in the form of naturally saline aquifers, as well as other coastal regions where aquifers are affected by saltwater intrusion. Although brackish groundwater containing total dissolved solids up to 3000 mg·L⁻¹ (equivalent to an EC of ≈ 4.7 dS·m⁻¹) are allowed for irrigation use in Texas (TWDB, 2013) and other states, these present the biggest management challenges and the greatest potential for long-term adverse effects on plants and ecosystems (Duncan et al., 2009). Practical and sustainable use of moderately brackish waters (Table 1) for landscape irrigation will expectedly require blending with less saline water sources, application to salt-tolerant species, and use of judicious irrigation practices and leaching methods modulated by systematic salinity monitoring (Boland, 2008; Cassaniti et al., 2012; Duncan et al., 2009; Farnham et al., 1985; Grieve et al., 2012; Miyamoto et al., 2001; Niu and Cabrera, 2010; Tanji et al., 2007). A sensible integration of these practices should help moderate undesirable osmotic and specific-ion effects and minimize their concentration and deposition in the immediate and surrounding soil and water environments (Duncan et al., 2009). Although we are unaware of any large-scale direct uses of brackish water resources for urban irrigation in the United States, there are municipalities actively tapping into brackish water aquifers as primary water sources for desalination and incorporation into their potable water systems (e.g., El Paso, TX, and San Antonio, TX).

MUNICIPAL RECLAIMED WATER.

This is defined as municipal wastewater that has been partially or fully treated to meet specific quality criteria suitable for various (largely nonpotable) uses or discharged into surface bodies of water [National Research Council (NRC), 2012]. In some parts of the United States (e.g., California) and abroad, MRW is also referred as “recycled” water [California Ag Water Stewardship Initiative (CAWSI), 2018; California Department of Water

Resources (CDWR), 2016; Parsons et al., 2010; Rahman et al., 2016]. This water source has been a long-standing and viable alternative for both agricultural and urban irrigation, among other reuse applications (NRC, 2012; USEPA, 2012). In urban settings, MRW has a well-established use record in golf courses and large corporate and municipal parks and landscapes, particularly in the southeast and southwest regions of the United States. A recent survey indicates that 25% of the estimated total volume of irrigation water (1.859 million acre-ft) used by U.S. golf courses in 2013 came from reclaimed water sources (Golf Course Superintendents Association of America, 2015).

Availability of reclaimed water has been limited to a few end users, as collection of raw sewage influents, their treatment, and discharge of effluents or their subsequent distribution for reuse are strictly regulated (Drinan and Spellman, 2013; USEPA, 2012), using a separate delivery pipeline system (identified by its code-mandated purple color and companion signage). Until recently, the reuse of MRW for urban irrigation was, thus, limited, cost-effective, or both for only large-sized customers, such as golf courses and managed landscapes in industrial or municipal parks [Mattson, 2006; San Antonio Water System (SAWS), 2013]. Within the past decade, there has been an increase in local and state mandates to provide more residential and commercial service connections for MRW, which is effectively leading to a significantly higher use of this source for urban irrigation in these cities and regions (CAWSI, 2018; CDWR, 2016; Parsons et al., 2010).

Reclaimed waters could have similar drawbacks as mildly brackish water, with relatively high levels of total salinity and undesirable specific ions (Duncan et al., 2009). The quality and composition of MRW depend largely on the nature and uses of primary source water(s), influent types, collection system(s), treatment(s) used, and even the natural precipitation characteristics of each location (Drinan and Spellman, 2013). This is highlighted by some of the most critical chemical quality parameters of MRW produced, and intensively/ extensively used for urban irrigation uses

(≥15 years), in some representative municipalities from across the country (Table 1). It is readily apparent that the MRW from the two listed Texas locations, which rely heavily on saltier groundwater primary water sources, have salinity (EC), Na, Cl, and HCO₃ alkalinity values that are higher than those from the California, Florida, and New Jersey locations, which have higher average annual precipitation, rely also on primary surface water sources with better (less saline) quality, or both. The irrigation quality parameters of the two Texas MRW are slightly to moderately undesirable, higher than those recommended for typical woody ornamental shrubs and trees, but still adequate for most annuals and turfgrasses (Duncan et al., 2009; Farnham et al., 1985; Nackley et al., 2015; Niu and Cabrera, 2010). The MRW from U.S. east coast locations (Florida and New Jersey) are of considerable better irrigation quality and, in fact, better than municipal potable tap water from many southwestern U.S. municipalities.

The treatment level and disinfection standards required for MRW to be used in sites with a high potential for periodic/direct contact with humans (Drinan and Spellman, 2013; NRC, 2012) override any considerations with respect to their irrigation water quality (Duncan et al., 2009). In some instances, MRW even match the potability (e.g., no pathogenic/biological activity) of municipal tap water (Table 2). This primary concern for human health is reassured by redundant, reliable, and robust treatment processes for elimination/removal of pathogenic organisms and some key contaminants (NRC, 2012). This approach has led to no reported incidents of human illness, nor legal claims or lawsuits associated with the use of MRW for irrigation in public and private landscaped sites with heavy human traffic, not even when used in the conventional and organic-certified production of vegetable and fruit crops (Parsons et al., 2010).

RESIDENTIAL GRAYWATER. Defined as the wastewater from laundry, showers, and bathtubs, graywater can constitute up to 60% of all wastewater from a household, yielding 90–100 gal/d (32,850–36,500 gal/year)

from an average U.S. household (Roesner et al., 2006; Sheikh, 2010). This means that a steady reuse of graywater for residential irrigation could represent substantial savings of potable water supplies, potentially providing 10.5–11.7 inches of annual irrigation for an average-sized (5000 ft²) residential mixed (turf and plants) landscape (Cabrera et al., 2013). In Texas, for example, there are at a minimum 1.7 million acres of managed landscapes, with the majority being residential landscapes irrigated mostly with municipal potable water resources, placing urban irrigation as the state's third largest water user after agricultural irrigation and other urban uses. Thus, initiatives leading to a widespread institution of graywater ordinances and effective usage of this alternative source could lead to significant potable water savings (Cohen, 2009; Sheikh, 2010).

Laundry effluents, which represent as much as 40% of residential graywater (Sheikh, 2010), can be easily and inexpensively used for landscape irrigation by simple diversion, compared with the need for expensive retrofitting plumbing to capture all graywater effluents in existing houses (Allen et al., 2010; Hartin and Faber, 2015). Simple laundry-to-landscape graywater systems are typically driven by both the pumping capacity of the washing machines and gravity, discharging the nonfiltered effluents, via hoses or tubing, into mulched basins or branched manifold drains directed at shrubs and trees (Hartin and Faber, 2015). Alternatively, the laundry effluents can be discharged into portable drums with hose

Table 2. Biological quality comparison of several water sources based on their level of total coliform bacteria (from Sheikh, 2010).

Water source	MPN/100 mL ^z
Drinking water	<1
Disinfected tertiary reclaimed water	<2.2
Disinfected secondary reclaimed water	<23
Undisinfected reclaimed water	20–2,000
Graywater	100–100 million
Raw wastewater	Millions to billions

^zMPN = most probable number; 1 MPN/100 mL = 0.2957 MPN/fl oz.

attachments, to be then moved around the residential landscapes as needed. Larger, complex, treated and filtered, and energy-driven (pressurized) graywater systems are economically justifiable in multifamily housing units, and commercial and public buildings generating substantial graywater effluents and having more extensive landscaped areas. Design and management of these complex systems will invariably require permits and professional installation by licensed plumbers or irrigation/landscape contractors.

The chemical composition of graywater is highly variable because it depends on its sources and variability among and within household water uses (Cohen, 2009; Eriksson et al., 2002). From an irrigation quality perspective, laundry graywater can have salinity levels comparable with milder reclaimed water sources from regions with moderate to high annual precipitations (Table 1). However, graywater effluents are likely to have undesirably higher Na, B, and HCO₃⁻ concentrations (Gross et al., 2005; Negahban-Azar et al., 2012), as these ions are constitutive in surfactants,

builders (technically these are water softeners), and peroxide bleaching agents, the three main ingredients (>70% by weight) in laundry detergents (Smulders et al., 2002). In addition, the use of hypochlorite bleach in laundry produces graywater effluents with chlorine (Cl₂) levels in excess of those used for surface disinfection (Smulders et al., 2002), which can cause plant toxicity (Cabrera et al., 2015) and negatively affect soil microorganism populations (Table 3). Therefore, it is recommended to avoid the use of laundry effluents laden with hypochlorite bleach for irrigation and to divert these directly into the household sewer system (Hartin and Faber, 2015).

By definition graywater is untreated wastewater and, even with the exclusion of toilet and kitchen effluents, almost invariably it will have concentrations of fecal coliform (Table 2) and other biological indicators that exceed the maximum limits allowed by local, federal, and international standards for water uses involving human contact (Roesner et al., 2006; Sheik, 2010). This fact—a priority concern for public health—has been one of the

leading reasons for a restrained promotion and permitting of residential graywater systems in the United States compared with other countries (Allen et al., 2010; Sheik, 2010). In those U.S. locations where graywater is officially permitted for irrigation, the potential risks of its microbiological quality have been addressed by requiring the use of systems and delivery methods that prevent human exposure. This includes prohibitions to its storage which requires it to be used as it is produced or diverted back to the sewer system, restrictions to its irrigation application method (surface, drip, and subsurface methods—never in pressurized sprinkler systems), and strict precautions to avoid cross-connection with potable water sources (Allen et al., 2010; Hartin and Faber, 2015; Sheik, 2010).

Management and environmental considerations for alternative irrigation waters

An increasing demand and competition for fresh and high-quality water resources challenge the need

Table 3. Soil microbial community composition, assessed by phospholipid fatty acid (PLFA) analyses, in soil samples collected from the root zone of ‘Radrazz’ Knock Out rose plants drip irrigated for 12 weeks with well water and laundry graywater. The soil texture from the research plot, located in Uvalde, TX, was clay (Hyperthermic Aridic Calcicustolls) of the Uvalde series (R.I. Cabrera, unpublished data).

Soil microbial parameters ^z		Well water ^y	Laundry graywater ^x	
Total living microbial biomass [PLFA (ng·g ⁻¹)] ^w		1,644 (Avg) ^y	883 (Poor)	
Functional group diversity index (from <1.0 to >1.6)		1.57 (Very good) ^y	1.2 (Slightly below avg)	
Functional group ^u	Biomass [PLFA (ng·g ⁻¹)]	(%)	Biomass [PLFA (ng·g ⁻¹)]	(%)
Total bacteria	881	54	439	50
Gram (+)	539	33	310	35
Actinomycetes	213	13	101	11
Gram (-)	342	21	130	15
Rhizobia	12	0.7	0	0
Total fungi	183	11	25	3
Arbuscular mycorrhizae	52	3	0	0
Saprophytes	131	8	25	3
Protozoa	21	1	0	0
Undifferentiated	558	34	419	47

^zPLFA analyses based on procedures by Findlay (2004), performed by Ward Laboratories, Inc. (Kearney, NE). The results shown are from composite soil samples; 12 soil cores [1 inch diameter × 6 inches long (2.5 × 15.2 cm)] per irrigation treatment; 1 ng·g⁻¹ = 1 ppb.

^yThe well water had the following water quality parameters: pH of 6.9, EC of 0.5 dS·m⁻¹, 18 mg·L⁻¹ of sodium, 30 mg·L⁻¹ of chloride, and sodium adsorption ratio of 0.5; 1 dS·m⁻¹ = 1 mmho/cm, 1 mg·L⁻¹ = 1 ppm.

^xGraywater made with well water and fabric detergent (Tide; Procter & Gamble, Cincinnati, OH), fabric softener (Downy; Procter & Gamble), and hypochlorite bleach (Clorox; Clorox Co., Oakland, CA) mixed at the manufacturers’ recommended rates for a large laundry load in a conventional washing machine [producing 42 gal (159.0 L) of combined washing and rinsing effluent].

^wThe total amount of PLFA detected in a soil sample determines its total viable microbial biomass, as it correlates well with other measures of microbial biomass (Willers et al., 2015).

^uCategories for total living microbial biomass: very poor (<500 ng·g⁻¹), poor (500+ to 1000 ng·g⁻¹), slightly below average (1000+ to 1500 ng·g⁻¹), average (1500+ to 2500 ng·g⁻¹), slightly above average (2500+ to 3000 ng·g⁻¹), good (3000+ to 3500 ng·g⁻¹), very good (3500+ to 4000 ng·g⁻¹), and excellent (>4000 ng·g⁻¹). Categories for functional group diversity index: very poor (<1.0), poor (1.0+ to 1.1), slightly below average (1.1+ to 1.2), average (1.2+ to 1.3), slightly above average (1.3+ to 1.4), good (1.4+ to 1.5), very good (1.5+ to 1.6), and excellent (>1.6). Categorical ratings by Ward Laboratories (Kearney, NE).

^vSpecific phospholipid fatty acids are used as signature biomarkers for specific microbial groups, allowing for an estimate of their biomass (Willers et al., 2015).

and uses of traditional water sources for urban landscape irrigation (Duncan et al., 2009; Rahman et al., 2016; Roesner et al., 2006; Tanji et al., 2007). Despite any discussion in favor of the ecosystem services and societal benefits of vegetated landscapes in urban areas, there is an urgency to change and adapt their designs to be less reliant on scarce and valuable potable water resources. We support the contention that plant materials for urban landscapes will have to become adaptable to minimal or zero supplemental irrigation (USEPA, 2013b). Furthermore, as one of the main challenges from the use of most alternative water sources is a high salinity level, selection of plant materials should heavily emphasize salt tolerance (Bernstein et al., 1972; Costello et al., 2003; Grieve et al., 2012; Nackley et al., 2015; Niu and Cabrera, 2010; Tanji et al., 2007; Wu and Dodge, 2005).

The sustainability of landscape sites irrigated with any saline water sources—including brackish water, MRW, and graywater (Table 1)—requires holistic/comprehensive management plans that include site (soil and water) assessments, plant selection, irrigation design and management, soil and water amendments, cultural practices, and systematic monitoring (Boland, 2008; Duncan et al., 2009; Rahman et al., 2016; Roesner et al., 2006; Tanji et al., 2007). In addition to overall salinity, these programs need to address concurrent challenges of sodicity, alkalinity, and high B concentrations also found (Table 1) in these sources (Boland, 2008; Gross et al., 2005; Holgate et al., 2011; Miyamoto, 2012; Negahban-Azar et al., 2012; Parsons et al., 2010). From a salt/ion dilution approach, intensive applications of alternative water sources in regions receiving lesser precipitation, using minimal leaching and having heavier (e.g., clay) soil textures, will likely exacerbate resulting issues of soil salinity, sodicity, and loss of soil structure; plant growth reductions; and tissue toxicities/disorders leading to the loss of aesthetic value and plant death (Miyamoto, 2012; Negahban-Azar et al., 2012; Pratt and Suarez, 1990; Tanji et al., 2007).

The irrigation method and scheduling practices chosen to apply alternative water sources become more

relevant than with traditional waters (Boland, 2008; Grieve et al., 2012; Tanji et al., 2007). These need to strive for an overall management that reduces foliage wetting (particularly in woody species; Grieve et al., 2012; Miyamoto and White, 2002; Niu and Cabrera, 2010), applies volumes that maintain control of salinity changes and limits within the root zone (i.e., suitable leaching; Miyamoto, 2012; Pratt and Suarez, 1990; Tanji et al., 2007), and reduces runoff losses of irrigation water and its chemical constituents, which can potentially impact downstream ecosystems (Duncan et al., 2009; Rahman et al., 2016). The use of alternative waters also requires frequent monitoring and adjustment of irrigation system distribution uniformity as to avoid the buildup of salts in localized areas throughout the landscape (Costello et al., 2003; Duncan et al., 2009). Furthermore, the use of poor-quality irrigation waters also imposes the need for enhanced filtration, and systematic monitoring and cleaning/maintenance of emitters (drippers, nozzles, and heads) and other irrigation system components (Costello et al., 2003; Duncan et al., 2009).

Blending of different water sources for landscape irrigation could be considered a potential management option for the dilution of challenging water quality parameters (Glawe, 2013). However, the strict protection of potable water in the United States limits its potential blending only with active rainwater systems (into a collection cistern, with the tap water refill pipe protected by an air gap), albeit it could be used as a backup water supply provided it minimizes or eliminates any potential for cross-connections and backflows into the potable water system (USEPA, 2013a). Most, if not all, existing regulations also address cross-connection and backflow prevention procedures that ensure complete separation of MRW from potable water supplies (NRC, 2012). Because of public health concerns with the microbiological quality of untreated graywater (Table 2), its blending with other sources is not permitted in the United States, nor is it allowed to be stored unless treated and disinfected (Allen et al., 2010; Sheik, 2010).

A priority interest has been and is placed on the potential effects of

alternative water sources such as MRW and graywater on human health, but considerably little attention has been devoted to their environmental impacts and risks (Mattson, 2006; NRC, 2012; Rahman et al., 2016; Roesner et al., 2006). A recent comprehensive report by the National Research Council of the National Academy of Sciences addresses the need for research evaluating the environmental impacts of these water sources in “sensitive ecological communities” (NRC, 2012), whereas others claim that a lack of this information is what has slowed their extensive reuse for urban irrigation applications (Roesner et al., 2006). Paradoxically and historically, disposal of the bulk of municipal wastewaters (treated and untreated) has been carried out by discharging them into surface bodies of water, thus already impacting their immediate and surrounding ecosystems. Interestingly, the National Research Council report simultaneously calls for research to assess the potential use of reclaimed water and impacts of its contaminants on habitat restoration projects, and the use of environmental buffers (including soil aquifer treatment systems, wetlands, rivers, and reservoirs) for the attenuation of contaminants in wastewaters for the design and optimization of future potable reuse systems (NRC, 2012).

At a smaller, residential and public/commercial urban landscape scale, there is a substantial body of information on the effects of irrigation with MRW sources on soil physical and chemical characteristics, such as salt accumulation, sodicity, alkalinity, loss of soil structure, and heavy metal deposition, to name a few (Duncan et al., 2009; Gross et al., 2005; Holgate et al., 2011; Miyamoto, 2012; Nackley et al., 2015; Negahban-Azar et al., 2012; Pratt and Suarez, 1990; Tanji et al., 2007). As the sum of managed residential landscapes can approach, or even rival, the areas devoted to irrigated agriculture in some regions (Cabrera et al., 2013), an extensive irrigation use of MRW and graywater could significantly impact not only these (built) urban environments, but also extend to the surrounding and (hydrological) downstream natural ecosystems.

In addition to effects on the physical and chemical properties of soils, there is an increasing interest in the impact alternative irrigation sources have on the biological status of soils (Roesner et al., 2006). Long-term irrigation (4–26 years) with MRW in carbon-limited sandy agricultural soils has induced a significant increase in soil microbial abundance by providing a carbon and mineral (nitrogen, phosphorous, and potassium) nutrient source (Hidri et al., 2010). Short-term irrigation (20 weeks) of perennial ryegrass (*Lolium perenne*) turfgrass with various water sources has produced differential effects on soil microbial community composition (Holgate et al., 2011), with significantly alkaline-sodic municipal tap water and two graywaters (all averaging: pH = 8.6, EC = 0.7 dS·m⁻¹, Na = 197 mg·L⁻¹, HCO₃⁻ = 361 mg·L⁻¹, and SAR = 27) increasing some soil bacterial and fungal biomarkers and decreasing others. Preliminary results from ongoing studies on the irrigation of landscape plants with laundry graywater point to adverse effects on total microbial biomass and diversity in soil samples collected from the root zone of ‘Radrazz’ Knock Out landscape roses (*Rosa*) compared with irrigation with good-quality well water (Table 3). The significant reductions in all soil microbial communities, including annihilation of beneficial rhizobia and arbuscular mycorrhizae, is attributed to the inclusion of hypochlorite bleach in this laundry graywater treatment. These effluents had Cl₂ levels in excess of those used for surface disinfection (Cabrera et al., 2015), producing fairly short-term (within 12 weeks) adverse effects on soil microorganism populations. These observations lend support to the recommendation to avoid irrigation with laundry effluents containing hypochlorite bleach, diverting them instead directly into the household sewer system (Hartin and Faber, 2015).

Concluding remarks

An accelerated dwindling availability of potable and good quality water supplies for urban landscape irrigation is significantly changing the future of urban landscapes and vegetated spaces, requiring a major shift to designs and plantings that

require minimal irrigation, and this to be accomplished with nontraditional water sources. The successful use of alternative irrigation sources requires knowledge on their challenging chemical properties and their temporal variability (assessed by continuous monitoring), allowing to implement and modulate the best management practices that successfully deal with their quality and effects on plants and soils. In addition, to safeguard any impacts to human health, plants, and soils, there is a pressing need to assess the potential effects they might have on the built and natural ecosystems neighboring the areas where they are applied.

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