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REVIEW

Pivotal role of municipal wastewater resource recovery facilities in urban agriculture: A review

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Abstract

Urban agriculture provides a promising, comprehensive solution to water, energy, and food scarcity challenges resulting from the population growth, urbanization, and the accelerating effects of anthropogenic climate change. Their close access to consumers, profitable business models, and important roles in educational, social, and physical entertainment benefit both developing and developed nations. In this sense, Urban Water Resource Reclamation Facilities (WRRFs) can play a pivotal role in the sustainable implementation of urban agriculture. Reclaimed water as a recovered resource has less supply variability and in certain cases can be of higher quality than other water sources used in agriculture. Another recovered resource, namely, biosolids, as byproduct from wastewater treatment can be put to beneficial use as fertilizers, soil amendments, and construction material additives. The renewable electricity, heat, CO_2, and bioplastics produced from WRRFs can also serve as essential resources in support of urban agriculture operation with enhanced sustainability. In short, this review exhibits a holistic picture of the state-of-the-art of urban agriculture in which WRRFs can potentially play a pivotal role.

Practitioner Points

- Reclaimed water can be of higher quality than other sources used in urban agriculture.
- Biosolids can be put to beneficial use as fertilizers, soil amendments, and construction material additives.
- The renewable electricity, heat, CO_2, and bioplastics produced can also serve as essential resources in support of urban agriculture.

KEYWORDS

energy, food, resource recovery, sustainability, urban agriculture, wastewater

Zhi-Wu Wang is a member of the Water Environment Federation (WEF).

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INTRODUCTION

With the rapid population growth and urbanization, it would be difficult for rural agriculture industry to meet the food supply need of urban societies by 2050 due to the limited arable land, water scarcity, as well as energy-intensive farming and transportation (Heinemann et al., 2011; Tilman et al., 2001). Urban agriculture (UA) offers close access to consumers, profitable business models, and educational, social, and physical benefits to urban communities, making UA a comprehensive solution to water, energy, and food problems facing modern urban societies (Astee & Kishnani, 2010; Hou et al., 2009; Lohrberg et al., 2016; Lovell, 2010). In order to make the UA concept viable, Water Resource Reclamation Facilities (WRRFs) serving urban municipal wastewater treatment needs could play a pivot role in that they simultaneously recover water, nutrients, and energy, the three essential resources required for UA (Tian et al., 2018). To shed light on this importance, this study reviewed publications collected in Web of Science and Google Scholar between 1983–2021 with regard to the essential roles that WRRFs could play in the context of UA. It is anticipated that the knowledge presented in this study can enlighten researchers and practitioners in the fields of UA and WRRFs to work together for enabling a circular economy along the effort to mitigate the accelerating effects of anthropogenic climate changes.

UA AND ITS BENEFITS

Definition of UA

As illustrated in Figure 1, Mougeot (1996) defined UA as an economic activity correlated to the production of food or nonfood products in urban or peri-urban areas. The definition of UA was expanded in the following years; for example, the processing and distribution of food, livestock breeding, aquaculture, and horticulture were added in 1999 (Bailkey & Nasr, 1999); urban waste and resources reuse, human and public services, and intensive production methods were added in 2000 Mougeot (2000); wide-ranging activities, economic and business vitality, and individual and community well-being were added in 2002 (Butler & Maronek, 2002). The definition of UA currently encompassed the technological innovations of novel food-production and material reuse methods that maximize output in urban areas (Abelleira-Pereira et al., 2015; Hodgson et al., 2011).

Benefits of UA

More than three-quarters (76%) of world's population resided in urban areas in 2015 (Congress, 2020). Urbanization and population growth have brought new issues related to land shortages, environmental pollution, and urban food security. Thus, more arable land for food supplies is demanded. For example, 40% of the arable land in the United Kingdom would have to be dedicated to meeting the food demand of London which houses 12% of the UK population (Miccoli et al., 2016). Moreover, with the growth of urban resident income, the food demand could add further pressure to the global food supply systems. To meet this demand it is projected that the world's agricultural production will increase by between 70% and 100% over current levels (2050 Heinemann et al., 2011). Global arable land availability is scarce and is only projected to increase by 12% compared to today (Heinemann et al., 2011; Tilman et al., 2001).

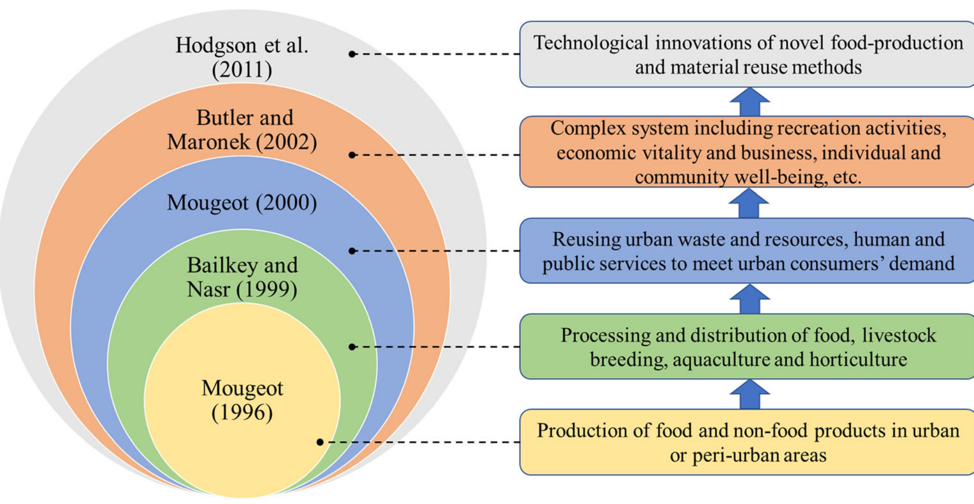

FIGURE 1 Development of urban agriculture (UA) definition

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Although there is a possibility to meet the food supply demand by increasing the unit productivity of arable land through using fertilizers and pesticides, cultivating newly selected crops, and employing energy-intensive or water-demanding practices (Tilman et al., 2001), rural-centric farming approaches alone are environmentally unsustainable due to fossil fuel consumption, greenhouse gas (GHG) emissions, exploitation of soil and water resources, and aquatic contamination from agricultural runoff (Vandermeer et al., 2009). In addition, the present retail sale and food distribution systems heavily rely on road and air transportation which have high embedded environmental costs due to elevated fuel consumption and air pollution associated with these transport mediums (Paxton, 1994).

To ensure urban food security, one option is to split large-scale, centralized rural farming to a multiplicity of small-scale farming systems that interconnects producers, processors, distributors, and consumers in close distance (Brown et al., 2003). UA can offer such an option to address these urbanization challenges through (i) transportation distance reduction, (ii) profitable business model creation, (iii) individual and social well-being improvement, (iv) climate and ecological effects mitigation, and (v) urban waste and spaces utilization.

Reduce transportation distance

Food products typically travel between 1500 and 2500 miles from rural farms to urban points of sale, which is 25% farther than food products traveled back in 1980 (Halweil, 2002). Fruits and vegetables often spend 7 to 14 days in transit before arriving in the supermarket (Brown et al., 2003), which led to 3% to 37% higher grocery price in urban supermarkets (Hunger, 1990). UA can reduce transportation distance and energy footprint, as well as shorten supply chains. It also has the potential to minimize product packaging, storage, and distribution time/cost; reduce price differentials between producers and consumers; and improve food accessibility to local markets (Aubry et al., 2008). Localizing food production in urban areas can also solve food waste problems due to food spoilage over long distance transporting (Stuart, 2009).

Profitable business model creation

Investment in UA tends to have a good return rate per dollar input, for example, every $1 invested in a community garden plot yields approximately $6 worth of vegetables (Bellows et al., 2003). Urban farmers can earn up to $90,000 and $136,000 per acre in Ohio and Philadelphia, respectively, by selecting the proper crops and growing techniques (Mogk et al., 2010). Other successful examples of UA include Detroit, where the UA grossed $200 million in sales with approximately 5000 jobs created (Mogk et al., 2010). UA with proper implementation can support economic vitality by providing neighborhood residents with job opportunities and clean public spaces rooted in community involvement (Lovell, 2010).

Improve individual and social well-being

The visibility of trees, grass, and flower gardens as a result of UA can provide easy access to nature and improve general life satisfaction of people in urban areas, reduce irritability and mental fatigue, and restore calmness and the ability to concentrate (Kaplan & Kaplan, 1990). In addition, UA can create green spaces for social gathering, build a sense of community, and reduce stress, anger and even blood pressure (Ulrich, 1992). Blair and Nichaman (2002) found that gardening three to four times a week approximately equals moderate walking or bicycling in terms of health benefits. Honeyman (1992) found that urban built environment with vegetation produced more mental rehabilitation than those without vegetation and that vegetated urban scenes had more positive psychological impacts than nature only scenes by comparing human response to remote nature scenes and urban built environment scenes. UA have additional functions such as youth education, tourism, and community development through school programs, work programs, and other agriculture-related activities, which can improve the life qualities of current residents and make the city more attractive to new residents (Mogk et al., 2010).

Mitigate climate and ecological effects

As a result of urbanization, the area of impervious surfaces such as roads, driveways, sidewalks, parking lots, and rooftops through which water cannot infiltrate has expanded rapidly (Lu & Weng, 2006). In the United States, the total impervious surface area of the 48 states and Washington DC was estimated to be $112{,}610 \pm 12{,}725\ km^2$ (Elvidge et al., 2004). This transformation of urban green land to impervious surface area reduced the amount of carbon fixed through photosynthesis by 1.6% of the pre-urban values and caused urban heat islands effect (Lamptey et al., 2005). Practicing UA

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can effectively alleviate this phenomenon in various ways. For example, by providing vegetation and facilitating carbon sequestration, UA can potentially increase evapotranspiration and produce cooling, moderate urban climate, and reduce urban heat island effect (Zasada, 2011). In addition, concomitant water usage such as irrigation could reduce air temperature, increase relative humidity, and promote the extent of living comfort by increasing evaporation (Lamptey et al., 2005). Moreover, urban green spaces can play a role in species preservation for birds and butterflies by providing food, resting spaces, and protection along migratory flight paths and increase a city's biodiversity with plant variety and beneficial soil microorganisms, insects, and animals (Brown & Jameton, 2000). Plants in UA can not only absorb soil contaminants through their root systems to reduce soil erosion and ground water contamination but also reduce air pollution by absorbing pollutants through their foliage (Brown & Jameton, 2000).

Utilize urban waste and spaces

WRRFs can recycle and reuse urban waste to support UA. One of the most efficient ways is through recycling purified wastewater for irrigation. Cheema et al. (1996) reported that California saves 759,000 m^3/day fresh water by recycling and reusing treated effluent from its 200 WRRFs. UA can decrease the burden of WRRFs using a combined system that also catches and reuses stormwater runoff. Biosolids from WRRFs and composting urban organics such as food waste, leaves, and lawn clippings can also be applied as fertilizers in UA (Brown et al., 2003).

UA also can function as a substitute for abandoned or vacant lots and yield multiple profits in a short time. The US Accounting Office identified 130,000 to 425,000 contaminated vacant industrial sites or brownfields that could be safely converted to agricultural purposes if properly redeveloped (Bailkey & Nasr, 1999). UA regenerates and revitalizes a city by transforming these vacant lots from eyesores into bountiful and beautiful gardens which can feed peoples' souls and bodies (Brown et al., 2003). This has been well illustrated by a food project in urban and suburban Boston that staff and volunteers annually raise more than 120,000 pounds of fresh vegetables and 12,000 pounds of other food on vacant lots located less than 2 miles from downtown Boston (Washington, 2000). Thirteen times more food yield per acre than rural farms was produced from urban gardens by utilizing raised beds, soil amendments, and season extenders such as row covers and hoop houses (Washington, 2000).

UA IN DEVELOPED COUNTRIES

Although the status of UA in developing countries has been reviewed in several previous studies (De Bon et al., 2010; Hamilton et al., 2014; Orsini et al., 2013), to our best knowledge, its applicability in developed countries has not be reviewed except for some individual project reports in Singapore, North America, and Europe (Astee & Kishnani, 2010; Hou et al., 2009; Lohrberg et al., 2016; Lovell, 2010). To fill this knowledge gap, this section focuses on the various UA types adopted in developed countries. Different taxonomies have been derived to classify the existing forms of UA in developed countries. Features such as temporary or permanent, horizontal or vertical, monocultures or polycultures, the farm area and annually product selling area, locations and the nature of lands, operating characteristics, capital inputs, and how they interact with urban systems can be used to categorize UA. This review used a taxonomy mainly based on forms, scales, locations, and functions of the UA primarily used in developed countries (Table 1).

Commercial urban farm

Commercial urban farms sell their products directly through farmers' markets (Table 1). In the United States, it was reported that the largest direct sales of commercial urban farm products occur in California, New York, Pennsylvania, Michigan, and Ohio (Timmons & Wang, 2010). Smaller markets with several vendors are also around in many US urban cities. Some urban farmers even go from door to door with their produce (Brown et al., 2003). These urban farms provide convenience for inner-city residents to access food, while also increasing their own income.

There are 11,760 ha of commercial urban farmland in Greater London (Defra, 2011). In terms of production, commercial urban farms account for the overwhelming majority of food grown in London. Garnett and Gillie (1999) reported that the Lower Lea Valley straddling the northeastern boundary of Greater London is the heart of commercial horticulture in London, which has hosted market gardens since the eighteenth century. With the glasshouses and other technologies, Lower Lea Valley is well-known for the production of not only ornamental flowers but also vegetables such as cucumbers, tomatoes, and peppers.

Community garden

Community gardens are places where plants or foods are grown in communal settings (Agustina & Beilin, 2012).

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TABLE 1 Comparison of different UA types

Categories	Features	Benefits	Examples	References
Commercial urban farm	Directly from farm to markets	Economic benefits for farmers; food provision and other additional values	Smaller markets; door-to-door peddling with trucks	Brown et al. (2003), Timmons and Wang (2010)
Community garden	Communal plots for each household or community use	Food budget saving; promotion of healthy diet; social and communal benefits	"The peoples garden (USDA); homeless garden project" (Santa Cruz, CA); "just food" (new York City) Community-supported agriculture	Agustina and Beilin (2012), Johnston and Baker (2005), Mok et al. (2014), Saldivar-Tanaka and Krasny (2004)
Backyard garden	Plots around homes, the smallest scale	Food and nutrient supplement, esthetic, life quality improvement	Balconies, decks, and courtyards	Brown et al. (2003), Kortright and Wakefield (2011)
Allotment garden	An area subdivided into small plots, which are rented under a tenancy	Reducing poverty, improving self-provision	Statutory, temporary, and private allotments	Lohrberg et al. (2016)
Green roof	Roofs with a vegetated surface and substrate	Increasing food production; stormwater management and building temperature regulation	Intensive and extensive green roofs	Engelhard (2010), Mok et al. (2014)
Vertical farming	Growing plants are vertically stacked in layers	Hight output, water and space saving	Hydroponic, aeroponic, or aquaponic; building-based vertical farms and shipping-container vertical farms	Birkby (2016), Mok et al. (2014)
Rooftop greenhouse	A greenhouse built on the building roof	Increased crop yields; space and energy saving	Gotham greens, the vinegar factory, and Lufa farms in New York	Cerón-Palma et al. (2012), Despommier (2011), Sanyé-Mengual et al. (2015)
Educational and therapeutic garden	Gardens with educational and healing effects	Raising public awareness; treatment to mental and physical disorder	School gardens; physical and mental health care institution gardens	Lohrberg et al. (2016)

Abbreviation: UA, urban agriculture.

In most cases, community gardens (Table 1) are large land divided into smaller plots for each household's use and can be owned by a municipality, an institution, a community group, a land trust, or private ownership (Brown et al., 2003). The Urban Gardening Program, established by US Congress in 1977, provided annual grants of US $150,000–250,000 through the Cooperative Extension Service for such gardens (Brown & Jameton, 2000; Hynes & Howe, 2002). The USDA started a new initiative, called "The People's Garden", to promote community gardening in 2009 (Mok et al., 2014). Currently, the American Community Gardening Association estimates that there are over 18,000 community gardens in the United States and Canada, including neighborhood, public housing, and school gardens (Kortright & Wakefield, 2011). Nonprofit organizations and government agencies, such as the "Homeless Garden Project" in Santa Cruz, California, 'Just Food' in New York City, and 'Food-Share' in Toronto, Canada, work with community gardens to encourage food production and to distribute food to those in need (Johnston & Baker, 2005; Saldivar-Tanaka & Krasny, 2004).

Backyard garden

Urban backyard gardens are plots around homes, including balconies, decks, and courtyards (Table 1). Produce can be grown or raised even in the simplest of containers (Brown et al., 2003). Backyard food gardeners are

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motivated by different factors such as cooking with fresh ingredients, teaching children about nature, environmental sustainability, connecting to cultural or past identity, esthetics, personal hobby, and neighborhood exchange (Kortright & Wakefield, 2011). In much of North America, the harvest from the backyard garden has eased the food budgets of low-income families and their network of family and friends.

Allotment garden

An allotment garden is an area subdivided into small plots, which are rented under a tenancy agreement (Table 1). They usually stem from municipal initiatives on public land, and their regulation is highly formalized, sometimes following specific regional or national laws. They can also be managed by an organized group or even established as an allotment garden association. Allotment gardens emerged in the 18th century to cope with urban poverty, and the First World War prompted their expansion (Lohrberg et al., 2016). Nowadays, their functions have shifted from self-provision to food supplement, education, and social and physical entertainment.

Green roof

Green roofs, roofs with a vegetated surface and substrate, provide ecosystem services in urban areas, including improved stormwater management, better regulation of building temperatures, and increased food production (Table 1). Rooftop food production is a more recent addition to the mix of urban food production systems in North America. There are both commercial-scale and community-focused projects in existence, as well as household-scale rooftop gardens (Engelhard, 2010). In May 2009, Toronto passed a bylaw requiring green roofs to be installed on new commercial, institutional, and residential developments, and while not specifically addressing the issue of rooftop agriculture, it listed "opportunities for local food production" as one of the many environmental benefits (Kaill-Vinish, 2009). There are various plans for large-scale rooftop food production, such as the Brooklyn Grange project in New York which entails a 100,000 ft^2 hydroponic greenhouse and grows over 80,000 lbs. of organically cultivated produce per year, making it the largest rooftop farm in the world (Plakias, 2016). In extensive green roofs, plant species are typically restricted to those that are drought-tolerant and shallow-rooted, minimizing both irrigation and depth of planting media to limit weight (Mok et al., 2014).

Vertical farming

Vertical farming (Table 1) sometimes called sky farming is based on the concept that the most efficient growing conditions can be built around the crop rather than trying to adapt to the natural environment (Mok et al., 2014). In vertical farming, growing plants are vertically stacked in layers that may reach several stories tall, in controlled indoor environments, with precise light, nutrients, and temperatures. Some common choices of structures to house vertical farming systems include abandoned warehouses in cities, new buildings built on environmentally damaged lands, and even in used shipping containers from ocean transport (Birkby, 2016). Vertical farms come in different shapes and sizes, but all vertical farms use one of three soil-free systems for providing nutrients to plants, namely, hydroponic, aeroponic, or aquaponic systems (Birkby, 2016). Hydroponic system involves growing plants in nutrient solutions that are free of soil. In aeroponic systems, plants grow in an air/mist environment with no soil and using up to 90% less water (Birkby, 2016). An aquaponic system takes the hydroponic system one step further by combining plants and fish in the same ecosystem.

Rooftop greenhouse

Rooftop greenhouse hrs (Table 1) as a new form of UA consist of a greenhouse built on the roof of a building that typically generates produce via soilless culture systems (Cerón-Palma et al., 2012). These structures are considered a component of the "building-based UA" movement, which also covers vertical farming or sky farming (Despommier, 2011). Gotham Greens and Lufa Farms are local producers based in New York and Montreal that have built Rooftop greenhouses ranging in size from 830 to 2900 m^2 (Sanyé-Mengual et al., 2015). Cerón-Palma et al. (2012) performed a preliminary assessment of Rooftop greenhouses. Energy modeling results illustrated the environmental benefits of energy flow exchange between rooftop greenhouses and office buildings. The results showed that the introduction of residual heat from the greenhouse into the building on an ideal winter day could substitute 87 kWh of the heating demand.

Educational and therapeutic garden

Educational gardens are gardens located in educational institutions that provide garden-based learning to their community (schools, kindergartens, etc.) or gardens

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developed by environmental or social centers that offer educational services to visitors. School gardens are the most common form spreading environmentally and climate-friendly gardening ideas and practices (Lohrberg et al., 2016). Therapeutic gardens are typically located inside the city, at physical and mental health care institutions, applying basic healing effects through gardening and agriculture (Table 1). With a variety of plants and flowers carefully chosen, these gardens can stimulate the sight, smell, and touch, awakening the senses, memories, and emotions. They can be used for the treatment of mental disorders, autism, Alzheimer's disease or cerebral paralysis, addiction to drugs and alcohol, and so on.

PIVOTAL ROLES OF WRRFS IN UA

WRRFs in the United States alone process an estimated 32 billion gallons of wastewater per day (Pabi et al., 2013). WRRFs discussed herein are centralized in urban areas receiving municipal wastewater with typical inflow compositions reported for developed countries (Davis, 2010). Given their centralized nature in the city, drought-proof water supply, and the significant infrastructure already in place, WRRFs can play a pivotal role in promoting the concept of UA. The water, nutrient, and energy are three essential resources required for supporting agricultural activities (Tian et al., 2018). WRRFs can recover these three essential resources from domestic wastewater in support of UA (Coats & Wilson, 2017; Guest et al., 2009; Takashi et al., 2007). Indeed, resource recovery from wastewater has been aggressively advocated (Cluster, 2015) and is the reason why traditional wastewater treatment plants have been rebranded as WRRFs to recognize and reinforce the significant resource recovery potential that exists in wastewater streams (Coats & Wilson, 2017). Nowadays, industry professionals have broadly agreed that WRRFs should evolve into resource recovery factories that valorize valuable commodities out of wastewater (Coats & Wilson, 2017). To this end, an array of technologies has been developed as illustrated in Figure 2 to produce UA relevant commodities such as irrigation water, fertilizers, heat, electricity, carbon source, as well as construction and packaging materials.

Reclaimed water for UA

UA irrigation with reclaimed water

Reclaimed water is becoming increasingly attractive as an alternative source to surface and ground water for UA

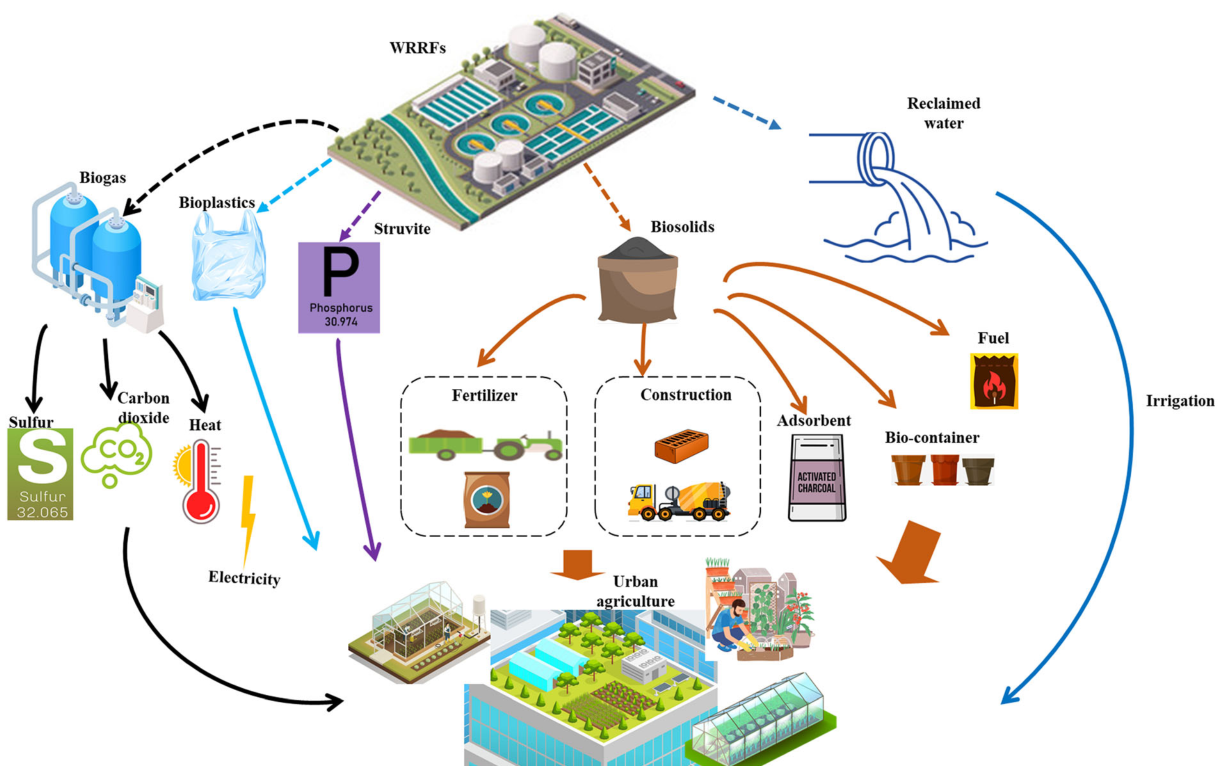

FIGURE 2 Schematic illustration of the pivotal roles of Water Resource Reclamation Facilities (WRRFs) in support of urban agriculture (UA)

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irrigation, because of its high quality, stable supply, and potential energy savings.

High-quality irrigation water

The reclaimed water from advanced WWRFs is often higher quality than the surface water to which it is discharged (Bureau of Reclamation, 2010; EPA, 2012; Ruetten et al., 2004). For example, Upper Occoquan Service Authority located in Centreville, Virginia, has been practicing indirect potable reuse by generating reclaimed water with quality higher than that in the Occoquan reservoir where it discharges (EPA, 2012). The primary chemical constituents to consider for UA irrigation include salinity, sodium, trace elements, chlorine residual, and nutrients, and excess concentrations of those can cause negative effects on crop growth and human health (EPA, 2012). A range of treatment options are available in WRRFs to deliver almost any level of water quality depending upon the expected use of the reclaimed water (EPA, 2012).

Stable supply

The generation of wastewater is actually not significantly lower during droughts (EPA, 2012). This means that reclaimed water is virtually a drought-proof water supply. Properly planned, this drought-proof supply can be used to increase the reliability of the overall water supply system (Ruetten et al., 2004). In addition, municipal wastewater generation is relatively constant throughout the year and typically increases with population growth in urban area (EPA, 2012). Moreover, reuse of municipal water produced and reclaimed all in the same place within a limited geographical range requires less resources for pumping and transport to UA sites.

Energy saving

Water reuse for UA can reduce energy consumption by eliminating additional potable water treatment and associated water conveyance because reclaimed water typically offsets potable water use and is used locally. For example, about 20% of California's electricity is consumed by water-related energy use, including potable water conveyance, storage, treatment, and distribution and wastewater collection, treatment, and discharge (Klein et al., 2005). The amount of energy required for treatment and transport of potable water is generally much greater than the energy required to treat wastewater for reclamation in southern California. The estimated net energy savings from switching from potable water treatment and transport to wastewater reclamation and reuse could range from 0.7 to 1 TWh/year. At a power cost of $0.075/kWh, the savings would be on the order of $50 to $87 million per year (Schroeder et al., 2012).

Applications of reclaimed water for UA irrigation

Water treatment technologies (combined with disinfection) offer a ladder of increasing water quality, and choosing the right level of treatment should be dictated by the end application of the reclaimed water for achieving economic efficiency and environmental sustainability (Council, 2012).

Unrestricted irrigation for food crop

Unrestricted use of reclaimed water is the irrigation (surface or spray irrigation) for food crops that are intended for human consumption, including crops eaten raw without processing (Schroeder et al., 2012). An example of large-scale reclaimed water irrigation for raw-eaten food crops is in Monterey County, California. More than 5000 ha of lettuce, broccoli, cauliflower, fennel, celery, strawberries, and artichokes has been irrigated with recycled water for more than a decade (EPA, 2012). The World Health Organization (WHO) guidelines and regulations (Organization, 2006) have specified treatment processes, water quality standards, and monitoring regimes that minimize risks for use of reclaimed water for irrigation of crops that are ingested by humans. In the United States, 27 states or territories have developed rules, regulations, or guidelines regarding this type of reclaimed water use by 2012 (EPA, 2012).

Restricted irrigation for food crop

Restricted use of reclaimed water is the use of reclaimed water to irrigate crops that are either processed before human consumption (those prior to sale to the public or others have undergone chemical or physical processing sufficient to destroy pathogens) or not consumed by humans such as seed crops, industrial crops, fodder crops, orchard crops, and alfalfa plants (EPA, 2012). It reduces opportunities of human exposure to the water, resulting in less stringent treatment and water quality requirements than other forms of reuse (EPA, 2012). By 2012, 43 states or territories have developed rules, regulations, or guidelines regarding this type of reclaimed water use, and there are several states that even do not require disinfection if certain parameters are met (EPA, 2012).

Other UA relevant reclaimed water application

Reclaimed water can also be used with biosolids for the production of bricks, concrete, and the construction of gardens and other facilities related to UA. In addition, it can be used to control dust generated by these construction sites. Likewise, the reclaimed water requires a minimum of secondary treatment and disinfection prior to use. The details of relative criteria in different states of

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the United States or other countries were summarized elsewhere (EPA, 2012).

Biosolids for UA

An estimated eight million dry tons of nutrient-rich biosolids are produced annually from WRRFs in the United States (Coats & Wilson, 2017). Biosolids are the stabilized and processed residuals from wastewater treatment are nutrient- and energy-rich materials and can be utilized in many ways in support of UA (Møller et al., 2009). In 1993, the US Environmental Protection Agency (USEPA) promulgated CFR Title 40 Part 503, establishing federal standards for the use or disposal of biosolids. These regulations defined pathogen reduction requirements for municipal sewage sludge resulting in Class A and B biosolids standards. Class B biosolids are treated by processes to significantly reduce, but not totally eliminate, pathogen concentrations and therefore require special handling, whereas Class A biosolids are treated by additional processes to further reduce pathogens to very low concentrations that do not require special handling or other restrictions (USEPA, 1990). In addition, Part 503 rule also uses the term Exceptional Quality (EQ) to characterize biosolids that meet low-pollutant and Class A pathogen reduction (virtual absence of pathogens) limits and that have a reduced level of degradable compounds that attract vectors (USEPA, 1990). EQ biosolids are considered a product that is virtually unregulated for use, whether used in bulk, or sold or given away in bags or other containers. The Class A requirements shall be met when biosolids are applied to a lawn or a home garden or sold or given away in a bag or other container for application to the land, while Class B requirements shall be met when biosolids are applied to agricultural land, forest, a public contact site, or a reclamation site (USEPA, 1990). Different types of biosolids can be used as fertilizers and soil amenders.

Direct farmland application as fertilizers

Biosolids generally contain 1%–5% phosphorous and 1%–10% nitrogen as well as significant levels of trace metals required by crops for healthy growth (Crocker et al., 2018). Direct land application of biosolids can represent an interesting strategy for improving crop yield by increasing nitrogen and phosphorus contents, soil organic matter contents, and fertility (Alvarenga et al., 2015). This blend of nutrients, from an economic standpoint, makes biosolids a very attractive organic alternative to conventional chemical fertilizers (Darvodelsky & Bridle, 2012).

Globally, land application of biosolids is considered to be the most sustainable use of this material (Darvodelsky & Bridle, 2012). Additionally, biosolids use in UA can offset the use of chemical fertilizers which can have nonrenewable origins and require large amounts of energy to produce. Based on carbon footprint values of N and P, for every dry ton of biosolids used in agriculture, approximately 192.5 kg of CO_2 emissions is abated from avoided fertilizer production (Goh et al., 2018). Consequently, direct farmland application of biosolids could play an important role in UA especially in commercial farm or community garden for its ability to improve soil properties, nutrient condition, and crop growth.

Soil amendment

Because of foot and wheel traffic destroying vegetative cover and compacting the surface soil (Craul, 1985; Gregory et al., 2006), a crust on or within several centimeters of the ground surface is typically formed on urban soil, which deteriorates the soil properties. Biosolids have been used to improve soil physical properties, such as reducing bulk density (Punshon et al., 2002), increasing total porosity (Chang et al., 1983) and water-holding capacity (Punshon et al., 2002), and improving pore size distribution (Pagliai & Antisari, 1993); chemical properties such as cation exchange capacity and organic matter contents, (Ozores-Hampton et al., 2011; Stoffella et al., 2001); and biological properties such as microbial communities, soil fauna, and aboveground plant biomass yield (Sydnor & Redente, 2002). Various types of biosolids, that is, amended biosolids, limed biosolids, composted biosolids, heat-dried biosolids, and hydrochars from hydrothermal carbonization of biosolids, can be applied towards bringing these positive properties to the receiving soil.

Amended biosolids

A blending biosolids with other recycled or reused materials can improve the biosolids properties for UA use. For example, blending biosolids with organic (e.g., sawdust, biochar, and lignite) or inorganic (e.g., sand or quarry rock "fines") materials have been practiced for reducing the moisture content for urban soil use (Ervin et al., 2014; Yu et al., 2013). These organic materials can also reduce nutrients (i.e., nitrogen) leaching from soils amended with biosolids, sorb contaminants, and mitigate negative impacts on the environment (Paramashivam et al., 2017). For example, biochar produced from the pyrolysis of organic waste such forestry, garden, and

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agricultural wastes can sorb xenobiotic organic contaminants (Spokas et al., 2009; Wang et al., 2010) and metals (Uchimiya et al., 2010). Sawdust can remove pathogens by the toxicity of tannin compounds (Banegas et al., 2007). Lignite-amended soil had higher C content (35%), N content (33%), and a C/N ratio than control soil (Budaeva et al., 2006). Hence, lignite mixed with biosolids could reduce heavy metal uptake by plants (Simmler et al., 2013) and inorganic N leaching from soil (Paramashivam et al., 2016). District of Columbia Water Authority (DC Water) has replaced Class B lime-stabilization processing system with a thermal hydrolysis pretreatment enhanced anaerobic digestion process to produce Class A/EQ biosolids (Johnston et al., 2019). The resulting biosolids were successfully blended with woody (e.g., shredded mulch and sawdust) and mineral (e.g., sand) substrates to create byproducts with ideal moisture contents, allowing their handling and application in urban environments.

Limed biosolids

Limed biosolids are biosolids blended with liming agents such as calcium oxide or calcium hydroxide to destroy residual pathogens and minimize odor, as well as limit the attraction of disease vectors such as insects and other living organisms that can transport biosolids-borne pathogens away from site. Therefore, limed biosolids can become Class A biosolids useful in UA as fertilizers or topsoil. To obtain limed biosolids, the pH of the biosolids should be raised with these alkaline materials to a specified level for a specified time according to USEPA promulgated CFR Title 40 Part 503 (USEPA, 1990). By increasing pH accordingly, the metal hydroxide precipitation has been formed, and metal leachability has been reduced. This results in products which have higher alkalinity and low heavy metal bioavailability, which are well suited for use in heavily cropped fields or acidic soils requiring alkaline amendments (Goh et al., 2018).

In Western Australia, the Water Corporation produces lime-amended biosolids from their Subiaco WRRF (Stamatelatou & Tsagarakis, 2015). Although this material is suitable to market as-is, blending it with red clay has yielded a product known as Lime-amended BioClay (LaBC®). In addition to functioning as an organic fertilizer for acid sandy soils, LaBC® also overcomes water repellence, a significant issue of arid soil types like those found in Australia (Shanmugam et al., 2014). On the market since 2011, LaBC® is sold to farmers as a three-in-one product to offset the use of agricultural lime and chemical fertilizers as well as to decrease the natural water repellence of sandy soils and increase their water-holding capacity (Stamatelatou & Tsagarakis, 2015).

Composted biosolids

Composted biosolids are biosolids blended with organic green waste and composted to achieve volatile solids reduction, odor mitigation, and residual pathogen inactivation. Composted biosolids are suitable for horticultural and domestic use and used as top dressing for fields (Goh et al., 2018). Vietor et al. (2010) found that composted biosolids can enhance turfgrass establishment and soil properties. In Australia, Sydney Water has 20% of all biosolids produced ends up in compost with approximately two thirds of all compost products in Sydney containing a blend of biosolids (Darvodelsky & Bridle, 2012).

Heat-dried biosolids

Heat-dried biosolids are biosolids that have undergone some form of drying which effectively kills all pathogens, concentrates the nutrients in the biosolids, and significantly reduces the volume of material to be transported to farmland. Potential odor and pathogen issues are mitigated during heat-drying, and the resulting products are essentially dried biosolids in a concentrated and commonly pelletized form. This thermally treated biosolids normally meets the requirement of Class A biosolids defined by USEPA. To implement this thermal treatment, the four time–temperature regimes have been proposed by USEPA (USEPA, 1990). Treatment temperature and time vary with the solid contents. For example, biosolids with 7% solids or greater in the form of small particles can be heated through the contact with either warmed gasses or an immiscible liquid with temperature of 50°C or higher for 15 s or longer (USEPA, 1990).

These heat-dried biosolid products can be applied in any agricultural, horticultural, or domestic application requiring fertilizer use (Goh et al., 2018). A well-established and often cited example is the heat-dried product known as Milorganite® produced in Milwaukee, Wisconsin, USA. Touted as the gold-standard of biosolid fertilizers, Milorganite® was first marketed back in 1926 and as of 2016 had been sold for 90 years. In 2006, the sale of 41,500 dry tones of Milorganite® throughout the United States and Canada resulted in a USD $5.85 million net revenue for the Milwaukee Metropolitan Sewerage District (LeBlanc et al., 2009).

Hydrochars from hydrothermal carbonization of biosolids

Hydrothermal carbonization (HTC) could be a preferable option for biomass with high moisture. It is typically carried out at a temperature of 180–220°C, with pressure in the range of 2.5–5 MPa, and for a residence time of 15–120 min in an inert or oxygen-free environment (Kambo & Dutta, 2014). HTC required less input energy than conventional drying methods for sewage sludge or

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biosolids since the hydrothermal treatment process does not involve evaporation of water (Escala et al., 2013). The other advantages of HTC include complete sterilization and destruction of organic pollutants such as pharmaceutically active compounds in sewage sludge or biosolids (Funke & Ziegler, 2010; Libra et al., 2011).

The renewable products could be produced during HTC. For example, the peat or lignite-like product derived from HTC of sewage sludge or biosolids is called hydrochar or sewchar (Melo et al., 2019). Hydrochars from sewage sludge or biosolids could be used as soil conditioner or ameliorant (Libra et al., 2011), reducing the demand of synthetic inputs (Tasca et al., 2017). Hydrochar application has been shown to reduce the tensile strength, increase the hydraulic conductivity, enhance the soil water-holding capacity, immobilize, and later release the soil nitrogen (Zhang et al., 2014), reducing the leaching potential of this element to groundwater and improve its availability once the crop is present on the field (Bargmann et al., 2014). Nevertheless, the potential of marketing sewchar for UA should be explored.

Biosolid-based construction materials

Biosolids reuse for producing bricks and cement-like materials can be applied in the construction of UA facilities.

Bricks

Brick manufacturers generally incorporate waste into bricks to achieve quarry activities reduction and costs saving (Association, 2009). The incorporation of biosolids into the bricks, possibly blended with other materials (e.g., fly ash, agricultural wastes, and forest wastes), was proposed (Alleman & Berman, 1984). The world's first full-scale sewage brick plant started back in 1991 in Tokyo with a capacity of 5500 bricks a day out of 15,000 kg of incinerated sludge ash. Bricks made with sludge are a resilient material and have not been demonstrated to leach heavy metals (Spinosa, 2004). Bricks containing sludge have been shown to perform better under structural tests than traditional bricks. These tests include compression strength, water absorption rates, abrasion strength, and bending strength (Spinosa, 2004). Sewage bricks have been applied in public walkways and gardens (Spinosa, 2004).

Cement

Biosolids produced in WRRFs contain CaO, SiO_2, Al_2O_3, and Fe_2O_3 which are four major oxides of Portland cement clinker. The Portland cement clinker is a dark gray nodular material made by heating ground limestone and clay at a temperature of about 1400–1500°C, and it can be used as the binder in many cement products (Valderrama et al., 2013). Biosolids can be used in the cement kilns to produce cement clinker through co-firing (Stasta et al., 2006). To achieve this technology, dry matter content in the sludge can reach approximately 92% in industrial practice by decreasing the water content in a sludge drier (Stasta et al., 2006). Manufacturers accept incinerated ash, dried sludge, or dewatered sludge cake as raw materials of cement production. The amount of biosolids that can be added as raw material substitute with controlled ratio ranging from 5 to 15 wt% depending on the organic contents to avoid undesirable changes in the mechanical and rheological properties of pastes and mortars (Collivignarelli et al., 2019; Johnson et al., 2014).

Biosolid fuels for heating in greenhouses

Greenhouses normally utilize direct combustion of natural gas, liquefied petroleum gas, or fuel oil to provide heat for boilers, water heaters, and unit heaters for maintaining the temperatures necessary to grow plants and to increase carbon dioxide (CO_2) levels to induce higher plant growth (Esen & Yuksel, 2013). The high cost of heating a greenhouse with traditional fuel necessitates switching to low-cost, alternative fuels. The composition of digested sludge is about 67% carbon, 5% hydrogen, 25% oxygen, 2.2% nitrogen, and 0.8% sulfur (Stasta et al., 2006). Dried digested sludge has a similar composition and calorific value (12.0–20.0 MJ kg^{-1}) to brown coal (14.6–26.7 MJ kg^{-1}) (Manara & Zabaniotou, 2012; Stasta et al., 2006).

In addition, the bio-oil or biocrude converted from municipal sludge and biosolids via hydrothermal liquefaction (HTL) is another alternative. HTL is the process of heating and pressurizing an aqueous biomass slurry under 200–375°C and 15–22 MPa (2200–3200 psi) (Anthony, 2015). The produced bio-oil contains higher portions of hydrogen and carbon than the initial feedstock, giving it a maximum heating value of 31.46 ± 0.37 MJ kg^{-1} with a conversion yield of $39.42 \pm 1.4\%$ that can be nearly as high as traditional crude oil (Anthony, 2015). Moreover, HTL can mitigate the environmental impacts of sludge and biosolids by sterilizing pathogens, removing almost all heavy metals (Peterson et al., 2008), achieving dechlorination and denitrogenation, removing contaminants of emerging concerns (CECs), and reducing costs associated with anaerobic digestion and the transportation of biosolids (Anthony, 2015).

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Adsorbents for air purification in greenhouses

Biochar can be produced through biosolids pyrolysis (Collivignarelli et al., 2019; Smith et al., 2009). The produced biochar can be used as a low-cost adsorbent for air purification (Hadi et al., 2015). Typical usage examples include the adsorption of volatile organic compounds (Anfruns et al., 2011), NO_x (Pietrzak & Bandosz, 2008), and H_2S (Bandosz & Block, 2006) generated in greenhouse.

Bio-containers as plantable pots

Biosolids with an appropriate mix of additives can yield solid bio-containers that could be marketed based on end use characteristics (Stone, 2017). In this application, transplant is not necessary because the bio-containers can degrade over time. This practice also reduces transplant shock, saves the transplant time and cost, and avoids disposal of the containers (Nambuthiri et al., 2013). This can be advantageous for UA greenhouses and nurseries to begin growing their plants in a controlled environment and then move the plant with container to the garden. Multiple mixes of biosolids, cardboard, cellulose, starch, polymer, and natural glue were developed to provide overall pot stability and structural strength (Stone, 2017). The fiber additive could provide rigidity and help increase marketability as a legitimate commercial product. In all cases, plants grown in bio-containers made by biosolids and additives produced more biomass than plants grown in Peat Pots (Stone, 2017).

Bioplastic production

The potential applications of bioplastics as replacement for petroleum-based plastics are gaining popularity in various fields involving packaging, coating, and construction materials useful for UA. Potential UA application include food wrapping, encapsulation of seeds, encapsulation of fertilizers for slow release, biodegradable plastic films for crop protection, and biodegradable containers for greenhouse facilities (Alcaraz Cercós, 2015). There are current two pathways for bioplastic production from WRRFs, that is, fat, oil, and grease (FOG)-based and polyhydroxyalkanoates (PHAs)-based bioplastic production.

Recent studies showed that the FOG in sewer systems can be recovered for bioplastic production (Michael et al., 2012). For example, the free fatty acids constituent of FOG can be used as a potential substrate for biopolymer synthesis, replacing plastics from petrochemical sources in many applications (Ruiz et al., 2014). Another constituent of FOG, namely, crude glycerol, is also a viable feedstock for value-added conversion into biopolymers or biochemicals (Luo et al., 2016). Cui et al. (2019) comprehensively reported the paths and mechanism of bioplastic synthesis from components of FOG such as fatty acid and crude glycerol.

PHAs are biodegradable polymers produced by many species of naturally occurring bacteria (Arcos-Hernández et al., 2015). Both pure and mixed culture PHA production strategies reduce production costs through the use of low-value carbon-rich raw material feedstocks like municipal and industrial wastewaters (Nikodinovic-Runic et al., 2013). Activated sludge utilized for PHA production is anticipated to be more cost-effectively advantageous than pure culture methods because reactor sterilization is not needed (Yadav et al., 2020). PHA production with municipal wastewater and sludge treatment occurs in four-stage process: (i) removal of readily biodegradable carbon from wastewater coupled to the selective growth of biomass with PHA storage capacity (Collivignarelli et al., 2019), (ii) sludge acidogenic fermentation for the production of a VFA-rich liquid stream, (iii) PHA accumulation from the VFA-rich liquid stream using enriched biomass from step (i), and (iv) PHA recovery and characterization (Morgan-Sagastume et al., 2014). Morgan-Sagastume et al. (2014) reported the first pilot-scale system treating municipal wastewater and producing biomass with enhanced PHA storage capacity.

Phosphorus recovery as a fertilizer for UA

As a finite resource and an essential fertilizer element, the price of phosphorous (P) increased tenfold from 1950 to 2000. In 2007 alone, it grew by about 200% (Tan & Lagerkvist, 2011). For these reasons, interests are drawn towards P recovery from obviously available but often unexploited P sources such as municipal wastewater (Cieślik & Konieczka, 2017; Egle et al., 2016). Technically, P can be removed from municipal wastewater, sewage sludge, and sewage sludge ash from thermal disposal such as incineration.

Currently, the most widely used technologies for P recovery from WRRFs are those based on precipitation of phosphoric minerals from sludge or leachates (Cieślik & Konieczka, 2017). Phosphoric minerals can be precipitated in the form of struvite ($NH_4MgPO_4{\cdot}6H_2O$), hydroxyapatites [$Ca_5(PO_4)_3(OH)$] or calcium phosphates [$Ca_3(PO_4)_2$]. Up to 85% of dissolved P can be recovered from digested supernatant by crystallization or instant

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precipitation (Egle et al., 2016). The most important advantage of this precipitation method is the ability to obtain high-quality phosphoric minerals, which can be applied in agriculture directly. Precipitated hydroxyapatite materials contain very low concentrations of heavy metals and are thus considered to be safe for the environment (Nakakubo et al., 2012). Furthermore, struvite is poorly soluble and represents an eco-friendly mean even with a high load of P (Kataki et al., 2016). Therefore, these high-quality phosphoric minerals can be recovered as excellent fertilizer in service for UA.

Biogas for UA

Sulfur recovery from biogas as a fertilizer

Raw biogas from sewage digesters mainly contains 55% to 65% methane (CH_4), 35% to 45% carbon dioxide (CO_2), and trace amounts of other components such as 0.005% to 2% hydrogen sulfide (H_2S) (Andriani et al., 2014; Rasi et al., 2007). H_2S is harmful to the environment and corrosive to the metallic parts of engines, pumps, compressors, gas storage tanks, and valves and reduces the lifespan of process equipment (Huertas et al., 2011; Ryckebosch et al., 2011). THIOPAQ™ is a biotechnological process capable of absorbing H_2S into a mild alkaline solution followed by the oxidation of the absorbed sulfide to elemental sulfur (S) by naturally occurring microorganisms such as *Thiobacillus* (Center, 2004). The conversion efficiency of H_2S to S is expected to be between 95 and 97% (Center, 2004). The obtained S is an ideal slow release S fertilizer. Although S is unavailable for plant uptake, it is oxidized to the plant available sulfate-S form by soil micro-organisms (Boswell & Friesen, 1993). Therefore, S fertilizer produced from the technologies mentioned in this section can have a huge application potential in UA. Sulfur can be applied alone in the form of dried pellets or particulates of different sizes or can be mixed with superphosphate, bentonite clay, urea, and other fertilizers to enhance fertilizer efficiency (Boswell & Friesen, 1993).

Electricity and heat production from biogas

Wastewater treatment is the eighth largest anthropogenic source of CH_4 emissions (12.8 million metric tons of CO_2 equivalent) in the United States (USEPA, 2014). Therefore, efficient biogas production and utilization at WRRFs can reduce the carbon footprint, minimize external energy requirements, and provide financial benefits for WRRFs (Agency et al., 2007; McCarty et al., 2011). Perhaps the most readily adaptable approach to reduce external energy requirements with existing treatment plants is to make full use of the CH_4 produced from conventional anaerobic digesters through the use of combined heat and power (CHP) systems. CHP is the simultaneous production of electricity and heat from a single fuel source, such as biogas, natural gas, coal, or oil. CHP is not a single technology but an energy system that can be modified depending on the needs of the energy end user. The electric power produced from CHP can offset all or most of a WRRF's power demand, and the surplus can be directly utilized by the equipment or facilities in UA. Likewise, the thermal energy produced by the CHP system can be used to meet digester heat loads and, in some cases, for space heating (Bastian et al., 2011) such as greenhouses and buildings.

Capture of carbon dioxide for utilization in UA

Raw biogas from sewage digesters or fume from CHP contains a large amount of CO_2 which has potential to be utilized in various UA scenarios. For example, gaseous CO_2 can be used for crop growth stimulation and pest control in greenhouses (Byrns et al., 2013; Song, 2006) and for packaging gas of foodstuffs (Byrns et al., 2013, Song, 2006). Liquid CO_2 can be used as cryogenic fluid in chilling or freezing operations or as dry ice for temperature control during the storage and distribution of foodstuffs.

The amount of CO_2 that can be produced and captured from municipal sludge anaerobic digesters is tremendous. Byrns et al. (2013) estimate that the annual biogas production in the United Kingdom is 342 million m^3 which contains 0.27 million tons of CO_2. The global market for merchant CO_2, measured by the amount of CO_2 sold in the market, which excludes in-plant CO_2 utilized by manufacturers, is estimated to be $3.2 billion/year back in 2003 (Gobina, 2004). The CO_2 capture efficiency depends on the separation technologies which can be divided into physical, chemical, or biological methods. Examples include cryogenic, membrane, high-pressure water scrubbing, amine and alkaline absorption, and biofixation (Byrns et al., 2013). These technologies have been reported in detail by Khan et al. (2017) and De Hullu et al. (2008).

Life cycle assessment of the WRRF contribution to UA

Life cycle assessment (LCA) has been performed to evaluate the environmental impacts of using products

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recovered from WRRFs for UA. For irrigation using reclaimed water, LCA conducted by Pasqualino et al. (2011) based on the data from a WRRF located in Spain on the Mediterranean Coast showed that reclaimed water for irrigation with tertiary treatment has a lower environmental impact than the ocean discharge option in terms of all of the categories such as acidification potential, global warming potential, and eutrophication potential. Similarly, the results of LCA conducted by Romeiko (2019) indicated that replacing groundwater with reclaimed water as the irrigation source significantly decreased life cycle global warming, acidification, ozone depletion, smog formation, and respiratory impacts of corn, soybean, and wheat systems in Northern China. The LCA conducted by Canaj et al. (2021) further revealed that tertiary reclaimed water as a supplementary source of irrigation water could reduce the net environmental impact by 23.8% due to lower consumption of irrigation water (−50%), electricity (−27.7%), and chemical fertilizers (−22.6%) compared with conventional groundwater irrigation system in Italy.

For using biosolids as solid amendment, LCA conducted by Peters and Rowley (2009) shows that the reuse of biosolids products can be environmentally beneficial but transportation distances can change the preferences between technologies, and drying biosolids using biogas produced in situ in WRRFs rather than petrochemical methane could significantly improve environmental performance. Also, biosolids for kiln application such as bricks or cement production perform better than landfill, farmland application, composting, lime amending, and heat-drying (Peters & Rowley, 2009). LCA conducted by Sablayrolles et al. (2010) indicated that composted biosolids are more beneficial to the environment than heat-dried biosolids in terms of resource depletion, acidification, eutrophication, greenhouse effect, summer smog, and ecotoxicity. LCA conducted by McDevitt et al. (2013) revealed that direct farmland application of biosolids has the least environmental impact than landfill, composting, and the amendment with biochar, while biosolid composting has the most environmental impact. The results of the LCA by Mohajerani et al. (2018) further confirmed that the incorporation of biosolids into bricks is environmentally favorable and is a promising alternative approach with respect to most of the environmental impacts except water depletion, which is mainly due to the higher water demand of biosolids-amended bricks during the shaping process.

For using biosolid as fuels, Mills et al. (2014) found that creating a solid fuel with postanaerobic digestion biosolids to displace coal was the most sustainable solution economically and environmentally in terms of LCA results. By converting raw sewage sludge to refinery-ready biocrude oil with HTL technology nationwide, almost 4.5 million barrels of upgraded biocrude oil could be produced per year, while about 330,000 metric tons of CO_2 equivalent GHG emissions could be offset annually (Bond, 2015). For adsorbents such as biochar production, Miller-Robbie et al. (2015) reported that the addition of biosolids biochar production adds little to the overall energy use but provides substantial (26%) reduction in GHG emissions for the national case in terms of LCA results, largely due to the recalcitrant carbon storage in biochar. In contrast, Thompson et al. (2016) reported that biosolids biochar adsorbent negatively affect environmental quality at the highest level than wood biochar and coal-based powdered activated carbon, attributing to energy consumption for biosolids drying, manufacture of mineral fertilizer to substitute biosolids applied for soil amendment, and the need for supplemental adsorbent.

For bioplastic production, Vogli et al. (2020) compared the LCA results of PHA-based biopolymers from anaerobically digested sewage sludge with other polymers. It can be noted that the PHA-based biopolymers tend to have both lower nonbiogenic GHG emissions and lower fossil energy demand values than the fossil-based polymers (Vogli et al., 2020). For P recovery, according to the LCA conducted by Bradford-Hartke et al. (2015), recovering P using struvite precipitation resulted in positive environmental impacts due to energy and chemical use being offset by operational savings and avoided mineral fertilizer production. In contrast, for Pradel and Aissani (2019), the overall assessment remains unfavorable for P recovery comparing with mineral P fertilizer use due to the low P yields, low P concentration in the sludge, and the large amounts of energy and reactants needed to recover the P. For sulfur recovery from biogas, LCA by Cano et al. (2018) indicates that biological H_2S scrubbing technologies are much more favorable than physical–chemical technologies in most environmental impact categories such as terrestrial acidification, freshwater eutrophication, human toxicity, water depletion, and photo-oxidant formation and, as expected, are more favorable than the nontreatment scenario.

For electricity and heat production from biogas, LCA by Singh et al. (2020) shows that biogas plant has negative GHG emissions ($-0.2385\,\text{kg}\ CO_2\ \text{eq/m}^3$) compared with coal-based electricity plant which indicates that sewage sludge-based biogas plant has beneficial impacts on the environment. For CO_2 capture, Starr et al. (2012) evaluated the LCA of different technologies. The results show that alkaline absorption with regeneration technology had an 84% negative impact in all LCA categories than high-pressure water scrubbing largely due to the energy-intensive production of the alkaline reactants (Starr et al., 2012). For alkaline absorption with

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regeneration technology, it was determined that using NaOH instead of KOH improves its environmental performance by 34% (Starr et al., 2012).

Economic viability in leveraging WRRF for UA

Reclaimed water for irrigation has been widely accepted in developed countries (Organization, 2006). For example, in Monterey County, California, more than 5000 ha of raw-eaten food crops has been irrigated with reclaimed water for more than a decade (EPA, 2012). Lv et al. (2021) developed a method to quantitatively analyze eco-economic benefits of urban reclaimed water for irrigation. The results show that the cost of using reclaimed water to irrigate urban green spaces was 46% of the cost of using tap water (Lv et al., 2021). The total benefit of using reclaimed water to irrigate green spaces was three times the benefit of using tap water (Lv et al., 2021). Yet, the cost of moving reclaimed water between centralized WRRFs and UA sites was not reported.

Limed and heat-dried biosolids have been commercialized. For example, in Australia, limed biosolids blended with red clay have yielded a product known as Lime-amended BioClay (LaBC®) (Stamatelatou & Tsagarakis, 2015). A heat-dried biosolid product known as Milorganite® produced in Milwaukee, Wisconsin, USA, has also been commercialized in North America. Hydrochars from HTC of biosolids have been commercialized by enterprises in several developed countries such as AVA-CO2 Schweiz AG in Switzerland and TerraNova Energy in Germany. Class A/EQ biosolids, amended, and composted biosolids have also been widely accepted in developed countries as fertilizers (Darvodelsky & Bridle, 2012; Johnston et al., 2019). Similarly, biosolid-amended bricks and cement have also been commercialized (Collivignarelli et al., 2019; Johnson et al., 2014; Spinosa, 2004).

Technological economics analysis (TEA) performed by Bond (2015) showed that economic impacts of biosolid-based bio-oil or biocrude production through a HTL project have the potential to be favorable. In the case of implementation at a WRRF, this project could save the City of Austin on the scale of $32 million (Bond, 2015). This technology has been commercialized by several utilities and companies. For example, Genifuel®, a company in Utah conducted biosolid HTL based on systems developed by the Pacific Northwest National Laboratory (PNNL), part of the US Department of Energy, which implements reactions at 350°C and 200 bars, with residence times of approximately 45 min (Castello et al., 2018; Genifuel, 2020). The whole system is even smaller than an anaerobic digester, which can convert biosolids with 20% total solids into biocrude oil and sterile water (Castello et al., 2018; Genifuel, 2020). Biosolid-based adsorbents, bio-containers, and bioplastic production are still in lab or pilot scales, and the possibility of their commercialization is still under investigation. For example, a biosolid-based bioplastic production pilot plant has been operated in Sweden by Veolia subsidiary and project partner AnoxKaldnes (Bioplastic, 2015). While its current production capacity is small, for example, a few kilos per week, the idea is to scale this up to include the total treated wastewater volume and ultimately resulting in a production capacity of 2000 metric tons/year (Bioplastic, 2015).

Currently, about 50 technologies have been developed to recover P from byproducts of WRRFs (e.g., digester supernatant, sewage sludge, sewage sludge ash) including crystallization, precipitation, wet chemical processes, and thermochemical processes (Egle et al., 2016). Many crystallization and precipitation processes have been commercialized including PhoStrip, PRISA, DHV Crystalactor, CSIR, Kurita, Ostara, Phosnix, Berliner Verfahren, and FIX-Phos (Mehta et al., 2015). For example, in 2016, Ostara and the Metropolitan Water Reclamation District of Greater Chicago opened the world's largest nutrient recovery facility with its Waste Activated Sludge Stripping to Remove Internal Phosphorus (WASSTRIP®) and Pearl® technologies (Ostara, 2017). The technologies offer many benefits including a reduction in sludge production, up to 50% P removal, and a revenue stream with the sale of Crystal Green®, a slow release P fertilizer. As of 2017, the Ostara technology had 15 commercial installations located in the United States, Canada, the Netherlands, Spain, and the United Kingdom (Ostara, 2017). For P recovery as struvite from digested sludge, Life Cycle Cost Analysis (LCCA) performed by Saerens et al. (2021) indicates that an overall positive economic result is observed in most scenarios as a result of the operational benefits (mainly from the improved dewaterability of the sludge) although struvite is currently undervalued.

After upgrading, biomethane was utilized for electricity and heat production through grid injection. All examined upgrading technologies (membrane separation, water scrubbing, chemical absorption with amine solvent, and pressure swing adsorption) have demonstrated substantial mitigation of the overall environmental and economic impacts (Ardolino et al., 2021). Membrane separation provides the best performances in terms of LCCA. Technology readiness level (TRL) for the production of biomethane by means of biogas upgrading reached a value equivalent to the market availability, which is the maximum level of development of a technology

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(Ardolino et al., 2021). For sulfur recovery from biogas, THIOPAQ™ and the Shell-Paques processes commercialized by Shell and Paques can remove up to 99% H_2S from biogas and convert it to elemental sulfur (Center, 2004). The system application range is approximately 200 pounds to 40 tons of sulfur per day (Center, 2004). However, the commercialization of carbon capture from biogas has rarely been reported.

CURRENT UNDERSTANDING OF THE HUMAN HEALTH CONCERNS

The human health concerns of utilizing reclaimed water for irrigation and biosolids for land application have to do with the presence of CECs and trace metals. CECs comprise a diverse collection of chemicals, including pharmaceutical, personal care products (PCPs), surfactants such as Per- and polyfluoroalkyl substances (PFASs), polycyclic aromatic hydrocarbons (PAHs), flame retardants, metallic trace elements, pesticides, and endocrine disruptors (Daughton & Ternes, 1999; Samal et al., 2022). Majority of CECs used domestically are conveyed daily to WRRFs (Samal et al., 2022). The average wastewater retention time is generally shorter than the degradation half-lives of many organic CECs that enter a WRRF (Halling-Sørensen et al., 1998; Tchobanoglous et al., 2003), resulting in their discharge via effluents and biosolids. Reported levels of CECs and their degradation products in WRRF effluents ranged from ng/L to μg/L (Deblonde et al., 2011; Focazio et al., 2008; Loos et al., 2013; Pal et al., 2010) and in biosolids ranged from ng/kg to mg/kg (Boix et al., 2016; Clarke & Smith, 2011; Daughton & Ternes, 1999; Halling-Sørensen et al., 1998; Heidler & Halden, 2008; Kinney et al., 2006; Semblante et al., 2015; Tchobanoglous et al., 2003; USEPA, 2009, 2021a; Xia et al., 2005).

To date, there is no regulatory or recommended concentration limits for CECs in both WRRF effluent and biosolids (Samal et al., 2022). Land application of biosolids and WRRF effluent for agricultural production has been a widely accepted practice in the United States and many other countries. The human health impacts of CECs associated with land application of biosolids and reclaimed water are considered to be de minimis according to Kumar et al. (2017). In addition, some studies have shown that although some CECs can be taken up by plants (Al-Farsi et al., 2017; Liu et al., 2020; Miller et al., 2016; Reyes et al., 2021), human health impacts due to consumption of food grown in biosolids- or reclaimed water-applied fields are considered to be negligible comparing with the daily household exposure to CECs (Hundal et al., 2011; Kumar et al., 2017).

In contrast to CECs, the health impacts of trace metals are much better understood (Briffa et al., 2020; He et al., 2005; Tchounwou et al., 2012). The USEPA Part 503 Biosolids Rule (USEPA, 2018) has regulatory limits on a list of 10 trace metals for biosolids (USEPA, 2021b) and requires that biosolids should be used in accordance with approved management practices including operational standards, monitoring, recordkeeping, and reporting. Biosolids for farmland application, soil amendment, and bio-containers are not allowed if the concentration of any of the 10 trace metals exceeds its celling concentration limit. The USEPA's guidelines for reclaimed water use for irrigation have recommended concentration limits for 17 trace elements (EPA, 2012). On the other hand, no health concerns were reported for the biosolid-based construction materials, fuels and adsorbents, recovered P fertilizer, and biogas utilization. As for bioplastics, its health concerns are similar to petroleum-based plastics, which have been documented elsewhere (Bernard, 2014; Momani, 2009).

FUTURE DIRECTIONS AND OUTLOOKS

Utilizing resources recovered from municipal wastewater for UA is still a controversial topic. On one hand, wastewater represents a new source of water, energy, nutrients, and renewable materials and thus should no longer be considered as waste but as a new resource to be handled in a circular economy-type loop. On the other hand, the use of recycled materials for UA may have an adverse impact on public health and the environment, depending on treatment and regulatory expectation. Current scientific knowledge in agronomic and environmental sciences, as well as in the economic and social sciences, can be integrated and used to lower the associated risk through the effective management and design of WRRFs. The following questions should be addressed in future works: (i) How to increase public acceptance of UA using WRRF materials for safe environmental reuse within an adapted risk assessment framework? (ii) What socioeconomic models can render this integrated approach sustainable? (iii) What WRRF treatment systems can be used to ensure public health? and (iv) What economical methods should be developed to transport resources between centralized municipal WRRFs and decentralized UA sites.

AUTHOR CONTRIBUTIONS

Jiefu Wang: Conceptualization; data curation; investigation. **Yuepeng Sun**: Conceptualization. **Xia Kang**: Conceptualization. **Allison Deines:** Conceptualization.

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Ross Cooper: Conceptualization. **Karen Pallansch:** Conceptualization; project administration; resources. **Zhiwu Wang:** Conceptualization; data curation; investigation; methodology; project administration; resources; supervision.

DATA AVAILABILITY STATEMENT

The data that support the findings of this study are available from the corresponding author upon reasonable request.

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