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The third route: A techno-economic evaluation of extreme water and wastewater decentralization

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ABSTRACT

Water systems need to become more locally robust and sustainable in view of increased population demands and supply uncertainties. Decentralized treatment is often assumed to have the potential to improve the technical, environmental, and economic performance of current technologies. The techno-economic feasibility of implementing independent building-scale decentralized systems combining rainwater harvesting, potable water production, and wastewater treatment and recycling was assessed for six main types of buildings ranging from single-family dwellings to high-rise buildings. Five different treatment layouts were evaluated under five different climatic conditions for each type of building. The layouts considered varying levels of source separation (i.e., black, grey, yellow, brown, and combined wastewater) using the corresponding toilet types (vacuum, urine-diverting, and conventional) and the appropriate pipes and pumping requirements. Our results indicate that the proposed layouts could satisfy 100% of the water demand for the three smallest buildings in all but the aridest climate conditions. For the three larger buildings, rainwater would offset annual water needs by approximately 74 to 100%. A comprehensive economic analysis considering CapEx and OpEx indicated that the cost of installing on-site water harvesting and recycling systems would increase the overall construction cost of multi-family buildings by around 6% and single-family dwellings by about 12%, with relatively low space requirements. For buildings or combined water systems with more than 300 people, the estimated total price of on-site water provision (including harvesting, treatment, recycling, and monitoring) ranged from \$1.5/m^3 to \$2.7/m,^{3} which is considerably less than the typical tariffs collected by utilities in the United States and Western Europe. Where buildings can avoid the need to connect to centralized supplies for potable water and sewage disposal, water costs could be even lower. Urine-diversion has the potential to yield the least expensive solution but is the least well developed and had higher uncertainty in the cost analysis. More mature layouts (e.g., membrane bioreactors) exhibited less cost uncertainty and were economically competitive. Our analysis indicates that existing technologies can be used to create economically viable systems that greatly reduce demands on centralized utilities and, under some conditions, eliminate the need for centralized water supply or sewage collection.

1. Introduction

Freshwater resources face unprecedented pressure due to population growth, climate change, and poor management (Arora et al., 2015; Damania et al., 2017; Doell et al., 2009; Sedlak, 2014). The historic reliance on a linear approach to water infrastructure, imported water,

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centralized water distribution, energy-intensive treatment, and disposal to surface waters is being questioned because of its environmental and economic shortcoming as well as its lack of resilience (Daigger, 2009; Vázquez-Rowe et al., 2017). Approaches integrating source-separation and resource recovery, together with radical improvements in water use efficiency and energy-positive concepts, have been suggested as alternatives that could improve economic and environmental sustainability, reliability, and resilience of urban water systems (Garrido-Baserba et al., 2018; Larsen et al., 2016; McCarty et al., 2011; Piratla and Goverdhanam, 2015; Remy, 2010; Roefs et al., 2016).

Despite the attestations from proponents of alternative approaches that conventional urban water infrastructures are becoming obsolete and that the benefits from a paradigm shift in water and wastewater infrastructure would outweigh potential risks, change has been slow in coming. In part, the reluctance to change can be attributed to the challenge of technology lock-in—an inability to transition to better practices due to institutional inertia and a lack of willingness to take risks on unproven approaches (Larsen et al., 2016; Tchobanoglous et al., 2004; van Loosdrecht and Brdjanovic, 2014). However, even in places where lock-in is less of a problem (and alternatives have inherent advantages), such as rapidly urbanizing countries or in parts of cities where urban infill or suburban expansion is putting strain on urban water systems, alternatives have been slow to develop (Gikas and Tchobanoglous, 2009; Setegn and Donoso, 2015). The slow progress in alternative approaches is largely due to regulatory impediments as well as the challenges of operating less conventional systems, adding risk, and slowing down their implementation. Furthermore, there is a historic correlation between project size and transaction costs for project management that favours to larger projects (Haaskjold et al., 2021). Rabaey et al. (2020) provided a comprehensive discussion on existing bottlenecks towards decentralization, including confidence in emerging technologies, the sunk cost of existing infrastructure, and other key aspects. Nevertheless, increasing recognition of the potential advantages of alternative approaches is leading to major efforts from both public and private institutions (NYC_WRR, 2022; OCWD, 2022; TokyoWRC, 2022; USWaterAlliance, 2022). The widespread uptake of reuse initiatives and on-site water systems, and successful long-term operation prove that past impediments are starting to disappear. At the same time, cheaper and more reliable sensors and actuators are enhancing technology performance, while the costs of key technologies such as MBR, RO, and other types of equipment continue to drop (SustainableWater, 2022; WaterTech, 2022). Similarly, with new technological advances in connectivity and remote control, decentralized initiatives are no longer limited to decentralized management. Multiple facilities and operations can be remotely managed in a centralized fashion (OpsCTRL, 2022).

Among the emerging paradigms for urban water systems, extreme decentralization—the practice of integrating building-scale water recycling within cities with existing centralized water systems—is attractive because it is compatible with existing water governance because it provides a means for cities facing water stress to adjust to water stress without abandoning existing infrastructure or making large investments to rapidly transition away from existing approaches. Rabaey et al. (2020) assessed the viability of implementing a household-scale water system that relied on rainwater capture and greywater reuse as an alternative to expansion of existing centralized water infrastructure or trucked water. For ease of adoption for a single household, the authors considered a relatively simple process train consisting of a membrane bioreactor (MBR) and ultraviolet (UV) disinfection unit for greywater treatment and a reverse osmosis (RO) system followed by a second UV unit for production of drinking water. Under conditions typical of most locations, the system had an initial payback time of around 10 years. In addition to costs savings and independence from the drinking water network, the household-scale water system could lead to the emergence of new features, such as the possibility of adjusting water quality for specific uses (e.g., softened water for cleaning), improved aesthetics (e.g., adjusting ion composition to suit personal taste) or, even, accessing health benefits through the addition of supplements to drinking water.

Acknowledging the potential benefits of decentralized urban water systems (Larsen et al., 2013; Li et al., 2021; Rabaey et al., 2020; Singh et al., 2015; Sun et al., 2020. Sun et al., 2020; Yang et al., 2021), here we have extended the analysis by providing a detailed economic analysis of systems that are likely to be considered at the scale of individual homes and multi-family dwellings using standardized cost assessment methods. To account for more complex types of distributed wastewater treatment technologies (e.g., anaerobic and membrane technologies that might be viable in multi-family buildings) as well as resource recovery approaches that require source-separation, we have included a range of new and emerging technologies under different housing scenarios.

The technical and economic feasibility of five personalized treatment trains (Table 1) has been evaluated in dwellings ranging from a 2.3 to 300 people equivalent through (i) a detailed economic analysis considering capital and operating expenditures and revenues for the in-house sewer system, pumps, and treatment modules; (ii) an estimation of the required space for tanks and technology; (iii) an assessment of the final water quality. The overall costs of implementing the systems in already existing buildings are presented, as well as the fraction of the total construction investment they represent for new buildings. This multidisciplinary effort illustrating the current state-of-the-practice and potential benefits of decentralization is meant to provide a rigorous baseline for further development of sustainable urban water systems, and it does not represent a conclusive case on extreme decentralisation. Rather, it is likely that new technologies and better controls will lead to other, more efficient configurations.

2. Methodology

2.1. Research approach

2.1.1. Housing types

To evaluate the technical and economic feasibility of implementing decentralized urban water facilities, we considered six housing types (i.e., single-family homes, low-rise dwellings, low/medium-rise, medium-rise, medium/high-rise, and high-rise buildings) representative of residential housing in many cities (Fig. 1). An occupancy rate typical of Europe (i.e., 2.3 inhabitants per dwelling) was assumed, with an average apartment floor space per capita of 30 m^2 (Negro and Economidou, 2014; Roefs et al., 2016). The assumed area per story (i.e., the area available for rooftop rainwater collection) representing the total floor space (i.e., including the space for staircases, lifts, hallways) was estimated for typical European buildings through Eq. 1, where A_s and A_{hh} are the areas per story and household, respectively [m^2], γ is a constant (equal to 1.15), and N_{hh} is the number of households in the designed building. The extra surface assigned to the average area destined to common spaces is that of 30 m^2 per story.

$$A_s = \gamma^* A_{hh}^* N_{hh} + 30 \tag{1}$$

Table 1
Summary of the proposed scenarios.

Scenario	Toilet type	Stream types	Treatment train
V1	Vacuum	Black Grey (+ treated black)	UASB + OLAND reactor + struvite reactor MBR + RO + UV
V2	Vacuum	Black Grey (+ treated black)	OLAND reactor + struvite reactor anMBR + RO + UV
C1	Conventional	Black Grey (+ treated black)	OLAND reactor + struvite reactor MBR + RO + UV
C2	Conventional	Black Grey (+ treated black)	OLAND reactor + struvite reactor anMBR + RO + UV
UD	Urine-diverting	Yellow Grey + Brown	electrochemical cell + MABR anMBR + RO + UV

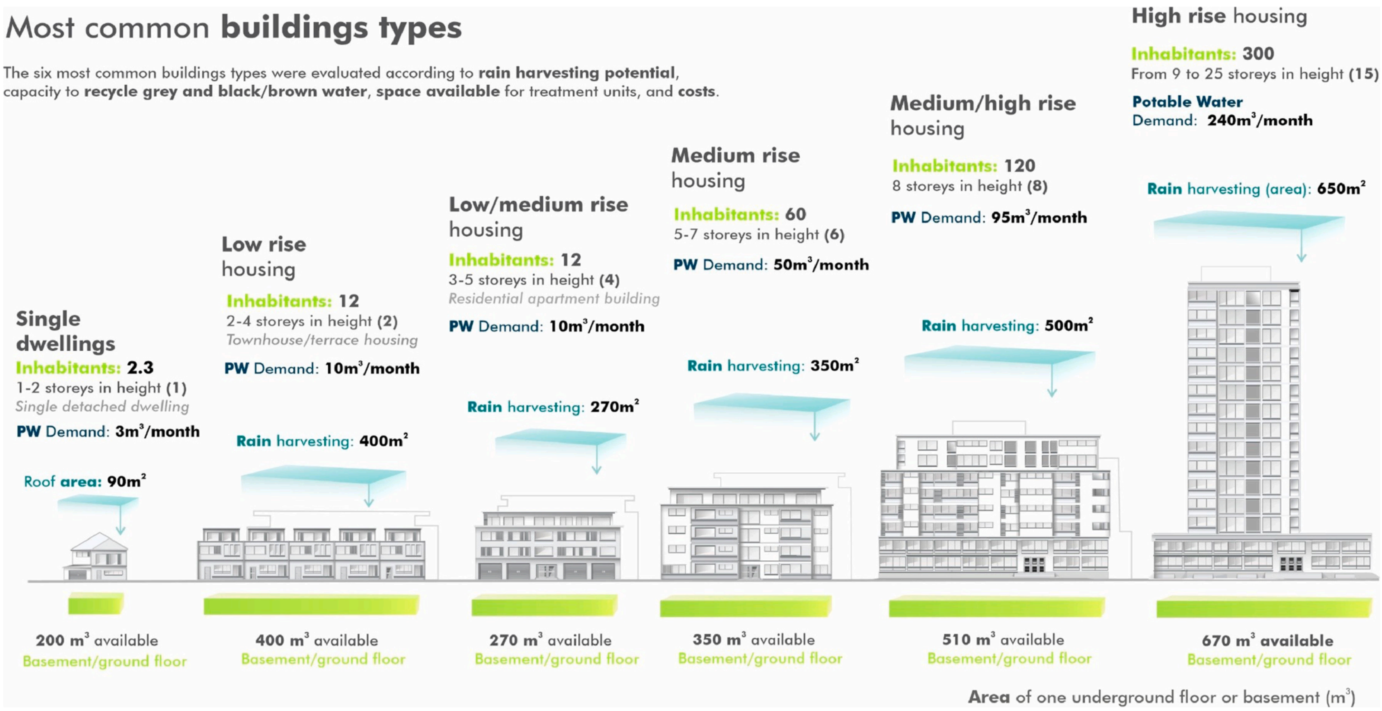

Fig. 1. Characterization of the most common buildings according to their number of inhabitants (using an average of 2.3 habitants per household), number of stories (used number of floors in this study), total potable water demand (estimating 26/l per person pe r day); available roof area that could be used for rain harvesting (90% of available roof area), and total space (as volume) of a typical basement or underground floor for each type of building.

2.1.2. Treatment layouts

Five treatment layouts were compared involving the use of vacuum, urine-diverting, and conventional toilets (Table 1). The first two layouts employed vacuum toilets (V1 and V2), while the second two scenarios involved a more conservative approach using conventional toilets (C1 and C2). The last scenario considered the potential benefits of source-separation of urine (UD). Thus, the use of toilets diverting urine within its corresponding treatment train was also assessed. **Error! Reference source not found.** summarizes the principal characteristics of each of the proposed scenarios.

Two main conceptual approaches based on the different wastewater qualities were evaluated for the five source-separated approaches (Fig. 2). The main treatment processes employed for black water and grey water, functioning in series (Fig. 2a). This concept was implemented in four of the five source-separated approaches (V1, V2, C1, and C2). The only exception is the urine-diverting scenario (UD), where the black water stream was divided in a yellow and a brown water subsystem instead (Fig. 2b).

In both scenarios, harvested rainwater is used to supply potable water, while recycled water is used for home appliances, toilet flushing, and showering. Biological anaerobic treatment was included in all cases except for one scenario, which included conventional toilets and aerobic wastewater treatment (C1). Anaerobic treatment results in the conversion of organic matter into biogas, while struvite reactors and the electrochemical cell in the urine source-separation scenario enable the recovery of phosphate. Nitrogen that is not incorporated into struvite is eliminated through the anammox process. This approach based on the biological removal of nitrogen was chosen over ammonia recovery by stripping because it has proven to be more cost-effective (Garrido-Baserba et al., 2018). In the urine-diverted (UD) system, nitrogen was converted into nitrate because nitrification occurs in the membrane aerated biofilm reactor (MABR). The liquid solution obtained in this process can be used as fertilizer. In all scenarios, the fraction of the treated wastewater that is not recycled is suitable for landscape irrigation. For the purpose of this analysis, the treatment systems discharge excess water to the centralized sewer system as needed. Future permutations of this system could use the treated wastewater for irrigation during seasons when there is demand and discharge to surface or groundwater at other times of the year. Tanks and required equipment for each treatment train were accordingly seized (see supplementary information 4-5), and the basement of each building was assumed as the location for the water storage and treatment modules.

2.1.3. Water usage

Water use per PE was derived from a reported global indoor average of 108 L*day^{-1}. The per capita potable water demand for potable uses exclusively (i.e., drinking, kitchen sink, and bathroom) was 26.5 L*day^{-1} in all scenarios (Rabaey et al., 2020). To assess the sensitivity of this assumption, the performance of the rainwater capture system was also assessed by assuming a lower potable water demand of 15 L*person^{-1}*day^{-1} (WHO, 2011). In terms of wastewater production, the per capita volume of grey- or blackwater depended on the type of toilets employed in the dwellings. 40 L*day^{-1}, 15 L*day^{-1}, or 5 L* day^{-1} of black water were considered depending on the use of conventional, urine-diverting, or vacuum toilets, respectively (De Graaff et al., 2010; Larsen et al., 2013).

Despite having higher capital and operational costs, vacuum toilets may offer economic and environmental benefits because less water is required and more concentrated blackwater is obtained. Their use is coupled with the treatment of the collected wastewater both in: a) two separated biological treatment stages (anaerobic and aerobic treatment in a UASB and an MBR); or b) a single anaerobic stage (in an anMBR).

The feasibility of a configuration involving the use of conventional toilets, which yield more diluted blackwater, was also assessed. The additional flushing water would decrease constituents loads, which would allow the biological treatment system to achieve the required BOD and total phosphorus objectives without the anaerobic stage of the MBR. However, vacuum-based systems coupled with nutrient recovery typically yield the lowest energy footprint (Kjerstadius et al., 2015).

Finally, urine source-separation was considered because yellow

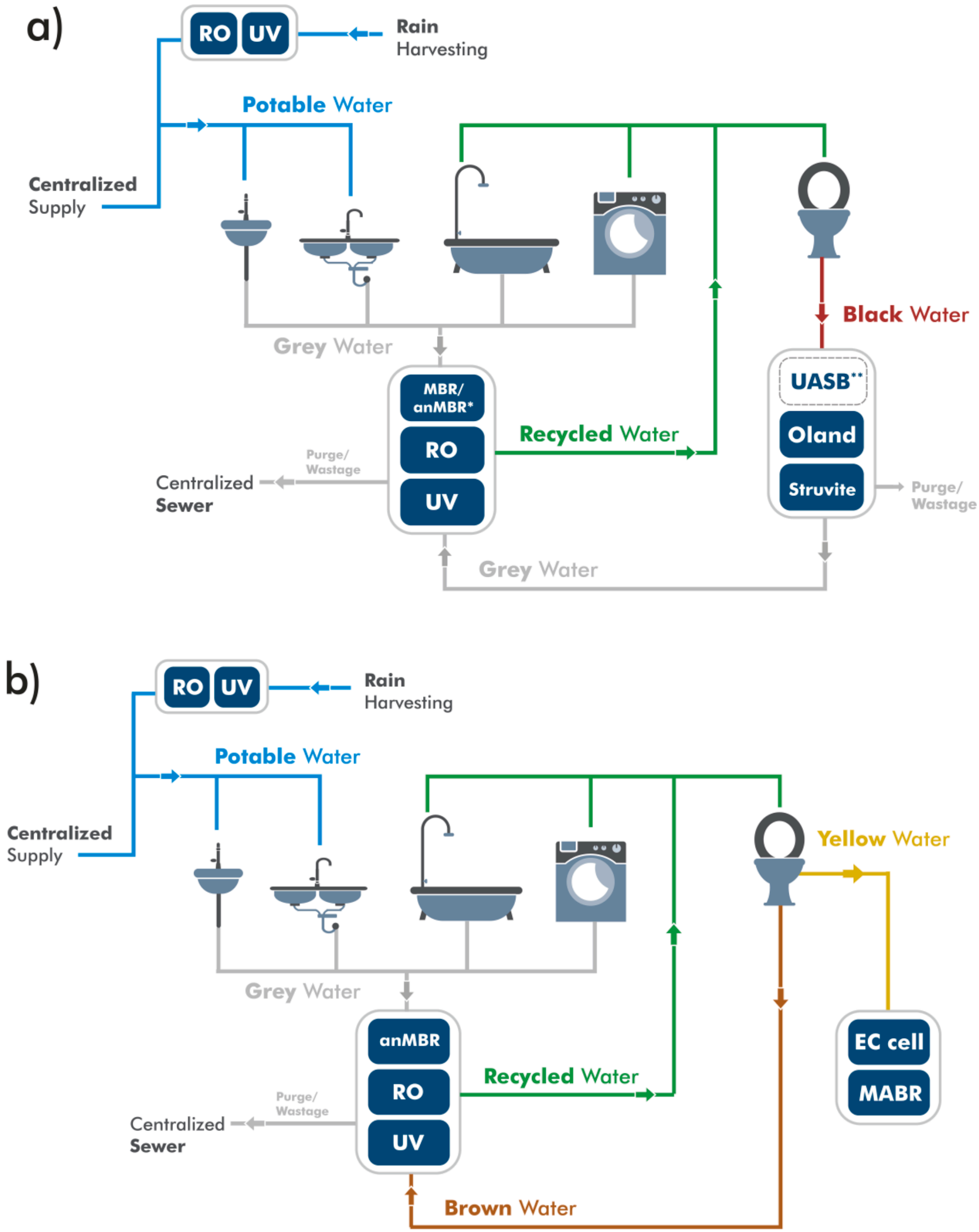

Fig. 2. Flow diagrams of a fully decentralized approach for both black and grey water streams. The upper flow diagram (a) corresponds to vacuum and conventional toilets while (b) corresponds to the urine-diverting layout. *Indicates that MBR will be used for scenarios V1 and C1 while anMBR will be considered in V2 and C2. **No UASB will be considered in those scenarios including anMBR (V2 and C2).

water contributes 80% of N and 50% of P to residential wastewater (Jimenez et al., 2015; Rossi et al., 2009). The collection and recovery of nutrients from urine could provide a revenue stream and reduce energy use in the treatment system (De Paepe et al., 2018; Larsen et al., 2015; Lienert and Larsen, 2010, 2010b; Maurer et al., 2006; Remy, 2010). Existing technologies for on-site urine treatment are still not sufficiently mature to yield reliable results. However, we estimated the costs of an innovative treatment scheme involving the electrochemical treatment of urine coupled with nitrification in an MABR to determine the merits of further development of on-site urine treatment and resource recovery systems.

2.1.4. Influent and effluent composition

Constituents concentrations in grey and black water are shown in Table 2 and are calculated using average daily constituent load per capita, as described by Larsen et al. (2013). The results are in concordance with Kujawa-Roeleveld & Zeeman (2006); Dhadwal (2020); Dhadwal et al. (2021); Sun et al. (2020). Urea hydrolysis (i.e., 90% of total nitrogen converting to ammonia) is assumed in all scenarios except for UD, hence the high total ammonia concentration in BW (Gao et al., 2019; Larsen et al., 2013).

Influent variability is assumed to be low as the collected wastewater originates solely from residential dwellings. However, deviations in influent concentrations stemming from seasonal or specific events may occur.

For the UD scenario, we considered that BW consists of a mixture of yellow and brown waters (i.e., urine and faeces plus toilet paper, respectively). In the modeled scenario, urine and faeces were assumed to be collected separately using urine-diverting toilets (Laufen, Switzerland), which use 3L per simple flush. A urine production of 1.5 L/p*d and a recovery efficiency in households of 75% of the flow considering technological and performance limitations to complete recovery were assumed (Lienert and Larsen, 2010; Rossi et al., 2009). All non-recovered urine was assumed to mix with brown water; thus, the latter's concentration has been calculated accordingly. The per capita daily volume of brown water produced is assumed to be 15 L/p*d.

Table 3 shows the concentration of the resulting flow, yellow and non-mixed brown water.

Effluent concentrations and performance were obtained based on reported removal efficiencies (supplementary information, tables 4-25). Each of the treatment layouts was also modelled using the software SIMBA# (ifak, 2021). The simulations were used to evaluate both the performance and feasibility of the process flow diagrams. The effluent characteristics obtained from both the modelling approach and the reported-based calculations (supplementary information, tables 4-25) were compared to ensure the correctness of the calculations. Fig. 3 shows the model of the V1 layout to treat black and grey water elaborated using SIMBA#.

2.2. Sewer infrastructure

The required pipe lengths were calculated using the Urban Water Infrastructure Model (UWIM) for sewers (Maurer et al., 2013). Pipes were assumed to originate at the center of every housing ground floor story. Lengths were calculated based on housing density and settlement area, the latter of which was adapted to the floor space of each building. The model parameter (f_2) representing the housing shape factor was estimated for each scenario (Maurer et al., 2013). Sewer methodology and calculation are detailed in supplementary information section 2.

Greywater and rainwater pipes (made of unreinforced concrete) were assumed for gravity water collection. In contrast, blackwater pipes were assumed to be operated either by vacuum (as required in the V1 and V2 scenarios) or by gravity (C1, C2, and UD), and were assumed to be composed of HDPE. Costs associated with vacuum pumps and pumps needed to return treated water to the apartments were included in the economic analysis (see supplementary information section 6). A constant vacuum system (CVS) was included for blackwater pumping from vacuum toilets because it has proven to be more cost-effective than a vacuum on-demand system (VOD), especially when pump prices and capacities are compared in multi-family dwellings (Dometic, Sweden; Jest Vacuum AS, Norway). A double-pipeline was assumed for black water in the urine diversion scenario.

Table 2
Influent composition of Blackwater (BW) and Greywater (GW) (Larsen et al., 2013; Kujawa-Roeleveld & Zeeman, 2006; Dhadwal, 2020; Dhadwal et al., 2021; Sun et al., 2020; Gao et al., 2019).

	BW (mg/L)	GW (mg/L)
COD	10500	472
BOD	3560	175
TN	2000	8
NH4-N	1800	3
N-NO_3	0.2	6
TP	260	5
TSS	8360	175
VSS	6690	64

Table 3
Composition of undiluted fresh urine, brown water (incl. faeces, toilet paper, and water), and the mixture of brown water and non-recovered urine (Kirchmann and Pettersson, 1994; Larsen et al., 2013; Lindeboom et al., 2020; Maurer et al., 2006; Remy, 2010).

mg/L	YW (1.5 L/p*d)	BrW (15 L/p*d)	BrW + non-recovered YW
COD	10,400	2,490	2,660
BOD	3,870	800	869
TN	8,800	80	278
NH4-N	463	72	80.9
TP	800	34	51.5
TSS	0.0	2,790	2,720
VSS	0.0	2,230	2,180

2.3. Treatment technologies

2.3.1. Organic matter removal and gas production: upflow anaerobic sludge blanket reactor (UASB)

The UASB reactors considered in scenario V1 enabled the uncoupling of the solids retention time (SRT) from the hydraulic retention time (HRT). The system was assured to operate with an HRT of 7.15 days and a SRT of 30 days. The common practice in full-scale anaerobic digestion facilities is to convert a fraction of the produced biogas into heat to satisfy heat requirements (while producing energy) via gas microturbines or combined heat and power (CHP). We adopted this approach because it also has been employed in small-scale applications (Adami et al., 2020; Baccioli et al., 2018; Zhang et al., 2021). One of the challenges when operating UASBs and anMBRs is the high percentage (up to 50%) of methane gas dissolved in the effluent that could escape, which could lower energy recovery efficiencies and increase greenhouse gas (GHG) emissions (Cookney et al., 2016; Shin et al., 2014; Shoener et al., 2016; Velasco et al., 2018a). Thus, a degassing membrane contactor was included in the treatment train, with a separated reactor separating the gas and liquid phases (Henares et al., 2016; Velasco et al., 2018b). UASB design calculations, removal efficiencies, and costs can be found in supplementary information section 4.1.

Depending on scenario conditions (e.g., location, space availability, budget constraints, legislation), different biogas management options are possible (Rodero et al., 2018; Salihu and Alam, 2015; Yentekakis and Goula, 2017). Arguably the most common solution in biogas-producing treatment plants involves upgrading (i.e., purifying) the biogas to achieving 99% methane purity and selling it as fuel for electricity production or transport applications (gas or liquified). Both natural gas and CO_2 have a value market that could allow additional revenues to offset some of the system operating costs.

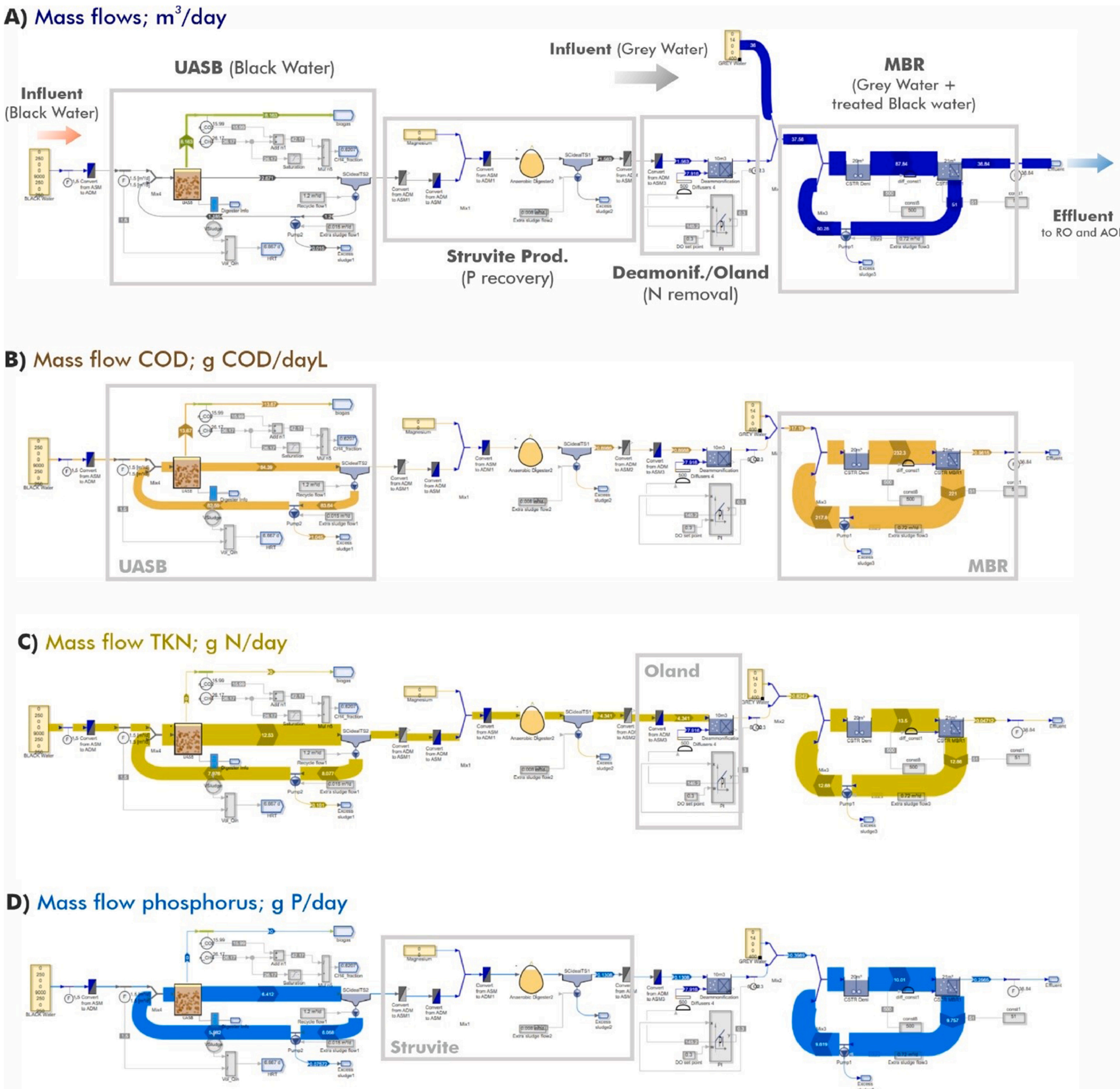

Fig. 3. Main constituents mass flows of one of the five treatment layouts (V1) with its corresponding Sankey diagrams obtained by the modelling software SIMBA#. The sanky diagrams represent a) volumetric mass, m3/day (blue); b) COD mass flow, g COD/day (brown); c) TKN mass flow, g N /day (green); and d) Phosphorus mass flow, g P/day (light blue). The five models corresponding to the five scenarios under consideration can be found at section 9 of the supplementary information.

2.3.2. Organic matter removal and gas production: anaerobic membrane bioreactors (anMBRs)

Alternative configurations were considered in which anMBRs were employed. These systems provide the advantages of anaerobic wastewater treatment (e.g., production of biogas, lower energy consumption, and less excess sludge production) while overcoming the technology's main challenge of inferior organic carbon removal (Foglia et al., 2020; Guo et al., 2016; Li et al., 2020; Lin et al., 2013; Muñoz Sierra et al., 2019). The combination of membrane technology and anaerobic treatment allows for the decoupling of HRT from SRT, which helps lower overall costs while continuing to provide adequate treatment with shorter HRTs (Ariunbaatar et al., 2021; Foglia et al., 2020; Ribera-Pi et al., 2020; van den Berg et al., 2020). anMBRs modeled in this study were sized according to their HRT. The low-strength wastewater (i.e., greaywater) could be successfully treated using anMBR technology (Gouveia et al., 2015; Martin Garcia et al., 2013; Shin et al., 2014; Wang et al., 2018b, 2018a). It must to be noted, that as anMBRs achieve a high solids separation, the clogging of the pores is not expected to be major concern (Membrana, Wuppertal, Germany); microporous contactors designed for low flow rates (instead than non-porous membranes) appear to be the best approaches for higher methane recovery yields.

2.3.3. Organic matter removal: aerobic membrane bioreatcors (MBRs)

A flat sheet reverse osmosis (RO) membrane (DuPont de Nemours, Inc., Delaware, USA) was assumed as a means of meeting current guidelines for phosphorus concentrations in wastewater reuse. The selected membrane offered the highest removal efficiency for pollutants (including NH_4-N & P) relative to similar models (Van Voorthuizen et al., 2005). The use of the RO membrane improved the final water quality and provided another barrier against pathogen exposure in showers, baths, or incidental contact with recycled water.

The MBRs were assumed to use a rejection side-stream configuration and included a combination of biological treatment in an aeration tank and ultrafiltration in a crossflow, multi-tube membrane loop (e.g., polyethersulphone membranes; Berghoff GmbH, Eningen, Germany) prior to RO. The hydraulic residence time in the reactors was assumed to be 15.8 h for scenario V1 and 40 h for scenario C1. The volume of the system was calculated using Eq. 57 in the Supplementary material (Fletcher et al., 2007; Wen et al., 1999; Xing et al., 2001).

2.3.4. Nitrogen removal: oxygen limited autotrophic nitritation/ denitrification (OLAND) process

The removal of nitrogen was assumed to take place in a single stage OLAND process, suitable for wastewater with high loads of organic matter in decentralized, small-scale treatment systems (Larsen et al., 2013; Lv et al., 2011; Nhu Hien et al., 2017; Windey et al., 2005). The OLAND process removal efficiencies and the calculation for the required biofilm surface is provided in supplementary information section 4.2. Nitrate production during the OLAND process was assumed to be negligible.

2.3.5. Phosphorus removal: Struvite recovery

Small-scale phosphorus recovery in the form of struvite, which would be accompanied by ammoniacal nitrogen removal, was assumed to take place in a crystallizer followed by a decanter (Ali, 2005; Kataki et al., 2016; Shaddel et al., 2020). Removal efficiencies, design calculations, energy demand, and the mass of added Mg(OH)2(s) needed to maintain supersaturation at a pH of 8 were calculated to obtain the effluent concentrations and operational costs (see Supplementary Information §4.3). Phosphorus recovery through struvite recovery is likely to become more important to sustainability in the water sector; however, this approach still presents operational challenges, even in large facilities. Its implementation may only take place as operators gain more experience and pilot-scale treatment systems demonstrate reliabel system perfornance. However, even if the struvite recovery only addresses the prevention of capacity loss due to detrimental struvite precipitation inside reactors and piping, significant infrastructural sustainability goals would still be achieved.

2.3.6. Urine treatment

As part of the urine diversion scenario, yellow water was assumed to be treated through a combination of two technologies. First, we assumed that the urea in the urine will be allowed to hydrolyze, resulting in the production of ammonia and bicarbonate. As a result of the pH increase associated with this process, the precipitation of struvite and other minerals (e.g., Mg, Ca, K, and Na salts; Maurer et al., 2006) occurs, increasing the risk of clogging pipes and equipment. Odor nuisance, ammonia loss through volatilization are other challenges associated with urea hydrolysis (Christiaens et al., 2017; De Paepe et al., 2020; De Paepe et al., 2020b Maurer et al., 2006; Rossi et al., 2009). To avoid the need to add chemicals (i.e., caustics, acids) to inhibit microbial activity and thus prevent urea hydrolysis, an electrochemical cell similar to the unit proposed by De Paepe et al. (2020) was included (see supplementary information 4.7).

2.3.7. Storage tanks

Tanks used to store recycled water or potable water were assumed to be composed of FPP (flexible polypropylene) liners, whereas tanks used for storage of black, grey, brown water or sewage were assumed to use PVC (polyvinyl chloride) liners. Smaller tanks were assumed for rainwater collection (since smaller volumes are required) and larger steel tanks with the indicated liners were employed for grey and black water management (Enduramaxx Ltd, Lincolnshire, United Kingdom; Power Plastics Ltd, North Yorkshire, United Kingdom). A rotary screen was included in the design before the bioreactor to reduce membrane fouling and clogging. Seizing and costing calculation can be found in supplementary information 4.11.

2.3.8. Rainwater treatment

Rainwater treatment employed a combination of pre-filters, such as downspout diverters and first flush filters (RainHarvestSystems.LLC, 2021), in-tank filtration, RO followed by UV treatment combination. The rainwater disinfection and treatment systems were selected from among the smallest models available. Further details can be found in supplementary information 4.8.

2.4. Rainwater capture

The total demand of drinking water (m^3/month) for potable uses (i.e., drinking and cooking) for each type of building was calculated considering a per capita water consumption of 26.5L/day (Rabaey et al., 2020) multiplied by the number of building inhabitants for 31 days per month. The daily amount of 26.5L represents an approximate 30% of the overall water usage (80 L/day) and does not consider other uses such as flushing toilets (25%), cleaning (20%), washing clothes (15%), gardening, etc. To estimate the supply of rainwater (m^3/month), average monthly rainfall (mm) data was extracted from Climate Data Org (2021). The authors acknowledge that rainfall regimes and overall extreme variability are being impacted by climate change (IPCC, 2022, IPCC, 2021; Slater et al., 2021; UNEP, 2021) and that estimates based on historic averages or return periods might be inaccurate in the coming decades. To assess the potential of rainwater systems to complement the existing access to a centralized water supply, it would be more appropriate to consider a particularly dry period (e.g., the driest year in a 20- or 50-year period). Nevertheless, according to the latest IPCC report, this approach could be very short-lived as extreme variability is expected to change drastically in the upcoming decades (IPCC, 2022). Future studies on rainwater capture capacity should include these most recent forecasts. Furthermore, other options for storing precipitation generated by shorter, more intense storms (e.g., managed aquifer recharge) should be considered.. For this analysis, five cities were selected to cover a variety of climates according to the Köppen-Geiger climate classification, which considers temperature and precipitation patterns: i) Miami (North America), which exhibits an equatorial climate; ii) Santiago de Chile (South America), which was representative of an arid climate; iii) Barcelona (Europe), which has Mediterranean climate with a dry summer; iv) Hong Kong (Asia), which is a warm climate with dry winters; and v) Toronto (North America) which is a representative temperate climate zone. Precipitation patterns (i.e., average monthly values) are included for each case study in the SI section [*X – need to transfer the excel spreadsheet to SI*].

Rainfall values were multiplied by the roof surface of each dwelling type (m^2), and a correction factor for roof losses of 0.9 was included. Drinking water demands and rainwater accumulations were estimated for each type of building and location. Rainwater accumulated in the storage tanks was carried over to between months.

2.5. Economic analysis

Information from technology manufacturers was used to calculate the capital (CapEx) and operational (OpEx) costs of the water and wastewater treatments systems for all building types and scenarios (Courtens et al., 2014; Etter et al., 2011; Fletcher et al., 2007; Lo et al., 2015; Lozano et al., 2007). Values obtained from the cost-estimation simulator CapdetWorks were employed for the assessment of the UASB reactor. Benefits from the recovery of nutrients and biogas were estimated and included in the economic analysis (Deng and Hägg, 2010; Etter et al., 2011).

$$OPEX = \sum_{t=1}^{T} \frac{OPEX_t}{(1+r)^t} \tag{2}$$

$$I = \sum_{t=1}^{T} \frac{I_t}{(1+r)^t} \tag{3}$$

$$Total\ Cost\ (TC) = \sum_{t=1}^{T} \frac{OPEX_t}{(1+r)^t} + CAPEX \tag{4}$$

OpEx and incomes (I) were calculated considering an interest rate (r) of 5% and a time horizon (T) of 30 years (Roefs et al., 2016). Eq. 2 and Eq. 3 yield the total discounted lifetime *OpEx* and *I*, where $OpEx_t$ and I_t are the costs at time *t*. Total costs (TC) shown in Fig. 5 were calculated according to Eq. 4.

3. Results and discussion

3.1. Rainwater capture and domestic water recovery potential

The extent to which rainwater can satisfy potable water demand depends on location and rainfall patterns (Fig. 4). For single-family dwellings and the two smallest multi-family dwellings, rainwater alone could supply potable water for all locations other than the driest location (i.e., Santiago de Chile). For the taller buildings, rainwater was only adequate for several months per year. Buildings in Hong Kong, Miami, and Toronto could rely upon harvested rainwater for almost half of the year.

Although significant savings can be obtained from rainwater harvesting for potable water supply, the most significant reductions in water consumption were attributable to the continuous recycling of the black and grey water. Wastewater recycling satisfied the water demand of toilet flushing, home appliances, and showering. Fig. 4 illustrates the recovery capacity per building type and indicates the volumes of grey and black water recycled by the treatment systems. On an annual basis, a minimum of about 74% of the total water demand (including potable water) can be satisfied in the worst scenarios (high-rise buildings without enough rain). Smaller buildings and buildings located in climates with rainfall regimes that are more conducive to harvesting will be able to fully satisfy an even greater fraction of the total water demand through the combination of rainwater harvesting and wastewater recycling in most of the scenarios, because potable water only accounts for about 17-20% of overall household water use (EPA, 2021).

3.1.1. Rainwater treatment scheme

The use of rainwater as a secure source of potable water is a proven concept that has found worldwide acceptance (Campisano et al., 2017; Liu et al., 2021; Peterson, 2016; Smit, 2019). Different treatment schemes can meet the strict health standards in wealthy countries (Alim et al., 2020; Piemontese et al., 2020; Suleiman et al., 2020).

In this study, a RO/UV system (with customizable remineralization) was used to treat rainwater (Dirisu et al., 2019; Rabaey et al., 2020). This configuration was chosen for its reliable performance and its potential to satisfy health guidelines. The ability of RO to remove nearly all

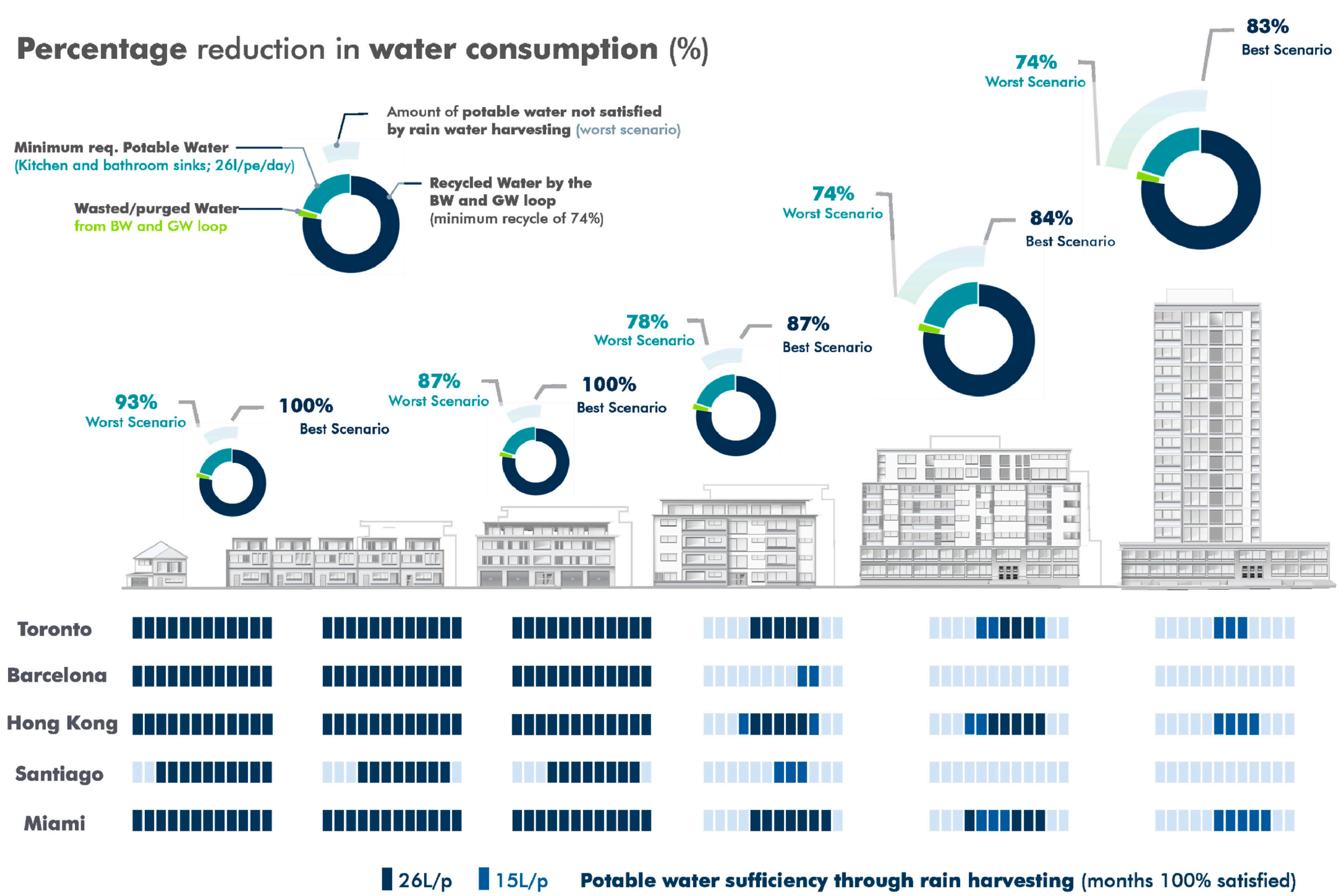

Fig. 4. Water recovery capacity per type of building, showing the amount of grey and black water recycled by the treatment loop, the percentage of water purged, and the performance of the rainwater capture installation. The latter is depicted using coloured bars where each fraction represents a month of the year (from January to December), when demand can either be satisfied or not depending on the location and total monthly rainfall (V1)

constituents of rainwater might necessitate remineralization with ions such as magnesium, calcium, and fluoride to protect health and improve taste (Naser et al., 2017; Sedlak, 2019). Many of the RO commercially available today already include remineralization filters (e.g., Waterdropfilter, 2021).

Bottled water obtained through RO-based filtration has steadily grown in the last decades (Chen et al., 2021; Eke et al., 2020; Hawkins, 2017; SustainableWater, 2022). An increasing portion of the population prefers bottled water over tap due to its security perception, taste (Qian, 2018; Wu et al., 2021), and, likely, good marketing. Therefore, existing and popular RO filtration schemes ensuring the recommended level of minerals and appealing taste could be easily adopted to ensure public acceptance.

3.1.2. Recycled water scheme (from black and grey water loops)

The proposed treatment trains for the recovery of the generated wastewater produced an effluent that meets the guidelines of US EPA (EPA/600/R-12/618), WHO (GDWQ, 2011 & 2006), and EU regulation 2020/741 for reuse in urban and bathing applications. The summary of the effluent quality obtained for each layout can be found in supplementary information section 10.

In terms of water volumes, bathing applications (i.e., showers and baths) are the most critical uses of recycled water in this study. Recycling domestic water for drinking purposes is still in its infancy but already serving millions daily. Mass-scale implementations include examples such as Singapore, Orange County (CA), Windhoek (Namibia), Wichita Falls (TX), and Altamonte Springs (FL), among others (OCWD, 2021; Van Rensburg, 2016; Tortajada and Nambiar, 2019; WRF, 2019; Watereuse, 2019). Recent years have also seen the appearance of specialized companies satisfying the private demand for domestic recycled water for drinking purposes (Tangent, 2021), together with a wide plethora of research studies (Kehrein et al., 2021, 2020; Khan, 2013; WRF, 2016; WRRF, 2015; Wu and Englehardt, 2016) stating domestic recycling as a reliable alternative. The use of recycled water for toilet flushing, cooling towers, landscaping, and industrial purposes is widely accepted (Amaris et al., 2020; Ilemobade et al., 2013), especially among leading technology companies (Microsoft, 2017; Salesforce, 2021). Although potable water reuse is still not widely accepted in many countries, using recycled water in showers and baths is likely to engender less hesitancy and was therefore considered in this study. Without this application, the demand for the rainwater tanks would have been considerably greater, and the system would have been less attractive. If recycled water is not considered acceptable, other alternatives for reducing demand on the rainwater tanks (e.g., recirculating showers) could be considered.

The treatment train employed in our analysis includes RO followed by disinfection with an ultraviolet (UV) lamp after MBR treatment. This approach is predicted to satisfy drinking water requirements, including an ammonia concentration lower than 1.5 mg/L and a COD of $<$10 mg/L (Agrawal, 2009; Hespanhol and Prost, 1994; WHO, 2011). This MBR-RO-UV combination of technologies has been proven to eliminate bacteria and viruses (Friedler and Gilboa, 2010; Ghernaot et al., 2019; Tang et al., 2018).

One of the most significant risks to public acceptance of showering or bathing in recycled water involves undesirable smells. Previous research indicates that some odorous compounds can pass through RO systems when treating sewage streams (Agus et al., 2011), suggesting that complementary treatments such as advanced oxidation processes (AOPs) may be necessary to ensure public adoption. Different authors have already demonstrated the efficacy of AOP for potable reuse (Barazesh et al., 2015; Weng et al., 2020; Wu and Englehardt, 2016).

3.2. Space requirements

3.2.1. Holding capacity

The space requirements of on-site water systems depend on the desired degree of autonomy of the building. The sizing of the tanks in our analysis was designed according to the desired duration (or capacity) for holding the produced wastewater and treated water before its reuse or treatment, respectively. Similarly, the sizing of the solids holding tanks was also designed according to the desired duration for storing the produced sludge.

Two main degrees of autonomy for the recycling systems were evaluated. In the first approach, half of a week of autonomy was evaluated. In this approach, equalization, conditioning, and storage tanks were sized to hold wastewater for 3.5 days. Solids holding tanks were sized to allow for the accumulation of a week's worth of sludge production. In a second approach, all equalization, conditioning, and storage tanks water tanks were sized to hold the wastewater and sludge for only over 24 hours.

After sizing all the required units and equipment for the two degrees of autonomy, the theoretical space requirements as a percentage of the basement floor were calculated (Table 5, Fig. 5). A basement floor with a ceiling height of 3 m was assumed as the location of the treatment units and related equipment, plus other building equipment that is normally put in the basement level (e.g., electrical equipment, HVAC, elevators).

Results of our analysis (Fig. 5) indicate that the water storage and recycling equipment will occupy approximately between 8 and 40% of the basement (or a further underground floor) space under the lowest degree of autonomy scenario. The estimated area ranges from 15 to 120% of the basement area under the larger storage capacity scenario (Table with all space requirements per building and layout can be found see supplementary information section 11). The single-family dwelling and the two largest apartment buildings sacrificed the greatest fraction of the basement space to the treatment systems. The highest floor occupancy corresponds to HR buildings and is 91.4% and 28.7% for longer and shorter water holding periods. The remaining basement space could be reserved for future plant changes or destined for other uses, such as vehicle parking or additional storage. Further details can be found in supplementary information; Sizing section 4.1 to 4.9; space requirement discussion 11.

3.3. Economic assessment

3.3.1. Alternatives economic comparison

Total costs (CapEx and OpEx) were estimated for the six types of buildings (Fig. 5). Among the five treatment trains, the V1 scenario (i.e., vacuum toilets combined with anaerobic UASB and aerobic MBR) was the most expensive, costing about 50% more than each of the other scenarios. The V1 option provides a smaller footprint (due to the small volume of wastewater from vacuum toilets) and incorporates the most proven technologies, providing the most reliable data on sizing, costing, and performance. Conversely, anMBRs allows for considerable cost reductions, both in terms of CapEx and OpEx (V2, C1, and C2). The urine-diverting configuration achieves the lowest OpEx but shows higher CapEx than all other configurations except for V1. The novelty of the treatments involved in the UD scenario causes uncertainty in the cost calculation of this layout, which could be easily reduced with further development (i.e., MEC and MABR), positioning the UD scenario as the layout with the highest potential in terms of cost-effectiveness and resource recovery.

3.3.2. The economics of extreme decentralization

The total costs of the six types of buildings were also evaluated in terms of $/m^3 and $/per capita (Fig. 6). The economies of scale show a significant reduction in the cost of water as the number of people serviced increases (Fig. 6). Fig. 6 also illustrates how the total costs of the five treatment trains for each of six types of buildings would compare with current water prices in different regions of the world. These cost estimates do not consider the avoided costs benefits from both rainwater harvesting and recycling water (74-100%), as discussed in section 3.36 - Income Assessment.

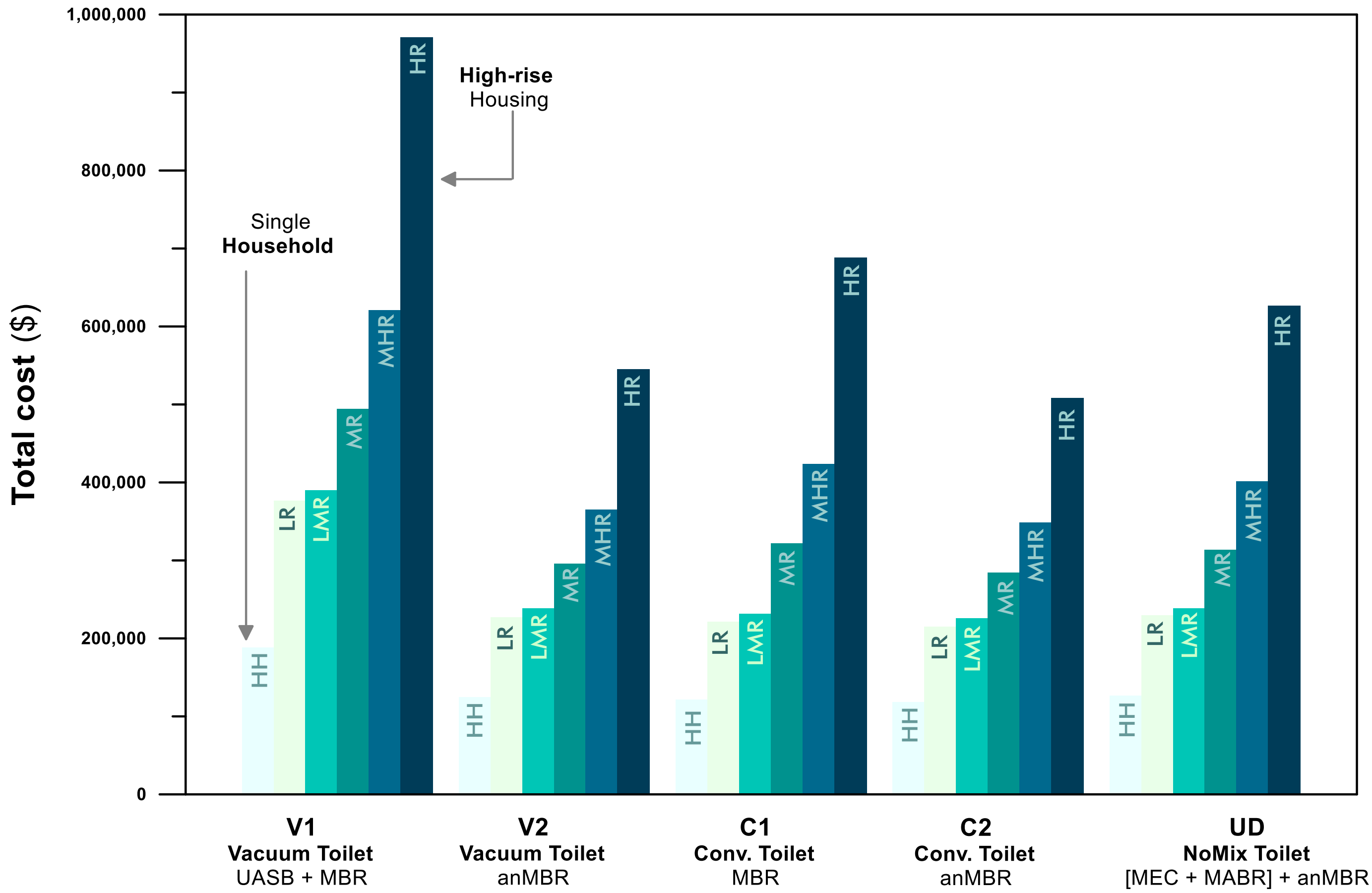

Fig. 5. Total costs, including monitoring and costs per capita for the five scenarios and building types.

Estimates shown in Fig. 6 illustrate how the most expensive of the presented decentralized systems (V1) in buildings with more than 100 inhabitants could be competitive with the cost of water in wealthier regions, such as North America and Western Europe (4.3-3.9$/m^3). Similarly, for buildings larger than 300 inhabitants, such as large buildings, districts, or city blocks, with populations between 300 to 2,000, could become the most cost-effective decentralized solutions.

This study indicates that large buildings (or interconnected buildings) could result in the most cost-effective approach when considering current technologies, monitoring requirements, and infrastructure costs (i.e., pipelines), especially when compared with existing centralized systems. Nevertheless, decentralized district-scale approaches with smaller buildings (resulting in lower population densities) may not achieve similar economic advantages. In previous studies estimating the costs of decentralized approaches covering neighborhoods or districts, pipeline-related costs (i.e., installation, maintenance and pumping) accounted for more than 60% of total costs (Garrido-Baserba et al., 2018; Roefs et al., 2016). The present study, which was focused on independent apartment blocks, did not consider interconnected sewer networks between neighboring buildings, which limited the length of the pipeline and reduced its impact on the overall CapEX and OpEX. Future studies should identify the most economically feasible density ratios for different district organizations, such as large buildings, interconnected buildings in city blocks (with short pipeline connections and modular treatment designs), or smaller housing at district-scale.

3.3.3. The cost of including decentralized water systems in new building construction

The total cost of the most expensive decentralized solution (V1) was compared to the total construction costs for each type of building under study (Fig. 7). Fig. 7 shows the percentage of the total construction costs for scenario V1; however, the relationships will be similar for the other scenarios.

The installation cost could increase construction costs by 6-9%. The OpEx over a 30-year period was equivalent to 3-24% of construction costs. See supplementary information section 5 - construction costs.

3.3.4. Cost breakdown

The most expensive among the five treatment trains, the V1 scenario (i.e., vacuum toilets combined with anaerobic UASB and aerobic MBR), was used to illustrate the cost breakdown of decentralized treatment in buildings. An extended discussion covering all the main cost contributors explaining the main differences between treatment trains (e.g., sewer pipes, dual reticulation-piping, specialized toilets, treatment technologies) can be found in supplementary information section 12. Similarly, the breakdown of costs (and obtained benefits) for all treatment configurations can be found in the supplementary material (see section 7 and section 8).

Fig. 8 shows that the recycling of grey water and black water accounts for about 65-80% of the overall cost, with a higher percentage of the costs in larger buildings. Within the treatment systems, the CapEx and OpEx costs are of a similar magnitude, partially due to the necessity of vacuum pumping to evacuate the highly concentrated waters and the high costs of operating the UASB reactors. Nevertheless, the overall deployment of the grey water sewer pipeline is more costly than black water sewer since pipe diameters are larger to hold bigger volumes.

The remainder of the costs were related to monitoring and rainwater harvesting. Monitoring costs were high, accounting for the majority of the remaining costs (Fig. 9). See supplementary for monitoring break down costs and discussion (Section 13 - Process Monitoring). As discussed, a conservative approach with the highest standards on monitoring was assumed in this analysis. Monitoring is a crucial and

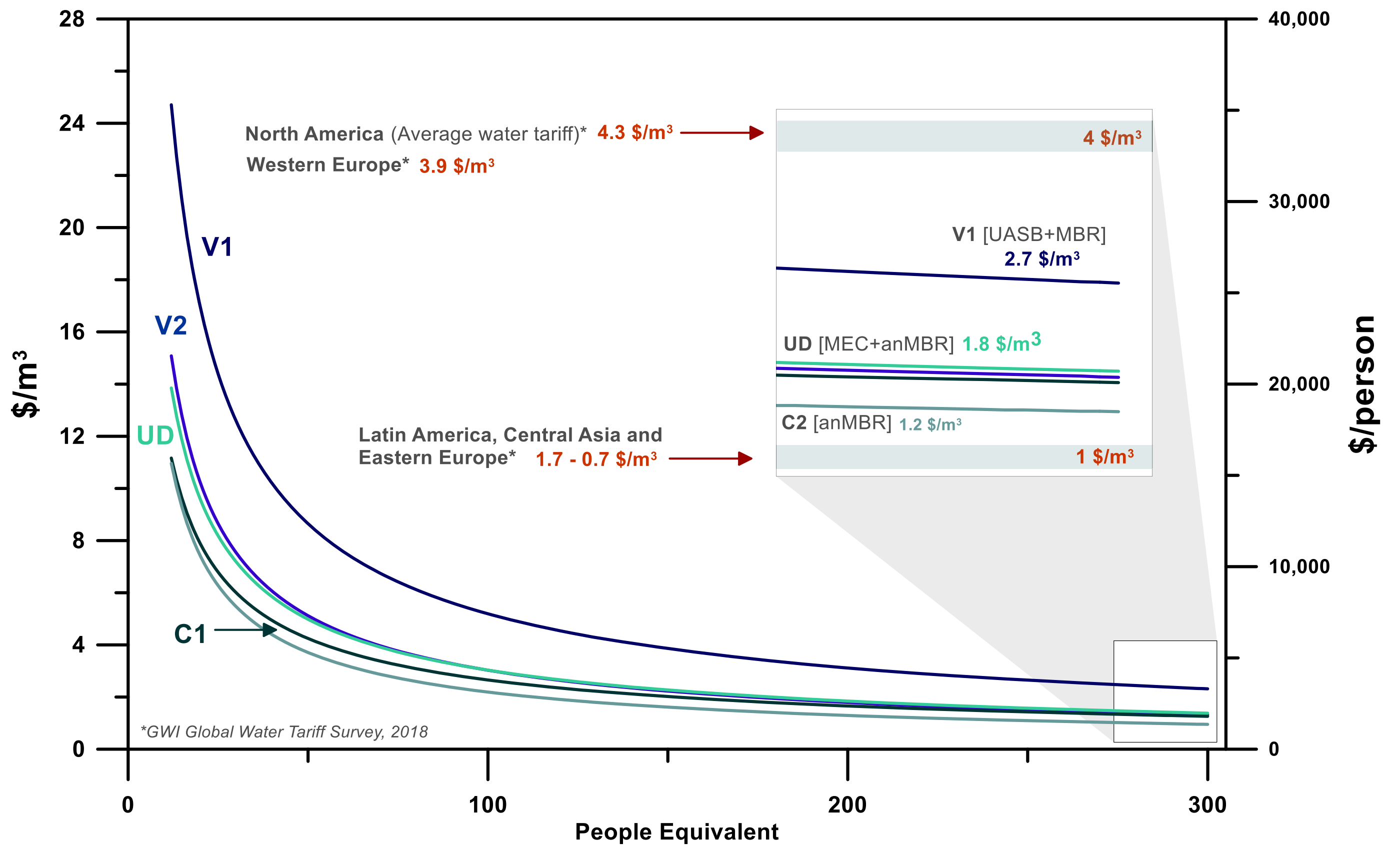

Fig. 6. Summary of the treated water price ($\$/m^3$) and cost per person for the five different alternatives (V1, V2, C1, C2, and UD) depending on the number of inhabitants (and without considering water or sewer connection savings). A closer look at the high-rise housing Building (i.e., 300 people) shows that the water price could be potentially lower than many regional averages (i.e., North America and Western Europe). The right axis corresponds to $/person in a 30 years horizon.

expensive process (sensors, sampling, control, LIMS, personnel, etc.) and is standardized to assure health and safety, therefore similar requirements were deemed necessary independently of the size. Therefore, no significant economies of scale were attributable to monitoring; and the monitoring costs for larger buildings (14%) increased substantially for smaller buildings (50%). Research is needed to identify approaches for reducing monitoring costs, especially in smaller systems, through the use of new or soft sensors, or performance-based monitoring strategies.

Rainwater collection and treatment accounts for a relatively small fraction of the overall cost (5-8% across all buildings and scenarios), entailing low capital investment and low operating and maintenance costs. RO and UV treatment consume large amounts of energy (Judd, 2017; Sanz, 2013), but the volume of rainwater requiring treatment was small, which limited the OpEx of the rainwater system to 1-2% of total costs for all scenarios.

3.3.5. Process monitoring

To ensure public acceptance, the highest standards on monitoring need to be satisfied. Water reuse should satisfy the most stringent of the quality requirements, demanding an advanced, robust, and safe treatment system able to comply with the tighter health and monitoring standards.

To avoid a negative perception towards this type of approach, the presented system must yield the highest water qualities (such as potable water), even if none of the reused water will be recycled as a direct potable source. A very conservative estimation of the monitoring requirements was purposely chosen to demonstrate that the most rigorous and advanced monitoring routine could be achieved, avoiding concerns about safety and control. The monitoring costs were calculated as in three main blocks: 1) Laboratory tests (outside lab); 2) Sensors CapEx and OpEx (external service contract); 3) Real-time remote monitoring (external service contract). (See section 8 in supplementary information)

The real-time remote monitoring was selected to enable remote monitoring of these decentralized systems, allowing the necessary technical and operational expertise to operate and maintain in real-time such complex systems. A centralized management of these different decentralized units was considered to allow experts to control and coordinate with technicians.

The laboratory test routine designed follows the guidelines from the sampling workbook "Sampling Questionnaire and Matrix" developed by the Water Environment Federation (WEF) task group on Wastewater Treatment Plant Data Collection. The workbook indicates the required sampling locations within the treatment process and the type and frequency of analyses required to meet the process control goals (including the requirements for process simulators). The compiled laboratory costs were obtained from several commercial laboratories, and costs should be updated and adapted accordingly. It must be noted that the cost of using outside laboratories is the most expensive option, and it was selected to reflect the worst economic scenario and to assume no need for further investment or space requirements.

The operating budget could be significantly reduced by analyzing in situ some of them and, especially, decreasing the number of lab tests from the recommended sampling routine once a stable performance is obtained. In parallel, some service contracts could be reduced depending on process performance, maintenance routine, etc.

Nevertheless, we anticipate monitoring costs to further decrease in upcoming years with cheaper and better sensors capabilities, new development of reliable soft sensors, and more breakthroughs in control theory (Vanrolleghem and Lee, 2003).

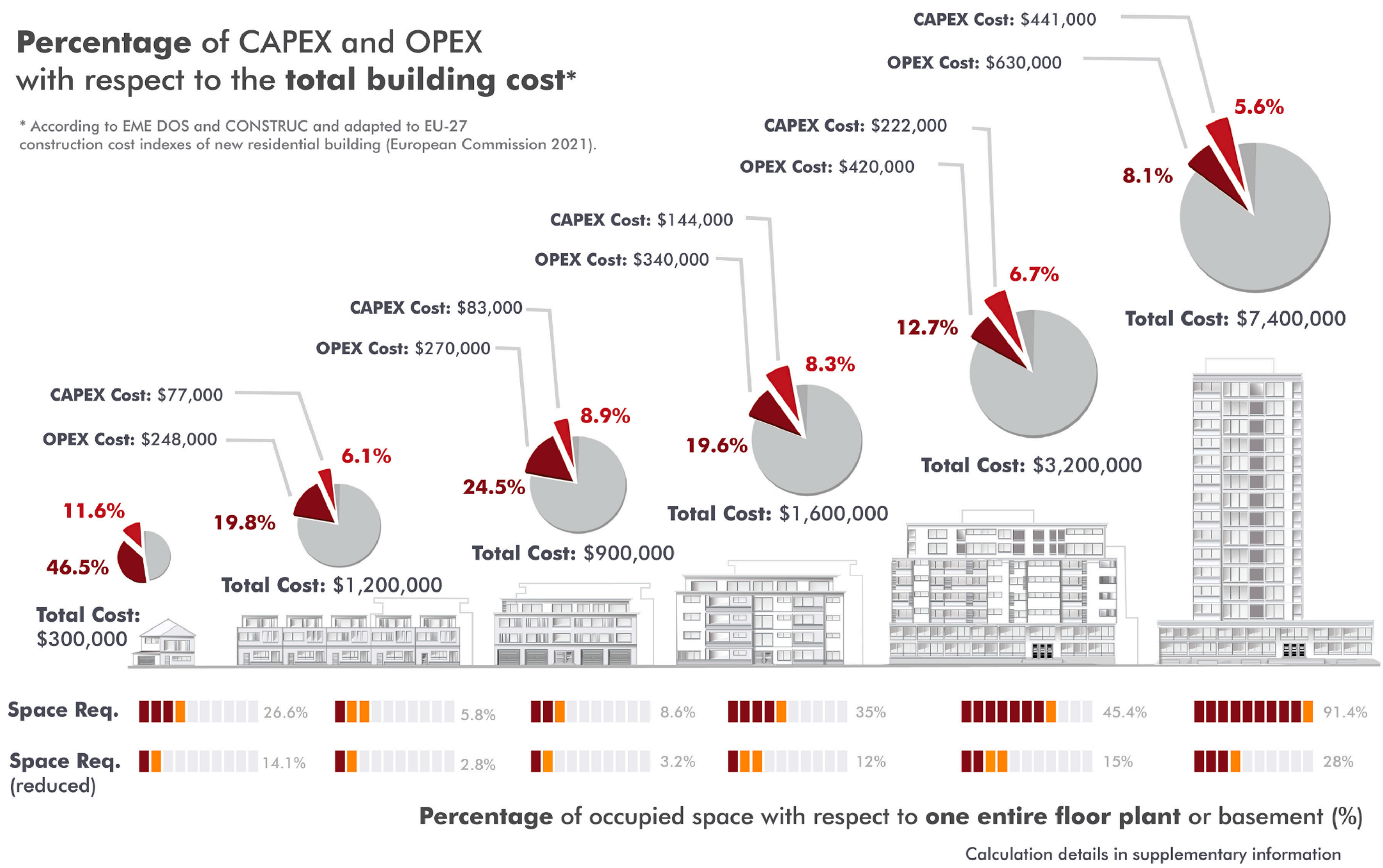

Fig. 7. Percentage of the total construction costs of the buildings that the CapEx and OpEx of the treatment plant represent. The estimated OpEx of the buildings are shaded in darker grey in each pie chart. Below the buildings, the percentage of the basement destined to tanks and technologies occupied by those is shown.

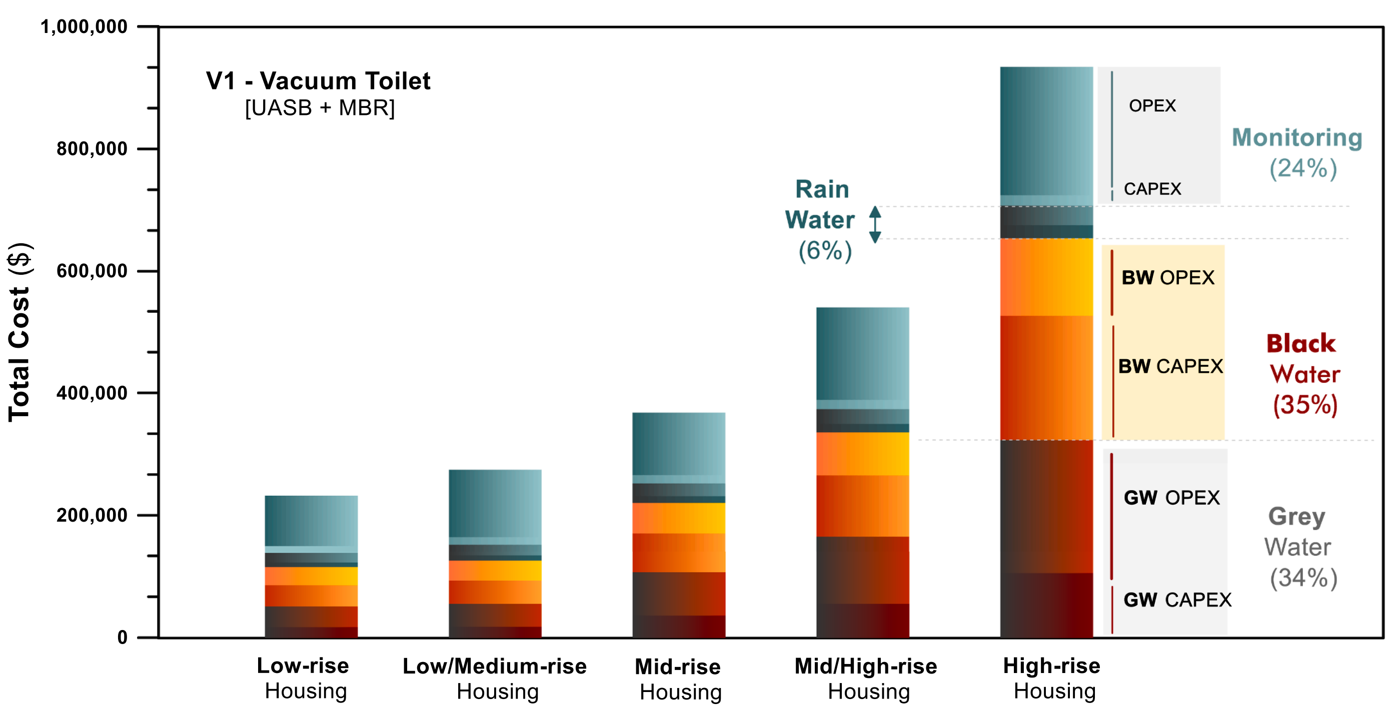

Fig. 8. Capital expenses (CapEx) and discounted lifetime operational expenses (OpEx) for the five larger buildings (V1) for a 30 years horizon. Dark, red-colored sections correspond to grey Water, orange-colored section to black water, darker blue corresponds to pipeline and sewer connection-related costs, and greenish-colored categories correspond to rainwater-related unit costs. Note differentiated OpEx and CapEx for each stream (BW, GW, and RW).

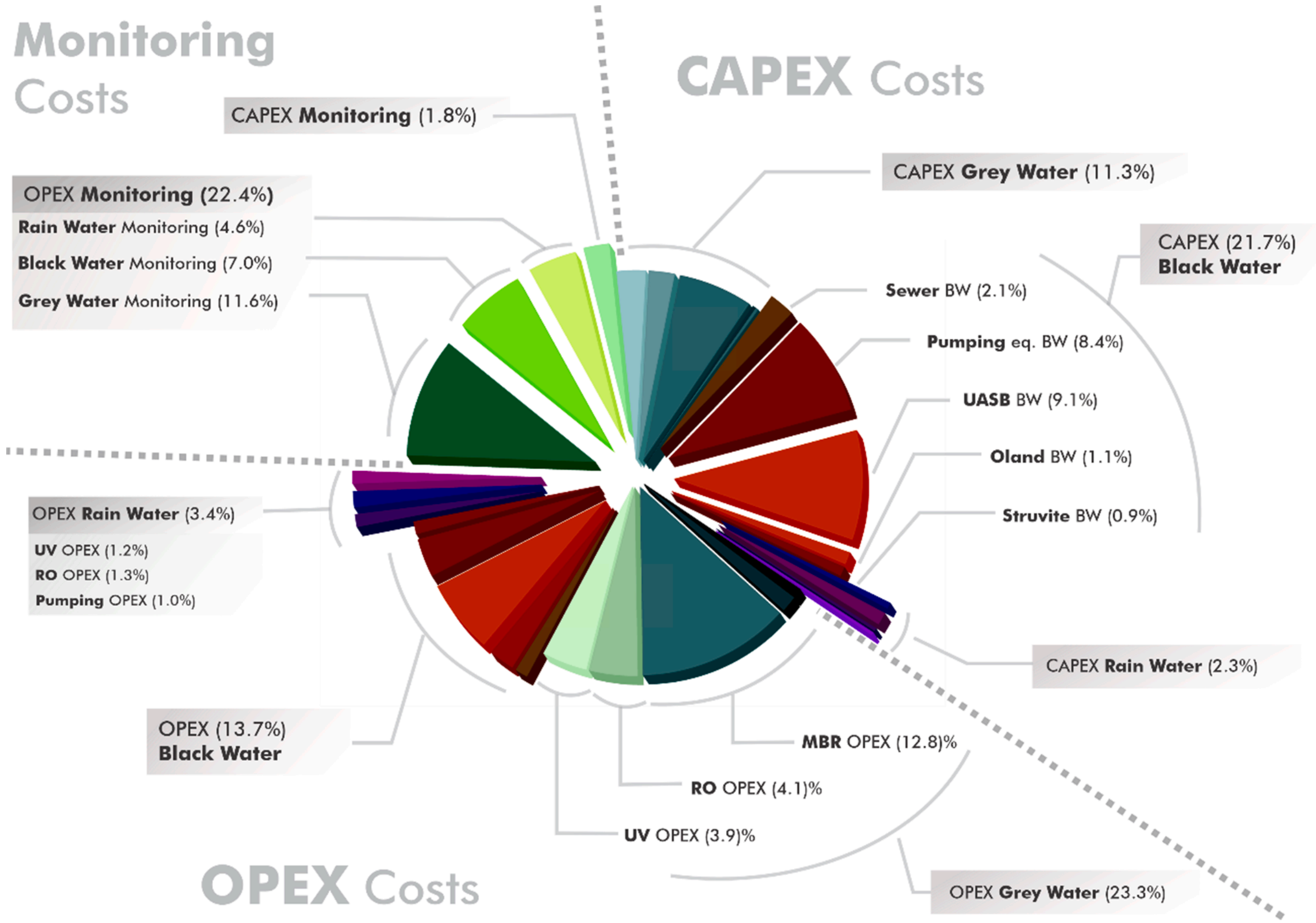

Fig. 9. Cost breakdown for the largest Building (HR), where costs from GW treatment. BW and RW are shown in blue, red, and purple, respectively. CapEx, OpEx, and monitoring costs are separated to compare the three categories (V1).

3.3.6. *Income assessment*

The recovery of resources by the water recycling system could offset the cost of installing and operating the systems (Section 3.3.2). Each treatment train considered in the analysis enabled the recovery of energy and several also included nutrient recovery. Although the production of fertilizer might be popular in community gardening initiatives and the produced biogas could help offset project greenhouse gas emissions, these two sources of revenue account for less than 2% of the total estimated OpEx (see supplementary information section 4.1.4 for details). Instead, the most most important offset was related to the decrease in water demand from the centralized water system, which could offset a substantial fraction of the annual operational costs (i.e., 74-100%; Figure).

Fig. 10 compares the theoretical savings of reducing the dependency of utility- water with the total CapEx and OpEx of each treatment train (i.e., water consumption charges minus infrastructure charges, which are accounted for separately and assuming an average water price of 2€/m^3), (Ajuntament de Barcelona, 2019). Fig. 10 shows that water savings account for 57% to 135% of total costs even with a theoretical increase in the water price lower than the inflation.

The proposed systems are assumed to have connection to the centralized sewer system to provide a means for disposing of excess water and providing redundancy if the treatment system is taken out fo service for repairs. Nevertheless, there may be sceanrios in which it is advantageous to avoid connecting the system to a centralized the sewer system (e.g., buildings located on the edges of cities where sewers have not yet been installed, places where landscape irrigation and the use of treated wastewater for ecological pruposes or groundwater recharge are attractive. The construction and maintenance costs of the connection to the water main (110 mm; PE) were also included and obtained using architecture and engineering prices simulation software CYPE. The described considerations result in capital and monthly savings of approximately 8-17% (See supplementary information section 6).

4. Conclusions

A detailed techno-economic and feasibility evaluation of the costs of extreme decentralization indicates that costs to builders and residents will be comparable to or lower than those of current systems, while also reducing per capita water demand. Using existing on-site treatment technologies and approaches that are likely to become mature in the next two decades we made several observations:

- The complete set of tanks and treatment equipment required for wastewater treatment and rainwater capture could fit into space that is available in existing basement space. For the smallest buildings, the equipment occupied a considerable fraction of the available basement space.
- The effluents of the presented treatment line configurations meet current quality regulations in terms of greywater reuse and drinking water. Public perception and limiting legislation, therefore, remain the principal impediments to wastewater recycling for domestic and even drinkning purposes.

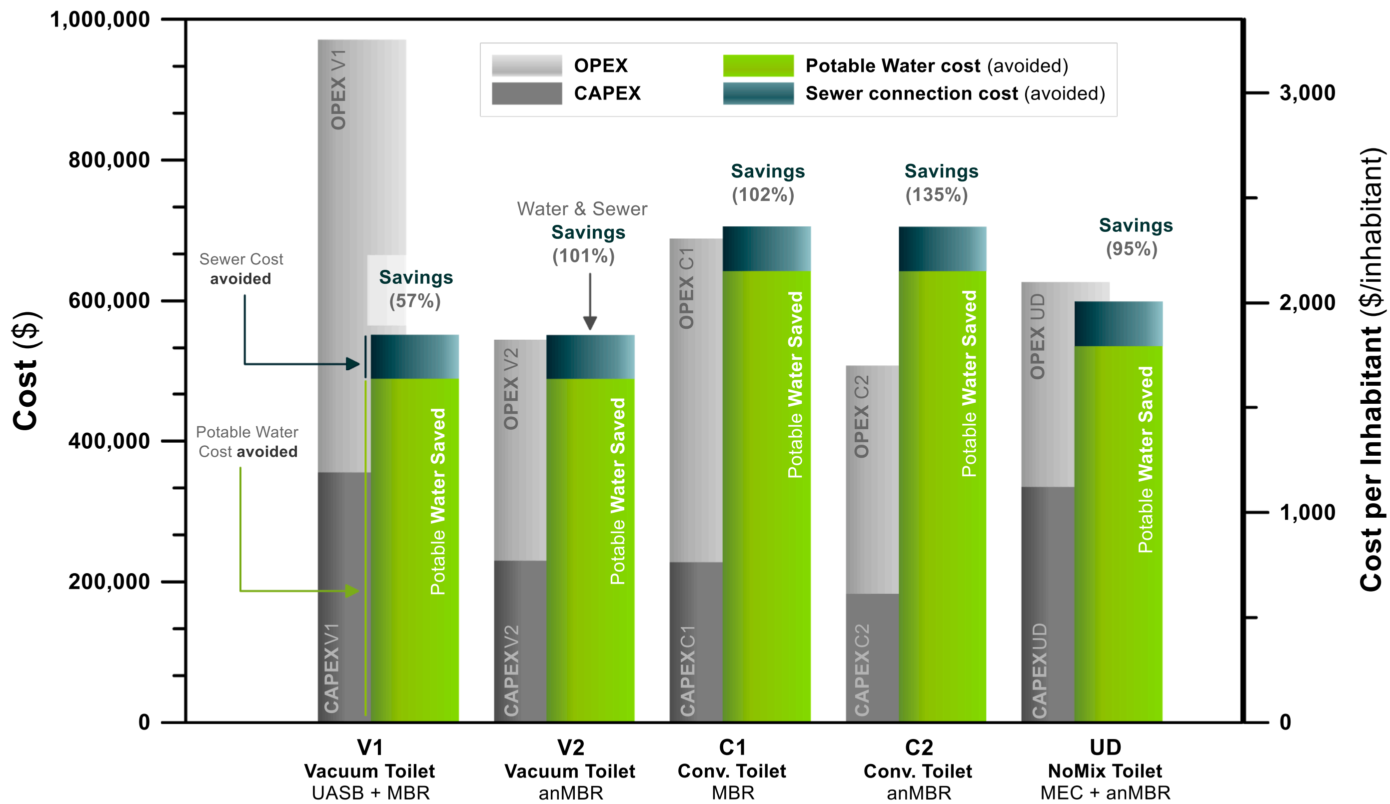

Fig. 10. Savings (avoided cost) in front of CapEx and OpEx of the plants, where sewer connection biogas and water savings are quantified (struvite and biogas benefits are considered negligible).

- The capacity of the installations for covering potable water demand from rainwater capture in average households depends on the building size and regional climate. Rainwater could only meet water demand for the entire year on the small buildings in the wetter climates. However, greywater could be recycled multiple times, assuring that building equiped with on-site treatment sysyems would have a substantially smaller per capita water demand than other types of housing.
- Decentralized plants allow for a drastic reduction in sewer disposal costs. Biological treatment units (i.e., MBRs, anMBRs, and UASB reactors) and black water pumping account for the largest capital and operational costs, while struvite recovery, UV, and RO systems provide significant benefits to the plants' performance and final water quality, requiring lower investments.
- During the construction of new residential blocks, only 5.6-11.6% of additional investment would be required to incorporate the required pipeline network and equipment installation. Exhaustive monitoring and maintenance is necessary to ensure health safety.
- The substitution of UASB reactors with anMBRs together with the consideration of source-separation of urine approaches can drastically lower the plants' operational costs, especially in large buildings, and offer an opportunity to minimize costs and maximize resource recovery.

5. Authors CREDIT Statement

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Declaration of Competing Interest

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

Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.watres.2022.118408.

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