

A novel concept to integrate energy recovery into potable water reuse treatment schemes

Nils Horstmeyer, Max Weißbach, Konrad Koch and Jörg E. Drewes

ABSTRACT

Potable water reuse applications can provide a safe and sustainable water supply where conventional freshwater resources are limited. The objectives of this study were fourfold: (i) to analyse existing potable water reuse applications regarding operational characteristics and energy demands, (ii) to determine the theoretical energy potential of wastewater and identify opportunities for energy recovery, (iii) to define design requirements for potable water reuse schemes that integrate energy recovery and (iv) to propose strategies for more energy efficient potable water reuse schemes. Existing potable water reuse schemes commonly utilize conventional wastewater treatment processes including biological nutrient removal followed by advanced water treatment processes. While meeting high product water quality, these treatment schemes are characterized by relatively high specific energy demands (1.18 kWh/m^3). Given that the theoretical energy potential of municipal wastewater is approximately two times higher (2.52 kWh/m^3), opportunities exist to integrate energy recovery strategies. We propose three alternative potable water reuse schemes that integrate energy recovery from carbon via methane and nitrogen via either the coupled aerobic–anoxic nitrous decomposition operation process or partial nitrification/anammox. Compared to conventional potable water reuse schemes, the energy requirements of these schemes can be reduced by 7–29% and the overall energy balance by 38–80%.

Key words | energy efficiency, energy recovery, potable water reuse, resource recovery, water reclamation

Nils Horstmeyer
Max Weißbach
Konrad Koch
Jörg E. Drewes (corresponding author)
Chair of Urban Water Systems Engineering,
Technical University of Munich,
Am Coulombwall 3, Garching 85748,
Germany
E-mail: jdrewes@tum.de

INTRODUCTION

Population growth and demographic shifts, climate change impacts, uneven distribution of freshwater resources, water scarcity and emerging water quality issues are becoming more and more pressing worldwide (Mekonnen & Hoekstra 2016; United Nations 2017). Various options exist to provide sufficient water supplies locally. These options depend on local availability and other site-specific factors and range from conventional surface water supplies (from rivers and lakes) or groundwater to brackish water or seawater

desalination and importing water from other watersheds. Depending on the type of water source, the required energy demands for treatment (no water conveyance and distribution included) are typically $0.05\text{--}0.37 \text{ kWh/m}^3$ for conventional water supply from surface water, $0.19\text{--}0.58 \text{ kWh/m}^3$ for conventional water supply from groundwater, $0.26\text{--}2.6 \text{ kWh/m}^3$ for brackish water desalination and $3.7\text{--}4.4 \text{ kWh/m}^3$ for seawater desalination (Appelbaum 2002; Cooley & Wilkinson 2012; Arzbaeher *et al.* 2013; Schimmoller & Kealy 2014). Importing water from other regions is a unique solution for certain regions and depends mainly on transport distance and elevation difference (Arzbaeher *et al.* 2013). The State Water Project in

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California, USA requires about 0.79 kWh/m³ for water conveyance alone, excluding energy requirements for water treatment (Cooley & Wilkinson 2012).

Many regions around the world have already established potable water reuse applications (Drewes & Khan 2011; NRC 2012) or are considering it as a future water supply option. The energy demands of existing potable water reuse schemes employing advanced treatment processes vary in the range 1.15–2.0 kWh/m³ (Tchobanoglous *et al.* 2011; Cooley & Wilkinson 2012; NRC 2012). However, these potable water reuse schemes represent a more cost-efficient option to augment local water supplies than brackish water or seawater desalination. Nevertheless, compared to conventional water supplies their specific energy demands are still significant.

Potable water reuse has been practised for more than 50 years and provides reliable and safe drinking water. Significant improvements in individual treatment processes and improved water quality monitoring have resulted in an increased confidence in potable water reuse practices worldwide (Drewes & Khan 2015; Schimmoller *et al.* 2015).

However, the widespread implementation of potable water reuse schemes is hindered by the lack of public confidence and regulatory uncertainty but also by high energy consumption and subsequent high operational and maintenance costs (O&M). With increasingly more stringent water quality requirements, it is expected that additional treatment steps for the removal of emerging microbial and chemical contaminants will be required that will further increase the energy demand of water treatment systems in the future.

In the past, significant process optimizations have resulted in improved energy efficiency of the overall treatment scheme (mainly by improved aeration systems for activated sludge processes, employment of energy recovery devices for high-pressure membranes and by utilization of anaerobic treatment processes). However, the current design philosophy of water reclamation facilities is still focused on initial biological carbon and nutrient removal followed by advanced treatment with little attention to simultaneous energy recovery. A new paradigm has emerged that is shifting the perception of wastewater as a disposal issue to an opportunity to continuously recover resources including nutrients, energy, heat and water (McCarty *et al.* 2011; Hering *et al.* 2013; Gao *et al.* 2014). Verstraete *et al.*

(2009), Remy *et al.* (2014) and Batstone *et al.* (2015) have proposed physical and biological treatment technologies to integrate improved energy and nutrient recovery in water reuse. Schaum *et al.* (2015) analysed the water cycle (water flows and energy demands) of metropolitan areas.

Untreated municipal wastewater contains potential, thermal and chemical bound energy. The potential energy (E_{pot}) of wastewater is proportional to its elevation and can be calculated with:

$$E_{\text{pot}} = m \times g \times h \text{ (J)} \quad (1)$$

where m is the water quantity/mass (kg), g the gravitational acceleration of the earth (9.81 m/s²) and h the elevation (m). In theory, potential energy can be utilized in the sewerage system or the influent and/or effluent of wastewater treatment plants (WWTPs). In many settings, this energy is not considered a significant energy source due to gravity-driven conveyance systems and dependency on the local topography.

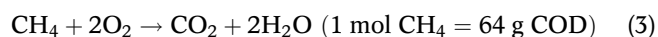
The amount of recoverable thermal energy (E_{therm}) can be calculated by:

$$E_{\text{therm}} = c_p \times \Delta T_m \times m \text{ (kJ)} \quad (2)$$

where c_p is the specific heat capacity of water (4.18 kJ/(kg × K)), ΔT_m the temperature gradient (K) and m the water quantity/mass (kg). Typical temperatures in wastewater streams vary from 10 °C to 20 °C (Dürrenmatt & Wanner 2014). In colder areas heat recovery can be a viable option for heat generation (McCarty *et al.* 2011). However, the maximum recoverable thermal energy depends strongly on the point of heat recovery (e.g. most feasible in a sewerage system) and potential heat consumers. Additionally, studies already indicated the negative side-effect of heat recovery in the sewerage system on downstream biological processes of WWTPs, in particular nitrification efficiency (Wanner *et al.* 2005). Due to these drawbacks, the thermal energy will not be considered a viable on-site energy source in this study.

As well as thermal energy, electrical energy can be generated from the chemical energy potential. This is the preferable option in terms of subsequent energy utilization as it is more versatile and can be transported almost without loss. The chemical energy potential of waste streams is

commonly assessed by the chemical oxygen demand (COD). The energy content per g COD can be determined using the overall enthalpy (ΔH of products – ΔH of reactants) expressed in the following reaction, assuming methane (CH_4) is the organic substrate:



with $\Delta H (\text{CH}_4) = -74.87 \text{ kJ/mol}$, $\Delta H (\text{O}_2) = 0 \text{ kJ/mol}$, $\Delta H (\text{CO}_2) = -393.50 \text{ kJ/mol}$ and $\Delta H (\text{H}_2\text{O}) = -285.83 \text{ kJ/mol}$. The overall enthalpy results in $\Delta H = -890.29 \text{ kJ/mol}$ or -13.91 kJ/g COD , which equals 3.86 kWh/kg COD .

Municipal raw wastewater contains typically COD concentration in the range of 250–950 mg/L (Tchobanoglous *et al.* 2003; Remy *et al.* 2014). Thus, the considered energy potential of wastewater depends on the assumed COD load and usually varies in the range $1.66\text{--}1.93 \text{ kWh/m}^3$ (COD = 430–500 mg/L) (McCarty *et al.* 2011; Scherson & Criddle 2014) up to $3.09\text{--}3.86 \text{ kWh/m}^3$ (COD = 800–1,000 mg/L) (Remy *et al.* 2014; Batstone *et al.* 2015). For comparison, the energy requirements of a conventional activated sludge (CAS) process with nitrification is in the range $0.35\text{--}0.80 \text{ kWh/m}^3$ (Appelbaum 2002; Fricke *et al.* 2015). Thus, it is worthwhile investigating treatment alternatives that can further reduce the energy demand to make potable water reuse not only a technically feasible but also a more energy efficient solution and to integrate resource recovery as a cornerstone of integrated water management concepts.

The objectives of this study were to: (i) analyse systematically existing potable water reuse applications regarding operational characteristics and energy demands; (ii) determine the theoretical energy potential of wastewater and identify opportunities for energy recovery; (iii) define design requirements for potable water reuse schemes that integrate energy recovery; and (iv) propose strategies for more energy efficient potable water reuse schemes. Hence, we designed three alternative potable water reuse schemes that integrate energy recovery from carbon and nitrogen. The alternative potable water reuse schemes are compared with each other and to a benchmark regarding: (i) overall energy balance, (ii) greenhouse gas potential, (iii) effluent water quality and (iv) process stability. Finally, necessary investigations which need to be addressed in the future are proposed.

EXISTING POTABLE WATER REUSE SCHEMES

Potable water reuse is being practiced worldwide using a broad variety of treatment process configurations. This variety is due to a wide range of possible treatment options and different regulatory requirements. The trend to meet more stringent water quality criteria over the last 20 years has resulted in incremental improvements of unit processes in potable water reuse treatment train (TT) design.

Treatment train characteristics

Existing potable water reuse schemes can be classified into membrane-based or non-membrane-based TTs and treatment schemes with or without an environmental buffer (Figure 1). Membrane applications are a robust process for organic carbon removal and an effective barrier to pathogens and trace organic contaminants. High-pressure membrane applications (e.g. reverse osmosis – RO) have become the backbone of many potable water reuse schemes worldwide (Drewes & Khan 2011; Gerrity *et al.* 2013). The efficient removal of dissolved solids, pathogens and trace organic compounds results in the fact that more than half of the existing potable water reuse schemes worldwide and 80% of the schemes in California, USA, utilize RO as a key treatment process (Drewes & Khan 2011; Drewes & Horstmeyer 2015). In particular, potable water reuse schemes located in coastal areas favour RO and in California and Australia are also combined with advanced oxidation processes (AOPs), e.g. ultraviolet light–hydrogen peroxide (UV– H_2O_2) (Gerrity *et al.* 2013). The Orange County Water District in Southern California established Water Factory 21 in 1976, which was later replaced by the Groundwater Replenishment System (GWRS) representing the largest potable water reuse project worldwide with a capacity of $348,400 \text{ m}^3/\text{d}$ (Drewes & Horstmeyer 2015). The GWRS treatment scheme consists of the core processes of microfiltration/reverse osmosis/advanced oxidation process (UV– H_2O_2). Due to the lack of brine disposal options, inland potable reuse projects are considering low-pressure membrane filtration (e.g. ultrafiltration – UF), granular activated carbon (GAC) adsorption, chemical oxidation (e.g. ozonation), and natural treatment systems (e.g. soil aquifer

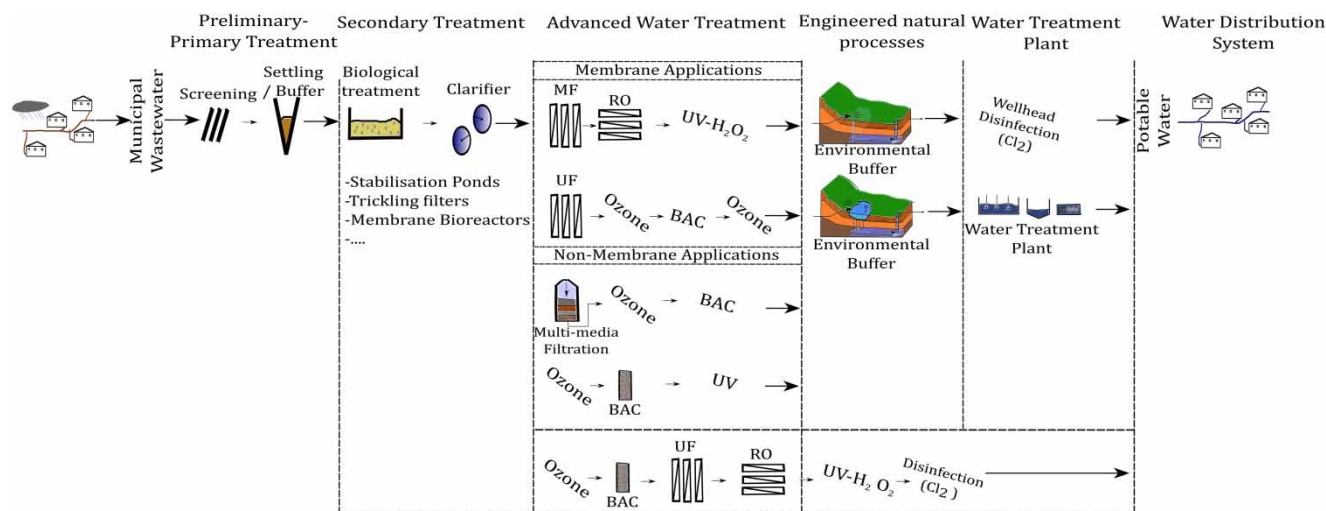


Figure 1 | Design of selected potable water reuse schemes worldwide. MF, microfiltration; RO, reverse osmosis; UV, ultraviolet light; H_2O_2 , hydrogen peroxide; Cl_2 , chlorine; UF, ultrafiltration; BAC, biological activated carbon.

treatment, riverbank filtration, wetland treatment) (Drewes & Horstmeyer 2015). Managed aquifer recharge options are only applicable if the geological conditions are suitable. Advanced processes like GAC or RO (with the exception of suitable brine disposal options) can be employed practically everywhere. Non-membrane based reuse schemes consider activated carbon as a robust barrier for organic contaminants. If GAC is applied, usually RO is not considered. Many locations utilize GAC for the removal of bulk and trace organic compounds, other treatment schemes consider biological activated carbon (BAC) in combination with ozonation (Gerrity *et al.* 2013; Drewes & Khan 2015).

Certain technologies have been used over the past few decades (e.g. ozonation), while other technologies have become more and more economically feasible in the recent past (e.g. RO, UV-AOP) (Gerrity *et al.* 2013). As a result, a number of ozone-based alternatives (e.g. ozone/BAC) are increasing in popularity throughout the world (Gerrity *et al.* 2013). It is expected that the attention to a broader spectrum of contaminants will increase the diversity and/or the treatment complexity of treatment units for potable water reuse applications. Recently, more attention has been given to schemes that facilitate direct potable water reuse without the use of an environmental buffer (Drewes & Khan 2015; Lahnsteiner *et al.* 2017). Several decades of operational experience with potable water reuse

applications demonstrated the reliability and robustness of existing treatment schemes.

Energy demand, greenhouse gases and carbon footprint

Augmentation, distribution and (waste)water treatment require energy at all stages. The reported energy demands in this study are only for the (waste)water treatment. Water collection, transport and distribution are not included in the reported energy demands. These factors depend mainly on the distance and net elevation (Cooley & Wilkinson 2012). Depending on the specific treatment configuration, degree of treatment (wastewater characteristic, effluent criteria), regional and site-specific factors, and the facility's capacity, existing potable water reuse schemes require between 1.15 and 2.0 kWh/m³, while the energy demand of CAS (with nitrification) can vary from 0.35 to 0.80 kWh/m³ (Appelbaum 2002; Fricke *et al.* 2015), and the advanced water treatment can vary between 0.85 and 1.2 kWh/m³ (Tchobanoglous *et al.* 2011; Cooley & Wilkinson 2012; NRC 2012). The GWRS treatment scheme represents a characteristic membrane-based treatment configuration (including MF/RO/UV- H_2O_2) and is considered as a benchmark for comparison of alternative potable water reuse schemes as the largest potable water reuse project worldwide. The average energy demand of this scheme is 1.18 kWh/m³

(Figure 2). With 0.70 kWh/m^3 , the use of the integrated membrane system (MF/RO) consumes more than 90% of the advanced water treatment (AWT) energy demand (Holloway et al. 2016). While these membrane processes provide very efficient treatment barriers in potable water reuse, they are characterized by high energy requirements (Schimmoller & Kealy 2014). As a result, the entire AWT process train represents 181% of the CAS energy demand. The energy demand for AWT can vary between 0.85 and 1.2 kWh/m^3 (Cooley & Wilkinson 2012; Schimmoller & Kealy 2014).

Beside the direct energy demand of single treatment processes and total treatment scheme configuration, the generation of greenhouse gases (GHGs) and the resulting overall carbon footprint needs to be considered. The main contributor to GHG emissions is indisputably the energy demand (Cornejo et al. 2014; Schimmoller & Kealy 2014). The carbon footprint for CAS treatment with biological nutrient removal ranges from 0.13 to $0.69 \text{ kg CO}_2\text{-equivalent (CO}_2\text{e)/m}^3$ (Lazarova et al. 2012; Cornejo et al. 2014; Remy et al. 2014). Membrane-based potable water reuse schemes are in the range 0.60 – $2.40 \text{ kg CO}_2\text{e/m}^3$ (Vince et al. 2008; Stokes & Horvath 2009; Kalbar et al. 2013; Cornejo et al. 2014; Niero et al. 2014). Holloway et al. (2016) analysed the GWRS benchmark showing that the GHG emissions range from 1.15 to $1.50 \text{ CO}_2\text{e kg/m}^3$, depending on the RO concentrate disposal pressure (ocean disposal or deep well injection with 69 bar). The direct and accurate comparison of different GHG calculations is challenging due to the variability in location, technologies, life-cycle stages and parameters

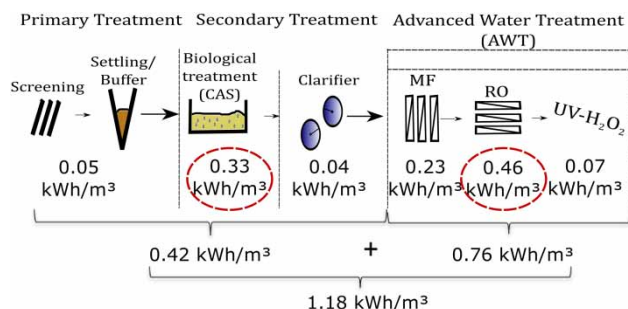


Figure 2 | Energy demand of the characteristic benchmark Groundwater Replenishment System (GWRS) example. Main energy sinks of the primary, secondary treatment and AWT are marked with red circles. MF, microfiltration; RO, reverse osmosis; UV, ultraviolet light; H₂O₂, hydrogen peroxide; AWT, advanced water treatment (Arzbaecher et al. 2013; Fricke et al. 2015; Holloway et al. 2016). Please refer to the online version of this paper to see this figure in colour: <http://dx.doi.org/10.2166/wrd.2017.051>.

included (Cornejo et al. 2014). Another approach to consider the full financial, environmental, and social elements of a potable water reuse project is the triple bottom line framework (Schimmoller & Kealy 2014; Schimmoller et al. 2015).

Additionally, it has been recognized that CAS systems directly emit GHGs. Beside CO₂ and CH₄, nitrous oxide (N₂O) has come to the forefront in the last decade (Kampschreur et al. 2008, 2009; Desloover et al. 2011; Daelman et al. 2013). With a CO₂e of 300 (Ravishankara et al. 2009), the contribution of N₂O to the operational carbon footprint of CAS systems is estimated to be on average $0.40 \text{ kg CO}_2\text{e/m}^3$ (range 0.02 – $1.18 \text{ kg CO}_2\text{e/m}^3$) (assumption $0.081 \text{ kg N}_2\text{O}/(\text{population-equivalent (PE)} \times \text{a})$) (Daelman et al. 2013) and potentially increases the total carbon footprint in the range 0.15 – $1.87 \text{ kg CO}_2\text{e/m}^3$.

Presently, considerable additional nitrogen loads are released to the system by ammonification of organic nitrogen during anaerobic digestion (AD). Applying main-stream treatment of reject water imposes additional ammonia loads to the nitrification/denitrification (N/DN) process elevating aeration energy demand for nitrification and COD demand for denitrification. Eventually, a lack of organic reducing power by intensive use of AD potentially leads to anoxic nitrous denitrification (Weißbach et al. 2018) and ultimately to higher N₂O emissions than during conventional treatment without AD.

To mitigate this drawback to main-stream nitrogen removal, the partial nitrification/anammox (PN/A) process was developed, because it facilitates complete autotrophic nitrogen removal (and thus higher process efficiency). However, considerable N₂O emissions need to be accounted for in GHG calculations. Kampschreur et al. (2008) and Desloover et al. (2012) quantified conversion of 2.3 – 3.8% of the reactor nitrogen load to N₂O. The unintended production of N₂O and associated CO₂e emissions possibly compromise the benefits of higher process efficiency.

MUNICIPAL WASTEWATER AS ENERGY AND NUTRIENT SOURCE

Municipal raw wastewater contains a considerable chemical energy potential (see Introduction). Thus, the energy and nutrient content of municipal raw wastewater was characterized through weekly measurements over a period of one

year at the WWTP at Garching, Germany. Wastewater influent was collected after a 4 mm drum screen. Average values were determined as follows: flow rate $4,644 \pm 724 \text{ m}^3/\text{d}$, temperature $15.3 \pm 2.4 \text{ }^\circ\text{C}$, pH 8.24 ± 0.06 and electrical conductivity $1,438 \pm 241 \text{ }\mu\text{S}$. The theoretical energy potential has been determined to vary between 1.9 and 3.14 kWh/m^3 (mean 2.52 kWh/m^3) (Figure 3 and Table SI 1 in the supplementary material, available with the online version of this paper).

ENERGY RECOVERY PLATFORMS (FROM CARBON AND NITROGEN)

Raw wastewater represents a high theoretical energy potential (2.52 kWh/m^3). Combined heat and power (CHP) units, for instance combustion of biogas harvested from anaerobic treatment, are common examples of transforming chemical energy into electrical usage via energy recovery platforms.

Anaerobic main-stream treatment results in approximately twice the methane yield compared to main-stream CAS (100 L/m^3 via main-stream AD compared to 50 L/m^3 via AD of waste activated sludge) (McCarty *et al.* 2011). Additionally, less waste sludge (1.2 L/m^3) is produced in anaerobic treatment than in aerobic treatment (2.6 L/m^3) (McCarty *et al.* 2011). Considering accumulated losses of, for example, waste heat, friction and kinetics during energy conversion, the total yield of electrical energy is approximately 6–11% ($0.15\text{--}0.28 \text{ kWh/m}^3$) (Lazarova *et al.* 2012; Remy *et al.* 2016). This yield represents approximately 67% of the required energy demand of a CAS system and only 24% of a CAS system including AWT processes. Puchongkawarin *et al.* (2015) estimate that approximately only 25–50% of the CAS system energy demand and only 11–22% of the CAS

system including downstream AWT can be recovered. Hence, the limitations of existing schemes are evident in terms of achieving neutral net energy operation.

Temperature is a key factor for efficient AD. Anaerobic processes are most effective in mesophilic ($28\text{--}40 \text{ }^\circ\text{C}$) and thermophilic ($50\text{--}57 \text{ }^\circ\text{C}$) conditions. Nevertheless, full-stream anaerobic treatment plants operate also commonly under psychrophilic conditions ($<20 \text{ }^\circ\text{C}$) (Martinez-Sosa *et al.* 2011; Petropoulos *et al.* 2017) and with high strength wastewater. Inoculated reactors with cold-adapted biomass from low temperatures regions over evolutionary time-scales (e.g. from the high arctic or alpine lakes) showed promising results in establishing anaerobic treatment for low temperature ranges ($10\text{--}30 \text{ }^\circ\text{C}$) and low-strength wastewater (Tauseef *et al.* 2013; Petropoulos *et al.* 2017). Other challenges associated with anaerobic treatment are: (i) corrosion by sulfuric acid, (ii) nitrogen removal and (iii) capturing residual methane in effluents (25 times CO_2e). Particularly low temperatures increase the solubility of methane and as a consequence result in higher GHG emissions (Tauseef *et al.* 2013; Crone *et al.* 2016). Up to 6.9% of dissolved methane losses are possible (Crone *et al.* 2016).

For nitrogen removal, nitrification–denitrification is still the most established and traditional process in CAS (Lackner *et al.* 2014). Nitrification describes the two-step process where ammonia (NH_4^+) is oxidized to nitrite (NO_2^-) and nitrite is oxidized to nitrate (NO_3^-). The process is stoichiometrically expressed as (Schmidt *et al.* 2003):

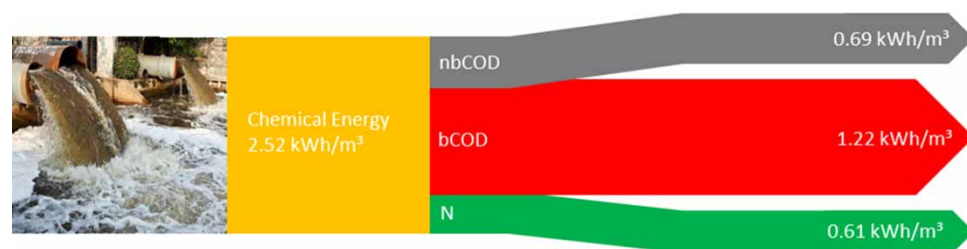
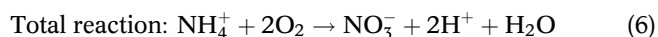
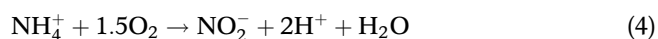


Figure 3 | Chemical bound energy potential of raw municipal wastewater (COD = 496 mg/L , TN = 81 mg/L , nbCOD = 179 mg/L , bCOD = 317 mg/L). Based on complete COD oxidation and 3.86 kWh/kg COD and on the (higher) heating value for ammonia nitrogen. bCOD, biodegradable COD; iCOD, inorganic (non-biodegradable) COD.

Considering the energy demand, 4.67 kg O₂/kg NH₄-N are necessary for nitrification resulting in an aeration energy demand of 2.38 kWh/kg NH₄-N (Sobieszuk & Szewczyk 2006). To mitigate the significant aeration energy demand for nitrification, technologies and strategies have been applied to increase oxygen mass transfer into the mixed liquor and prevent excess aeration. However, similar to the PN/A process, oxygen limitation can trigger aerobic nitrous denitrification resulting in increased N₂O emissions. Beside the energy demand of nitrification, 7.60 kg COD/(kg N) are necessary for complete denitrification (Sobieszuk & Szewczyk 2006). Hence, a considerable amount of the energy potential of COD is lost by applying conventional nitrogen removal. To enable more efficient nitrogen removal, novel processes have been proposed focusing on short cuts in the microbial metabolism or completely alternative microbial pathways. These processes (e.g. N/DN, PN/A and the coupled aerobic–anoxic nitrous decomposition operation (CANDO)) potentially facilitate more efficient nitrogen removal, because they combine both lower oxygen and COD demand (Table 1). Considering oxygen demand, PN/A is theoretically most efficient, because only 37.5% oxygen is required compared to N/DN (50% of NH₄⁺ is oxidized to NO₂[−] with 25% less oxygen demand). Furthermore, the process is completely autotrophic saving theoretically 7.60 kg COD/kg N for additional 10.9 kWh/kg N removed with PN/A (64% biodegradable COD (bCOD) and 2.25 kWh/kg COD based on data from the Garching WWTP, see Table SI 2 in the supplementary material, available with the online version of this paper).

In contrast, the CANDO process (Scherson *et al.* 2014) enables energy recovery from nitrogen. It is a two-stage biological process for intentional N₂O production, harvesting and subsequent energetic utilization as co-oxidant during biogas combustion. The oxygen demand represents 75% of

the N/DN oxygen demand. Weißbach *et al.* (2018) could demonstrate a conversion of 63% of NH₄-N to N₂O-N under real feed-stream conditions treating digester effluent. The partially heterotrophic process was operated at a COD/N ratio of 4.0 kg COD/kg N (Weißbach *et al.* 2018). The degradability of 64% resulted in a considerably lower bCOD demand of 2.60 kg COD/kg N compared to CAS (7.60 kg COD/kg N) (Table 1). As a result, 5.0 kg COD/kg N removed are available for biogas production (7.20 kWh/kg N, based on 64% bCOD and 2.25 kWh/kg COD) and 0.53 kWh/kg N (0.083 kg N/m³, 82.3 kJ/mol N₂O, conversion efficiency NH₄-N to N₂O-N 63%, 23.2 mol N₂O/kg N), resulting in additional 7.73 kWh/kg N that can potentially be generated.

ALTERNATIVE POTABLE WATER REUSE SCHEMES

This section provides the design requirements that integrate energy recovery and explains and compares the alternative TTs with each other and with the defined benchmark for potable water reuse schemes. The overall energy balance, the GHG potential, the effluent water quality and the process stability are considered.

Design requirements

The design of alternative potable water reuse schemes requires a thorough understanding of (i) source water characteristics, (ii) regulatory and water quality requirements, (iii) maintaining proper performance of treatment processes and their combinations, and (iv) storage and blending requirements. Maintaining system reliability (by establishing redundancy, robustness and resilience) is a very important design element for potable water reuse

Table 1 | Comparison of N/DN, PN/A and CANDO processes

Process	O ₂ demand (kg O ₂ /kg N)	COD demand (kg COD/kg N)	Energy demand (kWh/kg N)	Energy potential from COD (kWh/kg N)	Energy potential from N (kWh/kg N)	Overall balance (kWh/kg N)
CAS N/DN	4.67	7.60	2.38	–	–	2.38
PN/A	1.75	–	0.90	10.9	–	–10.0
CANDO	3.50	2.60	1.80	7.20	0.53	–5.93

N/DN, nitrification/denitrification; PN/A, partial nitrification/anammox; CANDO, coupled aerobic–anoxic nitrous decomposition operation.

schemes and is defined as the probability of providing adequate performance for defined time periods while meeting predefined water quality conditions (Drewes & Horstmeyer 2015). System reliability can be achieved by various combinations of treatment processes considering expected log-removal efficiencies of single unit operations for pathogenic microorganisms as well as chemicals of concern. However, overtreatment should be avoided (Schimmoller & Kealy 2014).

So far, the concept of process optimization to increase energy efficiency in potable water reuse schemes has primarily targeted savings for individual unit operations and processes. However, integrating energy recovery platforms into potable water reuse schemes offer opportunities for significant savings but might also result in a reappraisal of the design of current potable water reuse schemes. These novel approaches embrace the concept of lowering energy requirements while increasing energy recovery. The substitution of energy intensive CAS treatment by coupling biological, chemical and physical treatment processes could reduce the overall energy demand while increasing the organic material present in primary effluent for intensified biogas production and more energy efficient nitrogen removal via processes like CANDO or PN/A.

Alternative treatment trains

Three alternative TTs for potable reuse schemes are proposed (Figure 4). TT I consists of an improved pretreatment with coagulation, flocculation and microsieving (100 µm) to increase carbon removal during primary sludge removal. The primary sludge is diverted to an anaerobic digester to convert carbon to methane. The primary effluent is subject to downstream physical treatment including ultrafiltration (UF) prior to subsequent RO treatment as an additional membrane separation step. The RO also serves as a concentration step for residual dissolved organic constituents and ammonia. The UF retentate and RO concentrate is diverted to the anaerobic digester to utilize as much as possible of the dissolved organic carbon for methane production. The concentrated ammonia is transferred after the anaerobic treatment (via the digester centrate) to the CANDO process as a second energy recovery platform. The CANDO process converts the ammonia in

a double-stage biological process to N_2O . Methane and N_2O are subsequently energetically utilized in CHP. The RO permeate is further treated by a biological activated carbon filter (BAF) to remove the remaining low-molecular organic carbon and nutrients (<100 Dalton). Subsequent water treatment is applied as pathogen barriers and for any residual trace organic chemicals in two chemical processes via an AOP (UV- H_2O_2) and final chlorine disinfection.

TTs II and III (TT II and III) are based on anaerobic treatment with intensified biogas production via an anaerobic membrane bioreactor (AnMBR). Raw municipal wastewater is directly fed into the AnMBR after a fine screen (4 mm) to remove coarse particulate matter. The AnMBR represents the first energy recovery platform for methane production from carbon. The MBR filtrate is diverted to downstream RO treatment followed by BAF and UV/AOP and final disinfection like TT I. In TT II, the RO concentrate (elevated in ammonia) is utilized in the CANDO process (intentional N_2O production for subsequent co-combustion with methane. In TT III, the RO concentrate is subject to PN/A treatment to convert ammonia to N_2 with final release into the atmosphere. CANDO and PN/A were chosen for nitrogen removal due to the better overall energetic balance compared to conventional N/DN (N/DN: 2.38 kWh/kg N; CANDO: -5.93 kWh/kg N; PN/A: -10.0 kWh/kg N, see Table 1). The comparison of TT II and TT III will provide an assessment of whether the approximately two times better energy balance of PN/A compared to the CANDO process influences the overall energy balance.

Previous studies have demonstrated the viability of microsieving at pilot-scale to remove particulate matter (Remy *et al.* 2014). Combined with coagulation/flocculation, this solids separation process removed more than 95% of suspended solids. Moreover, the microsieve removed 70–85% of organic matter (COD). Nearly 95% of particulate COD (pCOD) and 20% of the dissolved COD (dCOD) from raw wastewater was removed (Remy *et al.* 2014). This increased primary sludge removal can result in a higher methane yield and with up to 40% higher energy recovery compared to CAS treatment (Remy *et al.* 2014). Additionally, the organic load on the subsequent purification steps is reduced (95% TSS removal). However, downstream UF treatment of the microsieved pretreated

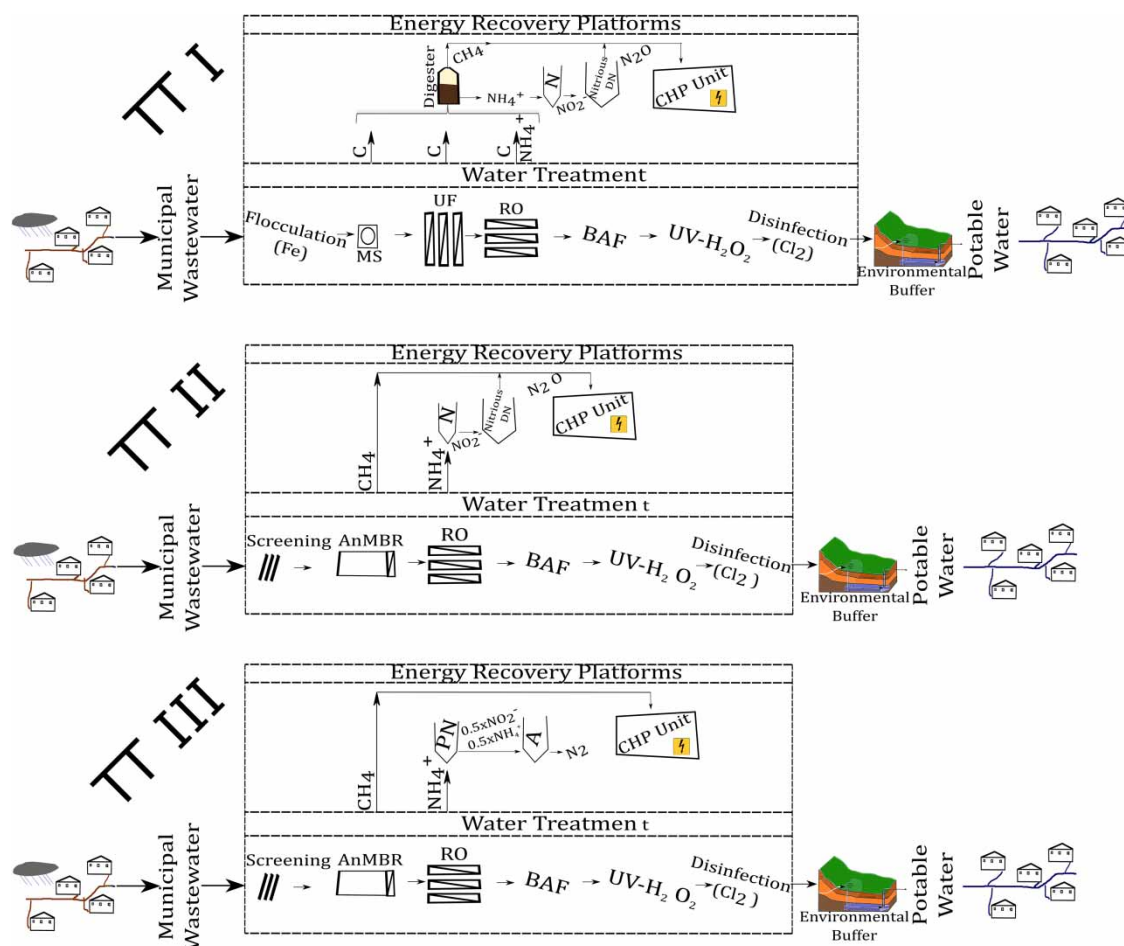


Figure 4 | Alternative potable water reuse schemes (TT I, II and III). TT, treatment train; N, nitrification; DN, denitrification; C, carbon; CHP, combined heat and power plant; Fe, ferrous iron; MS, microsie; UF, ultrafiltration; RO, reverse osmosis; BAF, biological activated carbon; UV-H₂O₂, ultraviolet light-hydrogen peroxide; Cl₂, chlorine; AnMBR, anaerobic membrane bioreactor; PN, partial nitrification; A, anammox.

raw wastewater as proposed for TT I is still challenging due to the remaining high organic carbon load and associated high fouling propensity of the UF membrane. Advanced fouling mitigation strategies are required to maintain a sustainable UF operation. Kim *et al.* (2011) analysed a new fouling mitigation strategy by using fluidized GAC in a two-stage anaerobic system with the second stage operating as AnMBR. The system was operated continuously over 120 days at laboratory scale at 35 °C with synthetic wastewater (average COD = 513 mg/L) while achieving a permeate flux of 10 L/m² hour (LMH). The total energy demand for both stages was 0.06 kWh/m³. The approach was also successfully established at pilot scale by Shin *et al.* (2014).

Comparison of the three proposed alternative TTs (TTs I–III) against the benchmark potable water reuse scheme (CAS + AWT) reveals that the energy requirements can be decreased by 7% (TT I) and 29% (TT II and III), respectively (Figure 5(a); for detailed data see Table SI 3 in the supplementary material, available with the online version of this paper). TT II and III consist of an AnMBR which lowers the overall energy requirements for wastewater treatment to 0.84 kWh/m³. The calculation of the overall net energy balance even results in higher energy efficiencies for TTs I–III (Figure 5(a)). The chosen nitrogen removal strategy is the main contributor to achieving an overall more energy efficient potable water reuse schemes. PN/A and the CANDO process can increase the energy

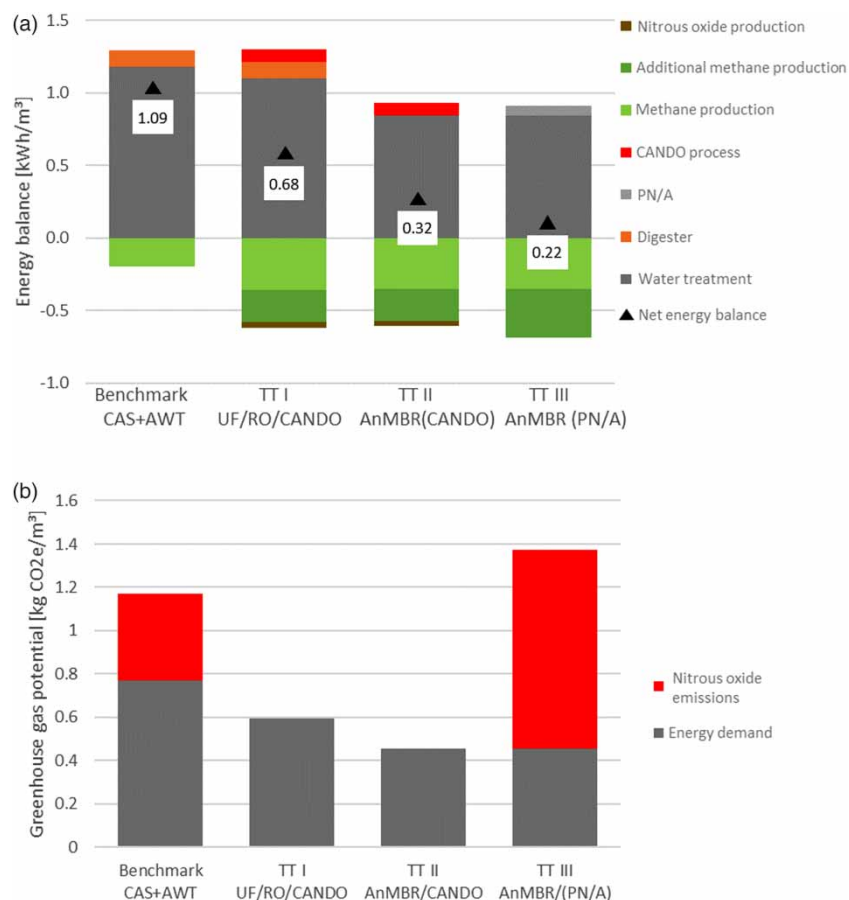


Figure 5 | (a) Energy balance of benchmark and treatment trains (TT) I, II and III; (b) nitrous oxide emission and energy demand as greenhouse gas potential of benchmark and treatment trains (TT) I, II, III based on 0.40–0.92 kg CO₂e/m³ for N₂O emissions and an energy demand equivalent of 0.54 kg CO₂e/kWh. CAS = conventional activated sludge, AWT, advanced water treatment; UF, ultrafiltration; RO, reverse osmosis; AnMBR, anaerobic membrane bioreactor; CANDO, coupled aerobic-anoxic nitrous decomposition operation; PN/A, partial nitrification/anammox.

production via additional methane production by 0.22 and 0.34 kWh/m³ (64% bCOD, 2.28 kWh/kg COD and η_{el} = 38%), respectively. Additional N₂O utilization via the CANDO process (0.04 kWh/m³) results in a more energy efficient balance for TT I and II. TT III with PN/A has the best overall energy balance with 0.22 kWh/m³. However, process stability and considerable N₂O emissions are of concern, especially when aiming for complex and varying wastewater compositions or full-stream applications (Laureni et al. 2016). Kampschreur et al. (2008) and Desloover et al. (2012) quantified N₂O emissions of a full-scale PN/A process with 2.3–3.8% of the reactor nitrogen load. While the CANDO process captures all N₂O emissions and converts it to energy, the additional GHG from a PN/A process due to N₂O emissions to the atmosphere account

for 0.92 kg CO₂e/m³ (0.081 kg/m³ N, N₂O CO₂e = 300) (Figure 5(b)). Considering both energy recovery and GHG emissions, TT II is the most sustainable potable water reuse scheme with a net energy balance of 0.32 kWh/m³ and 0.45 kg CO₂e/kWh based on the assumptions of this analysis. However, additional research is needed to demonstrate feasibility, reliability and water quality of the proposed alternative water treatment as well as the CANDO implementation in a main-stream application.

The main requirement of all potable water reuse schemes is the reliable and continuous generation of high-quality water in compliance with drinking water regulations. A wide range of naturally occurring and anthropogenic trace organic and inorganic contaminants, residual nutrients, total dissolved solids, residual heavy metals and pathogens

Table 2 | Log-removals efficiencies of treatment trains (TT) I, II and III compared to the benchmark (CAS + AWT)

	Benchmark ^a	TT I ^b	TT II and III ^b
Enteric virus log-removal	14.0	18.0	18.0
<i>Cryptosporidium</i>	12.0	12.5	12.5
Total coliform bacteria log-removal	18.5	21.0	21.0

Log-removal efficiencies of treatment steps according to the Treatment Train Toolbox from Trussell et al. (2017).

TT, treatment train; CAS, conventional activated sludge; AWT, advanced water treatment; SRT, solids retention time; UF, ultrafiltration.

^aAssuming CAS with SRT of 7 days, UV dose of 300 mJ/cm², Cl₂ of 450 mg-min/L and a contact time of 90 min with an initial Cl₂ demand of 0 mg/L.

^bAssuming UV dose of 300 mJ/cm², Cl₂ of 450 mg-min/L and a contact time of 90 min with an initial Cl₂ demand of 0 mg/L and additional log-removal for UF treatment.

needs to be properly removed by individual treatment units. Each alternative potable water reuse scheme fulfils the minimum removal requirements for microbial contaminants (9-log removal for total coliform bacteria, 12-log removal for enteric viruses and 10-log removal for protozoa (*Cryptosporidium*)) for potable reuse projects in California, USA (California SWRCB 2013; Trussell et al. 2017) (Table 2).

CONCLUSIONS

Environmental sustainability is one of the most critical challenges in contemporary water and wastewater management. The aim of technologies employed to treat water and wastewater should not only be to remove relevant contaminants, but also to achieve high energy efficiency by recovering useful resources from wastewater. The theoretical energy content in raw wastewater is 2.52 kWh/m³. Conventional potable water reuse schemes (CAS + AWT) currently require an average of 1.18 kWh/m³. Integrating anaerobic treatment (e.g. AnMBR) and CANDO offers the opportunity to establish more energy efficient potable water reuse schemes with significant lower GHG emissions. However, more detailed investigations of these alternative treatment schemes are needed including a detailed technical feasibility study. In particular, the feasibility of the proposed physical treatment processes (microsieving, UF and RO) coupled with subsequent biological (BAF) and disinfection processes require detailed research to overcome operational issues, demonstrate long-term operation and guarantee final water quality for drinking water purposes.

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