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Brackish Water Desalination using Reverse Osmosis and Capacitive Deionization at the Water-Energy Nexus

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Abstract

In this article, we present a critical review of the reported performance of reverse osmosis (RO) and capacitive deionization (CDI) for brackish water (salinity < 5.0 g/L) desalination from the aspects of engineering, energy, economy and environment. We first illustrate the criteria and the key performance indicators to evaluate the performance of brackish water desalination. We then systematically summarize technological information of RO and CDI, focusing on the effect of key parameters on desalination performance, as well as energy-water efficiency, economic costs and environmental impacts (including carbon footprint). We provide in-depth discussion on the interconnectivity between desalination and energy, and the trade-off between kinetics and energetics for RO and CDI as critical factors for comparison. We also critique the results of technical-economic assessment for RO and CDI plants in the context of large-scale deployment, with focus on lifetime-oriented consideration to total costs, balance between energy efficiency and clean water production, and pretreatment/post-treatment requirements. Finally, we illustrate the challenges and opportunities for future brackish water desalination, including hybridization for energy-efficient brackish water desalination, co-removal of specific components in brackish water, and sustainable brine management with innovative utilization. Our study reveals that both RO and CDI should play important roles in water reclamation and resource recovery from brackish water, especially for inland cities or rural regions.

Keywords: electrokinetics; energy consumption; water productivity; brine utilization; carbon footprint; hybridization.

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1. Introduction

Due to rapid population growth and resource depletion, freshwater stress and scarcity are one of the most severe challenges around the world, especially in countries such as Saudi Arabia (Aljohani 2017), Jordan (Qasim et al. 2018), and Tunisia (Walha et al. 2007). Seawater (a salinity ~35 g/L) is considered an infinite water resource, and seawater desalination by reverse osmosis (RO) using semipermeable membranes has been practiced at commercial scales for decades at numerous countries such as Israel (Segal et al. 2018), Australia (Linge et al. 2013), Spain (Quevedo et al. 2011), and the US (Rao et al. 2018), and is believed to be the most optimized technology for seawater desalination (USDOE 2014). Existing seawater RO (SWRO) plants operate near the thermodynamic limit, where the applied pressure is only 10−20% higher than the osmotic pressure of the concentrate (Elimelech and Phillip 2011). The energy consumption of an industrial-scale SWRO is typically on the order of 2.5−4.0 kWh/m^3 (Shannon et al. 2008, Subramani et al. 2014, Yangali-Quintanilla et al. 2011), producing freshwater at an average cost of 0.5–1.2 USD/m^3 (Kim et al. 2010a, Nanda 2018). However, RO becomes less energy efficient for brackish water desalination due to the high energy consumption per ion (Amy et al. 2017).

Brackish water (salinity of 1−10 g/L (Elimelech and Phillip 2011)) is an essential water desalination alternative when compared to seawater, potentially enabling the movement of the water-energy nexus away from the seacoast further inland. This allows for production of clean water at lower operating cost and lower energy burden. Already, a shift to using brackish water as the water source has been sharply increasing in water-stressed regions, such as Egypt (Allam et al. 2002). Sources of brackish water include groundwater (naturally saline aquifers), rivers, wastewater (e.g., hydraulic fracturing, cooling water, human activities and industrial processes), and irrigation return flow. So far, brackish water desalination represents over 21% of the total worldwide desalination capacity (Jones et al. 2019). Compared to seawater desalination, most desalination plants using brackish water have much lower production capacity between 500 and 10,000 m^3 per day (Jaber and Ahmed 2004), even up to 76,000 m^3 per day in the US (Mickley 2001). Despite the relatively lower production capacity, the inland brackish water desalination plants can provide a vital solution to solve the water stress and scarcity in remoted areas.

Electrokinetic desalination technologies, such as capacitive deionization (CDI), have attracted great attention for water reuse due to their energy-efficient separation of ionized or ionizable species from solutions. CDI are broadly defined as a category of ion separation technologies, where electrodes are cyclically charged and discharged, regardless of deionization mechanisms or electrodes (Biesheuvel et al. 2017). It is noteworthy that non-cyclical characteristics have been achieved in recent CDI design, such as flow electrode CDI (He et al. 2018, Zhang et al. 2020). In CDI, charged ions are electrically separated through the formation of electrical double layers (EDLs) to produce "fit-for-purpose" water (Delgado et al. 2019, Pan et al. 2018a). As a result, the energy consumption in CDI correlates with the quantity of ions removed. CDI also could apply an electrical current to drive Faradaic reactions, such as redox (cathode-anode) and intercalation reactions (Zhang et al. 2018). This provides opportunities of simultaneous oxidation and separation for other contaminants in brackish water. However, CDI is typically not effective to remove uncharged organics and biological species, which is the biggest difference to RO.

As climate change and population increase continue well into the 21st century, water stress will continue to be a major concern and will thus drive increases in brackish water desalination. Technologies for brackish water desalination will consume a significant amount of energy, and the energy intensity of desalination facilities will affect the cost effectiveness and environmental impacts of the plant. To the best of our knowledge, little-to-none research has been focused on addressing the water and energy nexus for brackish water desalination using RO and CDI. In this article, we review the recent advances of the performance of RO and CDI for brackish water desalination from the energy, economic and environmental aspects. We first illustrate the criteria and the key performance indicators to evaluate the performance of brackish water desalination. Next, we systematically gather technological information of RO and CDI for brackish water desalination published in the recent years. A summary on the effect of key parameters on desalination performance, as well as energy-water efficiency, economic costs and environmental impacts are included. We provide in-depth discussion on the interconnectivity between desalination and energy, and discuss the trade-off between kinetics and energetics for RO and CDI. We also compare the results of technical-economic assessment for RO and CDI plants in the context of large-scale deployment, and discuss the engineering design, construction and operation from the lifetime-oriented considerations.

Finally, we illustrate the challenges and opportunities of desalination for brackish water.

2. Criteria and Key Performance Indicators for Brackish Water Desalination

The performance of a desalination facility depends on the types of the applied materials (e.g., electrodes and membranes) in desalination facilities and the operating conditions (e.g., flow rate, applied current and pressure). When determining the performance of a desalination facility, we should establish a standard evaluation procedure that first defines feed salinity (C_o) and required salinity removal ratio (η), and then measure three indicators that are largely related to the cost-effectiveness of a process: (i) specific energy consumption, (ii) water recovery ratio, and (iii) productivity of clean water from the processes. This consideration has been critically addressed and discussed in the review paper published by Hawks et al. (2019). Eq (1) gives the removal ratio of salinity (η) quantified by the fraction of ions removed through the desalination facilities.

$$\eta\ (\%) = \frac{C_o - C_i}{C_o} \times 100 \qquad (1)$$

where C_o and C_i are the salinity (g/L) of feed and effluent streams, respectively.

In this part, we briefly illustrate the definition of several key performance indicators, including the treatment capacity, water recovery, specific energy consumption, and operation and maintenance (O&M) costs. We also discuss the importance of the environmental impacts and carbon footprints when evaluating the performance and cost-effectiveness of a desalination facility.

2.1 Treatment Capacity and Recovery

Treatment capacity (Q_i) and water recovery (R) are essential to evaluating the cost effectiveness of a desalination plant. The R value of a desalination plant also directly determines the volume of the rejection brine, which requires subsequent treatment and management. The R value can be determined by eq (2):

$$R\ (\%) = \frac{Clean\ Water\ Production}{Treatment\ Capacity} = \frac{Q_o}{Q_i} \times 100 \quad (2)$$

where Q_i and Q_o are the flow rate of feed water and permeate (clean water), respectively. The R value is dependent on the ion removal mechanisms of a desalination facility, as well as the types of the applied materials (e.g., electrodes and membranes). Other factors, such as losses from flushing, can also influence the R value.

Additionally, the productivity of the desalination units is a metric of evaluation that is frequently determined and compared. Usually, it is directly proportional to the size (land footprint) of the desalination plant, and can be evaluated in a unit of the amount of clean water (m^3) per m^2 of membrane per treatment time. In RO processes, the concentrate factor (CF) of the concentrate stream is a useful indicator to the overall concentrate salinity. CF is a function of the removal ratio of salinity (η) and the water recovery, as shown in eq (3):

$$\mathrm{CF} = \left(\frac{1}{1-R}\right)[1 - R(1-\eta)] \quad (3)$$

CF also represents the ratio of the concentrate salinity to the feed salinity. CF increases exponentially as recovery increases: when recovery increases from 70% to 90%, the CF increases dramatically from 3.3 to 9.9 (assumed a 99% salt removal ratio). This reveals several potential issues primarily found in brackish water desalination using membrane processes, such as (i) precipitation of sparingly soluble salts (e.g., SiO_2, $CaCO_3$, $CaSO_4$, and $BaSO_4$), which could cause scaling and fouling, as well as (ii) the costs of subsequent brine disposal. Thus, this foreshadows the need for alternative brackish water desalination processes.

2.2 Specific Energy Consumption (SEC) of a Desalination Process

Specific energy consumption (SEC, kWh/m^3) is the electrical energy used to product a unit of clean product water. It is considered as the most important parameter characterizing the performance of the desalination plant. The SEC of a desalination process can be approximately determined via eq (4):

$$SEC = \frac{Total\ Electricity\ Consumption}{Clean\ Water\ Production} \tag{4}$$

The total electricity consumption of a desalination process could be attributed to the use of high-pressure pumps (for RO), the use of external electrical energy (for CDI), and the water pumps (for CDI). For cyclical operations of most CDI units, the desorption (discharge) step should be taken into consideration when calculating SEC, as shown in eq (5):

$$SEC = \frac{\int_0^{t_{TOT}} I \times V \, \mathrm{d}t}{Q_i \times t_c \times R} \tag{5}$$

where V is the applied voltage (e.g., the range of voltage in the desorption process for CDI), I is the current, t_{TOT} is the total operation time, including adsorption and desorption time, and t_{c} is the treatment time (e.g., adsorption time in the charging phase for CDI). Hawks et al. (2019) have proposed a framework for systematically quantifying the performance of CDI, in terms of SEC and throughput productivity. When evalauting different desaliantion facilities (e.g., RO and CDI), the use of throughput productivity, instead of an average removal rate such as average salt adsorption rate (ASAR), is much meaningful.

2.3 Operation and Maintenance (O&M) Costs

Total costs for producing clean water include capital (e.g., installed process equipment, civil works, buildings), operation (e.g., energy and chemicals), maintenance (e.g., membrane and electrode replacement), labor, and miscellaneous (e.g., insurance, interest, and project management). Operations of the various parts of a desalination plant, including (i) the feed water intake, (ii) pretreatment, (iii) the main desalination unit, (iv) post-treatment of clean water and (v) brine disposal facility, would contribute to the total costs of a desalination plant. The operation and maintenance costs ($C_{\mathrm{O\&M}}$) are running expenses and largely depend on the design of the main desalination unit, which can be calculated by eq (6).

$$C_{O\&M} = C_{en} + [C_{cc} + C_{mem}] \tag{6}$$

where C_{en}, C_{cc} and C_{mem} are the costs of electrical energy (denoted as the "water cost" in this paper), chemical cleaning and membrane replacement, respectively. The cost of electrical energy can fluctuate and become complex since the price of electricity normally changes from year to year. Additionally, electrical energy tariffs can lead to sharp increases in operation costs. Maintenance costs are generally associated with membranes and membrane-related materials. If membrane replacement is periodically carried out, the costs of personnel, chemicals (e.g., antiscalants), cartridge filter will be fairly constant over the years (Ruiz-García and Ruiz-Saavedra 2015). It is noteworthy that we use the term "water costs" in this paper, to express only the operation costs due to the use of electrical energy, such as pumps and external power.

2.4 Environmental Impacts and Carbon Footprints

The environmental impacts and carbon footprints of a desalination plant can be determined by conducting life cycle assessment (LCA). LCA determines the environmental impacts of a product or a technology throughout the entire life-cycle, i.e., from the design, materials extraction, manufacture, distribution, use and final disposal (end-of-life). As suggested by ISO 14040–14044 (International Organization for Standardization 2006a, b), LCA includes four steps, viz goal and scope definition, data inventory, life-cycle impact assessment, and interpretation. **Fig. 1** illustrates the assessment of environmental impacts and carbon footprints for a desalination plant. The scope for LCA could be generally cradle-to-gate, cradle-to-grave, or cradle-to-cradle (Lior 2017), where cradle, gate and grave usually represent creation (resource extraction), factory gate and disposal phase, respectively. In this stage, the functional unit of a desalination plant also should be clearly defined, typically "one m^3 of freshwater produced from the plant." For the stage of data inventory, all inputs (e.g., energy and chemical uses) and outputs (e.g., freshwater, brine, and other pollutants) within the scope of LCA should be quantified.

<Figure 1>

With a sound data inventory, the environmental impacts of a desalination facility can be determined by various types of methodologies, e.g., ReCiPe (Goedkoop et al. 2009). For a desalination facility, the energy intensity or carbon footprint (categorized as global warming potential in LCA) is extremely important to address the water-energy nexus. Other key environmental impact categories include land occupation (so-called land footprint), natural resources depletion (resource footprint), and ecosystem quality (ecological footprint). To the best of our knowledge, the environmental (and/or ecological) impacts determined from LCA have been combined with different methods, such as techno-economic analysis (Kim et al. 2018) and exergy analysis (Blanco-Marigorta et al. 2014), to provide an insight of system optimization for desalination systems.

2.5 Summary

The specific energy consumption of processes directly determines the operation costs, while the water recovery ratio and the productivity of clean water are largely related to the post-treatment cost of the brine stream and the land costs (land footprint), respectively. The determination of key performance indicators for the cost-effectiveness of RO and CDI should be based on the same removal ratio of salinity. For instance, in the case of a feed salinity of 5.0 g/L (brackish water), a removal ratio of salinity above 90% is required to meet the standard of clean water for human consumption (salinity < 0.5 g/L) (Australian Government 2004). For other purposes, the salinity for treatment to the irrigation water standard should below 0.9 g/L (Drewes et al. 2009). In other words, comparison between RO and CDI must be based on the feed salinity, removal ratio, throughput productivity, and water recovery ratio.

3. Performance Evaluation of RO for Brackish Water Desalination

RO typically operates using multiple stages (the concentrate of one is the feed of the next) or passes (the permeate of one is the feed of the next) in series. The system design of brackish water reverse osmosis (BWRO) is conceptually different from that of seawater reverse osmosis (SWRO). BWRO usually operates using multiple stages (Blair et al. 2017, Wei et al. 2017), while SWRO using multiple passes (Greenlee et al. 2009). The choice of the number of passes or stages depends on several parameters, such as feed water quality, energy cost and desired recovery ratio. Typically, a two-stage configuration is used for BWRO to achieve high recovery ratios with substantial energy savings: The water recovery of each stage in BWRO is above 50%, thereby leading to an overall water recovery of over 70% (Greenlee et al. 2009). In this part, we compile the recent advances of RO processes for brackish water desalination.

3.1 Effect of Membranes and Configurations on Desalination Performance

Membranes are responsible for rejecting inorganic components from feed stream to another compartment. Table S1 (see the Supplementary Information) presents the characteristics and operating conditions range of different membranes for BWRO. Thin-film composite membranes using aromatic polyamide as selective layer are the most prevalent membranes in commercial RO plants due to their high salt rejection, water permeability, and chemical/thermal/mechanical/biological stability, as well as their wide operation temperature and pH ranges. Asadollahi et al. (2017) provided a comprehensive review on the performance of commercial polyamide thin-film composite RO membranes, and then evaluated the effect of membrane fouling and chlorine attack on their performance. Commercial polyamide thin-film composite membranes exhibit water permeability around 0.001–0.004 $m^3\ m^{-2}\ bar^{-1}\ hr^{-1}$ at salt rejections of 98.0–99.5% for brackish water feed (a salinity of 2.0 g/L). Similarly, Otitoju et al. (2018) conducted an overview on the technology progress of interfacial polymerization and surface modification for RO membranes. They pointed out that the structure–property relationships and kinetic performance of composite membranes should be future research directions to explore broader niche applications.

Recently, various types of new generation membranes for desalination, such as nanocomposite

(Kurth et al. 2014), cellulose nanocrystals (Asempour et al. 2018), aquaporin (Wang et al. 2012), nanotube (Yang et al. 2013), and graphene-based (Cheng et al. 2016, Joshi et al. 2014) membranes, have been developed. These innovative RO membranes can generally provide beneficial advances including (i) low fouling, (ii) enhanced boron-rejection, (iii) improved thermal-stability and mechanical properties, and (iv) inorganic-organic nanocomposites with purported higher permeability (Alvarez et al. 2018, Amy et al. 2017, Otitoju et al. 2018). In RO processes, the production capacity (i.e., permeate flux and water recovery) is highly dependent on the control of membrane fouling. Despite the rapid development of core membrane technology and innovative system design, membrane scaling and fouling still inhibit RO to achieve high recovery greater than 95% (Greenlee et al. 2009). To effectively control membrane fouling, several good engineering practices on feedwater properties should be followed, including, perhaps most importantly, keeping the turbidity of feed less than 0.2 NTU (Wilf and Bartels 2006). To provide effective biofouling control in RO desalination, Al-Abri et al. (2019) also analyzed the feasibility of alternatives to chlorination, such as ozone, ultraviolet and nano-photocatalytic materials. These alternatives could effectively mitigate the degradation of RO membranes due to the attack by chlorines. Especially for the ozone, the pH of the solution should be carefully controlled as the self-decay rate of ozone is relatively low at the acidic condition, which might attack the RO membrane.

An alternative to increasing the overall water recovery is to pretreat the feed stream (e.g., through compact accelerated precipitation softening) (Oren et al. 2001) or treat the concentrated brine (Qu et al. 2009). With these side treatment units, the overall water recovery of BWRO theoretically could increase to 93−98% (Gabelich et al. 2007, McCool et al. 2013). However, additional pretreatment or post-treatment processes will increase the overall energy consumption. Qiblawey et al. (2011) found that the introduction of a softener as a pretreatment prior to the primary BWRO unit increases the energy consumption from 1.9 to 13.8 kWh/m^3, illustrating the delicate balance between water recovery and energy consumption in BWRO.

3.2 Energy Consumption and Throughput

Table 1 compiles the technology information of BWRO plants around the world, pertaining to the operation capacity, water recovery, energy consumption and costs. BWRO has been practiced at large scales for decades at numerous countries, such as Israel, Spain, Tunisia, the US and the UK. The typical energy intensity of BWRO processes is approximately 0.8–2.5 kWh/m^3 (as shown in **Fig. 2**), depending on the salinity of feedwater and effluent requirement. The primary energy use in BWRO is for the initial pressurization (pumping) of the feed, which is relevant to the desired pressure and flow rate. Karabelas et al. (2018) found that the incorporation of energy recovery device (ERD) could further lead to achievable energy consumption as low as 0.4−0.7 kWh/m^3 for BWRO. The use of ERD can effectively reduce the SEC and thus the operating cost for desalination. In general, ERD utilizes and converts the remaining pressure of the brine into the forms of mechanical energy via either turbine systems or pressure exchanges. The efficiencies of energy recovery for turbine systems and pressure exchanges are typically ~90% (Aljundi 2009) and 96−98% (Fritzmann et al. 2007), respectively. Alghoul et al. (2009) observed that a small-scale RO without ERD would consume 2−3 times more energy, compared to that with ERD. Kim et al. (2019) also conducted a critical review on the performance of ERD, and found that the pressure exchanger should be the most efficient ERD with a reported efficiency of over 95%.

<Table 1>

<Figure 2>

In addition to ERD, numerous studies had been conducted to evaluate the feasibility and stability of renewable energy assisted (e.g., PV-powered) BWRO systems, especially in remote areas. The durability of renewable energy assisted systems to supply the load without any significant disturbances has been demonstrated in many places, such as PV-powered RO in Malaysia (Alghoul et al. 2016). Khan et al. (2018) also critically reviewed the current status of renewable energy assisted systems, and evaluated the effect of energy supply profiles on the economics of RO plants. The analyses indicated that the water costs of PV-powered and wind-powered BWRO systems were approximately 0.9−8.5 (with water production capacities of 1−2,000 m^3/d) and 0.7−1.7 (with capacities of 22−3,720 m^3/d) USD/m^3, respectively. Therefore, BWRO should play an important role in water reclamation from brackish water for inland cities or rural regions.

System capacities for BWRO range from less than 0.4 m^3 per d for prototype units and up to 700,000 m^3 per d for full-scale plants. RO recovery varies from 35% to 85%, depending on feedwater properties, type of pretreatment and post-treatment, and concentrated brine disposal options. The water recovery (throughput) directly affects the size of plants and the operating costs (energy consumption), as well as the costs of brine treatment. Sarai Atab et al. (2018) found that the production cost would linearly increase with the increase of feed salinity (ranging from 8,000−14,000 mg/L). The concentration of rejected brine from SWRO also increases with the increase of feed salinity, as well as applied pressure and feedwater temperature (Sarai Atab et al. 2018).

3.3 Costs Estimation

For BWRO plants, both the production capacity and the energy source play major roles in the cost of produced water (USD/m^3). In this section, we compile the recent advances on the cost estimation for a BWRO plant, in terms of plant sizes, energy sources, membrane replacement and total costs.

3.3.1. Plant sizes

Table 2 presents the water production cost of the main BWRO processes, in terms of plant sizes. According to a survey of six BWRO plants in Texas (4,500–104,000 m^3/d), the total production cost of water ranges between 0.29 and 0.63 USD/m^3 product, with approximately 40–70% of costs attributed to O&M (Arroyo and Shirazi 2012). For a smaller size BWRO plant, the costs of water with a plant capacity of 20–1,200 m^3/d range from 0.78 to 1.33 USD/m^3, while that for a plant capacity less than 20 m^3/d are even higher than 5.66 USD/m^3 (Karagiannis and Soldatos 2008). In the case of a plant capacity of 30–100 m^3/d, the O&M costs are contributed to about 60–93% in the entire costs (Jaber and Ahmed 2004).

<Table 2>

3.3.2. Energy sources

In additional to fossil-fuel-based electricity, the supply of energy for BWRO desalination can come from various sources, such as solar, wind, geothermal and ocean energy. Ghermandi and Messalem (2009) critically evaluated a total of 79 solar-powered BWRO desalination units worldwide. They found that the water costs of PV-powered BWRO systems with ERDs were in the range of 2.3−35.9 USD/m^3, while the costs of hybrid solar (with fuel or wind power) systems were in the range of 0.9−31.8 USD/m^3 (Ghermandi and Messalem 2009). The hybrid solar systems were beneficial for BWRO since the complementary aspects of two energy sources could be exploited. Moreover, instead of using batteries (higher costs) with PV to drive SWRO, it is recommended to implement battery-less PV systems, such as direct connection of PV-DC motor, using supercapacitor as electrical regulator, or using controlled DC/DC converter (Shalaby 2017).

3.3.3. Membrane replacement

The operating costs of RO will increase due to the frequent replacement of the membranes. In general, McCool et al. (2013) suggested the average life time of 3 and 10 years for RO and MF elements, respectively. **Table 3** compares the membrane lifespan and replacement costs of different RO processes for brackish water desalination. Shaaban and Yahya (2017) found that the permeate flux of brackish water membranes are more affected by the temperature of feedwater rather than the feed pressure and concentration. For BWRO, Drewes et al. (2009) assumed a 7-year membrane lifetime and suggested that the cost of membrane replacement should be in the range of 0.012 to 0.015 USD/m^3. Similarly, Greenlee et al. (2009) suggested a membrane replacement rate of 5% per year (every 5−7 years) with the permeate flux of 12−45 L per m^2 per h (i.e., hydrostatic pressure of 600−3000 kPa) at a water recovery of 75−90%.

<Table 3>

Pretreatment of feedwater by low-pressure membrane filtration (e.g., ultrafiltration) could reduce the long-term costs since it would increase the lifespan of the RO membrane by 20−30% (Jamaly et al. 2014). Pearce (2008) found that, using the filtration processes as a pretreatment, the cleaning frequency would be reduced to twice (or even once) per year. Similarly, Ruiz-García et al. (2018) highlighted that, with an appropriate conventional pretreatment, it could preserve BWRO membrane (e.g., BW30-400) elements in service for up to 11 years. In their another study (Ruiz-García and Ruiz-Saavedra 2015), they found that the most cost-effective scenario should be the operation for the first ten years without replacing membranes, even considering new generation membranes (e.g., ECO-440i DOW™). However, in this case, the chemical cleaning would be more often after approximately 70,000 h of operation.

3.3.4. Total costs

In the above sections, we provide the information on costs estimation of BWRO plants, in terms of plant sizes, potential energy sources, and membrane replacement. Here, we compile the total plant costs for BWRO plants (**Table 4**). Normally, the water cost of BWRO should be generally less than that of SWRO, even though the treatment capacity in the BWRO plant is significantly lower than that in SWRO (Almulla et al. 2003).

<Table 4>

3.4 Carbon Footprint and Life-cycle Environmental Impacts

The variability in life-cycle stages considered and methodology, as well as the location and operational parameters of RO plants would result in a significant range of carbon footprint. **Table 5** presents the carbon footprint of BWRO plants according to the analyses from recent studies. Cornejo et al. (2014) conducted an overview on carbon footprint of desalination technologies, and they found that the BWRO plants (0.4–2.5 kg CO_2-eq/m^3) generally have a lower carbon footprint than SWRO plants (0.4–6.7 kg CO_2-eq/m^3). Similar analysis by LCA also indicated that the carbon footprint of BWRO plants (0.84–1.60 kg CO_2-eq/m^3) is lower than that of SWRO plants (1.54–2.81 kg CO_2-eq/m^3) (Muñoz and Fernández-Alba 2008). Zhou et al. (2011) examined the life-cycle environmental impacts of desalination for high-salinity brackish water (~15 g/L) with a plant capacity of 10,000 m^3/d and an average electricity consumption of ~2.0 kWh/m^3 from different approaches of impact assessment characterization. The considered life-cycle stages included the infrastructure (e.g., construction and land preparation), operational (e.g., chemicals, membranes and electricity), and dismantling (e.g., used construction materials and membranes) phases. The system boundary of LCA excludes both the pretreatment of raw water and brine treatment. The results indicated that the global warming potential for producing 1 m^3 of pure water was ~1.58 kg CO_2-eq (Zhou et al. 2011).

<Table 5>

The electricity consumption of a BWRO plant is responsible for more than 90% of the contribution in environmental impact categories (Muñoz and Fernández-Alba 2008, Tarnacki et al. 2012). Therefore, the types of energy sources for desalination are crucial to the environmental impacts as well as carbon footprint. Raluy et al. (2005) noticed that the combination of BWRO with renewable energies could result in a significant decrease in the airborne emissions, such as CO_2 (an average of 80% reduction) and SO_x (an average of 60% reduction). Similarly, the global warming potential of BWRO could decrease by 98% if the desalination plant is integrated with wind power, instead of using electricity from the Spanish grid mix (Tarnacki et al. 2012).

3.5 Summary

BWRO is a mature technology which has been widely deployed for decades around the world. According to our analysis based on the recent literature, the energy intensity of BWRO processes is approximately 0.8–2.5 kWh/m^3, depending on the salinity of feedwater and effluent requirement. A high water recovery of over 90% could be achieved by BWRO. The energy consumption of BWRO is responsible for >90% of the contribution in life-cycle environmental impacts, and the primary energy use in BWRO is for the initial pressurization of the feed. The incorporation of energy recovery devices could further lead to achievable energy consumption as low as 0.4−0.7 kWh/m^3. For inland cities or rural regions, BWRO powered by renewable energy could provide a solution to water reclamation. Appropriate pretreatment of feedwater by low-pressure membrane filtration could effectively increase the lifespan of the RO membrane by 20−30%, and thus reduce the long-term costs.

4. Performance Evaluation of CDI for Brackish Water Desalination

CDI can be generally described as the cyclic processes of electro-adsorption (known as a “charge” stage) and electro-desorption (known as a “discharge” stage) using porous or high surface area electrodes through the formation of EDLs. CDI is particularly effective for treating non-traditional waters of relatively low ionic strength since it operates at ambient conditions without the needs of extensive chemicals use. In this part, we compile the recent advances of CDI processes for brackish water desalination.

4.1 Effect of Electrodes and Configurations on Desalination Performance

CDI typically relies on the formation of EDLs to store charges onto electrodes. The design, synthesis and fabrication of electrode materials are the key in developing CDI processes. Critical reviews on fabrication (Jia and Zhang 2016), electrosorption behavior (Huang et al. 2017), and surface modifications (Ahmed and Tewari 2018) of various CDI electrodes, as well as their applications in capacitive technologies (Ratajczak et al. 2019) have been published. Pan et al. (2018a) also conducted an overview of various electrode materials, including activated carbon composite, flow suspension electrode (e.g., AC/MnO_2 suspension) and battery electrode ($Na_4Mn_9O_{18}$), for water desalination. Here we provide a brief summary on the performance of different carbon-based electrodes in CDI from recent studies, as presented in Table S2 (see Supplementary Information). Most of the CDI electrodes are carbon materials (i.e., mainly composed by element carbon), such as activated carbon (Hu et al. 2018), carbon aerogel (Zhu et al. 2018), ordered mesoporous carbon (Chen et al. 2018), activated carbon cloth and nanotubes (Li and Park 2018), graphene family (e.g., graphene sponge (Xu et al. 2015) and graphene hydrogel (Ma et al. 2018)), and carbon composite (Nie et al. 2012), Carbon electrodes are well polarizable and typically with high specific surface area; however, their electrical conductivity strongly depends on the thermal treatment, microtexture, hybridization and content of heteroatoms. As presented in Table S2 (see Supplementary Information), the salt adsorption capacity (SAC) of carbon-based electrodes in conventional CDI varies between 5.0 and 49.3 mg g^{-1}. Physico-chemical properties of electrode materials, such as specific surface area (SSA), mean pore diameter (D_p), carbon graphitization degree (I_D/I_G) and charge-transfer resistance (R_{ct}), play important roles in SAC and capacitance, especially for the application of CDI. For instance, previous studies have reported that a higher I_D/I_G value should be beneficial to the charge transfer in the adsorption process (Nie et al. 2012). The R_{ct} value of electrodes reflects the fact of charge transfer in the adsorption process. To determine the impedance information on carbon materials, a Randles circuit model (Roberts and Slade 2010), considering R_{ct}, series resistance (R_s), Warburg open diffusion resistance (W_O), and constant-phase element (Q_1), can be applied. **Fig. 3** illustrates the performance of different carbon-based electrodes in CDI, in terms of SAC (mg/g) and deionization rate (mg/g/s). A larger SAC of electrodes generally represents a smaller size of CDI stack (land footprint as one of the key technical-economic measures), while a greater deionization rate implies a high productivity of clean water. In fact, the ragone plot sometimes can not provide a comprehensive comparison among different electrodes. The SAC, which is highly specific to CDI systems, should be corresponded to technical-economic measures, while the deionization rate should scale with the actual productivity ($L/hr/m^2$) of CDI systems (Hawks et al. 2019). In addition to carbon-based electrodes, various types of CDI electrodes, such as porous silicate network (Metke et al. 2016) and battery electrodes (Ahn et al. 2020), have been developed. For instance, the use of battery electrodes could significantly improve the adsorption capacity of CDI; for instance, the Ag/AgCl electrodes provide an SAC of up to 85 mg g^{-1} at a low voltage of 0.2 V (Ahn et al. 2020). However, efforts towards improving the long-term stability of battery electrodes (e.g., used in real brackish water) should be emphasized.

<Figure 3>

Design and configurations of CDI cells also play crucial roles in desalination performance. Aside from the conventional CDI, novel architectures on cell and/or electrode designs (**Fig. 4**), such as wire-shaped electrode CDI (Mubita et al. 2018), membrane CDI (Lee et al. 2006, Qian et al. 2015), rocking-chair CDI (Lee et al. 2018), flow-electrode CDI (Jeon et al. 2013), flow-through electrode CDI (Suss et al. 2012), flow-by electrode CDI (Hemmatifar et al. 2015), pressurized CDI (Caudill 2018), inverted CDI (Gao et al. 2015), honeycomb-shaped CDI (Cho et al. 2017), and even desalination batteries/generators (Chen et al. 2017, Pasta et al. 2012, Suss et al. 2015b), have been developed. These novel approaches aim to provide a higher salt adsorption capacity and charge efficiency with stable cycling, even avoiding separate regeneration step. For instance, in membrane CDI (MCDI), ion exchange membranes are adjacent to the surface of the electrodes in a conventional CDI cell for avoiding the co-ion and ping-pong effects, and thus improving the charge efficiency and electrode life. Omosebi et al. (2017) further modified MCDI cell using asymmetric electrodes that consist of a pristine anode and oxidized cathode, together with a single anion exchange membrane. The modified system exhibited a high salt adsorption capacity of 16.6 mg g^{-1}, and cut nearly half the cost of conventional MCDI. Other novel architectures, such as flow-through electrode, are trying to reduce hydraulic resistance by using high surface area electrodes with a hierarchical and porous structure.

<Figure 4>

Membrane fouling by inorganic scaling, organics, colloids, and biomass would be a challenge in membrane CDI (Chang et al. 2002). Brackish waters are known to contain numerous inorganics, such as calcium carbonate and ferric ions, as major contributor to scaling (Mossad and Zou 2013). Wang et al. (2019b) indicated that brackish water containing ferric ions would cause a significant decrease in CDI performance, where Fe_2O_3 precipitate was found to be the predominant foulants. Research with natural brackish waters shows mixed results for the role of organics on the performance of membrane CDI (Kalfa et al. 2020, Suss et al. 2015a). For instance, Gabelich et al. (2002) found that the organic matter in river water reduced the sorption capacity of electrodes (carbon aerogel electrodes). Zhang et al. (2013) also showed similar reduction in efficiency with organic content. Wang et al. (2019b) suggested that the presence of natural organic matter would alleviate the ferric-species scaling, thereby decreasing the salt adsorption capacity. However, Lee et al. (2006) found no reduction in performance of membrane CDI cell over 500 desalination cycles. Similarly, Xu et al. (2008a) found no significant fouling over several hours of desalination of brackish water from a natural gas generation site, using a carbon aerogel type CDI, but the water samples had oils and grease in low concentrations. Kim et al. (2010b) indicated that brackish water containing 5–10 mg L^{-1} of oils (e.g., octane) did not affect the performance of CDI. Therefore, future research is needed to fundamentally understand the effect of chemical composition and organic material concentration, hardness, and other water properties on the fouling of membrane CDI. Available solutions to fouling in membrane CDI include pretreatment of membranes (Mikhaylin and Bazinet 2016, Suss et al. 2015a), and using modification methods, such as powdered/biologically activated carbon (Ng et al. 2010), nanocomposites from reduced graphene oxide with TiO_2 catalysts (Zhang and Jia 2015), cellulose-derived graphenic nanosheets (Pugazhenthiran et al. 2015), and starch-derived porous carbon nano-sheets (Wu et al. 2017b).

4.2 Energy Consumption and Throughput

Both cell designs (e.g., type of water flow and thickness of channels and pair spacers) and operating conditions exhibit a remarkable effect on energy consumption and clean water throughput. **Table 6** compiles the performance of different designs of CDI, in terms of energy consumption and clean water productivity, from the literature. To minimize Faradaic reactions, CDI usually operates under a voltage lower than 1.2 V (Cai et al. 2017, Li et al. 2011). The typical energy intensity of conventional CDI is approximately 0.1–1.5 kWh/m^3, depending on the salinity of feedwater and effluent requirement. Voltea B.V., a company based in the Netherlands, has successfully developed a so-called CapDI technology to deionize water with moderate salinity of < 4.0 g L^{-1} for industrial and commercial applications. Voltea’s Industrial System (IS12) can produce purified water at a capacity of 100 L min^{-1} with the energy consumption of only 0.1–0.2 kWh m^{-3} (Voltea 2015). Since 2009, a large-scale CDI facility with a treatment capacity of 3600 m^3 d^{-1} has been deployed by EST Corp. (China) for treatment of cold-rolling wastewater at a treatment cost of 0.069 USD m^{-3} (EST 2009). Other novel CDI architectures, such as membrane CDI (MCDI) and flow-electrode CDI (FCDI), could effectively reduce the energy intensity, compared to conventional CDI. For MCDI, when treating brackish water with a salinity of 1.10–4.65 g L^{-1}, it required 0.17–3.45 kWh m^{-3} to recover

50% of the feed at a permeate salinity of 0.5 g L^{-1} (Zhao et al. 2013). Tan et al. (2020) also evaluated the performance of pilot-scale MCDI, and found it required around 0.36 kWh m^{-3} to desalinate the brackish water with a salinity of 1.9 g L^{-1} down to 1.2 g L^{-1} at a water recovery of ~49%. Several studies (Zhao et al. 2013) revealed that the pumping energy in CDI/MCDI is rather small to the total energy consumption, ranging from ~0.2−3.0% of total energy consumption.

<Table 6>

The kinetics of deionization (adsorption-desorption) are the key factors determining the energy consumption of CDI technologies. Alencherry et al. (2017) found that increasing the electrical conductivity and hydrophilicity of electrodes significantly enhances the deionization rate and kinetics of CDI. However, the kinetics of electro-adsorption during desalination step were found to be independent of the thickness of AC electrodes in CDI, while slow desorption kinetics during regeneration were observed for thicker electrodes (Zornitta et al. 2016). To standardize the performance metrics for CDI, Suss et al. (2015b) recommended the use of the total cycle time (the duration of both charging and discharging steps) for determining the deionization (salt adsorption) rate of the static electrode CDI (inherently a two-stage process).

In CDI, the energy efficiency largely depends on the circuit resistance, including electric resistance of electrode (transport losses of electric charges) and ionic resistance of feed streams (transport losses of ionic charges). Qu et al. (2015) found that, in a CDI system, the major circuit resistance should be attributed to the contact resistance between electrodes and current collectors. In the case of MCDI, Dykstra et al. (2016) found that the main resistance is from the spacer channel and the external electrical circuit, rather than the electrodes. In other words, the thickness of carbon electrodes (i.e., ~260 μm) could further increase without a significant increase of energy consumption. However, this practice would result in an increase in the capital cost. The corresponding energy consumption was about 6.1 kJ per mole of salt removed (Dykstra et al. 2016). In addition to the system resistance, the improvement of electrode materials, and thus, their capacitance could increase the energy efficiency of CDI. However, Qin et al. (2019) found that further increases of capacitance above 300 F g^{-1} may only have little gain in overall energy efficiency. For instance, with the significant increases of electrode capacitance from 300 to 1,000 F g^{-1}, the energy efficiency of CDI slightly increases from 3.5% to 6.2% in the case of a water flux of 10 L m^{-2} h^{-1}. In particular, as the water flux increases, the effect of electrode capacitance on energy efficiency of CDI becomes less remarkable.

4.3 Costs Estimation

In comparison to RO, CDI technologies are relatively new and the information on cost estimation for large-scale operations is not widely available. In this section, we compile the recent advances on the costs estimation for a CDI plant, in terms of energy consumption, electrode/membrane replacement and total costs.

4.3.1. Energy consumption

As aforementioned, the energy consumption of a CDI process depends on the circuit resistance. In other words, the salinity of the feedwater exhibits a significant influence on the energy consumption of CDI. Extensive studies have also reported that deploying an energy-recovering device, such as a supercapacitor (Chen et al. 2019) and a converter-battery system (Tan et al. 2020), with CDI during regeneration could compensate the energy consumption required for desalination. For instance, Tan et al. (2020) found that incorporation of energy recovery system with membrane CDI could reduce the total energy consumption by 30–40%, with a water recovery of ~87%. It is noted that methods to avoid polarization could effectively minimize energy losses during regeneration; for instance, using a long regeneration time (slow discharge). However, this might also increase the operating cost and complexity of the CDI design. For MCDI, to facilitate kinetics while increasing water throughput, the reverse polarity by consuming nominal energy during regeneration has been largely applied (Weinstein and Dash 2013).

4.3.2. Electrodes and membranes

The manufacturing cost and scalability of the electrodes are the key factors to the total costs of CDI systems. Biomass, such as sugar cane bagasse (Zornitta et al. 2018), could be utilized as low-cost carbon precursor for the synthesis of porous electrodes. For maintenance, Table S4 (see Supplementary Information) presents the electrode lifespan and the associated water recovery for

CDI and MCDI. According to the literature, the cost of carbon electrodes ranges between 4 to 50 USD per kg (Omosebi et al. 2017, Zuo et al. 2008), depending on purity and sophistication. Caudill (2018) identified that an improvement of electrode lifespan to more than 1,000 hours could provide a drastic reduction on operating costs. For long-term operation, anodic oxidation due to undesirable Faradaic reactions is a critical issue since it leads to electrode deterioration and declined performance. Zhang et al. (2018) proposed several strategies, such as developing novel electrodes and deploying alternative cell configurations (e.g., MCDI) or operations, to mitigate the electrode deterioration by Faradaic reactions.

For MCDI, the costs for cell replacement were the largest contributor (81%) to the total O&M costs, while the costs for maintenance, energy, chemicals and wastewater treatment were all below 10% (Huyskens et al. 2015). This was attributed to the high capital cost for MCDI cell with the assumed short cell lifetime of about 2 years, especially for biomass hydrolysates applications (Sata 2004). For the membrane cost, in practice, membrane material is directly coated onto the electrodes, which is much cheaper than using free-standing membranes. The cost of ion exchange coatings/polymers can be lower than 100 USD per m^2, compared to the higher costs of UF (350 USD/m^2) or Nafion (1,400 USD/m^2) membranes (Zuo et al. 2008).

4.3.3. Total costs

The major costs of a CDI stack usually involve capital, electricity-based operational (e.g., pumps and external power), and replacement of electrodes and membranes. **Table 7** summarizes the cost breakdown for a CDI/MCDI plant. For MCDI, the cell cost was found to be the largest contributor (~58%) of the total capital costs, while the maintenance costs were typically assumed to be 5% of equipment costs (Huyskens et al. 2015). Metzger et al. (2020) also reported similar findings that the costs of ion exchange membranes shared around ~80% of the total capital costs for MCDI. Caudill (2018) successfully developed a pressurized CDI (PCDI) to increase the lifetime of electrodes by ~87%, compared to conventional non-pressurized CDI. If prolonging the lifetime of electrodes from 500 hours to 2000 hours, the total O&M costs would significantly reduce from 1.70−1.80 to 0.50−0.55 USD/m^3 (Caudill 2018). For PCDI, although the electricity cost for pumps increases for the sake of maintaining an additional 60 psig, the cost saving from the prolonged lifespan of electrodes far outweighs the cost incurred by the electricity cost.

<Table 7>

4.4 Carbon Footprint and Life-cycle Environmental Impacts

Only few studies on LCA have been conducted to determine the life-cycle environmental impacts of CDI technologies. For instance, Yu et al. (2016) determined the environmental impacts of CDI equipped with activated carbon/carbon black electrodes for brackish water (i.e., salinity of 0.584 g/L) desalination. They found the carbon footprint of CDI was about 1.43 kg CO_2-eq per m^3 of clean water, where 56.6% of the carbon footprint was attributed to the electrode materials, and the remaining 23.1% and 21.0% were from the energy consumption and chemical components, respectively. Shiu et al. (2019) evaluated the environmental impacts of different designs and scales of CDI techniques, including MCDI, using LCA. They found that material and chemical usages had the greatest overall impact (52–90%), while electricity consumption exhibited a relatively lower impact (as low as 9.7%). Also, the change of CDI housing materials from aluminum plates to plastic castings could effectively decrease the overall environmental impacts. A recent analysis by Metzger et al. (2020) estimated the amount of CO_2 emission for BWRO and CDI-based technologies. They found that, in the case of Middle East and North Africa, a transition from BWRO to CDI-based technology would reduce approximately 130 tons of CO_2 emissions per day for desalinating the brackish water at a salinity of 3.0 g/L.

4.5 Summary

CDI is a relatively new technologies in the field of brackish water desalination. To the best of our knowledge, only a few large-scale CDI plants have been reported, and most available studies are conducted at a lab scale. The types of electrodes and configurations play important roles in the performance of CDI. According to our analysis on the recent literature, the typical energy intensity of conventional CDI is approximately 0.1–1.5 kWh/m^3, depending on the salinity of feedwater and effluent requirement. Other novel CDI architectures, such as MCDI and FCDI, could further reduce the energy intensity of desalination, compared to conventional CDI. The water recovery of CDI systems is generally around 70–90%. Studies on the LCA of brackish water desalination using CDI are still limited. The results from available studies indicate that the manufacturing of the electrode materials and the use of chemicals should be the major contributor in life-cycle environmental impacts.

5. Trade-off between Kinetics and Energetics

The importance and significance of the energy consumption (energetics) and productivity (kinetics) can be illustrated from the thermodynamic perspective (Pan et al. 2018a). The minimum energy requirement for desalination, regardless of the salt removal mechanism, generally increases with the water recovery ratio according to the thermodynamics. A superior energetics (low energy consumption) results in a low operation cost; however, a slow kinetics (low productivity) would lead to a huge reactor size, thereby increasing the capital and land costs. Thus, for system optimization of brackish water desalination, there should be a trade-off between kinetic and energetic efficiencies. **Fig. 5** illustrates a trade-off between capital costs and energy consumption at a given production rate of fresh water for practical desalination systems from the thermodynamic point of view. The best design of processes for achieving the minimum costs is not necessarily the most energy efficient design. Systems operating with perfect energy efficiency (thermodynamic reversibility) represents a very slow frictionless event (an ideal system), thereby requiring the largest making resources (i.e., capital costs) (Spiegler and El-Sayed 2001). As the systems depart from ideality (becoming irreversible), the operating costs (e.g., energy cost) would gradually increase while typically leading to a significant reduction in the capital cost. The total cost for producing fresh water could be minimized as there is a trade-off between capital costs and energy costs. In practice, other design parameters, such as dimension and weight, should be considered to optimize the plant design (Miller 2003).

<Figure 5>

For the configurations of RO systems, Lin and Elimelech (2017) derived analytical expressions to quantify and optimize the average water flux (kinetic efficiency) and SEC (energetic efficiency), as shown in **Fig. 6**. The O&M costs, especially energy consumption and membrane replacement, would influence their contributions to the total cost of the plant. For long-term operation, RO with a high water flux is not suitable because of the great potential of fouling and scaling (Lin and Elimelech 2017, Sablani et al. 2001). The system configurations also exhibited significant influence on the energetics and kinetics, as well as the economic costs of the auxiliary processes, such as pretreatment, energy recovery devices and brine treatment. These auxiliary processes would in turn affect the overall techno-economics of a desalination plant.

<Figure 6>

Similarly, for the CDI systems, Wang and Lin (2018) established a systematic approach to determining the tradeoff between kinetic and energetic efficiencies (see **Fig. 7**). The kinetic efficiency depends on several factors such as average salt adsorption rate (for CDI), thereby affecting the clean water productivity. The energetic efficiency is directly related to the specific energy consumptions. Since the operating costs of desalination rely on both kinetic and energetic efficiencies, the optimal trade-off between kinetics and energetics should be located at the place where the total costs are minimized. Thus, Hemmatifar et al. (2016) suggested two performance indicators, i.e., average salt adsorption rate (ASAR) and energy-normalized adsorbed salt (representing energy loss per ion removed), to characterize the performance of CDI, in terms of clean water throughput (kinetics) and energy efficiency (energetics). These two indicators provide a powerful tool for balancing resistive and parasitic losses, thereby optimizing the overall energy efficiency.

<Figure 7>

Especially for the energetic efficiencies between RO and CDI, Qin et al. (2019) provided an estimate of the energy consumption in brackish water desalination using mathematical models. They found that CDI exhibits greater energy efficiency for a salt rejection of less than 25% with a high water recovery, compared to RO. However, particularly at high salt rejections (>50%) and moderate-to-high brackish water salinities (2–10 g/L), RO is significantly more energy efficient than CDI. Ramachandran et al. (2019a) further modified the important scaling values and resistance parameters based on Qin's models. They found that a reasonably high salt rejection over 70% with a high water recovery of > 80% could be achieved by CDI in an energy efficient manner. On the same base of assumptions, Porada et al. (2020) found that the energy consumption of MCDI would be lower than that of RO, in the case of a feed salinity of ~2.3 g/L (40 mM) at a water recovery of 93.5%, a salt rejection of 80%, and a total flux of clean water of 11.9 L/m^2/h. In fact, each technology has its own niche area of applications, depending upon the goals, objectives and targets of the treatment. For instance, CDI should exhibit better energetic efficiency for desalination of lower salinity brackish water (i.e., <2 g/L) than RO. However, for the sake of high salt rejection (>80%), BWRO has demonstrated superior long-term performance with a lower capital cost (Qin et al. 2019). If the feedwater contains silica, organic matter and pathogens, RO would produce an overall better water quality than CDI (Drewes et al. 2009).

6. Technical-Economic Assessment

For technical-economic assessment, both BWRO and CDI must be able to meet a certain salinity standard (e.g., 0.5 g/L for drinking water or 1.0 g/L for irrigation water) for water reclamation. On the same basis, the criteria used to assess RO and CDI for brackish water desalination include (i) lifetime-oriented total costs, (ii) thermodynamic energy efficiency and clean water productivity, (iii) pretreatment for prolonging the lifetime of key components, and (iv) post-treatment for sustainable brine management. In this part, we discuss the above key components for determining the technical-economic performance of BWRO and CDI.

6.1 Lifetime-Oriented Considerations to Total Costs

Lifetime-oriented considerations to total costs are crucial for the techno-economic analysis of brackish water desalination. A number of studies have reported the cost estimates based on parameterized process models to project fixed and variable costs at a large scale for RO and CDI (Bales et al. 2019, Hand et al. 2019, Metzger et al. 2020). However, only a few studies provided the information of total costs based on real plant operations using RO and CDI, as presented in **Table 8**. For an RO plant, the capital and energy use costs per unit of capacity generally decrease as the size of the plant increases. For a BWRO plant, the capital costs can account for about 35−42% of the total plant costs. The second large cost is the energy costs, which is up to 25% of the total costs. Other O&M costs include consumable chemicals (~14%), membrane replacement (~12%), labor (~5%), and miscellaneous (8%). Similar findings were observed in the literature that the unit water price ranges between 0.10 and 1.00 USD/m^3 (Miller 2003, Sethi 2007). The associated costs include capital (~54%), energy (~11%), chemicals (~10%), labor (~9%), maintenance (~9%), and membrane replacement (~7%) (Miller 2003). Koyuncu et al. (2001) suggest that membrane cost usually represents approximately 20–30% of the total capital cost.

<Table 8>

For CDI, the capital costs are the large portion to overall water costs, accounting for 27−73% of total costs. CDT Inc. (Texas, USA) has estimated the total costs of CDI desalination for brackish water with a salinity of 6.4 g/L to meet the irrigation water standard (i.e., 1.0 g/L). Hand et al. (2019) noticed that lifetime should be the primary factors of water costs for CDI and membrane CDI due to the relatively high portions of capital costs to total costs, and it would be more pronounced at a greater reduction in salinity between feed and effluent water. For instance, for improving the lifetime of CDI from 2 to 5 years, the reduction in water costs at 1.46 and 2.92 g/L would be 0.07 and 0.17 USD/m^3, respectively, based on a parameterized process model. In other words, prolonging the lifetimes of CDI systems at more than 2 years would be crucial to realize the cost effectiveness of brackish water desalination. Similarly, Drewes et al. (2009) suggested that the total costs of desalination using CDI was in the range of 1.93−2.60 USD/m^3, depending upon the operating flow rate from 0.7 to 3.0 L/min per module. If the module lifetime is assumed to extend (up to 20 years), the total cost of desalination would be reduced to 1.76−1.97 USD/m^3 under the same range of flow rate (Drewes et al. 2009). EST Water (China) also deployed several large-scale CDI plants in Mainland China for desalinating various types of produced water. The estimated costs for energy consumption and module maintenance were 0.055 USD/m^3 and 0.014 USD/m^3, respectively (EST 2009).

Improved engineering design, such as less pretreatment, efficient desalination unit and effective brine management, is the key to lower overall desalination costs. BWRO requires more professional staff for operating high-pressure pumps, heaters and clean-in-place systems, while CDI usually operates at ambient pressure and temperature. In the case of BWRO, Anqi et al. (2015) have conducted numerical simulations to evaluate the desalination performance. They found that the Sherwood number is strongly dependent on the Reynolds number, as well as the configurations of spacers (especially for local Sherwood number). In other words, both the pressure drops and the arrangement (e.g., spacing) of turbulators should be optimized for a given range of Reynolds numbers.

6.2 Energy Efficiency and Clean Water Productivity

In response to the water and energy nexus under global climate change, a major reconsideration of desalination technologies regarding the energy efficiency and the impact of the concentrated brine on the discharge environment has occurred. In fact, thermodynamic energy efficiency could help us to understand the limit for further improvement on specific energy consumption among different desalination technologies. Wang et al. (2019a) also suggested to consider both thermodynamic energy efficiency and specific energy consumption when comparing different desalination systems and processes. From the thermodynamic point of view, CDI is more energy efficient for brackish water desalination compared to RO. However, the total energy consumption of the current RO system (mostly on a plant scale) cannot usefully be compared to the CDI values, which are often at the lab scale. As CDI technologies are relatively new, compared to RO, the currently available data of energy consumption from large-scale CDI operations is similar to that of RO. The improvements of both RO membranes and energy recovery devices have made significant breakthrough on the energy consumption of BWRO being close to the thermodynamic limits. In contrast, the electrode materials and cell design of CDI are still under extensive research, resulting in improved CDI performance with each novel material. Conversely, high energy consumption of CDI is largely due to application of high currents; however, larger treatment capacities will overcome this limitation. To achieve an industrial scale operation, CDI technologies should balance the capital cost (the number of modules) and the energy cost (the effective area of electrodes). Hand et al. (2019) found that energy consumption is not very relevant for CDI, which can be a small fraction of total costs especially for membrane CDI.

Aside from energy efficiency, the water productivity and recovery ratio are essential to overall water costs as they are associated with the subsequent brine management. Regardless the mechanisms of desalination, various quantities of brine (the concentrate stream) would be produced from processes as a by-product. For the same treatment capacity, the desalination plant with a lower recovery ratio will generate a greater amount of concentrate brine. The quality and quantity of the brine would determine the optimum approach to subsequent management and utilization. Drewes et al. (2009) found that the salinity of brine from RO (20−22 g/L) was much higher than that from CDI (7−8 g/L). According to the data from large-scale operation, the quantity of brine from RO was ten times less than that from CDI (Drewes et al. 2009). This was attributed to the large amount of water for electrode regeneration and rinsing in CDI due to the slow electro-desorption kinetics. Recently, the significant improvement on water recovery of CDI has been achieved. Ramachandran et al. (2019b) developed a new scheme based on variable flowrate operation to increase water recovery for CDI with a minimal additional cost. They successfully demonstrated a high water recovery ratio of ~90% while improving thermodynamic efficiency by at least 2-fold. This would produce a significant less volume of brine solution, compared to conventional constant flowrate operation. For flow-electrode CDI, Ma et al. (2019) achieved an extreme water recovery of 95–99% with the brine concentration of 20–50 g/L, though the charge efficiency is compromised.

6.3 Pretreatment for Prolonging Life-time of Key Components

Dynamics of feedwater characteristics (e.g., salinity and temperature) must be considered in designing a brackish water desalination system. For instance, temperature variance may result in membrane scaling, especially when the concentrations of silica and bicarbonate in the feedwater are high (Alghoul et al. 2009). The fouling and scaling would be potential issues to membrane operations in BWRO, while conventional CDI has relatively less issues with fouling and scaling. For membrane CDI, scaling and fouling would still be a challenge (Chang et al. 2002). BWRO systems typically require pretreatment to prevent membrane scaling and fouling, including pH adjustment, dosing system of antiscalants and disinfectants (optional), and microfiltration. CDI systems need a cartridge filtration as the pretreatment, and might require additional pre-treatment to remove organic matter. A number of studies have reported that the content of organic matter would reduce the sorption capacity of CDI electrodes by fouling (Gabelich et al. 2002, Zhang et al. 2013) or ferric-species scaling (Wang et al. 2019b). Surface modification on CDI electrodes might be potential solutions to improve the resistance of CDI systems against fouling and scaling (Pugazhenthiran et al. 2015, Wu et al. 2017b, Zhang and Jia 2015). For energy-efficient RO optimization, a specific concentration limit on the product stream must be adhered, while applying suitable feed pressure to minimize spatial variance in flux (Wei et al. 2017). The life span of an RO membrane increases by using a low feed pressure; however, this results in compromising water recovery ratios.

The key components affecting the desalination performance of RO or CDI include membranes and electrodes. The life cycle of an RO plant is approximately 20 years, where the major maintenance is membrane replacement at about every 5−7 years. However, both membrane and electrode lifetime could be significantly shortened by severe fouling due to improper pretreatment of feedwater. For membranes, manufacturers will normally provide detailed instructions for standard operation and maintenance procedures of their membrane products. Proper pretreatment of feedwater and periodical membrane cleaning are required to maximize the efficiency of desalination and ensure the life time of membranes (Asadollahi et al. 2017, Avlonitis et al. 2003). For life-time of electrodes in CDI, Welgemoed and Schutte (2005) estimated that electrodes could last for 10 years in the case of carbon aerogel electrodes. Wang et al. (2019b) also suggested that the foulant caused by ferric ions (e.g., Fe_2O_3) were irreversible once formed on the electrodes, which could be difficult to be entirely removed by backwash

The necessary and extent of pretreatment relies on the quality of feedwater, the plant location and the intake system. However, one of the challenges in leveraging brackish water is the dynamics of feedwater quality. Surface water, such as seawater and wastewater, typically contains readily available nutrients (natural organic matter) and oxygen for bio-respiration. Therefore, when surface water is involved, pretreatment is essential to ensure separation efficiency and avoid biological fouling and scaling (due to the presence of multiple ions such as carbonates and sulfates, barium, magnesium and calcium) for membrane technologies. Conventional pretreatment, typically for RO processes, includes chemicals addition (chemical pretreatment) and generally contains flocculation, sedimentation and filtration (physical pretreatment) to mechanically remove algae, colloids and particles. Other pretreatments include ultrafiltration (UF) (Gao et al. 2016), cartridge filtration (Farhat et al. 2020), microfiltration (Wu et al. 2017a), forward osmosis (Khanzada et al. 2017), and dissolved air flotation (Henthorne and Boysen 2015). These alternatives are also considered as effective approaches to reducing membrane fouling potential and cleaning frequencies.

Table 9 presents the costs of various pretreatment methods for RO and CDI plants. According to the site measurements, the conventional pretreatment can achieve a water recovery of ~99% with the electricity consumption as low as 0.025 kWh per m^3 of feedwater (Vince et al. 2008). The reported costs of conventional pretreatment were approximately 0.13 USD/m^3 (Pearce 2008). However, the conventional pretreatment is chemicals/labor intensive and space consuming (van Hoof et al. 2001). The land footprint of conventional pretreatment is approximately 35–40 m^2 per 1000 m^3 permeate per day (Wilf 2004). In contrast, the land footprint for UF pretreatment is only 30–60% of conventional pretreatment (Wolf and Siverns 2004). The reported UF treatment costs vary from 0.21 USD/m^3 to 0.52 USD/m^3 (Glueckstern and Priel 2003, Jurenka et al. 2001). On the other hand, the typical lifetime of media filter and UF membranes is 20–30 and 5–10 years, respectively (Wolf and Siverns 2004). Thus, there is an optimal choice of technology to use for pretreatment based on operating costs and lifetime. For the media filtration, the reported costs were approximately 0.51

USD/m^3 (Glueckstern and Priel 2003), which was similar to that of UF pretreatment. For CDI, a number of studies have indicated that CDI needs only simple pretreatment, such as cartridge filtration (Xu et al. 2008b). The reported O&M costs for cartridge filtration were approximately 0.015–0.021 USD/m^3 (Farhat et al. 2020). The low micron range cartridge filtration could be utilized for RO plants as a protection for the subsequent high-pressure pumps.

<Table 9>

6.4 Post-treatment Options for Sustainable Brine Management

For both BWRO and CDI, post-treatment could represent a significant portion of the total water production costs as these two technologies will generate a high concentration brine. Metzger et al. (2020) also highlighted that the brine generated from desalination facilities would pose severe environmental impacts and becomes an increasing economic concern. There are different options for the disposal or treatment of brine from a desalination plant. the costs of concentrated brine disposal. The selection of brine disposal methods represents a compromise between technology availability, total cost, local resources, and environmental impacts. **Table 10** compiles the costs of various concentrated brine disposal methods. Surface water (i.e., ocean, river, lake and lagoon) discharge is the most common management practice since it is the least expensive option among other available brine disposals. However, it is often limited to coastal desalination plants. It may also change the salinity of the receiving water, depending upon water recovery and concentrate factor, thereby changing the water chemistry (e.g., dissolved gases and lack of oxygen) and affecting aquatic animals. Several studies (Purnama et al. 2005, Smith et al. 2007) showed that an increase in costal desalination installations at the Arabian Gulf would increase the salinity in the Gulf, leading to local variations in dissolved oxygen concentration and temperature. Therefore, for surface water discharge, Mickley (2004) suggested a standard limit of the salinity difference between the concentrate stream and receiving water less than 10%. If the feed of brackish water desalination is groundwater, the concentrate brine must be treated before disposal since it typically contains high concentrations of gases, such as CO_2, ammonia and H_2S. These dissolved gases are harmful and toxic to aquatic life.

<Table 10>

Besides surface water disposal, combined sewer disposal (if available) is usually the next option as a relatively low-cost disposal method. Huyskens et al. (2015) reported the costs of brine disposal in wastewater treatment plants at about 1.26 USD/m^3. Beside further treatment, the brine reject can be directly utilized via various approaches, e.g., mixed with raw water to provide the use of irrigation purposes (Peñate et al. 2014). Also, salinity gradient energy (so-called blue energy) in brine could be harvested by various promising technologies, such as pressure-retarded osmosis (Benjamin et al. 2020), reverse electrodialysis (Nam et al. 2019), capacitive mixing (Simoncelli et al. 2018), and 2D nanopore diffusio-osmosis (Siria et al. 2017).

6.5 Summary

The total costs of a desalination plant for producing clean water include capital, O&M, labor, and miscellaneous costs. Other important adjunct units to desalination, such as pretreatment, post-treatment, brine discharge and waste management, also should be considered in the total costs. These costs largely depend on local conditions, such as the method of desalination, source water quality, clean water productivity (capacity), and availability of concentrate-disposal sites. In fact, it is difficult to comprehensively compare the total costs of BWRO and CDI as, at the same removal ratio, the water recovery ratios of BWRO and CDI are quite different. As we discussed, lifetime-oriented considerations (including pretreatment of feedwater, desalination facilities, and post-treatment) to total costs are important for the techno-economic analysis of brackish water desalination. In general, for RO, the major cost components include capital costs of modules and energy consumption due to high-pressure pumps. For CDI, the major cost components include the capital costs of modules and electrode replacement. RO needs periodical membrane cleaning to control fouling and scaling, while CDI uses electrochemical reactions (or electrostatics) to regenerate the saturated (or fouled) electrodes. Thus, the cost for chemicals use in RO (for membrane regeneration) would be generally higher than that in CDI, and the cost for electrode regeneration in CDI is embedded in the cost of its energy consumption. In addition, the quality and quantity of the brine solution generated from BWRO and CDI are quite different. Therefore, the costs of brine management would be the major concern when evaluating the total costs of brackish water desalination.

7. Perspectives, Prospects, and Priority Research Directions

CDI has shown better energy efficiency for desalination of lower salinity brackish water (i.e., <2 g/L) than RO. However, for the sake of high salt rejection (>80%), RO has demonstrated superior long-term performance with a lower capital cost (Qin et al. 2019). RO also can produce an overall better water quality than CDI, as RO is effective to remove silica, organic matter and pathogens (Drewes et al. 2009). Because of these special pros and cons, we believe that each technology has its own niche area of applications, depending upon the goals, objectives and targets of the treatment. In this part, we suggest three prior research directions, from the aspects of (i) more cost-effective desalination, (ii) high selectivity and (iii) sustainable brine management, to optimize the desalination of brackish water and address the challenges and opportunities in water and energy nexus.

7.1 Hybridization for Energy-Efficient Brackish Water Desalination

RO is the most commonly used technology since it can tackle the entire range of saline waters up to seawater, although it is not energy efficient at a low salinity (<3 g/L TDS). From the thermodynamic point of view, CDI or other electrokinetically-driven techniques, such as electrodialysis reversal (Liu and Wang 2017) and electrodeionization (Pan et al. 2018b), should be energetically more efficient for brackish water desalination, compared to pressure- or thermally-driven techniques. Although we focused on only CDI in this review, each desalination technology should exhibit its own best operation with the highest energy efficiency at a certain range of feedwater quality. Therefore, hybridization could provide the synergetic solution to energy-efficient and “fit-for-purpose” water for developing a sustainable water supply, especially for brackish water reuse. For instance, Sarai Atab et al. (2018) analyzed the hybridization of RO with an adsorption cycle for providing large quantities of water for irrigation (24,000 m^3/d) and high quality water for domestic use. The proposed hybrid plant has the minimum specific energy about 0.8 kWh/m^3 at RO recovery of 45%, with a production cost of 0.56 USD/m^3 (Sarai Atab et al. 2018). However, compared to existing municipal water sources, desalinated water still comes at substantially higher costs (Carter 2015). The choice of desalination mechanisms (techniques) and their configurations depend on numerous factors, such as the quality of the feed, targeted quality and productivity of reclaimed water, and options for brine disposal. These factors sometimes are related to the local regulatory standards and requirements. For identifying the best available design and operation strategies, Li and Noh (2012) suggested that a relationship between water recovery and membrane lifetime (especially for BWRO) should be established to incorporate capital and operating costs, along with the system optimization. In the future, the hybridization of different separation technologies incorporated with renewable energy for energy-efficient brackish water desalination should be evaluated.

In addition to hybridization for energy-efficient brackish water desalination, the development of district water (e.g., desalination) and energy (e.g., renewables) supply center provides great opportunities for advancing overall energy and water efficiency. Inland brackish water desalination can be coupled with renewable energy to augment freshwater supply sustainably, especially at remote areas that lack access to a reliable electricity grid. Production of clean water also could be considered as an option of energy storage for intermittent renewables. Therefore, the design of desalination plants should be considered along with the planning procedure of district energy supply system for supporting ancillary services in wholesale energy markets. This could ensure the security and sustainability of water and energy supply. Several field tests have been conducted to demonstrate its feasibility and reliability, such as solar photovoltaic electricity at Pakistan (Khanzada et al. 2017), and Malaysia (Alghoul et al. 2016). Kim et al. (2016) also conducted a dynamic performance analysis to evaluate the feasibility of integrated hybrid energy systems with RO desalination plants and identify its dynamic characteristics. Similar to RO, the energy demands of CDI can be met by renewable energy sources, such as solar PV. Coupling a CDI unit with PV makes the water desalination system self-sufficient in energy demands, and could be deployed in remote off-grid locations (Mehrabian-Nejad et al. 2017). Distributed, modular CDI have been combined with solar cell modules in recent reported research (Tan et al. 2018, Wu et al. 2017b).

Adequate infrastructure is essential to address long-term water and energy scarcity challenges. Centralized desalination plants are usually practiced in larger scales including urban areas and cities, while decentralized plants are employed for rural or remote regions that lack access to centralized systems (Silva Herran and Nakata 2012). Vakilifard et al. (2018) examined the role of water-energy nexus in optimizing water supply systems, and they found that there is a research need in the optimization of the decentralized water-energy supply system independently, or as an integral part of a centralized system in urban areas. In addition to the optimization of water supply systems, the research efforts covering instrumentation, control and automation (ICA) of energy-efficient water technologies for brackish water desalination systems should be addressed in the context of future smart cities.

7.2 Co-removal of Specific Components in Brackish Water

Development of scalable, affordable and robust CDI electrodes with a high ion adsorption capacity can facilitate the deployment of desalination for widely available brackish water sources. Although CDI is effective to remove salts from water, a comprehensive study on electrosorption of competing ions in brackish water is needed to understand the behavior of CDI electrodes. Huang et al. (2017) strongly recommended applying a three-electrode cell for examining the electrosorption behaviors of carbon materials. In practice, brackish water may contain various contaminant ions, such as arsenic, fluoride, boron, phosphate, lithium, iodide, copper, cadmium, ferric, and nitrate ions. Some of the above-mentioned ions in brackish water are classified as precious metals (e.g., lithium), which could be further precisely separated and recovered by electrokinetic methods. On the other hand, more regulations on effluent water quality have resulted in that boron and arsenic are becoming of main interests since they are typically difficult to remove by RO. Despite the recent advances of membranes, boron (Br) and arsenic (As) rejection remains low in comparison to other inorganic components, such as sodium chloride (Teychene et al. 2013). In addition to removal of inorganic ions, co-removal of organic matter in brackish water has been the focus of intense scientific and practical efforts. For instance, Lester et al. (2020) developed a novel configuration of CDI with activated carbon electrodes to remove both salt and trace organic, hydrophobic compounds (such as bisphenol A and estrone) from wastewater.

To maximize the removal of other compounds of interest, plant design (e.g., membrane selection) and optimization warrant significant attention. Technological improvement and breakthrough for both membranes and electrodes could enhance economical separations to drive market penetration for brackish water desalination. A number of studies have suggested that nanotechnology-enable materials, such as ion exchange membranes, could facilitate the wide adoption of water desalination (Alabi et al. 2018, Mauter et al. 2018). For membrane desalination, effective removal of divalent cations at ultrafast water flux and low-pressure operation, as well as in-situ regeneration for fouling and scaling control are important. Novel membrane materials, such as graphene oxide membranes (Mi 2014, Zheng et al. 2017), have been under developed and investigated. This could support the use of brackish water in cooling systems with minimal scaling and brine production. Future research on development of brackish water desalination also should focus on the removal of other regulated contaminants of emerging concern, such as disinfection byproduct, pharmaceutical and personal care products, and endocrine disrupting compounds.

7.3 Sustainable Brine Management with Innovative Utilization

Desalination typically generates concentrated brine that contains relatively high concentrations of salts, organic matter, and inorganic constituents (e.g., boron and copper). One of the critical concerns to inland brackish water desalination is concentrated brine disposal due to its potentially high costs. As aforementioned, if surface water disposal is used for brine treatment, its environmental impacts on local water bodies also should be critically evaluated via an LCA. When ocean disposal of such streams or deep-well injection is not available, zero liquid discharge (ZLD) strategies are usually required to reduce the volume of concentrate while simultaneously removing contaminants from the brine. ZLD typically involves thermally drying of concentrated brine, such as thermal concentrate crystallization (Choi et al. 2018) and capillary crystallization (Abahusayn 2011, Sobhani et al. 2012), which are deployed at high energy and capital costs (Gray et al. 2011). Therefore, there is still an urgent need for innovative brine management and utilization to allow economic use of inland brackish water resources. For instance, engineered natural systems via biological approaches might become an alternative to manage the concentrated brine, as well as to handle the issues of organics, nitrate, and other contaminants. Similarly, when low-grade heat (such as residual heat from power plants or geothermal energy) is nearby accessible, membrane distillation could be deployed to increase water recovery prior to brine crystallizer (Deshmukh et al. 2018). These innovative approaches of brine utilization could provide the potential to further recover nutrients, minerals and energy for realizing a circular economy.

Aside from the end-of-facility treatment, optimization of chemicals and antiscalants dosing is of strategical importance to improve the cost of subsequent brine management. Sweity et al. (2015) found that several antiscalants (e.g., polyacrylate-based or polyphosphonate-based) could significantly contribute to the biofouling of RO membranes, where the biofouling enhancement potential should be critically screened along with their antiscaling activity. Similarly, both energy efficiency (related to energy consumption) and clean water productivity (related to water recovery) are critical parameters to realize the ZLD of a desalination plant. Taking the example of BWRO, the achievable water recovery is typically 60−85% and thus results in a concentration factor of 2.5−6.7. However, with a high water recovery of >97%, the concentration factor would increase to more than 33.3, leading to relatively high costs for ZLD processes. In other words, both the energy consumption and water recovery should be practically balanced with the consideration of subsequent brine disposal to achieve an overall cost-effective scenario.

8. Concluding remarks

Brackish water is an important alternative to fresh water resources, potentially enabling the movement of the water-energy nexus away from the seacoast further inland. Technology maturity determines the extent of practical deployment: RO has been fully commercialized for decades; whereas CDI has not yet achieved widespread market adoption. Further improvements on desalination technologies would provide significant potential to ensure the availability, accessibility, and affordability of fit-for-purpose fresh water from brackish water. The quality of feedwater plays a critical role in the selection, design and operation of desalination technologies because one of the challenges in exploitation of brackish water is the dynamics of feedwater quality. To facilitate the exploitation of brackish water, we suggest three prior research directions for the optimization of brackish water desalination while addressing the challenges and opportunities in water and energy nexus, including hybridization for energy-efficient brackish water desalination, co-removal of specific components in brackish water, and sustainable brine management with innovative utilization. For both BWRO and CDI, development of sustainable brine management with innovative utilization would effectively mitigate the environmental impacts and reduce the O&M costs. Along the way, achieving as high as possible on water recovery would directly decrease the amount of the brine generation from desalination facilities, which would further address the concern of the energy requirement for post-treatment.

Conflicts of interest

There are no conflicts to declare.

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References

Abahusayn, M., 2011. Salt brine capillary crystallization, US.

Ahmed, M.A. and Tewari, S., 2018. Capacitive deionization: Processes, materials and state of the technology. Journal of Electroanalytical Chemistry 813, 178-192.

Ahn, J., Lee, J., Kim, S., Kim, C., Lee, J., Biesheuvel, P.M. and Yoon, J., 2020. High performance electrochemical saline water desalination using silver and silver-chloride electrodes. Desalination 476.

Al-Abri, M., Al-Ghafri, B., Bora, T., Dobretsov, S., Dutta, J., Castelletto, S., Rosa, L. and Boretti, A., 2019. Chlorination disadvantages and alternative routes for biofouling control in reverse osmosis desalination. npj Clean Water 2(1).

Al-Karaghouli, A. and Kazmerski, L.L., 2013. Energy consumption and water production cost of conventional and renewable-energy-powered desalination processes. Renewable and Sustainable Energy Reviews 24, 343-356.

Al-Obaidi, M.A., Alsarayreh, A.A., Al-Hroub, A.M., Alsadaie, S. and Mujtaba, I.M., 2018. Performance analysis of a medium-sized industrial reverse osmosis brackish water desalination plant. Desalination 443, 272-284.

Alabi, A., AlHajaj, A., Cseri, L., Szekely, G., Budd, P. and Zou, L., 2018. Review of nanomaterials-assisted ion exchange membranes for electromembrane desalination. npj Clean Water 1(1).

Alencherry, T., Naveen, A.R., Ghosh, S., Daniel, J. and Venkataraghavan, R., 2017. Effect of increasing electrical conductivity and hydrophilicity on the electrosorption capacity of activated carbon electrodes for capacitive deionization. Desalination 415, 14-19.

Alghoul, M.A., Poovanaesvaran, P., Sopian, K. and Sulaiman, M.Y., 2009. Review of brackish water reverse osmosis (BWRO) system designs. Renewable and Sustainable Energy Reviews 13(9), 2661-2667.

Alghoul, M.A., Poovanaesvaran, P., Mohammed, M.H., Fadhil, A.M., Muftah, A.F., Alkilani, M.M. and Sopian, K., 2016. Design and experimental performance of brackish water reverse osmosis desalination unit powered by 2 kW photovoltaic system. Renewable Energy 93, 101-114.

Aljohani, M.S., 2017. Synergistic efficiency of the desilication of brackish underground water in Saudi Arabia by coupling γ-radiation and Fenton process: Membrane scaling prevention in reverse osmosis process. Radiation Physics and Chemistry 141, 245-250.

Aljundi, I.H., 2009. Second-law analysis of a reverse osmosis plant in Jordan. Desalination 239(1-3), 207-215.

Allam, A.R., Saaf, E.-J. and Dawoud, M.A., 2002. Desalination of brackish groundwater in Egypt. Desalination 152, 19-26.

Almulla, A., Eid, M., Cote, P. and Coburn, J., 2003. Developments in high recovery brackish water desalination plants as part of the solution to water quantity problems. Desalination 153, 237-243.

Alsarayreh, A., Majdalawi, M. and Bhandari, R., 2017. Techno-Economic Study of PV Powered Brackish Water Reverse Osmosis Desalination Plant in the Jordan Valley. Int. J. of Thermal & Environmental Engineering 14, 83-88.

Alvarez, P.J.J., Chan, C.K., Elimelech, M., Halas, N.J. and Villagran, D., 2018. Emerging opportunities for nanotechnology to enhance water security. Nat Nanotechnol 13(8), 634-641.

Amy, G., Ghaffour, N., Li, Z., Francis, L., Valladares Linares, R., Missimer, T. and Lattemann, S., 2017. Membrane-based seawater desalination: Present and future prospects. Desalination 401, 16-21.

Anqi, A.E., Alkhamis, N. and Oztekin, A., 2015. Numerical simulation of brackish water desalination by a reverse osmosis membrane. Desalination 369, 156-164.

Arroyo, J. and Shirazi, S., 2012. Cost of Brackish Groundwater Desalination in Texas, p. 8, Texas.

Asadollahi, M., Bastani, D. and Musavi, S.A., 2017. Enhancement of surface properties and performance of reverse osmosis membranes after surface modification: A review. Desalination 420, 330-383.

Asempour, F., Emadzadeh, D., Matsuura, T. and Kruczek, B., 2018. Synthesis and characterization of novel Cellulose Nanocrystals-based Thin Film Nanocomposite membranes for reverse osmosis applications. Desalination 439, 179-187.

Australian Government, 2004. National Water Quality Management Strategy, Australian Drinking Water Guidelines 6, p. 615, National Health and Medical Council, Agricultural and Resource

Management, Council of Australia and New Zealand, Australia.

Avlonitis, S.A., Kouroumbas, K. and Vlachakis, N., 2003. Energy consumption and membrane replacement cost for seawater RO desalination plants. Desalination 157, 151-158.

Bales, C., Kovalsky, P., Fletcher, J. and Waite, T.D., 2019. Low cost desalination of brackish groundwaters by Capacitive Deionization (CDI) – Implications for irrigated agriculture. Desalination 453, 37-53.

Benjamin, J., Arias, M.E. and Zhang, Q., 2020. A techno-economic process model for pressure retarded osmosis based energy recovery in desalination plants. Desalination 476.

Biesheuvel, P.M., Bazant, M.Z., Cusick, R.D., Hatton, T.A., Hatzell, K.B., Hatzell, M.C., Liang, P., Lin, S., Porada, S., Santiago, J.G., Smith, K.C., Stadermann, M., Su, X., Sun, X., Waite, T.D., van der Wal, A., Yoon, J., Zhao, R., Zou, L. and Suss, M.E., 2017. Capacitive Deionization - defining a class of desalination technologies. arXiv, 1-3.

Blair, D., Alexander, D.T., Couperthwaite, S.J., Darestani, M. and Millar, G.J., 2017. Enhanced water recovery in the coal seam gas industry using a dual reverse osmosis system. Environmental Science: Water Research & Technology 3(2), 278-292.

Blanco-Marigorta, A.M., Masi, M. and Manfrida, G., 2014. Exergo-environmental analysis of a reverse osmosis desalination plant in Gran Canaria. Energy 76, 223-232.

Cai, W., Yan, J., Hussin, T. and Liu, J., 2017. Nafion-AC-based asymmetric capacitive deionization. Electrochimica Acta 225, 407-415.

Carter, N.T., 2015. Desalination and Membrane Technologies: Federal Research and Adoption Issues, p. 15, Congressional Research Service.

Caudill, L.S., 2018. Pressure-Driven Stabilization of Capacitive Deionization, University of Kentucky, USA.

Chang, I.-S., Le Clech, P., Jefferson, B. and Judd, S., 2002. Membrane fouling in membrane bioreactors for wastewater treatment. Journal of Environmental Engineering 128(11), 1018-1029.

Chen, F., Huang, Y., Guo, L., Sun, L., Wang, Y. and Yang, H.Y., 2017. Dual-ions electrochemical deionization: a desalination generator. Energy & Environmental Science 10(10), 2081-2089.

Chen, Y.-W., Chen, J.-F., Lin, C.-H. and Hou, C.-H., 2019. Integrating a supercapacitor with capacitive deionization for direct energy recovery from the desalination of brackish water. Applied Energy 252.

Chen, Z., Zhang, H., Wu, C., Luo, L., Wang, C., Huang, S. and Xu, H., 2018. A study of the effect of carbon characteristics on capacitive deionization (CDI) performance. Desalination 433, 68-74.

Cheng, C., Jiang, G., Garvey, C.J., Wang, Y., Simon, G.P., Liu, J.Z. and Li, D., 2016. Ion transport in complex layered graphene-based membranes with tuneable interlayer spacing. Sci Adv 2(2), e1501272.

Cho, Y., Lee, K.S., Yang, S., Choi, J., Park, H.-r. and Kim, D.K., 2017. A novel three-dimensional desalination system utilizing honeycomb-shaped lattice structures for flow-electrode capacitive deionization. Energy & Environmental Science 10(8), 1746-1750.

Choi, Y., Naidu, G., Jeong, S., Lee, S. and Vigneswaran, S., 2018. Effect of chemical and physical factors on the crystallization of calcium sulfate in seawater reverse osmosis brine. Desalination 426, 78-87.

Chung, H.J., Kim, J., Kim, D.I., Gwak, G. and Hong, S., 2020. Feasibility study of reverse osmosis–flow capacitive deionization (RO-FCDI) for energy-efficient desalination using seawater as the flow-electrode aqueous electrolyte. Desalination 479.

Cornejo, P.K., Santana, M.V.E., Hokanson, D.R., Mihelcic, J.R. and Zhang, Q., 2014. Carbon footprint of water reuse and desalination: a review of greenhouse gas emissions and estimation tools. Journal of Water Reuse and Desalination 4(4), 238-252.

Delgado, A.V., Jiménez, M.L., Iglesias, G.R. and Ahualli, S., 2019. Electrical double layers as ion reservoirs: applications to the deionization of solutions. Current Opinion in Colloid & Interface Science 44, 72-84.

Deshmukh, A., Boo, C., Karanikola, V., Lin, S., Straub, A.P., Tong, T., Warsinger, D.M. and Elimelech, M., 2018. Membrane distillation at the water-energy nexus: limits, opportunities, and challenges. Energy & Environmental Science 11(5), 1177-1196.

Drewes, J.E., Xu, P., Heil, D. and Wang, G., 2009. Multibeneficial Use of Produced Water Through High-Pressure Membrane Treatment and Capacitive Deionization Technology, Bureau of Reclamation, U.S. Department of the Interior, Colorado.

Dykstra, J.E., Zhao, R., Biesheuvel, P.M. and van der Wal, A., 2016. Resistance identification and rational process design in Capacitive Deionization. Water Res 88, 358-370.

Elimelech, M. and Phillip, W.A., 2011. The future of seawater desalination: energy, technology, and the environment. Science 333(6043), 712-717.

EST, 2009. Electrosorption Technology for Desalting, p. 40, EST Water and Technologies Co., Ltd., China.

Farhat, N.M., Christodoulou, C., Placotas, P., Blankert, B., Sallangos, O. and Vrouwenvelder, J.S., 2020. Cartridge filter selection and replacement: Optimization of produced water quantity, quality, and cost. Desalination 473.

Fritzmann, C., Löwenberg, J., Wintgens, T. and Melin, T., 2007. State-of-the-art of reverse osmosis desalination. Desalination 216(1-3), 1-76.

Gabelich, C.J., Tran, T.D. and Suffet, I.M., 2002. Electrosorption of inorganic salts from aqueous solution using carbon aerogels. Environmental science & technology 36(13), 3010-3019.

Gabelich, C.J., Williams, M.D., Rahardianto, A., Franklin, J.C. and Cohen, Y., 2007. High-recovery reverse osmosis desalination using intermediate chemical demineralization. Journal of Membrane Science 301(1-2), 131-141.

Gao, L.X., Rahardianto, A., Gu, H., Christofides, P.D. and Cohen, Y., 2016. Novel design and operational control of integrated ultrafiltration — Reverse osmosis system with RO concentrate backwash. Desalination 382, 43-52.

Gao, X., Omosebi, A., Landon, J. and Liu, K., 2015. Surface charge enhanced carbon electrodes for stable and efficient capacitive deionization using inverted adsorption–desorption behavior. Energy & Environmental Science 8(3), 897-909.

Ghermandi, A. and Messalem, R., 2009. Solar-driven desalination with reverse osmosis: the state of the art. Desalination and Water Treatment 7, 285–296.

Giwa, A., Dufour, V., Al Marzooqi, F., Al Kaabi, M. and Hasan, S.W., 2017. Brine management methods: Recent innovations and current status. Desalination 407, 1-23.

Glueckstern, P. and Priel, M., 2003. Comparative cost of UF vs. conventional pretreatment for SWRO systems. Desalination & Water Reuse 12(4), 34-39.

Goedkoop, M., Heijungs, R., Huijbregts, M., De Schryver, A., Struijs, J. and Van Zelm, R., 2009. ReCiPe 2008. A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level 1.

Graves, M. and Choffel, K., 2004. Economic Siting Factors for Seawater Desalination Projects along the Texas Gulf-Coast, p. 18, Texas Water Development Board.

Gray, S., Semiat, R., Duke, M., Rahardianto, A. and Cohen, Y., 2011. Treatise on Water Science. Peter, W. (ed), pp. 73-109, Elsevier B.V., Oxford.

Greenlee, L.F., Lawler, D.F., Freeman, B.D., Marrot, B. and Moulin, P., 2009. Reverse osmosis desalination: water sources, technology, and today's challenges. Water Res 43(9), 2317-2348.

Hand, S., Guest, J.S. and Cusick, R.D., 2019. Technoeconomic Analysis of Brackish Water

Capacitive Deionization: Navigating Tradeoffs between Performance, Lifetime, and Material Costs. Environ Sci Technol 53(22), 13353-13363.

Hawks, S.A., Ramachandran, A., Porada, S., Campbell, P.G., Suss, M.E., Biesheuvel, P.M., Santiago, J.G. and Stadermann, M., 2019. Performance metrics for the objective assessment of capacitive deionization systems. Water Res 152, 126-137.

He, C., Ma, J., Zhang, C., Song, J. and Waite, T.D., 2018. Short-Circuited Closed-Cycle Operation of Flow-Electrode CDI for Brackish Water Softening. Environ Sci Technol 52(16), 9350-9360.

Hemmatifar, A., Stadermann, M. and Santiago, J.G., 2015. Two-Dimensional Porous Electrode Model for Capacitive Deionization. The Journal of Physical Chemistry C 119(44), 24681-24694.

Hemmatifar, A., Palko, J.W., Stadermann, M. and Santiago, J.G., 2016. Energy breakdown in capacitive deionization. Water Res 104, 303-311.

Henthorne, L. and Boysen, B., 2015. State-of-the-art of reverse osmosis desalination pretreatment. Desalination 356, 129-139.

Hu, C.C., Hsieh, C.F., Chen, Y.J. and Liu, C.F., 2018. How to achieve the optimal performance of capacitive deionization and inverted-capacitive deionization. Desalination 442, 89-98.

Huang, Z.-H., Yang, Z., Kang, F. and Inagaki, M., 2017. Carbon electrodes for capacitive deionization. J. Mater. Chem. A 5(2), 470-496.

Huyskens, C., Helsen, J., Groot, W.J. and de Haan, A.B., 2015. Cost evaluation of large-scale membrane capacitive deionization for biomass hydrolysate desalination. Separation and Purification Technology 146, 294-300.

International Organization for Standardization, 2006a. ISO 14040 — Environmental management — Life cycle assessment: Principle and Framework, Switzerland.

International Organization for Standardization, 2006b. ISO 14044 — Environmental management − Life cycle assessment: Requirements and guidelines, Switzerland.

Jaber, I.S. and Ahmed, M.R., 2004. Technical and economic evaluation of brackish groundwater desalination by reverse osmosis (RO) process. Desalination 165, 209-213.

Jain, A., Kim, J., Owoseni, O.M., Weathers, C., Cana, D., Zuo, K., Walker, W.S., Li, Q. and Verduzco, R., 2018. Aqueous-Processed, High-Capacity Electrodes for Membrane Capacitive Deionization. Environmental Science & Technology 52(10), 5859-5867.

Jamaly, S., Darwish, N.N., Ahmed, I. and Hasan, S.W., 2014. A short review on reverse osmosis pretreatment technologies. Desalination 354, 30-38.

Jeon, S.-i., Park, H.-r., Yeo, J.-g., Yang, S., Cho, C.H., Han, M.H. and Kim, D.K., 2013. Desalination via a new membrane capacitive deionization process utilizing flow-electrodes. Energy & Environmental Science 6(5).

Jia, B. and Zhang, W., 2016. Preparation and Application of Electrodes in Capacitive Deionization (CDI): a State-of-Art Review. Nanoscale Res Lett 11(1), 64.

Jones, E., Qadir, M., van Vliet, M.T.H., Smakhtin, V. and Kang, S.M., 2019. The state of desalination and brine production: A global outlook. Sci Total Environ 657, 1343-1356.

Joshi, R.K., Carbone, P., Wang, F.C., Kravets, V.G., Su, Y., Grigorieva, I.V., Wu, H.A., Geim, A.K. and Nair, R.R., 2014. Precise and ultrafast molecular sieving through graphene oxide membranes. Science 343(6172), 752-754.

Jurenka, R., Martella, S. and Rodriguez, R., 2001. Water treatment primer for communities in need, Bureau of Reclamation.

Kalfa, A., Shapira, B., Shopin, A., Cohen, I., Avraham, E. and Aurbach, D., 2020. Capacitive deionization for wastewater treatment: Opportunities and challenges. Chemosphere 241, 125003.

Karabelas, A.J., Koutsou, C.P., Kostoglou, M. and Sioutopoulos, D.C., 2018. Analysis of specific energy consumption in reverse osmosis desalination processes. Desalination 431, 15-21.

Karagiannis, I.C. and Soldatos, P.G., 2008. Water desalination cost literature: review and assessment. Desalination 223(1-3), 448-456.

Khan, M.A.M., Rehman, S. and Al-Sulaiman, F.A., 2018. A hybrid renewable energy system as a potential energy source for water desalination using reverse osmosis: A review. Renewable and Sustainable Energy Reviews 97, 456-477.

Khanzada, N.K., Khan, S.J. and Davies, P.A., 2017. Performance evaluation of reverse osmosis (RO) pre-treatment technologies for in-land brackish water treatment. Desalination 406, 44-50.

Kim, J., Park, K., Yang, D.R. and Hong, S., 2019. A comprehensive review of energy consumption of seawater reverse osmosis desalination plants. Applied Energy 254.

Kim, J.E., Phuntsho, S., Chekli, L., Choi, J.Y. and Shon, H.K., 2018. Environmental and economic assessment of hybrid FO-RO/NF system with selected inorganic draw solutes for the treatment of mine impaired water. Desalination 429, 96-104.

Kim, J.S., Chen, J. and Garcia, H.E., 2016. Modeling, control, and dynamic performance analysis of a reverse osmosis desalination plant integrated within hybrid energy systems. Energy 112, 52-66.

Kim, S.J., Ko, S.H., Kang, K.H. and Han, J., 2010a. Direct seawater desalination by ion concentration polarization. Nat Nanotechnol 5(4), 297-301.

Kim, Y.-J., Hur, J., Bae, W. and Choi, J.-H., 2010b. Desalination of brackish water containing oil compound by capacitive deionization process. Desalination 253(1), 119-123.

Koyuncu, I., Topacik, D., Turan, M., Celik, M.S. and Sarikaya, H.Z., 2001. Application of the membrane technology to control ammonia in surface water. Water Science and Technology: Water Supply 1(1), 117–124.

Kurth, C.J., Koehler, J.A., Zhou, M., Holmberg, B.A. and Burk, R.L., 2014. Hybrid nanoparticle TFC membranes, Nanoh2O, Inc., US.

Landaburu-Aguirre, J., García-Pacheco, R., Molina, S., Rodríguez-Sáez, L., Rabadán, J. and García-Calvo, E., 2016. Fouling prevention, preparing for re-use and membrane recycling. Towards circular economy in RO desalination. Desalination 393, 16-30.

Lee, J.-B., Park, K.-K., Eum, H.-M. and Lee, C.-W., 2006. Desalination of a thermal power plant wastewater by membrane capacitive deionization. Desalination 196(1-3), 125-134.

Lee, J., Jo, K., Lee, J., Hong, S.P., Kim, S. and Yoon, J., 2018a. Rocking-Chair Capacitive Deionization for Continuous Brackish Water Desalination. ACS Sustainable Chemistry & Engineering 6(8), 10815-10822.

Lee, M., Fan, C.S., Chen, Y.W., Chang, K.C., Chiueh, P.T. and Hou, C.H., 2019. Membrane capacitive deionization for low-salinity desalination in the reclamation of domestic wastewater effluents. Chemosphere 235, 413-422.

Lee, S., Choi, J., Park, Y.-G., Shon, H., Ahn, C.H. and Kim, S.-H., 2018b. Hybrid desalination processes for beneficial use of reverse osmosis brine: Current status and future prospects. Desalination.

Lester, Y., Shaulsky, E., Epsztein, R. and Zucker, I., 2020. Capacitive deionization for simultaneous removal of salt and uncharged organic contaminants from water. Separation and Purification Technology 237.

Li, H., Pan, L., Lu, T., Zhan, Y., Nie, C. and Sun, Z., 2011. A comparative study on electrosorptive behavior of carbon nanotubes and graphene for capacitive deionization. Journal of Electroanalytical Chemistry 653(1-2), 40-44.

Li, M. and Noh, B., 2012. Validation of model-based optimization of brackish water reverse osmosis (BWRO) plant operation. Desalination 304, 20-24.

Li, M. and Park, H.G., 2018. Pseudocapacitive Coating for Effective Capacitive Deionization. ACS Appl Mater Interfaces 10(3), 2442-2450.

Lin, S. and Elimelech, M., 2017. Kinetics and energetics trade-off in reverse osmosis desalination with different configurations. Desalination 401, 42-52.

Linge, K.L., Blythe, J.W., Busetti, F., Blair, P., Rodriguez, C. and Heitz, A., 2013. Formation of halogenated disinfection by-products during microfiltration and reverse osmosis treatment: Implications for water recycling. Separation and Purification Technology 104, 221-228.

Lior, N., 2017. Sustainability as the quantitative norm for water desalination impacts. Desalination 401, 99-111.

Liu, Y. and Wang, J., 2017. Energy-saving “NF/EDR” integrated membrane process for seawater desalination. Part II. The optimization of ED process. Desalination 422, 142-152.

Ma, J., Wang, L. and Yu, F., 2018. Water-enhanced performance in capacitive deionization for desalination based on graphene gel as electrode material. Electrochimica Acta 263, 40-46.

Ma, J., Ma, J., Zhang, C., Song, J., Collins, R.N. and Waite, T.D., 2019. Water Recovery Rate in Short-Circuited Closed-Cycle Operation of Flow-Electrode Capacitive Deionization (FCDI). Environ Sci Technol 53(23), 13859-13867.

Ma, J., Ma, J., Zhang, C., Song, J., Dong, W. and Waite, T.D., 2020. Flow-electrode capacitive deionization (FCDI) scale-up using a membrane stack configuration. Water Res 168, 115186.

Mauter, M.S., Zucker, I., Perreault, F.o., Werber, J.R., Kim, J.-H. and Elimelech, M., 2018. The role of nanotechnology in tackling global water challenges. Nature Sustainability 1(4), 166-175.

McCool, B.C., Rahardianto, A., Faria, J.I. and Cohen, Y., 2013. Evaluation of chemically-enhanced seeded precipitation of RO concentrate for high recovery desalting of high salinity brackish water. Desalination 317, 116-126.

Mehrabian-Nejad, H., Farhangi, B. and Farhangi, S., 2017. Application of PV and Solar Energy in Water Desalination System. Journal of Solar Energy Research 2(2), 13-18.

Metke, T., Westover, A.S., Carter, R., Oakes, L., Douglas, A. and Pint, C.L., 2016. Particulate-free porous silicon networks for efficient capacitive deionization water desalination. Sci Rep 6, 24680.

Metzger, M., Besli, M.M., Kuppan, S., Hellstrom, S., Kim, S., Sebti, E., Subban, C.V. and Christensen, J., 2020. Techno-economic analysis of capacitive and intercalative water deionization. Energy & Environmental Science.

Mi, B., 2014. Materials science. Graphene oxide membranes for ionic and molecular sieving. Science 343(6172), 740-742.

Mickley, M., 2004. Review of Concentrate Management Options, Texas Water Development Board.

Mickley, M.C., 2001. Membrane Concentrate Disposal: Practices and Regulation, U.S. Department of the Interior, Bureau of Reclamation.

Mikhaylin, S. and Bazinet, L., 2016. Fouling on ion-exchange membranes: Classification, characterization and strategies of prevention and control. Advances in Colloid and Interface

Science 229, 34-56.

Miller, J.E., 2003. Review of Water Resources and Desalination Technologies, p. 54, Sandia National Laboratory.

Mossad, M. and Zou, L., 2013. Study of fouling and scaling in capacitive deionisation by using dissolved organic and inorganic salts. Journal of Hazardous materials 244, 387-393.

Mubita, T.M., Porada, S., Biesheuvel, P.M., van der Wal, A. and Dykstra, J.E., 2018. Capacitive deionization with wire-shaped electrodes. Electrochimica Acta 270, 165-173.

Muñoz, I. and Fernández-Alba, A.R., 2008. Reducing the environmental impacts of reverse osmosis desalination by using brackish groundwater resources. Water Res 42(3), 801-811.

Nam, J.Y., Hwang, K.S., Kim, H.C., Jeong, H., Kim, H., Jwa, E., Yang, S., Choi, J., Kim, C.S., Han, J.H. and Jeong, N., 2019. Assessing the behavior of the feed-water constituents of a pilot-scale 1000-cell-pair reverse electrodialysis with seawater and municipal wastewater effluent. Water Res 148, 261-271.

Nanda, H., 2018. Reverse osmosis - the evolution which never stops, pp. 12-13.

Ng, C.A., Sun, D., Zhang, J., Wu, B. and Fane, A.G., 2010. Mechanisms of fouling control in membrane bioreactors by the addition of powdered activated carbon. Separation Science and Technology 45(7), 873-889.

Nie, C., Pan, L., Liu, Y., Li, H., Chen, T., Lu, T. and Sun, Z., 2012. Electrophoretic deposition of carbon nanotubes–polyacrylic acid composite film electrode for capacitive deionization.

Electrochimica Acta 66, 106-109.

Omosebi, A., Gao, X., Holubowitch, N., Li, Z., Landon, J. and Liu, K.L., 2017. Anion Exchange Membrane Capacitive Deionization Cells. Journal of The Electrochemical Society 164(9), E242-E247.

Oren, Y., Katz, V. and Daltrophe, N.C., 2001. Improved compact accelerated precipitation softening (CAPS). Desalination 139, 155-159.

Otitoju, T.A., Saari, R.A. and Ahmad, A.L., 2018. Progress in the modification of reverse osmosis (RO) membranes for enhanced performance. Journal of Industrial and Engineering Chemistry.

Pan, S.-Y., Snyder, S.W., Lin, Y.J. and Chiang, P.-C., 2018a. Electrokinetic desalination of brackish water and associated challenges in the water and energy nexus. Environmental Science: Water Research & Technology 4(5), 613-638.

Pan, S.-Y., Snyder, S.W., Ma, H.-W., Lin, Y.J. and Chiang, P.-C., 2018b. Energy-efficient resin wafer electrodeionization for impaired water reclamation. Journal of Cleaner Production 174, 1464-1474.

Pasta, M., Wessells, C.D., Cui, Y. and La Mantia, F., 2012. A desalination battery. Nano Lett 12(2), 839-843.

Pearce, G.K., 2008. UF/MF pre-treatment to RO in seawater and wastewater reuse applications: a comparison of energy costs. Desalination 222(1-3), 66-73.

Peñate, B., Subiela, V.J., Vega, F., Castellano, F., Domínguez, F.J. and Millán, V., 2014.

Uninterrupted eight-year operation of the autonomous solar photovoltaic reverse osmosis system in Ksar Ghilène (Tunisia). Desalination and Water Treatment, 1-8.

Porada, S., Zhang, L. and Dykstra, J.E., 2020. Energy consumption in membrane capacitive deionization and comparison with reverse osmosis. Desalination 488, 114383.

Pugazhenthiran, N., Sen Gupta, S., Prabhath, A., Manikandan, M., Swathy, J.R., Raman, V.K. and Pradeep, T., 2015. Cellulose derived graphenic fibers for capacitive desalination of brackish water. ACS Applied Materials & Interfaces 7(36), 20156-20163.

Purnama, A., Al-Barwani, H.H. and Smith, R., 2005. Calculating the environmental cost of seawater desalination in the Arabian marginal seas. Desalination 185(1-3), 79-86.

Qasim, T., Obeidat, M.S. and Smadi, H., 2018. Technical and Economical Evaluation of a Fresh-Water Production from Zero-Wastewater Reverse Osmosis System: A Feasibility Study in Jordan. Review of European Studies 10(2).

Qian, B.Q., Wang, G., Ling, Z., Dong, Q., Wu, T.T., Zhang, X. and Qiu, J.S., 2015. Sulfonated Graphene as Cation-Selective Coating: A New Strategy for High-Performance Membrane Capacitive Deionization. Advanced Materials Interfaces 2(16).

Qiblawey, H., Banat, F. and Al-Nasser, Q., 2011. Performance of reverse osmosis pilot plant powered by Photovoltaic in Jordan. Renewable Energy 36, 3452-3460.

Qin, M., Deshmukh, A., Epsztein, R., Patel, S.K., Owoseni, O.M., Walker, W.S. and Elimelech, M., 2019. Comparison of energy consumption in desalination by capacitive deionization and reverse osmosis. Desalination 455, 100-114.

Qu, D., Wang, J., Wang, L., Hou, D., Luan, Z. and Wang, B., 2009. Integration of accelerated precipitation softening with membrane distillation for high-recovery desalination of primary reverse osmosis concentrate. Separation and Purification Technology 67(1), 21-25.

Qu, Y., Baumann, T.F., Santiago, J.G. and Stadermann, M., 2015. Characterization of Resistances of a Capacitive Deionization System. Environmental Science & Technology 49(16), 9699-9706.

Quevedo, N., Sanz, J., Ocen, C., Lobo, A., Temprano, J. and Tejero, I., 2011. Reverse osmosis pretreatment alternatives: Demonstration plant in the seawater desalination plant in Carboneras, Spain. Desalination 265(1-3), 229-236.

Raluy, R.G., Serra, L. and Uche, J., 2005. Life cycle assessment of desalination technologies integrated with renewable energies. Desalination 183(1-3), 81-93.

Ramachandran, A., Oyarzun, D.I., Hawks, S.A., Campbell, P.G., Stadermann, M. and Santiago, J.G., 2019a. Comments on “Comparison of energy consumption in desalination by capacitive deionization and reverse osmosis”. Desalination 461, 30-36.

Ramachandran, A., Oyarzun, D.I., Hawks, S.A., Stadermann, M. and Santiago, J.G., 2019b. High water recovery and improved thermodynamic efficiency for capacitive deionization using variable flowrate operation. Water Res 155, 76-85.

Rao, P., Morrow, W.R., Aghajanzadeh, A., Sheaffer, P., Dollinger, C., Brueske, S. and Cresko, J., 2018. Energy considerations associated with increased adoption of seawater desalination in the

United States. Desalination 445, 213-224.

Ratajczak, P., Suss, M.E., Kaasik, F. and Béguin, F., 2019. Carbon electrodes for capacitive technologies. Energy Storage Materials 16, 126-145.

Richards, B.S., Capão, D.P.S. and Schäfer, A.I., 2008. Renewable Energy Powered Membrane Technology. 2. The Effect of Energy Fluctuations on Performance of a Photovoltaic Hybrid Membrane System. Environmental Science & Technology 42, 4563-4569.

Richards, B.S., Capão, D.P.S., Früh, W.G. and Schäfer, A.I., 2015. Renewable energy powered membrane technology: Impact of solar irradiance fluctuations on performance of a brackish water reverse osmosis system. Separation and Purification Technology 156, 379-390.

Roberts, A.J. and Slade, R.C.T., 2010. Effect of specific surface area on capacitance in asymmetric carbon/α-MnO2 supercapacitors. Electrochimica Acta 55(25), 7460-7469.

Ruiz-García, A. and Ruiz-Saavedra, E., 2015. 80,000h operational experience and performance analysis of a brackish water reverse osmosis desalination plant. Assessment of membrane replacement cost. Desalination 375, 81-88.

Ruiz-García, A., Melián-Martel, N. and Mena, V., 2018. Fouling characterization of RO membranes after 11 years of operation in a brackish water desalination plant. Desalination 430, 180-185.

Sablani, S.S., Goosen, M.F.A., Al-Belushi, R. and Wilf, M., 2001. Concentration polarization in ultrafiltration and reverse osmosis: a critical review. Desalination 141(3), 269-289.

Sarai Atab, M., Smallbone, A.J. and Roskilly, A.P., 2018. A hybrid reverse osmosis/adsorption desalination plant for irrigation and drinking water. Desalination 444, 44-52.

Sata, T., 2004. Ion exchange membranes: preparation, characterization, modification and application, Royal Society of Chemistry, Cambridge.

Schäfer, A.I., Broeckmann, A. and Richards, B.S., 2007. Renewable Energy Powered Membrane Technology. 1. Development and Characterization of a Photovoltaic Hybrid Membrane System. Environmental Science & Technology 41(3), 998-1003.

Segal, H., Birnhack, L., Nir, O. and Lahav, O., 2018. Intensification and energy minimization of seawater reverse osmosis desalination through high-pH operation: Temperature dependency and second pass implications. Chemical Engineering and Processing 131, 84-91.

Sethi, S., 2007. Desalination Product Water Recovery and Concentrate Volume Minimization (DRAFT), US Department of the Interior, Bureau of Reclamation.

Shaaban, S. and Yahya, H., 2017. Detailed analysis of reverse osmosis systems in hot climate conditions. Desalination 423, 41-51.

Shalaby, S.M., 2017. Reverse osmosis desalination powered by photovoltaic and solar Rankine cycle power systems: A review. Renewable and Sustainable Energy Reviews 73, 789-797.

Shannon, M.A., Bohn, P.W., Elimelech, M., Georgiadis, J.G., Marinas, B.J. and Mayes, A.M., 2008. Science and technology for water purification in the coming decades. Nature 452(7185), 301-310.

Shiu, H.-Y., Lee, M., Chao, Y., Chang, K.-C., Hou, C.-H. and Chiueh, P.-T., 2019. Hotspot analysis and improvement schemes for capacitive deionization (CDI) using life cycle assessment. Desalination 468.

Silva Herran, D. and Nakata, T., 2012. Design of decentralized energy systems for rural electrification in developing countries considering regional disparity. Applied Energy 91(1), 130-145.

Simoncelli, M., Ganfoud, N., Sene, A., Haefele, M., Daffos, B., Taberna, P.L., Salanne, M., Simon, P. and Rotenberg, B., 2018. Blue Energy and Desalination with Nanoporous Carbon Electrodes: Capacitance from Molecular Simulations to Continuous Models. Physical Review X 8(2).

Siria, A., Bocquet, M.-L. and Bocquet, L., 2017. New avenues for the large-scale harvesting of blue energy. Nature Reviews Chemistry 1(11).

Smith, R., Purnama, A. and Al-Barwani, H.H., 2007. Sensitivity of hypersaline Arabian Gulf to seawater desalination plants. Applied Mathematical Modelling 31(10), 2347-2354.

Sobhani, R., Abahusayn, M., Gabelich, C.J. and Rosso, D., 2012. Energy Footprint analysis of brackish groundwater desalination with zero liquid discharge in inland areas of the Arabian Peninsula. Desalination 291, 106-116.

Spiegler, K.S. and El-Sayed, Y.M., 2001. The energetics of desalination processes. Desalination 134(1-3), 109-128.

Stokes, J.R. and Horvath, A., 2009. Energy and air emission effects of water supply. Environ Sci

Technol 43(8), 2680-2687.

Subramani, A., Voutchkov, N. and Jacangelo, J.G., 2014. Desalination energy minimization using thin film nanocomposite membranes. Desalination 350, 35-43.

Suss, M., Porada, S., Sun, X., Biesheuvel, P., Yoon, J. and Presser, V., 2015a. Water desalination via capacitive deionization: what is it and what can we expect from it? Energy & Environmental Science 8(8), 2296-2319.

Suss, M.E., Baumann, T.F., Bourcier, W.L., Spadaccini, C.M., Rose, K.A., Santiago, J.G. and Stadermann, M., 2012. Capacitive desalination with flow-through electrodes. Energy & Environmental Science 5(11), 9511.

Suss, M.E., Porada, S., Sun, X., Biesheuvel, P.M., Yoon, J. and Presser, V., 2015b. Water desalination via capacitive deionization: what is it and what can we expect from it? Energy Environ. Sci. 8(8), 2296-2319.

Sweity, A., Zere, T.R., David, I., Bason, S., Oren, Y., Ronen, Z. and Herzberg, M., 2015. Side effects of antiscalants on biofouling of reverse osmosis membranes in brackish water desalination. Journal of Membrane Science 481, 172-187.

Tan, C., He, C., Tang, W., Kovalsky, P., Fletcher, J. and Waite, T.D., 2018. Integration of photovoltaic energy supply with membrane capacitive deionization (MCDI) for salt removal from brackish waters. Water Research 147, 276-286.

Tan, C., He, C., Fletcher, J. and Waite, T.D., 2020. Energy recovery in pilot scale membrane CDI treatment of brackish waters. Water Res 168, 115146.

Tarnacki, K., Meneses, M., Melin, T., van Medevoort, J. and Jansen, A., 2012. Environmental assessment of desalination processes: Reverse osmosis and Memstill®. Desalination 296, 69-80.

Teychene, B., Collet, G., Gallard, H. and Croue, J.-P., 2013. A comparative study of boron and arsenic (III) rejection from brackish water by reverse osmosis membranes. Desalination 310, 109-114.

Triki, Z., Bouaziz, M.N. and Boumaza, M., 2013. Techno-economic feasibility of wind-powered reverse osmosis brackish water desalination systems in southern Algeria. Desalination and Water Treatment 52(7-9), 1745-1760.

USDOE, 2014. The Water-Energy Nexus: Challenges and Opportunities, p. 262, U.S. Department of Energy, USA.

Vakilifard, N., Anda, M., A. Bahri, P. and Ho, G., 2018. The role of water-energy nexus in optimising water supply systems – Review of techniques and approaches. Renewable and Sustainable Energy Reviews 82, 1424-1432.

van Hoof, S.C.J.M., Minnery, J.G. and Mack, B., 2001. Dead-end ultrafiltration as alternative pre-treatment to reverse osmosis in seawater desalination: a case study. Desalination 139(1-3), 161-168.

Vince, F., Marechal, F., Aoustin, E. and Bréant, P., 2008. Multi-objective optimization of RO desalination plants. Desalination 222(1-3), 96-118.

Volfkovich, Y.M., Rychagov, A.Y., Mikhalin, A.A., Kardash, M.M., Kononenko, N.A., Ainetdinov, D.V., Shkirskaya, S.A. and Sosenkin, V.E., 2018. Capacitive deionization of water using mosaic membrane. Desalination 426, 1-10.

Voltea, 2015. Voltea's Technical Bulletin: Technology Comparison. Voltea B.V. (ed), p. 8, The Netherlands.

Walha, K., Amar, R.B., Firdaous, L., Quéméneur, F. and Jaouen, P., 2007. Brackish groundwater treatment by nanofiltration, reverse osmosis and electrodialysis in Tunisia: performance and cost comparison. Desalination 207(1-3), 95-106.

Wang, H., Chung, T.S., Tong, Y.W., Jeyaseelan, K., Armugam, A., Chen, Z., Hong, M. and Meier, W., 2012. Highly permeable and selective pore-spanning biomimetic membrane embedded with aquaporin Z. Small 8(8), 1185-1190, 1125.

Wang, L. and Lin, S., 2018. Intrinsic tradeoff between kinetic and energetic efficiencies in membrane capacitive deionization. Water Res 129, 394-401.

Wang, L., Dykstra, J.E. and Lin, S., 2019a. Energy Efficiency of Capacitive Deionization. Environ Sci Technol 53(7), 3366-3378.

Wang, T., Zhang, C., Bai, L., Xie, B., Gan, Z., Xing, J., Li, G. and Liang, H., 2019b. Scaling behavior of iron in capacitive deionization (CDI) system. Water Res 171, 115370.

Wang, Y., Vázquez-Rodríguez, I., Santos, C., García-Quismondo, E., Palma, J., Anderson, M.A. and

Lado, J.J., 2019c. Graphite felt 3D framework composites as an easy to scale capacitive deionization electrode for brackish water desalination. Chemical Engineering Journal.

Wei, Q.J., McGovern, R.K. and Lienhard V, J.H., 2017. Saving energy with an optimized two-stage reverse osmosis system. Environmental Science: Water Research & Technology 3(4), 659-670.

Weinstein, L. and Dash, R., 2013. Capacitive Deionization: Challenges and Opportunities, pp. 34-37.

Welgemoed, T.J. and Schutte, C.F., 2005. Capacitive Deionization Technology™: An alternative desalination solution. Desalination 183(1-3), 327-340.

Wilf, M., 2004. Fundamentals of RO–NF technology, Limassol.

Wilf, M. and Bartels, C., 2006. Integrated Membrane Desalination Systems -- Current Status and Projected Development.

Wolf, P.H. and Siverns, S., 2004. The new generation for reliable RO pre-treatment, Limassol.

Wu, B., Suwarno, S.R., Tan, H.S., Kim, L.H., Hochstrasser, F., Chong, T.H., Burkhardt, M., Pronk, W. and Fane, A.G., 2017a. Gravity-driven microfiltration pretreatment for reverse osmosis (RO) seawater desalination: Microbial community characterization and RO performance. Desalination 418, 1-8.

Wu, T., Wang, G., Dong, Q., Zhan, F., Zhang, X., Li, S., Qiao, H. and Qiu, J., 2017b. Starch Derived Porous Carbon Nanosheets for High-Performance Photovoltaic Capacitive Deionization. Environmental science & technology 51(16), 9244-9251.

Xie, Z., Shang, X., Yan, J., Hussain, T., Nie, P. and Liu, J., 2018. Biomass-derived porous carbon anode for high-performance capacitive deionization. Electrochimica Acta 290, 666-675.

Xu, P., Drewes, J.E., Heil, D. and Wang, G., 2008a. Treatment of brackish produced water using carbon aerogel-based capacitive deionization technology. Water Res 42(10-11), 2605-2617.

Xu, P., Drewes, J.E., Heil, D. and Wang, G., 2008b. Treatment of brackish produced water using carbon aerogel-based capacitive deionization technology. Water Research 42(10), 2605-2617.

Xu, X., Pan, L., Liu, Y., Lu, T., Sun, Z. and Chua, D.H., 2015. Facile synthesis of novel graphene sponge for high performance capacitive deionization. Sci Rep 5, 8458.

Xu, X., Wang, M., Liu, Y., Lu, T. and Pan, L., 2017. Ultrahigh Desalinization Performance of Asymmetric Flow-Electrode Capacitive Deionization Device with an Improved Operation Voltage of 1.8 V. ACS Sustainable Chemistry & Engineering 5(1), 189-195.

Xu, X., Allah, A.E., Wang, C., Tan, H., Farghali, A.A., Khedr, M.H., Malgras, V., Yang, T. and Yamauchi, Y., 2019. Capacitive deionization using nitrogen-doped mesostructured carbons for highly efficient brackish water desalination. Chemical Engineering Journal 362, 887-896.

Yang, H.Y., Han, Z.J., Yu, S.F., Pey, K.L., Ostrikov, K. and Karnik, R., 2013. Carbon nanotube membranes with ultrahigh specific adsorption capacity for water desalination and purification. Nat Commun 4, 2220.

Yangali-Quintanilla, V., Li, Z., Valladares, R., Li, Q. and Amy, G., 2011. Indirect desalination of Red Sea water with forward osmosis and low pressure reverse osmosis for water reuse.

Desalination 280(1-3), 160-166.

Yu, T.-H., Shiu, H.-Y., Lee, M., Chiueh, P.-T. and Hou, C.-H., 2016. Life cycle assessment of environmental impacts and energy demand for capacitive deionization technology. Desalination 399, 53-60.

Yun, T.I., Gabelich, C.J., Cox, M.R., Mofidi, A.A. and Lesan, R., 2006. Reducing costs for large-scale desalting plants using large-diameter, reverse osmosis membranes. Desalination 189(1-3), 141-154.

Zhang, C., He, D., Ma, J., Tang, W. and Waite, T.D., 2018. Faradaic reactions in capacitive deionization (CDI) - problems and possibilities: A review. Water Res 128, 314-330.

Zhang, C., Wu, L., Ma, J., Wang, M., Sun, J. and Waite, T.D., 2020. Evaluation of long-term performance of a continuously operated flow-electrode CDI system for salt removal from brackish waters. Water Res 173, 115580.

Zhang, W., Mossad, M. and Zou, L., 2013. A study of the long-term operation of capacitive deionisation in inland brackish water desalination. Desalination 320, 80-85.

Zhang, W. and Jia, B., 2015. Toward anti-fouling capacitive deionization by using visible-light reduced TiO2/graphene nanocomposites. MRS Communications 5(4), 613-617.

Zhao, R., Porada, S., Biesheuvel, P.M. and van der Wal, A., 2013. Energy consumption in membrane capacitive deionization for different water recoveries and flow rates, and comparison with reverse osmosis. Desalination 330, 35-41.

Zheng, S., Tu, Q., Urban, J.J., Li, S. and Mi, B., 2017. Swelling of Graphene Oxide Membranes in Aqueous Solution: Characterization of Interlayer Spacing and Insight into Water Transport Mechanisms. ACS Nano 11(6), 6440-6450.

Zhou, J., Chang, V.W.C. and Fane, A.G., 2011a. Environmental life cycle assessment of reverse osmosis desalination: The influence of different life cycle impact assessment methods on the characterization results. Desalination 283, 227-236.

Zhou, J., Chang, V.W.C. and Fane, A.G., 2011b. Environmental life cycle assessment of brackish water reverse osmosis desalination for different electricity production models. Energy & Environmental Science 4(6).

Zhu, G., Wang, H.Y., Xu, H.F. and Zhang, L., 2018. Enhanced capacitive deionization by nitrogen-doped porous carbon nanofiber aerogel derived from bacterial-cellulose. Journal of Electroanalytical Chemistry 822, 81-88.

Zornitta, R.L., Lado, J.J., Anderson, M.A. and Ruotolo, L.A.M., 2016. Effect of electrode properties and operational parameters on capacitive deionization using low-cost commercial carbons. Separation and Purification Technology 158, 39-52.

Zornitta, R.L., Srimuk, P., Lee, J., Kruner, B., Aslan, M., Ruotolo, L.A.M. and Presser, V., 2018. Charge and Potential Balancing for Optimized Capacitive Deionization Using Lignin-Derived, Low-Cost Activated Carbon Electrodes. ChemSusChem 11(13), 2101-2113.

Zuo, Y., Cheng, S. and Logan, B.E., 2008. Ion exchange membrane cathodes for scalable microbial fuel cells. Environmental Science & Technology 42(18), 6967-6972.

Figure

Figure 1.

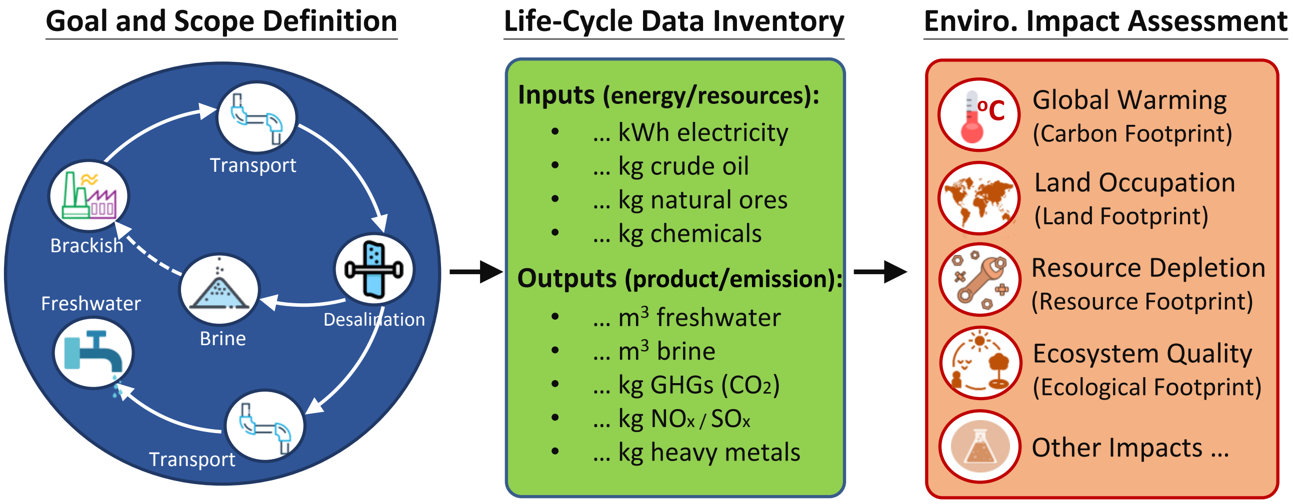

Fig. 1 Assessment of environmental impacts and carbon footprints for a desalination plant.

Figure 2.

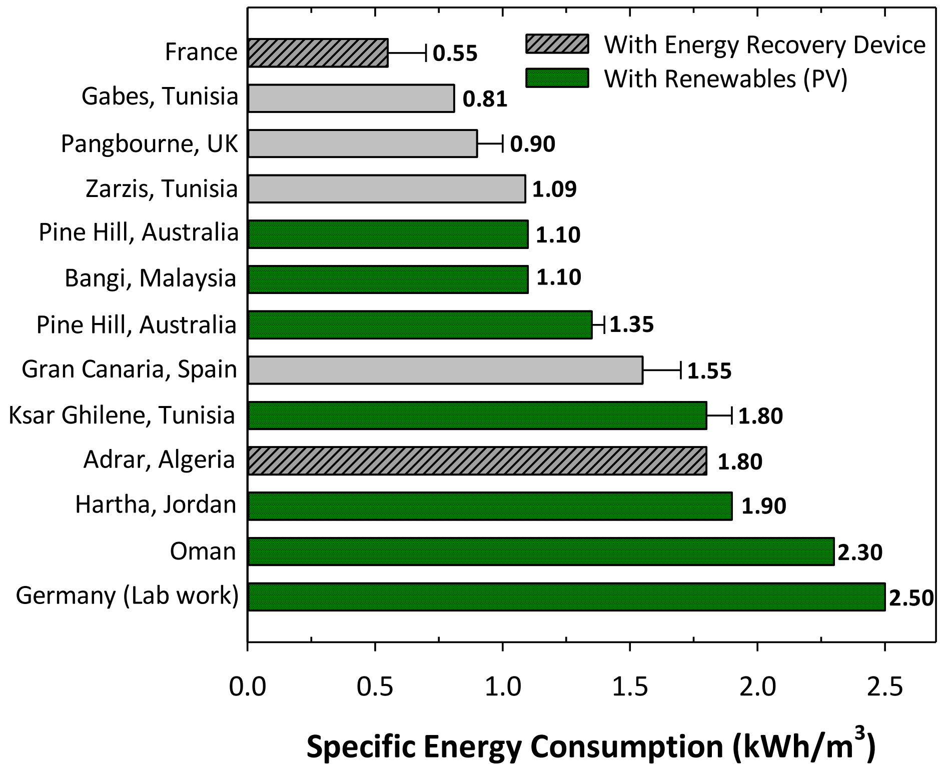

Fig. 2 Specific energy consumption of several BWRO desalination plants around the world. Data source: France (Vince et al. 2008); Gabes, Tunisia (Walha et al. 2007); Pangbourne, UK (Pearce 2008); Zarzis, Tunisia (Walha et al. 2007); Pine Hill, Australia (Schäfer et al. 2007); Bangi, Malaysia (Alghoul et al. 2016); Pine Hill, Australia (Richards et al. 2008); Gran Canaria, Spain (Ruiz-García and Ruiz-Saavedra 2015); Ksar Ghilène, Tunisia (Peñate et al. 2014); Adrar, Algeria (Triki et al. 2013); Hartha, Jordan (Qiblawey et al. 2011); Oman (Alghoul et al. 2009); Germany (Richards et al. 2015).

Figure 3.

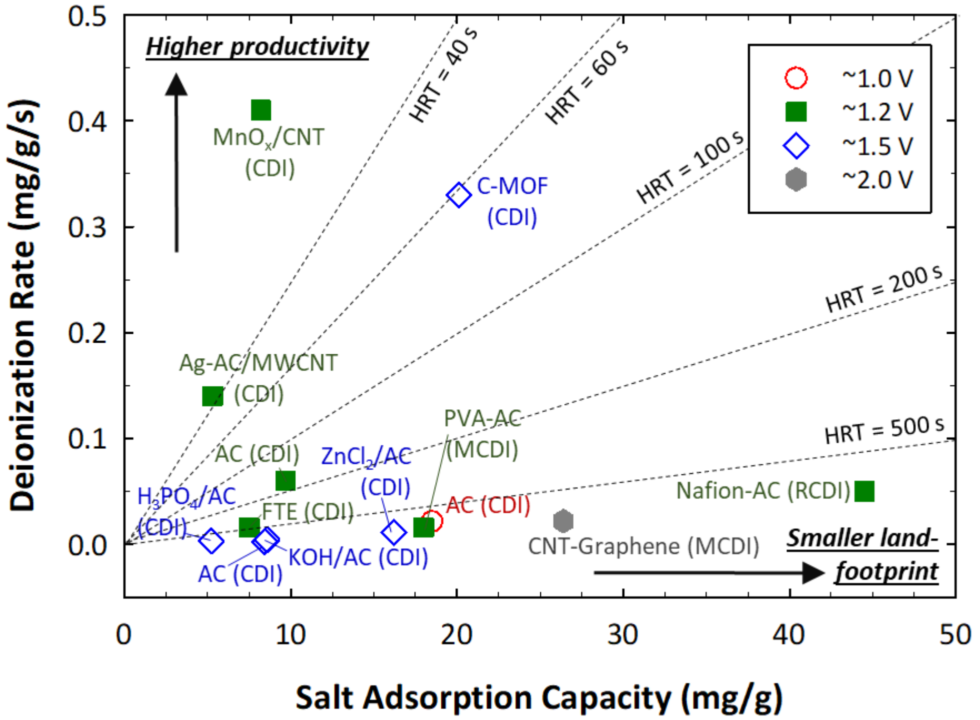

Fig. 3 CDI ragone plot (Kim-Yoon plot) for salt adsorption capacity (mg/g) and deionization rate (mg/g/s) at different applied voltages. HRT: hydraulic retention time (sec) for adsorption step (a half cycle of CDI operation). See operating details and associated references in the Supplementary Information (Table S3).

Figure 4.

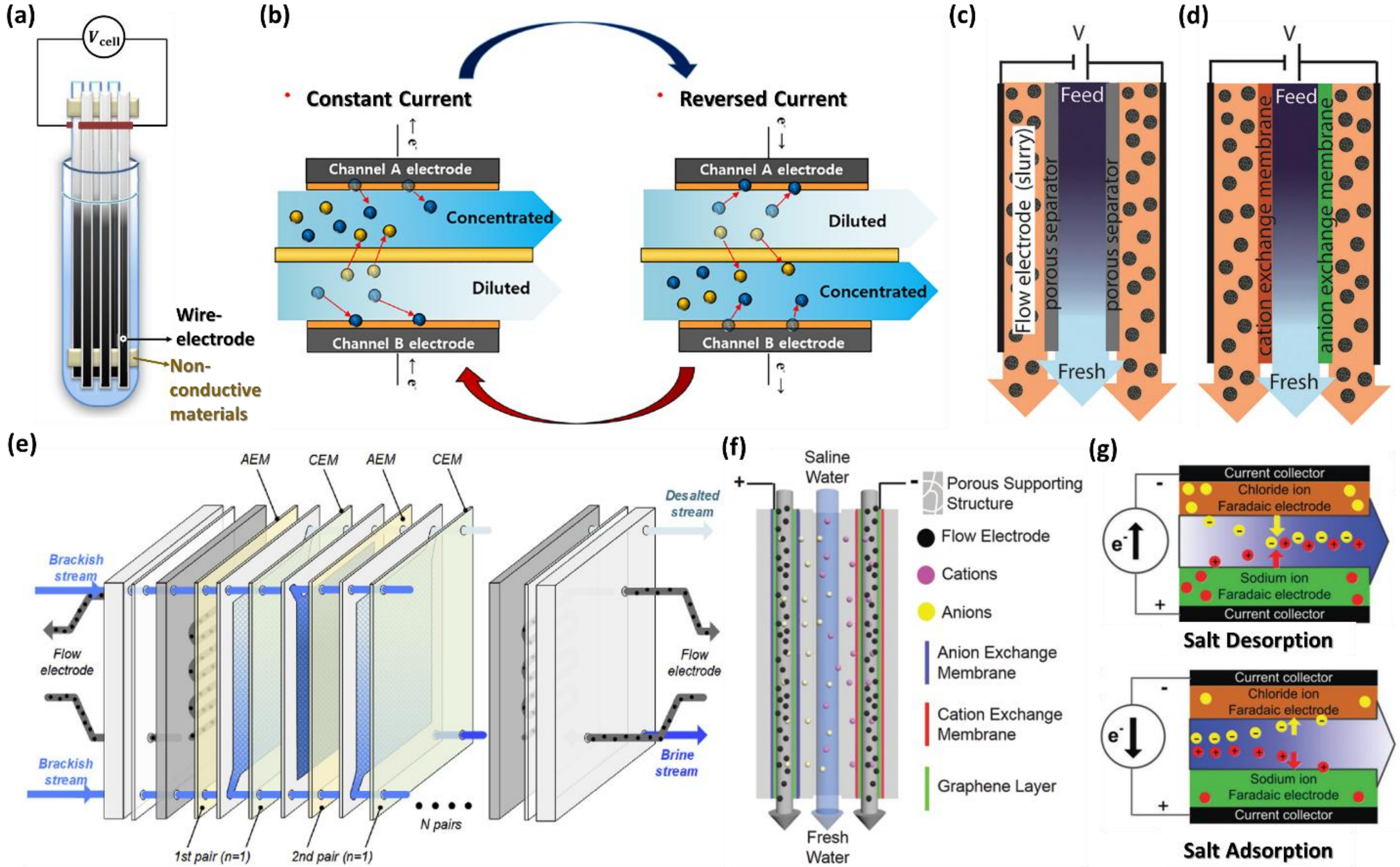

Fig. 4 Schematic diagrams of novel CDI architectures: **(a)** wire-shaped electrode CDI. Adapted from the ref. (Mubita et al. 2018) **(b)** rocking-chair capacitive deionization (RCDI). Adapted from the ref. (Lee et al. 2018a) **(c-d)** flow electrode CDI. Adapted from the ref. (Suss et al. 2015b) **(e)** flow electrode CDI with ion exchange membranes in stacks. Adapted from the ref. (Ma et al. 2020) **(f)** honeycomb-shaped CDI. Adapted from the ref. (Cho et al. 2017) **(g)** desalination batteries/generators. Adapted from the ref. (Chen et al. 2017)

Figure 5.

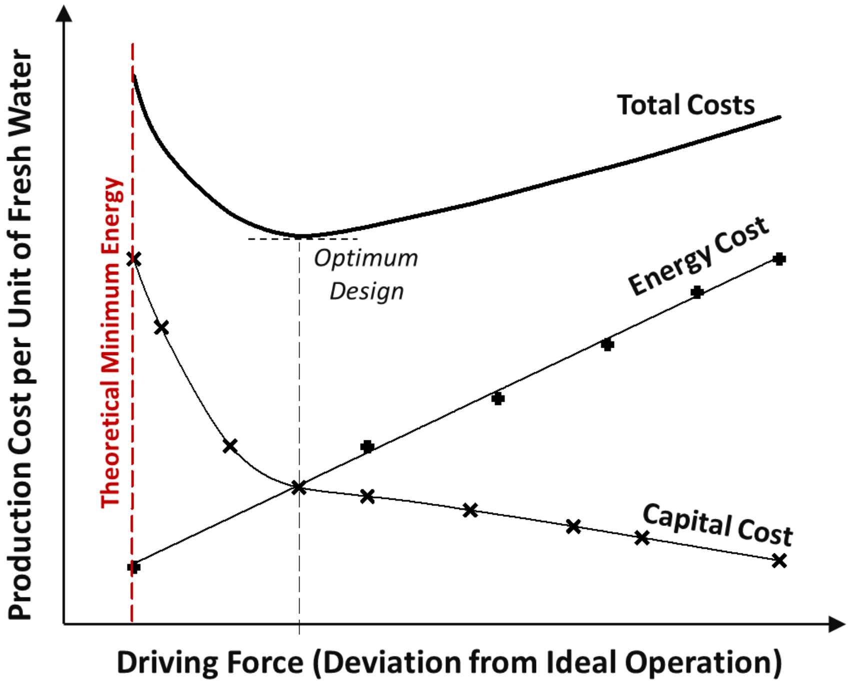

Fig. 5 Trade-off between capital costs and energy consumption at a given production rate of fresh water for practical desalination systems from the thermodynamic point of view. The desalination system becomes a reversible process when the driving force (abscissa) approaches zero. Adapted from the ref. (Spiegler and El-Sayed 2001)

Figure 6.

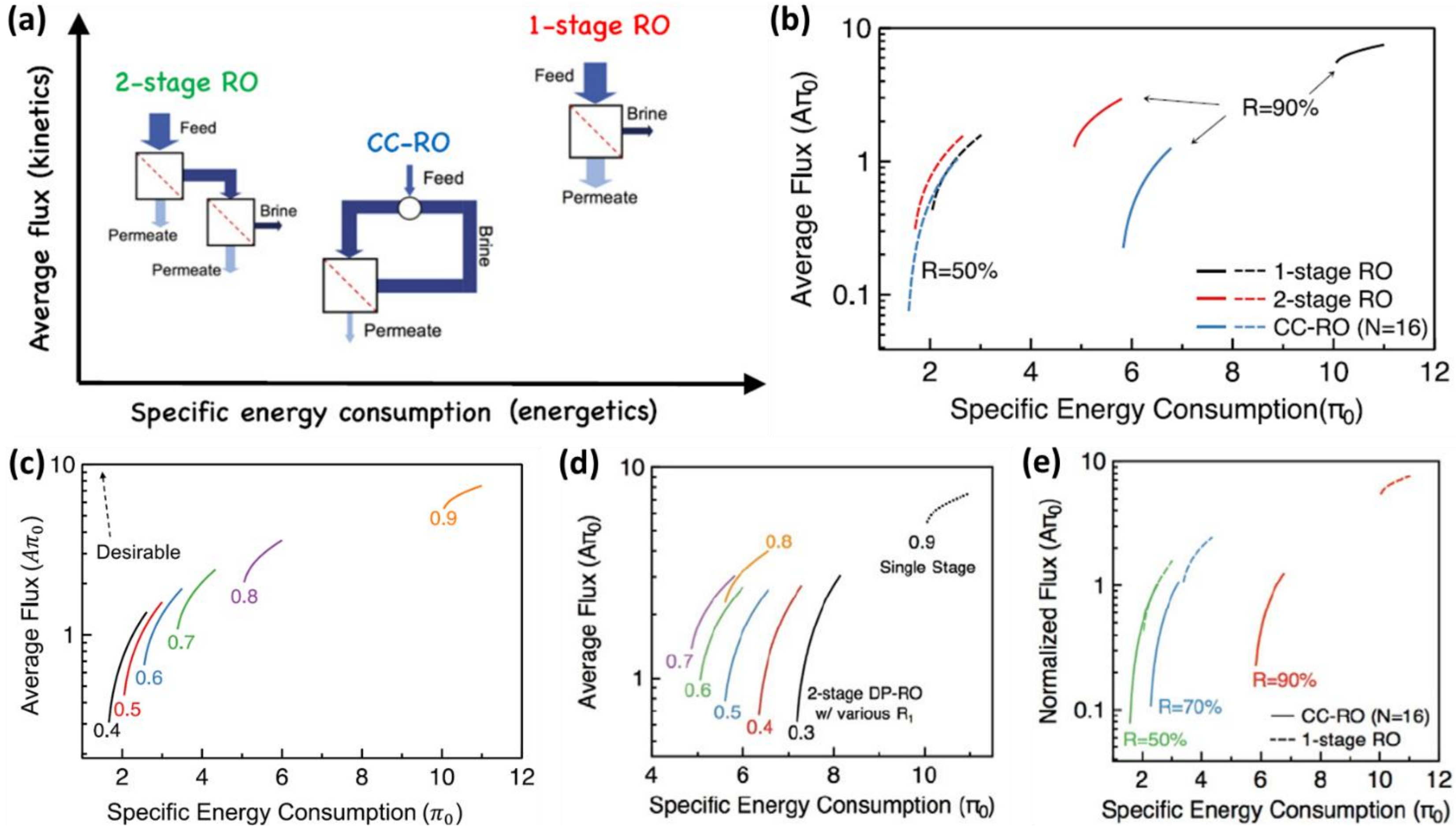

Fig. 6 (a) Relationship of kinetics and energetics for different RO configurations. (b) relationship of average water flux for different RO configurations at recovery ratios (denoted as R) of 50% and 90%. (c) a single-stage RO process with different recovery ratios from 40% to 90%. (d) a 2-stage RO process with different recovery ratios from 30% to 80%. (e) a closed-circle RO process with different recovery ratios from 50% to 90%. Adapted from ref. (Lin and Elimelech 2017) with permission from Elsevier.

Figure 7.

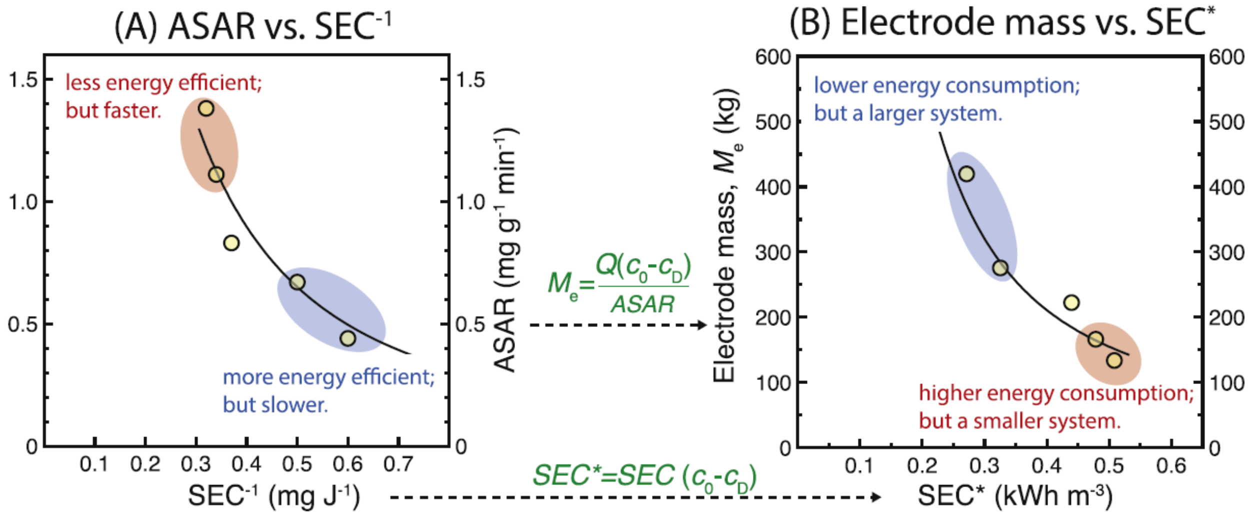

Fig. 7 Relationship of (A) the trade-off between average salt adsorption rate (ASAR) vs. SEC^{-1}, and (B) the trade-off between the scale of an MCDI system and energy consumption. Experimental conditions: c_0 is 20 mM, c_D is 10 mM, and the electrode material is activated carbon particle adsorbent composite (PACMM^{TM}, Irvine, CA, USA). The treatment capacity is about 263 L min^{-1}. Adapted from ref. (Wang and Lin 2018) with permission from Elsevier.

Table
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Table 1. Performance and economics of reverse osmosis (RO) for brackish water desalination plants. [a]

Location	C_i (g/L)[b]	C_0 (g/L)[b]	Capacity (m^3/d)	Recovery (%)[c]	P_f (bar)	SEC (kWh/m^3)	Water cost (US$/$m^3$)	Power cost (US$/kWh)	Year of data access	Types of Membrane	ERD	RE
Metropolitan CA, USA (Yun et al. 2006)	0.7	0.5	700300	85	n/a	n/a	0.12−0.13	n/a	2005	ULP-G3	no	no
Pangbourne, UK (Pearce 2008)	0.9–2.2	n/a	n/a	n/a	n/a	0.8–1.0	n/a	n/a	2008	n/a	no	no
Oman (Alghoul et al. 2009)	1.0	0.03	5	65−70	12	2.3	n/a	n/a	2008	n/a	n/a	PV
Bradford, UK (Al-Obaidi et al. 2018)	1.1	0.002	1200	68	9−10	n/a	n/a	n/a	2018	TMG20D	no	no
Hartha, Jordan (Qiblawey et al. 2011)	1.7	0.01	0.33	22	3.3	1.9	n/a	n/a	2007	TFM-100	no	PV
UKM, Bangi, Malaysia (Alghoul et al. 2016)	2.0	0.05	4.6	>90	8.3	1.1	n/a	n/a	2013−2014	BW30-4040	no	PV
South Shouneh, Jordan (Alsarayreh et al. 2017)	2.0	0.43	960	60	20	n/a	0.31–0.38[d]	n/a	2017	n/a	no	PV
Adrar, Algeria (Triki et al. 2013)	2.1‡	~0.15‡	8660	60		1.8	0.66	0.14–0.18	2014	n/a	yes	WN
Gabes, Tunisia (Walha et al. 2007)	2.7	0.33	n/a	66.6	15	0.81	n/a	n/a	2007	Nanomax 95	n/a	no
Sharjah Emirate, UAE (Almulla et al. 2003)	3.0	0.07	31420	83*	18−20	n/a	0.37	0.055	2002	n/a	yes	no
France (Vince et al. 2008)	3.0	0.3	35000	74–82	10–17	0.4–0.7	0.22–0.27	0.03–0.04	2006	BW30LE	yes	no
Gran Canaria, Spain (Ruiz-García and Ruiz-Saavedra 2015)	3.1–7.8	0.50−0.15‡	360	~60	14−24	1.4−1.7	n/a	n/a	2004−2015	BW30-400	no	no
Ksar Ghilène, Tunisia (Peñate et al. 2014)	4.0–4.5	n/a	50	68−70	12−13	1.7−1.9	n/a	n/a	2006–2013	n/a	no	PV
Pine Hill, Australia	4.1‡	0.9‡	1.87	58	7	1.3–1.4	n/a	n/a	2005	TFC-S	no	PV

(Richards et al. 2008)												
Germany (Lab work) (Richards et al. 2015)	5.0	1.0	n/a	~20	<11	2.5	n/a	n/a	2015	BW30	no	PV
Pine Hill, Australia (Schäfer et al. 2007)	5.3	0.26	4.0	~55	10	1.1	n/a	n/a	2007	ESPA4	no	PV
Zarzis, Tunisia (Walha et al. 2007)	5.3	0.64	n/a	50	15	1.09	n/a	n/a	2007	Nanomax 95	n/a	no
San Joaquin, USA (McCool et al. 2013)	6.7–14.4	n/a	n/a	83−93	n/a	n/a	0.67–0.74	0.11	2012	BW30-400	no	no

[a] "n/a" denotes "not available". [b] Salinity presented with † are converted from ppm (assumed that 1 ppm equals 1 mg/L), while ‡ are converted from conductivity (assumed that 1 mS/cm equals 0.5 mg/L). [c] "*" denotes "with water recovery device. [d] Assumed 1.000 € equals 1.258 USD. Acronyms: C_i: feed salinity; C_o: effluent salinity; P_f: feed pressure; ERD: energy recovery device; RE: renewable-assisted system; PV: photovoltaic systems; WN: wind-powered systems.

Table 2. Total costs for producing one m^3 of clean water by BWRO.

Plant size	Production capacity (m^3/d)	Total cost (USD/m^3) [a]	Reference
Large	~700,000	0.12−0.13 [b]	(Yun et al. 2006)
	4,500–104,000	0.29−0.63	(Arroyo and Shirazi 2012)
	40,000–46,000	0.26−0.54	(Al-Karaghouli and Kazmerski 2013, Karagiannis and Soldatos 2008)
Medium	35,000	0.22−0.24	(Vince et al. 2008)
	~31,420	0.37	(Almulla et al. 2003)
	3,785	0.26−0.35	(Drewes et al. 2009)
	20−1,200	0.78−1.33	(Al-Karaghouli and Kazmerski 2013, Jaber and Ahmed 2004, Karagiannis and Soldatos 2008)
	30−100	1.99−2.23	(Jaber and Ahmed 2004)
Small	<20	5.66−12.98	(Karagiannis and Soldatos 2008)
	Few m^3	0.56−12.99	(Al-Karaghouli and Kazmerski 2013)

[a] Assumed 1.000 € equals 1.258 USD. [b] Costs include capital, operation and maintenance.

Table 3. Membrane lifespan and replacement costs of RO for brackish water desalination.

Membrane materials [a]	Water recovery (%)	Lifespan of membrane (year)	Replacement rate (% per year)	Cost of membrane replacement (USD/m^3)	Reference
n/a (Polymer)	-	5−10	10	-	(Landaburu-Aguirre et al. 2016)
Polyamide TFC	75	7	-	0.012–0.015	(Drewes et al. 2009)
Polyamide TFC	75−90	5−7	5	0.01	(Greenlee et al. 2009)
Polyamide TFC	> 80	3	-	-	(McCool et al. 2013)
Polyamide TFC	74−82	5	-	0.027–0.043	(Vince et al. 2008)

[a] n/a: not specified; TFC: Thin-Film Composite.

Table 4. Comparison of total costs for a brackish water RO plant.

Categories	Critical factors	Costs (per m^3 of produced water)[b]	Reference
Capital	Daily production capacity	150−165 USD	(Drewes et al. 2009)
		240−400 USD	(Greenlee et al. 2009, Karagiannis and Soldatos 2008)
		400−455 USD	(Vince et al. 2008)
Energy	Operation configuration, etc.	• 0.26−1.33 USD (small units) for CS • ~2.48 USD (small units) for GT • 5.57−12.78 USD (small units) for PV	(Karagiannis and Soldatos 2008)
		0.13−0.14 USD (medium units) for CS	(Vince et al. 2008)
Chemicals for CIP [a]	Fouling and scaling due to raw water quality, cleaning frequency, membrane type, regulations, etc.	0.008−0.050 USD	(Greenlee et al. 2009)
		0.04 USD	(Vince et al. 2008)
		0.113−0.200 USD	(Drewes et al. 2009)
Membrane replacement	End-of-life replacement	0.050−0.430 USD (0.04−0.34 € [c])	(Avlonitis et al. 2003)
		0.008−0.050 USD	(Greenlee et al. 2009)
		0.027−0.043 USD	(Vince et al. 2008)
Maintenance	Instrumentation, electricity, equipment, pumps, accessories, etc.	0.01 USD	(Vince et al. 2008)
		~8% of total costs	(Wilf 2004)
Labor	Plant capacity, etc.	0.013 USD (in France)	(Vince et al. 2008)
		0.028 USD (in USA)	(Drewes et al. 2009)
		1−5% of total costs	(Wilf 2004)
Miscellaneous	Insurance, etc.	0.5% of the total capital cost	(Vince et al. 2008)

[a] CIP: Clean-in-place. [b] CS: conventional source of energy such as gas, oil, and coal. GT: geothermal energy. PV: electricity from photovoltaics. [c] Assumed 1.000 € equals 1.258 USD.

Table 5. Carbon footprint of RO brackish water desalination.

Plant background [a]	Life cycle stages [b]	Energy source / mix	Carbon footprint (kg CO_2-eq)	Reference
Q_0= 10000; C_i=15; SEC=2.0	CS, OMS, and DS	US electricity	1.58	(Zhou et al. 2011b)
Q_0= 10000; C_i=15; SEC=2.0	CS, OMS, and DS	Singapore electricity	1.15	(Zhou et al. 2011a)
Groundwater	CS and OMS	California electricity	1.628	(Stokes and Horvath 2009)
Q_0= 20000; C_i=15.3; SEC=2.0	CS, OMS, and DS	Spain electricity	0.84–1.60 [c]	(Muñoz and Fernández-Alba 2008)
Q_0= 45500; SEC=4.0	CS and OMS	Different energy types	0.08–1.78	(Raluy et al. 2005)
Q_0= 100; C_i=2.0; SEC=1.0	PT and OMS	-	0.624	(Tarnacki et al. 2012)

[a] Q_0: plant capacity (m^3/d); C_i: feed concentration (g/L); SEC: specific energy consumption (kWh/m^3). [b] CS: construction stage (e.g., construction and land preparation); PT: pretreatment of feedwater; OMS: operation and maintenance stage (e.g., chemicals, membranes and electricity); DS: dismantling stage (e.g., used construction materials and membranes). [c] based on the uncertainty analysis with Monte-Carlo simulation with a 95% confidence.

Table 6. Performance of various capacitive technologies for brackish water desalination.[a]

Type [b]	Location	C_i (g/L) [c]	C_0 (g/L) [c]	Treatment Capacity (m^3/d)	Operation time (hr) [d]	Water Recovery (%)	Cell voltage (V)	R_d (mg/g/s)	SEC (kWh/m^3)	Water cost (US$/$m^3$)	Types of Membrane	Published Year
CDI	Spain (Wang et al. 2019c)	2.1	1.5	0.002	CT	50	-	0.068	0.60	-	none	2019
	Taiwan (Yu et al. 2016)	0.58	-	0.010	n/a	60	1.3	-	0.4	-	none	2016
	Taiwan (Chen et al. 2019)	0.58	0.39	0.014	CT	-	1.2	0.0112	0.09	-	none	2019
	Germany (Zornitta et al. 2018)	0.58	-	0.014	CT	-	1.0	0.0217	0.29 (1.8 kJ/g-salt)	-	none	2018
	India (Alencherry et al. 2017)	0.6	-	0.014	1 (BT)	-	1.2	0.1401	-	-	none	2017
	Shanghai, China (Xie et al. 2018)	0.5	-	0.018	BT	-	1.5	0.0112	-	-	none	2018
	Nanjing, China (Xu et al. 2019)	0.58	0.42	0.029	0.5 (BT)	-	1.2	0.012	-	-	none	2019
	Kentucky, USA (Caudill 2018)	4.0	1.0	5.76	CT	-	0.9	-	-	0.005	none	2018
	The Netherland (Voltea 2015)	4.0	1.0	158	CT	90	-	-	0.1–0.2	-	none	2015
	Shandong, China (EST 2009)	0.80–1.86 ‡	0.21–0.48 ‡	2400	CT	69–75	1.5–2.0	-	1.04–1.52 (1.33)	-	none	2006
	Shanghai, China (EST 2009)	0.67 ‡	0.14 ‡	3600	CT	78.5	1.5–2.0	-	0.55	0.069	none	2009
MCDI	Texas, USA (Jain et al. 2018)	0.565 ‡	0.388 ‡	n/a	0.28 (BT)	n/a	1.2	0.0161	-	-	CMI-7000S	2018
	Russia (Volfkovich et al. 2018)	0.373	0.146	0.007	n/a	n/a	1.4	-	0.06	-	Mosaic (film and fibrous)	2018
	Taiwan (Lee et al. 2019)	0.28	0.05	0.014	CT	-	1.2	-	0.12	-	AEM/CEM (Beijing)	2019
	The Netherland	1.10–	0.5	0.043	0.028	50	~1.4	-	0.17–3.45	-	Neosepta	2013

	(Zhao et al. 2013)	4.65			(BT)						AMX/CMX	
	Australia (Tan et al. 2020)	1.9	1.1–1.3	3.9–4.3	CT	46–87	0.4–0.8	-	0.28–0.37	-	Voltea	2020
FCDI	Korea (Chung et al. 2020)	0.55	0.15	0.007	CT	-	1.0	0.039	0.09	-	AEM/CEM	2020
	Australia (Zhang et al. 2020)	2.0	0.5	0.0012	CT	92	-	-	0.50–0.56	-	Fujifilm AEM/CEM	2020
	Australia (Ma et al. 2020)	1.00	0.15	0.0053	1 (SCT)	84.3	-	-	0.14–0.32	-	Fujifilm AEM/CEM	2020
AFCDI	Shanghai, China (Xu et al. 2017)	5.84	1.29	0.072	2 (BT)	>85	1.8	0.161	-	-	Neosepta AMX/CMX	2017
FTE-CDI	Livermore, USA (Suss et al. 2012)	5.84	2.34	0.0007	0.5 (BT)	n/a	1.25	0.016	-	-	none	2012
RCDI	Korea (Lee et al. 2018a)	2.92	n/a	0.003	CT	n/a	± 1.2	0.02−0.05	0.33 (23.9 kJ/mol)	-	AEM (AMV, Japan)	2018

[a] C_i: salinity of feedwater (g/L); C_o: salinity of feedwater (g/L); R_d: deionization rate (mg/g/s); SEC: specific energy consumption (kWh/m^3). Water costs only include operation costs (e.g., pumps and external power).

[b] FTE-CDI: flow-through electrode capacitive deionization; FCDI: flow-electrode capacitive deionization; AF-CDI: asymmetric flow-electrode capacitive deionization; RCDI: rocking-chair capacitive deionization.

[c] Salinity presented with † are converted from ppm (assumed that 1 ppm equals 1 mg/L), while ‡ are converted from conductivity (assumed that 1 mS/cm equals 0.5 g/L).

[d] BT: batch operation; CT: continuous operation; SCT: semi-continuous operation.

Table 7. Cost breakdown for a CDI/MCDI plant.

Categories	Critical factors	Costs [a]
Capital	Daily treatment capacity	CDI: 0.514−1.891 USD/m^3 (Drewes et al. 2009) MCDI [b]: 0.014 USD/m^3 (Huyskens et al. 2015)
Electricity	Pumps and external power	CDI: 0.005 USD/m^3 (Caudill 2018)
		MCDI: 0.377 USD/m^3 (Huyskens et al. 2015)
		PCDI: 0.020 USD/m^3 (Caudill 2018)
Cell replacement	Electrodes and/or membranes	CDI: 0.468 USD/m^3 (Caudill 2018)
		MCDI: 2 years lifetime (Huyskens et al. 2015)
		PCDI: 0.298 USD/m^3 (Caudill 2018)
Maintenance	Chemicals, etc.	CDI: 0.171−0.629 USD/m^3 (Drewes et al. 2009) MCDI: 5% of capital cost (Huyskens et al. 2015)

[a] Assumed 1.000 € equals 1.258 USD. [b] A typical total area of 10 m^2 per cell, with a total number of 40 cells with a treatment capacity of 1,500 m^3 per day. The plant life was assumed to be 10 years in this review, and we estimated the capital costs based on the information reported from the ref. (Huyskens et al. 2015).

Table 8. Comparison of plant costs for brackish water desalination using RO and CDI-based technologies.

Item	Units	RO (Metropolitan, 2006) (Yun et al. 2006)	RO (France, 2008) (Vince et al. 2008)	RO (USA, 2009) (Drewes et al. 2009)	CDI (EST Water, China, 2009) (EST 2009)	CDI (Texas, USA, 2009) (Drewes et al. 2009)	MCDI (Belgium, 2015) [e] (Huyskens et al. 2015)
Capacity of clean water	m^3/d	700,300	35,000	3,785	2,700	3,785	1,500
Water recovery	%	85	74−82	75	75	25−33	-
Feed concentration	g/L	0.750	3.0	5.52±0.72	-	0.52−6.40	3.4
Effluent concentration	g/L	0.015	0.3	0.15−0.33	-	0.378−5.90	0.1
Plant life	year	-	25	20	-	20	10
Capital (fixed)	USD/m^3	0.041−0.057	0.090−0.100	0.040−0.043	-	0.514−1.891	0.014 [f]
Energy	USD/m^3	0.029	0.029−0.037	0.057−0.076	0.055	0.055−1.221	0.377
Labor	USD/m^3	0.007	0.130−0.140	0.028	-	0.028	-
Chemicals (antiscalants)	USD/m^3	0.016	0.065−0.073	0.113−0.200	-	-	0.001
Modulus Replacement	USD/m^3	0.014−0.016 [a]	0.027−0.043 [a]	0.012−0.015 [a]	0.014 [b]	0.171−0.629 [d]	0.068 [f]
Miscellaneous	USD/m^3	0.010	-	-	-	-	-
Total	USD/m^3	0.116−0.134	0.223−0.240	0.256−0.354	0.070 [c]	1.933−2.602	-

[a] Modulus replacement is for the RO membrane replacement. [b] Membrane replacement is for the filter of feedwater as a pretreatment. [c] Assuming 1.000 CNY equals 0.150 USD. [d] Modulus replacement includes the electrode replacement. [e] The process was applied for biomass hydrolysates. [f] The plant life was assumed to be 10 years in this review, and we estimated the capital costs based on the information reported from the ref. (Huyskens et al. 2015).

Table 9. Specifications and costs of various pretreatment methods for RO and CDI desalination plants.

Item	Content	Unit	Media filtration [a]	Ultrafiltration (UF) [a]	Cartridge filter [b]
Specs	Process description	-	Sand filter followed by micron-filter	Semi-permeable membrane	Through a 5-µm, dead-end filtration mode
	Function	-	Removal of large particles at a high permeate flux	Contaminant removal with an acceptable permeate flux	Removal of particles and colloidal materials
	Water production	m^3/day	90,000	90,000	3,830–5,800
	Land footprint	m^2/Km^3/d	35–40	10–25	-
	Lifetime	years	20–30	5–10	-
	Energy consumption	kWh/m^3-feed	0.03–0.20	-	-
Costs	Capital	USD/m^3	0.22	0.23	-
	Energy	USD/m^3	0.16	0.16	0.008–0.010
	Chemicals	USD/m^3	0.05	0.03	-
	Maintenance	USD/m^3	-	-	0.007–0.013
	Miscellaneous	USD/m^3	0.07	0.09	-
	Total	USD/m^3	0.51	0.52	0.015–0.021

[a] An electric power price of 0.045 USD/kWh. The technical data for land footprint was gathered from ref. (Wilf 2004, Wolf and Siverns 2004); Lifetime was gathered from ref. (Wolf and Siverns 2004); energy consumption was gathered from ref. (Pearce 2008, Vince et al. 2008). The breakdown costs were gathered from ref. (Glueckstern and Priel 2003). [b] Assumed 1.000 € equals 1.258 USD. The electric power price of 0.002 USD/kWh. All information was gather from ref. (Farhat et al. 2020).

Table 10. Costs of concentrated brine disposal methods including operations and maintenance costs.

Option	Cost (USD/m^3)	Description
Surface water disposal	0.026−0.264 (Graves and Choffel 2004) [a]	• The least expensive option for coastal plants. • May change the salinity of receiving water. • Standard limit of salinity difference should < 10% (Mickley 2004).
Combined sewer disposal	0.30−0.66	• Considered as a relatively low-cost disposal method. • This option is often not available.
Evaporation pond	1.18−10.04	• Typically for small size plant (<400 m^3/d) (Mickley 2004). • Hot climate and land availability. • Regulations on soil and groundwater from salts and other chemicals. • Need for pond leakage monitoring.
Deep well injection	0.33−2.64	• Considered the most economical solution for inland plants. • Might eventually exhibit risks of leaching into the above aquifers.
Brine concentrator	< 26.41	• Capable of high salinity from 70,000 to 165,000 mg/L. • High capital cost and energy intensive (10−15 kWh/m^3) (Lee et al. 2018b). • Typically the most expensive option.
Crystallization	-	• Energy consumption of capillary crystallization at 4.0 kWh/m^3 (Abahusayn 2011).
Electrodialysis metathesis	0.64−11.21 (Giwa et al. 2017) [b]	• Energy consumption at 1.1−1.8 W per cell • Cost is dependent on the TDS of feed • Not affected by inorganic or organic membrane fouling
Pressure-retarded osmosis	-	• A power density of 16.7 W per m^2 of membrane. • The unit cost of USD10,085 per kW of installed capacity. • Net specific energy production of 0.25 kWh per m^3 of mixed brine solution. (Benjamin et al. 2020)

[a] The cost is estimated based on a distance offshore from 1.6 to 32.2 km. [b] The cost is estimated for electrodialysis metathesis followed by a crystallizer for per m^3 of recovered water.