A comprehensive review of saline effluent disposal and treatment: conventional practices, emerging technologies, and future potential

Parul Sahu

ABSTRACT

An ever-increasing volume of saline effluents from industries, oil–gas fields, and desalination plants has resulted in an enormous amount of pollutants with undesirable effects on the environment and human health. Adequate disposal and treatment of these effluents remains a persistent problem and poses significant technical as well as economic challenges. Saline effluents can have considerable environmental impacts, depending upon the sensitivity of the surrounding ecosystems. Conventional disposal techniques mostly suffer from direct or indirect contamination of water/soil and are no longer preferred. As a result, several advanced treatment methods are being considered for sustainable saline effluent management in recent times. In this context, a comprehensive and updated review of conventional methods, along with emerging technologies for disposal and treatment of saline effluent, is presented. Existing treatment approaches, including membrane operation, thermal processes, chemical techniques, and biological methods, are discussed. The application of innovative hybrid processes (combining two or more treatment methods) aiming at lower energy demand and higher treatment efficiency has also been evaluated. Subsequently, emerging sustainable strategies like waste minimization and water recovery, zero liquid discharge, and resource recovery for saline effluents have been examined. The prospect of integrating the renewable energy sources with energy-intensive saline treatment methods towards energy–water–environment nexus is also explored.

Key words | effluent treatment, desalination, disposal & treatment, resource recovery, saline effluent, zero liquid discharge

HIGHLIGHTS

- Saline effluent disposal and treatment methods are discussed.
- Hybrid technologies for sustainable treatment are presented.
- Emerging saline waste management strategies are evaluated.
- Water and resource recovery from waste effluent is assessed.

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INTRODUCTION

The large volume of saline effluents and concentrated brine from various sources like desalination plants, oil and gas drilling rigs, mining, industries, etc. have become a serious cause of pollution and threat to coastal, as well as surface water ecosystems (Iwama 1991). Saline wastewaters, such as concentrated brine, oil-field brine water, make-up saline solutions, etc., are high in salts, which can include scale formation and naturally occurring radioactive material (NORM)-contaminant salts (Bader 1994). With rising population, industrialization, and urbanization, these saline effluents are posing serious disposal problems with growing human health and environmental concerns. The disposed saline effluents harm the marine environment, i.e., generating anoxic condition on the seabed, changing light conditions, and influence on marine species and seagrass (Gacia et al. 2007). There are about 16,000 operational desalination plants, producing nearly 142 million m³/day of hypersaline concentrate (brine) worldwide (Jones et al. 2019). While such seawater desalination plants exist near the coast, the number of inland plants has been increased for domestic and industrial applications (Ahmed et al. 2000).

Growing amounts of brines from desalination plants and industrial effluents and their disposal have raised major ecological concerns globally. Common effluent disposal practices like the discharge of the effluent into the water bodies, disposal in evaporation ponds/bores, deep-well injection, etc. affect the surroundings adversely. Emergent challenges in terms of increasing concentrate volume, regulatory pressures, public awareness, and environmental issues have led to the need for adequate handling of saline wastes using cost-effective and innovative techniques. Consequently, the interest has grown towards sustainable saline effluent treatment strategies in recent times (Ahmed et al. 2002). Over the years, several physico-chemical, biological, ecological, and mechanical methods have been researched for the treatment of saline and industrial wastewater. Innovative hybrid approaches, which combined two or more different methods, are gaining interest due to higher treatment efficiency and improved performance over the stand-alone method. Other viable effluent-management alternatives like waste volume minimization (Subramani & Jacangelo 2014), zero liquid discharge (ZLD) (Xevgenos et al. 2015; Xevgenos et al. 2016), and recovery of salt/minerals (Abdul-Wahab & Al-Weshahi 2009; Kim 2011) are also being explored lately.

Many reviews focusing on different aspects of saline water treatment have been reported in the literature. Some of these publications presented the state-of-the-art technologies for treating waste brine generated from desalination plants (Afrasiabi & Shahbazali 2011; Pérez-González et al. 2012; Randall & Nathoo 2015). A large segment of review work centered around modern separation methods, i.e., forward osmosis (FO), membrane distillation (MD), membrane crystallization (MCr), electrodialysis (ED), electrodialysis reversal (EDR), capacitive deionization (CDI), eutectic freeze crystallization (EFC), and advanced oxidation processes (AOPs). These emerging technologies offer the advantage of high water recoveries (90–98%) from concentrated brines. Removal of organic pollutants from saline effluents has also been discussed in the literature (Lefebvre & Moletta 2006; Xiang et al. 2019). Anaerobic/aerobic biological treatment and AOP were found feasible.
for removing carbonaceous, nitrogenous, and phosphorous pollution at high salt concentrations.

Segregated reviews focusing on the treatment of concentrated brines (primarily from the desalination plant) with less emphasis on the spectrum of saline effluents from different sources have been reported. This article aims to supplement the existing state of the art for the management of saline effluents from various sources by presenting an updated review of the developments, opportunities, and challenges associated with multiple disposal and treatment techniques. Emerging effluent management strategies like waste minimization and the transformation of saline waste streams into valuable products have been examined, and a few illustrations are presented in detail. The prospect of utilizing renewable energy sources for saline effluent treatment technologies is also discussed.

CHARACTERISTICS OF SALINE EFFLUENTS

Primarily, saline effluent is a wastewater stream containing a combination of dissolved salts, hardness ions, organic content, nutrients, or other metals. Such highly saline streams are produced mainly from industries (agro-food, oil and gas, tannery, chlor-alkali, pulp and paper, etc.), acid mine drainage, and desalination plants. For instance, in the food industry, saline waste streams are generated due to the usage of brine solutions and NaCl salt for obtaining the final product. The brine discharges from desalination plants contain highly concentrated salts (>5 wt%), pre-treatment chemicals, and heavy metals resulting from corrosion of tubes, pipes, and equipment (Abdul-Wahab & Al-Weshahi 2009). Industries, such as mining, steel and electroplating, discharge a high level of heavy metals (Ni, Cd, Pb, Zn, Cu) in addition to salt ions (Cl⁻, SO₄²⁻, CO₃²⁻, NO₃⁻) (Moosavirad et al. 2015).

Characteristics, chemical compositions, and volume of any effluent stream (concentrate, retentate, or brine) vary, depending on the feed water quality, produced water quality, and the method/process by which it is generated (Pérez-González et al. 2012). Based on a literature study, Jones et al. have reported 41 types of feed water for different desalination technologies. This number is expected to increase further, considering saline effluents from other industries.

An assessment of constituents of the feed effluent streams (salt, metal, organic, and inorganic contents) is

<table>
<thead>
<tr>
<th>Source</th>
<th>Type</th>
<th>Characteristic</th>
</tr>
</thead>
<tbody>
<tr>
<td>Inorganic industries</td>
<td>Industrial effluent having heavy metals</td>
<td>TSS: 1,605–14,500 mg/L (Moosavirad et al. 2015)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Heavy metals: 0.5–2.5 mg/L (Moosavirad et al. 2015)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Salt ions: 1,400–11,000 mg/L (Moosavirad et al. 2015)</td>
</tr>
<tr>
<td>Oil and gas operations</td>
<td>Produced water</td>
<td>TDS: 3,000–10,000 mg/L (Lawrence et al. 1999)</td>
</tr>
<tr>
<td>Discharge from SO₂ scrubber</td>
<td>Industrial effluent</td>
<td>TDS: 80,000–100,000 mg/L (Tun &amp; Groth 2011)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>TSS: &lt;50 mg/L (Tun &amp; Groth 2011)</td>
</tr>
<tr>
<td>RO-based desalination plants</td>
<td>Reject brine water</td>
<td>TDS: 10,114–57,935 mg/L (Ahmed et al. 2000)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>TDS: 8,747–48,618 mg/L (Ahmed et al. 2001)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>TDS: 79,660 mg/L (Melián-Martel et al. 2013)</td>
</tr>
<tr>
<td>Multi-stage flash (MSF) based</td>
<td>Brine</td>
<td>TDS: 70,000 mg/L (Al Mutaz &amp; Wagialia 1990)</td>
</tr>
<tr>
<td>desalination plants</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Other desalination plants</td>
<td>Discharge brine</td>
<td>TDS: 50,000–80,000 mg/L (Hashim &amp; Hajjaj 2005)</td>
</tr>
<tr>
<td>Wastewater treatment plant</td>
<td>Municipal wastewater</td>
<td>TSS: 1,367–1,939 mg/L (Cartagena et al. 2013)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>COD: 1,868–2,540 mg/L (Cartagena et al. 2013)</td>
</tr>
<tr>
<td>Industry</td>
<td>Saline water</td>
<td>TDS: 48,000–80,000 mg/L (Tuin et al. 2006)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>COD: 53,000–89,000 mg/L (Tuin et al. 2006)</td>
</tr>
<tr>
<td>Petrochemical industry</td>
<td>Spent brine from ion-exchange regenerant</td>
<td>TDS: 30,000 mg/L (Osman et al. 2019)</td>
</tr>
<tr>
<td></td>
<td>and RO</td>
<td>TOC: 60 mg/L (Osman et al. 2019)</td>
</tr>
</tbody>
</table>

TDS, total dissolved solids; TSS, total suspended solids; COD, chemical oxygen demand; TOC, total organic carbon.
critical for the selection of suitable treatment methods. Table 1 shows the major components detected in effluent streams from different sources.

While the significant constituents of saline effluents are inorganic salts, a considerable amount of chemicals along with other reaction products used in pre-treatment, chlorination, and de-chlorination can also be found in untreated saline effluent (Alberti et al. 2009). The costs for effluent disposal and treatment systems are mainly driven by volume, salinity, and constituents of the concentrate. Prior to the selection of a treatment method for the given effluent stream, a qualitative, as well as a quantitative analysis of different components present in such flows, is needed. Depending on the nature of contaminations, i.e., organic, inorganic, radioactive, etc., suitable pre- or post-treatment methods can be employed to fit the end use.

**EFFLUENT DISPOSAL METHODS**

The most widely used methods for disposal of effluents from inland plants include deep-well injection, pumping into evaporation ponds, discharge into surface water bodies or municipal sewers, concentration into solid salts, irrigation of plants tolerant to high salinity (halophytes) (Ahmed 2004; Afrasiabi & Shabhazali 2011). The nature and constituents of effluents influence the selection of an effluent disposal method and the associated cost involved with the method. Typically the cost of effluent disposal ranges from 5 to 35% of the total cost of the plant from which it is generated (Younos 2005). Some of the conventional effluent discharge methods are discussed briefly in the following sections.

**Deep-well injection**

This method is commonly used for the disposal of the produced water from oil and gas operations along with toxic and hazardous wastes (e.g., NORM contaminant salts) (Reeder 1981). The contaminated waste stream is injected into specially designated horizons (wells) geologically isolated from potential underground sources of drinking water. Dependence on these injection wells for long-range and environmentally sound disposal of brine is becoming complex and costly, given environmental protection and water quality considerations (Brandt & Tait 1997). Weller (1993) developed an underground injection cost model for salt-water disposal (SWD). This model provides the capital and operating costs and optimizes those costs based on hydraulic fracture design under specified reservoir conditions. The model utilizes primary inputs and predicts surface pressures, injection rates, and disposal costs for deep-well injection. Cost predictions using this model can be compared prima facie with the associated cost of other alternatives like reverse osmosis (RO), thermal evaporation, etc. For high volume salt streams, these methods become uneconomical and inappropriate. With promising auxiliary technologies like pre-treatment, pre-concentration, effective partial demineralization, and improved treatment systems (i.e., de-oiling, dissolved organic removal), injection technologies can be made more feasible.

**Evaporation ponds**

Evaporation ponds are artificially constructed with a large surface area to allow water evaporation using solar radiation (Velmurugan & Srithar 2008). Solar energy-based evaporation ponds are a potential alternative for effluent disposal, especially in arid and semi-arid areas with high evaporation rates and low land costs. Evaporation ponds offer several advantages like easy construction and operation, low maintenance, low cost, etc. During the design of solar ponds, the lining of the pond using suitable impervious material (clay, synthetic polymer) is essential to prevent leakage. Proper sealing of adjacent sections and joints avoids the adverse effects on soil salinity and potential underlying water contamination through seepage from ponds (Shammas et al. 2010). Application of evaporation ponds/basins has also been demonstrated for concentrating the rejected brine in desalination plants to reduce its volume and to produce salts as part of the treatment process (Abdulsalam et al. 2017). Complying with the accepted engineering standards while designing these ponds is vital to minimize the environmental risks associated with the saline disposal basins. Gilron et al. (2003) have developed a wind-aided intensified evaporation (WAIV) to address the issue of large area requirements, generally associated with evaporation ponds. Vertically mounted
and continuously wetted evaporation surfaces with high packing densities were designed to reduce the land area required for brine disposal drastically. A large-scale WAIV test unit constructed in this study allowed the evaporative capacity per area footprint by a factor of >10 to be increased. Evaporation ponds (when properly designed and managed) can be a viable means of saline effluent disposal, especially for inland areas.

**Discharge into surface water/municipal sewer**

Discharge of effluent to a surface water body (river, lake, ocean, etc.) is found to be the most diffuse management practice for disposal of saline water due to the lowest associated cost in comparison to other conventional methods for disposal (Afrasiabi & Shahbazali 2011). However, such discharges into surface water bodies are suitable, provided that the volume of the reject water is not large relative to the volume of surface water. The compatibility of the concentrate to be released with the receiving water needs to be established prior to discharge. Effluents having salinity above the permissible limits of surface water discharge can be diluted using suitable low salinity water from wastewater treatment plants, cooling towers, etc. Enhancement of dissolved oxygen (DO) levels of saline water, before discharge, can avoid negative impacts on receiving water bodies (Mickley 2001).

Natural dispersion of brine through long sea outfalls is being practiced as an economical disposal strategy for coastal desalination plants. Nevertheless, selecting the appropriate configuration and location of the water outfall system through hydrodynamic studies of the brine effluents is important to avoid harmful effects on marine flora and fauna (Nikiforakis & Stamou 2015). Shao et al. (2008) have developed a mathematical model on the cumulative impact of brine discharge into shallow coastal waters with a flat seabed. In their study, the placement of the outfall further offshore has been found to reduce the shoreline salinity impact in a coastal region. Fernández-Torquemada et al. (2009) investigated the area of influence of the hypersaline plume formed by brine discharge in the shoreline or near beaches. Monitoring of the effluent dispersion from three seawater reverse osmosis (SWRO) desalination plants has been carried out with field campaigns and data acquisition. The expanse of the plume behavior was found to depend on discharge concentration, level of production, and season. It was observed that the area of influence of the discharge could be reduced by increasing the dilution using seawater as diluent. Usage of a diffuser to increase the velocity of discharge and to mix in the near field was also suggested to maximize brine dilution and consequently reduce the area of influence. González et al. (2011) have investigated new discharge systems to ensure a lesser environmental impact. The study dealt with mixing brine with a thermal effluent in a pipeline to obtain better dilution to minimize the effect on the area. The effects of salinity increase on the growth and survival of the seagrass were also considered. The outcomes from these studies, involving adequate discharge system design and pipe-design for homogeneous mixing, were found to be promising with fewer environmental impacts.

Another disposal method adopted by some small plants involves rejecting the effluent in municipal sewerage systems. However, such discharges are feasible only if the saline effluents are non-toxic and do not adversely affect the clarifier settling operation. Saline effluent discharge in the sewer has the advantage of lowering the biochemical oxygen demand of sewage (Ahmed et al. 2001). However, combining high salt concentration brines with sewage effluents can cause sewage contaminants and other particulates to aggregate in particles of different sizes. This, in turn, affects the sedimentation rate and could interfere with the transference of light in the water body, which would diminish the productivity of phytoplankton (Hashim & Hajjaj 2005).

A comparison of the associated cost for disposal methods discussed above is presented in Figure 1.

Although the budget estimations are with reference to the desalinated brine (Panagopoulos et al. 2019), the relative cost impact can be extrapolated to a saline wastewater treatment. It is evident that the direct discharge of saline effluents into surface water or sewers has a lower cost impact over deep-well injection and solar evaporation; however, such practices cause adverse effects on the aquatic environment. Discharge of salt beyond a limit can affect the performance of the biological treatment of the sewage system. Therefore, before direct disposal of saline effluents into surface water, suitable pre-treatment should be carried...
out, and the self-purification capacity of the receiving water body must be considered and not extended. Such discharges, however, should be strictly avoided in environmentally sensitive areas.

**EFFLUENT TREATMENT METHODS**

The adequate treatment of saline waste to minimize its volume is a sustainable alternative for handling such effluents. Some commonly used methods/techniques adopted over the years for saline and industrial wastewater treatment are shown in Table 2. These remediation methods result in treated water that can be reused for potable or non-potable purposes. A brief description of each method is presented in view of their involvement in sustainable effluent management processes (sections Effluent treatment methods and Emerging sustainable effluent management techniques).

### Membrane-based methods

Membrane operations are gaining attention due to their effective separation capabilities in the area of wastewater treatment for water reuse applications. A typical membrane-based separation involves the passage of the effluent stream through a semi-permeable membrane such that the molecules are separated into retentate and permeate based on their size and charge of differential interactions (Rathore & Shirke 2011).

The selection of membrane having appropriate characteristics, i.e., pore size, molecular weight cut off, is very important for any membrane-based separation. Membranes and membrane housings/modules constitute the consumable part of the filtration operation. Conventional membrane modules are in the form of flat sheet, hollow fibers, and spiral wound (Baker 2004). Advancements in novel filter materials and tailored pore structures are being deliberated for improving filter performance to achieve higher efficiency. Commonly used membrane materials are polyvinylidene fluoride, nylon, polyether-sulfone, polysulfone,

![Figure 1 | Comparison of cost for disposal methods.](image)

**Table 2 | Classification of saline effluent treatment methods**

<table>
<thead>
<tr>
<th>Membrane-based methods</th>
<th>Thermal operations/methods</th>
<th>Chemical conversion processes</th>
<th>Biological methods</th>
</tr>
</thead>
<tbody>
<tr>
<td>Filtration (micro-, ultra-, nano-)</td>
<td>Multi-effect distillation (MED)</td>
<td>Ion-exchange</td>
<td>Aerobic treatment</td>
</tr>
<tr>
<td>Reverse osmosis (RO)</td>
<td>Multi-stage flash (MSF)</td>
<td>Precipitation of insoluble compounds</td>
<td>Anaerobic treatment</td>
</tr>
<tr>
<td>Electrodialysis (ED)</td>
<td>Mechanical vapor compression (MVC)</td>
<td>Bio-oxidation</td>
<td>Nitrification and denitrification</td>
</tr>
<tr>
<td>Membrane bio-reactor/filtration</td>
<td>Spray evaporation/drying (SE/D)</td>
<td>Advanced oxidation processes (AOPs)</td>
<td>Biological reduction</td>
</tr>
<tr>
<td>Membrane distillation (MD)</td>
<td>Freeze crystallization (FC)</td>
<td>Coagulation/electrocoagulation</td>
<td></td>
</tr>
<tr>
<td>Forward osmosis (FO)</td>
<td>Eutectic freeze crystallization (EFC)</td>
<td>Chemical softening</td>
<td></td>
</tr>
<tr>
<td>Membrane distillation (MD)</td>
<td>Forward osmosis (FO)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Membrane distillation (MD)</td>
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</tbody>
</table>
polyamide high/low-pressure film, and regenerated cellulose. Ceramic membranes in hollow fiber and flexible sheet modules are also being used widely (Maddah et al. 2018). Some of the important membrane processes for saline wastewater treatment are discussed in detail.

**Pressure-driven membrane process**

The pressure-driven membrane processes (micro-, ultra-, and nano-filtration and reverse osmosis) has been conventionally exploited for wastewater treatment, water conditioning, and production of drinking water worldwide (Apel et al. 2019). Table 3 shows the general characteristics of different membrane processes in effluent treatment applications (Gude 2017).

Pressure-driven membranes allow filtration across their porous structure by the application of a positive hydrostatic pressure to force water to leave high concentration and flowing into the desalted zone. While RO membranes are capable of rejecting small molecules like monovalent ions (Na⁺, Cl⁻), other membranes, including NF, UF, and MF, can remove molecules of increased sizes (organic matter). Although NF, UF, and MF are not suitable for treating high salinity water, they are becoming attractive pre-treatment techniques for RO as they result in reduced turbidity, bacteria removal, and production of quality water at low-pressure levels (Teuler et al. 1999; Brehant et al. 2005). UF- and MF-treated water can be used for low-grade industrial uses. NF can efficiently eliminate the organic substances (e.g., micro-pollutants & organism, multivalent ions) present in saline effluents. Further, the application of MF, UF, and NF as pre-treatment operation for RO can result in higher recovery and higher flux, with a significant reduction in overall treatment cost (M’nif et al. 2007).

RO is one of the most promising desalination technologies and is being exploited at a commercial scale. Using the RO process for brine treatment, high-quality permeate (ultra-pure and boiler-feed quality water) with recovery up to 70% can be achieved. However, for high salinity feed, the energy cost increases sharply due to higher pressures and lower recovery. Also, the operation is restricted to an applied pressure of >100 bar due to limits on pressure vessels employed for RO membrane modules. The high pumping energy requirement associated with the RO process can be compensated using energy recovery devices (energy recovery turbines, pressure exchangers, turbo-charger) (Esmaeilion 2020). Frequent membrane replacement due to scaling and fouling is another limitation of RO processes. The addition of anti-scaling materials to feed saline water is being considered to increase the water recovery and removal efficiency of inorganic matter (Pramanik et al. 2017).

Low-pressure reverse osmosis (LPRO) systems operating at remarkably low pressures (<10 bar) with elevated product water flux and salt rejections are being investigated for saline water treatment. Successful demonstration of treating agricultural drainage water (total dissolved solids (TDS): 15,000 ppm, rich in gypsum), desalinating red-sea water, and high potential fouling feed water using LPRO are reported (Lee et al. 2003; Yangali-Quintanilla et al. 2011; Park & Kwon 2018). Provided suitable membrane selection (based on the feed water quality), the LPRO system enables energy savings and enhanced permeate flux.

**Forward osmosis**

FO is an osmotically driven membrane process caused by the different solute concentrations across the membrane. It offers advantages of lower hydraulic pressure and reduced energy demand. The FO process (shown in Figure 2) is the opposite of RO in practice. When solutions of varying solute concentrations are separated by a semi-permeable

<table>
<thead>
<tr>
<th>Membrane process</th>
<th>Particle capture size</th>
<th>Typical contaminants removed</th>
<th>Typical operation pressure range</th>
</tr>
</thead>
<tbody>
<tr>
<td>Microfiltration (MF)</td>
<td>0.1–10 μm</td>
<td>Suspended solids, bacteria, protozoa</td>
<td>0.1–2 bar</td>
</tr>
<tr>
<td>Ultra-filtration (UF)</td>
<td>0.01–0.1 μm</td>
<td>Colloids, proteins, polysaccharides, most bacteria, some viruses</td>
<td>1–5 bar</td>
</tr>
<tr>
<td>Nano-filtration (NF)</td>
<td>0.001 μm</td>
<td>Viruses, natural organic matter, multivalent ions</td>
<td>5–20 bar</td>
</tr>
<tr>
<td>Reverse osmosis (RO)</td>
<td>0.001 μm</td>
<td>Almost all impurities, including monovalent ions</td>
<td>10–100 bar</td>
</tr>
</tbody>
</table>
membrane, the solvent (i.e., water) moves across the membrane from the lower solute concentration side to higher solute concentration (i.e., ‘draw solution’). A recovery process is then employed to separate pure water from a diluted draw solution. In comparison to RO desalination, the FO process extends the advantages of high-quality product water, reduced membrane fouling, and low energy input (Jiao et al. 2015).

The membrane modules and devices used for FO include plate and frame, spiral-wound, tubular, and hydration bags (Cath et al. 2006). A report by the WaterUse Foundation explored the feasibility of applying FO using commercially available membranes to dewater RO concentrate and also investigated innovative draw solutions. Sodium chloride and dendrimers were found to be promising draw solutions (Chekli et al. 2012). Based on a life cycle cost analysis conducted by Valladares Linares et al. (2016), cost comparison for the FO, RO, and LPRO processes is presented in Table 4.

While FO offers lower expenditure than RO and LPRO, the issues of concentration polarization, membrane fouling, and reverse solute diffusion need further consideration. The commercial success of this process for saline effluent treatment will be subjected to the development of high-performance membranes and easily separable draw solutions (Zhao et al. 2012). Also, one of the major technological barriers of FO technology is the separation, re-concentration, and recovery of the product water from the draw solution.

**Electrodialysis and electrodialysis reversal**

Electrodialysis (ED), an efficient method to treat saline concentrate, employs electrical potential-driven ion-exchange membranes to attract dissolved ions, without the movement of water molecules across the membrane. An ED stack consists of a series of alternate cation exchange membranes and anion exchange membranes between a cathode and an anode. An applied electrical potential drives cations and anions in opposite directions resulting in diluate (low salinity) and concentrate (high salinity) (Al-Amshawee et al. 2020). ED has demonstrated its potential in treating industrial effluents and the removal of heavy metal from wastewater (Gmar et al. 2019). In a pilot-scale study, Korn-gold et al. (2009) investigated the treatment of brine discharged from an RO plant using ED with a separated gypsum precipitator to reduce the scaling problem in ED. In another study, Zhang et al. (2012) performed a techno-economic analysis and environmental impact evaluation of the RO concentrate treatment process using ED. Both a lab-scale and a pilot-scale ED system were investigated for this purpose, and among the various parameters studied, applied current density and feed flow rate were found to influence the salt removal rate. At high feed salinities, the productivity and efficiency of the ED process decrease due to a reduction in the chemical potential difference of salt in the diluate and concentrate. The application of a smaller ratio of the electrical current to its limiting value was recommended to make the ED process potentially feasible for the desalination of high salinity water (McGovern et al. 2014).

Electrodialysis reversal (EDR) is a more advanced ED process with the same electrochemical principle as ED.

<table>
<thead>
<tr>
<th>Table 4</th>
<th>Cost in USD for 100,000 m³/day capacity plant</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cost component</td>
<td>FO</td>
</tr>
<tr>
<td>Equipment + material + membrane</td>
<td>$40,866,655.8</td>
</tr>
<tr>
<td>Construction</td>
<td>$37,874,492.2</td>
</tr>
<tr>
<td>Total cost</td>
<td>$78,741,148</td>
</tr>
</tbody>
</table>
EDR uses electrode polarity in a reverse manner by reversal of DC voltage to reverse ion transport, which in turn self-cleans the membrane surface and minimizes scaling/fouling. A few EDR systems for RO brine treatment and volume minimization to recover water have been reported. These systems have demonstrated high water recovery (85%) with significant volume reduction (~ 6.5 times). Saltworks Technologies Inc. has designed EDR systems capable of handling saltwater feed up to 8 wt% (Panagopoulos et al. 2019). These promising results encourage the development of EDR technology for large-scale implementation.

Membrane distillation

MD is an evaporative process based on vapor–liquid equilibrium, where only volatile components are transported through the membrane resulting in separation. This process is driven by the difference between the partial pressure of a solution contacting one side of a porous hydrophobic membrane and its partial pressure on the other side of the membrane (Susanto 2011). In comparison with other separation processes, MD offers advantages of higher solute rejection, lower hydrostatic pressure, and fewer pretreatment requirements. Amongst several MD configurations investigated, direct contact membrane distillation (DCMD) is the most widely used arrangement for desalination. Song et al. (2007) have studied large-scale DCMD modules employing porous hydrophobic polypropylene hollow fiber membranes for desalination. The effect of variation in distillate temperature, brine salt concentration, and temperature on vapor flux was investigated. Water vapor fluxes were found to increase with brine temperatures. An increase in salt concentration resulted in a reduction of flux. Abdelkader et al. (2019) have employed DCMD for the treatment of saline dairy effluent. Macro-filtration and ultra-filtration methods were used as a pretreatment to reduce membrane fouling in MD. Notably, the pre-treatment of effluent before DCMD led to improved product quality with stable flux and better thermal efficiency. Osman et al. (2019) have conducted a computational modeling and experimental study for the desalination of petrochemical industrial effluents using DCMD. Model predictions were able to obtain the water flux successfully and can be used as potential process design tools for a similar type of industrial effluent. Vacuum membrane distillation is another MD variant wherein a vacuum is applied at the permeate side to lower the vapor pressure resulting in reduced heat loss and increased permeate flux. This configuration can be useful for removing volatiles from an aqueous solution (Camacho et al. 2015). Although at the developing stage, MD, in conjunction with low-cost pretreatment, could be an exciting alternative to treat saline effluents. The possibility of using waste heat in the MD treatment needs to be further explored.

In comparison with the RO process, FO and MD methods have the advantage of treating effluents with high TDS levels (up to 100,000 mg/L). Also, these two processes can utilize waste heat sources to heat the feed water to MD or regenerate draw solution in FO (Subramani & Jacangelo 2014). Recent advances of four key membrane processes (e.g., RO, FO, ED, and MD) in membrane materials and process designs have been reviewed by Duong et al. (2019). The focus was on improving fouling resistance, water flux, and energy efficiency. Aquaporin membranes and polyamide membranes embedded with nanoparticles have shown excellent performance for RO and FO processes. In the case of MD, super-hydrophobic membranes have exhibited great potential. The latest developments in RO and ED process design have been towards control of membrane fouling by exploring novel energy-saving membrane-based pre-treatment and reversal operations, respectively. A comparative chart of specific energy consumption (SEC) associated with the various membrane processes discussed is presented in Figure 5.

While the majority of membrane processes (i.e., low- and high-pressure RO, FO, ED, and EDR) have low SEC, the MD process requires high energy for vaporization. Further technological advancement and integration of waste heat or solar thermal sources with MD can reduce the energy demand and make the process more viable in the future.

Thermal operations

Generally, thermal technologies utilized for desalination are based on evaporation (mechanically or naturally) and crystallization. Separation of one component from others, present in the liquid phase, is carried out by creating a new phase, i.e., vapor or solid using thermal energy. Some
of the conventional thermal-based separation methods are discussed below.

Multi-stage flash and multi-effect distillation

Multi-stage flash (MSF) and multi-effect distillation (MED) have been used over the years for desalting processes due to their simple layout and reliable performance. MSF involves flash evaporation of saline water, which is repeated in consecutive stages. Purified water vapor is condensed and cooled using feed water for energy recovery. MED is based on the process of film boiling for vapor generation in successive stages operating at decreasing levels of pressure. The steam generated in one stage or effect is used to heat the concentrate in the subsequent stage operating at a lower temperature or pressure. The performance of both these operations depends on the number of stages or effects. While both these processes are energy-intensive, MED involves low capital cost with higher efficiency in comparison to MSF. The relatively large thermal energy consumption associated with MSF and MED can be addressed by utilizing waste heat and cogeneration (Borsani & Rebagliati 2005; Ophir & Lokiec 2003). New advancements in low-temperature evaporation systems (with thermal storage) driven either by low-temperature waste energy or solar power need to be leveraged (Stengler et al. 2018).

Mechanical vapor compression

Mechanical vapor compression (MVC), an alternative method for the treatment of highly concentrated wastewater, requires mechanical power as energy input to drive the compressor to produce condensable water vapor and wet salt. No steam is required except for preliminary heating to raise the plant to working temperature. A vertical tube falling film concentrator followed by a forced-circulation crystallizer is a commonly used arrangement of mechanical evaporation systems (Subramani & Jacangelo 2014). MVC consumes less energy than thermal evaporation and can be driven by renewable energy sources (i.e., solar, wind, etc.) (Marcovecchio et al. 2010).

Spray evaporation/drying

The spray evaporation or drying (SE/D) process is a thermal process that converts saline water into a dry powder of mixed solid salts. Spray dryers consist of a drying chamber and a centrifugal atomizer through which the concentrate slurry is sprayed. The dry solid is blown by hot air and collected into a solid storage chamber (Subramani & Jacangelo 2014). While spray dryers for concentrate management require high capital cost and energy, utilizing waste process heat can make this process feasible. An example of rapid spray evaporation using waste heat described in a patent for removing dissolved salts from saline water is shown in Figure 4 (Hartman et al. 2004).

Saltwater from a saline water source is preheated in the condenser and then atomized using a nozzle to form microdroplets of 1–100 μm. These saltwater droplets are sprayed into an air heated chamber to undergo rapid evaporation, resulting in the separation of salt solids from vaporized water. The mixture of suspended solids and water vapor is filtered to remove the salts, and the vapor is condensed by
exchanging heat from incoming saltwater, to recover the salt-free water. The application of spray/atomization with air-heating enhances the evaporation rate to remove salt particles and produce salt-free water from saltwater.

**Freeze crystallization and eutectic freeze crystallization**

Freeze crystallization (FC) processes have shown their potential for juice concentration applications. The FC process can be classified into two categories: direct freezing and indirect freezing (Randall & Nathoo 2015). While the indirect process involved separating saline water from the refrigerant by a solid metallic heat transfer surface, the direct contact process is characterized by the absence of such a physical barrier. The FC process involves the cooling of saline water to a temperature where water freezes and salt remains in solution. The ice crystals formed are subsequently separated from residual concentrate followed by washing to recover products, which can be used in a variety of applications. The techno-feasibility of FC-based desalination using available low-cost refrigeration has been demonstrated in a recent work (Sahu 2019).

Eutectic freeze crystallization (EFC) is a low-temperature thermal technique, wherein the aqueous solution is separated into pure ice and salt crystals by freezing it down to eutectic temperature. By virtue of the density difference between ice and salt crystals, they can be readily separated by the force of gravity (van der Ham et al. 1998). In comparison to evaporative thermal methods, EFC offers the possibility of a complete conversion of feed into water (from ice) and salt crystals (Kim 2011). van der Ham et al. (1999, 2004) have investigated a cool disk column crystallizer to recover copper sulfate and mono-ammonium-phosphate crystals from their respective aqueous solutions. Using EFC, Himawan et al. (2006) have tested a batch glass crystallizer at lab-scale to recover magnesium sulfate heptahydrate from industrial solution. Recovery of salable quality products from saline effluents using EFC processes can make them an attractive option.

The cost for thermal systems, from a capital and operating (energy) perspective, is typically higher than the cost for membrane-based systems (Mabrouk et al. 2007). A comparison of SEC associated with various thermal processes is presented in Figure 5 (Panagopoulos et al. 2019). Thermal technologies tend to be beneficial where energy costs are low, such as in the Middle East and Gulf countries. The integration of renewable energy sources like concentrated solar power and geothermal energy with these systems can overcome the large energy requirements, making the process energetically more attractive. Both the cooling-based processes, i.e. FC and EFC, have large energy demands due to...
associated refrigeration requirements. The practical application of these processes is subjected to the availability of lost-cost refrigeration and efficient crystallizer designs (Sahu et al. 2020).

Precipitation and reaction-based processes

Different chemical reaction-based processes have been utilized for the treatment of saline effluent and to recover selective salt/chemical/metal of interest. Ion-exchange-based reactions using suitable resins have been extensively used to recover salts (sodium, magnesium, calcium, chloride, sulfates) and heavy metals (Cu, Cd, Zn, Ni, Pb) from waste streams in recent years (Oztekin & Yazicigil 2007; Millar et al. 2015; Moosavirad et al. 2015). In this process, strong acid cation exchange resin in the hydrogen form converts dissolved salts into their corresponding acids, and strong base anion resin in the hydroxide form removes these acids (Brandt et al. 2017). The ion-exchange method can selectively reduce the toxic heavy elements in wastewater subject to the choice of suitable resin. After separating the loaded resin, the metal in a more concentrated form is recovered by elution using regenerative reagents (Moosavirad et al. 2015). Application of magnetically enhanced anion exchange resins for the removal of dissolved organic matter and synthetic organic chemicals from wastewater (including greywater and landfill leachate) was reviewed by Boyer (2015). The magnetic ion-exchange (MIEX) resins have also been reported to handle a wider range of dissolved organic carbon (DOC) better than coagulation and activated carbon adsorption. The prospect of integrating MIEX resin with physical–chemical processes like coagulation, membrane technology, activated carbon, ozone disinfection, lime softening, and combined ion-exchange processes was also explored. Such integrations have shown multiple benefits including very high DOC removal, reduction in coagulant dose, and membrane fouling during wastewater treatment.

The application of organic solvent to precipitate salts from saline water and solutions was patented and reported by Bader (1995). Their process involved adding a miscible organic solvent to saline water to precipitate the salts and subsequent separation of the salt from the solvent using stages of hydro-cyclones. Interestingly, the proposed process can potentially be used for precipitation and separation of salts, scale salts, and NORM contaminant salts from saline effluents as well as for the remediation of contaminated soils. Treatment of radioactively contaminated water using the coagulation–sorption method with acquired magnetic properties has also been reported (Timoshenko et al. 2009). In a recent study, Zhang et al. have investigated the performance of precipitation (using Fe$^{3+}$ and Al$^{3+}$) and Electrocoagulation (EC) for removal of silica (inorganic foulant in membrane process) as pretreatment of saline water. Al$^{3+}$ ions were found more effective than Fe$^{3+}$ ions in removing dissolved silica in water. In EC studies, 90% silica removal efficiency was reported (Zhang et al. 2019).

Other applications of reactive precipitation for saline effluent treatment involved the reactive absorption of CO$_2$ into sodium hydroxide solutions to precipitate calcium carbonate from nano-filtration retentate in an integrated membrane process (Drioli et al. 2004). In another lab-scale experimental campaign using reactive crystallization, the potential for magnesium recovery from concentrated brines was assessed. Semi-batch and continuous crystallizers operated by a reactive precipitation process using NaOH solutions were adopted. High purity (98–100%) magnesium precipitates were obtained from the brine by maintaining a stoichiometric injection of alkaline reactant (Cipollina et al. 2015).

Application of AOP like ozonation, Fenton process, photocatalysis/photooxidation, electrochemical oxidation, etc. for treating organic pollutants has been reported in a recent review (Pérez-González et al. 2012). These methods have been used to remove DOC, chemical oxygen demand (COD), and total ammoniacal nitrogen in saline waters before discharge. Deng & Zhao (2015) have examined the use of hydroxyl radical- and sulfate radical-based AOPs in wastewater treatment of selected landfill leachate and biologically treated municipal wastewater as model samples.

More recently, integrated electrochemical processes were adopted for treating textile industry wastewater. EC alone and combined with electro-Fenton (EF), anodic-oxidation, and peroxide-coagulation were investigated to treat dye-laden wastewater from textile industries using a batch reactor. Combined EC and EF methods were found to be most efficient with ~98% removal of COD, total organic carbon (TOC), and metal contents (Afanga et al. 2020). The Integrated EC–EF technique can be used for treating textile-like saline wastewater effluents.
Interestingly, precipitation and reaction-based processes as discussed above can be applied at an intermediate stage of treatment for selective extraction of a particular salt/ion/metal from saline wastewater to improve the overall removal efficiency.

**Biological methods**

Advanced biological methods can offer an alternative to treat saline and industrial effluents abundant with organic pollutants and nutrients (Semenova et al. 2013; El-Sheekh et al. 2014; Narayanan & Narayan 2019). Biological treatment of organic pollutants in wastewater stream generated from industries, i.e., food processing, leather, and petroleum, has been examined in a review. Anaerobic and aerobic biological treatment methods were compared with prevalent physico-chemical treatment methods and found to be more suitable for the treatment of organic matter (Lefebvre & Moletta 2006). The stand-alone activated sludge process has not been successful in treating saline wastewater due to the adverse effect of high concentration inorganic salts on the removal of carbonaceous oxidation, nitrification, denitrification, and phosphorous. Therefore, bio-augmented systems using salt-tolerant (halophilic) organisms and biofilms have been exploited to improve the performance of conventional activated sludge processes in saline effluent treatment (Wu et al. 2008). Several biotreatment configurations, namely fed-batch reactor, rotating bio-disc contactors, sequencing batch reactor, moving bed biofilm reactor, etc. have been researched over the years (Kargi & Dincer 1996; Kargi & Uygur 1997; Wu et al. 2008; Bassin et al. 2011, 2012). These systems perform well for salt concentrations <1 wt%, however, they become inefficient in the presence of high salt contents due to the alteration of the biomass properties and growth, resulting in instability and reduction in organic removal due to salt build-up in the sludge (Abdelkader et al. 2019).

Application of membrane bioreactors (MBR) for treating various industrial wastewaters has emerged recently. MBR, a combination of filtration with biological treatment, favors prolonged hydraulic retention time and increased biomass in reactors. MBR is an advanced technology that offers several advantages like high-quality effluent, high loading capacity, small space requirement, and easier operation over conventional activated sludge processes (Barreiros et al. 1998; Luo et al. 2019; Abdollahzadeh Sharghi et al. 2020). The design and operation of MBR plants for treating municipal and industrial wastewater have already been demonstrated at the pilot scale (Alnaizy et al. 2011; Deowan et al. 2019). The treated water can be reused for non-potable purposes (e.g. toilet flush, garden, agriculture). Another variant of the membrane and biological system, namely the osmotic membrane bioreactor (OMBR) has been researched recently for wastewater treatment and resource recovery. Integrating biological processes and FO, the biologically treated water is transported from the mixed liquor through a semi-permeable FO membrane by an osmotic pressure gradient, into a highly concentrated draw solution. As compared to the MBR system, OMBR offers advantages of better product water quality, lower fouling propensity and high fouling reversibility (Wang et al. 2016).

Recently, a novel application of electro-biochemical reactors for removing salts and organic pollutants from wastewater has been explored. Bio-electrochemical systems consist of anode and cathode separated by an ion-exchange membrane, capable of converting the chemical energy embedded in bio-degradable materials (wastewater, sludge, and sediments) into electrical energy (Jegathambal et al. 2019).

**Hybrid treatment methods**

The application of hybrid methods, which combine two or more methods, has been emerging rapidly in recent times to synergize the waste management with environmental protection. Some of the significant research advancements in this area are assessed in the following subsections.

**Integrated membrane processes**

A combination of pressure-driven membrane operations (i.e., MF, UF, NF) with RO is gaining interest for industrial and saline wastewater treatment. The application of MF and UF systems prior to the RO leads to an increased permeate flux, improved reliability, and better plant-economies (Wilf & Schierach 2001). Drioli et al. (2006) have attempted integrated MF–NF–RO membrane systems with MD/ MCr
units to improve water recovery and cost-effectiveness in the context of seawater desalination. Using permutation-combinations, four different configurations of integrated systems (as shown in Figure 6) have been proposed.

The first configuration (FS1) exploits MF and NF for pre-treatment to the RO unit. Such configurations are characterized by water recovery up to 50%, an energy saving of 25-30%, and a reduction in discharge volume. The second and third configurations employ MCr after NF and RO retentate, respectively, in the FS1 system. Before MCr unit, Ca^{2+} ions need to be precipitated and separated to avoid scale formation. In the fourth configuration, an additional MD unit was integrated on the RO retentate in addition to the FS2 configuration. With the addition of MCr and MD units, thermal energy requirements increase on top of electrical energy requirements for MF–NF–RO.

In another illustration, Nadjafi et al. (2018) demonstrated the feasibility of integrated MF–UF and UF–RO systems for the treatment of refinery wastewater. A pilot-scale hybrid membrane system has been utilized to reduce harmful and damaging components in refinery wastewater with the aim of reusing at boilers and cooling towers. In comparison to the MF–UF system, the UF–RO hybrid system has shown better removal efficiency (~98%) for salts and organic pollutants. The product water quality obtained after treatment was found to meet the minimum standards to be reused at boilers and cooling towers or to be discharged to the environment (Nadjafi et al. 2018).

Cartagena et al. (2013) have combined a membrane bioreactor (MBR) pilot plant with an NF/RO pilot plant to reduce emerging micro-pollutants (EMPs), organic matter, nutrients, and salinity of a municipal wastewater plant. Flat sheet and hollow fiber modules were tested, and the results demonstrated an efficient removal of suspended solids (100%) and COD (98%) with each module. In a recent study, MBR effluents with and without NF/RO treatment were evaluated for reuse in agricultural-irrigation considering the salinity, nutrients, ion toxicity, sodium/potassium adsorption ratio, etc. as water quality parameters (Jalilnejad Falizi et al. 2018). Besides, MBR effluents and RO permeated were mixed in different proportions and analyzed for irrigation standards. Among the tested samples, RO permeate-MBR effluent mixture in 2:1 ratio was found to be the most suitable for irrigation purposes, with the dual advantage of suppressing the harmful effect of MBR effluent salinity and amending the adverse impact of RO permeate infiltration.

**Combined RO and thermal operation**

The combinations of membrane-based methods and thermal operations as a hybrid system for desalination effluent management are being considered as an excellent economic alternative to a stand-alone membrane or thermal plant. A patent by Brandt & Tait (1997) presented a method of treating and subsequent disposal of saline wastewater (from oil and gas wells) using a combined RO and combustion heat evaporation unit (shown in Figure 7).

The first stage of this combined process employed an RO unit, which divides the feed streams into two streams. One of the streams (i.e., potable water) possesses a lower salt concentration than the incoming brine and can be used for domestic or industrial applications. The other stream (concentrated brine) having a higher salt concentration than the incoming brine stream is subjected to combustion evaporation for further concentration. This highly concentrated brine stream from the evaporator can take the form of a salt mixture or salt slurry.

Hamed (2005) studied the hybrid desalination processes with the power generation system in the context of the UAE and presented the state-of-the-art of simple and fully integrated hybrid systems. For a prescribed 100% power load condition, the SEC was reported to be 34 and 15 kWh/m³ of product water for dual-purpose-Power/MSF and Hybrid power/MSF/RO system, respectively. Further research endeavors were carried out by the R&D center of these plants to develop dihybrid NF/MSF and trihybrid NF/RO/MSF systems.

More recently, the concept of Membrane Crystallization (MCr), which combines MD with crystallization, has also been evolved to treat water and recover minerals from saline concentrates. While the MD process results in production water and concentration of the feed, crystals are formed from a supersaturated solution using the crystallization process. MCr has been applied to recover several minerals including sodium, magnesium, barium, strontium, and lithium from various process effluents and concentrates. Multiple factors including temperature, flow rate, and composition of the feed were found to affect the performance.
Figure 6 | Different integrate membrane system configurations. (a) FS 1: Integrated MF–NF–RO system, (b) FS 2: Integrated MF–NF–RO system with an MCr unit operating on the NF retentate, (c) FS 3: Integrated MF–NF–RO system with an MCr unit operating on the RO retentate, (d) FS 4: Integrated MF–NF–RO system with an MCr unit operating on the NF retentate and an MD unit on the RO retentate.
of this process. The fouling in MCr is expected to be less severe than other pressure-driven membrane processes. However, the variation in nutrient compositions could be a barrier for its widespread application and needs to be addressed (Pramanik et al. 2016).

Based on the above development, the membrane-thermal-based integrated systems are expected to result in high water productivity and enhanced thermal performance with lower energy consumption and construction costs, especially in the regions with abundant thermal energy sources.

**Combined chemical, ion-exchange and RO module**

Integration of chemical and ion-exchange processes with membranes have been attempted to circumvent the fouling and scaling issue associated with membrane processes by selective removal of ions. The ion-exchange membranes take advantage of their electrochemical properties to regenerate protons and hydroxide ions by applying a separate bed of cation- and anion-exchange resins (Grabowski et al. 2006). For coupling ion-exchange with membranes, regeneration of resin is essential, and the type of resin used determines the regeneration potential (Bornak 2014). A combination of high boron rejection membrane and ion-exchange resins were applied for selective removal of boron in brackish water source. An overall reduction of boron from 0.62 to 0.3 ppm was obtained (Glueckstern & Priel 2007). A comprehensive economic analysis was also presented using an optimization tool and considering the additional operating cost and investments associated with the RO and ion-exchange system. The stand-alone high boron rejection system was found to be more economical than the combined membrane and ion-exchange resin system (Gebreeyessus 2019). In another study, the advanced separation process for boron by combining boron sorption on a fine-powdered boron selective resin with microfiltration membranes was put forward by Bryjak et al. (2008). The use of sorbent in the form of fine particles resulted in higher boron-uptake and thus improved separation efficiency. This approach can be extended for potential adsorption of specified components present as trace amounts in an effluent, subjecting to specific selective resins. In a recent review (Gebreeyessus 2019), a coupling of the ion-exchange process with membrane filtration (RO/UF/ED) was explored, and the advances in composite membrane materials were discussed. Two variants of ion-exchange membrane coupled technology were presented (see Figure 8).

Depending on the purpose of specified rejections, ion-exchange can either be placed before (Process A) or after (Process B) the membrane unit. These hybrid systems result not only in a reduction of energy but also in membrane area requirements. Sustainability and feasibility of large-scale operations, including the cost of ion-exchange resin regeneration, need to be addressed in the future. While most of the investigations deal with desalination, the application of these hybrid systems can be extended to treat shale oil/gas wastewater and other solids with efficiency up to 70% (Kim et al. 2016).

As part of a mini review, Xiang et al. proposed an integrating process comprising FO, pre-coagulation, AOPs, and post-biological treatment for treating organic contaminants present in RO concentrates at lower cost and energy usage (Figure 9). FO has been used to reduce the volume of feed concentrate for subsequent treatment steps. Potassium chloride solution was used as a draw solution. Pre-coagulation was effective as a pre-treatment of APO to improve the biodegradability of concentrate from FO. A novel rotating advanced oxidation contactor equipped with the zeolite/TiO2 composite sheets was used for...
the removal of target contaminants and its transformation by-products. A post-biological treatment resulted in significant energy savings and efficient elimination of organic matter. The overall process was found to be an energy-efficient and cost-effective alternative to RO concentrate treatment (Xiang et al. 2019).

Biotreatment coupled with membrane and thermal processes

Saline effluents which contain organic compounds in addition to high concentrations of salt are difficult to treat using membrane- or thermal-based operations. An additional biological treatment stage becomes desirable for treating effluents having COD and organic carbon. Conventional aerobic biotreatment of saline water requires 10–20 times dilution of feed to enable biodegradation, which makes the process less attractive. Removal of these organics without dilution requires the use of halophilic organisms adapted for working at high salinities (Woolard & Irvine 1995). Tuin et al. (2006) worked on improving the treatability of saline wastewaters through the combination of aerobic treatment and pretreatment. Nanofiltration and crystallization were used as a pretreatment to remove and recover up to 80% of the salts as crystallized Glauber salt. Anaerobic pretreatment was carried out in a sludge reactor to remove
sulfate and organic load partially. Combined aerobic and anaerobic treatments were found to be feasible with two to three times dilution or more.

Based on a detailed understanding of various physico-chemical and biological techniques applied to treat saline wastewater, Lefebvre & Moletta (2006) proposed a generic sequence of the treatment chain (shown in Figure 10) for the removal of salt and organic matter from industrial saline wastewater. Pretreatments using pH adjustment, nutrients balancing along with coagulation–flocculation are essential for the removal of colloids followed by filtration (NF, RO) for removing salts. In the following stage, the biological treatment ensures the removal of nutrients (carbon, nitrogen, and phosphorus) present in pretreated wastewater. Finally, the post-treatment using RO and evaporation removes the remaining salts and produces treated effluent.

More advanced membrane-based biological treatment systems, i.e. MBR and OMBR (section Biological methods), have also been integrated into other processes including MF, UF, ED, and MCr to control salinity build-up in the bioreactors and reduce waste generation. However, these novel systems suffer from issues like membrane stability and fouling, low water production, contaminant accumulation in the draw solution, and high energy consumption. Future studies of these hybrid systems can be directed to robust and anti-fouling membranes, process optimization, and utilization of renewable energy (Li et al. 2018). A comparative table of noteworthy methods used for the treatment of saline and industrial effluents is presented (Table 5). Advantages, limitations, technology status, potential applications, and future prospects of each method are summarized.

**Capacitive deionization-based hybrid systems**

CDI was developed as a nonpolluting, energy-efficient, and cost-effective alternative desalination technology. This technology involves flowing a saline solution through an unrestricted capacitor module consisting of high surface-area electrode-pairs in series. Polarization of each electrode pair by a DC power source results in electro-adsorption of anions and cations present in saline solution. Subsequently, the saturated electrode undergoes regeneration by desorption of the ions under the reverse electric field (Subramani & Jacangelo 2015). A schematic of adsorption–desorption mechanism in CDI and a potential application for saline effluent treatment is shown in Figures 11 and 12, respectively.

Coupling of CDI with RO can complement the limitation of disadvantages of membrane fouling and high energy consumption associated with high-pressure membrane operations. Two types of hybrid systems, namely the RO-CDI pass system and RO-CDI stage system, have been employed for brine treatment and water softening applications. The feasibility of selective metal removal with improved energy efficacy of CDI-based hybrid process has been demonstrated at bench-scale (Choi et al. 2019). Further research efforts toward module design, scale-up operation, and sustainability analysis for CDI will be the key drivers for commercial application of this cutting-edge technology.

**EMERGING SUSTAINABLE EFFLUENT MANAGEMENT TECHNIQUES**

Potential alternatives like ZLD, wealth generation from waste (i.e. salt/mineral/chemicals recovery), constructed wetlands (CWs) techniques, etc. are getting attention for sustainable management of saline effluent. The use of saline effluents in other applications like shrimp production for aquaculture and resource recovery has also been suggested by researchers (Sorgeloos et al. 2001; Quist-Jensen et al. 2016). Some of the potential saline effluent management strategies have been discussed in the following subsections.
<table>
<thead>
<tr>
<th>Method</th>
<th>Advantages</th>
<th>Limitations</th>
<th>Future research prospects</th>
<th>Technology status</th>
<th>Potential application</th>
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</thead>
<tbody>
<tr>
<td>Reverse osmosis (RO)</td>
<td>Lower capital cost, High treatment efficiency, Easy scale-up, and Control</td>
<td>Need for pre-treatment system</td>
<td>Membrane material improvements, Scale inhibitors, Renewable energy integration</td>
<td>Commercial scale</td>
<td>Seawater/brackish water desalination (Greenlee et al. 2003)</td>
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<td></td>
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<td>Limited for hypersaline water</td>
<td>(photovoltaic solar power)</td>
<td>(leading technology)</td>
<td>Industrial and saline wastewater treatment</td>
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<td>High electric energy consumption</td>
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<td></td>
<td></td>
<td>Need for pre-treatment system</td>
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<td>Membrane fouling</td>
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<td></td>
<td>High capital cost, Corrosion and scaling</td>
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<td>Thermal multi-stage flash</td>
<td>Handle high salinity, Simple layout, Reliable performance</td>
<td>Energy-intensive, High capital cost, Corrosion and scaling</td>
<td>More efficient and low-cost construction materials, Use of renewable energy and cogeneration power plants</td>
<td>Commercial scale</td>
<td>Seawater or brackish water desalination (Al-Sahali &amp; Ettouney 2007)</td>
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<td>evaporation</td>
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<td>Ion-exchange process</td>
<td>The capability of treating concentrated brackish water, Selective removal of</td>
<td>Resin cost and regeneration, transfer of impurities to sludge, High maintenance, and operational costs</td>
<td>Process intensification for large-scale application</td>
<td>Commercial scale</td>
<td>Inorganic industrial wastewater, Textile wastewater (Moosavirad et al. 2015)</td>
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<td>Electro dialysis (ED)</td>
<td>High water recovery and salt removal, High segregation of metals, Relatively low energy consumption</td>
<td>Scaling, Membrane fouling, High capital cost, Organic matter, colloids, Inefficient for silica removal</td>
<td>Selection of membrane materials and stack to suit compatibility with the feed</td>
<td>Commercial scale</td>
<td>Brackish water desalination, RO brine (Korngold et al. 2003; Zhang et al. 2012), Electroplating industry (Al-Amshawee et al. 2020)</td>
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<tr>
<td>Membrane bioreactor (MBR)</td>
<td>Low footprint, handle high TSS and organic pollutants, High-quality effluent</td>
<td>Salinity pre-treatment, Fouling</td>
<td>Selection of membrane materials and stack to suit compatibility with the feed</td>
<td>Commercial scale</td>
<td>Vegetable oil refinery wastewater (Abdollahzadeh Sharghi et al. 2020)</td>
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<tr>
<td>treatment</td>
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<td>High strength wastewater (fat, oil, grease) (Jalinejad Falizi et al. 2018)</td>
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<tr>
<td>Membrane distillation (MD)</td>
<td>Low electrical energy consumption, High salt rejection, Handle high salinity (up to 20 wt%)</td>
<td>Membrane fouling and wetting</td>
<td>Potential of integrating with a waste heat source and solar energy, Fabrication of super-hydrophobic membrane materials</td>
<td>R&amp;D scale</td>
<td>Petrochemical industrial effluents (Osman et al. 2019), Dairy saline effluent (Abdellkader et al. 2019), RO concentrates (Camacho et al. 2015), Metabolic wastewater (Susanto 2011)</td>
</tr>
<tr>
<td>Forward Osmosis (FO)</td>
<td>Less energy-intensive than RO, Draw solution flexibility</td>
<td>Issues of reverse salt flux and concentration polarization</td>
<td>Development of suitable membranes and draw solution</td>
<td>R&amp;D scale</td>
<td>Landfill leachate treatment (Cath et al. 2006), Concentration of industrial waste (Cath et al. 2006; Zhao et al. 2012)</td>
</tr>
</tbody>
</table>
Waste minimization and water reclamation

This approach aims to produce a lower volume of concentrates by improving the water recovery from the saline effluent treatment process and simultaneously reducing the concentrates before ultimate disposal. However, the costs become substantial due to extensive treatment and severity of the operation of handling concentrates (Gude 2017). Treated water can be reused for potable and non-potable applications (as illustrated in Figure 13) to augment the water supply and offset the cost of treatment.

Osman et al. (2016) used direct contact membrane distillation (DCMD) in an attempt to concentrate RO and electrodialysis brine originating from difficult-to-treat petrochemical effluents. Overall, water recovery up to 80% and a salt rejection of up to 99.5% were achieved in the study. A mixture of sodium sulfate (Na$_2$SO$_4$), sodium chloride (NaCl), calcium sulfate (CaSO$_4$), calcium carbonate (CaCO$_3$), and glauberite (Na$_2$Ca(SO$_4$)$_2$) crystals was also detected in the MD brine.

Proactive management strategies focusing on the reduction of wastes and effluents at the origin can be of great importance. Improvements in operational aspects through automation, process intensification, and green technologies can help reduce waste at the source. For example, at seawater desalination plants, the brine volumes can be reduced by further concentration, applying alternative membranes and modules, and increasing recovery of the RO unit (Liyanaarachchi et al. 2014).
Zero liquid discharge (ZLD) is a progressive effluent treatment process, aiming to eliminate all liquid discharge from a system and produce a clean-water stream suitable for reuse. Processing of saline effluent to reach a dry end product using ZLD can be crucial from an environmental and resource recovery viewpoint. Also, safe disposal of dried end products into mine, bores, or sea is much more economically feasible than liquid disposal. Membrane processes are used to concentrate the pretreated waste stream and return the permeate to the process. Thermal evaporators, crystallizers, dryers, etc. are subsequently used to further reduce the liquid concentrates to a solid product for adequate usage/disposal. A few pilot-scale projects (e.g. SAL-PROC™, ZELDA) adopted the ZLD principle for sequential/selective extraction of dissolved elements in the form of valuable salts and chemical compounds (mineral, slurry, liquid form) from saline waters (ZELDA (Zero Liquid Discharge Desalination) Project n.d.). The ZELDA project has demonstrated the technical feasibility and economic sustainability of reducing the overall impact of desalination systems using brine management strategies based on the use of electrodialysis metathesis (EDM) and valuable compound recovery to reach ZLD. EDM involves the interchange of cations and anions between salts by employing additional solution compartments and ion-selective membranes in an electrodialysis unit. This configuration separates EDM concentrate into two streams of highly soluble salts i.e. Na-mixed stream and Cl-mixed stream with diminishing membrane fouling potentials of scalants (Panagopoulos et al. 2019). The high concentration liquid salts obtained from EDM are subjected to crystallization and evaporation to recover valuable compounds.

Mohammadesmaeili et al. (2010) proposed a sequential three-stage process as shown in Figure 14 to recover mineral salts and freshwater from RO concentrate on achieving ZLD. In the first stage, a modified lime-soda treatment, including acidification, lime addition, and soda addition was performed on feed (RO retentate) to remove...
magnesium, calcium, and silica. The second stage involved applying the secondary RO process to recover 80–90% of the freshwater and residual concentrate. In the final stage, the evaporation of residual liquid followed by crystallization was conducted to recover sodium sulfate (Na₂SO₄), glaserite (K₃Na(SO₄)₂), and salt mixture in three fragments. Recently, Xevgenos et al. (2015, 2016) investigated a solar energy-driven ZLD pilot-scale plant (Figure 15) under a European project (SOL-BRINE) for brine management.

The innovative features of the system include high water recovery (90%) with complete brine elimination, useful end products (distilled water and dry salt), and renewable energy-based operation with state-of-the-art technology (vacuum evaporation, solar drying). The demonstration plant at Agios Fokas comprised a two-effect vacuum evaporator, a vacuum crystallizer, and a solar dryer unit. The results showed the prospect of recovery of CaSO₄, CaCO₃, NaCl, MgSO₄, etc. from brine effluent with a salinity of ~7 wt% released from the desalination plant. The system was demonstrated to achieve high-quality water and was optimized for better energy performance.

**Resource recovery**

Saline effluents are being considered as an important source for the extraction of salts, chemicals, and minerals. Using suitable separation and purification techniques (described above), specific salt and minerals can be recovered from different sources like industrial saline wastewater, rejected brine, bitterns, etc. For instance, caustic soda production from brine using a diaphragm cell is an established technology and can be extended to use rejected brine of desalination plants. From among the various techniques to recover salts and minerals from waste brine, distillation (evaporation and cooling), membrane-based separation, electrodialysis, ion exchange, and eutectic freezing have been extensively used and reported in the literature (Korngold & Vofsi 1991; Karelin et al. 1996; Kılç & Kılç 2005; Ortiz et al. 2005).

In the context of the Gulf Cooperation Council countries, Alberti et al. (2009) have explored the possibility of reusing brine from desalination plants to produce sodium chloride for commercial use. Depending on the constraints and requirements of a different scenario, four configurations were presented, combining various technologies and processes, as shown in Figure 16. In addition, the costs, requirements, and outputs of technologies were compared and tabulated (see Table 6).

Although solar configurations (completely and partially) have a lower energy requirement, the salt purity is limited and an additional purification step may be needed to improve the product quality. The hybrid and thermal solution can be well suited wherever additional water production and limited land area are edging parameters.

A process for reuse and valorization of the brine from SWRO plants (Gran Canaria, Spain) for the chlor-alkali industry was developed by Melián-Martel et al. (2013). A block diagram of the proposed process is shown in Figure 17.

Primary purification was carried out using chemical precipitation, clarification, and filtration to remove impurities present in the brine. Subsequently, brine is preconcentrated using multi-effect evaporators to reach a desirable saturation level before carrying out the electrolysis. A subsequent purification step was performed using ion-chelating exchange resin to eliminate calcium and magnesium along with remaining metal impurities. Purified brine was sent to membrane cells for electrolysis of NaCl for the production of chlorine (Cl₂), caustic soda (NaOH), and hydrogen (H₂). The depleted brine from the electrolyzer was recirculated back for mixing with the purified brine. For an equivalent of 1,000 m³/day feed brine, the amount of Cl₂, NaOH, and H₂ obtained was 36.2, 126.6, and 1 ton/day, respectively. The energy
consumption was estimated to be 2,150 kWh/tonne of NaOH. The authors have suggested that the by-product hydrogen can be used in situ for gaining electrical energy to make this process self-sustained. However, the sludge generated at different stages of this process must be managed on the ground and solid waste due to the associated use of membranes. Another study on brine management via on-site sodium hypochlorite production was carried out by Abdul-Wahab et al. using an electrolysis process. On applying the low voltage DC, electrolysis occurs and sodium hypochlorite is generated instantaneously. This method eliminates the hazards of gaseous chlorine and provides a safer disinfectant chemical (Abdul-Wahab & Al-Weshahi 2009).

In a comprehensive literature review, Shahmansouri et al. (2015) have first briefly reviewed the extraction of several potential compounds, namely, NaCl, LiCl, CaCO₃, Rb₂CO₃, Mg(OH)₂, Cl₂, NaOH, Br₂, etc. from seawater and RO concentrates. The majority of the extraction processes were a combination of membrane and thermal processes discussed in the sections Membrane-based methods and Thermal operations. Subsequently, the economic assessment of commodity extraction of magnesia, chlorine-sodium hydroxide-hydrogen, sodium chloride, and bromine, magnesium hydroxide-sodium chloride was conducted considering capital costs, operation-maintenance cost, and sale revenues. The profitability was found to depend on commodity pricing and final product purity. The

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### Table 6 | Comparison of operational parameters (redrawn from Alberi et al. 2009)

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Completely solar</th>
<th>Partially solar</th>
<th>Hybrid</th>
<th>Completely thermal</th>
</tr>
</thead>
<tbody>
<tr>
<td>Land usage</td>
<td>High</td>
<td>High</td>
<td>Medium</td>
<td>Medium</td>
</tr>
<tr>
<td>Electrical energy usage</td>
<td>Very low</td>
<td>Medium</td>
<td>High</td>
<td>High</td>
</tr>
<tr>
<td>Thermal energy usage</td>
<td>Nil</td>
<td>Medium</td>
<td>High</td>
<td>High</td>
</tr>
<tr>
<td>Capital costs</td>
<td>0.7 M$</td>
<td>1.5 M$</td>
<td>2.5 M$</td>
<td>2 M$</td>
</tr>
<tr>
<td>Operational costs</td>
<td>$ 10,000/y</td>
<td>$ 40,000/y</td>
<td>$ 70,000/y</td>
<td>$ 60,000/y</td>
</tr>
<tr>
<td>Produced salt purity</td>
<td>Moderate</td>
<td>Moderate</td>
<td>High</td>
<td>High</td>
</tr>
<tr>
<td>Sweet water production</td>
<td>No</td>
<td>Yes</td>
<td>Yes</td>
<td>Yes</td>
</tr>
</tbody>
</table>

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![Figure 16](image_url)
economic analysis indicated that the extraction of bromine, chlorine, sodium, magnesium, potassium, and uranium is feasible for commercial purposes. Recovery of trace elements like rubidium, cesium, lithium, strontium, cobalt, etc. present in saline effluents was, however, not found to be economical.

Quist-Jensen et al. (2016) presented a novel integrated MD system (shown in Figure 18) combining Nano-Filtration (NF), RO, MD, and Membrane Crystallization (MCr) operations for simultaneous production of potable water and minerals from seawater. Thermodynamic simulations and experiments were performed for this purpose. Multivalent components, such as barium, strontium, and magnesium, were found to recover readily from NF retentate. KCl and NiCl₂ could be recovered from both NF and RO brine, whereas lithium was found to be recovered only from RO brine. The prospect of recovering copper and manganese from desalination brine was also reported.

In a recent study, the prospect of synthesizing magnesium–ammonium–phosphate fertilizer from sea bittern and an industrial waste stream containing ammonium carbonate and polyphosphoric acid has been presented. The composition of waste streams is reported in Table 7.
The stoichiometric ratio of $\text{Mg}^{2+}: \text{NH}_4^+: \text{PO}_4^{2-}$ in 1:1:1 resulted in high purity MAP completely removing nitrogen and phosphate from the effluent. Based on the pilot-scale experiments, a techno-economic analysis was also performed for a 1 t/d capacity plant resulting in a capital cost of 88 lakhs with a payback period of 4.75 years. This case study establishes a cost-effective and easy-to-implement method for producing fertilizer rich in ammonia and phosphate from saline waste effluents (Sanghavi et al. 2020).

Sengupta et al. have reviewed the recovery of nutrients (nitrogen and phosphorus) from wastewater, based on physical, chemical, and biological principles. Nitrogen in the form of ammonium ion ($\text{NH}_4^+$) or gaseous ammonia ($\text{NH}_3$) was recovered using various methods, namely ion-exchange and adsorption-based bio-electrochemical process, air stripping of ammonia and membrane-based schemes. With regard to phosphorous removal (in the form of elemental phosphorus or phosphoric acid), processes/methods including physical filtration and membrane processes, chemical precipitation, high-temperature acid hydrolysis, biological assimilation, physical–chemical adsorption, and ion-exchange have been reported. The recovery of macro-nutrients of nitrogen and phosphorous as synthetic fertilizers was recommended by the authors (Sengupta et al. 2015).

Resource recovery from saline effluents offers the dual advantage of managing effluents effectively and producing products with additional value. While the economic feasibility of recovering salts of sodium, magnesium, potassium, and bromine has been established, further research developments toward extracting trace elements are still needed.

**Table 7** Ionic composition of wastewater streams (Sanghavi et al. 2020)

<table>
<thead>
<tr>
<th>Ionic composition (%)</th>
<th>Waste stream containing ammonium carbonate</th>
<th>Waste stream containing polyphosphoric acid</th>
<th>Sea bittern</th>
</tr>
</thead>
<tbody>
<tr>
<td>$\text{NH}_4^+$</td>
<td>5.75</td>
<td>–</td>
<td>–</td>
</tr>
<tr>
<td>$\text{Mg}^{2+}$</td>
<td>0.04</td>
<td>0.04</td>
<td>7.87</td>
</tr>
<tr>
<td>$\text{Na}^+$</td>
<td>5.75</td>
<td>0.15</td>
<td>–</td>
</tr>
<tr>
<td>$\text{Ca}^{2+}$</td>
<td>0.04</td>
<td>0.09</td>
<td>–</td>
</tr>
<tr>
<td>$\text{Cl}^-$</td>
<td>0.13</td>
<td>0.09</td>
<td>25.00</td>
</tr>
<tr>
<td>$\text{SO}_4^{2-}$</td>
<td>0.02</td>
<td>0.02</td>
<td>4.19</td>
</tr>
<tr>
<td>$\text{PO}_4^{2-}$</td>
<td>–</td>
<td>20.5</td>
<td>–</td>
</tr>
<tr>
<td>$\text{K}^+$</td>
<td>–</td>
<td>–</td>
<td>1.80</td>
</tr>
</tbody>
</table>

**Constructed wetlands technique**

The concept of constructed wetlands (CWs) is gaining interest as an economical alternative for saline wastewater treatment. CWs are artificially engineered ecosystems, constituting plants, microorganisms, and soil-substrate, to remove contaminants/pollutants from waste effluents through physical, chemical, and biological processes. Halophytes or salt-tolerant plants, which possess the characteristics of excellent adsorption of salts and minerals, are used effectively under high salt stress conditions. These haloduric plants assimilate nutrients, accumulate contaminants into their tissues, and provide adsorption sites for microorganisms (Liang et al. 2017). The critical elements of CWs design include flow-path (horizontal or vertical), hydrology (surface-flow and subsurface-flow), plant-selection and density (plants per unit area). Pilot-scale CWs have been demonstrated for the removal of heavy metals from industrial saline wastewater. Some macrophytes (i.e., Typha domingensis, Typha angustifolia) have been found suitable for the treatment of wastewater of high conductivity and pH enriched with heavy metals (Hadad et al. 2006; Arivoli et al. 2015). Various factors, such as feed composition and salinity, climatic conditions (temperature, humidity), hydraulic retention time and microbial activities, influence the performance and efficiency of CWs. A careful combination of halophytes, halophilic-microorganisms, and optimal substrates is critical for achieving acceptable treatment efficiency of saline wastewater in CWs (Liang et al. 2017).

**Integration of renewable energy sources**

As the energy (thermal/electrical) consumption imparts a major fraction of treatment expenditure, the integration of renewable energy for saline effluent management is attractive to synergize the overall treatment process. A scheme for integrating solar and geothermal-based renewable energy sources to power membrane and thermal technologies is presented in Figure 19. It can be perceived that such integrations also help reduce the CO$_2$ emissions (due to fossil fuel burning) associated with these technologies.

In a recent study, Lu et al. (2019) modeled and designed a green energy powered zero liquid discharge desalination (ZLDD) system consisting of FC, MD, and crystallization. A lab-scale ZLDD system having 72 kg seawater feed was...
supported by a 50.5 m² solar panel (for MD) and 207 kg LNG refrigeration (for FC and crystallization). Further techno-feasibility and economic analysis however are required for implementing such amalgamated systems.

Another excellent example of utilizing solar power, of circumventing high thermal energy associated with the spray evaporation system (discussed in the section Spray evaporation/drying) for saline wastewater treatment, has been presented by Yu et al. (2019). Their system utilizes solar energy as a heating source for air and wastewater and combines the spray concentrator and separator to treat the high saltwater in two stages. Further studies on process optimization and economic evaluations on the entire system will warrant the practical applicability of this interrelated process scheme.

While solar-based systems (photovoltaic, solar concentrators, etc.) are being incorporated with treatment technologies, more efficient energy conservation and storage devices need to be designed in the coming days. Also, the potential of other renewable resources like ocean thermal energy, wind energy, etc. is still underdeveloped. Utilization of these potential resources for saline effluent treatment can be exploited for plants located near sea-shores or having exposure to strong wind.

CONCLUSIONS AND FUTURE OUTLOOK

Both conventional and emerging technologies on saline effluent treatment and disposal are critically reviewed. Conventional disposal of effluents has an adverse impact on marine ecosystems/groundwater-quality and is no longer preferred. More saline effluent treatment and disposal practices with lower environmental footprint are required than are used at present. Based on the characteristics of saline effluent under consideration, an adequate hybrid (membrane/thermal/biological) or stand-alone treatment system can be adopted. Membrane processes are promising at different stages of saline effluent management; however, further R&D efforts toward membrane material and design are needed for enhancing the performance and economy of the overall treatment. Application of emerging technologies, namely, low-pressure membrane processes, waste heat/solar energy-based thermal operations and low-cost biological systems towards saline effluent treatment, need to be demonstrated.

Since many effluent streams contain an array of pollutants, multiple treatment technologies are required in an optimized arrangement. A guiding flow diagram for saline effluent management is illustrated in Figure 20.
The hybrid/integrated systems presented in the review are capable of exploiting the full benefits of technologies involved with improved efficiency and operating costs. Nonetheless, these novel hybrid systems need to be comprehensively evaluated for economic viability and practicality, not only for setting up new plants but also for those already installed. The establishment of a common facility for collective treatment of saline effluent from clusters (industries/plants/rigs) can be a cost-effective approach due to the large-scale operation. Recovery of resources (water, salt, minerals, etc.) towards better saline effluents management will facilitate the generation of new business opportunities. Further progress towards implementing these strategies for converting waste effluents into wealth using renewable energy will strengthen the energy–water–environment nexus.

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DATA AVAILABILITY STATEMENT

All relevant data are included in the paper or its Supplementary Information.

REFERENCES


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