Water–energy sustainability synergies and health benefits as means to motivate potable reuse of coalbed methane-produced waters

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Abstract Management of coalbed methane (CBM)-produced water is a crucial part of the water–energy nexus, especially as CBM is projected to play a key role as a bridge fuel in major economies. In this paper, we consider one management technique, i.e., desalination of CBM-produced water to generate potable water. We discuss a confluence of geographic, sociotechnical, regulatory, and other circumstances that could make this concept viable for select coal-bearing regions. Having said that, for maximizing benefits, it is prudent to take a synergistic view targeting multiple objectives (water access, health, environmental impacts, and ease of waste management). Thus, we make design recommendations and suggest a system-evaluation framework for making sustainable decisions related to *produced-to-potable* water systems. For instance, a key question is whether such systems should be centralized or decentralized—and this paper highlights crucial tradeoffs that are present in both the cases.

Keywords Coalbed methane · Point-of-use · Produced water · Reverse osmosis · Water–energy nexus

INTRODUCTION

A critical challenge for our times is finding solutions to the so-called “water–energy nexus,” whereby, water is consumed to produce energy and energy is consumed to produce potable water (Schnoor 2011). Global climate models project dramatically increased water scarcity by the end of this century, arising from climate change impacts and mitigation efforts including energy decarbonization (i.e., reduction of net greenhouse gas [GHG] emissions for production of a unit amount of power) (Arnell 2004; Mall

et al. 2006; IPCC 2008; Immerzeel et al. 2010). Many low-carbon energy pathways are heavily water consuming, e.g., concentrated solar power (CSP), nuclear power, and CO₂ capture and storage (CCS) (Liu et al. 2015; Fricko et al. 2016; Srinivasan et al. 2017; Sharma and Mahapatra 2018). Thus, it is imperative to develop strategies to making water and energy systems less impactful on each other, especially in water-stressed developing economies with rapidly growing populations and surging energy demand.

The Paris Climate Agreement lays the framework for reducing the GHG intensities of major economies as means to reduce overall global emissions, and it is anticipated that China and India will play especially critical roles in this effort. Together, these two countries account for approximately a third of the world’s population and global GHG emissions. Under the Paris Agreement, they have committed to reductions in GHG emissions per unit gross domestic product by 60–65, and 33–35%, respectively, (compared with a 2005 baseline) by the year 2030. Rapid growth in renewable energy, most notably solar and wind energy, are predicted to contribute toward achievement of these goals. However, significant technology transfer and low-cost international financing (e.g., from the Green Climate Fund [GCF]) will be necessary to support the growth of these technologies. The significant uncertainty surrounding GCF funding makes it likely that clean-coal technologies will play a key role in decarbonization efforts (Shukla and Dhar 2016; Akash et al. 2017). In addition, both countries remain heavily reliant on coal for energy production, and this is not expected to change in the near future (Garg et al. 2017; Spencer et al. 2018). Although the ultimate goal for long-term sustainability will be widespread commercialization of nonfossil renewable energy platforms, it is of utmost practical importance to

understand how clean-coal alternatives can help bridge the transition from the current day to long-term future.

Coalbed methane (CBM)¹ will play an important role in decarbonizing the economies of China and India over the next several decades, as they transition to longer-term solutions. Aggressive developing of CBM resources will help both countries can make progress toward their GHG emissions reduction goals and simultaneously reduce their import dependence on natural gas. Figure 1 shows how the widespread commercialization of CBM can contribute to energy decarbonization in three critical ways:

1. *Reducing grid GHG emission factor* Since natural gas (methane) produces power at significantly lower CO₂ emission factor than coal, the overall grid emission factor—which is the weighted average of the coal-fired and gas-fired power plants—reduces substantially.
2. *Reducing fugitive methane emissions from virgin and operating coal seams* Fugitive emissions from underground coal mining contribute to one-sixth of the global warming potential (GWP) of the net coal-life cycle. Because the methane is combusted instead of getting released into the atmosphere, there is reduction of net emissions, since methane is a much more potent GHG than CO₂.
3. *Making carbon capture and sequestration (CCS) more appealing*² Incremental methane recovery from coal seams (known as enhanced CBM) can improve the economic prospects of the CCS process by supplementing significant part of the overall CCS-chain costs.

These benefits make CBM development attractive for near-term decarbonization in coal-rich developing economies. It is noteworthy that the typical lifespan of CBM wells is ~ 30 years, which closely corresponds to the service life of gas-fired power plant units. It is appealing that these assets have a relatively short timeframe, close to one generation, insofar as CBM is not intended as a long-term solution. As will be discussed in a later section, recurrent cost–benefits analysis and/or life-cycle assessment (LCA) will play an important role in evaluating the appropriateness of near- and long-term options over time, and determining when various technologies should be adopted and then ultimately phased out. The relative preferability of various technologies will change over time based on their economic and environmental performances.

¹ In this paper, CBM extraction is meant to connote a generic process of methane utilization from coal resources, either from virgin coalbed methane (VCBM) or coal mine methane (CMM) extraction, which involves simultaneous utilization of coal and methane.

² It is anticipated that CCS will need to account 1–2% of basic emission mitigations in the absence of large-scale renewables deployment (UNEP 2017).

In accelerating development of unconventional, lower-carbon fossil fuels as means to decarbonize their economies in the near-term future, China and India will be emulating the United States, which has achieved dramatic reductions in both GHG emissions and energy import dependence within the last 10 years—via rapid transition from coal to shale gas. Indeed, China and India have made significant progress in this vein thus far. CBM has become a critical component of China's gas portfolio (Dodson 2014), with production reaching around 4×10^9 cubic meters in 2014 (Mu et al. 2015). Similarly, CBM production in India has increased 20-fold in the last 7 years (Standing Committee on Petroleum and Natural Gas 2016) and is expected to account for 5% of total gas production by the end of 2017 (Thomas 2016).

Finally, China and India face significant challenges arising from chronic, severe regional water stress, together accounting for some 1.9 billion persons facing some type of water shortage (Mekonnen and Hoekstra 2016). Therefore, these countries must identify novel means of navigating and harmonizing interrelated energy, climate, and water objectives. This paper evaluates possible reuse of CBM-produced waters for potable water supply in water-stressed, coal-producing regions in China and India (Moore 2012). We contend that a confluence of social–technical factors (e.g., geographic, sociotechnical, regulatory, etc.) could make this approach uniquely appealing. As we will discuss, several key enabling technologies are already in place; however, integration of the various system components and formulation of supportive social policy will constitute key implementation challenges.

THE CARBON–WATER–ENERGY NEXUS IN CBM BASINS

In this section, we discuss the possible negative environmental impacts arising from widespread CBM development.

The state of water in coal-bearing areas

Geographically, many coal-producing regions in the developing world are in locations that experience chronic water scarcity. For example, it has been projected that more than half of the people living in the vicinity of the Jharia coalfield region do not have access to sufficient drinking water.³ Overuse of ground water sources has led to depletion of limited resources and lowered the water table elevation (Ghosh 1993). Mine water now constitutes

³ Jharia, located in eastern India is the only Indian coalfield which produces prime coking coal (required for steel production).

Fig. 1 Schematic illustrating three-fold GHG benefits arising from CBM utilization. Case (a) shows (underground) coal mining-to-combustion route. Case (b) shows utilization of methane over the first case. Case (c) shows capture of CO₂ from electricity generation and its disposal in coal seams to produce additional methane. The brown arrows depict CO₂ flow, while the blue arrows depict methane flow. The thickness of the arrows is meant to connote the relative flux in terms of global warming potential

their key potable source (Singh 1994; Mahato et al. 2017). In the future, existing regional water scarcity could be exacerbated by unchecked global climate change and implementation of high water-use technologies such as CCS (since power plants are often located close to coal mines to reduce transportation costs), thereby increasing the number of people without adequate drinking water access.

Referring to water quality, pristine groundwater in coal-bearing regions frequently contains high concentrations of sodium and calcium ions, and various geo-mining activities increase the concentrations of these substances (Singh et al. 2007). Backfilling⁴ tends to cause heavy metal contamination within the water (Mahato et al. 2008). Sediment runoff and leaching from overburden⁵ (OB) dumps pose significant contamination threat to surface and ground waters in India and elsewhere (Tiwary 2001). Acid mine drainage (AMD)⁶ also leads to severe water contamination and shortages in mining areas producing high-sulfur coal, such as some parts of Brazil (Galhardi and Bonotto 2016), China (Haibin and Zhenling 2010), and South Africa (Shiao and Maddocks 2014). Given that many coal-producing regions are geographically predisposed to high water scarcity and that water stress has historically been

exacerbated by water-quality impacts caused by mining operations, CBM-producing regions are in need of innovative strategies to secure sustainable potable water supply. Thus, CBM development ought to be seen as an attempt for near-term decarbonization effort, which can enable continued energy security for coal-rich countries. The risk of stranded assets for such projects is limited since the lifespan of such wells (~ 30 years) closely corresponds to that of gas-fired power plant units, which means it may be treated as a mitigation effort for a duration close to one generation. Further, as pointed out in Box 1, the produced water chemistry is amenable to conventional desalination treatment, and thus, the additional energy and cost requirements for produced water management do not immediately make CBM development a nonfeasible approach.

CBM challenges pertaining to produced water quantity

Although CBM offers several key benefits for energy decarbonization (i.e., reducing the GHG emissions per unit energy output), it is not without significant environmental drawbacks; most notably, its dramatic water footprint. Widespread deployment of CBM to achieve climate change mitigation goals could intensify existing water access challenges in coal-bearing areas. Large volumes of water are extracted during CBM production: on the order of 10³–10⁴ liters per day per well. Thus, a single well produces as much water as is consumed by several hundred

⁴ Filling mined open stopes by waste rock or sand to avoid subsidence (Rustan et al. 2010).

⁵ Material of low economic value covering the actual material of interest (Rustan et al. 2010).

⁶ Metal-rich water formed as a result of reaction of water with sulfur-rich minerals (U.S. Environmental Protection Agency 1994).

persons.⁷ To date, 725 wells have been drilled in India (Standing Committee on Petroleum and Natural Gas 2016). This number will grow exponentially as the CBM market matures from its current nascent state.⁸ As additional wells come online, more and more groundwater will be extracted. Continuous, long-term reservoir dewatering (Elder and Trevits 1975) remains the most popular technique for gas recovery, despite various methodological enhancements (e.g., in drilling, logging, monitoring, etc.) that have occurred over the last few decades.

There is very high variability in the rate of water extraction over time during CBM operation; whereby, most of the water produced from a single well is generated within the first 2 years of the well's operation (Fig. 2). Because individual wells are brought online rapidly during the initial phase of CBM operation, water production rate is very high for the first 5 years of a wellfield's useful life (Singh and Hajra 2018). It then decreases markedly over a roughly 20-year time horizon. Taking the average over this duration, water production rate is roughly on the order of 10^{-1} – 10^1 m³-water for 1 000 m³-gas produced⁹ (USGS 2000). Over the life of a typical CBM project, the net effect on regional water withdrawals is very pronounced—around 5 million cubic meters. In addition, sustained pumping can lead to significant groundwater depletion, with drawdowns of more than 3 m expected at distances of 8–16 km from the project site (Wheaton and Metesh 2002). Thus, widespread deployment of large-scale CBM production could further intensify water stress in coal-producing regions, by making groundwater even less available as drinking water supply.

CBM challenges pertaining to produced water quality

CBM production could also further threaten water quality in coal-producing regions. Previous studies have documented very high concentrations of total dissolved solids (TDS) in CBM-produced waters from different locations. Sodium (Na⁺) and bicarbonate (HCO₃⁻) ions generally account for the highest contributors to overall TDS (Table 1), though other species are also often present. Chloride (Cl⁻) ions may also be present at highly variable quantities, especially in mature CBM fields, often as result of frack fluid usage. Excessively high Cl⁻ concentrations (e.g., up to 200 000 mg L⁻¹) have been observed in some

CBM-produced waters (Flores 2014), which signals marine influence on the origin of the coalfield (Bouska 1981). To protect the drinking water supply, it is imperative that CBM-produced waters not be allowed to intermingle with regional groundwater and surface water sources. This can be challenging and cost-prohibitive. Re-use for irrigation purposes is theoretically appealing; however, prolonged application of CBM-produced waters can result in salt alkalization and salinization of agricultural soils, and these effects are exacerbated by significant transpiration and/or water stagnation due to poor drainage (Mallants et al. 2017; Wang et al. 2018). Other disposal mechanisms are expensive—costing as high as US\$ 0.3 per m³ (where corresponding value of gas is of the order \$ 10²–10³) (Boysen et al. 2002).

CONVERTING LIABILITY INTO ASSET: DIRECT POTABLE REUSE OF CBM-PRODUCED WATER

Rationale

As discussed above, the high volume and poor quality of CBM-produced waters render them an environmental liability. Simultaneously, many coal-producing regions are experiencing extreme water scarcity and have no viable drinking water supply. Accordingly, we posit that it may be worthwhile to evaluate the feasibility of CBM-produced waters for direct potable reuse; as means to convert an environmental liability into valuable assets and contribute to improved human health. Many of the factors that contribute to the appeal of this approach are unique to developing economies with high population densities. Thus, conventional wisdom from developed economies with historically good water access (e.g., US and EU) may not be directly relevant.

RO as an enabling technology: Opportunities

We posit that treated CBM-produced waters may be suitable for potable reuse in highly water-stressed coal-bearing regions without viable alternative water sources. In an analysis of CBM-produced waters from three of the gassiest coalfields in India (Jharia, Bokaro and North Karanpura), Singh and Hajra (2018) found that water samples were highly saline but relatively free from contaminating bacteria and other constituents of concern. Related work has found that CBM-produced waters tend to be relatively low in organic content; e.g., < 10 mg L⁻¹ in all CBM basins in the United States, compared to 24–5 804 mg L⁻¹ for shale gas-produced waters from the Marcellus shale formations (Orem et al. 2014) and have pH either in or very close to the preferred range of 6.5–8.5

⁷ Potable water consumption in this region is ~ 2 to 5 L per person per day (Hossain et al 2013).

⁸ For example, in the US, > 56 000 wells were managed simultaneously (U.S. Environmental Protection Agency 2010).

⁹ Water–gas ratio of 1 barrel per thousand cubic feet of gas corresponds to 5.6 m³-water per 1 000 m³-gas.

Fig. 2 Simulated gas and water generation profiles over the lifetime of a typical CBM reservoir (Salmachi and Özgen Karacan 2017). The water production rate in the primary axis is given in STB or stock tank barrel (which means 1 barrel in surface temperature and pressure conditions) and 1 STB = 0.159 m³. Similarly, the secondary axis shows the gas production rate in million metric standard cubic feet, and 1 MMSCF = 28 252 m³. Note that total lifetime of the well is considered to be 8 000 days, i.e., just over 20 years. Reproduced with permission

Table 1 Summary of TDS concentrations for CBM-produced waters from various regions, with representative levels of several principal ion species. *Source* Compiled from Kinnon et al. (2010), Dahm et al. (2011), Gui-Hong and Sijursen (2013), Singh (2017)

Coalfield/basin	Country	TDS mg L ⁻¹	(Na ⁺) mg L ⁻¹	(HCO ₃ ⁻) mg L ⁻¹	(Cl ⁻) mg L ⁻¹
GGR	USA	610–6230	240–2400	524–2870	4.5–2190
Powder River		252–2768	12–1170	236–3080	BDL–282
San Juan		150–39 260	36–7384	117–13 900	BDL–20 100
Bowen Basin	Australia	2650–8090	1100–3060	300–2860	747–4530
Jincheng	China	691–3409	119–945	346–2281	11–149
Jharia	India	550–2280	326–2273	1622–3021	57–344

(Kinnon et al. 2010; Dahm et al. 2011; Gui-Hong and Sijursen 2013). Thus, CBM-produced waters could become an appealing water source in regions with high water scarcity if the high TDS levels can be efficiently remediated.

It is our view that the recent dramatic growth of reverse osmosis (RO) in CBM-producing countries represents a key contributor to the viable reuse of CBM-produced waters for drinking purposes. Box 1 highlights the technical considerations of RO related to treatment of produced waters. Their relative absence of organic compounds and microbial organisms reduces the likelihood of biofouling (i.e., bacteria-mediated clogging), which is the principal operational concern for membrane-based treatment technologies including RO (Ang et al. 2011). It is also advantageous that the TDS in CBM-produced waters is typically dominated by sodium ions (Na⁺), insofar as single valent (+1) ions induce less fouling than multi-valent

ions (e.g., calcium [Ca²⁺] or magnesium [Mg²⁺]) (Mo et al. 2008).

The Asia-Pacific region is home to three of the major CBM-producing countries: Australia, China and India. This same region is also experiencing a dramatic surge in demand for RO and associated membranes, thereby stimulating worldwide growth in these industries. The global RO market was valued at US\$ 6.5 billion as of 2017 but is projected to grow to US\$ 9 billion by 2022 (Research and Markets 2017). In some urban areas in India, household-level RO penetration is as high as 50% (O'Connor 2016). Domestic water purification, of which home-use RO constitutes as major share, is a billion-dollar industry in India (Rawat 2013). We contend that the already high and still growing demand for household-level RO could be a good indicator that this technology is culturally appropriate for direct reuse of CBM-produced waters in coal-producing developing regions.

Box 1 Reverse osmosis and its suitability for treatment of CBM-produced waters

RO is a membrane-based treatment for removing dissolved inorganic salts from water. The raw water is filtered through a semipermeable membrane at high pressures (up to 6 900 kPa). For brackish waters with initial TDS on the range 2 000–10 000 mg L⁻¹, RO produces treated effluents with TDS < 500 mg L⁻¹. Yield is 75–80% by volume (i.e., 7.5–8 L of treated water is produced from 10 L of raw water). For increasing initial TDS (e.g., seawater with TDS > 10 000 mg L⁻¹), recovery is reduced to 60%.

RO is highly energy consuming. This has historically made it cost-prohibitive for developed countries with abundant water resources or viable treatment alternatives (e.g., the US or EU) (Tchobanoglous and Burton 1991; Li 2011). However, the dramatic decrease in RO treatment costs over time (Fig. 3) has made it increasingly cost-competitive for treatment of select water chemistries in domestic use applications. Even so, RO remains technologically infeasible and/or economically impractical for some raw waters. For example, shale gas-produced waters exhibit TDS > 75 000 mg L⁻¹ and require higher-than-allowable transmembrane pressures (Shaffer et al. 2013). Other industrial and geological-produced waters containing appreciable dissolved iron are unsuitable for RO treatment because of significant problematic scaling. Thus, RO appears to be uniquely suitable for treatment of CBM-produced waters. This is reflected in the cost comparison for RO relative to other management strategies that could be considered (Fig. 4).

When thinking about the appropriateness of RO treatment, it is informative to consider other alternatives to achieve the same goal. For direct potable reuse of CBM-produced waters, the system serves dual functions: managing large volumes of highly saline “wastewaters” and simultaneously creating potable drinking water. Alternative *management* options include reinjection, surface discharge, or reuse in irrigation purposes. None of these is especially appealing. Reinjection has high costs (Fig. 4) due to significant energy investment for pumping. There are also regulatory issues related to avoiding groundwater contamination. Similarly, reinjection is not suitable for low-permeability coal seams. It is most widely used within the US, for enhancing oil production in locations where oil and coal formations are located in close physical proximity. China and India have few sites, if any, that are suitable for this prospect. Finally, surface discharge and irrigation reuse are much less expensive than reinjection; however, each presents contamination risks (Ramirez 2005). Irrigation-based reuse may cause lasting soil damage because of high sodium carbonate contents, and few crops can be irrigated with high-TDS waters (Poudyal et al. 2016). With respect to drinking water treatment, other desalination technologies do exist on the market, e.g., ion exchange. These physicochemical treatments that are very expensive are not yet widely commercialized compared to RO. It may also be noted that these desalination technologies are different in chemistry and intent from other commonly deployed point-of-use treatment methods such as activated charcoal-based filtration—which is more useful for removal of organics. Thus, utilization of a developed technology like RO bodes well for public/industrial acceptance.

Fig. 3 RO-treatment costs over time for raw waters of various qualities. Inset depicts detailed ranges of energy-consumption and treatment costs for brackish waters and seawater (Cohen et al. 2017). CBM-produced waters are characterized as “brackish” based on TDS

Finally, it is noteworthy that many developing countries, including China and India, have relatively flexible drinking water regulations compared to most developed countries (e.g., USA and Australia). For instance, Australia mandates TDS below 600 mg L^{-1} for palatability and requires sodium below 180 mg L^{-1} (NHMRC, NRMCC 2011). India has a similar TDS limit (500 mg L^{-1}); however, levels up to 2000 mg L^{-1} are acceptable for areas without any alternative sources, and there is no specific sodium limit (BIS 2012). Based on the typical composition of CBM-produced waters (e.g., Table 1) and the typical design specifications of current RO technology, it is not unreasonable to imagine that households could treat CBM-produced waters to a level that satisfies pertinent drinking water regulations. Increased access to safe, healthy potable water would have substantive positive impacts on quality of life in affected regions.

RO as an enabling technology: Challenges

All technologies have drawbacks, and RO is no exception. A principal disadvantage of RO is that it creates highly concentrated brine wastes. These wastes contain all of the salts that were removed from a water sample during desalination. For example, use of RO to achieve 60–85% TDS removal for a typical CBM-produced water (Table 1)

could generate brines with TDS concentrations above 10000 mg L^{-1} . Such wastes are highly noxious and require careful management. Thus, brine treatment becomes an important operational consideration for the proposed scheme. This question is also highly relevant for the burgeoning RO industry in Asia-Pacific, beyond what we are proposing in this paper. It is most desirable to remediate RO brines in a way that produces valuable materials, as means to offset collection and treatment costs. The composition of CBM-produced waters has been of considerable research interest in recovery of sodium, chloride, and other salts from high-TDS wastes. A detailed perspective on such options is provided in Box 2.

Another major issue associated with RO is its high energy consumption. Currently, China's and India's energy consumption for water treatment, on a per-capita basis, is relatively high compared to the rest of the developing world, yet considerably lower than United States and the Middle East (Liu et al. 2016). Increasing reliance on RO to enhance water access will increase energy consumption for water treatment and could make China and India more comparable to the developed world. This trend is not likely to be significantly offset by anticipated process improvements, though there is strong interest in configuration adjustments to minimize overall energy use (Shrivastava et al. 2015; Mazlan et al. 2016; Werber et al. 2017).

Regardless, it should be emphasized that the CBM industry would be required to expend energy for produced water management even in the absence of any desire to use it as potable water supply. We posit that it is, therefore,

worthwhile to use RO for treatment of CBM-produced waters even though it is relatively a higher energy-consuming process.

Box 2 Highlighting technologies for material recovery from concentrated brine

Materials recovery from high-TDS brines is an actively evolving area. The strong interest in this topic has been spurred by strong growth in advanced desalination in the Middle East and also the shale gas revolution the United States. While the commercial appeal of such processes may not be immediately obvious at present, there have been several recent, noteworthy advancements. A few highlights are summarized as follows:

- Ahmed et al. (2003) studied salts recovery (e.g., gypsum, sodium carbonate, bittern) from RO brines with high sodium and bicarbonate content through the proprietary SAL-PROC method.
- Macedonio et al. (2011) studied recovery of calcium and sodium chlorides from concentrated brines using wind-aided intensified eVaporation (WAIV) and membrane crystallizer (MCR) techniques.
- Ji et al. (2010) estimated that membrane distillation-crystallization boosts RO recovery to more than 90% along with NaCl recovery of up to 17 kg m⁻³.
- Boopathy and Sekaran (2017) evaluated NaCl recovery via passing of hydrochloric acid through brine residue after evaporation. They have found a cost reduction by 57% through this method as compared to disposal into a landfill.
- El-Naas et al. (2010) have proposed a unique means of fusing brine management and CO₂ capture, whereby, ammoniated brine is reacted with CO₂ to precipitate TDS. Carbon capture was almost 90%. Sodium removal was ~ 45% at 20 °C, while calcium and magnesium were almost completely removed. A process for simultaneous CO₂ capture (at 99% efficiency) and using the modified solvent process without ammonia has also been suggested (El-Naas et al. 2017).

Materials recovery is doubly appealing as part of the proposed CBM produced-to-potable water scheme. First, it will be necessary to treat the RO waste brines before they can be safely disposed. Second, production of a valuable material generates a revenue stream that could offset the costs of brine treatment and/or RO. But the latter is only true if the economic value of the recovered materials is greater than the cost of the recovery technique. Interestingly, a recent survey of brine materials recovery methodologies shows that this is indeed true for several proposed technology options (Fig. 5).

Fig. 5 Total desalination costs and value of recovered salts for selected integrated brine management and materials recovery processes. Integrated process scheme: (1) ED-MSF-Crystallization; (2) UF-NF-MSF-Crystallization; (3) UF-NF-RO-MSF-Crystallization; (4) MF-NF/MCR-RO/MCR; (5) MF-NF/MCR-RO/MD; (6) Precipitator/MF/drying bed/NF/RO/ED/MED/Precipitator/Evaporator-crystallizer. ED electro-dialysis, MSF multi-stage flash, UF ultrafiltration, NF nanofiltration, MF microfiltration, MCR membrane crystallization, MD membrane distillation, MED multi-effect distillation Source Gede Wenten et al. (2017). Individual technologies are described in greater detail in the original paper from which this figure was adapted (Gede Wenten et al. 2017)

ANTICIPATING THE COSTS AND BENEFITS OF AN INTEGRATED CBM PRODUCED-TO-POTABLE SYSTEM

We have discussed the conceptual merits of our proposed scheme; however, it remains to be seen how the system should be configured to best deliver the desired benefits at the lowest possible cost. How do we evaluate the best possible design choice for such concepts? This question especially becomes relevant since some companies have already started installing pilot RO plants for CBM-produced water, and there is an urgent need to understand possible design tradeoffs before any one option becomes widely commercialized. The meaningfulness of this evaluation will hinge upon selection of suitable metrics to address all components of the so-called sustainability triple bottom line: economic, social, and environmental acceptability. In this section, we elucidate some important management considerations that will impact the design of CBM produced-to-potable water systems in specific developing-world contexts and discuss various possible performance metrics of interest.

Overarching design framework: Centralized versus decentralized configuration

One of the most fundamental design considerations related to deployment of RO-based desalination of CBM-produced waters is selection of a centralized or decentralized treatment configuration. In a centralized system, the raw water is transported to a single, large-scale treatment facility, and the potable effluent is delivered to individual households via a piped network. Most households in the US and EU are serviced by a centralized water system. In a decentralized system, the raw water is delivered to individual households via a piped network. Each household then treats its own drinking water using an appliance-scale device. Decentralized treatment is sometimes also referred to as “point of use” (POU) treatment. Figure 6 illustrates a comparison of centralized and decentralized water treatment configurations. Subsequent paragraphs summarize several principal advantages and disadvantages of each configuration.

Centralized systems offer good economies-of-scale with respect to construction and operation/maintenance costs. For this reason, centralized desalination has become extremely popular in the Middle East. The capital investment required per unit output of treated water is much lower in a centralized desalination system than in decentralized desalination system serving the same population size. Another beneficial aspect of centralized systems is the tunability of the process for various raw water qualities. As discussed above, there is dramatic variability in CBM-produced water quality, especially in TDS. For raw waters

with very low salinity, it is possible to apply very minor treatment at very low cost to make the water drinkable. For example, some CBM waters from the Jincheng basin in China could be easily treatable by RO or even nano-filtration (which is less energy-consuming than RO) to satisfy the $\text{TDS} \leq 1\,000\text{ mg L}^{-1}$ regulatory requirement for Chinese drinking waters (Ministry of Health of China 2006). This is especially significant because Chinese drinking-water standards are among the most rigid in the developing world (Qu et al. 2012). For higher-salinity raw waters (e.g., $\text{TDS} = 1\,000\text{--}20\,000\text{ mg L}^{-1}$), the RO system can be tuned to deliver similarly treated water of high quality, though at a greater economic cost. In this way, it is possible to make use of higher-quality raw waters first with the increasing expenditure to deliver larger treated volumes only as needed. Finally, RO treatment of increasingly saline raw waters will create increasingly concentrated waste brines (e.g., $50\,000\text{ mg L}^{-1}$). Thus, another principal advantage of the centralized treatment configuration is that it facilitates efficient brine collection and management. Brine treatment may be significantly easier in a centralized system because the waste will be produced at a single location and can be remediated using large-scale unit operations managed by trained personnel. For highly concentrated waste brines, material recovery may become a viable, appealing option (Box 2), and these technologies are generally more economically attractive at large scale.

For decentralized or point-of-use systems, a principal advantage is the already high market penetration of domestic water purifiers in relevant regions. Also, while the costs per unit water treated can be higher, there is also greater certainty that the final consumed water will be of the desired quality as compared to the centralized system. In decentralized systems, the treated water is consumed very quickly after treatment without having to traverse a piped distribution network. It has been well documented that treated water can become microbially contaminated as it travels through the transmission systems, especially in developing countries (Bahl 1976; Semenza, et al. 1998; Bhunia et al. 2009; Reese et al. 2017). However, the risk of contamination is dramatically reduced with reduced intermittency of water supply (Kumpel and Nelson 2013; Adane et al. 2017). Ren et al. (2013) demonstrated that POU treatment may be especially beneficial for developing country scenarios due to convergence of economic, social and environmental criteria.

The main drawback of a decentralized configuration is the relative lack of process control. This undermines both reliability and efficiency. With respect to reliability, it is difficult to assess the actual quality of the treated water because the RO devices are operated in private homes by untrained users. With respect to efficiency, there is no economy of scale or opportunity to tune the extent of RO

Fig. 6 Proposed treatment and distribution network schematics for **a** centralized treatment and **b** decentralized treatment

treatment based on raw water quality (i.e., salinity) (Fig. 7). Also related to efficiency, it is more difficult to manage waste brines within the decentralized configuration, because they are created in very small quantities over a large geographic footprint. These logistical challenges also reduce the attractiveness of some material recovery techniques for brine management.

Quantifying anticipated economic costs

Economic cost is a key metric for evaluating any project. For the proposed scheme, it will be relevant to compare leveled capital and operating investments for treating some fixed volume of water over an assumed wellfield lifetime of 20 years. Looking at Fig. 7, if the amounts of supply of various water qualities from coal-based formations (produced water and mine water) could be determined, a cost-supply framework could be developed which could yield the amount of water through such sources if the willingness to pay is known. Alternatively, if the population to which the water needs to be provided is fixed, the area of such a curve could indicate the overall investment required for such processes. It may be noted that because of ease of process

control and regulation for a centralized system, three levels of water treatment may be performed, incurring different costs as shown in Fig. 7. The decentralized systems can be assumed to have constant costs since the investments shall be majorly composed of purchasing expensive POU RO units. The different costs that may be computed here are the capital cost for the RO equipment, transmission cost, and operation cost (mostly comprising membrane replacement, energy, and labor), and the cost of water management averted. For a centralized plant, the capital costs would largely be upfront. However, for the decentralized system, owing to lifetime of 6–8 years, there would be multiple capital-cost-addition periods. Apart from that, the other costs such as energy consumption and management charges averted can be expected to be similar.

It is not enough to compute economic costs of various proposed systems. We must also quantify the anticipated benefits as means to normalize what values various investments could yield. This is often done by valuing benefits (discussed in the next section) and establishing cost-effectiveness thresholds of various measures, which are often used by decision-makers (Eichler et al. 2004; Neumann et al. 2014).

Fig. 7 Hypothetical cost curves for centralized and decentralized systems for combined water sources. Here, the low-TDS CBM water has a TDS < 1 000 mg L⁻¹; the mine water has a variant TDS between 300 and 1 300 mg L⁻¹ (but with other major contaminants) (Singh et al. 2010), and the brackish CBM water has TDS > 2 000 mg L⁻¹. There is a scope for colocation of treatment facilities for both mine water and CBM water since they are likely to be found at similar geographic locations. For an enhanced discussion about mine water, the reader may refer to this review (Pinto et al. 2016)

Quantifying anticipated social and environmental benefits

Evaluating costs is relatively straightforward but in terms of the benefits, a system framework needs to be developed for the monetized benefits of reduced water-borne diseases. Water-borne diseases which often stem from untreated water, occur in large numbers in India. These include diarrhea, typhoid, cholera, viral hepatitis and acute encephalitis syndrome. These problems are often compounded in coal mining regions. Providing clean drinking water to such areas can lead to significant health benefits, which may be measured in terms of several well-known metrics such as averted disability adjusted life years (DALYs). It would be worthwhile to consider a systems-level analysis focusing on the health benefits arising out of such diseases, since costs of treatment for these can be highly considerable (NCMH 2005). Since medical aid in many of these regions/sectors is socialized, the monetary advantages accrued through averting water-borne diseases can be invested for setting up desalination systems. Having said that, the cost-benefits ratios for such systems is not known. A number of factors here are at play. Firstly, how do we monetize health benefits. Secondly, while the cost investments and the health benefits may be monetized, another question is the willingness of the consumers or the government to pay for such a program. Drinking water has been known to be a major contributor to severe health crises in these countries (Srinath Reddy et al. 2011; Lu

et al. 2015; Perry et al. 2015). It is understandable that the government of these countries have initiated several reforms to achieve universal health care (Marten et al. 2014) and there is significant coverage for the poorer population and those serving in government enterprises. For example, the *Rashtriya Swastha Bima Yojana* (National Health Insurance Scheme) specifically covers many unorganized-sector workers including miners especially. In addition to that, it has been seen that the health costs of produced water management from unconventional fields also are comparable (smaller by a factor of 2–3) to the disposal costs (Bartholomew and Mauter 2016).

As Zapata et al. (2017) indicate, clean energy can lead to public health savings in the state of California. We contend that given the extremely high population density in the areas under consideration, the health benefits of clean drinking water to many people can supplement the costs for treating produced water and the public-sector enterprises have an incentive to do these things, as shown by the active involvement of industries for mine water treatment.¹⁰

¹⁰ Mine water treatment efforts have finally led to commercialization of these prospects by Coal India Limited with costs of treatment at ~ US\$ 3/m³ (NML 2017). As per local media reports, another private sector company JUSCO, in association with the local municipality is mulling investment of US\$ 15 million to convert pit water to potable water. In fact, in a recent international meeting of various urban governance bodies, this was featured as one of the major achievements of the city of Dhanbad (City Report of Dhanbad, India 2018).

Having said that, incorporating social factors into traditional environmental assessments or creating altogether new social assessments has been suggested by research to improve the feasibility of decisions on the end user (Sureau et al. 2018; Zimdars et al. 2018). This calls for improved dialogue with the beneficiary population—without which the full benefits of such an initiative may not be reaped. This has to do with proper management of water at household levels. While piped water supply does lead to reduction in diarrheal incidences, education and awareness levels have been shown to be the key to ensuring health benefits—even with clean water (Jalan and Ravallion 2003; Brick et al. 2004). Improper water storage is a crucial challenge, about which awareness needs to be generated at household levels. This may be compounded with boiling, filtering, or chlorination to inhibit microbial growth (Sobsey et al. 2003; Clasen 2015). Another useful feature of dual-direction societal interaction is the advantage of low-cost interventions that can be brought by local entrepreneurs. Annala et al. (2018) discuss one such case of frugal RO systems in western India—costing 42% less than branded ones. Thus, capitalizing on the end-user's understanding of such systems can be instrumental in large-scale interventions in developing countries.

Decision-making frameworks should also encompass environmental sustainability parameters, notably energy consumption and solid/liquid waste generated. The efficiency improvement in RO systems has stabilized owing to complete technological commercialization, while waste generation may be calculated through simple mass balances. An increasing stress toward making the right choices is also in terms of resilience, i.e., the ability of systems to maintain *status-quo* in changing environmental conditions. This also has interesting repercussions in terms of produced-to-potable decisions. For instance, what if stronger climate targets and/or accelerated renewable technology development eliminate the need and scope for CBM development altogether, and in such a case, which system would be more operable with changed water sources? In such a case, it is prudent to evaluate the more modular design which can spread risks along multiple systems (Ahern 2011). It is prudent to consider reusability of pipelines/treatment systems that are installed, for other purposes past the project life. Sustainability and resilience have some similar overarching goals (human–nature harmony, community well-being) but also have considerable differences.

Assessing government support for choices

While we do refer to economic profitability and social acceptance as important evaluative measures, government support is also crucial in predicting the more anticipated

design choices. In countries such as India, the government does constitute a single, centralized stakeholder with possible strong interests in designing an integrated energy–water system to maximize public health and other outcomes in socialized economies. It is interesting to note that China and India have a socialist aspect of the government, even though the structure of the governments are somewhat different. For instance, the Chinese coal production is dominated by a handful of state-owned companies and while there is some degree of autonomy in the Indian coal and gas sectors, the Ministries of Coal and Petroleum and Natural Gas play a rather important overseeing and supervisory role in guiding the policy of the energy sector. Thus, we argue that the government and its owned enterprises have much influence as far as framing of operations is concerned. This leads, in a way, to an advantageous situation to considering beneficial uses of CBM-produced water given a lesser number of stakeholders. However, such choices may not be always perceived as the most acceptable by the public but, because of the considerable control of the governmental agencies, schemes may be implemented with considerable societal weaknesses. For instance, within the sanitation sector, the *Swachh Bharat Abhiyan* (Clean India Mission with an annual budget of over US\$ 500 million) has received some criticism for overlooking social inequities (Gatade 2015), lack of training within implementers (Routray et al. 2017), etc.

System integration is another criterion for such a concept which relies on their inherent synergies to be perceived as advantageous. Figure 8 showcases the synergies that can be achieved via systems integration, but integration requires purposeful coordination across stakeholders. On the government front, it is also required to assess the scope and probability of forging intersectoral collaboration since economic and environmental sustainability for multiple criteria may be easier achieved with a centralized stakeholder in the government. For instance, produced water may be derived from active coal mines, and from virgin coal seams. Integrating the water resources from different sectors would not be a mere connection of pipelines; rather may reflect a legal stipulation as well. Moreover, for water-quality standards, fossil-fuel industries will need to have environmental clearance from the specific ministries/departments which ensure proper potability standards. This is not trivial, especially when different factions have different priorities among competing social, environmental, and economic objectives. Thus, the notion of “public good” is central to the ideas forming the base of this paper. We hypothesize that countries such as India and China, where public sector enterprises (PSEs) have significant backing of the government, may be better able to implement systems integration approaches that are

Fig. 8 Possible synergies for produced-to-potable concepts identified in this paper

more likely to make decisions to enhance quality of life for people in such areas.

CONCLUSIONS

Energy decarbonization in China and India will enhance incentives for rapid expansion of the CBM industry, potentially creating tradeoffs among climate, energy, and water objectives. We have highlighted several sociotechnical factors that could make direct reuse of RO-treated CBM-produced waters an appealing means of harmonizing various sustainability objectives. While methane was seen largely as a mine hazard in the context of coal mining, the development of drilling and utilization technologies have made it a resource. Similarly, we argue that produced water may also be seen as an economic capital in the right conditions, especially when human health and other social benefits are seen as monetized quantities. In this perspective paper, we have tried to describe several synergies that may aid in solving the coal–water nexus through the produced-to-potable concept (summarized in Fig. 8). Possible

questions of interest for future related work in this vein will include quantification the financial costs and social/environment benefits arising from centralized or decentralized RO treatment of CBM and other produced waters; assessing current willingness of various stakeholders (e.g., coal and gas companies, households, etc.) to try out this and similar initiatives as means to fund sociotechnical initiatives designed for the greater good. We assert that produced water “resources,” in addition to gas resources, have the potential to serve as social capital if technology and policy can be appropriately aligned.

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REFERENCES

- Adane, M., B. Mengistie, G. Medhin, H. Kloos, and W. Mulat. 2017. Piped water supply interruptions and acute diarrhea among under-five children in Addis Ababa slums, Ethiopia: A matched

- case-control study. *PLoS ONE* 12: e0181516. <https://doi.org/10.1371/journal.pone.0181516>.
- Ahern, J. 2011. From fail-safe to safe-to-fail: Sustainability and resilience in the new urban world. *Landscape and Urban Planning* 100: 341–342. <https://doi.org/10.1016/j.landurbplan.2011.02.021>.
- Ahmed, M., A. Arakel, D. Hoey, M.R. Thumarukudy, M.F.A. Goosen, M. Al-Haddabi, and A. Al-Belushi. 2003. Feasibility of salt production from inland RO desalination plant reject brine: A case study. *Desalination* 158: 109–117. [https://doi.org/10.1016/S0011-9164\(03\)00441-7](https://doi.org/10.1016/S0011-9164(03)00441-7).
- Akash, A.R., A.B. Rao, and M.K. Chandel. 2017. Relevance of carbon capture & sequestration in India's energy mix to achieve the reduction in emission intensity by 2030 as per INDCs. *Energy Procedia* 114: 7492–7503. <https://doi.org/10.1016/j.egypro.2017.03.1882>.
- Ang, W.S., A. Tiraferri, K.L. Chen, and M. Elimelech. 2011. Fouling and cleaning of RO membranes fouled by mixtures of organic foulants simulating wastewater effluent. *Journal of Membrane Science* 376: 196–206. <https://doi.org/10.1016/j.memsci.2011.04.020>.
- Annala, L., A. Sarin, and J.L. Green. 2018. Co-production of frugal innovation: Case of low cost reverse osmosis water filters in India. *Journal of Cleaner Production* 171: S110–S118. <https://doi.org/10.1016/j.jclepro.2016.07.065>.
- Arnell, N.W. 2004. Climate change and global water resources: SRES emissions and socio-economic scenarios. *Global Environmental Change* 14: 31–52. <https://doi.org/10.1016/j.gloenvcha.2003.10.006>.
- Bahl, M.R. 1976. Impact of piped water supply on the incidence of typhoid fever and diarrhoeal diseases in Lusaka. *Medical Journal of Zambia* 10: 98–99.
- Bartholomew, T.V., and M.S. Mauter. 2016. Multiobjective optimization model for minimizing cost and environmental impact in shale gas water and wastewater management. *ACS Sustainable Chemistry and Engineering* 4: 3728–3735. <https://doi.org/10.1021/acssuschemeng.6b00372>.
- Bhunia, R., R. Ramakrishnan, Y. Hutin, and M.D. Gupte. 2009. Cholera outbreak secondary to contaminated pipe water in an urban area, West Bengal, India, 2006. *Indian Journal of Gastroenterology* 28: 62–64.
- BIS. 2012. Indian Standard: Drinking Water-Specification. New Delhi: Bureau of Indian Standards. Retrieved November 2, 2017, from <http://cgwb.gov.in/Documents/WQ-standards.pdf>.
- Boopathy, R., and G. Sekaran. 2017. Studies on process development for the separation of sodium chloride from residue after evaporation of reverse osmosis reject solution. *Separation and Purification Technology* 183: 127–135. <https://doi.org/10.1016/j.seppur.2017.04.008>.
- Bouska, V. 1981. *Geochemistry of coal-coal science and technology 1*. Amsterdam: Elsevier.
- Boysen, D.B., J.E. Boysen, and J.A. Boysen. 2002. Strategic produced water management and disposal economics in the Rocky Mountain region. Retrieved April 16, 2018, from http://www.gwpc.org/sites/default/files/event-sessions/Deidre_B_Boysen_PWC2002_0.pdf.
- Brick, T., B. Primrose, R. Chandrasekhar, S. Roy, J. Muliyl, and G. Kang. 2004. Water contamination in urban south India: Household storage practices and their implications for water safety and enteric infections. *International Journal of Hygiene and Environmental Health* 207: 473–480. <https://doi.org/10.1078/1438-4639-00318>.
- City Report of Dhanbad, India. 2018. Eighth Regional 3R Forum in Asia and the Pacific. *Achieving clean water, clean land and clean air through 3R and resource efficiency-A 21st*. Indore, India. Retrieved May 1, 2018, from http://www.uncrd.or.jp/content/documents/6185City%20Report_India_Dhanbad.pdf.
- Clasen, T. 2015. Household water treatment and safe storage to prevent diarrheal disease in developing countries. *Current Environmental Health Reports* 2: 69–74. <https://doi.org/10.1007/s40572-014-0033-9>.
- Cohen, Y., R. Semiat, and A. Rahardianto. 2017. A perspective on reverse osmosis water desalination: Quest for sustainability. *AIChE Journal* 63: 1771–1784. <https://doi.org/10.1002/aic.15726>.
- Dahm, K.G., K.L. Guerra, and J.E. Drewes. 2011. Composite geochemical database for coalbed methane produced water quality in the Rocky Mountain Region. *Environmental Science and Technology* 45: 7655–7663. <https://doi.org/10.1021/es201021n>.
- Dodson, S. 2014. Coalbed methane in China: Challenges and obstacles. *World Coal*. Retrieved October 13, 2014, from <https://www.worldcoal.com/cbm/22082014/cbm-in-china-special-report-cbm100/>.
- Eichler, H., S.X. Kong, W.C. Gerth, P. Mavros, and B. Jonsson. 2004. Use of cost-effectiveness analysis in health-care resource allocation decision-making: How are cost-effectiveness thresholds expected to emerge? *Value in Health* 7: 518–528. <https://doi.org/10.1111/j.1524-4733.2004.75003.x>.
- Elder, C.H., and M.A. Trevits. 1975. *Gas production from the Pocahontas No.-3 coalbed, Island Creek Mines, Buchanan County, WV*. Bruceton Station, PA: Bureau of Mines.
- El-Naas, M.H., A.F. Mohammad, M.I. Suleiman, M. Al Musharfy, and A.H. Al-Marzouqi. 2017. A new process for the capture of CO₂ and reduction of water salinity. *Desalination* 411: 69–75. <https://doi.org/10.1016/j.desal.2017.02.005>.
- El-Naas, M., A.H. Al-Marzouqi, and O. Chaalal. 2010. A combined approach for the management of desalination reject brine and capture of CO₂. *Desalination* 251: 70–74. <https://doi.org/10.1016/j.desal.2009.09.141>.
- Flores, R.M. 2014. *Coal and coalbed gases*. Waltham, MA: Elsevier. <https://doi.org/10.1016/B978-0-12-396972-9.00008-2>.
- Fricko, O., S.C. Parkinson, N. Johnson, M. Strubegger, M.T.H. van Vliet, and K. Riahi. 2016. Energy sector water use implications of a 2 °C climate policy. *Environmental Research Letters* 11: 034011. <https://doi.org/10.1088/1748-9326/11/3/034011>.
- Galhardi, J.A., and D.M. Bonotto. 2016. Hydrogeochemical features of surface water and groundwater contaminated with acid mine drainage (AMD) in coal mining areas: A case study in southern Brazil. *Environmental Science and Pollution Research* 23: 18911–18927. <https://doi.org/10.1007/s11356-016-7077-3>.
- Garg, A., V. Tiwari, and S. Vishwanathan. 2017. Relevance of clean coal technology for India's energy security: A policy perspective. *IOP Conference Series: Earth and Environmental Science* 76: 012001. <https://doi.org/10.1088/1755-1315/76/1/012001>.
- Gatade, S. 2015. Silencing caste, sanitising oppression: Understanding Swachh Bharat Abhiyan. *Economic & Political Weekly* 50: 29–35.
- Gede Wenten, I., D. Ariono, M. Purwasasmita, and Khoirudin. 2017. Integrated processes for desalination and salt production: A mini-review. *AIP Conference Proceedings* 1818: 020065. <https://doi.org/10.1063/1.4976929>.
- Ghosh, R. 1993. Remote sensing for analysis of ground water availability in an area with long unplanned mining history. *Journal of the Indian Society of Remote Sensing* 21: 119–126. <https://doi.org/10.1007/BF02992108>.
- Gui-Hong, L., and H.P. Sjursen. 2013. Characteristics of produced water during coalbed methane (CBM) development and its feasibility as irrigation water in Jincheng, China. *Journal of Coal Science and Engineering (China)* 19: 369–374. <https://doi.org/10.1007/s12404-013-0317-y>.

- Haibin, L., and L. Zhenling. 2010. Recycling utilization patterns of coal mining waste in China. *Resources, Conservation and Recycling* 54: 1331–1340. <https://doi.org/10.1016/j.resconrec.2010.05.005>.
- Hossain, M.A., M.M. Rahman, M. Murrill, B. Das, B. Roy, S. Dey, and D. Chakraborti. 2013. Water consumption patterns and factors contributing to water consumption in arsenic affected population of rural West Bengal, India. *Science of the Total Environment* 463: 1217–1224. <https://doi.org/10.1016/j.scitotenv.2012.06.057>.
- Immerzeel, W.W., L.P.H. van Beek, and M.F.P. Bierkens. 2010. Climate change will affect the Asian water towers. *Science* 328: 1382–1385. <https://doi.org/10.1126/science.1183188>.
- IPCC. 2008. *Climate change and water*. Geneva: Intergovernmental Panel on Climate Change. <https://www.ipcc.ch/pdf/technical-papers/climate-change-water-en.pdf>.
- Jalan, J., and M. Ravallion. 2003. Does piped water reduce diarrhea for children in rural India? *Journal of Econometrics* 112: 153–173. [https://doi.org/10.1016/S0304-4076\(02\)00158-6](https://doi.org/10.1016/S0304-4076(02)00158-6).
- Ji, X., E. Curcio, S. Al Obaidani, G. Di Profio, E. Fontananova, and E. Drioli. 2010. Membrane distillation-crystallization of seawater reverse osmosis brines. *Separation and Purification Technology* 71: 76–82. <https://doi.org/10.1016/j.seppur.2009.11.004>.
- Kinnon, E.C.P., S.D. Golding, C.J. Boreham, K.A. Baublys, and J.S. Esterle. 2010. Stable isotope and water quality analysis of coal bed methane production waters and gases from the Bowen Basin, Australia. *International Journal of Coal Geology* 82: 219–231. <https://doi.org/10.1016/j.coal.2009.10.014>.
- Kumpel, E., and K.L. Nelson. 2013. Comparing microbial water quality in an intermittent and continuous piped water supply. *Water Research* 47: 5176–5188. <https://doi.org/10.1016/j.watres.2013.05.058>.
- Li, M. 2011. Reducing specific energy consumption in Reverse Osmosis (RO) water desalination: An analysis from first principles. *Desalination* 276: 128–135. <https://doi.org/10.1016/j.desal.2011.03.031>.
- Liu, L., M. Hejazi, P. Patel, P. Kyle, E. Davies, Y. Zhou, L. Clarke, and J. Edmonds. 2015. Water demands for electricity generation in the U.S.: Modeling different scenarios for the water–energy nexus. *Technological Forecasting and Social Change* 94: 318–334. <https://doi.org/10.1016/j.techfore.2014.11.004>.
- Liu, Y., M. Hejazi, P. Kyle, S.H. Kim, E. Davies, D.G. Miralles, A.J. Teuling, Y. He, and D. Niyogi. 2016. Global and regional evaluation of energy for water. *Environmental Science and Technology* 50: 9736–9745. <https://doi.org/10.1021/acs.est.6b01065>.
- Lu, Y., S. Song, R. Wang, Z. Liu, J. Meng, A.J. Sweetman, A. Jenkins, et al. 2015. Impacts of soil and water pollution on food safety and health risks in China. *Environment International* 77: 5–15. <https://doi.org/10.1016/j.envint.2014.12.010>.
- Macedonio, F., L. Katzir, N. Geisma, S. Simone, E. Drioli, and J. Gilron. 2011. Wind-Aided Intensified eVaporation (WAIV) and Membrane Crystallizer (MCR) integrated brackish water desalination process: Advantages and drawbacks. *Desalination* 273: 127–135. <https://doi.org/10.1016/j.desal.2010.12.002>.
- Mahato, M.K., G. Singh, P.K. Singh, A.K. Singh, and A.K. Tiwari. 2017. Assessment of mine water quality using heavy metal pollution index in a coal mining area of Damodar River Basin, India. *Bulletin of Environmental Contamination and Toxicology* 99: 54–61. <https://doi.org/10.1007/s00128-017-2097-3>.
- Mahato, M.K., P.K. Singh, A.K. Tiwari, and A.K. Singh. 2008. Risk assessment due to intake of metals in groundwater of East Bokaro Coalfield, Jharkhand, India. *Exposure and Health* 8: 265–275. <https://doi.org/10.1007/s12403-016-0201-2>.
- Mall, R.K., A. Gupta, R. Singh, R.S. Singh, and L.S. Rathore. 2006. Water resources and climate change: An Indian perspective. *Current Science* 90: 1610–1626.
- Mallants, D., J. Simunek, and S. Torkzaban. 2017. Determining water quality requirements of coal seam gas produced water for sustainable irrigation. *Agricultural Water Management* 189: 52–69. <https://doi.org/10.1016/j.agwat.2017.04.011>.
- Marten, R., D. McIntyre, C. Travassos, S. Shishkin, W. Longde, S. Reddy, and J. Vega. 2014. An assessment of progress towards universal health coverage in Brazil, Russia, India, China, and South Africa (BRICS). *Lancet* 384: 2164–2171. [https://doi.org/10.1016/S0140-6736\(14\)60075-1](https://doi.org/10.1016/S0140-6736(14)60075-1).
- Mazlan, N.M., D. Peshev, and A.G. Livingston. 2016. Energy consumption for desalination: A comparison of forward osmosis with reverse osmosis, and the potential for perfect membranes. *Desalination* 377: 138–151. <https://doi.org/10.1016/j.desal.2015.08.011>.
- Mekonnen, M.M., and A.Y. Hoekstra. 2016. Four billion people facing severe water scarcity. *Science Advances* 2: e1500323. <https://doi.org/10.1126/sciadv.1500323>.
- Ministry of Health of China. 2006. *Standards for drinking water quality*. National Standard of the People's Republic of China (GB 5749-2006).
- Mo, H., K.G. Tay, and H.Y. Ng. 2008. Fouling of reverse osmosis membrane by protein (BSA): Effects of pH, calcium, magnesium, ionic strength and temperature. *Journal of Membrane Science* 315: 28–35. <https://doi.org/10.1016/j.memsci.2008.02.002>.
- Moore, T.A. 2012. Coalbed methane: A review. *International Journal of Coal Geology* 101: 36–81. <https://doi.org/10.1016/j.coal.2012.05.011>.
- Mu, F., W. Zhong, X. Zhao, C. Che, Y. Chen, J. Zhu, and B. Wang. 2015. Strategies for the development of CBM gas industry in China. *Natural Gas Industry B* 2: 383–389.
- National Research Council. 2010. *Management and effects of coalbed methane produced water in the Western United States*. Washington, DC: The National Academies Press.
- NCMH. 2005. *Report of the national commission on macroeconomics and health*. New Delhi: Ministry of Health and Family Welfare, Government of India. Retrieved January 16, 2017, from <http://www.who.int/macrohealth/action/Report%20of%20the%20National%20Commission.pdf>.
- Neumann, P.J., J.T. Cohen, and M.C. Weinstein. 2014. Updating cost-effectiveness: The curious resilience of the \$50,000-per-QALY threshold. *The New England Journal of Medicine* 371: 796–797. <https://doi.org/10.1056/NEJMp1405158>.
- NHMRC, NRMCC. 2011. *Australian drinking water guidelines paper 6 national water quality management strategy*. Canberra: Commonwealth of Australia. Retrieved November 2, 2017, from https://www.clearwater.asn.au/user-data/resource-files/Aust_drinking_water_guidelines.pdf.
- NML. 2017. Coal mine water reclamation. *CSIR-NML*. Retrieved November 4, 2017, from http://www.nmlindia.org/download/Tech_HB/files/assets/downloads/page0011.pdf.
- O'Connor, C.L. 2016. *Decentralized water treatment in urban India, and the potential impacts of reverse osmosis water purifiers*. SM Thesis, Cambridge, MA: Massachusetts Institute of Technology. Retrieved February 4, 2018, from <http://hdl.handle.net/1721.1/106259>.
- Orem, W., C. Tatu, M. Varonka, H. Lerch, A. Bates, M. Engle, L. Crosby, and J. McIntosh. 2014. Organic substances in produced and formation water from unconventional natural gas extraction in coal and shale. *International Journal of Coal Geology* 126: 20–31. <https://doi.org/10.1016/j.coal.2014.01.003>.
- Perry, M., S. Wyllie, V. Prajapati, J. Menten, A. Raab, J. Feldmann, D. Chakraborti, et al. 2015. Arsenic, antimony, and Leishmania:

- Has arsenic contamination of drinking water in India led to treatment resistant kala-azar? *Lancet* 385: S80. [https://doi.org/10.1016/S0140-6736\(15\)60395-6](https://doi.org/10.1016/S0140-6736(15)60395-6).
- Pinto, P.X., S.R. Al-Abed, D.A. Balz, B.A. Butler, R.B. Landy, and S.J. Smith. 2016. Bench-scale and pilot-scale treatment technologies for the removal of total dissolved solids from coal mine water: A review. *Mine Water and the Environment* 35: 94–112. <https://doi.org/10.1007/s10230-015-0351-7>.
- Poudyal, S., V.D. Zheljajkov, C.L. Cantrell, and T. Kelleners. 2016. Coal-bed methane water effects on dill and its essential oils. *Journal of Environmental Quality* 45: 728–733.
- Qu, W., W. Zheng, S. Wang, and Y. Wang. 2012. China's new national standard for drinking water takes effect. *The Lancet* 380: e8. [https://doi.org/10.1016/S0140-6736\(12\)61884-4](https://doi.org/10.1016/S0140-6736(12)61884-4).
- Ramirez, P. 2005. *Assessment of contaminants associated with coal bed methane-produced water and its suitability for wetland creation or enhancement projects*. U.S. Fish and Wildlife Service. Retrieved August 22, 2018, from <https://www.fws.gov/mountain-prairie/contaminants/papers/documents/R6721C05.pdf>.
- Rawat, V.S. 2013. Water purifier market to touch Rs 7,000 cr by 2015. *Business Standard*. Retrieved October 14, 2017, from http://www.business-standard.com/article/companies/water-purifier-market-to-touch-rs-7-000-cr-by-2015-113041700369_1.html.
- Reese, H., P. Routray, B. Torondel, G. Sclar, M.G. Delea, S.S. Sinharoy, L. Zambrano, et al. 2017. Design and rationale of a matched cohort study to assess the effectiveness of a combined household-level piped water and sanitation intervention in rural Odisha, India. *BMJ Open* 7: e012719. <https://doi.org/10.1136/bmjopen-2016-012719>.
- Ren, D., L.M. Colosi, and J.A. Smith. 2013. Evaluating the sustainability of ceramic filters for point-of-use drinking water treatment. *Environmental Science and Technology* 47: 11206–11213. <https://doi.org/10.1021/es4026084>.
- Research and Markets. 2017. *Global reverse osmosis (RO) systems market—forecasts from 2017 to 2022*. Knowledge sourcing intelligence LLP. Retrieved October 14, 2017, from https://www.researchandmarkets.com/research/sgzn9k/global_reverse.
- Routray, P., B. Torondel, M.W. Jenkins, T. Clasen, and W. Schmidt. 2017. Processes and challenges of community mobilisation for latrine promotion under Nirmal Bharat Abhiyan in rural Odisha, India. *BMC Public Health* 17: 453. <https://doi.org/10.1186/s12889-017-4382-9>.
- Rustan, A., C. Cunningham, W. Fournay, K.R.Y. Simha, and A.T. Spathis. 2010. *Mining and rock construction technology desk reference*. Boca Raton: CRC Press.
- Salmachi, A., and C. Özgen Karacan. 2017. Cross-formational flow of water into coalbed methane reservoirs: Controls on relative permeability curve shape and production profile. *Environmental Earth Sciences* 76: 200. <https://doi.org/10.1007/s12665-017-6505-0>.
- Schnoor, J.L. 2011. Water–energy nexus. *Environmental Science and Technology* 45: 5065. <https://doi.org/10.1021/es2016632>.
- Semenza, J.C., L. Roberts, A. Henderson, J. Bogan, and C.H. Rubin. 1998. Water distribution system and diarrheal disease transmission: A case study in Uzbekistan. *The American Journal of Tropical Medicine and Hygiene* 59: 941–946. <https://doi.org/10.4269/ajtmh.1998.59.941>.
- Shaffer, D.L., L.H. Arias Chavez, M. Ben-Sasson, S.R. Castrillón, N.Y. Yip, and M. Elimelech. 2013. Desalination and reuse of high-salinity shale gas produced water: Drivers, technologies, and future directions. *Environmental Science and Technology* 47: 9569–9583. <https://doi.org/10.1021/es401966e>.
- Sharma, N., and S.S. Mahapatra. 2018. A preliminary analysis of increase in water use with carbon capture and storage for Indian coal-fired power plants. *Environmental Technology & Innovation* 9: 51–62. <https://doi.org/10.1016/j.eti.2017.10.002>.
- Shiao, T., and A. Maddocks. 2014. Finding solutions for South Africa's coal-fired water and energy problems. *The Guardian*. Retrieved October 14, 2017, from <https://www.theguardian.com/sustainable-business/2014/sep/18/south-africa-coal-water-energy-problems-mining>.
- Shrivastava, A., S. Rosenberg, and M. Peery. 2015. Energy efficiency breakdown of reverse osmosis and its implications on future innovation roadmap for desalination. *Desalination* 368: 181–192. <https://doi.org/10.1016/j.desal.2015.01.005>.
- Shukla, P.R., and S. Dhar. 2016. India's GHG emission reduction and sustainable development. In *Enabling Asia to stabilise the climate*, ed. S. Nishioka, 41–54. Singapore: Springer.
- Singh, A.K., and P.N. Hajra. 2018. *Coalbed methane in India: Opportunities, issues and challenges for recovery and utilization*. Cham: Springer. <https://doi.org/10.1007/978-3-319-66465-1>.
- Singh, A.K., G.C. Mondal, S. Singh, P.K. Singh, T.B. Singh, B.K. Tewary, and A. Sinha. 2007. Aquatic geochemistry of Dhanbad, Jharkhand: Source evaluation and quality assessment. *Journal of Geological Society of India* 69: 1088–1102.
- Singh, A.K., M.K. Mahato, B. Neogi, and K.K. Singh. 2010. Quality assessment of mine water in the Raniganj Coalfield Area, India. *Mine Water and the Environment* 29: 248–262. <https://doi.org/10.1007/s10230-010-0108-2>.
- Singh, G. 1994. Augmentation of underground pumped out water for potable purposes from coal mines of coal mines of Jharia coalfield. In *5th international mine water conference*, 679–689. Nottingham, UK: IMWA.
- Singh, U. 2017. *Design and development of experimental and simulation framework for emerging challenges in Indian natural gas industry*. BTech Thesis, Department of Mechanical Engineering, National Institute of Technology Rourkela, Rourkela, India.
- Sobsey, M.D., T. Handzel, and L. Venczel. 2003. Chlorination and safe storage of household drinking water in developing countries to reduce waterborne disease. *Water Science and Technology* 47: 221–228.
- Spencer, T., M. Colombier, O. Sartor, A. Garg, V. Tiwari, J. Burton, T. Caetano, F. Green, F. Teng, and J. Wiseman. 2018. The 1.5 °C target and coal sector transition: At the limits of societal feasibility. *Climate Policy* 18: 335–351. <https://doi.org/10.1080/14693062.2017.1386540>.
- Srinath Reddy, K., V. Patel, P. Jha, V.K. Paul, A.K. Shiva Kumar, and L. Dandona. 2011. Towards achievement of universal health care in India by 2020: A call to action. *Lancet* 377: 760–768. [https://doi.org/10.1016/S0140-6736\(10\)61960-5](https://doi.org/10.1016/S0140-6736(10)61960-5).
- Srinivasan, S., N. Kholod, V. Chaturvedi, P.P. Ghosh, R. Mathur, L. Clarke, M. Evans, et al. 2017. Water for electricity in India: A multi-model study of future challenges and linkages to climate change mitigation. *Applied Energy* 210: 673–684. <https://doi.org/10.1016/j.apenergy.2017.04.079>.
- Standing Committee on Petroleum and Natural Gas. 2016. *Production of Coalbed Methane—Fourteenth Report*. New Delhi: Lok Sabha Secretariat. http://164.100.47.193/isscommittee/Petroleum%20&%20Natural%20Gas/16_Petroleum_And_Natural_Gas_14.pdf.
- Sureau, S., B. Mazjin, S.R. Garrido, and W.M.J. Achten. 2018. Social life-cycle assessment frameworks: A review of criteria and indicators proposed to assess social and socioeconomic impacts. *The International Journal of Life Cycle Assessment* 23: 904–920. <https://doi.org/10.1007/s11367-017-1336-5>.
- Tchobanoglous, G., and F.L. Burton. 1991. *Wastewater engineering: Treatment, disposal and reuse*. Boston: Irwin McGraw-Hill.

- Thomas, T. 2016. CBM to contribute 5% of India's gas production by 2017: Pradhan. *Business Line*. Retrieved October 13, 2017, from <http://www.thehindubusinessline.com/economy/cbm-to-contribute-5-of-indias-gas-production-by-2017-pradhan/article8614589.ece>.
- Tiwary, R.K. 2001. Environmental impact of coal mining on water regime and its management. *Water, Air, and Soil pollution* 132: 185–199. <https://doi.org/10.1023/A:1012083519667>.
- U.S. Environmental Protection Agency. 1994. Washington, DC: Technical Document—Acid Mine Drainage Prediction. Retrieved May 1, 2018, from <https://www.epa.gov/sites/production/files/2015-09/documents/amd.pdf>.
- U.S. Environmental Protection Agency. 2010. Washington, DC: Coalbed Methane Extraction: Detailed Study Report. Retrieved August 14, 2018, from <https://nepis.epa.gov/Exe/ZyPURL.cgi?Dockey=P100MQJE.TXT>.
- UNEP. 2017. The emissions gap report 2017—A UN environment synthesis report. Nairobi. Retrieved November 2, 2017 from https://wedocs.unep.org/bitstream/handle/20.500.11822/22070/EGR_2017.pdf.
- USGS. 2000. Water produced with coal-bed methane. Retrieved April 14, 2018, from <https://pubs.usgs.gov/fs/fs-0156-00/fs-0156-00.pdf>.
- Wang, Z., X. Tian, and X. Wu. 2018. Hydrochemical characteristics and quality assessment of shallow groundwater and CBM co-produced water in the Shizhuangnan block, Qinshui Basin, China. *Environmental Earth Sciences* 77: 57. <https://doi.org/10.1007/s12665-017-7212-6>.
- Werber, J.R., A. Deshmukh, and M. Elimelech. 2017. Can batch or semi-batch processes save energy in reverse-osmosis desalination? *Desalination* 402: 109–122. <https://doi.org/10.1016/j.desal.2016.09.028>.
- Wheaton, J., and J. Metesh. 2002. *Potential ground-water drawdown and recovery from coalbed methane development in the Powder River Basin, Montana*. Butte: Montana Bureau of Mines and Geology. <http://citeseerx.ist.psu.edu/viewdoc/download?doi=10.1.1.521.5320&rep=rep1&type=pdf>.
- Zapata, C.B., C. Yang, S. Yeh, J. Ogden, and M.J. Kleeman. 2017. Low carbon energy generates public health savings in California. *Atmospheric Chemistry and Physics* 18: 4817–4830. <https://doi.org/10.5194/acp-2017-796>.
- Zimdars, C., A. Haas, and S. Pfister. 2018. Enhancing comprehensive measurement of social impacts in S-LCA by including environmental and economic aspects. *The International Journal of Life Cycle Assessment* 23: 133–146. <https://doi.org/10.1007/s11367-017-1305-z>.

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