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Chemical Contamination of Drinking Water in Resource-Constrained Settings: Global Prevalence and Piloted Mitigation Strategies

Susan E. Amrose,¹ Katya Cherukumilli,²
and Natasha C. Wright³

¹Department of Mechanical Engineering, Massachusetts Institute of Technology, Cambridge, Massachusetts 02139, USA; email: samrose@mit.edu

²Department of Civil and Environmental Engineering, University of Washington, Seattle, Washington 98185, USA

³Department of Mechanical Engineering, University of Minnesota, Minneapolis, Minnesota 55455, USA

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Keywords

chemical contamination, drinking water, heavy metals, pilot, prevalence, cost

Abstract

Chemical contamination of drinking water (including salinity) puts more than one billion people at risk of adverse health effects globally. Resource-constrained communities are the most affected and face unique challenges that require innovative safe water solutions. This review focuses on arsenic, fluoride, nitrates, lead, chromium, total dissolved solids, emerging organic contaminants, and, to a lesser extent, manganese, cadmium, selenium, and uranium. It covers contaminant prevalence, major health effects, and treatment technologies or avoidance strategies that have been proven effective in realistic water matrices and conditions. The review covers the leveled costs of water for pilot- and full-scale systems most relevant to resource-constrained communities, with a focus on component costs, including the costs of power systems, lifting water, waste management, and labor. These costs are not universally reported, but can be significant. The findings

are analyzed and discussed in the context of providing sustainable safe water solutions in resource-constrained settings.

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1. INTRODUCTION

Chemical contamination in the environment has a significant negative impact on human health (1). This impact is expected to rise substantially over the next few decades due to climate change (2), increased industrialization (3), and our increased capacity to quantify the health burden attributed to specific chemicals (1). Drinking water is an increasingly large source of chemical exposure, though the majority of drinking water treatment research focuses on microbial contamination due to its large disease burden and high associated childhood mortality (4). The communities most impacted by chemical contamination in drinking water are socioeconomically disadvantaged and marginalized communities (referred to here as resource constrained), whether they are in high-income countries (HICs) or low- and middle-income countries (LMICs). This has been attributed to a complex set of factors, including increased physical exposure, a reduced likelihood that water will be tested and treated, poorer health outcomes due to reduced health care access, and an increased underlying burden of malnutrition and disease (5, 6).

Water treatment solutions designed to serve resource-constrained communities must address numerous constraints, such as low and irregular incomes, low-quality education (resulting in increased rates of illiteracy and innumeracy), low managerial and technical capacity, weak physical infrastructure, limited access to financial services, and weak or absent formal regulatory institutions (7). Water treatment to address chemical contamination is particularly challenging under these constraints. Adverse health effects caused by low levels of chemical exposure typically found in drinking water tend to be chronic, imperceptible, and uncertain, limiting physical and social cues that would otherwise act as an incentive for behavior change (8). Awareness and risk perception may be especially low in resource-constrained areas because of inadequate testing, language barriers, low-quality education, and poor information dissemination by government agencies (5, 9, 10). Once motivated to act, resource- and capacity-limited communities are often faced with

Resource constrained: refers to communities which are socioeconomically disadvantaged or marginalized

High-income countries (HICs): countries with gross national incomes of more than US\$12,055 per capita (as defined by the World Bank)

Low- and middle-income countries (LMICs): countries with gross national incomes of less than US\$12,055 per capita (as defined by the World Bank)

chemical treatment technologies that are expensive and highly operationally complex (5, 8). These barriers may then be compounded by the need to safely manage and dispose of the steady stream of contaminant-enriched waste that results from most chemical water treatment processes (11, 12). Many locations are faced with multiple contaminants in a single water source (13), which further increases operational complexity and costs of treatment.

This article reviews the recent literature on safe water treatment or approaches that have been demonstrated to address some of these challenges in resource-constrained settings. As affordability is a primary constraint in resource-constrained settings, we limit our review to safe water approaches that have demonstrated efficacious operation and have been used to estimate water provision cost in real or realistic model waters at the pilot scale. Bench-scale studies are less reliable as a basis for establishing the cost of achieving a given level of performance in real source waters and environmental conditions (14). Pilot studies also uniquely probe the suitability of a safe water approach to some of the many nontechnical constraints imposed by highly constrained environments.

We review recently reported levelized costs of water (LCOWs) for all relevant approaches. We find that the cost literature is not uniform, and often not transparent, in its reporting of pilot-scale LCOWs. In most cases, the total water cost reported by different research groups considers different components, which are not universally defined or always fully described. We therefore review key significant contributions to cost arising from components that are not universally considered, including the power system; lifting water from a well; waste management; and other costs, such as operator labor and equipment installation. We also find that, throughout the cost literature, the terms technologies, interventions, systems, and options are often used interchangeably, which can make it difficult to understand exactly what is being evaluated, compared, or priced. For this review, we follow Amrose et al. (15) and consider a safe water system as encompassing all components “from source to sip,” composed of a series of stages including treatment technologies, protection technologies, delivery models, and last-mile labor before consumption. Costs are frequently reported for single stages or different combinations of stages, which we refer to as safe water approaches to denote that their boundaries start and end at different points on the source-to-sip chain. Comparing costs between approaches rather than systems has the potential to significantly distort results, particularly when the costs of unreported stages are substantial. These differences are noted and discussed where appropriate.

The existence of efficacious safe water approaches with low LCOWs is often not enough to sustainably deliver clean water in resource-constrained settings (16). A growing literature emphasizes that innovative delivery models are also needed, both to support long-term viability and to facilitate short-term behavior change (17, 18). A strong enabling environment of policies, legislation, regulatory institutions, and social norms to facilitate the success of the intervention is also emphasized (8, 17). We refer the reader to recent reviews of delivery models and enabling environments relevant to LMICs (17–19) and small water systems in industrialized countries (5). A full discussion of the integration of delivery models and safe water approaches to form viable safe water systems is beyond the scope of this review, which largely reflects the scarcity of this discussion in the literature. However, this integration will be critical to turn the safe water approaches reviewed here into sustainable and viable safe water solutions capable of effectively addressing chemical contamination in resource-constrained settings.

A wide variety of chemical contaminants exist in drinking water. This review focuses on arsenic (As), chromium (Cr), fluoride (F), lead (Pb), nitrate (NO₃⁻), total dissolved solids (TDS), emerging organic contaminants (EOCs), and, to a lesser extent, cadmium (Cd), manganese (Mn), selenium (Se), and uranium (U). This selection of contaminants is based on the significance of their impact on marginalized populations, or their potential for future impact. This article contextualizes

Safe water approach: a subset of stages from source to sip, generally including a water treatment process or contaminant avoidance approach

Levelized cost of water (LCOW): the cost per unit volume of product water produced by a water treatment process or service

Safe water system: a complete series of stages from source to sip, including treatment technologies, protection technologies, delivery models, and last-mile labor before consumption

Emerging organic contaminants (EOCs): a diverse array of manufactured chemicals that are a rising concern for freshwater quality and for human and aquatic health

this impact by compiling a high-level review of known human health effects from drinking water exposure, combined with recent estimates of human populations at risk of exposure to chemical contamination in drinking water globally. EOCs have recently caused global concern due to their ubiquitous presence in a variety of aqueous environments; they include personal care products (PCPs) (e.g., insect repellants, preservatives, sunscreen additives), pharmaceuticals (e.g., antibiotics, hormones, anti-inflammatory drugs, antiepileptic drugs), pesticides, perfluorinated compounds, flame retardants, gasoline additives, and anticorrosives (20). Little is known about the health effects of chronic exposure to low levels of EOCs, particularly in complex mixtures (21). However, EOCs are included because health effects cannot be ruled out (22), and because EOCs are a particular concern in LMICs, where more than 70% of industrial waste and most wastewater is dumped untreated into surface waterways (23, 24).

Several recent reviews of chemical contamination in drinking water have focused on LMICs, looking at heavy metals [occurrence (25) and removal (26)] and the geogenic contaminants As and F (17). Other reviews have focused globally or on HICs and on the occurrence and/or removal of single contaminants or contaminant groups [e.g., heavy metals (27), As (14, 28, 29), Cd (30, 31), Cr (32), F (33, 34), Pb (35), Mn (36), NO_3^- (37), Se (38), U (39), TDS (40–42), and EOCs (20)]. This review complements and builds on this research by (a) including a wide variety of contaminants, which can frequently co-occur, in a single review of prevalence, health, technologies, and cost; (b) focusing a review of safe water approaches on features most relevant to applications in resource-constrained settings, for some contaminants for the first time; (c) considering factors relevant to resource-constrained areas in HICs in addition to LMICs; (d) providing a comprehensive comparison of LCOWs across contaminants and safe water approaches; and (e) examining the separate contributions to the cost of key factors, such as the power system, lifting water, waste management, and other costs.

2. HEALTH IMPACTS

The recent literature on human health impacts due to chronic exposure at levels commonly found in contaminated drinking water has been comprehensively reviewed for each of the contaminants As (43), Cd (31), Cr (32), F (34), Pb (44, 45), Mn (36), NO_3^- (46), Se (47), U (48), TDS (49), and EOCs (50). Common drinking water sources and primary health impacts due to drinking water exposure are summarized in **Table 1**.

Health impacts vary widely between contaminants. Chronic As exposure is known to be associated with various forms of cancer (e.g., skin, lung, liver, bladder), along with skin lesions and skin thickening (e.g., hyperkeratosis), liver diseases (hepatotoxic effects), and diabetes mellitus (43). Cardiovascular, renal, reproductive, and neurological effects due to As intake have also been reported, but further study is needed to confirm these results (43). Oral Cd exposure targets primarily the kidneys and bones and can cause renal dysfunction, reduced bone density, fractures, and osteoporosis (31). Exposure to Cr [specifically Cr(VI)] has been linked to an increased risk of lung cancer in humans, and animal studies suggest links to other cancers (32). Chronic ingestion of excessive F leads primarily to mottled tooth enamel and severe bone deformities (e.g., dental and skeletal fluorosis) (34). Pb exposure is associated predominantly with cognitive, behavioral, and developmental impacts, which are particularly harmful to young children (44, 45). Pb exposure can also lead to hypertension, as well as cardiovascular, respiratory, renal, digestive, and reproductive effects (44, 45). Exposure to excessive Mn has been linked to cognitive impairment, motor disturbances, and speech difficulties, as well as neurological effects in children (36). Ingestion of NO_3^- is most commonly associated with methemoglobinemia, with increasing evidence of correlations with colorectal cancer, thyroid disease, and neural tube defects (i.e., birth defects of the

Table 1 Major human health impacts, primary sources, and MCLs and guideline values for various agencies of select chemical contaminants^a

Contaminant	Sources in drinking water ^b	Primary effects on human health	MCLs ^c		
			WHO (39)	EPA (40)	BIS (41)
Arsenic	Geogenic (mining, smelting)	Cancer, skin effects, liver disease, diabetes	10 µg/L	10 µg/L	10 µg/L
Cadmium	Geogenic, plumbing, mining, industry, landfills, sewage sludge, fertilizers, urban areas	Kidney dysfunction, bone toxicity	3 µg/L	NA	3 µg/L
Chromium	Geogenic, electroplating/textile industries, wood treatment, corrosion control	Cancer	50 µg/L	100 µg/L	50 µg/L
Fluoride	Geogenic (fertilizers, aluminum refineries)	Mottled tooth enamel, bone deformities (fluorosis)	1.5 mg/L	4 mg/L	1 mg/L
Lead	Corroding pipes, faucets, and plumbing fixtures (geogenic)	Cognitive and developmental impairment, hypertension	10 µg/L	15 µg/L	10 µg/L
Manganese	Geogenic, industry, mining, landfills	Cognitive, motor, and speech impairment	400 µg/L ^d	50 µg/L	100 µg/L
Nitrate	Fertilizers, manure (septic tanks, wastewater)	Methemoglobinemia, cancer, thyroid disease, birth defects	50 mg/L	10 mg/L	45 mg/L
Selenium	Geogenic, coal mining, combustion	Parkinson's disease, ALS, cancer	40 µg/L	50 µg/L	10 µg/L
Uranium	Geogenic, mill tailings, nuclear activities, fossil fuel combustion, fertilizers	Kidney inflammation, cancer	30 µg/L	30 µg/L	NA
TDS	Geogenic, seawater intrusion (drainage from irrigated fields)	Hypertension, (pre)eclampsia	NA	500 mg/L	500 mg/L
EOCs	Pesticides, inadequate wastewater treatment, hospital discharge	Unknown; no consensus	Varying values for specific organic parameters		

^aData compiled from References 17, 30, 33, 48, and 51–57.

^bSources in parentheses are much less prevalent. If no sources appear in parentheses, no relative prevalence between sources is implied.

^cThe reported values are MCLs unless otherwise noted (see footnote d) for guideline values.

^dThis WHO guideline value for manganese is based on secondary aesthetic thresholds (taste, staining, odor) and not on health.

Abbreviations: ALS, amyotrophic lateral sclerosis; BIS, Bureau of Indian Standards; EOC, emerging organic contaminant; EPA, US Environmental Protection Agency; MCL, maximum recommended contaminant level; NA, not applicable; TDS, total dissolved solids; WHO, World Health Organization.

brain, spine, or spinal cord) (46). Although Se is an essential nutrient in trace quantities, excessive exposure has been found to increase the risk of Parkinson's disease, amyotrophic lateral sclerosis, and certain types of cancer (e.g., melanoma, lymphoid neoplasms, kidney, colorectal) (53). U exposure has been associated with renal dysfunction as well as certain cancers (e.g., bone, kidney) (48). High levels of TDS can increase the risk of hypertension, which is particularly concerning for pregnant women because it increases their risk of coronary heart disease, (pre)eclampsia, and other pregnancy-related complications (49). Moderate levels of TDS can also affect the taste of water, which can indirectly lead to adverse health effects by limiting water quantity or causing a community to switch to an alternative water source that is of lower microbial quality (58).

Maximum recommended contaminant level (MCL): a threshold limit on the amount of contaminant allowed in a water source, often based on health considerations

Guideline value: a threshold limit on the amount of contaminant allowed in a water source that is based on aesthetic standards or nonhealth considerations

The health impacts of EOCs in drinking water are still largely unknown (50), though some effects have been established. Endocrine disruptors, some fire retardants, several widely used industrial chemicals (such as bisphenol A), and some pesticides are known to interfere with endogenous hormone systems (59). Perfluorinated compounds and endocrine-disrupting pesticides have been linked to an increased incidence of cancers, reproductive issues, and growth retardation (50, 60). Most pharmaceuticals appear unlikely to cause health impacts at the low levels found in drinking water supplies, but some cytotoxic drugs are known to be carcinogenic (60). Additional research is needed to fully understand the overall health impacts from chronic exposure to the low average concentrations of EOCs commonly found in drinking water (20, 50). Key challenges to determining these specific health outcomes include the extremely large and diverse array of new chemicals entering the environment, the formation of metabolites (often at larger concentrations than their parent compounds), and the frequent co-occurrence of multiple contaminants (20, 50).

The severity of the health outcomes listed above tends to increase with the dose and duration of exposure (31, 32, 36, 47). Health outcomes can also be affected by age, gender, nutritional status, genetic factors, lifestyle, and the presence of other diseases (43). Speciation of a contaminant can affect its toxicity as well. For example, hexavalent selenate [Se(VI)] is more toxic than tetravalent selenite [Se(IV)] (38). Hexavalent chromium [Cr(VI)] is toxic, whereas trivalent chromium [Cr(III)] is mostly benign (32). Similarly, trivalent arsenite [As(III)] is commonly reported as being more toxic than pentavalent arsenate [As(V)], though evidence remains uncertain (29). As many of these factors vary by person or are difficult to measure (e.g., speciation), assumptions are used to define health risks in terms of a single variable that is correlated to the overall level of exposure—the total contaminant concentration. Regulatory agencies use this metric to define a maximum recommended contaminant level (MCL) based on health considerations, or a guideline value based on aesthetic considerations. MCLs and guideline values for several agencies are listed in **Table 1**.

The cost of remediating contaminated drinking water tends to increase substantially when very low product water concentrations are required. For resource-constrained populations, it is often not clear whether priority should be placed on reducing health risks by significantly reducing contaminant levels, or on increasing affordability. Affordable water can translate into improved health care access and nutrition, which might reduce susceptibility to the poor health outcomes caused by higher levels of water contamination. Exacerbating this dilemma, the evidence for adverse health effects at lower levels of contaminant exposure tends to be weaker and more contested than the evidence at higher levels of exposure [e.g., for As (61)]. Different agencies approach this debate with different perspectives and priorities, under different political and lobbying pressures and lobbying efforts, resulting in MCLs and guideline values that vary widely (**Table 1**). The definition of chemically safe drinking water is therefore highly location specific, and may reflect the current cost of treatment, political capital of resource-constrained communities, propensity of the regulatory body to favor affordability over health risk, and strength of evidence for health effects, among other factors. The LCOW for a safe water approach is tied to this location-specific definition because it is used to specify the required purity of the product water for the LCOW calculation. It is therefore important to understand this definition, and the political considerations behind it (including how it might change in the near future), to fully understand the LCOW in the context of providing a sustainable safe water solution for a specific community.

3. GLOBAL CHEMICAL CONTAMINATION

Figure 1 presents the geographical distribution of global populations at risk of exposure to chemical contaminants in drinking water, as reported in the literature. We present only population

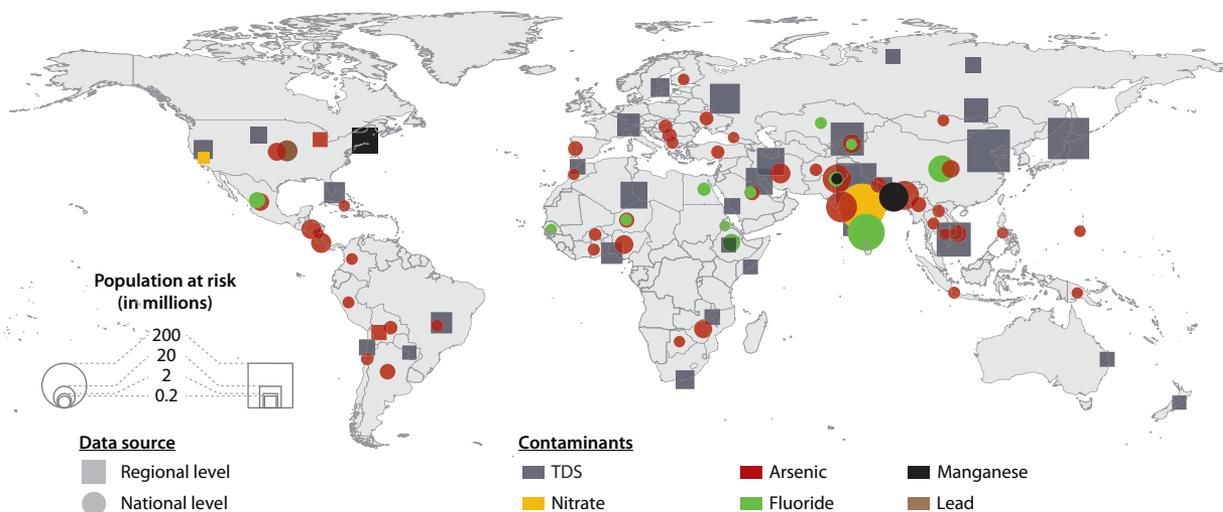


Figure 1

Geographical distribution of global populations at risk of exposure to chemical contamination in drinking water, as reported in the literature (54, 55, 62–80). We present only population estimates that (a) consider a contamination level threshold greater than or equal to the World Health Organization maximum recommended contaminant limit (WHO MCL) and guideline values presented in **Table 1** and (b) for which the estimated population at risk of exposure exceeds 50,000. For total dissolved solids (TDS), which have no health-based WHO MCL or guideline value, the US Environmental Protection Agency guideline is used. Further considerations are discussed in the **Supplemental Material**, which includes a table of exposed population estimates by country and contaminant.

estimates (a) that consider a contamination level threshold greater than or equal to the World Health Organization (WHO) MCL and guideline values of **Table 1** and (b) for which the estimated population at risk of exposure exceeds 50,000. For TDS (a contaminant category without a WHO MCL or guideline value), the US Environmental Protection Agency (EPA) guideline is used. Further considerations are discussed in the **Supplemental Material**, along with a table of exposed population estimates by country and by contaminant.

It is clear from **Figure 1** that India and Bangladesh bear a large burden of chemical contamination in drinking water, with significant populations at risk of exposure to As, F, nitrates, Mn, and TDS. Notably, India and Bangladesh have the two largest populations exposed to geogenic As in groundwater (an estimated 115.4 million people combined are at risk), resulting in an estimated 24,000 adult deaths annually in Bangladesh alone (63).

Globally, most of the burden of chemical contamination falls on LMICs, with some significant contamination in the United States. Within LMICs, many of the people exposed (particularly in sub-Saharan Africa and Asia) live in rural areas, with little water infrastructure and limited financial resources (8, 81, 82). Within the United States, Pb, NO₃⁻, and As affect both urban and rural populations, with the impact felt disproportionately by persons who are socioeconomically disadvantaged and marginalized (5, 10, 83).

As a rough indicator of global prevalence for each contaminant with available data, the most recent or well-established population estimates for each country can be summed, producing the following estimates of global population at risk: As (332 million), F (183 million), NO₃⁻ (226 million), Pb (40 million), Mn (101 million), and TDS (1,182 million). Many contaminants co-occur; thus, these values themselves cannot readily be summed to offer a lower bound on the global population affected by chemical contamination in drinking water. However, the largest value (for TDS) suggests that well over one billion people globally are at risk of exposure to potential negative impacts due to chemical contamination.

Supplemental Material >

Efficacious pilot study: a study conducted in real environmental conditions for which the final product water contaminant concentration meets a specified standard

While these summations act as a rough guide of exposed populations, they have several important limitations. Some values, notably those for TDS, may overestimate the exposed population because they incorporate population estimates that reflect geographic regions of known contamination weighted by population density factors, as opposed to estimates considering community-level water extraction and usage data (54). Other values are likely significant underestimates, as **Figure 1** reflects only published studies of populations at risk that meet our criteria. For example, one excluded study suggested that the European Union may have lead pipes in as many as 25% of homes, putting up to 120 million people at risk of Pb exposure (84). This estimate is not included in **Figure 1** because no threshold level for exposure was set, making it difficult to interpret; however, it suggests that the population at risk of exposure to Pb in drinking water may be far greater than 40 million. In addition, there are several cases for which significant localized drinking water contamination has been reported, but a preliminary literature review revealed few to no estimates of populations at risk. For example, high levels of Pb were reported in drinking water in Karachi, Pakistan (85), and there are reports of high Mn in groundwater in Cambodia and China (79); high Se in groundwater in the United States, Jordan, and India (38); and high U in groundwater throughout the United States, Asia, Africa, Brazil, Australia, and parts of Europe (48). More than 100 pharmaceutical substances have been recorded in water sources across the United States and Europe, and more than 30 have been found in Latin America and Asia (56). There have also been numerous global occurrences of pesticides and their metabolites (59). These combined reports suggest chemical contamination of drinking water is much more prevalent than reported in **Figure 1**. However, it is difficult to speculate about the size of the populations at risk without careful study.

These estimates represent only the population potentially exposed to the various contaminants in drinking water above their respective threshold value. A further estimate of the adverse health impacts due to these exposures would require information about the specific contaminant concentrations, exposure duration, water intakes, population nutritional and health status, and dose–response relationships for various health effects. This is beyond the scope of this review.

4. TREATING OR AVOIDING CHEMICAL CONTAMINATION

In this section, we review safe water approaches to either (*a*) treat water to reduce the concentration of chemical contamination below a prescribed health standard or (*b*) avoid chemical contamination by developing an alternative water source that is free from (or has reduced levels of) the chemical contaminant. Detailed descriptions of the treatment or avoidance processes underlying each approach are widely available elsewhere (e.g., 17, 27), along with general advantages and disadvantages of each approach. Here, we focus on the key attributes of the approach in the context of its application in or relevance to resource-constrained communities, along with contaminants for which each approach has demonstrated efficacy at the pilot scale in real waters or realistic water matrices.

For the purpose of this review, we consider only approaches that have been demonstrated to be efficacious according to the WHO MCL and guideline values in **Table 1** (i.e., the final product concentration must fall below the WHO values in **Table 1**). For TDS, the EPA guideline is used. EOC studies for contaminants without corresponding WHO MCLs are considered efficacious if they meet a potable drinking water standard or health risk target described within the reported study.

4.1. Treatment Approaches

Water treatment approaches that have been demonstrated at pilot or full scale and are suitable for resource-constrained settings have been based on the following treatment processes: membrane

processes, adsorption, ion exchange, precipitation/coagulation, electrocoagulation (EC), advanced oxidation/reduction processes (AOPs/ARPs), biological processes, and solar stills. Several other treatment approaches have been proven in pilot- or full-scale studies, but are based on processes that are more appropriate for larger-scale systems with relatively high operational capacity and resources. These include multistage flash (MSF) distillation, multiple-effects evaporation (MEE), and mechanical vapor compression (MVC) thermal-based desalination processes. These approaches are not reviewed here.

Some treatment approaches may include a pretreatment step to prefilter the source water or to oxidize or reduce a contaminant to a state that is more amenable for removal and/or less toxic [e.g., As(III) to As(V), or Cr(VI) to Cr(III)]. Common oxidants include air, chlorine, and Mn compounds, and a common reductant is ferrous sulfate (17). Pretreatment steps must be considered (in terms of cost, logistical complexity, performance enhancement, waste generation, etc.) when comparing safe water approaches.

4.1.1. Membrane processes. Membrane processes separate a contaminant from water on the basis of its size or charge, and include ultrafiltration (UF), nanofiltration (NF), reverse osmosis (RO), and electrodialysis (ED). UF is frequently used to separate biological contaminants, whereas chemical contaminants tend to require the smaller pore size of RO or NF. RO is a membrane process in which a pressure greater than the osmotic pressure of the feed stream is used to move water through a semipermeable membrane. ED uses an electric potential across a series of alternating anion and cation exchange membranes to draw charged contaminants into alternating channels of diluate (product) and concentrate (brine) streams. Membrane processes are highly effective at removing a wide range of contaminants over a wide pH range, making them particularly versatile and suitable for cocontamination (17). Efficacious community-scale pilot installations of RO and NF systems have recently been demonstrated to remove a variety of contaminants in many LMICs as well as rural areas of the United States and Europe [e.g., TDS (86–93), As (94), F (95, 96), NO_3^- (97), and Cr (98)]. The literature reports fewer efficacious community-scale ED pilots in LMICs than NF/RO pilots [e.g., TDS (99), F (95), and NO_3^- (100)]. In Israel, Oren et al. (101) recently utilized a community-scale hybrid RO/ED plant to increase water recovery and reduce brine volume while removing TDS.

Pilots of membrane systems targeting marginalized or low-income communities have revealed the following. First, there is a trade-off between capital cost and operational cost that must be considered when different system designs are compared (102). For example, for a single RO membrane element (defined by desired salt rejection and permeability), the capital cost of an RO plant can be reduced at the expense of increasing operational costs by decreasing the membrane area and increasing the applied pressure, achieving the same product water output. Similar trade-offs also exist for other membrane processes. Thus, common performance metrics reported in the literature, such as contaminant rejection (in percent), specific energy consumption (in kilowatt hours per cubic meter), LCOW (in dollars per cubic meter), total membrane area (in square meters) and recovery (as the ratio of product water to feed water), depend on operational decisions and trade-offs, and must be evaluated holistically when comparing designs. Second, small pumps are less efficient than large pumps, and this factor must be considered before assuming that specific energy performance metrics from a large plant will translate to a small community or household level (102). Third, a cost-optimized system design requires a complete understanding of the water source (89); the maximum recovery of a membrane-based system may be limited primarily by membrane fouling agents such as sparingly soluble salts (e.g., CaSO_4 and CaCO_3), which can foul membranes at high recovery rates. Obtaining water quality analyses in individual communities may be difficult for widespread desalination programs but should be considered in the

Pretreatment:

a step included prior to treatment either to prefilter the source water or to oxidize/reduce a contaminant to a state that is more amenable for removal and/or less toxic

Regeneration:

the process of using thermal, electrochemical, ultrasonic, or chemical methods to partially restore some adsorbent capacity of an adsorbent media

delivery model and cost analysis if they are required to achieve target performance. Fourth, intensive hands-on training may be necessary to ensure proper procurement and safe handling of the chemicals required to prevent membrane fouling and ensure increased water recovery (89, 99). Membrane systems also require maintenance to change prefilters and backflush membranes, all of which may contribute to the complexity and cost of the approach.

4.1.2. Adsorption and ion exchange. Adsorption and ion exchange are processes that transfer ions from a liquid phase to the surface of a solid phase (i.e., an adsorbent medium or ion exchange resin) (17). Commercially available adsorbents include a wide variety of materials with varying performances, material costs, and local availability; these can include naturally occurring biosorbents (e.g., chitosan, eggshell powder), clays (e.g., zeolites, bentonite, kaolinite, montmorillonite), inorganic oxides and hydroxides containing metal centers (e.g., iron, aluminum, silicon, titanium), industrially refined materials (e.g., activated carbon, biochar, activated alumina, ion exchange resins), and metallic zero-valent iron (17, 26). Adsorption processes are widely used in water treatment because of their convenience, ease of operation, simplicity of design, relatively low costs, and applicability at both small and large scales. Synthetic ion exchange resins are manufactured to specifically target cationic or anionic contaminants by reversibly exchanging with similarly charged loosely bound ions present on the media (e.g., Na^+ , K^+ , OH^-) (103). Ion exchange processes can be highly reliable and effective and have the ability to target specific contaminants. Like adsorbents, they can be used on multiple scales.

Adsorption and ion exchange have been demonstrated in resource-constrained settings within household filters, community filters, and small water systems to remove numerous contaminants [e.g., As (14, 104–110), Cd (111), Cr (112–114), F (115), Pb (111, 116), and EOCs (117)]. Subsurface-flow constructed systems (e.g., wetlands and wells injected with reactive media), reliant on in situ processes to adsorb or degrade contaminants, have been demonstrated at the pilot scale to successfully remove As, NO_3^- , U, and some EOCs [e.g., BTEX chemicals (benzene, toluene, ethylbenzene, and xylene), phenol] (118, 119).

While adsorption and ion exchange approaches are commonly used in resource-constrained settings, there are several challenges to their sustainable use in these environments. Over time, the saturated media or resin must often be either discarded or regenerated using thermal heating or chemical washing with acids or bases (14, 17, 114), which can add to the cost and logistical complexity and may require special training. For some community-scale systems, the pH of the source or product water must be controlled to achieve peak performance (17, 26), requiring a chemical supply and a relatively high level of operator skill. Although many adsorbent materials are proposed as low-cost alternatives for resource-constrained settings (e.g., 26), few papers consider the cost of acquiring the precursor materials for processing (120) or the cost of maintaining a supply chain of the processed material such that it is readily accessible in remote rural areas, both of which can be significant. Safe water systems that have demonstrated sustainable operation over time (≥ 2 years) have used a combination of technology and business model innovation to address these challenges [e.g., for As mitigation (110)].

4.1.3. Precipitation/coagulation. Precipitation/coagulation processes rely on the addition of salts and coagulants to regulate solution pH and enhance the formation of flocs, which are later removed through sedimentation or filtration. Precipitation processes reduce contaminant concentrations through the precipitation of solid minerals containing the contaminant (e.g., calcium arsenate), whereas coprecipitation processes generate solid particles that act as an adsorbent for the contaminant [e.g., aluminum or iron (hydr)oxides precipitating and then acting as an arsenic adsorbent] (17).

Precipitation/coagulation approaches have been used efficaciously in resource-constrained settings in India, Argentina, Ethiopia, and the United States in recent pilot- or full-scale installations to remove As (68, 121), F (17), and Cr (114). The Nalgonda technique is an established precipitation/coagulation approach developed specifically for resource-constrained settings that targets F through the addition of lime and alum (34). While this technique has been widely promoted in India, it has not been scaled up significantly because of challenges that are common to many precipitation/coagulation approaches, such as operational complexity, residual aluminum or salts left in solution, and a tendency to produce large quantities of waste sludge (17, 34). Air oxidation followed by in situ coprecipitation with naturally occurring iron addresses some of these challenges by eliminating external chemical addition, and is commonly used in Vietnam to remove As (17). However, to be effective, the contaminated aquifer must contain a high enough concentration of naturally occurring iron to coprecipitate with the contaminant, limiting its applicability.

Passivation:
the buildup of surface layers on an electrode surface over time that reduces contaminant removal efficiency of EC processes

4.1.4. Electrocoagulation. EC is a coprecipitation process that generates in situ adsorbents through the application of an external voltage to metal sheets (typically aluminum or iron) in contaminated water (122, 123). The sacrificial anode material is electrochemically corroded, which oxidizes the zero-valent source metal, generating precipitates with a strong affinity for adsorbing numerous aqueous contaminants. Typically, coagulants such as alum or polyaluminum chloride are then added to help remove the solids. Pilot installations of iron- and aluminum-based EC reactors of varying scales have been built across India, China, and Bangladesh to remediate As (124–127) and F (128, 129).

EC is advantageous because it is highly effective in diverse feed waters, produces little waste sludge compared with other removal methods, and does not typically require the sourcing and procurement of hazardous chemicals or specialized adsorbent media (122, 123). These advantages can be especially beneficial in remote resource-constrained settings with a lower capacity to handle hazardous materials, higher last-mile distribution costs for specialized media, and higher transport costs for waste disposal. Key challenges facing EC systems include passivation of the metal cathode, the need to replace the sacrificial anode, and the need for electricity (122, 123). An electrochemical As remediation plant in India, based on EC processes, has used a viable business model to sell As-safe water at a locally affordable price in a remote village for several years (127). This demonstrates that an EC-based process can consistently meet water demand in a region with intermittent electricity, and that its high contaminant removal efficiency can offset the added electricity cost.

4.1.5. Advanced oxidation/reduction processes. AOPs/ARPs refer to processes that generate strong oxidative or reductive species in water, which are then used to degrade a wide variety of organic and inorganic contaminants into carbon dioxide, water, and either less toxic species or species that are easier to separate from the system in a subsequent treatment step (130). The oxidants in AOPs are typically hydroxyl radicals (sometimes sulfate or chlorine radicals), generated through ozone (O_3), UV, electrochemical, catalytic, physical (e.g., sonochemical), and hybrid processes. A wide variety of catalysts have been tested, including metals, clays, carbon materials, composite materials, and semiconductors (e.g., TiO_2 , ZnO , WO_3 , Cu_2O). These differ with respect to raw material cost, market availability, catalytic effectiveness, and ability to accelerate reaction time. The most common combinations of processes that are established at full scale involve O_3 , O_3 and hydrogen peroxide (O_3/H_2O_2 , called the peroxone process), UV, UV and chloride (UV/ Cl_2), and Fenton's reagent processes with iron species [Fe(II) or Fe(III)] using either UV or solar radiation (130). A major advantage of these approaches is the lack of a contaminant-rich waste that must be disposed of at the end of the process.

Capital costs: costs related to acquiring, upgrading, and maintaining physical infrastructure for the safe water approach

Multiple pilot- and full-scale demonstrations have been reported for AOPs to remove pharmaceuticals and PCPs (e.g., 131, 132) and pesticides (e.g., 133), as well as ARPs to remove Cr (134). Most demonstrations occurred in the United States, Spain, and China, and targeted large-scale water treatment plants ($\geq 30 \text{ m}^3/\text{day}$). Key barriers to implementing AOPs/ARPs in resource-constrained settings include high costs, operational complexity, potentially harmful oxidation by-products, and local availability of catalysts and other chemicals. Costs are driven by the material costs of chemicals and catalysts as well as the electricity costs, which are determined by the energy efficiency. In a recent comprehensive review of the AOP literature, Miklos et al. (130) found that the energy efficiency of an AOP is influenced by water matrix, process capacity, and various energy-independent process parameters, such as oxidant or catalyst dose. This has resulted in a wide variation in reported energy efficiencies of similar AOPs. Despite this variation, the energy required to mitigate contamination tends to be higher for ultrasound, UV/catalyst, and electrochemical processes than for photo-Fenton, UV-based, and O_3 -based processes, translating into relatively higher costs for these processes when scaled up. Given the ubiquity of EOC contamination, there is a need to explore the economic viability and applicability of these approaches for remote and resource-constrained settings.

4.1.6. Biological processes. Biosorption processes use microbes to promote accumulation of a contaminant in a biomass, which is then disposed of. Biotransformation processes use microbes to transform contaminants into a more desirable, usually less toxic state [e.g., reduction of Cr(VI) to less toxic, less mobile Cr(III)]. Biological processes are advantageous because they tend to have lower operating costs and limited chemical inputs, handle multiple contaminants, and produce minimal waste sludge (135). However, they tend to have relatively high capital costs and can be sensitive to environmental conditions, resulting in increased operational complexity (135).

The use of biofilters to remove pharmaceuticals and PCPs in drinking water is well established in Europe and is beginning to emerge in the United States (136), with recent pilot-scale demonstrations to remove Mn in China and the United States (137, 138). Slow sand filters (SSFs) are a type of biofilter colonized by microbial populations that attach to sand grains and grow on the steady influx of nutrients (e.g., ammonia, methane) in the source water (136). They have been used to remove pesticides, pharmaceuticals, and other organic contaminants in the United Kingdom and United States (24, 139). SSFs have also been used to remove moderate levels of As in Vietnam (140), though this removal is attributed primarily to coprecipitation with the naturally occurring iron in the feed water (140). Biological reduction processes have been used at the pilot scale to efficaciously remove Cr in South Africa (141) and nitrates at numerous locations in the United States and Europe (summarized in 142). Most pilot- and full-scale systems have been large scale ($> 35 \text{ m}^3/\text{h}$), though a small-scale ($0.15 \text{ m}^3/\text{h}$) system was piloted efficaciously in Iran (143). Few long-term pilot studies or economic assessments have been completed in conditions relevant to resource-constrained communities.

4.1.7. Solar stills. Distillation is the process of evaporating water and condensing the resulting vapor within the same system. While distillation processes include MSF, MEE, and MVC, these technologies are only affordable at capacities above $3,000 \text{ m}^3/\text{day}$ and for higher salinities than those typically present in groundwater (87). Thus, this section focuses specifically on solar stills in which feed water contained in a sealed basin is evaporated using solar energy transmitted through a plastic or glass cover, and the condensate flows along the underside of the cover to a collection trough. Efficacious household solar still pilots have been performed to remove TDS in Mexico (144), in India (145, 146), and in Bangladesh (147), and to remove As in India (145). However,

distillation does not remove chemicals with boiling points lower than or near that of water, such as chlorine and volatile organic compounds.

Reported benefits of solar stills include the ability to run off-grid, removal of almost all contaminants, and a modular design. Reported drawbacks include high capital costs relative to other desalination technologies (87) and significant maintenance requirements, including removal of algae and of sand and dust from the still cover. Solar stills are more attractive for small production volumes because the required land area scales linearly with the production rate. Large systems may additionally require pumps to transfer feed, product, and brine streams, requiring an external energy source. A recent review found that still efficiencies range from 23% to 60% (148). However, efficiency will vary annually and by the hour. Therefore, it is important to note the climatic conditions and duration used in the efficiency calculations. Hundreds of solar still designs have been proposed and tested, with recent designs focused on enhancing the traditional design to include energy storage materials and/or wick materials (146, 147, 149).

Brine: the highly concentrated by-product of desalination processes

4.2. Waste Management

Most chemical water treatment processes produce significant amounts of solid and liquid wastes that are contaminant enriched and must be disposed of, monitored, or processed to recover resources. Solid wastes include spent adsorbents or ion exchange resins, as well as slurries containing coagulated salt flocs. Liquid wastes are produced from the backwash, regeneration, and rinse steps of adsorption, ion exchange, and precipitation processes as well as the concentrate streams from membrane processes. The characteristics of the waste are determined largely by the concentration and composition of the feed water and the efficiency of the removal process (11, 150). Highly efficient removal processes, or membrane processes with a high water recovery ratio, tend to produce a more highly concentrated, and therefore more hazardous, waste stream. If waste is not properly managed, it can become a new source of chemical contamination, potentially polluting previously uncontaminated aquifers or accumulating in soil and plants (151). Despite their importance, the disposal practices used in pilot- and full-scale studies are only rarely reported or described in detail.

In LMICs, solid wastes and sludge produced by community-scale and household filters are often disposed of in nearby ponds or on open fields with minimal site preparation and essentially no monitoring (11, 152). In HICs, contaminant-rich sludges are often disposed of in landfills, though there has been some debate about long-term stability (153). Recently, As- and F-laden wastes were successfully solidified and stabilized in construction materials, such as mud bricks (e.g., 154), cement (155), and concrete (e.g., 156, 157). However, these methods have not been scaled up, and some authors have argued that the stigma likely to be associated with a construction project using materials enriched with hazardous contaminants would pose too great a risk for any builder to agree to use the materials (158). A few recent studies have reported on waste disposal methods used in sustained operations of As removal plants, in which treated water is consumed by users. One study reported storing sludge in a monitored, aerated pit to maintain oxic conditions so that the As will remain bound to the media (159). Another study reported collection of the sludge by an environmental management company registered with and permitted by the state government for handling of hazardous wastes, and sending it for storage at an approved hazardous chemical waste site (127).

The large quantities of concentrated brine produced by membrane technologies is a major burden for membrane approaches, particularly for inland treatment plants (12). Brine is generally disposed of through surface water discharge, sewer discharge, deep-well injection, evaporation ponds, or land application (12). The choice of a management option can depend on the quantity, quality, and composition of the waste; the location of the disposal site; the availability of a

Avoidance

approaches: safe water approaches that avoid contamination through blending or seeking and developing new, less contaminated water sources

receiving site; local regulations and permissions; public acceptance; costs of waste transportation and management; and the capacity of the facility for future expansion (12, 160). Surface water discharge has low operating costs and energy demand, but it can have large negative environmental impacts on the aquatic environment and salinity of semiclosed areas (161). Discharge into sewage has been widely adopted for small-scale brackish water desalination, but it can hinder the biological processes of the wastewater treatment plant if the dilution from other incoming sources is not high enough (12). Deep-well injection is generally considered in the absence of other options, and it requires careful prior study to avoid pollution of nearby aquifers that could be used as a source of drinking water (162). Evaporation ponds are the most expensive option and are used primarily in dry and semidry areas with an abundant source of solar energy and land availability (163). Land application includes spray irrigation of brine on salt-tolerant plants and grasses and is generally used for low volumes of brackish water brine, restricted by climactic conditions, seasonal demands, availability of suitable land, and groundwater conditions (164). In LMICs, brine is frequently discharged on land, in open sewers, or in surface water (89, 165). The water treatment literature currently lacks comprehensive studies on brine disposal in resource-constrained settings, including descriptions of existing practices and proposed solutions.

4.3. Avoidance Approaches

Avoidance approaches develop alternative safe water sources by blending contaminated water with different uncontaminated or less contaminated sources or by abandoning contaminated sources and seeking or developing new sources (including rainwater or surface water). Avoidance approaches can also include consolidating with another community water system or purchasing water from a nearby system or vendor. Most of the literature related to chemical contamination of drinking water in resource-constrained communities focuses on avoidance approaches that develop or seek new sources of water, including rainwater harvesting (RWH), alternative groundwater sources, and alternative surface water sources. We focus on these approaches below.

4.3.1. Rainwater harvesting. During the wet season, it is possible for communities to collect rainwater through household- or community-level RWH systems and use it for drinking, cooking, and other domestic needs. The primary advantages are available water near the point of use, minimal maintenance costs, and ease of operation and maintenance. In the past 2 decades, there have been significant developments and interest in RWH in LMICs, particularly in urban and periurban developing areas, largely to address issues of water scarcity. However, RWH has also been heavily promoted as an alternative safe water source for As- and TDS-affected regions, particularly in Vietnam (166), Nepal (167), Bangladesh (168), Palestine (169), and Bolivia (170).

Harvested rainwater is generally free from chemical contaminants, with some exceptions (170, 171). However, microbial contamination is common (166, 172, 173) and may come from the rooftop or catchment or from animals living near the cistern (169). As a result, point-of-use treatment to reduce microbial contamination is often recommended. The lack of calcium, magnesium, sodium, and potassium in rainwater can also be detrimental to health, as surface and groundwater serve as a major dietary source of these minerals (174). Assessing these potential health trade-offs is a key aspect of implementing RWH as a safe water approach.

4.3.2. Alternative groundwater sources. Identifying an alternative groundwater source is a widely used and preferred mitigation strategy for any contaminant, both in HICs and in LMICs. This method can be significantly cheaper than chemical treatment if an alternative safe aquifer exists nearby with an available access point. However, it can be costly if a new groundwater access

point needs to be installed, which also bears the risk of having new unforeseen water quality problems. In addition, all alternative sources bear the risk that the water quality will change over time.

In Bangladesh, where a significant fraction of the rural population is exposed to As in groundwater, alternative groundwater sources have had the greatest impact on reducing exposure to date (81). This is largely due to the widespread testing and marking of hand-pumped shallow tubewells (STWs), which have enabled households to switch to nearby low-As STWs (81) or a nearby community deep tubewells (DTW). DTWs are generally able to tap deeper aquifers with lower levels of As (175). Jamil et al. (176) recently estimated that the cost per reduced As exposure as a result of well testing that resulted in well-switching to an STW was nearly 10 times lower than the cost of installing a DTW or a piped water supply. Surveys in India have suggested that well-switching could be an effective mitigation strategy for F and NO_3^- contamination there as well (81).

With well-switching, the time spent collecting water tends to increase significantly; in one study in Bangladesh, water collection time increased 15-fold (177). This cost is borne primarily by women and girls (178), potentially exacerbating inequalities and reinforcing gender stereotypes (179). This burden must be weighed against the social risk of arsenicosis, which is also borne disproportionately by women and girls (180). A more distant water source could also mean that more water is stored in the home for longer periods, which could increase its risk of becoming microbially contaminated if it is not stored properly. A recent study found that households that had switched to As-safe tubewells (which tended to be further away) exhibited a 46% increase in childhood mortality compared with neighboring households that did not switch (181). All of these risks and trade-offs must be weighed against the risk of not avoiding the chemical contamination.

4.3.3. Alternative surface water sources. Another alternative safe water approach is to switch to a surface water source, either protecting or treating the source to avoid or reduce any microbial contamination. For small water treatment systems in HICs, this approach could include connecting to a public water supply or combining with multiple communities to form a larger system (182). In LMICs, this might include a community or household water treatment filter or building a protected dug well (DW). Technologies used to treat microbial contamination in low-income regions at the community and household scales have recently been reviewed (15).

In Bangladesh, thousands of SSFs, pond sand filters (PSFs), and DWs were installed by the government and nongovernmental organizations before 2005 in response to widespread As contamination (16, 173). However, subsequent studies showed high frequencies of bacterial contamination in the PSFs and DWs, particularly in the dry season (173), and nearly a quarter of the systems were quickly abandoned (16). Even when they were functional, studies showed that only approximately 50% of people with access to these options were actually using them (183). Key factors correlated with high functionality of the community-managed systems were similar to factors correlated with high functionality of community-managed As remediation filters; a sense of community ownership in terms of contributions of labor, time, or money and the presence of a paid caretaker all increased the likelihood of functionality (16). This highlights the importance of an appropriate delivery model to ensure financial viability, pay key staff, and involve the community as a key stakeholder (17). It also highlights the importance of sociopsychological factors, such as a sense of community ownership, that both significantly affect performance outcomes and can be affected by the implementation approach or behavior change strategy of the implementing organization (18).

5. COSTS OF TREATING OR AVOIDING CHEMICAL CONTAMINATION

In this section, we review recent studies of the cost of provision of safe drinking water to address chemical contamination. We have included only efficacious studies (as defined in Section 4) in

Tubewell: a type of water well in which a long stainless steel tube or pipe is bored into an underground aquifer, generally affixed with a hand pump

Shallow tubewell (STW): a tubewell that is less than approximately 50 m deep

Deep tubewell (DTW): a tubewell that is greater than approximately 150 m deep

Well-switching: a household with an unsafe tubewell switching to a nearby safe tubewell, often in association with blanket contaminant testing efforts

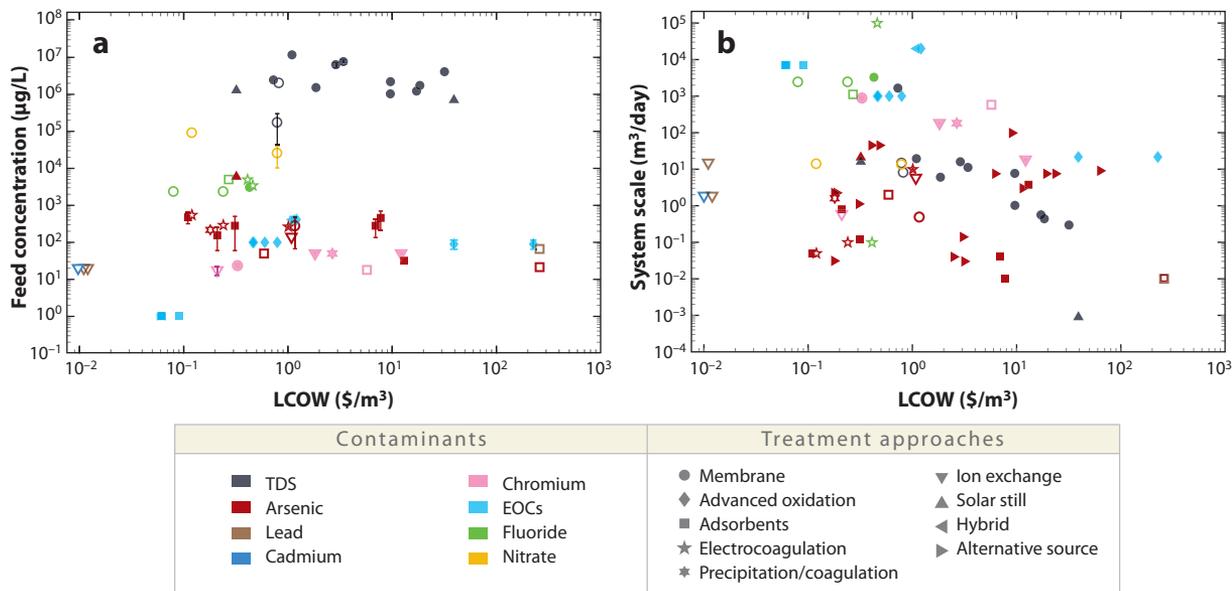


Figure 2

The levelized cost of water (LCOW) versus (a) feed water concentration and (b) system size for various treatment approaches (indicated by symbol shape) and contaminants (indicated by symbol color). The LCOW may consider only operational and maintenance costs (indicated by open symbols) or may consider operational plus fixed costs, such as capital, installation, and/or other costs (indicated by filled symbols). In all cases, final contaminant concentrations were less than the World Health Organization Maximum Contaminant Limit and guideline values presented in Table 1, the US Environmental Protection Agency guideline value for total dissolved solids (TDS), or a standard as defined in the study for emerging organic contaminants (EOCs). Reported LCOW values vary by up to three orders of magnitude, even when considering the same contaminant or technology approach and similar feed water concentrations and system scales. The Supplemental Material includes a table summarizing each data point as well as parallel figures separated by individual contaminants. This figure was compiled from data in References 86–93, 95–101, 104, 106–108, 110–117, 121, 124, 126, 128, 129, 131, 133, 144, 145, 168, 184, and 185.

Supplemental Material >

real or realistic water matrices at the pilot scale. Following the literature, we focus mainly on the water supply cost (as borne by the provider) as opposed to water supply prices (as seen by the consumer). However, a few authors report market prices rather than costs for household filters. We also note that, in some cases, the consumer bears a portion of the cost of water provision. For example, the consumer may bear the cost of safe conveyance of water to the home, up to the so-called last mile, contributing a critical component to complete the safe water system (15). Prepaid meters or microcredit can be important mechanisms to help finance these costs; however, we do not review this literature here. We present the reported LCOWs of safe water solutions based on pilot studies, followed by a review of power system costs, the cost of lifting water from the ground, waste management costs, and other reported costs.

Recurring operating and maintenance costs:

ongoing expenses related to operating and maintaining the safe water approach

5.1. Levelized Cost of Water

The most commonly reported metric across technologies and contaminants is the LCOW (in dollars per cubic meter). This is intended to be a holistic measure of the costs, ideally accounting for the full capital and recurring operating and maintenance costs of a system over its useful life, allowing for a comparison between alternative approaches. Figure 2 shows the LCOW as a function of feed water concentration (Figure 2a) and system scale (Figure 2b) for various safe water

approaches and contaminants. Pilots are further disaggregated into specific adsorbent materials or processes (e.g., RO versus EDR within the membrane category) in the **Supplemental Material**. This also includes a complete table of the LCOWs, pilot locations, feed and product concentrations, factors included in the cost estimation, and study citations, along with some additional information about the pilots.

In **Figure 2** we observe the range of contaminants, approaches, feed concentrations, and scales that have been used in pilot tests with LCOW estimates to address chemical contamination in drinking water, allowing for the identification of data gaps. In our preliminary review, the literature includes significantly more pilots for the treatment of As compared with other contaminants. Treatment of Cd, Pb, and nitrates is minimally represented, and no pilots meeting our criteria were found for Mn, Se, or U. Most of the studies for a given contaminant consider similar feed concentrations, which are meant to reflect the average or upper quartile of concentrations found in the environment at specific locations; more study is needed to understand how cost changes with feed concentrations found in different locations. Notably, certain approaches (e.g., AOPs) and contaminants (e.g., F, EOCs, Cr) have largely been studied in real and realistic water matrices only at relatively large scales (>100 m³/day capacity). This can make it difficult to extrapolate results and costs down to scales most relevant for small water systems serving resource-constrained communities. We also note that the LCOW does not compare the approaches by any of the other critical dimensions (beyond cost) that are necessary for viability.

The reported LCOW values in **Figure 2** vary by 1.5–3 orders of magnitude within each approach category and for most contaminants. The variations in reported LCOWs with contaminant or approach category do not appear to follow a trend (at this level of aggregation) with feed concentration or system scale, with the exception of membrane-based treatment of TDS, for which the LCOW decreases with increased system scale. This partially reflects the diversity of conditions represented within each approach category, including, for example, different catalysts and oxidants (for AOP approaches) or different adsorbent materials (for adsorption approaches). However, it also reflects the wide variation in the calculation methodology and components considered within the LCOW, and the limitations of using the LCOW as a comparative metric across studies. For example, the authors reporting the lowest LCOW values considered only the material cost of ion exchange media and its commercial waste management (111), while the LCOWs for some membrane systems considered the capital cost of the treatment process, water pumps, piping, storage tanks, and renewable power system, along with the cost of installation, consumables, maintenance, and component replacement (91). To explore this limitation further, **Figure 3** presents the LCOW value and the components included in its calculation for two case studies: one approach category with a single contaminant (adsorbent approaches removing As) and one specific process within an approach category with a single contaminant (RO membrane processes removing TDS).

For the adsorbent approaches shown in **Figure 3**, there is wide variation in the LCOWs of both household-scale and community-scale systems. Two studies (105, 107) state costs with no indication of what is included, making it difficult to fully identify trends. One very high LCOW for the household filter reported by Tomlinson et al. (116) represents the market price (rather than cost) of consumables for an off-the-shelf pitcher filter sold in the United States intended for low-income users. This large variation may reflect a large profit margin, highlighting the difficulty of comparing LCOWs based on market prices to those based on costs. The variation also likely reflects differences in the filters' performance; the pitcher filter has additional features, such as higher throughput, that consumers in HICs may be willing (and able) to pay for compared to filters designed for LMICs. Similar feature differences can be obscured in comparisons of LCOWs among filters or treatment systems that are all designed for LMIC markets; these filters may still

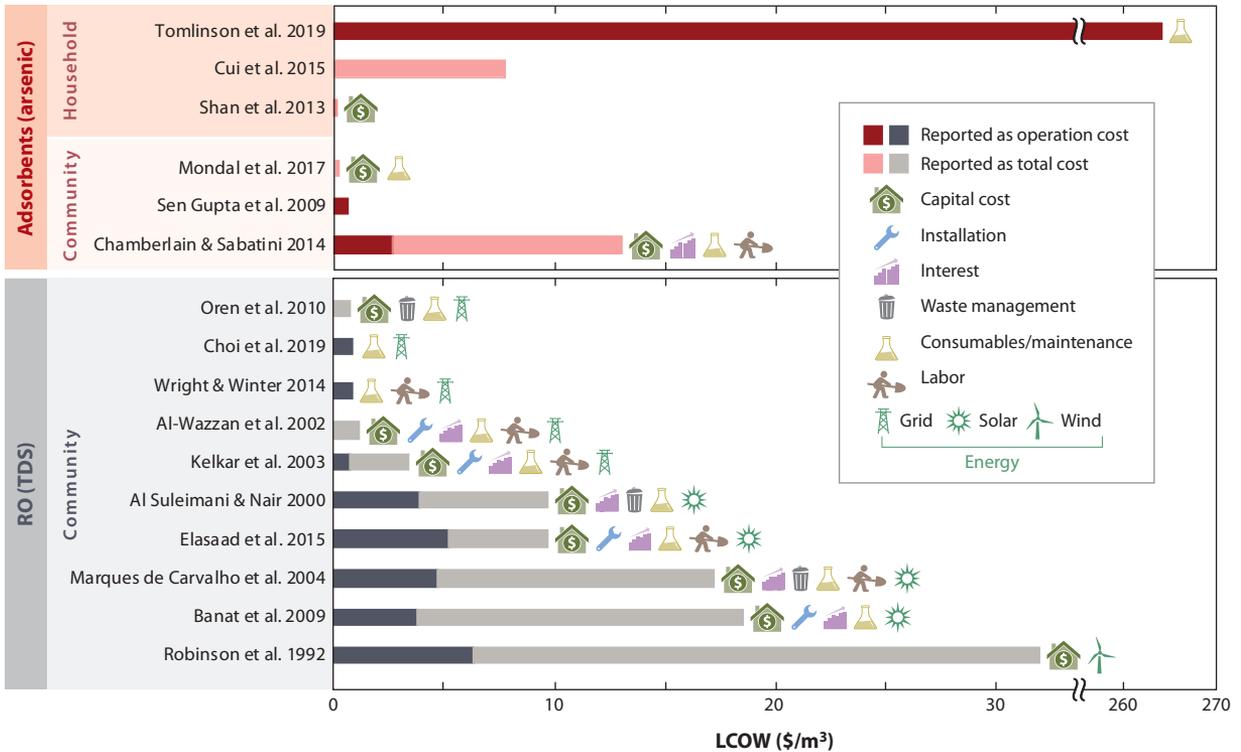


Figure 3

Comparison of the reported levelized cost of water (LCOW) and principal components considered in the LCOW calculation for two specific treatment approach/contaminant combinations: adsorbent arsenic removal and reverse osmosis (RO) total dissolved solids removal. The solid portion of each bar represents the operational costs, if operational costs were separately reported per unit volume of water produced. Adsorbents were used in either household-scale filters or community-scale treatment plants. All RO systems were community scale. References in the figure correspond to References 86–93, 97, 101, 104–108, and 116.

differ significantly in terms of ease of use, ease of maintenance, treatment time, ease of water conveyance, or other factors.

The RO case study in **Figure 3** offers a clearer picture of the variability in the type of components that are often considered in the calculation of the LCOW. As expected, reported costs tend to be higher for more comprehensive estimates that include more diverse components. Capital equipment and energy are the most frequently reported cost components, with waste management as the least commonly reported. No single paper includes all components. Some trends can be discerned from a few studies that include the same components; LCOW estimates for distributed, small-scale renewable energy powered RO are higher than those for installations connected to grid-based energy sources. However, it is difficult to draw conclusions across the breadth of studies available due to the lack of consistent LCOW methodology.

In addition to the variation regarding which specific components were taken into consideration, we observed significant variation in how each component cost was calculated. For example, capital costs were calculated in at least six different ways: actual purchase price by researchers, actual purchase price by end users, projected cost at scale, quotes from a system/component supplier, estimates based on previous literature (dollars per cubic meter or dollars per component from other articles), use of a proxy for capital cost (e.g., dollars per square meter of membrane area),

or no basis given. Similarly, energy costs were calculated as follows: actual consumption over the course of the pilot, actual consumption extended to a different system scale or assuming higher-efficiency components, assumed consumption based on previous literature, or no basis given. We also found no standard methodology for selecting interest rates or system lifetime; for the RO case study, interest rates varied between 3% and 12% over assumed system lifetimes of 3 to 20 years. One study (93) analyzed the effect of system lifetime on estimated costs. These variations make comparing costs between studies difficult and often inappropriate. We note that the diversity of conditions studied adds breadth and depth to the literature and that not all LCOW values are meant to be directly compared. However, we also note that LCOW values are often compared between common processes without due consideration or transparent reporting of variations in the underlying calculation methodology.

5.2. Power System Costs

An off-grid renewable energy system may be of interest in low-income communities due to lack of local grid availability or the desire to reduce annual operating costs for the community. Off-grid renewable energy solutions are also of interest because of the potential for environmental and human health benefits (87). In this section, we discuss current literature pertaining to off-grid and renewable energy systems directly tied to potable water treatment.

One concern with renewable energy-based off-grid systems is the high up-front capital cost, particularly in low-income communities, which tend to be sensitive to these costs (87). Field pilots that include cost estimates for solar-powered chemical remediation were reviewed for the treatment of F (128) and TDS (87, 89–91, 93, 99). Wright & Winter (87) interviewed personnel at Tata Projects, a large supplier of on-grid community-scale RO systems in India. While the company had also installed pilots of a photovoltaic (PV) system, the capital cost of this system was reported to be 58% higher than that of the on-grid system. The interview further revealed that local financial institutions were not willing to increase the payback period to account for this added capital cost, despite the fact that the operating costs would drop by 54% (87). Al Suleimani & Nair (93) estimated a capital cost increase of 48% when switching from an off-grid diesel system to an off-grid solar power system, translating to a 25% decrease in the LCOW over a 20-year lifetime. Bilton et al. (186) provided a theoretical feasibility study of community-scale PV-RO systems as a function of location and water demand and provided results in the form of global maps indicating the estimated cost of water (in dollars per cubic meter) and array area (square meters) required. The western half of Mexico, northern Africa, southern Africa, much of the Arabian peninsula, and northern Australia are some of the areas identified in which they predicted a lower LCOW for a solar-powered system than for a diesel-powered system (186). Other authors have completed theoretical design and optimization studies of PV, wind, diesel, and hybrid schemes (summarized in 187).

A few studies looked specifically at the environmental impact of small-scale renewable energy-powered chemical remediation. Ranjan & Kaushik (188) completed an economic assessment of 10 solar still designs using a standard economic model that accounted for the equivalent cost of environmental degradation and high-grade energy savings. Ramanujan et al. (165) completed a theoretical design and environmental sustainability analysis for a PV-EDR brackish water desalination system for a community in the Gaza Strip. They found an order-of-magnitude reduction in the carbon footprint of the use-phase energy consumption and a reduced environmental burden on groundwater extraction versus the existing on-grid RO systems, but pointed out the need for further analysis of the land use for the PV panels and the ecological impacts of brine discharge (165).

5.3. Lifting Water Costs

The cost associated with lifting water from a groundwater well to the treatment plant, or from a surface water source to an overhead tank, is rarely considered in the reviewed literature. Of the 70 pilot studies that include cost estimates shown in **Figure 2**, only 3 studies explicitly consider the cost of lifting water (92, 106, 110). For some, this is because there is no obvious economic cost; for example, Sarkar et al. (109) retrofit an existing hand pump with As-selective adsorbents and do not include the human labor required to use the hand pump in their analysis. Other studies assume that this cost is negligible relative to the cost of treatment. However, this is not always the case. Mondal et al. (106) estimated the cost of lifting water for their adsorbent-based As remediation pilot at 30% of the LCOW. Analyzing technologies for the removal of TDS from groundwater, Nayar et al. (189) compared the thermodynamic least energy to complete desalination with the energy required for lifting the feed water from a groundwater source. The study found that for brackish water wells for which the pumping height exceeds 50 m, the energy demand for the lift could be larger than the energy required to desalinate. Wells of this depth are not uncommon; 43% of wells in the United States have depths greater than 50 m (190), and simulations have suggested that approximately 16% of the global land area has a water table depth of >40 m (191). Lifting energy could be a major component of overall energy consumption, and cost, in these areas.

Another plausible reason to exclude the cost of lifting water is the argument that this cost will be the same for all technologies, and so will not have an impact on technology selection. However, membrane-based technologies, such as RO, may be implemented with vastly different recovery ratios [25% (89) versus 97% (101)], which will impact the amount of water that must be lifted from the ground. This trade-off should be considered along with the energetic and capital costs of choosing a higher recovery design.

5.4. Waste Management Costs

Waste management costs for many chemical water treatment processes are highly dependent on specific operating parameters and location, making them difficult to predict and generalize. Local contaminant concentrations in the feed water will affect total waste volume. In most chemical treatment processes, operating parameters (often in addition to system design parameters) control the amount of contaminant removed per process cycle, which affects cost in two opposing ways: More efficient processes tend to produce less waste (reducing disposal costs), but more highly concentrated, and therefore more hazardous, wastes (increasing disposal costs). Disposal costs tend to increase linearly with the waste volume, but increase suddenly and significantly once a certain hazardous threshold of contaminant concentration is met. Hazardous disposal costs are based on location-specific factors, such as local regulatory disposal and handling requirements, the availability of skilled personnel to handle the waste, and the distance to suitable receiving sites. Many process-specific waste management methods depend on location as well; for example, the capital costs of building evaporation ponds depend on local land prices, and their performance depends on local solar insolation and wind conditions. Similarly, the cost of installing a deep injection well depends on the local geology. All of these factors make it difficult to predict waste management costs to within a wide margin of error until the location and operating conditions of a treatment plant are known.

The challenge of highly variable waste management costs is compounded by the fact that these costs often account for a significant fraction of the LCOW. Frey et al. (192) estimate that the cost of handling wastes generated by As removal processes (coagulation, lime softening, activated alumina, anion exchange, and RO) accounts for 12% to 88% of the total water cost for US small water systems, assuming nonhazardous disposal. The cost of sludge treatment during operation

of a small pilot electrocoagulation system (12 L/h) using aluminum electrodes to remove F in Australia was estimated as 22.2% of the total water cost (129). Results from a pilot nanofiltration system to remove F in South Brazil were used to project costs for a 3,200 m³/day system that would dispose of concentrate in the municipal sewer; for initial F concentrations of 2–4 mg/L, the cost of waste disposal accounted for 32.4% to 47% of the total water cost (96). The cost of disposing of As-laden sludge through a professional hazardous waste handling company was estimated as up to 5% of the total water cost (a conservative maximum estimate) (125); the low value was attributed to the very low quantities of sludge produced during the treatment process. Chen et al. (193) examined the cost trade-offs in California between using an As adsorbent media to exhaustion, resulting in hazardous waste, versus less-efficient use of the media, which avoided the hazardous waste designation. They found that the cost of media replacement dominated over the costs of waste transport and disposal, regardless of whether the waste was designated as hazardous or non-hazardous. Brine management has been estimated to account for between 5% and 33% of the total water cost (194). Other estimates have suggested a range of \$0.40–1.80/m³ of concentrate (195).

5.5. Other Costs

Various categories of costs have been considered in the calculation of the up-front capital expenditure and recurring operational expenses reported in Section 5.1. For capital expenditure, most papers consider only the equipment necessary for the treatment process. Other reported costs appearing in a minority of pilots include, as a percentage of the LCOW (with the countries of the pilots): civil structure [7.8–12.2%, India and United States (88, 97)], engineering costs [including contractor overhead and engineering, legal, and administrative costs; 3.5–4.5%, Spain (133)], water meters [2.6%, Brazil (96)], and the costs of installing a well [1.8%, Brazil (96)]. Additional other costs as a percentage of the process equipment capital include manufacturing overhead [16.4%, India (125)]; technology installation [11–25%, India and Mexico (89, 91)]; valves and piping [15%, United States (114)]; electrical systems, instrumentation, and controls [15%, United States (114)]; and construction-related costs [site work, earthwork, labor, engineering costs, etc.; 57.5%, United States (114)]. Remaining other costs are minor (<1% of LCOW or capital).

For recurring operating and maintenance expenditure, most studies consider only the operating costs, consisting of consumable chemicals and electricity required for the treatment process itself. Additional materials costs for maintenance, generally composed of chemicals and materials required for cleaning, are considered in a small subset of studies, though these costs are rarely disaggregated from the treatment chemical and material costs. An additional small subset of studies consider replacement costs for process equipment (e.g., membranes, UV lamps, batteries, solar panels). Additional other costs in operating and maintenance include, as a percentage of operating and maintenance costs, operator labor [including labor taxes; 9.7–74%, India, Brazil, and Mexico (88, 89, 96, 110)] and water testing and analysis [10.1%, Mexico (89)].

The wide variation in operator labor costs is likely, at least partially, a reflection of economies of scale. A single plant of 3,200 m³/day has a labor contribution of 9.7% (96), whereas three plants of ≤30 m³/day have a labor contribution of 52–74% (88, 89, 110). Notably, however, of the three membrane-based treatment processes in this subset, only the 3,200 m³/day plant considers the cost of concentrate management, which accounts for 45% of the operating and maintenance costs. This demonstrates the importance of considering all costs to understand the true cost drivers of a safe water approach.

The single study reporting water testing and analysis (89) costs targets NO₃⁻ removal and includes monthly testing for pH, alkalinity, chlorine, hardness, iron, copper, and nitrates with bacteriological testing twice per year. Several authors have noted the critical importance of an

enabling environment that includes a network of reliable analytical labs offering high-quality water testing at an affordable price to the success of safe water approaches (8, 17). The significant size of these costs relative to other operating and maintenance costs (>10%) demonstrates the need for innovation to produce accurate, robust, low-cost analytical techniques to support affordable safe water approaches.

Several notable costs are not typically discussed in the literature of safe water approaches, though they may significantly influence the cost of delivering water and can be affected by aspects of the technology and the business model design. One of these is the cost required for training to build local capacity to a level where operators and skilled technicians are available to operate, maintain, and repair the equipment required to consistently deliver safe water. Many studies stress the need for suitable training programs when designing for resource-constrained communities (see 7 and references therein). We could not find reports of costs of a training program associated with delivering a safe water solution to resource-constrained communities. Chamberlain & Sabatini (104) estimated a cost of \$0.40/m³ for expenditures on direct support, which they defined as all activities associated with a new (or unfamiliar) technology or process. However, they did not specify the activities supported by the costs or their marginal impact on outcomes. Without such reports, one cannot link the technical complexity of a safe water approach (which may be affected by design decisions) to the resulting cost of required training, limiting feedback from what may be a significant cost driver. Similarly, a growing body of research (reviewed in 18) has identified factors that determine whether or not certain behaviors are adopted or maintained for safe drinking water consumption, and demonstrated that these factors can be used to guide implementation strategies that produce improved behavior change outcomes (183). However, there is not yet a literature that documents the programmatic costs of implementing strategies that target specific factors on the basis of different behavior change techniques.

6. CONCLUSION

Chemical contamination in drinking water is a global issue affecting more than one billion people, placing them at risk of adverse health impacts and water scarcity. This global prevalence of exposure to chemical contamination leads to increased health impacts, including cancers, cognitive deficits, neurological disorders, hypertension, bone deformities, endocrine disorders, and cardiovascular disease. Many safe water approaches to address chemical contamination have been proven to be efficacious in pilot- or full-scale demonstrations targeting use in resource-constrained settings. These include treatment approaches to remove chemical contamination and alternative strategies that utilize chemically safe water sources to blend or replace contaminated sources. The reported LCOW of these options varies by three orders of magnitude between different safe water approaches, between different pilot demonstrations within a single approach, and within reports of a single treatment process removing a single contaminant. In many cases, the LCOW is not calculated on the basis of the same cost categories or components. This highlights both the need to move away from direct LCOW comparisons and the need for careful, contextual reporting on the scope of LCOW measurements.

SUMMARY POINTS

1. More than one billion people globally are at risk of exposure to potential adverse health impacts from unhealthy or unsafe levels of chemical contaminants, including TDS, in their drinking water.

2. Treatment to remove chemical contamination is particularly difficult because most treatment technologies produce contaminant-enriched waste and multiple contaminants frequently co-occur. Both factors increase logistical complexity and cost.
3. India and Bangladesh are particularly strongly affected by chemical contamination in drinking water, with significant populations at risk of exposure to As, F, Mn, nitrates, and TDS.
4. Use of an alternative source can lead to new health impacts from other sources of contamination (e.g., microbes) that must be considered relative to the risk of health impacts from the original chemical contamination.
5. The LCOW of efficacious safe water approaches demonstrated at pilot or full scale varies by three orders of magnitude across approaches and contaminants targeted; there is no dominant, discernible trend with feed water concentration or system scale.
6. Large variations in the methodology used to calculate the LCOW and the cost components considered make it difficult to meaningfully compare LCOW values across studies, limiting the ability to discern trends across diverse field conditions from multiple studies.
7. Additional costs that are less commonly considered in the LCOW can contribute significantly to the end cost to the consumer. These include the costs of the power system; lifting water from a groundwater well; wastewater management; and other costs, such as labor, installation, engineering fees, and water analysis.
8. Additional costs are commonly left out of the LCOW in comparisons of safe water approaches under the argument (implicitly or explicitly) that they will be constant across approaches. However, these costs can be strongly influenced by features of different approaches and may have performance trade-offs within a given approach.

FUTURE ISSUES

1. Increased transparency is needed in the reporting of costs, including enough information for full replication of the reported cost calculation.
2. Innovation is needed to produce reliable, accurate, low-cost analytical techniques to monitor water quality.
3. Research is needed to understand the costs of training programs associated with delivering specific safe water solutions to resource-constrained communities as a function of technical complexity and of evidence-based behavior change strategies.

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