Review in Advance first posted online on September 11, 2015. (Changes may still occur before final publication online and in print.)

Safe Drinking Water for Low-Income Regions

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Annu. Rev. Environ. Resour. 2015. 40:9.1–9.29

The *Annual Review of Environment and Resources* is online at environ.annualreviews.org

This article's doi:
10.1146/annurev-environ-031411-091819

Keywords

disinfection, arsenic, water quality, health outcomes, user preference, willingness to pay, household water treatment and safe storage

Abstract

Well into the 21st century, safe and affordable drinking water remains an unmet human need. At least 1.8 billion people are potentially exposed to microbial contamination, and close to 140 million people are potentially exposed to unsafe levels of arsenic. Many new technologies, water quality assessments, health impact assessments, cost studies, and user preference studies have emerged in the past 20 years to further the laudable goal of safe drinking water for all. This article reviews (*a*) the current literature on safe water approaches with respect to their effectiveness in improving water quality and protectiveness in improving human health, (*b*) new work on the uptake and use of safe water systems among low-income consumers, (*c*) new research on the cash and labor costs of safe water systems, and (*d*) research on user preferences and valuations for safe water. Our main recommendation is that safe water from "source to sip" should be seen as a system; this entire system, rather than a discrete intervention, should be the object of analysis for technical, economic, and health assessments.

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INTRODUCTION

Safe drinking water is essential for a life of health and dignity and has been recognized as a human right by the community of nations (1). A detailed meta-analysis comparing the economic benefits of universal access to safe water services (with chlorine) to the cost of such access finds a high benefit-cost ratio of between 5.7 and 6.3 for Africa, and between 6.5 and 9.9 for South and Southeast Asia (2). In low-income regions throughout the world, however, consumers continue to rely on unsafe drinking water sources. Low-income regions themselves are heterogeneous: Poorer rural consumers have lower access to safe water than richer urban consumers (3), and piped water is in general safer than nonpiped sources (4).

There are many biological and chemical contaminants in drinking water (5, 6), and we limit the scope of this article to microbial and arsenic contamination. Microbial contamination is by far the greatest drinking water hazard in low-income areas (5); at least 1.8 billion people lack reliable access to affordable and clean water (7). Almost 1,000 child deaths per day result from diarrheal diseases caused by unsafe water, inadequate sanitation, and poor hygiene (8). Arsenic is the most hazardous chemical contaminant that significantly—and often naturally—occurs in drinking water (9, 10). An estimated 140 million people are potentially exposed to excessive arsenic (9), which leads to skin lesions, cancers, reproductive problems, and impaired cognitive function in children (9, 10). Efforts to mitigate microbial and arsenic contamination bring up a range of contaminant-specific issues (such as the removal of waste from arsenic remediation), but they also face similar implementation challenges, some of which we point out in this review.

This article reviews the recent literature on (*a*) safe water approaches with respect to their effectiveness in producing safer water or protectiveness in improving health outcomes; (*b*) the uptake and use of safe water systems among low-income consumers; (*c*) the costs of providing (and using) safe water systems; and (*d*) experimental and observational findings on user preferences and willingness to pay (WTP) for, or walk to fetch, safe water. This broad scope acknowledges that technologies, their scale, their delivery models, their costs, user preferences, and usage rates jointly determine the safety of water in the drinking cup. The review summarizes these literatures, highlights their convergences and debates, and calls out key issues for future research.

The drinking water literature often uses the terms technologies, options, interventions, and systems interchangeably, and this has made it difficult to understand exactly what is being evaluated, compared, or priced. We consider safe water systems from "source to sip" as a series of stages including treatment technologies, protection technologies, delivery models, and "last mile" labor before consumption. The research literature mostly covers technological approaches in discrete stages between source and sip, i.e., in treatment, storage, or conveyance within a safe water system. Several evaluations of these technologies analyze their costs of provision and adoption, including the supply cost to the provider and the willingness and ability to pay of the consumer. Smaller bodies of literature cover educational and social marketing interventions, whose goal is to induce consumers to switch from unsafe water to a safe water system, and waste management approaches from arsenic removal. A handful of papers have evaluated the impact on water quality from specific management techniques, such as Water Safety Plans (WSPs) or utility service upgrades. We review the main trends in all of these literatures.

We organize systems by three scales of delivery: (*a*) centralized piped and treated systems, most prevalent in the urban core; (*b*) community-based or small-networked systems; and (*c*) household-based safe water systems that call on consumers to treat their water at home on a regular basis. Within these scales, all the technologies included in our review are efficacious, i.e., they have been shown to produce safe water when correctly used in the laboratory. Their effectiveness under less-controlled field conditions has been varied. Where the literature exists, we review technological approaches at the three scales of delivery with respect to how effective they are in producing safe water in the field, or how protective they are in producing positive impacts on health. As the literature shows, positive health outcomes may not result even when microbial or arsenic loads have been reduced to acceptable levels.

We define microbiologically safe water in line with the World Health Organization (WHO) guidelines, which say that no *Escherichia coli* should be detectable in a 100-ml sample. Although the no-detectable-*E. coli* measure indicates that adequate water safety measures are in place, the WHO argues that this is an indicator of low risk rather than a primary indicator of safe water (5). We include interim interventions, such as safe storage or household water treatments (5, p. 84), which significantly and measurably reduce *E. coli* counts in low-resource settings, even if these are not reduced to zero. We define safe water for arsenic contamination in line with the WHO guidelines (5) and the US Environmental Protection Agency Maximum Contaminant Levels, which call for no more than 10 μg/L (10 ppb) of arsenic in drinking water. However, national arsenic standards for drinking water are in some cases less stringent, e.g., 50 μg/L in Bangladesh and other developing countries (10). Therefore, we include interventions that aim to reach national standards.

We include approaches for treating unsafe water as well as those that allow households to avoid contaminated water. For each treatment, we review any recent research on how well the technology, given the scale of delivery, works in field conditions. Where possible, we review its impacts on human health, observed user preferences in the field, observed adoption rates, and usage rates over time. We review the costs of each approach (given its specific delivery model), and the last mile cost: what the end user will pay in cash and how much labor she must expend. Where the literature permits, we review the cost-effectiveness of safe water systems, bearing in mind that cost-effectiveness depends on production costs, the delivery model, implementation costs, and consumer uptake. This is an explicitly techno-social framing and builds on earlier assessments of safe water treatment technologies (e.g., Reference 11).

This review does not include interventions that are primarily aimed at improving water quantity, sanitation, or hygiene, all of which are arguably as important for human health as safe water is (12). It does not include interventions to improve the quality of natural water bodies such as lakes and rivers, or to augment local sources through, e.g., rainwater harvesting (except as an explicit arsenic avoidance measure); we do, however, consider sources such as deep wells and protected springs that are specifically intended to provide safe(r) drinking water. Finally, we do not discuss environmental sustainability: This aspect of safe water systems, while very important, is beyond the scope of this review.

SAFE DRINKING WATER SYSTEMS: FROM SOURCE TO SIP

We develop a conceptual source-to-sip model (see **Figure 1**) that starts at the water source and ends at the point of consumption. All safe drinking water systems contain five stages: (*a*) source, (*b*) conveyance and storage (and sometimes treatment) from the source, (*c*) a public or private access point for the household, (*d*) conveyance and storage (and sometimes treatment) beyond the access point, and (*e*) consumption (sip). Treatment before access must be implemented by utilities or communities; after collection treatment may be done by the household. Between these treatments the water is conveyed through pipes and pumps or hauled using buckets barrels, and trucks. All stages together determine the system's effectiveness and its cost, although safe water interventions can occur at one or more stages. We review all interventions that are aimed at improving water quality at one or more of these five stages in low-income regions of the globe. Our review is skewed towards technological interventions along the source-to-sip pathway, reflecting the skew in the safe drinking water literature.

TREATING OR AVOIDING *MICROBIAL* CONTAMINATION

In developing countries, many do not have access to piped water, and of those that do, many receive water of dubious quality (7). Recognizing that even piped systems may not provide safe water, new household water treatment and safe storage (HWTS) options were introduced, and existing ones evaluated in the field, between 1990 and 2000. These include chlorination (13), solar disinfection (SODIS) (14, 15), ceramic pot filter (15), and combined coagulation-disinfection (PuR) (16). By 2001, articles and reports began emphasizing quality over just access, especially for rural communities (17–19). By 2007, the WHO had explicitly advocated HWTS for households without access to reliable piped water supplies, stating that HWTS could be effective in preventing diseases (20). In 2010, Clasen (21) argued that HWTS do not improve access, and that progress

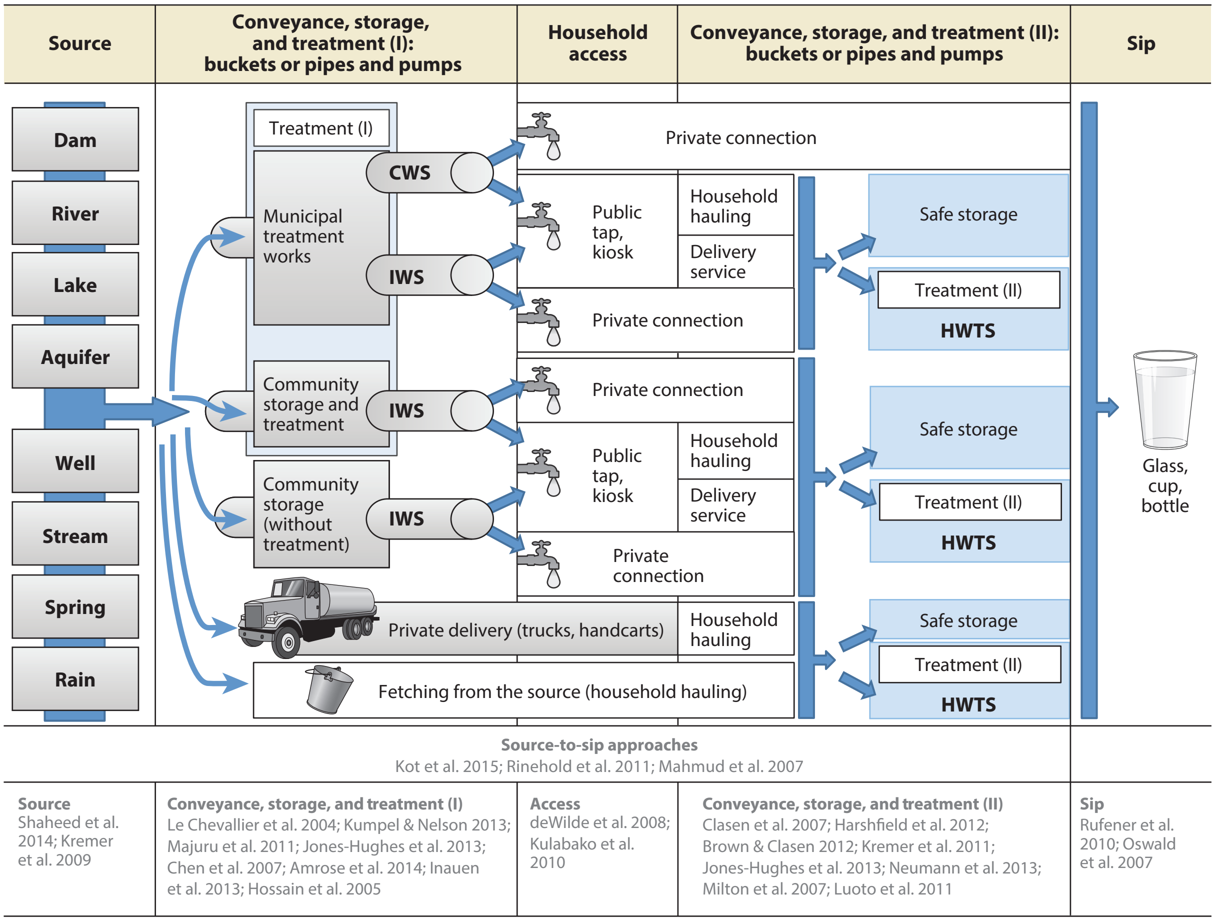

Figure 1

Source-to-sip model. Interventions may focus on one stage in the system, but they always have interactions with the rest of the system into which they were introduced. Examples are given of papers that demonstrate contamination or evaluate an intervention at a specific stage of the system. Piped water delivery systems have intermittent water service (IWS) or continuous water service (CWS). Treatment I refers to any municipal-, utility-, or community-level water treatment. Treatment II refers to any household water treatment and safe storage (HWTS). Treatment at the household level may be necessary if water quality at the access point is compromised; otherwise, safe storage is sufficient.

on access must come from expanding urban networks or small-community systems. Our review reflects the distinction between quantity and quality as found in the literature; articles on small-community or centralized piped interventions tend to focus on quantities and frequency of delivery, whereas the HWTS papers emphasize water quality and health impacts.

Centralized Piped Network and Community-Based Approaches

This section merges two scales because the treatment technologies (although not the management) are the same for piped networks and small communities. Many technologies for pathogen control are not specific to lower income countries, small-community systems, or centralized piped networks; local resources determine which are feasible for any given situation (22). Many of these technologies, for example chlorination (or UV) disinfection, have previously been reviewed in the *Annual Review of Environment and Resources* (formerly the *Annual Review of Energy and the Environment*) (11). Since 2004, there have been advances in membrane filter technology for small-community systems (23). Community-based membrane filters have been analyzed with respect to challenges specific to developing countries, such as finding decentralized energy sources (24) or providing treated water in a kiosk model (25).

The WHO counts piped water access in the user's dwelling, plot, or yard as the most improved form of access; globally 56% of people had piped water in 2012 (3). When nonpiped sources are of inferior quality, increasing the number of households connected to an urban piped water network can be an effective safe(r) water intervention (26). Improving municipal treatment, protecting water quality in the distribution network, and converting from intermittent water service (IWS) to continuous water service (CWS) all improve drinking water quality for connected households (27–29). The enormous literature on water utility efficiency in developing countries is mainly focused on volumes delivered; a notable exception is Lin (30), who incorporates percentage of water receiving treatment and continuous service into a model for Peruvian utilities.

In addition to piped water access, public taps or standpipes, tube wells or boreholes, protected dug wells, protected springs, and rainwater collection are also classified as improved. Globally, 33% of the population had access to these in 2012 (3). Sources that are considered improved may not be free of fecal contamination: In a review of 319 studies on water sources, 38% of the studies reported improved sources that had fecal contamination more than 25% of the time (31). Water quality interventions in community systems often focus on discrete stages of a source-to-sip system, for example, the creation of new sources, source protection, treatment, or improved distribution networks. Systems that provide several of these steps resemble small utilities, and may therefore take on some of their characteristics, such as the professionalization of operators, managers, and investment in some of the same treatment technologies.

WSPs have been developed and applied in several settings but have rarely been evaluated for water quality or health impacts in developing countries. According to the WHO, WSPs contain three components: (*a*) system assessment and design, (*b*) operational monitoring of control measures, and (*c*) management plans (32). WSPs analyze risks for the entire system from source to sip, with the aim of creating an improved risk management strategy (33). Community readiness can be included in the design of WSPs (34), and Rinehold et al. (35) recommend that WSPs include household storage and treatment, emphasizing the role that end-users play in minimizing health risks even in community- and utility-scale systems.

Household-Based Approaches

Any household without continuous piped water must store its drinking water. If the water is safe at the point of access, then safe storage may provide some protection against contamination in the home (36–38). The US Centers for Disease Control's (CDC's) definition of a safe storage container includes (*a*) a small opening with a lid or cover and (*b*) a spigot or small opening for safe access to the water without hands or dipping cups or ladles having to touch the water (39).

Some version of household water treatment is in use by more than 1 billion people worldwide. Different regions of the globe have widely different HWTS usage rates, from 66.8% in the Western Pacific to only 18.2% in Africa. The vast majority of users (possibly two-thirds globally) practice boiling; chlorine disinfection is the second most common HWTS, with 5.6% of all user households (40). Significant contamination occurs at the sip (drinking cup) stage regardless of the disinfection mechanism (41–43).

We review dilute sodium hypochlorite, tablets of sodium dichloroisocyanurate (NaDCC), and solid calcium hypochlorite; all deliver free chlorine (44). PuR™, a Procter and Gamble sachet product, combines coagulation with disinfection (16). Filters include biosand filters, ceramic filters treated with colloidal silver, and the Lifestraw™ filter. The ceramic and biosand filters are neither standardized nor patented (45); Lifestraw filters combine physical filtration with chemical disinfection and are patented and standardized (46). SODIS exposes water in polyethylene or polyethylene terephthalate bottles to direct sunlight for 2–30 h (the range found in the literature for 3-log inactivation of *E. coli*) (47).

Not all HWTS are created equal. Treatment time, efficacy, the appearance of treated water, and reliability vary with HWTS and source water quality. Only chlorine treatments offer residual protection. Higher turbidity decreases the effectiveness of chlorination while also increasing the risk of chlorinated organic compounds (48). The health effects of indoor air pollution from boiling using solid fuels are potentially serious (49). The effectiveness of SODIS is reduced by increased cloud cover and turbidity (47). UV lamps require electricity and relatively clear water to operate (50). Overall, each HWTS has its own pros and cons; there is no best solution for all contexts.

Water Quality and Health Outcomes: Centralized Piped Network and Community-Based Approaches

Several studies have shown that improved sources have better water quality than unimproved sources, but do not guarantee safe drinking water without additional treatment. For example, in Cambodia 47% of piped water sources and 30% of nonpiped stored water met the *E. coli* count of <1 per 100-ml sample criterion (51). In Vietnam, the mean adjusted longitudinal prevalence ratio for diarrhea for households with a piped water connection, compared to those without piped water, was 0.57 (52). Wolf et al. (53) pooled data from 61 interventions and, through a meta-regression, found a modest but statistically insignificant effect on diarrhea from moving from unimproved sources to improved (point) sources.

We found only three evaluations of interventions in centralized piped networks that reported water quality or health impact from a developing country. Semenza et al. (26) found nonpiped access with household treatment to have the lowest rates of diarrheal illness in Uzbekistan, but piped access had superior health outcomes compared to nonpiped access with no treatment. A matched comparison study from India found that whereas 31.7% of tap samples from intermittent water supply areas tested positive for *E. coli*, only 0.7% of samples from continuous supply areas did (28). Galiani et al. (27) found that expanded network coverage in Argentina, especially in poor areas, led to an 8% decrease in child mortality. It was not clear how much increased access versus improved quality contributed to this health impact. The meta-regression by Wolf et al. (53) found a protective effect from continuous piped water access compared to all other types of access, but interventions that provided basic, intermittent piped water access also improved health outcomes when compared to access from unimproved sources.

Fewtrell et al. (12) identified six studies on the health impact of community-based supply interventions, including public standposts and private connections; they estimated that the (mean) relative risk of illness from supply interventions was 0.75. In rural South Africa, Majuru et al. (54) found the all-ages incidence rate ratio for diarrhea for two intervention villages to be 0.43, as compared to a neighboring control village. It was unclear what level of water treatment occurred within these small-community systems.

In Costa Rica, Madrigal et al. (55) compared four small-community systems, two that produced higher water quality matched to two systems that provided low water quality. They usefully identified key characteristics of better performance: working rules governing operation and maintenance, engaged local leaders, local accountability, a sense of ownership, and a willingness to pay the cost of a properly managed system. These characteristics reflect Ostrom's (56) classic work on how to govern common resources, but the overall published evidence is mixed on whether the local public goods approach to safe drinking water systems has been effective.

WSPs, where implemented, have been found to improve water quality in community systems. Mahmud et al. (57) found that WSPs developed for rural communities in Bangladesh successfully reduced sanitary risks and improved water quality in public dug wells, pond sand filters, and tubewells. We found no other evaluation of WSP effectiveness in a low-income setting.

Water Quality and Health Outcomes: Household-Based Approaches

Several meta-analyses have estimated the mean health impact (on diarrhea) of HWTS: all-ages relative risk of 0.65 across 12 randomized controlled trial (RCT) studies (12); 0.43 across 6 RCT studies (58); and 0.56 across 28 studies, including RCTs and non-RCTs (59). The meta-regression conducted by Wolf et al. (53) estimated a protective effect of 0.55–0.62 for HWTS with filters, but no significant risk reduction for HWTS using chlorine when adjusted for nonblinding bias. Safe storage practices even without treatment can provide water quality and health benefits, for example in Benin (36) and Bangladesh (37).

In general, the protective effect of HWTS in the field shows a high degree of heterogeneity. Even technically effective HWTS can reduce or prevent diseases only if drinking water is a dominant source of pathogens and if they are correctly and consistently used (5, 12). Meta-analyses have estimated significant reductions in the risk of diarrhea for HWTS using chlorine, PuR, and ceramic filters impregnated with silver (58, 60). LifeStraw Personal filters appear less protective than ceramic filters; the biosand filter and ceramic pot filters show similar levels of protection (46, 61, 62). An RCT of a SODIS intervention in Cambodia found the mean incidence rate ratio for nondysentery diarrhea to be 0.38 (63). A cluster randomized trial of an in-home UV tube system in rural Mexico showed significant declines in *E. coli* in treatment compared to control households (43); however, a companion study found no effect on diarrhea from the same intervention, possibly because the baseline incidence rates were already low (64).

Effectiveness in the field has only recently been measured for boiling. Reductions of 86.2%, 99%, and 97% of thermotolerant coliforms were observed for boiled and stored drinking water in rural Guatemala, peri-urban India, and rural Vietnam, respectively. The actual concentrations of coliforms in stored water after treatment were similar in all three studies (65–67). We found no studies that compared boiling to other HWTS for effectiveness or protectiveness.

With these highly variable research designs and field results, many researchers have called for evaluating HWTS in blinded trials to minimize bias, and for using objective (as opposed to reported) outcome measures (58, 68, 69). Blinding a SODIS, boiling, or liquid chlorine trial would require complicated logistics (70). But several studies have been able to blind, or even double- or triple-blind some HWTS. None of these found any effect on diarrheal incidence in low-income settings (71–73).

There continues to be a lack of information about health impacts over the long term and in nonintervention settings (12, 58). One example is Harshfield et al. (74), who randomly chose 201 households from a sodium hypochlorite program that had been running in rural Haiti for eight years, compared them with 425 control households, and estimated a (mean) relative risk of 0.41 for diarrheal incidence in children under the age of five.

The extensive literature on the health outcomes from HWTS use seems to have concentrated on chlorination (by itself or combined with coagulation). In contrast, no papers were found evaluating the health outcomes from boiling. This is surprising, given that an estimated two-thirds of people who currently use any HWTS are boiling their water. The variation in health impacts reflects variations in the implementation models across these studies, and also the multiple-pathway nature of diarrheal diseases (12). Furthermore, consistent and sustained use after the implementation of HWTS interventions has generally been poor.

Sustained Use

It is generally assumed that households with small or piped networks, once connected, will not choose one day to disconnect from the system. No such assumption is possible for point-of-use or point-of-collection interventions. In this section, we review evidence on the sustained use of community and household treatments; sustained is loosely defined as continued use over five months or more since the end of intervention activities. The protective impact of a safe water system, whether it is community-based or household-based, is dependent on rates of uptake, as well as on correct and consistent usage. Expected health benefits drop when a household reverts to untreated water for even one day per year (75); for high and moderately high contamination levels, a decline in adherence from 100% to 90% reduces predicted health gains by 96% (76).

Many factors contribute to whether or not a household adopts and uses an HWTS. These include the flow rate, water quality, ease of use, financial costs, and supply chain requirements (77). Other important factors include taste and smell (often conflated with water quality), affordability, and cultural practices (78). Social marketing, education, and outreach methods may affect sustained use (see, e.g., 79, 80); this is an emerging area of research in the field. Psychological-social factors, such as knowledge, risk perceptions, and beliefs about health, also determine uptake and use (81).

Follow-up studies looking at HWTS usage rates six months or more after the end of an intervention have found little or no residual effect of the intervention. In a meta-regression, Hunter (82) finds that SODIS, chlorination, and coagulation-chlorination interventions lose effectiveness after 12 months, whereas ceramic filters remain effective. They suggest user drop-out, failure of the HWTS, and inability to purchase additional product as the reasons (82). Reductions in use have been found for PuR, boiling, chlorination, and SODIS after six months (83); similar reductions for the ceramic filter in rural Cambodia were mostly attributed to breakage (84). One HWTS exception is a LifeStraw Family filter intervention that targeted HIV-positive mothers in Zambia: Twelve months later, 90% of households reported using the filter (85).

With limited evidence of sustained use (and thus effectiveness) of HWTS in general, Schmidt & Cairncross (68) argued that the benefits for HWTS were low, and the acceptability to target populations was unclear, whereas both were likely high for interventions that increased drinking water access. Clasen et al. (69) responded that HWTS (mostly boiling) were already used by 850 million people in 58 low and middle income countries, indicating that other barriers were restricting scale-up and sustained use. Clearly, a better understanding of when and why households discontinue the use of HWTS after initial adoption is essential for effective interventions; absent this, serious doubts will remain on its scalability as a safe water approach.

Point-of-collection treatment at community systems may be more sustainable, although here, too, the evidence is mixed. Kremer et al. (86) found that more than 50% of users had confirmed chlorine residuals in their stored drinking water from a community chlorine dispenser in rural Kenya, two and a half years after the end of promotional activities [see also Pickering et al. (87), who provide a shorter follow-up period in Bangladesh]. However, deWilde et al. (88) found low use (and no impact on diarrheal incidence) in a point-of-collection UV intervention in rural Mexico, five years after the program began.

TREATING OR AVOIDING ARSENIC CONTAMINATION

Compared to poor microbial water quality, widespread arsenic contamination is relatively new, emerging significantly in the literature over only the past 20 years (10). Many excellent papers have synthesized various aspects of this literature in the past few years. For example, there are reviews of arsenic prevalence (9, 10, 89, 90); causes of arsenic mobilization, possible health effects and toxicology, regulatory limits, exposure routes, and mitigation options (9, 10); arsenic removal technologies (in laboratory and field work) (9, 10, 91–95); stakeholder and user preferences (89); and lessons learned from existing mitigation interventions (10, 89)

We focus on community-scale and household systems. The operation of centralized piped treatment systems for arsenic mitigation is underreported, although some work exists for Latin America (mostly in Spanish; see 90, 92). Small-community piped networks may be promising (96), but only a few systems have been implemented or reported on (97, 98), and little is known about their financial viability or near-term applicability to arsenic mitigation (10).

A clear emerging theme, with parallels in the microbial contamination literature, is that no single mitigation system will work across social, economic, cultural, and institutional contexts (10, 91, 92, 94), or across different business models (9, 99). The vast majority of arsenic mitigation studies have been conducted in rural Bangladesh or West Bengal, India [two-thirds of the arsenic-affected population resides in these two countries (9)]. Most studies compare systems comprised of a few, very similar business models [e.g., the communities-as-beneficiaries of Gebauer & Saul's (99) microwater treatment plants model, or highly subsidized household filters]. Our review is similarly biased toward these settings.

Community-Based Approaches

Community-based safe water approaches for arsenic mitigation include either (*a*) using an alternative arsenic-safe water source or (*b*) reducing the arsenic concentration of an arsenic-unsafe source. Switching to a nearby arsenic-safe tubewell commonly includes tubewell testing and labeling (e.g., safe/unsafe), and promotional campaigns encouraging users to switch (100). Well switching, deep tubewells (typically defined as a >150-m depth in the Bengal Delta), dug wells, and rainwater harvesters all attempt to use water sources that meet local standards without added treatment. Water vendors are a common source of arsenic-safe water in Cambodia, selling 10–20 L packaged water at a "low" cost (95), and are becoming more common in South Asia.

We review only arsenic removal processes that have been tested and found efficacious in the field. The vast majority of these at the community scale have been column filters containing media such as activated alumina, granular ferric hydroxide, or hybrid anion exchange media (94, 101), most of which require periodic regeneration (94). Pilot studies of small-community plants using zerovalent iron (93), subterranean in situ arsenic remediation (94), and an electrolytic technology, Electrochemical Arsenic Remediation (ECAR) (102), have shown promising results in Argentina, Bangladesh, and India, respectively, but are not yet widely deployed.

All arsenic mitigation options (including avoidance and removal) have different trade-offs with respect to source water sensitivity; complexity of operation and maintenance tasks; amenability to automation; and aesthetic water quality (e.g., taste, color, and smell) (94, 98, 103). For example, systems that include a water treatment step (e.g., arsenic removal processes) tend to be more complex than systems with no water treatment step (e.g., deep tubewells, dug wells) (98) but do not rely on the existence and verification of a naturally potable water source. The complexity of treatment can be a barrier to success for community managed systems (98, 104) but could potentially be overcome [e.g., within a community kiosk model (99)].

Household-Based Approaches

The most used household arsenic removal (HHAsR) systems are based on zerovalent iron (ZVI) (94). The SONO filter has been widely deployed in Bangladesh and uses ZVI filings treated in a proprietary process to produce composite iron matrix material (10). It is one of few filters officially approved by the Bangladesh government (105). In Nepal, the Kanchan filter has used a design based on iron nails (94). HHAsR filters frequently have low flow rates (1–5 L/h) with some exceptions (e.g., passive sedimentation sand filters, with a flow rate of 60 L/h), and most filters have problems with periodic clogging (94).

Arsenic-Bearing Waste

Unlike methods to remove pathogens, arsenic removal methods produce arsenic-bearing byproducts, most commonly as a solid waste that can contain 0.1 to 7,500 mg As/kg (106). Regeneration processes and routine backwashing of some arsenic removal systems can also produce acidic, caustic, and/or arsenic-rich liquid waste (107). With few exceptions (see, e.g., 107), arsenic-bearing byproducts are disposed of in drains, ponds, roads, and open fields with minimal site preparation and no monitoring (94, 104, 106, 108–110). Very little is known about the environmental risk of these disposal practices (106) or the human risk of handling the wastes. The Toxicity Characteristics Leaching Procedure (TCLP) is the most common test to characterize arsenic-bearing waste, and passing the TCLP is often used to claim that a specific waste is environmentally benign. However, the TCLP was developed for US landfill conditions (106)—it was not designed to determine environmental risk under vastly different conditions or to classify waste for human handling.

Some researchers have proposed stabilization of arsenic-bearing solid wastes in bricks or concrete (106). Recently, cement stabilized arsenic-bearing iron oxide waste from an ECAR plant in West Bengal, India, was shown to leach <0.4% of its total arsenic over 406 days in chemically simulated rainwater (111). More work is needed before these methods can present an acceptable level of risk to a company or a community.

Contaminant Swapping and Relative Risk Assessment

All arsenic-safe alternatives and arsenic removal systems are susceptible to microbial and fecal contamination (95, 112–114). Therefore, arsenic mitigation interventions run the risk of trading negative health impacts of arsenic for negative health impacts of microbial contamination. Howard et al. (112) compared the burden of disease expected from the measured microbial contamination of installed arsenic mitigation options in Bangladesh to the burden expected from local arsenic contamination; they found that the microbial disease burden was comparable or higher than the arsenic burden for options other than deep tubewells in the dry season. In Bangladesh, Wu et al. (113) found a statistically significant increase in reported diarrhea episodes for children 2–5-years old drinking from shallow arsenic-safe tubewells compared to those drinking from shallow arsenic contaminated wells.

In addition to microbial contamination, new drinking water sources can expose the population to industrial or agricultural runoff, pesticides, nitrates, manganese, fluoride, or other contaminants (9, 115). The health risk due to all contaminants must be assessed (through testing for known potential contaminants) and compared to the risk of continuing to use arsenic-unsafe sources (112–114). In the case of deep groundwater, if contamination or water of undesirable quality is found at one depth, testing may reveal an alternative depth with water that meets WHO guidelines and aesthetic preferences (115), making the most use of available arsenic-safe sources.

Effectiveness: Alternative Safe Water Sources

Alternative arsenic-safe sources mostly provide water with <10 μg/L arsenic, meeting the WHO guideline (95, 110, 112, 116). In the dry season, a small fraction of dug wells in Bangladesh (∼20%) contained >10 μg/L arsenic in a randomized survey of sources (112). The vast majority of deep tubewells provides water with <10 μg/L arsenic (10, 116), with some exceptions (89, 117).

There has been some debate over the long-term vulnerability of deep tubewells to arsenic intrusion from contaminated shallow aquifers, particularly if these aquifers are used heavily for irrigation (89, 116). However, no arsenic intrusion has been definitively measured in the Bengal basin (116), and several researchers have argued that the urgent need for arsenic mitigation justifies the risk of deep aquifer arsenic intrusion in the distant future (89, 116).

Effectiveness: Arsenic Removal

The efficacy of most arsenic removal processes is sensitive to source water characteristics [e.g., pH and concentrations of arsenic, iron, phosphate, silicate, and calcium (10)] that vary throughout many arsenic-affected regions. Field studies have only recently begun to (*a*) report on a wider variety of source water characteristics, beyond arsenic and iron concentrations, (*b*) discuss how those concentrations compare to the ranges typically found in the region of interest, and (*c*) target field studies across diverse aquifers representative of the region (e.g., 105). The contingent nature of arsenic removal efficiency makes generalization extremely difficult. Furthermore, effectiveness studies are often conducted by research groups with a direct interest in the technology's success; this may result in unintended bias. Although data on source water and possible conflicts of interest are needed to fully interpret efficacy results, for brevity and to discuss trends, we focus here on post-treatment arsenic concentrations.

A systematic review of research from 1980–2011 on field effectiveness for arsenic remediation technologies resulted in a review of 51 studies (44 with efficacy data), evaluating 50 technologies across 90 interventions (94). The efficacy of an intervention was rated as excellent if ≥95% of the reported post-treatment samples contained <10 μg/L arsenic, good if ≥95% of samples contained <50 μg/L, and poor otherwise. Oxidation-filtration and ion exchange interventions mostly showed poor evidence of efficacy. Coagulation-coprecipitation-filtration interventions were mixed—approximately half showed poor and half good evidence. ZVI and adsorption technologies mostly showed good evidence, with one adsorption technology (an activated alumina filter) showing excellent evidence. The most studied individual technology was the SONO filter, with one intervention showing poor, and nine interventions showing good, evidence of field efficacy.

Two notable results emerged from Jones-Hughes et al.'s (94) review: (*a*) Up to August of 2011, only one arsenic removal technology was reliably highly effective under field conditions in developing countries; and (*b*) of 51 published studies, 50 were appraised as weak in their research design. Studies were weak for a variety of reasons, ranging from small or vague samples to differences between the outcomes presented in the methods versus those actually reported. There is clearly a need to gather stronger evidence of arsenic removal efficacy in the field.

In the past two years, field effectiveness has been reported for several emerging and existing arsenic removal technologies. Community-scale ECAR was piloted over 3.5 months in West Bengal, India (102); post-treatment samples consistently showed <10 μg/L arsenic, with <4 μg/L achievable after some system modifications. Granular titanium dioxide ($GTiO_2$; 118), treated laterite (TL; 119), and Hybrid Anion Exchange (HAIX; 101) media filters were operated in China, West Bengal, and Cambodia; post-treatment arsenic concentrations were consistently <10 μg/L

($GTiO_2$), <6 μg/L (TL) and <50 μg/L (HAIX) for 9 months–2 years. Oxidation-coagulation at optimized pH was tested in Assam, India (120); post-treatment arsenic concentrations were <10 μg/L for 30 consecutive days. Eleven SONO filters were field tested in Bangladesh (105); post-treatment arsenic concentrations were <10 μg/L in most cases for new and used filters after operating for 15 months to >8 years. Thus, field data on reliably effective arsenic removal technologies are beginning to emerge, although viable delivery systems (from source to sip) are still poorly understood.

Effectiveness: Functionality Under Field Operation

For a safe water system to be effective, it first has to be functional. In a survey of 1,060 randomly selected arsenic mitigation technology installations in Bangladesh, only 64% were functional and 55% of these suffered from periodic breakdowns (98). Deep tubewells were the most likely to be functional (90%), and arsenic iron removal plants, the least (17%). The implementation model also has an effect; Kabir & Howard (98) found that community contributions and functionality were positively correlated; and Johnston et al. (114) attributed successful operation of a community filter to the presence of paid caretakers. Community participation is often cited as essential to successful mitigation (10, 97), but the overall evidence demonstrates that it is not sufficient and that community-run arsenic treatment can easily become defunct (109). In contrast, Ravenscroft et al. (116) reported on 43 deep tubewells in Bangladesh and found that 40 were still in use 13 years later, supporting the high long-term functionality of deep tubewells.

Protectiveness: Reducing Arsenic Exposure

Arsenicosis symptoms develop after 5–10 years of exposure (10), making it difficult to measure the health impacts associated with a specific intervention. Urinary arsenic concentrations have been used to estimate the reduction in recent arsenic exposure. In Bangladesh, Milton et al. (110) and Norton et al. (121) reported lower than expected reductions in urinary concentrations among adopters of household arsenic remediation systems, after 12 months and 12 weeks of reported use, respectively. This result was attributed to exposure from other sources (e.g., alternative drinking water sources or food), ineffectiveness of the intervention, or the release of arsenic stored in body tissues from previous exposure (110, 121). These studies suggest that the health impacts of household arsenic removal systems might be lower than expected.

Several studies have measured a significant reduction in arsenic exposure associated with the use of arsenic-safe water interventions. Milton et al. (110) measured a substantial decrease in urinary arsenic metabolites among individuals who reported drinking from the community dug wells 100% of the time. However, such full compliance was rare. Chen et al. (100) measured a 46% reduction in average urinary arsenic concentrations among ∼6,000 participants that had switched from an arsenic-unsafe well (>50 μg/L arsenic) to an arsenic-safe well over a two-year period, compared to those who did not switch.

Adoption and Sustained Use

In 2006, it was estimated that ∼29% of the entire arsenic-affected population in Bangladesh had switched to a nearby arsenic-safe (arsenic < 50 μg/L) shallow tubewell and ∼12% were using an arsenic-safe deep tubewell (122). Recently, Inauen et al. (103) found that among 1,200 households in Bangladesh, only ∼62% of those with access to an arsenic mitigation option regularly used it, suggesting that estimates of arsenic-safe water coverage (based on the presence of supply) may be overestimated. Deep tubewells showed relatively low adoption (54%) in spite of high acceptability, possibly because of distance from the average home. Well sharing showed the opposite trend (71% adoption, low acceptability), indicating that users may perceive no other option.

In Bangladesh, the fraction of participants who receive a household arsenic removal filter and are still using it 1 to 5 years later varies widely: ~20% at 12 months (110), 72% at 2 years (108), 75% at 1–2 years (114), 93% at 1–5 years (103), and 0% at a few weeks (97). This variation could reflect differences between the technologies (although in several cases, the same filter was studied), or could reflect the variety of ways in which the technologies were introduced and supported. Certain promotional activities and factors, such as whether participants contributed to the filter cost, can affect adoption rates (103). These details are often not reported, making it difficult to understand what the sustained use (or lack thereof) should be attributed to, i.e., the technology or the system within which it is embedded. It is also difficult to interpret self-reports of use. Neumann et al. (105) were only able to recover 7% of the arsenic expected based on self-reported use in a SONO filter operated over 8 years, indicating it was used much less frequently than reported.

Abandonment rates of community arsenic removal filters also vary widely (97, 98, 101, 103, 109, 114), with little data on ongoing usage rates for longer than two years. Management choices, such as whether a caretaker is paid, can affect functionality outcomes (98, 114) and confound the results; it is difficult to distinguish a system that was abandoned because it fell into disrepair from a system that fell into disrepair because it was abandoned. This problem also has its parallels in community-based interventions against microbial contamination (e.g., 88).

COSTS OF SAFE DRINKING WATER

In this section, we review recent studies on the cost of provision as well as the end-user's cost of safe, or at least safer, drinking water, for our three scales of delivery. The cost of safety up to the so-called last mile, whether borne by institutions, individuals, or a combination of these, is a key element of a safe water system. We do not review the literature on financing these costs, such as prepaid meters or microcredit, although these are potentially important ways of easing up cash or credit constraints.

Costs of Treating or Avoiding Microbial Contamination

Low-income populations need low-cost access to safe drinking water, even if some combination of public and private sources is willing and able to subsidize the cost of provision. "Low cost" is a frequently used and infrequently explained term in the safe water literature. Low-cost systems may be labeled as such by comparison with piped water from the source to access point (e.g., 11, 123). Alternatively, they may be considered low cost when compared to a benchmark of affordability (e.g., 124). Studies suggest that 3%–6% of a household budget can be considered affordable, but this refers to overall water, sanitation, and hygiene expenditures and not to the incremental cost of better water quality.

Centralized Piped Networks

The literature assessing piped network reforms mainly reports changes in volumes, coverage, and tariffs, and rarely mentions water quality indicators. On the basis of limited data, piped water quality seems to have improved with private sector participation and its associated higher tariffs in OECD countries (125). For low- and middle-income countries, the evidence on water quality post utility reforms is thinner and more varied. Network reforms have resulted in near-universal coverage with improved water quality in Phnom Penh (126), significant infrastructure fees (127) and significant improvements in network coverage and in child mortality (27) in Buenos Aires, and continued poor quality and reliance on vended water in Jakarta (128). To highlight the combined technical, financial, and institutional mechanisms that underlie water quality improvements in low-income cities, we look in some detail at Kampala (Uganda) and Hubli-Dharwad (Karnataka, India).

In the late 1980s, the median formal and informal costs of getting a new water connection in Kampala added up to \$197, in a city where the (average) total monthly expenditure for low-income households was \$170 (129). The next decade saw a major overhaul of the public utility, with performance-based managerial compensation and an adjustment of tariffs with reduced connection fees (130); this was overseen by strong commitment and investment from the state. By 2006, customer perceptions of service quality and water quality were high (130), and 28,000 new connections had been added (131). The average tariff of US \$0.65 per m^3 (2006 US dollars), indexed to inflation, covered the operating costs of the National Water and Sewerage Corporation (131, 132). The poor without private connections still had access to public standposts, many with prepaid meters charging \$0.01 per 20-L jerrycan; this was more than four times cheaper than vended water in the city (133). However, the drinking water quality from the public standposts in low-income neighborhoods remains poor and discolored (134, 135), and some residents with private taps sell potable water to their neighbors.

The literature on urban water in India uniformly agrees that its network systems are characterized by intermittent water supplies, poor water quality, and high coping costs (136). In line with this, in 2002, tariffs in Hubli-Dharwad covered 19% of operating costs, and water deliveries were as infrequent as once a week (137). In 2003, when a public-private partnership upgraded 10% of the city to continuous water supply, private connections were metered, and effective management passed from the city to the state. However, against the advice of pro-poor reformists, the free public standposts were shut down. Recent research finds that water quality measurably improved in the continuous supply zones (28). Tariffs also rose. For some households, these changes resulted in a 300% rise in monthly water charges (137), and in 2012, more than 50% of households with large unpaid bills were low income. However, coping costs have dropped sharply, and the poor had had to pay the most in coping costs from unreliable and poor quality supplies (138). These examples show that piped network systems may need to make several near-parallel changes to provide reliably safer drinking water.

Community-Based Approaches

Small-community system costs are even less well documented than piped-network water costs. The reported cost of provision is frequently limited to installation costs (e.g., the devices) and partial operating costs (e.g., electricity usage). It is rare to include what could be called the "enabling costs" of provision, such as social marketing, maintaining the supply chain, or community mobilization. User cash costs are usually zero, because community water systems are treated as public health interventions rather than as water supply interventions (139); therefore, the assumption is that they will be subsidized. However, much of the mobilizing, hauling, and treating in these "free" systems is carried out by female labor, with the unpaid body, in effect, substituting for pipes (140, 141).

In an experimental study of protected springs in Kenya, the cost of the protection effort averaged \$1,000 per spring, with an estimated \$55 per year for maintenance, and an average user base of 46 households per spring (142). This excludes the local costs of promoting safe water, reminder visits to households, community organization costs, etc., all of which were presumably necessary.

Even without counting these enabling costs, the authors found a near-zero social return to the protection efforts.

In a study of free chlorine dispensers at the community well in Kenya, Kremer et al. (86) found an uptake of 60% within six months and of more than 50% 30 months into the intervention. It is easy to chlorinate when collecting water, and part of the disinfection gets done during the walk back from well to home. As cost-effectiveness depends on rates of uptake, this research confirms the high cost-effectiveness of chlorine (see 15, 60), at least when it is free to the user, convenient, and no other safe water choices are on offer.

Small safe water enterprises, as opposed to free community water, allow for cost recovery from the users, and thus are demand-driven rather than top-down. The sale of UV disinfected water through WaterHealth International kiosks (143), with a long payback period in low-income communities, is expected to recover capital and operating costs at the (very) low price of $0.20 to $1.00 (in 2008 US dollars) per person per year. Cost recovery is, however, dependent on high uptake, and consistent uptake is a challenge for community-based systems. An observational study in rural Mexico found that five years post-installation, households preferred buying bottled water delivered to their doors to fetching free UV water (88). A small study in Ghana found that three years post-installation, only 38% of households were using UV-treated water (144); many continued to use sachet water, despite its higher cost and uncertain quality. Both examples highlight the high value that the poor give to the convenience factor.

Household-Based Approaches

The HWTS literature is similarly incomplete when it comes to cost of provision or cost-effectiveness of safe water systems. Yet these technologies have been vigorously promoted because of their potential to prevent disease while being "low cost." A full cost accounting would include annualized costs including capital (for a durable technology) and all cash, labor, and promotional inputs until replacement in a range of settings (145). In reality, the literature offers only partial costs from a few studies.

Boiling, the most common HWTS in use (40), has varied costs because of the differences in fuel costs and stove efficiencies (66). Research from peri-urban India finds a low mean cost of $0.7 per month for wood stoves (65). A more detailed estimate from rural Vietnam finds that buying wood for boiling costs, on average, 1% of monthly income, with another 2%–3.5% added as the value of labor (67). Boiling is more expensive than chlorination, with its running costs of $0.3 to $0.4 per month; however, household chlorination has been hard to sustain because of taste and smell (see, e.g., 15, 146).

Of the other HWTS reviewed, SODIS has no cash costs, in effect. Ceramic filters impregnated with silver have been promoted in rural Cambodia, where they cost $4–$8, with replacement costs of $2.5–$4, in an area where 31% lived on $1 per day (84). Procter and Gamble's PUR sachets, unless subsidized, are more expensive; research from Guatemala finds a commercial cost of $0.14 to treat 10 L of water (147). Household models using UV tubes are viable only for middle-income regions; the full cost in 2008 was $50 (50) for a locally manufactured device in Mexico. Finally, there is limited evidence that stand-alone safe storage is a cost-effective intervention (e.g., 36, 43).

Overall, considering both community- and household-level approaches, and looking at the health benefits of microbiologically safe water, the literature comparing the (partial) cost-effectiveness of several water disinfection methods suggests that chlorination dominates source water protection or other disinfection systems. Household chlorination costs have been estimated at $53 per disability-adjusted life year (DALY) averted (123), whereas community source water chlorination costs could be as low as $20–$25 per DALY averted (142). Even with modest uptake, and with marketing and reminder visits included, chlorine appears cost-effective relative to the public health benchmark of $100 per DALY saved (see 139). The literature is not conclusive about how sustained the use of chlorine can be, as this depends on overcoming negative user perceptions of chlorine and on all of the unaccounted-for costs discussed above. Moreover, blinded studies have not shown protective benefits from chlorination (53, 71, 72).

Costs of Removing or Avoiding Arsenic in Drinking Water

The costs of arsenic removal or avoidance reported in the literature are subject to all the caveats discussed in the microbial contamination section above: They are usually presented without accounting for the numerous, and necessary, socially borne costs of deployment, uptake and consistent use, and sludge disposal. This is a new literature compared to the costs of providing microbiologically safe water, and we do not yet have estimates of the cost per DALY averted, for example, from various approaches to arsenic mitigation versus avoidance.

In an extensive review of costs and effectiveness of arsenic removal options, mainly from India and Bangladesh, Jones-Hughes et al. (94) find that community filtration options are generally low cost when they rely on locally sourced materials. Variants of activated alumina have a wide range of flow rates (42–1,000 L/h) and of capital costs. (The median was 3,000 US dollars, converted from Australian dollars.) Operating and enabling costs were rarely, if ever, mentioned in these studies. Household filters, such as the SONO system or Kanchan filter, cost between $10 and $36 to buy; the year to which these prices are indexed is unclear, however. The filter flow rates ranged from 1–5 L/h; this means that the slower variants could not satisfy the UN-recommended 20 L per person per day for the average household.

Johnston et al. (114) find that, once community filtration is installed, rural Bangladeshi communities could pool together the approximately $300 needed to replace the filters, but most simply fail to do so. In rural Cambodia, the cost of arsenic removal with absorptive media has been found competitive with piped water supplies (148), but hand-dug wells and vendor-supplied water remain the cheapest short-term options. Roy (149) models the reductions in sick days, medical expenses, and avoidance costs (meaning walking to a safe source) once arsenic-safe sources of drinking water were made available in two districts of West Bengal. She finds that the total monthly benefits for an average household are almost $7 a month, but that most households prefer piped water to arsenic-removal options (see also 97).

In sum, the literature on the cost of providing or acquiring safe drinking water is growing, but it remains partial and difficult to use for generalized conclusions. A comprehensive view of the cost per household of safe drinking water would include installation costs, normal operating costs, repairs and replacements, interest paid on borrowed money, education to induce a shift from unsafe to safer sources, marketing, community mobilization and labor (if needed), and planning and policy changes at local or regional levels (145, 150). In addition, costs of water quality testing and monitoring, and (for arsenic removal) of sludge management and monitoring, should all be included (148). However, in much of the supposedly low-cost literature it is not clear what has been included and what has been excluded, and who bears these excluded costs.

Though household safe water systems have generated a voluminous literature, their reliability, and therefore their cost-effectiveness, depends on their consistency of use. This is, at present, low for most safe water interventions, at least given their accompanying pricing and delivery models. The costs to the provider and the user of every intervention depend on the delivery (or business) model, and these are almost never clarified in the cost of safe water literature. In a discussion of business models for arsenic-safe water that is also applicable to microbial safety, Gebauer & Saul (99) show that different business models (e.g., free water for the indigent, or prepaid water "ATMs") have different cost structures, stemming from differences in their economic versus their social goals. Making business models transparent seems a necessary step before the cost- effectiveness of safe water systems can truly be compared.

USER PREFERENCES AND WILLINGNESS TO PAY FOR SAFE DRINKING WATER

User preferences for safe water products and their willingness to pay for these are relevant considerations for the effectiveness and financial sustainability of safe water systems. Preferences can be seen as (partial) predictors of user uptake and sustained use, especially as user perceptions are often uncorrelated with actual water quality (see, e.g., 151). WTP studies are useful for estimating the potential for (partial) cost recovery from the user and for estimating the subsidies needed for widespread uptake. For microbial contamination, we review preferences and WTP for household treatment, as we found no studies on small-community or piped systems that unbundled water supplies from water quality. The WTP for access to enough water is almost certainly higher than that for safe water. For arsenic contamination, preference studies cover both community- and household-scale systems.

Microbial Contamination

Thus far, in experimental and survey-based preference studies, filters seem to fare better with users than other HWTS. This comparative preference has been found in research from rural Kenya (152) and urban Bangladesh (153). In a non-experimental study from India, Poulos et al. (154) also report that filters were preferred to chemical additives. Overall, chlorine additives show low rates of user preference in studies from East Africa and (especially) South Asia when users are given a choice of HWTS.

A body of research on how to raise user preferences and adoption rates for safe water systems has now emerged. Ahuja et al. (139) provide a good review of education and social marketing efforts aimed at increasing rates of usage of safe water systems. These include dissemination of information regarding local water quality (155, 156), commitments from and reminders to community members (157), and messages showing the health and social desirability outcomes from using safe water. Mosler et al. (158) tested several different types of outreach and promotional activities and found household visits by trained promoters to be the most effective. (In line with the discussion above, however, no costs of outreach were mentioned.) A review of outreach methods for safe water finds that current evidence is equivocal on what are the most persuasive methods (159). Dreibelbis et al. (160) argue that individual-level psychological factors should not be emphasized at the expense of technological and other contextual factors.

As might be expected from the preference studies, consumer WTP for safe water products is generally low, even when the offer price is technically "affordable." Low-income consumers routinely incur coping and averting costs (in cash and labor) to avoid drinking unsafe water, and these can exceed 1% of household income (see, e.g., 161). These coping costs may not be reflected in WTP studies on safe water. However, some studies based on the contingent valuation survey method (or stated preferences) find a WTP of up to 1.2% of mean household income in peri-urban Cambodia (151); at least 3% in Éspirito Santo, Brazil (162); and between 1.8% and 7.5% in Parral, Mexico (163). Of these three, Brazil and Mexico are middle-income countries; only Cambodia is low income.

Survey-based methods are, by design, hypothetical in nature. WTP studies using travel cost or real-auction evaluation methods (i.e., revealed preferences) allow researchers to observe at least a one-time real payment. Kremer et al. (142) estimate the willingness to pay for safe water in Kenya by estimating the average travel cost to protected springs versus closer but unsafe sources. They find an average WTP of 12.7 (female) workdays, which, when monetized, yields a WTP of \$9 per year per household—a low figure even in a region with a wage rate of <\$2/day. In an auction experiment from Bangladesh, Luoto et al. (164) found that the WTP for chlorine products actually went down with use. Revealed WTP methods consistently show lower WTPs for safe water compared to stated methods. The only generalizable conclusion about preferences from the literature is that safe drinking water has a high benefit-cost ratio, depending on the technosocial context of provision, but that households undervalue the private benefits of safe water. On the basis of these and other studies, Ahuja et al. (139) conclude that, when considering the benefit-cost aspect of safe water, there is a strong case for subsidization or free provision even if usage rates remain modest—especially for point-of-collection chlorine.

Arsenic Contamination

When asked to compare arsenic mitigation options, community members and institutional stakeholders strongly prefer piped water systems, followed closely by deep tubewells (89, 95, 97, 103). Studies show a preference for community over household options in general (89); this finding is commensurate with the findings, reported above, that household systems show high rates of failure and disuse. However, distance is a barrier to deep tubewell use (103, 116) and piped water systems have not been installed in many rural regions due to their high capital cost. Well sharing (i.e., switching to arsenic-safe shallow tubewells) continues to be one of the most used options, although it is not considered desirable; this may reflect the paucity of available alternatives (103).

Preferences for household arsenic removal filters have been mixed but are generally low due to complaints or perceptions of clogging, low flow rate, breakage, and bad taste (89, 94). Community arsenic removal filters also tend to receive low ratings due to difficult operation and maintenance (103, 108; but see 94). Operational and maintenance (O&M) difficulties reflect both the technology and the implementation model (e.g., a trained kiosk operator might perceive these differently from a volunteer), and issues of taste and O&M costs are specific to technology type as opposed to arsenic remediation per se. Overall, no study has compared the O&M costs or collection time burden required to provide the same level of reliability and water quality (including microbial and chemical water quality) across technology and implementation model combinations.

As with safe water systems for microbial contamination, behavior change campaigns have been used to increase well-switching behavior in arsenic-affected areas of Bangladesh (89, 100). Inauen et al. (103) found that technology-specific psychological factors could influence behavior change interventions. This implies that different technologies will require different levels of investment to achieve a comparable level of adoption, a cost that should be considered for technology comparisons (148). Most studies do not provide enough information about the educational or motivational programs that accompanied technology interventions, but descriptions of repeated meetings, focus groups, skits, and songs (100, 110) suggest that these costs could be significant.

The poor performance of arsenic removal technologies and lack of coordination among implementers has resulted in a persistent negative association hindering future adoption. Hoque et al. (165) point out that communities are confused when they begin using one promoted technology only to be asked to change their behavior again by another group. Das & Roy (109) note the widespread frustration with many arsenic removal technologies that quickly become defunct, making future arsenic removal interventions unwelcome; this phenomenon has also plagued microbial contamination removal efforts. These historical and locally specific effects might increase the cost of arsenic remediation in affected areas.

CONCLUSION

At least 1.8 billion people do not have affordable and reliable access to drinking water free of microbial contamination, and approximately 140 million people are exposed to dangerous levels of arsenic through their drinking water sources. A range of current technologies at urban utility, small-community, and household scales are efficacious against microbial or arsenic contamination. However, effectiveness in providing safe water or protection against waterborne diseases in the field remains highly varied. Arsenic mitigation or avoidance is the more recent literature of the two, and stronger research designs are needed here to confirm the effectiveness of some heavily promoted technologies.

Piped water is considered the most improved form of access, and users prefer piped water to other options for arsenic mitigation; access to piped water, however, is only slowly being expanded in low-income regions. Uptake, consistent use, and affordability for the poor remain major challenges for non-piped systems. New research indicates that social marketing, frequent reminders, and other enabling activities can increase user preferences and valuations for safe water. Overall, protectiveness (in terms of human health) and affordability (for society or the user) are achievable, but remain highly context specific. Despite existing advocacy of household water treatment methods to mitigate both microbial and arsenic contamination, the literature suggests that most HWTS-based systems, with the possible exception of boiling, are unlikely to be transformative at larger scales.

Finally, technology descriptions and assessments dominate the reviewed safe water literature, but technologies are only part of a safe water system from source to sip. Effectiveness or lack thereof, and low costs of provision or lack thereof, which are routinely attributed to technological interventions are, in fact, characteristics of the technology plus its delivery (or business) model, and its accompanying marketing and mobilization activities. Assessments and comparisons of efficacy and cost-effectiveness are only meaningful along a specific source-to-sip pathway. The entire range of costs incurred—including the socially borne enabling costs of outreach, mobilization, failures, and transitions from unsafe to safe water—should be transparent to researchers, safe water advocates, and policymakers. To reach the goal of safe and affordable water for all, a systemic approach to safe water services is more useful than intervention-by-intervention assessments.

SUMMARY POINTS

1. Safe drinking water from "source to sip" consists of a series of interactions between technologies, their delivery models, their scales and costs of production, and consumer uptake and consistent use. Safe drinking water is a system, not a product or an intervention.
2. Users prefer (safe) piped water to other options, community-based arsenic removal to household removal options, and community-based chlorine dispensers to household chlorination.
3. Of household treatment and safe storage systems, users seem to prefer ceramic filters to other options; chlorine is generally disliked on account of its taste and smell when other choices are available.
4. Boiling is the most used water treatment globally, but studies rarely include it as an option in multi-HWTS comparisons of effectiveness or user preference.

5. It seems unlikely that household treatment and safe storage systems—with the possible exception of boiling—can be transformative at scale under current prices, delivery models, and preferences, but they are effective and protective in specific contexts.
6. Cost analyses for "low-cost" systems are usually reported on a partial basis, with installation costs and some operational costs included. The enabling costs of social marketing, mobilization, education, reminders, and community- or household-based unpaid labor are mentioned but not explicitly accounted for.
7. Delivery models and business models significantly affect costs and uptake, at all three scales of service. Yet they are rarely made explicit.
8. Safe water systems can be highly effective, but consumers undervalue drinking water quality and have low willingness and/or ability to pay for safety. This is a particular challenge for arsenic mitigation or avoidance, as arsenicosis is only evident after several years of exposure.
9. From both a human health and a social welfare perspective, it may be necessary to subsidize systems with high benefit-to-cost ratios (e.g., those with low costs per DALY averted).

FUTURE ISSUES

1. Research should consider a pathway along the safe water system, as opposed to a specific safe water intervention, as the unit of analysis for a realistic account of effectiveness and scalability in low-income regions.
2. More independent assessments, as opposed to designer-led assessments, of safe water interventions are needed, especially as interventions progress beyond the pilot stage.
3. More research on piped systems and community-based systems is needed, relative to that on household-based systems. The safe water literature is skewed toward HWTS at present.
4. New research on community or household safe water systems should focus on user convenience and ease of dissemination; these remain understudied relative to assessments of technical efficacy and WTP.
5. Future research on low-cost safe water should aim for a fuller accounting of costs (and thus of actual benefit-cost ratios) from the water source to the drinking cup, to make transparent the total costs of safe water delivery (145).
6. The transition time to move populations to safe systems from unsafe water is significant for new (and even existing) technologies. This time and effort are largely unaccounted for and may lead to overoptimistic assessments of safe water interventions.
7. The most vulnerable populations in low-income regions are arguably migrants, refugees, and the homeless. Little scholarly literature exists on robust methods of safe water provision for these groups.

DISCLOSURE STATEMENT

The authors are not aware of any affiliations, memberships, funding, or financial holdings that might be perceived as affecting the objectivity of this review.

ACKNOWLEDGMENTS

We thank the reviewers for their thoughtful comments and suggestions. These have significantly improved the clarity of this paper.

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