Risky responsibilities for rural drinking water institutions: The case of unregulated self-supply in Bangladesh

Alex Fischer[a,*], Rob Hope[a], Achut Manandhar[b], Sonia Hoque[a], Tim Foster[c], Adnan Hakim[d], Md. Sirajul Islam[e], David Bradley[f,g]

[a] School of Geography and Environment and Smith School of Environment and Enterprise, University of Oxford, OUCE, South Parks Road, Smith School of Geography and Environment, Oxford OX1 3QY, United Kingdom
[b] Department of Engineering, University of Oxford, United Kingdom
[c] Institute for Sustainable Futures, University of Technology Sydney, Australia
[d] United Nations Children's Fund (UNICEF), Bangladesh
[e] Laboratory Sciences and Services Division, International Centre for Diarrheal Disease Research (icddr,b), Bangladesh
[f] Department of Zoology, University of Oxford, United Kingdom
[g] Department of Disease Control, London School of Hygiene and Tropical Medicine, United Kingdom

A B S T R A C T

The drinking water sector is off track to reach Sustainable Development Goal (SDG) 6.1 with over a quarter of the world's population lacking safe and reliable services. Policy approaches are shifting away from provision of access towards managing the multiple risks of water supply and quality. By considering how infrastructure, information, and institutional systems evolved in Bangladesh, this article identifies the unintentional consequences of reallocating management responsibility for rural water services away from government agencies towards individuals and households.

Between 2012 and 2017, we estimate up to forty-five unregulated tubewells were installed privately for every publicly funded rural waterpoint. This growth rate more than doubled total national waterpoint infrastructure since 2006. The scale of growth is reflected in the declining ratio of households per tubewell from over fifty-seven in 1982 to less than two in 2017, potentially approaching market saturation. This scale of growth aligns to an observed decrease in the real price of private market shallow tubewells by seventy percent between 1982 and 2017. In 2018, we estimate households invested up to USD253 million in tubewells, nearly sixty-five percent of the total national water and sanitation sector's household-level finance. In effect, household investments became critical to achieve the Millennium Development Goal (MDG) target of improved infrastructure access, but now pose challenges for meeting targets of safely managed services. The scale of continued private investment provides an opportunity for policymakers to explore blended public finance models to meet emerging consumer preferences, while at the same time introducing regulatory and monitoring systems.

1. Introduction

The challenges of achieving and sustaining safely managed rural drinking water for all is being redefined at national and global levels by expanding the set of standardized targets embedded within Sustainable Development Goal (SDG) 6.1. Although the day-to-day experience of water consumers has not changed, the additional indicators widen our understanding of the global gaps in institutional performance. In 2019 the United Nations Joint Monitoring Programme reported that 785 million people were in need of improved drinking water infrastructure access compared to over 1.9 billion who required safe and reliable

drinking water services (UNICEF/WHO, 2019). The SDG targets reflect how global policy drivers are aiming to move beyond a model defined by provision of goods—specifically improved infrastructure access targets—to one that also includes managing the multiple risks factors that undermine safe and reliable services, a key distinction in water security literature (Bradley and Bartram, 2013). This raises questions of how to adapt dominant institutional models designed around providing access to drinking water infrastructure towards ones that manage the uncertainties, vulnerabilities, and hazards identified across the SDG 6.1 dimensions of water quality, variability of supply, reliability of infrastructure functionality, and behavioral complexity of decisions and

---

investments. In many cases, this requires reflexive confrontation of the unintended consequences of past decisions and the existing institutional allocation of risk responsibilities.

Global and national monitoring systems are constantly evolving enhanced tools and methods to evaluate sector and institutional performance, as well as to identify how risk and exposure is distributed across geographies and subpopulations (Bartram et al., 2014). However, these monitoring systems provide limited insights into how institutions are designed. Knowing who makes decisions and how responsibility is allocated within institutions is a core component of understanding risk management. Institutional theorists North (1990) and Ostrom (2010a) explain that institutions are designed around the formal and informal rules, norms, and shared strategies that order, replicate, and thus shape individual and collective decisions. Risk-management approaches seek to reduce the uncertainties of the decision-making processes by using information and rules to increase predictability of the decision outcomes.

The shift in what is measured in SDG 6.1 reinforces the growing policy and academic debate around the limited capacity of existing service models to reduce exposure and vulnerability to water safety and supply reliability risks. Analysis of formal responsibility within existing water and sanitation service models focuses largely on multi-user systems, and largely within the context of urban utilities (Bakker, 2013). This positions many rural and small systems outside regulatory mandates and discussions. Recent policy research around the role of private financial capital in rural contexts suggests that privately managed systems are increasing in many regions of the world but have only recently been formally recognized within a small number of national planning frameworks (Butterworth et al., 2013; Lockwood et al., 2017; Sutton, 2017). This indicates a growing policy gap for managing risks extending from privately funded and individually managed infrastructure. These self-supply systems that serve a limited number of users remain outside existing regulatory systems and are overlooked by national monitoring systems, specifically in emerging market contexts.

Bangladesh offers a case study reflecting global patterns of how drinking water responsibility has shifted between public and private spheres. This article quantifies the extent to which self-supply service models have increased access to domestic water supplies for millions of households while also identifying the unintended challenges created by unregulated decisions and individualization of risk-management responsibilities. While there is a significant body of literature quantifying exposure, vulnerability, and hazards related to groundwater quality and supply in rural Bangladesh (George et al., 2012a; Johnston et al., 2014; Kabir and Howard, 2007; Smith et al., 2000), we suggest there is a more limited analysis of how institutions adapt to and manage these risks. Achieving the SDG targets on water safety, maintenance, affordability, and equal access will be a challenge under current institutional designs. The SDG indicators contribute to making these performance gaps visible.

To explore these gaps, the article reconciles multiple data sets with new surveys of rural infrastructure systems to contribute a multi-decadal perspective on public and private investment into tubewell and handpump technology in rural Bangladesh. The analysis applies machine learning techniques to adjust for uncertainty, adapts mean-crowding metrics developed by population ecologists to understand clustering of growth patterns, and draws on spatial analysis to analyze the rate and scale of growth of tubewell infrastructure across different regions of Bangladesh. Data from three drinking water infrastructure audits located in different regions of Bangladesh are compared to identify the scale of growth and ratio of tubewell installations per household. One of the study areas is further contrasted with household socioeconomic surveys. After applying the growth rates to a national scale model, we discuss the implications of the findings for private finance of infrastructure, national monitoring systems, and risk-management approaches.

1.1. Intersection of risk theory and institutional design to achieve SDG 6.1

Rural drinking water service provision models are changing as the social and political risk logics of institutions evolve. For decades, institutions in many countries have been designed around positive logics of acquisition of access to improved drinking water infrastructure. These institutional mandates are now refocusing more explicitly on mitigating and managing the distribution of uncertainty, hazards, vulnerabilities, and exposure in relation to water quality, supply, reliability, and equitable access. This phenomenon is seen as part of wider societal shifts of risk logics from ones focused on the distribution of "goods" such as income, social services, health, and infrastructure into one increasingly focused on the distribution and prevention of "bads" by managing uncertainty to reduce hazards, vulnerabilities, and exposure (Beck, 2013, Beck, 1992). This negative logic places attention on the allocation of responsibility linked to the formal and informal constraints on decision-making systems (Beck, 1992; Bergkamp, 2017; Ewald, 1991; Matten, 2004). The SDGs have reinforced concerns that many states are unable to ensure safe and reliable drinking water in rural contexts if the current institutional models remain focused on positive logics of access.

The differences between these risk logics are immediately understood by the way formal rules, embedded in legal, policy, and administrative governance structures, respond. This process transforms how institutions are designed and where the formal and informal rules assign responsibility for making certain decisions (Garrick et al., 2018; Ostrom, 2010b). Many national governments have retreated from previous public mandates by ceding responsibility to local municipalities, private sector providers, and civil society. As the multi-leveled government system struggles to deliver satisfactory services, specifically in rural contexts, many decisions are shifted to individuals acting outside formal or collective processes.

These governance transitions in the rural drinking water sector follow the general process of devolution of institutional responsibility. Lockwood et al. (2017) categorize these service delivery models into five groups: community-based; direct local government provisioning; public utility; private sector; and self-supply. Under all these models, central government agencies maintain some form of overarching mandate to ensure provision of drinking water, often financing the larger infrastructure components. In rural contexts, this is often structured around unregulated community management models (Harvey and Reed, 2007). With the wide-scale concerns of underperforming functionality and reliability of community-managed systems, there is increasingly a push for professionalized service delivery approaches (Hope, 2015; Moriarty et al., 2013). In the absence of many of these systems, and with delivery failures occurring, a fully privatized self-supply model is emerging in many areas (Sutton, 2017). In the case of rural settings, these results in the transfer of responsibility and accountability to non-government actors, most often individuals or groups of households. However, these responsibilities have not been formalized or reconciled with collective risk-sharing mechanisms or regulatory guidelines. This sequence facilitates what Koehler et al. (2018) identify as increasingly pluralist allocations of responsibility, or ones which simultaneously occur at different institutional levels.

Risk theory provides an analytic framework to interpret how institutions are shifting responsibility away from states and towards individuals. Risk-based management approaches help focus on where decisions are made within polycentric institutions. Attention to decision-making highlights how both formal and informal rules place constraints on individual and collective choices in order to reduce uncertainty. Beck (1992, p. 35) argues that modern institutions are neither designed for nor adapting to management of the risk logics that deviate from provision of goods and services. Institutions are breaking from the previous linear processes of modernization by monitoring and confronting unintended, harmful, and unpredictable side effects of previous routine economic, social, and techno-scientific decisions (Lash,
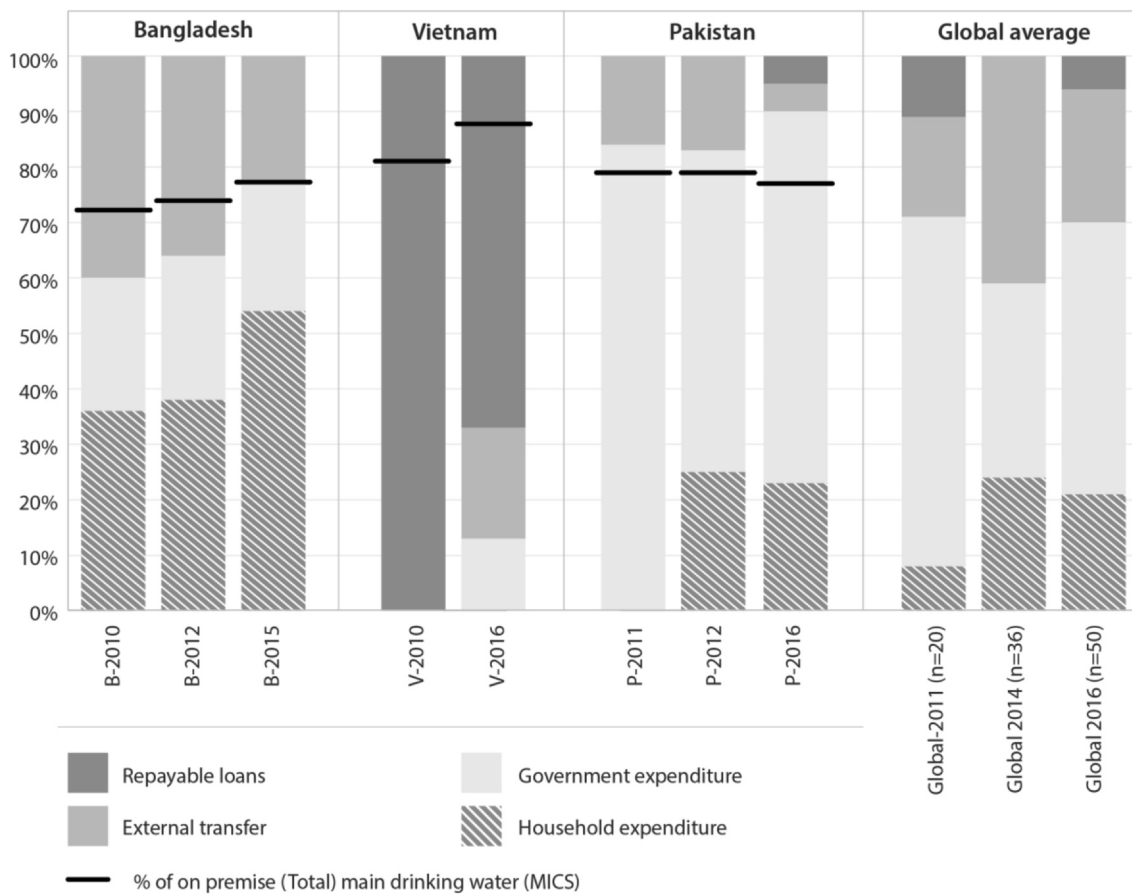
Fig. 1. Proportion of WASH sector financing sources compared to percent of on-premise drinking water coverage. Data: WHO (2012), WHO (2014), WHO (2017), and BBS (2015).

2003). This self-awareness, or reflexive confrontation of past decisions, challenges previous risk narratives and perceptions of state capacity to deliver public goods. Rayner (2007, p. 166) further argues that the rise of these risk politics has caused the "state to retreat from many responsibilities and prerogatives that it took for granted in the mid-20th century, ceding them to the private sector and to civil society in the form of non-governmental organizations." This is reflected in the increasingly diffuse institutional power structures that are promoting decentralization, marketization, and privatization of responsibility (Bakker, 2002; Beck, 2013; Mythen, 2005).

In response to changing allocations of risk responsibility, rural institutions are gradually being redesigned, largely informally, away from collective responsibility structures towards individuals and households. This is often pronounced in places where the public sector is ceding its role or has been unable to deliver. This article identifies how and where these institutional transitions in the rural drinking water sector, as observed by Rayner (2007) and Bakker (2002), have extended beyond civil society and private utilities assuming responsibilities where states have ceded that role, to that of the individual outside any public regulation or collective shared decisions.

1.2. Uncertainty of the scale of self-supply and private investment

Self-supply, or privately financed and managed systems, often with limited number of users, are not a new phenomenon in rural areas. Recent work based on household surveys by Grönwall et al (2010, p. 14) estimate that 269 million urban residents across forty-three countries rely on self-supply, while country-specific studies in Lebanon (Korfali and Jurdi, 2007) and Madagascar (MacCarthy et al., 2013) suggest even greater scales of uncounted private systems. Sector policy

makers have started to formalize self-supply as part of wider sector strategies in a few countries, including Zambia (Sutton, 2004), the Self-supply Acceleration Programme in Ethiopia (Butterworth et al., 2013), the Upgraded Family Well Programme in Zimbabwe (Olschewski, 2016), and widely adopted rain-water harvesting in Thailand (Saladin, 2016). These recent policy initiatives reframe self-supply as an opportunity to expand access beyond limitations of government public provision models.

In many countries, including those with advanced regulatory systems, self-supply also produces unintended challenges. There is no formal recognition of private water systems and limited regulatory oversight or enforcement of formal standard for small-scale infrastructure systems. This includes limited or no oversight of water quality or water abstraction rates relative to wider supply reliability, specifically for groundwater. The challenges to managing self-supply are evident in the United States where an estimated 42.5 million individuals, or roughly thirteen percent of the total population, rely on private self-supply from groundwater sources for drinking water (Dieter et al., 2018). Drinking water systems in the U.S. with fewer than twenty users are defined as outside federal regulatory mandates, and thus do not require compliance with national water quality standards (Flanagan and Zheng, 2018). There is also variation between states in regulating aquifers with limited monitoring of abstraction rates or supply sustainability.

In other country contexts, particularly Eastern European countries, the only direct oversight to rural water systems classified as private is through permitting of drillers and construction standards (Olschewski, 2016). In 2014, the European Union introduced the Drinking Water Directive, which mandates water quality testing and alignment with water safety plans to address gaps between urban and small systems,

although it does not specify self-supply infrastructure for fewer than fifty users (Hulsmann and Smeets, 2011). While self-supply is an increasingly prominent policy option in many countries, the lack of regulatory oversight leaves millions of individuals and households at risk of being exposed to unsafe water through unlicensed and unmonitored infrastructure.

These transitions toward self-supply remain largely invisible in national measurements of institutional performance. The current SDG 6.1 monitoring systems in Bangladesh, as in other countries, are not designed to track private systems or their performance. With households as the primary unit of analysis, the indicators are not designed to reflect how risks are managed, but instead to focus on coverage and proxies for safety including improved infrastructure and distance to dwelling. The most comprehensive perspective on the distribution of responsibility is gained through the Global Analysis and Assessment of Sanitation and Drinking-Water (GLAAS) program run by the World Health Organization (WHO) in conjunction with participating countries' statistical offices. The GLAAS reports offer a globally comparative analysis of national water and sanitation policies and estimated annual national sector expenditures (WHO, 2017, 2014, 2012). The results, summarized for the period from 2010 to 2016 in Fig. 1, show a significant variation in the share of the total national Water, Sanitation and Hygiene (WASH) expenditure provided by households. The disaggregation of sector finance provided by household, government, and donor funding provides an indication of the role of the private sector in managing water and sanitation services, however it remains limited by lack of differentiation between activities or capital versus operational expenses. This is a starting point to understanding how responsibilities shift in relation to financial inputs.

1.3. Evolution of drinking water services and responsibility in Bangladesh

Bangladesh provides a context to consider how rural drinking water services have transformed and evolved around the changes to public risk narratives. The country has been frequently portrayed as one of the most complex social-ecological systems for institutions to negotiate in order to deliver safe water to the country's 160 million inhabitants, of whom 105 million are considered rural (Hoque et al., 2019; World Bank, 2018a). This portrayal emerges from the confluence of environmental hazards and the unintended consequences of infrastructure choices (Atkins and Dunn, 2007; Sultana, 2012). Bangladesh's physical topography combines with the annual monsoon cycles to create significant intra-annual variation of water supply, from flooding events to declining river levels during the dry season (Rahman and Salehin, 2013). Growing demand for groundwater resources and reduced recharge has increased concerns about declining aquifers, particularly during the dry season (DPHE, 2015a). Driven by the government's post-independence promises to provide citizens with functioning infrastructure and their renewed commitment to global development targets, drinking water access and safety has been an enduring public policy challenge in Bangladesh.

The transitions in infrastructure, governance, and risk narratives are clustered into three periods by Yasmin et al. (2018) for urban context and Fischer (2019) for the rural setting. The first period, from 1972 until 1992, reflects the focus on reducing exposure to cholera, diarrhea and fecal contamination through a shift of household water sources from ponds and other surface water to groundwater abstraction through tubewells. Between 1972 and 1985, the government and international development partners installed an estimated 408,000 public shallow tubewells and 15,000 public deep tubewells (Black, 1990). The design and manufacturing of cast-iron handpumps reduced the production costs while also improving their durability. The growth of domestic production capability paired with the release of government control over supply chains enabled the growth of local markets, many of which were initially financed by demand from the irrigation sector (Black, 1990; Frink and Fannon, 1974). These technological advances and

domestic supply chains put in place the enabling conditions for the now-ubiquitous private handpump market.

The national transition towards handpump infrastructure facilitated institutional shifts starting in the late 1970s as the central state administration decentralized service provision responsibility to *upazila,* or sub-district county level. The decisions about where to install tubewells were reallocated from the national agencies to the Union *Parishads,* which are the lowest administrative units of local government. By the late 1980s, the local councils further transferred operational and maintenance responsibilities to community-based and self-assembled collectives of users, after tubewells and handpumps had been installed. This local collective organizing was reinforced by the government's Department of Public Health and Engineering (DPHE), which required that all publicly funded tubewells went to collectives of ten or more individuals who applied as a group. The applicants had to declare the water point would be available to anyone from that group. These DPHE water points were installed after a procurement process that required a formal tender under a set of defined installation standards, which, as of 2012, required independent water quality testing for each new tubewell installation (DPHE, 2015b). This provided one form of regulation for public water points, however they remained unmonitored after installation.

As the government relinquished control over supply chains, it was replaced by the growth of the informal private sector local well-drillers, or *mistri,* who installed shallow tubewells (STW). These shallow tubewell depths are defined as less than thirty meters (100 feet) deep. These are often hand-drilled and are not regulated or licensed by the government. The rapid growth of handpump manufacturers and local informal mechanic markets arguably enabled the informal shift towards the "self-help" or consumer control model of maintenance (Black, 1990). This private market growth also resulted from an increased availability of spare parts, overcoming the previous government supply delays and improving the functionality of infrastructure.

The government and media publicly recognized arsenic contamination in groundwater systems in 1993. This marked the transition into the second period of drinking water narratives, which continued until the National Arsenic Mitigation Information Centre (NAMIC), funded under the Bangladesh Arsenic Mitigation Water Supply Project (BAMWSP), released the results of the national arsenic blanket-testing, reducing some of the uncertainties around arsenic distribution and risk. This phase is characterized by public-sector initiatives to shift household's primary drinking water infrastructure towards newly defined arsenic-safe infrastructure, primarily deep tubewells. The chronic nature of arsenic pathogenesis and uncertainty of its spatial distribution was different from the previous acute and immediately visible characteristics of diarrheal diseases, particularly cholera. Risk of arsenic contamination challenged the previous period's narrative of rural water sector progress, which had been facilitated by the wide adoption of handpumps on shallow tubewells (Caldwell et al., 2003). The risk of arsenic contamination was also markedly different from other risks, as Atkins and Dunn (2007) notes, because the knowledge of arsenic contamination is contingent upon experts and scientific testing to render the problem visible. The resulting techno-political debate around arsenic concerns focused on causality, vulnerability, and where to allocate blame for the contamination (Smedley and Kinniburgh, 2002; Smith et al., 2000; van Geen et al., 2002). The policy response therefore focused on defining safe infrastructure alternatives to mitigate exposure (Kabir and Howard, 2007; Sultana, 2013; van Geen, 2008). While arsenic-safe technologies dominated national attention, policy solutions were not directed towards the institutional implications of growing private markets driving the increasing household reliance on the shallow tubewell infrastructure.

The 2006 national arsenic mapping resulted in tubewells labelled as either red or green as a way to warn users of safety. The testing enabled an inventory to produce more granular hotspot analysis to support policy responses. This resulted in what van Geen et al. (2016) identify

as the significant rate of additional investment in public water points, predominantly deep tubewells, installed in regions with higher rates of arsenic contamination. While financial resources were provided through national budgets, specific site selection for new water points remained the responsibility of local water committees and *Pourshavas* (Sultana, 2009). Mobarak and van Geen (2019) suggest that the distribution of new infrastructure points appeared to favor local elites and politicians over technocratic processes designed to distribute safe water point access based on risk to population and demonstrated need. Although poorly tracked, the private market continued to play a role in filling gaps in public provision.

In contrast to the public water points, private systems did not require drilling permits, registration, or testing for compliance with national water safety standards. Policy documents, starting with the 1998 National Policy for Safe Water Supply and Sanitation (Government of Bangladesh (GoB), 1998) and followed by consecutive government documents including the National Arsenic Mitigation Plan (GoB, 2004a), did not specify rules for private self-supply infrastructure. The government's sector strategies in both 2004 and in 2011 (GoB, 2004b, 2011), focused on how to leverage private finance for larger multi-user piped networks. Despite the major shift of households away from surface water since the 1970s, only a small fraction of the rural population had access to formally regulated piped schemes. The majority remained reliant on tubewells. In 2017, the government estimated a national average of eighty-eight people per public operating water point (DPHE, 2017a) compared to UNICEF (1993) estimates for 1992 that placed the average at ninety-two people per public operating water point. Significant population growth may explain part of the continued challenges to service provision. The result is that the literature has arguably overlooked the conditions leading to increasing market privatization infrastructure, limitations of public service delivery, and informal allocation of risk management responsibility to households and individuals.

Part of this challenge is that tubewell growth rates have not been consistently or systematically measured over time, specifically when differentiating public versus privately funded and managed infrastructure. Estimates of national tubewell infrastructure can be traced back to 1972 when they were predominantly public water points, as discussed by Black (1990, p. 44). The 1993 UNICEF and DPHE water point status report identified the provision and growth of public infrastructure only, disregarding private installations (UNICEF, 1993). The national arsenic mitigation projects produced two of the most widely cited estimates of public and private tubewell infrastructure. The first, in the policy findings of the 2000 BGS/DPHE national study, estimated between six and eleven million tubewells, fifty percent of which were suggested to be private (BGS, 2001, p. 2). The second was an outcome of the early 2000s National Arsenic Mitigation Information Center (NAMIC) project, which tested and labeled nearly five million public and private water points for arsenic contamination and produced the national estimate of 8.6 million tubewells (Johnston and Sarker, 2007, p. 1890). Academic studies updated these figures in the early 2010s with estimates of national tubewell infrastructure numbering between 10 million and 11.5 million for both public and private infrastructure (Bangladesh Bureau of Statistics (BBS), 2010). These estimates were recently revised by Jamil et al. (2019) with estimates of eighteen million in 2013. These multiple figures and estimates suggest that the growth after 2005 is significant, however scale and rate of growth remains poorly understood outside the public provision system. This context creates uncertainty around the scale of population relying on self-supply systems and the associated risks of untested water quality and unmonitored pumping.

2. Methodology

2.1. Study design

This study is designed to triangulate infrastructure growth rates and investment decisions between two study sites that represent regions of the country with different risk profiles. The resulting field data is considered in relation to other public data including an inventory of public investment costs collected from government project archives. The empirical contributions of this paper are based on data collected through water infrastructure audits, or blanket inventories designed to map every infrastructure point in a defined geographic area and collect related information from the owners or managers of those points. This approach provides neighborhood-scale data of infrastructure coverage, source of investment, and management responsibility for the system. By mapping every physical water point the study avoids double-counting water points when co-owned or shared by two or more households, a problem identified in earlier studies (George et al., 2012b; Hanchett et al., 2014). The enumeration system identifies *bari* scale units. A *bari* is defined as a group of spatially clustered residential housing structures, generally linked by related families. These results produce comparable estimates on scale, rate, and source of growth from different regions of Bangladesh.

Several analytical tools are applied to provide interdisciplinary understanding of the results. To address uncertainty we apply the Bayesian machine learning Gaussian process (Rasmussen, 2004) to model the growth trend of tubewells. This statistical approach provides probabilistic predictions based on uncertain data and is suitable for incorporating the uncertainty of the reported age and costs. Second, to understand the density and rate of growth across neighborhoods, mean-crowding metrics were applied at the village and *bari* scale across three periods. This method, borrowed from ecologists, was originally devised by Lloyd (1967) to reveal the patchiness of micro-scale distribution and competition experienced by an individual within a defined population. Mean crowding metrics measure the density from the perspective of the individual entity in a replicable and defined area. This is applied to the Matlab tubewell inventory to capture the density per area (*bari* scale). Mean crowding values around or below one imply randomly dispersed tubewells with fewer than one tubewell per *bari* while higher values imply densely clustered units, i.e. crowding within the *baris*. This metric provides a quantitative way to understand average density dependencies of users-to-infrastructure at large spatial scales, which often disguise non-random spatial distributions and obscure inequities in access, patterns of investment, and risk exposure (Wade et al., 2018).

2.2. Study sites and methods

The two water audit studies were undertaken consecutively, starting at Matlab, Chandpur district, from February until April 2017, followed by the second research site in Polder 29 of Khulna Division from December 2017 until January 2018. The field data collection studies were granted research ethics permissions from the lead university's Central Research Ethics Committee. In addition, the work in Matlab received additional permission from the Research Review Committee (RRC) and the Ethical Review committee (ERC) at International Centre for Diarrheal Disease Research, Bangladesh (icddr,b) which includes external reviewers.

2.3. Matlab study site

The Matlab work was designed, managed, and supervised by the authors and coordinated by a trained project officer and a field manager based in Matlab. The study population is dispersed across 142 enumerator areas, roughly aligned to the lowest level of administrative unit of the ward. Within these wards, or villages, are the *baris*. Alam et al. (2017) and the annual reports of icddr,b (Haq et al., 2018) provide

further information on the study site. Within the overall 142 villages, this study purposefully selected ten villages for the infrastructure water audit, due to budget and time limitations. Infrastructure in the area included tubewells with different models of No. 6 handpump; no other handpump types were found in this area of Bangladesh. The selected villages represent a population of 25,617 people in 6036 households, according to the 2016 icddr,b demographic data (Haq et al., 2018). The ten villages were selected based on two criteria: 1) half of the villages were to be located inside the flood protection embankment, and 2) include a heterogenous sample of primary drinking water infrastructure based on the icddr,b 2014 household survey (Haider et al., 2016).

The data were collected by ten female enumerators who were hired directly from each of the ten villages and trained over the course of a ten-day period, including piloting data collection and reviewing reliability. Mobile network-enabled tablet computers were programmed with ONA software (https://ona.io/) to enable enumerators to locate and map all water point infrastructure in the selected villages and to complete a questionnaire through an integrated survey tool, with data immediately displayed on an online platform. Enumerators went to every icddr,b-defined *bari* in the villages to visually identify tubewells and asked residents to locate all domestic water points that are in use, nonfunctional, or previously abandoned, including ones located inside the walls of a built structure. Water points solely used for irrigation were not included in the study. The enumerators identified water points and interviewed their owners or managers. The data was cleaned each day, using automated logic checks and manual review to ensure reliability of entry. All respondents were over eighteen years of age and provided oral consent to be interviewed about their tubewell infrastructure.

The Matlab water audit questionnaire comprised six thematic sections after obtaining consent, including: (1) basic identifiers and physical observations; (2) previous and current use; (3) management responsibilities; (4) water quality monitoring and safety perceptions; (5) investment and affordability; (6) functionality, reliability, and maintenance. The questionnaire was designed to build on previously standardized instruments and was refined through key informant interviews and field-testing prior to implementation.

2.4. Khulna study site

The second field study was conducted by a separate field team under the same research grant using the same blanket sampling strategy and a shortened version of the Matlab water infrastructure audit questionnaire. The site is located 160 km south-west of Matlab in an embankment, or polder system, covering five unions across Dumuria and Batiaghata *upazilas* (sub-districts), with 58,000 inhabitants living in 17,000 households. The Khulna water audit was conducted in the south and central regions of the polder, covering 34,639 residents. The survey identified 2805 tubewells and nineteen pond sand filters. The data provides insights, distinct from those of Matlab, through a different hydro-climatic context in a coastal zone, and decision-responses to high levels of saline intrusion. Further details can be found in the article by Hoque et al. (2019).

2.5. Secondary data

This paper draws from the results of several other studies and an inventory of DPHE project documents. A third geographic region was identified in order to triangulate findings with the two water infrastructure audits. This third site is represented in published research emerging from nearly two decades of longitudinal data collection in the Health Effects of Arsenic Longitudinal Study (HEALS)in Araihazar. The Araihazar study area is located 51 km north of Matlab and provides two blanket surveys of tubewell infrastructure, completed in 2000–2001 and 2012–2013. For full study details see van Geen et al. (2003, 2014) and recent analysis by Jamil et al. (2019).

The public infrastructure installation cost data was extracted from sixty-seven different DPHE project proforma documents covering government of Bangladesh investments from 1980 to 2016. This inventory identified 150 unique budget items recorded as both initial tender costs and actual reported costs at time of installation. This represents every project proforma document identified by researchers in the DPHE archives and is the most comprehensive standardized historic record of unit costs available. These DPHE project proforma proposals had indicative project budgets that specified prices under categories of deep versus shallow tubewells, not the specific drilling cost per feet of depth, which is more often indicative of actual cost. Of the 150 identified cost figures in these documents, twenty-nine entries were standardized for shallow tubewell comparisons and thirty-six for deep tubewells. This analysis uses the initial tender quote aligned to the year provided. The results of yearly costs were averaged when multiple project costs were identified in the same year. As of 2017, DPHE (2017b) provided a standardized list of public sector installation costs by infrastructure type, however this did not exist for previous years.

3. Results

3.1. Differing interpretations of Matlab's drinking water infrastructure growth and transitions

The Matlab household socioeconomic household survey data provided by iccdr,b across five survey years indicates that by 1980, tubewells had overtaken surface water as the reported main drinking water source in the Matlab villages. The proportion of households relying on tubewells (both shallow and deep) peaked sixteen years later at ninety-five percent and was mirrored by the fall of surface water sources to less than ten percent, as seen in Fig. 2(A). After 1996, no other improved water infrastructure gained significant use. Piped water schemes installed in the market areas in the past decade are anticipated to increase as a primary source in future surveys, however unreliability was reported as a major deterrent of use (Haider et al., 2016; Nahar, 2007; Razzaque et al., 1996).

The 2017 Matlab water audit reveals a different trajectory of infrastructure growth. The water audit identified a total of 3830 water points in the ten villages, of which 3734 were identified as domestic water sources and used for the analysis. Abandoned tubewells, defined as points that owners have no plans to rehabilitate, were identified as eight percent of the tubewells (n = 295), while non-functional tubewells at time of visit were 1.5 percent of the total (n = 58). In order to account for the respondent's recall bias observed in the spike of reported tubewell installations each five years from date of interview, a Gaussian process (Rasmussen, 2004) was used to model the uncertainty in the data, represented in Fig. 2 (B) as upper and lower bounds of the ninety-five percent confidence interval. While the proportion of households using tubewells as their main source of drinking water declined temporarily during the height of the arsenic crisis in the early 2000s, the rate of installations continued to grow exponentially. As seen in Fig. 2(B), the majority (sixty percent) of tubewell growth (n = 2234) occurred after icddr,b's blanket arsenic-testing campaign completed in 2005, which labeled tubewells red or green in accordance with the government of Bangladesh's arsenic safety threshold of 50 ppb (Jakariya et al., 2007). Further, out of the tubewells installed post-2005, eighty percent were reported as used for drinking water. The results show that the increase in number of new tubewells did not slow after the arsenic contamination was discovered, or after the first blanket testing of every tubewell in Matlab was completed. The growth suggests households' preference for individual and private sources located on premises.

The coverage rate of primary source does not reflect the wider infrastructure context. The rate of change shows that the reliance on tubewells plateaued in the late 1990s, however new installations continued to increase exponentially, as shown in Fig. 3. Applying density of
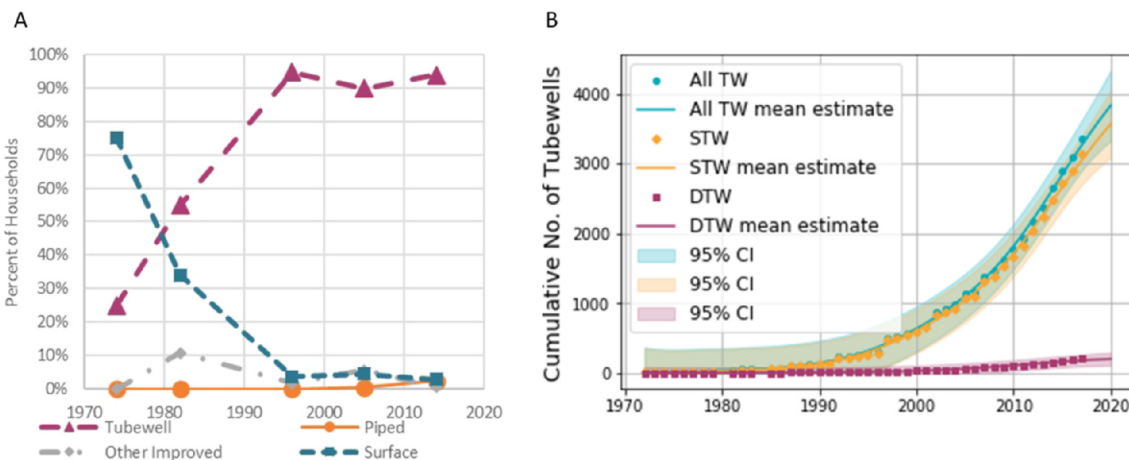
Fig. 2. Main source and growth rates A) Households main source of drinking water from 1974 to 2014 across 142 villages of icddr,b enumeration area. B) Number of tubewell installed each year as reported by owners in 2017 water audit for ten villages. N = 3734.

households to tubewell or users to tubewell ratios provides an important perspective on changing infrastructure conditions. The ratio revealed that the average coverage rates increased from over fifty-seven households per tubewell in 1982 to less than two households per tubewell in 2017.

The trajectory of the total number of tubewells in the ten villages is approaching the same number of households who report tubewells as their main infrastructure. If the 2015–2016 growth rate of nine percent and number of households are held constant, it suggests that within the next decade Matlab villages could reach a point of market saturation of one tubewell per household. This is reinforced by the water audit findings that sixty-seven percent of shallow tubewells were reported as being used by a single household compared to twenty percent of deep tubewells. When considering use as comprising three or fewer households, this number increased to ninety percent of the shallow tubewells and thirty percent of the deep tubewells. This indicates the market has shifted towards private and largely self-supply water points.

This ratio also reveals further village-level characteristics. By 2017 the differences in infrastructure coverage between villages, seen in Fig. 4, had diminished significantly. This study revealed a decrease in standard deviation of 1.63 in 2005 to 0.38 in 2017. This metric is significant when considering the total population in the study area has grown ten percent since 1996. Notably, the number of households has grown seventy percent in that same period, reflecting even greater demand for new water points from households with smaller average family sizes. Further, the two villages with historically lower coverage reported the largest increases. These two village are assessed by icddr,b's asset index as experiencing the lowest wealth levels of the ten villages surveyed (Haider et al., 2016). The two villages with the lowest asset scores showed the greatest relative increase of new tubewells. Within the newly installed tubewells, less than 1.5 percent were identified as publicly funded, indicating demand and willingness to pay among private households. The villages in this small sample are not representative, however the results indicate potential for future
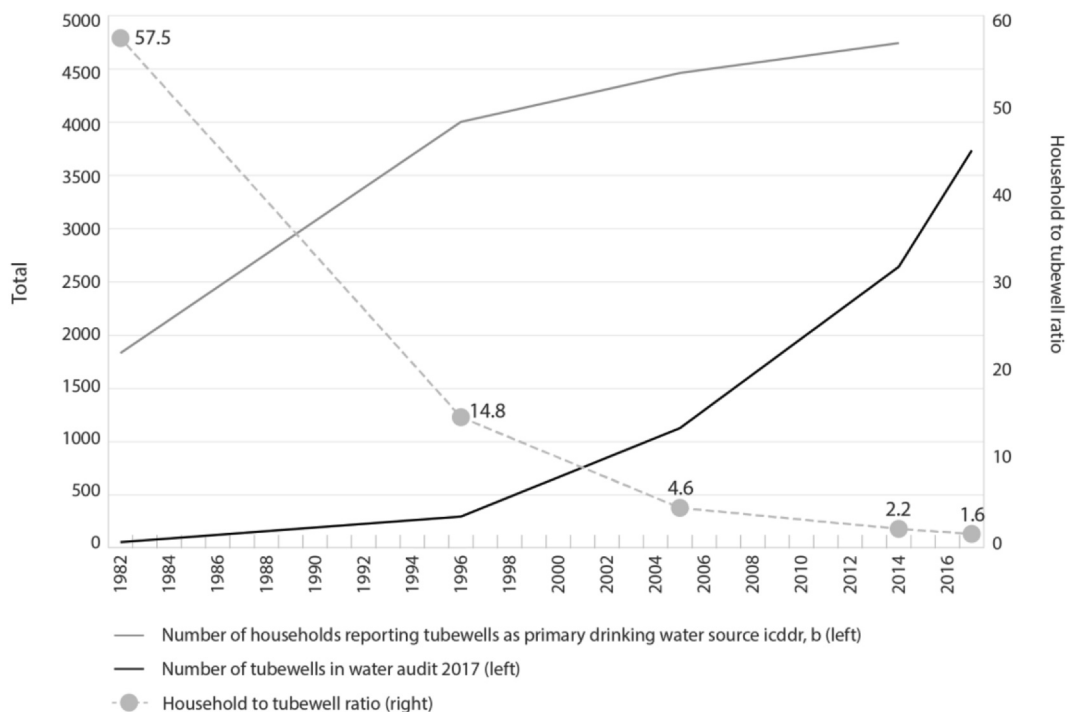
Fig. 3. Comparison of tubewell growth from icddr,b household surveys and water infrastructure audit results, 1982-2016.
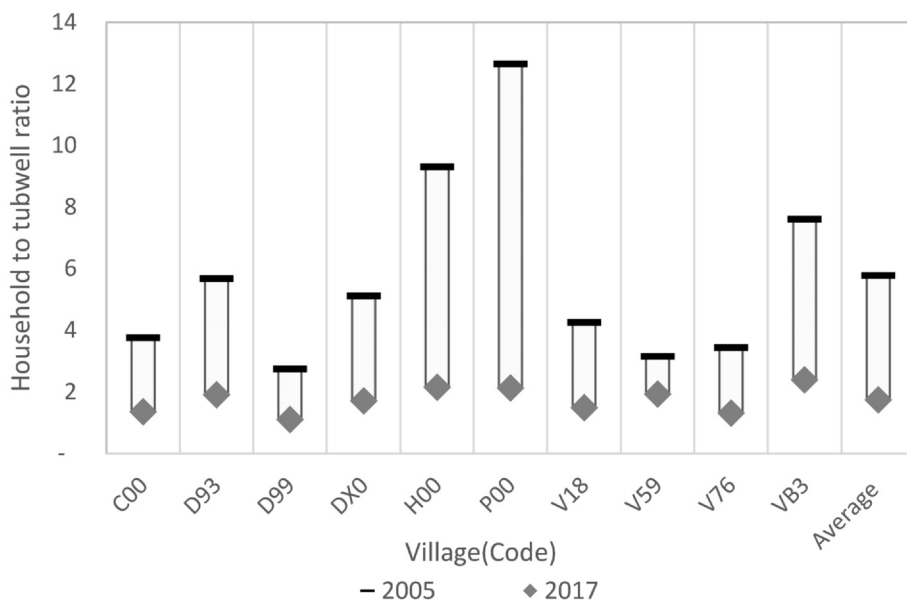
Fig. 4. Village-level comparison of household to tubewell ratio, 2005 to 2017.

research.

The village ratios are one new perspective, however they still obscure the *bari*-scale growth patterns. The spatiotemporal changes in dependencies on water points remain poorly understood. By applying the mean crowding metric to one village), illustrated through a map of tubewell locations and year of installation in Fig. 5, it provides an empirical analysis of this transition from public multi-user water points to private self-supply systems. The village has 227 tubewells for 480 households and a population of 2026 people as of the 2017 survey. Before 1996, seven percent of the *baris* had a tubewell, reflecting a

mean crowding of less than one. Despite the very small portion of *baris* having tubewell infrastructure, the prominence of tubewells as the primary drinking water source of households increased from twenty percent in 1982 to fifty-six percent in 1996 (Razzaque et al., 1996). By 2005, seventy percent of the households reported using tubewells as their main drinking water source (Nahar, 2007) but only thirty-three percent of the *baris* had tubewell infrastructure. The 2005 mean crowding remained less than one. By 2017, eighty-two percent of the *baris* reporting having at least one tubewell and the mean crowding rose to four. While density increased, the village reported only a small
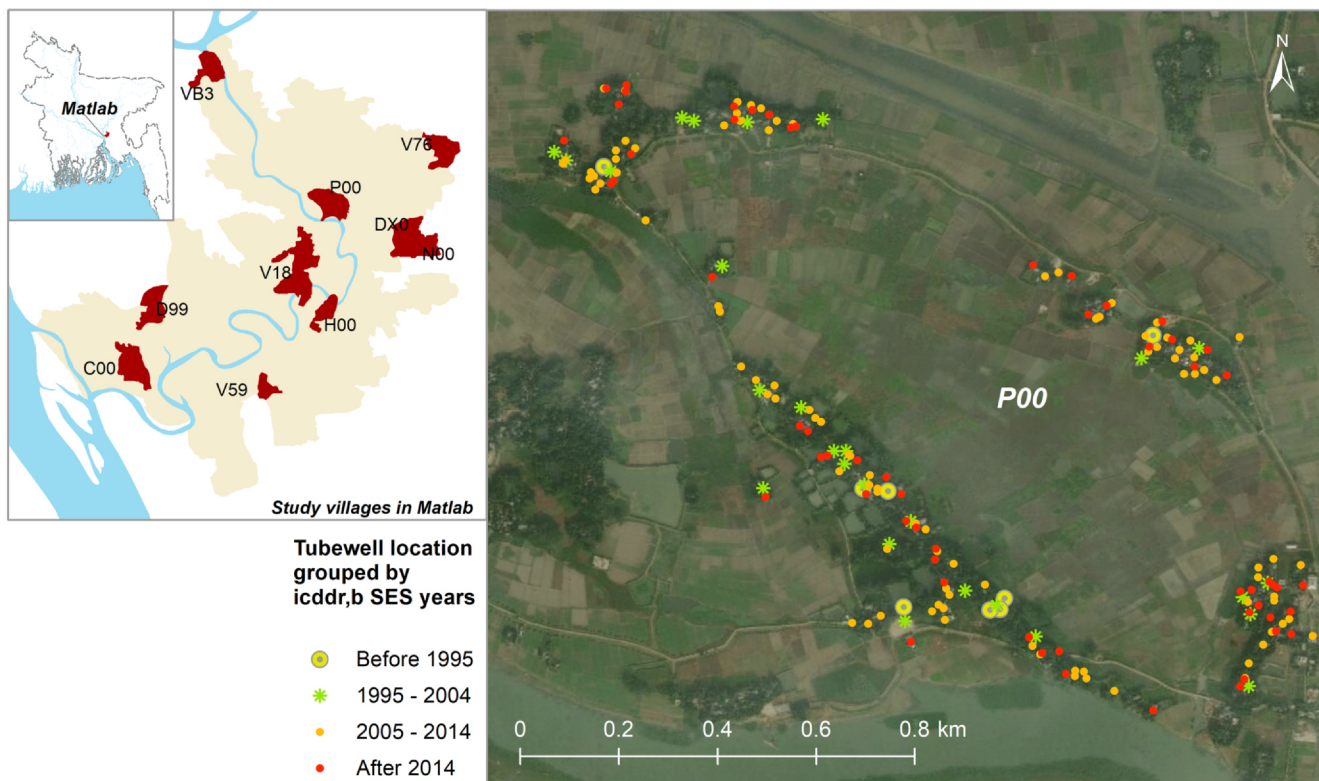

Fig. 5. Spatial distribution of tubewell growth by year of installation in one village, group by icddr,b survey years.

increase (+ four percent) in numbers using tubewells as their primary drinking water source. This reflects the increased density or crowding of tubewells in each *bari*, a shift in demand for household-level water points.

This progression shows the transition in household preferences for on– or in-premises water point infrastructure. Households' main drinking water infrastructure between the 1970s and 1990s reflects a period of pooled risk through shared water points under collective or public management models (Ahmed, 1998). By 2005, the reliance on multi-user shared tubewells located in nearby *baris* diminished and switched to a system of tubewells privately managed by a collective of multiple users, often located within individual *baris* as indicated by higher crowding metrics. Most recently, ownership has shifted to individual water points per household, diminishing reliance on shared tubewells within the same *bari*. The rates of this change vary between villages, but all show similar trajectories towards self-supply of between one and two households per tubewell, driven by private finance.

Beyond the aggregate growth rate and spatial patterns there are two additional trends which are not widely quantified or identified by previous studies but consistent across both Matlab and Khulna sites. The first is the specific role of private financing in this growth trajectory and the second is changing technology preferences increasingly shifted towards private deep tubewells, electric pumps, and in-premises handpump access.

The increased installation of individual and private water points is linked to changing ownership and management structures. In Matlab, ninety-five percent of the shallow and seventy-two percent of deep tubewells were reported as privately owned, financed, and managed. This allocates decision-making responsibility for water safety and the financial burden of repairs onto private households.

The second observation is a shift of private investment towards deep tubewells and electric pumps. The Matlab water audit identified that five percent of the tubewells (n = 173) were reported to be deeper than 150 m; of these ninety percent were installed after 2005 and two-thirds privately financed. Further, one in ten tubewells reported using electric pump systems (n = 343) instead of No. 6 handpumps. There was a further shift in household preferences: nearly sixty percent of identified No. 6 handpumps installed between 2006 and 2017 were located inside a built structure, defined as located within an enclosure of four walls. These observed trends suggest the market is in the early stages of changing preferences and redefining demand both for deep tubewells, defined as safer, and for the convenience of service delivery, directly into households.

3.2. Comparing regional growth trends

Previous blanket inventories provide an opportunity for insight into the regional comparability of the tubewell market's growth and variation of risk responses across different environmental and socioeconomic contexts. While recognizing that the majority of the population has obtained access to improved infrastructure, previous studies identified significant variation in tubewell density between southern coastal areas and other regions of the country (UNICEF, 1993). As seen in Fig. 6, the gaps between these regions have diminished significantly in the past seventeen years. The recent ratios range from 12.4 people per tubewell in Khulna in 2017; 6.7 in Matlab in 2017; 7.8 in Araihazar in 2013 as reported in Jamil et al. (2019) and 9.2 nationally using government estimates of rural population per tubewell ratios (BBS, 2015).

4. Contrasting risk drivers

The largest scale of growth was observed in the coastal belt. In this region, the presence of salinity in the upper shallow aquifer, which extends to a depth of 100 m, has high spatial variability similar to arsenic contamination (Hoque et al., 2019). Data from groundwater monitoring wells installed by Bangladesh Water Development Board

(BWDB) show that electrical conductivity in the shallow and main aquifers typically exceeds 8,000 μS/cm and 6,000 μS/cm during the dry pre-monsoon season (Zahid et al., 2013). In Bangladesh, the official permissible threshold level of salt in groundwater for the coastal districts is set at 1000 ppm or 1500 μS/cm, which is higher than the standard set at 600 ppm for the rest of the country (GoB, 2011).

In this coastal region, there is greater diversity of infrastructure options than in Matlab. In the areas adjacent to the Polder 29, households have a greater reliance on alternative sources such as pond sand filters, small piped schemes, rainwater and informal vendors for drinking water (Hoque and Hope, 2019). Yet, in the past decade, the number of tubewells in the Khulna study site quadrupled, with seventy-eight percent of the 2443 functional tubewells in 2018 being privately funded and thirty-seven percent being used for drinking, though one-third of the latter exceeded the salinity threshold of 1500 μS/cm, as shown in Fig. 7. In contrast, the tubewell numbers doubled in Matlab during the same period, with over seventy percent reportedly being used for drinking purposes. This reflects a critical difference in the type of groundwater concerns and risk responses, as unlike arsenic, salinity can be detected by taste. The rate of growth of privately financed tubewells remains similar to Matlab.

5. National growth beyond public provision models

This section estimates the national tubewell infrastructure stock using the 2017 population per tubewell ratios from Matlab and Polder 29 to determine the high and low bounds of estimated rural tubewell coverage. The selection of Matlab as the upper boundary and Khulna as the lower boundary corresponds to previous UNICEF and DPHE studies that identify these areas of the country as having the highest and the lowest infrastructure density (DPHE, 2016; UNICEF, 1993). Future studies would ideally refine these estimates with additional blanket inventories from around the country. The World Bank population data is linked to density of infrastructure to provide an estimated total tubewells in the rural areas for 2017 of between 8.8 million and 15.8 million tubewells. These are consistent with Jamil et al. (2019) estimates.

Projecting total national tubewell growth was difficult due to the absence of data on public and private tubewells in urban areas, an issue identified across multiple other studies. The urban level of households reporting use of tubewells as their main drinking water source in census data dropped from sixty-four percent in 2001 to forty-five percent in 2011 (BBS, 2012). However, this drop is not directly associated with the infrastructure investment in tubewells in use or dependency ratios, as many urban residents access multiple sources or report access to piped water while using privately funded handpumps to lift that water from underground reserve tanks. The World Bank reports that seventy percent of urban households in Bangladesh use tubewells as a water source (World Bank, 2018b). The 2011 Water Supply and Sanitation Sector Development Plan (WSSSDP) (GoB, 2011) only discusses urban areas covered by WASA areas, two city corporations, and 102 *Pourshavas* (small town municipalities) out of the total 308 *Pourshavas* (GoB, 2011, p. 18). The report provides a count of 172,077 tubewells (GoB, 2011, pp. 26–29). The report does not specify if this includes private tubewells. This estimate represents roughly half the total projected urban population, with an estimated 22.79 million people living in the remaining 206 *Pourshavas* as of 2011, areas where tubewell ownership and dependency is assumed to be higher. In order to determine a proxy estimate for the urban tubewell infrastructure stock, we use the 1998 National Policy for Safe Water Supply and Sanitation (NPSWSS) document (GoB, 1998), which sets the target ratio used to estimate the urban area tubewell infrastructure in relation to total population: fifty people per public water point and five people per private water point. To estimate total urban tubewells, the national public-to-private ratio from 2010 is applied to the World Bank's estimated urban population with the target number of people per tubewell. We recognize there are
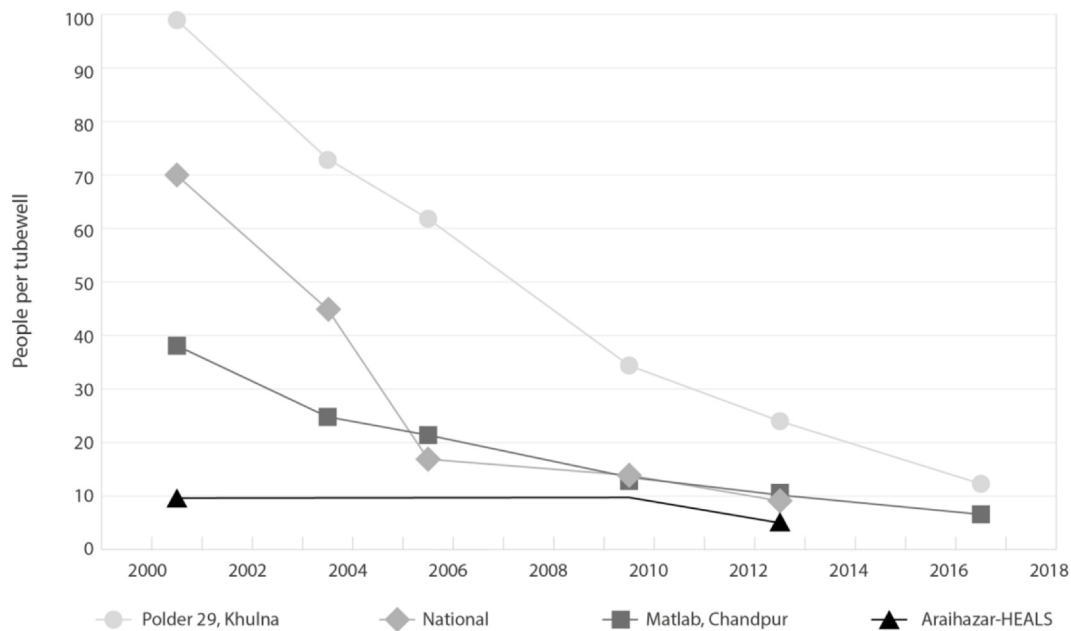
Fig. 6. Regional tubewell growth estimates through population-to-tubewell density across three blanket surveys and the national average. 2000–2017.
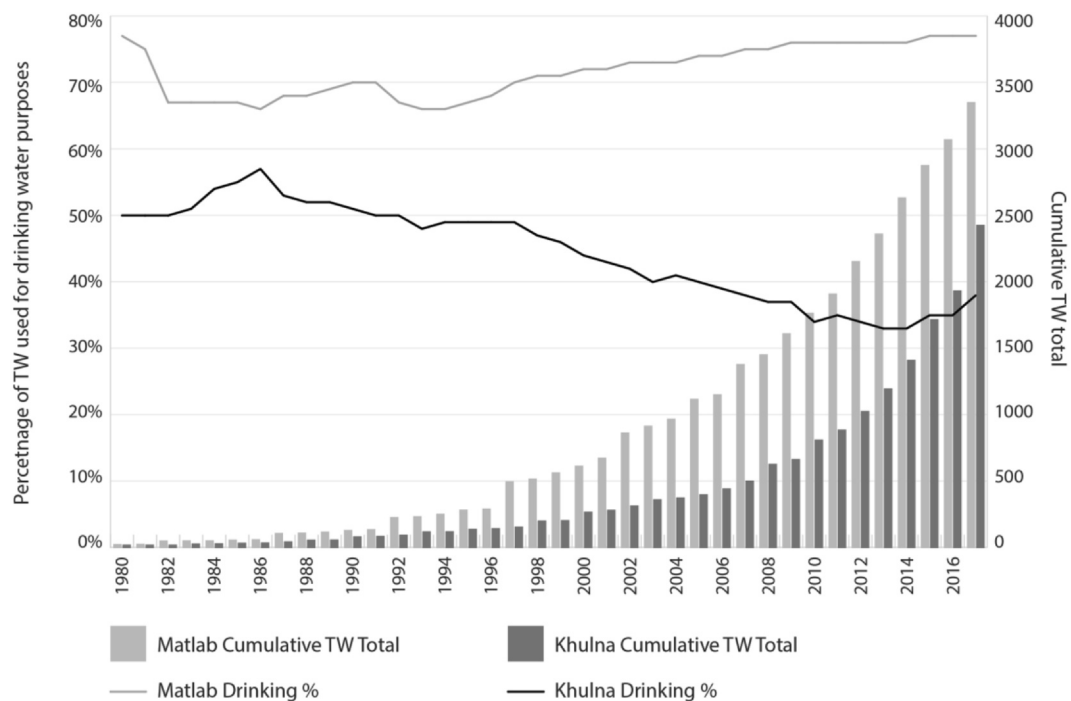

Fig. 7. Cumulative growth of tubewells as reported by age across two study sites and proportion of tubewells installed each individual year for drinking water purposes.

multiple limitations to this approach that can be modified in future studies.

These projections result with a total estimate of 2,666,446 tubewells in urban areas, although this is likely underestimated. When combined with the rural numbers, the projected national number of tubewells is between 11.5 million and 18.4 million tubewells, as shown in Fig. 8.

Within the total infrastructure, the ratio of public-to-private has changed significantly since 1972. Starting in 2012, the annual DPHE Circle Wise Water Source Status and Coverage Reports openly released data on publicly funded rural water infrastructure showing a steady, but lower, growth rate compared to private water points and the district-level spatial distribution of coverage (DPHE, 2017a). The

government estimates a national coverage rate of eighty-eight people per public water point in 2017 (DPHE, 2017a). This represents a total of 1,662,672 publicly-funded water points, of which ninety-one percent were reported as functional. These include 962,933 shallow tubewells, 335,538 deep tubewells, 292,936 Tara tubewells, 14,593 shallow shrouded tubewells, and the remaining split between pond sand filters, ringwells, rain water harvesting systems (DPHE, 2017a). Publicly provided water points are therefore estimated between eight percent and fifteen percent of the total non-piped infrastructure.
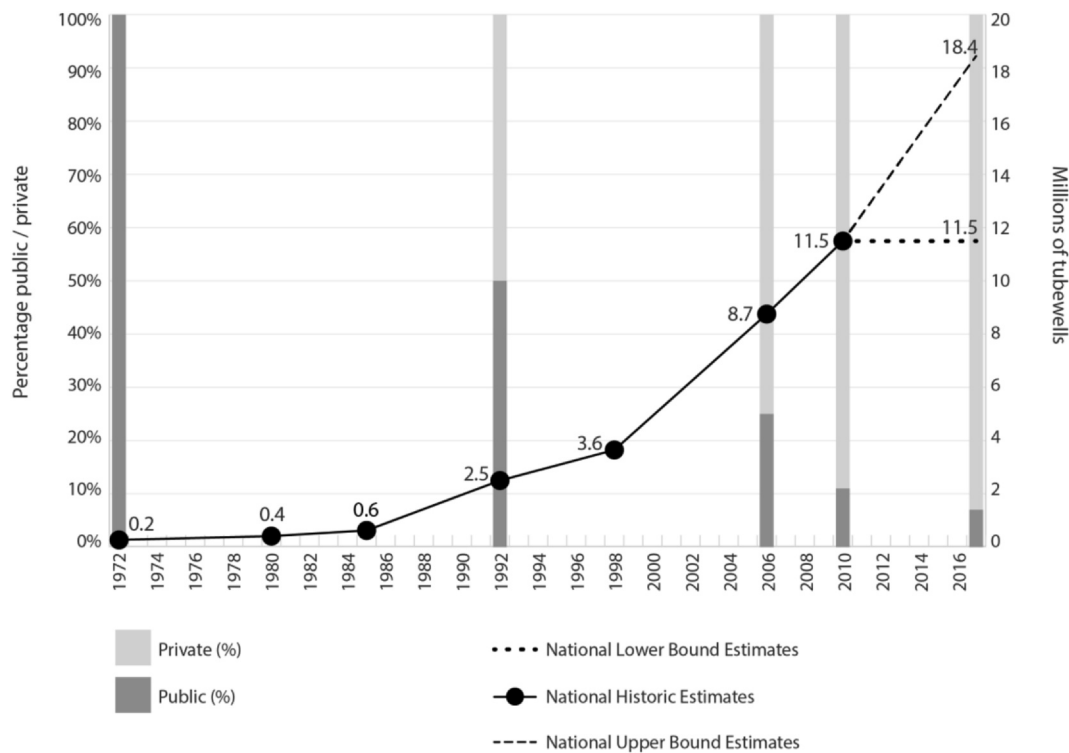
Fig. 8. Estimates of national tubewell growth from 1972 until 2016 and distribution of public versus private investment.

5.1. Infrastructure investment cost analysis

The owners and managers of drinking-water-only water points installed between 2006 and 2017 in Matlab were asked to identify the motivation for their private investments. Recognizing that the responses may be influenced by strategic recall bias, we find a clear majority installed new water points based on a preference for a private water supply (ninety-four percent). Investment driven by concerns for quality were far less frequent: well-switching away from an arsenic-contaminated source (ten percent) was less than well-switching for non-arsenic water-contamination concerns (twenty-four percent). A similarly low proportion of respondents identified financial motivations for investing, with twelve percent of households identifying an increase in income as motivation. More importantly, none of the households reported cost reduction of hardware as a motivation.

Using data from the Matlab water audit results and extracted from DPHE project proforma documents archives, Fig. 9 constructs a time series of reported public and private costs for installing shallow tubewells since the 1980s. These are adjusted for inflation to real 2017 prices using World Bank deflators and the World Bank Official exchange rate.

The results suggest households may not have directly identified a cost reduction but may have experienced a significant one when considering inflation and relative costs of other goods. The compiled historical price records suggest a seventy percent decrease in real price for private shallow tubewells installed between 1982 and 2017. In contrast, the figures suggest a forty-three percent fall in the real price of publicly provided shallow tubewells. While the inflation-adjusted real costs for both private and public installation costs have significantly and steadily decreased since the 1980s, the nominal prices have more than doubled. The nominal prices for publicly or privately installed shallow tubewells in the early 1980s were roughly equivalent, both around BDT 4500 (USD 184). By 2017, the costs for private installations had increased to BDT 10,123 (USD 121) while the costs for publicly installed systems increased to BDT 22,145 (USD 275). When these nominal prices are adjusted to real prices, there is a decrease from USD 444 in 1980 to USD

125 in 2017.

A similar pattern was observed for publicly funded deep tubewells, whereby nominal costs for private installers rose 168 percent from BDT 42,806 (USD 1383) in 1987 to over BDT 115,000 (USD 1454) in 2017. When adjusted to 2017 real inflation-adjusted costs however, the reverse occurred, with a drop from BDT 237,887 (USD 2,957) to BDT115,000 (USD 1,429). Although the nominal costs increased, the way consumers would perceive costs relative to other goods would feel less expensive in 2017 compared to earlier periods.

Beyond capital expenditure (CapEx) for installation, No.6 handpumps have annual operation and maintenance costs (OpEx) to factor into annual sector financial flows. The annual OpEx in Matlab show that costs vary by infrastructure age, with older handpumps reported as having higher average annual costs. The annual reported OpEx for both public and private shallow tubewells with handpumps installed in Matlab before 2007 averaged BDT 536 (USD 7.50) per year per handpump (n = 793). For those installed after 2007, there was a reported average yearly expenditure BDT 319 (USD 4.47) per year per handpump (n = 1,167). These reported expenditures identify households' willingness for recurring investment.

6. Private household investment as a major contributor to national sector finance

When considering both capital and operational expenses, there is a significant inflow of financial investment from the private sector not clearly accounted for in public planning models. Assuming a continued nine percent growth rate of tubewell infrastructure for rural areas based on the Matlab data, finding the projected national growth suggests a potential of between 793,005 to 1,420,307 new water points would have been installed in 2018, a year after this study. If most of the growth remained shallow tubewells, and the average private market costs remained around BDT 10,000 (USD 120), there is a potential of between USD 94.6 million (BDT 7.9 billion) to USD 170.1 million (BDT 14.2 billion) worth of new CapEx investments in 2018. Based on the infrastructure inventory count and age estimates, we estimate that the
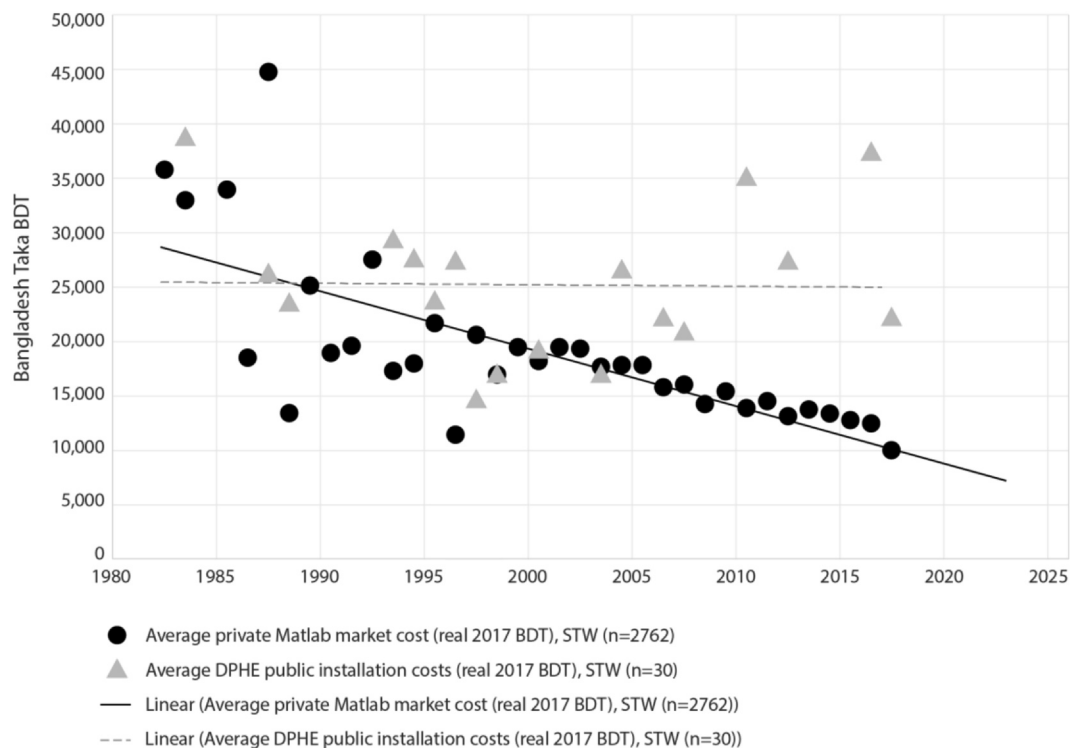
Fig. 9. Falling costs of private tubewells compared to public tubewells (adjusted to real prices, 2017) Sources: Matlab Water Audit and DPHE project proforma costs.

annual OpEx investment from rural households is upwards of BDT 6.9 billion or USD 83 million for 2018. The estimated total of both CapEx and OpEx investment for tubewells in 2018 is between USD 177.6 million and USD 253 million.

7. Discussion

Communities in rural Bangladesh have transitioned into a predominately self-supply and informal service delivery model. Privately financed and unregulated household infrastructure systems are being installed at rates and scales multiple times greater than formally regulated public systems. While this private investment has contributed to national success in achieving the Millennium Development Goals (MDG) of improved infrastructure access, it has not been accompanied by effective regulatory mechanisms or information systems to manage the risks related to untested water quality and unmonitored water supply. The estimated growth of tubewells at the three sites observed between 1970 and 2017 suggest that the total rural infrastructure stock is undercounted in national policy planning processes, thus potentially under-estimating population-level exposure to poor water quality. Neglecting to systematically quantify so great a development may appear remarkable, however it is not without precedent. In East Africa, the doubling rural per-capita use of hand-carried water, probably due to the shift from groundnut oil tins to plastic jerry-cans for carrying the water, was similarly missed until it was measured in a multi-country study (Thompson et al., 2001). These results further imply that the major shift in demand and service preferences could transform the rural water sector strategy by leveraging public–private investments in ways which enable pooled-risk through regulated service models.

The findings indicate that for every new publicly funded rural water point installed by the government between 2012 and 2017, an estimated forty-five new tubewells were privately installed. While the scale tubewell infrastructure has been roughly estimated in previous site-specific studies, including by Jamil et al. (2019), the national implications of the continued growth rate and privatized management has not been fully debated. The design of rural institutions, specifically how

risk-management responsibilities have been allocated, have not adapted in response to this infrastructure growth.

The drinking-water sector strategies have not recognized or been designed to manage the rapid shift of rural consumer preferences towards on- and in-premises private water infrastructure (GoB, 2011). This study suggests that the continued growth in number of built household structures for smaller family sizes is increasing the total infrastructure demand points. The focus on lower population growth rate in rural areas compared to urban areas distracts from the forty-three percent increase of households between the 2001 and 2011 censuses (BBS, 2012). This study shows that as the tubewells per household ratio is approaching a one-to-one ratio, it implies demand for tubewells could be approaching a point of market saturation.

Beyond increasing demand points, the preferences for in-premises services are also changing. The study identifies the important and unexpected increase of private household investment in deep tubewells. Policy strategies have long assumed that the costs of deep tubewells were unaffordable to households(GoB, 2004a). This study indicates rural families are in an early-stage shift of willingness to invest in deeper wells normally provided by government procurement and financing. Further, the increased consumer demand for on-premises access, delivered through mechanized pumping of groundwater, amplifies medium-term risk of increased extraction at a time that recharge rates and surface supply variability is increasing (DPHE, 2015a).

The findings from Matlab and Khulna field studies demonstrate the scale of household water supplies that operate outside any formal regulatory processes or monitoring structures. Since the public debate began around how to respond to widespread uncertainty about arsenic contamination of the groundwater system, scholars, including van Geen et al. (2002) and Pfaff et al. (2017), have documented the lack of testing at installation for private tubewells. The national blanket testing in the early 2000s provided a granular and robust inventory of distribution of arsenic exposure, however the more-than-doubling of untested new tubewells since 2006 leaves an even greater proportion of the population at risk of exposure. Government agencies have formalized regulations and processes for testing all public water points, while noting

continued questions of enforcement and reliability. However, these rules have not been extended to the rural private market, including drillers and private vendors.

This study reinforces previous analyses that suggest households do not directly invest in information on water quality (Ahmad et al., 2005; Balasubramanya et al., 2014; Barnwal et al., 2017). The Matlab survey did not identify any private household-level investment into testing, a cost which represents between one and three percent of the total shallow tubewell installation cost (Pfaff et al., 2017). This cost-point suggests it is not an issue of affordability relative to other investments but one perhaps of preference or low market availability of testing kits.

This study poses a contrasting question to previous research into risk-response behavioral dynamics. Previous studies explored households' behavioral responses to increasing awareness of arsenic and motivations for well-switching (Madajewicz et al., 2007). This study identified that fewer than a quarter of tubewell owners in Matlab recalled water quality concerns as the motivating reason for installing their private water point. Instead there is a clear preference to have an on- or in-premises water source. Despite Matlab being in an area with decades of NGO activities focused on cholera prevention activities and blanket arsenic-testing leaving visible red handpump handles as signifiers of risk, the continued exponential growth of untested tubewells suggests household preferences for self-supply is driving demand.

Previous studies offer insights into the costs of installing tubewells in specific years but not how they have changed between decades. Most of the previous cost analyses are based on a single-year estimate derived from NGO and government budgets, not from private market vendors (Balasubramanya et al., 2014; Jakariya et al., 2007; Jamil et al., 2019; Ravenscroft et al., 2014) By tracking the average costs of shallow tubewells on the private market and adjusting to 2017 real terms, this study suggests that consumers experienced the costs in 2016–2017 as nearly three times less expensive than those in the early 1980s. Although lower costs were not identified by consumers as a motivating factor for investment, perhaps because in nominal terms they have steadily increased, the installation costs are lower when considered in relation to other consumer goods. This reinforces early writing by Black (1990) stating that domestic production in the late 1980s and early 1990s enabled lower cost and larger-scale distribution of handpumps outside government supply chains.

These consumer preferences have significant policy and financial implications for a sector traditionally lacking reliable private capital flows. Previous studies in Bangladesh by Khan et al. (2014) show that households are, on average, willing to allocate five percent of their disposable income to safe tubewells and twenty percent to piped water schemes. This study seeks to expand beyond previous discussions of household-specific investment towards questions of what potential there is for sector-scale private capital to be leveraged as part of a blended finance model with public budgets.

This article estimates that for 2018 the total national rural private investment into tubewell infrastructure is upwards of USD 253 million including both capital and OpEx investments. This compares to 2017 GLAAS data on Bangladesh which reported a total WASH sector-wide expenditure of USD 1.157 billion across urban and rural geographies, of which USD 385 million was identified as investment by households, USD$336.8 million from government budgets, and USD 436.1 million from external aid (WHO, 2019). This implies that nearly two out of every ten dollars spent in the entire WASH sector is household spending on new tubewells. This study further suggests that the investment into tubewells could represent upwards of sixty-five percent of the total household investment in the WASH sector as estimated in GLAAS reports. These findings suggest that the level of household investment is increasing, however GLAAS data from 2014 and 2017 suggests that household investment has declined ten percent (USD 43 million) (WHO, 2019). Without more granular and standardized monitoring of private financial flows, the scale and potential for blended investment for overall sector needs remains uncertain.

The findings have advanced an argument that national development agenda progress, including achieving global MDG targets of improved infrastructure access, were enabled by the government investment programs launched in the 1970s and 1980s, but success was driven by private sector capital investment, specifically from households from all income and wealth brackets across multiple locations. The rural water-sector institutions have been designed around the risk logic of provision of access, a system which is undiscerning of the consequences of private assumption of responsibility. The role of private investment in Bangladesh has advanced national provision targets but has concealed the increased risks of exposure to unsafe water and supply variability. This has occurred while management responsibilities have been re-allocated outside direct government regulatory control or pooled risk mechanisms.

In order to achieve SDG 6.1, institutions are advancing beyond infrastructure provision logics to include risk logics focused on mitigation of uncertainty and managing the distribution of vulnerability and exposure. The rural drinking-water sector in Bangladesh provides insights into the process. The situation is now one of unregulated infrastructure with individualized responsibility for risk management. Although several policy reports suggest it is not feasible to retrospectively regulate private self-supply infrastructure (World Bank, 2018b, p. 22), recent articles by Jamil et al. (2019) argue there are cost-effective ways to optimize the spatial distribution of safe water points. Comparative country approaches also focus on licensing and standardization of drillers and the installation market. The tubewell density shown in the crowding metrics suggest spatial optimization models would offer further methods to facilitate consideration of the costs and benefits of small-scale distributed infrastructure networks, specifically as electric pumps are increasingly available to consumers. Further, the informal network of small vendors could be incentivized through a blended finance and performance-based service-delivery model. The household's preferences for in-premises water points indicates a potential willingness to pay. These new forms of shared risks underpin the redesign of institutional risk mentalities to more directly address risk management. The government could play the role of risk regulator without compromising the momentum of private finance in the sector by incentivizing households to pool finances and be accountable to the reduction of uncertainties of service delivery.

8. Conclusion

The SDG 6.1 of safely managed drinking water for all is substantively different from the MDGs of improved infrastructure access. Countries like Bangladesh saw their hard-won gains of achieving almost universal access to improved infrastructure being redefined because of gaps in safely and reliably managed services. While the success in achieving the MDG drinking-water goals was largely enabled by Bangladeshi families' private investment in new infrastructure, not government or donor funding, it is unlikely the SDGs will be achieved through the privatized risk-responsibility enabled by the self-supply systems.

With annual global financing needs for rural drinking water services estimated at USD 37.6 billion per year (Hutton and Varughese, 2016) the SDGs strategies could benefit from cost-sharing arrangements that harness this private demand in coordination with provision of public financial investments under forms of collective risk management. The government of Bangladesh has an opportunity to demonstrate this potential by harnessing the demand for improved services and infrastructure and leveraging the significant private investments by rural water-users. Early evidence from comparative rural public–private models is qualified by limited performance without accompanying institutional, regulatory, and financial architecture (Koehler et al., 2015). The challenge identified for Bangladesh in this article is to shift portions of the informal market towards regulated governance systems before the next phase of explosive growth of technology that could enable

greater extraction of either untested water quality or water from deep aquifers.

The exponential growth of self-supply tubewells in Bangladesh is a clear example of the need to transition the institutional design from an MDG focus on provision to an SDG design focused on managing for multiple intersecting risks. Since 1972 the Bangladeshi rural drinking water sector has shifted responsibility from being formally allocated to central government departments under a post-independence social contract that guaranteed infrastructure access, to the reverse, where individual families informally assume responsibility for managing their own infrastructure and services. This reallocation means that the responsibility for managing water quality and reliability remains unrecognized in formal rules of national law and policy. There is no current explicit regulation of private sector vendors or self-supply services.

The results of this study estimate that, after the completion of the national blanket arsenic testing in 2006, an estimated nine million tubewells were privately installed, the vast majority of which have not been tested for compliance with national water quality standards. With the rural tubewell market approaching a saturation point for households' demand, the high levels of private investment are not certain to continue.

The infrastructure audits indicate that consumer preference will shift towards higher-quality and more convenient services. Regulating risks of shallow tubewells is a major challenge but the shift towards electric pumps and in-premise piped systems offers an opportunity to align user demand and payments with regulated services. Shifting this risk-management approach would require the government and major stakeholders to rethink current and planned investments in ways that adopt a risk logic focused on managing the uncertainty of water quality and supply reliability. Risk theory provides a critical framework for interrogation of the institutional design of responsibility, decision processes, and the formal and informal rules.

The challenge remains that comparative global and national governance paradigms offer limited insights into how to regulate private actions in rural drinking water contexts, a sphere previously defined as outside the allocated authority of public action and government mandates. It is far more politically palatable to provide new infrastructure over regulating the use of existing infrastructures. The continued and growing concern for rural drinking water safety and reliability connects back to the limitations of modern institutions in managing risks when responsibility is allocated to individuals, not collectives. Without quantifying and debating institutional designs of shared responsibility through regulatory and monitoring approaches, the drinking water sector in the SDG era remains at risk of underperforming.

CRediT authorship contribution statement

Alex Fischer: Conceptualization, Methodology, Validation, Formal analysis, Investigation, Data curation, Writing - original draft, Writing - review & editing, Investigation, Visualization, Supervision, Project administration. **Rob Hope:** Conceptualization, Methodology, Writing - review & editing, Supervision, Funding acquisition. **Achut Manandhar:** Methodology, Formal analysis, Data curation, Writing - review & editing, Visualization. **Sonia Hoque:** Validation, Data curation, Writing - review & editing. **Tim Foster:** Validation, Methodology, Writing - review & editing. **Adnan Hakim:** Conceptualization, Methodology, Supervision, Writing - review & editing. **Md. Sirajul Islam:** Methodology, Writing - review & editing, Validation, Supervision, Project administration. **David Bradley:** Conceptualization, Methodology, Formal analysis, Writing - review & editing, Supervision.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

References

Ahmad, J., Goldar, B., Misra, S., 2005. Value of arsenic-free drinking water to rural households in Bangladesh. J. Environ. Manage. 74, 173–185. https://doi.org/10.1016/j.jenvman.2004.07.011.

Ahmed, S., 1998. Water management practices in rural and urban homes a case study from Bangladesh on ingestion of polluted water. Public Health 112, 317–321. https://doi.org/10.1016/S0033-3506(98)00262-5.

Alam, N., Ali, T., Razzaque, A., Rahman, M., Zahirul Haq, M., Saha, S.K., Ahmed, A., Sarder, A.M., Moinuddin Haider, M., Yunus, M., Nahar, Q., Kim Streatfield, P., 2017. Health and Demographic Surveillance System (HDSS) in Matlab, Bangladesh. Int. J. Epidemiol. 46, 809–816.

Atkins, P., Dunn, C., 2007. Environmental irony: summoning death in Bangladesh. Environ. Plann. 39, 2699–2715. https://doi.org/10.1068/a38123.

Bakker, K., 2013. Constructing "public" water: the World Bank, urban water supply, and the biopolitics of development. Environ. Plan. D Soc. Sp. 31, 280–300. https://doi.org/10.1068/d5111.

Bakker, K., 2002. From State to Market?: Water Mercantilización in Spain. Environ. Plan. A Econ. Sp. 34, 767–790. https://doi.org/10.1068/a3425.

Balasubramanya, S., Pfaff, A., Bennear, L., Tarozzi, A., Ahmed, K.M., Schoenfeld, A., van Geen, A., 2014. Evolution of households' responses to the groundwater arsenic crisis in Bangladesh: information on environmental health risks can have increasing behavioral impact over time. Environ. Dev. Econ. 19, 631–647. https://doi.org/10.1017/S1355770X13000612.

Barnwal, P., van Geen, A., von der Goltz, J., Singh, C.K., van Geen, A., Goltz, J.V., Der, 2017. Demand for environmental quality information and household response: evidence from well-water arsenic testing. J. Environ. Econ. Manage. 86, 160–192. https://doi.org/10.1016/j.jeem.2017.08.002.

Bartram, J., Brocklehurst, C., Fisher, M.B., Luyendijk, R., Hossain, R., Wardlaw, T., Gordon, B., 2014. Global monitoring of water supply and sanitation: history, methods and future challenges. Int. J. Environ. Res. Public Health 11, 8137–8165. https://doi.org/10.3390/ijerph110808137.

BBS, 2015. Multiple Indicator Cluster Survey 2012–2013. Socio-economic and demographic report Bangladesh Bureau of Statistics, Dhaka, Bangladesh.

BBS, 2012. Population and housing census 2011, Socio-economic and demographic report Bangladesh Bureau of Statistics, Dhaka, Bangladesh.

BBS, 2010. Multiple Indicator Cluster Survey 2009 Volume 1:, Technical Report. Bangladesh Bureau of Statistics, Dhaka, Bangladesh.

Beck, U., 2013. Why "class" is too soft a category to capture the explosiveness of social inequality at the beginning of the twenty-first century. Br. J. Sociol. 64, 63–74. https://doi.org/10.1111/1468-4446.12005.

Beck, U., 1992. Risk Society: Towards a New Modernity. SAGE Publications, Ltd, London, UK.

Bergkamp, L., 2017. The concept of risk society as a model for risk regulation–its hidden and not so hidden ambitions, side effects, and risks. J. Risk Res. 20, 1275–1291. https://doi.org/10.1080/13669877.2016.1153500.

BGS, 2001. Groundwater Quality : Bangladesh.

Black, M., 1990. From handpumps to health: the evolution of water and sanitation programmes in Bangladesh, India and Nigeria. United Nations Children's Fund (UNICEF), New York.

Bradley, D.J., Bartram, J.K., 2013. Domestic water and sanitation as water security: monitoring, concepts and strategy. Philos. Trans. R. Soc. London A Math. Phys. Eng. Sci. 371. https://doi.org/10.1098/rsta.2012.0420.

Butterworth, J., Sutton, S., Mekonta, L., 2013. Self-supply as a complementary water services delivery model in Ethiopia. Water Altern. 6, 405–423. https://doi.org/10.1111/j.1467-8721.2008.00557.x.

Caldwell, B.K., Caldwell, J.C., Mitra, S.N., Smith, W., 2003. Tubewells and arsenic in Bangladesh: challenging a public health success story. Int. J. Popul. Geogr. 9, 23–38. https://doi.org/10.1002/ijpg.271.

Dieter, C.A., Maupin, M.A., Caldwell, R.R., Harris, M.A., Ivahnenko, T.I., Lovelace, J.K., Barber, N.L., Linsey, K.S., 2018. Estimated use of water in the United States in 2015, U.S. Geological Survey Circular 1441. https://doi.org/https://doi.org/10.3133/cir1441.

DPHE, 2017a. Circlewise water source status and coverage- June 2017. Dep. Public Heal. Eng.

DPHE, 2017b. Unit Estimates for Village Water Supply Project of Department of Public Health Engineering.

DPHE, 2016. DPHE CircleWise Water Source Status and Coverage June 2016. Dep. Public Heal. Eng.

DPHE, 2015a. Feasibility Report on Rural Water Supply in Bangladesh, Department of Public Health Engineering. Dhaka, Bangladesh.

DPHE, 2015b. Guideline For Water Quality Testing and Sampling for Water Point Installed by DPHE. Dep. Public Heal. Eng.

Ewald, F., 1991. Insurance and Risk, in: Burchell, G., Gordon, C., Miller, P. (Eds.), The Foucault Effect: Studies in Governmentality. University of Chicago Press, Chicago, pp. 197–210.

Fischer, A., 2019. Constraining risk narratives: a multi-decadal media analysis of drinking

water insecurity in Bangladesh. Ann. Am. Assoc. Geogr. 109, 1433–1453. https://doi.org/10.1080/24694452.2019.1570840.

Flanagan, S.V., Zheng, Y., 2018. Comparative case study of legislative attempts to require private well testing in New Jersey and Maine. Environ. Sci. Policy 85, 40–46. https://doi.org/10.1016/j.envsci.2018.03.022.

Frink, D.W., Fannon, R.D., 1974. Progress Report: Field research and testing of a water hand pump for use in developing countries, Oct-Dec 1972.

Garrick, D.E., Schlager, E., De Stefano, L., Villamayor-Tomas, S., 2018. Managing the cascading risks of droughts: institutional adaptation in transboundary river basins. Earth's Futur. 6, 809–827. https://doi.org/10.1002/2018EF000823.

George, C.M., Graziano, J.H., Mey, J.L., van Geen, A., 2012a. Impact on arsenic exposure of a growing proportion of untested wells in Bangladesh. Environ. Health 11, 7. https://doi.org/10.1186/1476-069X-11-7.

George, C.M., Zheng, Y., Graziano, J.H., Rasul, S.B., Hossain, Z., Mey, J.L., 2012b. Evaluation of an arsenic test kit for rapid well screening in Bangladesh. Environ. Sci Technol 46. https://doi.org/10.1021/es300253p.

GoB, 2011. Sector Development Plan FY 2011-2025: Water Supply and Sanitation Sector in Bangladesh, Government of Bangladesh. Government of the People's Republic of Bangladesh, Local Government Division, Dhaka, Bangladesh.

GoB, 2004a. National Policy for Arsenic Mitigation, Government of Bangladesh. The Government of Bangladesh, Dhaka, Bangladesh.

GoB, 2004b. Sector Development Framework: Water Supply and Sanitation, Government of Bangladesh. Government of the People's Republic of Bangladesh, Local Government Division. https://doi.org/10.1016/B978-0-12-375678-7.00111-5.

GoB, 1998. National Policy for Safe Water Supply and Sanitation. Dhaka, Bangladesh.

Grönwall, J.T., Mulenga, M., Mcgranahan, G., 2010. Groundwater, self-supply and poor urban dwellers A review with case studies of Bangalore and Lusaka (No. 26), Human Settlements Working Paper.

Haider, M., Rahman, M., Alam, N., Ali, T., Saha, S., Mustafa, A., Alam, S., Barua, S., Streatfield, P., Nahar, Q., 2016. Household Socio-Economic Census 2014 (No. 132), Health and Demographic Surveillance System – Matlab, Scientific Report. Dhaka.

Hanchett, S., Monju, T., Akhter, K., Akhter, S., Islam, A., 2014. Water culture in South Asia: Bangladesh perspectives. Development Resources Press, Pasadena.

Haq, M.Z., Alam, S., Haider, M.M., Rahman, M.M., Mustafa, A.H.M.G., Saha, S.K., Barua, S., Alam, S.S., Ali, T., Alam, N., Nahar, Q., 2018. Registration of Health and Demographic Events 2016 (No. No. 138), Health and Demographic Surveillance System- Matlab, Scientific Report.

Harvey, P.A., Reed, R.A., 2007. Community-managed water supplies in Africa: sustainable or dispensable? Community Dev J. 42, 365–378.

Hope, R., 2015. Is community water management the community's choice? Implications for water and development policy in Africa. Water Policy 17, 664–678.

Hoque, S.F., Hope, R., 2019. Examining the Economics of Affordability through Water Diaries in Coastal Bangladesh. Water Econ. Policy. https://doi.org/10.1142/S2382624X19500115.

Hoque, S.F., Hope, R., Arif, S.T., Akhter, T., Naz, M., Salehin, M., 2019. A social-ecological analysis of drinking water risks in coastal Bangladesh. Sci. Total Environ. 679, 23–34. https://doi.org/10.1016/j.scitotenv.2019.04.359.

Hulsmann, A., Smeets, P., 2011. Towards a Guidance Document for the implementation of a Risk Assessment for small water supplies in the European Union Overview of best practices.

Hutton, G., Varughese, M., 2016. The Costs of Meeting the 2030 Sustainable Development Goal Targets on Drinking Water, Sanitation, and Hygiene - Summary Report.

Jakariya, M., von Brömssen, M., Jacks, G., Chowdhury, A.M.R., Ahmed, K.M., Bhattacharya, P., 2007. Searching for a sustainable arsenic mitigation strategy in Bangladesh: experience from two upazilas. Int. J. Environ. Pollut. 31, 415–430.

Jamil, N.B., Feng, H., Ahmed, K.M., Choudhury, I., Barnwal, P., van Geen, A., 2019. Effectiveness of Different Approaches to Arsenic Mitigation over 18 Years in Araihazar, Bangladesh: Implications for National Policy. Environ. Sci. Technol. https://doi.org/10.1021/acs.est.9b01375.

Johnston, R., Hug, S.J., Inauen, J., Khan, N.I., Mosler, H.-J.J., Yang, H., 2014. Enhancing arsenic mitigation in Bangladesh: findings from institutional, psychological, and technical investigations. Sci. Total Environ. 488–489, 477–483. https://doi.org/10.1016/j.scitotenv.2013.11.143.

Johnston, R., Sarker, M.H., 2007. Arsenic mitigation in Bangladesh: national screening data and case studies in three upazilas. J. Environ. Sci. Health. A. Tox. Hazard. Subst. Environ. Eng. 42, 1889–1896. https://doi.org/10.1080/10934520701567155.

Kabir, A., Howard, G., 2007. Sustainability of arsenic mitigation in Bangladesh: Results of a functionality survey. Int. J. Environ. Health Res. 17, 207–218. https://doi.org/10.1080/09603120701254904.

Khan, N.I., Brouwer, R., Yang, H., 2014. Household's willingness to pay for arsenic safe drinking water in Bangladesh. J. Environ. Manage. 143, 151–161. https://doi.org/10.1016/j.jenvman.2014.04.018.

Koehler, J., Rayner, S., Katuva, J., Thomson, P., Hope, R., 2018. A cultural theory of drinking water risks, values and institutional change. Glob. Environ. Change 50, 268–277. https://doi.org/10.1016/j.gloenvcha.2018.03.006.

Koehler, J., Thomson, P., Hope, R., 2015. Pump-priming payments for sustainable water services in rural Africa. World Dev. 74, 397–411. https://doi.org/10.1016/j.worlddev.2015.05.020.

Korfali, S.I., Jurdi, M., 2007. Assessment of domestic water quality: case study, Beirut. Lebanon. Environ. Monit. Assess. 135, 241–251. https://doi.org/10.1007/s10661-007-9646-x.

Lash, S., 2003. Reflexivity as non-linearity. theory. Cult. Soc. 20, 49–57. https://doi.org/10.1177/0263276403020002003.

Lloyd, M., 1967. Mean crowding. J. Anim. Ecol. 36, 1–30.

Lockwood, H., Mansour, G., Smits, S., Smets, S., 2017. Global study on sustainable service delivery models for rural water: evidence from 16 countries. In: 40th WEDC Int. Conf.

Loughborough, UK, pp. 1–7.

MacCarthy, M.F., Annis, J.E., Mihelcic, J.R., 2013. Unsubsidised self-supply in eastern Madagascar. Water Altern. 6, 424–438.

Madajewicz, M., Pfaff, A., van Geen, A., Graziano, J., Hussein, I., Momotaj, H., Sylvi, R., Ahsan, H., 2007. Can information alone change behavior? Response to arsenic contamination of groundwater in Bangladesh. J. Dev. Econ. 84, 731–754. https://doi.org/10.1016/j.jdeveco.2006.12.002.

Matten, D., 2004. The impact of the risk society thesis on environmental politics and management in a globalizing economy – principles, proficiency, perspectives. J. Risk Res. 7, 377–398. https://doi.org/10.1080/1366987042000208338.

Mobarak, A., van Geen, A., 2019. The politics of poison Elite capture of clean water in Bangladesh The Politics of Poison: Elite Capture of Clean Water in Bangladesh.

Moriarty, P., Smits, S., Butterworth, J., Franceys, R., 2013. Trends in rural water supply: Towards a service delivery approach. Water Altern. 6, 329–349.

Mythen, G., 2005. From 'Goods' to 'Bads'? Revisiting the Political Economy of Risk. Sociol. Res. Online 10, 1–13. https://doi.org/10.5153/sro.1140.

Nahar, L., 2007. 2005 Socio-Economic Census (No. 96), Health and Demographic Surveillance System – Matlab, Scientific Report. Dhaka, Bangladesh.

North, D.C., 1990. Institutions, Institutional Change and Economic Performance, Political economy of institutions and decisions. Cambridge University Press, Cambridge [England] ; New York.

Olschewski, A., 2016. Supported Self-supply – learning from 15 years of experiences. In: 7th RWSN Forum "Water for Everyone." pp. 1–14.

Ostrom, E., 2010a. The institutional analysis and development framework and the commons. Cornell Law Rev. 95, 807–815.

Ostrom, E., 2010b. Polycentric systems for coping with collective action and global environmental change. Glob. Environ. Change 20, 550–557. https://doi.org/10.1016/j.gloenvcha.2010.07.004.

Pfaff, A., Schoenfeld Walker, A., Ahmed, K.M., van Geen, A., 2017. Reduction in exposure to arsenic from drinking well-water in Bangladesh limited by insufficient testing and awareness. J. Water Sanit. Hyg. Dev. 7, 331–339. https://doi.org/10.2166/washdev.2017.136.

Rahman, R., Salehin, M., 2013. Flood Risks and Reduction Approaches in Bangladesh. In: Shaw, R. (Ed.), Disaster Risk Reduction Approaches in Bangaldesh. Springer, Japan, pp. 65–90.

Rasmussen, C.E., 2004. Gaussian Processes in Machine Learning BT - Advanced Lectures on Machine Learning: ML Summer Schools 2003, Canberra, Australia, February 2 - 14, 2003, Tübingen, Germany, August 4 - 16, 2003, Revised Lectures. In: Bousquet, O., von Luxburg, U., Rätsch, G. (Eds.), Springer Berlin Heidelberg, Berlin, Heidelberg, pp. 63–71. https://doi.org/10.1007/978-3-540-28650-9_4.

Ravenscroft, P., Kabir, A., Hakim, S.A.I., Ibrahim, A.K.M., Ghosh, S.K., Rahman, S., Akter, F., Sattar, A., 2014. Effectiveness of public rural waterpoints in Bangladesh with special reference to arsenic mitigation. J. WaterSanit. Hyg. Dev. 4, 545–562. https://doi.org/10.2166/washdev.2014.038.

Rayner, S., 2007. The rise of risk and the decline of politics. Environ. Hazards 7, 165–172. https://doi.org/10.1016/j.envhaz.2007.05.003.

Razzaque, A., Nahar, L., Sarder, A.M., van Ginneken, J., Shaikh, M.A.K., 1996. 1996 Socio-Economic Census (No. 83), Demographic Surveillance System- Matlab, Scientific Report. Dhaka, Bangladesh.

Saladin, M., 2016. Rainwater Harvesting in Thailand : Learning from the World Champions.

Smedley, P.L., Kinniburgh, D.G., 2002. A review of the source , behaviour and distribution of arsenic in natural waters 17, 517–568.

Smith, A.H., Lignas, E., Rahman, M., 2000. Contamination of drinking-water by arsenic in Bangladesh: a public health emergency. Bull. World Health Organ. 78.

Sultana, F., 2013. Water, technology, and development: transformations of development technonatures in changing waterscapes. Environ. Plan. D Soc. Sp. 31, 337–353. https://doi.org/10.1068/d20010.

Sultana, F., 2012. Producing contaminated citizens: toward a nature-society geography of health and well-being. Ann. Assoc. Am. Geogr. 102, 1165–1172. https://doi.org/10.1080/00045608.2012.671127.

Sultana, F., 2009. Community and participation in water resources management: gendering and naturing development debates from Bangladesh. Trans. Inst. Br. Geogr. 34, 346–363. https://doi.org/10.1111/j.1475-5661.2009.00345.x.

Sutton, S., 2017. Trends in sub-Saharan rural water supply and the essential inclusion of Self-supply to achieve 2030 SDG targets. Waterlines 36, 339–357. https://doi.org/10.3362/1756-3488.17-00013.

Sutton, S., 2004. Self Supply: A Fresh Approach to Water for Rural Populations, Water and Sanitation Programme Fieldnote.

Thompson, J., Porras, I., Tumwine, J., Mujwahuzi, M., Katui-Katua, M., Johnston, N., Wood, L., Development, I.I. for E. and, 2001. Drawers of Water II. International Institute for Environment and Development, London.

UNICEF/WHO, 2019. Progress on household drinking water, sanitation and hygiene 2000-2017: Special focus on inequalities, Launch version July 12 Main report Progress on Drinking Water , Sanitation and Hygiene. New York.

UNICEF, 1993. The Status of Rural Water Supply and Sanitation.

van Geen, A., 2008. Arsenic meets dense populations. Nat. Geosci. 1, 494–498. https://doi.org/10.1038/ngeo268.

van Geen, A., Ahmed, E.B., Pitcher, L., Mey, J., Ahsan, H., Graziano, J., Ahmed, K.M., 2014. Comparison of two blanket surveys of arsenic in tubewells conducted 12 years apart in a 25 km 2 area of Bangladesh. Sci. Total Environ. 488–489, 484–492. https://doi.org/10.1016/j.scitotenv.2013.12.049.

van Geen, A., Ahmed, K.M., Ahmed, E.B., Choudhury, I., Mozumder, M.R., Bostick, B.C., Mailloux, B.J., 2016. Inequitable allocation of deep community wells for reducing arsenic exposure in Bangladesh. J. WaterSanit. Hyg. Dev. 06, 142–150. https://doi.org/10.2166/washdev.2015.115.

van Geen, A., Allan, H., Ratan, K., 2002. Promotion of well-switching to mitigate the current arsenic crisis in Bangladesh. Bull. World Health Organ. 80, 732–737.

van Geen, A., Zheng, Y., Versteeg, R., Stute, M., Horneman, A., Dhar, R., Steckler, M., Gelman, A., Small, C., Ahsan, H., Graziano, J.H., Hussain, I., Ahmed, K.M., 2003. Spatial variability of arsenic in 6000 tube wells in a 25 km 2 area of Bangladesh. Water Resour. Res. 39, 1140. https://doi.org/10.1029/2002WR001617.

Wade, M.J., Fitzpatrick, C.L., Lively, C.M., 2018. 50-year anniversary of Lloyd's "mean crowding": Ideas on patchy distributions. J. Anim. Ecol. 87, 1221–1226. https://doi.org/10.1111/1365-2656.12854.

WHO, 2019. National systems to support drinking-water, sanitation and hygiene: global status report 2019. UN-Water global analysis and assessment of sanitation and drinking-water (GLAAS).

WHO, 2017. UN-Water global analysis and assessment of sanitation and drinking-water (GLAAS) 2017 report: financing universal water, sanitation and hygiene under the sustainable development goals. Geneva.

WHO, 2014. UN-Water Global Analysis and Assessment of Sanitation and Drinking-Water (GLAAS) 2014: Investing in Water & Sanitation: Increasing access, reducing inequalities. https://doi.org/9789241508087.

WHO, 2012. UN-Water Global Analysis and Assessment of Sanitation and Drinking-Water (GLAAS) 2012 Report: the challenge of extending and sustaining services.

World Bank, 2018a. World Development Indicators: Rural Population [WWW Document]. URL databank.worldbank.org/data/ (accessed 8.20.10).

World Bank, 2018b. Promising Progress, Promising Progress, WASH Poverty Diagnostic. World Bank. https://doi.org/10.1596/29450.

Yasmin, T., Farrelly, M.A., Rogers, B.C., 2018. Evolution of water governance in Bangladesh: an urban perspective. World Dev. 109, 386–400. https://doi.org/10.1016/j.worlddev.2018.05.003.

Zahid, A., Jahan, K., Ali, N., Islam, M.K., Rahman, A., 2013. Distribution of Groundwater Salinity and its Seasonal Variability in Coastal Aquifers of Bengal Delta, in: Zahid, A., Hassan, M., Islam, R., Samad, Q., Khan, M. (Eds.), Impact of Climate Change on Socio-Economic Conditions of Bangladesh. Dhaka, Bangladesh.