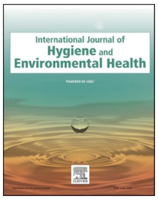
International Journal of Hygiene and Environmental Health 233 (2021) 113669

Contents lists available at ScienceDirect

International Journal of Hygiene and Environmental Health

journal homepage: www.elsevier.com/locate/ijheh



Modelling faecal pathogen flows and health risks in urban Bangladesh: Implications for sanitation decision making

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ARTICLE INFO

Keywords:
Faecal pathogens
On-site sanitation
Septic tanks
Systems modelling
Bangladesh
Sustainable development goal 6

ABSTRACT

Faecal-oral infections are a major component of the disease burden in low-income contexts, with inadequate sanitation seen as a contributing factor. However, demonstrating health effects of sanitation interventions – particularly in urban areas – has proved challenging and there is limited empirical evidence to support sanitation decisions that maximise health gains. This study aimed to develop, apply and validate a systems modelling approach to inform sanitation infrastructure and service decision-making in urban environments by examining enteric pathogen inputs, transport and reduction by various sanitation systems, and estimating corresponding exposure and public health impacts. The health effects of eight sanitation options were assessed in a low-income area in Dhaka, Bangladesh, with a focus on five target pathogens (*Shigella*, *Vibrio cholerae*, *Salmonella* Typhi, norovirus GII and *Giardia*). Relative to the sanitation base case in the study site (24% septic tanks, 5% holding tanks and 71% toilets discharging directly to open drains), comprehensive coverage of septic tanks was estimated to reduce the disease burden in disability-adjusted life years (DALYs) by 48–72%, while complete coverage of communal scale anaerobic baffled reactors was estimated to reduce DALYs by 67–81%. Despite these improvements, a concerning health risk persists with these systems as a result of effluent discharge to open drains, particularly when the systems are poorly managed. Other sanitation options, including use of constructed wetlands and small bore sewerage, demonstrated further reductions in local health risk, though several still exported pathogens into neighbouring areas, simply transferring risk to downstream communities. The study revealed sensitivity to and a requirement for further evidence on log reduction values for different sanitation systems under varying performance conditions, pathogen flows under flooding conditions as well as pathogen shedding and human exposure in typical low-income urban settings. Notwithstanding variability and uncertainties in input parameters, systems modelling can be a feasible and customisable approach to consider the relative health impact of different sanitation options across various contexts, and stands as a valuable tool to guide urban sanitation decision-making.

1. Introduction

Faecal-oral infections are a major component of disease burden in low-income contexts. Diarrhoeal disease linked to inadequate sanitation constitutes one percent of the total global disease burden and results in more than 400,000 deaths each year (Prüss-Ustün et al., 2019). Accordingly, sanitation investments by governments and development partners are primarily justified by health improvements. Despite strong theoretical grounds for supposing that sanitation improvements are necessary for health (Cumming et al., 2019), recent major studies

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https://doi.org/10.1016/j.ijheh.2020.113669
Received 16 April 2020; Received in revised form 19 November 2020; Accepted 24 November 2020
Available online 9 February 2021

evaluating the health effects of sanitation interventions have produced inconsistent results (Null et al., 2018; Luby et al., 2018; Humphrey et al., 2019). Concomitantly, in low-income urban contexts there is limited empirical evidence on how best to design interventions and direct available resources to maximise health gains (Mills et al., 2018). This situation compromises the effectiveness of sanitation programming. Significant funds are currently invested to install on-site sanitation systems and to improve emptying and sludge management services in densely populated low-income areas, however the fate and transport of liquid effluent discharging from these systems is often overlooked, despite the related potential health risk (Mitchell et al., 2016). There is therefore a growing recognition of the need to better understand and compare the health impacts of different sanitation options (WHO, 2018), particularly on-site systems (such as septic tanks or small-scale systems), which are given an important place in the emerging city-wide inclusive sanitation (CWIS) approach (Schrecongost et al., 2020).

The imperative to ensure sanitation decisions minimise public health risks in low-income urban settlements will only grow in importance. Globally, more than 600 million people living in urban areas still lack even basic sanitation, and the total urban population is expanding by around 80 million every year (WHO/UNICEF 2019), making it challenging to keep up with sanitation infrastructure requirements. Moreover, the Sustainable Development Goal target that seeks ‘universal access to safely-managed sanitation services’ represents an elevated ambition compared with previous global targets, and necessitates safe management of excreta along the entire sanitation chain from containment to final re-use or disposal (WHO/UNICEF 2018).

The sanitation challenges in urban Bangladesh are emblematic of those faced by many large cities in low- and middle-income countries. Dhaka, the focus of the study described in this paper, has all but eliminated open defecation, including from slums and informal settlements (BBS, 2015). Yet, despite significant investment in sanitation infrastructure, it is estimated that more than 99% of the city’s wastewater is discharged without treatment into drains and waterways (Ross et al., 2016). Around one-third of Dhaka’s population use pour-flush toilets that discharge directly to drains, with this number increasing to 40–50% amongst the poorest two quintiles (WSUP 2018). A further 20% use on-site septic tanks, which commonly discharge effluent into drains and surface water bodies, since dense clay soils prevent infiltration (DWASA, 2011). This situation poses a concerning health risk, as signified by high levels of faecal indicator bacteria in Dhaka’s open drains (Amin et al., 2019), as well as country-level disease burden estimates that attribute more than 23,000 deaths each year to inadequate water, sanitation and hygiene (Prüss-Ustün et al., 2019).

Global efforts to understand health effects of sanitation interventions have largely been in the form of impact evaluations. While on the whole such studies suggest sanitation improvements can reduce the risk of diarrhoeal disease (Norman et al., 2010; Wolf et al., 2018), individual study results vary. Randomized control trials (RCTs) and other quasi-experimental approaches applied to assess sanitation impacts have robust internal validity, but also come with limitations, particularly for densely populated urban areas. They produce results that may not be generalisable, are costly and struggle to disentangle complex urban sanitation contexts, in which people come into contact with excreta from multiple sources in multiple ways (Wang et al., 2017; Amin et al., 2019). Most recently, the inconsistent results produced by rigorous RCTs in rural Bangladesh, Kenya and Zimbabwe raised questions about the effect of sanitation investments, even when made alongside improvements in water supply and hygiene behaviours (Stewart et al., 2018; Luby et al., 2018; Humphrey et al., 2019). The equivocal results to date have been attributed to a range of factors, one of which is the importance of multiple and context-dependent pathways for the transmission of faecal pathogens (Cumming et al., 2019), calling our attention to examine these pathways in greater depth.

Various tools and approaches have emerged in recent years that characterise how excreta and its associated microorganisms move through the urban environment, though they do not estimate resultant health risks. Shit Flow Diagrams (SFDs) are increasingly used to illustrate the flow of both safely and unsafely managed faecal sludge and wastewater effluent in urban areas for advocacy purposes (Peal et al., 2014). SFDs highlight the scale of potential exposure risks at a city-wide scale. Limited understanding of pathogen removal by on-site systems – which are widespread in low-income urban areas – makes it difficult to move from a SFD to a more detailed risk-based sanitation planning process. The extensive variation in on-site sanitation technology, construction, management, discharge arrangements, and local context (e.g. soil type, groundwater depth etc.) results in different measured and predicted performance outcomes, and the potential for pathogen transmission and exposure via multiple pathways is not well understood. The recent SaniPath studies and exposure assessment tool examined a range of exposure pathways, but this approach did not extend to estimating health impacts of different sanitation interventions (Robb et al., 2017; Wang et al., 2017; Raj et al., 2020). Other urban sanitation planning tools exist, but a recent review noted their limitations in addressing this complexity, with some tools omitting health considerations altogether (Mills et al., 2018). What has been lacking, therefore, is an approach that can link sanitation options and scenarios with resultant health risks.

Systems modelling presents an alternative, but complementary, method for examining and predicting the context-specific health impacts of sanitation interventions (Mills et al., 2018). Systems modelling has been widely used to analyse and understand a range of complex cause-effect systems, including in fields of environmental health, public health and water management (Benedetti et al., 2013; Carey et al., 2015; Currie et al., 2018). Applying a systems modelling approach to sanitation interventions has the potential to link understanding of the magnitude of pathogen containment or release into the environment by various sanitation systems with potential for exposure and likelihood of illness, which in turn can better inform sanitation investments. Modelling the transport and fate of pathogens in water bodies and waterways has been carried out in a variety of contexts and scales (Haydon and Deletic 2006; Whitehead et al., 2016; Kroeze et al., 2016). Likewise, there is an emerging body of literature assessing exposure and risks associated with faecal pathogens in low-income urban areas, particularly through quantitative microbial risk assessment (QMRA) (Labite et al., 2010; Yapo et al., 2014; Katukiza et al., 2014). However, the extent to which these assessments have measured enteric pathogen concentrations in the environment has been limited. Combining these two approaches with sufficient, high-quality primary data, has been identified as a key priority for understanding health risks associated with faecal pathogens (Hofstra et al., 2019). Such systems modelling has the potential to support analysis of complexity, characterise causal pathways that might otherwise be difficult or costly to measure, identify key inter-relationships and evidence gaps, and evaluate multiple scenarios for sanitation service provision – including those that are novel or counter-intuitive (Mills et al., 2018).

In the context of a low-income neighbourhood in Dhaka, this paper applies a systems modelling approach to understand the local sanitation situation and its impact on health risks in relation to exposure to open drains. The specific objectives of this case study were to: (a) develop, apply and validate a systems modelling approach to inform sanitation infrastructure and management decision-making that weighs public health impacts; and (b) identify key gaps in the evidence base required to model pathogen flows and related health risks in urban environments. In so doing, the study presents a site-specific application of a conceptual approach outlined by Mills et al. (2018),[1] and seeks to address the lack of available tools that link sanitation options with health risks. The modelling focused on the flow of, and exposure to, five faecal pathogens and a faecal indicator bacteria in a bounded case study site, and explored a range of sanitation options and management scenarios. The intention of the work was to provide a health-based metric that can be used alongside other considerations – such as cost, environmental impacts, and potential for resource recovery – in a multi-criteria decision support tool.

2. Methods

2.1. *Study site*

The study focussed on a low-income neighbourhood in Mirpur, Dhaka. The site selection was based on five key criteria. Three criteria (high population density, low-income status, likely high exposure via open drains and surface water) were applied to ensure the study site was broadly representative of settings commonly found in low-income urban neighbourhoods in low- and middle-income countries. Two criteria (variation in sanitation technologies, and relatively uncomplicated hydraulic/drainage characteristics) were applied so the modelling could generate and compare results across different sanitation technologies, whilst simplifying hydraulic aspects of the modelling.

The chosen study site – which consisted of four parallel roads – was a contained catchment with no (or very limited) inflows of wastewater or excreta from elsewhere (Fig. 1). The eastern side was bounded by a main road, which represented the uppermost part of the catchment, while the western side of the site was bounded by a canal, which received all the wastewater via open drains from the four roads. The site's four roads were configured in an east-west direction (160–250m in length), with open drains running parallel along both sides of each road (average drain width: 30 cm). In two of the roads (Road B and Road D), the parallel drains converged into a single drain before the point of discharge into the receiving canal. Although slightly separated from the other roads, Road A was included in the study site due to its high coverage of septic tanks, which allowed for a useful point of comparison to Roads B, C and D. The road immediately north of Road D was initially included in the study due to the presence of two anaerobic baffled reactors (ABRs); however, this road had to be excluded because the ABRs were decommissioned just prior to the commencement of sampling. Instead, data collection for ABRs was conducted at a separate site (see Amin et al., 2020 for further information).

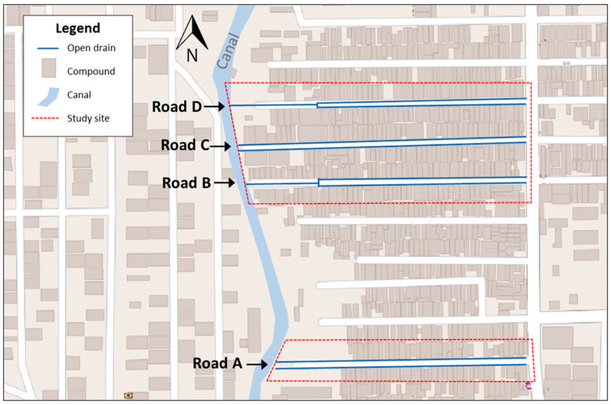

Fig. 1. Layout of four roads in the study site (bounded by dotted red perimeter). (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

The study site had a population of 4792 people living in 1493 households, which in turn were grouped into 176 compounds (Table 1) (80% government-owned compounds, 20% privately-owned). Most compounds were situated within 1 m of an open drain. Each compound included a sanitation facility. Pour-flush toilets discharging directly to drain (with no containment) were the most common sanitation facility, used by 71% of the population. One quarter of the population used pour-flush toilets with a septic tank[2] (all of which then discharged effluent into the drain), with road-wide coverage of septic tanks[3] ranging from 3% to 79% of the population. Almost all compounds had a metered water supply from Dhaka's municipal utility.

2.2. *Infrastructure assessment and household survey*

A household survey and infrastructure census were conducted to capture data for key modelling inputs. The infrastructure census identified and mapped all sanitation, water, and wastewater infrastructure in the study site, included a comprehensive enumeration of compounds and population, captured water meter data, and measured flowrates in each open drain. The household survey covered 350 households (approximately 30% of the population in the study site) and collected information about water and sanitation facilities and practices, exposure behaviours, and prevalence of diarrhoea. To estimate exposure frequency, each respondent was asked how many times they and their children came into contact with a drain in the previous week. This allowed calculation of separate exposure event distributions for adults and children (see Fig. S3 in Supplementary Material).

Table 1
Characteristics of study site.

	Road A	Road B	Road C	Road D	Total
Population	970	1194	1277	1351	4792
Number of households	357	344	404	388	1493
Number of compounds	34	43	48	51	176
Average household size (persons)	2.7	3.5	3.2	3.5	3.2
Population by age group					
Adults (>18 years) (%)	62	62	62	62	62
Children (5–17 years) (%)	24	25	24	24	24
Children (<5 years) (%)	14	14	14	14	14
Sanitation facilities					
Septic tanks (tank only, effluent to drain) (%)	79	11	3	15	24
Toilet direct to drain (%)	17	89	88	79	71
Other (%)[a]	4	0	9	6	5
Drinking water piped to compound (%)	100	89	98	100	97
% of compounds within 1m of open drain (%)	93	65	85	62	75
Avg. drain width (cm)	28	31	30	31	30

Source: Household survey and infrastructure census (described below). [a]'Other' sanitation facilities were single-chamber holding tanks or tanks of unknown type.

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[1] Although it is possible to define a tolerable health risk and then work backwards to identify acceptable sanitation solutions, the purpose of this study was to compare the predicted impact of different technologies and management strategies in a low-income urban context where physical, financial and social constraints present a range of trade-offs and require incremental improvements.

[2] These septic tanks were constructed by non-governmental organisations and consisted of two chambers with a baffle between the chambers.

[3] 'Road-wide' coverage of septic tanks refers to the percentage of the population in a particular road using sanitation facilities with a septic tank.

2.3. Choice of pathogens and transmission pathways

After reviewing enteric infection literature focussed on Dhaka and consulting an expert advisory panel of sanitation and public health specialists (see Acknowledgements), five reference pathogens were chosen: three bacterial pathogens (*Shigella, Vibrio cholerae, Salmonella* Typhi), one virus (norovirus GII), and one protozoa (*Giardia*).[4] Additionally, *E. coli* were selected as faecal indicator bacteria (FIB). Given the different patterns of removal, inactivation and survival of different pathogen types in sanitation systems and environmental compartments, representatives from each of the three pathogen groups were sought. When selecting reference pathogens, consideration was given to local infection prevalence, data availability to support modelling (including fate and transport, and dose-response), availability of sensitive and specific methods for detection in environmental samples and the likely importance of sanitation for control. Studies of diarrhoea patients in Dhaka have found a prevalence of infection of around 16% for Norovirus GII (Rahman et al., 2016), 12% for *V. cholerae* (Das et al., 2013), 8% for *Giardia* (Haque et al., 2005), and 3% for *Shigella* (Das et al., 2013); while the annual incidence rate for typhoid has been estimated at around 2 per 1000 persons (Naheed et al., 2010).

The study focussed on exposure to faecal pathogens via open drain water as a primary transmission pathway, as this was a key pathway relevant to sanitation in the study site, as in other low-income areas of Dhaka (Amin et al., 2019). Open drains were ubiquitous in the study neighbourhood, and young children were frequently observed to be sitting and playing near the drains. Ingestion of open drain water was modelled assuming hand contact with the drain water (either directly or via an object) would result in hand contamination and subsequent hand-to-mouth transfer of microorganisms attached to the hand (Wang et al., 2017). Pathways via groundwater contamination were not relevant because groundwater in Mirpur is deep (Akhter and Hossain 2017), and households are served by a municipal piped water supply. This study did not examine additional pathways for faecal-oral transmission of enteric pathogens (e.g. water supply, food hygiene, soil, person-to-person), although we recognize their importance and the need for wider multi-sectoral interventions and environmental modification to control transmission of faecal pathogens (Robb et al., 2017; Cumming et al., 2019).

2.4. Modelling approach

The model consisted of two connected sub-models following the logic previously outlined by Mills et al. (2018): (i) pathogen fate-and-transport sub-model to estimate reference pathogen and indicator concentrations at specific locations, and (ii) exposure-and-risk sub-model (Fig. 2). Model inputs comprised a combination of estimates based on relevant scientific literature as well as primary data collected from the study site (household survey and infrastructure census). Eight sanitation options were tested in the model to predict their expected effect on concentrations of microbes in wastewater and subsequent health risks from exposure. The model also tested the effects of different scenarios relating to climate, disease prevalence, and faecal sludge management.

2.4.1. Pathogen fate and transport sub-model structure and approach

The stochastic fate and transport sub-model was written in Python 3 and consisted of 165 nodes (representing compounds, drains and sanitation systems) to model wastewater volume and pathogens (see Fig. S1 in Supplementary Material). Conceptually, the model started with the households (or residents) as the source for pathogens, with pathogens flowing through and/or being removed by, various infrastructure (sanitation systems and drains) across the study site. Household survey results indicated that for 96% of households, faeces of young children were disposed of via the same sanitation system used by adult members of the household, and hence when assigning pathogen inputs by sanitation type the model did not differentiate between adults and children. Pathogen inputs from animals were not included in the model because of the small animal population in the study site, combined with the fact that four of the six target organisms were human specific.[5] The model did not consider possible pathogen inputs via greywater as the purpose was to examine relative outcomes across different sanitation options.

Model input variables included household water and toilet usage by time of day (based on household survey data), faecal mass excretion per person per day (point value = 243g) (Rose et al., 2015), and pathogen-specific prevalence of infection (Table 2). To account for the diurnal distribution in water usage, the model was run for three days with a 10-min time step. Stochastic inputs were modelled using random sampling (1000 Monte Carlo simulations). Pathogens and FIB were assumed to be uniformly mixed within each node. Concentrations from the final 24-h period (144 steps, allowing the first 2 days for burn-in) were pooled to create one random sample across the study site. A zero-inflated skewed normal distribution was then fitted to the random sample. In order to understand the impact on populations 'downstream' from the study site, the model also calculated the quantity of pathogens/FIB discharged into the canal.

Pathogens were removed from the system in three ways:

i) A log reduction value (LRV) was applied for the relevant sanitation system (Table 2 and Tables S6–S9 in Supplementary Material) at each time step, based on its expected performance in removing pathogens.
ii) A quantity of pathogens was removed at each time step according to literature-based assumptions on pathogen die-off in surface waters (see Table S10 in Supplementary material).
iii) A quantity of pathogens was removed at each time step according to literature-based assumptions on pathogen settling (see Table S11 in Supplementary material). Estimates for pathogen settling in drains considered the proportion of pathogens that might attach to particulate matter and the proportion in a free phase, with settling velocities applied to each fraction (Cizek et al., 2008).

In order to evaluate the plausibility of the estimated concentrations and the usefulness of the modelling approach for filling in gaps in environmental data, the random sample of estimated pathogen/indicator concentrations was compared with results from an environmental monitoring program conducted at the same site (Amin et al., 2020).

2.4.2. Exposure and risk sub-model structure and approach

The exposure-and-risk sub-model started with the predicted pathogen concentrations (from the fate-and-transport sub-model) and applied a QMRA with input parameters for exposure (contact frequency, ingestion volume), dose-response and probability of illness, to estimate the number of cases of illness and overall disease burden in terms of disability-adjusted life years (DALYs). Model inputs are summarised in Table 2 and described more fully in Supplementary Material. The model was focused on the impact of sanitation via exposure to open drains (with ingestion associated with either direct contact with drain water or contact with objects submerged in drains). Ingestion volume was estimated based on a methodology applied in previous SaniPath studies (Gretsch et al., 2016), which considered exposure frequency (based on data from the household survey) and range of variables influencing the degree to which drain water might be ingested via hands (see Tables S13

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[4] Resource limitations prevented inclusion of a helminth.

[5] In total, compounds in the study site housed 12 cows, 1 goat, 1 cat, 94 chickens and 64 pigeons. Of the six target organisms, only *E. coli* and *Giardia* are not human specific.

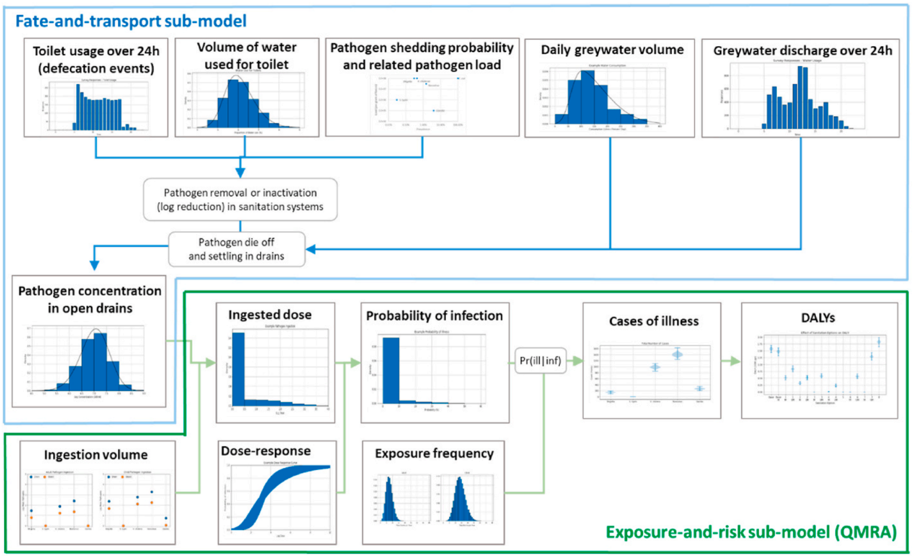


Fig. 2. Schematic of modelling approach and sub-models.

and S14). Exposure frequencies were inputted as a negative binomial distribution based on how many times survey respondents reported that they and their children had come into contact with an open drain in the previous week (see Fig. S3). Converting cases of illness to disability-adjusted life years (DALYs) was done by calculating the sum of years lived with disability (YLD) and the years of life lost (YLL) due to premature mortality, drawing on Global Burden of Disease studies for estimating disability weight, duration and distribution of cases across different levels of severity (Troeger et al., 2018; Stanaway et al., 2019) as well as Dhaka-based studies for case fatality estimates and the associated years of life lost (Tables S15 and S16) (Paul et al., 2016; Yu et al., 2018) The scope for examining health impact was contained to the study site population, and the risks to the wider population could only be characterised by quantifying pathogens exported (outflow) into the nearby waterway, to which other downstream populations may be exposed.

2.4.3. Sanitation options

The sanitation options assessed are outlined in Table 3 and illustrated in Fig. 3. Two key considerations informed the choice of these options: (i) realistic constraints in a low-income Dhaka context, and (ii) options representing a variety of scales. The realistic constraints included cost and complexity (leading to exclusion of high-tech options and inclusion of interim solutions such as deepening and covering drains); soil type (e.g. dense, clay soils that prevent infiltration of on-site effluent (DWASA, 2011)); and cultural appropriateness (e.g. preference for water-based sanitation). The options were drawn from sanitation technologies already present throughout Dhaka. Exceptions were Options 3 and 4 which involved decentralised constructed wetlands, solutions that are increasingly adopted in urban environments (ElZein et al., 2016; Russo et al., 2019; Stefanakis 2019); and Option 7 comprising sealed vaults, associated with container-based sanitation, which is also increasingly being adopted elsewhere (Russo et al., 2019). Variety in technology scale was intended to reveal differences in pathogen transmission via open drains, and included household, communal (up to ~400 households) and whole road (~1200 households) systems, as well as wastewater piped to centralised wastewater treatment beyond the study site. The options were selected in consultation with both local stakeholders and the expert advisory panel. 'Managed' and 'unmanaged' variants of each option were proposed based on the level of faecal sludge management (FSM) and other common management issues such as blockages and overflows that occur in practice. Detailed assumptions underpinning managed and unmanaged cases (and their associated LRVs) are described in Tables S6–S9 in Supplementary Material.

2.4.4. Scenarios and sensitivity analysis

The effects of six scenarios were examined: four climate-related scenarios (dry season, wet season but not raining, wet season and raining, wet season and flooding), a short-term sudden increase in infection/disease prevalence (outbreak), and the dumping of septic tank sludge in a drain (sludge dumping). In addition, sensitivity analysis was conducted to assess the effect of key parameters on the predicted impacts of different sanitation options. Plausible maximum and/or minimum values were inputted for: (i) disease prevalence, (ii) shedding load, (iii) immunity, (iv) ingestion volume, (v) exposure frequency (vi) LRVs of sanitation systems. The details of the scenarios and maximum/minimum values tested for each parameter are presented in Tables S4, S5 and S9 in Supplementary Material.

2.5. Environmental microbiology

In order to validate the transport and fate component of the model (see Section 2.4.1), faecal pathogen and FIB concentrations in drain water were measured at various locations in the study site. Samples were also collected from flood waters, drain sediments, the canal receiving all drain water, as well as septic tanks and ABRs (sludge, supernatant and effluent) to assess the plausibility of LRV assumptions for sanitation systems. Because the ABRs in the study site were decommissioned just prior to data collection, sampling of ABR sludge, supernatant and effluent took place in a neighbouring area (Amin et al., 2020). In total, 150 environmental samples were collected and both faecal pathogen and indicator concentrations were measured. *E. coli* concentrations were assessed using IDEXX Quanti-tray 2000 (IDEXX Laboratories, Westbrook, Seattle, WA), a method which quantifies the most probable number (MPN) of *E. coli* per 100 ml, while pathogens were detected and quantified using singleplex quantitative PCR. In-depth descriptions of the environmental sampling and laboratory methods can be found in an accompanying paper (Amin et al., 2020)).

3. Results

3.1. Environmental microbiology findings

The detailed findings from all the environmental samples are presented elsewhere by Amin et al. (2020), however the summarised results are presented in Table 4. Pathogens of all types were detected in almost

Table 2
Parameters included in model.

Variable	Parameter	Source(s)
Prevalence of diarrhoea (in previous week)	7.5%	Household survey
Infection state of each individual (1 = infected, 0 = not infected)[a]		
V. cholerae	Bernoulli ($p = 0.0043$)	Weil et al. (2009, 2014); Das et al. (2013)
S. Typhi	Bernoulli ($p = 0.0003$)	Ames and Robins (1943); Naheed et al. (2010); Gunn et al. (2014); Darton et al. (2016); Gauld et al. (2018)
Shigella	Bernoulli ($p = 0.0030$)	Das et al. (2013); George et al. (2015)
Norovirus GII	Bernoulli ($p = 0.0149$)	Partridge et al. (2012); Milbrath et al. (2013); Rahman et al. (2016); Wu et al. (2019)
Giardia	Bernoulli ($p = 0.040$)	Karim et al. (2018)
Shedding load (log$_{10}$ per gram of faeces)[b]		
E. coli	Normal ($\mu = 8$, $\sigma = 0.5$)	Wright (1982); Mara and Oragui (1985)
V. cholerae	Normal ($\mu = 8$, $\sigma = 0.5$)	Uddin et al. (2013)
S. Typhi	Normal ($\mu = 6$, $\sigma = 0.5$)	Expert opinion
Shigella	Normal ($\mu = 8$, $\sigma = 0.5$)	Assumption
Norovirus GII	Normal ($\mu = 7.5$, $\sigma = 0.5$)	Kirby et al. (2014); Teunis et al. (2015); Sabrià et al. (2016)
Giardia	Normal ($\mu = 5.5$, $\sigma = 0.5$)	Danciger and Lopez (1975)
Probability of illness given infection		
V. cholerae	0.53	Weil et al. (2014)
S. Typhi	0.70	Darton et al. (2016)
Shigella	0.19	George et al. (2015)
Norovirus GII	0.55	Teunis et al. (2008); Kirby et al. (2014)
Giardia	0.40	Ortega and Adam (1997); Platts-Mills et al. (2015)
Faecal mass (grams per day)	243	Rose et al. (2015)
Water usage (litres per day)	Road A: Skew normal ($\mu = 56.8$, $\sigma = 121.3$, $\alpha = 4.9$) Road B: Skew normal ($\mu = 25.4$, $\sigma = 99.2$, $\alpha = 5.0$) Road C: Skew normal ($\mu = 53.6$, $\sigma = 106.1$, $\alpha = 4.8$) Road D: Skew normal ($\mu = 41.4$, $\sigma = 120.0$, $\alpha = 4.6$)	Water meter data
Blackwater as proportion of water use	Normal ($\mu = 0.061$, $\sigma = 0.0072$)	Household survey
Sanitation system LRVs [managed, unmanaged]		
Holding tank	Bacteria [0.5,0.25], Virus [0.25,0.125], Protozoa [1,0.5]	See Tables S6–S9 in Supplementary Material
Septic tank	Bacteria [1,0.5], Virus [0.5,0.25], Protozoa [2,1]	
ABR	Bacteria [1,0.5], Virus [0.5,0.25], Protozoa [2,1]	
Constructed wetland	Bacteria [2,1], Virus [1.5,0.75], Protozoa [1.5,0.75]	
Waste stabilisation pond	Bacteria [6], Virus [4], Protozoa [4]	
Exposure frequency		
Adults	Negative binomial ($N = 3$, $P = 0.958$)	Household survey
Children	Negative binomial ($N = 1$, $P = 0.795$)	Household survey
Ingestion	Various (see Tables S13–S14 in Supplementary Material)	Various
Dose-response		
V. cholerae	Beta-Poisson ($\alpha = 2.5 \times 10^{-1}$, $N_{50} = 2.43 \times 10^{2}$)	Hornick et al. (1971)
S. Typhi	Beta-Poisson ($\alpha = 1.75 \times 10^{-1}$, $N_{50} = 1.11 \times 10^{6}$)	Hornick et al. (1966, 1970)
Shigella	Beta-Poisson ($\alpha = 2.65 \times 10^{-1}$, $N_{50} = 1.48 \times 10^{3}$)	DuPont et al. (1972)
Norovirus GII	Fractional Poisson ($P = 0.722$, $\mu = 1106$)	Messner et al. (2014)
Giardia	Exponential ($k = 1.99 \times 10^{-2}$)	Rendtorff (1954)
DALYs – diarrhoeal pathogens		
Distribution of cases by severity	Mild (0.243), Moderate (0.617), Severe (0.14)	Troeger et al. (2017)
Duration by severity (years)	Mild (0.0115), Moderate (0.0115), Severe (0.0115)	Troeger et al. (2017)
Disability weight by severity	Mild (0.074), Moderate (0.188), Severe (0.247)	Troeger et al. (2017)
Case fatality	0.001%	Paul et al. (2016)
Years of life lost per fatality	55	Paul et al. (2016)
DALYs – typhoid		
Distribution of cases by severity	Moderate (0.35), Severe (0.43), Severe with GI bleeding (0.05), Severe with other abdominal complications (0.17)	Stanaway et al. (2019)
Duration by severity (years)	Moderate (0.038), Severe (0.079), Severe with GI bleeding (0.076), Severe with other abdominal complications (0.079)	Stanaway et al. (2019)
Disability weight by severity	Moderate (0.051), Severe (0.133), Severe with GI bleeding (0.133), Severe with other abdominal complications (0.324)	Stanaway et al. (2019)
Case fatality	0.3%	Yu et al. (2018)
Years of life lost per fatality	32	Yu et al. (2018)

[a] Point prevalence of infection for *V. cholerae*, *shigella* and norovirus GII estimated by multiplying prevalence of infection among diarrhoeal patients visiting medical facilities in Dhaka by the proportion of population with diarrhoea, adjusting for duration of symptoms, duration of shedding, and probability of illness given infection. Point prevalence of infection for *S.* Typhi was based on an estimate of typhoid incidence in Dhaka, adjusting for duration of shedding, chronic carriage, and probability of illness given infection. Point prevalence of infection for *Giardia* based on a cross-sectional assessment of stools in Dhaka.

[b] Applied equally to the whole population.

[c] *Salmonella typhimurium* used as a proxy for *S.* Typhi.

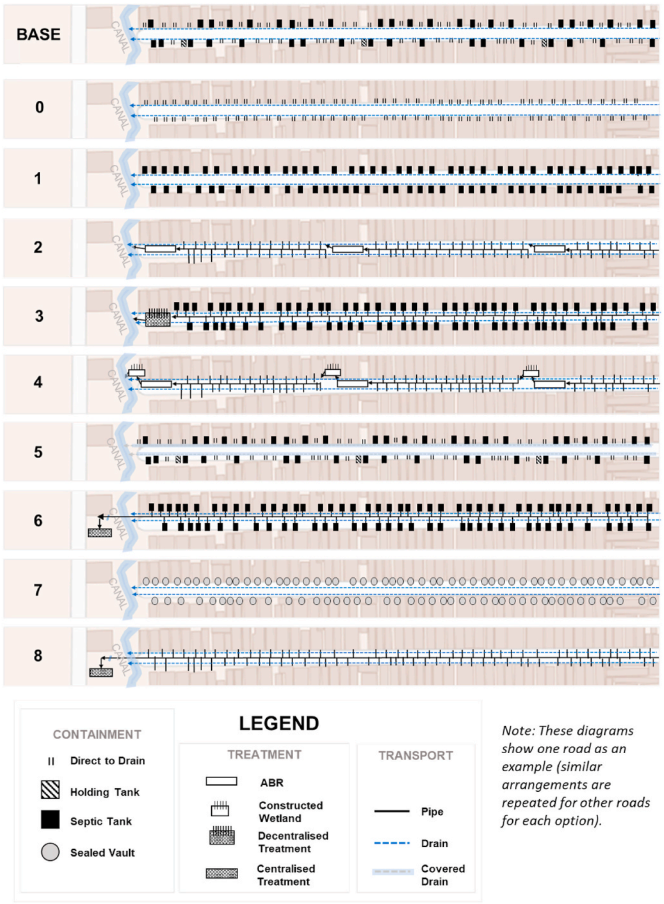
all sample types. *Shigella* and *V. cholerae* were the most commonly detected pathogens in drain samples (100% of samples), followed by norovirus GII (67%), *Giardia* (50%) and *S.* Typhi (27%). *Giardia* and *S.* Typhi were more prevalent in the wet season, though the converse was true for norovirus. *E. coli* had the highest geometric mean concentration in drain water, followed by *Shigella*, *V. cholerae*, *Giardia*, norovirus GII, and *S.* Typhi.

3.2. Comparison of sanitation options

Modelled disease burden associated with faecal pathogen exposure

Table 3
Sanitation options examined by the model.

Option	Description	Managed	Unmanaged
Base case	**Base case**: Represents the current sanitation infrastructure in the study site (combination of septic tanks, holding tanks and toilets discharging directly to drains).	Optimally managed holding tanks and septic tanks (regularly emptied and maintained)	Holding tanks and septic tank systems overloaded, unmaintained and not emptied. (represents the current situation).
Option 0	**No containment**: Hypothetically remove all sanitation systems (so all toilets discharge to drains) (included as a reference point)	NA	NA
Option 1	**Septic tanks**: Full septic tank coverage for all household compounds (two-chamber tank only without soak-away infiltration), septic tank effluent flows direct to drains	Optimally managed septic tanks (regularly emptied and maintained)	Septic tank systems overloaded, unmaintained and not emptied.
Option 2	**Communal primary treatment**: All toilets discharge to closed sewer and piped to decentralised primary treatment in anaerobic baffled reactors (ABRs) (three ABRs per road, approximately 400hh per ABR) which then discharge to the drain	Optimally managed ABR (regularly emptied and maintained)	ABR overloaded, unmaintained and not emptied.
Option 3	**Septic tanks with secondary treatment**: Full septic tanks coverage for all household compounds. All septic effluent collected and piped through small bore sewers to decentralised secondary treatment (constructed wetland) at the end of each road (for approx. 1200hh) discharging into adjacent canal	Optimally managed septic tanks (regularly emptied and maintained) and constructed wetland (proactively maintained)	Septic tanks overloaded, unmaintained and not emptied. Constructed wetland unmaintained, with clogged filter media, blocked or overflowing, plants unattended.
Option 4	**Communal primary and secondary treatment**: All toilets discharge to closed sewer and piped to a decentralised primary and secondary treatment (three ABR and constructed wetlands per road, approximately 400hh per system), which then discharge to drain	Optimally managed ABR (regularly emptied and maintained) and constructed wetland (proactively maintained)	ABR may be overloaded, unmaintained and not emptied. Constructed wetland unmaintained, with clogged filter media, blocked or overflowing, plants unattended.
Option 5	**Deepen and cover drains**: Sanitation systems remain as per base case but the open drains are all deepened and covered	Regularly maintained and flushed drains	Blockages (e.g. with solid waste and faecal waste) create overflows into the road as well as breakage of covers
Option 6	**Septic tanks with small-bore pipe to centralised tertiary treatment**: Full septic tank coverage with effluent piped through shallow small-bore sewer to centralised secondary and tertiary treatment (beyond the study boundary)	Optimally managed septic tanks and well-maintained small-bore sewer.	Septic tanks overloaded, unmaintained and not emptied, and small-bore sewer with broken pipes or overflowing into drains or road.
Option 7	**Fully sealed vaults**: All toilets discharge to fully sealed vaults or containers (with contents tankered to centralised faecal sludge treatment)	Optimally managed and emptied vaults (frequently safely emptied)	Poor emptying practices and broken tanks leading to some direct discharge into the open drains
Option 8	**Sewer system to centralised tertiary treatment**: Toilets discharge to a closed sewer and conveyed to centralised secondary and tertiary treatment (beyond the study boundary)	Optimally managed and maintained sewerage system	Poor maintenance and blockages result in local sewerage overflows into open drains or road



Note: These diagrams show one road as an example (similar arrangements are repeated for other roads for each option).

Fig. 3. Schematic of sanitation options.

via open drains varied widely under the different sanitation options (Fig. 4). Under the base case ('Base-UM'), the model estimated an annual disease burden associated with the five target pathogens to be 1.3 DALYs per 1000 people (540 cases of illness per 1000 persons per year). As a point of comparison, reverting all households to toilets that discharge

Table 4

Prevalence and mean $\log_{10}$ concentration per 100 ml for faecal pathogens and faecal indicator bacteria (*E. coli*) by sample type.

Sample type	n	FIB (*E. coli*)		Norovirus GII		*V. cholerae*		*Shigella*		*S.* Typhi		*Giardia*	
		% +ve	Mean	% +ve	Mean	% +ve	Mean	% +ve	Mean	% +ve	Mean	% +ve	Mean
Drain water	30	100	7.2	67	3.5	100	4.8	100	4.9	27	2.5	50	4.0
Canal water	4	100	6.9	0	–	75	3.6	100	4.0	50	2.8	0	–
Flood water	6	100	5.0	33	4.7	67	4.5	67	4.3	17	–	50	3.3
Septic sludge	10	100	6.1	90	6.5	0	–	70	5.3	10	4.7	0	–
Septic supernatant	8	100	5.9	25	2.5	0	–	13	3.5	0	–	0	–
Septic effluent	22	100	6.8	64	4.6	45	3.4	95	3.3	0	–	18	3.5
ABR sludge	8	100	8.0	100	7.1	63	5.9	100	6.9	0	–	25	7.1
ABR supernatant	7	100	7.9	71	4.2	57	3.7	100	4.5	0	–	29	3.3
ABR effluent	7	100	8.0	100	5.1	86	4.2	100	5.0	29	2.7	57	3.8

Note: Mean values represent the geometric mean and are based only on the positive samples.

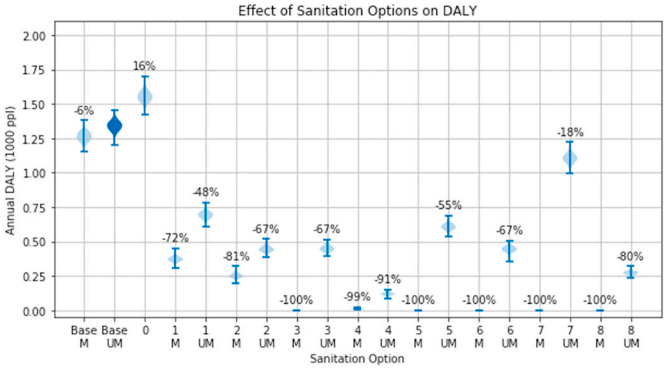


Fig. 4. Modelled effect of sanitation options on annual DALYs (relative to base case unmanaged – base-UM).

Note: Numbers above violin plots refer to percentage change in DALYs relative to the *unmanaged* base case. Base = sanitation status quo in study site; Option 0 = no containment (direct to drain); Option 1 = septic tanks; Option 2 = communal treatment in ABRs; Option 3 = septic tanks with effluent treated in constructed wetlands; Option 4 = communal ABRs and constructed wetlands; Option 5 = deepen and cover drains; Option 6 = septic tanks with centralised tertiary treatment offsite; Option 7 = fully sealed vaults (contents regularly removed by tanker and treated offsite); Option 8 = sewerage to centralised treatment offsite.

directly to drain (‘0’) was estimated to increase DALYs by 16% compared with the base case. Improving the management of existing septic tank infrastructure through regular emptying (‘Base-M’) was predicted to have only a 6% reduction in the disease burden for the population within the study site. This minimal impact is because under this scenario more than 70% of the population would still be using toilets discharging directly to the drain. Relative to the base case, comprehensive coverage of septic tanks was associated with a 72% reduction in DALYs when well-managed (’1-M’), and a 48% reduction when poorly managed (’1-UM’). Complete coverage of communal scale ABRs was predicted to have a greater impact of 81% and 67% for managed (’2-M’) and unmanaged (’2-UM’) situations respectively, since it also reduced exposure through closed pipe conveyance. An option with comprehensive septic tank coverage, with all effluent piped through small bore sewers to decentralised secondary treatment at the end of the roads, eliminated all of the disease burden for the study site population when well managed (’3-M’), since it prevents entry of all pathogens to the drain. However, when the septic tanks were unmanaged (’3-UM’), this option had some residual health risk, albeit still 67% lower than the base case. Communal ABRs installed alongside secondary treatment reduced the disease burden by 99% when well managed (’4-M’), and by 91% when unmanaged (’4-UM’). Deepening and covering all open drains (’5M’) and fully-sealed containment systems (’7-M’) also reduced the local disease burden associated with poor sanitation to zero when well-managed, though the former option exported high numbers of pathogens to ‘downstream’ neighbourhoods. Both of these options posed a significant health risk if unmanaged (’5-UM’, ‘7-UM’). Conveyance to centralised treatment (’6M’ and ‘8M’) also reduced the local disease burden to zero when well-managed, and by 67% (for ‘6M’) and 80% (for ‘8UM’) when unmanaged.

The results for all sanitation options need to also be considered in light of the exported pathogen load to ‘downstream’ neighbourhoods (see Fig. 5). Pathogens discharged to the open canal at the boundary of the study site could adversely affect other populations; though it was beyond the scope of this study to quantify those health risks. When well-managed, Options 6, 7, and 8 resulted in the lowest pathogen concentrations in exported wastewater since they avoid any local discharge. This was followed by: (i) Options 3 and 4, which include secondary on-site treatment in constructed wetlands; (ii) Options 1 and 2 comprising primary on-site treatment in septic tanks and ABR respectively; and finally (iii) Option 5, Option 0 and Base Case, which all exported drain water containing high pathogen concentrations (for example, concentration of greater than 5 $\log_{10}$ per 100 mL for *V. cholerae)*. However, when Options 6, 3 and 4 were unmanaged, pathogen concentrations in exported wastewaters were similar to those for unmanaged Options 1 and 2.

The relative contribution of each pathogen to DALYs for each sanitation option is shown in Fig. 6. Under the base case, norovirus GII was predicted to contribute 36% of the disease burden associated with the five target pathogens, followed by *V. cholerae* (35%), *Giardia* (25%), *Shigella* (5%) and *S. Typhi* (<1%). There was little variation in the percentage contribution of *V. cholera*, *Shigella* and *S.* Typhi across different sanitation options, though for a number of sanitation options (1, 2, 3-UM, 4 and 6-UM) the percentage contribution of *Giardia* to the disease burden was reduced, while the contribution of norovirus increased.

Ranges of predicted pathogen concentrations were generally wide – even on the $\log_{10}$ scale – yet predicted cases of illness fell within a relatively narrow band (Fig. 7). In terms of percentage reduction in cases of illness, the pathogen-specific impact of sanitation improvements tended to mirror the sanitation LRVs assigned to each pathogen type. Sanitation improvements had the greatest effect on Giardiasis, reflective of the sanitation system LRVs which were highest for *Giardia*. Cholera and Shigellosis exhibited similar percentage reductions in cases of illness across different sanitation options. The percentage reduction in cases of illness associated with sanitation improvements was lowest for norovirus. The effect of sanitation options on cases of typhoid was difficult to characterise, which is likely an artefact of the lower probability of *S.* Typhi being shed at any particular point in time.

3.3. Model validation

The model predicted a wide range of concentrations for all pathogens and *E. coli*, which generally agreed with empirical observations (Fig. 8). The model estimated concentrations that spanned the majority of observed concentrations for three of five pathogens (*Shigella*, *S.* Typhi, *V. cholerae*) as well as *E. coli*. When norovirus and *Giardia* were detected

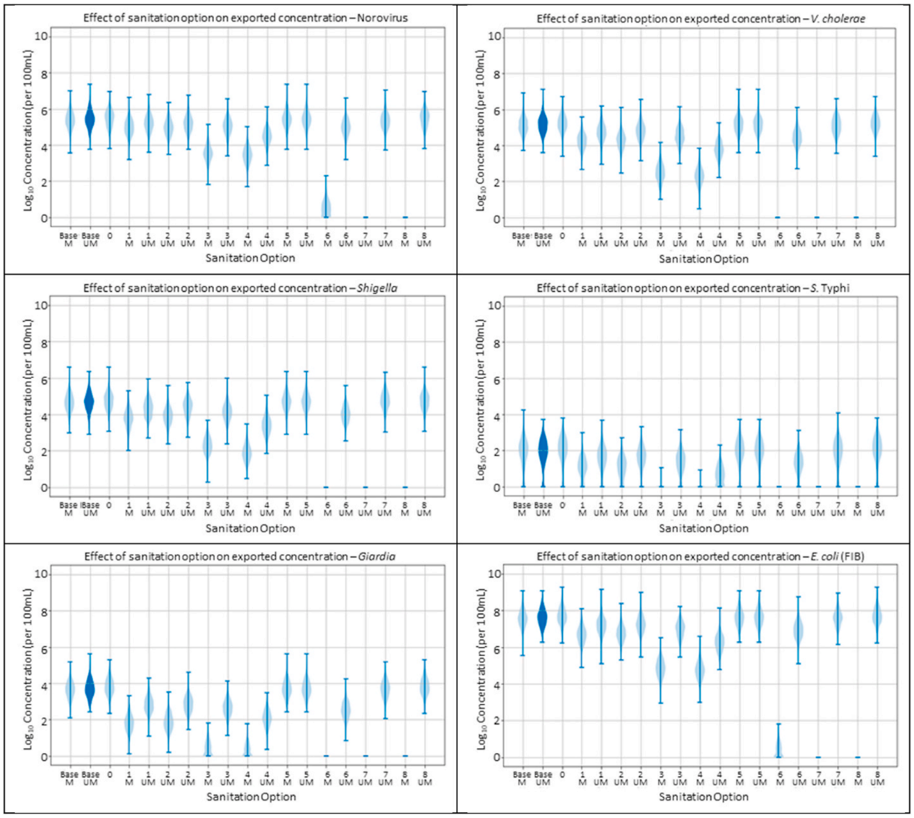


Fig. 5. Predicted concentrations of exported pathogens and FIB in wastewater discharged to canal by pathogen and sanitation option.

Note: Exported pathogen concentrations are in relation to 'downstream' communities nearby to the canal, hence Option 8M (well managed sewer system to centralised tertiary treatment) is assigned exported pathogen concentrations of zero, on the assumption that tertiary treatment (and its residual health risk) would occur elsewhere.

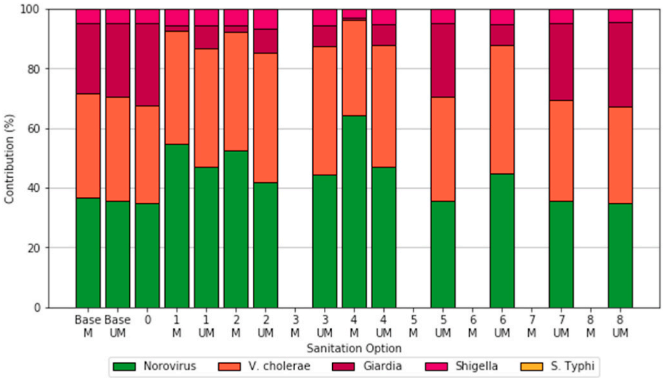


Fig. 6. Relative contribution of each pathogen to DALYs by sanitation option.

in drain water, the observed concentrations for these pathogens fell within modelled distributions. However, the sampling results had a high number of non-detects for *Giardia* and for norovirus that the model failed to reproduce. Conversely, *S.* Typhi was detected more frequently than the model predicted, although the observed concentrations (2–4 $\log_{10}$ concentration per 100 mL) were still within the range the model anticipated. Those pathogens with lower prevalence of shedding among the population but higher shedding load per gram of faeces (*V. cholerae*, *Shigella*) had the widest modelled ranges (0-7 $\log_{10}$ concentration per 100 mL), while the more prevalent microorganisms had narrower modelled ranges (e.g. *E. coli*, norovirus and *Giardia*).

Given the limited literature on LRVs for sanitation systems, it was important to review the chosen LRV assumptions against empirical data. On average, measured pathogen and FIB concentrations in ABR effluent were similar to modelled concentrations, but for septic tanks the measured effluent concentrations were discernibly lower than what the model predicted for three pathogens (those being norovirus, *V. cholerae* and *S.* typhi) (Fig. S4). Measured concentrations in septic tank effluent were lower than in ABR effluent for all five pathogens and for the FIB. While this may indicate differential LRVs, it could also relate to the smaller catchment population represented in septic tank samples (as compared with ABRs), which in turn may lower the probability of infected individuals being within the user population.

The predicted numbers of cases of infection for each pathogen are shown in Fig. S5. Among the five pathogens, norovirus was predicted to cause the highest number of infections annually (927, 36%), followed by *V. cholerae* (904, 35%), *Giardia* (637, 25%), *Shigella* (120, 5%), and *S.* Typhi (<1, <1%). Estimated case numbers for all pathogens were below the total case numbers one would expect from all transmission routes based on the prevalence inputs. Overall, predicted cases of illness from exposure to drains amounted to 30% of the estimated total cases of illness based on prevalence inputs. In other words, the model estimated that 30% of illnesses from the five target pathogens were attributable to exposure via open drains, while the rest of the cases were due to transmission via other exposure pathways. Pathogen-specific ratios were 25% (norovirus GII), 46% (*V. cholerae*), 25% (*Giardia*), 24% (*Shigella*), and <1% for *S.* Typhi. Children under 5 years were predicted to experience a disproportionately high disease burden, accounting for 39% of cases of illness despite constituting only 14% of the population. The proportion of estimated illnesses borne by children under 5 years was relatively similar for norovirus (40%), shigellosis (38%) and cholera (36%), but slightly higher for giardiasis (43%).

3.4. *Scenarios and sensitivity analysis*

Analysis of six scenarios and sensitivity analysis revealed the importance of several key input parameters in determining the disease burden, though generally speaking the relative effectiveness (rank order) of different sanitation options was not sensitive to changes in input parameters, except for the LRV input parameter.

Among the six scenarios tested, a flooding scenario – which reduces

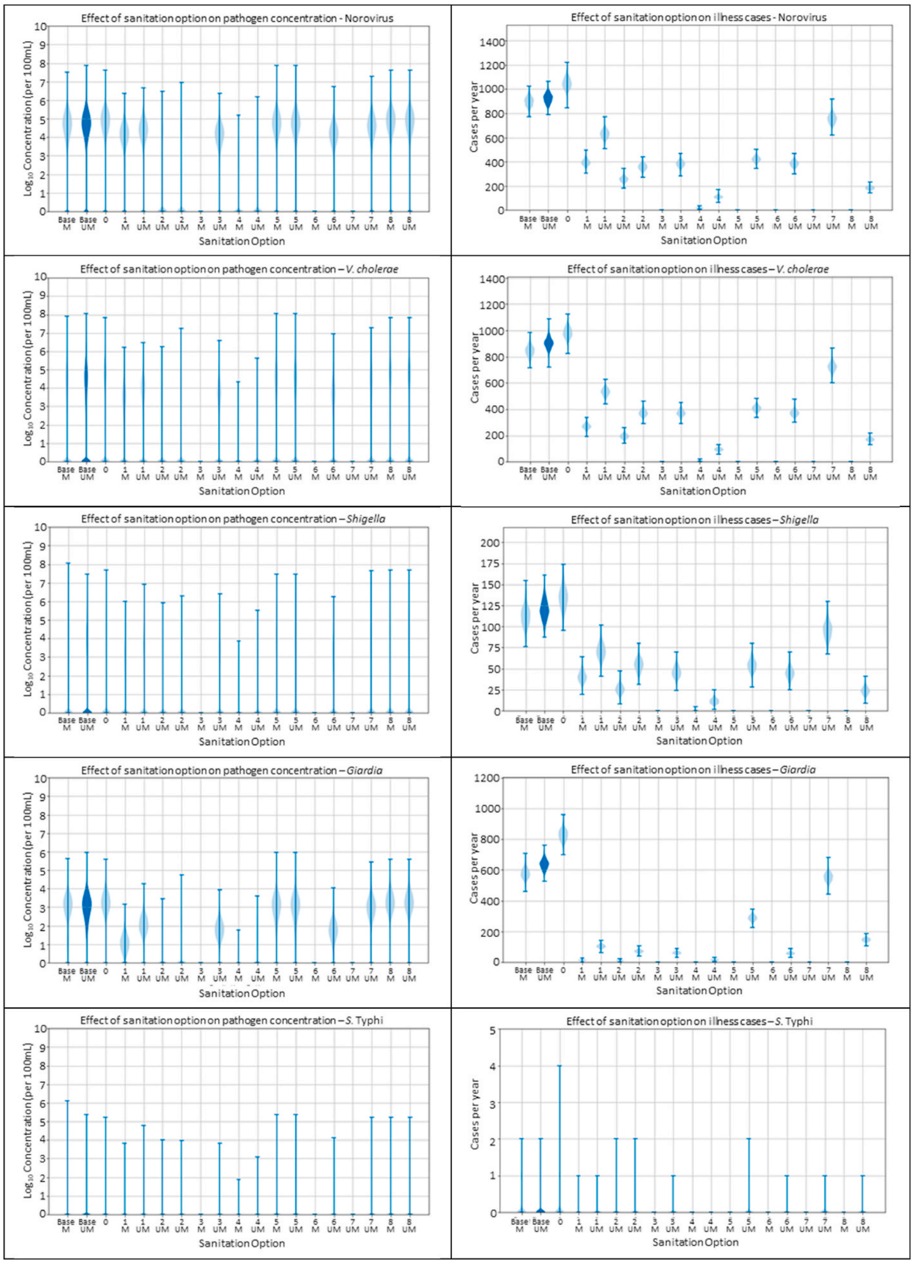


Fig. 7. Modelled impact of sanitation options on pathogen concentration in drains and associated cases of illness per year.

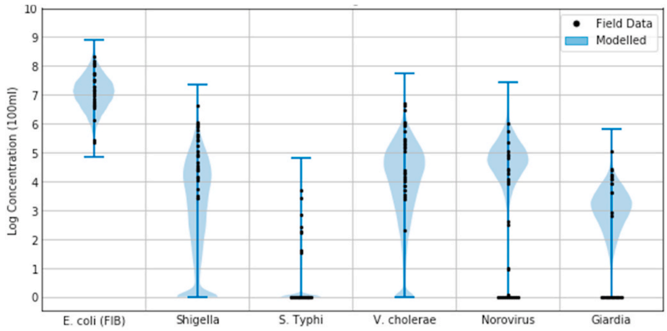


Fig. 8. Faecal pathogen and faecal indicator concentrations in drain water: Modelled vs Observed.

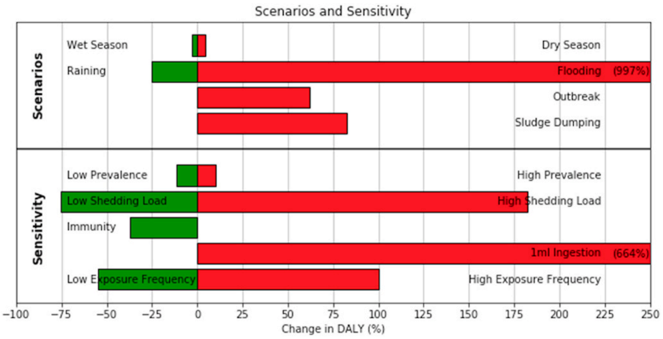


Fig. 9. Relative change in disease burden estimates for selected scenarios and sensitivity analyses for the base case (UM).

the pathogen concentration through dilution, but increases local exposure – had a major effect on disease burden, with base case DALYs increasing by an order of magnitude (Fig. 9). This effect was similar across all sanitation options. Inclusion of sludge dumping in the base case (as typically occurs in relation to on-site options) increased the estimated disease burden by 83%. The health risk associated with open drains under the base case was predicted to increase by 62% during an outbreak scenario when more infected people in the population would be excreting pathogens into the septic tanks and drains; conversely the overall disease burden was predicted to decrease during wet season, especially with moderate raining that did not result in flooding (increased dilution but no change in exposure). The reduced disease burden in wet season is linked to the predominance of norovirus, which was the only pathogen with assumptions suggesting prevalence decreased during this time.

Among the parameters for which sensitivity analysis was conducted, ingestion volume and shedding load emerged as the most influential in terms of impact on disease burden (Fig. 9). Modifying the ingestion volume from a distribution averaging 0.1 mL (base case assumption) to a fixed value of 1 mL (a typical value applied in QMRA studies) (WHO 2016) had a considerable effect on predicted disease burden, increasing the base case DALYs by 664%. However, increasing the ingestion volume in this way did not change the rank order of sanitation options. A one $\log_{10}$ increase in the shedding loads increased the disease burden under the base case by 182%. It should be noted that the effect for shedding may be exaggerated due the deterministic nature of the sensitivity analysis: shedding variables for all pathogens were modified simultaneously, so the figure shows the combined impact of all five shedding variables. In contrast, prevalence changes had a more moderate impact. High prevalence assumptions increased the base case DALYs by 10%. The model results were less sensitive to changes in assumptions about pathogen survival and settling, with maximum and minimum inputs having little impact on modelled concentrations in drains or disease burden.

The rank order of sanitation options in terms of health risk varied little across different scenarios and when substituting key inputs with maximum and minimum values. In other words, the prioritisation of sanitation options in terms of health risk was not sensitive to changes in most input parameters. There was one exception, however: the relative effectiveness of different sanitation options did change when applying different LRV assumptions to reflect how well or poorly systems were managed (Fig. 10).

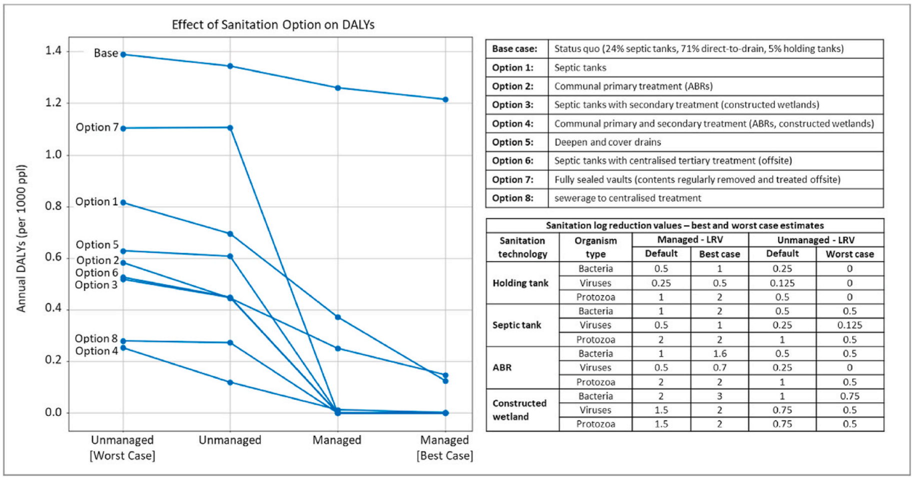
4. Discussion

The discussion covers three main areas: 1) the feasibility of the systems modelling approach; 2) implications for sanitation interventions and decision-making; and 3) key gaps in evidence, including important areas for further research and development.

Feasibility of the systems modelling approach: This study demonstrates the value and feasibility of a systems modelling approach for comparing sanitation options and estimating associated health risks. Modelled concentrations for most pathogens were relatively consistent with empirical observations, suggesting the fate and transport submodel generates realistic results. Full validation of the exposure and risk component of the model was not possible, though disease burden outputs in terms of number of cases and DALYs appeared plausible when compared alongside two key benchmarks. First, for the base case, the model predicted total DALYs of 1.3 per 1000 persons, which is equivalent to 25% of the WASH-attributable diarrhoeal disease burden in Bangladesh generally (5.3 DALYs per 1000 people) (Prüss-Ustün et al., 2019). This is what one might expect given exposure to drains represents just one of numerous possible exposure/transmission pathways and that the target pathogens constitute only a proportion of the total disease burden. Second, the number of cases of illness predicted for each pathogen was lower than the input prevalence, which is what one would expect for the same reason (i.e. exposure to open drains being one of multiple possible transmission pathways). Nonetheless, while the model produced plausible results, it is important to emphasise the outputs are theoretical and premised upon numerous assumptions and uncertainties.

The main deviation between modelled and observed pathogen concentrations was the proportion of samples with no detection of norovirus GII or *Giardia*. There are a number of possible explanations for this discrepancy. First, either the prevalence or shedding load may have been overestimated. By contrast, *S.* Typhi had a significantly lower prevalence input than both *Giardia* and norovirus, and the model successfully predicted a high proportion of samples with no detection. Other plausible contributors may be pathogen loss during sample processing, PCR inhibition, and limit of detection issues. Another important factor is that when pathogens enter the drainage system, the model assumes an instantaneous uniform distribution within a 'node' (drain section of ~45m). It is reasonable to expect significant dispersion of pathogens by the time they reach the end of a drain, but they may not be uniformly distributed. Hence, a grab sample of 400 mL may not contain the target pathogen even if the overall concentration within a drain section is quite high.

Implications for sanitation interventions and planning: The results have important implications for reliance on on-site sanitation in low-income urban contexts. Even with the highest possible removal in septic tanks and ABRs, there is still significant residual risk, and this worsens under poor management. For example, a quarter of the health



Base case:	Status quo (24% septic tanks, 71% direct-to-drain, 5% holding tanks)
Option 1:	Septic tanks
Option 2:	Communal primary treatment (ABRs)
Option 3:	Septic tanks with secondary treatment (constructed wetlands)
Option 4:	Communal primary and secondary treatment (ABRs, constructed wetlands)
Option 5:	Deepen and cover drains
Option 6:	Septic tanks with centralised tertiary treatment (offsite)
Option 7:	Fully sealed vaults (contents regularly removed and treated offsite)
Option 8:	sewerage to centralised treatment

Sanitation log reduction values – best and worst case estimates					
Sanitation technology	**Organism type**	**Managed - LRV**		**Unmanaged - LRV**	
		Default	**Best case**	**Default**	**Worst case**
Holding tank	Bacteria	0.5	1	0.25	0
	Viruses	0.25	0.5	0.125	0
	Protozoa	1	2	0.5	0
Septic tank	Bacteria	1	2	0.5	0.5
	Viruses	0.5	1	0.25	0.125
	Protozoa	2	2	1	0.5
ABR	Bacteria	1	1.6	0.5	0.5
	Viruses	0.5	0.7	0.25	0
	Protozoa	2	2	1	0.5
Constructed wetland	Bacteria	2	3	1	0.75
	Viruses	1.5	2	0.75	0.5
	Protozoa	1.5	2	0.75	0.5

Fig. 10. Estimated disease burden by sanitation option and level of management.

risk remains unaddressed when shifting away from all toilets discharging directly to drains to universal coverage of well-managed septic tanks. Likewise, analyses of the environmental samples from the northern drain of Road A, where 90% of the population's excreta was contained in septic tanks, indicated that *V. cholerae* and *Shigella* were still detected in all drain samples. Septic tanks systems should comprise a two-part system including a soak-away or drainfield with infiltration to soil to provide further pathogen removal and inactivation. However, in Dhaka the clay soil and space limitations prevent such a design. These findings may be widely relevant as recent research has reported that tanks connected to open drains are commonplace in Dhaka and in many other cities (Peal et al., 2020), and high levels of faecal contamination have been observed in open drains in many urban areas (Yapo et al., 2014; Katukiza et al., 2014; Gretsch et al., 2016; Berendes et al., 2018, 2020).

Although both ABRs and septic tanks were assigned the same LRVs, the disease burden associated with full coverage of ABRs was 30–40% less than the disease burden associated with full coverage of septic tanks. This result reflects differences in exposure rather than a differential effect on pathogen removal. A key design feature of the ABR systems is that the black water is conveyed from households in enclosed pipes, and so pathogens do not enter the drain until after the liquid effluent is discharged (at a location some distance 'downstream' from the user households). As a result, the modelling demonstrated how the reduced local exposure served to reduce the disease burden in the immediate area served by ABRs. This is in contrast to septic tanks, which in the study site discharged effluent to drains immediately outside user compounds. This indicates the potential for ABRs to serve as a first step on a ladder towards sewerage and a potential option for contexts such as Dhaka where effluent infiltration is not possible, so long as adequate management can be provided. Use of constructed wetlands further reduced the disease burden as compared with ABRs alone, but in practice requires both land area and proactive, dedicated management and hence may only be a viable option in certain contexts. A related solution is, over time and as it becomes available, to link ABR effluent to a wider sewerage network. An interim option of deepening and covering drains should be approached with caution; unless hydraulically designed to carry faecal matter during heavy rainfall, this could exacerbate local flooding and pathogen exposure.

Although the modelling produced outputs relating to localised disease burden, this overlooks the negative externality and downstream health risk created when pathogens are simply exported elsewhere rather than removed. Illustrating this is Sanitation Option 5 ("deepen and cover drains"); although it prevents exposure locally, wastewater with high pathogen content is still discharged into a receiving waterway at the western perimeter of the site. Most of the other options also include a residual pathogen load that enters that receiving waterway. Quantifying the health risk associated with pathogen export is not straightforward as it requires an increase in the spatial scale of modelling and associated inputs. Nonetheless, conducting modelling at a city-scale, such as in the hypothetical example presented in Mills et al. (2018), is an important next step, and could also support the inclusion of additional exposed populations such as sanitation workers.

In considering the implications of the results for on-going sanitation planning and investments in Dhaka, which include both off-site and on-site sanitation options, the following points need to be made. First, further implementation of septic tanks and ABRs to treat blackwater must include subsequent effluent treatment to reduce the pathogen load (given inability to infiltrate effluent) and should deliver comprehensive rather than piecemeal sanitation coverage. In addition, in other contexts such as Japan and the US, UV treatment or other disinfection steps are implemented before discharge of the effluent (US EPA, 2003; Gaulke 2006), and this provides a further option for consideration, noting that it introduces additional operation and maintenance requirements. Second, this study demonstrates that the use of on-site systems in Dhaka or elsewhere requires appropriate and robust management (such as overcoming inadequate emptying practices that result in sludge wash-out and short-circuiting, and avoiding sludge dumping in drains) – otherwise these systems fail to adequately protect public health. This is particularly true in locations that flood, as this scenario exhibited the potential for significant increase in disease burden during such times. To protect public health, strong institutionally-based management systems are required, including for decentralised systems such as ABRs, as has been demonstrated elsewhere (Willetts et al., 2020). Third, an important complement to consideration of health risks is weighing up the costs of different sanitation options as well as other criteria such as environmental considerations (Willetts et al., 2020).

Key evidence gaps: There is a need for more robust data on how on-site sanitation systems, as actually constructed and operated in low-income settings, perform under different conditions and management regimes. The sensitivity analysis revealed LRV as the key input parameter for differentiating between the effect of the various sanitation options. Yet, robustly determined LRVs for different types of pathogens (bacteria, viruses, protozoa and helminths) in septic tanks and ABRs are lacking, despite the ubiquity of these technologies in urban settings and the key role of on-site sanitation in city-wide inclusive sanitation strategies (Schrecongost et al., 2020). Although the LRV inputs were largely based on literature, available data points for septic tanks and ABRs are very limited and rarely presented consistently or with sufficient information. For example, some sources do not clarify whether the quoted LRVs refer to just a septic tank, or its use in combination with a soak-away for infiltration (and related treatment in the soil) (Adegoke and Stenstrom, 2019). Estimating LRVs for small-scale or on-site sanitation systems is fraught with methodological challenges. It is challenging to measure the 'influent' to a septic tank, given the periodic nature of that influent and because it contains a mixture of solid and liquid. Nonetheless, addressing this knowledge gap is an urgent priority, including examining the potential effects of different management regimes, as well as modifications to septic tank designs such as an effluent filter or addition of disinfection processes.

The sensitivity analysis also highlighted other key areas of uncertainty that have a significant impact on estimated pathogen concentrations and disease burden. Chief among those were shedding load for each pathogen – for which evidence remains relatively scant – and ingestion volume of drain water. This study followed the SaniPath approach to estimate ingestion volume (Gretsch et al., 2016; Robb et al., 2017) and extended it to incorporate adults and children >12 years. This resulted in ingestion volumes of approximately 0.1 mL, well below the 1–5 mL fixed estimates used in other studies. Nonetheless, the rank order of the sanitation options (in terms of their predicted impact on disease burden) remained unaffected by substitution of these and other key parameters with minimum/maximum estimates. Another evidence gap concerns exposure assessment. Data on exposure of children and adults to open drains and via other contamination pathways is limited (Gretsch et al., 2016), and yet essential for furthering systems modelling of this kind. Structured observation, rather than self-reporting (as undertaken in this study), is likely to generate more robust data for the purposes of characterising exposure behaviours.

4.1. Limitations

The study has a number limitations related to both the modelling and processes for detecting and quantifying pathogens.

Model parameters. While we sought to base model parameters on empirical evidence from the Dhaka context, this was not always possible. Estimates for shedding load were based on few empirical data points, and generally related to adults. For some pathogens (*V. cholerae*, *Shigella* and norovirus GII), we assumed that prevalence of infection did not differ between diarrhoea patients seeking medical care and those not seeking medical care. These estimates also relied on self-reported diarrhoea which may be subject to recall bias (Zafar et al., 2010),[6] and we applied estimates for duration of shedding and duration of symptoms that were not always Dhaka-specific. The estimates also did not fully reflect the complexities of co-infection with multiple pathogens, intermittent shedding or immunity due to previous infection. Exposure estimates were based on self-reported contact with drains with no structured observation, as would be preferable. Dose-response relationships were generally based on challenge studies involving adults from high-income countries, so it is uncertain how applicable they may be to children or to a low-income area of Dhaka more generally, where longer term impacts of repeated enteric infection may also play a role. Standardising DALY assumptions across all diarrhoeal pathogens also belies the differential impact of pathogen-specific diarrhoea cases. Moreover, the DALY calculations only consider acute impacts, and did not account for childhood growth impairment and other sequelae associated with repeated enteric infections (Troeger et al., 2018).

Model validation. Only partial validation of the model was possible based on an assessment of faecal pathogen and indicator concentrations in drain water. While we could check the plausibility of disease burden outputs, this only confirmed that results were within credible ranges. The microbiological methods underpinning the validation of pathogen and indicator concentrations also had a number of caveats, including uncertain limits of detection and the assumption that what was detected and quantified in the environmental samples was 100% infective. While most of the target organisms were human specific, animals are potential additional sources for *Giardia* and *E. coli*; however, the animal population in the study site was relatively small.

Generalizability of findings. The model set-up and its application had a localised focus, and hence there is a question around wider relevance of the findings. We chose a study site that was a relatively 'simple' catchment with limited inflows of wastewater and pathogens, but recognize that many urban environments in Dhaka and other large cities are significantly more complex. Hence, the complexity of modelling will increase when attempted in other contexts. The results produced by the model may be quite specific to the study site. Different contexts will vary in infrastructure, exposure behaviours, epidemiology and climate. Nonetheless, one of the strengths of a systems modelling approach is that it possesses flexibility to be applied to different contexts so long as the assumptions and inputs are tailored accordingly.

Additional transmission routes. By virtue of the study site, the modelling focused just on local transmission of faecal pathogens from sanitation systems via open drains. In Dhaka – as in other urban centres – high numbers of faecal pathogens and indicators have been detected in drinking water, on fresh produce, and soils (Amin et al., 2019), while person-to-person transmission also represents an important route for infection for some of these pathogens. These transmission pathways may well be impacted by local sanitation interventions. In the study site, rubbish and sediment were frequently removed from the drain and piled up on the roadside, and the pathogen content in drain sediment samples indicate that these could be an important exposure pathway (Amin et al., 2020). Similarly, throughout the study site, overflowing or backed-up toilets were observed, and visibly damaged water supply pipes were submerged in drains. Underground storage tanks for municipal water were also observed and likely provided sites for contamination. However, in many cases the place at which contamination of food or water occurs may be geographically distant to the point of exposure, making it difficult to evaluate with a localised modelling of sanitation options. Similarly, a localised focus means exported pathogens cannot be easily converted into a health risk to populations outside of the study site. For the same reason, safe management along the whole sanitation chain could not be fully studied and incorporated. Comprehensively incorporating these additional considerations and transmission routes will help build a fuller understanding of faecal pathogens in the urban environment, but will also require more data, more extensive validation and application of modelling to a larger geographic scale.

5. Conclusion

Systems modelling can help build a more complete and evidence-based picture of urban sanitation outcomes in specific contexts and more generally. In this paper, we have applied a systems modelling approach to investigate pathogen flows in a low-income urban area in Dhaka, Bangladesh. The study demonstrated that it was feasible to compare the impact of different sanitation options by modelling expected outcomes, despite the extensive variability and uncertainties associated with many of the parameters involved. This approach can therefore complement other planning and decision tools used to weigh sanitation options and understand how excreta and its associated microorganisms move through the urban environment. Using QMRA to compare the relative effect on annual DALYs of eight sanitation options, our results demonstrated that some options that reduce exposure (for example, blackwater piped directly to ABRs) may provide slightly greater reduction in health risk than septic tanks, and options that include constructed wetlands for effluent treatment can further reduce health risk. However, when poorly managed, such as through inadequate sludge emptying (leading to short-circuiting and wash-out of sludge) or sludge dumping, all of the sanitation options we examined provide inadequate pathogen removal to protect public health.

The modelling approach developed and applied in this study could be strengthened in a number of ways. As this study focused on a bounded area, an important consideration was export of wastewater with high pathogen concentrations to 'downstream' neighbouring areas, pointing to the need to conduct modelling at a city rather than neighbourhood scale, and to include a larger number of transmission pathways. Further research is critically needed to fill the multiple evidence gaps we identified, including: 1) more robust LRVs for sanitation systems of different types and under different management regimes; 2) pathogen flows under flooding conditions; and 3) pathogen shedding and exposure assessments in typical low-income urban environments. Such improved evidence will assist in furthering the potential for systems modelling to provide practical guidance on how sanitation interventions can best be designed and managed to protect the health of urban populations.

Acknowledgements

The following staff members at icddr,b are recognised for their essential contributions to the fieldwork and laboratory work that underpinned this study. This includes: Dr Rashidul and Md Mamun Kabir for supporting the project and providing laboratory space, Dr. Rubhana Raqib for providing laboratory space, Md. Rana Miah and Golam Bashir Ahmed for laboratory procedures, Jamilur Rahman, Raju Ahmed, Md. Rayhan and all data collectors for carrying out field activities. We are grateful to Dr. Mami Taniuchi from University of Virginia, USA, for sharing the TaqMan card during the pilot phase.

WSUP staff in Dhaka, Bangladesh, in particular Farzana Begum, are thanked for their support with background information, networks and inputs to the research.

Simon Fane of UTS Institute for Sustainable Futures is thanked for his valuable guidance on the research and helpful comments on an early version of the manuscript.

The Expert Group is also thanked for their inputs to the study methodology, to a teleconference sharing the preliminary findings and, for selected members, review of this paper. The Expert Group comprised: Barbara Evans (University of Leeds), Peter Hawkins

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[6] The self-reported prevalence of diarrhoea based on a one-week recall was estimated to be 7.5% across all ages, which aligns with previous surveys based on a 48 h recall (Najnin et al., 2019).

(independent consultant), Sean Tyrrel (Cranfield University), Alexandria Boehm (Stanford University), Nynke Hofstra (Wageningen University), Nick Ashbolt (University of Alberta), Khairul Islam (WaterAid) and Tim Julian (EAWAG).

We also gratefully acknowledge the helpful feedback from four anonymous reviewers.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ijheh.2020.113669.

Funding

This work was funded by Water and Sanitation for the Urban Poor (WSUP), as part of the Urban Sanitation Research Initiative (USRI) (www.wsup.com/research) supported by the UK Department for International Development.

Ethics

Ethical approval was provided by the University of Technology Sydney (UTS HREC REF NO. ETH18-2599). The study protocol was also approved by the International Centre for Diarrhoeal Diseases Research, Bangladesh (icddr,b) scientific and ethical review committees (protocol number 19011).

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