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Quantifying averted disease burden as a performance indicator for water quality interventions: A review of current methodologies and challenges

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Abstract

Quantifying averted disease burden as a performance indicator for water quality interventions: A review of current methodologies and challenges

By Darcy Anderson

Access to a safe, sustainable drinking water supply protects against infectious disease and promotes overall health. While considerable progress has been made towards increasing water access, poor quality and sustainability remain a challenge. Traditional financing and delivery methods pay implementers based on activities and inputs, which offers minimal incentive for ensuring water quality monitoring and sustainable operation. Pay for performance offers an alternative financing strategy that delivers all or a portion of payment based on performance indicators of desired outputs or outcomes, increasing incentives for delivering actual improvements in health. Averted disability adjusted life years (ADALYS) have been used as a performance indicator for environmental health interventions to measure the aggregate burden of disease morbidity and mortality averted as a result of project intervention. Water-related disease burden can be measured for application as an ADALYs performance indicator following either a comparative risk assessment or quantitative microbial risk assessment approach. Comparative risk assessment models disease burden using water source type as a proxy indicator of microbial water quality, while quantitative microbial risk assessment uses contamination levels of indicator pathogens to model disease burden. This paper describes and compares the indicators and risk assessment methodologies. It also summarizes limitations of applying these approaches towards quantifying ADALYs as a performance indicator for water quality interventions.

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1. Background

1.1. Water and health

Access to a safe and sustainable drinking water supply is essential for health. Safe drinking water reduces exposure to pathogens, decreasing morbidity and mortality from infectious disease [1]. Poor water quality is also associated with a range of long-term health impacts, as repeated enteric infections during childhood can lead to malnutrition and permanent sequelae such as stunting and impaired cognitive development [2]. Water quantity is also a critical component of safe water access. Water is necessary for personal hygiene and domestic use, and insufficient water quantity for hygiene is associated with increased risk of dermatologic, ocular [3], and respiratory [4] infections.

Beyond immediate health outcomes, water access has social and economic impacts. For a majority of households in rural areas of low- and middle-income countries, water must be collected from communal sources outside the home, and the burden collection falls disproportionately on women and children [5]. In areas with limited access, women and children may dedicate multiple hours per day collecting water. For children, water collection as a domestic chore has been shown to reduce time spent in school [6], and for women, time dedicated to water collection could otherwise be used for leisure or income generating activities [5].

Safe water access has been recognized by as a human right [7] and established as a development target within both the Millennium Development Goals (MDGs) [8] and the Sustainable Development Goals (SDGs) [9]. While water access has grown substantially in recent decades, an estimated 663 million people globally still lack access to safe drinking water, and significant disparities persist both between and within countries [10]. Furthermore, challenges in maintaining sustainable water supplies likely mean

that access estimates over represent the true proportion of the global population with access to safe, sustainable drinking water [11].

1.2. Challenges for ensuring safe and sustainable access

The United Nations estimates that the water access target under the MDGs was met five years ahead of schedule [12]. However, the indicators for monitoring progress toward the MDG target fall short in measuring water safety and sustainability. The MDGs measure access to safe and sustainable water using indicators created by the World Health Organization (WHO) and the United Nation Children's Fund's (UNICEF) Joint Monitoring Program (JMP). JMP indicators track service level as a proxy for water quality without directly measuring microbial safety [13], although the JMP itself has recognized that indicators of improved/unimproved are poor proxies for microbial safety [14]. Water quality data from multiple countries has indicated that improved water sources often do not meet criteria for microbial safety [15], and Bain *et al.* [16] estimate that more than a quarter of improved sources contain levels of fecal contamination that exceed WHO guidelines.

JMP indicators under the SDGs aim to include microbial safety. New classifications for drinking water under the SGDs define the highest level of drinking water supply as 'safely managed,' where the source is 'located on the premises, available when needed, and free of fecal and priority contamination' [17]. However, consensus on how these new indicators with be measured is lacking, and how microbial safety monitoring will be incorporated into existing monitoring efforts remains unclear [18].

In addition to issues concerning water quality monitoring, high levels of non-functionality and discontinuity of service across water sources have likely resulted in overestimates of coverage [19]. Research has suggested that as many as a third of communal boreholes in sub-Saharan Africa are nonfunctional at any given time [20]. This figure is widely cited, though some have argued that this

estimate is unverified. In practice, functionality is infrequently monitored, and the true proportion is largely unknown, though the consensus within the literature suggests that a significant proportion of water sources are likely nonfunctional at any given time [21].

As water supply systems age, failure to perform necessary routine maintenance, inability of sources to meet growing demand, and failure to monitor source performance all contribute to decreased functionality [22]. Sustainability is being increasingly recognized as a critical element in water supply [23-25]. Despite this recognition, many projects are still implemented without any clear definition of sustainability or explicit provisions to improve long-term functionality and continuity of water sources [23], and most rural communal sources are not regularly monitored for functionality in low- and middle- income countries [26].

2. Incentivizing health gains over service delivery

2.1. Current delivery approaches

Since the 1980's the dominant paradigm for water supply management has been community-based, in which operation, maintenance, and collection of user fees are the responsibility of a local governing body. This paradigm grew out of the theory that national governments had insufficient resources to manage the highly dispersed network of rural water supplies and that local governance was more cost effective [23]. Community-based management also appealed to bilateral aid and non-governmental donors, who were able to implement a project over the course of months to years, before shifting responsibilities to the community and relinquishing any control or responsibility for long-term operation and maintenance [27].

Community managed systems should, in theory, function with local labor and funding user fees to support operation and maintenance [24]. However, multiple studies have shown that user fees for communal sources are often not regularly collected, leading to insufficient funds for routine operation and maintenance. Furthermore, appropriate replacement parts and skilled labor are not always reliably available, limiting capacity for repairs [28-30]. In response to the shortcomings of community managed models, a growing number of studies have criticized the current paradigm of water delivery as insufficient to ensure the safe and sustainable access to water supplies—the prerequisites to genuine health impacts. This has resulted in, calls for alternative models in which implementers provide technical assistance and support in monitoring, operation, and maintenance [24, 27, 31, 32].

2.2. Pay for performance financing

Pay for performance, also called results-based financing, is one alternative financing scheme that addresses the challenges of traditional financing and program delivery [32]. Traditional financing and program delivery models link funding to the completion of specific activities or the provision of specific inputs. Under such a system that emphasizes inputs over outputs and outcomes, there is minimal incentive for long-term monitoring or improving program delivery [33, 34]. In contrast, under pay for performance financing, funders deliver all or some proportion of payment conditional upon implementers achieving pre-specified performance targets. Rather than incentivizing inputs, pay for performance incentivizes outputs or outcomes. Funders and implementers agree to a contract that defines performance indicators and a methodology for determining payment based on achievement of those performance targets, and implementers accept the financial risk of achieving those targets [27].

Pay for performance is designed to improve service through financial incentives for achieving and maintaining indicators that are more closely aligned with the ultimate goals of the project—improved

health and wellbeing [33]. Compared to traditional financing, pay for performance also offers the potential for reduced corruption through improved transparency of payment systems. Pay for performance also can allow for greater autonomy and innovation among implementers, as the focus is on achieving specific objectives rather than the means by which those objectives are achieved [32].

3. Adapting pay for performance financing to water quality interventions

3.1. Designing pay for performance incentives

Precise and transparent definition of performance indicators and payment methodologies is critical for successful pay for performance financing. Clearly defined indicators and incentives help to align risks and goals for all parties, so that when implementers meet established performance targets both parties are satisfied, and project benefits are maximized for the target population [27]. Research has shown that pay for performance systems can lead to unintended distortions in project implementation if performance incentive are poorly defined or misaligned with project goals [35].

Performance indicators must be specific and measurable so that incentives may be delivered precisely and reliably [33, 34]. Furthermore, indicators should be selected to measure outputs or outcomes that lie along the causal chain to achieving project objectives. Where the relationship between performance targets and desired outcomes is not robust, pay for performance financing may promote inefficient or even counterproductive use of resources by project implementers [32].

3.2. ADALYs

Averted disability adjusted life years (ADALYs or DALYs averted) have been proposed as a performance metric for evaluating environmental health interventions to improve indoor air pollution [36]. DALYs are commonly used to measure the burden of morbidity and mortality associated with various health conditions and can also be used to estimate the health impacts of environmental risk factors [37-41]. ADALYs measure disease burden in DALYs averted as a result of a particular health intervention [36].

The DALY metric totals the number of life-years lost due to premature death, plus the burden of reduced quality of life by adjusting time spent living with a disease using a disability weight. Because DALYs are not specific to particular health conditions, the disease burden of multiple conditions can be combined to assess the overall health impact of a single environmental hazard [42]. This makes ADALYs well suited to serve as a performance indicator to measure the aggregate health impacts of a single intervention and allow for broad comparability and applicability across different types of interventions [36].

3.3. Developing pay for performance financing for water quality interventions

Two distinct approaches exist within the literature for evaluating performance of water quality interventions and estimating water-related disease burden, and both may be applied to measuring ADALYs as a performance indicator. Interventions to improve water quality and access in low- and middle-income countries have commonly been evaluated using JMP indicators [13], and the Global Burden of Disease studies [37-40] have used JMP indicators to estimate disease burden using a comparative risk assessment approach.

Alternatively, water quality in supply and distribution systems supply systems have also been evaluated using microbial contamination indicators, following safety standards established in the WHO's

Guidelines for Drinking-water Quality [43]. Microbial indicators can be used to estimate water-related disease burden using quantitative microbial risk assessment (QMRA) [44].Global data on microbial water quality are scarce, so assessments of the global disease burden using microbial data are infeasible [16], although QMRA has been previously used to measure disease burden as a performance indicator of water supply systems in low-income settings [45].

Both methodologies can be applied towards estimating ADALYs as a pay for performance indicator. Subsequent sections of this paper discuss the indicators and risk assessment methodologies used to estimate disease burden under these two approaches. The limitations of the indicators and risk assessment methodologies are described, as well as the challenges associated with assessing exposure to water-borne disease agents. Finally, the feasibility of applying each of these approaches to estimate ADALYs as a pay for performance indicator for water quality interventions is discussed.

4. Indicators for monitoring drinking water quality

4.1. Proxy indicators under the WHO/UNICEF's Joint Monitoring Program

The JMP classifies water sources as either improved or unimproved based on whether sources are "protected from outside contamination" [13]. Systematic reviews have further refined JMP categorizations by separating piped household connections from other improved community sources or by considering additional improvements in water quality associated with point-of-use household treatment and safe storage [46-51].

Compared to collection of water samples and microbial testing, JMP indicators serve as a rapid and relatively simple water quality monitoring tool. However, without testing microbial safety, JMP

indicators may mistakenly classify improved water sources as safe despite the presence of fecal contamination [16], and the converse is also possible, where unimproved water sources may actual comply with microbial safety guidelines.

4.2. Microbial indicators in the WHO's Guidelines for Drinking-water Quality

For individual water supply systems, the WHO's *Guidelines for Drinking-water Quality* outline performance indicators based on concentrations of pathogenic indicator species [43]. Testing for all possible water-borne pathogens is infeasible, so monitoring of microbial safety of water quality is conducted by measuring exposure to specific indicator species, or reference pathogens. Indicator species a subset of water-borne pathogens known to persist in the environment and be sensitive to removal or deactivation by treatment, and for which sufficient data are available to characterize a dose-response relationship [43].

Monitoring a subset of indicator species to represent overall disease risk is done under the rationale that adequate control of indicator species will also result in adequate control of other pathogens [43]. However, distribution and concentration of indicator species relative to other pathogenic species differs across environmental contexts [45], and different water treatment strategies are not equally efficacious across viruses, bacteria, and protozoa [52, 53]. Studies of diarrhea etiology conducted across seven countries in sub-Saharan Africa and South Asia identified rotavirus, *Cryptosporidium,* enterotoxigenic *Escherichia coli*, and *Shigella* as the four most common etiologic agents across all sites, although the fraction of diarrhea attributable to these four agents was generally less than half of all cases at most sites, and etiologic agents forming the remainder of disease burden varied significantly across all sites [54]. Monitoring and disease burden estimates derived using a reference pathogens fail to capture disease burden contributed by non-indicator species [45].

5. Risk assessment approaches to quantify water-related disease burden

5.1. Overview of existing risk assessment methodologies

Water quality-related disease burden is commonly quantified in one of two ways: using JMP classifications or microbial contamination indicators. JMP indicators have been used by the Global Burden of Disease studies following a comparative risk assessment approach [37-40]. Alternatively, the WHO's *Guidelines for Drinking-water Quality* use microbial indicators to set performance targets for water safety and model disease burden using QMRA [43].

Disease burden models differ across approaches, but each methodology yields a burden estimate in DALYs associated with water quality exposure. In both cases models estimate diarrheal disease burden only [41, 55]. The precise burden of all water-related disease is difficult to estimate. A variety of adverse health conditions related to microbial and chemical contamination, as well as poor access to sufficient water quantity, all contribute to the total burden of water-related disease, and existing data are insufficient to quantify their precise burden [56, 57]. However, diarrheal diseases are considered to form the majority of water-related disease burden, and approximately 94% of diarrheal burden is attributable to poor water, sanitation, and hygiene (WASH) conditions [57]. The following methodologies are therefore expected to underestimate the overall water-related burden of disease.

5.2. Comparative risk assessment methodologies

Comparative risk assessment quantifies disease burden by assessing the risk of current exposure scenarios relative to an ideal hypothetical minimum risk scenario. Relative risk assessments are derived from systematic reviews comparing risk across different levels of JMP service provision [41]. Prüss-Üstün *et*

al. [41] estimate the global burden of diarrheal disease in 145 low- and middle-income countries following a comparative risk assessment approach. Point-of-use filtration or boiling plus safe storage is considered the ideal minimum exposure scenario, while non-continuous basic piped water and improved and unimproved community sources are considered as alternative exposure scenarios. Relative risks are used to calculate a percent fraction of total diarrheal disease burden attributable each component of WASH as follows [41]:

$$PAF = \frac{\sum_{i-1}^{n} p_i(RR_i - 1)}{\sum_{i-1}^{n} p_i(RR_i - 1) + 1} \qquad (1)$$

where p_i represents the proportion of the population exposed, RR_i the relative risk associated with each exposure category compared to ideal exposure, and n the total number of exposure categories. Total attributable water-related disease burden *(AB)* estimates are obtained by multiplying total national diarrheal disease burden in DALYs (*B*) by the percent fraction attributable to poor water quality *(PAF)*:

$$AB = PAF * B \qquad (2)$$

Relative risks are derived from a systematic review by Wolf and coauthors [50], who measure risk across six categories of source and treatment type: unimproved, improved community source, basic piped water, higher quality piped water, household chlorination or solar treatment, and household filtration. Piped water supplies are divided into basic service (i.e. non-continuous service that requires storage within the home) and higher-quality piped water (i.e. continuous supply of safe quality with no storage required) to reflect the low reliability of piped distribution systems in many low- and middle-income countries.

Additional benefits of household water treatment plus the effects of safe storage are also considered, with safe storage represented as a binary covariate when interventions provided a safe storage container or

contained a storage vessel as an integrated component of the treatment mechanism. Effect estimates were subsequently adjusted for bias associated with lack of blinding in included studies (Table 1). Prüss-Üstün *et al.* exclude comparisons with higher-quality piped water, as effect estimates were based on a single study [58]. Instead, household filtration plus safe storage is considered as the ideal baseline exposure scenario.

Table 1. Effect estimates for epidemiologic studies of water quality interventions, adjusted for non-blinding. Adapted from Wolf *et al.* (2014) [50]

	Intervention exposure				
Baseline exposure	Improved community	Piped, basic	Piped, higher-quality	Chlorination or SODIS *+ safe storage*	Filtration *+ safe storage*
Unimproved	0.89 (0.78-1.01)	0.77 (0.64-0.92)	0.21 (0.08-0.55)	0.99 (0.76-1.27) *0.84 (0.61-1.16)*	0.66 (0.47-0.92) *0.55 (0.38-0.81)*
Improved community		0.86 (0.72-1.03)	0.23 (0.09-0.62)	1.11 (0.85-1.44) *0.94 (0.68-1.30)*	0.74 (0.52–1.05) *0.62 (0.42-0.93)*
Basic piped			0.27 (0.10-0.71)	1.29 (0.95-1.74) *1.09 (0.76-1.56)*	0.85 (0.58-1.25) *0.72 (0.47-1.11)*

Relative risk estimates shown with 95% confidence intervals. Italics indicate the additional effects of safe storage after point-of-use treatment.

5.3. Limitations of comparative risk assessment

Comparative risk assessment approaches rely on the robustness of relative risk estimates from systematic reviews, and Wolf *et al.* [50], along with other systematic reviews [47, 49, 59], note that the quality of evidence to date is poor. Estimates of WASH-related disease burden vary significantly depending on the methodologies and effect estimates used, and effect estimates show considerable heterogeneity across meta-analyses [60]. Research on the effectiveness of water quality interventions has a history of disagreement, particularly with regard to household water treatment and the added value of household connections [59, 61, 62], and some have also questioned the contribution of water quality overall to the global disease burden [51, 63].

Central to this debate are questions surrounding the rigor of published studies. The body of literature on water quality contains a large number of observational studies and non-randomized trials. Inclusion of these studies increases the number of comparisons available for analysis but may bias effect estimates due to their lower methodological quality [60], although meta-analyses excluding observational studies have found similar effect sizes for household water treatment interventions [49]. Lack of blinding in a majority of studies remains a significant source of bias. Due to the nature of intervention delivery for community-based water quality improvement, blinding is rarely possible. However, lack of blinding combined with self-reported diarrhea outcomes raises concerns over reporting bias from participants and study personnel [62]. In a meta-analysis of blinded and unblended trials across a wide variety of disciplines, Wood *et al.* [64] estimate that self-report in unblinded trials may overestimate health outcomes by as much as 25%. In a 2015 review of point-of-use treatment by Clasen *et al.* [49], over 80% of interventions were non-blinded. While pooled effects estimates suggest that water quality interventions reduce diarrhea, blinded trials fail to demonstrate any effect.

Wolf and coauthors adjust for lack of blinding following Savovic *et al.* [65]. Adjustment for non-blinding results in effect sizes that are smaller but remain significant for most exposure categories. Similar findings after adjustment for non-blinding are presented in meta-analyses by Hunter [48] and Clasen *et al.* [49]. However, non-blinding adjustments should be interpreted with caution, as adjustments are derived from clinical studies [65], which may have limited applicability to environmental interventions [49, 50].

In addition to bias surrounding lack of blinding, low adherence and adoption, particularly for point-of-use treatment, remains a challenge. Research has shown that even occasional exposure to untreated drinking water can vitiate the protective effects of consistent water treatment in the home [66]. Acceptability and adoption of point-of-use household treatment has been demonstrated to be low in multiple settings and often declines over time [67-74]. Overall lack of reporting on measures of adoption and compliance limits the possibility to adjust for associated biases [50, 60]. While Wolf *et al.* attempt to control for low

adherence by excluding studies with less than 20% adoption, this relatively low exclusion threshold is unlikely to adequately control for low adoption across included studies.

5.4. Quantitative microbial risk assessment methodologies

QMRA quantifies disease burden using dose-response models to estimate disease risk at particular microbial contamination levels [44]. Microbial water quality indicators are used to estimate the dose of pathogens ingested, which can be converted into risk of diarrheal disease using published dose-response curves [43, 75]. An overview of the values and formulae used in QMRA models may be found in Table 2.

Table 2. Overview of values and formulae used in quantitative microbial risk assessment models. Adapted from Brown and Clasen (2012) [55].

	Units	Background value or formula		
Contamination level (C)	Organisms per liter	Measured in baseline/project scenario		
Daily intake (V)	Liters	Measured in baseline/project scenario		
Daily dose (d)	Organisms per day	Equation (3)		
Daily risk of infection ($P_{inf,d}$)	Per day	Equations (4) and (5)		
Yearly risk of infection ($P_{inf,y}$)	Per year	Equation (6)		
Yearly risk of illness	Per year	Equation (7)		
Yearly DALY burden	DALYs per person	Equation (8)		
		Pathogen-specific values		
		Cryptosporidium	*Campylobacter*	Rotavirus
Infectivity constant (r)		0.2	0.145	0.253
Median infective dose (N_{50})	Number of organisms	-	896	617
Probability illness given infection ($P_{ill\ |inf}$)		0.7	0.3	0.5
Disease burden per case (B)	DALYs per case	0.0015	0.0046	0.014 – 0.48
Susceptible fraction (S)	Proportion of population	1.0	1.0	0.06

Brown and Clasen [55] use dose-response relationships for *Cryptosporidium*, *Campylobacter jejuni*, and rotavirus to assess the burden of disease associated with water quality interventions. These indicator species represent protozoa, bacteria, and viruses, respectively, each of which has different levels of susceptibility to removal through different treatment options. Daily exposure is estimated as:

$$d = C_D VA + C_R(1 - A) \qquad (3)$$

where C_D is treated water quality, V daily consumption volume, A the proportion of drinking water consumed daily that is treated, and C_R the pre-treatment water quality. Using daily dose estimates, probability of daily infection is estimated for *Cryptosporidium* using an exponential model as:

$$P_{inf,y}(d) = 1 - e^{-rd} \quad (4)$$

where r is an infectivity constant and d the dose ingested. Daily infection probabilities for *Campylobacter* and rotavirus are estimated using Beta-Poisson models as:

$$P_{inf,d}(d) = 1 - \left(1 + \frac{d}{N_{50}}\left(2^{\frac{1}{r}} - 1\right)\right)^{-r} \qquad (5)$$

where N_{50} represents the median infectious dose, found to cause infection in half of exposed individuals. From daily infection probabilities, yearly infection probability is modeled as:

$$P_{inf,y} = 1 - \left(1 - P_{inf,d}\right)^{365} \qquad (6)$$

Annual infection risk is calculated as the yearly risk of infection multiplied by the probability of infection becoming active disease $(P_{ill|y})$

$$P_{ill,y} = P_{inf,d} * P_{ill \setminus inf} \qquad (7)$$

Finally, annual disease burden in DALYs is estimated using the yearly probability of disease multiplied by a per-case disease burden in DALYs *(DB)* and the proportion of the population estimated be susceptible *(S)*:

$$DALYs = P_{ill} * DB * S \qquad (8)$$

Background data infectivity constants and dose-response relationships used in QMRA models are drawn from primarily from challenge studies in which healthy adult volunteers are dosed with varying concentrations of *Cryptosporidium* [76], *C. jejuni* [44], or rotavirus [44] and subsequently assessed for symptoms of diarrhea. Disease burden models for other pathogens have also been developed and are described elsewhere [77].

5.5. Limitations of quantitative microbial risk assessment

QMRA was developed based on a chemical risk assessment paradigm [44]. However, individual responses vary more across microbial compared to chemical exposures, as virulence depends on characteristics of both the pathogen itself and the immune response of the host [78]. Constants for infectivity and virulence derived primarily from challenge studies of adult populations in high-income countries are applied universally across other populations [44, 75]. In low- and middle-income countries, where individuals may have compromised nutritional status or face additional immune challenges, responses to infection may be more severe [43]. Additionally, evidence from challenge studies in adults is likely insufficient to accurately estimate the burden of disease in children [79], who are at greater risk for infection and bear the majority of global diarrhea burden [80]. QMRA models are also likely insufficient for other vulnerable populations, including pregnant women, the elderly, and other immunocompromised individuals [81, 82].

In addition to variation in host response, virulence also varies by pathogen strain and dose ingested. Different strains of *Cryptosporidium parvum,* for example, have been shown to have a median infectious dose ranging from 9 to 1,042 oocysts, depending on the strain ingested [83]. Systematic reviews have

shown that significant variation in infectivity constants is common across a wide variety of food- and water-borne pathogens, often varying by several orders of magnitude across different strains of the same species [84]. Furthermore, statistical power at low doses is often limited, as large sample sizes are need to define the lower bound of the dose-response curve, but the logistics and expense of challenge studies often prohibits large samples. Wide variation across infectivity constants results in a high degree of uncertainty surrounding dose-response relationships and final disease burden estimates [82].

5.6. Challenges of exposure assessment

Exposure assessment remains a challenge under both QMRA and comparative risk assessment methodologies. In the absence of comprehensive WASH, populations receiving only water quality interventions are expected to be protected from pathogens transmitted through direct consumption of contaminated fluids but remain at risk through transmission pathways related to poor sanitation and hygiene [59, 85]. Fecal-oral transmission of diarrheal disease has been commonly conceptualized using the F-diagram, showing transmission of fecal pathogens through fluids, fingers, food, fields, and flies [86]. Interventions to improve water quality primarily target pathogens transmitted through fluids, while interventions to improve sanitation and hygiene target the remaining pathways [85]. Where WASH conditions are poor, disease risk posed by sanitation and hygiene pathways may overwhelm any potential benefits of water quality interventions, regardless of treatment efficacy [60, 87]. Heterogeneity in the effect estimates across studies is likely in part due to varying environmental contexts and the relative importance of multiple transmission pathways in different settings [87].

Comparative risk assessment methods rely on field studies of water quality interventions conducted in the context of existing external sanitation and hygiene conditions. Compared to clinical challenge studies and QMRA, field studies offer the potential for a more holistic assessment of diarrheal risk under actual exposure conditions [82]. Studies of water quality interventions delivered in community-based settings by

design include the effects of transmission through other non-targeted pathways [87], and disease burden estimates derived from epidemiologic studies therefore reflect the aggregate risk of diarrhea in the context of other unmitigated exposure routes. In contrast, QMRA models consider transmission of infectious agents only from a single point source [44]. While QMRA directly measures water quality, and therefore provides a more accurate assessment of the microbial safety of individual sources than JMP classifications [82], QMRA models do not capture the effects of external pathways, including transmission from person-to-person or through alternative environmental pathways or water sources [44].

6. Quantifying the effects of water quality interventions

6.1. Averted disability adjusted life years as a performance indicator

Comparative risk assessment has previously been used to assess disease burden as a performance indicator for environmental interventions. Anenberg *et al.* [36] describe a methodology to use ADALYs as a performance incentive for reducing indoor air pollution through improved cookstove interventions. Disease burden in DALYs is estimated at both pre- and post-intervention air pollution levels, modeling burden using dose-response curves. ADALYs are estimated by comparing pre- to post-intervention, and attributing reductions in disease burden as averted due to project intervention.

Modeling health impacts as ADALYs associated with reductions in environmental risk indicators allows for timely and cost-effective evaluation of health interventions. Measuring health outcomes directly requires lengthy and expensive longitudinal studies, but risk assessment models can be applied to approximate longitudinal health impacts using more rapid cross-sectional assessments of environmental exposure indicators [88]. ADALYs may be applied as a performance indicator for other interventions targeting environmental risk factors, where sufficient data exist to support disease burden models. Both

comparative risk assessment models using JMP indicators and QMRA models using microbial indicators may be applied to estimate ADALYs as a performance indicator.

6.2. Applying risk assessment methodologies as performance indicators for water quality interventions

Both comparative risk assessment and QMRA can be used to generate ADALYs estimates. JMP or microbial indicators, respectively, are used to estimate baseline and endline disease burden. The difference between pre- and post-intervention DALY estimates is adjusted for the proportion of the target population exposed to unsafe water *(E)* and the proportion using the intervention technology *(U)* to estimate DALYs averted due to project implementation [36]:

$$ADALYs = \left(DALYs_{post} - DALYs_{pre}\right) * E * U \qquad (9)$$

Comparative risk assessment models using JMP indicators also require estimates of national diarrhea burden, as in Equation $DALYs = P_{ill} * DB * S$ (*8*, to be adjusted to reflect burden only within the specific target population. Anenberg *et al.* [36] make this adjustment by calculating the target population size, by number of households targeted and average household size, and assuming that national disease burden is uniformly distributed across the national at-risk population. QMRA models generate burden estimates in DALYs per capita and may be adjusted to the absolute number of DALYs averted similarly adjusting for the number of households targeted and average household size.

7. Discussion

Pay for performance financing can be used to incentivize long term monitoring and continued operation and maintenance of water supplies to improve sustainability. Models and proxy indicators may be used to estimate desired outcomes, but performance indicators and methodologies for determining payment must be specific, well defined, and robustly linked to desired health outcomes to avoid creating perverse incentives [32]. ADALY estimates derived from environmental water quality indicators and risk assessment methodologies are one metric that can be used to measure aggregate disease burden averted as a performance indicator for water quality interventions.

Existing water quality indicators and risk assessment methodologies to calculate water quality-related disease burden have a variety of limitations. JMP indicators rely on source type as a proxy for safety, which is known to correlate poorly with actual microbial risk [16]. ADALYs estimates calculated using JMP indicators will overestimate health benefits where the post-intervention quality of improved sources is unsafe. In instances where baseline quality of unimproved sources may already meet safety guidelines, intervention may not be necessary, and a comparative risk assessment approach will overestimate health benefits. Alternatively, microbial indicators quantify concentrations of specific reference pathogens as a direct measure of disease risk. However, as etiologic agents of diarrhea vary significantly across different environmental contexts [54], monitoring only a select number of indicator species is unlikely to capture the entirety of diarrhea burden where etiologic agents vary widely and are not specifically measured within the suite of indicator species.

Revisions to JMP indicators under the SDGs have the potential to reduce uncertainty in comparative risk assessment models through direct measurement of microbial safety. The shift from a binary improved/unimproved classification to consider safe management and microbial contamination will likely

change how epidemiologic studies of water quality interventions measure exposure. As new indicators are incorporated into field studies, meta-analyses will generate new effect estimates, and inclusion of microbial safety into JMP indicators will help to address the shortcomings of current indicators that rely purely on source type as a proxy for quality. ADALYs estimates from using microbial indicators may prove more accurate where the etiologic agents responsible for a majority of the local burden of diarrhea can be identified, and indicator species specifically selected to reflect the most salient etiologic agents in the local context. Models for only three indicator species are described here, but other models for common diarrheagenic agents have also been developed [44].

Risk assessment methodologies under both approaches are limited by the background evidence on which models are based. Comparative risk assessment is limited by the poor quality of published literature. Lack of blinding and self-reported diarrhea outcomes remain a significant source of bias in effect estimates from epidemiologic evaluations of water quality interventions [62], and these biases subsequently compromise the validity of burden estimates derived from comparative risk assessment approaches. Development of objective measures of diarrhea outcomes remains a challenge within WASH assessment but would help to address bias in effect estimates and disease burden models.

Variation in both host response and virulence of diarrhea pathogens makes universal application of QMRA challenging. QMRA models rely on clinical challenge studies of adult volunteers from high-income countries, but these studies likely have limited applicability towards other populations in low- and middle-income settings. Children, especially those facing malnourishment or additional immune challenges in poor-WASH conditions, may likely have significantly different responses to microbial contamination, and this remains a significant source of uncertainty in ADALYs estimates generated from QMRA models [79]. This limitation is particularly relevant as the majority of diarrhea burden occurs in children [80]. While ethics considerations prevent challenge studies on children and other vulnerable

populations, disaggregated data from outbreak investigations and other natural experiments can help to improve understanding of differential responses to exposure among these populations [79].

Exposure assessment related to consumption of unsafe drinking water remains a challenge across both risk assessment approaches. Even where drinking water is free of contamination, individuals remain at risk of disease transmission related to poor sanitation and hygiene conditions [87]. QMRA models ignore the contribution of disease risk through non-water pathways [44], while comparative risk assessment models rely on epidemiologic studies that account for the contribution of non-water pathways but cannot be reliably generalized to contexts where sanitation and hygiene conditions differ significantly [82]. Wide variation across epidemiologic studies of water quality interventions is likely driven in part by the relative importance of these alternative pathways, although the precise relationship between different transmission pathways and their contribution to the total WASH-related disease burden remains poorly understood [87]. Further research to understand the relative importance and relationships of different components of WASH is needed to better quantify the effects of interventions in different contexts. Ultimately, ADALYs models may need to account for the effects of local sanitation and hygiene conditions to accurately quantify the effects of water quality interventions.

ADALYs models under both risk assessment approaches fail to account for disease burden beyond diarrhea or the effects of water quantity. Previous estimates of WASH-related global disease burden have applied the same relative risk estimates from systematic reviews of diarrheal disease studies to estimate disease burden for other fecal-orally transmitted diseases such as typhoid and salmonella, although there is no empirical evidence to support this assumption [40]. Additional research on the impacts of water quality on other health outcomes would allow for more accurate estimates of total water-related disease burden in ADALYs models.

ADALYs models also do not account for a variety of health and wellbeing outcomes associated with water quantity. Research has shown that greater distances to a water source and longer collection times are associated with adverse child health outcomes [89]. QMRA models do not account for water access, and comparative risk assessment models only rudimentarily control for the effects of water access by considering sources requiring more than a 30 minute round trip for collection time as unimproved [50]. However, hygiene activities are expected to be reduced even where collection time exceeds five minutes in total [90]. Water quantity also provides non-health benefits, such as reduced time spent collecting water, that are difficult to quantify in ADALYs estimates but contribute to overall wellbeing and warrant consideration when delivering water quality interventions.

8. Conclusions

Limitations of current indicators and risk assessment methodologies pose significant challenges for quantifying ADALYs as a performance indicator. The evidence base underpinning QMRA and comparative risk assessment models suffers from poor methodologic quality, lack of generalizability across populations in low- and middle-income settings, and limited understanding of the relationships between WASH-related transmission pathways. However, pay for performance financing does not necessarily require highly precise estimation of averted disease burden, so long as methodologies for determining payment are clearly defined and understood by all stakeholders, and that performance indicators are well aligned with project goals.

ADALYs models can be revised as new evidence is generated. Particularly in the post-2015 development era, the SDGs will change standards for water quality indicators and monitoring, and new evidence will be incorporated to change existing methodologies for quantifying water-related disease burden. As the methodological quality of WASH literature and understanding of WASH-related disease transmission

pathways improves, methodologies for estimating ADALYs can be modified and updated to address current limitations.

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