A systematic review of waterborne disease burden methodologies from developed countries
H. M. Murphy, K. D. M. Pintar, E. A. McBean and M. K. Thomas

ABSTRACT
The true incidence of endemic acute gastrointestinal illness (AGI) attributable to drinking water in Canada is unknown. Using a systematic review framework, the literature was evaluated to identify methods used to attribute AGI to drinking water. Several strategies have been suggested or applied to quantify AGI attributable to drinking water at a national level. These vary from simple point estimates, to quantitative microbial risk assessment, to Monte Carlo simulations, which rely on assumptions and epidemiological data from the literature. Using two methods proposed by researchers in the USA, this paper compares the current approaches and key assumptions. Knowledge gaps are identified to inform future waterborne disease attribution estimates. To improve future estimates, there is a need for robust epidemiological studies that quantify the health risks associated with small, private water systems, groundwater systems and the influence of distribution system intrusions on risk. Quantification of the occurrence of enteric pathogens in water supplies, particularly for groundwater, is needed. In addition, there are unanswered questions regarding the susceptibility of vulnerable sub-populations to these pathogens and the influence of extreme weather events (precipitation) on AGI-related health risks. National centralized data to quantify the proportions of the population served by different water sources, by treatment level, source water quality, and the condition of the distribution system infrastructure, are needed.

Key words | acute gastrointestinal illness, drinking, estimate, review, water, waterborne disease

INTRODUCTION
Waterborne illness continues to be a concern in industrialized countries. In Canada, since the Walkerton waterborne outbreak of Escherichia coli O157:H7 and Campylobacter jejuni in 2000, which resulted in seven deaths and >2,300 illnesses, more stringent regulations regarding water treatment and risk management have been developed and implemented in many municipal systems across Canada (Holme 2003; Rizak & Hrudey 2007). Municipal water treatment requirements in Canada are high; however, there are still thousands of small systems and private wells, some under the direct influence of surface water (GUDI), that are untreated and at risk for pathogen intrusion, and are serving populations who may be at risk for exposure to waterborne pathogens (Richardson et al. 2009; Charrois 2010; Hunter et al. 2011).

The true burden of acute gastrointestinal illness (AGI) due to drinking water in Canada is currently unknown. AGI for the purpose of this review refers broadly to AGI associated with endemic and/or epidemic exposures. A few crude estimates of AGI incidence attributable to tap water consumption have been compiled. Payment (1997) estimated that the burden of tap water related waterborne disease could be costing Canadians several million dollars annually due to the costs associated with lost work days, hospitalizations and costs of medications. In 2008, Environment Canada estimated that as many as 90,000 cases of AGI and 90 mortalities may occur annually in Canada as a result of waterborne disease (Edge et al. 2001). These estimates were based on a previous Centers for Disease Control and Prevention (CDC) estimate of 900,000
cases of AGI and 900 mortalities annually in the United States as a result of waterborne microbial infections (ASM 1999). Vinson (2012) produced a crude annual burden estimate of $2.7 billion due to waterborne disease (not AGI-specific) in Canada from recreational and drinking water exposures. Estimating burden or source attribution of waterborne disease is challenging as there are numerous data and knowledge gaps.

In 2006, Messner et al. (2006), from the United States Environmental Protection Agency (USEPA), proposed a method for developing a national estimate of waterborne disease attributable to the consumption of drinking water. This approach focused on community drinking water supplies, predicting that 16.4 million cases per year of acute gastroenteritis are a result of the consumption of drinking water in the United States. Additionally, Colford et al. (2006) employed a simpler approach, estimating that between 4.26 and 11.7 million cases of AGI are attributable to public drinking water systems in the United States on an annual basis.

The purpose of this research was to systematically review the approaches that quantify the burden of waterborne illness published in the peer-reviewed and grey literature, and to identify key knowledge gaps and critical data requirements for waterborne disease attribution or burden estimates in developed countries.

**METHODS**

**Definition of waterborne disease**

For the purpose of this review, the definition of waterborne disease burden is: the burden of AGI attributable to drinking water exposures. No strict definition of AGI was applied to the specific gastrointestinal symptoms as the goal of the review was to capture all studies that examined AGI burden associated with drinking water. Figure 1 illustrates the main pathways of exposure related to waterborne AGI in Canada attributed to drinking water supplies.

**Research questions**

Three research questions were addressed in this review:

Q1 What waterborne disease estimates (or models) have been published to date in the context of the developed world?

Q2 What epidemiological studies or risk assessments (quantitative studies) have been published to date that examine or quantify the risk of waterborne disease in the context of the developed world that could be used for a burden estimate?

Q3 What expert elicitation studies (qualitative studies) have been published to date that examine the risk of waterborne disease on populations in the context of the developed world that could be used for a burden estimate?

**Review protocol**

A review protocol was developed using the ‘Cochrane Handbook for Systematic Reviews of Interventions’ (Higgins & Green 2011). Literature was searched using the following electronic databases: PubMed/MEDLINE, CAB, and Scopus. Titles, abstracts and keywords were searched with relevant search terms (Table 1) using Boolean techniques. In an effort to capture grey literature and reduce publication bias, Google Scholar and ProQuest Dissertations and Theses were also used. The first 100 relevant articles based on inclusion and exclusion criteria (Table 2) retrieved in Google Scholar were exported to RefWorks (Freeman et al. 2009; Brooks et al. 2013).

A rapid review of the literature identified 1991 as the oldest relevant source year of publication; therefore, a date range of 1990 to present was selected. Only articles published...
Table 1 | Search terms used in systematic review of literature on waterborne illness burden methodologies and data sources to support national burden of waterborne disease estimates, 1990–2013

<table>
<thead>
<tr>
<th>Question</th>
<th>Water terms</th>
<th>Disease terms</th>
<th>Study terms</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. What waterborne disease estimates (or models) have been published to date in the context of the developed world?</td>
<td>Waterborne; ‘waterborne disease’; drinking water</td>
<td>Pathogen; pathogens; ‘gastroenteritis’; ‘AGI’; ‘gastrointestinal disease’; diarrhea; diarrhoea</td>
<td>Estimate; estimates; model; models; burden; intervention; interventions; ‘disability adjusted life years’ (DALY); ‘quality adjusted life years’ (QALY); ‘public health’; ‘risk assessment’</td>
</tr>
<tr>
<td>2. What epidemiological studies or risk assessments (quantitative studies) have been published to date that examine/quantify the risk of waterborne disease in the context of the developed world?</td>
<td>Waterborne; ‘waterborne disease’; drinking water; ‘well water’; ‘surface water’; groundwater; ‘public water’; ‘municipal water’</td>
<td>Pathogen; pathogens; ‘gastroenteritis’; ‘AGI’; ‘gastrointestinal disease’; outbreak; outbreaks; diarrhea; diarrhoea</td>
<td>Intervention; interventions; ‘randomized control’; household; community; ‘epidemiologic study’; ‘epidemiologic studies’; ‘public health’; ‘attributed risk’; incidence</td>
</tr>
<tr>
<td>3. What expert elicitation studies (qualitative studies) have been published to date that examine the risk of waterborne disease on populations in the context of the developed world?</td>
<td>Waterborne; ‘waterborne disease’; drinking water; ‘well water’; ‘surface water’; groundwater; ‘public water’; ‘municipal water’</td>
<td>Pathogen; pathogens; ‘gastroenteritis’; ‘AGI’; ‘gastrointestinal disease’; outbreak; outbreaks; diarrhea; diarrhoea</td>
<td>‘Expert opinion’; ‘expert elicitation’; estimate; estimates; model; models; ‘qualitative method’; ‘qualitative methods’; qualitative; ‘public health’; survey; surveys</td>
</tr>
</tbody>
</table>

Table 2 | Inclusion/exclusion criteria for a systematic review of literature on waterborne illness burden methodologies and data sources to support national burden of waterborne disease estimates, 1990–2013

<table>
<thead>
<tr>
<th>Criteria</th>
<th>Population</th>
<th>Study design</th>
<th>Disease estimates</th>
<th>Outcomes</th>
</tr>
</thead>
<tbody>
<tr>
<td>Q1a</td>
<td>Must be representative of the Canadian context: a) Restricted to the following regions/countries: North America, Europe, New Zealand, Australia, Japan, Singapore; b) Using any drinking water source (private, public, regulated, unregulated etc . . .) – (exclude articles that only examine recreational water sources)</td>
<td>Any design (qualitative or quantitative data or mixed methods)</td>
<td>Waterborne disease (do not restrict studies that examine foodborne and waterborne in same study)</td>
<td>Proposes a model or methodology for estimating the burden of disease on a population</td>
</tr>
<tr>
<td>Q2</td>
<td>Must be representative of the Canadian context: a) Restricted to the following regions/countries: North America, Europe, New Zealand, Australia, Japan, Singapore; b) Using any drinking water source (private, public, regulated, unregulated etc . . .) – (exclude articles that only examine recreational water sources)</td>
<td>Any epidemiological studies, analysis of outbreak data</td>
<td>Waterborne disease (do not restrict studies that examine foodborne and waterborne in same study)</td>
<td>Calculates an attributable risk/disease incidence as a result of drinking a particular water source</td>
</tr>
<tr>
<td>Q3</td>
<td>Must be representative of the Canadian context: a) Restricted to the following regions/countries: North America, Europe, New Zealand, Australia, Japan, Singapore; b) Using any drinking water source (private, public, regulated, unregulated etc . . .) – (exclude articles that only examine recreational water sources)</td>
<td>Any qualitative studies</td>
<td>Waterborne disease (do not restrict studies that examine foodborne and waterborne in same study)</td>
<td>Uses qualitative methods to estimate waterborne disease impact on a population</td>
</tr>
</tbody>
</table>

*Q1–Q3 refer to the questions outlined in Table 1.*
in English were included. As suggested by Higgins & Green (2011), all relevant journals and conference proceedings that were cited numerous times during the search were hand-searched to ensure no articles were missed. In addition, the reference sections of all ‘finalist’ articles were hand-searched to identify any further relevant studies (Sargeant et al. 2006).

All articles recovered were saved and exported to RefWorks. The search results were merged and deduplicated. All titles and abstracts of articles/publications were preliminarily screened for relevance based on the inclusion/exclusion criteria specified in Table 2. Any title and/or abstract that did not clearly meet the criteria were eliminated in screen 1. Three rounds of relevance screening were performed by three independent researchers on titles and abstracts only. Disagreements were resolved via discussion among reviewers. After the third screen, full articles were examined for relevance. All relevant articles after this fourth screen proceeded to the data extraction phase.

Search terms and inclusion/exclusion criteria

The search terms (Table 1) were divided into three categories: water-related terms, disease terms and study terms. When performing the searches, these terms were separated by the Boolean terms OR/AND. Preliminary inclusion criteria for screen 1 are presented in Table 2. The screening process and results are illustrated in Figure 2.

After the third screen, extraction was based on two questions. If an article did not meet the requirements for these two questions it was excluded from data extraction.

Question 1: Does the article estimate or propose a method to calculate a disease burden due to drinking water? (Include any indicators of disease burden e.g., QALY, disability adjusted life years (DALY), cost, lost work days, risk of infection/illness, etc.) (Rice et al. 2006)

Question 2: Does the article fall under one of the seven categories for acceptable types of studies (Table 3)?

Data extraction/analysis

Data extraction focused on the following fields: primary results, methodology/study design, key data sources, study population, study location, drinking water sources, pathogens of concern, definitions of disease, water quality data collected and study limitations. The data extraction process involved categorizing the types of studies that estimate burden (Table 3). Once the articles were classified, a specific set of data extraction questions associated with each category were used to extract relevant information into an Excel database. Results from the data extraction process were analysed by category in a qualitative manner by comparing and contrasting the methods and results.

Qualitative evaluation of two US burden methodologies

Two burden methodologies (Colford et al. 2006; Messner et al. 2006) were compared to understand the different approaches, identify the main model assumptions and data gaps, and examine the sensitivity of the models to data inputs. Three scenarios were developed using the Messner
between variable inputs and key outputs. Sensitivity analyses
were performed by examining the regression relationships
90% credible bounds are presented. Sensitivity analyses
run with 10,000 iterations and the mean, upper and lower
@Risk Software (Version 5.7) add-on. Simulations were
scenarios were created using the Microsoft Excel Palisade
(point estimate and stochastic models were produced). The
burden estimates or methodologies.

RESULTS

For the systematic review, the initial database searches
yielded a total of 17,981 references. Thirty-eight articles
met the selection criteria and were retained for data extrac-
tion (Table 3). The breakdown of articles by category was as
follows: 14 quantitative microbial risk assessment (QMRA)
articles, 13 epidemiological studies (seven randomized con-
trolled trials (RCTs), six sporadic/ endemic studies) and 11
burden estimates or methodologies.

Table 3 | Finalist studies retained for data extraction in the systematic review of waterborne illness burden methodologies and data sources that could be used to support national burden estimates, 1990-2013

<table>
<thead>
<tr>
<th>Category</th>
<th>Number of studies</th>
<th>Authors/publication year</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. Burden methodology: Proposes a methodology for a waterborne disease burden estimate (only methodology- no burden calculated)</td>
<td>3</td>
<td>Chick et al. (2003); Havelaar &amp; Melse (2003); Soller (2006)</td>
</tr>
<tr>
<td>2. Burden estimate: Determines/calculates a waterborne disease burden estimate for a population using existing data (based on AGI or acute gastroenteritis)</td>
<td>8</td>
<td>Colford et al. (2006); Corso et al. (2003); Halonen et al. (2012); Laursen et al. (1994); Messner et al. (2006); Morris &amp; Levin (1995); Payment et al. (1997); Vinson (2012)</td>
</tr>
<tr>
<td>3. Quantitative microbial risk assessment (QMRA)/Risk assessment: Include only QMRAs that use ‘real’ water quality or epidemiological data</td>
<td>14</td>
<td>Astrom et al. (2007); Cummins et al. (2010); Eisenberg et al. (2006); Hartnett et al. (2007); Hunter et al. (2011); Lambertini et al. (2012); Masago et al. (2006); Mena et al. (2008); Payment et al. (2000); Perz et al. (1998); Pintar et al. (2012); Ryu &amp; Abbaszadegan (2008); Signor et al. (2007); Smeets et al. (2007)</td>
</tr>
<tr>
<td>4. Outbreak studies: Analyses a waterborne outbreak in a manner that provides a burden of disease (exclude all that do not calculate a burden or produce an OR only)</td>
<td>0</td>
<td>Colford et al. (2002, 2005a, b, 2009); Hellard et al. (2001); Payment et al. (1991, 1997)</td>
</tr>
<tr>
<td>5. Randomized controlled trials: Intervention study that provides a disease incidence rate and/or relative risk due to exposure to a drinking water source</td>
<td>7</td>
<td>Borchardt et al. (2005, 2012); Eisenberg et al. (2002); Frost et al. (2009); Nygard et al. (2007); Goh et al. (2005)</td>
</tr>
<tr>
<td>6. Sporadic/Endemic epidemiological studies: Studies that look at endemic waterborne disease to determine a disease burden estimate (exclude studies that only produce OR only)</td>
<td>6</td>
<td></td>
</tr>
<tr>
<td>7. Qualitative methodology</td>
<td>0</td>
<td></td>
</tr>
</tbody>
</table>

Russell et al. (2006) and Colford et al. (2006) model frameworks
(point estimate and stochastic models were produced). The
scenarios were created using the Microsoft Excel Palisade
@Risk Software (Version 5.7) add-on. Simulations were run with 10,000 iterations and the mean, upper and lower 90% credible bounds are presented. Sensitivity analyses were performed by examining the regression relationships between variable inputs and key outputs.

Risk assessment and infectious disease transmission models

QMRA is a methodology that can be used to quantify and predict potential health risks and has been identified as a possible method to quantify the burden of waterborne disease (Soller 2006). In this review, 14 risk assessment studies and infectious disease transmission models were identified that quantified waterborne disease burden using site-specific water quality or epidemiological data (Perz et al. 1998; Payment et al. 2000; Eisenberg et al. 2006; Masago et al. 2006; Astrom et al. 2007; Hartnett et al. 2007; Signor et al. 2007; Smeets et al. 2007; Mena et al. 2008; Ryu & Abbaszadegan 2008; Cummins et al. 2010; Hunter et al. 2011; Lambertini et al. 2012; Pintar et al. 2012). All but one study collected or used water quality data as inputs into their assessments. In total, eight were single-pathogen studies, while six were multi-pathogen assessments (Table 4). Studies were performed in the USA, Canada, Ireland, Sweden, UK, France, Japan and Australia.
Two of the studies focused on distribution system risk, while the remainder focused on risk due to the source/treatment component of water supply. Cryptosporidium and Giardia were the most common pathogens (12/14) identified in this review.

Nine of the 14 studies focused on public surface water supplies exclusively (Payment et al. 2000; Eisenberg et al. 2006; Masago et al. 2006; Astrom et al. 2007; Hartnett et al. 2007; Signor et al. 2007; Smeets et al. 2007; Ryu & Abbaszadegan 2008). One study focused on the risk due to public groundwater supplies and the associated distribution systems (Lambertini et al. 2012). Mena et al. (2008) studied the risk associated with a cross-contamination event simulated in a pilot distribution system (pipe loop). One study focused exclusively on private groundwater supplies in the UK and France, where E. coli indicator data were used as a surrogate to estimate risk due to Cryptosporidium in these supplies (Hunter et al. 2011). An Irish study examined the risk due to private wells, public surface water and public groundwater supplies (Cummins et al. 2012). One final study of Cryptosporidium risk for New York City did not specify the nature of the water supply (Perz et al. 1998).

**Epidemiological studies**

Epidemiological studies were divided into three categories: RCTs, outbreak studies and sporadic/endemic epidemiological studies. Seven RCTs, six sporadic/endemic studies and zero outbreak studies were retained for data extraction in this review.

**Randomized controlled trials**

The RCTs included in this review are of published household drinking water intervention trials conducted in contexts relevant to Canadian drinking water systems.
these trials, one group of households was randomly assigned to use an in-home intervention; in these studies, a household water filter was installed under the tap. The other group of households served as the control group and had no intervention. Theoretically, the two groups of households are representative of one another with the only difference being the intervention. Under this assumption, the researchers attribute detectable differences in self-reported illness between the two groups to the presence/absence of the household water intervention (Rothman 2012). However, household intervention trials assume all exposures to water are through a particular tap in the home, therefore failure to account for different exposures can tend to err on the side of no effect of the treatment device (null hypothesis). Consequently, water-attributed risk may be underestimated in these studies.

Five of the seven RCTs in this review were included in previous reviews by Colford et al. (2006) and Messner et al. (2006) as background to their USA burden estimates and therefore will not be discussed in detail here. In addition to these five trials, one trial conducted by Colford et al. (2005a) on HIV+ populations was included in the present review. Since 2006 when the USA burden estimates were published, Colford et al. (2009) conducted a trial on an older adult population that was also included herein. The trials were conducted on households served by community/municipal drinking water supplies using surface water or groundwater under the influence of surface water. No household RCT done on public groundwater sources or private wells was identified. All seven trials are described further in Table 5. These studies represent the only identified published household water intervention trials conducted in developed countries over the past three decades. This is not surprising since these studies are costly and time-consuming (Colford et al. 2006). The more recent trials by Hellard et al. (2001) and Colford et al. (2005b) were blinded and thus represent the ‘gold standard’ of RCTs. Both trials reported no significant difference between the intervention and control groups (Hellard et al. 2001; Colford et al. 2005b). The tap water leaving the plants and distributed throughout the water distribution system during both trials was reported to be of high quality (Hellard et al. 2001; Colford et al. 2005b). In the Colford et al. (2005b) study, the authors suggest that the lack of significant difference in illness rates observed between the control and intervention groups was attributed to the fact that more stringent water treatment regulations were in place. As a result, the authors concluded that the risk to tap water consumers in the system studied was limited. Both the Hellard et al. and Colford et al. studies noted that to detect an expectedly low attributable risk to treated drinking water, a much larger sample size would need to be used, and this would result in substantially greater trial costs (Hellard et al. 2001; Colford et al. 2005b).

Conversely, Payment et al. (1991, 1997) reported that both of their studies documented significant differences between intervention and tap water groups. These studies, however, were not blinded, potentially biasing the results (Noseworthy et al. 1994; Colford et al. 2002). At the time of the studies, it has been reported that the City of Laval distribution system was prone to more low pressure events and main breaks, and maintained a lower chlorine residual than what is generally applied today, potentially contributing to the higher attributable risk reported (Besner et al. 2010).

**Sporadic/endemic epidemiological studies**

Six epidemiological studies that focused on sporadic (endemic) disease were included in the review (Table 6). The studies vary in design, purpose, water source and target population (Table 6). All six studies estimated waterborne disease burden due to a water-related intervention (intentional or unintentional in the case of distribution system events) or the studies linked a particular water source to an increased incidence of illness at the community level (Eisenberg et al. 2002; Borchardt et al. 2003, 2012; Goh et al. 2005; Nygard et al. 2007; Frost et al. 2009). Sporadic/endemic disease studies tend to test various hypotheses until significance is found, consequently suggesting that publication bias may be an issue and that any review of the published literature may miss studies where no effect was observed (and thus not published).

Borchardt et al. (2003) links septic system density with diarrhoeal disease incidence in children. The study suggests that there is an independent relationship between the occurrence of diarrhoea in children and consumption of water
### Table 5: Summary of randomized controlled household drinking water intervention trials examined in the present systematic review of waterborne disease burden methodologies which could serve as potential data sources for a national burden estimate

<table>
<thead>
<tr>
<th>Author/ publication date</th>
<th>Study area</th>
<th>Source water</th>
<th>Treatment</th>
<th>Use of blinding/ use of inactive (sham) treatment device</th>
<th>Study design</th>
<th>Recruitment and response rate</th>
<th>Study population</th>
<th>Study dates</th>
<th>Length of follow-up</th>
<th>Sample size (HH/ individuals)</th>
<th>Drinking water treatment methods evaluated</th>
<th>Attributable risk % due to drinking water</th>
</tr>
</thead>
<tbody>
<tr>
<td>Payment et al. (1996)</td>
<td>Laval, Canada</td>
<td>River water</td>
<td>Fluocacitum, rapic sand filtration, croation, chemical disfination</td>
<td>No</td>
<td>RCT</td>
<td>Randomly selected from study area directory and phoned; 3,741 households contacted/ 606 enrolled</td>
<td>General population: homemenes with one child 2–12 years old (did not exclude immunocompromised individuals)</td>
<td>Jan. 1988–June 1989</td>
<td>12 months</td>
<td>606/2,408</td>
<td>Reverse osmosis, tap water</td>
<td>AR (% of all GI cases): 3% excess in tap water consumer group</td>
</tr>
<tr>
<td>Payment et al. (1997)</td>
<td>Laval, Canada</td>
<td>River water</td>
<td>Fluocacitum, rapic sand filtration, croation, chemical disfination</td>
<td>No</td>
<td>RCT</td>
<td>Randomly selected from study area directory and phoned; 457 households contacted/ 1,360 enrolled</td>
<td>General population: homemenes with one child 2–12 years old (did not exclude immunocompromised individuals)</td>
<td>Sept. 1995–Dec. 1994</td>
<td>16 months</td>
<td>1,062/5,215</td>
<td>Tap water w/purge valve, bottled plant water, bottled purified water, tap water</td>
<td>AR: 12% excess cases in tap group</td>
</tr>
<tr>
<td>Hollard et al. (2003)</td>
<td>Melbourne, Australia</td>
<td>Surface water source from protected catchment</td>
<td>Chlorination</td>
<td>Yes</td>
<td>RCT</td>
<td>Households were mailed invitations; they contacted researchers if they wanted to participate; 10318 households mailed info/600 enrolled</td>
<td>General population: homemenes with one child 2–12 years old (excluding those with immunocompromising conditions)</td>
<td>May 2000–May 2001</td>
<td>12 months</td>
<td>714/988</td>
<td>UV and 1-micron filter, inactive device (control)</td>
<td>UV and 1-micron filter, inactive device (control)</td>
</tr>
<tr>
<td>Colford et al. (2002)</td>
<td>Contra Costa County, USA</td>
<td>Surface water source</td>
<td>Conventional treatment and chlorination (ammonation added during study)</td>
<td>Yes</td>
<td>RCT</td>
<td>Mailed flyer, households contacted researchers; 29,515 flyers mailed out/80 enrolled</td>
<td>General population: homemenes with one child 2–12 years old (excluding those with immunocompromising conditions)</td>
<td>March 1999–Oct. 1999</td>
<td>4 months</td>
<td>50 individuals</td>
<td>Fisher’s exact test, confidence interval, confidence interval, confidence interval</td>
<td>AR: 24% of all GI cases attributable to water; sample size too small to evaluate illness</td>
</tr>
<tr>
<td>Colford et al. (2005a, b)</td>
<td>Denverport, USA</td>
<td>River water</td>
<td>Conventional treatment with granular activated carbon/sand filters and chlorination</td>
<td>Yes</td>
<td>RCT (with cross-over)</td>
<td>Solicitations sent out to 38,353 households; 1,421 households contacted/456 enrolled</td>
<td>General population: homemenes with one child 2–12 years old (excluding those with immunocompromising conditions)</td>
<td>Oct. 2000–May 2002</td>
<td>12 months</td>
<td>456/1,296</td>
<td>Fisher’s exact test, confidence interval, confidence interval, confidence interval</td>
<td>AR: 0.000% to 0.08% of all GI cases attributable to water No sig. difference (IRR = 0.98)</td>
</tr>
<tr>
<td>Colford et al. (2005a)</td>
<td>San Francisco, USA</td>
<td>Surface water source</td>
<td>Chlorination or filtration and chlorination in some cases (18% of the water supply is filtered)</td>
<td>Yes</td>
<td>RCT</td>
<td>Mailed flyer, clinic visit or telephone calls made to recruit patients who were enrolled at Infectious Disease Clinic; 330 screened/50 enrolled</td>
<td>General population: homemenes with one child 2–12 years old (excluding those with immunocompromising conditions)</td>
<td>May 2000–May 2001</td>
<td>4 months</td>
<td>7/50</td>
<td>Fisher’s exact test, confidence interval, confidence interval, confidence interval</td>
<td>AR: 7% (0%–91%) Borderline sig. difference between groups; larger sample needed (OR = 3.34; CI = 0.99–11.21)</td>
</tr>
<tr>
<td>Colford et al. (2005b)</td>
<td>Sonoma County, USA</td>
<td>GUDI supply, chlorination</td>
<td>Yes</td>
<td>RCT (with cross-over)</td>
<td>Mailed flyer to two groups (existing cohort of elderly and from study area); 4,391 households screened/714 enrolled</td>
<td>General population: homemenes with one child 2–12 years old (excluding immunocompromising individuals)</td>
<td>Apr. 2001–Jul. 2006</td>
<td>12 months</td>
<td>714/988</td>
<td>UV and 1-micron filter, inactive device (control)</td>
<td>UV and 1-micron filter, inactive device (control)</td>
<td>AR: 12% excess cases in tap water group</td>
</tr>
</tbody>
</table>

**HH** = Households.  
**RCT** = Randomized controlled design.  
**AR** = Attributable risk.  
**Double-blinded study (study subjects and analysts).**  
**Sig.** = Significant.  
**IRR** = Incidence rate ratio.  
**Triple-blinded study (study subjects, analysts and plumbers that installed treatment devices).**  
**GUDI** = Groundwater under the direct influence of surface water.
<table>
<thead>
<tr>
<th>Author, publication date</th>
<th>Study area</th>
<th>Study design</th>
<th>Study population</th>
<th>Sample size</th>
<th>Water source(s)</th>
<th>Primary results (related to water exposure and health outcome)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Borchardt et al. (2005)</td>
<td>Wisconsin, USA</td>
<td>Case control trial</td>
<td>Children aged 1–18; immunocompetent only</td>
<td>Cases = 153; Controls = 274</td>
<td>Untreated private wells</td>
<td>Population AR*: 11% of diarrhoea in children attributable to drinking private well water positive with faecal enterococci</td>
</tr>
<tr>
<td>Borchardt et al. (2012)</td>
<td>Wisconsin, USA</td>
<td>Community intervention trial</td>
<td>Households, all ages except 13–18; immunocompetent only</td>
<td>440 HH(\text{b}^{1}) (413 adults, 765 children)</td>
<td>Public groundwater supplies</td>
<td>AGI AR due to viruses in groundwater = 6% and 22%, (authors suggest as high as 63% among children &lt;3 years of age when NoV-GI was abundant in drinking water)</td>
</tr>
<tr>
<td>Eisenberg et al. (2002)</td>
<td>California, USA</td>
<td>Cross-sectional survey</td>
<td>HIV + patients</td>
<td>226 HH/ 458 individuals</td>
<td>Boiled, bottled, tap water</td>
<td>AGI AR avoided from boiled water consumption = 64%; Result between always and never drinking boiled water; Adjusted RR(c) boiled = 0.61 (CI 0.29–1.31) AGI AR risk due to bottled water consumption = 26%; Result between always and never drinking bottled water was not significant; Adjusted RR bottled = 1.35 (CI 0.84–2.18)</td>
</tr>
<tr>
<td>Frost et al. (2009)</td>
<td>Northwest city, USA</td>
<td>Community interventional trial</td>
<td>Households with child 2–10 years old and/or adult &gt;65</td>
<td>Controls = 164 HH/361 individuals intervention = 277 HH/711 iIndividuals</td>
<td>Unfiltered chlorinated surface water, surface water treated conventionally with 1 and 2 disinfection</td>
<td>GI AR~ 11%; Incidence rate ratio (IRR) for GI in intervention and control households = 1.12 (CI 0.87–1.43); no significant difference in illness as a result of intervention</td>
</tr>
<tr>
<td>Nygard et al. (2007)</td>
<td>Norway</td>
<td>Cohort study</td>
<td>Household; all ages</td>
<td>Exposed = 616 HH; Unexposed = 549 HH</td>
<td>Study of drinking water distribution system events; water sources not specified</td>
<td>AGI AR: 37% for households exposed to distribution system episodes compared to those that were not exposed</td>
</tr>
<tr>
<td>Goh et al. (2005)</td>
<td>UK</td>
<td>Case control trial</td>
<td>Individuals, all ages</td>
<td>Cases = 175; Controls = 537</td>
<td>Unfiltered chlorinated surface water, membrane treated chlorinated surface water</td>
<td>AGI AR~ −79% (reduction due to membrane treatment); Adjusted IRR=0.207 (0.0099–0.431) of cryptosporidiosis for those exposed to improved water treatment compared to those that were not</td>
</tr>
</tbody>
</table>

\(\text{AR}^{*}\) – Attributable risk.
\(\text{HH}^{b}\) – Households.
\(\text{RR}^{c}\) – Relative risk.
from private wells contaminated by faecal enterococci. The second study by Borchardt et al. (2012) was a large-scale study conducted on public groundwater systems in Wisconsin. The study examined virus occurrence in treated distributed drinking water and pooled AGI-incidence data collected from household surveys. Households were considered ‘exposed’ to viruses if one or more of their community samples were positive for enteric viruses. This is the first study that has attempted to quantify the burden of waterborne AGI due to untreated groundwater. It is also one of the few studies that has measured viruses in groundwater and linked virus occurrence to health (Locas et al. 2008).

In a cross-sectional study of HIV+ patients that reportededly consumed boiled tap water, tap water directly or bottled water, no significant findings were reported regarding disease reduction between the different groups (Eisenberg et al. 2002). Frost et al. (2009) conducted a community intervention trial on a municipal surface water system and did not report any significant findings with respect to disease reduction in the intervention group compared to the control group. Goh et al. (2005) conducted a study on the effect of membrane filtration on cryptosporidiosis in the United Kingdom. It was found that membrane filtration may be responsible for up to a 79% reduction of cryptosporidiosis cases (Goh et al. 2005). The study by Nygard et al. (2007) on drinking water distribution systems in Norway, examined the effects of distribution system events on gastrointestinal health. They reported that up to 57% of diarrhoeal illnesses were attributed to events in the distribution system such as main breaks or maintenance work.

**Burden methodologies and estimates**

This systematic review identified 11 publications that propose and/or produce a waterborne disease burden estimate (Laursen et al. 1994; Morris & Levin 1995; Payment 1997; Chick et al. 2003; Corso et al. 2003; Colford et al. 2006; Messner et al. 2006; Halonen et al. 2006; Soller 2006; Havelaar & Melse 2003; Messner et al. 2006; Halonen et al. 2012; Vinson 2012). Three papers proposed methodologies rather than produce a true burden estimate (Chick et al. 2003; Havelaar & Melse 2003; Soller 2006). Chick et al. (2005) proposed the use of a stochastic infection model to forecast infection probability of pathogens in treated drinking water in the United States. The model accounts for secondary transmission (person-to-person and recontamination of water by infected individuals and recreational water exposure, etc.) which is explained in more detail by Chick et al. (2001). They propose that endemic disease incidence data collected by the CDC along with water quality data collected by the USEPA could be used with stochastic methods to estimate secondary transmission parameters. Cryptosporidium exposure in New York City was used as a case study to verify the approximations in the model. The paper proposes a modelling approach rather than producing an actual estimate of the burden of waterborne disease in the United States (Chick et al. 2003).

In 2003, Havelaar and Melse published a methodology with the World Health Organization (WHO) to apply DALYs as a metric to illustrate the magnitude of disease burden attributed to treated drinking water (Havelaar &
## Table 7 | Summary of published methodologies to attribute waterborne disease to treated drinking water supplies, including important data and knowledge gaps identified by the authors that will help inform future estimates

<table>
<thead>
<tr>
<th>Author, publication date</th>
<th>Location</th>
<th>Measure of burden</th>
<th>Endemic/ outbreak</th>
<th>Type of water supply</th>
<th>Burden approach</th>
<th>Model data inputs</th>
<th>Data/knowledge gaps</th>
</tr>
</thead>
<tbody>
<tr>
<td>Chick et al. (2005)</td>
<td>New York, USA</td>
<td>Risk of infection</td>
<td>Endemic</td>
<td>Community supplies</td>
<td>Propose the use of a stochastic/dynamic model (including secondary infection)</td>
<td>Community intervention trial data; Water quality monitoring data</td>
<td>Secondary infection parameters for microbes; Data on shedding of microbes; Simultaneous illness prevalence rates and water quality monitoring data</td>
</tr>
<tr>
<td>Soller (2006)</td>
<td>USA</td>
<td>Risk of infection/infection</td>
<td>Endemic</td>
<td>Community supplies</td>
<td>Proposes that a microbial risk assessment (MRA) model could be used to inform a national estimate</td>
<td>Pathogen occurrence data; Population at risk; Dose response; Relationships; Secondary infection rates</td>
<td>Not many MRAs on distribution systems; Risk associated with transient small water systems unknown</td>
</tr>
<tr>
<td>Havelaar &amp; Melse (2003)</td>
<td>WHO, global level</td>
<td>Disability adjusted life years (DALYs)</td>
<td>Endemic and outbreak</td>
<td>Not specified</td>
<td>Proposes using DALYs as a measure for disease burden (calculated from number of cases, severity and duration of the specific illness); Final step in a disease model</td>
<td>Cases of infection(s); Duration of illnesses; Severity of illnesses</td>
<td>Need for quality epidemiological data; Severity weights lacking for drinking water-related illnesses; Guidance on how to use limited epidemiological data in MRA is needed</td>
</tr>
<tr>
<td>Colford et al. (2006)</td>
<td>USA</td>
<td>Annual cases of AGI</td>
<td>Endemic</td>
<td>Community supplies</td>
<td>Point estimates based on multiplication of inputs</td>
<td>¾ of community water systems at high and low risk</td>
<td>Distribution systems AGI; Groundwater-related AGI; AGI due to private wells/small unregulated systems; Missing data on vulnerable populations</td>
</tr>
<tr>
<td>Mesnier et al. (2006)</td>
<td>USA</td>
<td>Annual cases of AGI</td>
<td>Endemic</td>
<td>Community supplies; Groundwater and surface water</td>
<td>Monte Carlo modelling approach</td>
<td>Data from Payment RCTs; Expert opinion, USEPA water quality monitoring data</td>
<td>Distribution systems AGI; Groundwater-related AGI; AGI due to private wells/small unregulated systems</td>
</tr>
<tr>
<td>Morris &amp; Levin (1995)</td>
<td>USA</td>
<td>Average number of GI infections</td>
<td>Endemic and outbreak</td>
<td>All tap water</td>
<td>Burden calculated by pathogen (bacteria, viruses, protozoa) using point estimates</td>
<td>Literature data from past epi studies, outbreak data and CDC surveillance data</td>
<td>Large variability in point estimates; Unknowns regarding viral diseases and emerging pathogens such as Cryptosporidium</td>
</tr>
<tr>
<td>Payment (1997)</td>
<td>Canada</td>
<td>Cost associated with GI illness</td>
<td>Endemic</td>
<td>Community supplies</td>
<td>Point estimates based on multiplication of inputs (# cases * cost/case)</td>
<td>Proportion of AGI that is due to tap water (Payment RCTs); Cost per illness; # cases of AGI annually</td>
<td>Health risks due to ageing distribution and treatment infrastructure</td>
</tr>
<tr>
<td>Vinson (2012)</td>
<td>Canada</td>
<td>Cost associated with waterborne illness</td>
<td>Endemic and outbreak</td>
<td>All waterborne disease</td>
<td>Point estimates based on multiplication of inputs (# cases * cost/case)</td>
<td>Data from literature regarding # of cases; UK, US, Canadian healthcare costs; Canadian productivity costs</td>
<td>US data used for estimate</td>
</tr>
<tr>
<td>Corso et al. (2005)</td>
<td>Milwaukee, USA</td>
<td>Cost of illness</td>
<td>Outbreak</td>
<td>Community supply</td>
<td>Phone survey of households affected by outbreak; Point estimates based on multiplication of inputs</td>
<td>¾ population affected by diarrhea (mild, moderate and severe) following an outbreak; Medical costs; Indirect costs (time lost due to illness; time lost due to caregivers to sick)</td>
<td>Inadequate data in hospital and financial records to account for all costs (e.g., physician visits, ambulance costs, self-medication); Illness among visitors not quantified; Data for costs to businesses, government, etc.</td>
</tr>
<tr>
<td>Halonen et al. (2012)</td>
<td>Nokia, Finland</td>
<td>Cost of lost work days</td>
<td>Outbreak</td>
<td>Community supplies (distribution system contamination)</td>
<td>Retrospective study of sick leave following a contamination event; Point estimate of burden (# cases * cost/case)</td>
<td>Sick leave data from workers before, during and following event; GPS coordinates of workers</td>
<td>Sick leave data could not be directly linked to GI incidence because these data were unavailable</td>
</tr>
<tr>
<td>Laursen et al. (1994)</td>
<td>Uggelose, Denmark</td>
<td>Cost of lost work days</td>
<td>Outbreak</td>
<td>Community supply (contamination of a well with sewage)</td>
<td>Retrospective household questionnaire following outbreak; Point estimate of burden (# cases * cost/case)</td>
<td># lost days of productivity; Cost per lost day</td>
<td>Healthcare costs; Questions around heavy precipitation and onset of GI symptoms in the community</td>
</tr>
</tbody>
</table>
The proposed approach was not country-specific, but rather offers a template for others to reproduce estimates for specific pathogens. They propose applying a DALY approach to the final step of a disease model by taking the final number of cases of illness (by pathogen) and multiplying them by severity weights and duration estimates, which they provide, for typically waterborne cases of AGI. This approach is therefore an add-on to the estimation of the cases of illness attributed to a treated drinking water supply, but provides a mechanism for quantifying the economic and public health impact of these cases of disease on a country.

In 2006, Soller proposed a microbial risk assessment (MRA) framework that could be used to inform a national estimate of drinking water-related AGI, provided all pathogens are accounted for in the risk assessment and summed to give an overall estimate. The proposed risk assessment approach involves dividing the model into a source/treatment component and distribution system component that can be modelled either statically or dynamically (including secondary infection).

### Articles that present a burden methodology and produce a country-specific estimate of waterborne disease attributed to treated drinking water systems

Eight studies were identified that provide a methodology and produced an estimate for waterborne illness attributed to treated drinking water supplies in the USA, Canada, Finland and Denmark (Laursen et al. 1994; Morris & Levin 1995; Payment 1997; Corso et al. 2005; Colford et al. 2006; Messner et al. 2006; Halonen et al. 2012; Vinson 2012). Five studies quantify burden on a national level (Morris & Levin 1995; Payment 1997; Colford et al. 2006; Messner et al. 2006; Vinson 2012), while three quantify the burden attributed to specific outbreak situation in Denmark, the USA and Finland (Laursen et al. 1994; Corso et al. 2005; Halonen et al. 2012).

Five studies produced economic burden estimates (following the calculation of cases) and three provided a number of cases of illness or infection (Table 8). The economic measures included costs attributable to lost work days, lost productivity, and medical costs associated with

### Table 8 | Summary of burden of waterborne disease estimates identified in this review

<table>
<thead>
<tr>
<th>Author, publication date</th>
<th>Location</th>
<th>Illness</th>
<th>Burden indicator</th>
<th>Type of burden estimate</th>
<th>Burden estimate</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Cases of illness</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Morris &amp; Levin (1995)</td>
<td>USA</td>
<td>Pathogens that cause GI illness</td>
<td>Average number of infections (by type of illnesses)</td>
<td>National</td>
<td>Moderate to severe: 560,000 cases; Mild to moderate: 7,100,000 cases</td>
</tr>
<tr>
<td>Colford et al. (2006)</td>
<td>USA</td>
<td>AGI</td>
<td>Annual cases of AGI</td>
<td>National</td>
<td>4.26–11.69 million cases</td>
</tr>
<tr>
<td>Messner et al. (2006)</td>
<td>USA</td>
<td>AGI</td>
<td>Annual cases of AGI</td>
<td>National</td>
<td>16.4 million cases</td>
</tr>
<tr>
<td><strong>Economic burden</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Corso et al. (2005)</td>
<td>Milwaukee, USA</td>
<td>Cryptosporidium-related GI illness</td>
<td>Cost associated with illness</td>
<td>Outbreak</td>
<td>96.2 million dollars (cost of Milwaukee outbreak)</td>
</tr>
<tr>
<td>Halonen et al. (2012)</td>
<td>Nokia, Finland</td>
<td>GI illness</td>
<td>Cost of lost work days</td>
<td>Outbreak</td>
<td>1.8–2.1 million Euros (due to outbreak)</td>
</tr>
<tr>
<td>Laursen et al. (1994)</td>
<td>Uggelose, Denmark</td>
<td>GI illness</td>
<td>Cost of lost work days</td>
<td>Outbreak</td>
<td>1.6 million Danish Kroner</td>
</tr>
<tr>
<td>Payment (1997)</td>
<td>Canada</td>
<td>AGI illness</td>
<td>Cost associated with illness</td>
<td>National</td>
<td>40.3–107.5 million dollars</td>
</tr>
<tr>
<td>Vinson (2012)</td>
<td>Canada</td>
<td>AGI illness and respiratory (Legionella)</td>
<td>Cost associated with illness</td>
<td>National</td>
<td>2.7 billion dollars</td>
</tr>
</tbody>
</table>
AGI illness. The studies were conducted in the USA, Canada, Finland and Denmark. The estimates for the United States ranged from 4.26 to 16.4 million cases of AGI attributable to tap water (Morris & Levin 1995; Colford et al. 2006; Messner et al. 2006). The economic burden estimates for Canada ranged from $40.3 million to $2.7 billion Canadian for tap water-related illnesses and all waterborne illnesses, respectively (Payment 1997; Vinson 2012). The remaining three articles estimated burden due to specific outbreak events. The burden due to outbreaks in the USA, Finland and Denmark were 96.2 million US dollars, 1.8–2.1 million Euros (~2.46–2.87 million US dollars) and 1.6 million Danish Kroner (~290,000 US dollars), respectively (Laursen et al. 1994; Corso et al. 2005; Halonen et al. 2012).

The three studies produced by Morris & Levin (1995), Colford et al. (2006) and Messner et al. (2006). Morris & Levin (1995) calculated the number of illnesses by pathogen (Salmonella, Shigella, E. coli, Campylobacter, Giardia, Cryptosporidium, viruses), while Colford et al. (2006) and Messner et al. (2006) calculated the number of cases of AGI due to drinking water. Colford et al. (2006) and Morris & Levin (1995) produced crude point estimates in terms of cases of illness per year. Messner et al. (2006) refined the methodology by incorporating a Bayesian approach to estimating some of the data inputs, and used Monte Carlo simulation to account for the uncertain variability inherent in the data inputs used. Key aspects of each study, and inherent assumptions and knowledge gaps that were identified are presented in Table 7.

Morris & Levin (1995) developed an estimate using surveillance/outbreak case numbers, values from the literature and expert opinions. Values from the literature were used to inform any gaps in surveillance/outbreak data or to substantiate any assumptions. For instance, they used numbers from Britain to estimate the number of total cases of Campylobacter annually in the USA. Expert opinions were used from a study of experts at the CDC (Bennett et al. 1987) who estimated for various pathogens (Salmonella, Shigella, E. coli, Campylobacter, Giardia, Cryptosporidium, viruses) the per cent of cases that were attributable to different sources including water. For each pathogen included in the estimate, they present low, high and crude point estimates for the annual number of cases of illness due to drinking water in the United States. They also predict the number of mortalities for each pathogen based on published mortality proportions for each pathogen. In addition, the pathogens were grouped by the severity of the infection, where a moderate-to-severe infection was defined as ‘case fatality rate of greater than or equal to 0.1%’. Moderate to severe illnesses included: Salmonella, Shigella, E. coli and Campylobacter, while mild to moderate cases included: Giardia, Cryptosporidium, and viruses. Within each category, the results were summed to give a total number of moderate-to-severe cases, and mild-to-moderate cases of illness due to drinking water.

The Colford et al. (2006) approach is based on the use of data from five randomized controlled trials (RCTs) that have estimated the proportion of cases of AGI attributable to drinking water (Payment et al. 1991, 1997; Hellard et al. 2001; Colford et al. 2002, 2005a, b). This proportion is then applied to the estimated number of cases of AGI per person annually in the USA. Using the percentage of the US population supplied by municipal systems and data on the number of systems that did not consistently meet regulatory requirements, they estimated a proportion of the population being served by ‘high risk’ and ‘low risk’ systems (a 10-fold difference in risk between these). Similar to the Soller (2006) approach, they separate risk into ‘distribution system risk’ and ‘source/treatment risk’. Various scenarios are modelled by adjusting the risk attributed to the distribution system component and source/treatment component as well as the population exposed to high/low risk source waters. Colford et al. (2006) note that the approach could be refined to account for vulnerable populations (the young, elderly and immunocompromised) as data become available.

The Messner et al. (2006) approach is similar to the Colford et al. (2006) approach. Specifically, Messner et al. (2006) divide the estimate into a source/treatment component and a distribution system component. They also use data from two RCTs (Payment et al. 1991, 1997) to estimate the risk of AGI attributable to tap water consumption. Using expert opinion, they subsequently rank US community drinking water supplies with respect to the system studied in the RCTs to establish AGI incidence distributions. Examination of US monitoring data along with expert opinion is used to quantify the variability in
source/treatment and distribution system AGI risk. The final estimate is calculated by applying these inputs to a lognormal distribution (researchers assume AGI incidence is lognormally distributed) to produce an estimated number of cases per person-year of AGI due to drinking water. The number of cases per person-year is translated to the proportion of the population served by community water supplies (CWS) in the United States.

As presented in Table 5, five studies included in this review calculate the economic costs of waterborne disease (Laursen et al. 1994; Payment et al. 1997; Corso et al. 2003; Halonen et al. 2012; Vinson 2012). Laursen et al. (1994) conducted an historical follow-up study with a structured postal questionnaire following an outbreak in Denmark. The data collected from study participants included: demographic status, consumption of water, symptoms of gastroenteritis, physician contacts and sick leave taken during a specified time period following the outbreak incident. They found that 1,658 sick days were taken as a result of the outbreak. Payment (1997) used data from the literature and from his two RCTs (from Laval, Canada) to make assumptions regarding AGI attributable to tap water exposure and to subsequently estimate costs associated with waterborne illness in Canada. Vinson (2012) used data from predominantly grey literature, and made numerous assumptions to calculate a crude estimate of non-AGI specific waterborne disease (legionellosis, outbreak and endemic AGI cases, mycoplasma pneumonia and toxoplasmosis) due to all exposures (drinking, recreational exposure, premise plumbing). Following the Milwaukee outbreak, Corso et al. (2003) conducted a phone survey of 613 households to determine the number of people with mild, moderate and severe illness. Average cost of illness was determined by adding direct medical costs and indirect costs associated with loss of productivity. Total costs were determined by taking the average cost of illness multiplied by the burden of illness (mild, moderate and severe). Halonen et al. (2012) conducted a retrospective study of sick leave of Finnish public sector employees prior to, and following, a waterborne outbreak for those exposed and unexposed to the contaminated water. The difference in number of sick days used between the unexposed and exposed was assumed to be attributed to the outbreak and cost of lost work days was calculated accordingly.

Comparison of burden methodologies

The burden approaches presented by Colford et al. (2006) and Messner et al. (2006) for the United States were further explored. These approaches were compared using a US population of 182 million served by community surface water supplies and 90.5 million served by groundwater supplies (Colford et al. 2006). Some of the initial inputs used by Colford et al. (2006) and Messner et al. (2006) were modified to consider relevant data reported since 2006. In addition, a third estimate was produced by a stochastic approach to Colford et al. (2006) method to consider uncertainty and variability of the inputs and output.

All assumptions are outlined in Table 9 for each of the published approaches. Table 10 presents a comparison of the final results of the approaches by Colford et al. (2006) and Messner et al. (2006). In comparing the estimates, the mean number of AGI cases attributable to drinking water from CWS for the United States ranged from 3.09 to 18.48 million per year. This represents a mean value between 0.011 and 0.069 cases per person-year for those people on CWS. Colford et al. (2006) and Messner et al. (2006) reported ranges 0.0156 to 0.0435 cases per person-year and 0.02 to 0.12 cases per person-year, respectively, among the population served by CWS in the United States. The low end of the scenario estimate was derived by applying the Modified Colford Approach. On the upper end, the scenario estimates exceeded the numbers produced by Messner et al. (2006) even though most of the same assumptions were used in the model. This occurred because, for the present analysis, the lognormal distribution was not truncated at two cases per person-year, as done by Messner et al. (2006). It was assumed that the extremes in the distributions modelled could represent vulnerable populations such as young children, the elderly and the immunocompromised and therefore truncating the model was not necessary.

The two Colford et al. (2006) approaches are most sensitive to the attributed risk due to drinking water (based on the values from Payment et al. 1991, 1997; Hellard et al. 2001; Colford et al. 2002, 2005a, b), the overall AGI incidence rate due to all causes and the per cent
of the population consuming water of high risk \( (r = 0.95, 0.17, 0.16) \). In the \textit{Messer et al. (2006)} approach, the model was most sensitive to the log range for source water/treatment variability and distribution system variability for both water types (surface water and groundwater) \( (r = 0.33, 0.29, -0.21, 0.16) \). In addition, the results were sensitive to the ranking of Laval (\textit{Payment et al. 1991, 1997}) and Wisconsin (\textit{Borchardt et al. 2012; Lambertini et al. 2012}) among the US population source/treated waters and distribution systems \( (r = -0.31, -0.32) \).

**DISCUSSION**

This review identified and compared all published methodologies that may help inform waterborne disease burden and source attribution estimates on a national level.
Additionally, relevant epidemiological data from the literature were found and important knowledge and data gaps highlighted by various authors were identified. In Table 11, the authors compiled a global summary of knowledge gaps, which includes specific data needs that may contribute to filling these gaps and help further inform a national estimate of waterborne illness.

The review focused primarily on two recent burden methodologies produced for the United States, as they are the only methodologies to date that quantify waterborne disease burden on a national level (Colford et al. 2006; Messner et al. 2006). These methodologies rely on numerous assumptions with varying degrees of uncertainty, outlined herein, which require further refinement, as acknowledged by the authors. One advantage of both the Colford et al. (2006) and Messner et al. (2006) estimates is that as new data become available, the models can be updated in a transparent manner, to demonstrate progress in our understanding of the greatest sources of risk for waterborne AGI.

The Colford et al. (2006) and Messner et al. (2006) approaches are reliant on the use of data from randomized controlled trials over data produced in other epidemiological studies. These assumptions are justifiable, given the lack of epidemiological data available and given that randomized controlled trials are considered the ‘gold standard’ of epidemiological studies, to attribute risk to exposures. In the Colford et al. (2006) approach, the mean of the attributable risk due to drinking water from all trials (Payment et al. 1991, 1997; Hellard et al. 2001; Colford et al. 2002, 2005a, b) was used as the key illness rate input into the model. In the Messner et al. (2006) approach, the attributable risk due to drinking water from the Payment et al. (1991) trial(s) was applied. The Laval system studied in this trial was subsequently ranked among US systems based on expert opinion (Messner et al. 2006). Given the sensitivity around these inputs (attributable risk and ranking of Laval), it is important to ensure that the assumptions made are representative of the country context. In addition, the issue of study design and the lack of blinding in the original Payment et al. (1991, 1997) studies is worth considering, although it would be difficult to estimate the direction of bias that may be introduced in non-blinded studies.

Due to a lack of data on the potential health effects of consuming groundwater versus surface water supplies, the two water sources were treated equally in the Messner et al. (2006) analysis. Lack of data on the potential health effects related to the consumption of groundwater supplies was identified by both Messner et al. (2006) and Colford et al. (2006) as a significant knowledge gap in the literature. There is a need for both epidemiological data on the health impact of consuming groundwater as well as pathogen occurrence in these supplies. A recent community intervention trial, identified in this review, that focused on small groundwater systems in Wisconsin, USA, is one example of a project that will help fill this data gap (Borchardt et al. 2012). Additional epidemiological studies are required that quantify the risk due to groundwater systems, particularly private wells and small systems.

In the absence of epidemiological data, and based on the literature recovered in this review, QMRA could be used in the development of a national waterborne disease burden estimate for certain systems, such as private wells or smaller

### Table 10

Comparison of Colford et al. (2006) and Messner et al. (2006) model results in terms of cases of AGI per year for a US population served by CWS

<table>
<thead>
<tr>
<th>Approach</th>
<th>Estimated AGI cases due to groundwater</th>
<th>Estimated AGI cases due to surface water</th>
<th>Estimated total cases of AGI per year</th>
<th>Published US estimatesa (cases of AGI per year)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. Colford et al. (2006)</td>
<td>1.39 to 4.04 million</td>
<td>2.93 to 7.81 million</td>
<td>4.32 to 11.85 million</td>
<td>4.26 to 11.69 million</td>
</tr>
<tr>
<td>2. Modified Colford approach</td>
<td>Mean = 0.91 million. Upper and lower 95% credible bounds: 0.62–1.20 million</td>
<td>Mean = 2.18 million. Upper and lower 95% credible bounds: 1.64–3.10</td>
<td>Mean = 3.09 million. Upper and lower 95% credible bounds: 1.82–4.43</td>
<td>N/A</td>
</tr>
<tr>
<td>3. Messner et al. (2006)</td>
<td>Mean = 5.60 million. Upper and lower 95% credible bounds: 1.5–14.43 million</td>
<td>Mean = 12.88 million. Upper and lower 95% credible bounds: 3.6–30.5 million</td>
<td>Mean = 18.48 million. Upper and lower 95% credible bounds: 590,000–6,030,000</td>
<td>16.4 million</td>
</tr>
</tbody>
</table>

aValues refer to those published by Colford et al. (2006) and Messner et al. (2006).
non-municipal systems (Soller 2006). One disadvantage of the QMRA approach is that it is pathogen-specific and in order to calculate an overall burden due to waterborne exposures, a risk assessment would need to be performed for all pathogens and for all exposure scenarios (water sources, consumption patterns, immunocompromised,
elderly, children, etc.) and then the results combined. This is both labour- and data-intensive, but is proposed as an alternative to the modelling approaches published by Messner et al. (2006) and Colford et al. (2006). In many cases, minimal data exist on the prevalence (and enumeration) of enteric pathogens in surface, groundwater, or groundwater under the direct influence of surface water (GUDI) waters. There are very little data on the concentrations of microbial pathogens in groundwater under non-outbreak conditions (Abbaszadegan et al. 2003; Pitkänen 2013; Hynds et al. 2014). This is particularly true for enteric viruses as the detection methods are labour-intensive and costly. Large volumes of water need to be concentrated to detect viruses in groundwater (Abbaszadegan et al. 2003). Enteric pathogen occurrence and concentration data are needed (at a minimum) for key AGI-related pathogens in groundwater: norovirus, Campylobacter, E. coli O157, Cryptosporidium and Giardia (Hynds et al. 2014).

Prevalence data are integral to the development of the exposure assessment component of a QMRA which is likely why only two of 14 studies in this review performed QMRAs on groundwater supplies (either private wells or public systems). One study by Hunter et al. (2011) used E. coli datasets from France and UK private water supplies to infer Cryptosporidium exposure in groundwater. The other (Lambertini et al. 2012) based their QMRA on results of water quality data collected during a community intervention trial in Wisconsin, USA, where virus samples were collected from wells prior to, and after, UV treatment at the wellhead and also at a number of households in the study communities. The samples were analysed using real-time polymerase chain reaction (qPCR) for enterovirus, norovirus GI and GII, adenoviruses, rotavirus and hepatitis A virus. The QMRA was performed using ‘all viruses’, enterovirus and norovirus data. Even in the absence of rigorous data, an advantage of the QMRA approach is that it formally addresses the uncertainty and variability inherent in the data inputs (Cummins et al. 2010).

The Messner et al. (2006) and Colford et al. (2006) approaches focus only on community water systems and neither approach considers the risk due to private wells and small private supplies. For an estimate to quantify burden due to all water supplies, a segmented approach by water system type should be considered. Reynolds et al. (2008), in their review of AGI and non-AGI waterborne burden estimates, suggest that groundwater supplies be addressed separately. In order for risk due to small private supplies to be quantified, information on the number of individuals served by these systems is necessary, in addition to the quality of water from these systems, and the subsequent attributable risk of AGI associated with consuming water from these supplies. Quantifying risk due to small water supplies and private wells is a challenge, given the lack of representative epidemiological data. The RCTs in this review were all performed on large municipal systems using surface water supplies; therefore, the results from these trials are not relevant for small systems and private wells. Only one of the ‘endemic/sporadic’ disease burden papers in this review had findings relevant to small systems, namely private wells (Borchardt et al. 2005). Given the lack of epidemiological data available, QMRA could be an appropriate approach to generate burden estimates for these types of systems.

In both US estimates, the authors quantify the burden due to source water/treatment and the distribution system. This approach is supported by the work of numerous other authors identified in this review (Payment et al. 1991, 1997; Nygard et al. 2007; van Lieverloo et al. 2007; Besner et al. 2010; Lambertini et al. 2012). Although both Messner et al. (2006) and Colford et al. (2006) identified that source/treatment risk and distribution system risk need to be treated separately, due to a lack of data, the authors made the same assumptions for distribution systems as they did for source water/treatment risk.

Since 2006, two studies have attempted to quantify the burden of disease due to drinking water distribution systems (Nygard et al. 2007; Lambertini et al. 2012). Drinking water distribution systems are ageing and there are different events in these systems under normal operation that could potentially cause risk to consumers, such as leaks in the system, main breaks and low pressure events (Kirmeyer et al. 2001; Nygard et al. 2007; Besner et al. 2010; Lambertini et al. 2012). These different types of events can allow the intrusion of pathogens from the surrounding environment into the drinking water distribution pipe network (Kirmeyer et al. 2001). Nygard et al. (2007) studied the health effects of distribution system events in a project in Norway. Lambertini et al. (2012) used a QMRA approach to quantify a
range of AGI attributable to drinking water distribution systems based on the results of the community intervention trial by Borchardt et al. (2012). Studies of this nature will help inform estimates of risk associated with drinking water distribution and the effects of ageing infrastructure. Although these studies are promising, more work is needed in this area to answer the questions around distribution system risk.

This review identified that North American studies that examine the health risks associated with different water distribution system characteristics, such as the effects of pipe materials, pipe age, system size, disinfectant residual, and water age are needed, particularly as our infrastructure systems below the ground age further. The Nygard et al. (2007) study was conducted in Norway on systems that do not maintain disinfectant residuals, and therefore are not representative of North American systems that do maintain residuals (Kirmeyer et al. 2001). In addition, the pipe materials of the systems studied by Nygard et al. (2007) were mostly constructed of steel and polyvinyl chloride (PVC) (Nygard personal communication 2013). It is unlikely that these systems would be representative of North American systems that are predominantly constructed of cast iron, ductile iron and PVC (Folkman 2012) and thus are not proposed for future waterborne disease burden estimation for North American systems.

During the review, numerous articles were retrieved that examined links between water-related illness and environmental factors, such as precipitation and temperature changes. Although this theme was not included in the review questions, weather can influence waterborne disease attribution methods. There are studies that link increases in precipitation to increases in waterborne disease occurrence. Curriero et al. (2001) and Thomas et al. (2006) found that increases in precipitation were linked to waterborne disease outbreaks in the United States and Canada over the previous 54 and 25 years, respectively. Curriero et al. (2001) found that 51% of outbreaks were preceded within a 2-month lag by extreme precipitation in the 90th percentile, while Thomas et al. (2006) linked rainfall greater than the 93rd percentile with a 2.283 odds increase in the chance of an outbreak. Drayna et al. (2010) found that any rainfall 4 days previously was significantly associated with an 11% increase in children’s AGI-related hospital visit. In a study in the north of Newfoundland and Labrador, Canada, high water volume (snowmelt and rainfall) 2 and 4 weeks previously significantly increased the number of weekly GI-related clinic visits by 1.34 and 1.31 times, respectively (Harper et al. 2011). Since extreme precipitation events are becoming more common as a result of changes in climate (IPCC 2007) and evidence linking increased precipitation with waterborne disease, researchers may need to consider how extreme precipitation events influence the attribution of waterborne disease to drinking water supplies, and how this may vary by type of water source, geographical regions of a country, and the susceptibility of treatment systems and distribution systems to weather events and public health risks.

In addition, this review did not include non-AGI illnesses attributed to water such as Legionella, Naegleria fowleri or Pseudomonas aeruginosa. It also did not examine illnesses related to recreational water exposures or exposures through premise plumbing. These pathogens and exposure routes contribute to overall waterborne disease burden in developed countries and are becoming increasingly important causes of outbreaks and endemic cases of disease as reported by US CDC surveillance data (CDC 2013). Consequently, for an all-encompassing burden of waterborne disease analysis, these pathogens and routes of exposure should be considered in future estimates.

Finally, vulnerable populations such as children, the elderly and immunocompromised individuals, is one important aspect that should be considered when quantifying burden of waterborne disease. Colford et al. (2006) pointed out that the estimate they produced did not account for vulnerable populations who are often more susceptible to waterborne disease. Consequently, more data are required regarding the susceptibility of these sub-populations. Colford et al. (2005a, 2009) have conducted two RCTs that focus on HIV+ individuals and the elderly, respectively, to try and fill this data gap.

CONCLUSIONS

The results of this review suggest that there are limited methods available to quantify the burden of AGI illnesses attributable to drinking water at a national level. These vary from simple point estimates, to QMRA, to Monte
Carlo simulations that rely on both assumptions and epidemiological data from the literature.

Additionally, this review highlighted a number of knowledge gaps and data needs that would help further refine inputs into the methodological approaches first proposed by Colford et al. (2006) and Messner et al. (2006). Knowledge gaps and data needs identified include the following.

1. Epidemiological studies that quantify:
   - the risk of AGI due to the consumption of water from small and private water systems;
   - the risk of AGI attributable to drinking water distribution system operation and events;
   - the risk of AGI due to the consumption of groundwater supplies.

2. Pathogen occurrence water quality data, particularly for groundwater supplies.

3. Centralized data on the population served by various water supplies (small systems, large systems, private supplies, transient systems).

4. Data on the state, condition and operation of drinking water infrastructure including:
   - treatment systems in place and treatment capabilities;
   - drinking water distributions’ conditions (age, pipe material(s), frequency of main breaks and repairs, residence times, water age, etc.);
   - condition of private water supplies (quality of well construction, maintenance, etc.).

5. Water consumption-related health effects on vulnerable sub-populations such as children, the elderly and immunocompromised.

6. Research that quantifies the risk of AGI associated with the effects of changing climate (increased precipitation) on drinking water supplies.

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