Using quantitative microbial risk assessment and life cycle assessment to assess management options in urban water and sanitation infrastructures:

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Using quantitative microbial risk assessment and life cycle assessment to assess management options in urban water and sanitation infrastructures: Opportunities and unresolved issues

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Abstract

16   Quantitative microbial risk assessment (QMRA) and life cycle assessment (LCA) are two distinct environmental
17   management techniques that can provide complementary perspectives when assessing management options for
18   urban water and sanitation infrastructure. While QMRA per definition concerns microbial risks, accounting for
19   pathogens in LCA has received little attention. A few case studies, however, have explored the concurrent use of
20   QMRA and LCA. These studies were motivated by the perceived need to address trade-offs between local health
21   burdens associated with pathogens and global health burdens associated with other stressors at different spatial
22   and temporal scales. Along with the LCA, the QMRA results were sought to provide the basis for addressing such
23   trade-offs, rather than for deciding whether pathogen-related adverse effects experienced by specific individuals or
24   populations are acceptable, or which scenario leads to the highest overall health burden for a given community, as
25   is traditionally the case in QMRA. This paper highlights opportunities and unresolved issues related to the
26   concurrent use of QMRA and LCA, such as assumptions in translating chemical and pathogen health impacts to a
27   common metric or other mode structure and parameterisation aspects. Our aim is to facilitate more consistent
28   design and transparent communication of future case studies of this type, and to highlight opportunities for
29   QMRA experts to contribute to LCA method development so as to include pathogen health impacts. While most
30   examples provided in this paper focus on water reuse, the findings apply more broadly and can also be
31   extrapolated to other pathogen exposures in the context of urban water and sanitation systems as well as other
32   contexts.
33
34   **Keywords:** LCA, QMRA, human health risks, pathogen risk, pathogen impact, trade-off

Introduction

36   Quantitative microbial risk assessment (QMRA) and life cycle assessment (LCA) are among the environmental

37   management techniques used to support decision-making regarding urban water and sanitation systems, including

38   water reuse (Xue et al. 2015). QMRA offers a structured approach to assess human health risks that arise from the

39   exposure to pathogens. QMRA models describe the cause-effect chain starting from the pathogen sources and

40   ending with the adverse effects of pathogen exposures on human health. The application of QMRA is

41   commonplace in the assessment of urban water and sanitation systems (e.g. Amha et al. 2015, Liu and Persson

42   2014, Sales-Ortells and Medema 2015, Schoen et al. 2014, Symonds et al. 2014, Xue et al. 2016). LCA offers a

43   structured approach to assess the potential environmental impacts of products (i.e. goods and services), where the

44   unit of analysis is the life cycle, or supply chain, of the product under consideration. Life cycle impact assessment

45   (LCIA) models describe cause-effect chains starting from resource use and emissions and ending with potential

46   impacts on various areas of protection (i.e. human health, natural environment, and natural resources). Also LCA

47   has been used extensively to assess urban water and sanitation systems, amongst others for water reuse (e.g.

48   García-Montoya et al. 2015, Hendrickson et al. 2015). While QMRA per definition concerns microbial risks, the

49   adverse effects of pathogens on human health are not routinely included in LCA, as no standard LCIA

50   methodology for pathogen impact potential is currently available. A few case studies (Aramaki et al. 2006, Harder

51   et al. 2014, Heimersson et al. 2014, Kobayashi et al. 2015a), however, have explored the concurrent use of QMRA

52   and LCA to assess management options for urban water and sanitation infrastructures in terms of the broader

53   adverse effects of pathogens and other stressors on human health and the environment. The main purpose of this

54   article is to highlight and discuss opportunities and unresolved issues related to assessing management options for

55   urban water and sanitation systems in terms of the broader adverse effects of pathogens and other stressors on

56   human health and the environment through concurrent use of QMRA and LCA. But first, we briefly introduce

57   LCA and QMRA at the conceptual level, and provide a summary and analysis of the studies that explored the

58   concurrent use of QMRA and LCA and that were reported in peer-reviewed scientific journals.

Background

60   LCA

61   LCA is a technique for the environmental assessment of products (i.e. goods or services) and generally covers the

62   entire life cycle of a product, from raw material and natural resource acquisition to final disposal. It is also referred

63   to as environmental LCA in order to distinguish it from social LCA and life cycle costing (LCC). The procedure

64   of performing an environmental LCA is described in the ISO standards 14040:2006 and 14044:2006. These ISO

65   standards describe LCA as a compilation and evaluation of the inputs, outputs and potential environmental

66   impacts of a product system. A product system hereby is a collection of processes (i.e. activities transforming

67   flows of material and energy) that models the life cycle of a product and performs one or more defined functions.

68   A key feature of LCA is the functional unit. It represents a quantification of the identified function(s) of the

69   studied product system and serves as a reference to which the inputs, outputs and potential environmental impacts

70   can be related. For example, the functional unit for water reuse scenarios could be the provision of 1 m^3 of non-

71   potable water.

According to the ISO standards, LCA consists of four stages, which interact with one another in an iterative manner. *Goal and scope definition* is concerned with stating the intended application of the LCA study, the reason for carrying it out, to whom and how the results are to be communicated, as well as a number of important modelling specifications including the functional unit, the system boundaries, cut-off criteria, allocation principles (i.e. how to partition the input and output flows of processes between the product under study and co-products), and which options to model. *Life cycle inventory analysis* (LCI) is concerned with quantifying the environmentally relevant resource use and emissions associated with a product system in relation to the selected functional unit. *Life cycle impact assessment* (LCIA) is concerned with translating the resource use and emissions estimated in the LCI into potential environmental impacts, also in relation to the selected functional unit. Since its emergence in the late 1970s, LCA methodology has developed considerably and several life cycle inventory (LCI) databases and LCIA methods are available (Baumann and Tillman 2004). LCIA methods cover a continuously expanding number of impact categories and corresponding characterisation models for the conversion of the resource use and emissions from a product system into potential environmental impacts (Hauschild et al. 2013). Common impact categories used in LCIA include global warming, acidification, human toxicity, land use, eutrophication, water use, land use, abiotic resource depletion, and many more. The models used to describe these impacts in LCA may be at a "midpoint" level (e.g. greenhouse gas emissions enumerated as kg of CO_2-equivalents) or a more meaningful but less accurate "endpoint" level (e.g. climate change impacts on human health estimated in disability-adjusted life years). Disability-adjusted life years (DALY) are a measure of overall disease burden that was developed in the 1990s (Murray 1994). A recent discussion of the concept is provided in Gao et al. (2015). *Life cycle interpretation* is concerned with interpreting the results in order to draw conclusions and is done in between all stages. Figure 1 provides a visual summary of LCA. Two broad types of LCA are attributional and consequential LCA. Attributional LCA describes the environmentally relevant resource use and emissions related to a given product, while consequential LCA describes how the environmentally relevant resource use and emissions will change in response to possible decisions (Finnveden et al. 2009).

QMRA

QMRA is a technique to evaluate the effects on human health resulting from the exposure to representative pathogen members, typically addressing viral, bacterial and protozoan members (known as reference pathogens). Two broad types of QMRA commonly distinguished are static and dynamic QMRA. In static QMRA the exposure of individual hosts to reference pathogen(s) through one or multiple exposure pathways is modelled without accounting for immunity and the secondary spreading of disease. Dynamic QMRA models take into account immunity and the secondary spreading of disease from person-to-person (Eisenberg et al. 2002, Eisenberg et al. 2004, Eisenberg et al. 2008, Soller 2009), and recently also include zoonotic (i.e. animals-to-human) spreading (Waters et al. 2016). Either static or dynamic QMRA models may also be undertaken with point estimates (deterministic) or with distributional (stochastic) parameter values (Medema et al. 2006). In this paper, we focus primarily on static QMRA.

In static QMRA, an obvious assessment endpoint is the single-event probability of infection ($P_{inf,event}$) or illness ($P_{ill,event}$). If specific individuals are exposed to several exposure events during a given period of time (usually a year), it is possible to have the annual probability of infection ($P_{inf,year}$) or illness ($P_{ill,year}$) associated with multiple exposure events as assessment endpoint. Another possible assessment endpoint would be the number of cases of

111 infection or illness per event or year ($N_{inf/ill,event/year}$). This assessment endpoint follows directly from the
112 probability of infection or illness per event or year through multiplication by the number of individuals exposed.
113 Single-event as well as annual probabilities or number of cases of infection or illness can be compared (separately)
114 with corresponding threshold risk values considered acceptable by the regulators/stakeholders. For example, a
115 given QMRA study might address dermal exposure of agricultural workers during irrigation (Al-Jassim et al.
116 2015), or the ingestion of irrigation water by farmers or children playing in fields irrigated with reused water
117 (Symonds et al. 2014). Different exposures (i.e. combination of the reference pathogen, the suite of exposure
118 pathways, and the host) may lead to similar health outcomes (e.g. a certain type of illness). In this case, it is
119 possible to compare different exposures and identify those with the largest human health impact. When different
120 exposures lead to different health outcomes, however, comparison and prioritisation would require weighting or
121 severity factors for each distinct health outcome. The increasing availability of severity factors (e.g. Havelaar and
122 Melse 2003, Havelaar et al. 2003, Kemmeren et al. 2006, Vijgen et al. 2007) enables the translation of a
123 probability or number of cases of infection or illness into a health burden (also referred to as burden of disease),
124 often expressed as DALY. Also health burdens can be compared with corresponding threshold values considered
125 acceptable, such as an annual target of one DALY per million as used by WHO (WHO 2006). Furthermore, health
126 burdens allow for comparison and prioritisation among hazard exposures with different health outcomes. Also, the
127 aggregation of health burdens related to different exposures becomes possible and meaningful. For instance,
128 several QMRA studies estimated the health burden related to a suite of reference pathogen exposures with water
129 reuse (e.g. Ayuso-Gabella et al. 2011, Barker et al. 2013a, Barker et al. 2013b, Chen et al. 2012, Forslund et al.
130 2010, Hamilton et al. 2007) or other water and sanitation systems (e.g. Katukiza et al. 2014, Schoen et al. 2014).
131 In these studies, the focus of the assessment shifted from the acceptability of individual exposures towards the
132 overall community impact associated with a suite of exposures associated with a given management option. Table
133 1 illustrates which assessment endpoints are meaningful in combination with certain assessment purposes.

134 Model structure of static QMRA

135 Beaudequin et al. (2015) proposed a conceptual model for the assessment of health risks associated with
136 pathogens in diverse water reuse scenarios through static QMRA. Here we draw on their work and extend it with a
137 particular emphasis on facilitating the subsequent discussion of concurrent use of QMRA and LCA. Following
138 Beaudequin et al. (2015), we distinguish four sub-models. The *technical system sub-model* (a generalisation of the
139 "pond operation and performance sub-model" of Beaudequin et al. 2015) takes into account key influences on the
140 concentration of pathogens in the water reuse scenarios under consideration. The *exposure sub-model* describes
141 the interactions between the reference pathogens and the environment (i.e. environmental fate and transport
142 between the point of emission and the point of exposure of the host), and between the hosts and the environment
143 (i.e. exposure route, exposure medium, exposure frequency, and exposure volume). The *dose-response sub-model*
144 represents the interaction between each reference pathogen and the host. Both pathogen characteristics and host
145 characteristics influence the response to a given pathogen dose and there is considerable variability regarding both
146 sets of characteristics. However, various host effects are generally not accounted for, such as host immunological
147 status or microbiome, both key determinants in pathogenicity (Hajishengallis et al. 2012, Havelaar et al. 2014,
148 Karlsson et al. 2014). The *risk characterisation sub-model* estimates a number of cases of infection or illness, or a

149 health burden in terms of DALYs based on the probability of infection or illness obtained through the dose-
150 response sub-model. The four sub-models and the relationships among them are visualised in Figure 2.

151   Regardless of the assessment endpoint and purpose, the assessment requires information about the emissions
152 of pathogens, their fate and transport in the environment, the exposure of different host groups to the pathogens, as
153 well as the effects of the pathogens on the hosts. The respective model parameters may vary as there is often a
154 range of operating and exposure conditions, pathogen and host characteristics, and courses of disease that are
155 possibly relevant. Every technical system scenario (e.g. a certain set of operating conditions) related to a given
156 water reuse scenario in principle gives rise to multiple exposure scenarios (i.e. a suite of exposures that each can
157 be parameterised in different ways). For every combination of technical system scenario and exposure scenario,
158 the appropriate (suite of) dose-response sub-model and risk characterisation sub-model should be chosen. Hereby,
159 particular consideration may be given to sensitive or otherwise unique life-stages (e.g. pregnant women, young
160 children, elderly people, other immunocompromised individuals) (Beaudequin et al. 2015). Sensitive or otherwise
161 distinct people may not only be more susceptible to an initial infection (thus requiring a different parameterisation
162 of the dose-response sub-model), but may also be more likely to become symptomatic (thus requiring a different
163 parameterisation of the risk characterisation sub-model). However, only a limited number of dose-response
164 relationships have been developed and published (Beaudequin et al. 2015). This means that one simply has to do
165 with the limited number of available dose-response relationships and document these limitations. All-in-all,
166 accounting for the effect of pathogens in urban water and sanitation systems in principle would require
167 consideration of a suite of QMRA model variants (see also Figure 2), akin to previous screening-level risk
168 assessments (e.g. Sales-Ortells and Medema 2014). Each model variant thereby reflects a specific
169 parameterisation of the technical system, exposure, dose-response, and risk characterisation sub-models. Strictly
170 speaking, all possible model variants would need to be considered in the assessment. This is not practical,
171 however, and only a limited number of model variants can realistically be considered (e.g. Petterson and
172 Stenström 2015).

Studies that have explored the concurrent use of QMRA and LCA

174 A search in the Scopus database (TITLE-ABS-KEY (QMRA OR "microbial risk assessment" OR "microbial
175 risk*" OR "pathogen risk*") AND TITLE-ABS-KEY (LCA OR "life cycle assessment" OR "life-cycle
176 assessment" OR "lifecycle assessment")) in December 2015 yielded three case studies investigating the concurrent
177 use of QMRA and LCA. These studies were motivated by the perceived need to address trade-offs between local
178 health burdens associated with pathogens and global health burdens associated with other stressors at different
179 locations and points in time.

180   Aramaki et al. (2006) contrasted the reduction of adverse health effects related to pathogen inactivation
181 resulting from the installation of an urban wastewater system (estimated based on QMRA) with the increase in
182 adverse health effects associated with other stressors resulting from construction and operation of the treatment
183 plant (estimated based on LCA). Although the QMRA results were directly compared to the LCA results, the
184 QMRA results were not presented as an LCA impact category. Harder et al. (2014) and Heimersson et al. (2014)
185 (one study reported in two parts) sought to account for adverse effects of pathogens in LCA of wastewater
186 management scenarios in order to compare the adverse effects of pathogens on human health and other adverse
187 effects on human health for two wastewater management scenarios. Hereby, the pathogen-related effects were

188 estimated based on QMRA (Harder et al. 2014) and the results were presented as an LCA impact category
189 alongside other impact categories based on LCIA models (Heimersson et al. 2014). Kobayashi et al. (2015a)
190 investigated a scenario where recycled water from a municipal wastewater treatment plant was used to replace
191 water released from a dam to maintain environmental flows in a nearby river. To this end, the yearly health burden
192 associated with the consumption of treated river water and recreational use of the river with and without
193 implementation of the large-scale water recycling project was estimated. The health burden associated with
194 pathogens (estimated based on QMRA) was then compared with human health impacts resulting from the
195 operation of the water recycling scheme and associated with stressors other than pathogens (estimated through
196 LCA). This comparison was intended to reveal trade-off relationships between local impacts (i.e. pathogen-related
197 effects estimated based on QMRA) and global impacts (i.e. other adverse effects on human health estimated
198 through LCA). The overall model structure is strikingly similar in the above three studies. Basically, QMRA was
199 used to estimate the pathogen-related health burden for a number of core processes, while LCA was used to
200 estimate the health burden related to other stressors for both the core processes and the supply-chain processes
201 (see Figure 3).

202 In the study by Aramaki et al. (2006), the pathogen-related health burden estimated based on QMRA
203 represents aggregate effects for the downstream community as a whole. In the study by Harder et al. (2014) and
204 Heimersson et al. (2014), the pathogen-related health burden estimated based on QMRA represents the aggregate
205 health burden for all people possibly exposed to pathogens as a direct result of wastewater management operations
206 (but not supply-chain processes). In the study by Kobayashi et al. (2015a), the pathogen-related health burden
207 estimated based on QMRA represents the aggregate health burden for all people possibly exposed to pathogens
208 through consumption of river water and recreational use of the river. Comparing and contrasting QMRA results
209 with LCA results is possible because the concept of DALY has been adopted in both QMRA and LCA. It was
210 recommended in the literature, however, that understanding the background information on how DALYs are
211 derived is crucial to ensure the consistency of DALYs used in quantitative risk assessment (QRA) and LCA
212 (Kobayashi et al. 2015b). Finally, it should be noted that the case studies that have explored the concurrent use of
213 QMRA and LCA all relied on static QMRA and attributional LCA.

Opportunities and issues related to concurrent use of QMRA and LCA

215 The recent efforts to look into trade-off relationships between pathogen-related and other impacts on human health
216 in the context of urban water and sanitation systems that are considered in this paper illustrate new opportunities
217 for the use of QMRA – amongst others for the assessment of water reuse. Not only can QMRA provide the basis
218 for deciding whether pathogen-related adverse effects experienced by specific individuals or populations are
219 acceptable, or which management option leads to the highest overall health burden for specific individuals or
220 populations – concurrent use of QMRA and LCA can also provide a basis for avoiding problem shifting between
221 pathogen-related and other health burdens. For example, if two water reuse options with different levels of
222 disinfection were considered, it would be possible to investigate whether the reduction of the local health burden
223 associated with pathogens might be offset by an increase of the global health burden associated with the operation
224 of the disinfection process. In the remainder of this paper, we discuss a number of issues to be aware of with
225 regard to concurrent use of QMRA and LCA.

QMRA model specification

When the purpose of a QMRA model is to provide comparison with a target risk level, it is important to identify the model parameterisations that impact the most on specific individuals, generally via some form of sensitivity analysis (e.g. Petterson et al. 2007). When the purpose of a QMRA model is to compare different management options, or to avoid problem shifting between pathogen-related and other health burdens, however, it is important to identify those model parameterisations that contribute most to the overall impact aggregated over all possible exposures, which means that the likelihood (distribution) of each parameter needs to be known or estimated (stochastic QMRA). The most sensitive model parameters may differ between the two cases as a model parameterisation with a lower health risk on a per individual basis may still lead to a higher impact on a population basis if a larger number of individuals are affected.

LCA typically considers routine operations under steady-state conditions when the technical systems operate according to the design specifications. Such practice is perfectly sensible when it can be assumed that routine operations indeed are responsible for the larger part of the impacts. In QMRA, as pathogens generally represent acute effects (one gets infected or not by one exposure), it is important to take into consideration seasonal variations and periods of non-routine operation (e.g. rain events, treatment upsets or sub-optimal performance) (Beaudequin et al. 2015, Nilsson et al. 2007, Signor et al. 2007). This means that, also when QMRA results are to be compared to LCA results, it might be appropriate to consider modelling not only routine operations (as typical for LCA) but to also account for hazardous event periods that may occur, despite the inconsistency that may be introduced if impact categories considered in LCA are estimated based on LCIA models accounting for routine operation scenarios only. Such inconsistency is warranted if short-duration events that may occur on relatively infrequent intervals dominate pathogen risks (e.g. Medema et al. 2006), while for other scenarios (generally well operated drinking water plants) routine operations may still present the largest impacts (e.g. Westrell et al. 2003).

When there is an existing QMRA study that was designed to support threshold comparisons, it may be tempting to convert the single-event or annual probabilities or numbers of cases of infection or illness into health burdens for subsequent aggregation and comparison with LCA results (as is the case for some exposures in Harder et al. 2014). Such practice could be problematic as the parameterisation of the original QMRA may not be according to what would make most sense for a QMRA that is designed explicitly to produce results that are used alongside LCA results.

Choice of dose-response relationships

In QMRA models, dose-response relationships are usually non-linear. As dose-response relationships in LCIA usually are linear, an obvious question is whether dose-response relationships in QMRA could be linearized in the case where QMRA results are to be compared with LCA results (Harder et al. 2016). The application of linear dose-response relationships in LCIA models for human toxicity of chemicals, for instance, is acceptable because the assessment is concerned with chronic effects and the doses considered in a given LCA study usually are on the lower end of the dose-response curve. The assessment of adverse effects of pathogens on human health, however, is mostly concerned with acute effects, and the doses a given host is exposed to can be further up the the dose-response curve (Harder et al. 2016). It is therefore generally not recommended to linearize dose-response relationships for pathogens in QMRA models, not even when the results are to be compared with LCA results (see also Harder et al. 2016).

265   Another important aspect regarding the choice of dose-response relationship is that some dose-response
266   relationships that have been shown to be inappropriate are still in use in the literature. For example, Harder et al.
267   (2014) used a beta-Poisson model for *Norovirus*, even though Teunis et al. (2008) recommended a hypergeometric
268   model, and Messner et al. (2014) provided a simplification of the hypergeometric model in the form of a fractional
269   Poisson model. Although the use of a more appropriate dose-response model by Harder et al. (2014) would have
270   had a negligible influence on the results and conclusions, it appears worthwhile nevertheless to make sure that
271   appropriate dose-response curves are selected and assumptions clarified.

272   Scaling of QMRA results to a functional unit

273   In LCA of urban water and sanitation systems, scaling to the functional unit usually takes place before impact
274   assessment. This is possible because of the linearity of LCIA. Given the nonlinear mathematical relationships
275   involved in QMRA, however, it makes little sense to scale pathogen emissions to a functional unit. In some of the
276   case studies featuring concurrent use of QMRA and LCA, the functional unit of the LCA was therefore chosen so
277   as to represent the full-scale emissions relevant for QMRA. For instance, the functional unit chosen by Aramaki et
278   al. (2006) was the treatment of 50,000 m^3 of wastewater per day during a year, and the functional unit chosen by
279   Kobayashi et al. (2015a) was the provision of 18 GL of reclaimed water per year. Both functional units represent
280   full-scale plant operations during a year. Having the emission inventory based on full-scale plant operations is
281   important (at least for pathogen emissions) because of the non-linear dose-response relationships for pathogens.
282   Nevertheless, it would in principle be possible to scale the QMRA results to any other functional unit, as long as
283   the QMRA itself is based on the full-scale emissions. However, some functional units may be more preferable
284   than others (Harder et al. 2015a). One pitfall in particular is worth highlighting. Say QMRA model calculations
285   are performed based on the full-scale operation of a water reuse facility, but the model results are scaled to a
286   functional unit representing per capita water supply. Even if the units (a health burden per person) might suggest a
287   health burden at the level of an individual, it actually is an average share of an aggregate health burden at the level
288   of a population and through a range of different exposures.

289   Coverage of pathogen emissions

290   The three studies analysed in this paper only considered pathogen emissions from treatment operations (i.e. the
291   "foreground system" in LCA terminology) but not from elsewhere in the supply chain that supports treatment
292   operations (i.e. the "background system" in LCA terminology). Presenting the QMRA results covering only the
293   foreground system in an LCA framework (as was the case in Heimersson et al. 2014 and Kobayashi et al. 2015)
294   might camouflage the fact that the pathogen impact potential presented does not cover pathogen emissions in the
295   background system.
296   In principle, it would be possible to cover pathogen emissions also in the background system, and also in the
297   expanded system when system expansion is applied in LCA. System expansion is one way to facilitate
298   comparison of alternatives with multiple functions in addition to the ones represented by the functional unit. For
299   example, a treatment plant in a water reuse scenario might not only provide water but also wastewater treatment
300   services as well as treatment residuals that can be used in soil improvement. An analyst may want to compare the
301   environmental impacts of supplying 1 m^3 of recycled water, or alternatively, water from a conventional surface
302   water source. For a fair comparison, the environmental impacts of a separate system for the delivery of the same

303   wastewater treatment services and soil improvement should be added to the environmental impacts of the
304   conventional system. This procedure is called "system expansion" or "substitution" and is illustrated in Figure 4.
305   The term system expansion means adding components to the system that does not supply as many functions as the
306   multifunctional system, although the algebra of comparison is just as apt if the environmental burdens of the
307   components are subtracted from the multifunctional system in the calculation.

308   However, pathogen emissions at different locations and points in time cannot be aggregated (and scaled to a
309   functional unit) because of the non-linearity of dose-response models. Rather, emissions at different locations and
310   points in time each require a separate QMRA, the results of which can very well be aggregated (and scaled to a
311   functional unit). Having said the above, it is important to realise that the specific locations and points in time of
312   emissions related to a specific product or service are not specified in LCA, and often cannot be specified due to
313   the nature of the analysis. This hampers the consideration of pathogens in processes other than the core processes
314   in the foreground system, unless the supply-chain is very well known and all the contributing processes can be
315   located. In other words, it is difficult to model the health impact of pathogens in the background system.

316   Returning to the previous example, the comparison might look as follows. For the conventional surface water
317   scenario, the estimation of the pathogen related health burden of the foreground system would encompass
318   pathogen exposure through water consumption. For the recycled water scenario, the health burden would
319   encompass pathogen exposure through water consumption as well as other exposure pathways such as recreational
320   exposure to the wastewater effluent and agricultural application of the soil amendment. Hence, after consideration
321   of the relative contributions for different exposure pathways (e.g. recreational exposure, consumption of
322   agricultural produce), it may well be that just one dominates risk (e.g. Schoen et al. 2014), and only one needs to
323   therefore be integrated with the LCA analysis. Following the LCA procedure for system expansion, the pathogen
324   related health burden of the dominant exposure pathway(s) of the system expansion should be added to the
325   conventional system or subtracted from the overall pathogen related health burden in the water-recycling scenario.

326   Beyond static QMRA and attributional LCA?

327   In this paper, the focus was on static QMRA, and the case studies that explored the concurrent use of QMRA and
328   LCA all relied on static QMRA and attributional LCA. Nonetheless, the concurrent use of dynamic QMRA and
329   LCA in principle should be possible as well. In a similar vein, it should in principle be possible to use QMRA
330   results alongside consequential LCA. This means that there are ample opportunities for further case studies
331   exploring these possibilities in more detail. In doing so, collaboration between LCA and QMRA experts would be
332   helpful in order to ensure that in-depth knowledge and expertise from both fields is taken into account sufficiently.

333   The importance of conscious design and clear communication

334   The three case studies that explored the concurrent use of QMRA and LCA exhibited different ways to frame the
335   concurrent use and relate QMRA and LCA results to one another. The intention here is neither to criticise any
336   particular study, nor to judge which of the studies considered in this paper provided the most concise description
337   of how QMRA and LCA results relate to one another. After all, in an emerging field that still is in the phase of
338   exploring new opportunities, it is no surprise that a clear terminology and best practice have yet to fully emerge.
339   The main intention here is to highlight the importance of consciously and carefully contemplating the use of
340   terminology, and the way in which such studies are designed and results from QMRA and LCA presented. The

341 conceptual model and discussion of issues to be aware of when concurrently using QMRA and LCA, as presented

342 in this paper, will hopefully facilitate more consistent design and more transparent communication of future case

343 studies assessing management options in urban water and sanitation infrastructures in terms of the broader adverse

344 effects of pathogens and other stressors on human health and the environment. Hopefully, this paper also inspired

345 QMRA experts to seize the opportunities to contribute to LCA method development.

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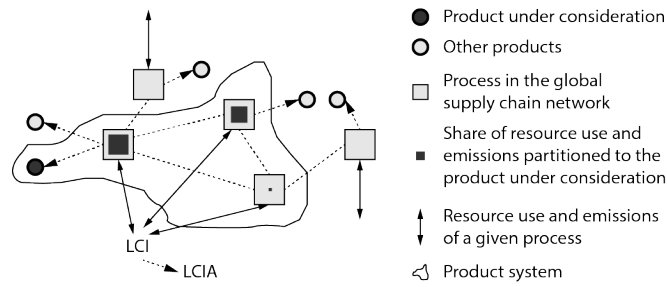
477 **Fig. 1** Visual summary of LCA. Broadly speaking, the task of a LCA study is to identify the resource use and
478 emissions of the (current or future) global supply system that are related to the product system under consideration
479 (LCI). Resource use and emissions are then translated into potential environmental impacts (LCIA).

480 QMRA (suite of model variants)   QMRA results
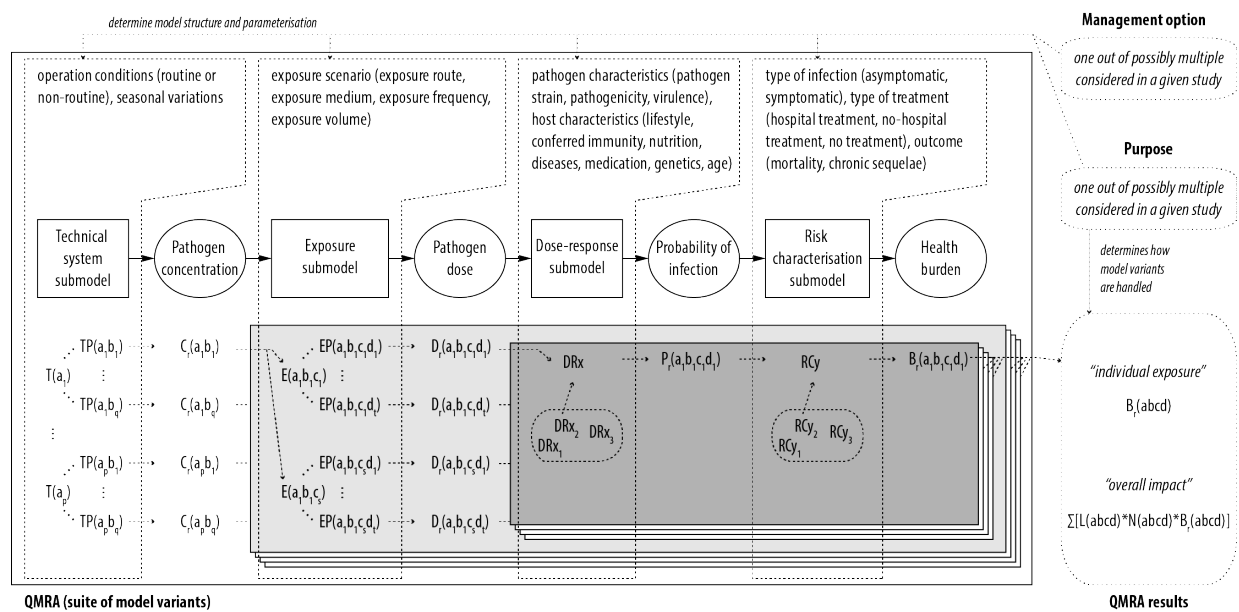
481 **Fig. 2** Conceptual model for the assessment of human health risks associated with pathogens. The conceptual
482 model consists of four sub-models and represents one possible out of multiple water reuse scenarios considered in
483 a given case study. The sub-models and the parameters influencing the sub-models are based on Beaudequin et al.
484 (2015). Given the different possible parameterisations of each sub-model, each QMRA in principle consists of a
485 suite of model variants, where each model variant represents different parameterisations of the four sub-models. T
486 = technical system scenario (a), TP = technical system scenario parameterisation (b), E = exposure scenario (c),
487 EP = exposure scenario parameterisation (d), DR = dose-response relationship, P = probability of infection, RC =
488 risk characterisation relationship, B = health burden, L = likelihood of model variant, N = number of people
489 exposed in model variant. Note that every pathogen considered (r) needs separate treatment and parameterisation.

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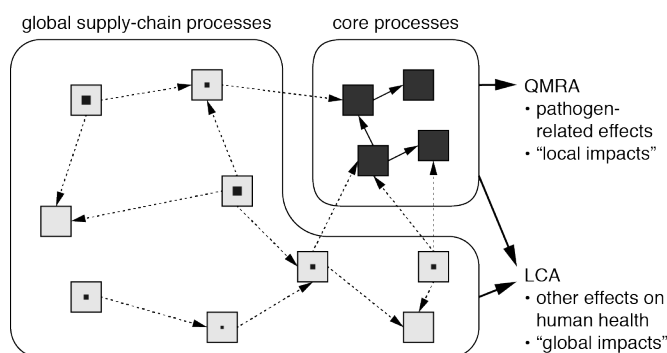
491   **Fig. 3** Overall model structure of the three case studies analysed in this paper. QMRA models are used to estimate

492   the pathogen-related health burden for a number of core processes. LCA is used to estimate the health burden

493   related to other stressors for both the core processes and the supply-chain processes based on LCIA models.

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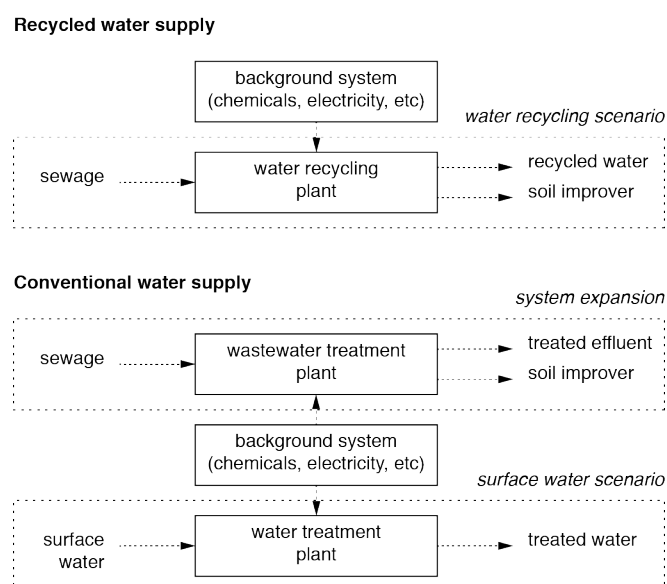
495   **Fig. 4** System expansion exemplified for two water supply scenarios: conventional water supply and water

496   recycling. Details are provided in the text.

497

498 **Table 1** Use of different assessment endpoints (top row) in relation to different assessment purposes (left column).

QMRA purpose	Probability of infection or illness (P_{inf}, P_{ill})	Number of cases of infection or illness (N_{inf}, N_{ill})	Health burden (B)
Threshold comparison	Useful	Useful	Useful
Comparison among different exposures	Useful for similar health outcomes	Useful for similar health outcomes	Useful for similar and different health outcomes
Comparison among different management options	Useful for similar health outcomes	Useful for similar health outcomes	Useful for similar and different health outcomes

499