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It's getting hot in here: Effects of heat on temperature, disinfection, and opportunistic pathogens in drinking water distribution systems

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ABSTRACT

As global temperatures rise with climate change, the negative effects of heat on drinking water distribution systems (DWDS) are of increasing concern. High DWDS temperatures are associated with degradation of water quality through physical, chemical and microbial mechanisms. Perhaps the most pressing concern is proliferation of thermotolerant opportunistic pathogens (OPs) like *Legionella pneumophila* and *Naegleria Fowleri*. Many OPs can be controlled in DWDS by residual disinfectants such as chlorine or chloramine, but maintaining protective residuals can be challenging at high temperatures. This critical review evaluates the literature on DWDS temperature, residual disinfectant decay, and OP survival and growth with respect to high temperatures. The findings are synthesized to determine the state of knowledge and future research priorities regarding OP proliferation and control at high DWDS temperatures. Temperatures above 40 °C were reported from multiple DWDS, with a maximum of 52 °C. Substantial diurnal temperature swings from ~30–50 °C occurred in one DWDS. Many OPs can survive or even replicate at these temperatures. However, most studies focused on just a few OP species, and substantial knowledge gaps remain regarding persistence, infectivity, and shifts in microbial community structure at high temperatures relative to lower water temperatures. Chlorine decay rates substantially increase with temperature in some waters but not in others, for reasons that are not well understood. Decay rates within real DWDS are difficult to accurately characterize, presenting practical limitations for application of temperature-dependent decay models at full scale. Chloramine decay is slower than chlorine except in the presence of nitrifiers, which are especially known to grow in DWDS in warmer seasons and climates, though the high temperature range for nitrification is unknown. Lack of knowledge about DWDS nitrifier communities may hinder development of solutions. Fundamental knowledge gaps remain which prevent understanding even the occurrence of high temperatures in DWDS, much less the overall affect on exposure risk. Potential solutions to minimize DWDS temperatures or mitigate the impacts of heat were identified, many which could be aided by proven models for predicting DWDS temperature. Industry leadership and collaboration is needed to generate practical knowledge for protecting DWDS water quality as temperatures rise.

1. Introduction

Extreme heat events are increasing in frequency around the world, against a backdrop of rising seasonal and annual average temperatures (Davariashtiyani et al., 2023; Fischer et al., 2021). One might assume that drinking water distribution systems (DWDS) are relatively insulated from excessive heat at the surface, as most DWDS infrastructure is buried. Yet significant heat transfer can occur to the typical DWDS buried depth, such that water temperatures can equilibrate with the surface within a matter of hours or days. Heat transfer to water also occurs during storage in above-ground clear wells, reservoirs, and towers, increasing the initial temperature of water entering or stored within the DWDS (Blokker and Pieterse-Quirijns, 2013). High DWDS temperatures degrade water quality by increasing the rate of chemical and biological processes, including disinfectant decay and microbial growth (Calero Preciado et al., 2021; Lai and Dzombak, 2021). Incidents of waterborne disease outbreaks associated with warm temperatures may foreshadow a more widespread problem in a rapidly warming climate (Rhoads et al., 2017).

The World Health Organization (WHO) recommends limiting drinking water temperature to below 25 °C to limit microbial growth (WHO, 2017). However, temperatures above 25 °C are routinely experienced

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in some DWDS (Del Olmo et al., 2021; Waak et al., 2018). Warm temperatures can promote the growth of opportunistic pathogens (OPs) such as *Legionella pneumophila (Lp),* which present high risk for immunocompromised and elderly populations. *Lp* is of particular concern due to high incidence of Legionnaires' Disease (Gerdes et al., 2023). *Lp* is typically considered a premise plumbing problem, in part because it thrives in the warm temperatures found in hot water systems (~35–50 °C) (Tolofari et al., 2022). However, *Lp* has also been detected in DWDS, with increased levels associated with higher temperatures (LeChevallier, 2019). Reports of real DWDS temperatures in hot climates are limited, but heat transfer models suggest that some DWDS could experience similar temperature ranges as premise hot water systems, including up to 50 °C in hot climates like Phoenix, AZ (Bondank et al., 2022).

Despite the potential for high DWDS temperatures that could encourage OP proliferation, limited research has focused on DWDS temperature, and few countries have drinking water temperature regulations (Agudelo-Vera et al., 2020). Maintaining a sufficient disinfectant residual level in a DWDS may be enough to control many OPs, despite high temperatures. However, the most common residual disinfectant is chlorine, which decays rapidly at high temperatures (Fisher et al., 2012). The effect of temperature on chlorine decay rate varies dramatically between waters, and concurrent phenomena at high temperatures, such as increased corrosion, can further accelerate residual loss. In chloraminating systems, higher temperatures promote growth of ammonia-oxidizing bacteria (AOB), which can cause breakpoint chlorination and loss of protective residual (Zheng et al., 2023).

To characterize the risk associated with high DWDS temperatures, the occurrence of high DWDS temperatures and the effect on disinfection and microbial phenomena must be understood. This critical review evaluates the state of knowledge in these areas, specifically: 1) The prevalence and effects of high DWDS temperatures above 25 °C, and the effect of high temperatures on 2) disinfectant decay kinetics and 3) OP persistence and growth. Findings are synthesized to identify key knowledge gaps and potential solutions to mitigate the impacts of high DWDS temperatures on drinking water microbial risk.

2. Methods for literature search

For each topic, three databases were searched with a single search string to query records (titles, abstracts, and keywords) in September 2023: Clarivate Web of Science, Scopus, and PubMed. Search strings are provided in Text S1. Duplicates, non-peer reviewed, non-English records and review articles were removed in Zotero. Reviews were conducted in Covidence in two phases: (1) title and abstract review, and (2) full text review. Records were removed during each phase according to the inclusion criteria described in Text S1. During phase one, two individuals reviewed all records. During phase two, one individual reviewed the records, and 10 % were reviewed by a second individual for quality control. Additional relevant literature was identified through citations.

Reviewed literature was also screened for DWDS water temperature data. Source water and finished water temperatures were excluded. Residential tap water measurements were also excluded, due to potential for additional heat sources in premise plumbing (Lautenschlager et al., 2010). Though tap water may be representative of DWDS with sufficient flushing time, the flushing times used in the literature vary widely, and most studies do not validate whether the flushing time was sufficient to completely clear the building plumbing and service line. The literature data is supplemented with previously unpublished DWDS temperature data from a U.S. utility. Temperatures were logged hourly by DWDS sensor stations during summer months (June–September) of 2021 and 2022. Corresponding daily average and daily maximum air temperatures were acquired from the National Oceanic and Atmospheric Administration Online Weather Data portal (https://www.weather.gov/wrh/climate). Köppen-Geiger climate classification descriptors were acquired from https://climateknowledgeportal.worldbank.org/. To facilitate analysis of the literature, these climate descriptors were further grouped into three categories based on their average monthly or annual temperatures, as reported in Table S6.

3. High temperatures and DWDS

3.1. Overview of heat and DWDS temperature

Seasonally high DWDS temperatures are observed in climates with warm or hot summers (Masters et al., 2016; Potgieter et al., 2018). Some DWDS in hot climates may have persistently high temperatures (Cruz et al., 2020; Montoya-Pachongo et al., 2018). Seasonally high or increasing temperatures are associated with higher incidence of DWDS pipe breaks (Wols et al., 2018) and pump failure (Bondank et al., 2018), as well as increased corrosion and mobilization of metals such as lead (Li et al., 2020) and increased turbidity (Calero Preciado et al., 2022). Warm temperatures are also associated with increased microbial activity, including growth of biofilms (Calero Preciado et al., 2022), and in chloraminating DWDS, growth of nitrifying communities (Potgieter et al., 2018). Formation of certain disinfection byproducts, such as trihalomethanes (THMs), can also increase with temperature, with elevated levels in summer (Furst et al., 2021). The effects of high DWDS temperatures on disinfectant decay and OP proliferation (discussed in subsequent sections) are of particular concern for drinking water safety.

Heat transfer from ambient air and solar radiation can occur throughout a water system, including at the source and treatment facility. However, source water temperature may have only limited impact on ultimate DWDS temperatures in some systems (C.M. Agudelo-Vera et al., 2015). The temperature of finished water entering a DWDS is often strongly correlated to ambient air temperature (Lai and Dzombak, 2021), while the temperature in further extremities of the DWDS may be largely determined by the soil temperature (Blokker and Pieterse-Quirijns, 2013). Heat transfer to a buried pipeline occurs by convection (bulk movement of water), and by conduction from the outer surface of the pipe in contact with the ground (Blokker and Pieterse-Quirijns, 2013). The ground temperature at DWDS depth is primarily controlled by the surface temperature, which is a function of the ambient air temperature, wind, and net solar radiation (C. Agudelo-Vera et al., 2015, 2017). The rate of heat transfer from the surface to DWDS is determined by the buried depth and material composition of the ground (Blokker and Pieterse-Quirijns, 2013). DWDS lines should be at least 12″ below the surface or frost line (International Code Council, Inc., 2021). In practice, depth varies from ~12″ in hot climates to ~7–10′ in cold climates. Heat typically transfers fastest through rock, followed by unconsolidated materials (e.g., sandy soil, clay) which are affected by factors such as moisture and density (Blokker and Pieterse-Quirijns, 2013). Heat transfer from the ground to DWDS water is controlled by the pipe or reservoir material, with cast iron possessing substantially higher thermal diffusivity than plastics like PVC (Díaz et al., 2023). For example, thermal conductivity is relatively high for heat-conductive pipe materials (60 W/m K for cast iron [CI]) and low for heat-insulating pipe materials (0.16 W/m K for polyvinyl chloride [PVC]) (Blokker and Pieterse-Quirijns, 2013).

Heat transfer to DWDS water is also constrained by hydraulic residence time (Blokker and Pieterse-Quirijns, 2013). Distal or stagnant portions of a DWDS typically experience the highest temperatures, exacerbating problems related to high water age (Machell and Boxall, 2014; Monteiro et al., 2017). Heating of DWDS water can also occur in aboveground storage infrastructure, depending on the material thermal diffusivity and insulation or shading from direct sunlight; such facilities may be affected more by elevated temperatures than buried pipelines. The urban heat island effect, in which the built landscape absorbs more heat than surrounding natural landscape, can substantially increase subsurface temperatures (Ferguson and Woodbury, 2007). In some urban areas, underground infrastructure such as building heating systems could also increase subsurface temperatures (Agudelo-Vera et al., 2017). Customers have an aesthetic preference against warm water from the "cold tap". Utilities may attribute reports of warm tap water to premise plumbing, without awareness of potential DWDS contributions (Novakova and Rucka, 2019).

The litany of DWDS issues reported due to seasonally elevated temperatures raises the possibility that such problems are persistent year-round in some hot climate DWDS and may become more widespread with increasing global temperatures and extreme heat events. The following sections aim to evaluate the current body of research on high or increasing temperatures on DWDS due to climate change and extreme heat and compile relevant DWDS temperature data. As a point of reference, "high temperatures" will refer to temperatures above the WHO guideline value of 25 °C.

3.2. Literature search results for DWDS at high temperatures

Twenty-one articles were identified that explicitly discuss the impact of high temperatures in DWDS beyond routine seasonal fluctuations (Table S1). Author affiliation nationality and geographic focus were primarily European. Most of the European nations represented feature relatively cool climates (temperate oceanic), with average maximum temperatures below 25 °C. Relatively hot climate nations (hot arid, hot semi-arid, or hot-summer Mediterranean) with average maximum temperatures above 25 °C were underrepresented. However, two articles focused on Phoenix AZ and Las Vegas NV, US cities with the hottest climate classification (hot arid).

Five identified articles are reviews. Agudelo-Vera et al. (2020) reviewed global drinking water temperatures and regulations, and concluded that critical knowledge gaps remain regarding DWDS temperatures. Zheng et al. (2023) reviewed the effect of temperature on nitrification in chloraminating DWDS. The remaining reviews only briefly discuss how rising temperatures may affect DWDS.

Of sixteen research articles, common subjects were heat transfer modeling (n = 5), physical component failure (n = 3), and biological aspects of water quality, including biofilms (n = 2) and invertebrates (n = 1). Two studies investigated disinfectant decay and microbial growth at elevated temperatures through analysis of historical data or modeling future projections. Kimbrough (2019) found median nighttime DWDS temperatures in Pasadena, CA increased by 1.6 °C between 1985 and 2000 and 2009–2016, which correlated with lower chloramine residuals and higher nitrite concentrations, indicating increased AOB activity. Lai and Dzombak (2021) estimated a 2.2 °C increase in average annual DWDS water temperatures between 2001 and 2020 and 2051–2070 across 91 US cities, causing significantly increased bacterial activity and chlorine decay rates. Bondank et al. (2018) modeled the failure rate of physical components and disinfection using projected ambient air temperatures for Phoenix, AZ, and Las Vegas, NV; between 2020 and 2050, predicted failure rates increased most for pumping stations (76 % ± 15 %) and chlorine residual loss (53 % ± 36 %).

Thirteen articles discussed measured or modeled temperature data for real water systems, most in relatively cool climates (Table S1). Only five articles reported real DWDS temperatures, four from Europe and one from Pasadena, CA. Of these, the highest reported DWDS temperature was 33 °C in the Czech Republic, where DWDS routinely exceeded regulatory guidelines of 8–12 °C (Novakova and Rucka, 2019). Other studies used temperatures from finished water, tap water, soil, or ambient air as either a proxy or to model DWDS temperature. The highest temperatures discussed were average maximum daily summertime air temperatures projected for Phoenix, AZ: by 2050, 44 °C (Bondank et al., 2018) and by 2100, 57 °C (Bondank et al., 2022); corresponding DWDS temperature predictions were not reported. Air temperatures above 50 °C have occurred in Phoenix and other hot desert climate regions (Mildrexler et al., 2011), but real DWDS temperatures under such high temperature conditions were not reported.

3.3. Occurrence of high DWDS temperatures

To document the occurrence and range of high DWDS temperatures, literature reviewed for Sections 4 and 5 were also screened for DWDS temperature data. Due to the focus of this critical review on high temperatures, which was reflected in the literature search terms, the collected data is not intended as a random sample. Twenty-three articles were identified, and the highest reported temperatures were recorded with descriptors of sampling campaigns, location and climate (Table S2). As observed by Agudelo-Vera et al. (2020), temperature is not always reported with the same rigor as other measurements. Four articles did not describe the number of sampling locations or sampling frequency. Documented campaigns varied by number of sampling sites (1–157) and frequency of measurements (e.g., hourly, monthly, single event). Only thirteen articles reported the maximum measured temperature.

Most articles reported temperatures above 25 °C (n = 19 of 23, including the present work), and temperatures over 30 °C were reported in all climate categories (Table 1). Climates classified as hot and dry had the highest range of maxima, from >25 to 50 °C, with three entries above 40 °C. By comparison, maxima ranged from 21.6 to 34ºC for tropical and subtropical climates (n = 7), and 9–33 ºC for cooler climates (n = 5). Five studies could not be classified by climate. The season in which the highest temperature occurred could be determined for 12 articles; all occurred in summer (or warm season) except one in tropical Nigeria (Ganiyu et al., 2022).

Among the literature reviewed, North American DWDS are strongly over-represented with 9 of 22 papers from the US or Canada, while 2–4 studies were identified from every other populated continent except South America. For drinking water temperature, climate is perhaps the more important factor. In this regard the data is relatively well balanced, with 5–7 papers from each of the three climate categories as defined in Table S6. However, within each category, there is significant climatic diversity which cannot be characterized with this small sample size.

To supplement the literature, DWDS temperature data is provided from a US utility and compared with ambient temperature (Fig. 1). Water temperatures were measured hourly in four underground DWDS sites (three storage reservoirs and one in-line) during summers of 2021 and 2022. At three sites, average daily water temperatures were consistently higher than average daily air temperatures (Fig. 1A), and maximum daily water temperatures increased proportionately with maximum daily air temperatures (Fig. 1B). At two sites, maximum daily water temperatures were consistently higher than maximum daily air temperatures. The maximum water temperature recorded, 52 °C, was higher than the maximum air temperature that day (46 °C). Further-

Table 1
DWDS temperatures reported in the literature (including the present study), categorized broadly by climate according to the Köppen classification as described in Table S6.

Climate category	Classification	n	Maximum temp. (°C)
Hot (dry)	Hot semi-arid, Hot arid, Hot-summer Mediterranean,	7	>25[a]–52
Tropical	Tropical, Humid subtropical	7	21.6–34[b]
Cool	Temperate oceanic, Continental, Warm-summer Mediterranean	5	9–33
Unclassifiable		5	29–34

[a] Kimbrough (2019) did not report a maximum, but reported the 75th percentile as 25 °C.
[b] This range excludes the mean and standard deviation (30.7 ± 4.31 °C) reported by Dong et al. (2022).

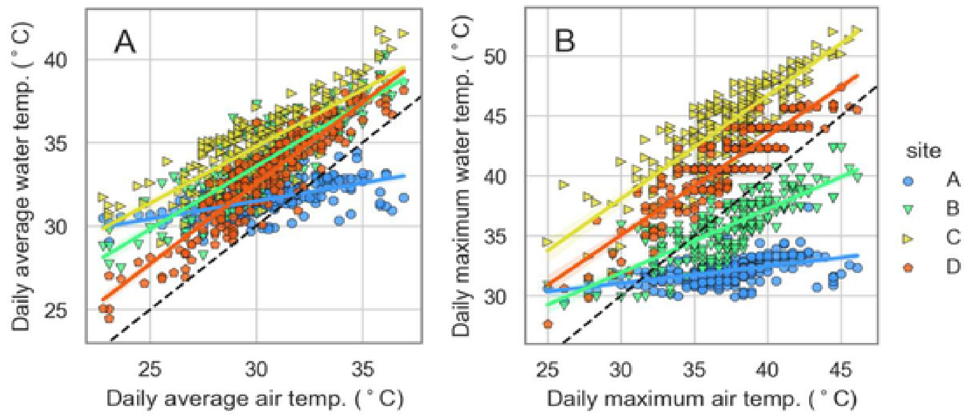

Fig. 1. Daily A) average and B) maximum temperatures for air versus water for four DWDS locations at a U.S. utility during the summer months of 2021 and 2022; the dashed black line indicates the line of equivalency. Sites A, B, and C are underground storage reservoirs; D is an in-line location.

more, this maximum water temperature was higher than the maximum air temperature (46 °C) at any time during the two summers of record. DWDS temperature could exceed air temperature due to the urban heat island effect, or there may have been variation in local temperature compared to the weather station several miles away.

At the two hottest locations (reservoir C and in-line site D), DWDS temperature exhibited substantial diurnal variation, closely tracking air temperature. The maximum change in DWDS temperature within a single day was 24 °C (site D). These observations are consistent with the shallow depth of sites C and D, perhaps in materials with high thermal diffusivity, such that water quickly reaches thermal equilibrium with the surface (Gunkel et al., 2022). Between sites, instantaneous temperature differences of ~20 °C were also observed. Thus, spatiotemporal variation in DWDS temperatures can be substantial, with potentially significant implications for chemical and microbial phenomena.

4. Residual disinfectant decay kinetics at high temperatures

4.1. Overview of disinfectant residual decay in DWDS

Chlorine-based disinfection has a long history of effectively reducing waterborne diseases such as cholera and typhoid (CDC, 1999). Residual disinfectants in DWDS provide protection against the growth and reintroduction of microbial pathogens during distribution. Chlorine is the most commonly used disinfectant. In the US, monochloramine and chlorine dioxide are also used to maintain DWDS residuals (USEPA, 2016). However, residual disinfectants may not be necessary in some nations where waterborne disease is effectively controlled by other measures, such as rigorous watershed protection programs and DWDS maintenance (Smeets et al., 2009).

Disinfectant decay results from reactions with numerous constituents in water; many of these reactions are temperature sensitive, such that overall decay rates increase with temperature (Fisher et al., 2012; Sathasivan et al., 2009). Decay rates are generally proportional to oxidant strength, with chlorine dioxide as the strongest, followed by chlorine and monochloramine (Copeland and Lytle, 2014). However, each disinfectant consists of multiple chemical species in a dynamic equilibrium determined by numerous factors, including temperature, pH and carbonate speciation (Hua and Reckhow, 2008; Rose et al., 2020; Vikesland et al., 2001). Reactants include inorganic species, some which react quickly (e.g., metals, iodide), and countless organic species, many which react slowly (Gallard and Von Gunten, 2002). Depending on disinfection contact time during water treatment, fast reactions are typically complete before distribution (Kiéné et al., 1998).

In DWDS, disinfectant decay is divided into "bulk decay" through reactions with dissolved constituents, and "wall decay" through reactions with constituents (e.g., scale, biofilms) on the interior pipe surface (Fisher et al., 2017; Rossman et al., 1994). Depending on pipe material, corrosion levels, and biofilm activity, wall decay can be more significant than bulk decay in DWDS (Kiéné et al., 1998; Zhang et al., 2008). Though monochloramine decays more slowly than chlorine, the presence of excess ammonia can stimulate growth of nitrifying organisms which oxidize ammonia to nitrite (Wilczak et al., 1996). If ammonia levels fall too low, breakpoint chlorination will occur, resulting in total loss of residual (Jafvert and Valentine, 1992). High temperatures promote increased nitrifier growth, as well as increased corrosion and biomass, such that wall decay rates of chlorine and chloramine are likely to increase significantly at high temperatures.

Due to the complexity of disinfectant decay processes, most modeling efforts are empirical and rely on experimental observations in real waters (Fisher et al., 2011). In most cases, the effect of temperature on reaction rate can be modeled with the Arrhenius equation (Eq. 1), with rate constant (k) a function of temperature (T), activation energy E_a, universal gas constant R, and pre-exponential factor A.

$$k = Aexp\left(\frac{-E_a}{RT}\right) \quad (1)$$

In theory, the activation energy coefficient (E_a/R) derived from experimental data can be used to predict the increase in decay rate (from k_1 to k_2) for an increase in temperature T_1 to T_2 (Eq. 2).

$$k_2 = k_1 exp\left[-\frac{E_a}{R}\left(\frac{1}{T_2} - \frac{1}{T_1}\right)\right] \quad (2)$$

Researchers and practitioners often use previously reported E_a/Rs to predict the effect of temperature on decay rate in a new water. As E_a/R varies significantly between waters, this practice can result in inaccurate predictions (Fisher et al., 2012).

As decay rates increase with temperature, disinfectant dosage can be increased to maintain a target DWDS residual. Some utilities have automated control systems which monitor and adjust conditions to maintain residuals throughout DWDS. However, small or under-resourced water systems in particular can have significant difficulty addressing residual loss in the DWDS. Furthermore, disinfectant dosage must be moderated to avoid exceeding regulatory limits for disinfectants as well as for THMs, which are regulated in many countries (Furst et al., 2019). Many US utilities adopted chloramine disinfection to minimize THM formation, but challenges with nitrification and chloramine loss are common, particularly in summer (Seidel et al., 2005). Thus, all of these tradeoffs are more difficult to manage at high temperatures. The following sections review literature reporting activation energy coefficients (E_a/Rs) to quantify the effect of temperature on chlorine, chlorine dioxide, and chloramine decay rates.

4.2. Chlorine decay at high temperatures

Chlorine decay results from a plethora of reaction mechanisms; a comprehensive review of chlorine reactions with organic and inorganic matter is provided by Deborde and von Gunten (2008). The effect of temperature on bulk chlorine decay arises from the individual effect of temperature on each reaction of chlorine, such that mechanistic modeling of chlorine decay has been infeasible. As such, effects to model the effect of temperature on chlorine decay have been entirely empirical (Fisher et al., 2011).

Fifteen articles reported primary E_a/Rs for chlorine decay under conditions relevant to DWDS. Experimental and modeling conditions were cataloged, along with source type, quality, and treatment level if reported (Table S3). The range of chlorine decay E_a/Rs is 290–14,300 K, where 290 K indicates little effect and 14,300 K indicates substantial effect of temperature on decay rate. Equation 2 was used to calculate the increase in decay rates (k_2/k_1) with increasing temperatures for nine surface waters without pre-chlorination that were tested at bench-scale. E_a/Rs for these waters range from 5,351 to 14,300 K (mean 8,270 K). For a moderate temperature increase from 25 to 35 °C, the mean and maximum E_a/Rs predict decay rate increases of 2.5x and 4.7x, respectively (Fig. 2A). For a substantial temperature increase of 25 to 50 °C (such as that described in Section 3.3.), the mean and maximum E_a/Rs predict decay rate increases of 8.6x and 41x, respectively (Fig. 2B). Thus, there is an immense range in the effect of temperature on decay rate, such that it would be inappropriate to select a random E_a/R from the literature to predict chlorine decay in a new water. Possible explanations for this variation include model and experimental design choices as well as differences in water quality.

4.2.1. Effect of modeling and experimental design on chlorine decay E_a/Rs

E_a/Rs were derived with several types of kinetic models. The simplest is the “single reactant” model (1RA), which is a first, second, or nth order decay equation. In 2RA models, two decay terms represent the fast and slow reaction phases, usually as parallel second order reactions. E_a/R can be calculated for each decay constant, or one for the whole model; either approach may be sufficiently accurate (Fisher et al., 2012; Monteiro et al., 2015). 1RA models are typically less accurate than 2RA models (Fisher et al., 2011). One 1RA model underestimated E_a/R by 50 % compared to a 2RA model (Jabari Kohpaei et al., 2011). Another class of models incorporate variable rate coefficients to account for the change in reactants over time; modest (< 10 %) differences in E_a/Rs between these and 2RA models were reported (Hua et al., 2015; Zhong et al., 2021). In seven of eight 2RA models reporting two E_a/Rs, the slow reactant phase had a larger Ea/R than the fast reactant phase, in some cases by more than 800 % (Jabari Kohpaei et al., 2011; Monteiro et al., 2015). This indicates that the slow reactant phase, which predominately occurs during residence in the DWDS, is more sensitive to temperature than the fast reactant phase, which is typically completed prior to entering the DWDS. Thus, choice of kinetic model affects the accuracy of E_a/R estimates as well as what can be learned about the underlying phenomena. However, controlled experiments would be needed to identify the underlying causes of variation in model parameters for different reactant phases and waters reported in the literature.

Experimental conditions may also affect the accuracy of E_a/R. Fisher et al. (2012) found that decay rate was slightly underestimated at the high end of an experimental temperature range (28 °C), suggesting that calculating reaction rates at temperatures beyond the experimental range may incur greater error. Most studies used maximum temperatures between 20 and 30 °C, though Hua et al. (2015) and Jabari Kohpaei et al. (2011) tested up to 40 °C and 50 °C, respectively. No studies directly investigated the effect of experimental temperature range on accuracy. Another experimental parameter is initial chlorine concentration (ICC). Ideally, decay models would be independent of ICC, but this is often not the case for 1RA models (Fisher et al., 2011). Li et al. (2016) found that low ICCs produced slightly higher (< 5 %) E_a/Rs than high ICCs with 1RA models, which may be unimportant compared to other sources of bias.

4.2.2. Effect of water quality on E/R

The overall effect of temperature on chlorine decay rate arises from the effect of temperature on each individual reaction contributing to chlorine decay. Thus, E_a/R must be related to the type and concentration of reactants, as determined by the source water quality, treatment processes, and contributions of DWDS pipe walls. Many studies reported no water quality information; others reported bulk water quality parameters, such as conductivity and total or dissolved organic carbon (TOC, DOC). Chlorine decay rates typically increase with increasing TOC levels (Cong et al., 2012; Kiéné et al., 1998). As the slow-reacting fraction is more temperature sensitive, and many organic reactants react slowly, one could hypothesize that TOC is correlated with E_a/R. Monteiro et al. (2015) tested three conventionally-treated, pre-

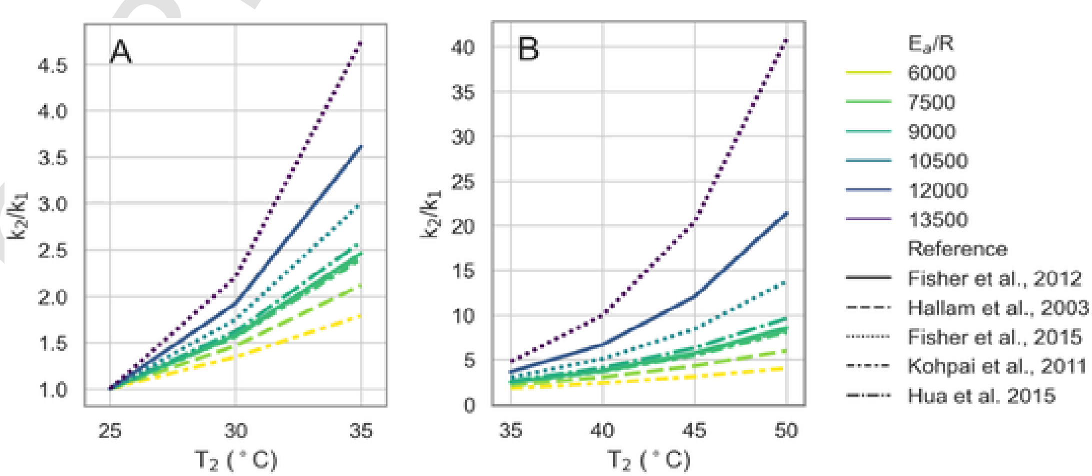

Fig. 2. Increase in theoretical chlorine decay rate constant k_2 over k_1 with increase in temperature from T_1 to T_2, where T_1 is 25 °C and T_2 is A) 35 °C or B) 50 °C, calculated using activation coefficients (E_a/R) from the literature, for nine surface waters that were not pre-chlorinated and were tested at bench-scale.

chlorinated surface waters, for which slow reaction E_a/Rs did increase with DOC levels (0.4–3.3 mg/L). However, of two reservoir waters with 2.5 and 1.5 mg/L DOC, the higher DOC water had the smaller E_a/R. Other articles did not report enough information to facilitate such comparisons.

Groundwaters typically have lower TOC than surface waters (Weisman et al., 2023), such that differences in E_a/R between groundwaters and surface waters might be expected. No peer-reviewed studies reporting groundwater E_a/Rs were identified, but a non-peer-reviewed study reported a slightly lower, but overlapping range of E_a/Rs for groundwater (5,000–8,000 K) compared to surface water (5,600–9,800 K) (Fisher et al., 2017). In surface water, organic matter and other water quality parameters can vary seasonally, such that E_a/R may vary seasonally as well. Three waters from the same region each exhibited seasonal variation in E_a/R of ~30–50 %, but with no discernable pattern (Lee et al., 2023). Controlled studies are needed to clarify the relationship of E_a/R and water quality.

Water treatment processes remove or transform chlorine-reactive constituents, such that trends in E_a/R might be expected according to treatment type. Studies evaluated waters treated with various processes, but none compared the effect of treatment on E_a/R for the same water. E_a/Rs reported for untreated surface waters were ~8,000 K, within the range for conventionally-treated surface waters (5,918–11,800 K). A granular activated carbon-treated surface water had an E_a/R of 6,889 K (Hallam et al., 2003). E_a/Rs for seawater treated by ultrafiltration/nanofiltration were 5,526–5,679 K, slightly lower than a conventionally-treated surface water (5,918–6,110 K) in the same study (Li et al., 2016). A reverse osmosis (RO)-treated seawater had a very low E_a/R (500 K), attributed to negligible DOC (0.06 mg/L) (Fisher et al., 2015). E_a/Rs for pre-chlorinated waters ranged from 291 to 10,396 K across four studies. In a comparison of conventionally-treated surface waters pre-oxidized with chlorine (n = 3) or ozone (n = 2), the ozonated waters had lower fast-reactant E_a/Rs, suggesting that ozone oxidized more fast reacting, temperature-sensitive constituents than chlorine (Monteiro et al., 2015). Controlled studies evaluating the effect of treatment on the same water would be needed to clarify how treatment affects E_a/R.

Wall decay rates may increase with temperature directly due to increased rate of reactions, and indirectly due to increased biomass, corrosion, and dissolution of material on pipe surfaces. Cong et al. (2012) evaluated the effect of temperature on wall decay by comparing decay rates in batch and pipe-loop experiments conducted with the same water. Wall decay rates increased with temperature; however, experimental conditions were unclear, and likely not representative of real DWDS. Additionally, two articles described temperature experiments conducted in pilot-scale (pipe-loop) systems, though neither attempted to quantify wall decay (Ozdemir et al., 2018; Kim et al., 2019). The experimental approach used in all three studies was contested in Fisher et al. (2017) for evaluating wall decay. Thus, the effect of temperature on chlorine wall decay rates remains a significant knowledge gap which requires rigorous experimental design to resolve.

4.3. Chlorine dioxide

Chlorine dioxide decay in DWDS is the result of auto-decomposition and reactions with organic and inorganic compounds (Aieta and Berg, 1986; Gan et al., 2020). Four studies were identified that quantitatively evaluated the effect of temperature on chlorine dioxide decay. Two found that decay rates increased systematically with temperature, in distilled water at 15–45 °C (Li et al. 2013) and in re-mineralized desalinated water at 20–45 °C (Ammar et al. 2013). E_a/Rs were calculated as 4,045 K and 2,709 K, respectively, representing modest increases in decay rate with temperature. Two studies did not find a clear effect of temperature. Grunert et al. (2018) tested chlorine dioxide decay in a pilot-scale pipe-loop with treated groundwater at three temperatures (15–25 °C), and found that decay rate first increased and then decreased with temperature. Zhang et. al. (2008) investigated chlorine dioxide reactions with corrosion scale in DI water and found that temperature (25–45 °C) did not significantly affect reaction rates because they were very fast. With few studies and a lack of real drinking water matrices, conclusions cannot be drawn regarding chlorine dioxide decay rates at high temperatures. However, the rapid decay of chlorine dioxide at moderate temperatures suggests it is not a suitable alternative to chlorine in high temperature DWDS.

4.4. Chloramine

Bulk decay of monochloramine in DWDS occurs through chemical reactions and biological activity. A comprehensive review of chloramine decay models is provided by Hossain et al. (2022). Six articles reported E_a/R values for mechanistic or empirical models of chloramine decay (Table S4). Mechanistic models of monochloramine decay focus on chemical pathways. In the absence of microbial factors, monochloramine bulk decay occurs through auto-decomposition by hydrolysis, substitution, and disproportionation. E_a/Rs have been reported for five auto-decomposition pathways. Of these, the most temperature-sensitive reactions are hydrolysis of monochloramine to free chlorine and ammonia (8,800 K) and bicarbonate acid-catalyzed disproportionation of monochloramine to dichloramine (22144 K) (Hossain et al., 2022). At neutral pH, the rate-limiting auto-decomposition reactions primarily form dichloramine; Vikesland et al. (2001) proposed that only these reactions need be considered to accurately model the temperature dependency of monochloramine auto-decomposition, but this was not experimentally verified.

Monochloramine decay also occurs through reactions with organic matter. A semi-mechanistic study examined the temperature dependency of reactions with organic matter in tap water (Zhang et al., 2017). The model assumes that free chlorine produced by monochloramine decomposition reacts with organic matter. The reported E_a/R (10,733 K) is moderately high compared to chlorine E_a/Rs. The reaction rate of chlorine with organic matter derived by Zhang et al. (2017) is almost as slow as the slowest auto-decomposition reaction producing dichloramine (Table S4). Dichloramine reacts with organic matter as well (Furst et al., 2018), but no articles investigated the temperature sensitivity of these reactions. It is not clear whether the temperature sensitivity of organic matter reactions should be considered for accurate monochloramine decay modeling.

In DWDS, chloramine decay is largely mediated by nitrifying microbial species. Sathasivan et al. (2009) empirically evaluated bulk chemical and microbial decay rates in samples from multiple sites in a DWDS. Overall chloramine decay rates were much faster in samples exhibiting evidence of nitrification compared to those without. In addition to increased microbial decay, substantially increased chemical decay was observed in nitrifying samples, presumably due to breakpoint chlorination as well as reactions with byproducts of nitrifying biofilms, including nitrite (Mitch and Sedlak, 2002) and proteins (Herath and Sathasivan, 2020). Microbial decay was more sensitive to temperature than chemical decay, with average E_a/Rs of 6,924 K vs. 3,551 K. Substantial variation in the microbial decay rate and E_a/R was observed between DWDS locations, consistent with highly variable microbial activity. Hossain et al. (2022) concluded that substantial knowledge gaps in the mechanisms of chloramine decay in DWDS hinder development of accurate predictive models for real systems. Indeed, these knowledge gaps are even greater at high temperatures.

The optimum range for nitrification is typically described as ~25–35 °C (Odell et al., 1996; Pintar et al., 2005; Wolfe et al., 1990). However, DWDS temperatures did not reach 35 °C in these studies, such that the high end of the nitrification temperature range is unclear. Nitrifiers are a diverse group of organisms, each with a different optimum growth temperature. Sarker et al. (2013) reported optimum growth temperatures for six AOB species ranging from 30 to 55 °C, most continuing to grow above 40 °C; however, these AOB, including *Nitrosomonas europaea*, may not be relevant for DWDS. Other *Nitrosomonas* species were reported in nitrifying DWDS (Hoefel et al., 2005; Lipponen et al., 2004), but their optimum growth temperatures were not determined. Further research is needed to determine the effect of high DWDS temperatures on nitrifier community composition and growth rates.

5. OP persistence and growth at high temperatures

5.1. Overview of drinking water OPs

Despite widespread adoption of water treatment and disinfection, OPs cause a substantial number of illnesses and deaths (NASEM, 2020). Drinking water OPs include bacteria from genera *Legionella* and *Mycobacterium*, and amoeba from genera *Acanthamoeba* and *Naegleria*. Some genera include multiple OPs which can co-occur in drinking water. *Lp* alone consists of at least 14 serogroups, with serogroup 1 typically associated with Legionnaires' Disease (van der Kooij et al., 2016). Premise plumbing can be an OP hot spot due to high pipe surface-to-volume ratio, high water age, low disinfectant residual, and high temperatures (30–60 °C) in hot water systems (Falkinham et al., 2015). Most research on *Legionella* has focused on premise plumbing. However, *Legionella* has been detected in many DWDS, even in the presence of disinfectant residuals (LeChevallier, 2019; Waak et al., 2018).

Increased DWDS temperatures are generally expected to favor the proliferation of bacterial and amoebal OP populations (Cope et al., 2019; Prest et al., 2016). Every OP has an optimal growth temperature range; temperatures below 50 °C are widely reported to favor *Legionella* persistence (Singh et al., 2022), but less may be known about other OPs. Along with temperature, other factors that can affect OP growth and persistence in a water system include pH, nutrient availability, and disinfectant residual. These factors can impact OPs directly, as well as indirectly by altering microbial community structure (Proctor et al., 2017). The microbial community structure affects OPs through complex ecological interactions (Hull et al., 2019; Zhang and Liu, 2019). Many OPs are thought to primarily reside within biofilms, which may provide nutrients and shelter (Falkinham et al., 2015; Lau and Ashbolt, 2009). Higher temperatures could promote biofilm growth, creating more shelter for OPs (Calero Preciado et al., 2021). Studies that only measure OPs in bulk water may not capture the true effect of temperature on OP persistence.

Amoebal drinking water OPs can cause illness directly as well as serve as reservoirs for bacterial OPs (van der Kooij et al., 2016). *N. fowleri* is of particular concern because infections are usually fatal, and its geographic range and routes of transmission are expanding with global warming (Cope and Ali, 2016). Although *N. fowleri* can be controlled in DWDS with chlorination, the combined threat of high temperatures and compromised disinfection may pose a challenge during heat waves (Bartrand et al., 2014). Amoebae are generally more resistant to chemical disinfection than bacteria (Loret et al., 2008), and when exposed to stressful conditions can transition from trophozoite (infectious) forms to encysted forms which are more resistant to chlorination and heat (Thomas et al., 2010). Intracellular bacteria inside amoeba can be shielded from chemical and thermal disinfection, where they can reproduce and ultimately recolonize a system (He et al., 2021; Ohno et al., 2008).

DWDS conditions can alter the biological state of OPs in a variety of other ways with implications for analytical methods as well as risk. For example, stress (e.g., via chlorination or heat) can induce a viable but not culturable (VBNC) state in legionellae, such that viable cells cannot be detected with culturing methods, but their nucleic acids can be detected with molecular methods (Kirschner, 2016). Furthermore, OP virulence can be affected by temperature. *Lp* displays maximum virulence at 37–42 °C (Mauchline et al., 1994), underscoring the potential public health implications of high DWDS temperatures. As such, the following literature review aims to consolidate evidence of OP survival, growth, and other behaviors at high temperatures, and identify pertinent knowledge gaps.

5.2. Effect of high water temperatures on OP abundance in DW systems

Sixty studies were identified that evaluate the effect of water temperatures above 25 °C on OPs in contexts relevant for DWDS (Table S5). Articles were categorized by field-scale ($n = 35$) or bench-scale ($n = 25$). Most full-scale studies were conducted in real or simulated premise plumbing systems ($n = 24$); studies in real or simulated DWDS were less common ($n = 13$). Reported temperatures ranged from 2.8 °C in a DWDS to 52 °C in cold water premise plumbing. The highest reported DWDS temperature was 43 °C. Fig. 3 shows the frequency with which each OP type was studied at field- or bench-scale. *Lp* was the most targeted species among field-scale ($n = 15$) and bench-scale ($n = 12$) studies. Numerous field-scale studies also measured across the genus *Legionella* (*Legionella* spp.) ($n = 15$), and species of genera *Mycobacterium* ($n = 7$). Eleven field-scale studies targeted amoeba, including *N. fowleri, Naeglaria* spp., *V. vermiformis*, or *Acanthamoeba* spp. Eight bench-scale studies co-cultured amoeba of various genera with *Lp* or non-*pneumophila* legionellae ($n = 1$). A variety of other bacteria and amoeba were targeted in ~1–3 studies each.

5.2.1. Bench-scale studies examining effect of DW temperature on OPs

Studies of *Lp* in sterile water matrices documented persistence between 30 and 37 °C over days or weeks. *Lp* multiplied in sterile tap water from 25 to 37 °C, with the fastest doubling time between 32 and 35 °C (Wadowsky et al., 1985). However, other studies found that temperatures of 32–37 °C decreased *Lp* longevity by days or weeks compared to lower temperatures (4–20 °C) (Kuchta et al. 1983; Dutka et al., 1984). All bench-scale *Legionella* studies in sterile waters used culture-based methods, which do not account for VBNC cells and thus likely underestimate the number of viable cells at high temperatures.

Studies of *Lp* in non-sterile waters reported persistence at even higher temperatures, despite also using mainly culture-based methods. In chlorine-free tap water at 35–65 °C, culturable concentrations of *Lp* serogroup 1 were only reduced by temperatures above 55 °C (Katz and Hammel, 1987). Environmental *Legionella* strains continued to multiply up to 42 °C in non-sterile tap water (Yee and Wadowsky, 1982). The role of biofilms in *Lp* persistence at high temperatures was also evaluated in several bench-scale studies. Rogers et al. (1994) found *Lp* survived temperatures as high as 50 °C in biofilms. Armon et al. (1997) documented prolonged survival of *Lp* serogroup 3 up to 36 °C in biofilms compared to sterile conditions, demonstrating the importance of testing OP thermotolerance under realistic ecological conditions.

Non-*Legionella* bacterial OPs were also found to persist at high temperatures. *Mycobacterium* spp. was reportedly more tolerant than *Legionella* to temperatures ≥50 °C (Schulze-Röbbecke and Buchholtz, 1992). In mesocosms with chlorine-free tap water, *S. aureus* was reduced by 2.2-logs with increasing temperature from 35 to 55 °C, but increasing to 65 °C did not further reduce concentrations (Katz and Hammel, 1987). Over 80 % of *Methylobacterium* spp. survived in 50 °C sterile tap water after 3 min (Szwetkowski and Falkinham, 2020), though its persistence over longer time scales is unclear. Concentrations of other bacterial OPs in biofilms, including *P. aeruginosa, S. maltophilia,* and *M. kansasii,* increased with water temperature to 30 °C even as ATP concentrations decreased, indicating a net decline in biological activity (van der Wielen et al., 2023). Overall, these studies suggest that many bacterial OPs can survive high DWDS temperatures of at least 50 °C.

Several studies reported the thermotolerance of individual amoeba species at bench-scale. For *V. vermiformis* in sterile tap water heated to 50 °C for 30 min, only ~0.04 % of trophozoites and 16 % of cysts survived (Kuchta et al., 1993). *N. fowleri* trophozoites remained viable at

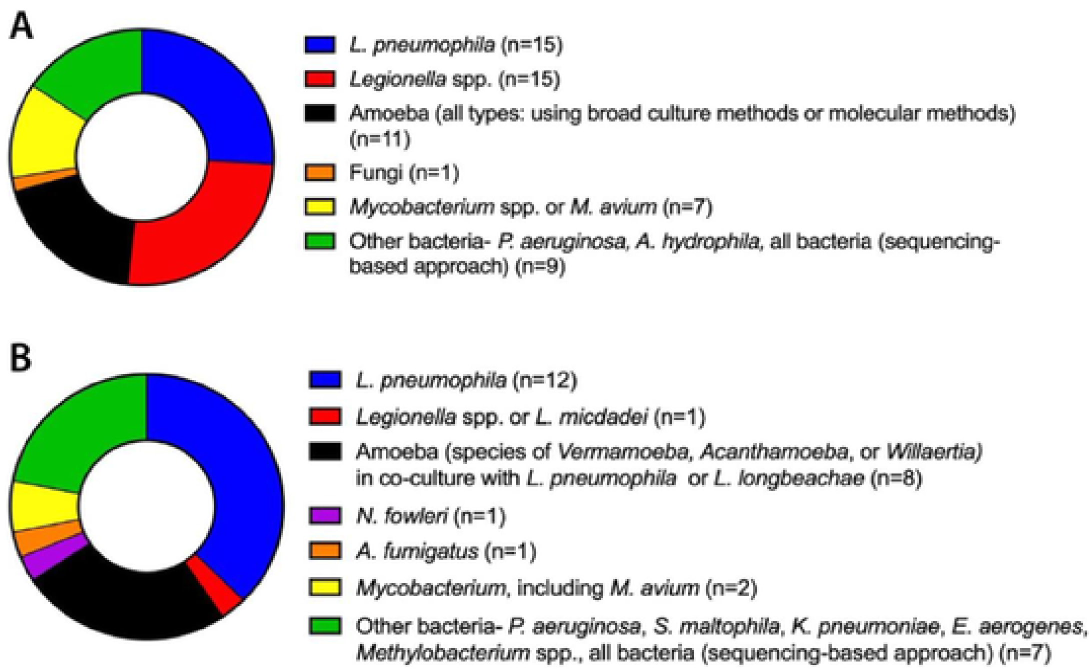

Fig. 3. Microorganisms targeted with molecular biology or culture-based techniques by number of (A) field-scale or (B) bench-scale studies. "All bacteria" refers to studies using 16S rRNA sequencing to measure all bacterial and/or archaeal organisms in a sample; "fungi" refers to studies using broad culture methods to measure all fungi. Studies that measured OPs in multiple categories are counted in both categories.

48 °C for up to 48 h in ultrapure water, but by 72 h were either nonviable or encysted; the upper limit for viability or encystation was 49 °C after 24 h (Lam et al., 2019). *A. polyphaga and W. magna* survived in a biofilm reactor at 40 °C for 4 weeks as cysts and returned to trophozoite form as temperatures decreased (Shaheen et al., 2019). Thus, there may be a significant range in the thermotolerance of different amoeba.

Multiple studies investigated survival or replication of *Legionella* in co-culture with amoeba. At 35 °C in sterile tap water, *Lp* reproduced intracellularly in trophozoite *A. castellani* (Ohno et al., 2008). A co-culture study with *V. vermiformis* in a biofilm reactor found ~20 °C promoted diverse *Legionella* speciation, but at 37 °C, *Lp* dominated; *V. vermiformis* did not seem to affect *Lp* persistence, and decreased to non-detectable after just one week at 37 °C (Buse et al., 2017). The amoebae may have been parasitized by *Lp,* as reported for *A. castellanii* at 35 °C (Ohno et al., 2008). *Lp* that transitioned to VBNC state at 42 °C were resuscitated by *A. castellani*, regaining cultivability (Ohno et al., 2003). A co-culture study of *Lp* with *W. magna* reported increased expulsion of vesicles in water at 30 °C relative to 22 °C and 40 °C (Shaheen and Ashbolt, 2018). *Lp*-containing vesicles can become aerosolized at the tap, increasing exposure risk. These studies suggest the combined effects of temperature and interactions with amoeba may largely determine the dominant bacterial OP species in DWDS.

5.2.2. Field studies of OPs at high temperatures

Lp was reported in DWDS in the US and Spain at water temperatures 18–32 °C, including in the presence of chlorine residuals ranging from 0.1 to >1.0 mg/L (LeChevallier 2019; Sanchez-Buso et al., 2015). Atkinson et al. (2022) found that water temperature was weakly correlated with *Lp* concentration in treated groundwater and DWDS; as temperatures were consistently above 18 °C, the authors concluded that this may be a threshold above which temperature is a less dominant factor. Garner et al. (2018) found *Legionella* spp. correlated with water temperature across four DWDS, with higher correlations observed for biofilm vs. water samples. The positive correlation of *Legionella* with temperature in biofilms confirms previous research which found *Legionella* was more abundant in biofilms at 40 °C compared to 20 °C (Rogers et al. 1994). Within one system, *Legionella* spp. was not directly correlated with temperature, but spiked in concentration following several short-term water temperature increases. Rhoads et al. (2017) reported that incidents of Legionellosis correlated with average and maximum DWDS temperature over several years in a US county, perhaps the best evidence that higher DWDS temperatures may increase *Legionella* exposure risk.

Among non-*Legionella* bacterial OPs, *Mycobacterium* spp. exhibited a stronger correlation with temperature than *Legionella* spp. in premise plumbing (Hu et al., 2021; Lu et al., 2017; Tang et al., 2020; Hozalski et al., 2020) and DWDS (Garner et al., 2018). Lu et al. (2017) found *Mycobacterium* spp. was positively correlated with temperature up to the highest temperature tested (49 °C), and was present at higher levels than *Legionella* in hot water premise plumbing samples, supporting previous research finding *Mycobacterium* is cultivable at higher temperatures than *Legionella.* In several studies, *M. avium* exhibited a similar or stronger correlation with temperature than *Mycobacterium* spp. (Hozalski et al., 2020; Tang et al., 2020). Garner et al. (2018) found the relationship between *P. aeruginosa* and temperature was not significant; Lu et al. (2017) appeared to find a positive relationship up to ~49 °C, but statistical significance was not reported. Other non-*Legionella* bacteria measured in field-scale studies could not be evaluated with respect to temperature.

Multiple field-scale studies documented survival of amoebae at high temperatures. For instance, culturable *Acanthamoeba* spp. was found in water as high as 48 °C in an Iranian hospital water supply (Bagheri et al., 2010). In a 34 °C service line water sample, culturable *N. fowleri* was detected and confirmed by PCR (Cope et al., 2015). Multiple DWDS in warm climates were reportedly colonized by *N. fowleri.* A DWDS in Louisiana (US) with water temperatures >30 °C was found to be colonized with *N. fowleri*, coinciding with a case of primary amoebic meningoencephalitis (Cope et al., 2015). In an Australian DWDS with water temperatures reaching 39 °C, *N. fowleri* and other amoeba were detected in water and biofilms (Miller et al., 2017). Australia considers

DWDSs with temperatures consistently above 20 °C at risk of *Naeglaria* colonization and recommends maintaining a CT (concentration x contact time) >30 mg-min/L in the DWDS (Trolio et al., 2008). Other DWDS studies examined seasonal changes in free-living amoeba populations and found mixed results. In a French DWDS, genera *Vermamoeba* increased while *Echinamoeba* decreased in relative abundance during summer vs. winter (Delafont et al., 2016). Seasonal effects are difficult to interpret due to multiple variables beyond temperature that can affect amoeba levels in DWDS (Morgan et al., 2016).

Some fungal OPs have been identified in drinking water (Anaissie et al., 2001). Buse et al. (2013) measured fungi in premise plumbing (cold water) via amplicon sequencing and found concentrations peaked in autumn. Hu et al. (2021) found no effect of season or temperature on culturable fungal biomass, though they found temperature drove overall variation in microbial community composition, similar to Buse et al. (2017). No studies were found that isolated the effect of temperature on specific fungal pathogens. However, those that can grow at temperatures above 37 °C may be more likely to be human pathogens (Robert and Casadevall 2009). Fungi can evolve thermotolerance to adapt to increasing environmental temperatures (Nnadi and Carter, 2021), and thus may become a bigger concern in drinking water as temperatures rise.

6. Critical knowledge gaps and future research needs

High temperatures have deleterious effects on DWDS, yet significant gaps in knowledge were identified that hinder understanding of the magnitude of the problems as well as development of solutions.

6.1. DWDS temperatures

This review presents to our knowledge the most extensive compilation of DWDS temperature data to date. Interpretation of this data is hindered by lack of robust reporting, as temperature is typically treated as an ancillary measurement. Most studies had low spatial and/or temporal resolution, and few reported ambient air temperatures, such that it cannot be determined if the DWDS temperature data represents typical or aberrant conditions for these systems. Seven studies (including the present work) reported temperatures for DWDS in relatively hot, dry climates. However, only three of these studies featured robust reporting of data from multiple sampling locations and events. Given the diversity of climates within this category, as well as the variation in DWDS design and operation around the world, more research is needed to establish the representativeness of these data. In particular, targeted research is needed to clarify the typical maximum temperatures and the extreme maxima possible during extreme heat waves in hot climates around the world.

Predictive models for DWDS temperature would be valuable tools for utilities and researchers, but existing models have not been validated against real DWDS water temperatures, with the exception of Blokker and Pieterse-Quirijns (2013), in which the model was validated against tap water which was flushed until temperatures stabilized. Current models may suffer from significant prediction error due to exclusion of the urban heat island effect, dynamic demand rates and storage infrastructure. DWDS storage reservoirs and tanks may be particularly vulnerable to heat, due to shallow or aboveground placement and long residence times, and thermal stratification in tanks is an additional complication that may need consideration. Different assumptions for hot vs. cool climate DWDS may be needed to generate accurate temperature predictions.

6.2. OP persistence and microbial ecology at high water temperatures

Some OPs survive or grow at temperatures beyond the maximum reported DWDS temperature (52 °C). However, substantial knowledge gaps remain regarding persistence of many OP species, their ecological interactions, and virulence at high water temperatures. Most studies used methodologies with significant caveats, not accounting for the effect of high temperatures on transitions between culturable, VBNC, and inactivated states, or on survival within biofilms or in amoeba vesicles which may facilitate aerosolized exposure. Few field studies were conducted in hot climate DWDS, and the focus on seasonal comparisons is difficult to interpret with respect to temperature. Other meteorological factors (e.g., precipitation) can also influence OP occurrence (Brigmon et al., 2020). Finally, exposure during extreme heat may be further exacerbated by changes in water use behavior, such as use of mist tents in cooling centers, which can produce contaminated aerosols (Masaka et al., 2021). Future studies should investigate how high DWDS temperatures modulate OP occurrence and health risks in the context of different climatic and exposure scenarios.

6.3. Chlorine decay at high DWDS temperatures

Despite the abundance of literature addressing the effect of temperature on chlorine decay, the quality of reporting of experimental conditions, sample water quality and treatment level was often insufficient. Reporting of experimental error, confidence intervals and model prediction error was also inconsistent. Few studies systematically evaluated the interaction of temperature and other variables, such that the reasons for substantial variation in E_a/Rs between waters cannot be conclusively determined. Chlorine decay modelling is necessarily empirical, but future research could couple modeling with targeted experiments to answer fundamental questions regarding the effect of increasing temperature on mechanisms and drivers of chlorine decay.

To enable accurate predictive models for chlorine decay in any water without extensive bench testing, we may need to identify which temperature-sensitive reactants or bulk water quality characteristics are predictive of the chlorine decay activation energy across waters. The effects of temperature, pH, and other factors on chlorine speciation may be important, particularly considering reactive species which occur in low concentrations (Rose et al., 2020). Additionally, improved methodologies are needed to determine wall decay rates under realistic conditions. As high temperatures can significantly aggravate DWDS pipe corrosion and biofilm growth, predicting wall decay rates may be the most critical need.

6.4. Nitrification at high temperatures

Few systematic studies address the effect of temperature on chloramine decay in real waters. Knowledge gaps in the mechanisms of chloramine decay in real DWDS hinder development of accurate predictive models (Hossain et al. 2022). The knowledge gaps are even greater at high temperatures. Nitrification is empirically associated with warm weather, but the potential for nitrification at temperatures above 35 °C is unclear. Other fundamental knowledge gaps include which organisms are primarily responsible for DWDS nitrification, and how temperature affects their growth rates. Shifts in nitrifier community composition at higher temperatures could have consequences for the efficacy of chloramine disinfection.

6.5. Effects of temporal variability of temperature on DWDS phenomena

The occurrence of extreme diurnal temperature swings in DWDS (> 20 °C) raises numerous questions about effects on chemical and biological processes. Most controlled experiments reported used constant temperatures, or one short pulse (3–10 min) at a high temperature, and few full-scale DWDS studies had robust reporting or analysis of temperature variability. The effect of repeated heating cycles on microbial community composition and OP persistence may be particularly profound. Future research should investigate effects of repetitive, transient exposure to high temperatures on OP persistence and ecological interactions.

6.6. Strategies to mitigate effects of high temperatures on DWDS

Designing DWDS to minimize heat transfer in hot climates will help protect water quality, along with some physical assets, from the deleterious effects of heat. Increasing the minimum recommended depth of DWDS (e.g., from 12″ to 1-m) is perhaps the most obvious way to delay or prevent some heat transfer; this would require updating local or national plumbing codes. Reducing water age might decrease maximum DWDS temperatures in some systems. Substantial water age reductions could be achieved in large water systems through decentralization, though this emerging approach faces numerous implementation challenges (Zodrow et al., 2017). Storage facilities that contribute to heating could be retrofitted with insulation, and placement of new facilities could be selected to minimize heat transfer. Finally, pipe materials could be selected for low heat transfer as well as material heat resistance. Iron alloys are strong heat conductors, and can corrode more rapidly at high temperatures (Lin et al., 2019), which exacerbates disinfectant decay and other water quality issues. However, cyclic heating and cooling could facilitate the degradation and chemical leaching of plastic polymers, and plastics like polyethylene have been found to support more opportunistic pathogens and higher bacterial diversity compared to ductile iron or stainless steel (Yu et al., 2010; Zhang et al., 2017). Recommendations for DWDS materials, as well as other adaptation strategies in hot climates, should be developed with consideration of affordability.

Municipal climate change adaptation efforts could also help cool DWDS, such as increasing shade coverage to disrupt the urban heat island effect. Agudelo-Vera et al. (2017) found shaded ground at 1-m depth had 5–10 ºC lower soil temperature compared to ground of the same depth in direct sunlight. Heat energy recovery from DWDS has been proposed to provide a sustainable energy source for municipalities, with the added advantage of cooling the water (Ahmad et al., 2021; Blokker et al., 2013). This approach is more sustainable than chilling the water, as reportedly done in some wealthy, hot arid climate nations in the Gulf.

Utility Water Safety Plans should address high DWDS temperatures that occur routinely or during extreme heat events. These efforts would be aided by proven temperature modeling tools that predict DWDS water temperature given weather forecasts (C.M. Agudelo-Vera et al., 2015). Ideally, these models should account for spatiotemporal temperature variability, and integrate with modeling tools for hydraulics, disinfectant decay and OP growth. This would enable identification of DWDS areas that are most susceptible to heat and degraded water quality, informing placement of specific interventions such as disinfectant booster stations. Such models could also aid in determining which regions of the distribution system to flush, and when, in order to prevent nitrifier and/or *Legionella* growth during periods of elevated temperatures (Cohn et al., 2015; CDC, 2021). Water systems that only occasionally experience high temperatures could deploy temporary treatment interventions during heat waves. For example, addition of powdered activated carbon at the treatment plant to increase NOM removal and minimize chlorine demand, or substituting chemicals to minimize nutrient residuals (phosphorus, nitrogen) which promote biological activity. These steps would help minimize disinfectant demand and improve biostability. Future research should also focus on application of alternative secondary disinfectants which are not susceptible to decay in the DWDS (e.g., metals, UV).

As ambient temperatures rise in many regions, more water systems may experience high temperatures. Understanding and resolving problems caused by high DWDS temperatures will require industry leadership and collaboration. Some utilities have years of temperature data, as well as robust hydraulic models and resources to invest in protecting DWDS water quality. However, other utilities have limited technical, managerial, or financial capacity to take preventive actions. In both cases, utility managers will benefit from improved understanding of how high temperatures could affect the operation and reliability of DWDS. Ideally, future research on high DWDS temperatures will be supported by industry and governmental partnerships that generate practical knowledge and solutions to protect drinking water quality as temperatures rise.

7. Conclusion

This review and analysis of original data present new insights on the temperatures expected in drinking water distribution systems under extreme heat stress, and their effects on disinfectant decay and opportunistic pathogen persistence. We identified temperatures in real systems that are much than temperatures typically studied at bench-scale, leaving pervasive gaps in knowledge on the effects of high temperatures on drinking water quality. While there are many chlorine decay models in the literature, the substantial variation in the effect of temperature on decay in different waters remains almost entirely unexplained. Additionally, while the relationships between drinking water temperatures and some OPs, particularly *L. pneumophila*, are well founded in existing literature, most thermophilic OPs that may occur in DWDS lack rigorous data to support predictions of their persistence under high temperature conditions. More systems-level research is needed to understand the effects of high temperature conditions on disinfectant decay and OP dynamics in drinking water systems, especially given variation in infrastructure and human behavior changes during such disasters in order to protect public health.

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CRediT authorship contribution statement

Kirin E. Furst: . **Katherine E. Graham:** Writing – review & editing, Writing – original draft, Visualization, Methodology, Investigation, Funding acquisition, Conceptualization. **Richard J. Weisman:** Writing – review & editing, Writing – original draft, Investigation. **Kadmiel B. Adusei:** Writing – review & editing, Investigation.

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests:

Kirin E. Furst reports financial support was provided by National Science Foundation. Katherine E. Graham reports financial support was provided by National Science Foundation. Kadmiel B. Adusei reports financial support was provided by National Science Foundation. If there are other authors, they declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

The authors do not have permission to share data.

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Supplementary materials

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