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Constructed Wetlands for Greywater Recycle and Reuse: A Review

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Abstract

Concern over dwindling water supplies for urban areas as well as environmental degradation from existing urban water systems has motivated research into more resilient and sustainable water supply strategies. Greywater reuse has been suggested as a way to diversify local water supply portfolios while at the same time lessening the burden on existing environments and infrastructure. Constructed wetlands have been proposed as an economically and energetically efficient unit process to treat greywater for reuse purposes, though their ability to consistently meet applicable water quality standards, microbiological in particular, is questionable. We therefore review the existing case study literature to summarize the treatment performance of greywater wetlands in the context of chemical, physical and microbiological water quality standards. Based on a cross-section of different types of wetlands, including surface flow, subsurface flow, vertical and recirculating vertical flow, across a range of operating conditions, we show that although microbiological standards cannot reliably be met, given either sufficient retention time or active recirculation, chemical and physical standards can. We then review existing case study literature for typical water supply disinfection unit processes including chlorination, ozonation and ultraviolet radiation treating either raw or treated greywater specifically. An evaluation of effluent water quality from published wetland case studies and the expected performance from disinfection processes shows that under appropriate conditions these two unit processes together can likely produce effluent of sufficient quality to meet all nonpotable reuse standards. Specifically, we suggest that recycling vertical flow wetlands combined with ultraviolet radiation disinfection and chlorine residual is the best combination to reliably meet the standards.

Keywords

nonpotable water reuse; low energy treatment option; wetland treatment performances; log₁₀ reduction targets (LRTs); disinfection

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1 Introduction

Growing concern in the urban water sector over water scarcity, resource use and aging infrastructure that is at capacity or outdated has motivated a large body of research into alternative water reuse strategies (NAS, 2016). Greywater (GW), or wastewater generated from households or buildings not including toilet water, has received attention as a candidate for reuse as it has the potential to carry less organics, nutrients and pathogens than mixed wastewater (i.e. including toilet water or “blackwater”) and is therefore thought to be easier to treat for reuse purposes (Abu Ghunmi et al., 2011; Eriksson et al., 2002; Ghaitidak and Yadav, 2013; Li et al., 2009; Mayer et al., 1999). Additionally, if treated to nonpotable instead of potable standards, less resource-intensive treatment processes may be used. Combined with reducing discharge to wastewater treatment plants and offsetting potable water demand, GW recycle represents a plausible system-level approach to achieve greater water sustainability and resiliency (Ma et al., 2015; Xue et al., 2015). Indeed many developed countries have shown interest in GW recycle systems including New Zealand (Leonard et al., 2016), Australia (GWA, 2010; ChristovaBoal et al., 1996; Tapsuwan et al., 2014) and Germany (Nolde, 2000; Nolde, 2005).

A number of technologies have been used for GW treatment, from simple physical filtration systems such as membranes and sand filters to highly automated and energy-intensive systems that include biological, chemical and physical treatment mechanisms (Friedler et al., 2005; Li et al., 2009; Winward et al., 2008a). Constructed wetlands are often suggested as an economically and energetically efficient way of treating various wastewater streams (Brix, 1999; Jasper et al., 2013; Kröpfelová, 2008; Liu et al., 2015; Vymazal, 2009), though the ability to treat GW specifically has mostly been assessed on a case-by-case basis. Not only are the individual case-study systems diverse, the available data on the ability of constructed wetlands to remove or inactivate pathogens of importance for human health risk (not merely pathogen indicators) are limited. Furthermore, although some authors have found evidence that constructed wetlands as a standalone technology can meet certain existing reuse standards (Gross et al., 2007; Jenssen et al., 2005), many more have found that they cannot do so in a reliable and consistent manner (Garcia et al., 2010; Jokerst et al., 2012), particularly when subject to the more stringent standards found in developed countries with advanced and evolving regulatory criteria.

This paper is intended to provide a quantitative case study review that evaluates the ability of constructed wetlands to treat GW for reuse at the house, school, neighborhood or commercial building scale. Of particular concern is the ability of treated effluent to meet the most advanced standards or guidelines for non-potable reuse, which are mainly driven by pathogen-related human health risk (Beck et al., 2013; NAS, 2016). In order to address the limited availability of specific pathogen removal data, the ability of wetlands to meet physical or chemical criteria that directly influence pathogen removal dynamics was reviewed. In addition, a quantitative review of common disinfection technologies appropriate for treating wetland effluent was performed with an emphasis on studies that disinfected treated GW.

2 Regulatory Criteria

Globally, water reuse standards are variable and governed by the intended use of the treated effluent. In general, limits are imposed on specific parameters to reduce nuisance odors and algal growth as well as to protect environmental and human health. For detailed reviews, see Li et al. (2009), Abu Ghunmi et al. (2011) or Ghaitidak and Yadav (2013).

Table 1 shows United States Environmental Protection Agency (USEPA) guidelines, California standards and Western Australia standards for a variety of end-use categories. These guidelines and standards were selected as they represent two countries at the forefront of water reuse and they include parameters that are typical of other global criteria (Abu Ghunmi et al., 2011; Li et al., 2009). In the United States, standards have been proposed on the state level as a response to the geographically-driven need for water reuse, though they vary by state and by intended use of the treated effluent. California, a state that experiences chronic water supply crises and thus has a strong incentive to reuse water, is often recognized as having some of the most stringent state-level standards mostly due to the requirement for a 5 log₁₀ reduction (LR) of poliovirus or similar virus (CDPH, 2010). At the national level, the USEPA has issued recommended guidelines (USEPA, 2012), though official standards have yet to be adopted. Outside of the US, Australia has been somewhat of a pioneer in water reuse implementation and regulation due to the extended and widespread drought conditions of the early 2000s (Floyd et al., 2014; Wong and Brown, 2009). In all cases, the limits for organics and microbiology are the focus, with treatment of the former often a prerequisite for the effective treatment of the latter.

A critical drawback of traditional standards and guidelines is that they are not risk-based and therefore lead to an unknown level of protection of human health (NAS, 2016). Recently, work based on quantitative microbial risk assessment (QMRA) for specific pathogens has been completed with corresponding recommendations for treatment requirements in the form of log₁₀ reduction targets (LRTs) (Jahne et al., 2017; Schoen et al., 2017). Unfortunately, few studies exist that have produced reliable data on either the presence or treatability of these specific pathogens, particularly with respect to GW and GW treatment wetlands. In the following sections, wetland performance was largely evaluated in the context of the conventional indicator concentration-based criteria, as they are what drive most regulatory and monitoring protocols, though LR performance of relevant pathogens and pathogen indicators is discussed where possible.

3 Greywater Characterization

As mentioned previously, GW is often cited as a lower strength, less polluted wastewater stream as compared to a typical mixed wastewater stream that includes toilet effluent. Greywater can also further be subdivided into what is commonly referred to as “light” and “mixed”, the former consisting of bath, shower and bathroom sink water and the latter including laundry and kitchen sink water. Numerous authors have provided detailed characterization of individual streams as well as combined streams (Eriksson et al., 2002; Jefferson et al., 2004; Rose et al., 1991) and still others have provided helpful compilations of the individual studies (Boyjoo et al., 2013; Friedler et al., 2011; Ghaitidak and Yadav,

2013; Li et al., 2009). Of importance for this review is both the variability in GW quality as well as the upper ranges to expect from the various sources. Accordingly, Table 2 shows the ranges reported by the previously mentioned review articles for light GW and mixed GW, mixed wastewater for comparison (Lowe et al., 2007; Metcalf, 2003), and parameter ranges for the case studies reviewed later in this article.

The physical and chemical parameter ranges given in Table 2 show that, as expected, light GW is generally of lower strength than mixed GW, and mixed GW is generally of lower strength than mixed wastewater (i.e. including blackwater). The larger values for biochemical oxygen demand (BOD) and chemical oxygen demand (COD) are often attributed to heavy detergent or food waste loads associated with laundry or kitchen sources (Ghaitidak and Yadav, 2013), and can be particularly extreme if unmixed with more dilute sources, even exhibiting similar or greater concentrations than mixed wastewater. Moreover, compared to the oxygen demand of mixed wastewater that stems in part from more readily degradable fecal material and food waste, that of GW can be due in part to less biodegradable surfactants from soaps and detergents (ChristovaBoal et al., 1996; Sharvelle et al., 2007). Thus, not only is a biological treatment step often necessary to meet BOD standards (Nolde, 2000), but also it must be metabolically diverse, capable of oxidizing more recalcitrant organic compounds.

TSS concentrations in GW are often less than mixed wastewater due to the absence of feces and bathroom tissue, which represent sources of larger solids. However, GW TSS concentrations can still be much greater than regulatory standards. More importantly, numerous authors have noted an association between suspended solids and larger pathogens including bacteria, bacterial indicators and protozoa (Falabi et al., 2002; USEPA, 2000; Winward et al., 2008b; Winward et al., 2008c). Thus, treatment systems that promote either the settling or filtration of solids have been shown to be effective in removing these larger pathogens (Falabi et al., 2002; Garcia et al., 2010; Gerba et al., 1999; Karim et al., 2004).

In terms of nutrients, ranges are again large. Phosphorus concentrations can be high, particularly in areas that have not adopted stringent legislation banning the use of phosphate-based detergents (Turner et al., 2013). Though not readily apparent in Table 2, past studies that have characterized individual GW streams have suggested the potential for nutrient deficiency (e.g. high C:N ratio), particularly if kitchen water is excluded (De Gisi et al., 2016; Jefferson et al., 2004; Li et al., 2009), which could reduce the efficiency of biological treatment processes. For example, Bergdolt et al. (2013) calculated the first order aerial removal rate for BOD from a FWS constructed wetland treating bathroom wash water (BOD:TN of approximately 6:1) and found it to be depressed relative to systems treating domestic or agricultural wastewaters. Of the ranges shown in Table 2 for the wetland studies reviewed, there is a potential for nutrient (nitrogen in particular) deficiency. Compared to a suggested BOD:TN ratio of 5:1 for biological processes (Metcalf et al., 1991), ranges for individual wetland studies reviewed here returned BOD:TN ratios of 3.3 to 49 (see Table S1).

With respect to the bacterial pathogens identified in Table 2, there is also a general increase in strength from light GW to mixed wastewater, as expected. Also, similar to the physical

and chemical parameters, it is clear that mixed GW fecal indicator concentrations often reach those of mixed wastewater, with ranges of up to 8 log (CFU/100 mL).

In addition to the large variability in bacterial pathogen data, Table 2 shows a clear lack of data regarding specific virus or protozoa counts in GW. Data are present for standard bacterial indicators (total coliform, fecal coliform), enteric-specific indicators (*Escherichia coli*, enterococci), an opportunistic bacteria associated with human skin and mucous membranes, (*Pseudomonas aeruginosa*, *Staphylococcus aureus*), a bacterium proposed as an indicator for the inactivation of enteric viruses and whose associated spores have been proposed as an indicator for parasitic protozoa (*Clostridium perfringens*) (Ottoson and Stenstrom, 2003; Payment and Franco, 1993), an aerosol-transmitted aquatic bacterium that can cause respiratory illness (*Legionella*), a gastrointestinal and sometimes pathogenic bacterium (*Salmonella*) and a surrogate for enteric viruses (Male-specific 2 (MS2) coliphage) (Havelaar et al., 1987). The lack of data for pathogens assessed by Schoen et al. (2017) is due to quantification cost, detection limits and occurrence; if an infected individual is not present in the house or building being sampled, the specific pathogen will likely not be detected.

For all parameters given, the range of influent concentrations from the wetland studies that will be discussed are mostly within the ranges given for light GW, despite the fact that a mix of source waters was treated. In addition to the fact that we reported ranges, not means, this discrepancy is also likely due to proper mixing; the majority of wetland studies reviewed incorporated an initial mixing stage (e.g. mixing basin, sump, etc.) that allowed temporary spikes in contaminant loading (e.g. laundry first rinse) to be dampened. It also allowed for dilution of high strength sources (e.g. laundry and kitchen) with lower strength sources (e.g. bathroom sink).

4 Greywater Treatment using Constructed Wetlands

Wetlands are complex systems that are home to myriad abiotic and biotic processes capable of removing, degrading or transforming many compounds considered to be pollutants (Garcia et al., 2010; Jasper et al., 2013; Kadlec and Wallace, 2008; Wetzel, 2001). Numerous authors have argued that a range of processes including biological treatment and physical filtration are necessary for the effective treatment of GW (De Gisi et al., 2016; Li et al., 2009; Nolde, 2000; Nolde, 2005), processes that are simultaneously present in wetlands. However, the diverse range of processes that makes wetlands uniquely suited to a wide range of removal processes also makes them variable in their treatment performance. Additionally, widely used modeling approaches and performance databases such as the P-k-C* model (Kadlec and Wallace, 2008) and the North American Treatment Wetland Database (NADB, 1998) used to predict performance are mostly based on systems treating domestic or agricultural wastewater, which are compositionally different from household GW (Bergdolt et al., 2013; Sharvelle et al., 2007; Todt et al., 2015). As such, this section is intended to review observed treatment dynamics of constructed wetlands treating GW. Specifically, we discuss the ability of constructed wetlands to reduce physical/chemical parameters that influence pathogen treatability, the ability for constructed wetlands to directly reduce or inactivate specific pathogens or pathogen indicators, and the ability of constructed wetlands

to produce effluent of suitable quality for disinfection by commonly used technologies including chlorination and UV radiation.

A total of 13 studies were reviewed, from which 38 individual system datasets were extracted. Table 3 summarizes the chemical, physical and pathogen performance by system type, while full details of individual studies can be found in Table S1. Wetland systems assessed included free water surface (FWS), green roof water recycle (GROW), horizontal subsurface flow (HSSF), vertical flow (VF), recycling vertical flow (RVF) and combination systems. For combination systems (e.g. FWS+HSSF), if enough data were available for individual components as well as the entire system, multiple system datasets were extracted (e.g. FWS, HSSF, FWS+HSSF). Studies were included only if data were provided that allowed for either the extraction or calculation of at least one parameter from each of the following categories: i) hydraulic loading rate (HLR, daily inflow volume divided by system area) and hydraulic retention time (HRT, volumetric capacity divided by daily inflow volume), ii) physical/chemical influent and effluent data, either BOD, TSS or turbidity, and iii) influent and effluent pathogen concentrations.

4.1 BOD, TSS and Turbidity

Figure 1 shows the resulting effluent concentrations of BOD, TSS and turbidity as a function of HRT. In each plot, USEPA guidelines for restricted and unrestricted use from Table 1 are represented with dotted and solid lines, respectively. Looking at all systems combined, there appear to be general trends for each parameter in which longer retention times produce lower effluent concentrations. However, looking at individual system types we see much variation with sometimes little to no relationship between HRT and effluent quality, indicating there are significant confounding factors to treatment performance predictability.

The ability of constructed wetlands to remove BOD has been well documented and is one of the more predictable treatment performance parameters (Kadlec and Wallace, 2008). Being a biological process, it is subject to prevailing redox conditions, the influence of temperature on metabolic rates, and the recalcitrance of the organic compounds being oxidized. In terms of redox conditions, we would expect more aerobic environments to increase the BOD removal rate as more oxygen is available to satisfy demand. While there are instances of each type of wetland producing low BOD effluent (Figure 1a), the VF and RVF most consistently produce low BOD effluent. Indeed, in looking at Figure 1a, we see that the VF and RVF systems achieve some of the lowest BOD effluent concentrations even at HRTs less than 5 days. This can also be seen by looking at Table 3, which shows an average removal rate for RVF and VF systems of 98% and 85%, while those for FWS, GRW and HSSF systems are 63%, 71% and 87%, respectively. Moreover, the RVF systems achieve low effluent concentrations (less than 5 mg/L) more consistently compared to single pass VF systems, as continuous circulation allows for enhanced oxygenation.

Temperature has been shown to affect BOD treatment performance as well and could be a confounding factor in the low correlation between intra-system treatment performance and HRT shown in Figure 1. Insofar as microbial metabolic processes in wetlands often proceed at greater rates in warmer temperatures (Reddy and DeLaune, 2008), wetlands located in colder climates may have a lesser capacity to reduce BOD concentrations, and treatment

performance may be diminished in winter months. One system, the FWS component of a hybrid FWS+HSSF system located in Colorado (Jokerst et al., 2012), only achieved an average annual effluent concentration of 31.7 mg/L, despite having a HRT of 6.8 days. This was due to lower treatment efficiency in winter months, as the water temperature within the wetland averaged only 4°C. Temperature also affected the treatment performance of the second stage of this hybrid FWS + HSSF system; the data points in Figure 1 with HRTs of 14 and 20 days are the HSSF component and combination system from this study, respectively. Although the effluent from the entire system averaged 12.7 mg/L for the year, during winter months it averaged 45.6 mg/L, well above even the restricted standard of 30 mg/L.

The chemical composition of GW also may affect its treatability. Compared to typical domestic or agricultural wastewaters, GW has the potential to be nutrient deficient and contain relatively recalcitrant surfactants, two factors that have been hypothesized to lower the efficiency of biologically mediated processes responsible for oxidizing organic matter and lowering BOD concentrations (Sharvelle et al., 2007). Although we do not have data to test these mechanisms directly, we can compare the performance data reviewed here to the NADB (NADB, 1998) which is composed of systems that treat predominantly mixed domestic effluent, agricultural wastewater, wastewater treatment plant effluent, stormwater and industrial effluent. Figure 2a shows data from this review plotted against monthly averaged BOD data from 49 different NADB wetlands. For a given loading, we generally see the GW treatment wetlands producing lower BOD effluent than the central tendency of the NADB wetlands. Again, we see the VF and especially the RVF systems performing the best, even outside of the space occupied by the NADB distribution. On one hand, the enhanced performance of the RVF systems is to be expected for BOD, as the systems documented in the NADB are mostly single pass, non-aerated systems. However, the tendency of the GW systems in general to perform better than the NADB systems gives evidence that is contrary to the previously suggested hypotheses; that nutrient limitation, recalcitrance of organics or both may lead to depression of treatment performance. Rather, Figure 2a would suggest that wetland design equations for BOD based on rate constants derived from the NADB (e.g. (Kadlec and Wallace, 2008), Chapter 8) are appropriate and may even be conservative for the design of GW treatment wetlands.

TSS is removed in constructed wetlands through a variety of mechanisms including settling, flocculation and filtration (USEPA, 2000), with different processes being more dominant in different systems. In FWS systems, low water velocities made possible by greater pore space allow for greater initial settling of larger solids (Gearheart et al., 1989), which can sometimes even occur on the order of hours (Wong et al., 2006). In HSSF and VF systems, these larger particles are also removed initially, though through more of a physical filtration or interception mechanism owing to their greater media surface area. Following removal of larger particles, filtration or adsorption of smaller and colloidal particles occurs which is directly related to the available surface area of media, vegetation or substrate (Garcia et al., 2010). For GW treatment, systems that can provide a range of removal mechanisms are most ideal as the suspended solids particle size distributions are often highly variable. For example, Ramon et al. (2004) studied bathroom sink and shower water and found that colloidal size particles were the dominant fraction in terms of number distribution with a

mean particle size of 0.1 μm while much fewer, larger particles made up most of the particle volume. This variability in size distributions has been noted by other authors as well who have found volume weighted mean particle sizes of GW ranging from 100–500 μm (Frazer-Williams et al., 2008; Ramon et al., 2004; Winward et al., 2008c).

Looking at Figure 1b, TSS shows a similar trend to BOD in that when taking all systems together, lower TSS effluent concentrations appear to be achieved as a function of greater HRT. However, if looking at individual system types, VF and RVF are the only system types without considerable scatter. For RVF systems in particular, even with HRTs of 3.5 days or less, effluent TSS concentrations are 1–3 mg/L, below any standard given in Table 1.

Comparing the GW-specific TSS dataset from this review to the equivalent dataset from the NADB in Figure 2b, we again see the GW treatment wetlands performing, in general, better than the NADB central tendency with the RVF systems performing best. As TSS removal is generally attributed to filtration processes, the continuous recirculation of the treatment volume through the RVF systems essentially means that for any “batch” of water applied to the system, its constituents are exposed multiple times to the filtration media increasing the probability of contact and adsorption. For all RVF systems reviewed, the recirculation rate tended to be approximately half the total system volume per hour, meaning that if plug flow were assumed the entire volume of the system would be recirculated (and re-exposed to the filter medium) once every two hours. Combined with the fact that the filter media in these systems are variable, including a layer of pebbles, tuff or plastic media and soil-based root zone, there are a variety of surfaces and for TSS to adhere to thus increasing the probability of obtaining particle/media interactions that are conducive to effective adsorption or filtration.

In terms of turbidity, although no system reliably meets the 2 NTU criteria for unrestricted reuse (restricted reuse does not have a turbidity requirement), many of the systems are capable of producing effluent with turbidity levels less than 20 NTU, particularly if HRTs of 3–5 days are achieved. As with BOD and TSS, we also see the RVF systems performing well, with the five RVF systems that report turbidity achieving effluent levels of less than 10 NTU. Again, this is likely attributed to the enhanced contact time with the range of media types found in these systems, particularly the lower, finer layers. For single pass systems, several authors have found that sand filtration as a polishing step is effective at removing turbidity of biologically treated effluent (Gerba et al., 1999; Li et al., 2009), likely due to the high surface area available for filtration and adsorption of smaller suspended, dissolved and colloidal organics. Although sand is not a commonly used media in the RVF systems reviewed, it may be that increased exposure time due to recirculation is an effective substitute for single-pass treatment through high surface area media.

4.2 Microbiological Parameters

Thus far we have shown that RVF systems can reliably meet Table 1 criteria for BOD and TSS and lower turbidity levels to less than 10 NTU. Additionally, we have shown that the GW constructed wetlands reviewed produce effluent that is generally comparable to or better than the larger treatment wetland performance literature. However, the ability of GW constructed wetlands to reliably meet microbiological criteria is far less robust or

predictable. Here we review the mechanisms by which constructed wetlands have been shown or suggested to remove pathogens or pathogen indicators and provide the results of the pathogen-related treatment performance of the GW treatment wetlands reviewed.

In general, pathogens, as they relate to water treatment and human health, are divided into three main groups with the differences in size, occurrence, persistence and treatability: bacteria, protozoa and helminths, and viruses (USEPA, 2012). Bacteria are microscopic organisms ranging from approximately 0.2 to 10 μm in size and can be removed or inactivated in constructed wetlands through a variety of processes including sedimentation of particle-associated bacteria, physical filtration by adsorption onto vegetation, media or biofilm, UV inactivation, attack by bacteriophage and predation by micro-zooplankton (Diaz et al., 2010; Garcia et al., 2010; Gerba et al., 1999; Morato et al., 2014; Stenstrom and Carlander, 2001; Vymazal, 2005). Protozoa and helminths can be excreted in feces as spores, cysts, oocysts or eggs which make them particularly resistant to environmental stresses including temperature extremes, sunlight and desiccation. They can be larger than bacteria, ranging in size from 1 μm to over 60 μm (USEPA, 2012), making physical removal processes including sedimentation and filtration effective (Garcia et al., 2010; Karim et al., 2004). Viruses are the smallest of the three groups, ranging in size from 0.01 to 0.3 μm . Due to their small size and ability to persist in the environment, physical removal processes can be less effective compared to bacteria and protozoa, necessitating a disinfection step for high levels of removal or inactivation. Although UV inactivation is a common method of virus inactivation, viruses generally require a higher dose of UV for inactivation compared to bacteria and protozoa, which, combined with an infectivity dose that can be orders of magnitude less than bacteria and protozoa, often makes them the focus of more stringent human health risk standards (Al-Gheethi et al., 2016; Asano, 2005; USEPA, 2012).

The second half of Table 3 summarizes the pathogen removal performance of the reviewed GW constructed wetlands, with full performance data provided in Table S1. All microorganisms listed in Table 3 are bacteria; *Staphylococcus aureus*, *Pseudomonas aeruginosa* and *Legionella* are pathogens capable of causing disease in humans, while the remaining bacteria are indicator organisms which are used to indicate the likelihood of human health risk occurrence (USEPA, 2012). No treatment performance data were found for virus, protozoa or helminths in GW treatment wetlands.

The ranges in Table 3 show that GW constructed wetlands are generally capable of providing up to 2 to 5 LR for bacteria, though LRs can also regularly be less than one. To test if HLR or HRT predicted any variation in LR for total coliform, fecal coliform or *E. coli* (these being the parameters with sufficient reported data), logarithmic regression analyses were performed using data in Table S1. Although an R^2 of 0.41 was found for the prediction of total coliform by HLR, an R^2 of less than 0.2 was found for all other relationships showing little ability to predict LR performance using standard system operating characteristics.

The LRs observed in Table 3 are generally consistent with larger treatment wetland databases, with bacterial LRs from the systems in Kadlec and Wallace (2008) showing a central tendency of 1–2 LR, though inter- and intra-system ranges can be quite large. Also,

in their respective reviews of removal processes in CWs, Vymazal (2005) and Garcia et al. (2010) found bacterial indicator removals to fall somewhat consistently in the 1–2.5 LR range. However, both cautioned against using CWs as a standalone unit process for the treatment of water with high bacterial concentrations as, despite consistent LR, effluent concentrations were best predicted by influent concentrations and often remained high ($>3 \log_{10}$).

To test if, as suggested above, influent bacterial indicator concentrations predicted effluent concentrations, we regressed the two for bacterial indicators having sufficient reported data. Figure 3 shows that influent concentration can explain some variation depending on the indicator, with a resulting R^2 of 0.91, 0.26, and 0.77 for total coliform, fecal coliform and *E. coli*, respectively. Based on these results as well as the lack of relationship with HLR or HRT, we would argue that the ability of constructed wetlands to meet indicator-based effluent standards is less a function of design and more of a function of the quality of water to be treated. Based on the large variability in effluent concentration of up to 8 log for total coliform and 6 log for fecal coliform and *E. coli*, the ability of GW constructed wetlands alone to produce effluent meeting even restricted reuse standards is unreliable.

While the ability of water treatment technologies to meet existing reuse standards (i.e. bacterial concentration-based) is important from an existing regulatory perspective, recent work in microbial risk assessment (Jahne et al., 2016; Schoen et al., 2016) and more progressive treatment standards (e.g. CA Title 22) have begun to argue for more pathogen-specific treatment targets and require demonstration of reduction of traditionally hard to treat non-bacterial pathogens. Although, to our knowledge, no GW treatment wetland study has looked at the removal of protozoa, viruses or virus indicators, we can gain some insight if we expand our scope to mixed wastewater treatment wetlands.

Due to their larger size, the removal of protozoa has been shown to be largely a physical process, attributed to either sedimentation in FWS systems (Falabi et al., 2002; Gerba et al., 1999) or filtration in HSSF and VF systems (Redder et al., 2010). Accordingly, removal of these pathogens, the most commonly reported being *Cryptosporidium* oocysts and *Giardia* cysts, can be reliable where sedimentation and filtration are promoted though not necessarily high. For FWS systems, LRs of *Cryptosporidium* oocysts and *Giardia* cysts of 0.4 to 1.7 have been reported (Falabi et al., 2002; Gerba et al., 1999; Karpiscak et al., 1996). For HSSF systems, LRs of the same pathogens of 0.2–3 have been reported (Nokes et al., 2003; Quinonez-Diaz et al., 2001; Redder et al., 2010).

For virus reduction, data from mixed wastewater treatment wetlands mostly deal with a variety of bacteriophages, which are used as virus indicators. These bacteriophages (a virus that infects and replicates within a bacterium) are similar in size and structure to enteric viruses, do not infect humans, are detectable by simple, rapid and inexpensive methods, and often are more environmentally persistent than enteroviruses (Burge et al., 1981; Havelaar et al., 1993; Havelaar et al., 1991; Kapuscinski and Mitchell, 1983). In order of increasing specificity, the most commonly reported bacteriophages include somatic and f-specific coliphages – types of bacteriophages that infect *E. coli*, the latter being closer in size to human enteric viruses (Cramer et al., 1976); male-specific 2 (MS2), an easily culturable type

of f-specific RNA coliphage; and PRD1, a double-stranded DNA that is easily culturable, similar in size to the human rotavirus and adenovirus (Olsen et al., 1974; Rusin et al., 2000) and has been shown to persist longer in the environment than MS2 (Vinluan, 1996; Yahya et al., 1993).

Similar to bacteria, adsorption onto plant and media surfaces is a primary mechanism of virus removal in wetlands (Garcia et al., 2010; Jackson and Jackson, 2008), often occurring within the first hours of entering the wetland (Hodgson et al., 2003; Quinonez-Diaz et al., 2001). Sedimentation is unlikely to be a significant virus reduction mechanism; an analysis of wastewater treatment pond influent and effluent showed less than 5% of viruses attached to particles greater than 180 μm (Symonds et al., 2014) and wetlands that rely on pathogen removal via sedimentation have shown poor virus removal performance (Falabi et al., 2002).

In VF and HSSF systems, coliphage LR_s have been shown to range from 0.2 to 2 (Falabi et al., 2002; Hench et al., 2003; Thurston et al., 2001; Torrens et al., 2009; Ulrich et al., 2005). Demonstrating the importance of media contact time, Torrens et al. (2009) observed roughly double the LR of both somatic and f-specific coliphages in VF systems with a depth of 65 cm compared to identical systems of depth 25 cm (0.7–1.3 and 0.2–0.7, respectively). They also observed larger LR of somatic coliphages compared to f-specific coliphages, supporting the notion that f-specific coliphages are a more conservative and appropriate indicator. Using seeded MS2, Chendorain et al. (1998) found an overall LR of 1.6 in a FWS system with a 9 day HRT and found removal rates to be bimodal with the majority of reduction occurring in the first 3m of the 30m long wetland. Gersberg et al. (1987) used seeded MS2 alongside seeded poliovirus in a HSSF system operated at a HRT of 5.5 days. They found overall LR of 2.7 for MS2, compared to 3.0 of poliovirus, supporting the conservative nature of the indicator. They also found greater removal of indigenous f-specific coliphages in vegetated vs. unvegetated cells, with observed LR_s of 2.0 and 1.3, respectively. Using indigenous MS2 as well as seeded PRD1 in floating macrophyte FWS systems, Karim et al. (2004) calculated decay rates for each indicator in both the water column and sediment. They found that wetland sediments prolong the survival of viral indicators and that PRD1 is a more conservative indicator, with water column and sediment decay rates of 0.397 d^{-1} and 0.107 d^{-1} for MS2 and 0.198 d^{-1} and 0.054 d^{-1} for PRD1. Lastly, Vidales-Contreras and colleagues have performed a number of tracer experiments with seeded PRD1 and bromine that largely corroborate the findings discussed above. They have shown that PRD1 persists for longer than MS2 in HSSF wetlands and is thus a more conservative indicator (Vidales et al., 2003), that overall reduction rates are greater than inactivation rates (decay rates of $-1.17^{-\text{d}}$ vs. $-0.16^{-\text{d}}$, respectively) suggesting that physical removal via initial adsorption is a more important than virus inactivation in HSSF systems (Vidales et al., 2003), and that plant and media surface area are critical to efficient virus reduction, observing greater reduction of PRD1 in HSSF than FWS systems (Vidales-Contreras et al., 2006), reporting a PRD1 LR of 2 in a 6 year old HSSF system operated at an HRT of 5.5 days. Taking into account the conservativeness of PRD1, this would suggest that for a properly designed HSSF system operated at an HRT of 3–5 days, 1–2 LR of virus can be achieved. Moreover, given the importance of media contact time, a greater virus LR could likely be achieved in an RVF system.

Above we have shown that GW constructed wetlands as a single unit process cannot reliably meet all reuse criteria; however, neither can traditional wastewater treatment unit processes, as evidenced by the need for final polishing or disinfection unit processes at wastewater treatment plants. Constructed wetlands can however generally provide 1–2 LR for most pathogen indicators and species including conservative viral indicators, particularly if media contact time is emphasized. Moreover, most of the reviewed system types were able to achieve at least the restricted use BOD and TSS criteria of 30 mg/L so long as operated with an HRT of at least 3–5 days. RVF systems in particular were able to reliably reduce BOD, TSS and turbidity levels to less than 5 mg/L, 5 mg/L and 10 NTU, respectively, even when operated with HRTs less than 5 days. Thus, we now turn our attention to disinfection unit processes that can be combined with constructed wetlands to reliably achieve the microbiological reductions required.

5 Greywater Disinfection Processes

Similar to the review performed for GW constructed wetlands, we searched the literature for studies evaluating the effectiveness of common disinfection technologies applied to treated GW. Disinfection technologies reviewed included chlorination, ozone and ultraviolet radiation (UV). Although we found only two cases that disinfected wetland effluent directly (El Hamouri et al., 2007; Winward et al., 2008c), we kept our scope limited to GW, with studies that evaluated raw and treated GW including treatment by fine filtration, coarse filtration, cartridge filtration, sand filter (SF), rotating biological contactor (RBC) membrane bioreactor (MBR) and horizontal subsurface flow constructed wetland (HSSF). A total of 9 papers were found from which 85 individual trials were extracted.

Tables 4 summarizes the results of the literature review, with Table S2 providing full experimental details of individual trials. In all, data were found for bacterial indicators (total coliform, fecal coliform, *E. coli* and *Enterococci*), a bacteriophage virus indicator (MS2 coliphage), an enteric bacterium (*Salmonella enterica*) and two opportunistic bacteria (*Staphylococcus aureus* and *Pseudomonas aeruginosa*).

Of the water quality parameters in Table 4, turbidity was the most frequently reported. In the context of disinfection, turbidity can be considered a general indicator of water quality that may influence disinfection technology effectiveness. For chlorine and ozone, turbidity can be an indirect measurement of the presence of organics, which can both quench the reactive species of the applied chemicals and, in the case of particle-associated pathogens, protect the pathogens from disinfection (Benami et al., 2016; Dietrich et al., 2007; Hunt and Marinas, 1999; Janex et al., 2000; Lechevallier et al., 1981; Xu et al., 2002). For UV, turbidity is a measurement of constituents that can attenuate the radiation making it less effective and, similar to ozone and chlorine, shield particle-associated pathogens from disinfection (Benami et al., 2016; Templeton et al., 2005; Winward et al., 2008c). Thus, producing effluent of low turbidity is important for maintaining disinfection technology effectiveness.

If we exclude the trials from Ekeren et al. (2016), which intentionally tested water of high organic carbon content, all disinfection trials treated water with turbidity values less than 10 NTU. If we compare these trials to the effluent produced by wetlands in Table 3, we see that

not all wetlands were able to produce water with these low turbidity values. However, as previously discussed, the RVF systems were able to reliably achieve turbidity levels of less than 10 NTU. This is not to say that effluents of greater turbidity cannot be disinfected but, as we will discuss below, the effectiveness of the disinfection technologies may be reduced, requiring either higher chemical dosages or greater radiation levels.

Figures 4 and 5 show the results of the reviewed disinfection trials for chlorination and UV (see Figure S1 for analogous ozone figures), plotting LR against dose. Although we included ozone trials in Table 4, we will only discuss in detail chlorine and UV as i) chlorine and UV have been more widely tested for GW applications (Winward et al., 2008b; Winward et al., 2008c), ii) ozone tends not to perform as well as chlorine and UV (Ekeren et al., 2016) and iii) life cycle costs for ozone disinfection systems are greater than chlorine and UV (Beck et al., 2013). Importantly, most trials reported full inactivation, meaning that the potentially achievable LR at that particular dose and under those particular conditions was likely not realized. These trials are distinguished by shading in Figures 4 and 5, enumerated in parentheses in Table 3, indicated by “>” in the text, and should be interpreted as “greater than” the LR shown.

5.1 Chlorine Disinfection Trials

Figure 4 shows the results for chlorine disinfection trials, which all used sodium hypochlorite specifically. LRs are plotted as a function of dosage, which is measured as chlorine residual multiplied by contact time (CT). Figure 4a shows bacterial indicators relevant to Table 1 criteria, while Figure 4b shows pathogenic bacteria and MS2 coliphage. In general, most trials conducted at dosages greater than 90 mg/L-min (the lowest prescribe dosage from Table 1) showed full inactivation. Comparing the trials illustrated in Figure 4a that did not result in full inactivation to Table 1 concentration-based criteria, only one of the three total coliform results from Winward et al. (2008b) resulted in an effluent concentration below the CA Title 22 restricted standard of 23 CFU/100mL while the other two trials were in exceedance (Table S2). For the two fecal coliform trials from Friedler et al. (2011) that did not achieve full reduction, effluent concentrations were approximately 2 CFU/100mL, less than the US EPA guideline for restricted reuse of 200 CFU/100 mL but above the non-detect guideline for unrestricted reuse. Below, we will first describe the factors that likely prohibited full inactivation before discussing implications for achievable LRs.

Total coliform trials that did not result in full inactivation were reported by Winward et al. (2008b), in which observed tailing at CTs of 60–150 mg/L-min was attributed to ‘robust particle shielding’ – TSS and turbidity of the tested water were 29 mg/L and 20 NTU, respectively. This was despite an initially high chlorine dosage of 80 mg/L. In contrast, Ekeren et al. (2016), who specifically tested high organic waters (Turbidity 26–37 NTU), found full inactivation of *E. coli* and *S. enterica* at CTs of 180–474 mg/L-min. Although they applied greater CTs, Ekeren et al. (2016) attributed the observed disinfection efficiency to a phenomenon observed by others (Dietrich et al., 2003; Winward et al., 2008b) in which water with greater organic content allows for greater initial chlorine dosages while not exceeding residual limits (a residual greater than 4 mg/L will damage metal fixtures), and

greater initial chlorine dosages increase chlorine particle penetration to inactivate particle-associated microorganisms.

Fecal coliform trials that did not result in full inactivation were reported by Friedler et al. (2011), who reported LR of 1.9 resulting from dosages of 15 and 30 mg/L-min. It should be noted however that these results were after a CT of 0.5 hours; after an extended CT of 6 hours in which residual chlorine remained between 0.5 and 1.0 mg/L, average effluent fecal coliform concentrations were reduced to less than 1 for both dosages, achieving a LR of 2.2 at full inactivation.

MS2 coliphage data come from Beck et al. (2013) and Ekeren et al. (2016), who tested relatively low organic and high organic GW, respectively. Whereas Ekeren et al. (2016) generally achieved more effective inactivation of bacteria species discussed above, they were only able to demonstrate LR of MS2 coliphage of 3.8 at a dosage of 474 mg/L-min compared to LR of >5–7 at dosages of 200–450 mg/L-min reported by Beck et al. (2013). Moreover, the one trial reported by Beck et al. (2013) in which full inactivation was not achieved was performed at a dosage of 90 mg/L-min and resulted in an LR of 4.4, greater than the 3.8 reported by Ekeren et al. (2016) at a dosage of 474 mg/L-min. This suggests that in terms of chlorine disinfection of viruses, removal of organics is critical in meeting standards such as the 5 LR required by California Title 22 (CDPH, 2010).

Shifting our focus to the LR effectiveness of chlorine disinfection, we again note that most all trials illustrated in Figure 4 were influent limited. However, we can summarize minimum expected LR which may be viewed as conservative predictions of reduction under similar conditions where influent concentrations are not limiting. For total coliform and *Enterococci*, influent limited results came from the low organic trials of Beck et al. (2013). They showed total coliform LR of >2.6–3.5 at dosages of 68–100 mg/L-min and *Enterococci* LR of >1.8–2.9 at dosages of 68–200 mg/L-min. For both parameters, additional trials were conducted at dosages of up to 460 mg/L-min using the same feed water, resulting in full activation as well. Fecal coliform was measured in several trials by Friedler et al. (2006; 2005; 2011), resulting in LR of >1.4–5.1 at dosages of 30–42 mg/L-min. Of note is the trial conducted at 42 mg/L-min that resulted in the largest LR of >5.1, as it treated SF effluent with relatively high organic content (TSS of 32 mg/L, turbidity 35 NTU). Lastly, Ekeren et al. (2016) demonstrated seeded *E. coli* LR of >5.2–7.1 at dosages of 180–474 mg/L-min (high organic water).

Figure 4b shows LR data for pathogenic bacteria and MS2 coliphage. Ekeren et al. (2016) showed that LR of *S. enterica* >7 could be achieved at dosages of 180–474 mg/L-min, and LR of *P. aeruginosa* of >7.4 could be achieved at a CT of 474 mg/L-min. Friedler et al. (2011) showed that lower dosages of 15–30 mg/L-min resulted in LR of the opportunistic bacteria *S. aureus* of >1.9 and *P. aeruginosa* of 1–1.3, noting that higher initial chlorine doses were more effective at inactivation of *S. aureus* than *P. aeruginosa*. Other studies have found *P. aeruginosa* to be less susceptible to chlorination than other enteric and indicator bacteria (Benami et al., 2015; Blanky et al., 2015). Friedler et al. (2011) showed that given a low initial chlorine dose (less than 5 mg/L), *P. aeruginosa* tended to persist after 6 hours of storage despite maintenance of a residual concentration of 0.5–1.0 mg/L. Lastly, but most

relevant to the 5-log reduction requirement in Table 1, are the MS2 coliphage trials by Beck et al. (2013). Using four different feed waters across a range of dosages, they showed LR_s of >5.5, >6.1, >6.2 at minimum dosages of 45, 90, 100 mg/L-min, demonstrating the effectiveness of chlorine at reducing virus loads in low organic water.

While chlorination appears to be an effective disinfection option for the types of water being discussed, the potential for disinfection byproduct (DBP) formation of wetland-derived effluent may be elevated. A study performed by Quanrud et al. (2004) compared the formation potential of carcinogenic trihalomethanes (THM, a common DBP) from wetland-derived organic matter and wastewater treatment effluent organic matter. They found that wetland-derived organic matter has a higher aromaticity, lower biodegradability, and higher chlorine reactivity than typical wastewater treatment plant effluent, resulting in greater THM formation potential following wetland treatment. Two important factors must be taken into account however. First, if RVF effluent is being used, organic matter concentrations will generally be lower than traditional wetland effluent, resulting in lower DBP concentrations. Second, the ultimate exposure pathway must be evaluated. For example, the greatest human health risk from DBPs is cancer from long term ingestion, i.e. drinking. As we are discussing nonpotable uses, inhalation or dermal exposure during high contact activities like showering would be the next pathways to consider, dermal exposure generally being of greater risk than inhalation during showering (Wang et al., 2007). Although certain analyses have shown the human health risk of DBP dermal exposure is less than more common microbial health risks associated with nonpotable reuse (Schoen et al., 2014), a process-specific risk assessment would likely be warranted if using chlorinated wetland effluent for high exposure activities like bathing or showering.

5.2 UV Disinfection Trials

Figure 5 shows the results of the UV disinfection trials. For dosages less than 100 mJ/cm², a number of trials did not reach full inactivation. Of the parameters subject to Table 1 concentration-based criteria, we see examples for each. For total coliform, four trials from Winward et al. (2008c) did not reach full inactivation, though one trial was still less than the CA Title 22 standard for unrestricted reuse. The trials in violation were conducted for a range of dosages (5.8, 69 and 277 mJ/cm²) though at a higher organics content than those not in violation; 19 mg/L TSS, 20 mg/L BOD and 18 NTU vs. 4, 6 and 2, respectively. The trial conducted at 277 mJ/cm² met the restricted standard of 2.2 CFU/100 mL, though not the unrestricted standard of 23 CFU/100 mL. In these trials, they used untreated GW from bathroom sinks and showers and found that particle shielding created a tailing effect, particularly for total coliform. While *E. coli* and *Enterococci* were fully inactivated at a dosage of 277 mJ/cm², total coliform persisted at a mean concentration of 1.0 log₁₀MPN/100mL. The authors even increased the dose to 1107 mJ/cm² and still observed a mean concentration of 0.9 log₁₀MPN/100mL (data not shown). Although the tailing of total coliform and not *E. coli* or *Enterococci* may be due to greater influent concentration of the former, the results give evidence to the strength of the particle shielding effect against UV inactivation.

For fecal coliform, the trials that did not reach full inactivation were conducted by Friedler et al. (2011) for low organic (turbidity 1.5 NTU, BOD 3.7 mg/L) rotating biological contactor effluent, though dosages of these trials were all less than 100 mJ/cm² and all concentrations were below the USEPA restricted guideline of 200 CFU/100 mL. Lastly, for *E. coli* that would be subject to Western Australian standards from Table 1, two of the three trials from Winward et al. (2008c), using the higher organic water and conducted at the lower dosages discussed above, did not achieve full inactivation. Effluent concentrations were however still less than the standard of 10 CFU/100 mL for surface irrigation. Three additional data points for *E. coli* are given in Figure 5b from Ekeren et al. (2016) showing LR_s of >5.5–5.6, however effluent concentration data were not given.

Figure 5b shows UV LR results for bacterial pathogens and MS2 coliphage. For *S. enterica* and *S. aureus*, all trials reviewed resulted in full inactivation. For *P. aeruginosa*, full inactivation was achieved for dosages above 100 mJ/cm², with mixed results at lesser dosages. Ekeren et al. (2016), using water with higher organics (discussed above), were able to achieve an LR of >7.1 (seeded) at 26 mJ/cm², while Friedler et al. (2011), using a lower organics water (discussed above), required at least 39 mJ/cm² to achieve the full inactivation LR of 2.1 (non-seeded). Complicating the unpredictable removals at lower dosages is the fact that UV disinfection, unlike chemical disinfection, does not entail a treatment residual. This becomes important when experimental setups do not measure disinfection effectiveness immediately following the UV reactor (e.g. using a sampling port in the reactor itself). Illustration of this was seen in the study by Friedler and Gilboa (2010) who sampled along a greywater treatment train and who measured UV disinfection effectiveness in a post-reactor flow equalization basin. The authors reported no statistical difference between average concentrations of *P. aeruginosa* from UV disinfected (69 mJ/cm²) and undisinfected effluent. Although a considerable fraction of samples taken from the UV disinfection trials were below the detection limit for *P. aeruginosa* (not so for undisinfected samples) indicating at least partial UV effectiveness, regrowth in the flow equalization basin of this opportunistic pathogen that is prone to biofilm formation masked these effects.

For MS2 coliphage, only two trials resulted in full inactivation, including one trial from Beck et al. (2013) at a dosage of 100 mJ/cm² and the single trial from the previously described treatment train of Friedler and Gilboa (2010). For the other trials conducted by Beck et al. (2013), lack of full inactivation was attributed to tailing despite low organic levels (turbidity 1.4–6 NTU). Even still, LR_s of 3.9–5.9 were reported at a dosage of 100 mJ/cm². Given that MS2 is generally more resistant to UV disinfection than poliovirus (Meng and Gerba, 1996; Shin et al., 2005), the LR_s achieved at 100 mJ/cm² would likely meet the California criteria for treatment. Additional testing of actual system (i.e. RVF+UV) would be warranted however as recent research on the mechanics of virus disinfection has shown how dynamic this process can be (Gerba and Betancourt, 2017; Gerba et al., 2017). Looking at results from lower dosages, trials conducted at 26 and 30 mJ/cm² by Ekeren et al. (2016) and Beck et al. (2013) support the need for higher dosages for viruses, as reported MS2 LR_s were 2.7 and 1.2–1.8, respectively.

Although similar disinfection performance can be achieved with both chlorine and UV, taking into account the heightened DBP formation potential of wetland-derived organic

matter (Quanrud et al., 2004) as well as the fact that UV may be cheaper than equivalent chlorination systems (Beck et al., 2013), UV seems particularly suited as a primary disinfection option. However, noting the reality that post-disinfection storage is likely required as well as the regrowth potential of certain opportunistic bacteria (e.g. see discussion of *P. aeruginosa* above), residual disinfection using chlorine would be recommended.

Lastly, we call attention to an important limitation in the linking of GW wetland effluent to the disinfection trials reviewed above. Comparing the microbial effluent concentrations given in Table 3 (and more importantly the ranges in Table S1) to the influent concentrations given in Table 4 we see examples of wetland effluent concentrations exceeding disinfection influent concentrations for almost every overlapping parameter. This means that although we have reviewed disinfection dosages and ambient water qualities conducive for full inactivation in this section, we cannot extrapolate LRs beyond the influent concentrations reviewed. Thus, while the experimental results we have reviewed suggest that UV disinfection + chlorine residual would be appropriate for treatment of wetland effluent with low organics, stress testing of actual systems (likely requiring seeded influent) is recommended.

6 Conclusions

The above review has shown that constructed wetlands, combined with appropriate disinfection unit processes, can likely treat GW and provide effluent appropriate for certain nonpotable reuse applications. If designed with a HRT of 3–5 days, single-pass constructed wetlands can generally meet restricted reuse chemical/physical standards. Also, with respect to BOD and TSS, GW wetlands tend to perform as well as, if not better than, would be expected using traditional design equations and rate constants based on treatment of agricultural or domestic wastewaters. If recirculation is added, as in the case of the RVF systems, effluent can be reliably produced in which BOD, TSS and turbidity are below 5 mg/L, 3 mg/L, and 10 NTU, respectively, which meets USEPA guidelines for restricted and environmental reuse and all Western Australian physical/chemical standards. Additionally, wetlands can generally provide 1–2 LR of bacteria, protozoa and viruses, though LRs are difficult to predict and effluent concentrations are best predicted by influent concentrations. Accordingly, if used as a standalone unit process, GW wetlands cannot reliably meet microbiological effluent standards.

Results from the reviewed GW disinfection experiments suggest that if organics are sufficiently removed from GW, a chlorine dosage of 100 mg/L-min or UV dosage of 100 mJ/cm² is likely appropriate for meeting all USEPA guidelines and all Western Australia guidelines, whereas higher dosages are likely required to meet California Title 22 criteria. In terms of pathogen LRs, chlorine dosages of 200–400 mg/L-min and UV dosages of 100–300 appear necessary to provide full inactivation of the influent concentrations reviewed. Full LR ranges of the reviewed disinfection experiments were largely influent limited however and stress-testing of actual systems is likely needed to ensure reliable treatment of high pathogen concentrations, i.e. the upper limits of wetland effluents reviewed.

Supplementary Material

Refer to Web version on PubMed Central for supplementary material.

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Figure 1.

Effluent concentration data as a function of hydraulic retention time (HRT), including any applicable reuse criteria, for a) biochemical oxygen demand (BOD), b) total suspended solids (TSS) and c) Turbidity. System types include free water surface (FWS), green roof water recycle system (GRW), horizontal subsurface flow (HSSF), recycled vertical flow (RVF), vertical flow (VF) and aerobic plus anaerobic (Aer. + Anaer.). For BOD and TSS, USEPA guidelines for unrestricted reuse are 10 mg/L, for restricted reuse are 30 mg/L. Turbidity guidelines for unrestricted reuse are 2 NTU.

Figure 2.

Comparison of GW wetland a) biochemical oxygen demand (BOD) and b) total suspended solids (TSS) performance with North American Treatment Wetland Database (NADB) datasets. System types include free water surface (FWS), green roof water recycle system (GRW), horizontal subsurface flow (HSSF), recycled vertical flow (RVF), vertical flow (VF) and aerobic plus anaerobic (Aer. + Anaer.). For BOD and TSS, USEPA guidelines for unrestricted reuse are 10 mg/L, for restricted reuse are 30 mg/L.

Figure 3. Log(effluent concentration) of a) total coliform (TC), b) fecal coliform (FC) and c) E. coli as a function of log(influent concentration). *Regression of all system types.

Figure 4. Log reductions from GW chlorination studies. Shading indicates influent limited LR and should be interpreted as "greater than".

Figure 5. Log reductions from GW chlorination studies. Shading indicates influent limited LR and should be interpreted as "greater than".

Table 1.

Physical, chemical and microbiological water quality guidelines (USEPA) and criteria (CA Title 22, Western Australia) for different end-uses of recycled water

Parameter	units	Unrestricted. ^d	US EPA ^a Restricted ^e	Environmental ^f	California Title 22 ^b Unrestricted ^d	Restricted ^e	Subsurface Irrigation	Surface Irrigation	Toilet Flushing
<i>Water Quality Parameters</i>									
pH		6.0–9.0	6.0–9.0						
TSS	mg/L		30	30			<30	<30	<10
BOD	mg/L	10 ^g	30 ^g	30 ^g			<20	<20	<10
Turbidity	NTU	2 ^h			2 ^o				
<i>Pathogen Criteria</i>									
Total Coliform	MPN/100mL				2.2 ^p	23 ^r			
Fecal Coliform	CFU/100mL	ND ^{i,j,k}	200 ^{j,m,n}	200 ^{j,m,n}					
E. Coli	CFU/100mL							<10	<1
Poliovirus or Surrogate					5 log inact. ^{l,q}				
<i>Disinfection Parameters</i>									
UV Disinfection	mJ/cm ²				100				
Chlorine Residual	mg/L	1 ^s	1 ^s	1 ^s					
Chlorine CT ^k	mg/L - min	90 ^s			450 ^{l,q}				

^a (USEPA, 2012).

^b ((CDPH), 2010).

^c ((GWA), 2010).

^d The use of reclaimed water in nonpotable applications in municipal settings where public access is not restricted.

^e The use of reclaimed water in nonpotable applications in municipal settings where public access is controlled or restricted by physical or institutional barriers, such as fencing, advisory signage, or temporal access restriction.

^f The use of reclaimed water to create, enhance, sustain or augment water bodies, including wetlands, aquatic habitats, or stream flow.

^g As determined from the 5-day BOD test.

h The recommended turbidity should be met prior to disinfection. The average turbidity should be based on a 24-hour time period. The turbidity should not exceed 5 NTU at any time. If SS is used in lieu of turbidity, the average SS should not exceed 5 mg/l. If membranes are used as the filtration process, the turbidity should not exceed 0.2 NTU and the average SS should not exceed 0.5 mg/l.

i ND = non detectable.

j Unless otherwise noted, recommended coliform limits are median values determined from the bacteriological results of the last 7 days for which analyses have been completed. Either the membrane filter or fermentation tube technique may be used.

k No fecal coliform detectable (ND). The number of total or fecal coliform organisms (whichever one is recommended for monitoring in the table) should not exceed 14/100 ml in any sample.

l After a minimum contact time of 90 minutes.

m Some stabilization pond systems may be able to meet this coliform limit without disinfection.

n The number of fecal coliform organisms should not exceed 800/100 ml in any sample.

o 10 NTU maximum, standards for media filter, membrane filter standards more stringent.

*p*₇ day median, not more than one sample to exceed 23 in 30 days, 240 maximum.

q Title 22 requires total coliform limit and either proof of viral removal or chlorine dose as given in table.

*r*₇ day median, not more than one sample to exceed 200 in 30 days.

s This recommendation applies only when chlorine is used as the primary disinfectant. The total chlorine residual should be met after a minimum actual modal contact time of at least 90 minutes unless a lesser contact time has been demonstrated to provide indicator organism and pathogen reduction equivalent to those suggested in these guidelines. In no case should the actual contact time be less than 30 minutes.

Table 2.

Physical, chemical and microbiological water quality of typical wastewater streams and reviewed treatment systems.

Parameter	Units	Light Greywater ^{a,b}	Mixed Greywater ^{a,b}	Mixed Wastewater ^d	Wetland Influent Ranges	Wetland Effluent Ranges	Disinfection Influent Ranges
<i>Physical and Chemical</i>							
TSS	mg/L	29–505	19–700	22–1690	4.9–120	0.3–52	4.0–32
BOD	mg/L	20–240	48–1056	112–1101	20–435	1.0–196	1.1–62
COD	mg/L	100–633	58–2950	1329–1650	77–646	6.0–363	17–130
Turbidity	NTU	12.6–240	19–444		15–254	0.3–114	0.2–35
Total Nitrogen	mg/L	3.6–19.4	1.1–74	9.0–240	5.2–15	1.7–14	2.8–4.1
Total Phosphorus	mg/L	0.11–48.8	0.062–500	0.2–32	0.8–9.3	0.47–5.2	
<i>Bacteria and Bacterial Indicators^c</i>							
Total Coliform	CFU/100mL	1–7.4	3.1–8.8	7.0–9.0	5.4–8.7	0.7–8.3	2.0–5.8
Fecal Coliform	CFU/100mL	0–6.9	2.0–8.0	4.0–8.2	3.7–8.0	0.0–6.0	1.4–5.1
<i>Escherichia coli</i>	CFU/100mL	2.3–5.7	3.6–6.7	4.0–7.9	2.8–6.7	0.0–6.4	2.6
Enterococci	CFU/100mL	1.9–3.4	2.4–4.6	4.0–5.0	2.8–3.8	0.5–3.3	1.8–3.8
<i>Pseudomonas aeruginosa</i>	CFU/100mL	2.6–3.5	2.3–4.3	3.0–6.0	3.7–6.8	0.2–6.0	2.1–3.8
<i>Staphylococcus aureus</i>	CFU/100mL	4.0–5.7	3.3–5.7		3.5	1.8	1.4–1.9
<i>Clostridium perfringens</i>	CFU/100mL	0.66		3.0–5.0	2.8–3.1	0.8–2.6	
<i>Legionella</i>	CFU/100mL	2.2	2.2–2.9		5.1	3.8–4.4	
<i>Salmonella</i>	CFU/100mL		3.7	2.0–4.0	0.7–1.3		
<i>Viral Indicators^c</i>							
MS2-Coliphage	PFU/mL			3.0–4.0			5.6–8.2

^aLight greywater includes bathroom sink and washbasin streams; mixed greywater includes bathroom sink, bathroom washbasin, kitchen sink and laundry streams.^b(Boyjoo et al., 2013; Friedler et al., 2011; Ghatidak and Yadav, 2013; Li et al., 2009).^clog10concentration.^d(Lowe et al., 2007; Metcalf, 2003), noting that values will vary with portion of population shedding at any given time.

Table 3.

Summary of GW constructed wetland treatment performance by system type. For results and attributes of each individual system, see Table S1.^a

Parameter	unit	FWS			GRW			HSSF			RVF			VF			Aer. + Anaer.								
		n	In	Out	Rem.	n	In	Out	Rem.	n	In	Out	Rem.	n	In	Out	Rem.	n	In	Out	Rem.				
Chemical and Physical																									
BOD	mg/l	1	84	32	63%	2	92	41	71%	17	196	25	87%	6	198	2.2	98%	4	99	10	85%	5	149	12.8	87%
TSS	mg/l	1	17	12	25%	2	61	12	84%	5	52	21	64%	4	66	2	98%	4	58	17	71%	5	52	14	64%
Turbidity	NTU	1	31	15	51%	2	44	15	77%	4	89	38	47%	5	65	3.9	97%	3	67	12	77%	5	81	8.8	88%
TN	mg/l	1	14	5.6	59%	0	--	--	--	3	7.2	4	44%	0	--	--	--	2	4.9	2.6	46%	4	13	7.9	46%
TP	mg/l	1	4.0	1.7	58%	0	--	--	--	3	2.7	2.3	24%	1	1.9	0.5	74%	1	5.2	2.3	55%	2	4.8	1.8	63%
Microbial ^b																									
Total Coliform	log ₁₀	0	--	--	--	2	6.4	3.8	2.6	3	7.1	5.1	2.1	0	--	--	--	3	7	4.2	2.8	2	8.2	6.1	2
Fecal Coliform	log ₁₀	0	--	--	--	0	--	--	--	14	7.2	4.4	2.8	2	5.7	3.9	1.9	0	--	--	--	3	5.7	2.8	2.9
<i>E. coli</i>	log ₁₀	1	4.0	2.9	1.1	2	3.3	0.85	2.4	4	4.0	2.6	1.4	1	4.7	0.1	4.6	3	4.3	2.1	2	2	3.5	3.8	1.5
<i>Enterococci</i>	log ₁₀	0	--	--	--	2	2.8	0.8	2	2	2.8	1.1	1.7	1	3.8	3.3	0.6	2	2.8	0.5	2.3	0	--	--	--
<i>Clostridia</i>	log ₁₀	0	--	--	--	2	3	2.3	0.7	2	3	1.7	1.3	0	--	--	--	2	3	1	2	0	--	--	--
<i>S. aureus</i>	log ₁₀	0	--	--	--	0	--	--	--	0	--	--	--	1	3.5	1.8	1.7	0	--	--	--	0	--	--	--
<i>P. aeruginosa</i>	log ₁₀	0	--	--	--	2	5.6	3.3	2.4	2	5.6	2.6	3	2	4	4.1	-0.1	2	5.6	1.6	4	0	--	--	--
<i>Legionella</i>	log ₁₀	0	--	--	--	0	--	--	--	0	--	--	--	1	5.1	4.4	0.7	0	--	--	--	0	--	--	--

^aFWS = free water surface wetland; GRW = green roof water recycle system; HSSF = horizontal subsurface flow wetland; PET = polyethylene; RVF = recycled vertical flow bioreactor; VF = vertical flow wetland; SF = sand filter; UV = ultraviolet radiation; Cl = chlorination; HRT = hydraulic retention time, or volumetric capacity divided by daily inflow volume; TSS = total suspended solids; BOD = biological oxygen demand; TN = total nitrogen; TP = total phosphorus.

^bvalues represent log₁₀ transformations of geometric means

Table 4.

GW disinfection technology characteristics and treatment performance

Parameter	unit	Chlorination						Ozone			UV		
		n	In	Out	Rem.	n	In	Out	Rem.	n	In	Out	Rem.
Chemical and Physical													
BOD	mg/l	9	15	--	--	--	--	--	--	14	7.3	--	--
TSS	mg/l	7	21	--	--	--	--	--	--	7	12	--	--
Turbidity	NTU	36	11	--	--	18	2.9	--	--	38	9.5	--	--
TN	mg/l	19	88	--	--	18	88	--	--	28	69	--	--
TP	mg/l	30	19	--	--	18	4.3	--	--	24	18	--	--
Microbial ^{b,c}													
Total Coliform	log ₁₀	10	3.8	2.4 (7)	3.1	12	2.9	1.7 (3)	1.6	14	3.4	1.4 (10)	3.0
Fecal Coliform	log ₁₀	6	3.4	0.3 (4)	3.3	--	--	--	--	8	2.9	1.5 (4)	2.2
<i>E. coli</i>	log ₁₀	3	6.3	0 (3)	6.3	--	--	--	--	3 ^d	2.6	0.2 (1)	2.5
<i>Enterococci</i>	log ₁₀	10	2.1	0 (10)	2.1	4	2.1	0 (4)	2.1	7	2.8	0.6 (5)	2.7
<i>S. enterica</i>	log ₁₀	3	7.8	0 (3)	7.8	--	--	--	--	3	7.9	0 (3)	7.9
<i>S. aureus</i>	log ₁₀	2	1.9	0 (2)	1.9	--	--	--	--	7	1.8	0 (7)	1.8
<i>P. aeruginosa</i>	log ₁₀	3	4.2	1.5 (1)	3.2	--	--	--	--	8	2.9	2.5 (6)	2.3
MS2 Coliphage	log ₁₀	9 ^d	5.9	0 (9)	5.9	10	5.9	2.8 (1)	3.6	13 ^d	5.8	3.1 (2)	3.1

^aTSS = total suspended solids; BOD = biological oxygen demand; TN = total nitrogen; TP = total phosphorus.^bvalues represent log₁₀ transformations of geometric means.^cValues in parentheses indicate number of trials resulting in full inactivation.^dTable S2 has additional trials, however only log reductions given.