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The status of potable water reuse implementation

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ABSTRACT

A review of the current status of direct and indirect potable water reuse (DPR/IPR) implementation has been conducted, focusing on the regulatory and practical aspects and with reference to the most recent published literature. The review encompasses (a) the principal contaminant types, their required removal and the methods by which their concentration is monitored, (b) regulatory approaches and stipulations in assessing/ratifying treatment schemes and maintaining treated water quality, and (c) existing full-scale installations. Analytical methods discussed include established in-line monitoring tools, such as turbidity measurement, to more recent polymerase chain reaction (PCR)-based assay methods for microbial detection. The key risk assessment tools of quantitative microbial risk assessment (QMRA) and water safety plans (WSPs) are considered in relation to their use in selecting/ratifying treatment schemes, and the components of the treatment schemes from 40 existing IPR/DPR installations summarised. Five specific schemes are considered in more detail.

The review reveals:

1 over half of the schemes identified employ reverse osmosis (RO) followed by UV disinfection, with UV-based advanced oxidation used in many modern schemes as the final step;
2 Whilst quantitative PCR appears to offer many advantages for microbial detection, due to its sensitivity and specificity, it nonetheless demands pre-concentration of the sample and is subject to interference leading to possible false positives;
3 QMRA studies suggest that the risk imposed by DPR and, in particular, IPR is very small compared with de facto reuse, the latter being subject to far less regulatory scrutiny;
4 There appears to be no evidence of acute conditions, and diarrhoeal disease specifically, from the few epidemiological studies which have been conducted; and.
5 IPR implementation becomes challenging if unbounded environmental waters are used as a buffer, since “zero deterioration” in environmental quality must then be demonstrated.

Whilst there are a number of ongoing projects where RO is not used because of the challenge imposed by disposal of RO concentrate, the prevalence of the sequential RO-UV combination implies the importance of quantifying the impact of process upsets on these unit operations.

1. Introduction

Indirect potable reuse (IPR) is the recovery of wastewater for potable use via an intervening environmental buffer, as opposed to direct potable reuse (DPR) where there is no such buffer or only limited dilution or storage time provided by such buffers. The environmental buffer, which may be a lake, river, or a groundwater aquifer, is considered to provide additional protection through dilution or removal by filtration (for aquifers), photolysis (for surface waters), or biological degradation (USEPA, 2017). Whilst this performance can be attained by additional unit process technologies for DPR, such supplementation does not address the loss in response time in the event of a process upset (such as membrane breaching or UV lamp failure). Such risks in implementing DPR must be addressed through advanced automation and real-time process performance monitoring.

A key aspect of implementation is regulation, and in this regard there are two identifiable elements:

- allocation of an explicit and quantitative contaminant removal to an individual unit operation within the water reuse facility, and
- quantification of risk associated with installation and operation of the water reuse facility.

The first of these provides minimum performance levels (or “credits”) attainable by individual process technologies for each key pollutant, and pathogens specifically along with a minimum number of

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sequential treatment steps which then provide multiple barriers to individual contaminants (AGWR, 2006, 2008; Olivieri et al., 2016; USEPA, 2017; Soller et al., 2018). The second element comprises a holistic health-based risk assessment and management for maximising the safety of drinking water delivery from source to tap (Davison et al., 2006; WHO, 2016, 2017). Both approaches demand input data pertaining to:

a) the feed and required output water quality,
b) process failure risk and the associated preventative/remedial measures, and
c) impacts of contamination.

It is of interest to appraise the current status of wastewater reclamation for potable water use from both a regulatory and practical perspective to elucidate the most productive forward path. The review is focused predominantly on the most recent publications (2016 onwards) pertaining to:

- contaminants of interest, their removal and their monitoring in the treated water,
- regulatory approaches to assessing and ratifying candidate treatment schemes and maintaining treated water quality, and
- existing full-scale installations.

2. Water reuse: key facets

Critically important parameters in reclaiming wastewater for potable supply comprise (Fig. 1):

a) the legislated maximum contaminant concentration (MCL) of chemical pollutants in the treated water (WHO, 2004, 2016, 2017), as well as the process performance in terms of pathogen removal expressed as the log removal value (LRV) or "credit" (WERF, 2016, SWRCB, 2018, CWB, 2021);
b) the minimum contaminant concentration measurable (MRL, the method reporting limit) (TWDB, 2015; WHO, 2017);
c) the frequency of events (process failure and mitigating factors, including environmental) leading to degradation of treated water quality or some other onerous outcome (such as pollution from residuals); and
d) the impact of treated water quality degradation, or possible pollutant release, associated with the above events.

2.1. Contaminants and MCL

The required pathogen removal, expressed as the LRV, is determined by:

i impact on life duration adjusted for disability (hence the disability-adjusted life years, DALYs), or
ii the infection rate limit (IRL)

A limit of 1 year reduction in projected life in 10^6 DALYs is stipulated by the World Health Organization, corresponding to a risk of 1 excess case per 100,000 people from lifetime exposure (WHO, 2004, 2016); an IRL of 1 per 10,000 people per year (10^{-4} pppy) has been originally proposed (Staatsblad, 2001; Smeets et al., 2009). Both approaches are constrained in their applicability to some extent: the DALY does not account for asymptomatic infection, and the IRL does not account for the infection severity associated with the different pathogens.

There are broadly two types of contaminants of concern in drinking water derived substantially from a municipal wastewater source (a) pathogens, and (b) dissolved organic and inorganic matter. Of the latter, concern has been focused on the so-called contaminants of emerging concern (CECs), a general term for organic and inorganic species present at low concentrations (also called micropollutants) but which may nonetheless impose a significant chronic health risk (Schwarzenbach et al., 2006). However, notwithstanding the concern of key contaminants such as PFAS (perfluoroalkyl/polyfluoroalkyl substances), DBPs (disinfection byproducts), and occasional industrial pollutants, the vast majority of CECs do not present any known human health risks at the levels found in WWTP effluent, and even less so following purification (Trussell and Trussell, 2015).

2.1.1. Pathogens

Historically, the key contaminants of concern in water reuse have been the pathogenic micro-organisms, or species indicating their presence (i.e. indicators). Indicator organisms most-commonly monitored comprise the thermotolerant faecal coliforms (TC) and E Coli specifically, the most prevalent TC in faecal matter and therefore considered

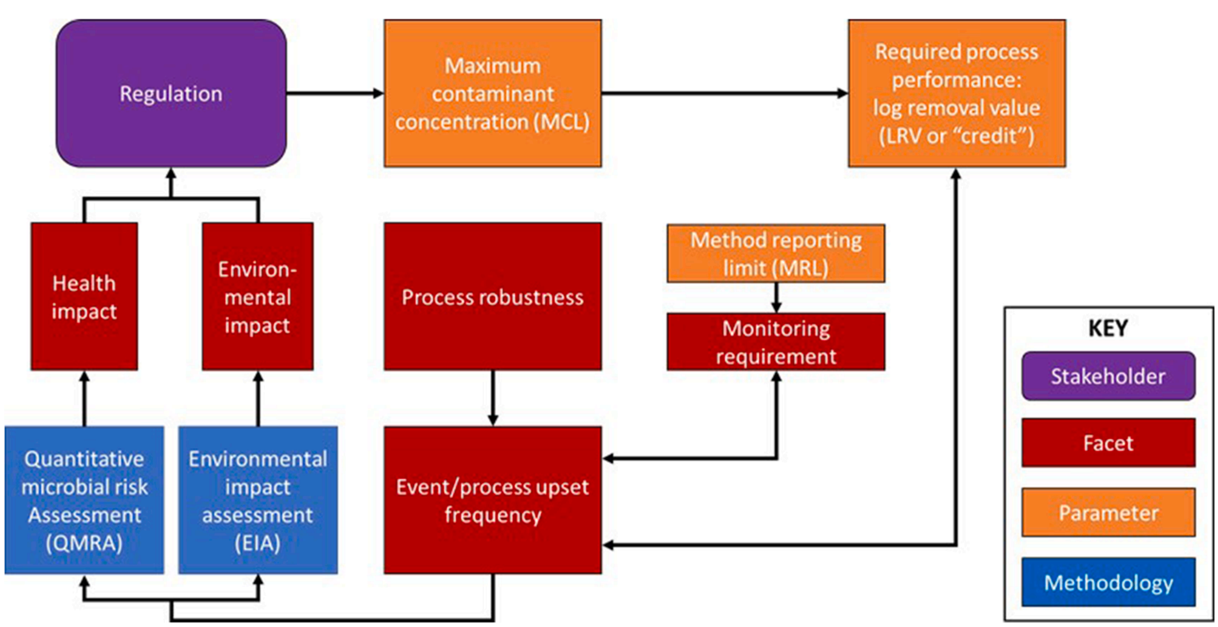

Fig. 1. Water reuse for potable supply: key facets and inter-relationships.

to provide a reasonable indication of faecal contamination (Holcomb and Stewart, 2020). Guideline MCL values and recommended measurement frequencies are in place for these pathogens for wastewater reuse for irrigation in many countries (Shoushtarian and Negahban-Azar, 2020).

For potable water reuse duties the pathogens of interest are enteroviruses and the protozoa Giardia and Cryptosporidium, including oocysts. Enteroviruses are relatively small (<0.05 μm) compared with E Coli (1–2 μm) or the protozoa (>4 μm), but are (a) largely associated with suspended solids (SS) – particularly in high-SS environments such as mixed liquors (Haun et al., 2014), and (b) less tolerant to chemical disinfection than the protozoa (Bitton, 2014) and cryptosporidium oocysts specifically. In potable reuse MCL values and system performance are often focused on enterovirus and protozoa removals and residual levels. This then takes account of removal on the basis of size exclusion (such as for permselective membranes) and physicochemical mechanisms (ultraviolet and chemical disinfection).

MCL values derive from toxicological and eco-toxicological evidence, quantified as the DALY. The resulting computed permitted pollutant concentrations are subsequently below the limit at which they can be quantitatively measured, as demonstrated in a number of studies of full-scale membrane-based systems (Ferrer et al., 2015; Purnell et al., 2016; Katz et al., 2017). This then means that the practically-measured LRV is governed by the ratio of the feed concentration to the limit of detection of the method used, and does not necessarily relate to the unit process, system performance or infection risk. The few early epidemiological studies conducted on water reuse (Nellor et al., 1985; Sloss et al., 1996; Sinclair et al., 2010) have indicated no impact from the supply of reused water to the local community, though some of these studies have been viewed as being flawed (Nappier et al., 2018) suffering from study design deficiencies and insufficient sensitivity to detect the relevant low-level adverse health effects.

The risk of pathogen breakthrough for specific unit processes can be determined by quantitative microbial risk assessment (QMRA) (Zhiteneva et al., 2020; Owens et al., 2020). This approach then relies on extensive data sets for both the source water pathogen concentrations and the performance of the unit process, and in particular the impact of the risk of process failure on downstream pathogen levels. Evaluation of the of the nature, frequency and impact of process failures (or “hazard events”) has been conducted originally in Australia (AGWR, 2006) and subsequently in the US (Salveson, 2018).

Although guidelines have been provided for its methodology (WHO, 2016), applying QMRA in practice is challenged by a lack of consensus with reference to (Zhiteneva et al., 2020; Owens et al., 2020):

1 the most representative values to be used for pathogen concentrations and their (range of) removals by specific unit technologies;
2 the validity of the use of point LRVs over probability distribution functions (PDFs), and subsequently the most apposite PDF for specific pathogens, applications or scenarios;
3 assumptions relating to pathogen-surrogate and pathogen-indicator ratios;
4 the basis for selecting the appropriate dose-response model; and
5 the specific risk parameter(s) used to represent the health burden threshold (i.e. average vs. median vs. specific percentiles, with or without associated confidence interval)

The ambient level of contaminants in the influent, required to determine the required LRV, is often unknown and has to be assumed or inferred from other data which may be site-specific or literature-based. Point value estimates of concentrations have in the past often been assumed in QMRA calculations, but these have been shown to underestimate the actual risk (Schmidt et al., 2020). Probability distribution functions, PDFs (based on probabilistic/stochastic methods using techniques such as Monte Carlo or Latin Hypercube sampling) more accurately reflect the range of risk incurred, but there is little consensus regarding the selection of the most appropriate PDF or statistical method. It was further noted (Owens et al., 2020) that the variability in the complexity of design was not necessarily commensurate with the requirement to assess the actual safety of the treated water as a drinking water supply.

Notwithstanding the issues surrounding the assumptions and precise methodology applied to QMRA, two recent analysis of representative IPR and DPR treatment schemes (Amoueyan et al., 2019; Soller et al., 2019) both concluded the risk values associated with such schemes to lie substantially below the IRL and/or DALY threshold values. In both studies, the risk associated with appropriately designed IPR and DPR schemes was significantly less than that for de facto (i.e. unintentioned) reuse. Outcomes were found to be highly dependent on either:

a) the residence time of the treated water prior to reuse, the IRL decreasing from 10^{-4} to 10^{-9} pppy on increasing the retention time from 90 to 100 days for IPR (Soller et al., 2019);
b) employing high-dose instead of low-dose UV (800 vs 12 mJ/cm^2) for DPR (Soller et al., 2019);
c) wastewater temperature, inactivation of viruses increasing significantly with temperature (Amoueyan et al., 2019);
d) wastewater pathogen concentration, treated water pathogen levels increasing with increasing feed concentrations (Amoueyan et al., 2019) - as may arise during outbreaks.

2.1.2. CECs

The environmental and health hazards imposed by CECs has long been recognised (Schwarzenbach et al., 2006), and the accumulation of the more persistent of these (and in particular the highly biorefractory perfluoroalkyl substances or PFAS, Podder et al., 2021) in crops irrigated with recovered wastewater increasingly reported (Mansilla et al., 2021; Ben Mordechay, 2021). Unlike pathogens, most CECs are not associated with faecal contamination. The MCL levels ascribed to non-reclaimed potable water sources can thus be considered equally germane to water recovered from municipal wastewater for IPR or DPR. The most recent proposed drinking water legislation (EU, 2020) encompasses some these compounds, specifically bisphenol A with beta-oestradiol and nonylphenol likely to follow.

Evidence suggests that there is limited removal at full-scale potable water treatment works for many endocrine-disrupting CECs (Valbonesi et al., 2021), including PFAS (Boone et al., 2019). However, a recent report of the possible chronic impacts of CECs assessed through in vitro bioassay indicated the measurable bioactivity – specifically the glucocorticoid, hydrocarbon and oestrogen receptor activity – in secondary effluent sampled across six US full-scale reuse installations (5 IPR + 1 DPR, all employing RO) to disappear in the final potable reuse water (Schimmoller et al., 2020). Moreover, actual total measured CEC levels in the treated reuse water were comparable to those measured in conventionally-sourced drinking water. Most studies have shown RO-based IPR and DPR systems to provide robust removal of both long and short-chain PFAS compounds, whilst non-RO systems, such as Ozone/BAC/GAC robustly remove long chain PFAS, down to below the limits pertaining to health standards.

2.2. Monitoring

The challenges to a consistent and representative QMRA underline the importance of real-time monitoring, which encompasses other key contaminants such as the CECs. The classical method for determining pathogen concentration is the laboratory-based method, which is low in cost but labour-intensive and demands an extended time period (~16 h). The more recent molecular methods target the microorganism gene, rather than the whole cell. Molecular methods can be more sensitive than culture-based ones (which include flow cytometry and other cell-level detection methods). However, no method can be considered sufficiently sensitive to directly and reliably detect pathogens at the limiting concentrations required on a continuous basis: sample enrichment is thus always required. This being the case, surrogate parameters have been identified for monitoring system performance on-line.

2.2.1. Physical and cell monitoring methods

A widely implemented monitoring method, and the one around which existing legislation for process performance monitoring is based, is turbidimetric (or nephelometric) measurement. Turbidity measurement is well-developed, well-understood, and robust. It has been shown to give a reasonable reflection of membrane integrity and the associated risk of pathogen breakthrough in some full-scale systems (Katz et al., 2017, Prado et al., 2019), though its sensitivity and reliability across different waters appears to be limited (Krahnstöver et al., 2019), particularly when air bubbles are present (Xin et al., 2019).

Laser extinction (Xin et al., 2019) has been reported as being able to detect and characterise both suspended microbubbles and particles, possibly offering some promise for on-line particle or microbe detection in permeate from air-scoured membrane processes such as MBRs. However, it is unclear as to whether this technique is sufficiently sensitive and robust for full-scale application. Particle counting technology, and nanoparticle counting in particular (Krahnstöver et al., 2019), has also been shown to infer microorganism cells downstream of membrane permeation (Fujioka et al., 2019; Lousada-Ferreira et al., 2016).

Flow cytometry, which employs a laser beam to characterise suspended particles according to their light scattering and fluorescence characteristics, is most often used for cell counting (Safford and Biscel, 2019). The captured scatter and fluorescence data infer cell characteristics such as relative size, complexity, and nucleic-acid content, and thus provide a unique cytometric "fingerprint" of the microbial community present in a water sample. Commercial instruments are available, and real-time monitoring is possible (Li et al., 2020). Although high in capital cost, analysis can be relatively low in operating cost if multiple samples are analysed (Safford and Biscel, 2019). It is nonetheless limited in sensitivity by interference from other non-cell fluorescing particulate or chemical species, as is the case with other fluorescence-based methods (Korshin et al., 2018; Sherchan et al., 2018; Sohrabi et al., 2021).

2.2.2. Biological, biochemical, and molecular monitoring methods

Identification and quantification of coliforms is most often through the laboratory-based defined substrate technology (DST) method, where a proprietary reagent system is used to enumerate specific target microbes from a mixture of bacteria. However, the drive towards greater speed, sensitivity and automation has led to the development of molecular methods which target highly specific genomic segments of the pathogen genetic material (Li et al., 2020). Of these, quantitative polymerase chain reaction (qPCR) assay has been the most widely tested and implemented, particularly for viruses (Haramoto et al., 2018; Farkas et al., 2020). It is sensitive, specific, quantitative in assaying for target microorganisms, and commercially available as a portable device. However, it is also relatively costly, complicated and time-consuming in relying on repeated thermal cycles over an extended time period (~5 h, Sohrabi et al., 2021) to complete the assay. qPCR is also often subject to interference and inhibited by organic substances, such as polyphenolic compounds, found in environmental samples (Farkas et al., 2020).

A recent development is loop-mediated isothermal amplification (LAMP), which amplifies the gene from the extracted nucleic acid material through a biochemical reaction. It is an isothermal analysis, and therefore less costly and more rapid than PCR, and is also usually more sensitive. It's combination with microfluidics allows it to be configured as a portable device for accessible for field application in environmental water samples (Li et al., 2020). However, it is not currently quantitative and, as with PCR, has not yet been configured for continuous monitoring and cannot provide the required sensitivity required without pre-concentration of the sample – specifically for the protozoa giardia and cryptosporidia.

A wide variety of portable biosensors have been developed to at least the proof-of-concept stage for a range of different target pathogens (Sohrabi et al., 2021). Biosensors can be either cell or molecular based. In general, for the most-commonly studied pathogen of E Coli the reported limit of detection is generally around 10 per 100 mL.

2.2.3. Chemical monitoring

There has been extensive study of the use of UV254 and fluorescence spectroscopy for monitoring the removal of organic matter generally and organic CECs (often referred to as trace organic chemicals, or TrOCs) specifically (Korshin et al., 2018, Sohrabi et al., 20,201, Song et al., 2021, Valbonesi et al., 2021). As with flow cell cytometry, fluorescence detection is subject to interference from naturally-occurring dissolved organic matters. Other examples of intrinsic water quality parameters used for monitoring RO integrity include conductivity, total organic carbon (TOC) and specific polyvalent species such as sulphate and calcium. However, these methods are generally insufficiently sensitive (Pype et al., 2016), with an LRV threshold of <3 (Hornstra et al., 2019), to detect minor breaches in membrane that nonetheless permit a minimal number of viruses to pass.

2.2.4. Monitoring: summary

Whilst there has been significant progress towards increasing the sensitivity and utility of water quality monitoring methods for application to water reuse, further progress is needed. Turbidity, the most widely monitored parameter in practice, and particle counting are both challenged by limited sensitivity, lack of specificity and false positives generated from suspended air (Xin et al., 2019). Flow cytometry appears to be the most advanced of the microbial cell-specific water quality monitoring tools but, as with all detection methods based wholly on physical/spectroscopic parameters, it is ultimately unable to differentiate between microbial cells and abiotic particles displaying the same spectroscopic properties – although fluorescence appears to be the property permitting the greatest degree of resolution (Korshin et al., 2018). Against this, established bulk parameters such as turbidity, as well as colour and UV254 (Lidén et al., 2016; Foschi et al., 2021), have been demonstrated as being capable of providing real-time monitoring of membrane integrity as an adjunct to off-line pressure decay test (Katz et al., 2017, Prado et al., 2019).

Molecular methods, including PCR and LAMP, would seem to offer the most promise as a sensitive and highly-specific tool for pathogen detection. qPCR-based methods have become the standard for off-line detection of viral genomes in concentrated water samples (Haramoto et al., 2018; Farkas et al., 2020; Gunnarsdottir et al., 2020). Currently, pre-concentration of samples is required to permit detection of many pathogens, with sample volumes as large as 1000 L being concentrated by up to 1500 times by filtration and then a further 100–150 times by specific adsorption (Gunnarsdottir et al., 2020), and the subsequent qPCR analysis then takes several hours to perform. LAMP is more rapid than qPCR, but currently not quantitative and is still subject to a pre-concentration step to capture the pathogens present at very low concentrations.

2.3. Process performance

Two key unit processes employed in most potable water reuse schemes are membrane separation and UV irradiation. UV advanced oxidation processes (AOP) achieve disinfection, direct photolysis, and advanced oxidation, whereas UV alone provides disinfection only. AOPs have been shown to degrade biorefractory trace organic chemicals - including some of the CECs (Kwon et al., 2020; Huang et al., 2020; Tan et al., 2020; Gao et al., 2020; Yeom et al., 2021).

In the case of disinfection efficacy specifically:

a UF can remove viruses, in the 99.99% range (Jacquet et al., 2021), but there is no current monitoring method to allow credits to be granted
b RO removes both protozoa and viruses, with various surrogates demonstrating up to 99.9% removal credit (Pype et al., 2016; Hornstra et al., 2019), and
c UV is a robust virus disinfectant, though adenovirus requires a higher dose per log reduction (Fig. 2); IPR and DPR systems employ UV dose values that far exceed that needed for 6-log reduction of all virus species, including adenovirus.
d Reuse treatment schemes achieve virus reduction of LRV 8–18 depending on the treatment scheme and operational parameters (Olivieri et al., 2016), but there are few reliable surrogates to demonstrate this reduction in real-time.

2.3.1. UV irradiation

A roughly linear response relationship exist between LRV and UV fluence (in mJ/cm^2) for inactivation of most pathogens, including viruses (Hijnen et al., 2006; Fig. 2). The transmission of UV irradiation (the UV transmittance, UVT) through water is nonetheless significantly impacted by colloidal and suspended solids (Carré et al., 2018). Since UV treatment is almost always employed downstream of membrane filtration in implemented IPR/DPR schemes, a possible loss of performance from a reduced UVT associated with solids or colloidal material is perhaps only practically germane to breaching of the membranes, though membrane breaching by particulates is not observed in most installations. Such breaching would also be expected to lead to increased permeate pathogen concentrations, which would then be accompanied by impaired salt, TOC and/or turbidity removal discernible at LRV values below 3 (Hornstra et al., 2019). Of greater practical relevance to UV process robustness is the risk of lamp and ballast failure, demanding duty/standby configuration.

More recently there has been increased research focus on light-emitting diode UV (LED-UV) technology for promoting energy efficiencies (Sholtes and Linden, 2019; Tan et al., 2020; Gao et al., 2020). In the case of LED lamps, failure has been associated with degradation of the optical power to significantly below the initial value, due to corresponding degradation of the semiconductor device or the silicon encapsulation (Arques-Orobon et al., 2020). Despite the progress in LED development, they are yet to be implemented for large-scale water disinfection.

The stipulated UV dose for achieving a target level of inactivation is generally based on the adenoviruses, since these are known to be the most resistant to UV light (Fig. 2). Other factors contributing to viruses

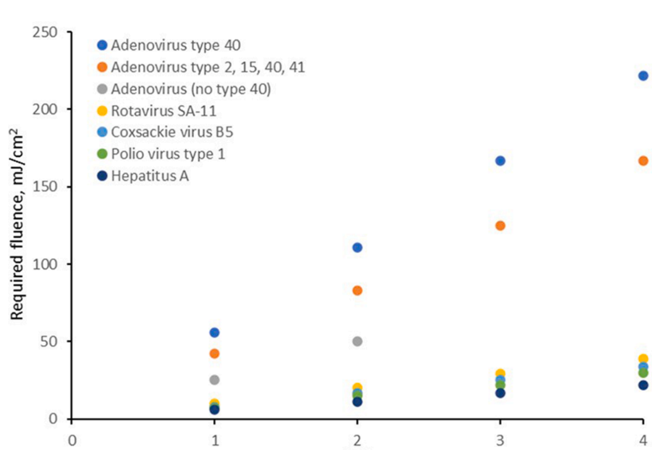

Fig. 2. Required fluence (or dose in mJ/cm^2) for virus inactivation by UV (Hijnen et al., 2006).

UV disinfection efficacy include the virus concentration and the degree of aggregation, aggregates being more resistant to disinfection (Gerba et al., 2018), though aggregation is minimal at the extremely low microbial concentrations germane to IPR and DPR treated water.

There has also been increased interest in the use of UV combined with chlorine and/or chloramines (UV-Cl) as an alternative to more conventional advanced photolysis methods (Kwon et al., 2020; Gao et al., 2020; Tan et al., 2020; Yeom et al., 2021). Recent studies have demonstrated improved removal by UV-Cl, compared with the conventional advanced photolysis (UV-H_2O_2), of recalcitrant CECs such as NDMA and 1,4-dioxane (Kwon et al., 2020), and phenacetin and acetaminophen (Tan et al., 2020). However, it is suggested that the photolytic degradation products of some CECs may themselves have associated toxicity (Yeom et al., 2021), and cytotoxicity or oestrogenic activity specifically (Huang et al., 2020).

2.3.2. Membrane technology

Virus rejection by membranes is often determined using surrogates since procedures required for the analysis of human viruses are time-consuming and cumbersome. Of these the most well explored is the bacteriophage MS2 (Amarasiri et al., 2017), which has a similar size. Whilst it has been shown (Purnell et al., 2016) that phages and enteric viruses do not necessarily co-exist, with seasonal variations in their relative concentrations, phages have been and continue to be used for assessing membrane and other treatment processes. These have included RO (Pype et al., 2016; Hornstra et al., 2019), direct membrane filtration (Jacquet et al., 2021) and membrane bioreactors (MBRs) (Zhu et al., 2021), as well as other barrier processes such as soil aquifer treatment (Morrison et al., 2020).

Whilst membrane technology, including ultrafiltration (UF), membrane bioreactors (MBRs) and reverse osmosis (RO), generally offer a robust barrier to the larger pathogens (bacteria and protozoa), viruses are not completely rejected. Studies or reviews of the capability of UF (Jacquet et al., 2021, Bray et al., 2021), MBRs (O'Brien and Xagoraraki, 2020; Zhu et al., 2021) and RO (Hornstra et al., 2019, Fujioka and Boivin, 2020) for rejecting viruses or their analogues have identified challenges relating to performance variability across different pathogen species, membrane characteristics and process operation.

Reported LRVs for virus rejection by UF membranes have generally been between 3 and 4. Jacquet et al. (2016) reported maximum LRVs of 3 or more for adenovirus and MS2 at high feed concentrations ($>10^8$ cells/mL), decreasing dramatically (<1) at influent concentrations below 10^4 cells/L Bray et al. (2021). reported mean LRVs of ~4 for total viruses, the value being lowest directly after cleaning. LRVs of ~4 for MS2 rejection were also reported by Lee et al. (2019), who also observed a decrease in LRV to <1.5 for aged membranes.

Studies of RO membrane rejection of MS2 phage in controlled challenge tests have indicated increased rejection (LRVs of 6–7, as reviewed by Hornstra et al. (2019) compared to UF. However, demonstrating removal of naturally-arising pathogenic viruses is challenged by their low feedwater concentration.

In the case of MBRs, membrane pore size dependency is demonstrated for some viruses, including enterovirus and human adenovirus, but not others (Table 1, O'Brien and Xagoraraki, 2020). The range of reported LRV values is also significant, both for single and across different species. This may arise from challenges posed by sampling and analysis (Gerba et al., 2018), but is also likely to reflect the impact of partitioning of the viruses between the solid and aqueous phases in the mixed liquor. Viruses favour the solid phase over the aqueous phase by up 200 times (Haun et al., 2014). Since the mixed liquor solids are almost quantitatively rejected, virus removal by MBRs tends to be greater than that provided by direct membrane filtration in the absence of such ancillary solids, as acknowledged in the most recently promulgated regulatory guidelines on MBR process credits (CWB, 2021).

A most recent pragmatic analysis conducted on behalf of the Water Research Foundation (Salveson et al., 2021) identifies performance

Table 1
Reported viral and phage log rejection by immersed membrane bioreactor technology, full-scale installations (adapted from O'Brien and Xagoraraki, 2020).

d_p, µm	Loc	Cap, MLD	Detect.	Nor.	Nor.1	Nor.2	H. ad.	F-sp C	Som C	Sap.	Ent.	Rot.	H sp C	Tot
0.4	Fr	0.39	qPCR		0–5.3	0–5.5								
0.1	MI	32	qPCR				4.1–5.6							
0.4	It	1.94	Plaque ass					6	4					
0.45	Fr	0.27	qPCR	3.3–6.8						1.8–4.1				
0.1	MI	32	qPCR				3.4–4.5				2.9–4.6			
0.1	MI	32	qPCR				4.1–6.3				4.1–6.8			
0.4	OH	12.9	Plaque ass					4.6–6	2.7–4.0					
			qPCR	1.5–3.3			2.4–4.9				2.2–4.7			
0.4	It	1.94	Plaque ass					5.77	4.35					
0.04	CA	5.7	qPCR			4.6–5.7	3.9–5.5	5.4–7.1						
0.04	UK	0.57	Plaque ass					3.5	5.3				3.8	
-*	CA	106	Fl cytom											4.0
0.4	KSA	1.6	dPCR				3.7				1.7			
0.4	Fr	1.8	qPCR	3.0						3.0		2.0		
0.05	Br	164	qPCR		1.1	1.2								
	Pore size dependency?			**N**			**Y**	**N**	**N**		**Y**			

d_p pore size; Loc Location; Cap capacity in megalitres/day; Detection method (quantitative PCR, Plaque assessment, Flow cytometry); Norovirus, Human adenovirus, F-specific coliphage, Somatic coliphage, Sapovirus, Enterovirus, Rotavirus, Human-specific coliphage, Total viruses; *sidestream MBR configuration

levels for specific types or "tiers" of MBR technologies. "Tier 1" MBRs refers to those already implemented at full scale and with demonstrated performance data. These MBRs provide LRVs of 1.0 for virus and 2.5 for protozoa, applicable to any hollow fibre or flat sheet MBR with pore sizes up to 0.4 µm and MBR filtrate turbidity values maintained at or below 0.2 NTU 95% of the time and never exceeding 0.5 NTU.

Almost all use schemes employ some sort of membrane separation technology, and many also include UV disinfection. The differing efficacies and susceptibilities of these two technologies reinforce the long-held principle of a multi-barrier approach in potable water reuse, as stipulated by regulatory bodies.

3. Regulatory guidelines and standards

3.1. Non-potable reuse (NPR)

Guidelines for wastewater NPR were originally provided by WHO (2006). These are taken as the minimum water quality required based on the direct use of the recovered wastewater with insignificant human exposure. A comprehensive review of standards and guidelines applied globally to recovered sewage (Shoushtarian and Negahban-Azar, 2020) has indicated that the bacteriological water quality required in all countries to be 1–6 orders of magnitude more stringent than those recommended by WHO (Table 2). The difference between the figures presented in the "Low" and "High" columns of Table 3 relate to the sampling schedule. For sampling over an extended period – normally one week – the guidelines refer to either (a) a maximum median value, or (b) a maximum value with an associated percentage compliance.

Within the EU the current standards for recovered wastewater for irrigation use (EU, 2020; ISO, 2020) categorise water quality classes for different uses. The most stringent standard is for *Class A*, and stipulates permitted maximum averaged concentrations for pathogens, BOD_5, TSS and turbidity of 10/100 mL, 5 mg/L, 5 mg/L and 3 NTU respectively (Table 3).

3.2. Indirect/direct potable reuse (IPR/DPR)

IPR demands significantly higher standards both of treated water quality and quality monitoring than NPR. This introduces a compliance issue, with essentially two elements to the sanctioning a candidate treatment scheme:

a) assigning of a maximum attainable performance to a specific unit operation or process technology, and
b) the assessment of risk.

Table 2
Selected global water quality standards, NPR (irrigation).

Origin	Indicator organism		Low	High	Ref
WHO	E Coli	***Concn,** restricted*	**$<10^5$**	**$<10^6$**	WHO, 2006
		***Concn,** unrestricted*	**$<10^3$**	**$<10^4$**	
USEPA	Faecal coliforms	***Concn***	**0**	**14**	USEPA, 2012
		Sampling conditions	7d median	single	
Texas	Faecal coliforms	***Concn***	**<20**	**75**	USEPA, 2012
		Sampling conditions	30d ave	single	
California	Total coliforms	***Concn***	**<2.2**	**<23**	CDPH, 2014
		Sampling conditions	7d median	30d	
New South Wales	Total coliforms	***Concn***	**<10**	**–**	Shoushtarian and Negahban-Azar, 2020
		Sampling conditions	7d median	–	

Figures refer to colony forming units (CFU) or most probable number (MPN).
WHO: *Restricted* crops not eaten raw; *Unrestricted* any crop.
WHO: Low/High values respectively relate to ignoring of/allowance for regrowth during storage.
EU: Low/High values respectively refer to "irrigation of crops to be eaten raw" and "non-potable, residential" respectively.

Table 3
EU *Class A* standards for irrigation (EU, 2020; ISO, 2020).

	TC[1]	BOD_5	TSS	Turbidity
Permitted average MCL[2]	≤10/100mL	≤5 mg/L	≤5 mg/L	≤3 NTU
Permitted single MCL[2,3]	≤100/100mL	≤10 mg/L	≤10 mg/L	≤6[4] NTU
Required sampling frequency	Weekly	Weekly	Weekly	Continuously
Indicative treatment performance: pathogen and its required removal[3]	E Coli ≥5LRV	TC ≥6LRV	CPS ≥4LRV	

Indicative treatment: physical, biological, filtration, disinfection (ISO, 2020).
[1] Thermo-tolerant coliforms (95%ile) for ISO, E Coli for EU.
[2] ISO (2020).
[3] EU (2020).
[4] ≤6 NTU EU (2020).

An established example of the former is that applied in the US State of California (WERF, 2016; SWRCB, 2018) and elsewhere (AGWR, 2006, 2008). The assessment of risk is the approach advocated by the WHO (2006), and is manifested in water safety plans (WSPs) in which quantitative microbial risk assessment (QMRA) is embedded (Zhiteneva et al., 2020).

3.2.1. Credits

The regulations widely recognised as being the most advanced and comprehensive globally are those in place in California. In this region the required performance in terms of removal of key pathogens is stipulated for:

a) the treatment goal (Table 4), and
b) individual unit operations (Table 5)

Whilst the California Title 22 standards have provided the benchmark for required process performance and product water purity, they are proscriptive in nature and do not necessarily account for the prevailing site-based circumstances and, most specifically, the associated risks. The incorporating of site-specific details which may impact on risk forms a key component of water safety plans.

3.2.2. Water safety plans

Water safety plans (WSPs) are essentially holistic risk assessments (RAs) applied to water management and supply, and are sanctioned by the WHO (WHO, 2006). They encompass the expected elements of an RA of hazard assessment, frequency and impact. Completion of WSPs is often an iterative process, where required modification of the system is revealed through close consideration of the risks, required control measures, and managerial aspects (Table 6).

The adaptation of WSPs for application to potable reuse schemes has the potential to replace a plethora of existing approaches which use various aspects of hazard and risk assessment (e.g Rodriguez et al., 2007.). The first documented attempt to assess a potable reuse scheme using a WSP approach was conducted by Dominguez-Chicas and Scrimshaw (2010) with risks illustrated using a heat map graphic. Subsequent advances have seen proposals for dedicated Water Reuse Safety Plans (suggested by Sanz and Gawlik (2014) and further developed in Goodwin et al., 2015) which draw on the strengths of the standard WSP agenda but with enhanced focus on integrating different water cycles and mechanisms to better account for uncertainty, risk interactions and risk prioritisation.

WSPs have also influenced national frameworks for drinking water quality and thereby indirectly impacted the design and management of (mostly indirect) potable reuse schemes. For example, in Australia, an early adopter of WSPs in 1999, recent instances of the federal drinking water guidance (NHMRC, 2016) have adopted the central principles of the WSP approach, putting more emphasis on the responsibility of utilities to monitor and properly manage processes in addition to conforming with delivered water quality standards.

Growth in the DPR sector is providing yet another level of challenge for regulators, standards setting bodies, and scheme operators as the distinctive nature of risks and uncertainties become more apparent. Concerns here centre around a perceived lack of appropriate analysis

Table 4
Water supply in US, total credits required.

Application	Enterov	Giardia	Crypto	FCs	Source
Potable water supply					
SDWA for DWTFs	4	3	–	–	USEPA, 2007
USEPA LT2ESWTR	4	3	>4–5.5*	*<1*	USEPA, 2007
IPR					
SWSAP or SWSA PWS[1], CA	8	7	8		SWRCB, 2018
SWSAP or SWSA PWS[2], CA	9	8	9		SWRCB, 2018
GW replenishment, CA	12	10	10		CDPH, 2014
DPR					
DPR, TX[3]	8	6	5.5		TWDB, 2015
DPR, CA	20	14	15		CWB, 2021
DPR[4]	12	10		9	NWRI, 2012

MCL; SWDA Safe Drinking Water Act; DWTF Drinking water treatment facility; LT2ESWTR Long Term 2 Enhanced Surface Water Treatment Rule; SWSAP or SWSA PWS Surface Water Source Augmentation Project or Public Water System.

* total removal/inactivation required (1–2.5 credit for filtration), credit depending on feedwater concentration.

[1] 100x dilution of treated recycled wastewater during any 24-hour period; excludes credits for conventional potable water treatment works.

[2] 10x dilution of treated recycled wastewater during any 24-hour period; excludes credits for conventional potable water treatment works.

[3] From treated WW (2ndary) to potable supply.

[4] 2ndary + advanced WW treatment stages, Enteric pathogenic bacteria, e.g. Salmonella spp.

Table 5
Californian Title 22 credits: pathogen log reduction assigned for unit processes.

Process	Virus[c]	Crypto	Giardia	Refs
Secondary activated sludge	1.9	1.2	0.8[a]	Olivieri et al., 2016
Microfiltration or ultrafiltration	0	4	4	Olivieri et al., 2016
MBR[d]	1	*2.5*	*2.5*	Salveson et al., 2021
Filtered and disinfected secondary	5	0	0	Olivieri et al., 2016
Reverse osmosis	1.5–2	1.5–2	1.5–2	Salveson et al., 2021, FPRC, 2019
Free chlorine post reverse osmosis	4	0	3	Olivieri et al., 2016
Ultraviolet/hydrogen peroxide AOP	6[b]	6[b]	6[b]	Olivieri et al., 2016
Surface applicn., 6 month retention time	6	10	10	Olivieri et al., 2016
Proposed totals, DPR[e]	**20**	**15**	**14**	CWB, 2021

[a] Subject to Bukhari (2017).

[b] For virus (including adenoviruses) & protozoa, assuming UV dose >300 mJ/cm^2 (AOP typically >900 mJ/cm^2).

[c] Additional LRV assigned for underground travel time, i.e. 1 LRV per month.

[d] Tier 1 (minimum) values: any FS/HF MBR, ≤0.4 μm pore size, permeate ≤0.2 NTU 95% of time ≤0.5 NTU absolute.

[e] At least four unit processes in treatment scheme.

Table 6
WSP process.

Item	Description
Team building	Assembly of WSP team
System description	Documentation and description of water treatment/ management process/system
Hazard assessment	Identification of hazards associated with the process/system
Risk characterisation	Definition of how hazards can enter into the water supply
Control measures	Identification of the means by which risks can be controlled
Monitoring	Identification of required monitoring of control measures, including the boundary conditions defining acceptable performance
Verification	Establishment of procedures able to verify the effective implementation of the WSP in meeting the defined health-based targets
Supporting programme	Development of the supporting programme to maintain safe operation (training, housekeeping, standard operating procedures, upgrade/improvement measures, research & development, etc.)
Management	Preparation of management procedures, including remedial measures, for routine and non-routine (process upsets) operation: this may lead to re-definition of the hazards
Documentation	Establish documentation and communication procedures

and contingency planning for catastrophic risks which have low probabilities of occurrence but high impact consequences. Enhanced safety cultures rooted in the principles which shape behaviours in other high reliability process industries (e.g. aviation, nuclear) combined with a regulatory philosophy which emphasises learning and experience sharing are argued to offer a more suitable operating environment for risk reduction and mitigation (Binz et al., 2020). An independent enquiry body that to investigate water quality and wider system failures would both build public confidence in reuse scheme operations and ensure that near misses are investigated and adequately responded to across the sector.

4. Installations

A summary of the global installations dedicated to wastewater reuse for potable supply (Table 7) indicates them to be predominantly US-based. There are additionally multiple installations established in Australia, Singapore, and Southern Africa. It is unclear as to whether any of the large number of water reuse plants in China provide an indirect supply of potable water (Zhu and Dou, 2018). However, it is evident from a number of studies that the de facto reuse of municipal wastewater for potable water supply presents a substantially greater risk to health than planned reuse either directly or indirectly (Nappier et al., 2018; Amoueyan et al., 2019; Soller et al., 2019).

In the US, where the water reuse regulations are set at State level, an investment of more than $11B in potable reuse is planned in California by 2035 to promote reuse of effluent currently discharged to sea. California has promulgated regulations for DPR (SWRCB, 2019), subject to review by an expert panel. Surface water augmentation in California has been limited because there were no specific regulations to allow the practice. However, this option is the primary focus for new projects in the State now that the regulations are in place. Whilst groundwater replenishment has been the historical focus in California, a number of surface water augmentation projects exist elsewhere in the U.S. (e.g., Nevada, Virginia, Georgia).

There are a number of largely common features across the IPR schemes:

- The environmental buffer in IPR schemes is most often groundwater (hence groundwater recharge, GWR), which generally allows extended times for attenuation of residual pollutant levels. Surface water augmentation (SWA) is also employed, but normally as a consequence of the GWR option being unavailable.
- The use of RO followed by UV irradiation, either for disinfection or occasionally as part of on AOP, is widespread. RO requires protection from channel blockage by fine solids and colloids, and this is most often achieved by MF/UF or, more recently, MBR technology.
- The few DPR schemes employ supplementary treatment steps and/or incorporate engineered storage to provide attenuation.

There appear to be fewer than 5 large DPR schemes currently (as at August 2021) delivering treated reused water to consumers. The schemes at Brownwood, TX and Cloudcroft, NM, have been approved but not constructed. The Wichita Falls scheme was piloted then decommissioned after one year of successful operation, and the El Paso plant is close to completion of the design phase. The 7.5 MLD Colorado River Municipal Water District scheme at Big Spring, TX, is apparently operational, however Since motivations for implementation of IPR/DPR differ regionally, it is instructive to examine a few of the installations in more detail – particularly the more established ones which have remained operational since first implemented. For the more numerous IPR plants, the regions most concentrated in large installations are Southern California, Singapore and Australia. However, one of the oldest installations is the New Goreangab Water Reclamation Plant at Windhoek, Namibia, a DPR facility which has been in substantially continuous operation since its commissioning in 2002.

4.1. Goreangab water reclamation plant, Windhoek, Namibia, DPR

The installation at Windhoek was motivated by a water crisis dating back to 1957, coupled with increased population growth and declining annual rainfall. Freshwater demand in the region increased by over 600 ML/y between 1982 and 2012 (Lafforgue and Lenouvel, 2015). In the absence of significant replenishable groundwater or river sources in the region, demand management and DPR represent the only viable options for conserving water.

Direct wastewater reclamation for predominantly potable use has been in operation in Windhoek since 1969. The original plant 3.3 MLD plant, designed to treat the treated effluent from the city's City's Gammams WwTP, was based on coagulation, DAF, RGF, GAC and chlorination prior to blending with the regular water supply from the surface reservoir provided by the Goreangab Dam. However, since the whole city and its informal settlements lie within the catchment of the dam, the reservoir water quality is often worse than the treated wastewater (Du Pisani, 2006). The capacity from this original plant was subsequently increased to 7.5 MLD in 1997, and currently provides irrigation-quality water.

The new 21 MLD capacity Goreangab water reclamation plant (NGWRP) was constructed at a cost of 12.5 m EUR (hence €0.60 m/MLD) and was opened in 2002. It is fed with tertiary-treated municipal and commercial wastewater, and comprises nine treatment steps (Fig. 3). In addition to online water quality monitoring, automated sampling at every process step is conducted daily, supplemented with concurrent manual sampling for microbiological samples and sampling of the final product water at multiple locations of the distribution network. Water quality guidelines used are an amalgamation of those of the WHO, USEPA, EU, and Namibia/South Africa and, as such, are based on the widely-accepted range of biological, physical and chemical parameters (Table 8).

Health impacts from water reuse were the subject of a 10-year long epidemiological study examining the relationship between diarrhoeal disease and potable reuse between consumers and non-users in Windhoek. Based on the study outcomes, it was concluded that potable reuse did not increase incidences of diarrhoeal disease from water-borne pathogens. Since no adverse health effects were detected the study was terminated in 1983 (Isaacson and Sayed, 1988). A more recent evaluation of the credits provided by the NGWRP (Law et al., 2015) revealed these to be 12.4–13.9 for viruses, 15.2–15.7 for bacteria, and 7.9–9.4 for protozoa – exceeding those demanded by the Australian standards (AGWR, 2006, 2008).

4.2. Langford water recycling scheme (LWRS), UK, IPR

The LWRS IPR scheme dates back to April 2000, when the UK Environment Agency (EA) granted licences to allow discharge of tertiary/quaternary-treated wastewater into the River Chelmer at Scotch Marsh, Essex, and subsequent abstraction downstream of this discharge (Fig. 4). Its implementation was preceded by 10 years of environmental data capture and collation at the behest of the EA. This scheme represents the first large-scale UK example of planned IPR, although there are many examples of de facto IPR in the country.

The WwTW concerned, Chelmsford Sewage Treatment Works (CSTW), treats the sewage by conventional primary settlement and secondary biological treatment using classical trickling filters and activated sludge to meet a discharge consent of 10 mgN/L ammonia, 20 mg/l BOD and 40 mg/l suspended solids. Yhe treated wastewater flows along a 15 km underground pipeline to be discharged into the tidal Chelmer - about a kilometre downstream of two potable water intakes (Langford WTW and the raw water pumping station to Hanningfield reservoir). Under the scheme wastewater is taken from the pipeline into the purpose-built recycling plant at Langford for further treatment. The treated recovered water is then discharged into the Chelmer upstream of the two potable water intakes, augmenting the river flow as well as

Table 7
Drinking water reuse installations (>1 MLD capacity, IPR unless otherwise stated).

ID	Project Name	State/ country	Year(s) op.	Status	MLD	Process	Unit process technology sequence
	US installations						
1	Montebello Forebay, County Sanitation Districts, LA County	CA	1962	Operational	166	GWR via SAT	Media Filtration → Cl
2	Water Factory 21, Orange County	CA	1976–2004	Decommissioned[1]	57	GWR via SB	LC → Air Stripping → RO → UV/AOP → Cl
3	Upper Occoquan Service Authority, Fairfax (UOSA)	VA	1978	Operational	204	SWA	LC → Media Filtration → GAC → IX → Cl
4	Denver Potable Reuse Demonstration	CO	1980–1993	- [2]	4	DPR[2]	LC → Recarbonation → Filtration → UV → GAC → RO → O3 → Cl
5	Huecco Bolson Recharge Project, El Paso Water Utilities	TX	1985	Operational	38	GWR via DI	LC → Media Filtration → O3 → GAC → O3 → Cl
6	Clayton County	GA	1985	Operational	68	SWA	Cl → UV
7	West Basin Water Recycling Plant	CA	1995–2014	Operational	66	GWR via DI	O_3 → MF → RO → UV/AOP
8	Gwinnett County	GA	1999	Operational	227	SWA	UF → O_3 → GAC
9	Scottsdale Water Campus	AZ	1999–2014	Operational	76	GWR via DI	Media Filtration → MF → RO → UV
10	Dominguez Gap Barrier, Terminal Island, City of LA	CA	2002–2014	Operational	23	GWR via DI	Media Filtration → MF → RO → UV/AOP
11	Alamitos Barrier, Water Replenishment District, So. CA, Long Beach	CA	2005	Operational	30	GWR via DI	Media Filtration → MF → RO → UV/AOP
12	Chino Basin Groundwater Recharge Project, Inland Empire Utility Agency	CA	2007	Operational	68	GWR via SAT	Media Filtration → Cl
13	Orange County Groundwater Replenishment System (GWRS)	CA	2008–2014	Operational	378	GWR via DI & SG	UF → RO → UV/AOP
14	Arapahoe County/Cotton wood	CO	2009	Operational	34	GWR via RBF	Media Filtration → RO → UV/AOP → Cl
15	Prairie Waters Project, Aurora	CO	2010	Operational	189	GWR via RBF	Riverbank Filtration → ASR → Softening → UV/AOP → BAC → GAC → Cl
16	San Diego Advanced Water Purification Demonstration Project	CA	2012[2]	Operational[2]	4		O3 → BAC → MF → RO → UV/AOP
17	Big Spring – Colorado River Municipal Water District (CRMWD)	TX	2013	Operational	7	DPR: Blending → CWT	MF → RO → UV/AOP → Conventional Treatment
18	City of Clearwater and the Southwest Florida Water Management District	FL	2013–14	Pilot	11	GWR via DI	UF → RO → UV/AOP
19	Wichita Falls – IPR & River Road WWTP & Cypress WTP DPR projects	TX	2014–15	Superseded by IPR scheme (2018)	26	DPR: Blending → CWT	MF → RO → UV → Storage → Conventional Treatment
20	Cambria Emergency Water Supply	CA	2014	Operational	2	GWR via DI	UF → RO → UV/AOP
21	Village of Cloudcroft	NM	Future	Approved	–	DPR: Blending → AWT	MBR → RO → UV/AOP → Storage → UF → UV → GAC → Cl
22	Hampton Road Sanitation District SWIFT project	VA	Future	Under design	454	GWR via DI	–
23	Franklin	TN	Future	Not yet built	30	SWA	–
24	San Diego Advanced Water Purification Facility	CA	Future	Under construction	68	DPR + SWA	MF → RO → UV/AOP
25	El Paso – Advanced Water Purification Facility	TX	Future	Under design	38	DPR	MF→ RO→ UV/AOP→ GAC→ Cl
	Global installations						
A	Vrishabhavathi Valley project, Bangalore	India	N/A	Studied	200	SWR	UF → GAC → Cl
B	Goreangab Water Reclamation Plant, Windhoek	Namibia	1969; *e*2002	Operational	21	DPR: Blending → AWT	PAC → O3 → Clarification → DAF → Sand filt → O3/AOP → BAC/GAC → UF → Cl
C	Torelle Reuse Plant, Wulpen	Belgium	2002	Operational	7	GWR via infiltration ponds	UF → RO → UV
D	NEWater, Bedok	Singapore	2003	Operational	87	SWA	UF → RO → UV
E	NEWater, Kranji	Singapore	2003	Operational	57	SWA	UF → RO → UV
F	Essex & Suffolk, Langford	UK	2003	Operational	30	SWA	Biological Filtration → UV disinfection
G	Western Corridor Project, Southeast Queensland (Bundamba, Luggage Point, Gibson Island)	Australia	2008	Intermittent operation, NPR	231	SWA → drinking water	
reservoir	UF → RO → UV/AOP						
H	George	S. Africa	2009	Intermittent[3]	10	SWA	Drum Screen → UF → Cl
I	NEWater, Changi	Singapore	2010; *e*2017	Operational	461	SWA	UF → RO → UV
J	Beaufort West	S. Africa	2011	Built	1	DPR: Blending → CWT	Sand filt → UF → RO → UV/AOP → Cl
K	Beenyup Groundwater Replenishment Reuse Trial, Perth	Australia	2011	Decommissioned	5	GWR via DI	UF → RO → UV
L	Beenyup Advanced Water Recycling Plant 1, Perth	Australia	2016 *e*2019	Operational	76	GWR via DI	UF → RO → UV

(*continued on next page*)

Table 7 (*continued*)

ID	Project Name	State/ country	Year(s) op.	Status	MLD	Process	Unit process technology sequence
M	Mexico City	Mexico	Ongoing	Incidental GWR[4]	2155	GWR	None
N	El Port de la Selva	Spain	2015		0.5–2.5	GWR via SAT	Media filtration → UV
O	Mörbylånga Drinking Water Treatment Plant	Sweden	2019	Operational	4	DPR (ind. effl/ BW blend)	Drum Screen → Coag/floc → DAF → SBR → CSF → Coag → UF → UV

"*e*" expanded; GWE Groundwater recharge; SWA/R Surface water augmentation/recharge; SAT soil-aquifer treatment; DI direct injection; SB seawater barrier; SG spreading ground; RBF riverbank filtration; C/AWT conventional/advanced water treatment; BAC biological activated carbon; Cl chlorination; DAF dissolved air flotation; GAC granular activated carbon; IX ion exchange; LC lime clarification; MBR membrane bioreactor; MF microfiltration; O_3 ozone Disinfection; PAC powdered activated carbon; RO reverse osmosis; UF ultrafiltration; UV ultraviolet radiation; CSF continuous sand filtration; BW desalinated brackish water.

[1] Superceded by GWRS.

[2] demonstration project: not put into service [3]Intermittent operation on demand.

[4] Untreated wastewater used primarily for agricultural irrigation.

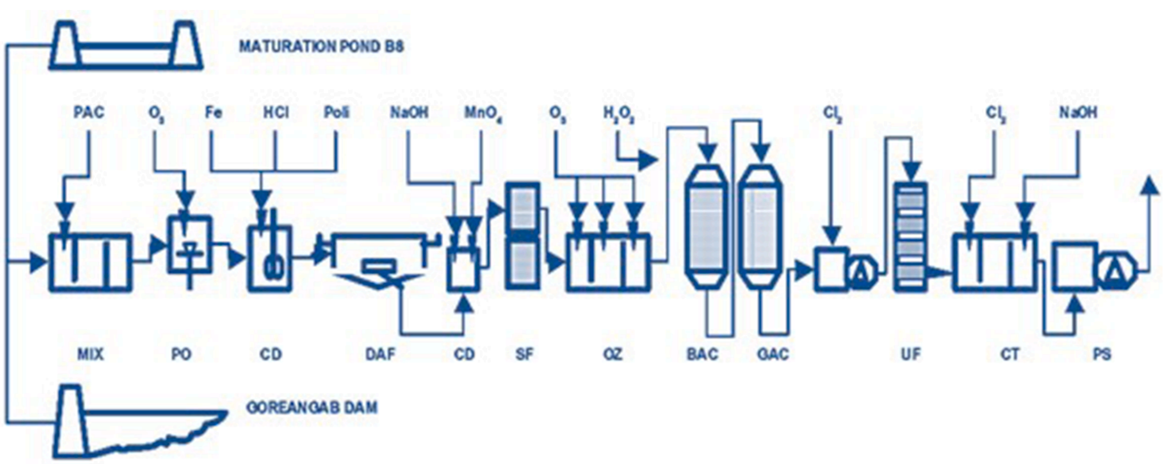

Fig. 3. The treatment scheme at Goreangab serving Windhoek.

Table 8
Water quality parameters monitored at NGWRP.

Microbiological	Chemical & DBPs	Compound/derived parameters
Total coliforms	Aluminium	Chemical oxygen demand
Faecal coliforms	Ammonia	Calcium carbonate precipitation potential
Escherichia coli	*Chloride*	Colour
Coliphages	Fluoride	Dissolved Organic Carbon
Enteric viruses	Iron	Total Dissolved Solids
Faecal streptococci	Manganese	Turbidity
Clostridium spores	*Nitrate & Nitrite*	Alkalinity
Clostridium	*Sulphate*	Total trihalomethane formation potential
Giardia	Total trihalomethanes	UV254
Cryptosporidium		
Chlorophyll a		

DBP Disinfection byproduct; *not removed by process*.

providing an indirect potable water supply.

The 20 MLD-capacity plant comprises advanced (ballasted) clarification, biological denitrification and nitrification, chemical precipitation of phosphorus, and UV disinfection. It was installed at a cost of £13 m, hence £0.65 m per MLD, and has operated since late 2002 on an as-needed basis: it is intended for use during drought periods, when treated volumes may represent up to 70% of the drinking water intakes.

The risk assessment and mitigation demanded extensive baseline environmental monitoring data to demonstrate compliance with the regulatory objective of "no deterioration" in environmental quality. Assessment was mainly based around the impact on downstream ecology from potential discharge of nutrients, endocrine disrupting chemicals (specifically nonylphenol), toxic metals and pathogens (Table 9). Quantitative ecological impacts encompassed invertebrate taxa environmental quality index ranges, fish sexual development and wildfowl diversity. Assessment for a further 8–10 years following implementation of the scheme demonstrated no significant difference in metrics either between (i) upstream and downstream samples, or (ii) sampling prior to or following the scheme implementation. Following expiration of the initial 10 year license, the EA granted an indefinite license to continue operation of the scheme.

In this instance the impact of the scheme on the safety of the drinking water supply was of secondary importance to the possible environmental impact on the receiving water bodies. Throughout the duration of the scheme's operation, the water quality of the recovered water has been substantially higher – based on every metric recorded – than the receiving environmental water which is subsequently abstracted for potable water supply.

The Langford scheme highlights three key issues:

1 Ownership of the water: Hanningfield reservoir is owned by the water company, but its management is subject to stipulations made by the UK regulator.
2 The quantification and subsequent proof of "no deterioration" in environmental quality: it is not possible to absolutely prove zero deterioration, and there is no consistency in the interpretation of this stipulation across the many UK environmental regulatory bodies and pressure groups.
3 Extent of responsibility: the water company has had to fund certain activities to mitigate against perceived impacts of abstraction of the wastewater which would otherwise flow to the downstream marina, such as the annual dredging of Maldon Dock which is subject to silting up - even though the scheme only operates during dry seasons.

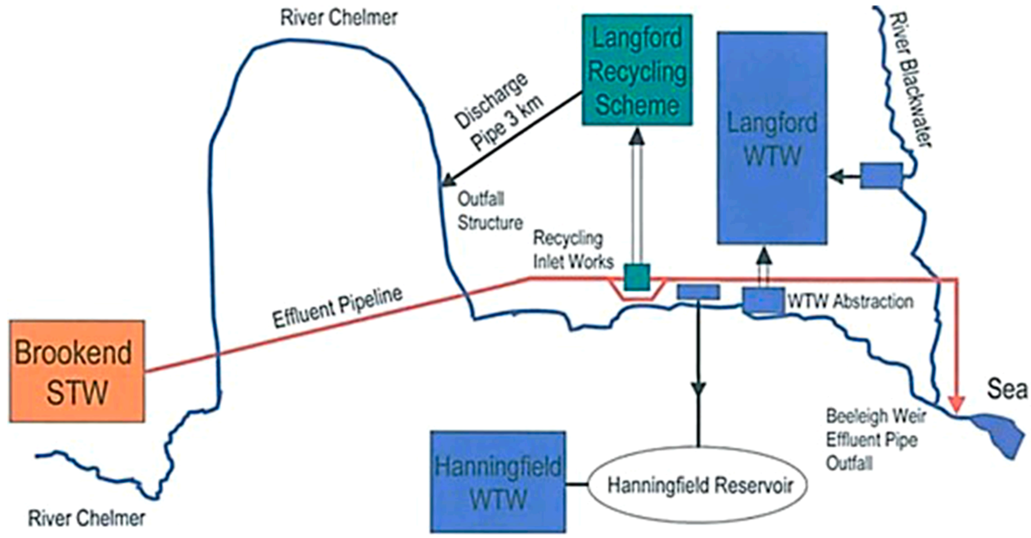

Fig. 4. River Chelmer, abstraction and discharge points of Langford scheme indicated (Herring, 2002).

Table 9
Environmental risk assessment of LWRS, summary of outcomes.

Parameter(s)	Outcome
Pathogens	99.3 - 100% removal; consistently higher bacteriological quality than receiving water
BOD, COD, TN, P, ave chlorophyl	No exceedances of UWWD benchmark. <3 exceedances ea. of licenced limit for BOD, soluble P, TN & Fe
Fe, Cu, Cd, nonylphenol	No exceedances of licenced limit other than one exceedance for Fe
Macrophyte mean trophic rank (MTR)	Annual results since 1994 largely stable[1,2]
Chlorophyll "a", Diatom Quality Index	Maximum chlorophyl level exceeded only in summer of 2007, both upstream and downstream of discharge
Invertebrates	No impact on average score per taxon[1]
Native fish (roach) male feminisation	No impact[1,3]
Water bird species population	Of 16 species evaluated, 14 remained stable or increased

[1] Applies to the period 7–9 years prior to and following scheme implementation.
[2] No alteration since P stripping at Shenfield STW in 1998: casts doubt on ability of MTR to detect changes in nutrients.
[3] Impact noted only for CSTW effluent: no impact from "stripped" river water (control) or LWRS effluent.

4.3. Singapore, IPR

Almost 40% of Singapore's water supply is currently made up of recovered municipal and industrial wastewater (termed "NEWater") by water reclamation plants (WRPs). The NEWater is primarily supplied to non-domestic sectors for industrial and cooling purposes and not directly to households for consumption. During dry months, however, a small proportion of NEWater is directed to reservoirs to supplement the drinking water reserves. The blended reservoir water then undergoes further treatment at the water treatment works prior to going into supply. A rigorous sampling and monitoring programme is in place for the NEWater (SFA, 2019b), encompassing over 300 parameters, to ensure that its quality exceeds benchmark international potable water quality standards (USEPA, 2009; WHO, 2017).

Water supply in Singapore is subject to the completion and implementation of a water safety plan and a water sampling plan by the supplier, the Singapore Public Utilities Board, in accordance with the WHO drinking water guidelines (SFA, 2019ab) Section 2. of Chapter 95 of the Singapore Environmental Public Health Act (SEPH, 2019a) requires the supplier to maintain records, report any incidents leading to the diminution in potable water quality to the Director General, and outline and enforce the necessary remedial measures.

The WRPs plants at Bedok and Kranji, both based on UV disinfection of membrane-filtered secondary wastewater, were commissioned in 2003 following a two-year demonstration programme at the Bedok site, encompassing:

- practical demonstration of the scheme based on operation of a 9.8 MLD capacity plant;
- extensive sampling and monitoring of the treated water for physical, chemical and microbiological parameters;
- assessment of health/toxicological impacts, based on rodents and fish, for carcinogenic and oestrogenic effects respectively.

The above was supplemented by independent expert panel review to scrutinise the performance and health data (PUB, 2002), as well as a public outreach programme which included a purpose-built visitor centre. The estimated CAPEX for the schemes was $0.58 m per MLD.

Singapore continues its drive towards water self-sufficiency. The planned WRP at Tuas (Fig. 5), to be based on MBR technology, is designed to recover most of the water and resources from 650 to 150 MLD of domestic and industrial wastewater respectively (Zheng, 2021), and will be one of the largest MBR installations in the world. The municipal effluent stream will, as with the earlier schemes, employ RO and UV as the downstream polishing technologies, but with MBR technology used for combined advanced biological treatment and clarification. The latent energy from the biosolids is to be recovered by anaerobic digestion, enhanced using upstream thermal hydrolysis.

4.4. Orange county water district groundwater replenishment system, IPR

The 265 MLD Orange County Water District (OCWD) groundwater replenishment system (GWRS) cost $481 m (hence $1.8 m/MLD) and has been online since January 2008. The plant clarifies, desalinates and disinfects secondary effluent using microfiltration (MF), reverse osmosis (RO) and UV/H_2O_2, with subsequent lime mineralisation (Fig. 6). Conditions imposed by the regulators to mitigate against the risk of contamination of the product water include (Dadakis et al., 2011):

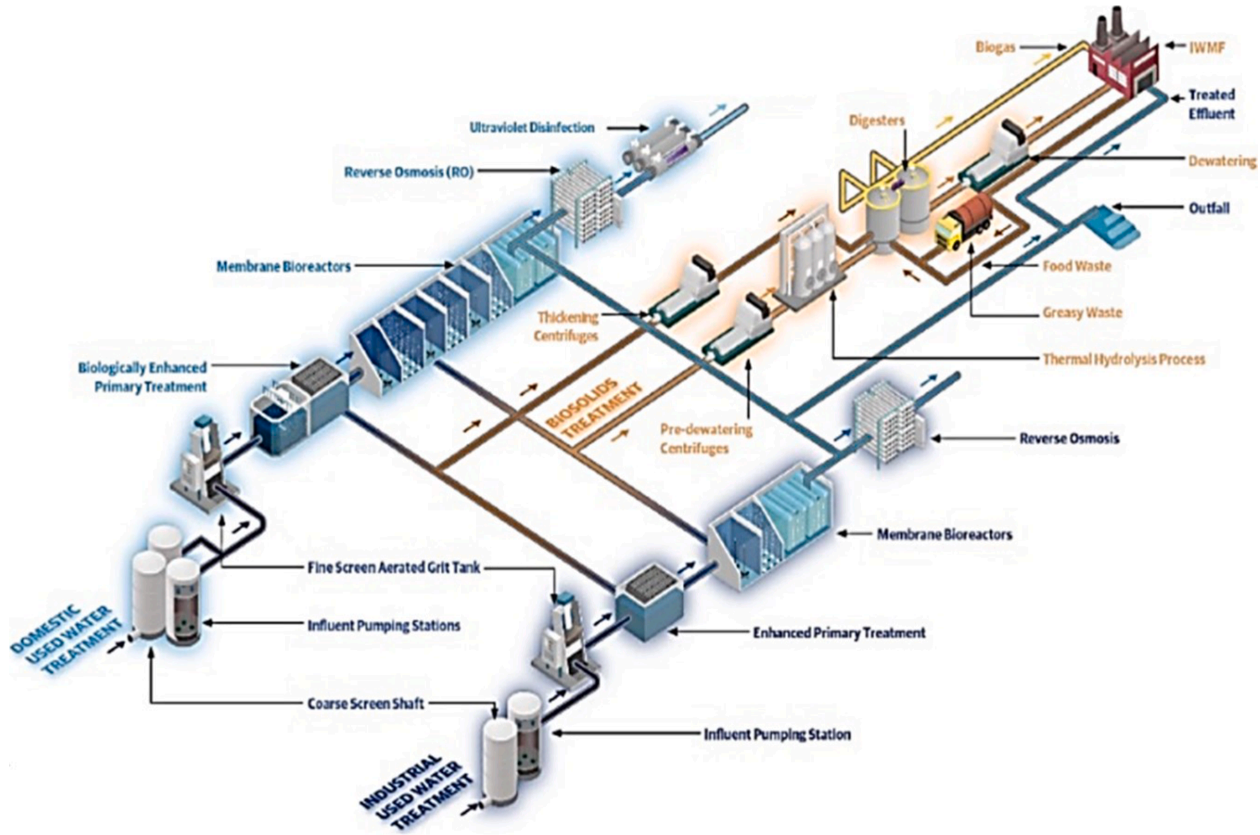

Fig. 5. The planned NEWater WRP at Tuas, Singapore (Zheng, 2021).

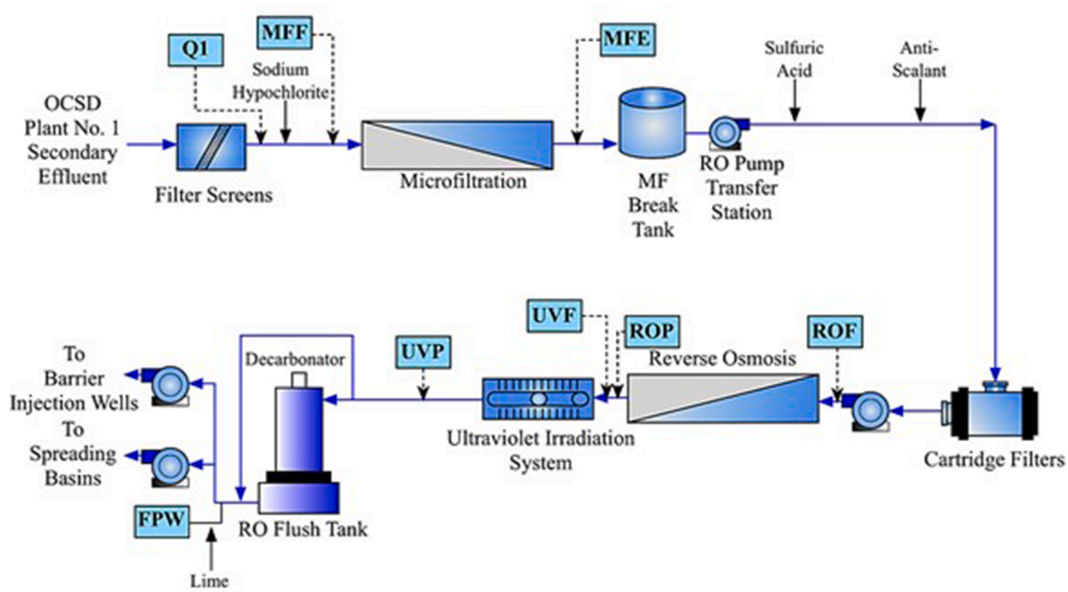

Fig. 6. The OCWD GWPS (from Stamps et al., 2018).

a) minimum retention times and separation distances between the both the surface spreading basins and the barrier injection point to the nearest down-gradient drinking water production well;
b) a maximum contribution of 75% from the recycled water stream to the total water stream at the surface spreading basin;
c) the requirement to monitor CECs (endocrine disrupting compounds (EDCs), pharmaceuticals and personal care products (PPCPs)) and, subject to advice from an Independent Advisory Panel, specific CECs on the basis of health impact or as an indicator of process performance (Table 10).

4.5. Wichita falls, TX, DPR

The 19 MLD demonstration DPR plant at Wichita Falls (Fig. 7) was operated continuously for 377 days from 9 July 2014. 1.8 m discrete water quality data were collected over the course of the operational

Table 10
Supplementary water quality parameters monitored at OCWD groundwater replenishment system.

CEC monitored due to health impact	CEC monitored for process performance
17β-oestradiol: steroid oestrogen naturally excreted by humans	N,N-diethyl-m-toluamide (DEET): insect repellent effectively removed (>90%) by RO
Triclosan: antimicrobial chemical found in toothpaste and hand soap	Sucralose: artificial sweetener effectively removed (>90%) by both RO Electrical conductivity* Dissolved organic carbon*
CECs monitored for both health impact and process performance	
N-Nitrosodimethylamine (NDMA): propellant and DBP, also found in cured meats and beer	NDMA: moderately removed (25–50%) by RO, but effectively treated by UV (>90% removal)
Caffeine: natural stimulant	Caffeine: well removed (>90%) by both RO and AOP
17β-oestradiol: naturally excreted steroid oestrogen	
Triclosan: antimicrobial found in PCPs	

*surrogate representing removal performance of RO membranes.

period (Nix et al., 2021), during which time over 7500 ML of wastewater was reclaimed as drinking water without water quality failures or plant shutdowns. No viruses were ever detected in the treated water and E. coli, Giardia, and Cryptosporidia were undetected downstream of the MF. The facility achieved 100% compliance with all primary and secondary drinking water regulations, as well as with the pathogen log removals stipulated by the regulator (Table 11). Management of the scheme was supplemented by a task force of all stakeholders, including the regulator and the regional health district, which met monthly to review operational data and receive updates from the health district's epidemiologist on any related health anomalies.

Record rainfall in May 2015 returned the region's reservoirs to 100 percent within three weeks. This contributed to the decision to transition to IPR.

5. Conclusions

A review of the status of wastewater reuse for potable supply, both direct and indirect, focused primarily on the regulatory and practical aspects has revealed:

1 More than half (22 out of 40) of the schemes for which information was captured (Table 7) are based on enhanced clarification – most often UF or MF – followed by RO and then UV for final disinfection, with modern DPR schemes tending to employ UV-based advanced oxidation as the final step.
2 The rapid development of quantitative polymerase chain reaction (qPCR) and loop-mediated isothermal amplification (LAMP) has led to their increased implementation for water quality monitoring due to their high specificity and sensitivity for pathogen detection.

Table 11
Key water quality determinants, Wichita Falls DPR scheme (Nix et al., 2021).

Parameter	Comment
Chemical concentrations	
Nitrate-N	Reduced from 20 mg/L in wastewater to 0.66 mg/L in recovered water supply
tTHMs	14.2 µg/L on average, 16.7 µg/L maximum, at distribution system point of entry
Pathogen LRVs	
Viruses	9
Giardia	6
Cryptosporidia	5.5

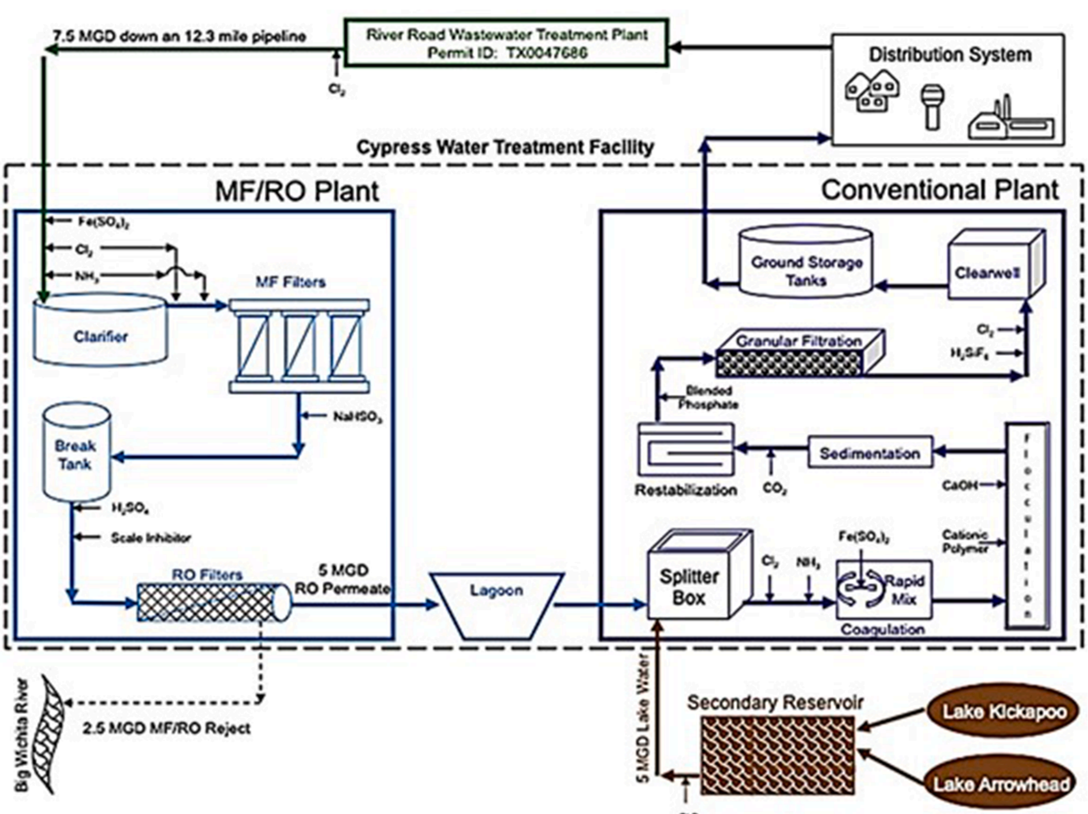

Fig. 7. Wichita Falls DPR scheme, membrane and conventional plant components (Nix et al., 2021).

3 Quantitative microbial risk assessment (QMRA) studies suggest that the risk imposed by DPR and, in particular, IPR is very small compared with de facto reuse, where the latter is not subject to the same level of scrutiny by the regulators.
4 The few epidemiological studies which have been conducted have related to early, more rudimentary IPR/DPR schemes, but these studies have nonetheless reported zero impact of municipal water reuse on the health of the consumer.
5 IPR is complicated if "zero deterioration" in the environmental quality of the water buffer has to be demonstrated, since this condition is not subject to consistent interpretation either internationally or nationally.

With respect to environmental impact on the buffer (point 5) extremely conservative conditions have been imposed, most graphically illustrated by the UK Langford scheme, which is not commensurate with the risk incurred and goes well above and beyond regulations pertaining to de facto water reuse. If, on the other hand, the environmental buffer is both contained and owned by the water provider then these stipulations don't apply.

QMRA (point 3) provides a mechanism for quantification of infection risk from pathogens, and can be extended to other pollutants for which quantitative data on influent concentration, removal by unit wastewater treatment technologies, and health impacts is available. However, it appears to be challenged by a lack of (a) appropriately detailed wastewater quality data, and (b) consensus relating to the methodology and assumptions used, which may limit the more widespread applicability of the findings from individual analyses. Notwithstanding such limitations and regardless of the noted inconsistencies and challenges, analyses conducted have all tended to indicate that the microbial risk imposed by the product water from existing IPR and DPR schemes to be significantly below the thresholds set by the regulators. The low risk is supported by the few epidemiological studies which have been conducted (point 4), This being the case, the most fruitful way forward would appear to be investing in ensuring process robustness.

Whilst incurring a relatively high energy demand, as well as generating a waste stream from the RO step which requires further management, the inclusion of RO with downstream UV in the majority of the treatment schemes (point 1) would seem to provide the appropriate level of efficacy when these processes are operating optimally. Maintaining this efficacy, i.e. the required low health risk, therefore relies on (a) reducing process failure risk and, in relation to this, (b) ensuring reliable on-line monitoring.

There has been extensive demonstration of qPCR assay for many duties, including reuse water quality assessment. The method appears to offer a reliable, precise and sensitive for pathogen identification and quantification (point 2) but, as with all microbial analytical methods for this duty, demand substantial enrichment of the water sample due to the extremely low pathogen concentrations in the treated water. Against this, research into the quantification of process failure – whether relating to UV or membrane technology – is sparse. This inevitably reflects negatively on the veracity of the outcomes from the various risk analyses conducted in this area.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Data Availability Statement

Data are available with the permission of refereed publications. The data that support the findings of this study are available from the corresponding author, SJ, upon reasonable request.

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