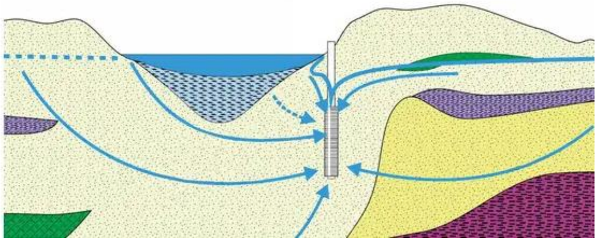

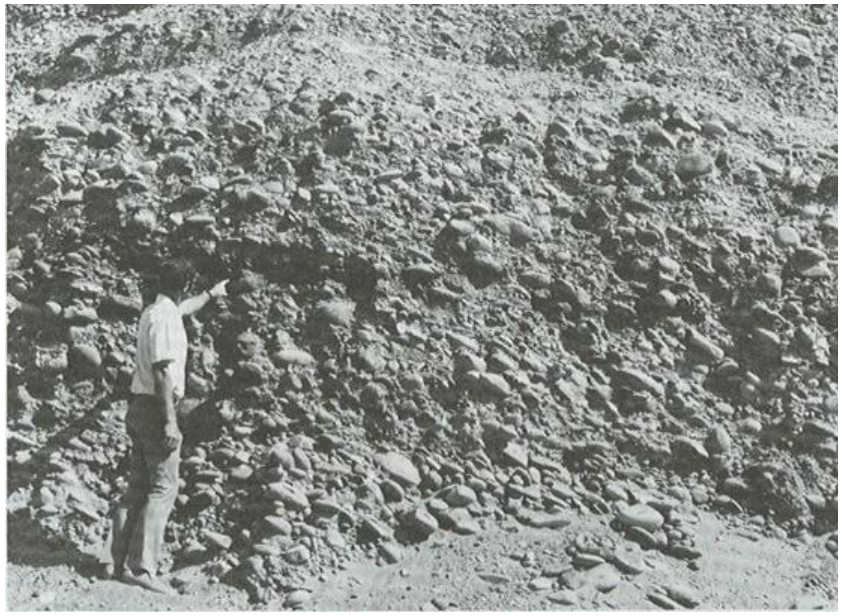

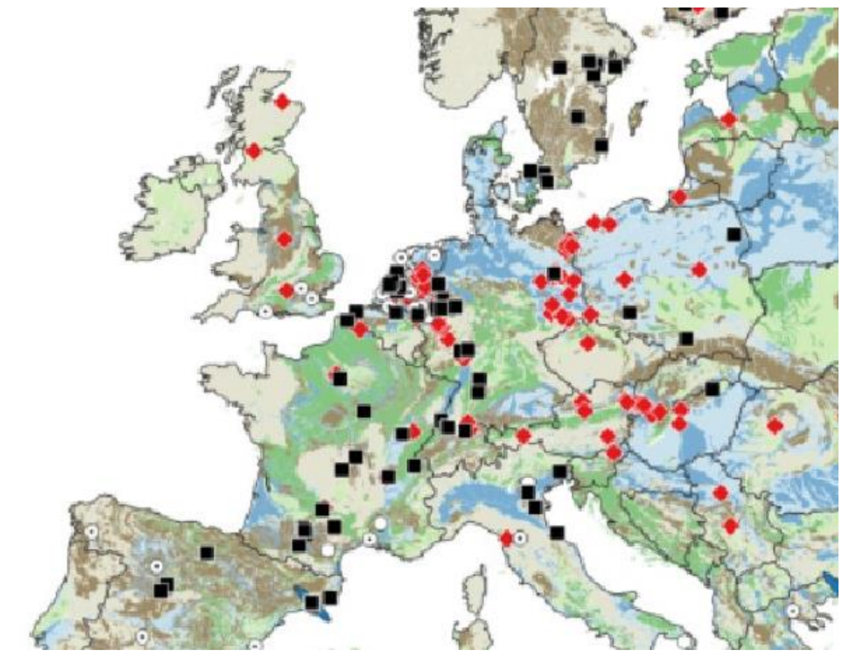

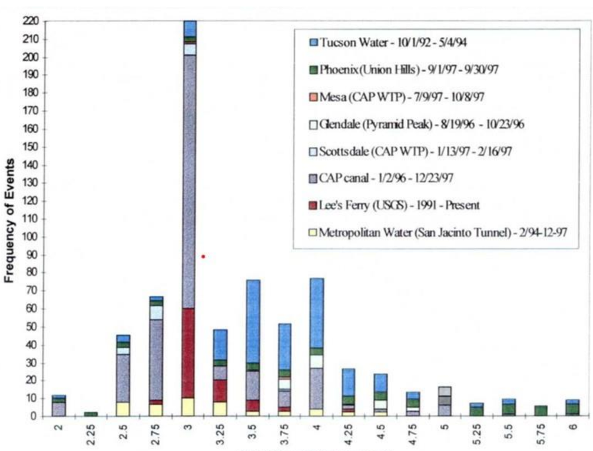

Fate and Transport of Pathogens, Organic Carbon and Nutrients in Soil Aquifer Treatment

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Introduction

Typical constituents of concern in treated wastewater effluent include pathogens (viruses, bacteria, parasites, etc.), organic compounds derived from natural, human and industrial, pharmaceutically active chemicals (endocrine disruptors) and soluble salts (i.e. nitrates, etc). These constituents may not be completely removed depending on the sewage treatment process. The use of water from effluent dominated streams is significant in the United States, a survey of more than two dozen major U.S. water utilities that use water from rivers that receive secondary treated effluent discharges amounting to more than 50% of the stream flow itself during low flow conditions did not appear to produce apparent adverse health effects on the population (Crook et al., 1999). Soil aquifer treatment (SAT) that occurs during managed aquifer recharge (MAR) can provide additional water treatment of treated effluent and with mixing with native groundwater should produce groundwater suitable for human consumption.

SAT uses natural processes to remove microorganisms including bacteria, protozoa, and viruses, organic compounds, and nutrients during recharge of treated wastewater through unsaturated soils to groundwater (NRC 1994). The fate and transport of pathogens in a recharge system depends on their characteristics affecting their interaction with groundwater and subsurface materials. i.e. charge density, isoelectric point (pH at which surface charge changes between negative and positive) and hydrophobicity (Rauch-Williams et al., 2023). Numerous studies have demonstrated the effectiveness of soil infiltration for removing pathogenic bacteria and larger organisms such as protozoa in arid and semi-arid regions via soil straining and inactivation near the soil surface (EPA, 2004; Ausland et al. 2002; Castillo et al. 2001; NCSWS 2001; Quanrud 1998; Bouwer 1996; Guessab et al. 1993; Kanarek et al. 1993; Gerba & Goyal, 1985; Gerba et al., 1975).

From a public health perspective, viruses are the organisms of primary concern in recharging groundwater with treated wastewater (Meschke & Sobsey 1998, Rauch-Williams, et al., 2023); field and laboratory studies indicate that virus removal with SAT is a function of distance traveled through the vadose and saturated zones (Knabe, D. et al. 2023; Betancourt 2014, 2019; Gotkowitz, M.B. et al. 2016; Taylor et al. 2004; Nasser, Glozman; and Nitzan 2001). SAT also reduces organic carbon and nutrients such as nitrate and phosphate, primarily within the first five to ten feet of infiltration (Hutchinson et al. 2017; Quanrud et al. 2005; Fox et al. 2000). Removal of constituents of concern (CECs) is variable and dependent on the CEC, pretreatment and operating conditions and travel distance (Sunyer-Caldu et al. 2023; Sallwey et al. 2020; Trussel, et al. 2018; Laws et al. 2011).

Finally, properly designed and operated SAT recharge facilities have been shown to be a sustainable effluent treatment process for the removal of pathogens and organic carbon (i.e. Murray 2020, Quanrud et al. 2005) and long-term monitoring indicates that SAT efficiency may increase over time (i.e. Elkayam et al., 2015, Regnery et al., 2015). Rauch-Williams, et al. (2023) state, "MAR processes can be very cost effective relative to other pathogen barriers, while not sacrificing water recovery. Thus, in terms of energy, sustainability, and longevity, it may be one of the most robust treatment processes for our use." The following review addresses SAT removal of bacteria, protozoa, viruses, organic carbon, nutrients and CECs.

SAT in Riverbank Filtration Systems

Riverbank filtration (RBF) is a process in which pumping wells located along riverbanks induce a portion of the river water to flow toward the wells. During RBF, river water contaminants are attenuated from a combination of SAT processes such as filtration, microbial degradation, sorption to sediments and aquifer sand, and dilution with background groundwater (Ray et al. 2003). RBF systems have been used extensively in Europe for over a century as pretreatment to remove pathogens and reduce contaminants from surface water initially affected by untreated, and now, treated sewage. The first documented RBF system was at the Glasgow Waterworks Company in Scotland in 1810 (Ray et al. 2003). As an example, Germany operates more than 300 RBF systems which supply over 16 percent of the drinking water supply (Schmidt et al., 2003); the Netherlands relies on RBF and MAR systems to pretreat for pathogens for approximately 20% of their water supply (Medema and Stuyfzand, 2002). In the USA, at least a dozen horizontal collector RBF systems operate in the United States with approval to remove *Giardia*, *Cryptosporidium*, and viruses (Ray et al. 2003).

In an RBF system, most of the SAT occurs in the hyporheic zone, the transition area between the surface water and groundwater in an alluvial aquifer, which is more biogeochemically (microbially) active than the surface or groundwater due to interactions of light, temperature, pH, oxygen, and organic matter (Jaramillo 2011). Beneath a river channel, the hyporheic zone is shallow, and creates, due to microbial oxygen consumption, an anaerobic zone in which reductive reactions also occur (Jaramillo 2011). Local conditions at an RBF site are among the determinants of the system's success in reducing pathogen and other contaminant concentrations in groundwater. Among the most critical factors are the aquifer thickness, river infiltration area (surface area in contact with infiltrating water), and length of the flowpath to the aquifer (Grischek et al. 2003).

The screened depth and distance of wells from the river, and hence travel distance and time, are important considerations for RBF efficiency in removing contaminants. Grischek et al.

(2003) compiled data on RBF site specifications from multiple facilities in the USA and Germany. At most sites in Germany, the distance between the riverbank and production wells is greater than 50 m and travel times are greater 50 days, in the USA, travel times are generally less than 50 days primarily due to the use of horizontal collector wells that may be within a few meters of the river channel. Due to varying well distances and subsurface heterogeneities, the efficiency of pathogen removal via RBF can vary. Knabe et al. (2023) reported that coliform bacteria and somatic coliphages removal in an active RBF system decreased as the Rhine River levels rose and travel times decreased, whereas adenovirus removal rates remained high. This was attributed to bacterial removal via straining (time-dependent) versus virus removal via adsorption (distance dependent) mechanisms. Likewise, one Ranney well (Collector 5) in the RBF system operated by Sonoma County Water Agency (SCWA) shows reduced efficiency during high winter flows; this lateral is not used during those periods (SCWA, 2013).

To date, there is no evidence that functioning RBF systems in alluvial aquifers have failed to prevent pathogenic disease (Ray, 2003, Medema and Stuyfzand, 2002). Schijven et al. 2003 reviewed disease outbreaks associated with RBF systems, in particular cryptosporidiosis, and concluded that other factors such as cross-contamination were likely responsible; the only unequivocal outbreaks have occurred in non-porous media hydrogeologic settings (i.e. karst and fractured rock aquifers) where SAT processes are minimal.

Pathogen Removal Efficiency Requirements

Pathogen "log reduction" is a term used by some regulatory agencies to refer to the physical or chemical removal or deactivation of pathogens. A 1-log removal is equivalent to a 90 percent reduction, a 2-log removal to a 99 percent reduction, etc. The Long Term 2 Enhanced Surface Water Treatment Rule (USEPA, 2006), requires water utilities serving surface water or groundwater under the influence of surface water to treat water and reduce *Cryptosporidium* by 3-log to 5.5-log, depending on the source water concentrations. Log reduction requirements for Giardia and viruses are 3-log and 4-log respectively. A 1.0-log reduction credit is given to RBF wells at least 50 feet from the river with additional credit given for site specific data; SCWA receives a 2.5-log reduction credit (SCWA, 2013). By comparison, Ray et al. (2003) states properly designed and operated RBF plants reach 4-log removal efficiency, other authors have reported log reduction rates of between 1.5-log to 5-log for bacteria and viruses in RBF and shallow SAT systems (Jamarillo 2011; Nagy-Kovács et al., 2019; Partinoudi and Collins, 2007; Elkayam et al. 2015).

U.S. states have differing regulations applicable to MAR, including those regarding removal efficiency, and no states have enacted MAR-specific regulations, but some include MAR systems in other programs (Rauch-Williams, et al., 2023). The California Groundwater Replenishment Reuse Projects regulations (CCR, 2014) require a demonstration of 12-log enteric virus reduction and 10-log Giardia and Cryptosporidium reductions for groundwater recharge of treated wastewater effluent. Up to half (5- to 6-log) of the reduction can be demonstrated via water treatment plant systems; a one-month log reduction credit for each month of demonstrated aquifer retention is effectively the standard criteria applied. Nevada also requires the same 12-log enteric virus/10-log *Giardia* and C*ryptosporidium* reductions between the point of entrance of raw sewage to the point of drinking water extraction (Rauch-Williams, et al., 2023).

Conversely, the Netherlands, which obtains approximately 20% of their water supply from RBF and MAR systems using river water with pathogen counts only two orders of magnitude lower than raw sewage (Zuurbier et al., 2018), uses a quantitative microbial risk assessment specific to each utility to establish log reduction requirements (Smeets et al., 2009). This results in a 4- to 6-log reduction total treatment requirement for enteric viruses and 6- to 8-log reductions for bacteria and protozoa (i.e. *Cryptosporidium);* Zuurbier et al. (2018) also reported that the RBF/MAR are operated to achieve a minimum residence time of at least 30 days, thus achieving over 9 log removals of viruses due to SAT and dilution.

Australia requires 9.5-log, 8-log, and 8.1-log reductions for enteric viruses, *Cryptosporidium*, and *Campylobacter*, respectively, for systems producing potable water from untreated wastewater (Australia NRMMC et al., 2008).

The World Health Organization (2017) set removal targets of 8.5-log for both enteric bacteria and enteric protozoa and a 9.5-log for viruses and reported a validated log reduction for SAT of 6-log removal for bacteria, viruses, and protozoa based on a review of literature values.

Recently, authors working with Polymerase Chain Reaction (PCR) DNA detection methods have suggested that traditional live culture methods may underestimate actual viral counts in wastewater and therefore log reduction requirements should be increased by two to three orders of magnitude (i.e. Gerba et al., 2017). However, PCR methods cannot distinguish between live and dead viruses, and the efficacy of pre-treatment methods to filter and concentrate viruses, and to digest dead nucleic acid (via nuclease) is uncertain (WRF, 2023). The primary conclusions of this study were that the of log reduction credits are limited, and potable water safety may be better addressed using long-term monitoring of the pathogenic viruses of concern to show removal rather than using analogues (i.e. non-pathogenic viruses). The WRF (2023) study also showed that the SAT/RBF treatment systems studies showed similar or better virus removal compared to the advanced water treatment with reverse osmosis systems studied.

SAT Removal of Bacteria and Protozoa

Removal of bacteria and protozoa in most soils is accomplished primarily by filtration (or straining) and die-off, and secondarily by adsorption and inactivation. Filtration occurs because the bacteria and protozoa are too large to move with water through the void spaces in soils and clogging layers. This filtration process also occurs in effluent-related clogging layers in the soil surface where microorganisms, algae, suspended solids, and organics accumulate (Houston et al., 1999). Microbial biofilms and clogging layers increase purification (Van Cuyk et al. 2001) by reducing the pore size, which strains the microorganisms, and increases predation of pathogens. Both column studies (i.e. Quanrud et al., 2003; Trussel et al. 2015, 2017) and field evaluations (i.e. Bouwer, 1996; Fox et al. 2006) indicate the complete removal of protozoa (i.e. *Cryptosporidium*) and bacteria over relatively short distance and time periods. Bouwer reported the vast majority of fecal organisms are restricted to surface layers as wastewater infiltrates through soils. Gerba (1975), reported fecal coliforms in secondary-treated effluent travelled a maximum vertical distance of 30 ft through a fine sandy loam to gravel soil. Trussel et al. (2015, 2017) reported over a 7-log removal of *Cryptosporidium* in a 3.7 m column packed with sandy aquifer material.

In addition to filtration, adsorption and inactivation play a role in removing pathogens from wastewater during SAT. Pathogens may adsorb to charged soil mineral and organic matter surfaces. Adsorption is influenced by soil texture, cation concentration, soluble organics, pH, virus type, infiltration rate, and soil moisture content (Gerba et al. 1975; Stevik et al. 2004). In some cases, adsorption can be negated. For example, microorganisms may sorb to clay particles, which can protect bacterial cells and possibly viral particles (Santamaria & Toranzos 2003) or organic matter in soil may compete with microorganisms for adsorption sites on soil particles and may increase transport through the soil (Blanc & Nasser 1996). However, the former process will tend to increase the bacterial retention time (and subsequent inactivation), whereas the latter process is most likely limited to the accumulation of organic matter that occurs in the near subsurface.

Microorganism survival is controlled by soil temperature, moisture content, pH, and organic matter, the type of organism, and the presence of other microorganisms (Campos et al. 2001; Gerba et al. 1975). For example, pathogenic enteric bacteria, among the most prevalent pathogens found in wastewater, are adapted to the conditions in the intestinal tract: high nutrients and a relatively high temperature (37ºC). When these organisms are introduced into wastewater or soil, they are not always capable of competing with indigenous organisms for scarce nutrients and their ability to reproduce and survive in soils tends to be limited.

SAT Removal of Viruses

Because of the small size of viruses, removal by filtration is not as effective as it is for larger bacteria and protozoa (Bitton & Harvey, 1992; Santamaria & Toranzos, 2003). Viruses can move through large soil/rock pores at a faster rate than average groundwater flow (Gerba, 2024). Viruses are also generally more resistant to inactivation than are most pathogenic enteric bacteria (Meschke & Sobsey 1998). However, virus adsorption to soil surfaces plays a major role in limiting viral transport in the vadose zone and groundwater (Gerba et al., 1975, Gerba, 1999; Schijven & Hassanizadeh 2000; Stevik et al. 2004). Table 1 summarizes the factors influencing the transport of viruses.

Soil type and moisture content, temperature, infiltration rate, travel time, and virus type are the major influences on virus transport (Yates & Gerba 1998, Knabe et al. 2023). Different strains of viruses have different isoelectric points which alter their ability to adsorb to soil particles. MS-2 and PRD-1 are bacteriophages that have been commonly used as viral surrogates in seeded virus studies because they are non-pathogenic. They adsorb less easily to soil particles than do enteric viruses (Goyal & Gerba 1979; Powelson et al. 1993; Van Cuyk et al. 2004), and therefore provide a “worst-case” model for viral transport (Havelaar et al. 1993).

Table 1. Factors Influencing the Transport of Viruses

Factor	Comments
Soil type[1]	Fine-textured soils generally adsorb viruses better than coarser-textured soils.
pH[1]	Generally, adsorption increases when pH decreases. However, other factors may alter this trend. pH affects inactivation of different viruses in different ways.
Cations[1]	Adsorption increases in the presence of cations, which help reduce the repulsive forces between viruses and soil particles. Cations can increase or decrease susceptibility to inactivation of different viruses.
Soluble organics[1]	Soluble organics generally compete with viruses for adsorption sites on soil particles, however, there is no significant competition at dissolved organics concentrations found in wastewater.
Virus type[1]	Adsorption to soils varies with virus type and strain because viruses have different isoelectric points.
Flow/infiltration rate[1]	Higher flow rates lead to lower virus adsorption to soil.
Saturated vs. unsaturated flow[1]	Virus movement decreases under unsaturated flow.
Redox conditions[3]	Viruses are inactivated more quickly in aerobic conditions due to greater levels of biological activity and predation
Climate[1]	Low-conductivity rainwater may desorb viruses from soil.
Moisture content[1]	Generally, low moisture content increases inactivation. Saturated soils show the lowest inactivation rates

Factor	Comments
Indigenous organisms[1]	Most reports indicate that indigenous aerobic microorganisms increase virus inactivation, but anaerobic microorganisms have no effect on viruses.
Soil physico-chemistry[3]	Particle size, porosity, sorting, grain chemistry, isoelectric points, and fractured bedrock affect retention and transport
Presence of a clogging layer[1]	Decreases transport of viruses. Generally increases inactivation.
Temperature[1]	Higher temperature generally increases inactivation.
Travel distance/retention time[2]	Removal may decrease with distance from the river (not constant or linear)
Precipitation[3]	In some climates, mobilization may increase with larger than normal precipitation events

[1]Source: Gerba and Goyal 1985
[2]Source: Knabe et al. 2023
[3]Source: Rauch-Williams et al. 2023

Many factors can affect virus adsorption/desorption, including soil porosity and grain size, virus size and surface charge, water flow velocity, temperature, pH, degree of soil mineralization and ion concentration and charge (Knabe 2023, Chu et al. 2001). Virus attachment to soil particles can be reversible and, if soil conditions and sufficient travel time or distance do not allow for inactivation, virus remobilization in groundwater can create a contamination risk (Pang et al. 2021, Knabe et al. 2023). However, other studies have shown most virus adsorption may be irreversible (e.g. Sasidharan et al., 2017).

Temperature is an important factor in the survival or inactivation of viruses in soil (Yates et al., 1985, Gerba 1999, Gerba et al. 1975; Nasser et al. 1993). Higher temperatures inactivate viruses; viruses are more stable at lower temperatures. At low temperatures (below 4ºC or 39 ºF), some viruses can survive for months or years. The die-off rate approximately doubles with every 10ºC (18ºF) increase in temperature between 5 and 30ºC (41 and 86ºF) (Gerba & Goyal 1985).

Other factors affecting pathogen survival are soil moisture content, pH, organic matter, pore size, type of organism, and presence of other microorganisms. Virus survival increases as the soil moisture content increases up to the soil saturation point (Blanc & Nasser 1996; Campos et al. 2001). Drying the soil will kill both bacteria and viruses (EPA 2004). Aerobic conditions in groundwater generally increase virus removal and decreased survival while anoxic conditions have been shown both to increase or decrease viral removal or inactivation (Gordon and Toze, 2003; Jansons et al., 1989, Rauch-Williams, 2023). Virus survival tends to decrease as pH increases (Campos et al. 2001). However, bacteria survival increases as pH increases (EPA 2004). The presence of dissolved or particulate organic carbon may result in longer virus survival in groundwater (Rauch-Williams et al., 2023). Due to their small size, viruses are transported more easily in subsurface material with smaller pores, where larger pathogens would be subject to straining and limited to transport in zones of preferential flow (Rauch-Williams, 2023). The presence of indigenous microorganisms may also cause inactivation and predation of viruses and *E. coli* (Gordon & Toze 2003; John & Rose 2005; Quanrud et al. 2003). Finally, virus type affects inactivation rate; virus survival varies for different viruses under the same conditions (Straub et al. 1992).

Phenomena related to climate change may influence virus fate and transport in recharge systems. Some factors are changes in localized precipitation amounts affecting surface water distribution and quality and natural and human-made recharge and changes in groundwater pumping; and fracturing of aquifers with potential for pathogen transport into groundwater (Rauch-Williams, 2023).

Groundwater contamination by viruses has primarily occurred with poorly treated, sometimes non-disinfected sewage (Vaughn et al. 1978). Table 2 summarizes the virus reduction and transport achieved in treated wastewater SAT field and laboratory studies. Virus removal from groundwater usually occurs in two phases, with greater removal in the shorter, initial phase and less, but sustained, removal in the later, longer phase (Drewes, 2024). Multiple laboratory studies have also shown complete removal of enteroviruses or MS-bacteriophage over relatively short distances (i.e. Trussel et al. 2015, 2017, Betancourt,et al. 2019). Betancourt et al. (2014), investigated three active MAR sites and concluded that complete enterovirus removal occurred within 15 days of aquifer retention time. The farthest vertical distance pathogenic enteroviruses traveled in any of the field studies was 37 feet in a coarse-textured soil (Vaughn et al. 1978); the maximum horizontal distance was 150 feet (Gerba, 1999). In finer-textured soils, enteroviruses travel much shorter distances (e.g. less than 5 ft.) (Duboise et al. 1976). Recently, Morrison et al., 2020, observed subsurface transport of adenoviruses and a bacteriophage (crAssphage) through an approximately 100 of vadose zone whereas enteroviruses were completely removed, indicating that enterovirus transport may more limited that other virus types. Similar results were replicated at this site in WRF, 2023.

Table 2. Summary of Field and Laboratory Studies of Virus Transport in Soils

Source	Study Characteristics	Virus and concentrations	Results
Jansons, et al. 1989	HLR: 0.9 – 30 ft/day; avg 14 ft/day	Echo virus 11	Max vert. transport: 30 ft.
	5-day wetting/ 9 day drying cycle		Max horiz. transport: 46 ft.
	Effluent: SE	Echo viruses 14,24,29,30	Max vert. transport: 10 ft.
	Soil: sand	Cocksackie virus B4	

Source	Study Characteristics	Virus and concentrations	Results
	Location: Canning Vale, Australia	Adenovirus 3	
		Poliovirus 1	Max vert. transport: 6.5 ft.
		Echovirus 6; Poliovirus 3	Max vert. transport: 3.3 ft.
		Reovirus	
		Poliovirus 2; Cocksackievirus B5	Max vert. transport: 1.7 ft.
Powelson, et al. 1993	IR: 3.3 – 3.9 ft/day	Bacteriophages PRD1	At least 99% removal by 15 ft with low infiltration rate. Higher infiltration rates resulted in greater virus transport.
	7-day wetting/ 7-day drying		
	Effluent: SE		
	Soil: coarse sand and gravel		
	Location: Tucson, AZ		
Gerba, et al. 1991	HLR: 0.886-1.083 ft/ day	Bacteriophage PRD1	Max vert. transport: 10 ft.
	Effluent: chlorinated SE		
	Soil: coarse sand and gravel		Max horiz. transport: 150 ft
	Location: Tucson, AZ		
Duboise, et al. 1976	HLR: N/A	Enteroviruses	Max vert. transport: 4.6 ft.
	Effluent: dechlorinated SE		
	Soil: loams to clays		
	Location: Kerrville, TX		
Vaughn, et al. 1978	HLR: N/A	Echovirus 12	Max vert. transport: 37 ft.
	Effluent: SE		
	Soil: coarse sand/fine gravel		Max horiz. transport: 10 ft
	Location: East Meadow, NY		
Gilbert, et al. 1976	HLR: 0.9 ft/day	Indigenous viruses at 1x 10^3 to 7 x 10^3 pfu/ 100L in municipal effluent	99.99% removal in 10-20 ft
	Effluent type: SE		
	Soil type: fine loamy sand		
	Location: Arizona		
Van Cuyk, et al. 2004	HLR: 0.016-0.089 ft/day	Spiked STE with indicator viruses at approximately 1 x 10^5 pfu/mL	99.9% removal in 2-ft
	Effluent type: STE		
	Soil type: medium sand		
	Location: Colorado		
Nicosia, et al. 2001	HLR: 0.105 and 0.207 ft/day	Spiked STE with bromide and indicator virus at 6.0 x 10^{10} or 1.6 x 10^{11} pfu/mL	96% removal at lower HLR
	Effluent type: STE		
	Soil type: fine sand		99% removal at higher HLR in 2-ft
	Location: Florida		
	HLR: 0.098 ft/ day		99.17% removal in 1-ft

Source	Study Characteristics	Virus and concentrations	Results
Higgins, et al. 1999	Effluent type: STE	Indigenous viruses at 3 x 10^3 pfu/mL and 7.8 x 10^3 in STE	98.45% removal in 2-ft
	Soil type: medium sand		99.79% removal in 5-ft
	Location: Massachusetts		
Oakley, et al. 1999	HLR: 0.027-0.217 ft/day	Indigenous virus at 1 – 1 x 10^4 pfu/mL	100% removal in 2-ft
	Effluent type: STE		
	Soil type: clay loam		
	Location: California		
Elkayam et al., 2015	HLR: 35 million m3/year 1 day of flooding and 1–3 days of drying		
	Effluent type: SE, no disinfection		
	Soil type: coastal sand dunes; approximately 30 m vadose zone (17 days)	Fecal coliform average = 1.9 × 105	ND all samples at all wells (accounting for false positives)
	Location: Tel Aviv, Israel		
Morrison et al. 2020	IR: basin assoc with Well EW-008A: 1.6 ft/day, basin assoc with Well WR-069B: 0.75 ft/day, no basin assoc with well WR-398A	crAssphage average 4.4 log10 gc/l	53% removal in 83.1 ft, >62% removal in 45.9 and 40 ft
	Effluent type: TE	Adenovirus average 3.6 log10 gc/l	>53% removal at 83.1 ft, >54% removal at 45.9 and 40 ft
	Soil type: not described	Enterovirus ND all samples	ND all wells
	Location: Tucson, AZ		
Betancourt et al. 2014	IR: Not stated	Adenovirus 1.07 - 3.22x10^5 copies/L, Enterovirus 3.19x10^3 - 5.27x10^4 copies/L, Aichi virus 1.05 - 4.73x10^4 copies/L, PMMoV 5.84x10^5 - 8.99x10^6 copies/L	All viruses decreased in wells by 90-99% with 5 and 15 days travel time. PMMoV detected in wells adjacent to river.
	Effluent type: activated sludge treatment, chlorination/dechlorination		
	Soil type: alluvial sand with some gravel and silts		
	Location: Prairie Waters Project, Brighton, CO		
	Residence time: 5 to > 15 days		
	Travel distance: 100-300 ft		
Betancourt et al. 2014	IR: ave. 3 ft/day	Adenovirus 9.37x10^3 copies/L, Enterovirus 3.46x10^4 copies/L, Aichi virus 4.76x10^4 copies/L, PMMoV 5.15x10^6 copies/L	Ademovirus and Enterovirus not detected in groundwater. Aichi virus and PMMoV reduced by 70% at 5 days travel time, not detected at 14 days travel time
	Effluent type: biotowers (trickling filter), chlorination-dechlorination		
	Soil type:coarse and and sandy gravel		
	Location: Tucson, AZ		

Source	Study Characteristics	Virus and concentrations	Results
	Residence time: 5 to 15 days		
	Travel distance: not stated		
Betancourt et al. 2014	IR: 2-3 ft/day	Adenovirus $8.07x10^{1}$ copies/L, Enterovirus $6.60x10^{1}$ copies/L, Aichi virus $6.60x10^{1}$ copies/L, PMMoV $6.60x10^{1}$ copies/L	Only Adenovirus detected in treated effluent, only PMMoV detected in wells with 3 or fewer days of travel time
	Effluent type: activated sludge (nitrification/denitrification), tertiary dual-media filtration (anthracite and sand, chlorination-dechlorination		
	Soil type: Not stated		
	Location: Los Angeles County, CA		
	Residence time: 0.45 to 128.5 days		
	Travel distance: not stated		
Betancourt et al. 2019	HRT: 0.5 days at 0.5 ft; 1 day at 1 ft; 2 days at 2 ft; 3 days at 3 ft; 6.1 days at 6 ft; 10.4 days at 10 ft; 15.3 days at 14.1 ft; 15.4 days at 14.4	Initial concentrations not stated	PMMoV 37% at 0.5 ft, 99,97% at 14.1 ft; Aichi virus 75% at 0.5 ft, 99.92% at 1 ft; Reovirus and Enterovirus/Adenovirus not detected at 0.5 ft
	Effluent type: non-disinfected secondary treated wastewater effluent/nonofiltration permeate (50:50 blend)		
	Soil type:95% sand, 4% silt, 1% clay		
	Location: column study with saturated, anoxic conditions		
	Residence time: 15.4 days		
	Travel distance: 14.4 ft		
WRF 2023	HRT: 1.1 ft/day	Mean log concentrations: Aichivirus 2, CrAssphage 3, Norovirus GI 2, Norovirus GII 3, Adenovirus 2, Reovirus 2, Enterovirus 2, PMMoV 3	Mean log concentrations: Aichivirus ND, CrASSphage ND, Norovirus GI ND/1, Norovirus GII ND, Adenovirus ND, Reovirus ND/-7, Enterovirus ND, PMMoV 1
	Effluent type: tertiary treated		
	Soil type: sandy loam		
	Location: Facility B Tucson, AZ		
	Residence time: could not be estimated		
	Travel distance: not stated		
WRF 2023	HRT: not stated		

Source	Study Characteristics	Virus and concentrations	Results
	Effluent type: tertiary treated Soil type:sand and gravel Location: Facility C Aurora, CO Residence time: 10 days Travel distance: not stated	Log_{10} GC/L: Aichi virus 4.1, Adenovirus 4.5, Norovirus GI 3.8, Norovirus GII 3.5, Enberovirus 4.9, Reovirus 3.7, Human Bocavirus 2.4	Log_{10} GC/L:Aichi virus ND, Adenovirus ND, Norovirus GI ND, Norovirus GII ND, Enterovirus ND, Reovirus 0.7, Human Bocavirus 0.2

Key: HLR – hydraulic loading rate, IR – infiltration rate, MPN – most probably number, SE – secondary effluent, STE – septic tank effluent, TE -tertiary effluent, pfu – plaque forming unit

SAT Removal of Organic Carbon

SAT removal of organic carbon occurs by biodegradation and sorption, with biodegradation being the primary mechanism, depending on the compound, under both aerobic and anaerobic conditions (Quanrud et al. 2005). Biodegradation occurs through redox reactions and may involve mineralization of organic compounds as they are oxidized by bacterial enzymes, or biotransformation as they are transformed into a lower-energy compound (Murray, 2020). Sorption (adsorption) is limited by site-specific characteristics of the source water, soil, and aquifer material, involving electrostatic and van der Walls attachments. As such, biodegradation is a sustainable and infinite process, whereas adsorption is finite and unsustainable over time (Murray 2020, Quanrud et al. 2005). Redox conditions in the vadose zone control the levels and types of microbial activity and microbe-dependent reactions. Organic carbon removal occurs quickly under unsaturated conditions (over days) and more slowly under saturated conditions, but total removal rates are similar under both conditions over a typical multiple-month SAT system residence time (Quanrud et al., 2005).

Quanrud (2005) reviewed work on recharge basin treatment of effluent and riverbank filtration of surface water demonstrating the potential for SAT removal of natural organic matter from Central Arizona project (Colorado River) water. This work included a focus on reduction of organic carbon to reduce formation of disinfection byproducts (DBPs), and the various forms of natural organic matter (NOM) contributing to DOC. Findings included:

- The highest organic carbon and nutrient removal rates by SAT typically occur between 0 to 5.0 feet below ground surface (bgs), where bacteria are most active and some carbon forms are filtered out by attachment to soil particles.
- Fox et al. (2000) investigated water quality at eight effluent recharge sites in Arizona, California, and Texas and found that most DOC was removed between the ground surface and 10 feet bgs and over time, the removal was similar for saturated and

unsaturated conditions. DOC removal rates up to 90 percent with DOC values of 1-2 mg/L in the recovered groundwater were observed at multiple sites.

- Riverbank filtration (RBF) investigations in Europe (i.e. the Rhine river) document century long water treatment with long-term biological degradation shown to be responsible for most TOC removal. A risk of potential release of sorbed hydrophobic organics during stagnation or low flow periods, as well as the potential for release of DOC during SAT site wet-dry cycling in basins was noted.
- Riverbank filtration research in the United States found DOC removals of 35-67 percent accompanied by a 50-80 percent decrease in DBP formation potentials.

These and other studies indicate that treated wastewater effluent DOC/TOC is composed of antecedent NOM from surface water/groundwater and biodegradable organic compounds and synthetic trace organics derived from wastewater; the former is removed via SAT, whereas refractory NOM not removed by SAT is associated with the original drinking water source (WRF 2023, Trussel et al., 2018; Amy and Drewes, 2007)).

Table 3 summarizes other laboratory and field studies conducted from the 1970s through the present that measured DOC/TOC and nitrogen removal rates via SAT. SAT in surface spreading basins or RBF provides sustainably reduces DOC/TOC levels by 35% to greater than 90% depending on the vadose zone depth and aquifer storage period. Except for one study which used chlorinated effluent, DOC/TOC removal rates of greater than 50% were consistently observed in studies that sampled recharged water within 5 to 10 feet bgs. DOC/TOC removal rates typically increase with increasing vadose zone thickness and/or short-term transport (i.e. weeks to months) in saturated aquifer material to rates in excess of 80%. For example, Murray (2020) described three facilities that achieve 75% (Northwest Water Reclamation Plant); 95% (Dan Region Reclamation Project (Shafdan), and 93% (Tucson Water Sweetwater Recharge Facilities (SWRF)) reductions in DOC/TOC through vadose zone depths of 5 ft, 60 to 120 ft, and 80 to 120 ft respectively (Table 3).

Multiple studies (i.e. Bouwer and Rice, 1984, Murray 2020) also indicate that organic carbon removal can be enhanced via wetting and drying cycles optimized to soil conditions on site. Periodic basin drying can be used to increase biodegradation under aerobic conditions in finer-grained soils. Moreover, unsaturated conditions (and aerobic biodegradation) may exist in sandier soils below flooded basins for extended periods if a clogging layer reduces the operational infiltration rate (Hutchinson et al., 2013, Hutchinson et al., 2017).

Organic carbon removal rates via SAT are also affected by the type of wastewater treatment. Several field and laboratory SAT studies indicate high DOC/TOC removal rates through short vadose zones (i.e. < 10 feet) for both primary and secondary treated effluent, though final DOC/TOC values were lower with secondary treated effluent (Bouwer and Rice, 1984; Rice and Bouwer 1984; Bouwer 1991). Column studies by Trussell et al. (2015) observed increased TOC removal with ozonated effluent compared to chlorinated effluent in 12-foot unsaturated columns, similar results were shown by Echigo et al., (2015) with 5-foot columns. Of note, disinfection of treated effluent prior to recharge is not practiced in either Israel or the Netherlands (Elkayam et al., 2015, Zuurbier et al., 2018).

Of note, the Trussel et al., 2017 study observed only minor decreases in TOC removal in columns mimicking saturated conditions after initial SAT in unsaturated columns (Table 3). These results conflict with multiple field studies that show additional DOC/TOC removal with relatively short residence times in groundwater (i.e. Hutchinson, 2023; Korich, 2023; Valhando, 2019, Murray, 2020; Fox et al., 2000).

Table 3. Summary of Field and Laboratory Studies of SAT Removal of Organic Carbon and Nitrogen Species

Source	Study Characteristics	Source Water Concentrations[1]	Groundwater Concentrations	Removal (%)
Bouwer and Rice, 1984	Location: 23rd Avenue Project, Phoenix, AZ	TOC 10.2 mg/L	1.9 mg/L	81
	Source water: secondary (activated sludge process) effluent from 23rd Avenue Sewage Treatment Plant in Phoenix, chlorinated			
	Configuration: Four 10-acre rapid-infiltration basins	Total N 18 mg/L	5.56 mg/L	69
	Soil: loamy sand over two-thirds of basin area with coarse sand with gravel and boulders and coarse sand alone in patches			
	Vadose zone depth: 10-65 ft varying with nearby Salt River flows and local pumping	Organic N 4 mg/L	0.15 mg/L	96
	Wet-dry cycling: 2 weeks wet, 2 weeks dry			
	Infiltration rate: 0.14-0.28 feet/day			
	Basin surface treatment: ripped to 4 inches bgs			
Rice and Bouwer, 1984	Location: Mesa Wastewater Treatment Plant, Mesa, AZ	Organic carbon 24 mg/L in PT effluent	13 mg/L in soil	45
	Source water: primary treated (PT) and secondary treated (ST) effluent	Organic carbon 19 mg/L in ST effluent	6 mg/L in soil aquifer	68
	Configuration: Constructed 3 m x 9 m infiltration basins (field study) and 2.75-m columns (lab study)	Organic N 4.3 mg/L in PT effluent	1.6 mg/L in soil	63
	Soil: ranging from loam sand to sandy loam	Organic N 4.0 mg/L in ST effluent	0.6 mg/L in soil aquifer	85
	Depth of travel: 3 feet	Organic carbon 75 mg/L in PT effluent	7 mg/L in soil column	91
	Wet-dry cycling in basins: 1 week wet, 2 week dry November - February; 1 week wet, 1 week dry March-October	Organic carbon 15 mg/L in ST effluent	7 mg/L I soil column	53

Source	Study Characteristics	Source Water Concentrations[1]	Groundwater Concentrations	Removal (%)
	Infiltration rate in basins: 3-9 feet/day	Organic N 5.0 mg/L in PT effluent	<1.0 in soil column	approx 100
	Basin surface treatment: scarification to 9 inches bgs	Organic N 2.0 mg/L in ST effluent	<1.0 in soil column	approx 100
Bouwer, 1991	Location: Flushing Meadows project installed in Salt River bed near Phoenix, AZ	Total N 27.4 mg/L	9.6 mg/L	65
	Source water: primary treated (PT) and secondary treated (ST) effluent			
	Configuration: Six parallel long, narrow infiltration basins 0.3-ac each			
	Soil: loamy sand underlain by sand and gravel layers	TOC 10-20 mg/L	5 mg/L	50-75
	Vadose zone depth:10 feet			
	Wet-dry cycling: 9 days wet, 12 days dry			
	Infiltration rate: 1-2 feet/day			
	Basin surface treatment: Not described			
Fox et al., 2000	Location: Mesa Northwest Water Reclamation Plant, Mesa, AZ	DOC 5.6 mg/L	2.2 mg/L	61
	Source water: activated sludge treatment with nitrification/denitrification, sand filtration and chlorination			
	Configuration: four recharge basins with total area of 30 acres			
	Soil: well or poorly graded sands or gravels, silty sand or gravels and silts, clayey sand or gravels and clays			
	Vadose zone depth: 10-20 ft			
	Wet-dry cycling: yes, timing specific to basin fill/drainage times	NO3-N 1-10 mg/L	1.3 - 2.1 mg/L	varies
	Infiltration rate: 0.22-0.79 ft/day			
	Basin surface treatment: not described			
Fox et al., 2000	Location: Sweetwater Underground Storage and Recovery Facility, Tucson, AZ	DOC 12-15 mg/L basin ponded water	5-6 mg/L first 10 feet of basin sediment	50-60
	Source water: filtered secondary effluent			
	Configuration: Eight infiltration basins (28 acres) with four on each side of Santa Cruz River			

Source	Study Characteristics	Source Water Concentrations[1]	Groundwater Concentrations	Removal (%)
	Soil: recent alluvium with gravel, sand and silt above clayey silt and mudstone to sand and gravel, clay lenses beneath basins			
	Vadose zone depth: > 100 feet	NO3-N source water for basins - value not stated	< 10 mg/L below 80 ft	>50
	Wet-dry cycling: varies, including 5 wet-7 dry, 2 wet-4 dry			
	Infiltration rate: average 2 feet/day decreasing over time to 1 foot/day			
	Basin surface treatment: annual plant removal, biennial disking (top 12 inches), ripping (top 30 inches) approximately every 6 years			
Korich, D. 2023, personal communication	Location: Sweetwater Recharge Facilities, Tucson, AZ	TOC 8.0 mg/L	1 mg/L	88
	Source water: tertiary-treated effluent as of June 2020	TKN 3.14 mg/L	0.68 mg/L	78
Fox et al., 2000	Location: Tres Rios Cobble Site in Tolleson, AZ	DOC 7.6 mg/L	3 mg/L shallow monitoring wells	61
	Source water: primary treatment,nitrified-denitrified effluent			
	Configuration: Two wetland basins, each 904 feet by 115 feet, with one "leaky" and one lined	Total N 6.0 mg/L	average decrease of 2.3 mg/L	38
	Soil: river run (cobble, gravel, sand) in leaky basin, 15-20 cm topsoil in lined basin			
	Vadose zone depth: up to 20 ft	NO3-N 2.8 mg/L	Mostly < 1 mg/L	mostly > 64
	Wet-dry cycling: not applicable			
	Infiltration rate: 0.72 feet/day (fully recharging wetland)			
	Basin surface treatment: not described			
Fox et al., 2000	Location: Rio Hondo Spreading Grounds, Los Angeles County, CA	DOC 5.75-18.5 mg/L	<2 mg/L	minimum 65-89
	Source water stormwater, imported surface water, reclaimed water via San Gabriel and Rio Hondo Rivers			
	Configuration: 20 basins, 10-20 feet deep, approximately 570 acres, single basin used for study			
	Soil: gravelly sand with thin discontinuous beds of fine-grained sediment in study basin			
	Vadose zone depth: 10-35 feet	NO3-N <4.66 mg/L-N	<6 mg/L-N	increased
	Wet-dry cycling:			
	Infiltration rate 2.11 feet/day			

Source	Study Characteristics	Source Water Concentrations[1]	Groundwater Concentrations	Removal (%)
	Basin surface treatment:			
Laws, et al., 2011	Location: Montebello Forebay, Los Angeles, CA Source water: tertiary treated wastewater Configuration: surface-spreading recharge basin, 6637 ft^2	TOC 7.8 mg/L	3.54 mg/L	55
	Soil: fine to coarse sand with clay lens 31 ft below basin Vadose zone depth: 8 ft Wet-dry cycling: none	NH_3 0.47 mg/L	<0.1 mg/L	near 100
	Infiltration rate: 2-3 ft/day Basin surface treatment: not stated	NO3 3.9 mg/L		slight increase
Abel, 2012	Location: column study Source water: primary effluent	DOC average 42.5 mg/L	approx 22 to approx 47 mg/L from 5° C to 25° C	17.7 -54.5 from 5° C to 25° C
	Configuration: Four 10-ft high double-walled soil columns at different temperatures to examine the influence of temperature variation and redox conditions on organics and nutrient removal Soil: silica sand Vadose zone depth: 10 ft	NH_4 32 average mg/L	not stated	8.8 - 99 from 5° C to 25° C
	Wet-dry cycling: not stated Infiltration rate: Basin surface treatment:	NO_3 average 0.7 mg/L	not stated	slight increase to 23.3 from 5° C to 25° C
Trussell et al., 2015	Location: Laboratory column study	DOC 6.0 mg/L chlorinated	3.7 mg/L	38
	Source water: chlorinated tertiary effluent and ozonated secondary effluent	NO3-N 2.93 mg/L chlorinated	0.37 mg/L	87
	Configuration: Two multi-column systems for each source water type (Cl2 and O3) simulating upper unsaturated and lower saturated zones; column: 4 m length, 14 cm diameter	TN 3.73 mg/L chlorinated	0.37 mg/L	90

Source	Study Characteristics	Source Water Concentrations[1]	Groundwater Concentrations	Removal (%)
	Soil: < 2mm, representative soil collected from a quarry in the same aquifer as the proposed spreading grounds, fine to medium grained sand with silt			
	Vadose zone depth: not applicable	DOC 5.4 mg/L ozonated	2.3 mg/L	57
	Wet-dry cycling: not described	NO3-N 3.18 mg/L ozonated	0.63 mg/L	80
	Infiltration rate: 5-day residence time vadose zone columns, 25- day residence time for saturated column	TN 3.68 mg/L ozonated	0.64 mg/L	83
	Basin surface treatment: not applicable			
Trussell et al., 2017	Location: Laboratory column study	TOC 8.14 mg/L	3.76 mg/L in vadose zone plus 1-month saturated	54
	Source water: Los Angeles Department of Water and Power recycled water		3.32 mg/L in vadose zone plus 6-month saturated	59
	Configuration: One column mimicking vadose zone with cyclic water application, two columns mimicking saturated zone with 1- and 6-month retention times, respectively; columns 12 feet length, 8-inch diameter	NO3-N	Not stated	35 in 1-month column, 88 in 6-month column
	Soil: < 2mm, representative soil sample purchased from mining ground within the vicinity of the spreading grounds, 2% clay, Ksat 3.5e10^{-4}, 0.3% organic content			
	Vadose zone depth: not applicable			
	Wet-dry cycling: not described	TN	Not stated	42 in 1-month column, 90 in 6-month column
	Infiltration rate: vadose zone column residence time 5 days, saturated column residence time 26 days			
	Basin surface treatment: not applicable			

Source	Study Characteristics	Source Water Concentrations[1]	Groundwater Concentrations	Removal (%)
Hutchinson, 2017 (data to 2016)	Location: OCWD Riverbed Filtration System (RFS), Fountain Valley, California	TOC 9.3 mg/L influent to RFS	4.8 mg/L effluent from RFS	48
	Source water: Santa Ana River water containing treated wastewater			
	Configuration: subsurface collector system placed approximately one meter below the surface with filtered water from the collector system is conveyed by gravity to the receiving recharge basin.	DOC 8.2 mg/L influent to RFS	4.3 mg/L effluent from RFS	48
	Soil: sand and gravel			
	Depth of travel: 3 feet in RFS	TKN 0.8-0.9 mg/L	Not stated	> 99
	Wet-dry cycling: not applicable			
	Infiltration rate: 2-5 ft/day			
	Basin surface treatment: not applicable			
Hutchinson, personal communication, 2023 (data from 2019)	Location: OCWD Riverbed Filtration System, Orange County, California	TOC 6.05 mg/L influent to RFS	2.34 mg/L monitoring well near recharge basin	61
	Source water: Santa Ana River water containing treated wastewater			
	Configuration: subsurface collector system placed approximately one meter below the surface with filtered water from the collector system is conveyed by gravity to the receiving recharge basin.			
	Soil: sand and gravel			
	Depth of travel: 3 feet,10-foot vadose zone in recharge basin			
	Wet-dry cycling: not applicable			
	Infiltration rate: not applicable			
	Basin surface treatment: not applicable			
Valhondo, et al., 2020	Location: Nord-East Spanish Mediterranean coast	DOC approx 17 mg/L	approx 8.5 mg/L at outlet	approx 50
	Source water: secondary treatment effluent			
	Configuration pilot recharge systems for WWTP facility: 50 x 50 ft^2, 5 ft deep with 8 ft x 50 ft long canals to emulate an aquifer with overlying 4 x 4 ft long box to emulate an infiltration basin.			
	Soil: homogeneous fine sand	NH_4 approx 80 mg/l	approx 10 mg/L at outlet	approx 88
	Depth of travel: 5 ft			
	Wet dry cycling: Four recharge periods from 27 to 118 days		approx 38 mg/L	increase

Source	Study Characteristics	Source Water Concentrations[1]	Groundwater Concentrations	Removal (%)
	Infiltration rate: 1.3 ft/day	NO3 approx 1 mg/L		
	Basin surface treatment: none			
Sallwey, et al., 2020	Location: column study	NH_4 0.4-0.7 mg/L	removed within first 3 ft of soil	100
	Source water: tertiary treated wastewater combined with rainwater			
	Configuration:20 ft high, 0.5 ft diameter column packed with different soil horizons of sand with silt/clay: 95% sand to 8.5 ft deep, 100% sand to 15 ft deep, 87% sand to bottom			
	Soil: see Configuration	NO_3 25-35 mg/L	26-24	increase
	Vadose zone depth:20 ft			
	Wet-dry cycling: variable with wetting for 60-120 minutes and drying from 100 to 720 minutes			
	Infiltration rate:	DOC 5-11 mg/L	approx 2 mg/L	5-42%
	Basin surface treatment: not applicable			
Dziura, 2020	Location: column study	TOC <4 mg/L	not stated	45-50 in 3-day columns, no additional in 1-month columns
	Source water: secondary clarifier effluent			
	Configuration: Two pairs of two columns at 3 days and 1 month of travel time through aerobic to anaerobic conditions.			
	Soils: from well construction, washed to remove clay and drilling mud, sieved to remove particles > 4mm	TN <12 mg/L	not stated	approx 3 in 3-day columns, up to 29 in 1-month columns
	Vadose zone depth: not applicable			
	Wet-dry-cycling: not applicable			
	Infiltration rate:			
	Basin surface treatment: not applicable			
Murray, 2020	Location: Sweetwater Recharge Facilities, Tucson, AZ	DOC 14.1 mg/L	0.98 mg/L	93
	Source water: chlorinated non-nitrified secondary effluent			
	Configuration: Eight infiltration basins over 20 acres along Santa Cruz River			
	Soil: sandy loam with porosity of 0.39			

Source	Study Characteristics	Source Water Concentrations[1]	Groundwater Concentrations	Removal (%)
	Vadose zone depth: 121 feet	DON 9.4 mg/L	below detection limits	approx 100
	Wet-dry cycling: 3 days wet, 3-4 days dry			
	Infiltration rate: 14 days vadose zone residence time	TN 23.3	not stated	61
	Basin surface treatment: not described			
Korich, D. 2023, personal communication	Location: Sweetwater Recharge Facilities, Tucson, AZ	TOC 8.0 mg/L	1 mg/L	88
	Source water: tertiary-treated effluent as of June 2020	TKN 3.14 mg/L	0.68 mg/L	78
Murray 2020	Location: Northwest Water Reclamation Plant, Mesa, AZ	DOC 6.1 mg/L	1.5 mg/L	75
	Source water: nitrified and denitrified, tertiary effluent			
	Configuration :Four recharge basins over 30 acres	DON 2.0 mg/L	0.2 mg/L	90
	Soil: fine clay lenses			
	Vadose zone depth: 5 feet			
	Wet-dry cycling: none	TN 23.3 mg/L	not stated	20-50
	Infiltration rate: 0.2 - 0.4 ft/day			
	Basin surface treatment: not described			
Murray 2020	Location: Dan Region Reclamation Project (Shafdan), Tel Aviv, Israel	DOC 11 mg/L	0.6 mg/L	95
	Source water: undisinfected secondary effluent			
	Configuration: not described			
	Soil: sand with interspersed sand and clay layers			
	Vadose zone depth: 66-131 feet			
	Wet-dry cycling: 24 hours wet, 48-72 hours dry	TN 2-20 mg/L	not stated	49-83
	Infiltration rate: 13-17 days vadose zone residence time			
	Basin surface treatment: not described			
Wilson et al. 1995	Location: Sweetwqter Underground Storage and Recovery Facility, Tucson, AZ Source water: chlorinated secondary effluent, tertiary effluent Configuration: 14 acres with four spreading basins excavated 10 ft bgs with alluvium extending 17 ft below basin floors Soil: permeable channel alluvium - mainly cobbles and gravelly sand with localized variations of silt and clay	DOC 12-15 mg/L	1.1-1.3 mg/L	90-92

Source	Study Characteristics	Source Water Concentrations[1]	Groundwater Concentrations	Removal (%)
	Vadose zone depth: approximately 120 ft below basin floors Wet-dry cycling: non-uniform but strive for 5-day wet, 7-day dry cycles in cool weather and 2-day wet, 4-day dry cycles in hot weather Infiltration rate: 2.5 to 4.2 ft/day Basin surface treatment: not described	Total N not stated	9 mg/L	57
Amy et al. 1993	Location: Sweetwater Underground Storage and Recovery Facility Source water: one test basin chlorinated secondary, other basin tertiary effluent Configuration: two walled test basins 12 ft x 12 ft constructed within a large newly excavated basin with multiple suction samplers at 1 to 20 ft bgs	DOC not stated	approximately 1-2 mg/L	40-63
	Soil: Vadose zone depth: approximately 110 ft below basin bottoms Wet-dry cycling: approximately 7-day wet, variable dry Infiltration rate: 2.7-19.7 ft/day Basin surface treatment: not described	NO3-N <1 - 8mg/L	7-15 mg/L	increased
Idelovitch and Michail 1984	Location: Dan Region Project, near Tel Aviv Israel Source water: partially treated effluent Configuration: four spreading basins, each divided into 4-5 subbasins, with net area of 60 acres Soil: relatively uniform, fine sand (<0.3 mm) with recovery wells on three sides	DOC 18 mg/L	3.2 mg/L	82
	Vadose zone depth: not stated Wet-dry cycling: 1 day flooding, 2-3 days drying Infiltration rate: 4.9-6.6 ft/day Basin surface treatment: not described	Total N 13 mg/L	7.2 mg/L	45
	Location: Whittier Narrows Groundwater Replenishment Project, Los Angeles County, California	TOC 3.2-16.4 mg/L	1 mg/L	69-94

Source	Study Characteristics	Source Water Concentrations[1]	Groundwater Concentrations	Removal (%)
Nellor, Baird, and Smyth 1985	Source water: treated wastewater blended with local stormwater runoff and imported Colorado River water/State Project water			
	Configuration: 689 acres with basins from 4 to 20 acres			
	Soil: permeable, sandy soil			
	Vadose zone depth: not stated	NO3-N 0.16-1.78 mg/L	2.06 mg/L	increased
	Wet-dry cycling: not described			
	Infiltration rate: approximately 0.9-2.7 ft/day			
	Basin surface treatment: not described			
Brooks, Weisbrod & Bar-Zeev 2020	Location: Shafdan WWTP, Tel Aviv, Israel (mid-scale study)	TOC average 14.3 mg/L	sand 11.9 mg/L, tuff 9.27 mg/L	sand 17, tuff 35
	Source water: sterilized secondary effluent			
	Configuration: Volcanic tuff buried in SAT infiltration basin for maturation/microbial treatment than moved to columns 16 inches long, 4 inches diameter. Compared with basin sand.			
	Soil: tuff, sand	TN average 7.2 mg/L	sand 6.29 mg/L, tuff 4.4 mg/L	sand 13, tuff 39
	Vadose zone depth: 16 inches			
	Wet-dry cycling:12-24 hours wet, 24-72 hours dry			
	Infiltration rate: 3.28 ft/day			
	Basin surface treatment: tuff			
Schmidt, et al., 2011	Location: Harkins Slough in Pajaro Valley Groundwater Basin near Watsonville, California	Pond water Phase 1 NO3-N <=10-350 ɥmol/L, Phase 2 <=50 ɥmol/L	Phase 1 <=10 - 350 ɥmol/L , Phase 2 <=25 ɥmol/L	>=50% Phase 1, >=80% Phase 2
	Source water: Harkins Slough surface water diverted and passed through sand filter			
	Configuration: recharge pond - modified natural depression underlain by clay, 7.4 acres			
	Soil: three discontinuous aquifer units separated by layers of fine silt or clay			

Source	Study Characteristics	Source Water Concentrations[1]	Groundwater Concentrations	Removal (%)
	Vadose zone depth: approximately 1 m to inverted water table			
	Wet-dry cycling: water diverted January - April, any remaining water pumped out June-July			
	Infiltration rate: Phase 1 (first 6 weeks) >=6.5 ft/day, Phase 2 (last 11 weeks) decline to <1 ft/day			
	Basin surface treatment: scraping after water pumped out June-July			

[1] DOC – dissolved organic carbon; DON – dissolved organic nitrogen, calculated as TKN – (NH4-N+NO3-N); TN – total nitrogen; NH4-N – ammonium nitrogen; Organic N – organic nitrogen; TOC – total organic carbon; NO3-N – nitrate nitrogen, TKN – total Kjeldahl nitrogen

SAT Removal of Nutrients

SAT removal of nutrients such as nitrogen occurs through the dual processes of nitrification, under unsaturated conditions where ammonia/ammonium is oxidized to nitrate, and denitrification, under saturated conditions where nitrate is reduced to nitrite and then to gaseous nitrogen (Murray 2020). During infiltration, the upper vadose zone may become anaerobic and molecular oxygen and organic carbon necessary for nitrification can become depleted, nonetheless, most secondary wastewater treatment employs nitrification processes such that denitrification in the vadose zone is more important. As with removal of bulk organics, Fox et al. (2000) found most of the nitrogen removal occurs within 5 ft bgs. In the field and laboratory studies reviewed in Table 3, nitrogen removal ranged from 35% to 100%, in general, longer residence times in groundwater, correspond to higher nitrogen removal.

Phosphorus removal by SAT is primarily by sorption and precipitation with some biodegradation and is should not be considered sustainable. Phosphate removal is less effective in sandy soils than in soils with fines to mediate these reactions (Abel et al. 2012). Multiple authors have reported minor phosphate removal rates, though Bouwer and Rice (1984) observed 93% removal of organic phosphate concentrations over multiple years of monitoring at the 23rd Avenue recharge site, which was attributed to precipitation of calcium phosphate in the aquifer.

Nutrient removal from groundwater will generally increase infiltration rates, due to decreased clogging, but may therefore result in increased transport of viruses downward through the subsurface (Rauch-Williams, 2023).

Contaminants of Emerging Concern (CECs)

Contaminants of emerging concern (CECs) in wastewater include, but are not limited to, chemical compounds in pharmaceuticals and personal care products (PPCPs). There are hundreds of synthetic or natural chemicals that, even at low concentrations in water, may cause toxic effects such as cancers, disrupted endocrine function, reduced fertility, and mortality in aquatic animals and plants and in humans and may bioaccumulate and become magnified through the food chain.

Wastewater treatment may reduce concentrations or remove some CEC compounds but the efficacy of these treatments are limited by the diversity of concentrations and physiochemical properties, such as hydrophobicity, polarity, and surface charge, these in turn affect SAT removal efficiency (Riley 2020, Sunyer-Caldu et al. 2023). For some CECs, SAT has shown the potential to significantly reduce their concentrations by 80 percent or greater, or to remove them to non-detectable levels from concentrations measured in treated wastewater (Yu, Bouwer, and Coelhan, 2006; Laws et al. 2011; Sallwey et al. 2020; Valhonda et al. 2019; Sunyer-Caldu et al. 2023).

During SAT, CECs may be removed from water by either sorption or biodegradation processes. Sorption is reversible under changing redox conditions and some CECs, (e.g. carbamazepine, ibuprofen) appear to have a low sorption capacity (Sallwey et al. 2022). Microbial degradation appears to be the primary driver of CEC removal, as indicated by concentrations of some CECs attenuating only after oxidizing dry periods during wet-dry cycling and not after periods of continuous infiltration (Sallwey et al. 2022).

Soil properties in an SAT system can have critical effects on CEC removal effectiveness. Due to smaller particle size and higher density of reactive sites, high clay content (10 percent or greater) soils provide greater contact time and a larger surface area for reactions between the contaminants and soil and can remove some CECs by between 50 and 90 percent (Riley 2020). Some CECs show increased removal under wet-dry cycling of recharge basins, indicating that wetting and drying of the vadose zone can also influence degradation and removal rates (Sallwey et al. 2020). The presence of soil biofilms, which reduce soil conductivity, can also increase removal rates via sorption and biodegradation, with different effects for different CECs (Munoz-Vega et al. 2023).

Recently, SAT enhancements have been shown to increase CEC removal efficacy. Valhondo et al. (2019) studied reactive barriers consisting of different organic materials (i.e. plant compost) installed within a several constructed SAT cells. The reactive barrier cells demonstrated higher rates of CEC degradation (40 to 100 percent) than the control (no barrier) sites. A related study in the same pilot project system measured concentrations of 56 CECs that showed greater removal rates via the reactive barriers (Sunyer-Caldu 2023). Of note, the maximum travel distance within these SAT systems was through 1 meter and 15 meters of unsaturated and saturated materials, respectively. In an investigation of wet-dry cycling effects (dissolved oxygen concentrations) on the removal of 27 CECs, some compounds were completely degraded under all dry periods tested (60 to 444 minutes), others showed greater attenuation during either shorter or longer dry period tests, and some had poor removals under all test conditions (Sallwey et al. 2022). Arad et al. (2022) tested direct subsurface air injection SAT as an alternative to wet-dry cycling and demonstrated variable removal of three different CECs. Shorter, more frequent dry periods and, potentially, air injection may allow for maximum CEC removal and minimal dry periods that compromise recharge efficiency (Sallwey et al. 2022; Arad et al. 2022). Echigo (2015) demonstrated increased removal of some PPCPs with ozonation of wastewater prior to SAT treatment, with removal percentages greater than 91 percent for carbamazepine, clarithromycin, supiride, and crotamiton. Trussell et al. (2018) observed removals of greater than 70 percent after 150 or 180 days residence time for atenolol, bisphenol a, carisprodol, cotinine, DEET, fluoxetine, gemfibrozil, iohexol, iopromide, meprobamate, TCPP, and TDCPP).

Although definitive classifications of compound properties and potential removal rates have not been determined for any CECs, different studies have observed similar patterns. Preimidone and carbamazepine have been shown in multiple studies to be resistant to degradation, whereas Ibuprofen, atenolol, gabapentin, and naproxen were well degraded (Dzuira 2020, Sallwey et al. 2022, Yu et al. 2006, Arad et al. 2022, Laws et al., 2011). Some CECs appear to attenuate significantly only after having moved into the aquifer (e.g. Ibufprofen, TCEP, TCPP) or after two weeks or more of travel time through the subsurface (e.g. naproxen, diclofenac) (Laws et al. 2011) suggesting degradation is occurring anaerobically. Perfluorinated alkyl substances (aka PFAS) have strong molecular bonds that make those compounds non-degradable, although they may potentially be removed from water by sorption (Dziura 2020).

Conclusions

SAT has been shown via numerous studies to remove pathogens and significantly reduce the concentrations of biodegradable constituents in treated wastewater effluent. Complete removal of microbial pathogens can be expected to occur within feet; viruses have been observed to travel farther, however, the farthest reported horizontal distance from a MAR site in literature for virus travel is 150 feet. Reductions in DOC/TOC concentrations of 50% or greater have been consistently measured over vadose zone depths of 3 to 10 feet. Additional reductions in DOC/TOC can be expected with greater vadose zone depths, and to a lesser extent due to transport in groundwater.

Reductions in nitrogen and phosphate of 35% to 100% can be expected depending on the chemical oxidation state and presence of aerobic or anaerobic conditions in the vadose zone and groundwater system. Removal of nutrients, especially nitrogen, can increase infiltration rates due to reduced clogging but may then result in greater transport of viruses.

The removal of CECs during SAT is highly dependent on the biodegradability of the compound and the SAT operating conditions. Biodegradable CECs can be reduced significantly via SAT, however, recalcitrant compounds such as PFOA/PFOS are not degraded.

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