

Biosand Reactors for Municipal and Industrial Wastewater Treatment: Status Quo, Challenges and Opportunities

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Abstract: Biosand reactors (BSRs), alternatively known as slow or biological sand filters, are passive systems that are used to remove contaminants from domestic wastewater, industrial wastewater and drinking water. This review focuses specifically on their application for remediation of industrial effluent and sewage-containing municipal and household effluent. The relationships between the physicochemical characteristics of the sand grains (size, size distribution, shape, chemical composition) and the hydraulic conductivity of the sand employed in BSFs are critically discussed in relationship to the achievable loading rates and hydraulic retention times. The modes of operation and influence of the functional microbial biomass as well as biodegradable and recalcitrant particulates on these parameters is comprehensively reviewed. Finally, the bioremediation of sewage-based and industrial wastewater is examined. This includes an account of the biotic and abiotic removal mechanisms and the limitations of BSRs for removal of pollutants such as phosphorus/phosphate and nitrates/nitrites. The removal mechanisms and removal efficiencies of macronutrients, micropollutants, fecal indicators and other microorganisms such as antibiotic-resistant bacteria in BSRs are discussed.

Keywords: biodegradation; hydraulic; organic loading; passive; slow sand filter

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1. Introduction

For decades, sand-based systems have been used ubiquitously for improving the quality of raw water for drinking purposes and for remediating municipal greywater and sewage from households or small communities [1–5]. More recently, they have been investigated and applied for remediation of industrial and agri-industrial wastewaters (WW) because of their longevity, low capex and running costs, low energy requirements, tolerance of seasonal input fluxes and ability to retain efficient performance after lengthy shut-down periods [6,7]. Different terms have been used to describe these fixed-film sand-based systems for WW treatment, including biosand filters [2], sand filters [8], biological sand filters [9] and biosand reactors (BSRs) [10]. The term ‘filter’ implies that the primary function is physical filtration, but removal of potential pollutants via substrate binding, precipitation [11,12], physicochemical catalytic breakdown of organics [13,14] and bioremediation [10] can also take place. As with most secondary WW treatment systems, the latter is arguably the most important removal mechanism. The term BSR is therefore used preferentially to describe the array of secondary and tertiary fixed-film sand-based WW treatment systems included in this review.

The main functions of the sand particles in BSRs are to trap solids and provide large surface areas and robust biofilm attachment sites for the microbial communities responsible for biodegradation of pollutants in WW [15]. It has been shown that the extensive rough surfaces of sand grains promote biofilm growth more effectively than synthetic biofilm attachment alternatives [15]. In BSRs, contact between the microbial communities and the pollutants takes place as the WW passively permeates through the interpore

spaces in the sand [16]. In contrast, active fluidization of sand particles is applied to enhance mass transfer between microbial populations and the WW in fluidized bed reactors that can also incorporate sand as a microbial support medium [17].

Similar passive principles of operation apply to BSRs and constructed/treatment wetlands (CW/TWs), most notably those containing sand as the primary matrix/medium [18]. The major difference is that CW/TWs contain plant species that may contribute to WW treatment via phyco-remediation [18]. The functional role of plants is incumbent on many factors, including the mode of operation viz. vertical flow (VF), horizontal subsurface flow (HSSF) or sub-surface flow (SSF), the organic loading rate (OLR), the hydraulic loading rate (HLR), the depth of the CW/TW, the type of WW, the type/s of plant/s and the climate [19,20]. Although there are many benefits associated with the addition of plants, their presence does not necessarily improve removal of solids, organics and/or macronutrients [8,20,21]. In some instances, plants are not desirable, for instance, if the WW is phytotoxic, dead plants will need to be removed to prevent nutrients leaching into the TW/CWs when they decay [19,20]. Another drawback is that the root systems of plants can contribute to surface clogging and/or cause ‘short-circuiting’ of WW due to the creation of preferential flow paths within the sand matrix [22,23].

This review focuses on BSRs applied for secondary and tertiary bioremediation of sewage, septic tank effluent, municipal and industrial wastewater and excludes fluidized bed reactors and CW/TWs, which have already been widely reviewed in the literature. The SCOPUS database was used to select the manuscripts for review using “sand bioreactor” and/or “sand filter” and/or “biosand reactor” and/or “slow sand filter” and/or “sand” and/or “wastewater” and/or “effluent” in the article title, abstract and keywords. Manuscripts describing treatment of raw water for drinking purposes and treatment of irrigation or household grey water were excluded. It must also be noted that the performance of BSRs may be improved by mixing sand with other substrates [24,25]. However, the intent of this review is to compare results from studies using sand as the primary physical substrate, so the literature describing the use of alternative substrates was largely excluded.

2. Hydraulic Performance of Biosand Reactors

During the start-up period, the achievable flow rates in BSRs are incumbent on the intrinsic hydraulic conductivity (HC) of the sand. The HC decreases during operation due to the formation of functional microbial biofilm that protrudes into the pore spaces of the sand matrix [16,26].

2.1. Sand Characteristics and Hydraulic Conductivity

It is challenging to capacitate BSRs during the design process. Not only is it difficult to predict the necessary changes in flow rates due to loss of HC after start-up, but models used to determine the HC of sands are typically based on particle size distributions and assume that the sand grains are perfectly spherical in nature, which is rarely the case [27,28]. The macroscopic features of grains that affect the flow in BSRs are shown in Figure 1. Particles that are ideal for modelling purposes are chemically inert, round, non-porous and uniform in size, in which case there are high positive and negative correlations between the porosity and grain size and grain size distributions, respectively [16,29,30]. If the grain size distribution is uniform, the particles pack together quite consistently and the geometry of the inter-grain pore channels (tortuosity) and hence flow of wastewater through the sand medium are relatively even [31,32]. Conversely, in sands with low uniformity, smaller grains sequentially fill the voids created by larger grains. This results in reduced overall porosity as well as blocking of flow channels (throats) between some grains, especially in the presence of fine particles [16,29,30]. Increased tortuosity created by uneven grain packing may result in the establishment of preferential flow paths and ‘short-circuiting’ [33]. If the flow through the sand matrix is not consistent, the hydraulic retention time (HRT) of the incoming wastewater can vary in different spatial areas, negatively affecting operational performance [16,33]. The situation

becomes more complex when other factors are also considered, most notably the grain shape, porosity and chemical make-up. These can have significant effects on the HC and flow paths of BSRs, rendering models based on particle sphericity inaccurate [16,31–35]. The shape of sand particles depends on their etiology viz. mineralogy, formation mechanisms and effect/s of weathering [32,33] and can be measured with indices such as circularity, curvature, angularity and roundness [16,29,32,36]. Non-spherical particles generally create more random packing arrangements, having a knock-on effect on the tortuosity of the void fraction [32,37–39]. The impact is variable, depending on the type/s of shapes and uniformity of the grains [32]. The mineral composition can also affect the HC of sand in BSRs because hydrated inter-particle bonds can be formed between grains composed of cementitious minerals and/or clays under certain conditions [31]. The chemical bonding processes affecting porosity, tortuosity and HC are complex, and a detailed discussion is beyond the scope of this review. Lastly, compaction processes can affect the HC, and water permeation may be unbalanced if compaction is not homogenous [40].

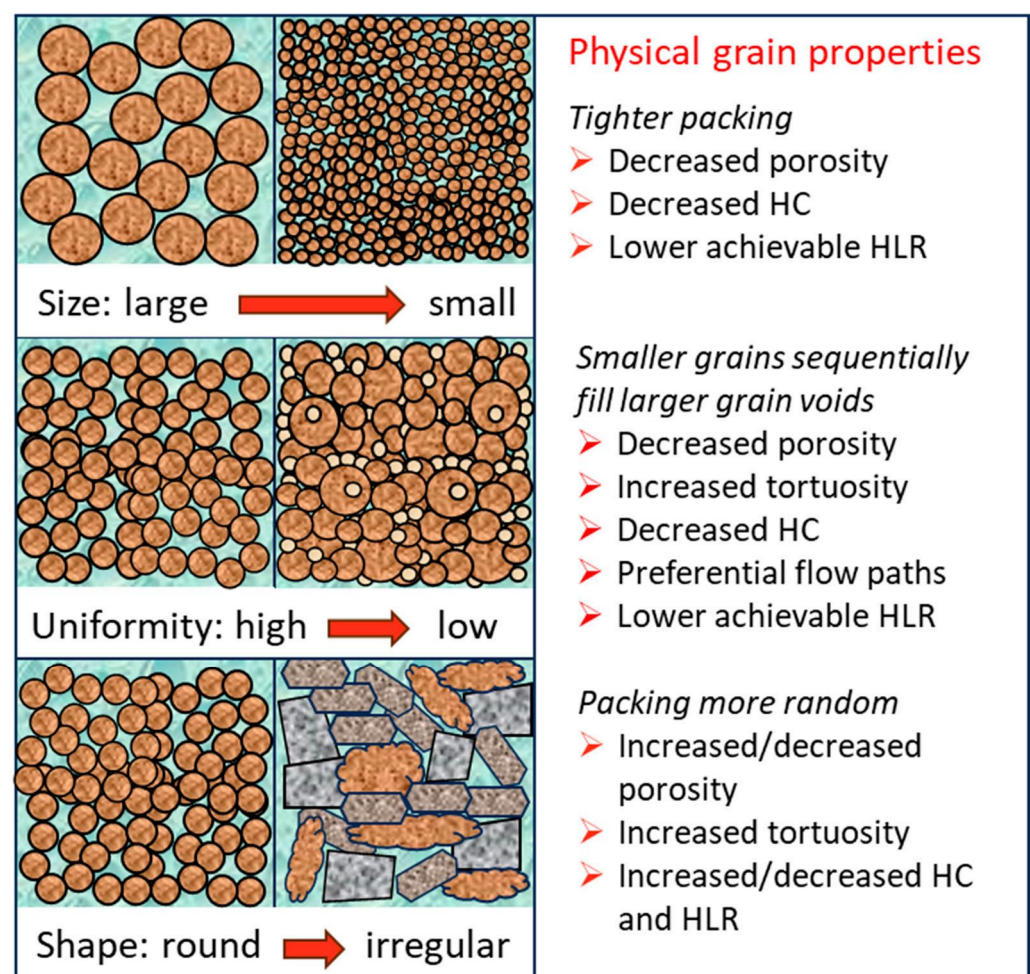


Figure 1. Diagram showing the effects of the primary physical characteristics of grains on the flow properties of biosand reactors. HC = hydraulic conductivity, HLR = hydraulic loading rates.

The gap between theoretical and actual efficiency of BSRs can benefit from applying detailed knowledge of sand characteristics during the design process. Of the thirty-one studies analyzed in this review, six reported on the origin of the sand (one beach sand, two river sand, two dune sand), five reported on the dominant mineralogy of the sand (quartz) and only one reported more detailed mineralogy. In terms of grain size, one manuscript reported only descriptors (fine, medium, coarse sand), six reported the range only, while seven provided only one grain size with no range and did not qualify whether this

was an average or whether the sand had been specifically sieved to obtain this exact sand fraction (which seems unlikely). Two manuscripts included only the d_{10} results viz. the size at which 10% of the grains have a finer grain diameter (ϕ). The thirteen remaining studies all included the average grain ϕ (d_{50}) or d_{60} sizes, and five of these also provided the grain ϕ ranges. Eleven studies reported the d_{60}/d_{10} ratio or uniformity coefficient (U) and two of these did not provide any additional grain data. Notably, only two studies included in Table 1 [10,29] reported particle shape indices. While the d_{50} is a useful parameter, the d_{10} provides valuable information on the fine particles that can block flow paths. The U value indicates how well-graded (uniform) sands are, with an ideal value of 1 if all the grains ϕ are the same [41]. Ideally, all these values should be included as a minimum in manuscripts pertaining to BSRs.

Most BSR experiments have been conducted using laboratory columns (Tables 1 and 2). These are typically gravity-fed and naturally mimic simple VF BSR systems. In the column experiments included in this review, the d_{50} of the grains ranged from 0.3 mm to 1.34 mm [42–44]. In pilot VF and HSSF BSR studies, the d_{50} ranges of the grain ϕ were typically smaller than in the column experiments (Table 1). Only a few studies have compared the effect of grain characteristics on hydraulic and/or operational performance of BSRs. Wang et al. [29] compared two different batches of river sands ($d_{50} = 0.82$ mm and 2.26 mm, $d_{10} = 0.38$ mm and 1.60 mm, $U = 2.78$ and 1.75, for fine and coarse sands, respectively) and crushed aggregates ($d_{50} = 1.60$ mm and 1.36 mm, $d_{10} = 0.44$ mm and 1.60 mm $U = 5$ and 10, for larger and smaller aggregates, respectively). The theoretical saturated HC ($T-HC_{SAT}$) was predicted using porosity results determined from infiltration tests as 0.825–0.953 mm/s, 0.279–0.288 mm/s and 0.749–0.973 mm/s for fine river sand, fine crushed aggregate and coarse crushed aggregate, respectively. Although the same HLR infiltration rates (HLR_{IR}) were applied in the comparative experiments, the large differences in HC resulted in different HRTs in the gravity-fed systems containing the different substrates, confounding comparison of performance results, discussed in more detail in Section 2.2 and Section 3, respectively. In pilot BSR columns packed with sand to 130 cm (h), researchers [45] found that the model optimal HLR_{IR} for 98% removal of *Escherichia coli* was high and similar for sand with grain ϕ of 0.3 mm and 0.5 mm ($5.28\text{--}8.4 \times 10^3$ mm/day and $4.56\text{--}8.64 \times 10^3$ mm/day, respectively), but no other performance observations were made. In another study, researchers found that the measured HC_{SAT} of raw dune sand increased 9-fold, from 0.285 mm/s to 2.50 mm/s by removing all grains <0.425 mm [10]. These results support the recommendation that d_{10} of sand grains in BSRs should range from 0.2 to 0.4 mm [30]. In another study, only low theoretical unsaturated HC ($T-HC_{UNSAT}$) of 0.077 mm/s was measured in gravity-fed columns, which also translated into low HLR_{IR} of 16–83 mm/day [46]. Unfortunately, only the d_{50} (1.2 mm) of the sand used in this study was provided.

The U values for five of the sands used in the BSR column experiments included in this review fell within the range (2–3) for beach sand, while five exhibited lower values, indicating that they were particularly well-graded. However, notably higher values of 5.85 [47] and 4.0 [48] have also been reported. In the latter, the poorly graded fine sand was packed to 46 cm (h) in the columns and then covered with 15 cm of more uniform coarse grains ($U = 1.3$). The high U and low d_{50} of the fine sand translated into low HLR_{IR} (60 mm/day) in full-scale HSSF systems containing the same sand [49]. Despite the poor hydraulic performance, the systems were kept hydraulically functional by rotationally resting BSR modules. In a semi-continuously operated pilot HSSF containing river sand with a U value of 8, porosity of 0.17 and large d_{50} (3.2 mm), another group of researchers [50] also obtained a low HLR_{IR} (110–130 mm/day) and high HRT (1.7 days). The results indicate that sizeable reactors with large spatial footprints would be required to treat the high volumes of effluent expected from industries in BSRs with such low HLR_{IR} .

Depending on the shape, sand containing mainly non-spherical grains may have higher or lower HC values than sand containing spherical grains. For example, higher HC values were predicted for sand containing more concave and less spherical grains [32]. In

a study comparing two sands (river and dune) with similar sizes and grain size distributions in BSR column experiments, significant ($p < 0.05$) differences in $M\text{-}HC_{SAT}$ were obtained (river = 0.154 mm/s and dune = 0.284 mm/s) [16]. It was found that these differences were mainly due to macro- and microscopic differences in the shapes of the grains. The dominant mineral in both sands was quartz, but it was present in higher amounts in the river sand (98%) than in the dune sand (81%), and the quartz grains in the river sand were more porous, more angular and less round than those in the dune sand [16]. The dune sand also contained 18% calcite grains which were more angular and less round than the quartz grains [16].

In summary, high HCs afforded by larger grain ϕ and/or more uniform grain size distributions, among other properties, translate into greater treatment capacities and mitigate against clogging. However, at high HLR, unless there are mechanisms in place to manipulate flow rates [10,26], the HRT may be too low for effective remediation [29]. This is discussed in more detail in Section 2.2. Going forward, it is recommended that researchers always conduct and report all simple grain size analysis results (d_{50} , d_{60} , U and range). If possible, the origin of the sand should be included, and the elemental and mineralogical composition should be determined wherever feasible. Researchers should consider determining shape indices for more in-depth studies. It is highly recommended that $M\text{-}HC_{SAT}$ values be obtained using simulated operational studies to assist in correctly capacitating full-scale BSR systems [16].

2.2. Operation Modes, Hydraulic Loading Rates and Hydraulic Retention Times

Eighteen of the twenty laboratory BSR column experiments reviewed were gravity-fed, while the remaining two were operated in upflow mode using pre-oxygenated WW (Table 1). Upflow mode minimizes bubble formation and allows more effective hydraulic control [51–53]. However, the energy costs of this type of operation in comparison to energy-neutral gravity flow need to be carefully considered at scale. Most of the laboratory column experiments and pilot BSR systems were operated intermittently or semi-continuously to mimic the inflow patterns of the WW in the ‘real world’ or to facilitate oxygenation. It has been shown in CW/TWs and BSRs that ingress of oxygen (O_2) via artificial aeration or intermittent operation promotes aerobic biodegradation and nitrification [42,54]. For instance, Ha and Ong [52] used artificial aeration and effluent recycling to create oxic (5 mg/L dissolved O_2) and anoxic zones for effective nitrification and denitrification, respectively. As with HC, the sand characteristics play a role in oxygenation of BSRs. During intermittent operation of laboratory column BSRs treating synthetic septic tank effluent, researchers found that the O_2 concentrations in the upper 35 cm remained close to ambient in columns containing coarser sand ($d_{10} = 0.8$ mm), while levels dropped to <2% after 10 of 14 cycles in columns containing finer sand ($d_{10} = 0.42$ mm) [40]. The existence of redox gradients within BSRs can be advantageous for removal of organics and total nitrogen (TN), as some organics are degraded preferentially under anaerobic/anoxic conditions, while others are more degradable under oxic conditions [55,56]. In high-COD WW, continuous operation may be preferable when the removal of the organics is higher under anoxic conditions [52,55]. For example, in pilot BSRs treating winery effluent, the HC, HLR and OLR were higher when the BSR systems were operated in continuous mode than when operated in pulse mode [7] (Tables 1 and 2).

Resting BSRs provides time for oxidation of residual organic matter and excess biofilm that could potentially cause clogging [8,26,49]. This can be achieved naturally in systems where WW generation is seasonal in nature [26] or systems can be designed with additional BSR modules that can be rested on a rotational basis [42,49]. It is also recommended that the surface layer be periodically removed or disturbed during resting periods to improve WW inflow [48,49].

The choice of VF or HSSF mode of operation depends on the type and volume of WW being treated. The maximum achievable HLRs in HSSF systems are lower than in VF systems containing the same substrate [7,26]. The hydrodynamics of a full-scale VF BSR was

investigated using computational fluid dynamics and it was established that the WW moved down the BSR in plug flow with no preferential flow paths [57]. Dead zones were found at the base, but constituted <3% of the BSR volume, and were reduced even further by using BSRs with smaller diameters operated in continuous rather than intermittent mode [57]. However, more studies are required, as these results cannot be confidently extrapolated to BSRs containing different sand, for reasons outlined in Section 2.1. If VF systems contain sand with high HC and are not prone to excessive clogging, high HLRs can be achieved (Table 1), which is an ideal scenario when large volumes of WW require remediation [26]. Such systems can also offer a high degree of operational flexibility if there are mechanisms in place to increase the HRT if required, for example, a novel BSR design with an inner and outer chamber filled with sand connected at the bottom via perforations where the height of the outlet, located in the outer chamber, can be adjusted to increase or decrease the hydraulic head according to flow requirements [26]. The capacity to treat large volumes of WW logically also translates into lower spatial, capex and operational requirements. Remediation efficiency can also be improved by decreasing the HLR in BSRs. For example, researchers found that removal rates of total suspended solids (TSS), chemical oxygen demand (COD) and coliforms improved when the HLR_{IR} was decreased from 269 mm/day to 134 mm/day [47]. Ultimately, for each BSR system, there will be an ideal balance between the functional flow rates and the HRT required for effective remediation of constituents present in influent WWs. In passively fed VF systems with no outlet flow control mechanism, higher HRTs can be achieved by compacting the sand [40] or increasing the filter depth [29], but the latter may be impractical at scale and neither of these strategies enable hydraulic flexibility. In addition, lower layers of BSRs may be functionally obsolete in some systems [58].

In the literature, the HLR used in BSRs has been expressed in three different ways. To avoid confusion, these have been defined for the purposes of this review as follows: (i) the HLR_{IR} , which has been standardized to mm/day; (ii) the surface loading rate (HLR_{SUR}), which is the volume of wastewater applied per unit sand surface area per unit time, standardized in $L/m^2_{sand} \cdot day^{-1}$; (iii) the volumetric loading rate (HLR_{VOL}), which is the volume of wastewater applied per unit sand volume per unit time, standardized in $L/m^3_{sand} \cdot day^{-1}$. For BSRs where most of the remediation takes place at the surface, the HLR_{SUR} may be relevant. However, the value will be the same for both deep and shallow systems, so it is useful to also know the HLR_{VOL} in reactors where a significant portion of remediation takes place under the surface. Currently, either the HLR_{IR} or the HLR_{SUR} is standardly applied to describe the HLR in BSRs. Of the thirty reviewed studies, nine reported HLR_{IR} , eleven reported the HLR_{SUR} , while two reported both the HLR_{IR} and HLR_{VOL} , and one reported the HLR_{SUR} and HLR_{VOL} . Of concern is that seven studies provided no HLR values. Given the simplicity of the calculations involved, it would be prudent for researchers to include all three values for readers to properly compare system functionalities. In addition, the overall HLR is skewed if BSRs are not operated in continuous mode, so it is important to also know the HLR during individual dosing episodes to make sure the systems are not transiently overloaded [5].

There were wide ranges in the HLR_{IR} and HLR_{SUR} reported for the laboratory column BSR experiments: from 16 mm/day [46] to 9360 mm/day [59] and 18 $L/m^2_{sand} \cdot day^{-1}$ [53] to 28,800 $L/m^2_{sand} \cdot day^{-1}$ [52]. The latter was operated using sand with large grains and a pump to force the WW upwards through the sand substrates, explaining the high HLR_{SUR} value that was obtained. The lower rates were all reported for intermittently operated systems, but the actual flow rates achieved during feeding were not provided. With continuous gravity-fed operation, rates of 288 mm/day using sand with $\phi = 0.77$ mm [60] and 1200 mm/day using well-graded sand ($U = 1.47$) with high d_{10} (0.86 mm) were reported [61]. Although more comprehensive sand grain characteristics were not included in these manuscripts, from the values that were provided, it may be assumed that the differences in the HLR_{IR} were related to the physical properties of the sand.

2.3. Temporal Changes in Flow Rates

After start-up, the HC and flow rates in BSRs can decrease due to (i) compaction of the sand and/or (ii) the formation of functional biofilm/biomass and/or (iii) collection of organic or inorganic particulates in the pore spaces of the sand matrix. Mechanical compaction may be performed deliberately when systems are being installed [40], but some unavoidable natural compaction also takes place over time [10,40].

The microbial biomass is the most important biotic component of BSRs [18,56]. Although the functional biomass can significantly reduce the HC of BSRs, this should arguably not be labelled as “clogging” unless the HC is reduced to below functional levels [16,18,56]. The biomass-associated reduction in HC is negatively correlated with the OLR because the heterotrophic microorganisms are reliant on organic substrates for growth [16,26,46,54]. Reductions in HC can therefore be reversed during periods of low organic loading [16,26]. There is an inflection point where the loading rates become insufficient to support microbial growth in the deeper layers of BSRs [54]. In systems operated at high OLR/HLR, substrate availability and biomass abundance are found in a continuum from the surface downwards [54,62]. Conversely, in BSRs operated with low HLR, such as those treating household sewage, the biolayer is formed on the surface [2–4,46,54]. This ‘schmutzdecke’ decreases ingress of WW into the substratum but can also increase and stabilize removal performance. In BSRs with high intrinsic HC where flow rates rely purely on gravity percolation, the WW treatment capacity is increased at high HLR, but the HRT may be insufficient for effective bioremediation [54]. This is not problematic in more sophisticated systems where the HRT can be adjusted according to need [26].

Researchers have measured similar losses in HC and/or flow rates due to build-up of functional biomass in BSRs. Reductions of 35–40% in flow rates were found after treating turkey processing WW in laboratory column experiments conducted over 2 years [48]. During remediation of pre-settled winery WW at OLR of 4326 gCOD/m³ sand·day^{−1}, the M-HC_{SAT} was reduced by 56% (from 0.285 mm/s to 0.129 mm/s) and 55% (from 2.50 mm/s to 1.11 mm/s), respectively, in BSR columns filled with raw sand and fractionated sand (≥0.425 mm grains only) [10]. In well-replicated experiments, researchers found that the M-HC_{SAT} of pilot intermittently operated BSFs treating pre-coagulated dairy WW reduced by 40% after 280 days when operated at high OLR (110 ± 10 gCOD/m² sand·day^{−1}) and HLR_{VOL} (20 ± 4 L/m² sand·day^{−1}) [62].

Entrainment of recalcitrant total suspended solids (TSS) may cause ‘true’ clogging which can be avoided or ameliorated by using primary treatment to remove or reduce the TSS before secondary treatment in the BSRs [50,62]. Primary settling is an obvious choice if the particles are settleable within a reasonable period, but other methods may be required if the particles remain suspended. Researchers showed that the reduction in M-HC_{SAT} could be improved from 85% to 40% by pre-coagulating dairy WW (440 mgFe/L dairy WW) to remove particulates [62]. The authors calculated a significantly lower spatial footprint (145 m² vs. 600 m²) and capex and operational costs for the hybrid (pre-coagulation) system (EUR 7920 and 1618, respectively) when compared with the conventional BSR treating raw, uncoagulated WW (EUR 32,845 and 4800, respectively). However, disposal of spent sludge was not accounted for and the possible impact of the increase in Cl after coagulation (155 ± 66 mg/L to 833 ± 145 mg/L) was not discussed.

When assessing the possible effects of TSS on flow rates, it is important to differentiate between organic and inorganic particulates because the latter do not degrade and can irreversibly reduce the HC [26,48,49]. Conversely, particulates in organic-rich WW such as winery WW [26] and turkey processing WW can mineralize within BSRs [49]. Intermittent feeding or resting can provide time for microorganisms to break down organic particulates [8,42,54]. A novel approach for restoring HC in BSRs is the use of burrowing earthworms at the surface to consume excess organic matter and create burrows for water infiltration [63], although it may be argued that the burrows could create preferential flow paths for the incoming WW.

In essence, efficient flow rates in BSRs can be maintained by using properly compacted sand containing grains with desirable flow characteristics, by employing primary removal of TSS relevant to the type of WW and applying operational parameters that have been optimized for each BSR system and type of WW [10,29,30,40,48]. Reduction in HC due to biomass formation should be factored in during the design phase [10]. By applying these considerations, BSRs can remain hydraulically functional for many years with no need to replace the sand or backwash the systems [7,26].

3. Wastewater Remediation Performance

Removal of organic and inorganic molecules in BSRs takes place via three major mechanisms, namely bioremediation, adsorption, and entrainment of WW particulates or precipitates. Biological sand reactors are not recommended for removal of recalcitrant particulates via entrainment because it can lead to irreversible clogging.

3.1. Abiotic Remediation and Removal Mechanisms

3.1.1. Adsorption and Entrainment

Molecules may adsorb onto the surface of sand particles or become part of the bio-films in BSRs. A few studies have shown increased pollutant removal when alternative adsorptive materials are mixed with sand. In laboratory column experiments, for example, biochar has been shown to improve the removal of macronutrients, chemicals, potential pathogens [61], TN [64] and pharmaceuticals [65]. As with entrainment, degradable molecules can be removed by both adsorption and biodegradation [14]. When these mechanisms are combined, increased removal of organics such as phenolics [14] and ammonia (NH₃) [66,67] can theoretically take place.

Table 1. Sand characteristics, operational and hydraulic parameters.

Laboratory Studies: Column Experiments			
Sand Characteristics and Porosity	Reactor Dimensions, Operation Mode and Retention Time	Hydraulic Loading Rates	Ref.
Beach sand: 14.2% FS, 66.5% MS, 18.2% CS DNP (size)	Columns: $\varnothing = 25$ cm, sand $h = 60$ cm Gravity-fed, intermittent operation (1:3 (feed:rest) days)	* $HLR_{SUR} = 26 \text{ L/m}^2_{\text{sand}} \cdot \text{day}^{-1}$ * $HLR_{VOL} = 8.4 \text{ L/m}^3_{\text{sand}} \cdot \text{day}^{-1}$	[8]
Grain $\varnothing d_{60} = 1.07$ mm, $d_{10} = 0.49$ mm, $U = 2.19$	Columns: $\varnothing = 25$ cm, sand $h = 20$ cm Gravity-fed, intermittent operation at 0.5 L/h during feeding	NDP	[68]
Grain $\varnothing d_{50} = 1.34$ mm (1.0–2.0 mm) * Fine ($d_{10} = 0.25$ mm) vs. coarse ($d_{10} = 0.86$ mm)	Columns: $\varnothing = 15$ cm, sand $h = 80$ cm Gravity-fed, intermittent operation	$HLR_{IR} = 70$ mm/day	[43,44]
Quartz sand grain $\varnothing = 0.210$ – 0.297 mm Porosity 0.36	Columns: $\varnothing = 5$ cm, sand $h = 29$ cm Upflow continuous mode, HRT 4–35 h	* $HLR_{SUR} = 288 \text{ L/m}^2_{\text{sand}} \cdot \text{day}^{-1}$	[51]
Grain $\varnothing = 0.7$ mm	Columns: $\varnothing = 4.4$ cm, sand $h = 23$ cm Gravity-fed, continuous mode	$HLR_{IR} = 288$ mm/day	[60]
Quartz sand grain $\varnothing d_{50} = 1.2$ mm	Columns: $\varnothing = 10$ cm, sand $h = 100$ cm Gravity-fed, intermittent operation	$HLR_{IR} = 16, 41, 83$ mm/day $T-HC_{UNSAT} = 0.077$ mm/s	[46]
Fine sand grain $\varnothing d_{50} = 0.3$ mm $U = 2.8$	Columns: $\varnothing$ NG, sand $h = 76$ cm Gravity-fed, intermittent operation	$HLR_{SUR} = 40 \text{ L/m}^2_{\text{sand}} \cdot \text{day}^{-1}$	[42]
Grain $\varnothing = 10$ mm (anoxic zone), 5 mm (oxic zone, porosity = 0.43, 0.36, respectively)	Column: $\varnothing = 10.4$ cm, $h = 249$ cm Upflow partially aerated, HRT 3 h, recirculation rate 200–300%	* $HLR_{SUR} = 7200$ – $28,800 \text{ L/m}^2_{\text{sand}} \cdot \text{day}^{-1}$	[52]
Fractionated Dune sand Grain $\varnothing d_{50} = 0.7$ mm (0.48–1.1 mm) $U = 1.7$	Columns $\varnothing = 10.5$ cm, sand $h = 145$ cm Gravity-fed, batch mode	$HLR_{VOL} = 2163 \text{ L/m}^3_{\text{sand}} \cdot \text{day}^{-1}$ $HLR_{IR} = 4110$ mm/day (at high OLR) $M-HC_{SAT} = 2.50, 1.15$ mm/s ($t = 0, 130$ days)	[10]
Quartz sand grain $\varnothing d_{10} = 0.86$ mm $U = 1.47$	Columns $\varnothing = 5$ cm, sand $h = 80$ cm Gravity-fed, continuous: saturated and anaerobic	$HLR_{IR} = 1200$ mm/day	[61]
Grain $\varnothing = 0.93$ mm (fine), 2.40 mm (coarse) $U = 1.50$ (fine) 1.35 (coarse)	Columns $\varnothing = 14.5$ cm, sand $h = 61$ cm (46/15 coarse/fine) Gravity-fed, intermittent operation (hourly)	$HLR_{SUR} = 264, 132, 66 \text{ L/m}^2 \cdot \text{day}^{-1}$	[69]
$U = 1.07$ Porosity = 0.494	Columns $\varnothing = 7.6$ cm, $h = 60$ cm Gravity-fed NDP	NDP	[70]
Quartz sand grain $\varnothing d_{50} = 0.8$ mm (0.6–1.2 mm) $U = <2.0$	Columns $\varnothing = 12$ cm, sand $h = 80$ cm Gravity-fed, aerated–anoxic in series	$HLR_{SUR} = 2000$ – $4000 \text{ L/m}^2 \cdot \text{h}^{-1}$ (for system), * 4.8 – $9.2 \times 10^4 \text{ L/m}^2_{\text{sand}} \cdot \text{day}^{-1}$	[71]

Fine sand grain $\phi < 1$ mm	Columns $\phi = 9$ cm, sand $h = 2, 12, 22$ cm Gravity-fed, batch operation (once-off)	HLR _{IR} = 6240, 7440, 9360 mm/day	[59]
Grain $\phi = 0.3$ mm (fine), 2.4 mm (coarse) U = 4.0 (fine), 1.3 (coarse)	Columns $\phi = 14.5$ cm, sand $h = 61$ cm (15/46 coarse/fine) Gravity-fed, intermittent operation (72 times/day)	HLR _{SUR} = 66 L/m ² _{sand} ·day ⁻¹	[48]
Grain $\phi = d_{10} = 0.42$ mm (fine), 0.8 mm (coarse) d_{50} (from GSD plot) = 1.0 mm (coarse), 0.65 mm (fine)	Columns $\phi = 36$ cm, sand $h = 50$ cm Gravity-fed, intermittent operation (1:3 (feed:rest) days) HRT = 5–6 h (fine), 2–3 h (coarse)	HLR _{SUR} = 3400 L/ m ² _{sand} ·day ⁻¹ (during feeding)	[40]
Grain $d_{10} = 0.15$ – 0.30 mm, $d_{50} = 0.45$ – 1.18 mm, d_{60} = 0.60– 1.20 , U < 4.0, porosity 0.38	Columns $\phi = 7.5$ cm, sand $h = 15$ cm Gravity-fed, intermittent operation (every 3 h), HRT = 2.8 days	HLR _{SUR} 20.5 L/m ² _{sand} ·day ⁻¹	[72]
River sand grain $\phi d_{10} = 0.41$ mm U = 5.85	Columns $\phi = 10$ cm, sand $h = 25$ cm Gravity-fed, intermittent operation (1 dose 5–6 h/day)	HLR _{IR} = 134, 204, 269 mm/day	[47]
River sand 1 $d_{10} = 0.38$ mm, $d_{50} = 0.82$ mm River sand 2 $d_{10} = 1.60$ mm, $d_{50} = 2.26$ mm	Columns $\phi = 30$ cm, sand $h = 15, 30, 70$ cm Gravity-fed, intermittent operation (8 doses/day every 3 h) HRT (for 70 cm (h) columns) = 12, 35 h	HLR _{IR} = 120, 200 mm/day Sand 1: T-HC _{SAT} 0.825–0.953 mm/s Sand 2: NDP	[29]
Quartz sand grain $\phi = 0.210$ – 0.297 mm Porosity = 0.469	Column $\phi = 2.5$ cm, sand $h = 33$ cm Upflow, continuous feeding, HRT = 12 h	HLR _{SUR} 18 L/m ² _{sand} ·day ⁻¹	[53]
Laboratory studies: vertical and horizontal flow and hybrid systems			
Sand characteristics and porosity	Reactor dimensions, operation mode and retention time	Hydraulic loading rates	Ref.
Particle $\phi = 0.25$ – 0.42 mm	HSSF BSR 32 × 13 cm (L × W), sand $h = 60$ cm	NDP for HSSF	[25]
Particle $\phi = 1.9, 1.6, 65, 0.45$ mm (bottom to top)	VF BSR: sand $h = 80$ cm	NDP	[73]
Grain $\phi = 1.7$ – 2.0 mm (sieved)	VF BSR: $\phi = 36$ cm, sand $h = 16$ cm HRT = 8, 12, 16, 20 24 h (UASB-BSR)	HLR: NDP	[74]
Pilot studies			
Sand characteristics and porosity	Reactor dimensions, operation mode and retention time	Hydraulic loading rates	Ref.
River sand grain $\phi d_{50} = 3.2$ mm, U = 8, Porosity = 0.17	HSSF bed, L × W DNP, sand $h = 110$ cm Semi-continuous operation, HRT 1.7 days	HLR _{IR} = 110–130 mm/day	[50]
Grain ϕd_{50} (i) 0.35 mm (ii) 0.7 mm (iii) 0.8 mm	HSSF BSR 800 × 150 cm (L × W), sand $h = 20$ – 70 cm HRT 4–8 days	NDP	[75]
Grain $\phi = 0$ – 3 mm	VF BSR L × W NG, $h = 85$ cm Intermittent operation (1 h intervals)	HLR _{SUR} = 96 L/m ² _{sand} ·day ⁻¹	[76]
Dune sand 81% quartz, 18% calcite	VF BSR 1 and 2: $\phi = 200$ cm (inner chamber = 140 cm)	HLR _{IR} = 206 mm/day	[16,26]

Grain ϕ d_{50} = 0.9 mm (0.003–2.2 mm)	Sand h = 1800 cm (outer chamber = 100 cm) Continuous operation	102–208 L/m ³ _{sand} ·day ^{−1} M-HC _{SAT} = 0.140 mm/s (operational)	
Fine sand grain ϕ = d_{10} = 0.17 mm	VF BSR: ϕ = 50 cm, sand h = 65 cm	HLR _{SUR} = 10 ± 2 L/ m ² _{sand} ·day ^{−1}	[62,77]
U = 2	Gravity-fed, intermittent operation (4 times/day)	T-HC _{SAT} = 0.310 mm/s (t = 0)	
Full-scale studies			
Sand characteristics and porosity			
Grain ϕ d_{50} = 0.3 mm (fine), 2.4 mm (coarse)	12× HSSF: 55 × 25 m (L × W), sand h = 60 cm (45/15 fine/coarse)	HLR _{IR} = 60 mm/day	[49]
U = 4.0 (fine), 1.3 (coarse)	Rotational feed and rest cycles between modules	5.11×10^5 L/day inflow for system	
Grain ϕ = 0.25–4.0 mm	HSSF BSR 27.5 × 6.9 m (L × W), sand h = 10–15 cm Gravity-fed during rainfall or irrigation, HRT = 4 days	HLR: DNP	[78]

* calculated from data provided in manuscript. NDP = no data provided; FS = fine sand; MS = medium sand; CS = coarse sand; UC = uniformity coefficient; HSSF = horizontal subsurface flow; L = length; W = width; ϕ = diameter; h = height of sand in reactor/column; IR = infiltration rate; HLR_{SUR} = surface hydraulic loading rate; HLR_{VOL} = volumetric hydraulic loading rate; HLR_{IR} = hydraulic loading rate infiltration rate (HLR reported in many studies has been changed to infiltration rate (IR, which is the infiltration velocity) in order to distinguish from HLR_{SUR} and HLR_{VOL}); T-HC_{SAT} = theoretical saturated hydraulic conductivity; T-HC_{UNSAT} = theoretical unsaturated hydraulic conductivity; M-HC_{SAT} = measured saturated hydraulic conductivity. All units have been converted to the same units for comparative consistency. Where a range of parameters were compared, the operational parameters for the optimal performance results are provided.

In the case of NH_3 , the hypothesis is that in cold climates, adsorption of NH_3 by materials such as zeolite [66] or clinoptilolite [67] in winter may be followed by increased nitrification with the advent of warmer weather going into spring. Studies detailing combined adsorption–biodegradation removal in BSRs are limited, and the subject merits further investigation.

In the case of TSS, these may be adsorbed onto the surface of sand particles or become entrained in the pores of the sand matrix in BSRs. As previously discussed, this is particularly problematic if the solids cannot be mineralized. Many factors influence removal of TSS, including the biodegradability of the materials, the sand characteristics and the HLR and OLR [40]. Many researchers have reported on removal efficiencies (RE) for TSS. The TSS concentrations and RE of primary and secondary wastewater/sewage and septic tank effluent reported in the literature range from 0.039 ± 0.011 g/L (RE 65–100%) in septic tank effluent [29] to 0.21 ± 0.19 g/L (RE 56%) in municipal sewage WW [47], with RE ranging from 56% [29] to 93% (influent TSS 0.086 g/L) in synthetic septic tank effluent [40] (Table 2). The TSS concentrations and RE of industrial WW reported in the literature are typically higher, with concentrations ranging from 0.17–1.5 g/L in turkey processing WW [49] to as much as 1.5 g/L (range: 0.6–2.0 g/L) with 76% RE (range 65–92%) in diluted olive mill WW [8] (Table 3). The number of variables involved in removal of TSS, including the operational parameters, the nature of the solids and the characteristics of the sand, makes it impossible to generically predict removal of TSS in BSRs, as each system is unique.

In contrast to readily biodegradable chemicals, recalcitrant elements that are removed by adsorption can leach from BSRs when finite binding sites become saturated and/or there are changes in redox status and/or pH and/or the biomass degrades [75]. This makes BSRs unsuitable for long-term removal of such chemicals, which includes total phosphorus (TP). Many studies have reported on removal of TP and/or phosphates (PO_4^{3-}) in BSRs. This may be misguided because P cannot be mineralized and will eventually leach from the systems [5,18,75]. In a pilot study focused on determining P removal from activated sludge effluent in BSRs, researchers obtained maximum removal rates of 9% and found that the P concentration in the treated effluent increased from the influent values in 33% of the samples [79]. At low HLRs, good P removal was obtained from WW from a septic tank (97%), a biofiltration tank (77%) and Polonite filter bag (92%) [75]. However, there was a >99.99% risk of leaching associated with simulated rainfall to the columns and almost all the P eventually leached into the column effluent. A meta-analysis of 329 samples from 10 studies of full-scale on-site BSR WW treatment systems in Finland and Sweden showed that while excellent organic removal rates were obtained, the effluent did not comply with discharge standards for P [1]. A previous meta-study on BSRs in Denmark yielded similar results (23–90% P removal), with good P removal (70–90%) only being found when the sand contained high concentrations of Fe [5]. The principal mechanism for removal of P occurs via binding of P to ferric oxides (FeO), calcium (Ca) or clay particles [80]. Unexpectedly, in a laboratory BSR column study comparing non-coated sand with sand coated with FeO, no significant differences in P removal efficiencies were noted, and the authors proposed that the electrostatic forces between the P and Fe were too weak [47].

In essence, if P removal is important for the type of WW being treated or to comply with discharge requirements, stand-alone BSRs cannot be relied on for consistent and long-term removal. In addition, because P is becoming depleted as a natural resource, recovery of TP from wastewater using physicochemical processes is becoming essential [81]. Pre- or post-coagulation or other means of capturing P for reuse supports a circular economy and has the advantage of rendering the treated effluent suitable for discharge to aquatic environments with minimal risk of eutrophication [62,77,81].

Table 2. Organic and nitrogenous removal performance of biosand reactors treating primary and secondary wastewater/sewage (real and synthetic) and septic tank effluent.

Descriptive Term for BSR Study Type and Working Volume	Wastewater (WW) Time: Run and (Start-Up)	Selected Parameters	Reduction/Outcome	Ref.
Intermittent sand filter Laboratory scale (7.9 L)	<i>Settled, filtered municipal WW</i> Duration not provided	Operational HC 0.1, 0.05, 0.02 HC Biofilm ‘clogging’ TOC: 0.11 ± 0.061 g/L NH ₄ ⁺ : 32 (range: 14–63) mg/L	Inadequate denitrification at low HC Reduced surface ponding at low HC TOC: 94 ± 6% NH ₄ ⁺ : 99 ± 2%	[46]
Intermittent sand bioreactor Laboratory scale (NG)	<i>Seawater salinity sewage</i> 21 months	CODs: 0.30 g/L NH ₄ ⁺ : 25 mg/L TN: 35 mg/L	CODs: 96% NH ₄ ⁺ : 96% TN: 75%	[42]
Partially aerated biological filter Laboratory (21 L)	<i>Synthetic raw municipal WW</i> NG (60 days)			[52]
Anaerobic biofilter Laboratory (1.6 L)	<i>Municipal WW</i> 70 days (0 days)	CODt: 509 ± 173 g/m ³ ·sand·day ^{−1} NH ₄ ⁺ : 42 ± 19 mg/L	CODt: 62 ± 12% NH ₄ ⁺ : 5%	[61]
Biological filter Laboratory (2.7 L)	<i>Synthetic saline sewage WW</i> 7 weeks (0 days)	CODt: 0.65–0.66 g/L NaCl: 0.66–1.49 g/L	COD: >80%	[70]
O/A two-stage BSF Laboratory (27 L)	<i>Secondary municipal effluent</i> Duration unclear (0 days)	TN: 22 (range: 20–26) mg/L	TN: <50%	[71]
Sand filter Laboratory (NG)	<i>Synthetic septic tank effluent</i> 1 week (0 days)	TSS: 0.086 g/L CODt: 0.46 g/L NH ₄ ⁺ : 56 mg/L	TSS: 93% CODt: 97% NH ₄ ⁺ : 62%	[40]
Sand filter Laboratory (0.65)	<i>Synthetic septic tank effluent</i> 40 days (acclimated sand used)	TN: 138 mg/L @ 0.28 mgN/cm ² ·day ^{−1} COD/N ratio: 6.2 Alk/NH ₄ ⁺ ratio: 7.1:1	Complete nitrification in upper 15 cm @ 17.0 gN/m ² ·day ^{−1}	[72]
Slow sand filter Laboratory (2.0 L)	<i>Municipal sewage WW</i> 100 days (0)	TSS: 0.21 ± 0.19 g/L CODt: 0.16 ± 0.11 g/L	TSS: 56% CODt: 56%	[47]
Packed bed reactor Laboratory (49 L)	<i>Septic tank effluent</i> 360 days	TSS: 0.039 ± 0.011 g/L CODt: 0.37 ± 0.10 g/L NH ₄ ⁺ : 46 ± 21 mg/L	TSS: 65–100% CODt: 81–98% NH ₄ ⁺ : 67–100%	[29]
Semi-saturated sand bed mesocosm Pilot (2.2 × 10 ³ L)	<i>Urban pond WW</i> “multi-seasonal”	TSS: 0.12 g/L CODt: 0.29 g/L TN: 39 mg/L	TSS: 81 ± 6% CODt: 82 ± 15% TN: 48 ± 15%	[50]

Where a range of parameters were tested, only the performance results from the ‘optimal’ reactor are provided in the table. Where necessary, further results are given in the text. In some instances, values have been extrapolated from figures or data provided in the manuscripts. NG = not given; TSS = total suspended solids; CODt = total chemical oxygen demand; CODs = soluble chemical oxygen demand; TOC = total organic carbon; TN = total nitrogen.

3.1.2. Increase in pH of Acidic Wastewater

It has been shown that the pH of acidic WW can be increased in BSRs. For example, the pH of simulated landfill leachate increased from influent to effluent in a BSR, while the effluent from a landfill bioreactor remained acidic [73]. Similarly, Achak et al. [8] found an increase in pH of olive mill WW ranging from 4–6 up to 7–9 in a BSR operated in front of a VF CW. Neither the elemental nor mineralogical composition of the sand was determined in either of these studies, but it was postulated that the consistent increase in electrical conductivity (EC) from influent to effluent (average 5.9–8.7 ms/cm) indicated that carbonates in the sand were mineralized [8]. This was supported by results obtained in a pilot HSSF BSR [7] and a VF BSR [26], where consistent increases in EC and pH were also measured during remediation of acidic winery WW. The major mechanism responsible for the increase in pH was later experimentally verified as being via dissolution of calcite [82,83]. This mineral form of calcium carbonate (CaCO_3) constituted 18% of the sand grains, and dissolution did not affect the hydraulic properties of the sand [83]. The theoretical abiotic neutralization capacity of the BSRs treating WW was predicted to be 37 years, which is beyond the projected lifespan of the system. An added advantage of calcite dissolution is that the presence of divalent calcium (Ca^{2+}) in the BSR effluent also protects receiving environments from sodicity from saline WW [26,82].

Microbial mechanisms of pH adjustment have also been seen in BSRs. In a study by Holtman et al. [83], it was found that while calcite dissolution was responsible for $95.5 \pm 0.16\%$ of the increase in pH, the remaining $4.5 \pm 0.13\%$ was due to biotic mechanisms. Interestingly, complete neutralization of highly acidic synthetic acidic mine drainage (pH 2) was also demonstrated in HSSF BSRs containing river sand with 98% quartz and only 0.1% calcite [9]. In this study, the synthetic acid mine drainage was supplemented with a carbon (C) source for biological sulfate reduction and the authors identified microbial sulfate and iron reduction and precipitation of Fe and S as the main neutralization mechanisms.

3.2. Bioremediation

3.2.1. Selection and Acclimation of Microbial Communities

The composition and function of microbial communities in BSRs are influenced by the type of WW being treated, climatic conditions, the physicochemical properties of the sand and the WW chemistry and redox status of spatial niches within BSR [9,55,84]. Examples of the character of the influent on the microbial composition include the selection of biological sulfate reducing bacteria [9] and diazotrophic bacteria [85,86] in BSRs treating acid mine drainage and high C to nitrogen (N) WW, respectively. In BSRs that are not overloaded, nutrient concentrations and hence microbial biomass typically decrease with depth in VF systems and depth and length in HSSF systems [55,72,87].

It has been recommended that BSRs be allowed to acclimate to the intended WW for around 100 days to allow the functional microbial communities to establish and stabilize before conducting experiments [88]. Applying WW in incremental concentrations during the acclimation phase has also been shown to have a positive effect on BSR performance [89]. However, many researchers do not factor in acclimation times, or they apply acclimation periods that could be considered insufficient (Tables 2–4). Of the 31 studies reviewed for bioremediation using BSRs, 10 did not allow any acclimation period. Three studies applied acclimation times that may be considered insufficient (1 = 4–6 days, 1 = 20 days, 1 = 3 weeks), while five applied acclimation periods ≥ 2 months. Thirteen studies did not report acclimation times, but nine of these were conducted for long periods (≥ 1 year), so it is unlikely that those results would be skewed due to lack of acclimation.

3.2.2. Bioremediation of Organics and Nitrogenous Compounds

Primary and Secondary Wastewater/Sewage and Septic Tank Effluent

Parameters used to assess nutrient removal performance in BSRs are inconsistent between studies, making inter-study comparisons challenging. Due to the relative simplicity and reproducibility of COD over other analytical methods such as biological oxygen demand (BOD) and total organic carbon (TOC), the former is commonly used as a proxy for determining removal of organics in WW. Of the eleven studies reporting removal performance of BSRs treating sewage-based WW (Table 2), eight reported on COD removal, one reported total organic carbon (TOC) removal and two only focused on nitrogen (N) removal. Nine studies reported on nitrification and/or denitrification.

Total COD (COD_T) was the most common parameter used for analyzing organic removal performance, being used in five of the studies. The remaining three analyzed organic removal using either soluble COD (COD_S), COD/N ratio or COD loading (gCOD/m³_{sand}·day⁻¹). The influent COD_T values ranged from 0.16 ± 0.11 g/L (RE = 56%) in municipal sewage [47] to 0.66 g/L (RE > 80%) in synthetic saline sewage [70] and high COD RE values were found in most of the studies (Table 2).

Many VF BSRs that have been extensively used to treat domestic WW (sewage and grey-water) have a conventional design consisting of gravel at the bottom, coarse sand in the middle and fine sand in the top layer. Most of the degradative processes take place in the top layer, which also controls the passive flow rate, making the lower layers somewhat obsolete [58]. Researchers studied the influence of grain size (Table 1) on COD_T RE and nitrification in intermittently fed laboratory columns treating synthetic septic tank effluent [40]. At lower HLR_{IR} (130 mm/day), COD removal efficiencies were similar (87–91%), but at higher HLR (300 mm/day), COD RE was higher in the columns containing the finer grains than in those containing the larger grains because the HRT was insufficient in the latter, a phenomenon already alluded to in Section 2.2. Nitrification was also higher (46–62%) in the columns containing fine sand than in those containing coarse sand (7%), despite lower O₂ (9–20%) being measured at 35 cm depth in the former than in the latter (17–21%). This was because the O₂ increased again to around 20% during the resting period. However, after the 10th feeding cycle, the O₂ in the columns containing fine sand decreased to around 1% at the end of the feeding cycle, suggesting that high nitrification rates may not be sustainable over time in BSRs containing fine sand. This is supported by historical data from full-scale BSRs treating domestic WW in Denmark where nitrification rates were found to be higher in systems containing coarser grains, ostensibly due to superior O₂ ingress, and nitrification rates decreased with increases in HLR [5]. Shahraki et al. [72] also demonstrated decreased nitrification efficiency with increased HLR, or when there was insufficient alkalinity. The authors measured O₂ partial pressures, not just at one depth, but at 6 cm, 14 cm, 22 cm and 30 cm in columns treating synthetic septic tank effluent with notably lower HLR_{SUR} (20.5 L/m²_{sand}·day⁻¹ [72] vs. 3400 L/m²_{sand}·day⁻¹) [40] and higher HRT of 2.8 days [72] vs. 5–6 h [40] under various N loading rates, alkalinities and HLR. They found O₂ concentrations comparable with atmospheric levels at all depths, albeit with a slight drop due to microbial utilization after dosing. At the low HLR_{SUR} applied, complete nitrification at 17.0 gN/m³·day⁻¹ took place in the upper 15 cm.

The presence of both oxic zones (for nitrification) and anoxic zones (for denitrification) is needed to ensure high TN removal rates in BSRs. In gravity-fed systems, denitrification may be promoted by using sand with smaller grain sizes to induce anoxic conditions, but this could then negatively affect nitrification [29]. Sea water is being used in water-stressed countries for toilet flushing [70]. A study reported reduced organic removal performance in BSRs operated when the salinity was increased from 0.065% to 0.265%; however, no replicates were included, and significant differences were not reported [70]. In contrast, high organic RE (94 ± 6%, influent TOC 0.11 ± 0.061 g/L) and nitrification (99 ± 2%, influent NH₃ 32 mg/L (14–63) mg/L) were achieved in gravity-fed, intermittently operated BSR column experiments with periodic resting and treating seawater salinity sewage with higher salt concentrations (3% NaCl) over a 21-month period [42]. Of concern was that denitrification was poor or non-existent (influent 0.1 ± 0.0 mg/L, effluent 43 ± 17 mg/L) [42]. The authors did not discuss denitrification and

performance was not compared with non-saline controls, but denitrification is typically quite poor in small BSRs treating domestic WW. For example, only 30–45% removal of TN was noted in full-scale systems in Denmark [5] and good nitrification but only negligible denitrification were obtained in BSR column experiments containing sand from an operational BSR treating septic tank effluent (Table 2) [72]. In column experiments treating low-strength municipal WW, there was also inefficient nitrate degradation and concentrations 2.9 ± 2.2 mg/L higher were measured in the effluent than in the influent [47]. While denitrification should theoretically be supported by the anoxic environment in the deeper layers of BSRs, this can only take place if there is an adequate supply of electron donors [56]. In some instances, denitrification can be retarded if levels of O_2 favor heterotrophic utilization of carbon substrates [54]. If there is sufficient biomass and the OLR is high enough, there should theoretically be sufficient soluble carbon electron donors available in the anoxic layers for denitrification [54], but it appears that this is difficult to achieve. To try and overcome limitations, Li et al. [71] applied a two-stage oxic/anoxic laboratory BSR with two aerated columns for nitrification followed by one anoxic column for denitrification of low-strength (COD average 41 mg/L) secondary municipal effluent at various HLR and C:N ratios for denitrification and did not manage to obtain >50% removal of TN. However, it did not appear that any acclimation phase was included in the study, and further investigations are needed to see if this strategy is a viable option for secondary treatment of higher-strength WW.

Overall, the study results demonstrate that there is a critical balance in each individual BSR system between the sand that is used, the attainable HRT and other operational parameters such as O_2 concentration on the RE of organics, nitrification and denitrification of sewage-based WW. Assuming typical organic loads and C:N ratios, the principles of how the hydraulic function of BSRs may affect nitrification and denitrification in gravity-fed VF BSRs with no flow-retarding mechanism are presented in Figure 2. As described in Section 2, the initial HRT is related to the HC afforded by the characteristics and packing of the sand grains, while after start-up, the HRT increases due to build up of biomass and entrainment of particulates. This typically occurs at higher rates in systems with high OLR/HLR. The HRT can therefore be manipulated to some extent by applying different HLR in systems with higher HC.

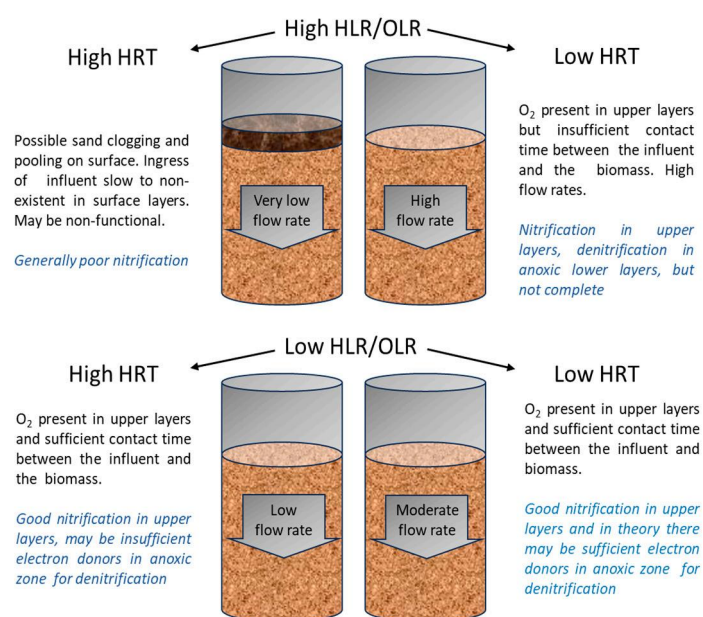


Figure 2. Basic principles showing how hydraulic loading rate (HLR), flow rate and related hydraulic retention time (HRT) typically affect nitrification and denitrification in gravity-fed biosand reactors treating primary and secondary sewage-based effluents.

Industrial Wastewater

Biosand reactors are slowly gaining traction as passive systems for remediation of industrial and agri-industrial wastewater. Laboratory column experiments, pilot and full-scale studies have been conducted (Table 3). In the reviewed studies, the organic loads in olive mill WW [8], winery WW [10,16], turkey processing WW [48,49,69], dairy wastewater [62,77] and synthetic landfill leachate [73] were considerably higher (Table 3) than those in sewage-based WW (Table 2). Long-term pilot studies yielded good organic removal rates over extended periods: 70% RE removal from winery WW (OLR: $279 \text{ gCODt.m}^3_{\text{sand}}\cdot\text{day}^{-1}$) over 2 years [26], 98% RE from dairy WW (OLR: CODt: $110 \pm 10 \text{ g/m}^2\cdot\text{day}^{-1}$) over 301 days [62,77] and 88% RE from synthetic landfill leachate (influent CODt: 32 g/L) over 18 months [73]. In addition, the BOD₅ in the effluent of a full-scale BSR treating turkey processing WW was reduced to $<0.010 \text{ g/L}$ (influent $0.57\text{--}2.0 \text{ g/L}$) over 2.5 years [49]. The high fat content of turkey processing WW can negatively affect secondary WW treatment systems, but good removal of fats, oil and grease (FOG) was obtained in full-scale BSRs [49] (Table 3). Overall, the study results indicate that BSRs are sustainable and effective options for remediation of certain types of organic-rich industrial WW. Research on other types of WW, including agri-industrial and food industry WW, is merited.

In addition to organic RE, the RE of other constituents relevant to the type of industrial WW has been determined. For example, good RE of potentially phytotoxic total (poly)phenolics (TPP) in olive mill and winery WW was obtained: 92% (influent 7.5 g/L (range: $6.0\text{--}8.2 \text{ g/L}$)) [8] and 75% (influent 40 mgGAE/L) [26], respectively, and 99.9% removal of fish pathogens (*Pseudomonas fluorescens*, *Aeromonas hydrophila*, *Aeromonas salmonicida* subsp. *Salmonicida* and *Yersinia ruckeri*) from fish farm WW [43,44].

The optimal operation and performance of BSRs treating industrial WW can differ considerably from those treating sewage-based WW and other industrial WW in different climates. For example, ion exchange was added to counter the reduction in nitrification during cold winters in BSRs treating turkey processing WW to comply with the 30-day discharge limit of 3 mg/L in Ohio, USA [49]. Contrary to treatment of sewage-based WW, higher winery WW COD loading rates were achieved with similar RE in VF BSRs operated in continuous mode when compared to intermittent (pulse) mode (Table 3) [26]. In the case of high-C:N wastewater, it has been shown that nitrogen-fixing bacterial species are selected over time, and that this natural nutrient balancing stimulates bioremediation of winery WW [86]. This nutrient self-regulation capability of BSRs may be relevant to other high-C:N wastewater types, and is an advantage advocating the use of BSRs over other secondary WW treatment systems.

Table 3. Organic and nitrogenous removal performance of biosand reactors treating industrial wastewater.

Laboratory Studies: Column Experiments					
Descriptive Term for BSR		Wastewater			
Study Type and Working Volume	Volume	Time: Run and (Start-Up)	Selected Parameters	Reduction/Outcome	Ref.
Sand filter Laboratory scale (1.18 × 10 ² L)		<i>Diluted (50%) olive mill WW</i> 1 year (NG)	TSS: 1.5 (range: 0.6–2.0) g/L	TSS: 76% (range: 65–92%)	[8]
			CODt: 28.0 (range: 25.6–30.8) g/L	CODt: 78% (range: 68–90%)	
			CODs: 20.7 (range: 17.0–24.3) g/L	CODt: 75% (range: 59–92%)	
			TPP: 7.5 (range: 6.0–8.2) g/L	TPP: 92% (range: 85–95%)	
Sand intermittent filter Laboratory scale (9.8 L)		<i>Mixed industrial WW</i> NG (4–6 days)	TSS: 0.31 ± 0.09 g/L	TSS: 67%	[68]
			CODt: 0.32 ± 0.09 g/L	CODt: 39%	
Biological sand filter Laboratory scale (18 L)		<i>Fish farm WW</i> 24 days (10 weeks)	NH ₄ ⁺ : 0.27 ± 0.16 mg/L	Fish pathogens: 99.9%	[43,44]
Biosand reactor Laboratory (1.3 × 10 ² L)		<i>Winery wastewater</i> 133 days (3 weeks, inoculated)	COD: 4326 g/m ³ ·sand·day ^{−1}	COD: 95%	[10]
Coarse/fine sand filter Laboratory (10.1 L)		<i>40% Turkey processing WW</i> 81 days (0 days)	CODt: 548 ± 314 g/m ³ ·sand·day ^{−1}	TOC: 70% (overloaded)	[69]
			CODt: 274 ± 157 g/m ³ ·sand·day ^{−1}	TOC: 77% (overloaded)	
			CODt: 137 ± 78 g/m ³ ·sand·day ^{−1}	TOC: >97%	
Sand bioreactors Laboratory (10.1 L)		<i>Turkey processing WW</i> 784 days	CODt: 0.84–3.9 g/L	CODt: >95%	[48]
			BOD ₅ : 0.46–2.26 g/L	BOD ₅ : >99%	
Pilot and full-scale studies					
Descriptive term for BSR		Wastewater (WW)			
Study type and working volume	Volume	Time: Run and (start-up)	Selected parameters	Reduction/outcome	Ref.
Biosand reactor Pilot (4.4 × 10 ³ L)		<i>Winery WW</i> 2 yrs (3 months)	Continuous: 279 gCODt·m ³ ·sand·day ^{−1}	COD: 70%	[26]
			Intermittent: 179 gCODt·m ³ ·sand·day ^{−1}	COD: 70%	
			TPP: 40 mgGAE/L	TPP: 75%	
Conventional intermittent sand filter Pilot (1.3 × 10 ² L)		<i>Dairy WW</i> 301 days	TSS: 1.80 g/L	TSS: 97%	[62,77]
			CODt: 110 ± 10 g/m ² ·day ^{−1}	CODt: 98%	
Sand channel Laboratory (NG)		<i>Synthetic landfill leachate</i> 18 months	CODt: 32.0 g/L	CODt: 88%	[73]
Slow sand filter Laboratory (16.2 L)		<i>Pulp and paper mill UASB effluent</i> 6 weeks (0 days)	NH ₄ ⁺ : 1435 mg/L	NH ₄ ⁺ : accumulation	[74]
			BOD, CODt, TSS, SO ₄ ²⁻ (Influent to BSR: values NG)	Hybrid UASB-BSR increased removal in comparison to UASB alone	
Sand bioreactor		<i>Turkey processing WW</i>	BOD ₅ : 0.57–2.0 g/L	Effluent BOD ₅ : < 0.010 g/L	[49]

Full-scale (8.3×10^7 L) $\times$ 12 units	2.5 yrs	TSS: 0.17–1.5 g/L FOG: 42–374 mg/L NH ₄ ⁺ : 18–44 mg/L	Effluent TSS: < 0.015 g/L Effluent FOG: 5 mg/L Effluent NH ₄ ⁺ : 0–3.8 mg/L
Where a range of parameters were tested, only the performance results from the ‘optimal’ reactor are provided in the table. Where necessary, further results are given in the text. In some instances, values have been extrapolated from figures or data provided in the manuscripts. NG = not given; WW = wastewater; TSS = total suspended solids; CODt = total chemical oxygen demand; CODs = soluble chemical oxygen demand; TPP = total (poly)phenolics; TOC = total organic carbon; TN = total nitrogen; BOD = biological oxygen demand; FOG = fats, oil and grease; UASB = upflow anaerobic sludge blanket.			

3.2.3. Removal of Fecal Indicator Organisms, Antibiotic-Resistant Bacteria and Micropollutants

It has been shown that BSRs can effectively reduce the number of fecal indicator organisms (*Escherichia coli*, coliforms), other enteric pathogens and micropollutants in sewage-based WW (Table 4). The RE of bacteria and micropollutants may be comparable with CW/TWs because researchers have found that plants have no significant effect on RE of *E. coli* in a variety of planted and unplanted VF and HSSF systems [76] and addition of root exudates does not improve the RE of micropollutants [60]. Good removal of bacteria resistant to sulfamethoxazole, tetracycline and erythromycin as well as the genes encoding for resistance to these antibiotics has been achieved in BSRs [90]. Notably, in this study, the authors determined that the removal mechanisms were a combination of entrainment and biodegradation. Based on the results obtained from a predictive model validated with historical literature data for VF BSRs with different sand grain ϕ , HLR_{IR}, depths and periods of resting, Sadiq et al. [45] cautioned that effluent from BSRs may not comply with standards for irrigation or groundwater recharge in terms of *E. coli* and fecal coliforms. They found that the risk of exceeding a fecal coliform standard of ≥ 100 most probable number/100 mL was approximately 50% at a 95% confidence level, and that the effluent would require tertiary disinfection for unrestricted agricultural use in Saudi Arabia. In a comprehensive array of laboratory, modelling and field experiments using fine, medium and coarse sand, residual heterotrophic bacteria were present in the treated septic tank effluent, albeit with a 1–2.5 log reduction in the heterotrophic bacterial count [90].

As with nutrient RE, it may be expected that the RE of enteric bacteria will be positively correlated with HRT, and that this is incumbent on the height of VF BSRs and the HC of the sand employed, as discussed in preceding sections of this review. This premise has been experimentally validated. In one such study, municipal sewage WW treated in VF BSRs only complied with WHO and US-EPA discharge guidelines for total and fecal coliforms when the height of sand was increased to 22 cm [59]. Although this study was short-term (30 days) and conducted with no acclimation period, the results are supported by additional research. A more in-depth study reported the removal rates of *E. coli* and MS2 virus in BSRs with different heights (13.2 cm and 54.5 cm) and containing sand with different average grain ϕ . Only a low HLR_{IR} (0.2 mm/h) could be achieved in the 13.2 cm columns containing fine sand. This was 4.5-fold less than the HLR_{IR} in the 54.5 cm columns containing coarse sand. Despite the difference in height, the HRT in the former was significantly higher and translated into higher RE of both *E. coli* (99.5% compared with 62.8%) and MS2 virus (73.9% compared with 27.6%) [58].

Remediation in BSRs has been proposed as a polishing step for the removal/reduction in micropollutants from domestic WW [53]. Removal efficiencies have been quite comprehensively studied, with different results being found for different chemicals. The micropollutant concentrations in the effluent of upflow BSRs operated in continuous mode were measured [51] and the highest REs were achieved at the highest HRT (range: 4–35 h) at a HLR_{SUR} of 288 L/m²_{sand}·day^{−1} (Table 1). Under these operational conditions, the REs of diclofenac, propranolol, iopromide, iohexol and iomeprol were 41%, 94%, 58% and 57%, respectively. When comparing the rates of degradation of micropollutants in pre-settled raw municipal WW spiked with pharmaceutical standards, Nord and Bester [91] recommended an HRT of 10–30 h for 90% removal of pharmaceuticals. The authors found that increases in OLR either increased (e.g., β -blockers metoprolol, propranolol and atenolol), decreased (e.g., iohexol) or had no effect (e.g., sulfamethizole, sulfamethoxazole, trimethoprim, iomeprol and carbamazepine) on the degradation rate. In contrast, the results of another study suggest that removal of some micropollutants may not be HRT-dependent, as found with tebuconazole (59% RE) and propiconazole (21% RE) [51]. Lastly, the effects of oxygenation and acetate addition (as a carbon source to stimulate microbial growth and activity) on the degradation of micropollutants in BSRs have been investigated [53]. Three different responses were noted; degradation of atenolol and iohexol was

initially enhanced by the addition of acetate, while metoprolol, iomeprol, diclofenac, propranolol and sulfamethizole were aerobically degraded and inhibited at acetate C concentrations >60 mg/L, and sulfadiazine, sulfamethoxazole and trimethoprim were removed independently of O₂ and acetate addition. Notably, carbamazepine, ditriazoic acid, iopromide, tramadol and venlavaxine were not removed.

4. Conclusions

Biosand reactors are simple, passive systems that have low capex and maintenance costs, do not require skilled operation and can adapt to variations in WW quality. They are well-suited to the intermittent flows of household sewage and seasonal agri-industrial WW and do not require lengthy start-up periods after being inactive. Provided the intrinsic HC of the chosen sand is high enough to allow reasonable achievable flow rates after the necessary accumulation of functional biomass, and the systems are not extensively used to remove recalcitrant particulates via entrainment, BSRs can operate effectively for many years with minimal intervention. With the application of optimized OLR/HLR and HRT for each system and WW type, excellent nitrification and organic RE can be achieved. Based on laboratory studies, the systems have also been proposed for tertiary treatment of WW for removal of various micropollutants and have shown effective reduction in ARB, ARG and some microbial pathogens in pilot or full-scale systems. However, denitrification rates may be insufficient to attain complete N removal and further research is required to mitigate against the presence of nitrates in BSR effluent. In addition, P removal rates can fluctuate, and because P is removed via adsorption only, it can leach into the effluent at any point. The systems are therefore not recommended when consistent P removal from WW is required, for instance, when the effluent is discharged into sensitive aquatic environments. The addition of ancillary processes to recapture this limited nutrient is an option that merits further investigation.

Table 4. Removal of pathogens, antibiotic-resistant bacteria, micropollutants and pharmaceuticals.

Descriptive Term for BSR Study Type and Working Volume	Wastewater (WW) Time: Run and (Start-Up)	Selected Parameters	Reduction/Outcome	Reference
Biofilm reactor/slow sand filter Laboratory scale (0.14 L)	<i>Final municipal WW effluent</i> 1 month (3 months)	Recalcitrant micropollutants	Micropollutants: 21–85%	[51]
Slow sand filter Laboratory scale (0.35 L)	<i>Final municipal WW effluent</i> 14 days (3 months)	Recalcitrant micropollutants	Root exudate addition: no effect	[60]
Biosand filter Laboratory (1.3 L)	<i>Municipal sewage</i> 30 days (0 days)	Total and fecal coliforms: 2.56–3.04 × 10 ⁸ CFU/mL	Total and fecal coliforms: 99%	[59]
Slow sand filter Laboratory (2.0 L)	<i>Municipal sewage WW</i> 100 days (0)	Total/fecal coliforms (MPN/100 mL)	Total/fecal coliforms: (90/99%)	[47]
Slow sand filter Laboratory (25 L)	<i>Rough filtered municipal WW</i> 20 days (20 days)	Total coliforms	Max 99% removal	[25]
Laboratory columns	<i>Secondary municipal effluent</i> Time = unclear	27 pharmaceuticals	Acetate improved degradation of some chemicals	[53]
Lateral flow sand filter Pilot (NG)	<i>Septic tank effluent</i> Operational since 2004	Antibiotic-resistance genes (ARG) Antibiotic-resistant bacteria (ARB)	2.2–5.5 log ₁₀ reduction, but ARG and ARB still present in effluent	[90]
Subsurface flow ecotechnologies Pilot (various)	<i>Domestic sewage</i> 16 months	<i>Escherichia coli</i>	1.5–2.1 log ₁₀ reduction, Effluent 4.5 × 10 ⁴ to 1.7 × 10 ⁵ MPN/100 mL	[76]
Horizontal slow sand filter Full-scale (2.3 × 10 ⁷ L)	<i>Horticulture leachate</i> 15 yr operation, snapshot study	<i>Fusarium oxysporum</i> f.sp. <i>lycopersici</i> propagules	99.9%	[78]

MPN = most probable; CFU = colony forming units; ARG = antibiotic-resistance genes; ARB = antibiotic-resistant bacteria.

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Abbreviations

Biosand reactor (BSR), constructed wetland (CW), diameter (ϕ), horizontal subsurface flow (HSSF), hydraulic conductivity (HC), hydraulic infiltration rate (HLR_{IR}), measured saturated hydraulic conductivity (M-HC_{SAT}), treatment wetland (TW), saturated HC (HC_{SAT}), surface hydraulic loading rate (HLR_{SUR}), theoretical saturated hydraulic conductivity (T-HC_{SAT}), theoretical unsaturated hydraulic conductivity (T-HC_{UNSAT}), removal efficiency (RE), uniformity coefficient (U), vertical flow (VF), volumetric hydraulic loading rate (HLR_{VOL}), wastewater (WW)

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