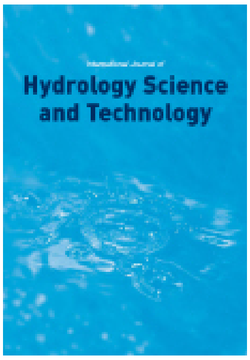

International Journal of Hydrology Science and Technology

ISSN online: 2042-7816 - ISSN print: 2042-7808
https://www.inderscience.com/ijhst

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DOI: 10.1504/IJHST.2025.10067874

Article History:

Received:	20 June 2023
Last revised:	14 December 2023
Accepted:	24 December 2023
Published online:	03 December 2024

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Abstract: This study investigated the applicability of different types of attenuation processes (i.e., aeration and stirring) with and without dilution in nutrients (nitrogen and phosphorous) and sulphide-polluted sediment cleanup via laboratory mesocosms. Attenuation refers to the decline in contaminant concentration, a phenomenon driven by processes like dilution, mixing, and dispersion. Dilution, a remedial method involving the blending of contaminated water with uncontaminated often happens with uncontaminated runoff or a tributary. Regardless of the seasons, aeration, stirring, combined aeration and stirring, and dilution generally resulted in better removal efficiency of pollutants. Aeration combined with stirring showed notable improvements across multiple water quality parameters, and parameters seemed to be treatment type dependent, but without any significant differences. Dilution reduced electrical conductivity and increased dissolved oxygen but did not influence ammoniacal nitrogen and phosphate. The energy consumption for a unit percentage improvement via aeration and stirring was 0.04–0.25 USD and 0.03–0.15 USD, respectively. Therefore, relying solely on attenuation processes without dilution is deemed economically infeasible in real or prototype applications. This research sheds light on potential applications including pros and cons, emphasising the need for a balanced approach, and setting the stage for future studies.

Keywords: aeration; dilution; energy; mesocosms; mesoscale physical habitats; sediment; stirring; water quality.

Reference to this paper should be made as follows: De Zoysa, S., Chandeep, K.A.T., Pathirathne, P.H.D.R. and Gomes, P.I.A. (2025) 'Evaluation of mesoscale physical habitats in sediment and water quality improvement – a mesocosm study for urban canals', *Int. J. Hydrology Science and Technology*, Vol. 19, No. 1, pp.93–115.

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1 Introduction

With rapid urbanisation and population growth, waterways (streams, rivers, and canals) are exposed to waste disposal, factory effluents, and other various anthropogenic activities. Importantly, such activities in the catchment areas, increase surface runoff, and sediment accumulation in waterways (Kiat et al., 2008; Gomes et al., 2014). It should be noted that many waterways in urban areas have been polluted for several decades or even centuries until legislative interventions brought some relief. Nevertheless, achieving unpolluted waterways still in many cases seemed to be difficult, and one major reason is the presence of severely polluted sediment (Zhang et al., 2016; Gomes et al., 2019). River sediment is solid and fragmented particulate matter consisting of silt, sand, and gravel (Dalezios et al., 2018) which is a vital storage medium for not only nutrients (nitrogen and phosphorus) (Dalu et al., 2019; Zhao et al., 2021), and heavy metals (Zhang et al., 2017), but also for organic compounds such as pesticides, herbicides, and hydrocarbons, (Che et al., 2020; Xiao et al., 2021). These inputs originate from diverse sources such as agriculture, industries, sewage, landfills, and urban runoff (Bhangaonkar and Patel, 2019; Savic et al., 2013) and eventually settle in sediments and act as a pollution source (Burton et al., 2004; Lv et al., 2021). In some cases, the sediment pollutant contents are 50 to 500 times higher than that in the overlying water (Cheng et al., 2014). Excessively contaminated sediments make achieving an unpolluted waterway challenging since pollutants (e.g., nutrients) could continuously diffuse into the water column (Müller et al., 2016). For instance, Sen et al. (2007) highlighted that the release of phosphorus from bottom sediments can serve as a noteworthy source of amplifying algal growth, contributing to eutrophic conditions, observed in Beaver Reservoir in Northwest Arkansas. Moreover, the findings of the study of Li et al. (2023) indicated that the sediment in the Biliuhe Reservoir in Liaoning Province, China serves as the primary source of nutrients, where the average concentration of NO_2-N in the overlying water was significantly higher than that in the interstitial water, indicating the predominant adsorption of NO_2-N in the sediments of the Biliuhe Reservoir. Also Zhang et al. (2012) has reported that contaminated sediment, without external pollutants, affects overlying water quality by releasing contaminants like phosphorous through sediment re-suspension. Concentrations near the sediment-water interface were about 1.5 times higher than those near the water surface.

The conventional strategy of cleanup of already polluted sediment is evacuation or dredging and supplying unpolluted sediment (Liu et al., 2015; Chen et al., 2020). However, this strategy is financially infeasible to many countries, especially developing countries (Burton, 2002). As an example, Ma et al. (2023) reported a cost of \$50–75 per cubic metre, translating to \$200,000 for removing approximately 4,000 m^3 of canal sediment. Moreover, Zhan et al. (2022) have mentioned that dredging removes the organic and nutrient-rich top sediment layer. Despite huge capital investments, many research suggest that dredging proves ineffective in meeting economic, environmental, and technical goals (He et al., 2013; Che et al., 2020). Also, the disposal of thousands of cubic meters of sediments in confined disposal areas is challenging (Burton, 2002). Therefore, dredging/evacuation is not an option in regions where the dredged sediments could promptly threaten both aquatic ecosystems and human health (Ren et al., 2021).

In this regard, strategies (e.g., alteration of instream physical habitats) that induce transport and mixing processes similar to natural streams are being discussed to be considered with the expectation that streams would trigger self-purification (Gomes and Wai, 2015). Many urban waterways such as canals are hydro-morphologically homogenous with prismatic cross sections as they are designed to convey the maximum discharge (Gomes et al., 2019). Several past studies have attributed the lack of physical heterogeneity as a reason for poor stream health (Randima et al., 2017; Wang et al., 2021). Most of those studies were field-based and conclusions were reliable on a probabilistic basis (e.g., Gomes et al., 2019). Only a few studies are available (e.g., Gomes et al., 2021) that were done in controlled environments such as in a flume to see the impact of heterogeneity or the relevance of inducing hydrodynamic processes on stream health. The study by Gomes et al. (2021) aimed to validate the hypothesis that strengthening physical heterogeneity in regulated urban waterways enhances water quality, by employing deflectors. Strategic placement of deflectors made of gabion baskets had led to depth and velocity variations, fostering turbulence, increasing DO, and facilitating the removal of H_2S, which was the primary focus of the investigation.

Unregulated streams have terrain-dependent mesoscale physical habitats (MPHs) such as riffles, pools, chutes, etc., and do perform a range of hydrodynamic processes such as mixing (stirring) and aeration, that are important in a healthy stream (Gomes et al., 2021). Riffles induce both aeration and stirring (mixing). As an example, water moving over a riffle develops shallow, fast, and wavy disturbed water surface with turbulences that diffuse oxygen to the stream from the atmosphere (Korpak et al., 2019). Meanders lead to higher retention time and aid the removal of nutrients like nitrogen (Craig et al., 2008). Okazawa et al. (2012) revealed that water contaminated with higher concentrations of nitrogen decreases when water flows in the meander sections, and the riffles and pools in meanders should have a role to play. Heidenwag et al. (2001) have illustrated the capability of self-purification in streams with the influence of dilution and mixing processes. Dilution could happen in a few ways and the stream, river, or canal receiving uncontaminated runoff or a discharge from an uncontaminated tributary (Gomes and Karunatilaka, 2022) are the two most prominent and is largely governed by catchment level management, but not instream factors. Investigation of the role of instream factors was the major motivation for this study.

Various treatment methods, such as aeration, oxygenation, mixing, dilution, and water transferring have been explored in the literature (Toffolon et al., 2013; Yu et al., 2022). Oxygenation focuses on increasing DO levels specifically providing oxygen, making it particularly effective for large-scale mesocosms targeting the bioremediation of specific contaminants (Cong et al., 2009; Liboriussen et al., 2009). Aeration, introducing air into water, is a cost-effective method that simulates natural processes, enhancing not only DO but also overall water quality addressing issues such as black colour, odour, chemical oxygen demand (COD), and biochemical oxygen demand (BOD) (Yu et al., 2022; Beam et al., 2023). In our study, which aims to simulate a natural canal, aeration proves to be a suitable approach. Dilution, a remedial method involving the blending of uncontaminated water with polluted sources, is effective in situations of heightened pollutant concentrations. By incorporating large volumes of low-nutrient water, dilution reduces nutrient loading, enhances flushing rates, contributes to improved water quality, and encourages the natural self-purification processes within the water system (Mosley, 2015). Therefore, this study analysed the effect of different treatment methods to upgrade water quality which usually happens in reactors but could also be related to natural hydrodynamic processes (aeration, mixing, and dilution) in streams. It should be noted that one popular design option in the ecological rehabilitation of streams is the placement of flow deflectors that would create pools, riffles, scour holes, etc. depending on the arrangement. Other rehabilitation design options also try to introduce MPHs to the stream, and the question is whether such hydrodynamic processes can alone improve the water quality. In this regard, a series of laboratory experiments were done using mesocosm prepared with field water and sediment to identify the roles and effectiveness of dilution and/or instream hydrodynamic processes such as aeration and mixing on water quality. Mesocosm studies provide a near-realistic environmental setting, allowing for a replicated design with well-defined treatments (Zhan et al., 2022). The necessity of these controlled experiments arises from the challenges associated with confounding factors in field studies (Pechlivanidis et al., 2011; Gomes et al., 2021) and offer notable advantages, including the ability to replicate trials, conveniently modify conditions, and obtain results prior to actual implementation (Gomes et al., 2021).

The objective of our research was to simulate hydrodynamic processes of mesoscale physical habitat in a controlled environment and investigate whether those alone can effectively enhance the water quality of polluted urban canals with sediments that have exceeded the pollution assimilation capacity. Concentrations in water and sediment can vary seasonally due to fluctuations in environmental conditions (Kumar et al., 2011). During the wet season, increased precipitation and runoff can elevate nutrient (nitrogen and phosphorus) inputs (Adeyemo et al., 2008) conversely, in the dry season, reduced rainfall and increased temperature concentrate nutrients potentially can lead to elevated concentrations in water bodies presenting much worse results (Mosley, 2015). Therefore, in the present study, it was hypothesised that nutrient concentrations of water and sediment are season-dependent and worst in the dry season. Through this comprehensive analysis of water quality parameters, we aimed to offer valuable insights into the potential of these hydrodynamic processes to contribute to the improvement of water quality and importantly to check whether the introduction of mesoscale physical habitats in stream, river, or canal rehabilitation is worth. Hence, this study served as an important initial step toward unravelling the role of natural hydrodynamic processes and how well the recreation of such processes would work in establishing a healthy stream.

2 Materials and methods

2.1 Study area

The canal network in Colombo has origins in the Dutch colonial period in the 17th century built for economic reasons as well as for flood control (Bandara et al., 2023). With a substantial population density of 18,350/km^2 within the Colombo municipality (Statistics.gov.lk, 2016), the canal network currently plays a major role in flood control. The network comprises five major sub-canals: Dehiwala, Kirulapone, Kinda, Heen and Dematagoda. Its tributaries including Kirulapone Canal, navigate through urban areas, receiving inputs from industries, sewage, and urban runoff. Approximately 43.5% of nearby households disposed of household waste into the canal and surrounding marshes nearly 25 years ago, a practice that, although to a lesser extent, continues today (Bandara et al., 2023). The contributing catchments of Dutch canals are heavily urbanised, with some sub-catchments exhibiting built area fractions surpassing 90%.

The study area experiences the wet season (May to September) and the dry season (December to March). The area is influenced by the northeast monsoon, with a tropical monsoon climate, which receives 2,000–2,500 mm of annual rainfall (Chandrasekara et al., 2018). The average annual temperature of 30–32°C in the study area contributes to a warm environment (Chandrasekara et al., 2018).

2.2 Experimental design

Sediment and water samples were collected from a representative location/reach of the canal – in front of The Open University of Sri Lanka, Nawala (6°53’18.6” N 79°52’53.3” E) (Figure 1). In this location water quality was about the average for nitrate species and phosphate in a comparative sampling conducted covering different locations of the Dutch canal network. In all cases, water was collected before sediment.

Dissolved oxygen (DO) and electrical conductivity (EC) were measured in situ using a DO meter (Milwaukee MW 600) and a conductivity meter (Mi 805), respectively. The canal water was collected into several 1 L polystyrene bottles and sediment was collected using a Van Veen Grab sampler and stored in a plastic container. The sampling depth was limited to a maximum of 125 cm from the canal bed surface, and the surface sediment is more contaminated in urban waterways such as canals (Savic et al., 2013).

Figure 1 Study area (see online version for colours)

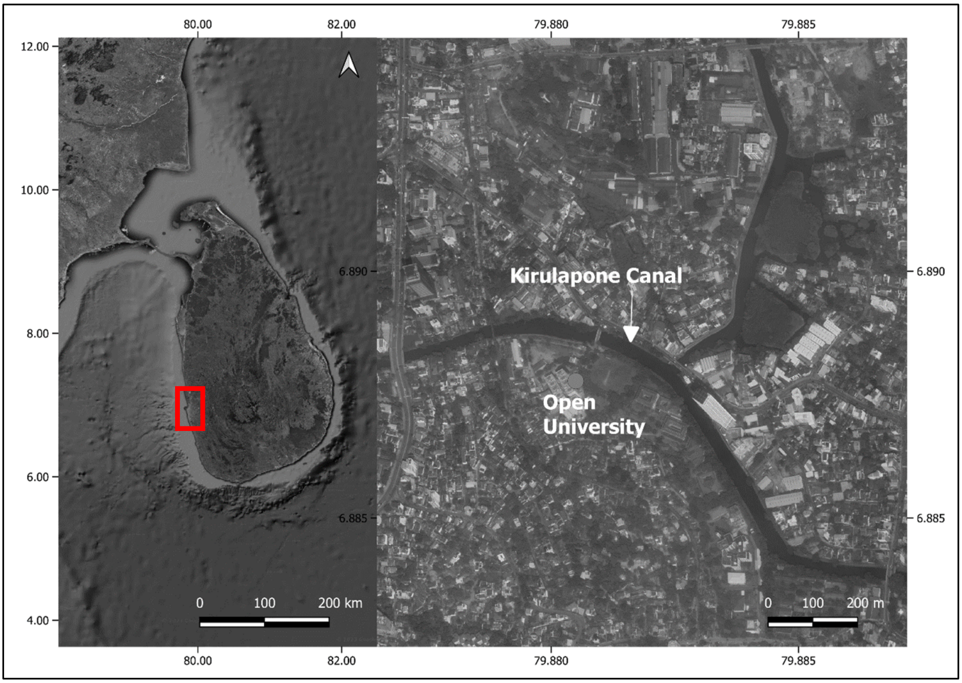
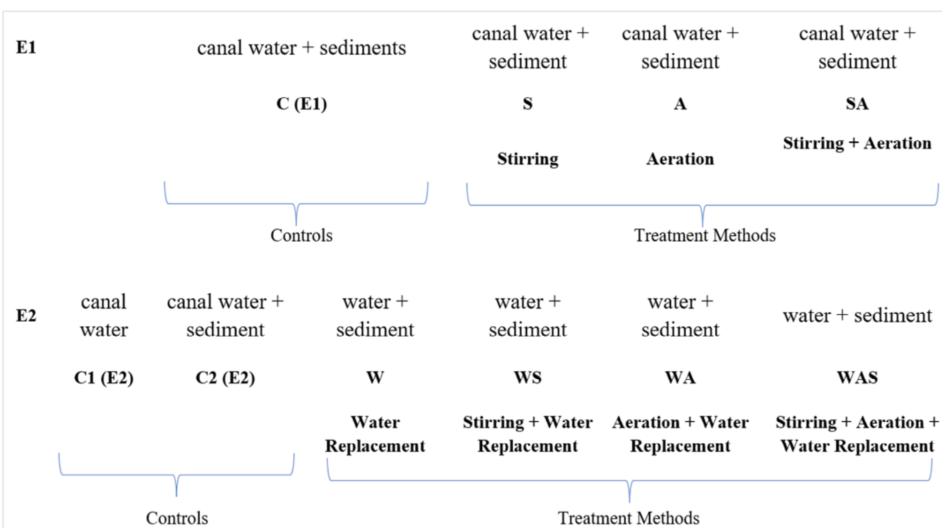


Figure 2 Treatment arrangements in experiment 1 (E1) and experiment 2 (E2) (see online version for colours)



The first experiment (E1) was conducted to quantify the role of the continuous stirring and aeration on dry season sediment cleanup for four weeks (Figure 2). The second experiment (E2) was conducted targeting both seasons to quantify the role of stirring and aeration with and without external dilution. In this regard, E2 involved water replacements wherein the weekly substitution of the entire water column with dechlorinated water from the public supply. Before these replacements, all machines were switched off for 24 hours. This allowed sediments to settle at the bottom and ensured a distinct separation between the water and sediment phases. This approach is supported by the principles outlined in the work of Coppock et al. (2017) underscoring the importance of minimising turbulence to facilitate sediment settling. Afterward, water was siphoned using a pipette, and new water was poured gently over a cellophane. Again, sediments were allowed to settle for 24 hours before switching on the machines.

To investigate the interaction between sediment and water in a canal environment, a common approach might be a shallow sediment layer (e.g., 2–5 cm) with a deeper water layer (e.g., 10–20 cm), that allows to observe sediment dynamics, including settling, resuspension, and nutrient exchange (Bromilow et al., 2006). This ratio was carefully chosen to mimic the sediment-water interface in typical aquatic ecosystems, providing an ecologically relevant setting for our experiment. Therefore, all mesocosms (Glassco-Boro 3.3 1 L beakers) were maintained to have the same depth: water ratio. The sediment depth was 2 cm, and water was filled up to 11 cm (900 ml of water) from the bottom of the beaker. Water was poured gently into the beakers over a cellophane that rested on sediment. Anyway, mesocosms were left under quiescent conditions for a day before water quality observations. We consider water quality in the water layer to be surrogative of sediment quality (Cai et al., 2021). EC and DO were measured before and after to observe any changes before and after replacing the water. In addition, other water quality parameters like ammoniacal nitrogen, nitrate, phosphate, and sulphide were observed weekly. Before the 31st day, the water of all 12 beakers was agitated for 1 min (62 rotations) using a 2 mm thick cylindrical glass rod with the expectation of aiding turbulent diffusion of pollutants in sediment and left to settle for a day followed by water quality measurements.

2.3 Water quality analysis

The water sampled was filtered using a filter paper with 0.45 µm pores before analysis. EC and DO were measured using a conductivity meter (WTW 720; inoLab, Germany) and a DO meter (HACH-sensIONTM) respectively. Nitrogen species were measured by HACH methods using HACH multi-parameter portable colorimeter (DR/900, USA) (Gomes and Wai, 2014). Nitrate was measured by the cadmium reduction method treated with the Nitra Ver 5 nitrate reagent powder pillows. Samples were reacted with the HACH-ammonia salicylate reagent (Cat No. 23953-66) and ammonia cyanurate (Cat No. 23955-66) to take the ammoniacal nitrogen levels. The resulting blue-green indophenol complex absorbance was measured using the colorimeter (DR/900, USA). Phosphate levels were measured by treating the samples with the PhosVer 3 phosphate reagent powder pillow following the ascorbic acid method (Murphy and Riley, 1962). The reaction resulted in the formation of a blue complex, and the absorbance of this solution was measured using the colorimeter (HACH-DR900). To measure the sulphide levels, samples were treated with sulphide 1 (zinc acetate) to trap hydrogen sulphide gas. Subsequently, sulphide 2 (methylene blue) was added to form the blue-coloured complex (APHA et al., 2005). The absorbance of the resulting solution was measured using the colorimeter (HACH-DR900).

2.4 Data analysis

All data are shown as average ± standard deviation. Statistical analysis was performed using the SPSS statistical package (IBM V22.0). 95% was considered as the percentage of confidence and the selection is rooted in the balance between precision and reliability. Any P value higher than 0.05 was regarded as a validation of the null hypothesis, with no significant difference between the variables. This P value is also standard practice for minimising type I errors while maintaining statistical sensitivity. Repeated measurements ANOVA was preferred for assessing any significant variation between the variables at different temporal scales.

3 Results

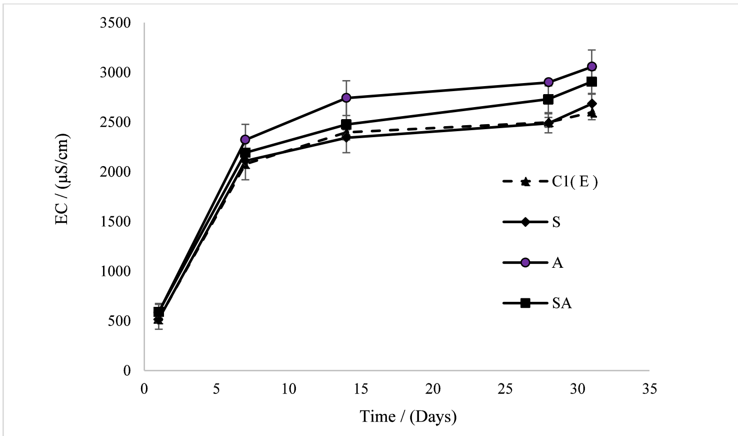
3.1 Experiment 1: application of attenuation processes without dilution

EC levels of all the mesocosms were almost similar (~500 μS/cm) on day 1 without a significant difference (RM ANOVA; $P > 0.05$) (Figure 3) and showed increasing trends up to the last day. Aerated mesocosms (A and SA) showed higher increments in EC levels than the mesocosm treated with stirring (S), but the differences were not statistically significant during the experimental period (RM ANOVA; $P > 0.05$). Ammoniacal nitrogen on the initial day was considerably high (> 0.25 mg/L) in all mesocosms [Figure 4(a)], however, all treatments reduced ammoniacal nitrogen by at least 90% (highest was by stirring with a 97% reduction). Initial nitrate-nitrogen concentrations were relatively low (<1 mg/L) and all mesocosms, except the stirred mesocosm, showed high percent increments relative to the initial [Figure 4(b)]. Aerated, and aerated with stirring showed approximately 110% and 362% increase, respectively. In contrast, there was a 38% reduction with stirring. Initial phosphate concentrations varied between 0.4 and 0.8 mg/l [Figure 4(c)]. Control and stirring showed a minor decrease in concentrations, whereas aeration, and aeration with stirring increased phosphate by 277% and 58%, respectively. This is an indication that aeration aided the diffusion of phosphates from the sediment. Initial sulphide concentrations were below 0.1 mg/L and temporally decreased in all mesocosms [Figure 4(d)]. Mesocosms treated with aeration and stirring showed a similar percentage (~50%) of removals, whereas the mesocosm that was subjected to aeration with stirring showed a higher removal percentage of 75%. Interestingly, the control gave the highest sulphide removal (78%), an indication that treatments given aided a certain amount of sediment-stored sulphide to diffuse into the water column.

3.2 Experiment 2: application of attenuation processes with dilution

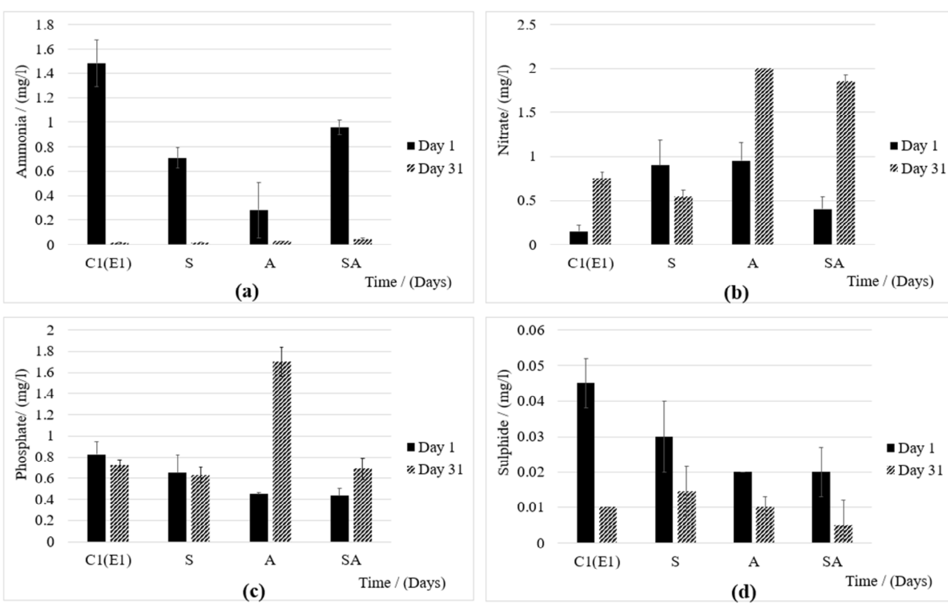
Initial concentrations of EC ranged from 310–1,600 μS/cm [Figure 5(a)]. In the control setups (C1 and C2), a gradual increment was identified in EC, and it was 98% and 140% respectively for C1 and C2. As expected, unlike in E1, all treated mesocosms in E2 followed fluctuation patterns with notable rises and falls in harmony with the dilution. Same as E1, no significant differences could be seen between mesocosms that were given a treatment (RM ANOVA; $P > 0.05$). A similar trend could be identified in the dry season [Figure 5(b)]. For both seasons, the treatment dilution provided the highest EC reduction of 70%. However, this is an ideal situation to assume a water column without any mixing, yet deep pool areas are possible.

Figure 3 Variation of EC in E1 treatment processes during the dry season



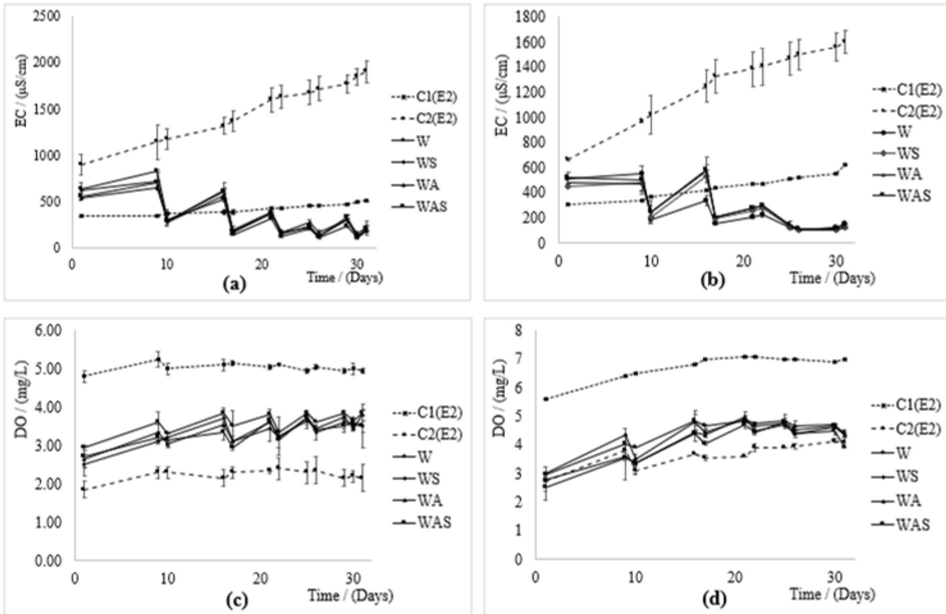
Note: C1 – canal water with sediments (control 1); C2 – water with sediments (control 2); S – canal water with sediments aided by stirring; A – canal water with sediments aided by aeration; SA – canal water with sediments aided by both stirring and aeration.

Figure 4 Variation of nutrients in E1, (a) ammonia nitrogen (b) nitrate nitrogen (c) phosphate (d) sulphide



Note: C1 – canal water with sediments (control 1); S – canal water with sediments aided by stirring; A – canal water with sediments aided by aeration; SA – canal water with sediments aided by both stirring and aeration.

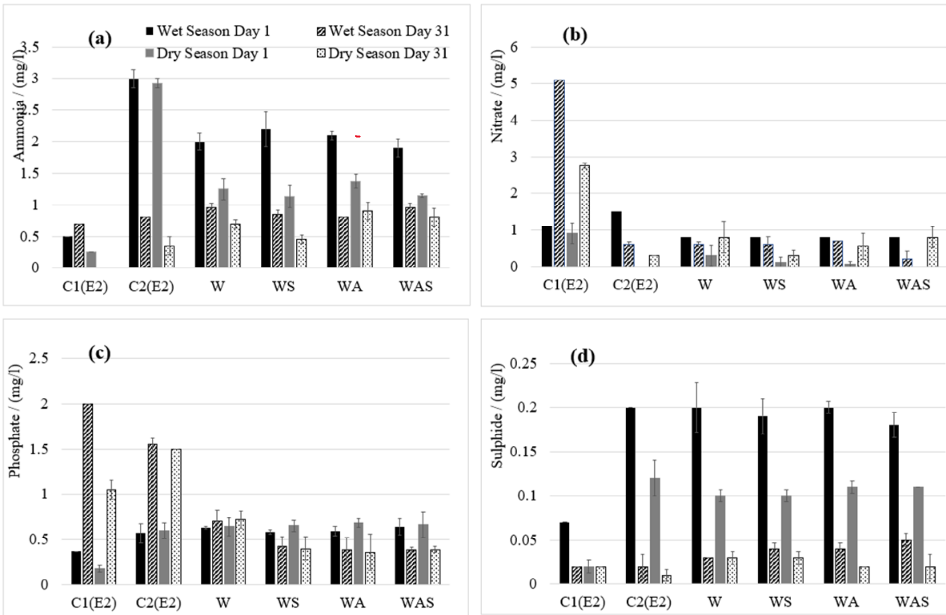
Figure 5 Seasonal variation of EC and DO in E2, (a) EC in the dry season (b) EC in the wet season (c) DO in the dry season (d) DO in the wet season



Note: C1 – canal water without sediments (control 1); C2 – canal water with sediments (control 2); W – canal water with sediments aided by dilution; WS – canal water with sediments aided by stirring and dilution; WA – canal water with sediments aided by aeration and dilution; WAS – canal water with sediments aided by stirring, aeration and dilution.

The DO concentrations in the treated mesocosms increased by the end of 31 days in both seasons and the trends were the same. Control without sediments (C1), showed the highest DO throughout while C2, canal water with sediment, and without any treatment showed the lowest DO throughout, and the differences were significant (RM ANOVA; $P < 0.05$). This was expected, as polluted sediment would consume more DO. In between the controls lie the mesocosms that were subjected to a treatment (attenuation processes with or without dilution), and these showed significant differences in many cases with the two controls (RM ANOVA; $P < 0.05$), an indication that treatment methods have an effect. Also in both seasons, mesocosms that were subjected to dilution only showed the lowest ultimate DO (3.9 mg/L) concentration and lowest improvement (32%) compared to other treated setups [Figure 5(c)] Aeration and stirring indicated the highest DO concentration (4.4 mg/L) and improvement (76%).

Figure 6 Seasonal variation of nutrients in E2, (a) ammonia nitrogen (b) nitrate nitrogen (c) phosphate (d) sulphide



Note: C1 – canal water without sediments (control 1); C2 – Canal water with sediments (control 2); W – canal water with sediments aided by dilution; WS – canal water with sediments aided by stirring and dilution; WA – canal water with sediments aided by aeration and dilution; WAS – canal water with sediments aided by stirring, aeration and dilution.

Figures 6(a)–6(d) show initial and final concentrations of ammoniacal nitrogen, nitrate, phosphate, and sulphide in E2 respectively. It is noticeable that initial nutrient contents in the wet season are comparatively higher than in the dry season, but in many cases without a significant difference (t-test; $P < 0.05$). Ammonium is the main nitrogen form discharged into the water from the sediment [Figure 6(a)]. On day 1, control setups without sediments (C1) showed the lowest ammoniacal nitrogen levels whereas the mesocosms with sediment showed four and three folds high ammoniacal nitrogen in wet and dry seasons, respectively. Among the treated mesocosms, mesocosm subjected to only stirring showed the highest reduction (60%) in ammoniacal nitrogen levels, and this was the case for both seasons. Hence, it could be considered the most effective setup for reducing ammoniacal nitrogen content, irrespective of the season. Initial nitrate levels in control setups C1 and C2 were considerably higher than other mesocosms in both seasons [Figure 6(b)]. This is an opposing trend compared to E1. Contrasting to ammoniacal nitrogen, the highest nitrate levels were indicated by the control C1 where there are no sediments. Nitrate contents were under 1 mg/l in both seasons in W, WS, WA, and WAS: where dechlorinated water was used [Figure 6(b)]. Similar to DO, in the treated mesocosms, nitrate contents were increased during the dry season while dropped during the wet season. No positive impacts could be seen through treatment methods: stirring, aeration, and water replacement on nitrate removal in both seasons. Initial phosphate contents fall below 1.0 mg/L in both seasons [Figure 6(c)]. Control setup C1: where no sediments were present indicated a fivefold rise in phosphate levels in both seasons while the control setup with sediments (C2) indicated a twofold increment. An overall decrease in phosphate levels irrespective of seasons could be observed in treated mesocosms excluding the mesocosm only aided with dilution. In contrast to the E1, the highest reduction in phosphate was obtained from aerated mesocosms and the result was similar in both seasons. The obtained result was not only due to the aeration process but also due to water replacement. However, in W where the only treatment method was water replacement, phosphate levels were not reduced in the dry season implying phosphate reduction cannot be obtained only through water replacement. Aerobic conditions must be satisfied for phosphate removal. Sulphide readings indicated a clear difference between the wet and dry seasons on day 1 [Figure 6(d)]; higher sulphide contents in the wet season contrasted with the dry season. Control mesocosm C1 had the lowest initial readings (0.07 mg/L in the wet season and 0.02 mg/L in the dry season) as there were no sediments, whereas the other mesocosms showed considerably higher initial sulphide contents. Aerated mesocosms (WA and WAS) were found to be the most effective mesocosm for sulphide removal in both seasons (~80%. However, results implied that sulphides do not require any treatment since control mesocosms also showed significant sulphide reductions (~90%) in both dry and wet seasons.

4 Discussion

4.1 Mechanisms of pollutant release from sediments

Sediments act as a sink for nutrients (phosphorus and nitrogen species) (Aigars and Carman, 2001), and such sediments in turn potentially depreciate water quality and result in continuous eutrophication in waterways (Cheng et al., 2014; Tang et al., 2014) when it starts to release pollutants to the water column, especially when the water column is disrupted such as by mixing. Release can take three major mechanisms: diffusion, convection, and resuspension (Zhao et al., 2018). Diffusion occurs when there is a concentration gradient of nutrients in the water-sediment interface and when sediment rests under static or running water. According to Srivastava et al. (2011) and Zhu et al. (2013), molecular or convective diffusion that happens when the water column is relatively static, governs the mass transfer of contaminants across the sediment-water interface, and there is a diffusive boundary layer where the mass transfer occurs. Zhu et al. (2013) further stated that convective diffusion occurs in the diffusive boundary layer. Unlike the static release which is dominated by molecular diffusion, frequent resuspension or settling of sediments controls the dynamic release (Fan et al., 2010). Resuspension increases the quantities of ammoniacal nitrogen, nitrate, and phosphorus ions added to the water column compared to the release by diffusion alone due to the direct release of these nutrients from the readily available pore-water and ion exchange fractions (Phillips et al., 2005). On the other hand, increased turbulence enhances water movement, promoting better contact between sediment and water. Turbulent diffusion, driven by irregular flow patterns, disperses and mixes substances in fluid, improving the distribution of dissolved materials in water (Ali et al., 2018).

4.2 Effect of treatment processes on water quality

Application of processes such as stirring, aeration, and dilution would trigger a two-step process on the fate of contaminants. Firstly, it promotes the diffusion of pollutants of sediment, and secondly, it transforms the original pollutant into less harmful species or even transports them out of the system (e.g., aeration removes dissolved gases to the atmosphere). It is necessary to have a delicate balance between these two, and the first process needs to be in harmony with the second unless the water column will be concentrated or oversupplied with pollutants. This is obviously unwanted and would create a situation worse than having polluted sediment, as an example exposing aquatic fauna to a much greater danger.

EC is a measure of charged ion concentration in water (Anhwange et al., 2012). Results suggest sediment acts as a catalyst, increasing EC. Control setups show gradual EC rise due to evaporation, enhanced by sediment presence. E1 results indicate increased EC irrespective of treatments due to getting concentrated with evaporation (Anhwange et al., 2012).

DO content is a vital factor in stream health, and 6.5 mg/L is the minimum acceptable as per Venkatesharaju et al. (1970) and Ranaraja et al. (2019) for the survival of many aquatic fauna. Thus, values observed in the range of 1–5 mg/L in our mesocosms are obvious indications of poor stream health. A less-than-proportional increase in the DO of mesocosms with sediment than mesocosms without sediments was because DO was consumed by microorganisms to decompose the organic matter (Hamid et al., 2020). Dilution improved DO more than aeration and stirring treatments, and dilution is known to reduce the chemical oxygen demand in river water (Cai et al., 2021).

Ammonium ions are the main nitrogen form discharged into the water from the sediment and this behavioural pattern was already reported by Cai et al. (2021). If ammoniacal nitrogen predominates in water, the nitrification process converts existing ammoniacal nitrogen into nitrite in the presence of oxygen and then to nitrate (Shen et al., 2016). The reduction of ammoniacal nitrogen contents irrespective of the treatment or season can be attributed to the fact that ammonia-oxidising bacteria generate nitrification by reacting with DO in the water (Zhang et al., 2021). Stirring eased ammoniacal nitrogen to leach out from sediments.

Unlike in EC and DO, dilution did not make a major impact on ammoniacal nitrogen removal. During the wet season, the initial ammoniacal nitrogen content was notably higher than in the dry season, primarily due to increased runoff during rainy periods, leading to the flushing of significant nutrient loads into the canal (Gomes et al., 2019; Gomes and Karunatilaka, 2022). However, the reduction in ammoniacal nitrogen levels during the dry season is comparatively lower than that observed in the wet season (Islam et al., 2015). This behaviour can be observed for all the nutrients considered.

Nitrate is an essential nutrient for reproduction, growth, and the survival of organisms. However, according to Johnson et al. (2000), nitrate levels higher than 1 mg/l are not good for aquatic life since it leads to eutrophication increasing algae growth. This ultimately reduces DO in the water (Murdoch et al., 2000). The increasing nitrate levels in E1 and E2 can be explained by ammonification-induced microbial death. The decomposing microbes under aeration and stirring enhance nitrification and increased the nitrate pool of the water column (Wang et al., 2012; Cai et al., 2021).

The maximum level of phosphate recommended for rivers and streams has been reported as 0.1 mg/L, while 0.025 mg/L could accelerate the eutrophication process in rivers (Adeyemo et al., 2008). Oxygen is an important factor affecting phosphate removal from sediments (House and Denison, 2002; Wu et al., 2014), by aiding phosphorus sorption and desorption at the sediment-water interface (House and Denison, 2002; Howell, 2010). Hence, E1 results are contrary to past research outcomes. Effective outcomes were gained through dilution in E2. With a combination of aeration or stirring together with water replacement, phosphorus contents were reduced. Phosphorus might have been leached from sediments due to stirring/aeration and was purified when water was replaced periodically. Therefore, in practical applications, achieving desired outcomes often necessitates the use of multiple treatment methods in parallel or sequential rather than relying solely on a single approach (Yu et al., 2022).

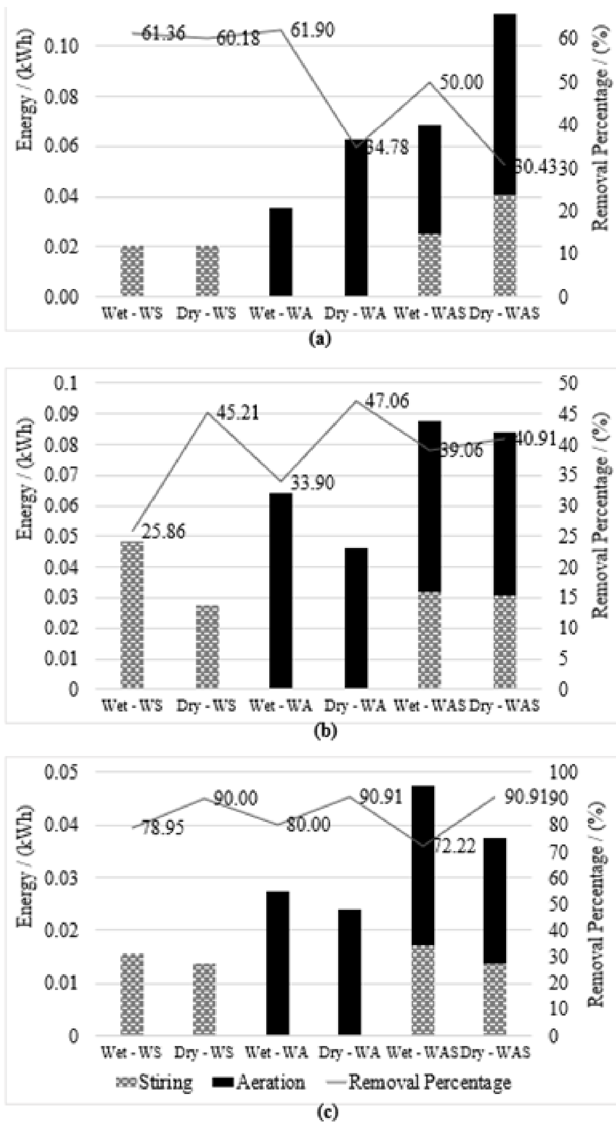
Sulphide presence in the water bodies comes partly from the decomposition of sulphur-containing organic matter and bacterial reduction of sulphates (Kularatne et al., 2003). Pore water usually contains 10–200 times higher H_2S concentration than the overlying water and it is presumed that H_2S produced in the surface sediment diffuses into the overlying water (Sakai, 2004). Kularatne et al. (2003) have stated that both bacteria and dissolved oxygen regulate sulphur-containing compounds in aquatic systems, and under oxygen-rich conditions, sulphide levels tend to reduce due to oxidation (Wilmot et al., 1988). Results in the present study showed that sulphides require no specific treatment as the control mesocosms also showed a significant sulphide reduction. The reason for this reduction might have been inherent microbial activities and the presence of oxygen at the water-sediment interface. However, the oxidation of hydrogen sulphide (H_2S) by oxygen is a gradual process that may require 2–4 hours or even days to complete (Tzvi and Paz, 2019).

4.3 Energy and water requirements, and cost implications

Cost-intensive dredging and desilting need to be replaced with energy and cost-effective processes that introduce heterogeneity to the regulated streams and rivers is a maxim in the current context. In our study aeration and stirring processes indeed aided sediment cleanup (surrogate with improved water quality), thus proving the technical feasibility. However, it is vital to know the financial commitment. Figures 7(a)–7(c) shows the energy requirement for a 1% improvement of the respective water quality parameter. Generally, aeration showed a higher energy requirement compared to the stirring process in mesocosms (Figure 7) and ranged between 0.03–0.06 kWh, whereas stirring ranged between 0.01–0.03 kWh. Therefore, the combination of aeration and stirring consumed 0.05–0.38 kWh. Ammoniacal nitrogen and phosphate removal are encouraged with a good oxygen level causing higher utilisation of energy in aerated setups. Sulphide removal required the least amount of energy compared to other nutrients [Figure 7(c)].

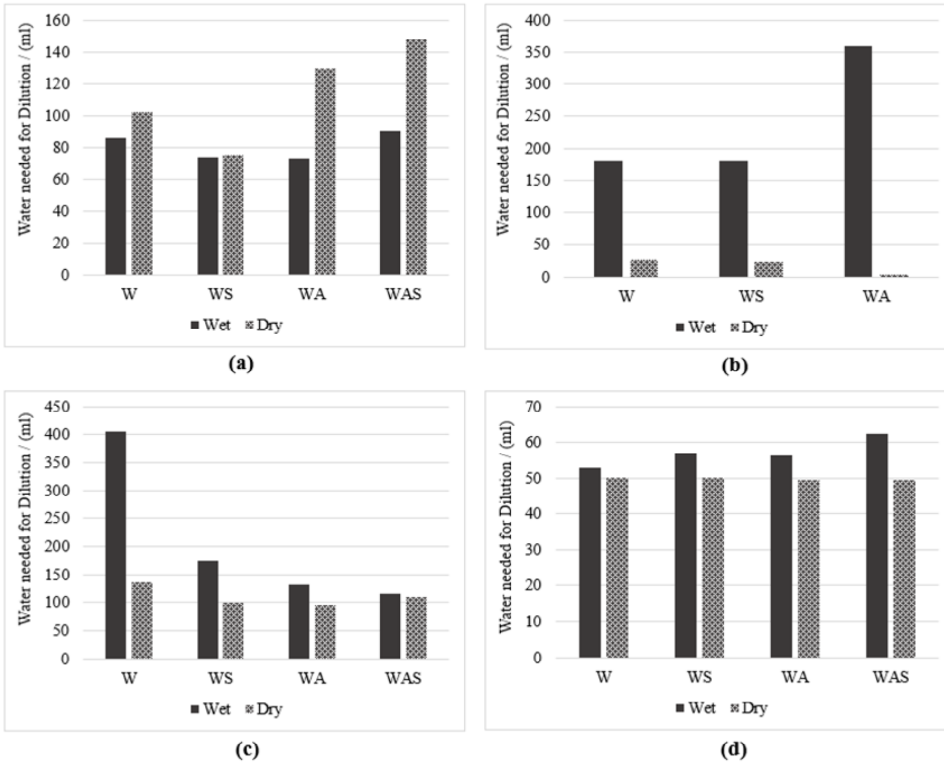
Looking at the water required for dilution purposes in the mesocosms for the removal of 1% of nutrients (or 1% improvement) [Figures 8(a)–8(f)], sulphide needed the lowest amount of water for the dilution compared to other nutrients. Even though ammoniacal nitrogen required a higher quantity of dilutions in the dry season, both nitrate and phosphate indicated a considerably higher requirement of water for the treatments in the wet season implying that the wet season produced critical conditions for water quality. When considering market rates in Sri Lanka, 1 kWh is 0.128 USD, and 1 L of freshwater (taken for construction works) is 0.65 USD and therefore costs of approximately ~0.05 USD, 0.04–0.25 USD, 0.06–0.27 USD and ~0.04 USD are needed for ammoniacal nitrogen, nitrate, phosphate, and sulphide removal respectively which are ranged within almost similar amounts. Dutch canal carries water with a flow rate of 8 m^3/s and therefore, an annual cost of ~14 million USD will need to be contributed to the canal water remediation.

Figure 7 Energy requirements in E2, (a) ammonia nitrogen (b) phosphate (c) sulphide



Note: WS – canal water with sediments aided by stirring and dilution; WA – canal water with sediments aided by aeration and dilution; WAS – canal water with sediments aided by stirring, aeration, and dilution.

Figure 8 Water requirements for dilution purposes in E2, (a) ammonia nitrogen (b) nitrate nitrogen (c) phosphate (d) sulphide



Note: C1 – canal water without sediments (control 1); C2 – canal water with sediments (control 2); W – canal water with sediments aided by dilution; WS – canal water with sediments aided by stirring and dilution; WA – canal water with sediments aided by aeration and dilution; WAS – canal water with sediments aided by stirring, aeration and dilution.

4.4 Limitations and recommendations

The study is constrained by a one-month timeframe, potentially limiting the assessment of long-term trends and seasonal variations in water quality dynamics. Focusing on a select set of water quality parameters, including DO, EC, nutrients, and sulphide, might result in the oversight of other potential indicators that could provide a more comprehensive understanding of the overall water quality. Moreover, confining the study to a single location along the Kirulapone Canal restricts the generalisability of findings. To address these limitations, the authors recommend that future research should consider extending the study duration, incorporating a more diverse set of water quality parameters such as heavy metals exploring multiple sampling locations to capture spatial variations, and investigating the influence of varying sediment-water depth ratios to assess how this factor influences water quality dynamics. Also using model rivers in flumes with flowing water should be considered.

5 Conclusions

This study critically examined the feasibility of attenuation processes induced by MPHs as a means of mitigating heavily polluted sediments in urban canals. While these approaches offer potential benefits in improving water quality to some extent, it becomes evident that relying solely on these strategies is insufficient to attain a healthy canal with good water quality. Dilution emerges as a pivotal factor, notably impacting EC, DO, and phosphate levels, leaving ammoniacal nitrogen unaffected. Intriguingly, the study revealed that ammoniacal nitrogen and sulphide removal can occur naturally without external interventions. A detailed analysis of energy and water requirements underscores that aeration, despite its higher energy consumption, should be carefully balanced with stirring for optimal results for ammoniacal nitrogen and phosphate treatments. The estimated cost of ~0.04–0.25 USD per unit percent purification for ammoniacal nitrogen, nitrate, phosphate, and sulphide remediation provides valuable cost-effectiveness insights.

The authors recommend extending the study duration, incorporating diverse water quality parameters, exploring multiple sampling locations, investigating varying sediment-water depth ratios, and using model rivers in flumes with flowing water to comprehensively understand and address the limitations, contributing to a more robust understanding of water quality dynamics in the urban canals. In essence, this study emphasises the vital role of dilution in effective water quality management for urban canals, paving the way for future research to explore complementary strategies and optimise existing approaches to ensure comprehensive water quality improvement.

Statements and declarations

Ethics approval

This manuscript includes an original work that has not been published elsewhere and has not been submitted to more than one journal for simultaneous consideration.

Consent to participate

All the authors of this work agree with the content and give their explicit consent to submit it.

Consent to publish

All the authors approve the version of this work to be published and agree to be responsible for all aspects to ensure that questions regarding the accuracy or completeness of any part of the work are properly investigated and resolved.

Funding

Financial support for this study was provided by the Faculty of Graduate Studies and Research (Grant No. FGSR-RG-2022-03).

Data availability statement

The data used in this research study are available from the corresponding author upon reasonable request.

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