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Floating Constructed Wetland for the Treatment of Polluted River Water: A Pilot Scale Study on Seasonal Variation and Shock Load

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ABSTRACT

This paper reports the performance of a pilot scale floating constructed wetland (FCW), employed for the treatment of polluted water collected from Buriganga river in Bangladesh. The FCW system included a tank for accommodating collected water and a floating mat with media, to support the growth of two macrophyte species Phragmites australis and Canna indica. Mean mass removal rates of 0.66, 0.76, 0.08, 0.51, 2.49 g/m²d were achieved for ammoniacal nitrogen (NH₄-N), total inorganic nitrogen (TIN), phosphorus (P), biochemical oxygen demand (BOD), chemical oxygen demand (COD), respectively by the FCW. Nitrogen removal was via nitrification-denitrification processes, whereas filtration–sedimentation appeared to influence phosphorus removal. The system achieved substantial Escherichia coli mortality rates, through protozoa predation and oxidation processes. Higher influent concentrations during dry period allowed greater removal of nutrients and E. coli. Hydraulic shock-loading experiment revealed critical interdependency between hanging roots maturity, input hydraulic, and pollutant loadings for maintaining stable performances.

Key words: Macrophyte; nutrients; organics; pollution control; treatment wetland.

1. INTRODUCTION

The discharge of various industrial and domestic effluents into surface waterways is severely damaging the aquatic environment in Bangladesh [1]. Natural purification is a viable route for maintaining the ecological health of the waterways [2]. However, in Bangladesh the strength of effluent discharge and pollutant load on the waterways are substantially higher than in many other countries [1]. Considering that the natural self-purification capacity of a waterway is limited, excessive pollutant loadings can cause severe and prolonged degradation of water
quality [3]. To protect the waterways, new regulations on wastewater treatment and discharge have been introduced. However, illegal wastewater discharge to local open channels has limited the effect of the new regulations. In addition to law enforcement, artificial strengthening of the purification ability of the water bodies is necessary for protecting the water environment. Constructed wetland, a low-cost green treatment technology is a viable option to enhance the pollutant removal abilities of open water channels. Several studies [4,5] reported the potential application of subsurface flow wetlands, for improving the water quality of polluted rivers. However, the implementation of subsurface flow wetlands for polishing polluted surface waters in Bangladesh is difficult, due to the lack of available land areas on the banks of rivers, as they are often used for agricultural activities.

Under such circumstances, FCW can provide a balance between the necessity of treating polluted river waters and shortage of land areas for agricultural activities. FCW systems include emergent macrophytes, supported by water buoyant mat structures [6]. The stems of the macrophytes remain above the water surface; their roots developing through water column beneath the floating mats [7]. Microbiological population develops around the hanging roots, forming a complicated network of roots and biofilms. Such complicated mesh produce net removal of pollutants as waterpasses through the network.

Different types of wastewaters had been studied for potential treatment by the FCW systems [7]. However, using FCW to treat polluted river water has not been thoroughly investigated. To date, only a limited number of studies have investigated the FCW systems for the treatment of polluted open water channels [8]. Significant variations of pollutant concentrations in the polluted river during dry and wet periods are common, as observed in surface water channels.
The influence of such variations on the effectiveness of FCW has not been critically examined. Shock loading pollutants is a common phenomenon in surface waters, caused by sudden increase of pollutant input by severe weather events in wet seasons, accidental pollutant spills or illegal discharges. Adverse impact of shock loading on the removal performances of subsurface flow wetlands had been reported [9], but previous studies have not carefully examined the effect of shock loading on FCW systems.

This study has been designed to assess the performance of a pilot-scale FCW, used for the treatment of polluted water collected from Buriganga river, which is one of the most polluted rivers in Bangladesh. The objectives of this study are two-folds: (a) to evaluate the routes and rates of the removal of nutrients, organics, and coliforms from the polluted surface water; and (b) to investigate the influence of seasonal variations and shock loadings on the removal.

2. MATERIALS AND METHODS

2.1 Collected river water

Polluted water was collected from Buriganga river near Najimuddirghat, Keraniganj, Bangladesh, and transported to the experimental site. The river water was stored in a feed tank, before being dosed into the experimental FCW system.

2.2 Construction of the pilot scale floating constructed wetland

The pilot scale FCW system had been established at Keraniganj industrial zone. Figure 1 represents engineering diagrams and pictorial plates of the pilot scale system. The length, width and depth of the tank (to accommodate the water volume collected from river Buriganga) were 3.60, 1.16, and 1.14 m, respectively (Figure 1a). The side walls (0.13 m thickness) of the water holding tank were constructed with brick, cement and sand materials. The bottom portion of the
tank had been water sealed with brick flat soil ing and cement concrete. The water holding capacity of the tank was 4.43 m$^3$ (4430 liter).
Locally available materials were selected for the construction of the floating mat. The outer edge of the floating mat was constructed with 4 inches diameter UPVC pipe (for achieving buoyancy). The length and width of the floating mat was 3.0 and 0.58 m respectively (Figure 1a). Nylon fiber mesh had been employed as the base material (of the floating structure) to support the load of media and macrophytes, and was further supported by four 1 inch diameter polypropylene random copolymer pipes (Figure 1a). Dry straw (0.01 m depth) overlaid by saturated clay soil (0.03 m depth) were employed as the media, to foster the growth of macrophytes. The floating mat covered 57.0 % of the free water surface inside the tank.

2.3 Macrophytes

Two types of macrophytes, *Phragmites australis* and *Canna indica*, were collected from local water channels, and planted into the media of the floating mat. After plantation the system was water logged for eight weeks, allowing the macrophytes to establish.
Figures 1b and 1c provide pictorial plates of the early stage (i.e. 1 month after plantation), matured stage (i.e. 8 months after plantation) macrophytes, and development of hanging roots (below the floating mat) during such stages. Headley and Tanner [10] reported maximum root depths of four macrophyte species (i.e. \textit{Carex virgata}, \textit{Schoenoplectus tabernaemontani}, \textit{Cyperus ustulatus} and \textit{Juncus edgariae}) within 0.57-0.87 m. The hanging root depths (beneath the mat) of \textit{Phragmites} and \textit{Canna indica} species in the experimental FCW was measured to be 0.84 and 0.76 m respectively, that fall within the reported ranges [10] of the root development of other species.

2.4 Operation of the system

After macrophytes establishment, the wetland was fed with Buriganga river water for a total period of 24 weeks (March - September). Water quality analyses across the system were performed during the last 21 weeks of this period.

Collected river water was dosed manually into the system once per day, 7 days a week. Collected river water flowed horizontally through the roots of the macrophytes towards outlet, maintaining a water depth of 0.99 m inside the tank (Figure 1a). The system received a hydraulic loading (HL) of 59.0 mm/d during the first 8 weeks of the analyses period. Input HL was suddenly increased (phase I shock loading) five folds (i.e. 295.0 mm/d) of the original HL (i.e. 59.0 mm/d) during week 9, followed by ten folds increment (i.e. 590.0 mm/d) of the original HL during week 10. Such sudden HL increments were performed, to study the impact of consecutive shock loadings on treatment mechanisms of the pilot scale FCW.

During week 11, HL was dropped to initial loading conditions (i.e. 59.0 mm/d) and was continued up to week 16. In week 17, HL was again suddenly increased (phase II shock loading)
7 folds (i.e. 413.0 mm/d) of the initial loading (i.e. 59.0 mm/d), followed by 14 folds increment (i.e. 826.0 mm/d) of the initial loading (i.e. 59.0 mm/d) in week 18. Such consecutive sudden increments were performed, to evaluate the efficacy of the FCW to encounter higher shock loadings (when compared with previous shock loadings), during later stages of the operational period. In week 19, HL was dropped to initial conditions (i.e. 59.0 mm/d), and was continued until week 21 (end week of water quality analyses campaign).

2.5 Water quality analyses

During experimental campaign, water samples were collected on a weekly basis from the inlet and outlet of the pilot scale FCW system. Forty two sets of samples were collected during 21 weeks of experimental analyses. Water samples were transferred immediately (after collection) to water quality testing laboratory, Department of Public Health Engineering, Government of Bangladesh for further analyses. For each sample, analysis was carried out for pH, dissolved oxygen (DO), alkalinity, total suspended solids (TSS), turbidity, ammoniacal nitrogen (NH$_4$-N), nitrite (NO$_2$-N), nitrate (NO$_3$-N), biochemical oxygen demand (BOD$_5$), chemical oxygen demand (COD), phosphorus (P) and faecal coliform (E. Coli). DO and pH values were measured by a HACH HQ40d probe. Alkalinity was measured by titrimetric method. TSS was measured with a portable HACH LXV (model 322.99.00002) meter, while turbidity was measured with a HACH 2100P turbidity meter. COD, P and nitrogenous compounds analyses were carried out using a HACH DRB200 reactor block and Shimadzu PharmaSpec UV-1700 spectrophotometer. BOD$_5$ measurement was carried out following 5 days incubation method with an incubator operated at 20°C. E. Coli was measured with Scharlau agar, and an incubator operated at 44.5°C.
2.6 Statistical analyses

Pollutant removal performances across the FCW system had been analyzed with statistical parameters minimum value, 25% percentiles, median, 75% percentiles, maximum value, mean and standard deviation. Minimum and maximum values denote the lowest and highest value (i.e. concentration/ removal percentages) for a given pollutant. The statistical parameter 25% percentiles indicate that one quarter of the total values (N) were lower than/equal to the value, as indicated by 25th percentile (for a given pollutant). 75% percentiles illustrate that three quarters of the total values (N) were lower than/ equal to the value, indicated by 75th percentile. Median value illustrates that half of the total values (N) were higher than the value as reported by median (for a given parameter). Standard deviation indicates the dispersion value of a particular data set for a given parameter.

The influence of seasonal variation (i.e. dry vs. wet season) on pollutant removal rates (g/m².d) in FCW system had been analyzed by t test. Prior to such test, different statistical analyses (i.e. Kolmogorov-Smirnov test, D’Agostino and Pearson omnibus normality test, and Shapiro-Wilk normality test) were performed, to examine whether data distribution approximated to normality (a prerequisite of t test). The results (data approximated to normality) were accepted when \( \alpha = 0.05 \). All statistical analyses had been performed using software GraphPad Prism (version 5.03).

The p values of t test compare the means between two groups with null hypothesis that both groups have same means. In this study the two groups had been categorized as the removal rates during dry and wet seasons for a given parameter (for example COD). A p value less than the threshold value (\( \alpha = 0.05 \)) illustrates null hypothesis rejection, indicating that the mean
difference is “statistically significant” between the two groups. In contrast, a p value greater than the threshold value (α=0.05) indicates that the difference between the means of the two groups is not statistically significant.

3. RESULTS AND DISCUSSION

3.1 Overall performance

Table 1 summarizes overall pollutant removal performances of the pilot scale FCW during the experimental campaign. The mean pH and DO values increased in the effluents of the FCW. Mean removal percentages of NH$_4^-$-N, NO$_2^-$-N, P and E. Coli were higher, compared with other parameters. Minor increase of NO$_3^-$-N concentration was observed in the effluent. Mean influent organic concentrations were lower, associated with lower removal percentages in the FCW. The experimental wetland was not efficient in terms of removing suspended solids (from river water), as illustrated by very low removal performances.

Table 1. Pollutant removal profiles across the pilot scale FCW system (N=21).
Biological removal of organics in wetland systems is influenced by biodegradability of input organic contents [11]. Organic matter biodegradability is expressed by BOD/COD ratio. A ratio of 0.5 or greater indicates that the wastewater contains easily biodegradable organic compounds, whereas, a ratio lower than 0.3 illustrates presence of slowly/hardly decomposable
organic contents [12]. The influence of biodegradability on organics removal routes (in the FCW system) has been demonstrated in Figure 2; showing correlation plots of input $\text{BOD}_5/\text{COD}$ ratios, organic ($\text{BOD}_5$, COD) removal rates (g/m$^2$.d), and effluent-influent ratio ($C_e/C_o$) of organic concentration, as the river water passed through the roots of the FCW.

The correlation plots of Figure 2 illustrate: (a) input $\text{BOD}_5/\text{COD}$ ratios across the FCW were extremely lower (<0.3); and (b) no clear correlation trends were observed between input ratios and measured organic removal rates; however lower organic effluent-influent ratio ($C_e/C_o$) values coincided with greater input ratios, indicating the dependency of biological organic removal routes on input biodegradability. Presence of recalcitrant organic compounds (due to lower input BOD/COD ratios - Figure 2) in the collected river water may have hindered the growth of organic degrading microbes (around the roots of the macrophytes), resulting in reduced organic removal performances (Table 1). Increment of mean dissolved oxygen concentration (Table 1) in the effluent (across the FCW) also supports these findings, as aerobic organic degradation is associated with consumption of dissolved oxygen [9]. Sun and Saeed [13] reported that microbiological organic removal routes are not dominant in constructed wetland systems, that receive wastewater with $\text{BOD}_5$ concentration $\leq$ 30 mg/L. Mean $\text{BOD}_5$ concentration of the collected river water was much lower (Table 1) than the reported value, which could have promoted other organic removal mechanisms over biological pathways [13].
Fig 2. Correlation plots of input BOD$_5$/COD ratios, organic removal rates (g/m$^2$.d), and organic effluent-influent concentration ratio (C$_e$/C$_o$).

NH$_4$-N removal performances were substantially higher (when compared with organic removals—Table 1), as the collected polluted river water passed through the root network. Major NH$_4$-N removal pathways in constructed wetlands include nitrification, plant uptake, ammonia volatilization, adsorption by substrate and biomass assimilation [14-16]. The influence of the latter three mechanisms on NH$_4$-N removal performances (in experimental wetland) might be ruled out, due to lack of free water surface, media with cation exchange properties and higher organic removal performances [14,17,18]. As such, the former two processes might have controlled NH$_4$-N removal pathways, during the passage of collected river water through macrophyte roots (in the pilot scale FCW).
Nitrogen uptake by plants differs according to the system configurations, loading ranges, wastewater types and environmental conditions. The contribution of plants, in terms of nitrogen removal has been reported within the range 0.5–40.0% of the total nitrogen removal [18]. Such removal might be an important factor in large scale wetlands with higher amount of plant biomass.

The removal of NH$_4$-N via nitrification process (in wetland systems) is dependent on two factors: (a) presence of DO and; (b) alkalinity content of the incoming wastewater [15]. Oxygen availability (inside the experimental FCW) was not a limiting factor to restrict nitrification, due to oxygen rich environment (Table 1). The influence of alkalinity on NH$_4$-N removals has been illustrated in Figure 3a, which depicts correlation plot of influent alkalinity (mg/L) and NH$_4$-N removal rates (g/m$^2$.d). As observed in Figure 3a, higher influent alkalinity values coincided with greater NH$_4$-N removal rates indicating enhanced removal rates in alkalinity rich conditions, and are in agreement with biological nitrification mechanism [15].

![Correlation plots of influent alkalinity (mg/L) vs. (a) NH$_4$-N removal rates (g/m$^2$.d); (b) TIN removal rates (g/m$^2$.d); and (c) Observed vs. theoretical N removal rates (g/m$^2$.d).]
Mean NH$_4$-N removal (in the FCW) was recorded to be 10.5 mg/L, whereas, NO$_3$-N accumulation (i.e. difference between effluent and influent values) was calculated to be 0.7 mg/L (Table 1). Such minor accumulation in the effluents (Table 1) further supports nitrification to be the major NH$_4$-N removal mechanism (from the collected river water), while flowing through the roots of the macrophytes.

It should be noted that increase of NO$_3$-N concentration in the effluent was not detected throughout water quality analyses campaign. NO$_3$-N removal performances (38.0%) were detected in the effluents of 15 weeks (of water quality analyses campaign), whereas, NO$_3$-N generation rates (-178.0%) were observed in the effluents of 6 weeks. Increase of NO$_3$-N in the effluents was observed, when influent NH$_4$-N concentrations (across the FCW) were substantial. Influent-effluent analyses of total inorganic nitrogen concentration (TIN- summation of NH$_4$-N, NO$_2$-N and NO$_3$-N concentrations), signify 46.0% mean TIN removal from river water employing FCW. These results imply that a majority portion of the nitrified NH$_4$-N (along with incoming NO$_3$-N-Table 1) had been removed. Increase of mean alkalinity concentration in the effluent (Table 1) supports that, denitrification might have controlled nitrogen removal (in the FCW), as such mechanism generates alkalinity [15]. In addition, the correlation plot (Figure 3b) of influent alkalinity (mg/L) vs. TIN removal rates (g/m$^2$.d) follows similar trend (i.e. coinciding of greater TIN removals with higher alkalinity values) of Figure 3a (influent alkalinity vs. NH$_4$-N removal rates correlation plot). Such observed similar trend illustrates that the nitrified products (i.e. NO$_3$-N) were subsequently denitrified in the experimental FCW.

Denitrification process is often limited by lack of organic carbon [19,20]. Biodegradable organic contents (of wastewater) might provide required organic carbon to facilitate denitrification in constructed wetlands [18], which could not be fulfilled by the hardly
decomposable organic contents of the collected river water (in this study). This is supported by
the correlation plot Figure 3c, that represents correlation trend between theoretical and observed
nitrogen removal rates (g/m².d). The calculation was based on the requirement of 5-9 g of BOD
for the removal of 1 g N [21]. The theoretical N removal rates were calculated on the basis of
observed BOD₅ removal rates. According to Figure 3c, no correlation was observed between
theoretical and observed nitrogen removal rates, indicating that incoming BOD load (of the river
water) was incapable to support nitrogen removal in the experimental wetland.

The contribution of incoming organic load to facilitate denitrification (in experimental
FCW) is further investigated with input COD (an indicator of organic carbon). Figure 4
represents correlation plots of input COD/NO₃-N ratio vs. NO₃-N removal rates (g/m².d),
effluent-influent NO₃-N ratio (Cₑ/Cₒ), and COD/TIN removal rates vs. TIN removal rates
(g/m².d), effluent-influent TIN ratios (Cₑ/Cₒ). According to Figure 4, the correlation plots did not
show any particular trends, supporting that incoming organic load did not support denitrification
as discussed in the previous paragraph.
Fig. 4. Correlation plots of input COD/NO$_3$-N ratios vs. NO$_3$-N removal rates (g/m$^2$.d), effluent-influent NO$_3$-N ratio (C$_e$/C$_o$), and input COD/TIN ratios vs. TIN removal rates (g/m$^2$.d), effluent-influent TIN ratios (C$_e$/C$_o$).

The independency of denitrification on the incoming organic load suggests that, the required organic carbon (for NO$_3$-N reduction) might have been provided from internal sources of the FCW. Plant derived materials such as rice husk, sugarcane bagasse [22,23] had been employed (as media) previously in subsurface flow wetlands, to support denitrification (via internal carbon leaching). As such, the hanging root network (Figure 1c) might have supplied required organic carbon, via root secretion and decomposing plant biomass [6,24] during the passage of river water through the rhizosphere.
It should be noted that, increment of DO concentration in the effluent (of the experimental FCW) could be linked to oxygen contribution from other sources. Considering the theoretical oxygen requirement for nitrification and limited organic degradation in the experimental FCW, such contributions were higher than DO concentration values detected in the effluents (Table 1). Atmospheric oxygen diffusion and oxygen leakage through rhizosphere are two major oxygen contributing sources in constructed wetlands. However, atmospheric oxygen diffusion might not have played the major role to enrich DO concentration (in the FCW), as the floating mat covered a majority portion of the open water surface (Headley and Tanner, 2012). As such, oxygen supply from the rhizosphere (of the FCW system) might have enriched DO content of the collected river water, while flowing through the hanging roots. However, oxygen supply by macrophytes is considered to be negligible in subsurface flow wetlands[25,26].

In subsurface flow wetlands, the matrix occupies a majority portion of the available wetland area, while roots of macrophytes develop through the void spaces of the employed media. In contrast, FCW systems lack of media and the hanging roots form a bulk rhizosphere; incoming water is forced to pass through such network. It could be possible that the dense rhizosphere (of the experimental wetland- Figure 1c) leaked substantial oxygen, thereby increasing DO content of the river water during its flow (through rhizosphere). Simultaneously, such hanging network might have promoted micro-gradient anoxic zones in the deeper portions (of rhizosphere), allowing coexistence of aerobic-anaerobic zones to support nitrification and denitrification [27].

3.3 Removal of coliforms and phosphorus

In subsurface flow and surface flow wetlands, coliform removals are achieved via protozoa predation, oxidation in aerobic environment, filtration/sedimentation of the suspended
solids containing adsorbed coliforms, and destruction via penetrated UV radiation in the water column [28-31. In the experimental FCW the latter two mechanisms might not have influenced observed coliform removals due to: (a) reduced suspended solids removal percentages (Table 1); and (b) coverage of water surface by the floating mat, hindering penetration of UV radiation inside the water column. As such, oxidation process (due to oxygen rich conditions -Table 1), along with protozoa predation could have controlled coliform removals in the FCW.

The removal of phosphorus in FCW systems is generally achieved via direct phosphorus uptake by macrophyte roots, phosphorus storage in plant biomass and filtration of phosphorus bound sediments by the hanging roots, followed by settling on the bottom portion [6,32]. Cui et al. [33] reported uptake of phosphorus by plant roots within the range 0.43-1.40% of the total phosphorus removal in wetland systems. As such, filtration (by roots) of P bound sediments, and subsequent settling process might have controlled phosphorus removal (Table 1) in the experimental FCW, and has been further explained in section 3.5 (of this paper).

3.4 Effect of seasonal variation

The overall water quality analyses campaign (i.e. 21 weeks) covered dry (March- middle of June) and wet seasons (Middle June- September), to observe the influence of seasonal variation on FCW performance. The temperature varied between 27 - 39°C within such periods. Table 2 denotes mean concentration and removal variation (across the experimental wetland) between dry and wet seasons. Subsequently, Table 3 represents statistical analyses (described in section 2.6) of such variations.

As observed in Table 2, nutrients (N, P), solids and E. Coli concentrations of the collected river were higher during dry period and lower in wet season. The FCW achieved higher NH₄-N
removal percentages during dry season (Table 2); the removal rates difference (between two seasons) was statistically significant (Table 3). Higher influent NH$_4$-N concentration might have enhanced removal performances in the experimental FCW, as greater input loadings often increase microbiological population inside biological reactors [34]. These results support the findings of Figure 3, strengthening that nitrification might be the major NH$_4$-N removal route in the FCW.

Increment and reduction of mean NO$_3$-N concentrations were observed in the effluents, during dry and wet seasons respectively (Table 2); such difference was not statistically significant (Table 3). Accumulation of NO$_3$-N concentration (during dry period) in the effluents does not indicate absence of denitrification in the FCW, as NH$_4$-N removal was higher and increase of NO$_3$-N concentration (in the effluent) was not substantial (Table 2). Total inorganic nitrogen removal analyses illustrate higher nitrogen reduction (from the collected river water) in dry period (average removal 21.4 mg/L N), when compared with wet period (mean removal 2.1 mg/L N). Difference of mean effluent alkalinity concentration increment (over incoming concentration) was higher (128.0 mg/L) during dry period, than that of wet period (61.0 mg/L), supporting the possibilities of higher denitrification rates in dry period. As such it could be stated that higher nitrification rates (Tables 2 and 3) might have produced substantial nitrates (during dry period), which were partially denitrified due to lack of sufficient internal carbon source.

A study by Breen [35] reported that lower input N loading might enhance nitrogen uptake by plants along with microbiological routes in constructed wetlands, due to competition between plants and microbes for available N metabolism. However, reduction of removal performances at
lower input loading (Table 2) during wet period suggest that, microbiological pathways might
have influenced nitrogen removals, as the river water passed through the hanging roots.

The removal performances of phosphorus and E. Coli were higher in the experimental FCW
during dry season (Table 2), following the trend of nitrogen removal. Phosphorus removal rate
difference was statistically significant between the two seasons, whereas, E. Coli removal rate
difference was not statistically significant (Table 3).

Influent organics concentration (Table 2) and removal performances (not statistically significant-
Table 3) were higher in the FCW during wet period. Mean input BODs/COD ratios in dry and
wet seasons were 0.16 and 0.20 respectively. Minor increase of such ratios (during wet season)
and influent concentration could have promoted limited organic biodegradation (due to lower
ratios) in the experimental FCW, resulting better performances. Mean DO concentrations
increment (i.e. difference between effluent and influent values) in wet and dry periods were 0.11
mg/L and 0.32 mg/L respectively, despite higher incoming DO concentration during wet season.
Such lower DO concentration increment (in the effluent) during wet period also supports the
existence of limited aerobic organic biodegradation route inside the reactor.

A few research studies investigated the performance of subsurface flow wetlands, in removing
pollutants from polluted river water. Dong et al. [4] observed 81.0, 87.0, 37.0 and 57.0% COD,
NH₄-N, P and TN removal percentages respectively in vertical flow wetlands, that received
heavily polluted river water with influent concentrations 65.0–158.0, 3.5–10.6, 0.6 –3.85, and
5.8–12.7 mg/L respectively. Mean concentrations of such pollutants (except COD) in the
collected river water (in this study) were higher during dry period (than the reported values)
(Table 2). Nevertheless, the removal percentages achieved by the FCW were almost similar to
the values reported by Dong et al. [4]. Another study by Wang et al. [5] observed 7.0 - 38.8, 40.2- 68.9, 54.2 -86.1, and 40.8 - 78.1% COD, NH$_4$-N, P and TN removal percentages respectively in bio-rack wetlands, that received low concentrated polluted river water with mean influent concentrations 8.27, 1.28, 0.154, and 3.05 mg/L respectively. The removal performances (during wet period) of NH$_4$-N and COD observed in the experimental FCW fall within such reported ranges (Table 2). These findings point out the potential application of FCW systems for the treatment of polluted surface waters in Bangladesh, and should be investigated further in future studies.

Table 2. Pollutant removal profiles across the experimental FCW during dry and wet periods.

<table>
<thead>
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Table 3. Statistical analyses of the removal rates in the FCW.

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<td>NH$_4$-N</td>
<td>g/m$^2$.d</td>
<td>1.48</td>
<td>0.13</td>
<td>0.0002</td>
<td>Yes</td>
</tr>
<tr>
<td>NO$_3$-N</td>
<td>g/m$^2$.d</td>
<td>-0.21</td>
<td>0.21</td>
<td>0.0709</td>
<td>No</td>
</tr>
<tr>
<td>P</td>
<td>CFU/m$^2$.d</td>
<td>0.16</td>
<td>0.04</td>
<td>0.0067</td>
<td>Yes</td>
</tr>
<tr>
<td>BOD$_5$</td>
<td>mg/L</td>
<td>0.02</td>
<td>0.81</td>
<td>0.0808</td>
<td>No</td>
</tr>
<tr>
<td>COD</td>
<td>mg/L</td>
<td>0.23</td>
<td>3.7</td>
<td>0.1985</td>
<td>No</td>
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<tr>
<td>TSS</td>
<td></td>
<td>0.17</td>
<td>-0.19</td>
<td>0.3560</td>
<td>No</td>
</tr>
<tr>
<td>E.Coli</td>
<td>CFU/m$^2$.d</td>
<td>8.8×10$^5$</td>
<td>2.4×10$^5$</td>
<td>0.1587</td>
<td>No</td>
</tr>
</tbody>
</table>
3.5 Impact of shock loading on system performance

Figure 5 illustrates the performance of the FCW during successive shock loadings in phase I (weeks 9 and 10) and phase II (weeks 17 and 18). A brief description on experimental hydraulic shock loadings is available in section 2.4 (of this paper).

According to Figure 5, NH$_4$-N removal percentages reduced significantly (in the FCW), with increase of successive hydraulic shock loadings during phase I shock loading. The deviation between input loading and removal rates was greater with amplification of hydraulic shock loading. These results are in agreement with [9], where reduced NH$_4$-N removal performances (from 91.7% to 61.6%) were recorded in tidal flow wetland, due to increase of influent NH$_4$-N concentration from 60.0 mg/L to 120.0 mg/L. The authors concluded that the contact time (between wastewater and microbes) was not adequate under greater loading, resulting diminished performances.

However reduction of NH$_4$-N removal performances were not observed (in the FCW) during successive hydraulic shock loadings in phase II, despite being operated under higher HL ranges (when compared with phase I-section 2.4). NH$_4$-N removals ranged between 40.0-56.0% within successive increase of hydraulic shock loadings (Figure 5), indicating stable performances. Phase II shock loading was applied during later stages of the water quality analyses campaign. It is likely that such stages might have allowed more matured growth of the hanging roots that provided a buffer against sudden HL increments, thereby maintaining adequate contact time. Lower input NH$_4$-N loading values (due to lower influent concentrations) might have additionally supported removal performances, despite greater sudden hydraulic inputs in phase II.
shock loadings. These results represent that the applied rapid loads were within the nitrification capacity ranges of the attached biofilms.

The removal percentages of NO$_3$-N and TIN (in the experimental wetland) followed NH$_4$-N removal trends in both phases of shock loading. Such phenomena signifies that denitrification (during shock loadings) was critically dependent on the production of NH$_4$-N (via nitrification process).

Organics (BOD, COD) removal rates and percentages increased sharply during successive hydraulic shock loadings in phase I (Figure 5). Increase of input organic loadings might have accelerated biological removal routes (in the FCW) which were observed to be limited in general, due to presence of hardly degradable compounds (Figure 2). Substantial input organics loadings during shock loadings of phase II (when compared with phase I) might have reduced organics removal performances (Figure 5); however such reduction was not significant.

The removal of phosphorus showed notable trends during both shock loading phases. Generation of phosphorus was recorded during consecutive hydraulic shock loadings (i.e. weeks 9 and 10-Figure 5) in phase I. In contrast, removal performances were observed during such consecutive HL increments (i.e. weeks 17 and 18) in phase II. It could be possible that, sudden application of substantial hydraulic loadings (during phase I shock loading) resulted turbulence inside the wetland reactor, fostering resuspension of previously settled phosphorus bound sediments.

Subsequently, the presence of matured bulk hanging roots (during phase II shock loadings) prevented such water turbulence and resuspension of sediments, thereby maintaining removal performances. These results further confirm that filtration (by rhizosphere), and subsequent
sedimentation of phosphorus bound sediments [32] were the main phosphorus removal mechanisms in the experimental FCW (section 3.3).

Greater reduction of E. Coli removal performances was observed during consecutive hydraulic shock loadings in phase II (Figure 5), which were not observed in the previous phase. It should be noted that, sudden input pollutant (i.e. E. Coli) loadings (during HL shock) were substantially higher in phase II (Figure 5). Such greater input values might have been responsible for observed diminishing performances.

In general, the hydraulic shock loading phases indicate critical interdependency between hydraulic loading increment, associated input pollutant loadings increase, and maturity of the hanging roots in the experimental FCW, for encountering the impact of sudden loading amplification within applied ranges.
Fig. 5. Pollutant removal performances during shock loading phases.
3.6 Correlations of pollutant mass loading and removal rates

Figure 6 illustrates correlations of nitrogen, organics, phosphorus (g/m².d) and E. Coli loading input vs. removal profiles, as the river water passed through the rhizosphere of the FCW. The correlation between input loadings and corresponding removal rates had been determined by statistical parameter coefficient of determination (R²). In general, positive correlations (as indicated by higher R² values- Figure 6) between input loading increments and removal rates, signify greater removal performances at higher loadings in the experimental FCW. Other research studies had also reported similar trends in subsurface flow wetlands [13,34].

It should be noted that NH₄-N removal rates deviated from the correlation trend line at upper input loading ranges (Figure 6). Such observations depict that nitrification rates (in the FCW) might have been affected by greater loadings in the experimental wetland. NO₃-N and TIN input loading and removals also followed similar trend of NH₄-N input- removal correlation plot (Figure 6). These results support that nitrate reductions were critically dependent on production rates (via nitrification), and are in agreement with the findings of Figure 3.

A study by Li et al. [8] recorded N and P volumetric removal rates (g/m³.d) as 0.46 g/m³.d and 0.12 g/m³.d respectively, from eutrophic lake water employing FCW systems. The experimental FCW system achieved 0.61g/m³.d NH₄-N, 0.75 g/m³.d N, and 0.08 g/m³.d P removal rates. Nitrogen removal rates achieved in the experimental FCW system are greater, whereas P removal rates are closer to such reported values. Another study by Tang et al. [3] reported 73.86-75.56% NH₄-N removal performances in a ditch–wetland–pond system (operated under similar HL ranges of the experimental wetland), employed for the treatment of polluted river water.
These findings also enlighten the potential application of FCW, to provide treatment of polluted surface water channels in Bangladesh.

Fig. 6. Correlation plots of input loading vs. removal profiles in the experimental FCW.
4. CONCLUSION

Nitrification-denitrification processes were the dominant route by the FCW to remove nitrogen from polluted river water. Denitrification was supported by internal supply of organic carbon from the hanging roots. Filtration of phosphorus bound sediments by the hanging roots, and subsequent sedimentation fostered phosphorus removals. The presence of non-biodegradable compounds in polluted river water restricted organics removal by the FCW. The FCW achieved greater removal of nutrient and E. Coli in dry period, and higher mass removal of organics in wet period due to higher influent concentrations. The performance of the FCW was critically influenced by the maturity of the hanging roots of its macrophytes, when the system experienced sudden increases of hydraulic and pollutants loads.

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REFERENCES


