The use of vertical flow constructed wetlands for the treatment of hyper-eutrophic water bodies with dense cyanobacterial blooms

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ABSTRACT

Eutrophication often leads to the periodic proliferation of harmful cyanobacterial blooms (HCBs), which threaten the sustainability of freshwater ecosystems and lead to serious environmental, health and economic damage. Hence, it is vitally important to take effective measures to manage HCBs and associated problems. In this study, vertical flow constructed wetlands (CWs) were operated under different hydraulic loading rates (HLRs) to treat a hyper-eutrophic water body with HCBs. Six sampling ports (representing different layers) were evenly distributed along the water flow direction to study the purification processes of CWs. With HLRs ranging from 0.2 m/d to 0.8 m/d, total nitrogen (TN), total phosphorus (TP), COD, total suspended solid (TSS) and Chlorophyll a (Chl.a) were efficiently treated by CWs, and they were mainly removed at the second layer of CWs. The concentrations of two cyanobacterial metabolites (geosmin and β-cyclocitrinal) in the effluent were mostly below their odorous threshold concentrations. As the HLRs increased, the treatment efficiencies of the CWs decreased gradually. There was no removal of TP, Chl.a, geosmin, or β-cyclocitrinal at an HLR of 1.0 m/d. Under suitable HLRs, this type of CW could provide a promising way to control HCBs and associated odorous problems in hyper-eutrophic water bodies.

Key words | constructed wetlands, harmful cyanobacterial blooms, hydraulic loading rates, odorous compounds, purification processes

INTRODUCTION

It is widely accepted that eutrophication favors periodic proliferation and the dominance of harmful cyanobacterial blooms (HCBs) (Paerl et al. 2011). The frequency, intensity, and duration of HCBs can be increased by global climate change due to global warming and higher frequencies of storms (Moe et al. 2016). In addition, cyanobacterial colonies with sheaths, mucilage and gas vesicles can prevent cells from zooplankton grazing, and improve buoyancy control, all of which lead to a competitive advantage over other phytoplankton genera (Yang et al. 2016). Therefore, under appropriate conditions of nutrients and weather, cyanobacterial colonies rapidly propagate and develop HCBs at the water surface.

The resulting HCBs are spreading globally and becoming an important environmental issue. The HCBs will form a dense mass that prevents incoming sunlight and depletes oxygen from water bodies (Piontkovski et al. 2012). This outcome will lead to poor energy flow to higher trophic levels due to their relatively poor edibility for zooplankton (Elliott 2012), and produce secondary metabolites (e.g. cyanotoxins and odorous compounds) that increase the cost of drinking water treatment (Ho et al. 2007) and cause great economic losses to the aquaculture industry (Tucker 2000); this outcome will also lead to anoxia near the sediment and encourage the releasing of phosphate and toxic metals (Stroom & Kardinaal 2016). In short, HCBs can threaten the sustainability of freshwater ecosystems, resulting in serious environmental, health and economic damage.

To avoid cyanobacteria proliferation, possible solutions including preventive measures and control measures need to be taken. The preventive measures are aimed at improving aquatic ecosystems, while the control measures are
used to reduce HCBs in a direct manner (Stroom & Kardinaal 2016). The improvement of the aquatic ecosystem could be realized by engineering measures such as sludge dredging, macrophyte restoration, increased flushing, and food web manipulation (Paerl et al. 2016), which will take effect following a long-term process. In an emergency situation, control measures are preferred to reduce the HCBs in a direct manner. The in situ application of algicides (Schrader et al. 2005), flocculants (Pan et al. 2006) and ultrasound (Hao et al. 2004) were reported to be effective in controlling HCBs. However, algicides may have side effects for the environment, specifically on water quality and ecological safety (Zhou et al. 2010). Although cyanobacterial cells will sink to the sediment after dosing of some flocculation agents, wind-induced mixing may result in their resuspension (Stroom & Kardinaal 2016). Ultrasound is employed to destroy gas vesicles to deposit the cyanobacterial cells on the bottom (Hao et al. 2004), but it consumes too much energy and might not efficiently inactivate algal cells. Compared with the above in situ methods, ex situ methods might have greater potential in the control of HCBs.

One of the ex situ methods, the constructed wetland (CW), has been gaining attention globally because of its high purification performance and low initial investment. Zhong et al. (2011) reported that a vertical flow CW system was able to manage HCBs in catfish ponds via three distinct processes: they could prevent excessive nutrient accumulation and improve water quality, remove cyanobacterial cells efficiently by filtration, and prevent the development of dense algal patches by water mixing effects. However, some studies also identified several disadvantages of CWs, such as clogging problems (Hua et al. 2010) and the large land areas required (Chang et al. 2012). In general, high suspended solids in the influent might lead to wetland clogging problems after long-term operation (Hua et al. 2010). However, whether CWs will be clogged by high influent cyanobacterial biomass has not been reported previously. A study of the purification processes of CWs is essential to locate the layer that may be clogged and to disclose the mechanisms of HCB management by CWs. The land area required for CWs is correlated with the hydraulic loading rate (HLR). High HLRS will contribute to decreases in required land area. Nevertheless, high HLRS might influence the purification performance of CWs that are treating hyper-eutrophic water bodies containing HCBs. A suitable HLR will be beneficial to maintain the balance between the land area required and the purification performance. This study will focus on the above two aspects and explore the possibility of treating hyper-eutrophic water bodies with dense cyanobacteria blooms using vertical flow CWs.

MATERIALS AND METHODS

Constructed wetland

A schematic diagram of two replicated vertical flow CWs used in this study is shown in Figure 1. The wetland cells consisted of concrete rectangular tanks with dimensions of 2.0 m (length), 1.0 m (width) and approximately 1.0 m (depth). Along the water flow, each wetland cell was separated into a down-flow chamber and an up-flow chamber. The chambers were filled to a depth of 80 cm with gravel, with diameters ranging from 10 mm to 20 mm, and were then planted with Canna indica. The filled chambers were evenly divided into six layers (W1-W6) to study the purification processes inside the CWs. The CWs were operated under an intermittent operation mode over four consecutive experimental periods (at HLRS of 0.2, 0.4, 0.8 and 1.0 m/d), each consisting of approximately 36 days. The influent water was pumped from a hyper-eutrophic water body suffering from annual HCBs.

Physicochemical analysis

Water samples were collected periodically (every 10 to 14 days) from the sampling ports in six layers of the two replicated CWs, between the hours of 9:00 a.m. and 10:00 a.m. Dissolved oxygen (DO), pH, conductivity (Cond) and water temperature (T) were measured in situ with the Thermo Orion 5 Star portable meter (Thermo-Orion Inc., Waltham, MA). Water samples were analyzed for chemical oxygen demand (COD), total nitrogen (TN), ammonium nitrogen (NH₄-N), nitrate (NO₃-N), total phosphorus (TP), soluble reactive phosphorus (SRP), Chlorophyll a (ChlA) and total suspended solids (TSS) within 24 hours in the laboratory according to standard methods (State Environmental Protection Administration of China 2002). Influent water samples were collected weekly in polyethylene bottles (capacity, 1.0 liter) and were preserved immediately with 1% Lugol’s preservative solution for cyanobacterial species identification. Unfiltered water samples from the CWs’ sampling ports in six layers were collected weekly in airtight glass bottles (capacity, 0.5 liter), without leaving headspace for odorous compound detection. Measurement of odorous compounds (geosmin and β-cyclocitrinal) and identification
of cyanobacterial species in the water samples was performed according to Zhong et al. (2011).

**Statistical analysis**

All data were presented as the mean ± SD (standard deviation), where n refers to the number of samples. The differences in water quality between the influent and effluent of the CWs were evaluated using paired t-tests. The relationship between odorous compound concentrations and the abundance of certain cyanobacterial species was explored using linear regression analysis. Linear regression analysis was also used to explore the relationship between the odorous compound pollutant mass loading rate (MLR) and the removal rate. The statistical analysis was performed using the SPSS 13.0 software package for Windows, and the statistical significance level was set to p < 0.05.

**RESULTS AND DISCUSSION**

The improvement of water quality by CWs

Characteristics of the influent and effluent of CWs at different HLRs are summarized in Table 1. DO concentrations were significantly lower in the effluent than in the influent (p < 0.05). For HLRs from 0.2 m/d to 0.8 m/d, concentrations of TN, TP and TSS in the effluent were significantly lower than those in the influent (p < 0.05). At an HLR of 1.0 m/d, significant differences were noted in DO, pH and COD values between the influent and effluent of CWs.

Furthermore, the water quality variations inside CWs at six layers (W1-W6), both along the water flow direction and under different HLRs, are shown in Figure 2 and Figure 3. For HLRs ranging from 0.2 m/d to 0.8 m/d, the effluent TN, nitrate, TP, COD, TSS and Chl.a concentrations at layer W6 (effluent) were obviously lower than those at layer W1 (influent). There were especially sharp decreases in the effluent COD, TSS and Chl.a concentrations at layer W2, compared to those at layer W1. It is noticeable that there were significant increases in NH₄⁺-N concentrations from layers W2 to W4, and the effluent NH₄⁺-N concentrations increased compared to those in the influent. Similar increases in SRP concentrations from layer W3 to layer W4 were observed; however, effluent SRP concentrations decreased to some extent. During the experiment, dense cyanobacterial blooms were observed, and the highest influent TSS and Chl.a concentrations were 87.6 mg/L and 212.2 μg/L, respectively. As a result, significant amounts of nitrogen and phosphate existed in particulate organic forms (i.e., algae biomass) in the influent. The accumulation

![Figure 1](https://iwaponline.com/wst/article-pdf/77/5/1186/249182/wst077051186.pdf)
and decomposition of algal biomass at layer W2 could be the main reason for the increase in NH₄-N and SRP concentrations from layers W2 to W4. In addition, the degradation of algal biomass will consume oxygen and reduce nitrification efficiency, which was ascribed to the competition for O₂ between the NH₄-N oxidizers and the heterotrophic microorganisms (Mbuligwe 2004; Zeng et al. 2013). From layers W4 to W6, the SRP concentrations decreased gradually, which could be due to the precipitation with Ca(II), Mn(II), and Fe(II) under anaerobic conditions (Cheng et al. 2017). As the HLR increased to 1.0 m/d, obvious increases in the SRP, TP, COD, TSS, and Chl.a concentrations were observed at layer W2. Only the SRP, COD and TSS concentrations decreased at layer W6, compared with those at layer W1. Hua et al. (2010) reported that the particulate suspended solids were trapped in the near-surface layer, which might contribute to the clogging that occurs in the upper layer (10–20 cm) of the wetland substrate (Hua et al. 2010). The obvious increase of TSS concentrations at the W2 layer indicates the risk of clogging when treating the hyper-eutrophic water bodies with HCBs at a high HLR. Nevertheless, the HCBs only appeared from May to October in this area, and were most serious from May to June in this study. During this period, growth rates of the bacteria and macrophytes in CWs were best under the optimal temperature. Bacteria flourishing in CWs might contribute to the decomposition of the algal cells trapped at layer W2. The macrophyte roots present in the wetland bed will increase the hydraulic conductivity of CWs, which could open paths for preferential flow in the substrate (Hans 1997). As a result, the temporary accumulation of cyanobacterial cells might not cause the clogging problems in this type of CW at a suitable HLR.

### The removal of cyanobacterial cells and associated odorous compounds by CWs

The algal-derived odorous compounds are receiving widespread attention because they can compromise the quality of drinking water and fishery products. Of major concern are secondary algal metabolites (e.g. geosmin, 2-MIB, and β-cyclocitral), which can impart tastes and odors, resulting in numerous consumer complaints (Ho et al. 2007; Young et al. 1999). It was reported that the incidence of 2-MIB problems has been much less frequent than that of geosmin (Oh et al. 2007). In this study, 2-MIB was seldom detected in the influent (data not shown). Thus, only the removal of geosmin and β-cyclocitral by CWs was analyzed.

The removal of geosmin and β-cyclocitral was observed at the second layer (W2) of CWs with HLRs ranging from 0.2 m/d to 0.8 m/d (Figure 4). Because the trapped particles, including cyanobacterial cells, were mainly accumulated at layer W2, the removal of intact cyanobacterial cells could be the reason for the good odorous compound removal efficiencies of the CWs. Similarly, conventional chemical-physical measures (e.g., coagulation-flocculation,
clarification and filtration) can be very effective for the removal of algal cells, and are capable of removing >98% of cyanobacterial cells (Westrick et al. 2010). However, the above treatment processes have been shown to lyse cyanobacterial cells, resulting in the release of intracellular odorous compounds, and the released odorous compounds
(extracellular metabolites) have been shown to be somewhat recalcitrant to the conventional measures (Ho et al. 2007).

In this study, the rebound rise in odorous compound concentrations was not obvious from layers W3 to W6 under HLRs from 0.2 m/d to 0.8 m/d. It indicated that CWs can not only remove the intracellular odorous compounds by intact cyanobacterial cell filtration, but also can possibly degrade the extracellular odorous compounds released from the trapped algal cells at suitable HLRs. Several studies indicated that odorous compounds (e.g., geosmin) are susceptible to biological degradation by a variety of microorganisms (Hoefel et al. 2003; Luo et al. 2009). It has been demonstrated that either sand (McDowall et al. 2009) or granular activated carbon filters (Elhadi et al. 2004) are suitable solid support matrices for the attachment of biofilm-associated bacteria involved in the biodegradation of geosmin. Similarly, the biofilm attached to the substrate of the CWs might also contribute to the biodegradation of the extracellular geosmin and β-cyclocitrinal.

On May 17 and May 9, the peak concentrations of the influent geosmin and β-cyclocitrinal were observed to be 197.8 ng/L and 5579.0 ng/L, respectively. Accordingly, the removal efficiencies of the geosmin and β-cyclocitrinal by CWs were 70.3% and 82.5%, respectively. Previous research reported that the odorous threshold concentrations (OTCs) needed for geosmin and β-cyclocitrinal to produce off-flavor in drinking water are in the ranges of 4–10 and 500–1,000 ng/L, respectively (Watson et al. 2000; Young et al. 1999). Except for the instances with the appearance of peak concentrations in the influent, the effluent concentrations of the two odorous compounds were below the OTCs at most times.

The relationships between the applied MLR and the corresponding mass removal rate (MRR) for geosmin and β-cyclocitrinal in CWs are presented in Figure 5. Good linear correlations between MLR and MRR were observed for geosmin and β-cyclocitrinal. The MLRs and MRRs for geosmin and β-cyclocitrinal were higher at HLRs of 0.4 m/d and 0.8 m/d than at the other two HLRs, during which time the influent odorous compound concentrations were relatively high. At an HLR of 1.0 m/d, the MLRs for geosmin and β-cyclocitrinal decreased because of low odorous compound concentrations in the influent. However, the MRRs for geosmin and β-cyclocitrinal were negative for several times at this HLR. This result indicated that this type of CW has great potential for odorous compounds removal only when the HLR is below 0.8 m/d.
Regression analysis was used to find the possible relationship between the odorous compound concentrations and certain cyanobacteria species abundances in the influent (Figure 6). Positive correlation between β-cyclocitril concentrations and Microcystis species abundances ($r^2 = 0.8511$) was noted, and geosmin concentrations and Oscillatoria species abundances ($r^2 = 0.7583$) were also positively correlated. Zhong et al. (2011) indicated that the abundance of Oscillatoria and Microcystis species were correlated with the occurrence of odorous compounds using canonical correspondence analysis. Thus, the variation of odorous compound concentrations in the effluent could be an indicator of cyanobacterial cell removal efficiencies by CWs. Although the cyanobacteria abundances were not measured in the effluent of the CWs, the low effluent concentrations of odorous compounds, TSS and Chl.a showed that the cyanobacterial cells had been filtrated by the CWs efficiently.

The results indicated that CWs can be used to treat hyper-eutrophic water bodies with dense cyanobacterial blooms at a suitable HLR, which can trap the cyanobacterial cells and remove the odorous compounds released.

**Influence of HLRs on the purification performance of CWs**

As the HLRs increased from 0.2 m/d to 1.0 m/d, the treatment efficiencies of the CWs decreased accordingly (Figure 7). There was no removal of TP or Chl.a by CWs at the highest HLR (1.0 m/d). Chang et al. (2007) evaluated the effect of HLRs (0.2 m/d to 1.2 m/d) on the purification performance of a similar CW treating moderately eutrophic water in a creek, and demonstrated positive relationships between HLRs and MRRs of NH$_4^+$-N, TP, COD and BOD$_5$. In this study, only the MRRs of COD increased as the HLRs increased from 0.2 m/d to 1.0 m/d (Table 2). The significant differences in MLRs between this study (hyper-eutrophic water bodies) and that conducted by Chang et al. (2007) (moderately eutrophic water bodies) could explain why the two studies reached opposite conclusions for similar HLR ranges. The average geosmin removal efficiency was highest (65.8%) for an HLR of 0.4 m/d, while the highest β-cyclocitril removal efficiency (99.7%) was observed at an HLR of 0.2 m/d, after which the odorous compound removal efficiencies gradually
decreased as the HLR increased further. There was no removal of either geosmin or β-cyclocitrinal as the HLR increased to 1.0 m/d. This result indicated that the highest HLR (1.0 m/d) could lead to the collapse of the CW system, which would result in poor purification performance when treating hyper-eutrophic water bodies.

CONCLUSIONS

For HLRs lower than 0.8 m/d, TN, TP, COD, TSS, Chl.a, geosmin and β-cyclocitrinal were efficiently treated by CWs. At the second layer of CWs, the efficient removal of intact cyanobacterial cells could explain the good purification performance. As the HLR increased to 1.0 m/d, the obvious increase of the TSS concentration at the second layer indicates the risk of clogging when treating hyper-eutrophic water bodies. The treatment efficiencies of CWs decreased as the HLRs increased. There was no removal of TP, Chl.a, geosmin, or β-cyclocitrinal for an HLR of 1.0 m/d. If vertical flow CWs are operated under suitable HLRs, they can provide a feasible way to improve water quality and control HCBs and related odorous problems in hyper-eutrophic water bodies.

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