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RESEARCH ARTICLE

Release characteristics of nitrogen and phosphorus from sediments formed under different supplemental water sources in Xi'an moat, China

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Abstract

The endogenous release of nutrients from sediments contributes to the eutrophication of landscape water to a certain degree, which depends on the characteristics of sediments. The study explored the characteristics of nitrogen (N) and phosphorus (P) released from two different sediments, which were deposited from reclaimed water (SRW) or surface water (SSW) respectively in Xi'an moat. This paper aimed to compare the effects of nutrient release from SRW and SSW on the water quality. Results showed that the maximum increase rates reached 1.21 mg TN/(L·day) and 0.11 mg TP/(L·day), respectively, in the overlying water of SRW, which were 1.6 and 2.8 times those of SSW. The released amounts of SRW were 0.192 mg TN/g and 0.038 mg TP/g, which were 4.1 and 12.7 times those of SSW. Meanwhile, the densities of benthic algae in SRW and SSW were 5.605×10^9 and 2.846×10^8 cells/L, respectively. Moreover, the species number and individual sizes of benthic algae in SRW were also larger than those in SSW, which played an important role in the nitrogen circulation. Unexpectedly, oxidation reduction potential (ORP) level of SRW was lower than that of SSW, although SRW has a higher dissolved oxygen level. Therefore, the N and P concentrations in the overlying water of SRW were considerably higher than those of SSW, which was mainly attributed to the higher nutrient contents and lower ORP in SRW.

Keywords Landscape water · Reclaimed water · Sediment · Endogenous pollution · Nutrient release · Benthic algae

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Introduction

The total number and area of urban landscape water bodies in China have increased with the development of the economy and the improvement of people's living standards in recent years (Qu et al. 2013). Urban landscape water can improve the microclimate, conserve water resources, beautify the city, and protect the environment. Urban landscape water can also play a key role in promoting the ecological environment, the cultural atmosphere, and people's entertainment. Thus, urban landscape water becomes an important component of city planning. However, more and more water transfer to urban landscape section undoubtedly aggravated the water supply crisis in the cities, especially which are located in arid and semi-arid regions (Sun et al. 2014, Warner et al. 2018).

Reusing the reclaimed water becomes a good option to solve this issue, because it has the advantages of sufficient water quantity, stable water quality, and relatively low production costs. Actually, reclaimed water has been widely used in landscape water supply in many countries and regions, such as China's Beijing Olympic Lake (Li et al. 2015) and the United

States' Elsinore Lake (Marks 2006). However, the urban landscape water supplied by reclaimed water is easily prone to eutrophication problems, such as water blooms and red tides, due to the shallow depth, slow flow velocity, and high concentrations of nitrogen (N) and phosphorus (P) pollutants (Ao et al. 2018a, Wang et al. 2018). Eutrophication problems often increase turbidity and decrease water transparency and dissolved oxygen (DO), which eventually lead to cloudy water with unpleasant odor. The deterioration of water quality not only reduces the landscape effect but also brings harm to the aquatic ecosystems and human health (Ao et al. 2018b, Hamilton et al. 2017, Pepper and Gerba 2018).

The strengthening of environmental pollution control and management (Li et al. 2018, Qin et al. 2010) has greatly reduced the exogenous pollutants in landscape water, but the endogenous pollution is difficult to eliminate (Gilabert-Alarcón et al. 2018). Endogenous pollution generally refers to the pollution loads from the sediments accumulated on the bottom of the surface water body. Nitrogen, phosphorus, and other pollutants can be released from the sediments and re-enter into the overlying water under certain physical, chemical, and environmental conditions. Thus, the sediments, being the largest source of endogenous pollution, can release various pollutants (especially N and P) to the overlying water through physical, chemical, and biological exchanges under certain conditions. The released N and P from the sediments may cause eutrophication problems in landscape water (Huang et al. 2016, Ni et al. 2016b). The degree of eutrophication of landscape water is usually connected with the characteristics of the sediment, which are determined by the water source. Generally speaking, different water sources imply different characteristics of water quality. For example, the concentrations of N and P in reclaimed water are often higher than those in surface water, while the content of clay or silt in surface water is higher than that in reclaimed water. There is no doubt that excessive N and P in water will lead to eutrophication, which can increase the contents of N, P, and pollutants in sediments through the deposition of algae and other pollutants.

The sediments of reclaimed water (SRW) and sediments of surface water (SSW) refer to the sediments formed under the different water supply, reclaimed water, and surface water respectively. Recently, considerable researches have been focused on the release characteristics of N and P from SSW. Koziorowska et al. reported the N flux from sediment ranged from 0.12 to 1.46 g/m^2/year, the P flux varied between 0.01 and 0.11 g/m^2/year (Koziorowska et al. 2018). Bolałek and Graca studied the N exchange at the water–sediment interface in Puck Bay found that ammonia nitrogen flux from sediment varied from 5 to 1434 umol NH_4^+-N m^2/day (Bolałek and Graca 1996). Ding et al. investigated the contribution of internal P to N limitation showed that the soluble reactive phosphorus (SRP) diffusion flux at the sediment–water interface ranging from −0.01 to 6.76 mg/m^2/day (Ding et al. 2018). Kim et al. found that the P release rate at the Jamsil submerged dam was approximately 105 mg/m^2/week under anaerobic conditions (Kim et al. 2003). However, research on nutrients released from SRW is scarce due to the short period during which reclaimed water has been applied and low amount of sediment accumulated in the corresponding landscape water. On the other hand, the research on benthic algae has a long history. Numerous scholars have studied the effect of benthic algae on the photosynthesis and nutrient cycle in sediments and water. Dalsgaard reported that benthic algae played an important role in the utilization and morphological transformation of nitrogen in the overlying water (Dalsgaard 2003). Zhang and Mei evaluated the effects of benthic algae on the release of SRP from sediments of shallow lakes and found that benthic algae could reduce the release of SRP from sediments directly and indirectly (Zhang and Mei 2015). However, there are few reports discussing the differences and functions of benthic algae between SRW and SSW. In particular, the effects of benthic algae on water quality and how much it affects the release of N and P from SRW are not yet understood comprehensively.

The novelty of this study is to investigate the release characteristics of N and P from sediments of reclaimed and surface water sources and explore the effects of benthic algae on water quality and the nutrient release from SRW.

The main objectives of this study are as follows:

- To investigate how much the overlying water quality are affected by the nutrients released from different sediments.
- To understand the characteristics of endogenous pollution released from various sediments due to the different water supply sources.
- To provide an important reference for managing the water quality supplied with reclaimed water and controlling the eutrophication caused by nutrients release from sediments.

Materials and methods

Source and characteristics of the sediments

The sediments in this experiment were taken from the South Gate and Southeastern corner zones of Xi'an moat. The South Gate zone has been supplied with reclaimed water, and the Southeastern corner zone has been supplied with surface water. The replenishment of both zones has been continuously implemented for more than 4 years, and the water quality of South Gate and Southeastern corner in Xi'an moat is shown in Table 1. The sediments of each zone were collected at 10 cm below the sediment–water interface using a stainless steel grab sampler. The sediments were homogenized for the subsequent

Table 1 Water quality of South Gate and Southeastern corner in Xi'an moat

Sample sites	TP (mg/L)	PO_4^{3-}–P (mg/L)	TN (mg/L)	NH_4^+-N (mg/L)	NO_3^-–N (mg/L)
South Gate	0.33 ± 0.13	0.18 ± 0.12	6.48 ± 1.43	0.13 ± 0.09	4.51 ± 0.96
Southeastern corner	0.13 ± 0.03	0.04 ± 0.02	4.40 ± 1.84	0.23 ± 0.06	2.05 ± 0.52

experiment on nutrient release after the debris, such as small stones and leaves, were filtered out by a coarse screen. At the same time, some of the sediments were centrifuged, freeze-dried, and ground to pass through a 60-mesh screen to determine total N (TN), total P (TP), and total organic carbon (TOC). The physicochemical properties of SRW and SSW are shown in Table 2.

Experimental design

Two parallel cylindrical glass reactors with a diameter of 24 cm, a height of 38 cm, and an effective volume of 10 L were used as the reactors. Three liters of homogenized SRW was added to one reactor, and the other reactor contained 3 L of homogenized SSW. Then, the same amount of tap water was injected into the two reactors slowly to the 10 L by siphoning. Both reactors were placed in an indoor environment (20.6 °C ± 1.8 °C), and the tops were sealed with rubber plugs. The light intensity in the reactors was kept at 30 Lux, and the experiment lasted for 56 days.

Sampling and analysis

Three water samples were simultaneously collected by siphoning at a depth of 5 cm below the water surface in the reactors as representative water samples for the overlying water of the simulated landscape water. The samples were collected at 9:00 a.m. every 7 days.

Temperature (T) and DO were measured at a depth of 5 cm below the water surface in the reactors by using a HACH Portable Multifunctional Water Quality Tester (HQ-30d, HACH, USA). The pH value was measured using a pH meter (pHS-3C, INESA, China). TN and ammonia nitrogen (NH_4^+-N), nitrate nitrogen (NO_3^--N), nitrous nitrogen (NO_2^--N), TP, and soluble reactive phosphate (SRP) were all analyzed using a spectrophotometer (UV-752N, XINMAO, China).

Sediment samples were collected using a peristaltic pump (BTOO-100M, Longer Pump, China) at the beginning and the end of the experiment in both reactors. The pH value was measured using a pH meter (pHS-3C, INESA, China) in the sediment, and oxidation reduction potentials (ORPs) at the sediment–water interface and the sediment at depths of 2, 4, and 8 cm were measured with an HACH portable multipurpose water quality analyzer (HQ-30d, HACH, USA). The concentration of the suspended solids (SS) in the sediment was measured using the gravimetric method.

The dried sediment was used to measure TOC, TN, TP, and inorganic P (IP). The TOC and TN of the sediment were analyzed by potassium dichromate oxidation spectrophotometry and the nesslerization spectrophotometric method, respectively. TP and IP were analyzed by the sodium hydroxide melting–molybdenum antimony colorimetric method and hydrochloric acid extraction–molybdenum antimony colorimetric determination. Moreover, the density and morphology of benthic algae were determined using a microscope (Nikon 50i, Nikon, Japan) at the end of the test.

Statistical analysis

The experimental data were statistically analyzed, calculated, and plotted using Origin 9.0 software. The average value, standard deviation, and variance of the data were analyzed using SPSS software (PASW Statistics 20.0). The homogeneity of variance among samples was tested using the *f* test method. The average value among samples was tested using the *t* test method at a confidence level of $p < 0.05$.

Results

Effect of different sediments on the water quality of the overlying waters

Time course of DO concentration in the overlying water

Figure 1 shows the time course of the DO concentration in the overlying water during the nutrient release process of different sediments. The DO concentrations in the overlying water of SRW and SSW decreased sharply from 9.73 to 2.20 and 1.77 mg/L, respectively, and then experienced slow declines

Table 2 Physicochemical indicators of the sediments formed under reclaimed water (SRW) and under surface water (SSW)

Sediment	Color	pH	TP (mg/g)	IP (mg/g)	TN (mg/g)	TOC (mg/g)
SRW	Black brown	7.52 ± 0.75	1.41 ± 0.20	1.09 ± 0.15	7.74 ± 0.59	52.60 ± 4.51
SSW	Yellow brown	7.11 ± 0.43	1.28 ± 0.14	0.84 ± 0.03	6.85 ± 1.02	33.74 ± 6.25

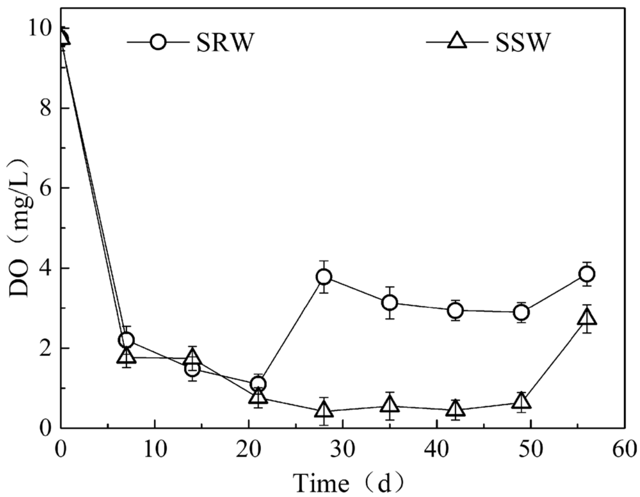


Fig. 1 Time course of DO concentration in the overlying water of SRW and SSW

to 1.10 and 0.76 mg/L at day 21 respectively. Subsequently, the DO concentration in the overlying water of SRW increased and then remained at 3.32 ± 0.46 mg/L from day 21 to the end of the experiment. However, the DO concentration in the overlying water of SSW decreased slightly and then remained at 0.52 ± 0.10 mg/L. The DO concentrations in the overlying water of SRW and SSW increased between day 50 and day 56, but the DO concentration in the overlying water of SSW was lower than that of SRW. Overall, the average DO concentration in the overlying water of SRW was 2.67 ± 1.00 mg/L, which was 2.36 times that of SSW. So, SRW maintained a higher DO concentration in the overlying water than SSW after 30th days although both kept a similar DO concentration before the initial 21st days.

Time course of N concentration in the overlying water

Figure 2 shows the time course of N concentration in the overlying water of the different sediments during the release process. In the first 7 days, the TN concentrations in the overlying water of SRW and SSW increased rapidly from 0 to 8.50 and 5.24 mg/L, respectively. Then, TN concentrations rose gradually to the peak values of 9.42 and 7.20 mg/L on day 21. Thereafter, the TN concentrations in the overlying water of SRW and SSW decreased gradually, reaching the lowest concentrations of 3.87 and 3.36 mg/L on day 56. During the entire experiment, the TN concentration in the overlying water of SRW was always higher than that of SSW. The maximum increase rates of TN concentration could reach 1.21 mg TN/(L·day) in the overlying waters of SRW, which was 1.61 times that of SSW (0.75 mg TN/(L·day). The average TN

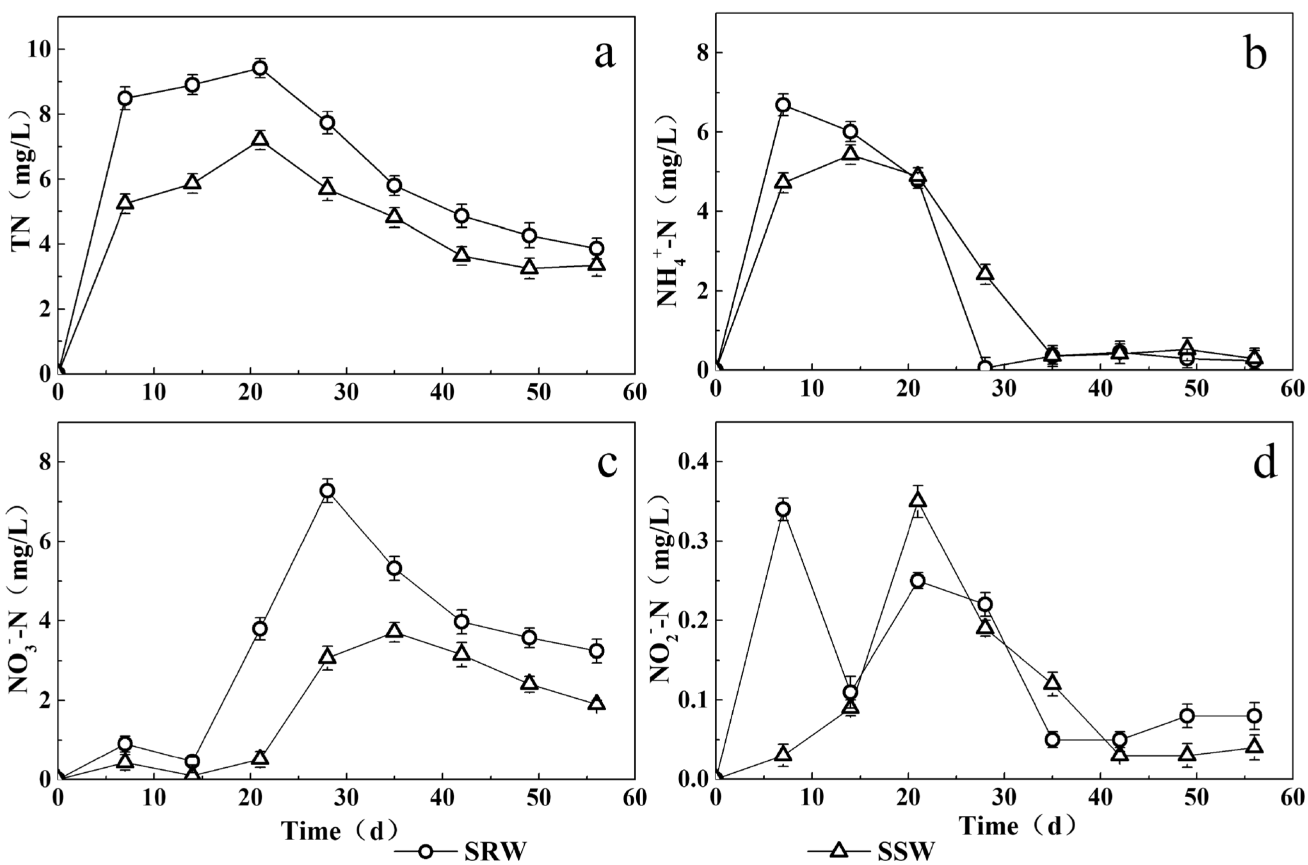


Fig. 2 Time course of N concentration in the overlying water of SRW and SSW

concentrations were 6.67 ± 2.23 mg/L and 4.88 ± 1.40 mg/L in the overlying waters of SRW and SSW, respectively. Overall, the amount of TN released from SRW was 0.192 mg TN/g which was 4.09 times that from SSW (0.047 mg TN/g).

As shown in Fig. 2b, the NH_4^+-N concentration in the overlying water of SRW reached the maximum value of 6.69 mg/L on day 7, whereas that in the overlying water of SSW reached a maximum of 5.43 mg/L on day 14 after a small plateau region. Then, the NH_4^+-N concentration in the overlying water of SRW and SSW decreased gradually and tended to be stable after 35 and 28 days, respectively. The NH_4^+-N concentrations in the overlying water of SRW and SSW were 0.23 and 0.29 mg/L, respectively, at the end of the experiment. The maximum increase rates of NH_4^+-N concentration in the overlying water of SRW and SSW could reach 0.96 and 0.67 mg NH_4^+-N/(L·day) respectively. The NH_4^+-N amounts released from SRW and SSW were 0.136 and 0.035 mg NH_4^+-N/g, respectively. The average NH_4^+-N concentrations in the overlying water of SRW and SSW were 2.36 and 2.38 mg/L, respectively.

As shown in Fig. 2c, the trends of NO_3^--N concentrations in the overlying water of SRW and SSW were stable at the early stage. The NO_3^--N concentration in the overlying water of SRW increased sharply from day 14 to the maximum value of 7.28 mg/L on day 28. Then, the NO_3^--N concentration decreased gradually to a minimum value of 3.24 mg/L on day 56. Similarly, the NO_3^--N concentration in the overlying water of SSW increased from day 14 to the maximum value of 3.71 mg/L on day 35, and then gradually dropped to a minimum value of 1.90 mg/L on day 56. Throughout the experiment, the NO_3^--N concentration in the overlying water of SRW was higher than that of SSW. The average NO_3^--N concentration was 3.57 ± 2.20 mg/L, which was 1.87 times that of SSW. The maximum increase rates of NO_3^--N concentration in the overlying water of SRW and SSW were 0.49 and 0.21 mg NO_3^--N/(L·day), respectively, which might be caused by the nitrification of NH_4^+-N. As shown in Fig. 2d, the NO_2^--N concentrations in the overlying water of SRW and SSW were low (< 0.35 mg/L) during the entire experiment.

The TN and NH_4^+-N concentrations in the overlying water of different sediments showed a similar trend. However, the peak and average concentrations in the overlying water as well as the maximum release rate of SRW were significantly higher than those of SSW.

Time course of P concentration in the overlying water

As shown in Fig. 3a, the TP concentrations in the overlying water increased in the first 21 days and then declined sharply before reaching stable concentrations for both SRW and SSW. The rate of TP concentration increased to the maximum value in the overlying water of SRW (1.87 mg/L) is higher than that of SSW (0.41 mg/L). The TP concentration in both overlying waters declined significantly from day 21 to day 35 and then stabilized. During the entire experiment, the TP concentration in the overlying water of SRW was significantly higher than that of SSW. The average TP concentrations were 1.19 ± 0.35 and 0.22 ± 0.11 mg/L (t test, $p = 0$) in the overlying waters of SRW and SSW, respectively. Furthermore, the maximum increase rates of TP concentration in the overlying water of SRW and SSW were 0.11 and 0.04 mg TP/(L·day). In addition, the TP amounts released from SRW and SSW were 0.038 and 0.003 mg TP/g, respectively.

The trends of SRP concentrations in the overlying water of SRW and SSW were similar to those of TP concentrations (Fig. 3b). The maximum SRP concentrations of 1.42 and 0.25 mg/L in the overlying water of SRW and SSW were

Fig. 3 Time course of P concentration in the overlying water of SRW and SSW

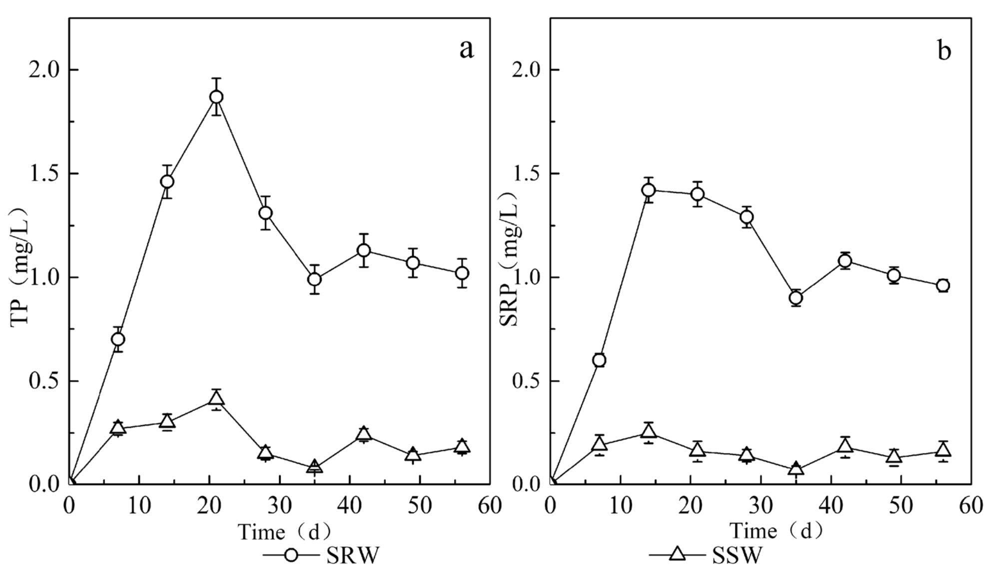


Table 3 ORP levels at different depths of SRW and SSW (on day 56) (unit: mV)

Depth (cm)	SRW	SSW
Water–sediment interface (0)	− 62.0	− 28.0
2	− 218.3	− 220.0
4	− 286.0	− 231.3
8	− 323.6	− 251.1

determined on day 14. Thereafter, the SRP concentration in the overlying water of SRW decrease rapidly and then stabilized after 35 days. However, the SRP concentration in the overlying water of SSW underwent a small decline and then fluctuated. During the entire experiment, the SRP concentration in the overlying water of SRW was significantly higher than that of SSW, and the average concentrations were 1.08 ± 0.28 and 0.16 ± 0.05 mg/L (*t* test, $p = 0$), respectively. The maximum increase rates of SRP in the overlying water of SRW and SSW were 0.10 and 0.03 mg SRP/(L·day), respectively. The SRP amount released from SRW was 0.029 mg SRP/g, which was 14.50 times that of SSW. So, the peak and average concentrations of P in the overlying water as well as the maximum release rate of SRW were significantly higher than those of SSW.

ORP levels and benthic algae characteristics of different sediments

ORP levels of different sediments

Table 3 shows the ORP levels at different depths of SRW and SSW on day 56. The sediment–water interface was adopted as the vertical origin of the coordinate. All the ORP values at different depths of dissimilar sediments were negative. The greater the depth of the sediment was, the larger the absolute value of ORP was. Therefore, the deeper sediment was more anaerobic than the shallower sediment regardless of the type of sediment. The ORP value in SRW was nearly always lower than that in SSW at the same depth, and the decrease gradient of ORP in SRW was 30.1 mV/cm, which was larger than that of SSW (23.4 mV/cm). This phenomenon indicated that SRW was more reducible than SSW, and SRW could supply a more reducible environment for pollutant release than SSW.

Characteristics of the benthic algae in different sediments

Figure 4 shows the micrographs of benthic algae in different sediments at the end of the experiment. As shown in Fig. 4, the individual sizes and number of species types in SRW were larger than those in SSW. The densities of benthic algae in SRW and SSW were 5.605×10^9 and 2.846×10^8 cells/L, respectively; the density of benthic algae in SRW was 19.7 times that in SSW.

Figure 5 shows the species distributions of benthic algae in SRW and SSW. More species of benthic algae were present in SRW than in SSW. Seven kinds of species were detected in SRW and *Eunotia* sp. was the dominant species (36%). Only three kinds of benthic algae were detected in SSW, and the dominant species was *Gomphonema* sp. (63%). Figure 6 shows the size distribution of benthic algae in SRW and SSW. The absolute amount of benthic algae with a larger size in SRW was markedly greater than that in SSW, where the size of benthic algae ranged from 2 to 50 μm. For example, the absolute amount of benthic algae with a size between 20 and 50 μm in SSW was smaller than that in SRW, although the benthic algae of 20 and 50 μm were dominant in SSW (with a percentage of 63%). This was similar to the case with the benthic algae larger than 50 μm. So, whether considering algal density or algal size, the benthic algae in SRW was superior to that in SSW.

Discussion

Characteristics of sediments in landscape water supplied with different water sources

Different concentrations of N and P appeared in the landscape water supplied with reclaimed and surface waters. Chen et al.

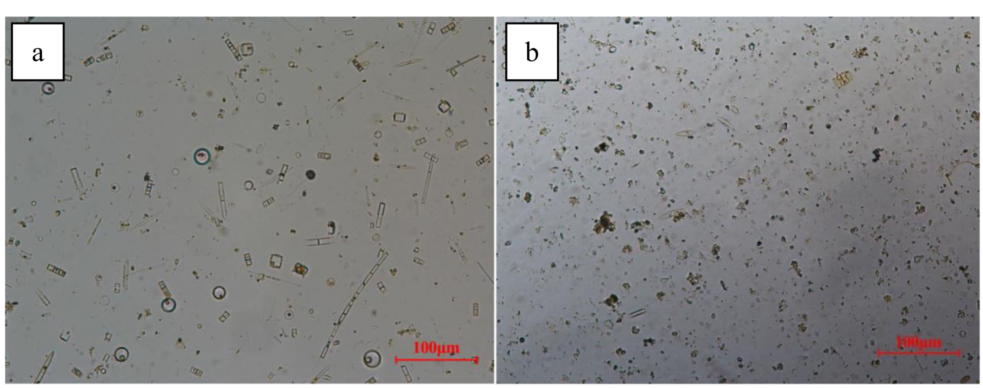
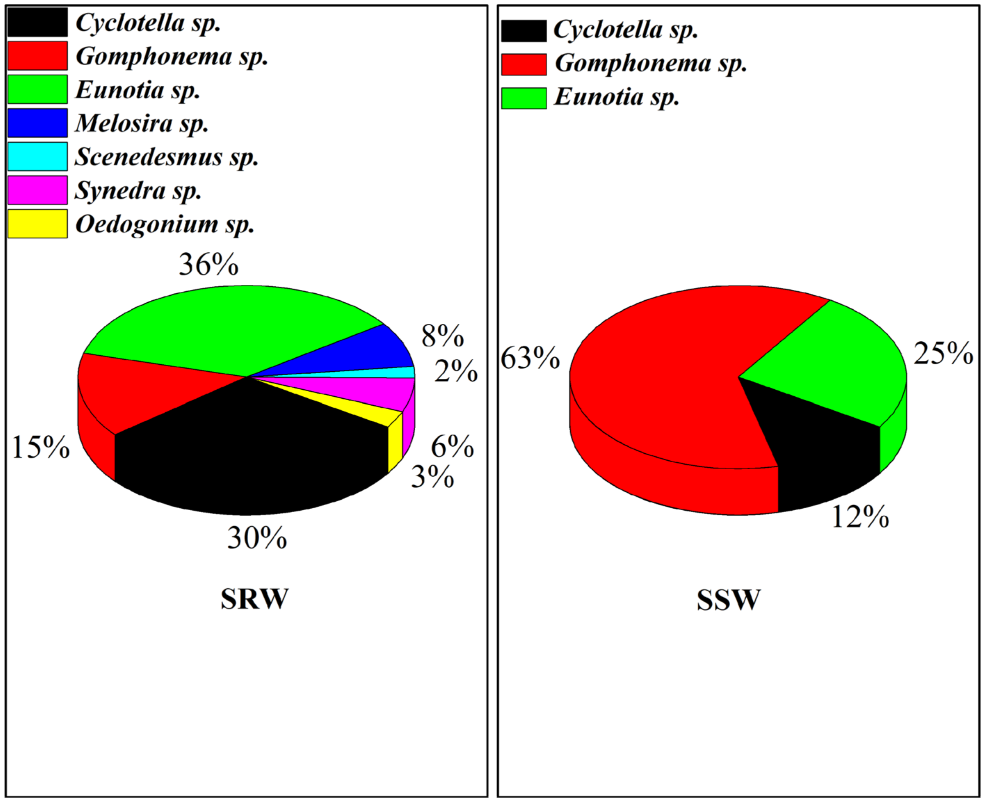


Fig. 4 Micrograph of benthic algae in SRW (**a**) and SSW (**b**) (× 200)

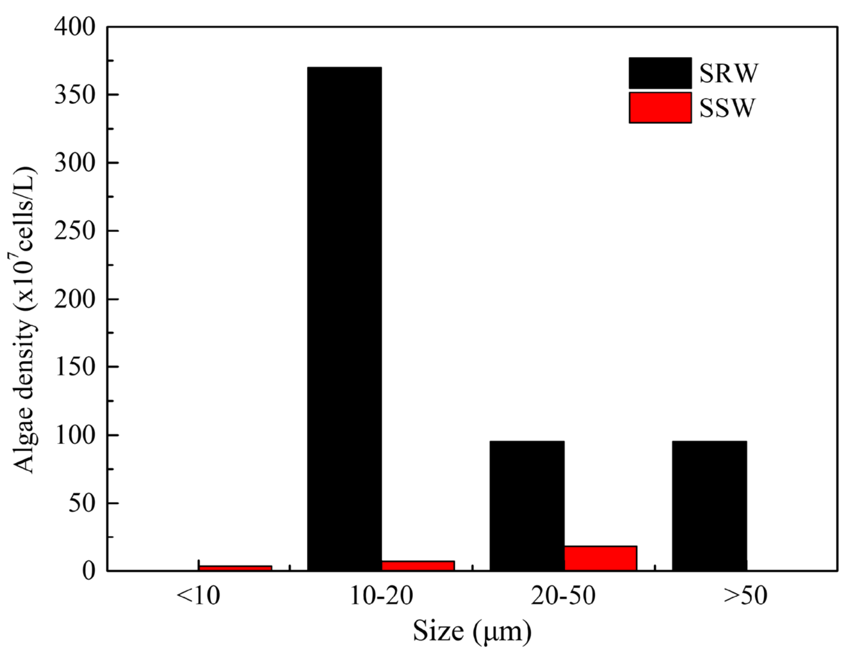
Fig. 5 Distribution of benthic algae species in SRW and SSW



reported that the TN concentrations can reach 9.78 mg/L and the TP concentrations can reach 0.58 mg/L in the overlying water when urban landscape water is supplied with reclaimed water (Chen et al. 2016). Ni et al. found that the TN and TP concentrations in Taihu Lake supplied with surface water are 4.22 and 0.54 mg/L, respectively (Ni et al. 2016a). In the current study, the reclaimed and surface waters have been used to supply Xi'an moat together for more than 4 years. Various sediments occurred in the Xi'an moat when supplied by reclaimed and surface water, which released different amounts of N and P. It eventually led to the different water qualities in the overlying waters. The results showed that the N and P concentrations in the overlying water of SRW were significantly higher than those of SSW.

The variations in the amounts of nutrients released from SRW and SSW were mainly due to their different nutrient

Fig. 6 Distribution of benthic algae size in SRW and SSW



contents and ORP levels of the sediments. As shown in Table 2, SRW was richer in N, P, and organic matters, which would cause the larger released amounts of TN and TP in per unit time under the same experimental conditions, comparing with SSW. On the other hand, the higher contents of N, P, and organic matters in SRW would create a favorable environment for the growth and reproduction of microorganisms. Under anoxic conditions, the microorganisms used organic matters and nutrients to grow and metabolize, thereby exacerbating anoxic conditions in the deep water bodies and sediments. Thus, the ORP of sediment was lower at deeper depths. Hanna et al. studied the nutrient release of continental sea sediment rich in organic matter (Hanna et al. 2013). Their result showed that the involvement of microorganisms in the aerobic and anaerobic degradation of organic matter exacerbates the anoxic status of sediments, which directly or indirectly promotes the release of nutrients from the sediment. The current study found that the ORP of SRW was lower than that of SSW even at the same sediment depth, and the ORP level decreased with the increase in the depth of both sediments. The strong reducibility of SRW promoted the release of N and P from the sediment to the overlying water, thereby resulting in the amounts of nutrients released from SRW being much greater than those from SSW.

Even though both sediments receive weak illumination, the benthic algae could still perform photosynthesis by using limited resources and release O_2 in the sediments. However, this study found that the DO concentration was nearly undetectable at a depth of 6 mm in the sediment, while the DO concentration in the overlying water of SRW was maintained at 2.67 ± 1.00 mg/L on average. This phenomenon illustrated that the released O_2 from benthic algae only improved the DO level in the overlying water but affect little on the DO level in the sediment through photosynthesis. This result was similar to that of (Higashino et al. 2008), who reported that the sediment has great resistance to oxygen molecule transport. Gu et al. examined the distribution and exchange of oxygen at the sediment–water interface and found that oxygen molecules have difficulty penetrating into the sediment (Gu et al. 2016). In the present study, the density, individual sizes, and number of species of benthic algae in SRW were larger than those in SSW. As a result, the DO concentration in the overlying water of SRW was higher than that of SSW. However, the high DO level in the overlying water of SRW only slightly affected the ORP of its sediment due to the limited penetration of oxygen molecules in the sediment. Therefore, benthic algae played an important role in improving the DO level of the overlying water, but had little effect on the ORP of sediment.

Briefly, the ORP of SRW was lower than that of SSW due to the different characteristics of sediments, but the DO concentration in the overlying water of SRW was higher than that of SSW because of the different algae at the sediment–water interface. This phenomenon is very unique in the nutrient release process of SRW determined by biological and chemical composition. Notably, SRW would affect the nutrients release of the sediment directly.

Variation in N concentration in the overlying water of different sediments

The oxidizing environment of the sediment was inevitably weakened by the drastic decline in DO concentration in the overlying water at the initial stage of this experiment. Given that the ORP level of SRW was lower than that of SSW, the anaerobic hydrolysis of organic matter in SRW was stronger than that of SSW. In addition, the amount of NH_4^+-N produced by the hydrolysis per mass of SRW was higher than that of SSW due to the high content of TN in SRW. Zhu et al. studied on the released N from sediment in eutrophic Lake and reported that the maximum NH_4^+-N flux from sediment was 48.76 mg m^{-2} day^{-1} (Zhu et al. 2019). Gu et al. investigated the contaminated lake sediments and showed that the NH_4^+-N flux from sediment could reach 67.4 mg/m^2/day (Gu et al. 2018). In this study, we found that the maximum NH_4^+-N flux from SRW was 147.9 mg/m^2/day, which was 1.4 times that of SSW. Yang et al. reported that low DO concentrations and ORP levels promoted the release of N-containing compounds in the sediment, with NH_4^+-N as the dominant pollutant (Yang et al. 2015). The lower ORP level and higher TN contents in SRW in the current study caused a higher concentration of NH_4^+-N in the overlying water, which was similar to the report of Yang et al. (2015).

The decreased NH_4^+-N and increased NO_3^--N concentrations in the overlying water of both sediments indicated that nitrification occurred in both systems in the middle stage of the experiment. However, the higher DO concentration in the overlying water of SRW was due to the number of species, and individual sizes of the benthic algae in SRW were larger than those in SSW. Davies and Hecky explored the photosynthesis and respiration of benthic algae in Lake Erie and showed that the total photosynthesis oxygen yield of benthic algae can reach 300 mg O_2/(m^2·h) (Davies and Hecky 2005). Therefore, the most important contribution for the DO concentration in the overlying water could be ascribed to the photosynthesis of benthic algae. So, it is reasonable that the DO concentration in the overlying water of SRW was always much higher than that of SSW. The concentration and increase rate of NO_3^--N in the overlying water of SRW were higher than those of SSW, although the nitrifiers in the overlying water of both sediments could convert NH_4^+-N to NO_3^--N under aerobic conditions. The high concentration and the increase rate of NO_3^--N in the overlying water of SRW might be attributed to the high concentrations of NH_4^+-N and DO.

The concentrations of TN and NO_3^--N in the overlying waters of both sediments decreased at the end of the experiment, which might be due to the existence of denitrifiers. The DO concentrations in the overlying waters of both sediments were kept relatively low during the entire experiment. The denitrification rates of the overlying water were measured (data not shown) in this experiment.

The low ORP would promote NH_4^+-N release from the sediment. In addition, the high TN contents in SRW increased the NH_4^+-N concentration in the overlying water. The large density, number of species, and individual sizes of the benthic algae in SRW would produce oxygen through photosynthesis, which improved the DO level and the nitrification in the overlying water but only slightly affected the ORP level in the sediment. At the same time, the concentrations of TN and NO_3^--N dropped because denitrification could occur at a certain depth of the sediment (Liu et al. 2018, Uusheimo et al. 2018). SRW was not only high in content of TN but also in the release rate of NH_4^+-N to the overlying water.

Variation in P concentration in the overlying water of different sediments

In this study, the concentration trends of TP and SRP in the overlying waters of both sediments were similar; they exhibited an increase at the beginning and then a decrease until a plateau was reached. Ruban and Demare proved that approximately 80% of the TP released from the sediment–water interface is SRP (most of which exists in the Fe-bound state) (Ruban and Demare 1998). The present study demonstrated that SRP also constituted the vast majority of TP released from the sediment. The SRP concentrations in the overlying waters of SRW and SSW accounted for more than 90% and 70% of the TP on average, respectively. Ding et al. investigated the P exchange across the sediment–water interface and indicated that the apparent diffusion flux of P was ranging between -0.2 and 0.65 mg m^{-2} day^{-1} (Ding et al. 2015). Steinman and Ogdahl evaluated the total maximum daily load for P in Bear Lake and reported that the maximum apparent TP release rate ranged from 0.41–6.69 mg/m^2·day under anoxic conditions (Steinman and Ogdahl 2015). The results of present study showed that SRW had a higher release rate and amount of TP and SRP than SSW, which led to a higher concentration in the overlying water of SRW. The maximum TP flux from SRW was 16.1 mg/m^2/day, which was 2.7 times that of SSW and obviously higher than that reported by other researchers.

This phenomenon was ascribed to the higher TP content and lower ORP in SRW than those in SSW. Howarth et al. investigated temperate estuaries and coastal marine ecosystems and showed that the chemical reactions of $Fe^{3+} \rightarrow Fe^{2+}$ easily occurred when the sediment had no oxygen (Howarth et al. 2011). They also found that the P combined with Fe^{3+} is easily released from the sediments. In the present study, SRP release might have a relationship with P combined with Fe^{3+}. Compared with SSW, SRW was more likely to release P to the overlying water due to its lower ORP. SRW also had a larger maximum release rate and amount of TP and SRP due to its higher content of TP than SSW.

Conclusion

The characteristics of nitrogen and phosphorus released from the sediments, which are deposited from different water supplies, were quite different in this experiment. The result showed that the release rate and amount of N and P from SRW were much higher than those from SSW. Meanwhile, NH_4^+-N and SRP were the main forms of N and P released from the sediments. Moreover, the maximum amounts of P released from SRW were 12.7 and 14.5 times those of SSW, while the maximum amounts of N released from SRW were 4.1 and 3.9 times those of SSW. In particular, the density, number of species, and individual sizes of benthic algae in SRW were much greater than those in SSW, which led to a higher concentrations of DO and NO_3^--N in the overlying water of SRW than those of SSW. However, the high concentrations of DO in the overlying water could not elevate the ORP in the SRW, which was rich in N, P, and organic matters owe to reclaimed water as a water source. So, water resource determined the characteristics of the sediment, which inevitably affected the release rate and amount of nutrients with the assist of benthic algae and led to the final results of the overlying water quality. Consequently, the endogenous pollution of SRW should be paid more attention to controlling and managing the water quality of landscape water. In addition, the detailed function and mechanism of benthic algae in the process of nutrient release from the sediments should be explored in the future.

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