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Effects of the Filter-Feeding Benthic Bivalve *Corbicula fluminea* on Plankton Community and Water Quality in Aquatic Ecosystems: A Mesocosm Study

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Abstract: The influence of filter-feeding bivalves on plankton communities, nutrients, and water quality in a given aquatic ecosystem is so profound that they can be considered ecosystem engineers. In a 70-day mesocosm experiment, we tested the hypothesis that *Corbicula fluminea* would change plankton community structure by reducing small zooplankton and large phytoplankton and improve water quality by reducing nutrients. We monitored levels of nitrogen and phosphorus, organic suspended solids (OSS), and light at the sediment surface. Within the plankton, phytoplankton biomass (as Chl *a*, >0.45 μm), the biomass of microphytoplankton (>20 μm), nanophytoplankton (2–20 μm), picophytoplankton (0.2–2 μm), and zooplankton were determined. Compared with the controls, *C. fluminea* reduced the abundance of rotifers and the biomass of phytoplankton, and picophytoplankton, thereby modifying the plankton community structure. We did not observe reductions in TN and TP concentration, but OSS concentrations were reduced, and light intensity at the sediment surface was increased as a result of the improved water transparency. Our research shows that colonization by *C. fluminea* may modify plankton community structure and improve water quality of eutrophic shallow lakes, shedding further light on the ecological roles of filter-feeding bivalves in aquatic ecosystems.

Keywords: *Corbicula fluminea*; phytoplankton; chlorophyll *a*; rotifers; cladocerans; water quality; aquatic ecosystem

1. Introduction

Filter-feeding bivalves often dominate the benthic fauna of aquatic ecosystems [1,2] due to their high fecundity and potential for rapid growth [3]. As ecosystem engineers, filter-feeding bivalves exert profound effects via their feeding [4], metabolic activity, and movements, which can cause bioturbation effects [5,6]. In turn, they are a food resource for fish, birds, and other large fauna [7], and their shells provide refuges for other benthic animals [8].

The filtering function of bivalve species, such as *Corbicula fluminea*, may potentially change the ecological conditions of aquatic ecosystems [1,3]. Filter-feeding bivalves feed on plankton, suspended particles, and organic debris [9,10]. By filtering matter from the water

column [10–12], they can contribute to a marked decrease in particle concentrations [6,13] and an increase in water clarity [14]. In Saginaw Bay, Lake Huron, colonization by zebra mussels increased water clarity by 60% [15], and bivalve activity elsewhere has been shown to propel ecosystems from turbid to clear water states [16].

However, filter-feeding bivalves are selective feeders, and a given species will only filter and remove particulate matter of an appropriate size [17,18]. For example, *Cerastoderma edule* selectively removes particles in the range 2–12 μm [19], *Venerupis corrugate* captures a range from 5 to 13 μm [20], while *Hyriopsis cumingii* is more likely to filter phytoplankton between 10 and 40 μm [21].

In addition to phytoplankton, filter-feeding bivalves can also filter small zooplankton [22]. Horsted et al. [23] found that *Mytilus edulis* reduced the number of tintinnid ciliates and rotifers. Pace et al. [24] observed a decline in rotifers and nauplii in the Hudson River after the arrival of invasive zebra mussels. Larger zooplankton, such as large cladocerans and copepods, are less affected due to their size and more effective escape strategies [25,26]. This selectivity in bivalve feeding results in their high potential for modifying plankton community structure [14].

Filter-feeding bivalves can also affect concentrations of nutrients in the water column [27] in a variety of ways. In some cases, the net effect may be a reduction in total water column nutrients, exerted by removal of plankton and particulate organic matter [6,28], or by transfer of solids from the water column to benthic habitat [29,30] in the form of feces and pseudo-feces [17,31]. Conversely, other studies have shown that bioturbation of sediment by filter-feeding bivalves causes particle resuspension and promotes nutrient release from sediment into overlying water [5,27,32]. Zhang et al. [33] showed that filter-feeding bivalves can promote the release of soluble reactive phosphorus (SRP) from the sediments to the water column. In addition, bivalves themselves excrete nutrients that can fertilize the aquatic system [34]. The nutrients excreted and released from sediments in these ways can increase the dissolved nutrient concentrations in the water column [35] and therefore affect the phytoplankton community.

Although plenty of research has been conducted on the effects of filter-feeding bivalves on aquatic ecosystems [36], including on eutrophication, nutrients availability [37], phytoplankton biomass [38], water clarity, and water quality [39], more information about *Corbicula fluminea* influence on plankton communities, especially on different sizes of plankton, including microphytoplankton, nanophytoplankton, picophytoplankton, and zooplankton, and corresponding water quality is still needed to elucidate its impact on freshwater ecosystems. Besides being widely distributed in Asia, Africa, and Australia, *C. fluminea* is one of the most invasive species worldwide, especially in American and European freshwater ecosystems [3], where it is still expanding its distribution range northward [40]. This has made the organism one of the dominant benthic animals in many lotic and lentic habitats, both pristine and polluted in different regions of the world [41]. The reasons for its success can be found in its high filtration rate on a per biomass basis with rapid growth, earlier sexual maturity, short life span, and high fecundity [3]. In addition, the species is easily maintained in the laboratory for several months, making it a good model bivalve to evaluate the effects of filter-feeding on plankton community structure, biomass, and water quality.

We hypothesized that the filter-feeding bivalves may modify the plankton community structure by reducing both small-bodied zooplankton and phytoplankton, and by reducing nutrient levels in the water. All these effects may lead to an overall improvement in water quality. Mesocosms were established to test the hypothesis and to evaluate the effects of *C. fluminea* on nutrient levels, light intensity, plankton community structure, and biomass. The results of this study may further clarify the role of filter-feeding bivalves in structuring the plankton community and contributing to the improvement in water quality in aquatic ecosystems.

2. Materials and Methods

2.1. Experimental Mesocosm Set-Up

The mesocosm experiment was carried out in eight circular plastic tanks (upper diameter = 57 cm, bottom diameter = 46 cm, height = 82 cm) containing sediment and 155 L

rainwater. Sediments were taken from a shallow eutrophic lake in Guangzhou City. The sediment was air-dried and sieved through a 0.5 mm stainless sieve to remove coarse debris and create uniformity [42]. The homogenized sediments were added in a 10-cm thick layer to each mesocosm [42]. The sediments were considered as suitable reservoirs of resting stages to inoculate the experimental mesocosms. All mesocosms were then filled with 155 L rainwater collected in advance (total nitrogen (TN) = 1.37 mg·L^{-1}, total phosphorus (TP) = 0.03 mg·L^{-1}). These were filled with rainwater to be sure that the zooplankton and microbial assemblages were similar in the different mesocosms. The mesocosms were exposed to natural sunlight and allowed to equilibrate for about two weeks, after which nutrient concentrations of the water were 0.59 ± 0.2 mg·L^{-1} for TN and 0.06 ± 0.01 mg·L^{-1} for TP.

Specimens of the filter-feeding bivalve *Corbicula fluminea* were bought from a market in Guangzhou City. Before being introduced into the mesocosms, the animals were kept in 200-L tanks with rainwater for about two weeks to adapt to the experimental environment. Five individuals (6.9 ± 0.4 g in average wet weight) were added to each of four treatment mesocosms, and the remaining four mesocosms without bivalves served as controls. No bivalve mortality was recorded in the course of the experiment, and the density of animals (167 ind·m^{-2} or 207 g·m^{-2}) used is comparable to that found in natural systems, including lake Taihu, a large shallow lake in China [43,44].

Nitrogen (N) and phosphorus (P) were added weekly to each mesocosm in the form of KNO$_3$ and NaH$_2$PO$_4$ at 1.5 mgN·L^{-1}·week^{-1} and 0.1 mgP·L^{-1}·week^{-1}, in order to simulate external loading [43]. Nutrients were added immediately after sampling [42], and further rainwater was added as required to maintain a constant water level during the experiment.

The experiment ran from 28 August to 6 November 2019, and the mesocosms were exposed to natural sunlight for all the experiment duration.

2.2. Sampling and Analysis

Water samples (1 L) were collected fortnightly from 20 to 30 cm below the water surface of each mesocosm using clean polyethylene bottles, and they were used for measurement of N, P, organic suspended solids (OSS), phytoplankton biomass as chlorophyll *a* (Chl *a*, >0.45 μm), and biomass (Chl *a*) of microphytoplankton (>20 μm), nanophytoplankton (2–20 μm), picophytoplankton (0.2–2 μm). Tow hundred milliliters of water was filtered using 20 μm filters (nylon net) for microphytoplankton, 2 μm filters (nuclepore) for nanophytoplankton, and 0.2 μm filters (nylon) for picophytoplankton biomass, respectively. Another 200 mL of water was filtered by cellulose acetate membrane (0.45 μm) for phytoplankton biomass. Chl *a* of phytoplankton was extracted using a 90% acetone/water solution for 24 h, and its concentration was measured spectrophotometrically [45]. The biomass (Chl *a*) of microphytoplankton, nanophytoplankton, and picophytoplankton was determined as for phytoplankton after size fractionation. Total nitrogen (TN) and total phosphorus (TP) levels were determined after persulfate digestion [46]. NO$_3$$^-$-N and NH$_4$$^+$-N were assessed according to Jin and Tu [47]. Samples for SRP were filtered through 0.45 μm cellulose acetate membrane filters and determined by UV spectrophotometry [46]. OSS was filtered by a Whatman GF/C fiber membrane, dried at 108 °C for 2 h, weighed, and ashed at 550 °C for 2 h, then weighed [48].

Light intensity at the sediment surface was measured between 9 a.m. and 12 p.m. with an underwater irradiance meter (ZDS-10W) before water samples were collected [43]. Meanwhile, water temperature, pH, and DO were measured using a YSI Model 556 multi-parameter probe [43].

At the end of the experiment (on day 70), zooplankton in each mesocosm was collected by filtering 5 L of water through a 64 μm mesh, after which the sample was fixed and stored in 5% formaldehyde. Species were identified according to Wang [49] and Chiang and Du [50], and their densities were counted under a microscope and calculated. The weight biomass of *C. fluminea* in each mesocosm also was recorded.

2.3. Statistical Analyses

Repeated measures analyses of variance (RM-ANOVAs) were used to determine any significant differences in TN, NO_3^--N, NH_4^+-N, TP, SRP, OSS, Chl *a*, light intensity, and plankton biomass at different size ranges between treatments, with time as the repeated factor. Independent sample *t*-tests were used to further examine significant treatment effects on each sampling occasion. Differences were considered significant when $p < 0.05$. All results are presented as mean $\pm$ SD. All statistical analyses were conducted using SPSS 26.0 software.

3. Results

3.1. Nutrients

The concentrations of TN, NH_4^+-N, TP, and SRP did not differ between the treatments (RM-ANOVAs, treatment effect, $p > 0.05$) (Figures 1 and 2). The concentrations of TN, NH_4^+-N, TP, SRP varied significantly over time (RM-ANOVAs, time effect, $p < 0.05$), while the concentration of NO_3^--N did not (RM-ANOVAs, time effect, $p > 0.05$). However, the concentration of NO_3^--N was higher in *C. fluminea* treatments than in the controls (RM-ANOVAs, treatment effect, $p < 0.05$) (Figure 1), specifically on days 56 and 70 (*t*-test, $p < 0.05$). The concentration of NH4$^+$-N did not differ between the treatments (RM-ANOVAs, treatment effect, $p > 0.05$) (Figure 1), but on day 70, it was lower in the *C. fluminea* treatments than in the controls (*t*-test, $p < 0.05$).
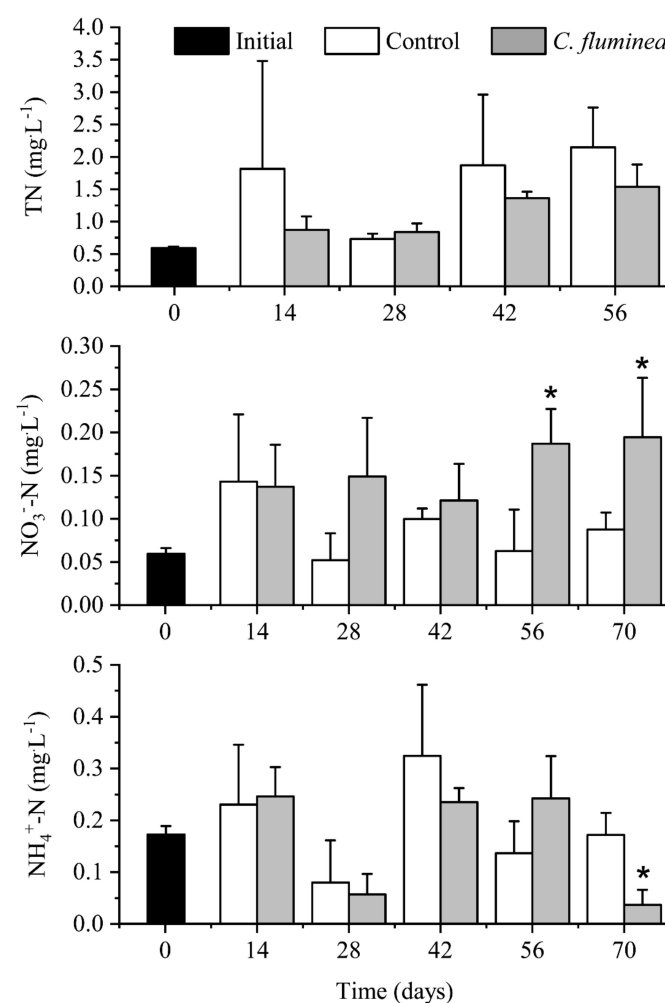

Figure 1. Total nitrogen (TN, mean $\pm$ SD), nitrate nitrogen (NO_3^--N, mean $\pm$ SD), and ammonium nitrogen (NH_4^+-N, mean $\pm$ SD) in different treatments over time. Asterisk indicates significant differences between the treatments and controls ($p < 0.05$).
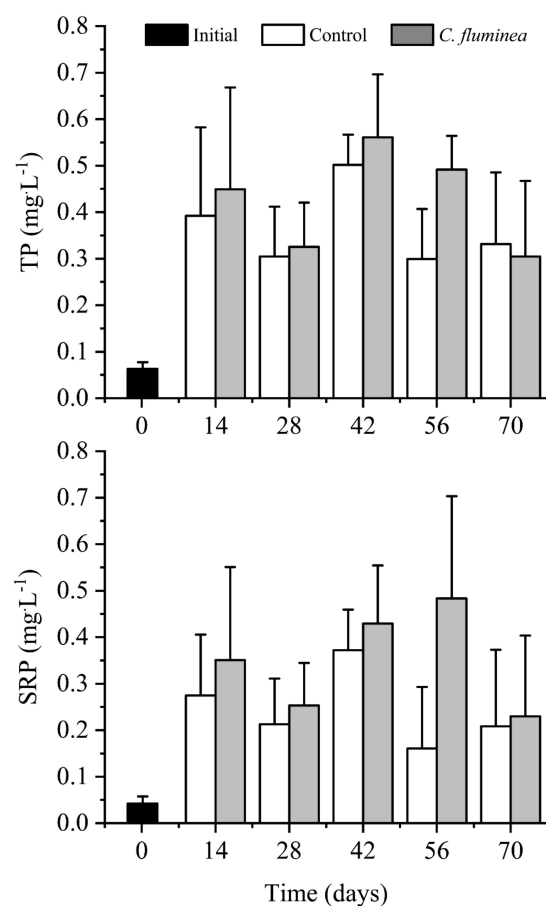
Figure 2. Total phosphorus (TP, mean ± SD) and soluble reactive phosphorus (SRP, mean ± SD) in different treatments over time. There is no significant difference between treatments at $p < 0.05$.

3.2. OSS and Light Intensity

Observed concentrations of OSS were lower in *C. fluminea* treatments than in the controls (RM-ANOVAs, treatment effect, $p < 0.05$), especially on day 70 (*t*-test, $p < 0.05$) (Figure 3). The concentration of OSS did not vary significantly with time (RM-ANOVAs, time effect, $p > 0.05$).
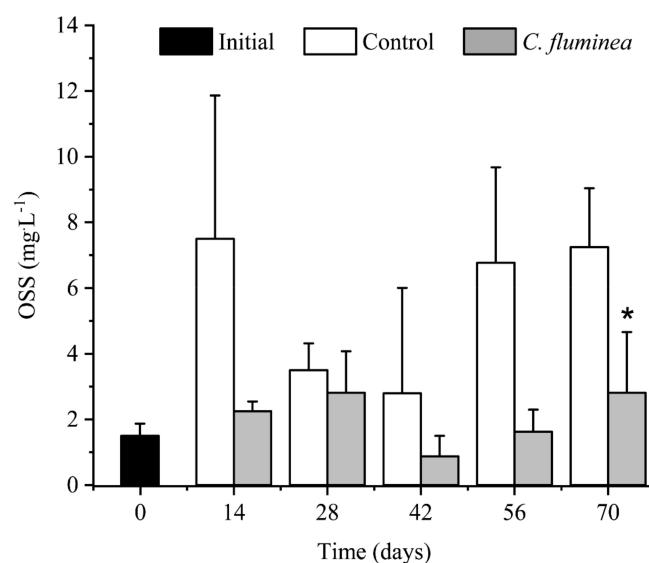
Figure 3. Organic suspended solids (OSS, mean ± SD) in different treatments over time. Asterisk indicates significant differences between the treatments and controls ($p < 0.05$).

The intensity of light at the sediment surface was higher in *C. fluminea* treatments than in the controls (RM-ANOVAs, treatment effect, $p < 0.05$), especially on days 14, 28, and 56 (*t*-test, $p < 0.05$) (Figure 4), suggesting that the presence of *C. fluminea* increased light penetration. Light intensities at the sediment surface varied significantly over time (RM-ANOVAs, time effect, $p < 0.05$).
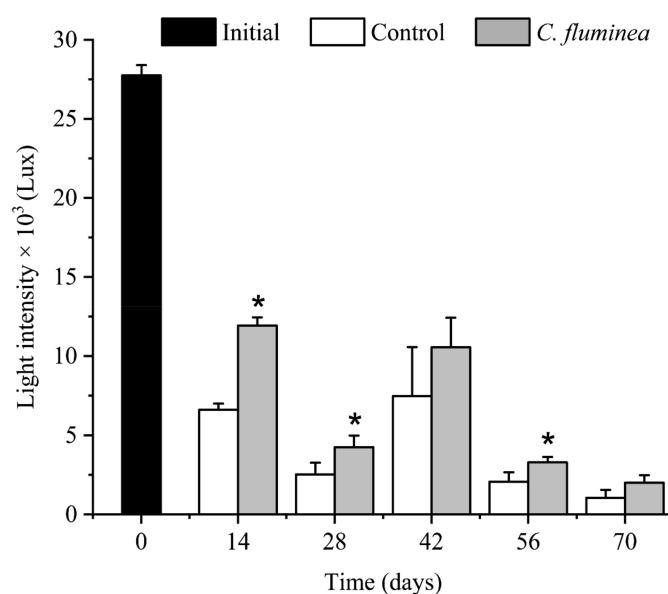
Figure 4. Light intensity (mean $\pm$ SD) in different treatments over time. Asterisk indicates significant differences between the treatments and controls ($p < 0.05$).
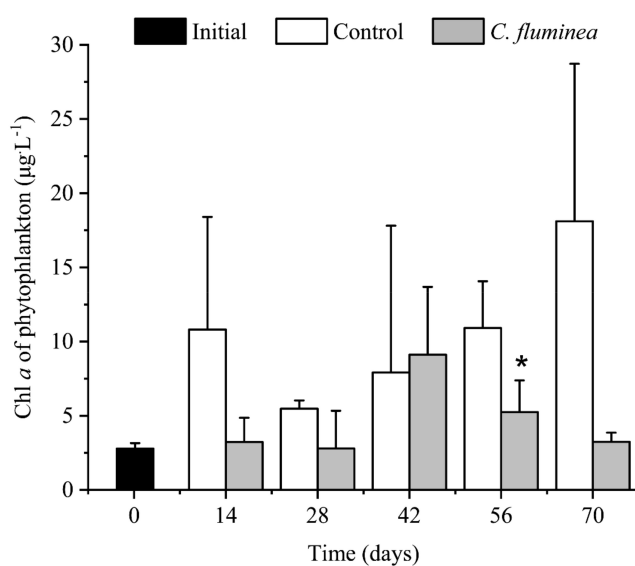
3.3. Phytoplankton Biomass and Community Structure

The biomass of phytoplankton (Chl *a*) was lower in the *C. fluminea* treatments than in the controls (RM-ANOVAs, treatment effect, $p < 0.05$), especially on day 56 (*t*-test, $p < 0.05$) (Figure 5). Chl *a* values did not differ with time (RM-ANOVAs, time effect, $p > 0.05$).

Figure 5. Chl *a* of phytoplankton (mean $\pm$ SD) in different treatments over time. Asterisk indicates significant differences between the treatments and controls ($p < 0.05$).

The picophytoplankton biomass (Chl *a*) was lower in *C. fluminea* treatments than in the controls (RM-ANOVAs, treatment effect, $p < 0.05$), especially on day 42 (*t*-test, $p < 0.05$), while the biomasses (Chl *a*) of microphytoplankton and nanophytoplankton

were similar between controls and *C. fluminea* treatments (RM-ANOVAs, treatment effect, $p > 0.05$) (Figure 6). The biomass of nanophytoplankton was lower in *C. fluminea* treatments than in the controls on day 56 (*t*-test, $p < 0.05$). The biomass of microphytoplankton, nanophytoplankton, and picophytoplankton remained relatively constant over time (RM-AVOVAs, time effect, $p > 0.05$), especially in the *C. fluminea* treatments.
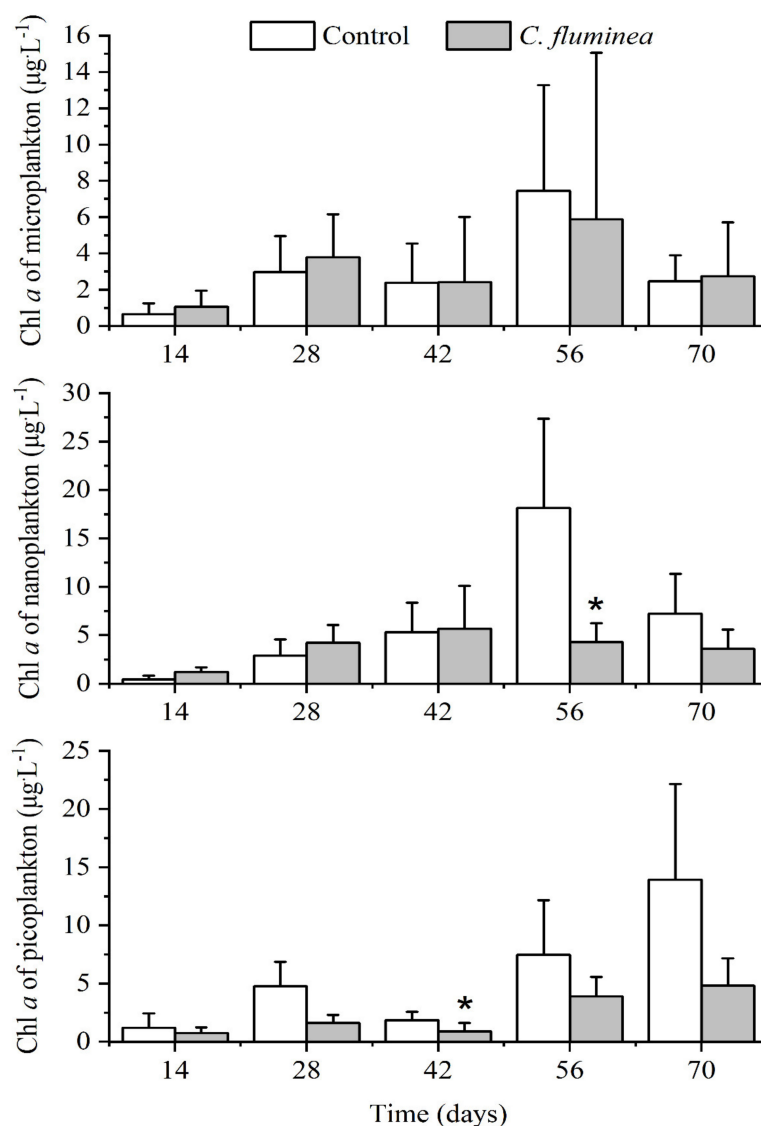
Figure 6. Chl *a* of microplankton (mean ± SD), nanoplankton (mean ± SD), and picoplankton (mean ± SD) in different treatments over time. Asterisk indicates significant differences between the treatments and controls ($p < 0.05$).

3.4. Zooplankton Community

At the end of the experiment, the abundance of rotifers in the *C. fluminea* treatments (144.36 ± 139.5 ind·L^{-1}) was lower than in the controls (1566.2 ± 763.85 ind·L^{-1}) (*t*-test, $p < 0.05$), but the abundance of cladocerans did not differ between *C. fluminea* treatments (52.62 ± 41.67 ind·L^{-1}) and controls (51.84 ± 8.79 ind·L^{-1}) (*t*-test, $p > 0.05$) (Figure 7).
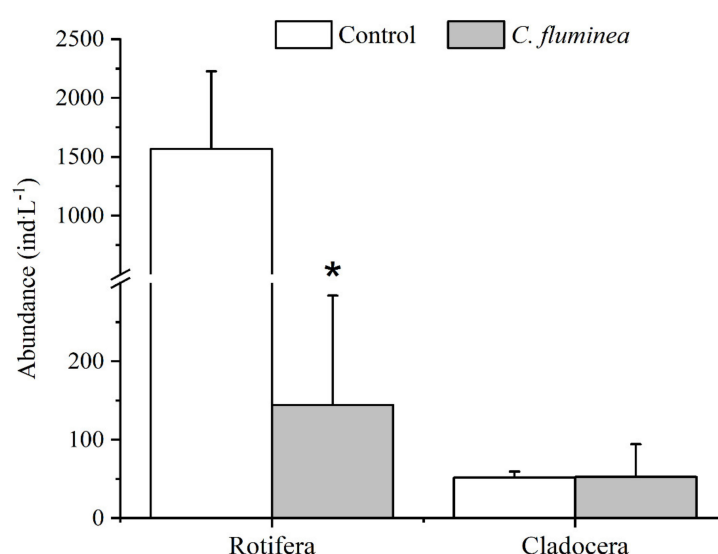
Figure 7. Abundance of zooplankton in different treatments (mean $\pm$ SD) at the end of the experiment. Asterisk indicates significant differences between the treatments and controls ($p < 0.05$).

4. Discussion

Corbicula fluminea is considered to be a very efficient ecosystem engineer, altering the structure and function of aquatic ecosystems where it is abundant [51,52]. Our results show that *C. fluminea* exerted a significant effect on the plankton community of experimental mesocosms, decreasing the abundance of rotifers and phytoplankton biomass and picophytoplankton biomass, reducing OSS, and increasing the light intensity at the sediment surface. Contrary to our initial hypothesis, *C. fluminea* did not reduce TN and TP levels in the water, but led to an increase in NO_3^--N concentrations.

C. *fluminea* can feed on phytoplankton and suspended particulate matter from the water column [53–55]. Their ability to reduce suspended particulate matter and phytoplankton biomass in the water column has long been recorded [56–60]. In this study, *C. fluminea* reduced the overall concentration of Chl *a* in the water column and the Chl *a* contributed mostly by picophytoplankton, while values for microphytoplankton and nanophytoplankton were not significantly altered.

The selective capture of suspended particles of different sizes is an important aspect of bivalve ecology [31,61]. *C. fluminea* has been shown to favor relatively fine particulate matter [62]. In this study, the biomass of picophytoplankton (mainly photosynthetic organisms with size between 0.2 and 2 μm) was significantly reduced in *C. fluminea* treatments, indicating that this animal may feed on picophytoplankton [63].

Bivalves feed not only on phytoplankton [64], but also consume microzooplankton [65]. In our experiment, the rotifers developed most probably from the sediments and could rapidly increase in number due to their r strategy [3]. This might be the key reason for the high abundance of rotifers in the controls. The abundance of rotifers was significantly reduced by *C. fluminea*, but the abundance of cladocerans was not significantly affected in the treatments compared to controls, indicating a selective negative effect of the bivalve on small zooplankton. One possible mechanism for reducing rotifer abundance is the direct effects of the bivalves by consuming rotifers. Some studies showed that bivalves could effectively prey on microzooplankton, including rotifer species [66–68] supporting this possibility. However, we could not rule out the indirect effects of bivalves on rotifers via the decrease in small phytoplankton. In this study, picophytoplankton biomass decreased in the enclosures with the bivalves. Small phytoplankton, including picophytoplankton, are known to important food for some rotifers. Accordingly, a decrease in the picophytoplankton abundance by the animal may have reduced rotifer abundance due to their food decreased, even though differences in particle-size-dependent feeding efficiencies

are closely related to rotifer species [69]. To clarify the relative importance of the bivalves' direct and indirect effects on rotifers, further studies are necessary.

Bivalves can also affect nutrient dynamics in the water column, not least by consuming organic material and moving undigested inorganic particles to the sediment surface in the form of feces or pseudo-feces [69,70], thereby reducing the total nitrogen and phosphorus in overlying water [71]. However, these effects may be offset by the disturbance and resuspension of sediment caused by burrowing, which may promote the release of nitrogen and phosphorus to the overlying water [33,72]. In addition, bivalves may release nutrients to overlying water via excretion [34], thereby contributing to increasing nutrients concentrations. However, to what degree the *C. fluminea* affected nutrient dynamics by consuming organic material, releasing undigested inorganic particles, and promoting the sediment release and resuspension, and excreting is unknown in this study. It was found that the concentration of NO_3^--N in water increased in the *C. fluminea* treatments, but there was no significant effect on the concentrations of TN, NH_4^+-N, TP, and SRP. Holland et al. [73] showed an increase in biodeposition and excretion by colonizing zebra mussels resulting in elevated NO_3^--N. The effects of bivalves on NO_3^--N is likely due to nitrification of excreted NH_4^+-N [74]. Yang et al. [75] and Li [76] found that although the presence of *Anodonta woodiana* reduces the biomass of phytoplankton, they exert no significant effect on nutrient concentrations in the overlying water.

Enhancement of water transparency is one of the major impacts of filter-feeding bivalves on the aquatic ecosystem [16,29]. Our results were consistent with previous findings in this regard, demonstrating that *C. fluminea* reduced OSS concentration in the overlying water and increased light intensity at the sediment surface.

Our study confirms the prevailing view of bivalves as important benthic animals in freshwater ecosystems with a role in maintaining clear water conditions in lakes and suggests that *C. fluminea* might be a tool in ecological restoration of eutrophic lakes within the natural area of distribution of the species. However, due to the strong invasiveness of C. *fluminea*, the introduction of this nonindigenous invasive species outside its biogeographical range can also bring some negative effects on the native biodiversity and ecosystem functioning [77], including competition for food resources; displacement of or reduction in available habitat for other species [31] and change the biogeochemical cycles. So, the introduction in ecosystems as a restoration tool of eutrophic lakes should be carefully considered.

In short, the presence of *C. fluminea* can reduce the abundance of rotifers and the biomass of phytoplankton, and picophytoplankton, causing substantial modifications in the structure of plankton communities. Although *C. fluminea* did not reduce TN and TP concentrations in this experiment, the species efficiently promoted a reduction in OSS concentrations, improved water transparency, and increased light intensity at the sediment surface.

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