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Demand for clean water has rapidly increased over the decade. This decade has been declared as the decade of water, which reflects the imperative of sustainable water and aquatic system management. Limnotek: *Perairan darat tropis di Indonesia*, one of the leading journals in Indonesia concerning the challenges over sustainable aquatic ecosystems, is answering the call by transforming to a global Journal entitled Journal of Limnology and Water Resources. This transformation is aimed to expand the communication among global scientific communities and stakeholders by highlighting further interdisciplinary issues related to both limnology and water resources sciences.

Although there has been a year's delay in our publication due to managerial issues and the transformation of the umbrella institution (BRIN), we are sincerely grateful that we can continue our new and polished publication this year. This current issue comprises critical topics such as the use of biotas to overcome the challenges of the reduction of polluted waters, the threat and potential control of alien invasive species, and research to aid the mitigation of hydroclimatic disasters.

Our continuing hard work can bring the journal to global acknowledgement as well as provide a better communication platform among the readers. Last but not least, this small step can lead to sustainable water and aquatic ecosystem management. We learn, we live, and we grow!

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Growth of the water fleas *Daphnia magna* (Straus, 1820) at different trophic levels of two small urban lakes in Indonesia

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Abstract: Nutrient enrichment in waters that has become a major environmental problem is related to excessive loading of nutrients into aquatic ecosystems. This nutrient enrichment, called eutrophication, favors phytoplankton growth, which can function as a natural daphnid feed. This study examined the growth performance of the water fleas *Daphnia magna* in water collected from small lakes (ponds) of different trophic levels. The water was taken from Situ Rawa Kalong, considered eutrophic from its dark green color, and the less eutrophic Situ Cibuntu with relatively clear water. Daphnids were grown in six aquaria filled with water from both ponds without artificial feeding with an initial density of five individuals/L. Samples of daphnids were taken every three to four days to observe their growth and reproduction, along with water samples to analyze the chlorophyll content and total suspended solids (TSS). The result showed that the eutrophic water of Situ Rawa Kalong favored phytoplankton growth, indicated by a consistently higher chlorophyll content in the water ranging from 35.3 to 140.7 µg/L compared to less eutrophic water of Situ Cibuntu with chlorophyll content ranging from 1.4 to 13.2 µg/L throughout the experiment. A much higher daphnid density of 151.7 individuals/L was achieved with more water chlorophyll content, meaning phytoplankton availability became a controlling factor for daphnid growth in the pond waters. This study reveals the functional relationships in the food chain between the water trophic level, the abundance of phytoplankton as the primary producer, and daphnids as the first-order predator. It also suggests that the open water trophic level can be managed to favor the daphnid growth, which can then be harvested for use as natural feed.

Keywords: *Daphnia magna*, eutrophic waters, nutrient, Situ Rawa Kalong, Situ Cibuntu

1. Introduction

The uncontrolled enrichment of nutrients in the aquatic environment, known as eutrophication, has become a common phenomenon not only in Indonesia, which is alleged to be a national problem that needs to be resolved immediately (Kementerian Lingkungan Hidup Republik Indonesia, 2011) but also throughout the world (Yang *et al*., 2008). These nutrients are mainly derived from anthropogenic activities, including the disposal of domestic and industrial wastes, fertilizers leaching from agricultural activities, and fishery cultivation activities (Haryani, 2013; Yang *et al*., 2008). For instance, a load of nutrient input from the upper Citarum River catchment area was around 34 tons N/day and 5.5 tons P/day (Garno, 2001). This load, together with more significant input from floating net cage fish cultivation, reaching 478 tons N/year and 68 tons P/year (Garno, 2002), leads to severe eutrophication problem in Saguling Reservoir, with N and P contents of 0.684–3.460 mg/L and 0.067–0.364 mg/L, respectively, while the chlorophyll content reached 5.364–71.126 mg/m^3 (Van der Gun, 2012; Hart *et al*., 2003). Eutrophication problems have also occurred in Lake Limboto, with the contents of N and P recorded at 0.89–1.66 mg/L and 0.12–0.64

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mg/L respectively, and the chlorophyll content of 18.43–42,18 mg/m^3 (Chrismadha & Lukman, 2008). Likewise, the waters of Lake Maninjau are considered to be under eutrophic conditions, with the TN and TP contents of 0.37–7.43 mg/L and 0.02–0.65 mg/L, and the chlorophyll content of 0.236–0.285 mg/L (Syandri *et al.*, 2014; Lukman *et al.*, 2013). A similar problem also occurred in several lakes of West Java, such as Situ Rawa Kalong, where the average contents of chlorophyll, TN, and TP were 254.23 µg/L, 11.13 mg/L, and 0.38 mg/L, respectively (Satya *et al.*, 2018).

Eutrophication leads to phytoplankton blooms (Chrismadha & Lukman, 2008; Sulastri *et al.*, 2015). This phenomenon is a logical consequence of the phytoplankton ecological role as autotrophic organisms that utilize abundant nutrients under available solar energy for their growth (Chrismadha *et al.*, 2012; Andriani *et al.*, 2017; Meirinawati & Fitriya, 2018). According to the ecological function of the food chain, phytoplankton bloom will be an abundant source of food for first-order consumers, one of which is daphnids. There have been studies that reported the ability of Cladoceran to grow by taking advantage of phytoplankton abundance in open water (Chen *et al.*, 2009; Zhang *et al.*, 2009; Pinto-Coelho *et al.*, 2003; Pandolfini *et al.*, 2000). On a laboratory scale, daphnids have also been shown to exploit phytoplankton abundance for their growth (Chrismadha & Widoretno, 2016).

The water flea *Daphnia magna* is a planktonic animal of the lower crustacean group, belonging to the class Crustacea and the order Cladocera (Bekker *et al.*, 2018). The animal is among various natural feeds commonly used in freshwater aquaculture. Several reports indicated superior nutrition values of daphnids, including high protein content with relatively complete amino acids and unsaturated fatty acids content (El-Feky & Abo-Taleb, 2020; Fahmi *et al.*, 2019; Herawati *et al.*, 2018;). Mass cultivation of daphnids usually involves organic wastes to fertilize water to stimulate their growth (Cheban *et al.*, 2018; Darmawan, 2014).

Eutrophication in lake water can be considered the same process as water enrichment by organic wastes and can, therefore, be utilized as a medium for the mass cultivation of daphnids. A high abundance of phytoplankton in eutrophic water has been reported by Vanni and Temte (1990) and Sulastri *et al.* (2015) and has been shown to have the potential to provide natural foods to support the growth of daphnids (Chrismadha & Widoretno, 2016). Until recently, however, there was still very little information on the potential utilization of eutrophic lake water for the mass cultivation of daphnids. Therefore, this study was intended to determine the potential utilization of water from eutrophic open waters as a growth medium for the water fleas *Daphnia magna*. In this case, the enriched water is designed to be converted into phytoplankton biomass, which is then available for grazing to support the daphnids' growth.

2. Materials and Methods

2.1. Location, Research Time, and Study Biota

The research was conducted at the Research Center for Limnology LIPI in November 2018. The biota studied was the water flea *Daphnia magna* Straus, 1820. The daphnid stocks were taken from the Research Center for Limnology LIPI collection, which were maintained in the water media enriched with catfish feed pellets (Hi-Pro-Vite 781^{TM}).

This study used six aquariums measuring 40 cm × 30 cm × 30 cm equipped with an aeration system and filled with 15 L of water from Situ Cibuntu dan Situ Rawa Kalong ponds. The aquariums were placed in a room with a transparent roof and a room temperature of 25–34°C.

2.2. Eutrophic Waters

The waters at two different trophic levels were taken from ponds around Bogor Regency: Situ Cibuntu and Situ Rawa Kalong, which were subsequently used as the growth media in this experiment. Situ Cibuntu represented lightly contaminated water based on the relatively clear water and the odorless water condition, while Situ Rawa Kalong represented heavily contaminated water due to its foul smell and dark green color. The water was taken from both ponds, collected in several plastic jerry cans, and then brought to the Research Center for Limnology LIPI, where it was poured into experimental aquaria. Subsequently, each aquarium was aerated overnight before the daphnid source was inoculated.

2.3. Daphnid Culture

Large adult daphnids obtained by filtering the animals from culture stock using a net with a mesh size of 2 mm were selected for sowing daphnid cultures. After inoculation with an initial density of five individuals/L, the daphnid population was reared for 19 days without artificial feeding. This research duration was set long enough for the daphnids to reproduce in two to three generations. For this experiment, daphnids filtered by a net with a mesh size of 2 mm were considered adults, while those who passed through the net were considered juveniles.

2.4. Observation

The growth of daphnids was observed in terms of their abundance five times: on day 5, day 8, day 12, day 15, and day 19. The aquaria water was stirred slowly to homogenize the daphnid distribution before 2 L water sampling. The water was filtered using a plankton net, and daphnid samples were collected into 20 mL plankton bottles while the remaining water was returned to the aquaria. Subsequently, the population of daphnids was calculated and extrapolated into individuals/L. Each sampling was conducted in three replications.

Water chlorophyll and total suspended solids (TSS) parameters were monitored to evaluate the presence of phytoplankton, considered the primary natural feed of daphnids. These two parameters were measured at the beginning of the experiment and five times during the experiment simultaneously as sampling for daphnid abundance. A 20 mL water sample was filtered using Whatman GF/C filter paper and then frozen for storage before analysis. The chlorophyll concentration was determined by the spectroscopic method using a UV-VIS spectrophotometer (Hach DR 2800) after extraction using 90% acetone (APHA, 2017).

Meanwhile, samplings for TSS measurement were also conducted following the gravimetric method (APHA, 2017) by filtering 100 mL of water samples through Whatman GF/C filter papers previously weighed. The filter papers and filtered solids were then heated to a temperature of 55°C until the weight was constant. The TSS concentration was determined by subtracting the weight of the filter paper containing the filtered solids from the weight of the filter paper and then extrapolating it into mg/L.

Water quality parameters were also monitored during the experiment to ensure a suitable water condition for daphnid growth. They included water temperature, pH, Dissolved Oxygen (DO), conductivity, Total Dissolved Solids (TDS), and turbidity, carried out twice a week using a Multiparameter Water Quality Checker Horiba U-50.

2.5. Data Analysis

The daphnid growth performance was evaluated by calculating the specific growth rate following the formula:

$$\mathrm{SGR} = \frac{\ln \mathrm{Wt} - \ln \mathrm{Wo}}{\mathrm{T}} \times 100 \text{ ....Eq (1)}$$

in which SGR = specific growth rate, ln Wo = natural log of daphnid abundance at day 0, ln Wt = natural log of daphnid abundance at day t, T = time (days). The above Equation 1 considered the sampling interval to determine the daphnids' growth responses to the phytoplankton development in the media.

The daphnids' responses to the water trophic status were also assessed regarding the adult-to-juvenile ratio, which assumed that the juveniles would dominate the daphnid population structure under appropriate growing conditions. It is generally known that under unfavorable conditions, the daphnids will turn their reproductive behavior from parthenogenetic to sexual mode, leaving the female adults carrying partially developed embryos that remain dormant until water conditions improve. If conditions are suitable for further development, the eggs will hatch, followed by the return of the offspring to the parthenogenetic pattern (Lawrence, 1981).

At the same time, the food availability related to the water trophic status to support daphnids' growth was evaluated in terms of water chlorophyll, TSS contents, and the chlorophyll to TSS ratio. TSS consisted of inorganic and organic materials. However, since there was no inorganic material in the experimental media, the TSS dynamics were thought to occur due only to organic materials formed by microbiotic development favored by organic contents in the waters. As Satya *et al.* (2018) reported, the eutrophic water of Situ Rawa Kalong contains nutrients and considerably high contents of total organic matter. Therefore, water can support phytoplankton growth and various microorganisms, from heterotrophic bacteria to tiny zooplankton such as rotifers (Wullur *et al.*, 2019; Yang *et al.*, 2008; Kritzberg *et al.*, 2006). Therefore, it is crucial to consider the presence of various microbiotics that could function as food for daphnids, represented by the parameter TSS, while the share of phytoplankton in TSS was calculated using the chlorophyll to TSS ratio.

The food availability was assessed by calculating the relative number of potential foods to the abundance of daphnids, represented by the chlorophyll to daphnids ratio and the TSS to daphnids ratio in this study.

3. Results and Discussion

3.1. Dynamics of Chlorophyll and TSS

The chlorophyll and TSS contents in the water media from both ponds are shown in Figures 1a and 1b. At the beginning of the study, the chlorophyll content in the water media of Situ Rawa Kalong and Situ Cibuntu were 108.11 and 13.17 mg/m^3, while the TSS contents were 13.33 and 6.67 mg/L, respectively. Both chlorophyll and TSS contents in the water samples from Situ Rawa Kalong were much higher than in the water from Situ Cibuntu.

The chlorophyll content decreased during the first five days of the study in both pond water media. After that, the chlorophyll content tended to increase to reach 140.67 mg/m^3 on day 12 in the Situ Rawa Kalong water, while the maximum chlorophyll content in the Situ Cibuntu water was 7.34 mg/m^{3,} which occurred on day 19. The TSS contents in the Situ Rawa Kalong water increased during the study and reached a maximum value of 60 mg/L on day 19, while in the Situ Cibuntu water, the TSS contents fluctuated at low concentrations in the range of 2.3–18.0 mg/L (Figure 1b).

During the study, the proportion of chlorophyll to TSS tended to be higher in the Situ Rawa Kalong water, which ranged from 0.11% to 0.83%, compared to the Situ Cibuntu water, which ranged from 0.08% to 0.30% (Figure 1c).

3.2. Daphnid Growth

In the first five days, the daphnid population had grown to 34–47 individuals in the Situ Cibuntu water, consisting of 27–35 juveniles and 5–12 adults, while in the Situ Rawa Kalong water, the population was 4–17 individuals consisting of 4–10 juveniles and 0–7 adults (Figure 2a). This phenomenon showed a more difficult adaptation phase in the more heavily polluted Situ Rawa Kalong water. Although both successfully reproduced immediately after stocking, the survival rates of both juveniles and adults were much better in less polluted water conditions. Figure 2c showed a better population growth rate in the Situ Cibuntu water; the rapid increase in the number of juveniles caused the proportion of adult individuals to decrease to only 20%. The decrease in the proportion of adult individuals also occurred in the water of Situ Rawa Kalong, but it was primarily due to the higher mortality rate of adult individuals.

After going through the adaptation phase, the daphnids grew better in the Situ Rawa Kalong water, and the population increased from 3–9 individuals (day 8) to 24–164 individuals (day12), 14–262 individuals (day 15), and 53–327 individuals (day 19). In the Situ Cibuntu water, the daphnids developed from 23–35 individuals (day 8) to 18–74 individuals (day 12), 15–81 individuals (day 15), and 18–85 individuals (day 19). In the Situ Cibuntu water, the daphnid growth was negative between days 5 and 8. During the study, other types of zooplankton were not found when observing the daphnid density, so it can be ascertained that the daphnids were the only consumers of phytoplankton.

The development of the proportion of adult individuals, especially the average adult proportion of 82% on day 8 in the Situ Rawa Kalong water, indicated that the daphnid population needs adaptation to take advantage of water fertility for its growth. This adaptability varies between individuals, as seen from the survival of adult individuals on day 8 (Figure 2c), resulting in wide variations of population development in the next growth phase.

Meanwhile, the slow development of the daphnid population in the Cibuntu Situ water showed the limited capacity of the water to support daphnid growth (Figure 2a). The ratio of chlorophyll to daphnids in the Situ Cibuntu water tended to be low, indicating that the daphnids could not grow due to limited phytoplankton abundance as the feed and, at the same time, showed the preference of daphnids to feed on phytoplankton (Figure 3a). In contrast, the abundant phytoplankton in the Situ Rawa Kalong water favored the daphnid population to increase and deplete the available phytoplankton, as indicated by the ratio of chlorophyll to daphnids, which continued to decrease until it approached the same value as in the water of Situ Cibuntu (Figure 3).

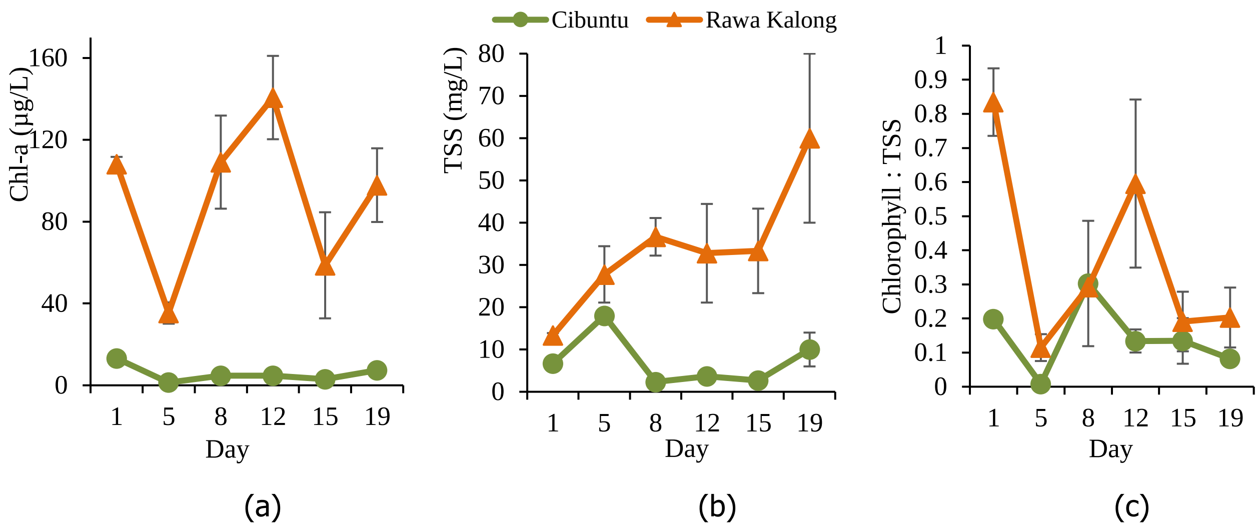

Figure 1. Chlorophyll content, TSS content, and Chlorophyll : TSS ratio in pond water with different trophic levels

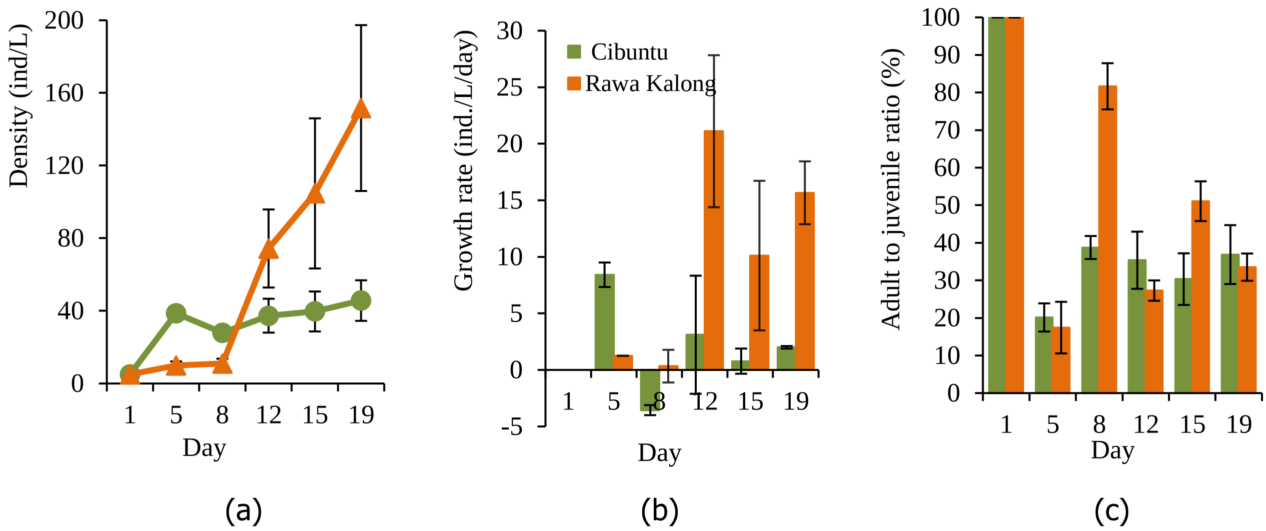

Figure 2. The daphnid population development in the water from both ponds: (a) Daphnid density; (b) Daphnid growth rate; and (c) Adult to juvenile ratio

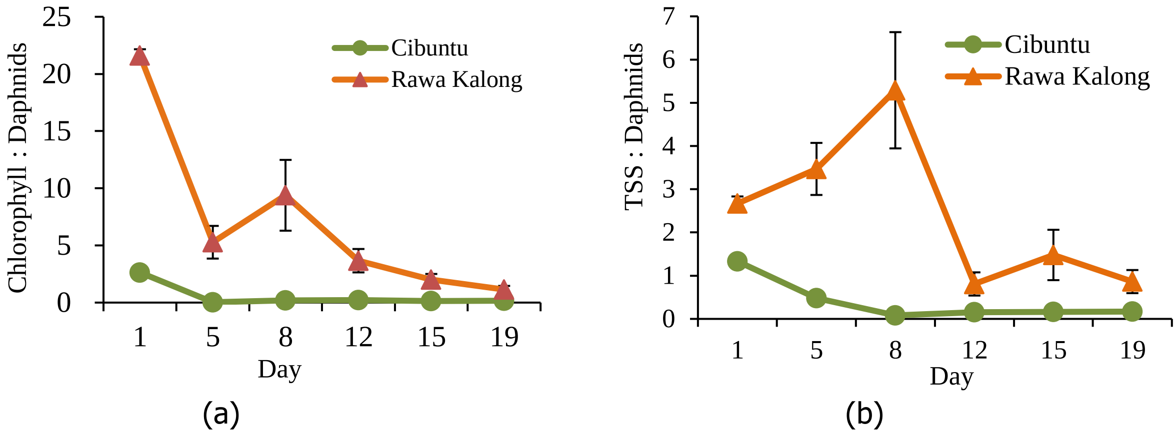

Figure 3. (a) The ratio of chlorophyll to daphnids; and (b) the ratio of TSS to daphnids to indicate feed availability in both pond waters

3.3. Water Quality Condition

The water temperature during the study, measured every morning, fluctuated in the range of 24.4–27.3°C. The pH of the Situ Rawa Kalong water tended to be higher than that of the Situ Cibuntu water, 7.9–9.6 and 6.9–8.2, respectively. The dissolved oxygen (DO) concentrations also fluctuated but were still in the range of 5.7–6.7 mg/L (Table 1). The DO values tended to decrease in the Situ Rawa Kalong water, while they increased in the Situ Cibuntu water during the final phase of the study.

The conductivity values of the Situ Cibuntu water were 53–68 µS/cm, lower than that of the Situ Rawa Kalong water at 132–162 µS/cm. During the study, the conductivity values of both pond water tended to increase. At the beginning of the study, the low turbidity value of 2.79 NTU was obtained from the Situ Cibuntu water, which continued to decrease until day 19 to 0.53 NTU. At the beginning of the study, the turbidity of the Situ Rawa Kalong water was high (54.49 NTU). It decreased on day 5 to 8.25 NTU, along with the formation of microorganism flocs at the bottom of the aquarium, then rose again to reach 40.22 NTU on day 19 (Table 1). The formation of flocs of microorganisms at the bottom of the aquarium occurs in all aquariums.

Table 1. Ranges of water quality parameters during the study

Parameter	Situ Cibuntu	Situ Rawa Kalong
Temperature (°C)	24.80–27.10	25.00–27.30
pH	6.94–8.22	7.94–9.54
Dissolved Oxygen (mg/L)	5.70–6.70	5.80–6.60
Conductivity (µS/cm)	53.00–68.00	132.00–162.00
Total Dissolved Solids (mg/L)	35.00–46.00	88.00–108.00
Turbidity (NTU)	0.44–2.79	8.25–54.49

Daphnids grew better in the eutrophic water of Situ Rawa Kalong than in Situ Cibuntu water, although a more difficult adaptation process was observed at the beginning of the culture. As shown in Figure 2, less population growth occurred in Situ Kalong water in the early phase of culture; however, after going through the adaptation process, a remarkable improvement in population growth was obtained so that it exceeded that of the Situ

Cibuntu water on day 12, and at the end of the experiment, the abundance of daphnids in the Situ Rawa Kalong water was higher by more than threefold. This superior growth can also be expressed in the specific growth rate, which reached 10–20% per day in the Situ Rawa Kalong water after day 8, while in the Situ Cibuntu water, it was only 2–5% per day (Figure 2). The growth of daphnids in the Situ Rawa Kalong water from this experiment was comparable to that reported by Chrismadha & Widoretno (2016) for daphnids grown in water enriched with fish pellet feed, where the maximum population size was 192 individuals/L. However, this was much lower than the total population of daphnids grown in the water medium enriched with various organic matters, where the peak population size was reported to be up to 10,000 individuals/L (El-Feky & Abo-Taleb, 2020; Darmawan, 2014; Izzah *et al.*, 2014). These findings could indicate that although eutrophic open water can support daphnid growth, it cannot yet be used for mass production.

A higher ratio of adults to juvenile daphnids in response to heavier water contamination was observed during the initial phase of culture, especially on day 8 in the Situ Rawa Kalong water compared to that in the Situ Cibuntu water (Figure 2). The lower proportion of juveniles in the Situ Rawa Kalong water during this phase can be attributed to the inability of juveniles to survive in the higher pollution conditions of the Situ Rawa Kalong water. After passing through the eight-day adaptation period, the juvenile proportion tended to increase.

This experiment also showed the population response to food shortage conditions, which was observed in daphnids reared in the Situ Cibuntu water, where juvenile blooms occurred immediately after inoculation until day 5. However, after that, the population experienced depletion due to the high mortality of juveniles, which could be particularly associated with low chlorophyll content in the water. Consistent with this observation, Ranta *et al.* (1993) and Nogueira *et al.* (2004) reported that the growth of daphnids was highly dependent on feed availability.

The initial chlorophyll concentration in Situ Rawa Kalong waters was 108.11 µg/L, while in Situ Cibuntu, it was 13.17 µg/L (Figure 1a). These values indicated that the trophic level of Situ Rawa Kalong was more than eight times higher than that of Situ Cibuntu, which was also supported by the higher content of TSS, conductivity, and turbidity values (Table 1). As pH measurements were conducted during the daytime, a higher pH was also observed in the Situ Rawa Kalong waters, which could be associated with the higher chlorophyll content. All water quality parameters (Table 1), including temperature and DO, were within the conditions suitable for daphnids' growth (El-Deeb Ghazy *et al.*, 2011; Ebert, 2005).

Chlorophyll content has been widely used to represent the abundance of phytoplankton in open water and is considered in assessing the trophic status of waters (Carlson & Simpson, 1996; Chapman, 1996). Sulastri *et al.* (2015) reported the dynamics of phytoplankton composition in Lake Maninjau from observations with an interval of four years from 2001 to 2014, mainly related to seasonal nutrient availability. In the waters of Situ Rawa Kalong, the high nutrient contents of 11.13 mg/L of TN and 0.38 mg/L of TP were reported by Satya et al. (2018). Meanwhile, considerably lower nutrient contents in Situ Cibuntu water, which ranged from 0.068 to 3.623 mg/L of $N\text{-}NO_3$ and 0.0007–0.101 mg/L of $P\text{-}PO_4$ was reported (Meutia, 2000), and an increase in the phytoplankton abundance with phosphorous enrichment was demonstrated (Chrismadha & Maulana, 2012). In addition, Figure 1 also showed a higher chlorophyll to TSS ratio in Situ Rawa Kalong water, which means a higher phytoplankton share in the TSS content in the Situ Rawa Kalong compared to that in the Situ Cibuntu.

The indications of a relatively stable chlorophyll to TSS ratio in the final stages of daphnid culture (Figure 1c) and the chlorophyll to daphnid ratio in the Situ Cibuntu water (Figure 3a) revealed the preference for daphnid feeding on phytoplankton. There was no population growth in this condition. In contrast, the higher phytoplankton abundance in the Situ Rawa Kalong water allowed the daphnid population to increase sustainably and to deplete the availability of phytoplankton as indicated by the ratio of chlorophyll to daphnids, which continued to decrease down to the values almost the same as in the Situ Cibuntu water

(Figure 3a). Although generally, daphnids are known as filter feeders with no preference feeds (Jensen *et al.*, 2001), some pieces of evidence have been reported to emphasize the dependence of the animals on phytoplankton to fulfill the nutritional requirement, particularly essential lipids (Cheban *et al.*, 2017; Martin-Creuzburg *et al.*, 2011). Yin *et al.* (2010) have also demonstrated the ability of *Daphnia magna* to select more nutritious algal cells for feed, while Lotocka (2001) and Mohamed (2001) reported the ability of the animal to reject toxic blue-green alga during feed filtration.

This experiment points out that the abundance of phytoplankton is highly dependent on the trophic status of water and becomes a controlling factor for daphnid growth in pond waters. It also shows that eutrophic open waters are not fertile enough to support a mass culture of daphnids, unlike in cultivation activities where measured organic fertilization is provided to enhance phytoplankton and other natural feed development. Further experiments are still required to elaborate the possibility of employing eutrophic waters to produce daphnid biomass, among others, by implementing a continuous culture system, in which eutrophic water is constantly supplied to the culture vessel to maintain adequate levels of nutrients over time, subsequently to support the growth of phytoplankton and daphnids.

4. Conclusion

This study reveals that trophic level determines the capacity of open waters for growing daphnids, and phytoplankton development is demonstrated to be the controlling factor. It is considered that open eutrophic water is insufficient to be used for mass production purposes of daphnids. It is suggested that controlling the water trophic level at an adequate level, such as by providing a continuous flow into the culture system, can overcome the insufficiency of water trophic level and make the eutrophic open water usable for growing daphnids. Further studies regarding this topic are still needed.

Data availability statement
All data included and used in this study is not confidential and available upon request.

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Conflict of interests
The authors state that we do not have any conflicts of interest to declare.

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Authors contribution
TC developed the experimental concept, provided the daphnid stock, analyzed the data, and prepared the manuscript. **LRT** collected the open water for experimental culture media, conducted the experiment, analyzed the data, and prepared the manuscript. **EN** carried out the water quality monitoring and chlorophyll analysis.

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A biological perspective for the fishery management of a small urban lake in Indonesia: a case study on the reproductive stage of the red devil (*Amphilopus citrinellus*) in Situ Cilodong, West Java, Indonesia

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Abstract: The presence and establishment of invasive alien fish species is one of the biggest threats to aquatic biodiversity. The red devil, *Amphilopus citrinellus*, is one of the emerging invasive species and its occurrence is massively detected in common water bodies in tropical areas such as Indonesia. However, the topic remains under-reported from the small urban lakes. This study aims to present the reproductive characteristics of the fish in Situ Cilodong, a small urban lake in the country, that can be used as a principal reference for population control. The sampling was conducted in June 2021 and May 2022 using a mix of seven mesh-sized gillnets. The results of the length-weight relationship reveal that both the female and male fish perform isometric growth type. The calculated Gonad Somatic Index (GSI) and the histological analysis confirmed that the fish is a multi-spawner species. The results imply that sustainable population control efforts must include intensive catch and engage a participatory approach between the legal authority and the local fishers.

Keywords: alien fish species, invasive fish, reproduction, biodiversity, small lake management

1. Introduction

Both globally and nationally, the biodiversity of freshwater fisheries is undergoing massive threats such as hydrological alteration, habitat degradation, overfishing, pollution, and invasive species domination (Dudgeon *et al*., 2006). The last threat, invasive species, is mostly caused by anthropogenic factors and has generated economic, ecological, and health disturbances (Krantzberg, 2019).

To mitigate such impacts, a structured population control method must be established, such as by preventing their translocation movement (Gherardi, 2010). However, if the species has been established in a new location, gradual eradication is advisory, such as by conducting intensive fishing on the sexually matured fish (Gherardi, 2010; Dina *et al*. 2022). Therefore, knowledge of their reproductive biology becomes important information that should be obtained by the resource managers.

In this study, we use the red devil (*Amphilopus citrinellus*) as a case study to show the importance of the knowledge of reproductive biology in planning a sustainable population control plan. The red devil, previously a valuable ornamental fish, is selected due to its rapid growth and establishment in common water bodies (Umar *et al*., 2015; Hedianto *et al*., 2022).

The fish originally from Nicaragua, Central America (Colombo *et al*., 2013), has been

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reported to massively proliferate in tropical common water bodies such as in Indonesia (Dina *et al.*, 2022). Its ability to adapt to new environments because of its phenotypic plasticity (Salzburger & Meyer, 2004; Machado-Schiaffino *et al.*, 2014) makes the fish categorized as an invasive species that can be harmful by The Regulation of The Minister of Marine and Fishery no. 19/ 2020 (MoMF, 2020).

There has been various research on the reproductive biology of the red devil in Indonesia; for examples: Jatiluhur reservoir (Purnamaningtyas & Tjahjo, 2010), Situ Panjalu (Warsa & Purnomo, 2013), Kedung Ombo Reservoir (Adjie & Fatah, 2015), Lake Sentani (Ohee *et al.*, 2020), Sangiran Reservoir (Santoso, 2019), and Sermo Reservoir (Hedianto, 2023). Nevertheless, those studies did not provide a histological assessment of the fish's gonad development. Furthermore, the study on the topic of urban small lakes is currently absent from academic literature although these types of lakes are crucial to support the livelihood and welfare of marginalized urban people.

Besides, providing water and livelihood for the locals. The urban lake in our case study also serves as a habitat for native fish species such as *Barbodes binotasus* and *Rasbora argyrotaenia* (Phadmacanty *et al.*, 2023). Therefore, the presence of the red devil may generate an adverse impact on biodiversity and human welfare, making thorough planning on the red devil population control vital. We expect that our results can provide basic information and reference for the development of such a plan considering the significance of the information on their reproductive biology.

2. Materials and Methods

2.1. Study site

Situ Cilodong is administratively located in Kalibaru County, Depok Municipality, West Java (Figure 1). Its extent reached a 9.5 ha area with a maximum depth of 3 meters (Pratiwi, 2013). The situ is surrounded by housing and agricultural areas (Figure 1).

The fish sampling was conducted in June 2021 and May 2022 by horizontally setting a 25 x 1.8 m gillnet with a mix of mesh-sized (¾, 1, 1¼, 1½, 2, 2½, and 3 inches) in the outlet area. The sampling site was selected to mitigate its disturbance on the tourism boat while still representing the extent and depth of the situ.

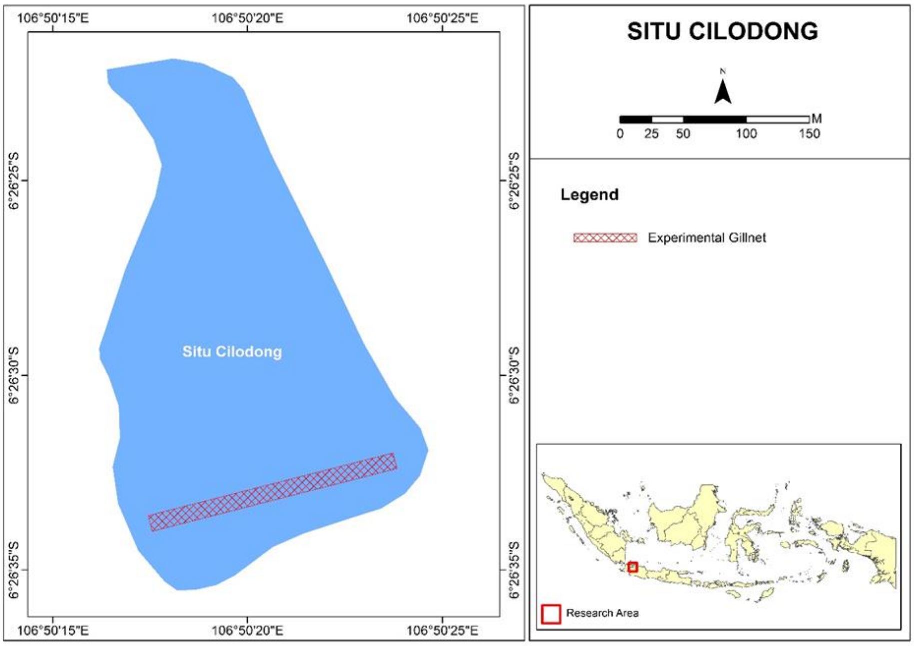


Figure 1. Study site and sampling location

2.2 Sample preservation and laboratory observation

The sampled fish was preserved in a 10% Formaldehyde solution (Silvano *et al.*, 2009), and then in the laboratory, they were wind-dried before being weighed and measured. Then, the samples were dissected to take their gonad to estimate their Gonad Maturity Level/ GML (Effendie, 1997).

Meanwhile, a 5% Formaldehyde solution was used to preserve a total of 15 gonad samples for the histological analysis, and hematoxylin-eosin (HE) was applied to color the specimens (Zulfadhli *et al.*, 2016). The specimens were observed referring to the work of Longenecker *et al.* (2020).

2.3 Data analysis

The length-weight relationship was calculated referring to the formula created by Pauly (1984) (Equation 1).

$$W = a\,L^b \quad \text{....Eq. 1}$$

where; W = fish weight (gram), L = fish length (mm), a and b= Constanta. The obtained b constant is further tested with the 95% confidence level t-test with the H_0: b=3 and H_1: b ≠ 3.

Gonad Somatic Index/GSI (Equation 2) was calculated following the work of (GSI) (Ohta *et al.*, 1996).

$$GSI = \frac{gonad\ weight}{body\ weight} \times 100 \quad \text{....Eq. 2}$$

3. Results and discussion

The range of the length of the fish samples was 66 – 185 mm and 65 – 200 mm for females and males respectively. Meanwhile, their body weighed 6.3 – 135 grams for female fish and 5.4 – 177 grams for males. These measurements show that the fish are larger than the fish sample obtained from Sermo Reservoir (see Hedianto *et al.*, 2022), which implies that the red devil in Situ Cilodong is currently in the reproduction and establishment stage (see Lawson & Hill, 2021).

From the length-weight relationship and the following t-test (Figure 2, Table 1), it can be concluded that the red devil follows isometric growth−equal growth between length and weight (Effendie, 1997). The growth pattern of fish is affected by age, body shape, GML, seasons, temperature, salinity, and food availability (Thulasitha & Sivashanthini, 2012). We suggest that the stable tropical environment and the abundance of food because of the situ's eutrophic condition are the principal factors affecting fish growth in the study area (see Aisyah *et al.*, 2021).

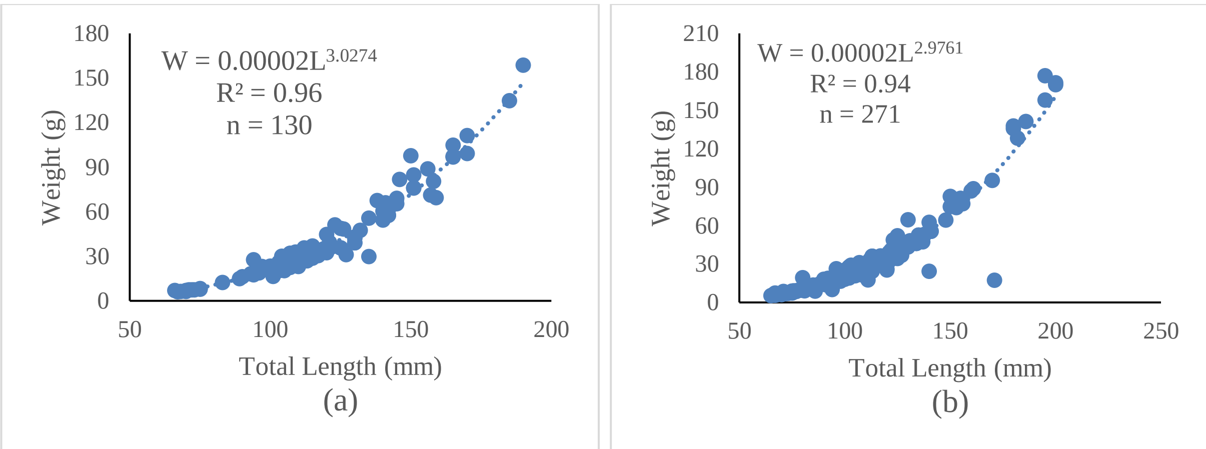


Figure 2. The length-weight relationship of the red devil in Situ Cilodong (a) Females, (b) Males

Table 1. t-test results for the red devil b-Constanta

Sex	t_{stat}	$t_{critical}$	Decision
Female	0.8033	1.9785	Failed to reject H_0: b = 3
Male	0.8553	1.9688	Failed to reject H_0: b = 3

Table 2. Length, weight, GML, and GSI of the red devil in Situ Cilodong

GML	n	Length (mm)	Weight (g)	GSI (%)
I	♂:80	♂: 89 ± 16	♂: 16.28 ± 8.87	♂: 0.02 ± 0.02
	♀:35	♀: 99 ± 19	♀: 22.18 ± 11.41	♀: 0.04 ± 0.02
II	♂:94	♂: 107 ± 13	♂: 25.78 ± 8.66	♂: 0.06 ± 0.04
	♀:37	♀: 109 ± 8	♀: 27.3 ± 5.94	♀: 0.09 ± 0.06
III	♂:30	♂: 135 ± 23	♂: 57.14 ± 30.64	♂: 0.12 ± 0.07
	♀:13	♀: 136 ± 21	♀: 57.28 ± 24.34	♀: 0.32 ± 0.26
IV	♂:3	♂: 198 ± 3	♂: 172.8 ± 3.7	♂: 0.35 ± 0.04
	♀:17	♀: 149 ± 19	♀: 77.31 ± 28.19	♀: 1.85 ± 0.87

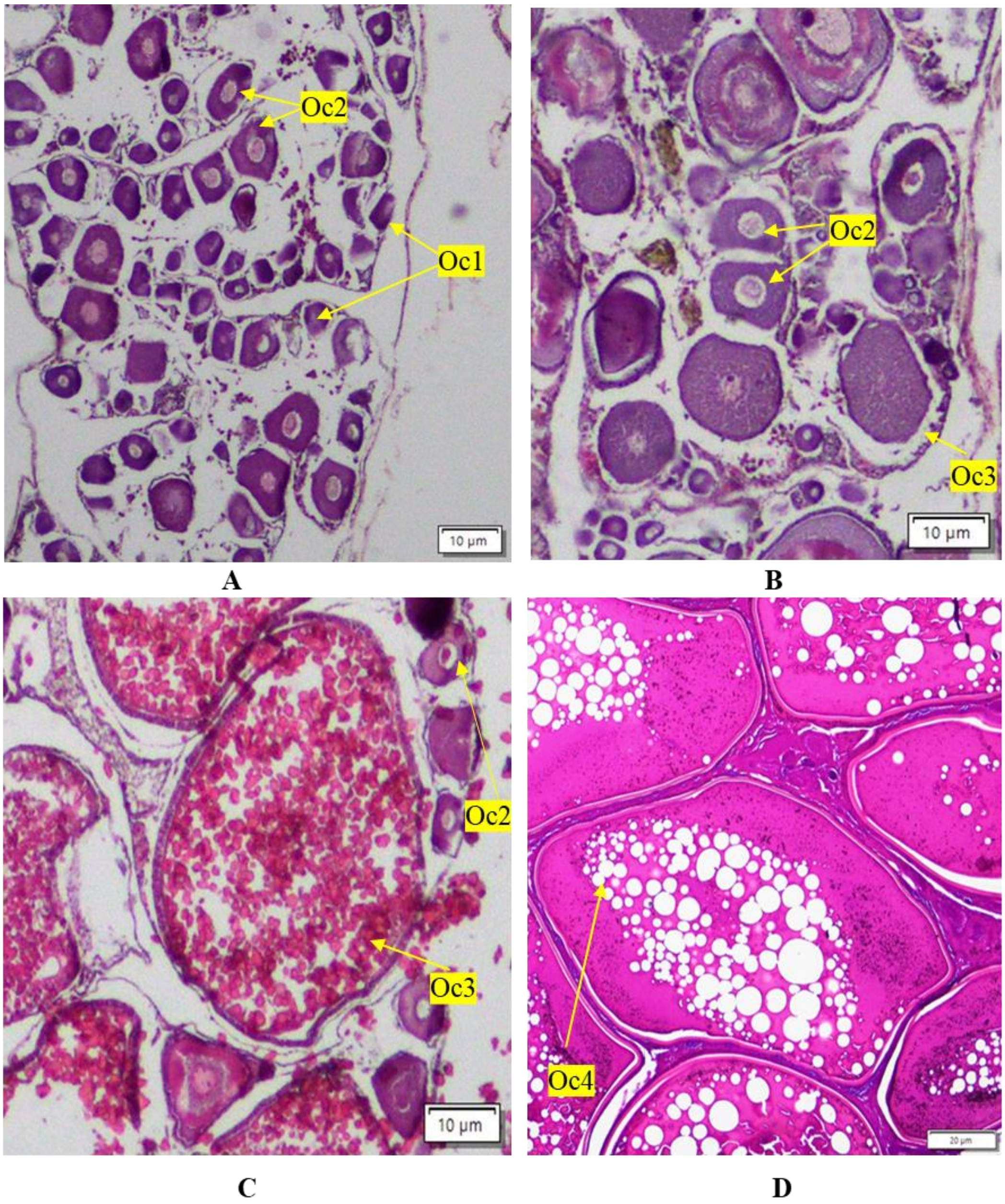
The calculated GSI for female fish is bigger than the male's (Table 2), which is a common phenomenon (Effendie, 1997). Further, the GSI data also reveal that the fish is a multi-spawner species since their GSI is smaller than 20% (Bagenal, 1978). Moreover, the data in the table confirm that four maturity levels are found in the sample, which also aligns with the histological observation (Figure 3 and 4).

According to the criteria proposed by Longenecker *et al*. (2020) and Nurhidayat *et al*., (2017), the histological results elaborate four GML for the female fish: GML I (Figure 3A), characterized by lots of primary oocytes; GML II (Figure 3B), exhibited more second-stage oocytes, thickened nucleus, and epithelial cells; GML III (Figure 3C), identified by the occurrence of third stage oocytes; and GML IV, hinted by the dominance of large sized oocytes.

The histological results for the male fish elucidate that: GML I (Figure 4A), hinted by the abundance of the spermatogonia and the primary spermatocytes; GML II (Figure 4B), shown by the emergence of both the primary and secondary spermatocytes; GML III (Figure 4C), identified with the occurrence of spermatid; and GML IV (Figure 4D), characterized by the spermatozoa equipped with flagella (ibid).

The histological analysis presents that the female red devils undergo asynchronous gonad development referring to the fact that there are several oocyte stages in the same GML. Thus, the fish is considered a multi-spawner species (Muchlisin, 2014; Purnamaningtyas & Tjahjo, 2010). This result corroborates the study conducted by Adjie & Fatah (2015), who observed the reproductive biology of the red devil in Kedung Ombo Reservoir.

Contextualizing our results with the study area, we suggest that eradication of the red devil in this situ using intensive catch as the most appropriate control program. Further, it is advised that the catch should be conducted at various times within a year considering that the fish can perform several spawning seasons. The use of selective fishing gear such as gillnet with appropriate mesh size is also recommended. In this case, there should be further discussion regarding the governance process because, in the current situation, the locals are only allowed to use hooks. Therefore, we recommend that the Watershed Agency of the Ministry of Public Works and Housing take the lead in the population control process. Furthermore, we advise the endorsement of a participatory approach connecting the government agencies and the local fishers.



Legend:
OC1 : Primary oocyte stage (oocyte size: < 7.43 µm)
OC2 : Cortex alveoler stage (oocyte size: 40-22.23 µm)
OC3: Vitelogenic stage (oocyte size: 23.74-58.93 µm)
OC4: Mature oocyte stage (oocyte size: >60.11 µm)

Figure 3. Histological observation on the female red devil: (A) GML I (*immature*), (B) GML II (*develop*), (C) GML III (*mature*), (D) GML IV (*ripe*)

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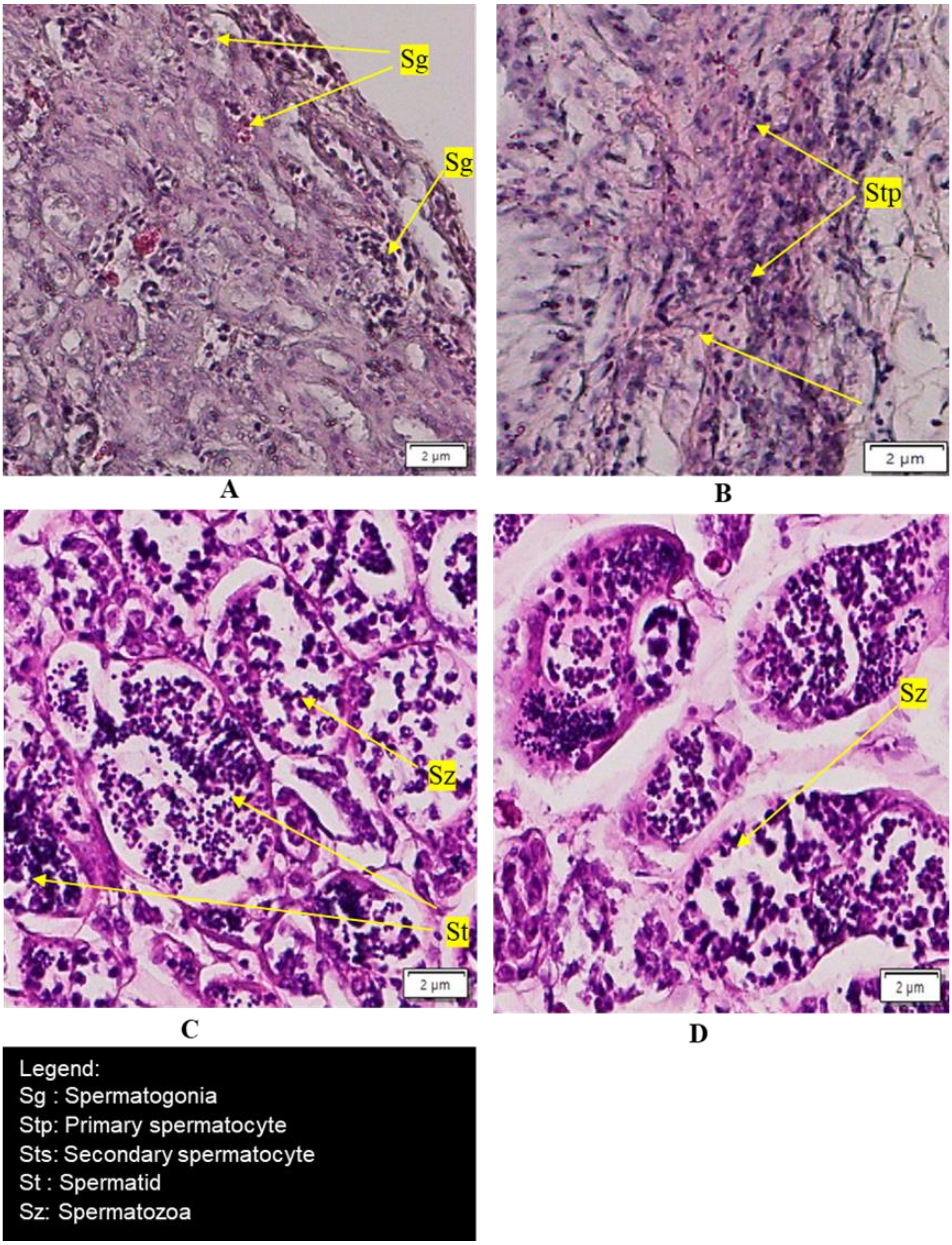

Figure 4. Histological observation on the female red devil: (A) TKG I (*immature*), (B) TKG II (*develop*), (C) TKG III (*mature*), (D) TKG IV (*ripe*)

4. Conclusion

We attribute the red devil as a multi-spawner species with several spawning seasons. Hence, there is a tendency that the fish can be a great biodiversity threat in the study area. We extend this knowledge can be a trend in other water bodies in Indonesia considering its climatic suitability. We suggest that a coordinated action plan can be performed by the authorities and local people to mitigate the impacts. Our main recommendations also include intensive catch and continuing monitoring programs in the area where the presence of the red devil has been acknowledged.

Data availability statement

We declare that all required data have been written and stated in this manuscript.

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Conflict of interests

The authors declare that there is no conflict of interest.

Author Contributions

IA and **RD** designed the topic and method of this research. **IA**, **GW**, and **AW** assisted in the fieldwork and data collection. **FSL**, **EN**, and **DO** processed the data and compiled articles. **IA**, **RD**, and **GW** helped improve the manuscript.

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Sorption kinetics of heavy metals from aqueous solution using *Spirogyra* sp.: a microcosm study

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Abstract: Understanding the mechanisms by which algae communities respond to disturbances in the lotic aquatic environment that is polluted by heavy metals is important, considering that algae is a biotic component of waters that acts as a producer in the aquatic food chain which has the potential to bio-magnify. This study examines the influence of time, biomass weight, heavy metal concentration, sorption capacity, and efficient removal on epilithic periphyton as a bio-accumulator of Cr, Pb, and Ni. The experiment was conducted on a laboratory scale using a canal system with a length and width of 1.2 and 1.0 meters, respectively. The canal system contains 132 L of water, has a 1.2 m^2 substrate and periphyton area, a depth of 0.09 – 0.10 m, and a current flow rate of 0.04 – 0.06 m/s. The dissolved Cr^{6+} initial concentration in the medium was 1.64 mg/L, Pb^{2+} and Ni^{2+} concentrations were 1.4 mg/L, and the adsorption process was studied for 24 hours. Based on microscope observations and functional group interpretation utilizing infrared spectra (FTIR), the periphyton community is dominated by *Spirogyra* sp., which has hydroxyl (O-H), carboxyl (C-H), and carbonyl (C-C and C=O) functional groups with the ability to binding heavy metals. The remaining quantities of Cr, Pb, and Ni in water were 0.43 mg/L (removal 69.29%), 0.05 mg/L (96.43% removal), and 0.03 mg/L (97.86% removal). Periphyton has a maximal sorption capacity of 1.019 mg Cr/g, 1.97 mg Pb/g, and 1.92 mg Ni/g. The sorption kinetics of Cr, Pb, and Ni follow a pseudo-second-order model with k_2 = 1.686 x 10^{-2} g/mg.min for Cr, 4.516 x 10^{-3} g/mg.min for Pb, and 2.259 x 10^{-2} g/mg.min for Ni, with R^2 of 0.965 for Cr and 0.971 for Pb and 0.972 for Ni. Periphyton can potentially play a role as a bio-accumulator in lotic habitats, adsorbing Cr, Pb, and Ni ions, according to this study.

Keywords: sorption capacity, sorption kinetics, Spirogyra, Cr, Pb, Ni

1. Introduction

Heavy metals are considered to be among the most serious ecological issues, as well as one of the most difficult to address. Heavy metals such as mercury, lead, arsenic, chromium, copper, cadmium, and nickel are frequently employed in the industrial sector, particularly in metal polishing and plating, as well as in products such as batteries and electronic equipment (Ali *et al.* 2021). Heavy metal-containing wastewater has been a serious cause of concern due to its toxicity, environmental persistence, bio-accumulative nature, and carcinogenic effects (Vertinsky, 2021). Even a trace amount may cause severe physiological and neurological consequences (Jaishankar *et al.* 2014; Ali et al. 2021). As a consequence, several attempts have been made to prevent or reduce this type of possible health threat.

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As a result, algae might serve a significant part in removing heavy metals from aquatic ecosystems and can contribute to environmental sustainability (Goswani *et al.* 2022). Algae are an enormous and diversified grouping of simple plant-like organisms that occur in freshwater, maritime, and wetlands areas, that vary from single-cellular to multi-cellular species. This bio-sorbent has received substantial research because of its widespread presence in nature. Algae has found uses as a compost, energy source, pollution reduction tool, stability substance, and nourishment, among other things. Recently, the metal adsorption capabilities of untreated and treated algae have been investigated. A lightweight, stiff cell membrane encloses the algal cells, with pores 3 – 5 nm wide that allow molecular-weight components such as water, ions, gases, and other elements to move freely across for growth and metabolism. Cell walls, on the other hand, appear to be impervious to bigger particles or macromolecules (Shamshad *et al.* 2014; Shamshad *et al.* 2016).

Algae in freshwater can acquire heavy metals via the sorption process, which comprises either physical as well as chemical sorption. The distinctive cell wall component structures in algal biomass, particularly through the cellular surfaces and cell wall spatial structure, determine the nature of metal bioaccumulation by algae (Znad *et al.* 2022). Through physical interactions and van der Walls forces, algae bind heavy metals to the surface of algae cells through physical adsorption (Yogeshwaran & Priya, 2022; Zeng *et al.* 2022). Algae cells' negatively charged surfaces can bind positively charged metal ions like chromium (Cr^{6+}), lead (Pb^{2+}), and nickel (Ni^{2+}). Chemical adsorption involves chemical bonding such as complexation, chelation, and exchange of ions between the outer layer of algae cells and metallic ions. Various functional groups in algal cell wall polysaccharides, such as carboxyl, hydroxyl, sulfate, sulfhydryl (thiol), phosphate, amino, amide, imine, thioether, phenol, carbonyl (ketone), imidazole, phosphonate, and phosphodiester, have the attributes to be associated with metal bonding (Omar, 2013; Ahmad *et al.* 2019; Spain *et al.* 2021). Some algae are capable of heavy metal absorption into specific organelles or intracellularly. High heavy metal concentrations, on the other hand, can harm the integrity of the algal cells (Ge *et al.* 2022).

Research using dried Spirogyra biomass as a biosorbent against heavy metals Pb, Cu (Lee et al. 2011), Cr (Onyancha et al., 2008), Ni (Guler & Sarioglu, 2013), Fe and Pb in fixed bed column (Yahya et al., 2020), Mn, Zn, Cd (Rajfur et al., 2010), and textiles dyes (Khataee et al., 2013) has been widely carried out, however, the sorption mechanism for the bioaccumulation of natural Spirogyra biomass in lotic waters and its potential for biomagnification in the food chain is still not well-informed. The purpose of this research was to assess the ability of the species of freshwater algae *Spirogyra* sp. to accumulate heavy metals Cr, Pb, and Ni ions. The effects of contact time, biomass weight, and initial level of heavy metals on capacity and biosorption efficiency were investigated and assessed. As a result, this work contributes to a better understanding of heavy metal pollution at compartment levels such as water and algae in lotic waters.

2. Materials and Methods

The experimental study was conducted at the Research Centre for Limnology and Water Resources, BRIN – Indonesia. This research includes several stages, (i) canal system development, (ii) colonization of periphyton, (iii) preparation of ion Cr^{6+}, Pb^{2+}, and Ni^{2+} solution, (iv) metals bioaccumulation test using periphyton.

The materials used were algae of *Spirogyra sp*, HNO_3 65%, standard solution of Cr, Pb, and Ni 1000 mg/L, NPK solution in 2 mg/L, and deionized water. Instruments used were Spectrophotometer UV-Vis 1800 Shimazu, GF-AAS Hitachi Z2000, Infrared Spectrophotometer Transformation Fourier (FTIR) Shimadzu IRPrestige-21, microscope Nikon Diaphot 300, analytical balance Ohaus, vacuum filters Eyela A-3S, oven Memmert, hotplate magnetic stirred Ika C-Mag HS-7, and glassware in the laboratory.

(i) The canal system was designed to simulate the stable condition of lotic water. The canal system was manufactured of acrylic and had dimensions of a length and width of 1.2 and 1.0 meters, respectively. It was filled with 132 L of water and had a periphyton area of 1.20 m^2. The water depth ranges from 0.09 to 0.10 m, with a current flow rate of 0.04 – 0.06 m/s.

(ii) The periphyton was grown in a canal system by spreading *Spirogyra* sp. seeds and adding a 2 mg/L NPK solution. The attached periphyton grew to the substrate for two weeks, assuming that periods are sufficient to determine the homogeneity of periphyton that grow in the lotic layer. To determine the prevalent periphyton algae species, samples of growing periphyton were taken during the acclimatization period, at the beginning and end of the observation, and examined under a microscope.

(iii) Ion Cr^{6+} solution was obtained at 1.64 mg/L, Pb^{2+} dan Ni^{2+} at 1.40 mg/L using a standard solution of ion Cr, Pb dan Ni 1000 mg/L. The concentrations used are ion Cr, Pb, and Ni effective concentration of 50% (EC50) (Yap *et al.* 2004).

(iv) The preparation of 1000 mL of 50 mg/L Cr^{6+} stock solution from $K_2Cr_2O_7$ was established by weighing 0.14144 grams of dry $K_2Cr_2O_7$, which was then weighed and dissolved in a 1000 mL volumetric flask with demineralized water. A 1.64 mg/L Cr solution was pipetting 3.28 mL of 50 mg/L Cr (VI) stock solution into a 100 mL volumetric flask and adding demineralized water to exactly 100 ml.

(v) A 50 mg/L Pb and Ni standard solution was made by pipetting 5 mL of a 1000 mg/L Pb and Ni standard solution and diluted in 100 mL of demineralized water using a volumetric flask. A 1.4 mg/L Pb solution was prepared by pipetting 2.8 mL of a 50 mg/L Pb and Ni solution into a volumetric flask and adding demineralized water to exactly 100 ml.

(vi) Cr^{6+} measurements were carried out by pipetting 3 mL of water samples and adding 0.15 mL of 50% H_2SO_4, 0.5 mL of 0.5% diphenyl carbazide, and 9 mL of demineralized water. Then the sample was left for 5 minutes and its absorbance was measured at 540 nm using a UV-Vis spectrophotometer.

(vii) Periphyton structures were analyzed before and after the bioaccumulation process. Periphyton samples were dried and mixed with KBr. The mixture was crushed until it became a fine particle and then pressed to form pellets. The pellets obtained were inserted into the sample holder and the infrared absorption spectrum was observed between 400 and 4000 cm^{-1} wavelengths. Bioaccumulation of metal ions using periphyton was observed for time periods of 0, 15, 30, 60, 120, 240, 480, and 1140 minutes after metal exposure to determine the sorption rate. Periphyton biomass and water samples were taken randomly. Water samples were digested with 65% HNO_3 according to standard methods (APHA, 2012). Periphyton biomass attached to the substrate was brushed and then dried at 40°C, weighed for its dry weight, and then digested with 65% HNO_3 according to standard methods (APHA, 2012). The solution was measured using AAS at a wavelength of 540 nm for Cr, 261 nm for Pb, and 232 nm for Ni. The heavy metal content in periphyton biomass was analyzed based on total Cr, Pb, and Ni.

The sorption capacity can be calculated by the formula in Equation 1:

$$Q = \frac{V(C_0 - C_t)}{m} \qquad \text{....Eq. 1}$$

The sorption efficiency can be calculated using the formula in Equation 2:

$$\text{Efficiency} = \frac{C_0 - C_t}{C_0} \times 100\% \qquad \text{....Eq. 2}$$

where:
Q = adsorption capacity per biomass weight (µg/g biomass)
V = volume of solution (mL)
C_0 = metal level at t 0 (mg/L)
C_t = metal level at t (mg/L)
m = periphyton biomass (g)

Kinetics of biosorption. The rate of pseudo-first-order (PFO) biosorption kinetics rate equation was proposed by Lagergren (1989) for the adsorption of a liquid-solid system derived from solid adsorption capacity. A PFO kinetic model's linearized equation is expressed as Equation 3 follows (Satya *et al.* 2020):

$$\frac{dq_t}{dt} = k_1(q_1 - q_t) \quad \text{....Eq. 3}$$

to get the k_1 and q constants, the equation above can be derived from Equation 4:

$$\ln(q_e - q_t) = \ln(q_e) - k_1 t \quad \text{....Eq. 4}$$

where:
k_1 = the constant of PFO (min^{-1})
q_e = the number of metallic ions adsorbed at equilibrium
q_t = the number of metallic ions adsorbed at t (mg/g)

The rate of pseudo-second-order (PSO) kinetics was evaluated from Equation 5 which may be written below (Satya *et al.* 2020):

$$\frac{dq_t}{dt} = k_2(q_e - q_t)^2 \quad \text{....Eq. 5}$$

where:
k_2 = the constant of PSO (g/mg.min^{-1})
q_e = the number of metallic ions absorbed at t (mg/g).

The formula can be modified into the passage that follows linear by separating the variables in the formula and fostering the equation under the constraints of t = 0 to t and qt = 0 to t:

$$\frac{t}{q_t} = \frac{1}{h} + \frac{1}{q_e} t \quad \text{....Eq. 6}$$

where:
h = the $k_2 q_e^2$ constant (mg/g.hr). The constant of PSO (k_2) was obtained through experiment by graphing t/qt against t.

3. Results and Discussion

3.1. The dominant type of algae community

Periphyton colonies formed on the rock substrate for two weeks (Figure 1a) until green filamentous algal periphyton were obtained (Figure 1b). Filamentous algal periphyton grows longitudinally and covers practically the entire rock surface, reaching a biomass density acceptable for bioaccumulation testing. The canal system's water temperature ranges from 25 to 35^{0}C, while the pH ranges from 7 to 9. The dissolved oxygen content measured ranged from 5 to 15 mg/L. This condition meets the parameters for periphyton growth, with temperatures ranging from 20 to 36^{0}C and pH ranging from 7.5 to 8.4 (Nybakken, 1993).

The periphyton colonies were dominated by filamentous algae from Chlorophyta, *Spirogyra* sp., unicellular algae *Cosmarium* sp., and diatoms. Spirogyra is a genus that is commonly found in freshwater environments. Microscope images demonstrate Spirogyra's unbranched form and spiral-shaped bands of *Spirogyra's* chloroplast (Lee & Chang, 2011).

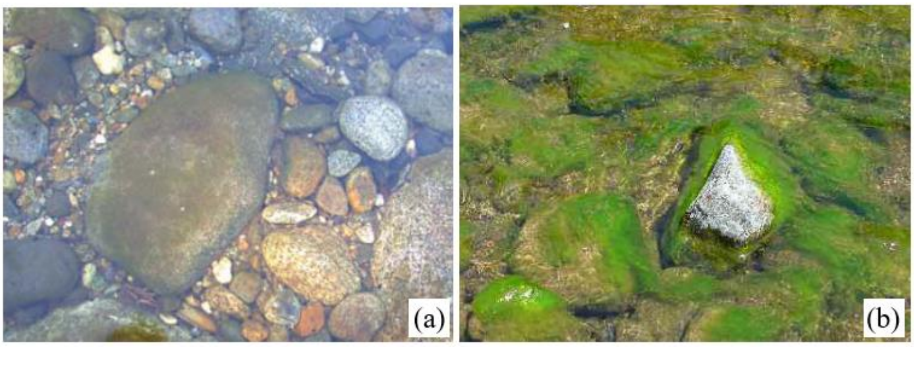

Figure 1. (a)Rock substrate used for periphyton growth, (b) periphyton colonies obtained after two weeks

3.2. FTIR Spectrum

Algal cells are comprised of polysaccharides with ion exchange properties such as cellulose, acid alginic, and sulfate (Loukidou *et al.* 2004; Turker & Baytak, 2004). This polymer has several groups with functions that can act as metallic ion binding regions. During or following the adsorption of the Cr, Pb, and Ni processes, the periphyton was analyzed using FTIR to observe changes in the functional group contained in the periphyton. The FTIR spectrum before the adsorption process shows similarities to the FTIR spectrum of Spirogyra (Table 1). The FTIR spectrum of Spirogyra displays hydroxyl (O-H) and amine (N-H), carboxyl (C-H), and carbonyl (C-C and C=O) functional groups (Onyancha *et al.* 2008).

Interaction of Spirogyra with heavy metals, carbonyl, and carboxyl groups in molecules such as proteins, amino acids, lipids, or carbohydrates can all play a part in heavy metal binding. Pb, Cr, and Ni may react with complexity with these groups, influencing the structure and function of algal molecules. The hydroxyl group contains oxygen, which can act as an electron pair donor in coordination bonds with heavy metals. Pb, Cr, and Ni ions can form bonds with hydroxyl oxygen, which is found on the surface of algae in molecules such as alcohol, phenol, or sugar. Lignin molecules in algal cell walls include phenol groups. Carboxylate groups also include oxygen, which can link to heavy metals. Heavy metals can make complicated interactions with oxygen carboxylates in fatty acids or amino acids in algae. This is often accomplished through coordination bonding, in which oxygen carboxylate functions as an electron donor to create bonds with metal cations. Algae amine groups can form coordination connections with heavy metals. Heavy metals can form strong coordination bonds with electron pairs on amine nitrogen. Although these interactions are often weaker than those via hydroxyl or carboxylates, amines can nonetheless contribute to heavy metal binding.

Table 1. Periphyton spectrum IR comparison before and after Cr, Pb, and Ni adsorption

Adsorption wave number (cm^{-1})			Chemical bounds
Periphyton before adsorption	Periphyton after adsorption	*Spirogyra* (Onyancha *et al.* 2008)	
–	–	3622	N–H
3339	3420	3341	O–H
2926	–	2925	C–H
–	–	2360	–CC–
1654	1653	1656	C=O and COOH
1425	1425	–	C–H
1037	1038	1038	C–O
876	–	–	C–N–S

Pb, Cd, Hg, and Zn have a great affinity for sulfide groups. Sulfide-heavy metal interaction can result in less solubility precipitation of heavy metal sulfides. Sulfide groups found in algal components such as cysteine and glutathione can help heavy metals bind via coordination interactions. A carbon atom is doubly linked to an oxygen atom to form the carbonyl group (C=O). Because of the difference in electronegativity between carbon and oxygen, this group displays substantial polarity. Coordination bonds, in which the oxygen atom functions as an electron pair donor to create bonds with metal cations, allow the carbonyl group to interact with heavy metals. Interactions between carbonyl groups in protein or carbohydrate molecules and heavy metals occur in Spirogyra. A carboxyl group is a complex structure made up of the group carbonyl (C=O) and hydroxyl (OH) which are both connected to one single carbon atom. This group offers organic compounds acidic properties along is capable of creating coordination bonds with metallic substances. In bonding with heavy metal cations, an oxygen atom that is part of the carboxyl acts as an electron pair. The carboxyl group is more polar in general and can interact with heavy metals in a variety of chemical compounds found in *Spirogyra* sp.

3.3. Bioaccumulation of Cr, Pb, and Ni using periphyton

The ideal biosorbent has to be one that can speedily adsorb large quantities of metals from wastewater and desorb them using chemical substances (Singh *et al.* 2007). The relationship of dissolved Cr, Pb, and Ni metal ion concentration in waters with time is presented in Figure 2.

Rapid bioaccumulation of Cr and Pb occurred in the first 480 minutes, and Ni occurred until 1440 minutes. At the adsorption time of 480 minutes, the remaining Cr and Pb concentrations in the water were 0.96 mg/L and 0.14 mg/L and at the adsorption time of 1440 minutes, the water's residual Ni content was 0.03 mg/L. During this time, ion exchange and physical adsorption rapidly that occurred are suspected on the periphyton surface cell wall. The adsorption rate of metals was very high during the first 8 hours, reaching about 85% of the total adsorption with Cr, Pb, and Ni concentrations remaining in water at 0.96 mg/L, 0.14 mg/L, and 0.09 mg/L, respectively. Then the adsorption rate starts to remain constant towards the equilibrium state. Ion Pb was adsorbed faster at the beginning because Pb's radius (0.175 nm) is bigger than Cr (0.139 nm) and Ni (0.072 nm), so the active site on the adsorbent surface saturates faster. The subsequent slow phase of adsorption may involve other mechanisms, such as saturation of the active site, complexation, or micro-precipitation (Lee & Chang, 2011; Onyancha *et al.* 2008). The concentration of Cr in the water at twenty-four hours of adsorption was 0.43 mg/L, Pb was 0.05 mg/L, and Ni was 0.03 mg/L. The value of Cd and Pb are still higher than the quality standard for Class C according to The Ministry of Environmental Decree No. 115/2003 amounting to 0.03 mg/L while for Ni there is no certain standard value for standard quality.

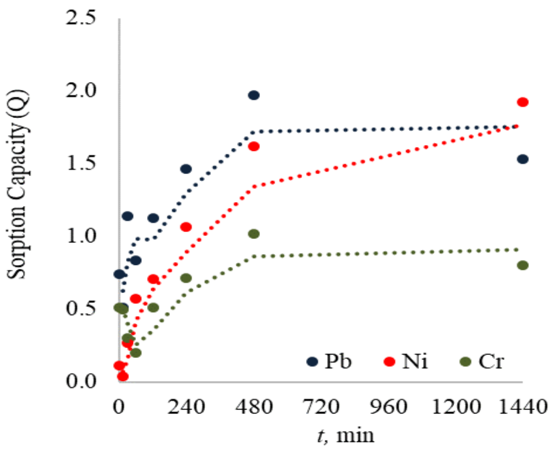

Figure 2. The sorption capacity of periphyton on Cr, Pb, and Ni

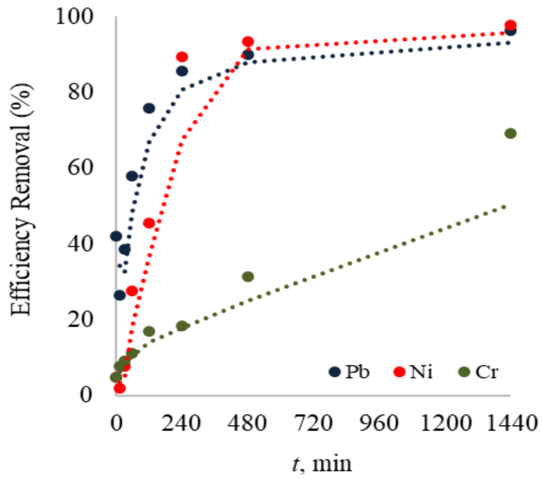

Figure 3. The sorption efficiency of periphyton on Cr, Pb, and Ni

Metal ion concentrations in periphyton biomass generally increase with time, but there is an inflection point on the Pb adsorption curve at 60 to 240 minutes. This decrease may occur depending on the circumstances or the process (Hill & Boston, 1991). In this study, the decrease could be caused by the presence of biological processes in periphyton which are living organisms. In addition, the random sampling process allows the extraction of stones of non-uniform thickness for each time of collection. The difference in nutrient consumption between periphyton colonies can also be the cause of the uneven metal adsorption process in the canal.

The composition of the type of periphyton growing in the canal system greatly affects the metal's capacity to bind in the waters. The differences in metal adsorption by various kinds of algae are mostly owing to variances in cell surface properties, particularly within the cell membrane. The outer layer of a cell is the main target of attaching metals in algae, and metal trapped on the surface frequently outnumbers metal accumulated in the internal compartment. (Andrade *et al.* 2005; Mehta & Gaur, 2005).

The biosorption capacity value was directly proportional to the biomass concentration. Biosorption was carried out at a media pH of 7 – 8, which is the optimum pH in the sorption process for Pb and Ni (Sing & Yu, 1998). Cr has a lower sorption capacity than Pb and Ni because Cr is more easily absorbed at pH which tends to be acidic (Imyim *et al.* 2016; Ding *et al.* 2022; Nafisyah *et al.* 2023). Pb, Ni, and Cr have the highest adsorption potential (Q_{max}) of 1.973 mg/g, 1.922 mg/g, and 1.019 mg/g, with the biosorption efficiency reduction Pb by 96.43%, Ni 97.86% and Cr 69.29% (Figure 3).

3.4. Sorption Kinetic

The bioaccumulation kinetics of Cr, Pb, and Ni were determined using the Lagergren equation. The Lagergren equation can be applied as pseudo-first-order (PFO) kinetics, assuming the number of metallic ions exceeds the percentage of active sites along the outer layer of the adsorbent. This formula becomes successfully utilized for modeling sorption kinetics data that occur in living microorganisms when concentrations are high and the process is constant (Loukidou *et al.* 2004; Gupta & Rastogi, 2008; Onyancha *et al.* 2008). Linear regression by passing log (qe – qt) against t will produce a PFO kinetics model with a constant value of k_1 (Figure 4).

The findings revealed the validity of the Cr, Pb, and Ni biosorption kinetics followed the PSO equation, indicated by a degree of determination coefficient (R^2) of 0.971, 0.972, and 0.965. According to Eq. 4, if the path is linear, the sorption mechanism is known as chemisorption. The PSO adsorption rate constants (k_2) for Cr, Pb, and Ni were 1.686 x 10^{-2}, 4.516 x 10^{-3}, and 2.259 x 10^{-2} g/mg. min, respectively (Figure 5).

The biosorption of metal ions in periphyton Spirogyra follows two phases. The first phase is rapid metabolism with adsorption on the outermost layer and cellular wall, the second phase is slow metabolism depending on transport across the cell membrane.

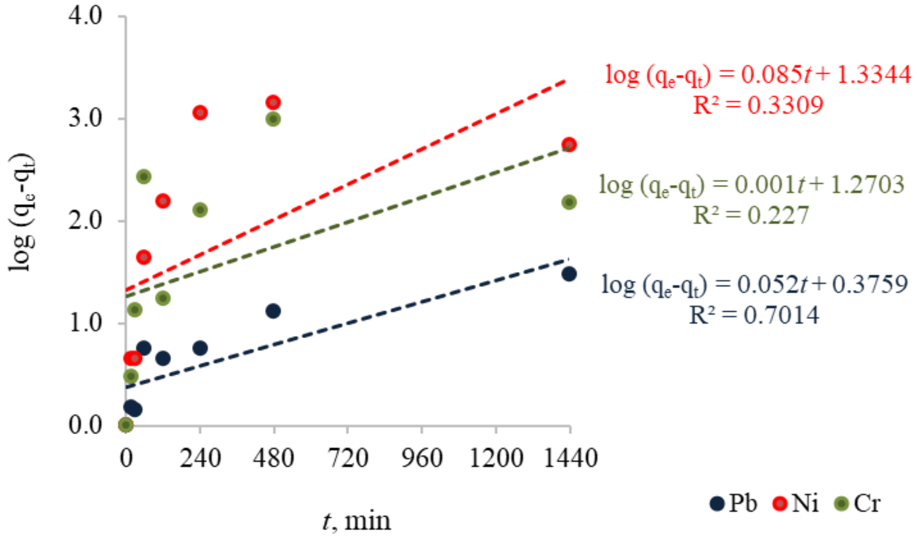

Figure 4. PFO sorption model of Pb, Ni, and Cr in periphyton biomass

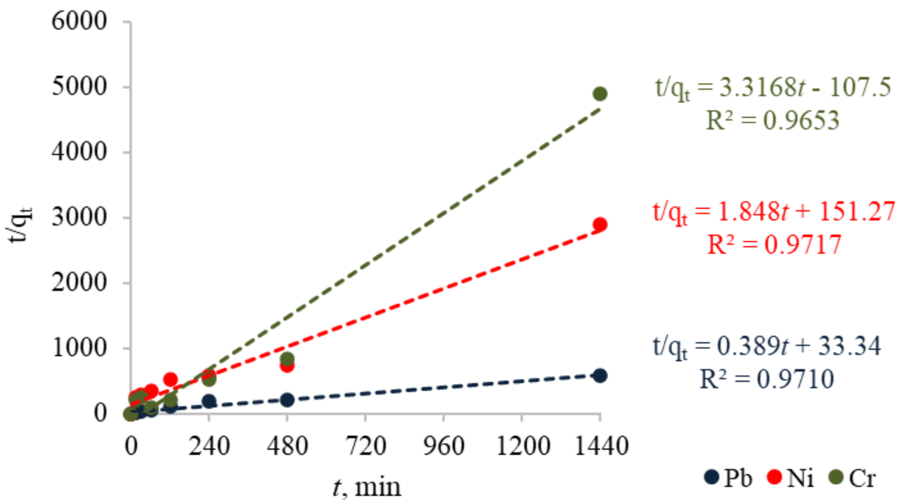

Figure 5. PSO sorption model of ion Pb, Ni, and Cr in periphyton biomass

4. Conclusion

Periphyton dominated by Spirogyra has the potential as a bio-accumulator for Cr, Pb, and Ni. The maximum biosorption capacity for Cr was 1.019 mg/g, Pb was 1.973 mg/g and Ni was 1.923 mg/g. The biosorption kinetics of Pb and Ni follow a pseudo-second-order reaction equation with a value of k_2 = 1.686×10^{-2} g.mg^{-1}.min^{-1} for Cr, k_2 = 4.516×10^{-3} g.mg^{-1}.min^{-1} for Pb and k_2 = 2.259×10^{-2} g.mg^{-1}.min^{-1} for Ni. The coefficient of determination (R^2) was 0.965 for Cr 0.971, for Pb, and 0.972 for Ni. The findings of this study can be used to characterize the bioaccumulation mechanisms of Cr, Pb, and Ni by periphyton Spirogyra in lotic waters.

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Author Contributions

ES and **FSL** as the main contributors conceptualized the study and data analysis, and wrote the original article. **RK**, **MRW**, **DO**, **EN**, and **NM** carried out the canal system construction, sampling, and analysis processes in the laboratory.

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Sediment capping technology for eutrophication control and its potential for application in Indonesian lakes: a review

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Abstract: Eutrophication occurs when the lakes become enriched with nutrients. Some nitrogen and phosphorus fractions will settle in sediment, and others will be released back into the overlying water column. Excess nutrients in water bodies resulting in hypoxic to anoxic conditions that can cause a mass fish death. Hence, we need a sediment management strategy to minimize resuspension and transport of sediment back into the water column. Sediment capping is a containment technology to reduce the release of nutrients from sediment as a strategy for eutrophication control. This study aims to provide insight into sediment capping technology, including several considerations in capping design, as well as information on several active materials that have been applied as capping materials and their efficiencies. Capping materials such as calcite, zeolite, bentonite, activated carbon, sludge, biochar, and gypsum from previous studies showed the efficiency of 54–99 % nutrient reduction with capping duration of 10–300 days in some eutrophic lakes. Sediment capping technology has successfully promoted lake ecosystem restoration in other countries, and this technology has the potential to be applied in Indonesian eutrophic lakes as a strategy for eutrophication control and sustainable management of lake ecosystems by considering the selection of the most effective, efficient, easy, inexpensive, and eco-friendly capping materials.

Keywords: sediment capping technology, eutrophication, Indonesian lakes

1. Introduction

Anthropogenic factors associated with industrial, urban, agricultural, domestic, and fish cultivation activities have led to increasing amounts of nutrients in aquatic environments, which led to a condition called eutrophication. Eutrophication occurs when a lake becomes nutrient-enriched (Wetzel, 2001). Some nutrient species like nitrogen and phosphorus fractions will settle in sediment, while other fractions which are redox-sensitive under anoxic conditions such as ammonia-nitrogen (NH_4^+-N), nitrate, organic nitrogen, and phosphorus bound to chemical compounds like iron (Fe) will be released back into the overlying water column (Phillips *et al.*, 2006; Zamparas *et al.*, 2014; Wang et al., 2018; Papera *et al.*, 2021). In this case, sediment acts as both carriers and long-term secondary sources of contaminants in aquatic ecosystems (Zhang *et al.* 2016). Excess nutrients in water bodies can lead to both overgrowth of algae and eutrophication. As dead algae decompose, oxygen is consumed in the process, resulting in low levels of oxygen (hypoxic) and anoxic

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conditions that can cause mass fish death (Jenny *et al*., 2016). In situ remediation technologies to prevent eutrophication have been studied such as floating treatment wetlands (Coveney *et al*., 2002; Tanner *et al*.,2011; Henny *et al*., 2020) that are only effective for water surface remediation. While in situ technologies for contaminated sediment such as dredging (Reddy *et al*. 2007 and Yu *et al*. 2017), chemical precipitation (Gonsiorczyk *et al*., 1998; Lürling and Oosterhout 2013), in situ chemical injection (Søndergaard *et al*., 2002; Engstrom et al., 2005; Wang and Jiang, 2016), and hypolimnetic oxygenation (Beutel, 2006; Liboriussen *et al*., 2009). However, these technologies have some weaknesses, including high cost, ineffective control of nutrient reduction, and toxicological risk to aquatic biota (Reitzel *et al*., 2013). Indeed, the management strategy for contaminated sediments has become one of the most challenging problems in the aquatic environment.

Sediment management strategies consist of five categories, which are selected based upon an evaluation of specific risks and goals (Apitz and Power, 2002): (1) no action if it is determined that sediment poses no risk; (2) natural recovery monitoring, if the risk is low enough that can be reduced naturally by self-purification; (3) in situ containment, in which sediment contaminants are in some manner isolated from target organisms, though the sediments are left in place ; (4) in situ treatment; and (5) dredging or excavation (followed by ex-situ treatment, disposal, and/or reuse).

The most common and straightforward strategy is dredging, which physically removes contaminants sediment from aquatic systems. However, the dredging strategy is not advisable due to the several disadvantages like the high cost of removal treatment (Hakstege, 2007), remobilization of contaminants that are trapped in the sediments (Martins *et al*., 2012), environmental degradation (Nayar *et al*., 2004) and the potential long-term threat for exposure from some remain contamination. No removal technology can remove every particle of contaminated sediment, and post-dredging residual contamination levels have often failed to reach the desired levels (Martins *et al*., 2012). Although dredging remains a potential strategy for contaminated sediment management, new technologies are needed to develop economical and effective ways to treat sediment contamination.

Sediment capping technology using in-situ capping (ISC) is one development approach that places a layer of clean material over contaminated sediments that is less energy-intensive, cost-efficient, and less disruptive to the environment. The objectives of ISC are to isolate the sediments from the overlying water column and biota (Zhang *et al*., 2016), and to reduce the contaminant flux of the sediment (Reible *et al*., 2003). Two types of caps, namely passive and active capping, can be used over contaminated sediments. Passive caps are the conventional type of caps commonly employing clean material like sand, silt, clay, and crushed rock debris. These materials are easily available at relatively low cost, although they have low adsorption capacity due to their dependency on physical retardation mechanisms than on chemical retardation (Eek *et al*., 2008). The thickness of passive caps is approximately 50 cm (Azcue *et al*., 1998). Therefore, they are inefficient for use for contaminant removal.

Active caps use chemical reactive materials that sequestrate and or degrade sediment contaminants to reduce their mobility, toxicity, and bioavailability (Zhang *et al*., 2016). Different from passive caps, active caps use thinner materials. The 12 mm thickness of active materials can theoretically replace 1 m of passive caps such as sand or soil (Olsta, 2007). Active caps can also be applied in areas under diffusion and advection-dominated conditions, thus effectively isolating contaminants in sediment from a bioactive portion of the cap for decades to centuries (Murphy *et al*., 2006). The objectives of this paper are to provide insights into sediment capping technology, including several considerations in selecting capping materials as the most essential part of sediment capping technology, as well as information on several active materials that have been applied as capping materials and their efficiencies. This study also reveals how this technology can be applied in Indonesian lakes.

2. Materials and Methods

The methods used in the literature review were conducted as follows: (1) searching and selecting appropriate articles regarding sediment capping technology, including theoretical presentations, review articles, and empirical research articles. We explored Google Scholar (https://scholar.google.com) using keywords such as sediment capping and capping material for nutrient removal in eutrophic lakes; (2) analyzing and synthesizing the collection of articles by identifying the important information, integrating them and determining the conclusion that can be drawn from the articles as a group; (3) finding differences in the types of capping materials and their efficiencies in removing nutrient-contaminated sediments. We used Mendeley Desktop (https://www.mendeley.com/) as a tool to organize and annotate all the references.

3. Results and discussion

3.1 Design Considerations for In-Situ Capping of Contaminated Sediments

The guidelines for in-situ capping (ISC) were described by Palermo et al. (1998) which was prepared for the U.S. Environmental Protection Agency (USEPA) under the Assessment and Remediation of Contaminated Sediments (ARCS) Program, administered by USEPA's Great Lakes National Program Office. A recommended sequence of steps involved with the design of an ISC is illustrated in a flowchart in Figure 1. To achieve the remediation goals, a capping project must be treated according to the considered design, construction, and monitoring. Considerations in the design process are summarized as follows:

1. Determination of remediation objective
   Once the objectives are set, the scope of the remediation effort can be defined, usually in terms of the areal extent of contamination, contaminant concentration, or volume of material to be remediated. The objective of contaminated sediment remediation may be quite site-specific. ISC is feasible to reduce uptake or toxic effects from a contaminant. However, ISC would not meet an objective to destroy or remove some particular sediment from the aquatic environment.
2. Evaluation of site characterization
   Varying site conditions indicate that sediments are subject to varying biogeochemical processes. Capping performance will be different based on some factors, i.e., water depths, bathymetry, temperature, dissolved oxygen concentration, redox potential, wind energies, current and flow, stagnant or fast-moving water bodies (Zhang *et al.*, 2016), waterways use (water supply, recreation, navigation, and wastewater discharge), geotechnical conditions (stratification of underlying sediment layers, depth to bedrock, and potential for groundwater flow), diffusion and advection (Palermo, 1998).
3. Evaluation of contaminated sediment characteristics
   The physical, chemical, and biological characteristics of the sediments should be determined both horizontally and vertically to determine the areal extent or boundaries of the site to be capped. The characteristics of contaminated sediments are primarily influenced by site-specific conditions. For example, the nature and level of the contamination, the concentrations and bioavailability of those contaminants and their pathways into the aquatic environment and their fate in the lake system. Depending on the type of contaminant, parameters of interest may include organic carbon content, pH, dissolved oxygen, redox potential, ionic strength, and salinity to determine the potential of migration through the capping layer. The physical parameters should include the determination of particle size distribution, organic matter content, water content, plasticity (Atterberg limits), undrained shear strength, slope stability and bearing capacity. In terms of biological parameters, they were focused on bioturbation and ensuring that the capped sediment remains isolated from aquatic biota (EPA, 2012). Moreover, turbulent flow conditions associated with

seasonal flooding can expose anoxic sediment to toxic conditions that may result in significant changes to contaminant speciation and the flux of contaminants from sediments (Riedel *et al.*, 1999). Also, groundwater discharge will cause significant widespread continuous flow through the sediment and lead to the release of contaminants (Liu *et al.*, 2001).

4. Determination of preliminary feasibility
Following the remediation objective, site and sediment characteristics, a preliminary determination of the overall feasibility of ISC at the target site should be conducted. The cost and effort involved in long-term monitoring and potential management actions should be evaluated as part of the initial feasibility study.

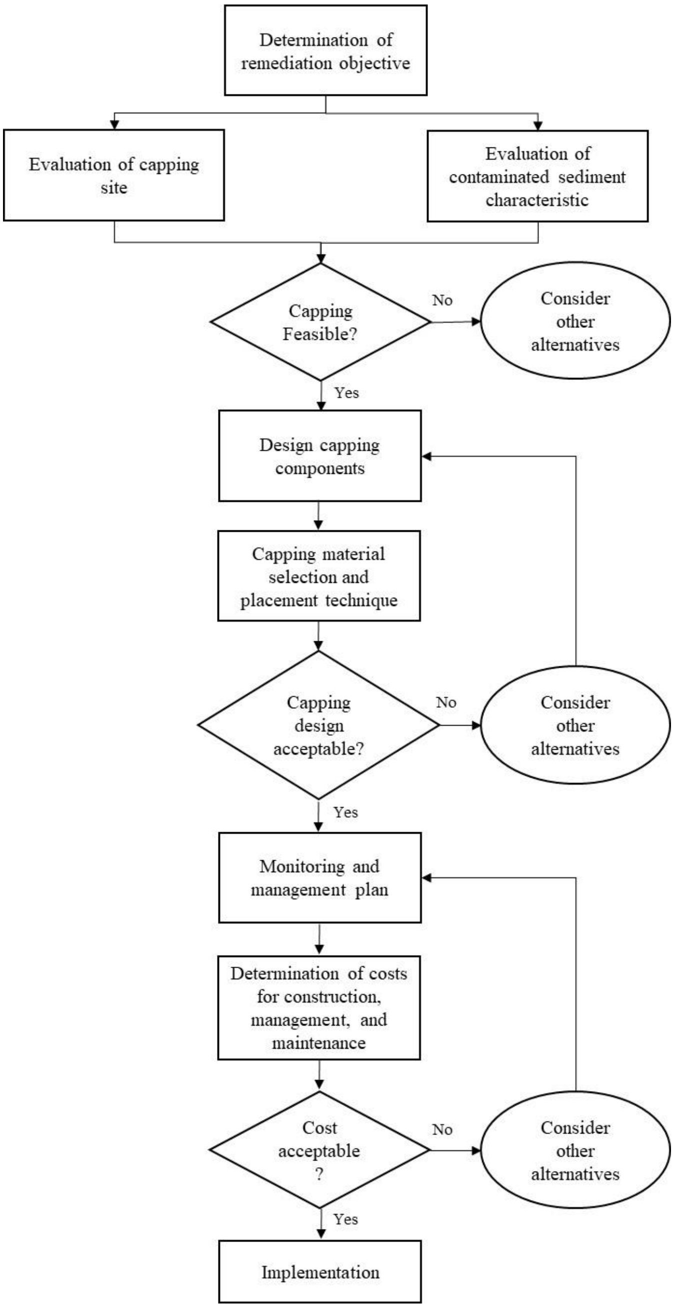

Fig.1 Flowchart showing the design sequence of an in-situ capping project (modified from Palermo, 1998)

5. Capping component design
   The composition and thickness of cap materials can be referred to as the cap design by considering physical isolation, sediment stabilization, and reduction of dissolved contaminant flux (EPA, 2012). The design must also be compatible with the available construction and placement techniques, consideration for effective short and long-term chemical isolation of contaminants, adsorption, bioturbation, consolidation, erosion, and other pertinent processes. The standard cap design for ISC is illustrated in Figure 2. The recent state-of-the-art cap designs involve a combination of laboratory experiments, knowledge of local species and their bioturbation behavior; wind forces circulation, analytical evaluations, hydrodynamic, sediment transport and erosion modeling (Palermo *et al.*, 1998), as well as advective and diffusive contaminant transport process modeling (Go *et al.*, 2009).

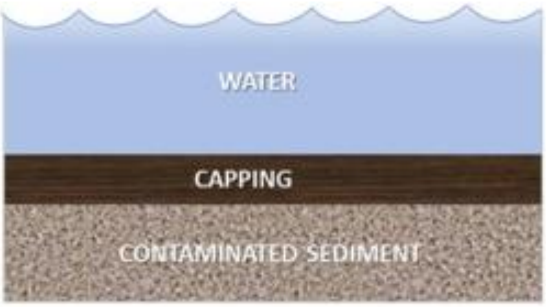

Figure 2. In-situ capping (ISC) design

6. Capping materials and placement technique
   The consideration for cap materials is the most important since these materials will generally represent the overall project cost. The selection among several potential cap materials must be determined by subsequent analysis using laboratory experiments. Most ISC projects have used sediment or soil materials, either dredged from nearby waterways or obtained from upland sources, including commercial quarries. Granular materials, i.e., sandy sediment or soil, should contain an organic fraction to act as an effective containment layer. Other materials, such as armor stone or geotextiles, should be considered in erosive environments (Palermo, 1998).
7. Monitoring and management plan
   When the capping design and materials have been accepted, then a monitoring program should be required to ensure that the cap is placed as intended and performing the basic functions (physical isolation, sediment stabilization and chemical isolation) as required to meet the remedial objectives. Specific parameters that may be monitored include cap thickness, cap consolidation, the need for cap nourishment, benthic recolonization, and chemical migration potential (Palermo, 1998). Furthermore, intensive monitoring is necessary at capping sites during and immediately after construction, followed by long-term monitoring at less frequent intervals.
8. Determination of costs for construction, management, and maintenance
   The important aspect that must be considered is the necessary costs for ISC, including material costs and long-term monitoring during ISC implementation. An economic study is required to consider the capping duration and the maintenance of materials.

3.2 Active Capping Materials

A summary of active capping materials for nutrient reduction applied in a number of previous studies is presented in Table 1. Apparently, their distinct characteristics depend on the type of material and adsorption capacity. Active materials play different roles in active capping technology, including target contaminant, capping duration, and their efficiencies in nutrient reduction (Zhang *et al.*, 2016).

Table 1. Several active capping materials

No.	Capping material	Contaminant	Capping duration	Finding	Application	Reference
1.	Calcite-zeolite mixtures	Phosphorus, ammonium	72 days	93 % reduction of the phosphorus fluxes and 99 % reduction of ammonium fluxes using batch and sediment incubation experiment	Sediment and water sample from a eutrophic, polluted small landscape waterbody in Shanghai, China	(Lin *et al.*, 2011)
2.	Rohrbach calcite	Phosphorus	70 - 230 days	80 % reduction of soluble reactive phosphorus flux using batch and sediment incubation experiment	Sediment and water sample from eutrophic Lake Epple and Lake Muggle, Germany	(Berg *et al.* 2004a)
3.	Manufactured calcite (U1)	Phosphorus	300 days	No phosphorus release in a 4.5 cm of U1 thickness using batch and sediment incubation experiment	Sediment and water sample from eutrophic Lake Epple and Lake Muggle, Germany	(Berg *et al.*, 2004)
4.	Calcite-modified Fe (FMCA)	Phosphorus	86 days	In batch experiment, FMCA show better adsorption process than unmodified-calcite, and the efficiency increase as well as Fe addition.	Sediment samples were collected from a eutrophic lake in Pudong, China	(Bai *et al.*, 2021)
5.	Calcite/Zeolite modified Fe	Nitrogen and phosphorus	135 days	77,8–99,7% of soluble phosphorus reduction and 54,0–96,7% of ammonium reduction using batch and microcosm incubation experiment	Sediments sample from a lake in Pudong New Area, Shanghai, China	(Zhan *et al.*, 2020)
6.	Fe-modified bentonite	Phosphate	90 days	68 % reduction of the phosphate flux from the sediment	Aitoliko Lagoon, Western Greece	(Zamparas *et al*, 2013)
7.	Bentonite humic-acid composite material (Bephos)	Phosphorus, ammonium	92 days	96.6% reduction of the phosphate flux and 75.2% reduction of the ammonium flux from the sediments	Aitoliko Lagoon, Western Greece	(Zamparas *et al.*, 2014)
8.	Bentonite clay and Bauxsol	Phosphorus	300 days	Bentonite clay effectively reduce phosphorus in oxic/anoxic condition (~82 %)	Lake Ainsworth, Australia	(Akhrust *et al.*, 2004)
9.	Bentonit, Illite, and Zeolite	Nitrogen and phosphorus	60 days	Illite showed the highest efficiency (90 %) in reducing phosphate and total phosphorus.	Highly eutrophic lake in Anseong City, Korea	(Gu *et al.*, 2019)
10.	Magnetite/bentonite modified fabric-wrapped zirconium (M-ZrFeBT)	Phosphorus	120 days	M-ZrFeBT can bind P with efficiency of 96.5–98.2%.	Eutrophic water body in Pudong New District, China	(Lin *et al.*,2020)
11.	Bentonite-modified zirconium (ZMBT)	Phosphorus	170 days	When the P concentration increased, ZMBT was able to prevent the released P with efficiency of 95 %	Shallow water body in Pudong District, China	(Zhan *et al.*, 2020)
12.	Zeolite-modified gypsum	Phosphorus	10 days	90 % of phosphorus release reduction using batch experiment	Artificial eutrophic water and sediment	(Yun *et al.*, 2007)

No.	Capping material	Contaminant	Capping duration	Finding	Application	Reference
13.	Zeolite, ceraicite and light porous media	Nitrogen	90 days	The highest efficiency of N reduction was performed by zeolite (90-100%), followed by ceraicite and light porous media (59 %)	Eutrophic lake in Xi'an, China	(Huang *et al.*, 2011)
14.	Zeolite, activated carbon and non-woven fabric mats	Nitrogen and phosphorus	60 days	Capping efficiency 94-98% for N and 74-79% for P	Eutrophic Lake in Anseong City, Korea	(Hong *et al.*, 2019)
15.	Dolomite and zeolite	Nitrogen and phosphorus	60 days	96-100 % prevent the release of N and P by considering the placement	Sediment and lake water samples from a highly eutrophic lake in Anseong City, Korea	(Alvarado *et al.* 2020)
16.	Zeolite-modified lanthanum (LMZ)	Phosphorus	20 days	LMZ as an inactivation agent to prevent P release from sediment (91 %)	Lake Taihu, China	(Li *et al.* 2019)
17.	Water clarifier sludge	Phosphorus, ammonium	60 days	The adsorption capacity of sludge sintered at 600 ^{o}C was 2.2 times higher than unsintered sludge (~80 %)	Mandai pond, a eutrophic pond in Osaka City, Japan	(Ichihara and Nishio 2013)
18.	Activated carbon and non-woven fabric mats	Nitrogen and phosphorus	210 days	The used of NFWM upper the capping material show more efficient to reduce nutrient (88–94%)	Sediment and lake water samples from lake in Anseong City, Korea	(Gu et *al.*,2017)
19.	Biochar	Ammonia-nitrogen	30 days	Reducing the ammonia in sediment up to 70.8 – 87.2 %.	Baiyangdian Lake, China	(Zhu *et al.*, 2019)
20.	Powdered-gypsum and granular gypsum	Phosphorus	45 days	Batch experiment show 80 % reduction of phosphorus for both powdered-gypsum and granular-gypsum	Eutrophic lake in Korea	(Kim *et al.*, 2007)[b]

3.3 Potential of sediment capping technology for Indonesian lakes

By considering the application of sediment capping technology using some materials in several lakes in other countries in Table 1, we summarized the positive and the negative impact of sediment capping technology as a scenario for eutrophication control. The positive impact of this technology includes good efficiency in reducing nutrients and preventing eutrophication; easy to apply by distributing uniformly over the surface of the waterbody or the area targeted for application; also, by knowing the duration of capping, the long-term monitoring during ISC implementation can be well-managed. Regarding the effect of sediment capping on the aquatic biota, several studies have proven that there is no lethal or sublethal toxicity produced by materials used such as activated carbon, apatite, zeolite, and organoclay (Özkundakci *et al.*, 2011; Paller and Knox, 2010; Rosen *et al.*, 2011). However, there was a change in feeding behavior and a decrease in growth rate using calcite and biopolymer materials for Rotifers, Cladocera and water insect species (Ghadouani *et al.*, 1998 and Galvez-Cloutier et al., 2012). The potential for toxicity to organoclays should not be overlooked due to their significant harmful effects on living organisms (Sarkar *et al.* 2013).

Furthermore, research conducted by Cho *et al.* (2009) observed no negative impact, while Cornelissen *et al.* (2011) and Jonker *et al.* (2009) reported the potential ecotoxicological minor impacts on benthic communities using activated carbon material. This is related to the characteristics of the sedimentary environment and the occurrence of physical or chemical changes in the capping material, such as changes in composition that depend on the type of activated carbon (raw or modified activated carbon) and particle size (75–300 µm) (Janssen and Beckingham 2013). Generally, sediment capping technology is an innovative proprietary water remediation technology with clear environmental benefits for healthy waterways to support economic, recreational and humanitarian well-being.

However, this technology has some negative impacts due to the limitations and undesirable effects of the technology. According to Public Service and Procurement Canada (Vallee, 2017), the primary disadvantage of sediment capping technology is that contaminants remain in place, resulting in an ongoing risk of contaminant loss, re-exposure, or disturbance of the contaminated sediment. Other limitations of using sediment capping as a remedial strategy as follows: (1) the risk of contaminant migration through diffusion and advection, particularly when contaminants easily transported through interstitial water and low association with sediment grain size; (2) the stability of a sediment cap can be disturbed by extreme weather events (such as storms, flooding and earthquakes); (3) local regulations may not allow capping in some areas; (4) long-term monitoring and maintenance of the cap is required. In addition, some temporary potential adverse effects include increased turbidity or suspended sediment within the water column, resuspension of contaminated sediments, and alteration of benthic habitat due to the placement of capping materials. To minimize the negative impacts, it is necessary to determine the most suitable and effective capping materials.

Sediment capping technology with various materials in Table 1 was applied in several lakes, including some batch experiments using water and sediment from the lakes. Those lakes have similarities with Indonesian lakes in terms of trophic state, except for surface area, depth and water volume. The trophic state of those lakes was eutrophic to hypereutrophic with the value of total nitrogen was > 750 µg/l, total phosphorus was > 30 µg/l, chlorophyll-a was > 5 mg/m^{3}, and Secchi depth was < 2.5 m according to trophic classification from Regulation of Ministry of Environment 28/2009. The trophic state was similar with several lakes in Indonesia that is eutrophic to hypereutrophic (Ministry of Environment Republic of Indonesia, 2014). Most of the lakes in Indonesia are experiencing environmental problems, water quality decline and eutrophication because of the enhancement of tourism, industry, agriculture/plantation, settlement/domestic and fish cultivation using floating net cages.

However, there has been no effective effort to restore the water quality up to this time, especially for eutrophication issue. Hence, sediment capping technology has the potential to be implemented for eutrophication control in Indonesian lakes, and it has been recommended in Yuniarti *et al*. (2021). It is necessary to carry out laboratory tests to assess the characteristics of water quality and internal loading of nutrients and to determine the most suitable capping material to reduce nutrients. In addition, it is necessary to consider the selection of the most effective, efficient, easy, inexpensive, and eco-friendly capping materials. The selection of capping material must consider the potential positive and negative effects before this technique is applied to more extensive field-scale studies.

4. Conclusion

Several types of active capping materials such as calcite, zeolite, bentonite, activated carbon, sludge, biochar, and gypsum can be used to reduce the release of nutrients from sediment with an efficiency of 54–99 % and capping duration of 10–300 days in some eutrophic lakes. Sediment capping technology showed a promising result for lake ecosystem restoration in other countries. Therefore, this technology has the potential to be applied in Indonesian eutrophic lakes as a strategy for eutrophication control and sustainable management of lake ecosystems by considering the selection of the most effective, efficient, easy, inexpensive, and eco-friendly capping materials.

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Author Contributions

AS conducted the investigation, formal analysis of the literature review and preparation of the manuscript, **PS** and **ABS** were involved in conceptualization as well as reviewed the manuscript. All the authors read and approved the final manuscript.

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Assessment of Flash Flood Vulnerability Index in a tropical watershed region: a case study in Ciliwung Hulu watershed, Indonesia

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Abstract: Flash floods, an unpredicted swift climatological disaster, frequently occur in Indonesia. However, there are limited vulnerability assessments, especially in urban and vital regions such as Bogor District. The study aims to assess the vulnerability index of Ciliwung Hulu Watersheds as one of the most susceptible areas in the district. Flash flood vulnerability index (FFVI) is selected to be calculated as the indicator. Data were obtained from the official government offices and processed using the FFVI formula referring to the work of Nasiri *et al.,* (2019) and Perka BNPB No. 2/ 2012 and then mapped using ArcGIS 10.3. The results and the maps show that the study area is categorized as highly to very highly vulnerable to flash flood disasters. The attained results help facilitate the governance interplay processes in building a more disaster-ready management plan and to construct a more resilient society.

Keywords: flash flood, Ciliwung Hulu, vulnerability index, watershed

1. Introduction

A flash flood is one of the most frequent disasters in Indonesia (Badan Nasional Penanggulangan Bencana/BNPB, 2018). The leading causes of the disaster in the country are high precipitation rates, steep topography, and the vast occurrence of barren land (Mahmood *et al.*, 2016). The flood is categorized as a climatological disaster that is unpredicted, swift, and severe; thus, the causalities level is usually significant (Hastanti and Miardini 2020; Rahman *et al.* 2016). Considering its massive impacts, an assessment of the vulnerability level of a location to the flood is imperative to be conducted. Vulnerability assessment is defined as the inability of a specific individual or community, and it can be used to mitigate the severity of flood causalities (Rijanta *et al.*, 2014).

Bogor, a district in West Java, Indonesia, is selected as the study site of our vulnerability assessment study. The district is an important supporting and satellite area for Jakarta, the capital of Indonesia, and is frequently accused as the flood sender to the capital city (Harsoyo, 2013). The district is known as one of the hot spots where flash flood frequently occurs due to its topography. The district typically has small upstream systems and is prone to experiencing landslides – the principal prerequisite to flash floods (BNPB, 2018).

Considering the importance of the district, the development of a flash flood disaster-ready is a necessity; thus, an assessment of flash floods in Bogor district is imperative. However, until recently, there have been only limited studies focusing on this sector in such cities in Indonesia as most vulnerability studies were conducted in major cities (cf. Azmiyati and Poernomo, 2019). To fill this gap, it is necessary to assess the flash flood vulnerability index as an input of disaster-ready management planning, where this study can generate a vital contribution (cf. Larsen *et al.*, 2001).

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In the district, we mainly focus on the area of Ciliwung Hulu Watershed, where a relatively recent colossal flash flood in the watershed area, especially in Gunung Mas, Tugu Selatan Village just happened. The flood had caused an emergency evacuation of 474 people and 134 households and destroyed their houses, bridges, and roads (Maulana, 2021). The repeated and the scale of the resulted damage make the area is suitable to be used as our case study (cf. Dewi and Abdi 2017).

2. Materials and Methods

2.1. Study site

Administratively, Ciliwung Hulu Watershed is an approximately 14-thousand-hectare areas, which comprises four sub-districts in Bogor District and Bogor Municipality (Ciawi, Cisarua, Sukaraja, Megamendung, and Bogor Timur) (Figure 1a). The watershed is dominated by dryland agricultural, dryland forest, and settlement areas covering about 47, 26, and 23 km^2 area, respectively, as presented by the land use and land cover map (LULC) provided by The Ministry of Environment and Forestry/MoEF (2020) (Figure 1b).

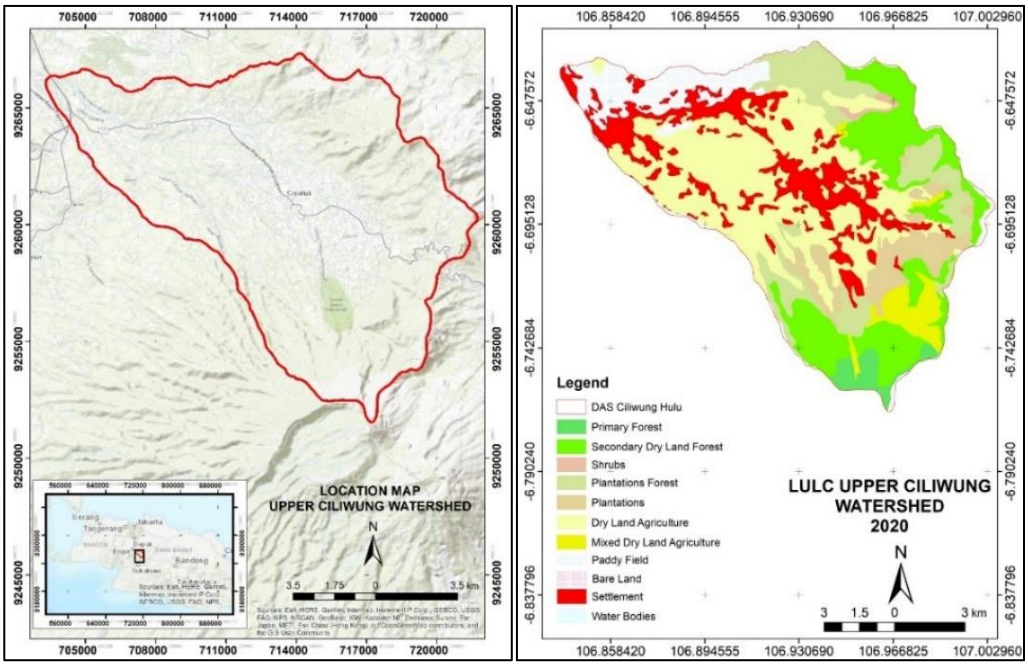

Figure 1. (a) Ciliwung Hulu Watershed; (b) Land use and land cover map of Ciliwung Hulu Watershed (MoEF, 2021)

The last flash flood that took place in 2021 impacted about 11 km^2 of agricultural area (BNPB, 2021). We could not obtain the economic impact of the 2021 flash flood, but as a proxy, a total of 11.2 billion IDR was estimated to be lost during the year due to repeated floods (ibid).

2.2. Data collection

We relied on the data published by the official websites of the sub-districts in Ciliwung Hulu Watershed, Bogor District, and West Java Provincial Government. We declare that there was no primary data collection was conducted to verify the obtained secondary data. Further, we also included the data which was extracted from the websites of the National Statistic Agency (Balai Pusat Statistik/BPS), the Provincial and District Disaster Prevention Agencies (Badan Penanggulangan Bencana/ BNPB), and MoEF. In addition, in case the data could not be obtained in the websites, a series of surveys to the sub-district and the district level government offices were conducted in June 2023.

To categorize the data, we refer to the methodology used by Hastanti and Miardini (2021) and Handoko *et al.* (2017). The data included: 1. The extent of the rice planting area and the average productivity; 2. The density of housing, public and emergency facilities; and 3. Housing market price.

2.3. Data analysis

The components of the index consist of social, economic, environmental, and physical vulnerability dimensions following the work of Nasiri *et al*. (2019), which was also used in the official document of BNPB, such as Perka BNPB No. 2/ 2012. *Social vulnerability* is defined as the level of openness of an individual or society to the social and environmental stressors that cause unpredicted disturbances in people's livelihoods (Adger, 1999).

The parameters used to estimate the social vulnerability index comprise population density, sex ratio, poverty, disability, and age group ratio (BNPB, 2012). The method to calculate each parameter and their definition of the ratios are elucidated in Table 1. Meanwhile, the formula to calculate the Social Vulnerability Index (SVI) (Equation 1) is defined as:

$$SVI = (0.6\ x\ population\ density) + (0.1\ x\ sex\ ratio) + (0.1\ x\ poverty\ ratio) + (0.1\ x\ disability\ ratio) + (0.1\ x\ vulnerability\ age\ group\ ratio) \text{ ....(Eq. 1)}$$

Whereas the Economic Vulnerability Index (EVI) is parameterized using the extent of the fertile land area (monetarized as 2021 Indonesian Rupiah/IDR) and the percentage of susceptible workers (Aisha *et al.*, 2019; BNPB, 2012). To identify the susceptible workers, we referred to the definition of susceptible work fields by Aisha (2019), which are farming, fishing, informal trade and service sectors, and daily workers.

The formulas to calculate the EVI (Equation 2) and the calculation of each parameter (Table 2) are:

$$\text{EVI}=(0.6 \times the\ monetary\ value\ of\ fertile\ land\ area\ ) + (0.4 \times the\ percentage\ of\ susceptible\ workers) \text{ ....(Eq. 2)}$$

Table 1. Parametrization of the Social Vulnerability Index (SVI)

Parameter	Definition	Weight (%)	Class range			Score
			Low	Medium	High	1: low, 2: medium; 3: high class range
Population density	The number of people who live in one square km area	60	<500 people/km^2	500 – 1000 people/km^2	>1000 people/km^2	
Sex ratio	The number of men per 100 women	10				
Poverty ratio	The percentage of people who live below the marginal poverty line in Bogor District	10				
Disability ratio	The number of disable people divided by the numbers of population in each sub-district	10	<20%	20 – 40%	>40%	Class/maximum score class
Vulnerable age group ratio	The number of people categorized in 0–14-year-old age group and more than 65-year-old divided by the number of populations in each sub district	10				

Source: (BNPB, 2012)

Table 2. Parametrization of the Economic Vulnerability Index (EVI)

Parameter	Weight (%)	Class range			Score (1: low, 2: medium; 3: high class range
		Low	Medium	High	
Fertile land	60	<50 million IDR	50 – 200 million IDR	>200 million IDR	Class/maximum score class
Susceptible workers	40	<20%	20 – 40%	>40%	

Source: (Aisha *et al.*, 2019; BNPB, 2012; Widyantoro & Usman, 2021)

The Physical Vulnerability Index (PVI) is a composite index consisting of housing density (permanent, semi-permanent, and non-permanent houses), the availability of public facilities, and the occurrence of emergency facilities (BNPB, 2012). Housing density is the result of the division of the number of houses and the extent of the area (e.g., villages). The result is then converted to the housing market price (Table 3). The formula to estimate the PVI (Equation 3) is written as:

$$\boldsymbol{PVI} = (0.4 \times \textit{the monetary vvalue of housing density}) + (0.3 \times \textit{the monetary value of public facilities}) + (0.3 \times \textit{the monetary value of emergency facilities})$$ ....(Eq. 3)

Table 3. Parametrization of the Physical Vulnerability Index (PVI)

Parameter	Weight (%)	Class range			Score (1: low, 2: medium; 3: high class range
		Low (million IDR)	Medium (million IDR)	High (million IDR)	
Housing density	40	<400	400 – 800	>800	Class/maximum score class
Public facilities	30	<500	500 – 1,000	>1,000	
Emergency facilities	30	<500	500 – 1,000	>1,000	

Source: (BNPB, 2012; Hastani & Miardini, 2021)

The Environmental Vulnerability Index (ENVI) includes the extent of land coverage by protected forests, natural forests, mangroves, bushes, and swamp areas (Table 4). The ENVI is calculated based on Equation 4 below:

$$ENVI = (0.3 \times protected\ forest) + (0.3 \times natural\ forest) + (0.3 \times Mangrove) + (0.1 \times bushes) + (0.2 \times swamps)$$ ....Eq. 4)

The Flash Flood Vulnerability Index value (FFVI), a composite index of SVI, EVI, PVI, and ENVI, is generated using the Analytic Hierarchy Process (AHP) by combining the indices mentioned above with their weight (BNPB, 2012) (Equation 5). The calculated FFVI is then used to categorize the level of vulnerability as revealed in Table 5.

$$FFVI = (0.4 \times SVI) + (0.25 \times PVI) + (0.25 \times EVI) + (0.1 \times ENVI)$$ ....(Eq. 5)

Table 4. Parametrization of environmental vulnerability index (ENVI)

Parameter	Weight (%)	Class range			Score 1: low, 2: medium; 3: high class range
		Low (ha)	Medium (ha)	High (Ha)	
Mangrove	30	<20	20 – 50	>50	Class/maximum score class
Natural forest	30	<25	25 – 75	>75	
Mangrove	10	<10	10 – 30	>30	
Bushes	10	<10	10 – 30	>30	
Swamps	20	<5	5 – 20	>20	

Source: (BNPB, 2012; Widyantoro & Usman, 2021)

Table 5. The categorization of the Flash Flood Vulnerability Index (FFVI)

Flash flood vulnerability index	Vulnerability level
0 – 0.6	Very low
0.61 – 1.20	Low
1.21 – 1.80	Medium
1.81 – 2.40	High
2.41 – 3.00	Very high

Source: Authors' creation based on the level of vulnerability categorization in Widyantoro & Usman (2021), Aisha *et al.*, (2019); Wahyuni (2015); Hastanti & Miardini (2021); and BNPB (2012)

3. Results and Discussion

The calculated SVI (Table 6) shows that the five sub-districts are included in the very high vulnerability level. The extremely high population density generates a 60% contribution to the SVI. The result indicates that the sub-districts are highly susceptible to environmental hazards (Das *et al.*, 2020; Armaş & Gavriş, 2016).

The second most influencing parameter to SVI is the sex ratio. The calculated ratio reveals that there are more men than women in the study area; thus, the vulnerability becomes lower since women generally require more time to resonate from the disaster impacts. This situation happens because, in general, women have higher pressures in child caring and bearing, and they receive lower income than men do (Viet Nguyen, 2015; Armaş & Gavriş, 2013).

Table 6. The calculated SVI

No	Sub-district	Population density		Sex ratio		Poverty ratio		Disability ratio		Vulnerable age group ratio		SVI class	
		Value	Score	Value	Score	Value	Score	Value	Score	Value	Score	Value	Class
1	Ciawi	1,481.06	3	106.7	1	7.69	1	0.03	1	36.92	2	2.30	High
2	Cisarua	2,700.11	3	108.2	1	7.69	1	0.02	1	33.40	2	2.30	High
3	Mega-mendung	1,448.36	3	110.0	1	7.69	1	0.02	1	37.09	2	2.30	High
4	Sukaraja	33,04.94	3	104.5	1	7.69	1	0.01	1	34.07	2	2.30	High
5	Bogor Timur	10,278.52	3	102.9	1	6.68	1	0.19	1	31.49	2	2.30	High

Meanwhile, the assessed EVI (Table 7) also elucidates that most of the study area is grouped into the high-vulnerability category except for Cisarua sub-district. The sub-district is categorized as a very high vulnerability condition.

The monetary value of the extent of fertile land area, the most influencing parameter, supports 60% of the EVI, which indicates that the decline or disappearance of fertile land will severely affect people's livelihoods. In all sub-districts, fertile land is categorized in the high-class range (score 3), which shows critical vulnerable conditions. On the contrary, the vulnerable worker ratio is categorized as low for all sub-districts except for Cisarua. The score of the ratio in the sub-district is included in the high level (Table 7).

At the same time, we found a more interesting finding about the calculated PVI (Table 8). The results show that all sub-districts are highly physically vulnerable. The very dense housing likely becomes the main factor behind the condition, as hinted by Aisha *et al.* (2019), who found that the level of causalities increases with the increment of housing density.

Meanwhile, the assessed ENVI reveals the different results (Table 9). Based on the ENVI, the environmental susceptibility of the study area is categorized as low level for Sukaraja and Bogor Timur and medium level for Ciawi, Cisarua, and Megamendung. The occurrence of protected forest areas in these last three districts (which is categorized in the medium class) becomes the principal factor explaining their relatively higher vulnerability compared to the first two sub-districts.

The results imply that the existence of protected forest areas in Ciawi, Cisarua, and Megamendung does not reduce the environmental susceptibility of the area, as hinted by Hastanti and Miardini (2021). The reason for this circumstance is that the calculation of the index is based on the conversion of the extent of the area, including forested area, into monetary value; therefore, the areas with a more considerable extent of forest area may suffer more significant economic loss. Nevertheless, reducing forest area is not a solution to reduce environmental susceptibility (ibid).

Table 7. The calculated EVI

No.	Sub-district	The extent of paddy field area (Ha)	The valuation of extent of fertile land area (Million IDR)	Score	The percentage of vulnerable workers (%)	Score	EVI class	
							Total score	
1	Ciawi	704	3,949.44	3	12.52	1	2.2	High
2	Cisarua	198	2,107.82	3	28.39	3	3.0	Very high
3	Megamendung	274	1,461.35	3	18.25	1	2.2	High
4	Sukaraja	80	108.65	3	7.53	1	2.2	High
5	Bogor Timur	57	765.99	3	6.94	1	2.2	High

Table 8. The calculated PVI

No.	Sub-district	Housing density		Public facilities			Emergency facilities			PVI class	
		House price (million IDR)	Score	Numbers	Price (million IDR)	Score	Numbers	price (million IDR)	Score	Total Skor	Kelas
1	Ciawi	1,035.53	3	232	46,400	3	13	3,250	3	3	Very high
2	Cisarua	1,939.96	3	228	45,600	3	15	3,750	3	3	Very high
3	Mega-mendung	1,153.81	3	234	46,800	3	11	2,750	3	3	Very high
4	Sukaraja	2,658.15	3	266	53,200	3	12	3,000	3	3	Very high
5	Bogor Timur	4,678.62	3	137	27,400	3	14	3,500	3	3	Very high

Table 9. The calculated ENVI

No.	Sub-district	Protected forest		Natural forest		Mangrove		Bushes		Swamps		ENVI	
		Area (Ha)	Score	Area (Ha)	Score	Area (Ha)	Score	Area (Ha)	Score	Area (Ha)	Score	Total score	Class
1	Ciawi	864.05	3	0	1	0	1	0	1	0	1	1.60	Medium
2	Cisarua	1,268.66	3	5.93	1	0	1	0	1	0	1	1.60	Medium
3	Mega-mendung	184.55	3	1.26	1	0	1	2.66	1	0	1	1.60	Medium
4	Sukaraja	0	1	0	1	0	1	0	1	0	1	1.00	Low
5	Bogor-Timur	0	1	0	1	0	1	0	1	0	1	1.00	Low

Overall, the mapping of the results of the calculation of the SVI, EVI, PVI, and ENVI shows that in almost all the sub-districts have low (the green area in Figure 2d) to very high vulnerability (the red area in Figure 2a-2d) to flash flood disasters depending on the calculated index. However, the calculation of the FFVI (Table 10 and Figure 3) indicates that all sub-districts are highly vulnerable to flash floods (the yellow area in Figure 3) except for the Cisarua Sub-district that has very high vulnerability (the red area in Figure 3).

The map in the figure illustrates the zonation of the flash flood vulnerability index within a 150-meter distance from the river. This distance, a result of an overlay between the flood hazard index map (BNPB, 2016) and the most recent flash flood events in the research area, is a particular area that should be cautioned during the flash flood. However, there is a possibility that other areas which are not mapped can be at a greater risk. Hence, a comprehensive assessment that considers potential high-risk zones beyond the mapped areas is a necessity for future assessment.

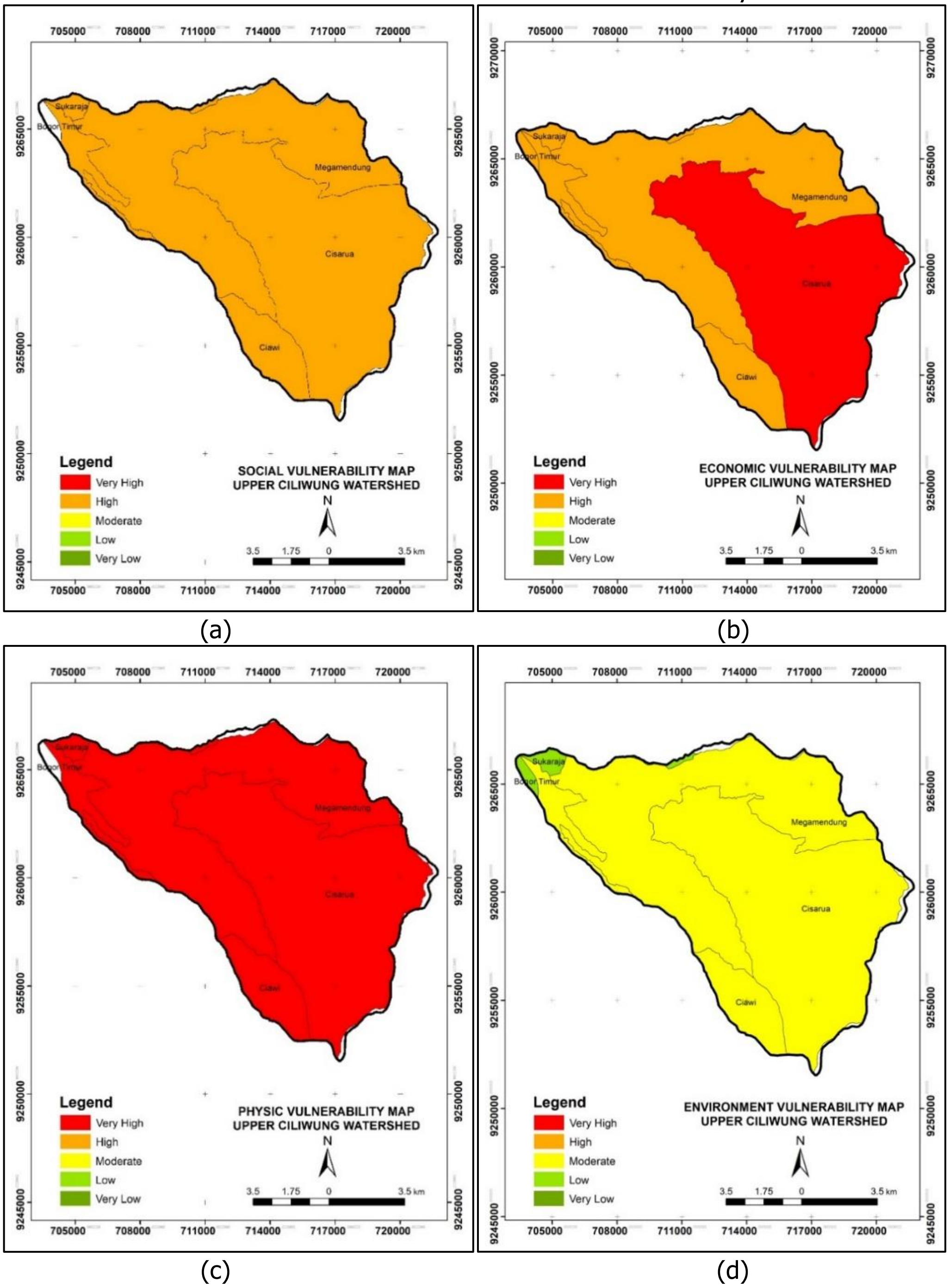

Figure 2. The mapping of (a) SVI; (b) EVI; (c) PVI; (d) ENVI

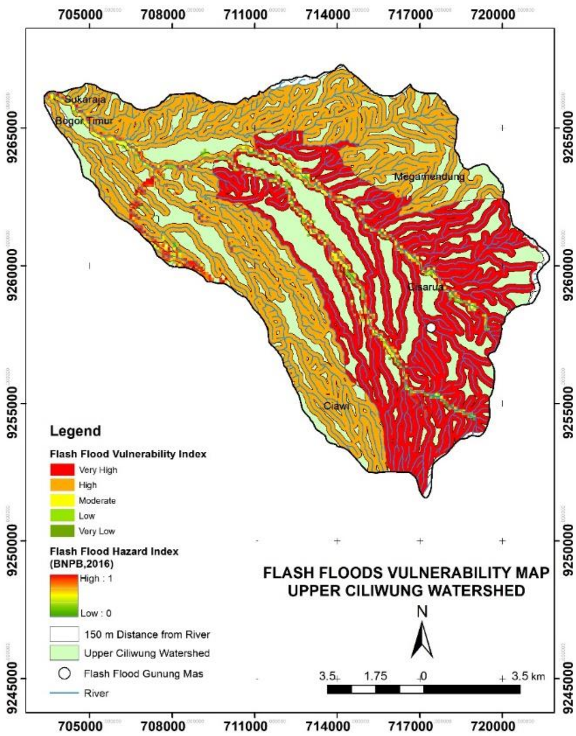

Figure 3. The map of the FFVI

Table 10. The calculated FFVI

No.	Sub-district	The flash flood vulnerability index	
		Score	Class
1	Ciawi	2.38	High
2	Cisarua	2.58	Very high
3	Megamendung	2.38	High
4	Sukaraja	2.32	High
5	Bogor Timur	2.32	High

The assessed FFVI and the map are crucial for prioritizing intervention management to effectively reduce and manage the risks of flash floods in the study area. The map can be used to establish a robust early warning system and to aid the development of evacuation routes, shelters, and community awareness programs. Furthermore, it is also helpful for the establishment of a post-flash-flood recovery plan towards a more resilient community.

4. Conclusion

The Ciliwung Hulu Watershed area is highly susceptible to the occurrence of flash flood disasters. Our results provide essential data for the government to plan disaster-ready management planning as well as raise the resident's awareness of the hazards. However, this is only the early step in the development of a flash flood resilience society. To aid further effort, we suggest that future research include the assessment of the mapping of vulnerability index in the larger areas. Further, to fully develop the disaster-ready management plan, the establishment of more coordinated cooperation between local, regional, and national authorities is essential. This is a process that requires interplaying governance processes aided by this study.

Data availability statement
We state that the source of all required data has been written in the manuscript. The secondary data can be found in the mentioned sources.

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RN and **FAW** (the principal contributors): data collection, analysis, illustration, writing the original draft, and revision. **EP** and **EGAS** (the supporting contributors): data collection

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