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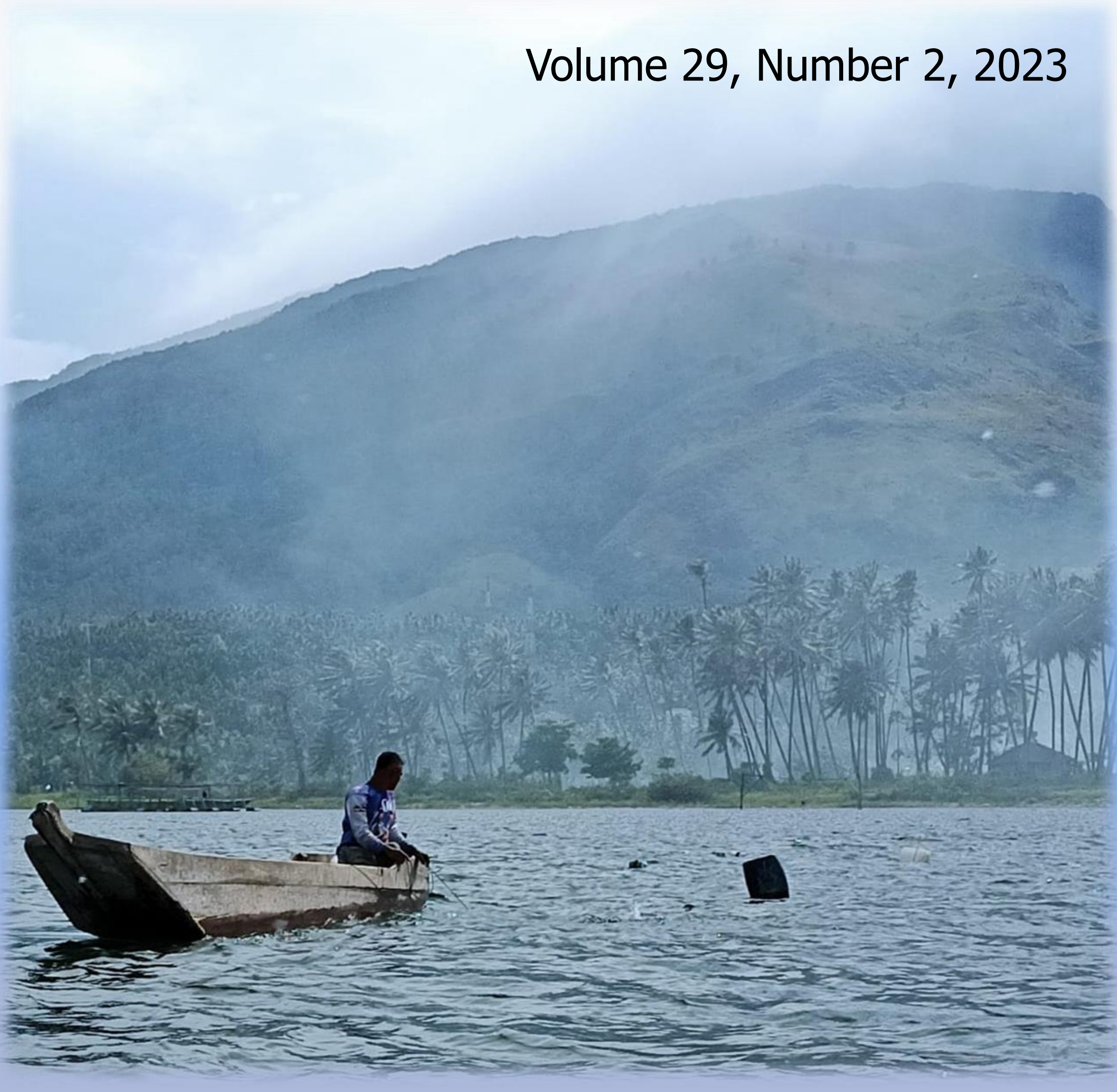

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Navigating Challenges in Sustainable Water Management

In this 2023 second edition, our journal delves further into challenges and opportunities in sustainable water and aquatic ecosystem management. The declared "decade of water" emphasizes the urgency of addressing global water issues, and this journal strives to answer this challenge.

We are grateful to present our refined publication this year, which features critical topics, such as catchment-scale erosion and the dynamics of hydro-chemical, pesticide accumulation on fish and sediment, floodplain lake biodiversity, and GHG emission mitigation in Asia.

This edition reflects the dedication of our contributors as well as our commitment to advancing water sciences, fostering global recognition for Limnotek *transforming into the* Journal Limnology and Water Resources, and serving as a communication platform. This small step is a part of our broader commitment to foster sustainable water and aquatic ecosystem management.

In the spirit of continual learning, living, and growing, we invite our readers to join us on this transformative journey. As we navigate the waters of global challenges, we remain steadfast in the pursuit of knowledge and solutions that contribute to a sustainable and harmonious coexistence with our aquatic environments. Together, let us strive for a future where clean water is not just a necessity but a shared responsibility and a testament to the resilience of the global scientific community.

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Assessment of soil loss using RUSLE method in Mrica Reservoir catchment, Central Java, Indonesia

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Abstract: The Indonesian government has identified the Serayu Watershed as a priority area for restoration within the National Mid-Term Development Plan. One of the significant challenges in this region is the high level of soil erosion, which threatens the overall ecosystem. This study aims to estimate the amount of soil loss in the Mrica Catchment using the Revised Universal Soil Loss Equation (RUSLE) Method. Various data sources were utilized, including soil type, rainfall, land cover, Digital Elevation Model, and conservation data. Geographic Information System (GIS) techniques were employed to calculate the critical factors required by the RUSLE Method, including soil erodibility (K), rainfall erosivity (Ri), slope length and steepness factor (LS), and cover management and conservation factor (CP). This research provides critical information for land management in Mrica Catchment. These factors were used to estimate soil loss in the Mrica Catchment, revealing a range between 62,553 tons per year (t/y) and 21,323,311 t/y, with an average value of 443.90 ton per hectare per year (t/ha/y). These results indicate high erosion potential based on the Classification of Erosion Hazard (HER). This study provides critical information for land management and offers suggestions for devising effective strategies to mitigate sedimentation impact in the Serayu Watershed. The highest soil loss values according to the RUSLE Method, both under the Environmental and Management Variable, are observed in the same location, namely, in the north of Mrica Catchment. The findings emphasize the urgent need for erosion control measures and sustainable land management practices in this priority restoration area.

Keywords: Mrica reservoir, RUSLE, soil loss

1. Introduction

Erosion and sedimentation are two significant problems that are influenced by hydrodynamic activity and sediment transport, which brings many hydrological changes to the watershed (Novico and Priohandono, 2012). Several factors affect the increase in erosion rates and accelerate its rate, including the increased population, anthropogenic needs, climate change, and the intensity of economic activity (Abu Hammad, 2011; Pambudi *et al.*, 2021). Soil erosion may be to blame for up to 80% of the degradation issue on agricultural land (Abu Hammad, 2011; Angima *et al.*, 2003). An increase in soil erosion has a linear relation with the increase in sedimentation, resulting in reservoir siltation (Abdul Rahaman

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et al., 2015). Reservoir sedimentation impacts the life of the reservoir, which can decrease by more than 65% and affect hydroelectric power purposes (Abdul Rahaman *et al*., 2015; Cantik *et al*., 2021; Chen and Tsai, 2017).

The negative impacts of erosion and sedimentation have been a concern of the Indonesian government. In 2009, the Indonesian government identified 108 critical watersheds in Indonesia to restore the condition of these watersheds (BAPPENAS, 2015). In line with the National Medium-Term Development Plan (RPJMN), of the 108 critical watersheds that have been observed, the Government of Indonesia will first prioritize 15 priority watersheds for restoration, one of which is the Serayu watershed (BAPPENAS, 2015). Soil and water conservation measures and efforts are needed to deal with this problem after analyzing the erosion hazard level of the Serayu watershed.

The Serayu River is one of the primary rivers that contribute significantly to the sedimentation of the Mrica Reservoir, a multipurpose reservoir. The Serayu River's route is heavily used for communities, agriculture, and other human activities, making it highly prone to its occurrence. Because of human activity, it is incredibly prone to significant contamination. One of the problems in the Serayu watershed that causes sedimentation in the Mrica Reservoir is damage in the upstream area, where there has been land degradation and an increase in erosion (Ainun Jariyah and Budi Pramono, 2013).

Problems in the Serayu Watershed are also triggered by land use. The existence of farming methods that ignore land conservation can also cause significant problems by increasing the rate of soil erosion (Eisenberg and Muvundja, 2020; Lesmana, 2020). In addition to land use and conservation issues, the catchment area of Mrica Reservoir Water is also a highland area with relatively high rainfall of roughly 4,000–4,500 mm/year, which might worsen erosion rates (Lembaga Kerjasama Fakultas Teknik UGM, 2015). This erosion problem caused a significant sedimentation increase in Mrica Reservoir. At the end of 2004, the remaining volume of the Mrica Reservoir was 78.05 million m^3 or around 55.26% of the initial condition (Cantik *et al*., 2021).

The RUSLE method calculates soil loss in places with significant surface flow but is not intended for areas without surface flow, while the MUSLE Method working principle is different from USLE. The erosion calculation produced by USLE Method is based on rainfall. MUSLE Method does not consider rainfall a source of erosion energy but uses runoff intensity to simulate the processes of erosion and sediment formation(Koirala *et al*., 2019). A comparison of erosion rate predictions utilizing the USLE, MUSLE, and RUSLE Methodologies reveals that the RUSLE Method produces more consistent results compared to all methods (Hanafi and Pamungkas, 2021). Using the RUSLE Method approach to assess erosion rates demonstrates that the RUSLE Method can provide effective erosion control strategies in severely damaged parts of the Serayu watershed (Lesmana, S.B., 2020). Therefore, the objective of this study is to predict the amount of soil loss that occurs in the Mrica Catchment using the RUSLE Method. The estimated soil loss can be used as a reference in considering effective and efficient efforts to be carried out to control reservoir sedimentation in Mrica Reservoir.

2. Materials and Methods

2.1. Study area

The research area of this study is Mrica Reservoir Catchment, which is located in the Southern Section of Central Java Province and covers an area of 98,703 Ha, with 54.36% of the land in Wonosobo Regency and 45.64% in Banjarnegara Regency. Three main rivers that play a significant role in contributing sediment to Mrica Reservoir are the Serayu River, Merawu River, and Lumayang River. The Mrica Reservoir is geographically located at coordinates 07°05′-07°4′N Latitude and 108°56'-110°05' E Longitude. The Mrica Reservoir is represented in Figure 1.

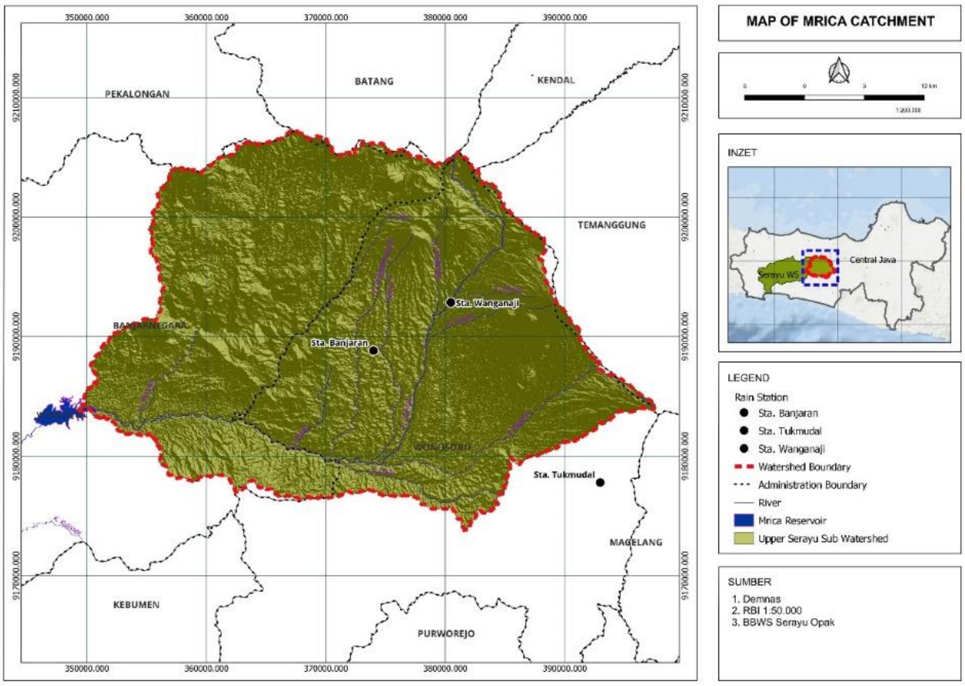

Figure 1. Mrica reservoir catchment area, the Upstream Serayu Watershed

2.2. Data collection

This study relied on secondary data and did not include any soil samples from the site. The secondary data used in this study are monthly and annual precipitation data from 2012 to 2016 at three rainfall stations, namely Banjaran, Tukmudal, and Wanganaji Rainfall Stations, with coordinates as shown in Table 1. We used Digital Elevation Model (DEM) data with 8 meters spatial resolution (Badan Informasi Geospasial, n.d.) obtained from DEMNAS. The DEM data was used to derive slope and slope length data, and it will also be used to delineate the watershed boundary. Soil type and land cover data were derived from vector data by Balai Besar Wilayah Sungai Serayu Opak dan Kementerian Lingkungan Hidup dan Kehutanan. The land management and conservation data (CP) were determined based on Chay Asdak, 1995 (Saputra *et al*., 2020). All these data will be processed using QGIS as Geographic Information System (GIS) software to calculate the soil loss using the RUSLE Method. Vector data is processed using GIS to be converted into raster data, and the scale is adjusted according to DEM spatial resolution.

Table 1. Coordinate of Rainfall Stations

Station	Coordinate	
	Latitude	Longitude
Banjaran	-7.3911	109.963
Tukmudal	-7.3282	109.930
Wanganaji	-7.2917	109.925

2.3 Soil Loss Analysis

In this study, the RUSLE Method estimates soil loss in the Serayu Watershed. The RUSLE, or Revised Universal Soil Loss Equation, is an erosion prediction method used to estimate the annual soil loss carried by water runoff from a particular land slope, considering a specific cropping and management strategy within a defined area. Its extensive utilization has underscored the effectiveness and credibility of the RUSLE method for achieving erosion-related objectives (Christanto *et al*., 2018). Equation 1, representing the RUSLE Method, is employed to compute soil loss (Renard, 1997).

$$A = R \times K \times LS \times C \times P \quad ....(1)$$

where A represents the annual average soil loss, expressed in metric tons per hectare per year (t/ha/y), LS denotes the factor that accounts for slope length and steepness, R stands for the rainfall erosivity factor, measured in megajoules per millimeter per hectare per hour per year (MJ mm/ha/h/y), P represents the land conservation factor, K signifies the soil erodibility factor, quantified in metric tons per hectare per hour per hectare per megajoule per millimeter (t/ha/h/ha/MJ/mm) and C represents the land cover management factor (Abdul Rahaman *et al.*, 2015).

To determine the value of R (erosivity) from the RUSLE Method, a formula devised by Wischmeier and Smith (1978) and then updated by Arnoldus (1980) (Panagos *et al.*, 2015) can be utilized (Eisenberg and Muvundja, 2020). Equation 2 shows the erosivity formula.

$$R = \sum_{i=1}^{12} 1.735 \times 10\left(1.5 \log_{10}\left(\frac{P_i^2}{P}\right) - 0.08188\right) \quad ....(2)$$

where the rainfall erosivity factor (MJ mm/ha/h/y) is represented by R, the annual rainfall (mm) is represented by P, and Pi represents the monthly rainfall (mm). Meanwhile, the soil erodibility factor (K) represents the ability of soil particles to weather and move due to precipitation's kinetic energy. Several physical and chemical features of the soil determine the ease with which soil erodes. These erodibility factors are indices that are used to forecast long-term average soil loss due to sheet and rill erosion under agricultural systems and conservation strategies. The main soil type at a location determines soil erodibility values.

Eq. 3 may be used to implement the LS factor (Bizuwerk *et al.*, 2003). In which FA indicates flow accumulation obtained from the GIS hydrology analysis tool, CS is cell size, m is 0.5 for slope angle >5%, 0.4 for slope 3%-5%, 0.3 for slope 1%-3%, and 0.2 for slope <1%, and s is the slope in percentage.

$$LS = \mathrm{Pow}\left(\frac{FA \times CS}{22.13}\right)^m \times (0.065 + 0.045s + 0.0065s^2) \quad ....(3)$$

GIS software is used to support the calculations and map renderings in this study. The results were processed in GIS using Inverse Distance Weighted (IDW) analyst tools to interpolate values. Geostatistical estimation and extensive calculations will be used, considering the statistical and spatial heterogeneity of the data. For rainfall erosivity factors, by examining point samples to create a continuous surface of erosivity, it studied rainfall erosivity variables throughout spatial space. This surface of erosivity is characterized in terms of both spatial continuity and a model of how it can vary.

There are two (2) components in the formulation of the RUSLE Method in this study, namely Environmental Variables, which consider that all variables in the calculation are constant, and Management Variables, which assume that factors C and P are two factors which can be seen in Table 2.

Table 2. Land management and conservation (CP) (Asdak, 1995 in (Saputra *et al.*, 2020))

No	Land Use	CP Value
1	Scrubland	0.30
2	Secondary Dryland Forest	0.01
3	Forestry plantation	0.05
4	Habitation	0.95
5	Plantation	0.50
6	Dryland farming	0.28
7	Dryland farming (mixed)	0.19
8	Field	0.01
9	Open space	0.95

3. Results and discussion

3.1 Rainfall Erosivity

Calculation of rainfall erosivity in Mrica Catchment using Equation 1 by calculating monthly and annual rainfall data from three stations as shown in Table 1. The rainfall data used is from 2012 to 2016. According to the result of this study, the rainfall erosivity factor in the Mrica Catchment ranges from 317.25 to 360.94 (MJ mm/ha/h/y).

3.2 Soil Erodibility

Soil or geological characteristics can affect the soil erodibility factor, i.e., porosity, parent material, texture, structure, etc. (Schwab *et al.*, 1993). The Mrica Catchment contains various soil types, including grumosol, regosol, alluvial, and latosol. The soil erodibility factor for each soil type is the smallest at 0.115 and the largest at 0.259, as shown in Table 3.

Table 3 Soil Erodibility

Soil Type	Area (Ha)	Soil Erodibility (K)
Alluvial	924.76	0.259
Grumusol	5,770.66	0.176
Latosol	88,878.43	0.115
Regosol	3,337.73	0.590

The largest soil type in the Serayu Subwatershed is latosol, with an area of 88.878.43 Ha and a K value of 0.115, followed by grumosol, with an area of 5770.6 Ha and a K value of 0.176, and regosol, with an area of 3337.3 Ha and the same K value as latosol, which is 0.115. In contrast, the smallest soil type is alluvial, with an area of 924.76 Ha and a K value of 0.259. In this study, the soil erodibility factor that has been determined becomes input data for GIS to obtain a soil erodibility map that can be seen in Figure 4.

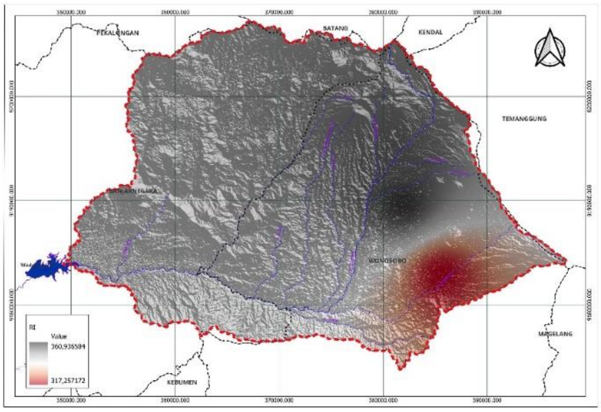

Figure 2. Annual Rainfall Erosivity

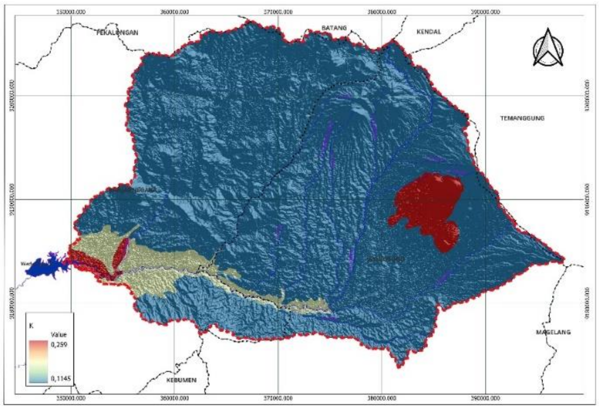

Figure 4. Soil Erodibility

3.3 Slope Length and Steepness Factor

The variable slope length and steepness factor (LS) is used to calculate how much soil is lost in the watershed, implying that the LS factor controls the volume of soil lost. According to Eq. 1, if the LS value is greater, the soil loss in the watershed will also be greater, and vice versa. Elevation data from the Mrica Catchment is required to calculate the LS factor.

From calculations on GIS, it was found that the value of the LS factor varied between 0-54. The LS map is processed using GIS as shown in Figure 5 below.

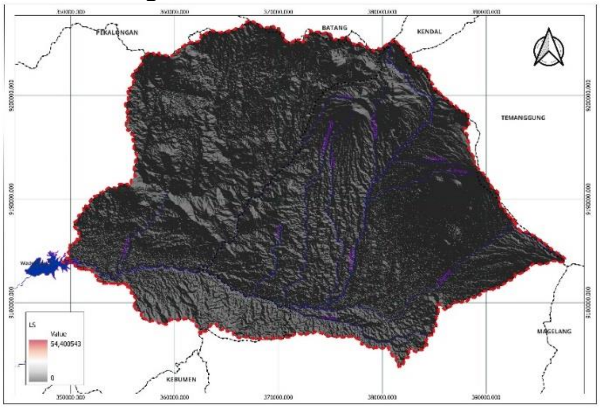

Figure 5. Steepness Factor and Slope Length

3.4 Conservation Factor and Cover Management

The factor values of conservation practice (P) and cover management (C) also determine the amount of soil loss in a watershed. After topography, land cover is the second most crucial factor determining soil loss since it can intercept rainfall and increase infiltration (Koirala *et al.*, 2019).

A higher C value indicates that the watershed is susceptible to significant soil erosion (Abdul Rahaman *et al.*, 2015). The P factor is reduced by employing conservation strategies that minimize runoff volume and velocity (Panagos *et al.*, 2015). The P factor is between 0 and 1, with a P value around 0 showing high-quality conservation practices (Morgan *et al.*, 1998). Soil and Water Conservation (SWC) practices must be developed and verified to establish the most acceptable SWC strategies suited for land cover (Tian *et al.*, 2021).

The data for land conservation and land cover in Mrica Catchment was obtained from the Ministry of Environment and Forestry. Table 4 shows the land cover in the Mrica Catchment. According to Table 4, the land cover with the most significant percentage is dryland farming, which is 39.3% of all total area in Mrica Catchment. The cover management and conservation factor (CP) varies between 0.01 to

0.95. It is then processed with GIS to produce the CP map shown in Figure 6.

Table 4. Land Cover

Classification	Area (Ha)
Shrubs	532.76
Secondary dryland forest	3,188.94
Forestry plantation	10,925.50
Habitation	3,766.31
Plantation	375.81
Dryland farming	38,823.22
Dryland farming (mixed)	34,251.73
Field	6,428.70
Open space	410.16
Total	**98,703.13**

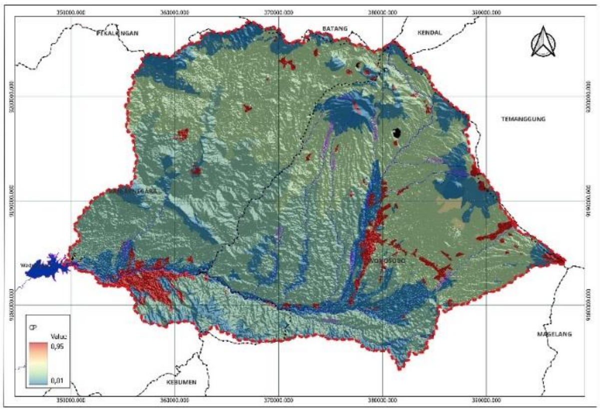

Figure 6. Cover Management and Conservation Factor

3.5 RUSLE's Soil Loss

The outcomes of soil loss will be classified as environmental variables and management variables. Environmental Variable is a map that shows soil loss taking into account the CP factor, while Management Variable is a map that takes into account the CP factor, which is a factor that can change at any time and cannot be used as a benchmark in calculating soil loss, so the CP value is ignored. Considering the CP (Environmental Variable) value, the soil loss map is presented in Figure 7.

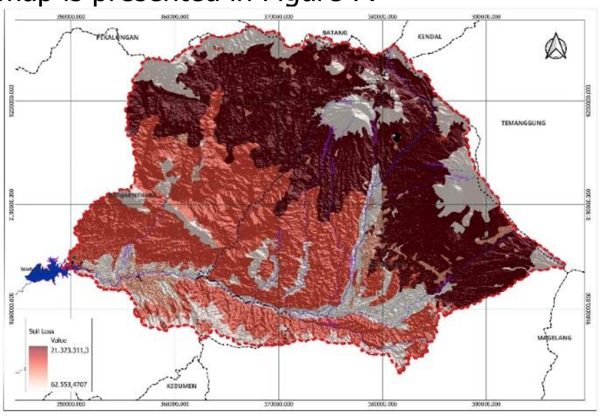

Figure 7. Soil Loss (Environmental Variable)

Based on Figure 7, it can be seen that soil loss values vary between 62,553 tons/y to 21,323,311 tons/y. The Mrica Catchment analyzed in this study had a total area of 98,703.13 Ha; hence, the total soil loss was 443.90 tons/ha/y. Following the Ministry of Environment and Forestry's Erosion Hazard Classification (EHR), soil loss in the Mrica Catchment is high.

On the Management Variable, the amount of soil loss without considering the CP value or the CP value is ignored. CP factor is related to land cover and conservation data, which are dynamic and cannot be assumed to be constant throughout the year. The soil loss map for the Management Variable can be seen in Figure 8.

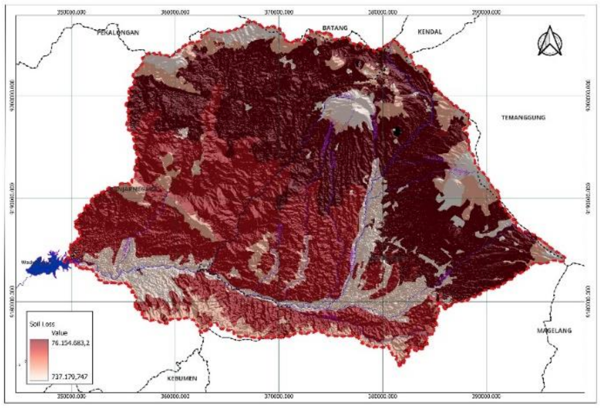

Figure 8. Soil Loss (Management Variable)

4. Conclusion

This study has demonstrated soil loss estimation in the Mrica Catchment using the Revised Universal Soil Loss Equation (RUSLE) Method. Based on the results of the soil loss analysis, the RUSLE Method emerges as an effective approach for predicting soil loss in watersheds when all relevant variables, including erodibility, rainfall erosivity, slope length, and steepness, as well as cover management and land conservation, have been accurately measured. The total soil loss, as determined by the RUSLE Method for an area of 98,703 hectares, amounts to 21,323.311 metric tons per year or 443.90 tons per hectare per year, particularly in regions dominated by dryland farming.

According to the RUSLE Method, the area with the highest soil loss values is reliably identified in both Environmental and

Management variables, notably dense in the northern area of Mrica Reservoir Catchment. It is important to note that repeated erosion of a certain severity can impact soil solubility and fertility. The soil loss estimates calculated in this study can serve as a valuable reference for devising effective and efficient strategies to control reservoir sedimentation in the Mrica Reservoir.

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Contributor Statement

BKPC, **RP**, and **EGAS** as principal contributors made primary contributions to this manuscript, taking on the responsibilities of conceiving the idea, data collection, developing the methodology, performing the analysis, and writing full manuscript.

DL as supporting contributor with responsibility to oversee the writing and the calculations. **KNA** also contributed in a supporting role, assisting the principal contributors in writing the manuscript.

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Organochlorine and pyrethroid residue in fish and sediment of Lake Singkarak, a tropical deep lake

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Abstract: Agricultural activities still involve the use of synthetic pesticides to support the increase of their products. On the other hand, the use of synthetic pesticides such as organochlorines and pyrethroids may contribute to the decline of aquatic ecosystem health due to the accumulation of their residues in sediments and organisms. The current study aimed to assess the levels of organochlorine and pyrethroids pesticide residue in endemic fish and sediment from Lake Singkarak. Bilih fish and sediment samples were taken in June 2021 at ten (10) sampling sites in Lake Singkarak. The samples were extracted and analyzed by gas chromatography. Seven organochlorine compounds were measured, including aldrin, endrin, dieldrin, DDT, heptachlor, lindan, and endosulfan. Meanwhile, three compounds chosen from the pyrethroid group, cypermethrin, permethrin, and α-cypermethrin, were also measured. Four organochlorine compounds, aldrin, dieldrin, DDT, and endrin, were detected in bilih fish in three different sites. Dieldrin had the highest concentration at nd-0.007 mg/kg, followed by DDT, endrin, and aldrin. Meanwhile, in the sediments, no organochlorine compounds were detected from all observed sites. Pyrethroid compounds were detected in bilih fish at six sites. The compound with the highest concentration was permethrin (nd-0.02 mg/kg), followed by cypermethrin and α-cypermethrin. The surface sediment from three sites contained two pyrethroid residues, permethrin and α-cypermethrin, at nd-0.002 and nd-0.001 mg/kg, respectively. Our findings show that the residual levels of organochlorine and pyrethroid in bilih fish still meet the standards set by the Codex Alimentarius Commission (CAC). Nevertheless, Bilih fish accumulate more pesticide than surface sediment, so it is essential to be aware of their potential accumulation in the human body as the final consumer. Restriction on synthetic pesticide application is necessary to reduce its residue input into the lake waters for ecological and human health.

Keywords: Bilih fish, Lake Singkarak, organochlorine, pyrethroid, sediment

1. Introduction

Farmers still use synthetic pesticides to eradicate pests and diseases in crops quickly and practically. However, its increasingly intensive use in agricultural or plantation areas can cause problems on land and aquatic systems. Generally, synthetic pesticides are toxic. Therefore, they become a potential source of pollution for the aquatic environment

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(Taufik, 2011). Pesticide residues can degrade water quality and accumulate in sediments and aquatic organisms (Lushchak, 2018; Shah & Parveen, 2023).

Organochlorine is a synthetic pesticide that began to be used worldwide in the 1950s to improve agricultural products. Although banned several decades ago, organochlorine residues are still detectable in soil, water, and agricultural commodities (Ardiwinata *et al*., 2020). To date, some farmers still use organochlorine obtained through illegal routes (Egbe *et al*., 2021). As a chemical with high environmental persistence, organochlorine has low solubility in water, high lipophilicity, and a low degradation rate (Jayaraj *et al*., 2016 or 2017). Organochlorine properties cause it to have a short residence time in the water, as it can be quickly adsorbed to suspended materials, sediments, and organisms (Fernández-Bringas, 2008). Therefore, organochlorine also contributes to bioaccumulation in the food chain (Sharma *et al*., 2009).

Pesticides in the form of pyrethroids have been widely used since the 1970s as an alternative to organochlorine (Yang *et al*., 2020). In contrast to organochlorines, pyrethroids are characterized by their low persistence, high degradability, and low toxicity to mammals (Costa, 2015). However, it is highly toxic to fish and non-invertebrate targets (Li *et al*., 2017). Like organochlorines, pyrethroids are also highly hydrophobic (Wang *et al*., 2023). This characteristic allows them to bind to sediment particles easily, which can potentially be a source of secondary contaminants if released into the water column (Li *et al*., 2014).

Lake Singkarak is the second largest lake on the island of Sumatera, with an area reaching 11,220 ha and a maximum depth of 270 m (Wils *et al*., 2021; Syawal *et al*., 2023). Locals use the lake water for various purposes. However, the agricultural activities around the lake cannot be separated from using insecticides to protect plants from pests. Pesticides from the pyrethroid group are also used in the lake to make it easier to harvest shrimp. Pesticide residue can potentially reduce the quality of lake water resources, including biotic components, such as the bilih fish, an endemic fish with high economic value (Triharyuni *et al*., 2022). However, information regarding the occurrence and pesticide pollutant levels in tropical lakes is still limited, especially studies involving endemic biota as a high-value economic food source. This study aimed to determine the levels of organochlorine and pyrethroid pesticide residues in bilih fish and sediments in Lake Singkarak.

2. Materials and Methods

2.1 Study Area and Sample Collections

The study was conducted in June 2021 at 10 sampling sites, including Batu Taba, Sumpur, Guguk Malalo, Tikalak, Ombilin, Sumani, Saniang Baka, Muaro Pingai, Paninggahan, and Tanjung Mutuih (Figure 1). The sampling location was determined by considering several factors such as water resources, water use, land use, irrigation system, type of crops, various pesticide use, and the agricultural area. As much as 50-100 g of bilih fish samples were taken from each study site. They were wrapped in aluminium foil and immediately cleaned. 500 g of sediment samples were taken from each location using a shovel or grab sampler. In the laboratory, the fish and sediment samples were stored at -25 °C and 4°C, respectively, until the extraction process was conducted.

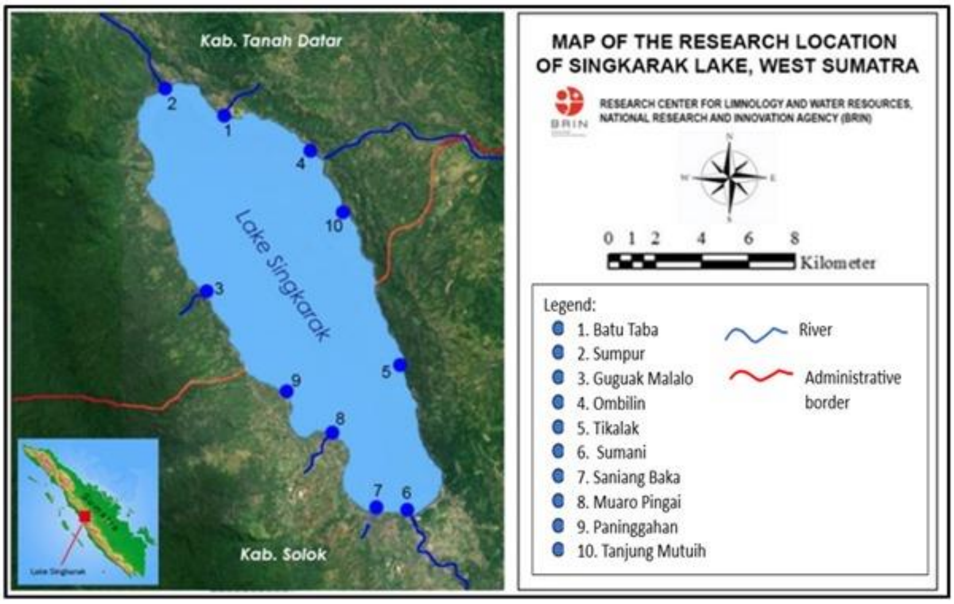

Figure 1. Map of sampling locations in Lake Singkarak

2.2 Sample Extraction-and Analysis

Sample extraction and analysis were carried out at the Laboratory of Agrochemical Material Residues in Bogor Regency. The procedure to extract sediment and bilih fish samples is shown in Figure 2 and Figure 3. The extraction was the step before the sample was measured in Gas Chromatography.

For chromatographic analysis, 1 µL of the solution sample was injected into the GC injector in the 450-Gas Chromatograph to determine the active ingredients of the pesticides classified as organochlorine. The temperature of the injector and detector were set at 300°C and 250°C, respectively. The initial temperature of the column was first set to 100°C for three minutes, then 250°C for 10 minutes (rate of 20°C/min), and then raised to 260°C for 20 minutes (rate of 2°C/min). The concentration of pesticide residues in the sample was calculated using the formula (Ardiwinata, 1999) as follows.

$$\text{Residue (mg/kg)} = (Ac \times Vis \times Cs \times Vfc) / (As \times Vic \times B \times R)$$

where:

Ac = Sample area (mm)
As = Standard area (mm)
Vic = Injection volume sample (µL)
Vis = Injection volume standard (µL)
Cs = Standard concentration (µg/mL)
B = Initial weight of sample (g)
Vfc = Sample final volume (mL)
R = Recovery (%)

Figure 2. Flowchart showing the extraction procedure of sediment samples for pesticide residue analysis (Rahmawati *et al.*, 2017; Oginawati *et al.*, 2021)

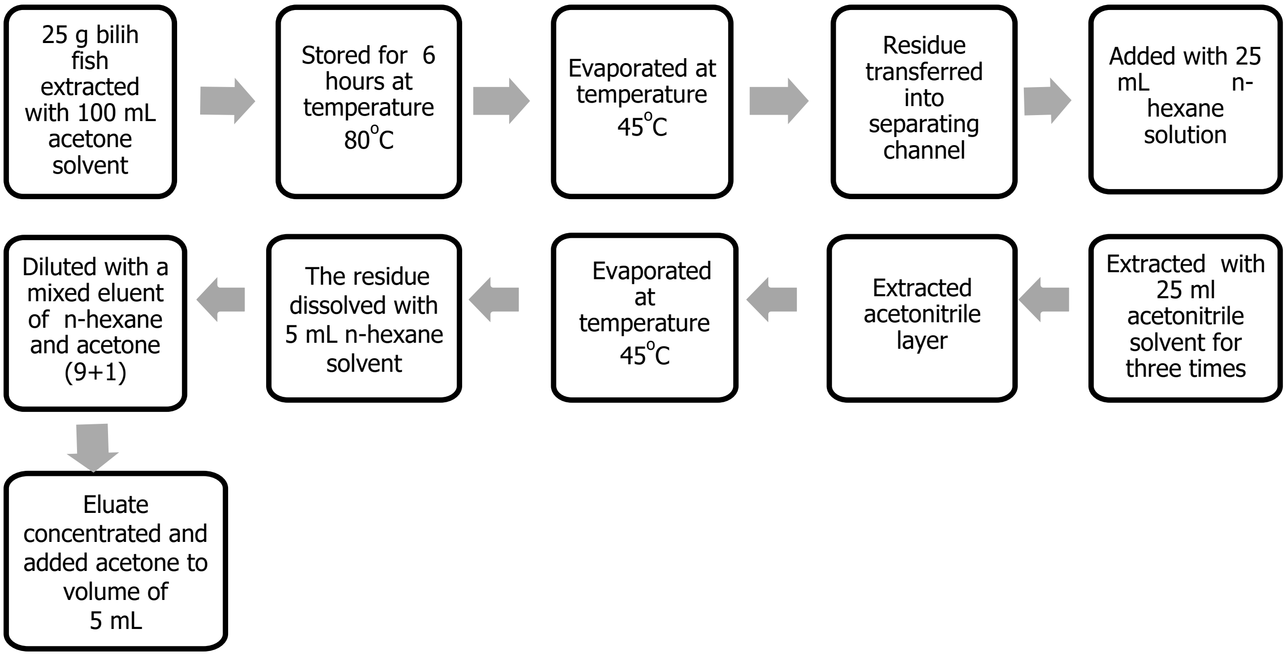

Figure 3. Flowchart showing the extraction procedure of fish samples for pesticide residue analysis (modified from Kanazawa, 1981)

3. Results and discussion

3.1. Organochlorine and Pyrethroid Residue of Bilih Fish

The types of organochlorine residues detected in bilih fish in Lake Singkarak during this study are shown in Table 1. Only four types of organochlorines were detected in fish, including aldrin, dieldrin, DDT, and endrin. Aldrin was only found in Batu Taba at 0.002 mg/kg, while endrin was only found in Sumpur at 0.005 mg/kg. Dieldrin and DDT were detected in bilih fish at the Tanjung Mutuih site at 0.007 mg/kg and 0.005 mg/kg, respectively. The three types of organochlorines (heptachlor, lindan, and endosulfan) were not detected in all observed sites. Meanwhile, at seven other sites, no organochlorine was detected.

The discovery of four types of organochlorines at three sites is thought to be caused by their use in the past and their high persistence in the environment. Their uncontrolled use also affects the presence of organochlorine, especially in Tanjung Mutuih sites where rice fields are found with channels (canals) that flow directly into lake waters. The presence of dieldrin in bilih fish in Tanjung Mutuih is thought to be due to the breakdown of aldrin compounds by bilih fish tissues. In addition, detecting aldrin in fish indicates the recent use of pesticides that contain this active compound (Mensah *et al.*, 2021).

This study found higher organochlorine levels in bilih fish than in the surface water of Lake Singkarak (Ibrahim *et al.*, 2022). In bilih fish, organochlorine levels were also higher than in the sediment. These high organochlorine levels are probably related to their low solubility in water and high lipophilic properties (Kafilzadeh, 2015). Taufik (2011) also reported that pesticide levels in pond fish are higher than in water and sediments. The lipophilic nature of organochlorines allows it to bind to the fatty tissue in fish easily.

The organochlorine levels in bilih fish in this study are generally smaller when compared to levels in tilapia and catfish in Lake Geriyo, Nigeria (Shinggu *et al.*, 2015). This rate is also still below the Maximum Residue Limit (MRL) based on the Codex Alimentarius Commission (CAC) 39th Session (2016). However, the dieldrin recorded in this study exceeds the levels detected in Lake Edko of 0.0004 mg/kg (Abbassy *et al.*, 2021).

Table 1. The concentration of organochlorine residue (mg/kg) in bilih fish from Lake Singkarak

Station	**Aldrin**	**Dieldrin**	**DDT**	**Endrin**	**Heptachlor**	**Lindan**	**Endosulfan**
Batu Taba	0.002	nd	nd	nd	nd	nd	nd
Sumpur	nd	nd	nd	0.005	nd	nd	nd
Guguak Malalo	nd	nd	nd	nd	nd	nd	nd
Ombilin	nd	nd	nd	nd	nd	nd	nd
Tikalak	nd	nd	nd	nd	nd	nd	nd
Sumani	nd	nd	nd	nd	nd	nd	nd
Saniang Baka	nd	nd	nd	nd	nd	nd	nd
Muaro Pingai	nd	nd	nd	nd	nd	nd	nd
Paninggahan	nd	nd	nd	nd	nd	nd	nd
Tanjung Mutuih	nd	0.007	0.005	nd	nd	nd	nd
LoD	0.001	0.001	0.001	0.001	0.019	0.015	0.017
MRL	0.2	0.2	5	-	0.2	0.01	0.2

Note: nd : not detected or below detection limit; LoD: Limit of Detection; MRL: Maximum Residue Limit

Three compounds from the pyrethroid group have been analyzed in bilih fish at ten sites in Lake Singkarak. Table 2 shows the results of the analysis of the pyrethroids accumulated in bilih fish. Permethrin is the dominant pyrethroid accumulated in bilih fish, followed by α-cypermethrin and cypermethrin. The permethrin level found in this study was nd-0.02 mg/kg. Meanwhile, the concentrations of cypermethrin and α-cypermethrin detected were nd-0.008 mg/kg and nd-0.007 mg/kg, respectively. Permethrin was found in three sites, with its highest concentration at Muaro Pingai. The highest levels of cypermethrin and α-cypermethrin were found at Tanjung Mutuih and Sumpur sites, respectively. The three compounds of pyrethroids analyzed in this study were not detected in four other sites.

The levels of pyrethroids measured in bilih fish was relatively higher than in sediments. The higher lipid content in fish leads to an increased potential absorption of hydrophobic compounds (Spacie & Hamelink, 1982). As mobile organisms, fish can be exposed to these compounds in other parts of the hydrological system. Similarly, the accumulation rate of pesticide residues increases with temperature (Charles *et al.*, 2000). Riaz *et al.* (2018) also reported that higher pyrethroids levels were detected in fish than in the Chenab River sediment. Similar results were seen in Mahboob *et al.* (2015) study on the Ravi River. Riaz *et al.* (2018) explained that fish are highly susceptible organisms to pesticide accumulation due to their lipophilic properties. Pesticide contamination in aquatic organisms is caused by direct uptake from water through gills or skin, via uptake of suspended particles, and even contaminated food (van der Oost *et al.*, 2003).

The pyrethroid (cypermethrin) levels in bilih fish in this study were lower than the levels detected in fish in the Red River Basin, Vietnam, at 0.109-0.802 mg/kg (Pham *et al.*, 2011). Mahboob *et al.* (2015) findings also showed more significant levels of cypermethrin in the Ravi River. Conversely, wild fish in the Iberian River Basin in Spain had a lower accumulation of cypermethrin due to the use of local insecticides (Corcellas *et al.*, 2015). Xie *et al.* (2022) reported slightly higher levels of permethrin and slightly lower levels of cypermethrin in fish compared to this study's findings. This difference may be correlated with the increase in population and pesticide use.

Regarding the CAC 39th Session (2016), the three types of pyrethroids contained in bilih fish in Lake Singkarak still meet the required standard values. Nevertheless, we must be mindful of the increased use of pesticides in Lake Singkarak as their potential accumulation in bilih fish can have lethal and sublethal effects. Lethal effects can be in the form of fish death, while the sublethal effect can be physiological and biochemical changes (Taufik, 2011).

Table 2. The concentration of pyrethroid residue (mg/kg) in bilih fish from Lake Singkarak

Station	Permethrin	Cypermethrin	α-Cypermethrin
Batu Taba	nd	0.003	0.004
Sumpur	nd	nd	0.007
Guguak Malalo	nd	nd	nd
Ombilin	nd	nd	nd
Tikalak	0.0040	0.0030	nd
Sumani	0.0020	0.0020	nd
Saniang Baka	nd	nd	nd
Muaro Pingai	0.02	nd	nd
Paninggahan	nd	nd	nd
Tanjung Mutuih	nd	0.008	0.0060
LoD	0.0053	0.0015	0.0010
MRL	1	2	2

Note: nd: not detected or below detection limit; LoD: Limit of Detection;
MRL: Maximum Residue Limit

3.2. Organochlorine and Pyrethroid Residue of Surface Sediment

Organochlorine residues were not detected in the sediments of all sampling sites of Lake Singkarak (Table 3), DDT metabolites could be detected due to the use of DDT in the past. Only DDT metabolites were found in sediments, whereas concentrations of organochlorine pesticides (heptachlor, dieldrin, DDT) were detected below detection limits in several different lakes (Dvoršćak *et al.,* 2019; Mergia *et al.,* 2022; Hu & Tao, 2023). Meanwhile, the sedimentary concentrations of organochlorine pesticides ranged from 0.00 to 14.83 ng/g dry weight in the Eldrin-dominated Iznik Lake (Aydın & Albay, 2022). Organochlorine pesticide residues in the sediment were also found in the Tono Reservoir with average levels of 0.09, 0.04, and 0.047 µg/g for aldrin, DDE, and DDD (Akoto *et al.*, 2016). The study also showed higher levels of aldrin in the sediment compared to fish and lower DDE and DDD levels in the sediment. Other studies showed organochlorines were found to be higher (the highest pesticide aldrin was 2-1438 µg/L) in sediments compared to water, molluscs, and fish (Barlas *et al.*, 2006; GA, 2016; Mensah *et al.*, 2021; Oginawati *et al.,* 2022).

Table 3. Concentration of organochlorine residue (mg/kg) in sediment from Lake Singkarak

Station	Aldrin	Dieldrin	DDT	Endrin	Heptachlor	Lindan	Endosulfan
Batu Taba	nd	nd	nd	nd	nd	nd	nd
Sumpur	nd	nd	nd	nd	nd	nd	nd
Guguak Malalo	nd	nd	nd	nd	nd	nd	nd
Ombilin	nd	nd	nd	nd	nd	nd	nd
Tikalak	nd	nd	nd	nd	nd	nd	nd
Sumani	nd	nd	nd	nd	nd	nd	nd
Saniang Baka	nd	nd	nd	nd	nd	nd	nd
Muaro Pingai	nd	nd	nd	nd	nd	nd	nd
Paninggahan	nd	nd	nd	nd	nd	nd	nd
Tanjung Mutuih	nd	nd	nd	nd	nd	nd	nd
LoD	0.001	0.001	0.001	0.001	0.019	0.015	0.017
TEL (ug/kg)	-	2.850	-	2.670	0.600	0.940	-

Note: nd: not detected or below detection limit; LoD: Limit of Detection;
TEL: Threshold-Effects Level

Table 4. The concentration of pyrethroid residue (mg/kg) in sediment from Lake Singkarak

Station	**Permethrin**	**Cypermethrin**	**α-Cypermethrin**
Batu Taba	nd	nd	nd
Sumpur	nd	nd	nd
Guguak Malalo	0.0020	nd	nd
Ombilin	nd	nd	nd
Tikalak	nd	nd	nd
Sumani	nd	nd	nd
Saniang Baka	nd	nd	nd
Muaro Pingai	0.0010	nd	nd
Paninggahan	nd	nd	0.0010
Tanjung Mutuih	nd	nd	nd
LoD	0.0053	0.0015	0.0010
TEL	-	-	-

Note: nd: not detected or below detection limit; LoD: Limit of Detection;
TEL: Threshold-Effects Level

We detected organochlorine compounds in sediments that can be assessed based on the freshwater sediment quality assessment guide. This is done to evaluate the possible ecological risks posed by pesticides. The organochlorine levels obtained can also be evaluated based on the Threshold-Effects Level (TEL), which indicates the level below which adverse biological effects are expected to occur rarely (Smith *et al.,* 1996).

Pesticide residues in surface water and sediments are critical because of their negative impacts on aquatic ecosystems and their implications for drinking water sources. Pesticides can accumulate in sediments through the indiscriminate use of pesticides that leads to their entry to the bottom sediments, where they are absorbed in the sediment's particles and, thus, may become the consistent source of aquatic pollution (Shah & Parveen, 2023).

Pyrethroids were analyzed in sediments to determine their possible contamination. The pyrethroid residues that were studied for their content in the Singkarak Lake sediments were permethrin, cypermethrin, and α-cypermethrin (Table 4). Permethrin was found at two sites, namely Guguak Malalo and Muara Pingai, with a concentration range of 0.001-0.002 mg/kg. Another type of pyrethroid found was α-cypermethrin at Paninggahan with a concentration of 0.001 mg/kg. Around the Guguak Malalo, Muara Pingai, and Paninggahan areas are agricultural areas. In that area, there are many rice fields with rice as the main crop, besides other agricultural products such as chilies, beans, corn, and shallots. It is possible that the sedimentary pyrethroid pesticides originated from pesticides in agricultural areas, were carried into the lake water column and accumulated in the sediments. Pesticides enter water bodies through runoff from agricultural areas that use many pesticides (Effendi, 2003).

Sediments at Sumpur, Batu Taba, Ombilin, Tikalak, Tanjung Mutuih, Sumani, and Saniang Baka sites were not detected with pyrethroid residue. Ombilin, as an outlet area of the lake quite distant from agricultural areas, resulted in the absence of residues in the sediment. While the other six locations, despite being agricultural areas, did not detect the presence of residues in the sediment. This absence of residues in the sediment is thought to be related to the pattern of pyrethroid use and the half-life of pyrethroid types in the sediment (Li *et al.*, 2017). Additionally, the surface area and organic content of sediment can influence the level of residue adsorption (Mensah *et al.*, 2021).

Compared to our findings, the pyrethroids levels in sediments in Lake Weija tend to be higher, where the levels of permethrin and cypermethrin were nd-0.0066 mg/kg and nd-0.0032 mg/kg, respectively (Afful *et al.*, 2013). Lake Weija is located in Ghana, where its condition is similar to Lake Singkarak, where agriculture is one of the primary land uses surrounding the lake. Meanwhile, this study's α-cypermethrin levels tend to be the same as the findings reported by Merga (2021) of <0.00071-0.00197 mg/kg in sediments in Lake Ziway, Ethiopia. Moreover, Lake Ziway got pressure from agricultural activities, besides its urbanization activities. Pesticides released from large- and small-scale agricultural activities are posing ecological risks to biotic components in Lake Ziway. Pyrethroids in sediment in this study align with Li et al. (2017) who state that permethrin concentrations in the sediment will always be high. Due to pyrethroids' high hydrophobic properties, in aquatic environments, they can be absorbed into particles, deposited into the sediment, and accumulate in the body of organisms.

The presence of residual pyrethroid pesticide in bilih fish and sediment in this study confirms the use of synthetic pesticides from that class in the area around Lake Singkarak. Similarly, the detection of organochlorines such as aldrin in bilih fish may indicate their current use through illegal routes despite being banned since 1970. Residual pesticide compounds in the lake waters, with increasing types and concentrations, will affect the health of aquatic biota and can also have long-term implications for human health. Proper use and restriction of synthetic pesticide use can contribute to minimizing pollutant inputs into the lake for the sake of a healthy lake ecosystem.

4. Conclusion

This study reveals the presence of organochlorine and pyrethroid pesticide residues in Lake Singkarak, with their levels higher in bilih fish compared to sediment. Our findings enrich information on the presence of pollutant compounds in tropical lake ecosystems, including in endemic biota that serve as a high-value economic food source. Future study is needed to measure the levels of other pesticide classes in various biotic and abiotic components in Lake Singkarak during the dry and rainy seasons. Moreover, the human health and ecological risks of synthetic pesticide application are also essential to investigate. This investigation is crucial to support efforts in the protection and management of tropical lake ecosystems amidst increasing anthropogenic activities.

Data availability statement

Data used in this study are available from the corresponding author upon request.

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Conflict of interests

The authors declare no conflict of interest.

Contributor statement

AI: Conceptualization, Investigation, Writing - original draft. **MSS**: Investigation, Methodology, Project Administration. **ANA**: Resources, Methodology, Formal Analysis. **S**: Writing - original draft, Writing - review & editing. **MAA**: Writing - original draft. **WF**: Writing - original draft. **RK**: Resources

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Hydrochemical dynamics of stream following rainfall events at agricultural catchments in New Zealand

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Abstract: One of the prerequisites for efficiently managing lake water quality is reliable data regarding the quantity and quality of inflows water, mainly the export of nutrients from the catchment area during rainfall events. We investigated the dynamic characteristics of hydrochemicals concerning rainfall events in agricultural stream waters flowing into eutrophic lakes situated on the North Island's central plateau of New Zealand. We utilized isotopic composition of water (δ^2H-H_2O and $\delta^{18}O$-H_2O) and nitrate ($\delta^{15}N$-NO_3^- and $\delta^{18}O$-NO_3^-) along with high-frequency hydrochemical data for source identification of water and nitrate during a drought period (2020). Our findings indicate that it is essential to initially grasp the fundamental mechanisms associated with rainfall events to formulate effective strategies for minimizing nutrient losses. The methodology outlined in this research integrates stable isotope hydrology with water quality monitoring initiatives, facilitating the understanding and managing the primary governing mechanisms behind diverse contaminant losses from land to adjacent water bodies, explicitly focusing on nitrates. This approach establishes a framework that can assist in devising measures for water quality improvement capable of anticipating the repercussions of substantial rainfall events more effectively.

Keywords: water isotopes, nitrate isotopes, agricultural catchment, event-based sampling, high-frequency data

1. Introduction

Although lakes constitute a minor fraction of surface water, they offer diverse ecosystem benefits such as supporting biodiversity, leisure and tourism, fisheries, hydroelectricity and climate change mitigation (Schallenberg *et al*., 2013). Nevertheless, the combined impacts of global change and human-induced factors persist in exerting environmental stress on lakes globally. A parallel trend is observed in New Zealand, with lakes facing deteriorating water quality. Specifically, around 46% of lakes in New Zealand with an area larger than 1 hectare are assessed to be in poor ecological condition, exemplifying the extent of the issue (Ministry for the Environment & Stats NZ, 2022).

The primary cause of diffuse contaminants, particularly nitrates, entering New Zealand's aquatic environment is pastoral agriculture, which stands as the predominant land use in the country (Howard-Williams *et al*., 2010). The rise in nitrate levels in New Zealand's water is notably linked to the increased application of nitrogen fertilizer in agriculture (Joy *et al*., 2022; Larned *et al*., 2020). This trend aligns with a global assessment highlighting the amplified livestock farming industry as a critical contributor to freshwater contamination (Mateo–Sagasta *et al*., 2017). Hence, lakes situated in catchment

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areas primarily characterized by pastoral land use often exhibit inferior water quality, a pattern observed in various studies, including those by Abell *et al*. (2010) and Verburg *et al*. (2010). Statistical data reveal a disproportionate contribution to nitrogen loads from livestock farming land, equivalent to 6.8% of the total land area but accounting for 37% of nitrogen loads. This underscores pastoral land as the primary source of land-based nitrogen in New Zealand (Elliott *et al*., 2005). Given the significance of agriculture to the national economy, addressing contaminant export necessitates finding mutually beneficial solutions that preserve agricultural production and profitability while upholding ecosystem function.

Studying water quality at the catchment level is increasingly complex due to the substantial surface runoff, accompanied by elevated levels of leached nutrients, discerned during specific meteorological conditions, such as rain events. Various studies, including those by Kozak *et al*. (2019), Arnell *et al*. (2011), and Tomer *et al*. (2010), have indicated that contaminant transport to the aquatic environment is heightened during rain events with increased discharge, leading to the identification of stormflows as hot moments (Wey *et al*., 2022; Sigler *et al*., 2020; McClain *et al*., 2003). Despite this recognition, questions persist regarding which rainfall characteristics contribute significantly to nitrate export. Therefore, it is crucial to comprehend the mechanisms controlling the generation of runoff from rainfall, particularly when considering nutrient management in catchments, as highlighted in studies by Kirsch (2020) and Monaghan *et al*. (2016).

Numerous investigations have focused on the hydrologic response of catchments to particular rainfall occurrence, such as those by Pavlin *et al*. (2021), Saffarpour *et al*. (2016), and Detty & McGuire (2010), or have reported on the impact of rainfall characteristics on runoff quantity and quality, as seen in studies by Sapač *et al*. (2020), Lintern *et al.* (2018), and Macrae *et al*. (2010). However, there are existing knowledge gaps regarding the relationships between hydrological responses, various rainfall events, and the reasons for variations in nutrient loads, particularly in pastoral catchments that contribute to lake inflows (Levine *et al*., 2021; Abell *et al*., 2013; Menneer *et al.,* 2004). Consequently, to enhance water quality in both inflows and receiving lakes, it is imperative to comprehend the dynamic nature of contaminant loading within the context of changing hydrological patterns.

Identifying and quantifying nutrient export to receiving waters is challenging due to the complex nature of terrestrial and in-stream biogeochemical processes. Relying solely on concentration data, as Barnes & Raymond (2010) emphasized, is insufficient. Using isotope data proves valuable in elucidating the pathways and occurrences of hydrogen and oxygen isotopes in water on a more comprehensive scale (McGuire & McDonnell, 2007). Environmental isotopes, including hydrogen, carbon, nitrogen, and oxygen, possess distinctive characteristics (Fry, 2006; Kendall & Caldwell, 1998) that make them effective tracers for understanding the cycling of water and nutrients in the environment.

Gaining insights into the hydrological pathways through which water reaches the stream is crucial for understanding flow generation and transporting soluble nutrients, particularly nitrate, from their sources. Nitrogen, primarily nitrate (NO_3^-), is a widespread concern for water quality in New Zealand (Singh *et al*., 2019; Davies-Coley, 2013). The use of isotope tracers for water ($\delta^2H–H_2O$ and $\delta^{18}O–H_2O$) offers a valuable initial indication of the origins, flow paths, and biogeochemical transformations of water contaminants, as discussed by Jung *et al*. (2019) and Abbott *et al*. (2016). In hydrologic studies, stable isotopes of water, along with hydrograph separation techniques, have been extensively employed to distinguish "old" (uniform) water from the more variable "new" water or the processes that gave rise to them, owing to their unique isotopic compositions. This approach has been utilized for an extended period, as demonstrated in studies by Li et al. (2020), Richey et al. (1998), and Sklash & Farvolden (1979). For instance, precipitation (representing new water) that initiates runoff often exhibits isotopic differences from the water already present in the catchment (representing old water), as observed in studies by Tan *et al*. (2021), Boutt *et al*. (2019), and Pionke & DeWalle (1992).

Isotopic tracers of nitrate are widely acknowledged as a highly promising tool for investigating the transport and destinations of nitrate. The isotopic composition of N in nitrate, expressed as $\delta^{15}N–NO_3^-$ and $\delta^{18}O–NO_3^-$ in per mil (‰), serves as distinctive markers enabling the differentiation of various sources and associated processes of nitrate, including atmospheric N_2, soil, chemical fertilizers, and nitrification (Xue *et al*., 2009). Isotope nitrate values exhibit significant variations between nitrogen fertilizers (typically close to or <0 ‰) and animal waste (generally > 10‰) (Nestler *et al*., 2011; Xue *et al*., 2009; Kendall, 1998) as well as between the isotopic composition of nitrate from precipitation and nitrate generated through nitrification (Kendall *et al*., 2007). Consequently, the stable analysis of nitrate in water is the predominant tool employed to discern nitrate sources and estimate their contributions to the enrichment of freshwater with nitrate (Voss *et al*., 2006; Wassenaar, 1995).

The motivation for this study stems from concerns about contaminant loading and its effects on the trophic status of a lake, chosen as a representative case for in-depth examinations of the hydrological processes influencing nutrient losses to freshwater originating from pastoral agriculture. This research advances our comprehension of the role of runoff in agricultural catchments. It furnishes detailed insights into nutrient-enriched lakes in the Central Plateau of the North Island, New Zealand.

2. Materials and Methods

2.1 Study areas

This research was conducted in the Bay of Plenty region situated on the North Island of New Zealand, as illustrated in Figure 1. The study focused on two sites within the Te Arawa Lakes Catchment, specifically Ōkaro and Ngongotaha. The Lake Ōkaro catchment, which feeds into the eutrophic Lake Ōkaro, has been extensively examined in previous studies (e.g., Santoso *et al*., 2021; Özkundakci *et al*., 2010; Forsyth *et al*., 1988), and its inflow catchment has been well-characterized (Hudson & Nagels, 2011; Özkundakci *et al*., 2011). Lake Ōkaro has established targets for enhancing its trophic state as per Environment Bay of Plenty (2006), particularly in a region where lakes may be partially nitrogen-limited (Abell *et al*., 2010). The Lake Ōkaro catchment has been a significant source of nutrient inputs to Lake Ōkaro, leading to frequent algae blooms in spring and summer (Paul *et al*., 2008). Consequently, Lake Ōkaro stands as the most eutrophic among the Rotorua Te Arawa Lakes, rapidly progressing in eutrophication (Özkundakci, 2011; Wood *et al*., 2009). The other site investigated in this study was the Ngongotaha stream catchment. These one of nine major stream tributaries flows into Lake Rotorua, a sizable eutrophic and polymictic lake located in the Bay of Plenty Region of New Zealand. The water condition in Lake Rotorua has deteriorated since at least the 1960s, primarily attributed to an excess of nutrient input, leading to eutrophication, undesirable algal blooms, and a prioritized need for remediation (Abell *et al*., 2013). Both catchments are predominantly characterized by pastoral agriculture, with areas comprising approximately 72% and 51% relative to the catchments for the Lake Ōkaro catchment (3.98 km^2) and Ngongotaha stream catchment (60 km^2), respectively.

2.2 Water sampling and analysis

This study was focused on understanding hydrochemical dynamics during rainfall events; thus, event-based samplings were conducted at two sites coinciding with winter events in July 2020. Water samples were collected utilizing a Manning VST portable vacuum sampler (Manning Environmental Inc, USA). The autosamplers were configured to operate at 1-h intervals, ensuring the rising and falling limb hydrograph coverage. To capture baseflow conditions, further discrete samples were obtained once in pre– and post–rain event periods. Precipitation samples were collected over a year (2019 – 2020) using free-evaporation bucket containers to develop a Local Meteoric Water Line (LMWL) for each site.

Before collecting samples, freshly acquired containers were washed with water from the sampling location. The sample containers and hoses of the autosampler underwent an acid-washing procedure using 10% nitric acid, followed by rinsing with high-quality deionized water prior to deployment. Following retrieval, water samples were subjected to filtration using a 0.45 µm cellulose-acetate membrane to assess dissolved inorganic nitrogen (DIN) and isotopes. Unfiltered samples were also obtained to assess total nitrogen (TN). The transported water samples were kept at a low temperature on ice during transportation to the laboratory, where they were promptly stored at four °C until subsequent analysis. The analysis encompassed the determination of nitrite–N (NO2—N) and nitrate–N (NO_3^-–N) as total oxidized nitrogen using a flow injection analyzer based on an automated cadmium reduction method (APHA 4500, modified 23rd ed. 2017). Nitrite–N was analyzed based on automated azo dye colorimetry. Thus, nitrate–N was calculated from total oxidized nitrogen minus nitrite–N. Total Kjeldahl Nitrogen (TKN) was analyzed based on the colorimetric method. Total nitrogen concentration was determined by adding the values for TKN, NO_2^-–N, and NO_3^-–N.

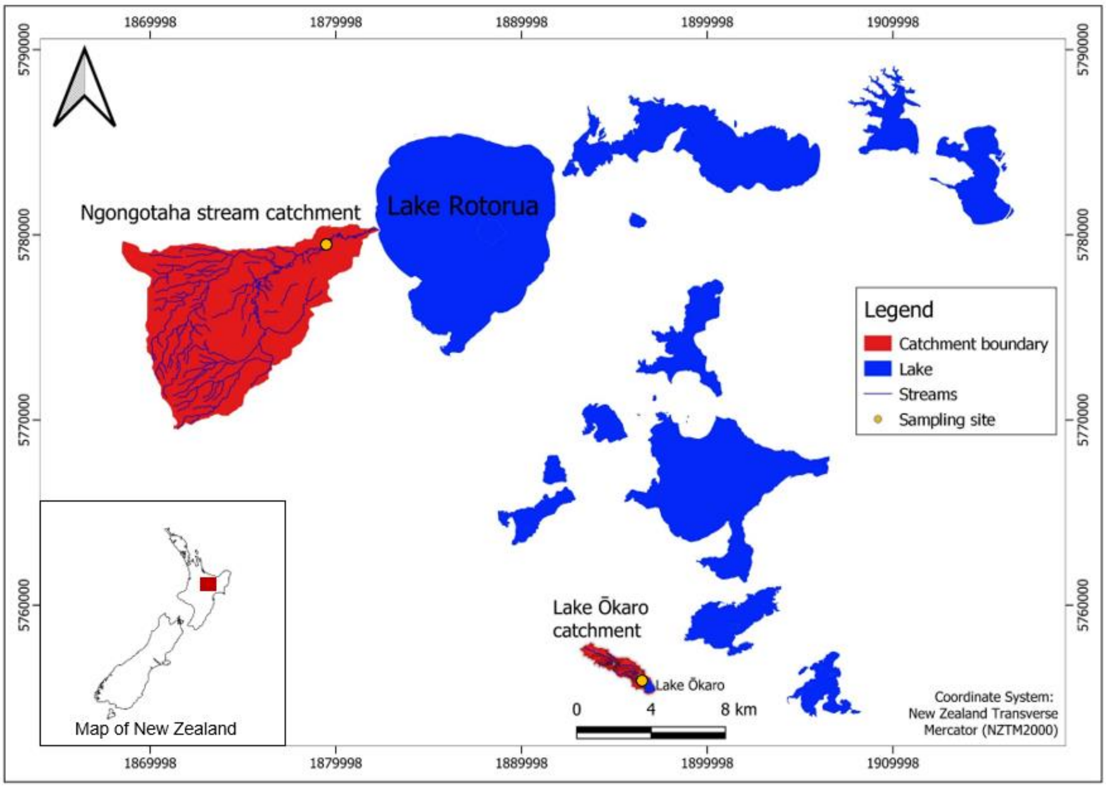

Figure 1. Map of study areas situated in a complex of Te Arawa Lakes, Rotorua, Bay of Plenty Region

2.3 Isotope analysis

Water samples underwent analysis for stable water isotopes, specifically oxygen ($\delta^{18}O$–H_2O) and hydrogen (δ^2H–H_2O). The isotopic compositions of the water samples, contained in 2 mL glass vials, were measured using a Los Gatos Research (LGR) TIWA laser spectrometer. The reported isotope ratios are presented in per mil (‰) relative to VSMOW–SLAP, referencing two internal working standards (AURORA2: δ^2H = +1.63‰, $\delta^{18}O$ = –0.8‰ and ANT01: δ^2H = +1.63‰, $\delta^{18}O$ = –0.8‰), which were previously calibrated with VSMOW2 ($\delta^{18}O$ = 0‰ and δ^2H = 0‰) and GRESP (δ^2H = –257.8‰, $\delta^{18}O$ = –33.39‰) international reference standards. Following the method described by Wassenaar *et al.* (2008), isotopic values were determined by averaging the last four out of seven injections to minimize memory effects. The analytical uncertainty, selected through an IAEA Water Stable Isotope Intercomparison test

(Wassenaar *et al.*, 2021), was approximately 0.2‰ and 0.09‰ for δ^{2}H and δ^{18}O, respectively. Additionally, an orthogonal regression analysis was employed to establish Local Meteoric Water Lines (LMWLs, IAEA, 1992), which were then compared to the Global Meteoric Water Lines (GMWLs, Craig, 1961) in a conventional δ^{2}H versus δ^{18}O diagram.

Nitrate stable isotopic composition (δ^{15}N and δ^{18}O) was examined at the National Isotope Centre (GNS Science) using the cadmium-azide method outlined in Wells *et al.* (2015). All results are reported to AIR for δ^{15}N and VSMOW for δ^{18}O, normalized against the international standards; USGS 34 (-1.8‰ for δ15N and -27.9‰ for δ18O), IAEA-NO3 (4.7‰ for δ15N and 25.6‰ for δ18O) and internal standard; KNO3b (10.7‰ for δ^{15}N and 11.7‰ for δ^{18}O). The analytical precision of these measurements is reported as 0.3‰ for both δ^{15}N and δ^{18}O.

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2.4 Additional data

Information regarding rainfall, soil moisture, and flow was obtained from the environmental data portal of the Bay of Plenty Regional Council (BoPRC) (https://envdata.boprc.govt.nz/Data). The recording of continuous data (15-minute intervals) of electrical conductivity (EC) and water level was conducted using a digital Mayfly data logger station developed by Stroud Water Research Center (Hicks et al., 2015; Hund et al., 2016). Water level measurements were transformed utilizing the rating curve established from each study area supplied by BoPRC.

3. Results and discussion

3.1 Hydroclimatic characteristics

The Lake Ōkaro catchment recorded an annual rainfall of 1250 mm, while the Ngongotaha stream catchment experienced a higher average of 1519 mm. The year 2020 was characterized as a drought year with an arid summer, deviating from the typical hydrological pattern of the Rotorua area. Notably, the rainfall distribution did not strictly adhere to seasonal patterns. Although substantial precipitation events usually occur in winter (June to August) and autumn (March to May), 2020 exhibited an interesting deviation with more intense spring rainfall.

By comparing the two sites, it was found that there was a difference in discharge patterns during the year 2020. The hydrograph at Lake Ōkaro catchment has more peaks and is flashier in response to rainfall. The hydrological characteristics of two catchments can differ markedly because of factors both natural and anthropogenic, including their size, topography, land use, and soil properties (Lei *et al.*, 2021; Stephens *et al.*, 2021; Omer *et al.*, 2020). Understanding these differences is crucial for efficiently managing water resources, preserving the environment, and promoting sustainable development in each catchment. Additionally, assessing the impact of anthropogenic changes on the catchment's hydrology can provide insights into potential areas for mitigation and sustainable water resource management.

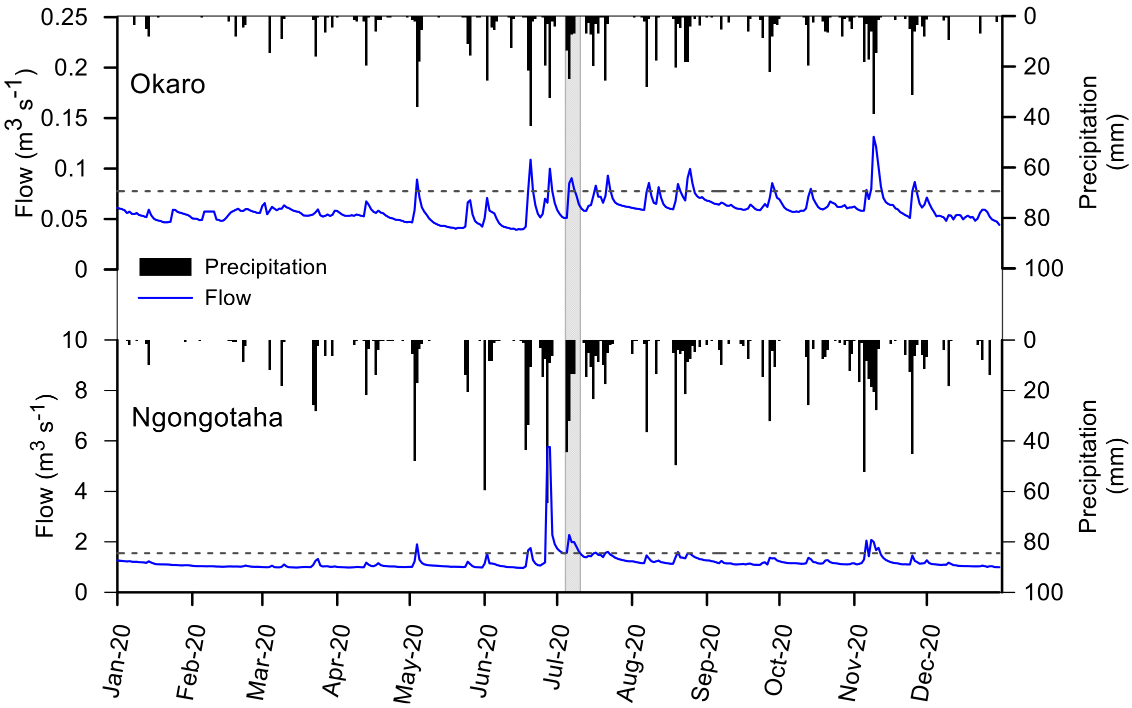

Figure 2. Daily rainfall and flow in the Lake Ōkaro and the Ngongotaha stream catchments during 2020. The shaded area indicates the event-based sampling examined in this study. The dashed line indicates the 95th flow percentile of flow.

3.2 Isotopic compositions related to rainfall events

Figure 3 shows the biplot of the isotopic composition of precipitation sampled during 2020 and stream water during July 2020 rainfall at the study areas. Rainfall δ^2H values spanned –67.9 to –4.8‰ and δ^{18}O spanned -9.81 to -1.91‰ for the Lake Ōkaro catchment. Meanwhile, for the Ngongotaha stream catchment, isotope values spanned –47.5 to –7.5‰ and –7.5 to –4.98‰ for δ^2H and δ^{18}O, respectively. This dataset reveals a seasonal fluctuation in rainfall's water isotope values. Winter exhibited the lowest δ^2H values, while the highest values were observed in summer. The LMWL trend, with a more downward slope and intercept of relative to GMWL, indicates additional evaporation following rainfall. However, it is important to acknowledge that the study period of the rainfall observation period is insufficient for constructing an LMWL, making it potentially unrepresentative of long-term precipitation patterns.

Figure 3 also illustrates that stream water samples were dispersed around LMWL, suggesting a relatively significant influence of δ^2H and δ^{18}O from rainwater. This influence was particularly evident during elevated runoff after rainfall events, indicating that the stream exhibited isotopic depletion and reflected the characteristics of recent precipitation. This observation aligns with the finding of Birkel *et al.* (2012), indicating that stream isotope values during increased streamflows exhibited fluctuations in the same direction as water isotope in precipitation. Similarly, von Freyberg *et al.* (2018) investigated temporal variations in water isotopes in stream water and rain, revealing that high-flow periods characteristically have elevated fractions of recently derived water from precipitation origins (new water).

In general, the isotopic compositions of nitrate within stream water from samples during the event ranged from 3.19 – 6.52‰ to δ^{15}N and –3.31 – 025‰ for δ^{18}O in the Lake Ōkaro catchment. In the Ngongotaha stream catchment, the corresponding ranges were from 2.38 – 5.09‰ for δ^{15}N and –5.98 – 3.63‰ for δ^{18}O. Typically, more depleted isotopic values of nitrate–N were noted during events than baseflow isotope values (Figure 4). Baseflow values were characterized by lower concentration and δ^{15}N–NO_3^- within the range associated with soil nitrogen-dominated sources (Xue *et al.*, 2009). Potential sources shifted from soil nitrogen to urine-urea sources during rainfall, as indicated by relatively depleted (≤ 4‰) nitrate isotopic composition.

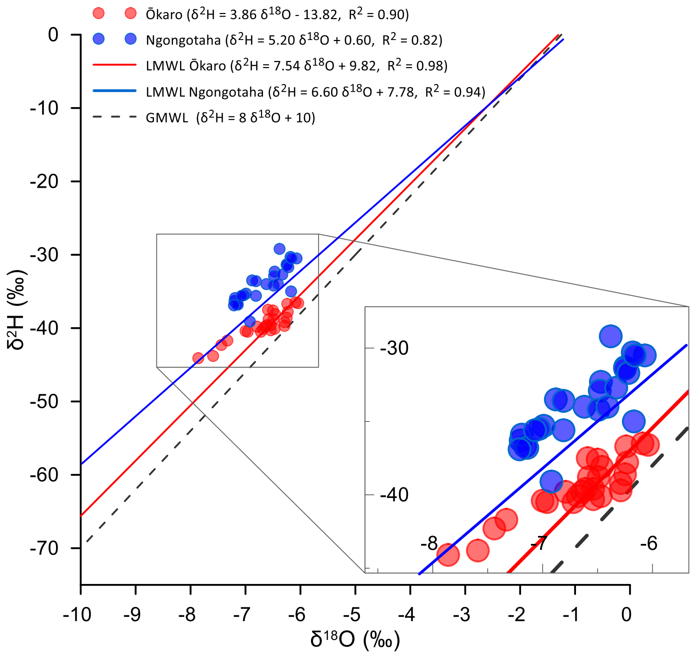

Figure 3. The distribution of stable isotopes in stream water for Lake Ōkaro and the Ngongotaha stream catchments is depicted. Regression equations and symbols are color–coded based on their association with the source of water samples. GMWL represents the global meteoric water line, and LMWL represents the local meteoric water line.

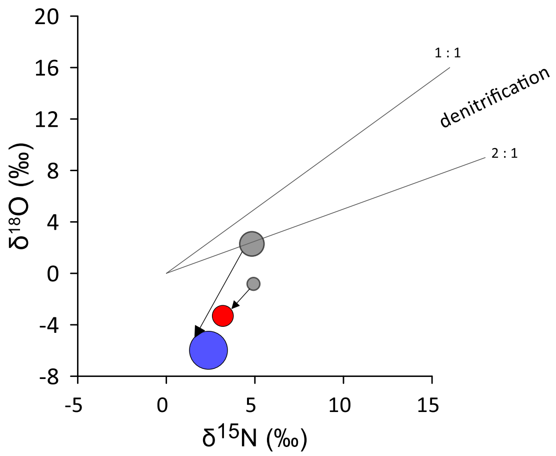

Figure 4. The plot of δ18O versus δ2H illustrates changes in nitrate isotopes from base flow (represented by a grey circle) to peak flow (represented by a colored circle) during rainfall events in July 2020. The red circles denote the Lake Ōkaro catchment, and the blue circles represent the Ngongotaha stream catchment. The movement along a 1:1 or 2:1 enrichment line suggests denitrification (solid line, Granger et al., 2008), referring to the stoichiometry of the reaction of nitrate that is reduced to nitrogen gas, indicating the molar ratios of nitrogen compounds involved in the denitrification process. The bubble size shows the proportion of nitrate–N concentrations in water samples ranging from 0.5 – 1.8 mg L-1.

3.3 Variation in hydrochemistry during events

The investigation into the variation in nitrogen concentration during the observed rainfall event yielded insightful findings. The hydrograph and chemograph overlay illustrated a lag time between rainfall initiation and corresponding peaks in nitrogen-related concentration, shedding light on the intricate dynamics of nitrogen mobilization within the watersheds (Figures 5a and 5b). Events in Ōkaro and Ngongotaha displayed slightly distinct responses during the ascending portions of the hydrograph; however, they exhibited comparatively similar trends during recession limbs following the end of rainfall. The peak flow in Ngongotaha was lagged of Ōkaro. Analysis of water isotope analysis indicates that augmented rainfall contributes to streamflow, particularly in instances of increased precipitation and wet antecedent hydrological conditions.

A discernible alteration in the concentration of N species coincided with the flow rising during rainfall events, emphasizing the complexity of nitrogen transformations during hydrological events. The data exhibited a nuanced temporal pattern in both sites, showcasing fluctuations in nitrogen concentrations and predominant TN compound throughout the event. Total nitrogen (TN) concentration ranged from 0.61 to 2.6 mg L^{-1} and 1.14 to 3.4 mg L^{-1} for the Lake Ōkaro and the Ngongotaha stream catchment, respectively. Overall, nitrate–N constituted the major portion of total nitrogen in stormflow in the Ōkaro site (50.44%) and in the Ngongotaha site (62.78%), and peaks in nitrate concentration were observed at specific intervals, especially during the rising limb of the hydrographs.

The time-series analysis of isotopic compositions of nitrate revealed dynamic shifts in response to changing precipitation patterns. Notably, a discernible increase in nitrate–N levels as flow increased coincided with the depleted values of nitrate isotopes, suggesting the potential nitrate sources for wash-off from surfaces. An apparent pattern was identified: the peak fraction of anticipated nitrate load originating from urine-urea sources coincided with the commencement of rainfall. Similar observations of depleted $\delta^{15}N$ with increased flow generation have been documented in other studies (Yue *et al.*, 2014; Kaushal *et al.*, 2011). The shift towards lighter $\delta^{15}N–NO_3^–$, coupled with relatively higher nitrate concentrations, suggests the oxidation of N–species with isotopically more lightweight compositions to nitrate (such as the nitrification of ammonium fertilizers and urine sources), as reported in other studies (e.g., Lin *et al.*, 2019).

Electrical conductivity (EC) exhibited swift reactions to the intensity of rainfall events, and the responses differed based on catchment conditions. In the Lake Ōkaro catchment, there was a rapid decline in EC during the event, indicating that the stormflow predominantly consisted of 'new' water (from rainfall), and EC returned to its pre-event level after reaching the peak of rainfall. In contrast, the event in the Ngongotaha stream catchment displayed an initial rise in stream water EC in response to the onset of rainfall.

A substantial variation in hydrochemistry between Ōkaro and Ngongotaha highlights the close connection between precipitation and stream hydrochemistry. Overall, these findings emphasize the necessity for thoroughly comprehending nitrogen's spatial and temporal patterns during rainfall events. This understanding is crucial for effective water quality management and well-informed land-use planning.

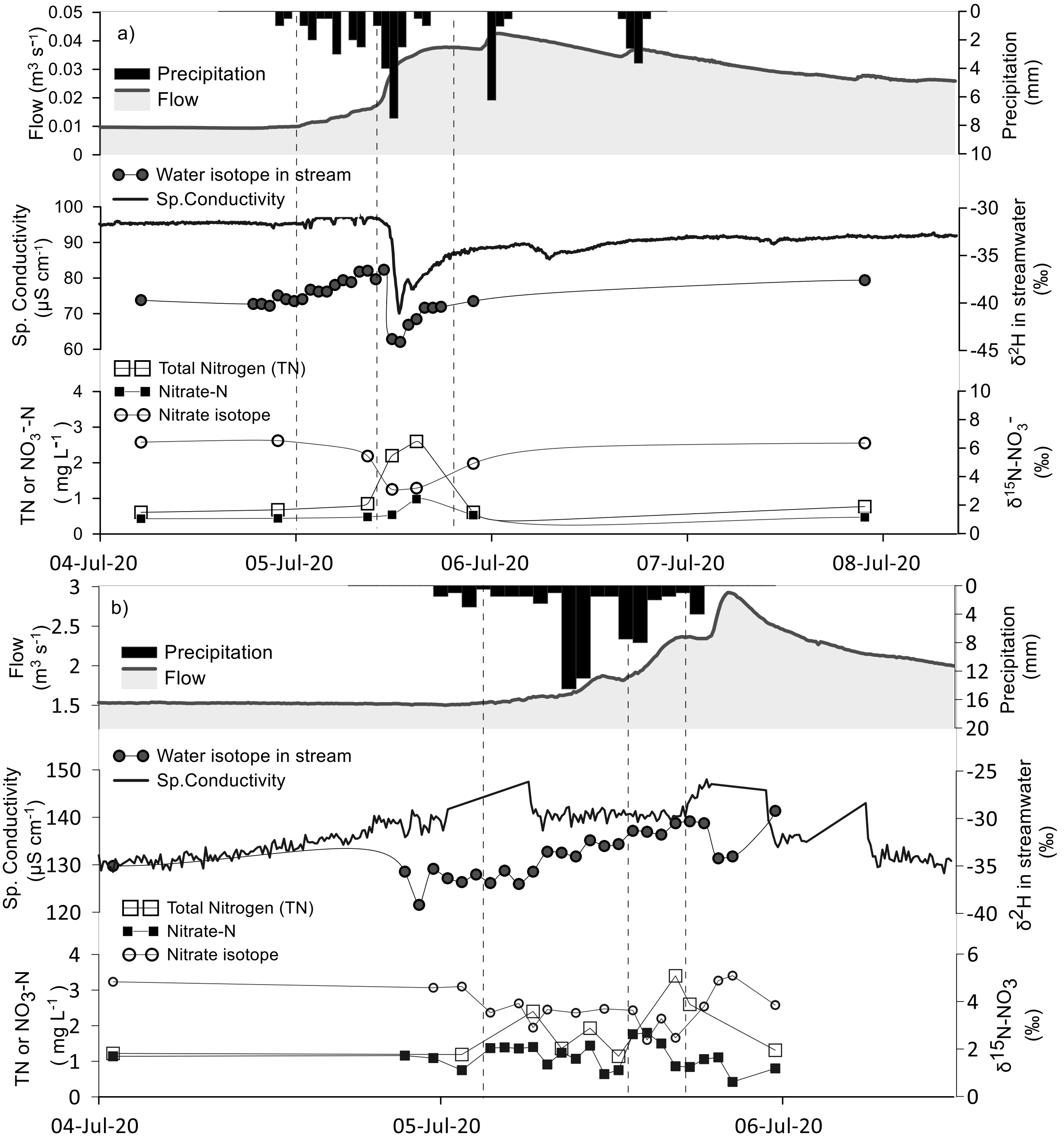

Figure 5. Temporal dynamics of hydrochemistry in July 2020 for Lake Ōkaro catchment (a) and Ngongotaha stream catchment (b). The dotted line highlights the dynamics of nitrate concentration, which increases when there is an increase in flow and decreases when the flow reaches its peak.

3.4 Implications for lake management

Understanding the processes of nutrient delivery in agricultural (i.e., pastoral) catchments is crucial for predicting and addressing excessive nitrogen (N) in surface waters (Galloway *et al.*, 2008). This study illustrates that heightened nutrient export is typically observed in wetter antecedent hydrologic conditions, primarily due to an amplified response to rainfall and elevated nutrient concentrations. This is likely a result of enhanced connectivity with the surface. The study underscores the importance of a more nuanced understanding of the influence of rainfall characteristics on the temporal dynamics of hydrochemical export in a pastoral environment. A broader implication is the recognition that rainfall and pre-rainfall conditions significantly determine nutrient export patterns.

The elucidation of the association between nitrogen concentration and runoff flow rate has advanced our comprehension of the joint influence of catchment hydrology and land use on nitrogen inputs into streamwater and downstream aquatic systems. Diverse factors, encompassing temperature shifts, alterations in vegetation structure, rate of fertilizer application, and the processes of nitrification–denitrification, have aggregated throughout all contributing flow paths (Duncan *et al.*, 2017), contribute to the complexity of nitrate concentration discharge variations (Lloyd *et al.*, 2016). The dominant controlling factor for in-stream nitrogen transformations is the seasonal variability in hydrological conditions. For example, Wong *et al.* (2018) demonstrated that nitrate concentration arising from agricultural activities exhibits significant elevation during increased precipitation.

Identifying nitrogen load peaks and their linkage to sources through intensive sampling implies the essential nature of high-resolution monitoring for a robust estimation of nutrient loads, particularly in assessing the impact of rainfall on water quality. These findings emphasize the benefits of transitioning from monthly monitoring to increased high-resolution event samplings, shedding light on the intricate relationship between rain events, catchment response, and the characterization of nitrogen loads in regions with intense rainfall on porous soils.

In cases where nitrate contamination is a concern, isotopic analysis can aid in identifying nitrate sources, providing crucial information for implementing targeted management practices to mitigate nutrient loading. This research adds to initiatives to regulate nutrient inputs and contamination control in the eutrophic lake. It emphasizes the benefits of integrating isotopic sampling into monitoring initiatives. Subsequent research endeavors focusing on the event-driven transport and transformation of nitrogen species can advance our comprehension of hydrological fluctuations and nitrogen cycling within the catchment of these eutrophic lakes. The broader significance of this study pertains to informing water quality management strategies designed to mitigate nitrogen losses from pastoral catchments. Implementing management practices to address excess nitrogen associated with urine and urea nitrogen deposition on pastures can substantially enhance water quality.

4. Conclusion

The findings from the study, which investigated nutrient dynamics in two agricultural catchments contributing to nutrient enrichment in lakes, underscore the critical role of understanding the complexities of nutrient transport during rain events. Hydrograph analysis, stable isotope tracers, and high-frequency hydrochemical data contribute to a more holistic view of nutrient transport processes. This knowledge is essential for formulating effective and timely mitigation strategies, ultimately aiding the sustainable management of agricultural landscapes and preserving water quality in associated water bodies.

Data availability statement

The data included and used in this study is not confidential and available upon request.

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Conflict of interests

The authors declare no conflicts of interest.

Author contribution

MY and **RM** contributed to data collection and analysis. **MY** wrote the manuscript and **RM** provided critical feedback and revision.

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The diversity and use of dwarf swamp forest vegetation in a tropical floodplain lake in West Kalimantan, Indonesia

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Abstract: To alleviate the consequence of severe biodiversity threats, fifteen national priority lakes to be rehabilitated have been declared in Indonesia. Lake Sentarum National Park (LNSP), one of the lakes, is a distinctive floodplain lake that exhibits significant vegetation and flora diversity. One particular ecosystem type in the area is dwarf swamp forest vegetation, which displays enormous amounts of floral vegetation in that area. This study intends to uncover vegetation data in the dwarf swamp forest habitat, which currently has relatively limited data series about its diversities. The vegetation specimens were collected using a 50 x 50 m line transect. Fourteen species from ten families were collected from six observation stations. The most prevalent vegetation is *C.* cf. *ensifolius*, *I. mentangis*, and *B. acutangula*. The species diversity index (H') is 1.78 (low category), and the small lake area has a greater species variety than the river area in the LNSP area. Furthermore, we found that dwarf swamp forest vegetation is mainly used as food for *Apis dorsata* honey bees to produce forest honey. The findings of this study will be helpful as a fundamental reference for future monitoring, research, and conservation efforts in the park.

Keywords: floodplain, Lake Sentarum, National Park, species diversity, dwarf swamp forest

1. Introduction

Indonesia, as a mega biodiversity country in the world, has abundant and unique biodiversity hotspots (Rintelen *et al.,* 2017). One of the spots is the tropical swamp forests located in Kalimantan (Muchlisin *et al.*, 2015). The swamp forests also play a significant role in preventing floods during rainy seasons and supplying fresh water for agriculture and aquacultural activities and settlements (Djufri *et al.*, 2016).

Research on flora biodiversity, especially the composition of vegetation, is essential as baseline data to plan better conservation and monitoring strategies. However, the data on plant species diversity in Kalimantan's DSF still needs to be upgraded, as the last research was conducted almost a decade ago (e.g., Randi *et al.* 2014). More importantly, according to our best knowledge there are scarce studies on the biodiversity of swamp forests, particularly in tropical areas.

In our case study, we selected a dwarf swamp forest (DSF) in Lake Sentarum National Park (LSNP), West Kalimantan, Indonesia, because it is located in one of fifteen national conservatory priority lakes in Indonesia (Ministry of Environment and Forestry/ MoEF, 2011). Further, it is also selected as it exhibits

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a wide and unique diversity of vegetation or flora (Anshari *et al.,* 2002). DSF is a type of vegetation composed of small trees and shrubs, about 5-8 meters tall, that can be inundated for up to eleven months each year (on average 9.5 months), with a maximum water level of 5.5 meters (Balai TNDS, 2008). Due to this unique and dynamic environmental condition, the forest is made up of several plant species that have adapted to areas that are almost always flooded with water throughout the year (Anshari *et al.*, 2002).

This study intends to uncover vegetation data in the DSF in LSNP area, where knowledge still needs to be discovered. Our research can fill the aforementioned knowledge gap as well as provide a theoretical contribution to the science of plant biodiversity. Furthermore, this study provides upgrades on the most recent biodiversity data in the study area, which can aid its monitoring and conservation efforts.

2. Methods

2.1. Study Site

LSNP is one of Indonesia's floodplain lakes classified as a Ramsar site since 1994 (Giesen, 1995; Anshari *et al.*, 2002). The forest area covers approximately 2.362 ha or 1.81% of the overall area of LSNP (Giesen & Anshari, 2016). The forest vegetation in the area has a relatively high diversity of species, and many of these species are endemic and specific to this forest ecosystem type (Randi *et al.*, 2014).

A number of endemic plant species can be found in LSNP, including *Casaeria* sp (limut), *Croton cf ensifolius* (melayak), *Dichilanthe-borneensis* (berus), *Eugeissona ambigua* (ransa), *Helicia cf petiolaris* (putat rimba), *Korthalsella cf germinans* (paha buntak), *Microcos cf stylocarpus* (tengkurung asam), *Rhodoleia spp* (insang dungan), *Ternstroemia cf toguian*, and *Vatica cf umbronata* (menungau) (Giesen, 1996).

As a floodplain lake, LNSP experiences frequent floods during wet seasons, when around 80 lakes are charged (Anshari *et al.*, 2001). However, during dry periods, the lakes are either partially or entirely emptied or completely (ibid). This unique condition creates a network of interconnecting seasonal lakes surrounded by swamp forests, peat swamp forests, and dry lowland forests on isolated hills (Andryannur *et al.*, 2022). Further, the specific hydrological patterns make the lakes contribute substantial parts to the water level of its main river, the Kapuas River (Hidayat, 2018; Hidayat, 2017).

Based on the average canopy heights, the forest ecosystem in LSNP is divided into seven types: dwarf swamp forest/rampak gelagah (Figure 1), short swamp forest/gelagah (kenarin-menungau-kamsia reed forests and kawi-kamsia reed forests), tall swamp forest/pepah forests (kelansau-kelansau pepah forests), and tall swamp forest/pepah forests (pepah kelansau-emang-melaban and pepah ramin-mentangur kunyit forests), riparian forests, hill forests, kerangas forests, and former agricultural land (Balai TNDS, 2008). The plant species diversity in each habitat is modest, but the overall plant diversity is very high, with a total of 262 species identified (Giesen, 2000).

2.2. Data Collection

Three 50 m x 50 m transects were set in six sampling sites in August and September 2016. The sites were tagged as Seriang River (Station 1), Majang Lake (Station 2), Tekenang (Station 3), Penyelawat Peat River (Station 4), Genali Lake (Station 5) and Sumbai Lake (Station 6). The sites were chosen using a purposive random sampling method, with locations representing different ecosystem types:

- The river area (Seriang River)
- The peat area (Penyelawat Peat River)
- The area of small lakes (Lake Majang, Lake Genali, and Lake Sumbai)
- The lake's middle area (Tekenang)

The numbers of representative sampling locations are different in each sampling time due to the water inundation level during the wet and dry seasons (Table 1).

Besides that, an interview process was also carried out in this research. A series of face-to-face interviews were conducted with the aid of four local field assistants and LSNP officers (4 people). Interviews were conducted to verify the laboratory-identified plant types locally and to collect information on the local use of the plants.

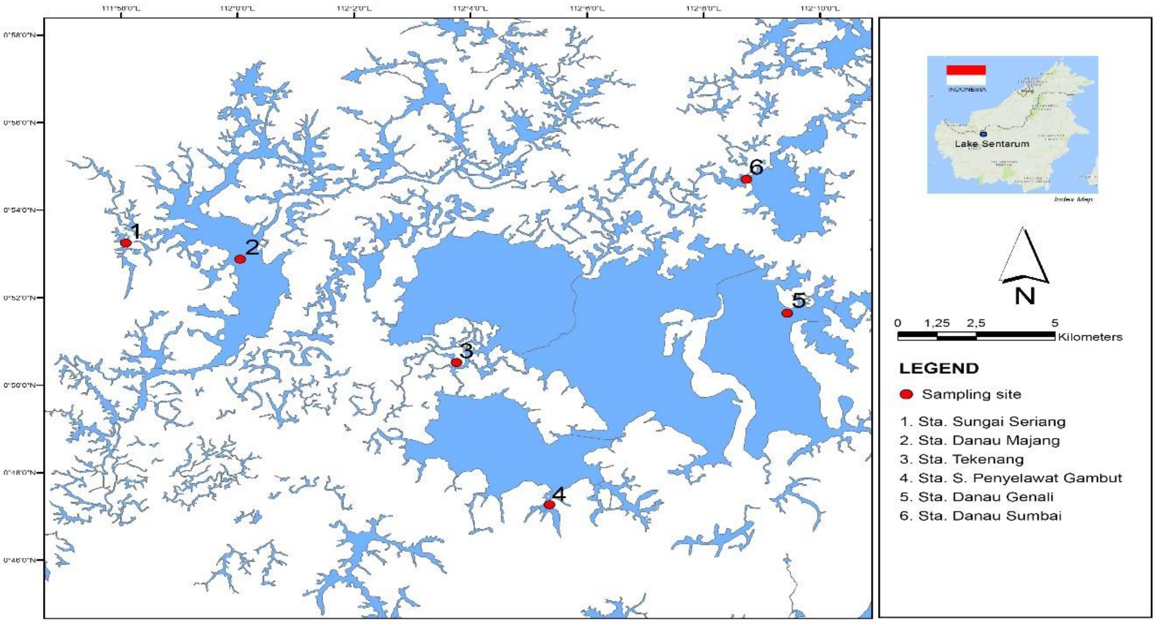

Figure 1. Sampling locations for dwarf swamp forest vegetation in the LSNP

Table 1. Sampling locations

Station	Sampling location	Geographic Coordinates	Ecosystem Type
1	Seriang River	00° 53′ 248″ N - 111° 58′ 076″ E	River area
2	Lake Majang	00° 52′ 785″ N - 111° 59′ 939″ E	Small lake
3	Tekenang	00° 50′ 512″ N - 112° 03′ 753″ E	Middle area lake
4	Penyelawat Peat River	00° 47′ 265″ N - 112° 05′ 347″ E	River area (Peat)
5	Lake Genali	00° 51′ 639″ N - 112° 09′ 431″ E	Small lake
6	Lake Sumbai	00° 54′ 701″ N - 112° 08′ 733″ E	Small lake

2.3 Data Analysis

At the observation site, the species, family, and number of individuals found for each plant sample were recorded. At the initial stage, the specimens are morphologically identified (flowers, leaves, and shoots) in the field. At the same time, we also recorded the number of individuals for each species. Then, the samples were made into herbariums and photographed using a digital camera. Next, the herbariums were sent to the Research Center for Biosystematics and Evolution - National Research and Innovation Agency (BRIN) for further identification.

Further, using the obtained data, we calculated the Important Value Index/ IVI (Equation 1) and the Shannon-Wiener Diversity Index/ H′ (Equation 2). The IVI is an index to represent the importance of a species shown by the level of the number of individuals, density, and frequency (Indriyani *et al.* 2017). The Index of Shannon-Wiener's Species Diversity (H′) was examined using the criteria according to Barbour *et al.* (1999) and Djufri (2002). The H′ value is categorized as: very high category (H′> 4), high category (H′≤ 3-4), medium category (H'≤ 2-3), low category (H'≥ 1-2), and very low category (H'< 1).

$$IVI = Species\ density + Species\ frequency\ of\ occurence \quad \text{....Eq. 1}$$

$$\mathrm{H}' = -\sum_{\mathrm{i}=1}^{\mathrm{n}} P_i (\ln \mathrm{pi}) \quad \text{....Eq. 2}$$

where:

H' : Shannon-Wiener Index of Species Diversity

P_i : Species-i relative abundance (ni/N)

n_i : Number of species-i

N : Total number of individuals

Meanwhile, species density and species frequency of occurrence are calculated using Equations 3 to 4 as follows:

$$\text{Species density (SP)} = \frac{\text{number of individuals for species} - \text{i}}{\text{the area of the sample plot}} \quad \text{.... Eq. 3}$$

$$\text{Relative density (RD)} = \frac{\text{density for species} - \text{i}}{\text{density all species}} \quad \text{....Eq. 4}$$

$$\text{Species frequency (SF)} = \frac{\text{number of plot samples of species}-\text{i is found}}{\text{the total number of sample plots}} \quad \text{.... Eq. 5}$$

$$\text{Frequency Relative (FR)} = \frac{\text{frequency of species}-\text{i}}{\text{the total number of all species}} x100\% \quad \text{....Eq. 6}$$

3. Results and Discussion

3.1. Vegetation Composition

There were fourteen plant species categorized into ten families that were collected in our study sites (Table 2). Among the identified species, most of them were sampled from Stations 2 and 5, followed by Stations 3, 6, 1, and 4, respectively. The most prevalent species were *C.* cf. *ensifolius*, *I. mentangis*, and *B. acutangula* revealed by the high percentage of individual numbers (25,63, 21,42 and 20,49%) and their occurrence in all sampling sites (Figure 2). The research results are in accordance with the work of Balai TNDS (2008) and Giesen (2016).

Further, the sites with the highest individual numbers were Sites 5 and 6 as opposed to Sites 1 and 4. Meanwhile, the sites with the highest numbers of species were Sites 2, 3, 5, and 6, as opposed to Sites 1 and 4. Thus, overall, the sites located in the small lake ecosystem type (sites 2, 3, 5, and 6) have higher diversities than the riverine ecosystem type (sites 1 and 4). This circumstance can be attributed to the mineral richness of the small lake ecosystem, which leads to high plant diversities (Furey & Tilman, 2021).

Another major finding that should be highlighted is that *C.* cf. *ensifolius*, *B. acutangula*, and *I. mentangis* were identified at all research stations (Figure 2). This is because these three plants are the species that generally structuring dwarf swamp forest formations (Balai TNDS, 2008). The research results are in accordance with what was found by Giesen (2016) elucidating that dwarf swamp forests are generally composed of *Croton* cf. *ensifolius* (melayak), *Ixora mentangis* (mentangis), *Barringtonia acutangula* (putat), *Memecylon edule* (kebesi), *Timonius salicifolius* (temirit), *Szygium claviflora* (kemasung), *Garcinia borneensis* (empanak), *Carallia bracteata* (kayu tahun), *Gardenia tentaculata* (landak), and *Pternanda teysmanniana* (gelagan).

We noted that the diversity of vegetation species in the dwarf swamp forest will have an impact on the water quality and biota. One reason underpinning this condition is that the plants that remain submerged in water for an extended period might give additional periphyton. Periphyton community growing on the submerged substrate in water could serve as a preferred natural food item for herbivorous and omnivorous fish (Biswas *et al.,* 2022). However, if the flood season is prolonged, it may cause the dying of the plants and their decaying may cause acidic waters (Utomo and Asyari, 1999). As a consequence, the fish biota that sustains in this environment is the type that is acid resistant and tolerant to low oxygen level; such as snakeheads and kissing gouramies (Xie *et al.*, 2017; Ahmadi, 2021).

In short, the dwarf swamp forests of LSNP exhibit unique ecosystem characteristics and support high plant and animal diversities, as hinted by Giesen and Anshari (2016). Furthermore, the ecosystem also operates as a buffer for the Kapuas River, absorbing ¼ of peak floods and maintaining water levels in the dry season by delivering 50% of dry season flow in the upper Kapuas River (ibid). Therefore, it is considered a valuable ecosystem to be preserved.

Table 2. The composition and distribution of dwarf swamp forest vegetation at each observation station in LSNP

No	Family/Species	Local Name	Station 1	Station 2	Station 3	Station 4	Station 5	Station 6	Total per Species	%
	Euphorbiaceae									
1	*Croton* cf. *ensifolius*	Melayak	216	427	138	261	1297	267	2,606	25.63
2	*Malotus sumatranus*	Belantik	-	-	1	-	-	-	1	0.01
	Lecythidaceae									
3	*Barringtonia acutangula*	Putat	393	10	17	838	1	824	2,083	20.49
	Rubiaceae									
4	*Ixora mentangis*	Mentangis	30	425	1,038	51	38	555	2,137	21.02
5	*Timonius salicifolius*	Temirit	320	146	124	19	28	-	637	6.26
	Melastomataceae									
6	*Memecylon edule*	Kebesi	-	20	-	-	1,897	13	1,930	18.98
	Myrtaceae									
7	*Szygium claviflora*	Kemasung	5	10	-	3	15	380	413	4.06
8	*Szygium durifolium*	Kayu ubah	-	4	-	-	-	2	6	0.06
9	*Szygium* sp.	Jijab	-	-	2	-	1	34	37	0.36
	Pandanaceae									
10	*Pandanus helicopus*	Rasau	-	20	-	-	200	-	220	2.16
	Rhizophoraceae									
11	*Carallia bracteata*	Kayu tahun	-	71	5	-	3	-	79	0.77
	Capparaceae									
12	*Crateva religiosa*	Punggu	-	-	2	-	-	9	11	0.11
	Clusiaceae									
13	*Garcinia borneensis*	Empanak	-	3	-	-	4	-	7	0.07
	Leguminosae									
14	*Crudia teysmannii*	Timba tawang	-	1	-	-	-	-	1	0.01
	Total number of individuals		**964**	**1,137**	**1,327**	**1,172**	**3,484**	**2,084**	**10,168**	**100**
	Total number of species		**5**	**11**	**8**	**5**	**10**	**8**		

(a) (b) (c)

Figure 2. The dominant vegetation of dwarf swamp forest in the LSNP area; (a) *Croton* cf. *ensifolius*, (b) *Ixora mentangis*, (c) *Barringtonia acutangula*

3.2. The importance and diversity of the vegetation

The calculated species frequency (SF), frequency relative (FR), species density (SD), density relative (RD), the important value index (IVI), and species diversity (H') (Table 3) present similar conclusions that *C.* cf. *ensifolius*, *I. mentangis*, and *B. acutangula* had the highest frequency/distribution. Further, we underline that *C.* cf *ensifolius* is an adaptive endemic species that can be found in all ecosystem types with a frequency value of 22.22 (Table 3). Meanwhile, other species are categorized as lower-range endemic species with a relative frequency value of less than 20. More importantly, *C.* cf. *ensifolius* is one of the most important pioneer species in LSNP (Giesen, 2000). At the same time, *C.* cf. *ensifolius*, *I. mentangis*, *B. acutangula*, and *M. edule* have the highest relative density values that show a strong pattern of adaptation to the environment.

Meanwhile, the obtained IVI (Table 3) demonstrates that *C.* cf. *ensifolius*, *I. mentangis*, *B. acutangula*, and *M. edule* have the highest IVI values. The plant species with high IVI values play a significant role in their communities (Asmayannur *et al.*, 2012). This result indicates that *C.* cf. *ensifolius*, *I. mentangis*, *B. acutangula*, and *M. edule* are the vegetations that have the most significant roles in the communities. On the contrary, *M. sumatranus* and *C. teysmannii* are non-dominant and poorly distributed at the observation sites.

At the same time, the calculated H' is 1,78, which indicates low category diversity (Barbour *et al.*, 1999; Djufri, 2002). This low diversity is because the vegetation found in the dwarf swamp forest type is just a few species in the LSNP area. A community is considered to have high species diversity if it contains numerous species. In contrast, a community is considered to have low species diversity if it contains just a few species. Setiawan *et al.* (2017) state that diversity is most valuable when all individuals come from different genera or types, while it is least valuable when all individuals come from the same genus or species.

Table 3. Values for dwarf swamp forest vegetation relative frequency, relative density, important value index, and species diversity index

Family	Species	SF	FR	SD	DR	IVI	H'
Euphorbiaceae	*C.* cf *ensifolius*	1.00	22.22	0.06	25.63	47.85	0.35
	M. sumatranus	0.06	1.23	0.00	0.01	1.24	0.00
Lecythidaceae	*B. acutangula*	0.67	14.81	0.05	20.49	35.30	0.33
Rubiaceae	*I. mentangis*	0.67	14.81	0.05	21.02	35.83	0.38
	T. salicifolius	0.50	11.11	0.01	6.26	17.38	0.17
Melastomataceae	*M. edule*	0.28	6.17	0.04	18.98	25.15	0.32
Myrtaceae	*S. claviflora*	0.39	8.64	0.01	4.06	12.70	0.13
	S. durifolium	0.11	2.47	0.00	0.06	2.53	0.00
	Szygium sp.	0.17	3.70	0.00	0.36	4.07	0.02
Pandanaceae	*P. helicopus*	0.22	4.94	0.01	2.16	7.10	0.08
Rhizophoraceae	*C. bracteata*	0.17	3.70	0.00	0.78	4.48	0.04
Capparaceae	*C. religiosa*	0.11	2.47	0.00	0.11	2.58	0.01
Clusiaceae	*G. borneensis*	0.11	2.47	0.00	0.07	2.54	0.01
Fabaceae	*C. teysmannii*	0.06	1.23	0.00	0.01	1.24	0.00
Total		**4.50**	**100**	**0.26**	**100**	**200**	**1.78**

SF = Species frequency, FR = Frequency Relative, D = Density, DR = Density Relative, IVI = Important Value Index, H' = Species Diversity Index

3.3. The use of the vegetations

The diverse uses of LSNP's plant species include wild animal consumption, human consumption, building materials, medicine, and other applications, with honey production as the most distinguished feature (Table 4). The results align with the conclusion of Sufardi (2015) that swamp woods give both ecological and economic benefits to the community, such as honey producers. In the study area, the use of forest honey is a common community economic activity in the LSNP area, particularly in traditional zones, in an effort to meet traditional community needs (Andryannur *et al*., 2022). Further, West Kalimantan is known as one of Indonesia's top producers of forest honey, with LNSP as the central producer (Kotimah *et al*., 2023). The abundance of lofty trees as a favorable home for forest honey bees (*Apis dorsata*) (Figure 3) endorses this benefit (Wardhani *et al.* 2022; Wijayanti *et al*., 2022; Ratnasari *et al*., 2022).

Table 4. The uses of dwarf swamp forest vegetation in LSNP

No	Species	Usage
1	*C.* cf*. ensifolius*	Produces forest honey
2	*B. acutangula*	Produces forest honey, medicinal plants
3	*I. mentangis*	Produces forest honey, medicinal plants
4	*T. salicifolius*	Building materials (piles)
5	*M. edule*	Produces forest honey, medicinal plants
6	*S. claviflora*	Produces forest honey
7	*S. durifolium*	Human consumption (fruit), medicinal plants, building materials (boards)
8	*Szygium* sp.	Wildlife consumption (the fruit is eaten by fish), a place for fish breed
9	*C. bracteata*	Produces forest honey
10	*C. religiosa*	Human consumption (leaves), wildlife consumption (flowers eaten by proboscis monkeys), animal feed
11	*G. borneensis*	Wildlife consumption (the fruit is eaten by fish)
12	*M. sumatranus*	Produces forest honey
13	*C. teysmannii*	Produces forest honey

(a) (b)

Figure 3. The uses of dwarf swamp forest vegetation by fauna in the LSNP area: a. Wild honey bees (*Apis dorsata*) as producers of forest honey, b. Fish nests as breeding places during high water/floods (in *C.* cf. *ensifolius* or *S. claviflora* plants) (source: www. jungledragon.com; Authors' personal documentation)

Besides serving as honey producers, the flora is also used as medicinal herbs. *B. acutangula*, *I. mentangis*, *M. edule*, and *S. durifolium* are among the medicinal plants found in DSF (Table 5). Moreover, around 40 species of medicinal plants are in the LSNP area, and they are used as medicinal plants by the locals (Balai TNDS, 2008; Ginting *et al.*, 2017).

Table 5. The uses of dwarf swamp forest vegetation as medicinal plants

No	Species	The utilization
1	*B. acutangula*	Scab medicine (shoot)
2	*I. mentangis*	Scab medicine (shoot)
3	*M. edule*	Canker sore medicine (fruits)
4	*S. durifolium*	Stomach medicine (fruits)

4. Conclusions

The dwarf swamp forest in LSNP supported low-level plant diversity, with several important species such as *C.* cf. *ensifolius*, *I. mentangis*, *B. acutangula*, and *M. edule*. Higher plant diversity occurred in small lake-type ecosystems and mainly were used as honey bee homes and medicinal sources. Considering its importance for both ecological and economic benefits, the preservation of the forest is crucial. This study arranges basic information for the development and monitoring of appropriate conservation planning. It can be applied as an input in the biodiversity data bank that can be used as a reference for future monitoring and research.

Data availability statement

Data used in this study can be requested from the corresponding author.

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Conflict of interest

The authors declare there is no conflict of interest.

Authors contribution

RK, ES, MSS, and **PL** (the main contributors): idea concept, data collection, conceptualization, data analysis, and writing the original draft. **EP, DO, AW, IGAAPP,** and **DA** (the supporting contributors): review and editing and writing methodology.

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Mitigating atmospheric methane emissions from Asian rice fields: a review of potential and promising technical options

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Abstract: Agriculture serves as a significant anthropogenic source of methane emissions. Numerous recent studies have examined the factors influencing methane emissions and have developed emission models. However, there is no a bridging review study related to methane emissions in Asia as one of the primary methane emitters. This review is divided into two manuscripts. In this first manuscript, I explore the process of methane emission and the factors that impact methane production and emissions. Meanwhile, the present state of studies conducted in various Asian countries and knowledge gaps are elaborated in the second manuscript. I elaborate several factors that influence methane production and their roles in the emission process. Further, I highlight that the gas is mostly produced in zero oxygen condition, although, a little concentration of methane also can be generated in oxic condition. This finding provides basic knowledge that contribute to the future research on methane emissions in rice field ecosystems. Eventually, I also explore various recommended technical solutions to reduce the gas emission.

Keywords: Methane, GHG emissions, rice fields, agriculture, Asia

1. Introduction

Methane is the second most significant greenhouse gas after carbon dioxide (van Dingenen et al., 2018). It is the primary hydrocarbon in the atmosphere that could significantly affect the earth's temperature due to its ability to absorb and emit infrared radiation up to 30 times higher than carbon dioxide within its short time in the atmosphere (Chen, 2021; Mer et al., 2001; Neue et al., 1996; Topp & Pattey, 1997; van Dingenen et al., 2018). About 530 million tons are released into the atmosphere every year (Ito, 2015). About 70% of the methane emission sources are emitted from anthropogenic factors such as agriculture, mining, natural gas uses, and other sources (Choi et al., 2017; Khalil et al., 1993; Mer et al., 2001; Minamikawa et al., 2006; Topp & Pattey, 1997). Agriculture is the main anthropogenic source, as domesticated ruminants and rice fields are responsible for up to 40% of the methane emissions (Mer et al., 2001).

In Asia, the majority of methane emissions are emitted from agriculture, specifically rice fields. About 90% of rice fields are inundated, and most of the rice production in the world comes from Asia (Wassmann et al., 2009). Flooded soil will allow methanogens to produce methane under anaerobic conditions (Ariani et al., 2021). As a response to the rapid growth of its population, methane emissions from the rice fields are also increasing in the region.

Various studies regarding factors affecting methane emissions to emission modeling have been produced recently (Schulz et al.,1997; Van Dengenen et al., 2018; Zhu et al., 2018; Conrad, 2020; Gwon et al., 2022; Mboyerwa et al., 2022; Ouyang et al., 2023). However,

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there is no comprehensive review that frames previous studies to elaborate on the recent knowledge gap in this topic in particular in Asia as the biggest methane producers. To fill the gap, I write this review, which includes the methane emissions process, factors that affect production and emissions, and the present state of the research on methane in Asian countries. Due to page limitation, the review is divided into two parts, with the first manuscript (this paper) includes the first two topics, and the second review comprises the third topic. This paper provides critical aspects to be addressed in future research to comprehensively understand global methane emissions and their environmental implications, particularly in Asian countries.

2. Method

The references were collected from the search engines such as Google Scholar and Web of Science (Clarivate) with several keywords, such as methane and climate change issues, methanogenesis, factors that influence methane emission in the aquatic ecosystems, and methods in methane research in the rice field ecosystems. I collected hundreds of literature that were refined with several criteria, such as Asian countries and rice field ecosystems without a specific time period. Finally, the results were compiled in the Mendeley reference manager.

3. Methanogenesis in the rice field ecosystems.

Methane is formed as a final product of the reductive process done by methanogens (archaea and bacteria) (Minamikawa et al., 2006) under anaerobic conditions due to flooded rice fields. Methanogens are strict anaerobic archaea and obligate methane producers that break down one-carbon compounds (carbon dioxide, carbon monoxide, methanol, methylamines, and methyl sulphides), acetate or coal to methane gas through one of several methanogenesis pathways (Buan, 2018). However, some methanogenic archaea in soils can tolerate oxygen for a short period; methane production is also reported to occur in oxic conditions (Wagner, 2017). A comparison study between aerobic and anaerobic conditions in the saturated soil to methane production rate showed that anaerobic conditions were able to produce methane at 66% higher than the methane production rate in aerobic conditions. Methane production in aerobic conditions is only half compared to anaerobic conditions at a water potential of -6 and -30 kPa. Therefore, it is understandable that methanogenesis in the flooded soil, as well as other wetlands, becomes rapidly limited by oxygen.

The carbon substrates used in the methanogenesis process are derived from the exudates and the sloughed tissues of the plants and soil organic matter (Minamikawa et al., 2006). Most methanogens are from domain archaea, with some species from anaerobic bacteria. There are two types of archaea based on the compound that they consume to produce methane: hydrogenotrophic methanogens (Methanosaeta and Methanosarcina) and Acetoclastic methanogens (Methanobacterium, Methanobrevibacter, Methanosprillum, Methanoculleus and Rice Cluster I (RC–I). The last type is a novel cluster of archaeal 16S rRNA genes sequenced from rice plant roots. Methanosarcinaceae, Methanobacteriales, Methanomicrobiales, and RC-I were mostly found in the soil of rice fields in China, the Philippines, Japan, and Italy (Conrad, 2007). In total, there are at least seven genera of archaea found living in rice fields (Mer et al., 2001; Minamikawa et al., 2006).

Methanogenesis (Figure 1) involves methanogenic species whose majority use acetate as C and an energy source to create methane, with less than a third of it requiring hydrogen as the energy source (Mer et al., 2001). There are two mechanisms used by methanogens to create methane in the rice field: acetate (acetoclastic methanogenesis) and hydrogen molecules (hydrogenotrophic methanogenesis). These mechanisms are able to help methanogens convert the molecules into methane with carbon dioxide and methane plus water (Conrad, 2007).

Soil redox potential (Eh), soil pH, and temperature are key factors that influence the methanogenesis process in the rice field because methanogens are generally mesophilic species that are active in temperatures ranging from 30 to 40 °C (Wihardjaka, 2016). Therefore, the decrease in temperature will result in low methane production (Topp & Pattey, 1997). Besides that, organic matter application will reduce Eh from below -200 to 150 mV and carbon sources supply. Generally, methane production begins at ± 150 mV (Minamikawa et al., 2006; Wihardjaka, 2016) by methanogens due to anaerobic conditions in flooded soil.

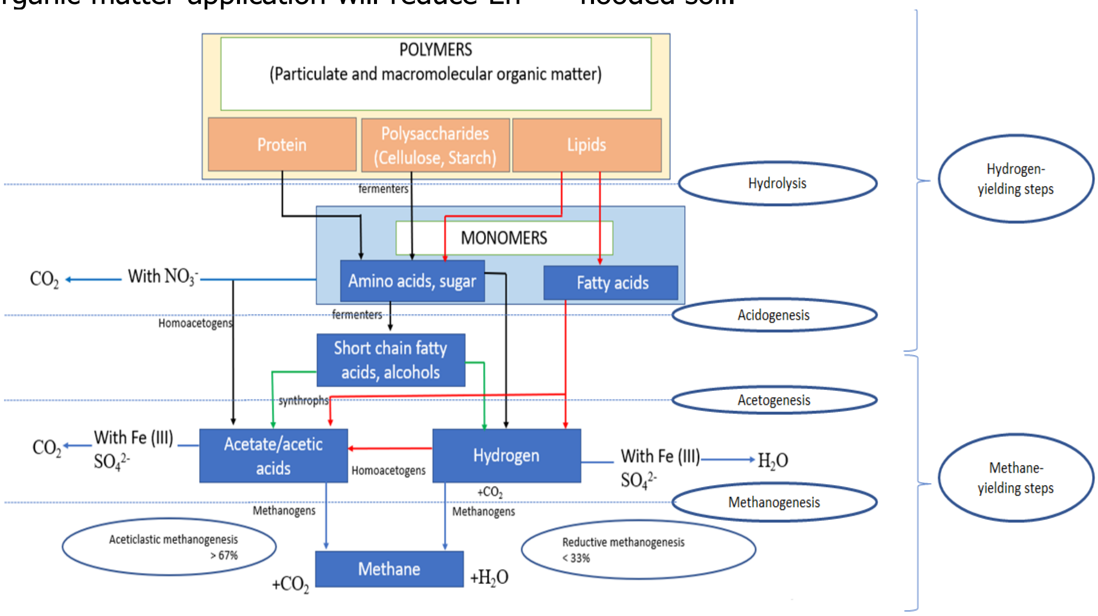

Figure 1. Pathways of the methanogenesis process (Source: Author's creation based on the modification of Conrad (2007), Parawira (2004), and Sikora et al. (2017)).

Some parts of the formed methane resulting from the process are consumed by methanotrophs under oxidative conditions in the roots of paddy or the aerobic organic layer of the soil (Minamikawa et al., 2006; Mer and Roger 2001; Mboyerva et al., 2022). Meanwhile, the rest of it is emitted into the atmosphere in 3 ways: diffusion, ebullition, and paddy's aerenchyma (Ariani et al., 2021; Topp & Pattey, 1997; Wihardjaka, 2016). Further, the majority of the gas is released through the paddy's aerenchyma (20-90%), while ebullition and diffusion are only 2-9% and 0.01-1%, respectively. About 0.1-4% methane infiltrates into the deep layers of soil, and the rest of it is oxidized by methanotrophs in the aerobic layers and is transformed to carbon dioxide (Fig. 2). Methanotrophs in the rice field soil belong to the genera Methylocystis, Methylosinus, Methylobacter, Methylomicrobium, Methylomonas, Methylocaldum, and Methylococcus (Minamikawa et al., 2006; Rahalkar et al., 2021). The population usually enlarges with the growth of paddy, however, irrigation and fertilizer do not affect the growth population of methanotrophs (Minamikawa et al., 2006).

There are four main habitats for both methanogens and methanotrophs in the rice field: the bulk soil, organic plant debris, rice plant roots, and shallow aerobic surface layer in the flooded rice field (Conrad, 2007). These habitats generally get organic matter sources to create methane from rice straw since about 80% of methane emissions come from it, decomposed root cells, and degradation of soil organic carbon (Conrad, 2007; Conrad, 2020). Studies showed both methanogens and methanotrophs can maintain their population under disadvantageous conditions, such as drainage for anaerobic methanogens and flood conditions for methanotrophs (Mer et al., 2001). In other words, methanogens and methanotrophs are interconnected through their roles in the methane cycle, where methanogens produce methane. In contrast, methanotrophs consume and mitigate their release into the atmosphere, which is influenced by water regimes.

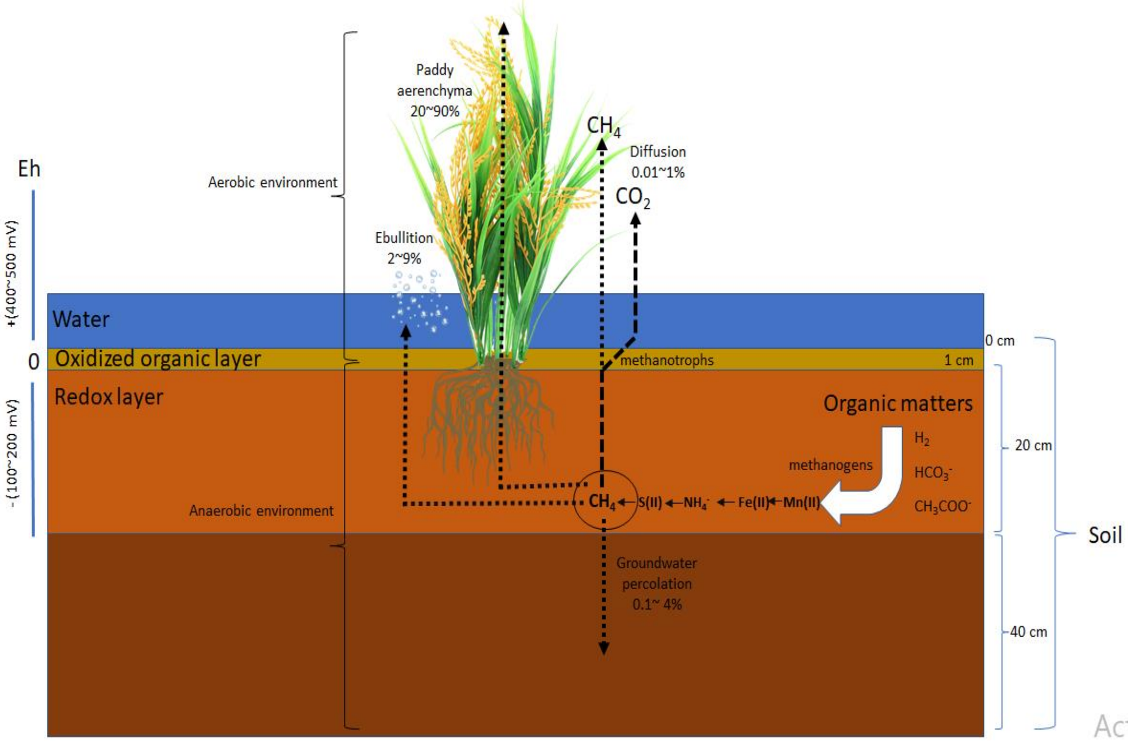

Figure 2. Methane production and emission in the flooded rice field (Source: author's creation based on the modification based on the diagram published by Ariani et al. (2021), Ito (2015), Topp & Pattey (1997), and Wihardjaka (2016))

4. Factors Influencing Methane Emission from Rice Fields: Role of Rice Cultivar and Environmental Conditions

Several factors that affect methane emissions from rice fields comprise rice cultivars, cultivation processes, climate/weather, and soil characteristics.

4.1 Rice cultivars

About 90% of methane is emitted from paddy aerenchyma; therefore, rice cultivation is one of the critical points that affect methane emissions from rice fields to the atmosphere. The formed methane in soil diffused to the passage through aerenchyma to evaporate. Aerenchyma in leaves, roots, and culms allows efficient gas exchange between the atmosphere and anaerobic soil. Moreover, the degrading roots and exudates are important carbon sources in methanogenesis (Neue et al., 1996). Therefore, the right choice of rice variety can lead to reduced methane emissions from soil. Many studies proved that rice-planted soil has higher methane emissions than fallow soil. The ability of rice to form methane is strongly dependent on aerenchyma cavities, number of tillers, rice biomass, root pattern, oxidizing ability, exudates, and microbial activities surrounding the roots (Arianti et al., 2022; Mer et al., 2001; Setyanto, 2006; Wihardjaka, 2016). Methane emission rates from paddy plants greatly depend on their traits because some varieties could emit less methane while producing a high-yielding crop (Ariani et al., 2021).

In a warmer environment, rice plants tend to produce more leaves area as well as tiller numbers to do photosynthesis, which leads to increases in emissions. GHG emission is positively correlated with leaf area, leaf number, tiller number, and root dry weight (Ariani et al., 2021). In addition, the amount of water and nutrients in the soil also affects the amount of GHG emissions, as plants need it to grow. Moreover, root exudate also affects the emission; the cultivar that has more arrangement of roots will form more methane because the exudate of roots acts as one of the carbon sources in methanogenesis (Wihardjaka, 2016). Finally, the length of the growing season also affects the amount of GHG emissions, as plants produce more GHG during longer growing seasons (Lu et al., 2000). Rice cultivar, which has more roots and higher stems because of the length of its growing period, tends to have more methane emission (Mer et al., 2001). Therefore, breeding new rice cultivars with low methane emission and high yield through traditional or biotechnology breeding techniques is one of the best ways to reduce methane emissions.

4.2 Cultivation process

The dynamics of methane emissions in different ecosystems are influenced by gas diffusion in relation with CH_4 transfer, microbial activities, methanogenesis process, and methane-mono-oxygenase activity (Mer et al., 2001). Most reports stated that methanogenesis takes place in anaerobic environments such as aquatic sediments, animal guts, flooded soils, peatlands, and coastal wetlands. The process is connected to the water availability during the flooding and draining periods when the soil is seasonally submerged (Conrad, 2020). As an example, methane emissions in the peatland and gley marsh decreased after the water was lowered to -10 cm, and after 7 days of drying, emissions in the peatland and gley marsh depleted by 28% and 10% respectively (Zhu et al., 2018). In addition, a study in rice growth related to the water regime showed that a significant level of methane emissions occurred soon after transplanting in the wet soil. This condition is mainly because of the increasing of trapped methane through diffusion in aerenchyma young rice plants (Trolldenier, 1995).

Further, a study in Korean upland rice fields found that intermittent irrigation reduces 25.1% GHG emissions compared to continuous flooded treatment (Lee et al., 2020). Flooded soil also lowers redox potentials that restrict the aerobic zone around the roots and allows anaerobic organisms to decompose the resulting exudate and debris to methane (Trolldenier, 1995). In Southeast China, methane emissions were reported to be 61% lower with intermittent draining systems than in continuous flooding irrigation systems, with no significant differences in rice biomass and grain yields (Lu et al., 2000; Wihardjaka, 2016). Overall, it shows that the water regime in an area plays an essential role in methane production as redox potential would deplete in a submerged environment, and it allows methanogens to proliferate in anoxic conditions. Proper drainage of the soil during the growing season could be an effective method without compromising the yields.

Another cultivation process that affects methanogenesis is organic matter, which is often used to increase the yield of rice. Some amendments such as rice straw, green manure, farmyard manure, and compost are more abundant in organic carbon than in chemical fertilizers (Mer et al., 2001). It is known that the rate of methane emission depends on the amount, kind, and prior treatment of the organic components; thus, methane emissions are highly dependent on the amount and condition of readily decomposable carbon contained in the additional substances (Sass, 2003). The application of organic amendments triggers methanogenesis, nitrification, and denitrification since the anaerobic fermentation of organic amendment produces an assemblage of organic matter that is not found in the oxic environment (Ariani et al., 2021; Neue et al., 1996). Organic amendments lower soil Eh and provide carbon sources for increasing menthane production in flooded soil (Wihardjaka, 2016). It is proven that in a hectare of rice field, five tonnes of rice straw could increase methane emission 10-fold compared with mineral fertilizers (Neue et al., 1996). In addition, silicate and zeolite could reduce GHG emissions by 14.1% and 21.7% in the rice field and additional salt, sulfate, N, Fe, gypsum, Mn4+, and SO42- -containing fertilizers also could reduce methane emissions (Lee et al., 2020; Minamikawa et al., 2006; van der Gon, 1996; Wihardjaka, 2016). More evidence from Japan supported that methane emissions vary in the different soil types with the same cultivation and irrigation types (Yagi & Minami, 1991). Further, the annual emission rates in the soil added with rice straw and mineral fertilizers are 1.8 to 3.5-fold higher than the soil that has mineral fertilizers only (Minamikawa et al., 2006; van der Gon, 1996; Wihardjaka, 2016).

4.3 Climate/weather

The next factor that influences methane production is temperature because temperature changes in the soil would affect the methanogens (Schulz et al., 1996; Chandrasekaran et al., 2022). Seasonal temperature variation strongly affects methane flux during the plantation period in various rice field locations in Japan (Yagi & Minami, 1991). Similarly, Chandrasekaran et al. (2022) reported that methane emissions increase linearly to the increase in soil temperature in India. Temperature affects methane emissions through anaerobic digestion reactors, which are done by mesophilic (20~42°C) or thermophilic (42~75°C) organisms such as methanogens (Parawira, 2004). At the same time, methanogenesis takes place at a temperature ranging from 30 to 40°C (Mer et al., 2001).

In temperate or cold regions, seasonal fluctuations in methane emissions were found to be closely associated with changes in soil temperature (Mer et al., 2001). In addition, temperature also affects methane transport through rice plants and daily emissions in the rice field. To be detailed, the production of methane from acetate reached its peak at the temperature of 36 to 40°C, and methane production from H14CO3- was only detected when the temperature reached at least 22°C and decreased after it reached 30°C (Schulz et al., 2006).

Other evidence that promotes the effects of temperature on methane emissions is obtained from The U.S. and Chile. In Winconsin, The U.S., methanogenesis achieved optimum level when temperatures are ranging between 35 and 45°C (Zeikus & Winfrey, 1976). Meanwhile, in Chile, it was found that under uncorrected conditions, 5°C temperature changes could double the methane production rate while the microbial community of the aquatic sediments is not changing (Lavergne et al., 2021).

4.4 Soil characteristics

Soil as a medium during methanogenesis plays a vital role in methane emissions from rice fields. Physicochemical soil properties such as soil Eh, oxygen availability, pH, soil texture, and mineralogy are the critical keys to managing methane emissions in rice fields (Mer et al., 2001). In addition, soil cultural practices also influence methane emissions, hinted by lower emissions and nitrous oxide concentration in the soil without tillage compared to the perfectly tillaged soil under the same water regime conditions (Wihardjaka, 2016; Yoo et al., 2016). This condition is related to the slow decay process of available organic matter, which makes less biomass returned to the soil.

However, decreasing methane emission by the no-tillage method is subjected to other soil characteristics, such as the depth of irrigation and the content of Fe(III) inside the soil. Thus, it is necessary to consider the long-term effects because of the accumulation of rice residues on and in the soil and the changes in soil physical properties (Minamikawa et al., 2006).

Meanwhile, soil texture affects the percolation rate in soil by providing the structure that allows water to pass through. The crack in the soil that is present during the drainage period allows the trapped methane to be released into the atmosphere. Thus, the texture of the soil can lead to maintaining the methane emissions in the rice field (Ariani et al., 2021). One supporting study conducted in Japan showed that Andosol soil has lower methane emissions than gley soil (Yagi & Minami, 1991) due to rapid percolation in Andosols, possibly reducing the emissions, while gley or peat soil tends to perform slow percolation. Whereas Inceptols, the most common soil type in rice fields in Indonesia, could lower methane emissions compared to Vertisol (Susilawati et al., 2015). A similar finding was also obtained from the country when comparing Inceptisol and Alfisol (Subadiyasa et al., 1997).

Not only are the chemical components of the soils that affect methane emissions, but their textures, pH, and oxygen levels are equally important. Soils rich in clays are good at retaining water and nutrients, whereas sandy, silty, and kaolinite soils are not. Meanwhile, the sandy soil density is less compact, slowing down pH, Eh variation, and organic matter decaying process (Mer et al., 2001).

Another vital element that affects methane emissions in aquatic ecosystems is the oxygen level (Conrad, 2020). The availability of light, which boosts phytoplankton photosynthetic activities, can increase the area of oxic soil layers, which then further leads to the rising methane oxidation reaction (Mer et al., 2001). In connection with this, soils with neutral pH tend to have the highest oxygen levels and methanogenesis is usually optimum within a neutral pH or slightly alkaline environment due to the sensitivity of methanogens to the change of pH level. In contrast, methanotrophs tend to be less sensitive to soil pH variations (Mer et al., 2001). To add more evidence, a study using three different varieties of rice plants in India showed that methane emissions diminish when soil pH rises beyond 7 because a higher pH would inhibit the hydrolysis and acidogenesis phase of methanogenesis (Chandrasekaran et al., 2022).

5. Conclusions

Paddy fields have a substantial potential for emitting methane into the atmosphere, which is necessary to provide accurate scientific information regarding GHG emissions; thus, it is imperative to study about the topic comprehensively. It is concluded that rice cultivars, cultivation processes, climate/weather, and soil characteristics are critical factors that affect methane emissions in the rice field. Furthermore, the impacts of some proper irrigation systems, rice cultivars, soil types, soil tillage techniques, and some fertilizer or amelioration are also essential to control the emissions. Therefore, future research must address the challenges of how to reduce emissions by regulating appropriate farming techniques. The recommended solutions are implementing appropriate water irrigation, crop varieties, amendments, and less tillage in a fit soil texture in order to reduce the methane emissions in the rice field ecosystems. Moreover, a wider framework and multidiscipline approach will be needed in order to enhance understanding the GHG emissions from various types of rice fields. It is expected that the results of this review can guide the decision-makers, scientists, and practitioners to formulate suitable plans to alleviate methane gas emissions.

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Data is available upon request.

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Author contribution

IGAAPP: Conceptualization, Writing-original draft, Methodology, Investigation, Data curation, and Writing – review and editing.

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