SCHOOL OF GRADUATE STUDIES

ASSESSMENT OF THE ECOLOGICAL CONDITION OF LAKE AREKIT USING BENTHIC MACROINVERTEBRATES AS A BIOLOGICAL WATER QUALITY INDICATOR, GURAGE ZONE, ETHIOPIA

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Assessment of the Ecological Condition of Lake Arekit using Benthic Macroinvertebrates as a Biological Water quality indicator, Gurage Zone, Ethiopia

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Thesis Paper Approval Sheet

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As members of the Board of Examiners of the Masters of Science Thesis open defense examination, we have read and evaluated this Thesis prepared by Abdisa Degafu and we examined the candidate. We hereby certify that, the thesis is accepted for fulfilling the requirements for the award of Masters of Science in Biology.

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Final approval and acceptance of the Thesis are contingent upon the submission of its final copy and the Council of Graduate Studies (CGS) through the Graduate Committee (SGC) of the candidate’s School.

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Abbreviation and Acronyms

%E	Percent Ephemeroptera Index
% CHIR	Percentage Chironomidae
% DT	Percentage Dominant Taxa
AUSRIVAS	Australian River Assessment System
BMWP	Biological Monitoring Working Party Index
BSI	British Standards Institute
DO	Dissolved Oxygen
EPA	Environmental Protection Agency
ESA	Ethiopian Standard Agency
HABQA	Habitat Quality Assessment
HFBI	Hilsenhoff Family-level Biotic Index
IBGN	Global Biological Normalized Index
MCI	Macroinvertebrate Community Index
OKAS	Okavango Assessment System
SASS	South African Scoring System
SDI	Shannon Diversity Index
SODIS	Solar Water Disinfection
TDS	Total Dissolved Solvent
UK	The United Kingdom
US	United States
WFD	Framework Directive
WHO	World Health Organization

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Abstract

The ultimate objective of this study was to determine the ecological condition of Lake Arekit using benthic macroinvertebrate communities and some selected physicochemical parameters. Both environmental and macroinvertebrate data were collected from three sampling sites between April and May 2023. Sampling sites were selected based on the degree of anthropogenic disturbance as high impaired (shore area/SS-site 1); moderately impaired (open area/OS-site-2); and less impaired (macrophyte area/vegetation area/MS-site-3). Water temperature, DO, pH, and conductivity, were measured using a portable mulitimeter probe. Turbidity and TDS were measured using measuring turbidometer. Inorganic nutrients were measured using APHA 1995 standardized method. Benthic macroinvertebrates were collected from the littoral region (sampling depth: 0-0.5 m) using standardized kick sampling with a hand D-frame net (28 cm x 30 cm in diameter) with a horizontal transect up to 10 meter from the shore towards the lake when the lake depth less than 0.5 m and diagonal transect when the lake depth greater than 0.5 m to maintain the consistency of sampling effort, a sample was obtained within 10 minutes at each site with three replicas. The results showed that the lake temperature and DO were ranged from 20.2^0*C (at OS) to* 20.8^0*C (at SS) and 2.59 mg/L (at OS) to 5.68 mg/L (at SS), respectively. The lake was alkaline with a pH ranged from 7.07 to 8.82. The maximum and the minimum pH of the study lake were recorded at SS. The water turbidity of the lake was varied from minimum values of 3.95 NTU (at MS) to a maximum value of 43.23 NTU (SS). Major inorganic nutrients of the Lake Arekit did not showed a significant spatial effect* $(p>0.05)$*. Nitrate and total phosphate (TP), correspondingly, were ranged between 2.71 mg/L and 108.18mg/L and 12.8 and 151.8 mg/L. In this study lake, a total of 620 benthic macroinvertebrate specimens comparing of six taxa were collected from all the three study sites. Notonectidae (39.19%) were comprised the highest taxa followed by Coenagrionidae (25.45%) and Chironomidae (20.64%). The occurrence of Hirunidae (5.8%), Gerridae (1.61%), and Begidae (0.64) were found low, all comprised less than 10% of the total taxa. All most all the benthic taxa found in the study lake were pollution tolerant, indicating that the lake water condition was highly impacted. The results of the physicochemical parameters and benthic invertebrates strongly suggested that the lake water of the study lake was polluted. This study proposes that management of anthropogenic activities, nonpoint sources of pollution, loss of natural riparian habitat, and occasional untreated point inflow are necessary.*

Key words/phrases*: Macroinvertebrate,* nutrients, *physicochemical parameters,* polluted.

CHAPTER 1. INTRODUCTION

1.1. Back ground of the study

Lakes and reservoirs are critical components of freshwater ecosystems and are an important habitat for aquatic species (Amusan *et al.,* 2024). Rapid industrial and urban development has, however, compromised the integrity of many reservoirs in developing nations due to pollution from industrial and domestic use (Qadri and Faiq, 2020). Pollutants such as nitrogen and phosphorus from such spillages may cause increased aquatic biological productivity, resulting in low dissolved oxygen (DO) and eutrophication of lakes and reservoirs and other such standing waters (Adams *et al*., 2020; Siziba *et al.,* 2021).

Eutrophication in turn reduces habitat heterogeneity, thus directly or indirectly affecting aquatic life (Dalu and Froneman, 2016). And reducing the health status of aquatic ecosystems, water quality and biological monitoring of lakes and reservoirs are critical given the increase in anthropogenic disturbances on those water bodies (Dalu *et al.,* 2012; Siziba *et al*., 2021). Eutrophication has been defined as "the enrichment of water by nutrients causing an accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance to the balance of organisms present in the water and to the quality of the water concerned" (OSPAR,2003). Macroinvertebrates are important indicators of water pollution caused by physical and chemical alteration of lake water and the surrounding habitat (Efitre *et al.,* 2001; Mwedzi *et al*., 2020). Due to their limited mobility and near sedentary lifestyle, macroinvertebrates can reflect conditions that are not present at the time of sampling (Baptista *et al.*, 2002). Thus benthic macroinvertebrates becoming effective indicators of current and long term water quality.

This is because macroinvertebrates respond to changes well before the manifestation of a problem (de Vries *et al*., 2021). As such understanding macroinvertebrate-environment relations is important in understanding the integrity of aquatic communities, especially within the context of habitat degradation. The diversity and composition of macroinvertebrates are important themes in aquatic ecology as they can be used to interpret the long-term effects of water pollution.

The response of macroinvertebrates to organic loading has been documented in rivers and streams (Benetti *et al.*, 2012; Akamagwuna *et al.*, 2021). And their use as water quality indicators in those water bodies is acceptable (Barbour and Paul, 2010; Dalu *et al.*, 2017). Using macroinvertebrates as indicators of water quality presents several advantages because they have narrow ecological requirements and great diversity of form and habit, making them useful bioindicators of aquatic ecological balance (Benetti *et al.*, 2012). The rationale for the use of ecosystem health macroinvertebrate metrics is based on the observation that some taxa, especially the Ephemeroptera, Plecoptera, and Trichoptera are sensitive to pollutants, while others such as Chironomidae and Oligochaeta are more tolerant (Masese *et al.*, 2014). Despite this general observation, macroinvertebrate communities may respond differently to chemical pollution in terms of both structure and function (Dalu *et al*., 2017).

Because of this, it is crucial to understand how different aquatic communities' macroinvertebrate populations respond to different environmental conditions. Knowing the long-term effects of water pollution in the face of environmental perturbation is important, and the use of benthic macroinvertebrates can be useful in water quality monitoring. Several studies on water quality assessments have focused on sampling physicochemical parameters (Nyamangara *et al*., 2013) Mhlanga *et al.*, 2020).

Although physicochemical analyses are important in assessing water quality, they do not reflect fluxes of effluent discharges in a water bodies as sampling is periodic and sporadic (Bere and Nyamupingidza, 2014). This means the use of an aggregate of physicochemical parameters and macroinvertebrates can seize the total spectrum of aquatic stressors. This study aimed at assessing the structure of benthic macroinvertebrate's communities at the family stage in relation to physicochemical parameters that symbolize the assemblage of benthic macroinvertebrates as biological indicator of Lake Arekit.

1.2. Statement of the problem

During the preliminary survey of this study, there were observed high different anthropogenic activities near and around the watershed of the Lake Arekit. The major anthropogenic activities observed in the lake Arekit were agriculture, urbanization, deposition of domestic and industrial effluents, misuse of pesticides and fertilizers, silt load from the catchment and car wash. As a result of these anthropogenic activities the water of Lake Arekit becomes polluted. Macroinvertebrates are among the most frequently applied groups in freshwater monitoring and assessment and have proven to be useful indicators in determining the health status of wetlands since differences in environmental requirements among taxa produce community characteristics that reflect ecological conditions (Gabriels *et al.*, 2010). Therefore, it’s important to understand the macroinvertebrate assemblage status to assess the water quality and productivity of Lake Arekit in order to provide the water quality management systems in place that may enhance the role of benthic macroinvertebrates in the livelihood of local societies living along the shores of the lake.

1.3. Objectives

1.3.1. General Objective

- The general objective of this study is to assess the ecological condition of Lake Arekit using benthic macroinvertebrate assemblages as a biological water quality indicator.

1.3.2. Specific objectives

- To determine the spatial patterns of benthic Macroinvertebrates in the Lake Arekit.
- To determine the diversity and abundance of benthic macroinvertebrates in the Littoral region of the Lake Arekit.

1.4. Significance of the research

Base line information for the next researchers and it's important to understand the macroinvertebrate status and provides the water quality management systems in place that may enhance the role of benthic macroinvertebrates in the livelihood of local societies living along the shores of the lake.

1.5. Expected outputs

- The spatial variation of benthic macroinvertebrate assemblages in relation to physical habitat and water quality parameters of Lake Arekit will be determined.
- The water quality status and productivity of Lake Arekit will be assessed using benthic assemblages.
- The diversity and abundance of benthic macroinvertebrates Lake Arekit will be determined.

CHAPTER 2. LITERATURE REVIEW

2.1. Macroinvertebrates as Bio monitoring in a global context

Many large-scale effects of pollution, habitat loss, climate change, and the introduction of alien species are currently harming ecosystems throughout the planet (Strayer, 2010). A set of international abatement targets, such as the UN Aichi Biodiversity Target 15 and the EU Biodiversity Strategy for 2030, have been established in response to the detrimental effects on biodiversity and ecosystem services. These targets aim to restore at least 15% of degraded ecosystems by 2020 (Balmford *et al.,* 2005) and to achieve favorable status for at least 30% of species and habitats that are not currently in that status by 2030 (Hermoso *et al.,* 2022).

Bioindicators play a crucial role in monitoring and measuring the effects on the environment (Carter and Resh, 2001; Niemi and Mc Donald, 2004). They also support policymakers who work to preserve and enhance freshwater ecosystems and the products and services they provide (Friberg *et al.*, 2011). It is clear that a global assessment system that tracks changes in ecosystem health and biodiversity using affordable bioindicators is needed (Feld *et al.,* 2010). This could be accomplished in the near future for riverine ecosystems because techniques that are similar in terms of scientific approach and underlying methodology are now being used globally (Tharme, 2003).

A significant body of knowledge on biological community responses to human-induced stress as well as a variety of assessment techniques with a high degree of commonality has been amassed over the course of the more than a century-long tradition of using bioindicators to assess environmental quality in freshwater science (Cairns and Pratt, 1993; Metcalfe-Smith, 1996, (Friberg *et al.,* 2011). The wide-spread use of biological indicators for monitoring that occurs today is an excellent example of the applied use of ecological knowledge that has helped many riverine ecosystems maintain and improve their environmental quality in recent decades (Birk *et al.,* 2010). It also has a significant potential to play a key role in many developing economies that have little experience using these tools and systems nationally.

2.2. Macroinvertebrates as Bio monitoring study in Africa

The South African Scoring System (SASS), a macroinvertebrate-based biotic index for river assessments, was developed (Weerts and Cyrus, 2008) after the British BMWP/ASPT System was modified in Africa. The SASS was meant to be a quick, low-cost, but reliable scientific method for identifying trends in water quality change over time or for detecting degradation of water quality. Currently, the River Health Program relies heavily on the SASS approach, which is the gold standard for the quick bio evaluation of South African rivers (Dallas, 1997; Dickens and Graham, 2002). In other parts of southern Africa, such as Zimbabwe (Phiri, 2000), Namibia (Namibian Scoring System NASS; Taylor and Palmer, 2004), and Botswana (Okavango Assessment System OKAS; Dallas, 2009), the SASS technique was more recently modified for use on regional fresh water macroinvertebrate taxa and Scores are allocated to specific indicator organisms at a particular taxonomic level based on specific requirements in terms of Physical and chemical conditions (Armitage *et al.,* 1983; Hilsenhoff, 1988).

While the value of the score system in determining the ecological river status has been well studied in the southern part of African countries (see above), only few studies have been conducted in the eastern and northern Africa. Based on two master theses, Obubu, (2010) for Uganda, and Koblinger and Trauner (2013) for Burkina Faso, the fact that the SASS approach fundamentally works in other parts of the African continent was confirmed.

Nevertheless, these scores might need to be modified before being applied to other areas. This is because certain macroinvertebrates might not exist in the area in question and might be replaced by other taxa, and some taxa might have varying levels of pollution tolerance depending on the region (Buss and Salles, 2006). The benefit of the biotic score is that managers and decision makers can more easily access information because it only requires qualitative sampling, negating the need to count abundances per taxon (Armitage *et al.,* 1983). The results can also be directly translated to water quality classes (Aura *et al.,* 2017). Furthermore, the scoring system is extensively utilized since it requires professionals with a modest level of training and allows for the examination of a large number of sites in a short amount of time and at a reasonably low cost (Rosenberg and Resh, 1993).

2.3. Macroinvertebrates as Bio monitoring in Ethiopia

Human activity was the cause of 65% of wetland disturbances in Ethiopia (Yilma Dugan, 1990). Some of the main factors that altered water quality were anthropogenic activities associated with agriculture and urbanization, such as the deposition of household and industrial effluents, increased nutrients as a result of improper use of pesticides and fertilizers, and silt load from the catchment (Romshoo and Rashid, 2012).

Variations in the physicochemical characteristics of water lead to variations in a biotope's species composition, abundance, and distribution (Ebenebe *et al.*, 2016). Since varying environmental requirements among taxa result in community characteristics that reflect ecological conditions, macroinvertebrates are among the most commonly applied groups in freshwater monitoring and assessment. They have also shown to be useful indicators in assessing the health status of wetlands (Gabriels *et al.*, 2010).

Moreover, all trophic levels of wetlands are home to a large number of macroinvertebrates (Culler *et al.*, 2014); the composition of these communities can reflect features of the soil (Armitage and Fong, 2004), vegetation (Verberk *et al.*, 2010), and hydrology (Culler *et al.*, 2014b). Furthermore, the sensitivity to stressors and habitat needs of macroinvertebrate taxa varies, leading to distinct assemblages depending on the wetland condition (Batzer, 2013). In addition, investigations that are completed quickly employ this assessment because it is less costly (Davies *et al.*, 2006). There is a long history of using macroinvertebrates as bioindicators of freshwater quality (De Pauw *et al.*, 2006).

In other case, there were a number of drawbacks to assessing water quality with a single index (Gabriels *et al.*, 2010). As such, the overall picture of aquatic ecosystems under a wide range of anthropogenic stressors is often not captured by a single metric. Consequently, due to its capacity to incorporate supplementary data from a wide variety of stressors, a more integrated approach such as the creation of a multimetric index has drawn greater attention recently (Vlek *et al.*, 2004). Lake Tana is one of the biggest freshwater Lakes in Ethiopia (Ayalew Wondie and Seyoum Mengistou, 2006). Large marshes and lowlands that flood occasionally encircle the Lake, offering a variety of benefits (Ibrahim Mohammed *et al.*, 2022).

However, despite the benefits, studies show that the sustainability of ecosystem services has been jeopardized because of factors such erosion, sedimentation, pollution, and pressure from a growing population in the watershed (Goraw Goshu *et al.,* 2010). Additionally, untreated wastewater was dumped into the wetlands, which decreased biodiversity and water quality (Negash Atnafu *et al.,* 2011). As a result, many wetlands are regarded as vulnerable, and some of the more heavily utilized ones have lost their ability to regenerate (Tadesse Mosisa and Bahilu Bezabih 2006).

2.4. Riverine macroinvertebrates are the key indicator group

The most often utilized riverine indicator group in contemporary freshwater biomonitoring is macroinvertebrates (Birk *et al.,* 2012; Carter *et al.,* 2017; Hellawell, 1986). They differ in their sensitivity to different stressors, which is a key requirement they share. They also have a number of useful benefits for bioassessments, including as a long lifetime, a widespread distribution in most rivers, and a sedentary behavior that offers good spatial resolution. Furthermore, they may be identified to an operational level in an economical manner and are simple to sample (Bonada *et al.,* 2006; Rosenberg and Resh, 1993).

Though biological reactions to water pollution have been documented since antiquity (Moog *et al.,* 2018), the field of modern biomonitoring began about a century ago in Europe and North America (e.g. Forbes and Richardson, 1913; Kolkwitz and Marsson, 1909), with the development of indicator systems that were primarily concerned with identifying organic waste inputs to rivers (Cairns and Pratt, 1993; Karr and Chu, 1999). It took another fifty years or so before river authorities began using these systems on a regular basis to evaluate environmental quality (Carter *et al.,* 2017; Hawkes, 1998).

The saprobic systems originated in central Europe in the early 20th century and assessed rivers by measures of saprobity, i.e., the dependence of aquatic organisms on decomposing organic substance as sole source of food, having species specific tolerances for bacteria, algae and fauna (Persoone and De Pauw, 1979; Sládeček, 1969). The saprobic systems were revised and modernized several times in the 1950s–1970s and became widely used in Central and Eastern Europe (Moog *et al.,* 2018). However, these systems did not become popular in the United States (US) and the United Kingdom (UK), mainly because they were considered specific for central Europe, their use was restricted to measuring organic pollution, and the collection and identifications of multiple organism groups was difficult and time consuming (Cairns and Pratt, 1993; Persoone and De Pauw, 1979).

Instead, rapid biomonitoring approaches using biotic indices for assessment were developed in the UK and the US, exemplified by the Trent biotic index (De Pauwand Vanhooren, 1983). And Beck biotic index (Beck, 1954). These systems were primarily designed to detect organic pollution on community levels but also reflected other environmental stressors (Armitage *et al.,* 1983; Lenat, 1993; Paisley *et al.,* 2014). The development of biotic indices along with the implementation of The Water Act in the UK in the 1960s, accelerated the use of macroinvertebrates for water quality monitoring as river authorities were now charged with biomonitoring responsibilities (Hawkes, 1998). Taxa- neutral diversity indices, i.e., numeric expressions of structural community composition based on both richness and abundances (Daly *et al.,* 2018) were popular in North America in the 1960–70s, as they quantified the heterogeneity of full assemblages, had high statistical power, and were not dependent on tolerance values (Karr and Chu, 1999).

Although diversity indices by passed some of the difficulties experienced by the saprobic and biotic indices, they were eventually considered unsuccessful for several reasons, partly because they required rigorous sampling and the observed response to degradation was often poor (Cairns and Pratt, 1993; Metcalfe, 1989). As the complexity of effluents increased with industrial activities and intensified land-use in the 1960s–1980s, many streams were affected by multiple stressors (Hellawell, 1986), and by the mid-1970s most European countries had changed their focus to biotic indices (Metcalfe, 1989).

The development of the BWMP systems in the UK from the mid-1970s (Hawkes, 1998; Paisley *et al.*, 2014) along with the predictive classification models RIVPACS/RICT (Wright *et al.*, 2000), has been instrumental for biomonitoring of rivers in the UK and has also been much used elsewhere (Birk *et al.*, 2010). In the US in the 1980s, biotic indices such as the family biotic index (Hilsenhoff, 1988; Lenat, 1993), were introduced for the purpose of rapid biomonitoring, together with structural and functional metric components, as the US Environmental Protection Agency (EPA) also called for efficacious methods to assess environmental quality of surface waters, as mandated by the Clean Water Act of 1972 (Barbour *et al.*, 1999).

Multi-metrics, in which simple metrics were combined to improve sensitivity, robustness and diagnostic capabilities of assessments, soon came into focus in the US (Cairns and Pratt, 1993; Karr and Chu, 1999). The implementation of the EU Water Framework Directive (European Community, 2000; here after EU WFD), at least in part, motivated for the use of multimetrics also in Europe as the directive require assessments based on multiple community components (Friberg, 2014; Hering *et al.*, 2006).

In Australia, by the mid-1990 s, a predictive model system called the Australian River Assessment System (AUSRIVAS) was developed and implemented under the National River Health Program, which was inspired by the systems used in the UK (Chessman, 1995; Simpson and Norris, 2000; Nichols and Dyer, 2013). In New Zealand, standardized methods for macroinvertebrate biomonitoring were introduced in 1999 with assessments based on the macroinvertebrate community index (MCI; Stark *et al.,* 2001).

2.5. Macroinvertebrates for Lake Biomonitoring

Benthic macroinvertebrates have been used for Lake biomonitoring since the turn of the 20th century (Cairns, 1983). Since then, their use as reliable and widespread bioassessment markers of the degradation or restoration of lake ecosystems has grown (Birk *et al.,* 2012; Dalu *et al.,* 2016). In aquatic food webs, they serve as a crucial link between primary producers, detrital deposits, and higher trophic levels (Gobiet *et al.,* 2013). Moreover, because their life cycles span many months, benthic macroinvertebrates reveal coupled and varied stressors that differ from those that can be reflected by plankton communities (Dalu *et al.,* 2016).

The potential indicator of benthic invertebrate fauna is highlighted in tiny Lakes, which are primarily found in nature with sparse macrophyte (Labat, 2017). Further, macroinvertebrates have a practical advantage over the other biological quality elements discussed because of their a) less variability in time compared to phytoplankton, which is only present during the vegetation period, b) response to habitat degradation compared to diatoms, and c) immobility compared to fish, which is less sensitive to pressures and dependent on Lake size (Ritterbusch *et al.,* 2022). Despite this, there hasn't been much usage of benthic macroinvertebrates in Lake evaluation programs (Birk *et al.,* 2012).

This is particularly evident in developing nations such as Ethiopia, where the absence of national legislation pertaining to bioassessment has impeded the development and calibration of techniques for monitoring and assessing impairment (Ruiz-Picos *et al.*, 2017). When evaluating the integrity of a water resource, managers cannot rely entirely on any one sign. Similar to this, the intricate biotic structure of fauna is noted as a characteristic that contributes to their robustness (Solimini *et al.*, 2006; Birk *et al.*, 2012).

CHAPTER 3. MATERIALS AND METHODS

3.1 Description of the study area

This study was conducted in Lake Arekit which is located in the Gurage drainage basins of Ethiopia. The Gurage Zone is located in Central-south Ethiopia with the location between 7040' to 8030' North and 37030' to 38040' East and covers an area of 5,932 km^2 (Yirga Enawgaw and Solomon Wagaw (2023). According to Yirga Enawgaw and Solomon Wagaw (2023), Gurage Zone is divided by four drainage basins, namely Awash, Rift Valley, Bilate and Omo-Gibe. The Western Gurage land (Watersheds) drains to Omo-Gibe Basin and cover large areas. In the western parts of Gurage land, several rivers and streams drain from northeast and east to west that is to Gibe River. With the exception of minor deviations at local level, the streams in the ecoregion have a dendrite drainage pattern (observation of this study). The major streams ultimately drain to west and south-west/east ward following the general inclination of the slope direction of the agro-region. The western watersheds of the Gurage land drain to Omo-Gibe Basin and cover large areas. In this part of the watershed, several inland waters are available, which drains from East to West (Yirga Enawgaw and Solomon Wagaw (2023).

Lake Arekit is one of the numerous naturally occurring inland water bodies in the Gurage drainage basin. The Lake is about 240 kilometers away from Addis Ababa, the country's capital. It is situated in the Gumar district, 70 kilometers from the settlement of Wolkite (the capital town of the Gurage zone). Lake Arekit is located between latitudes 70°59'30" and 80°16'00" N and 370°53'30" and 380°10'00" E, respectively. It is located at an elevation of 2820–2950 meters above sea level. The lake has a surface area of 130 hectares and an average depth of 1.5 meters (Figure 1). The Lake receives its water from direct rainfall and runoff from the nearby watershed. The lake has one outlet. It is a highland lake system (observation during this study).

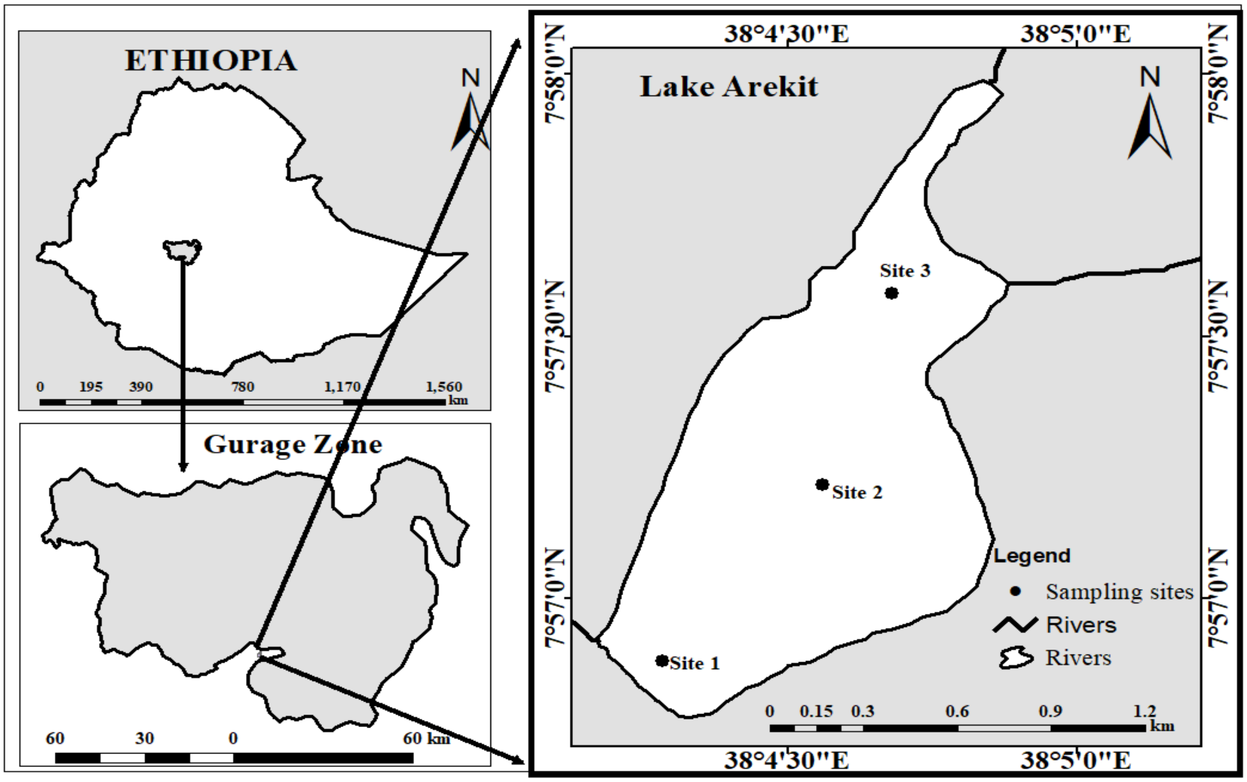

Figure 1: Map of the study area showing the sample stations (Lake Arekit, Ethiopia).

Table 1: Location of sites and their characteristics

Site	Elevation	N	E	Site features
Macrophyte	2882	07.95870°	038.06049°	Along the site, there is an evidence of high erosion and siltation. There are different natural habitats and macrophytes coverage. The site is considered as minimally impaired/disturbed.
Open	2882	07.42410°	038.06192°	There are no aquatic macrophytes, cover at the site and the location is relatively far from direct human impacts and moderately disturbed.
Shore	2878	07.94816°	038.06980°	This site is found in the main entrance road to get to gate Lake Arekit. The site is an indication of high shoreline modification. It is the primary fish landing area of the lake. Here, people engage in a variety of activities like car washing, swimming, and grazing cattle. Waste from Arekit town is prevalent.

3.2. Sampling design

3.2.1. Physicochemical parameters and Nutrients

Physicochemical parameters such as water temperature, DO, pH, and conductivity, were measured using a portable mulitimeter probe involves using a specialized device that can handle these specific measurements. Additionally, turbidity is measured by using turbidimeter and TDS is typically measured with a TDS meter. In other hand, Nutrients were measured using APHA (American Public Health Association) Standard Methods for the Examination of Water and Wastewater, 1995).

3.2.2. Macroinvertebrate sampling

A cross-sectional macroinvertebrate sampling design was used to collect specimens. Triplicate samples of macroinvertebrate assemblages were collected on a monthly basis from three sites at each station for four months from February to May 2023, covering part of the dry season (February to May). The general features of the Lake were observed along with two sampling stations (Shore and Open) sites are defined: In total three sites from the Lake were sampled.

3.3. Macroinvertebrates samples

Benthic macroinvertebrates were collected from the littoral region (sampling depth: 0 - 0.5 m) using standardized kick sampling with a hand D-frame net (28 cm x 30 cm in diameter) with a horizontal transect up to 10 meter from the shore towards the Lake when the Lake depth less than 0.5 m and diagonal transect when the Lake depth greater than 0.5 m to maintain the consistency of sampling effort, a sample was obtained within 10 minutes at each site with three replicas.

The collection of benthic macroinvertebrates was carried out based on Ontario Benthos Biomonitoring Network Protocol Manual Egertson *et al., (*2004). The collected macroinvertebtates were separated from sediment using 500µm sieves and they were stored in plastic bags, fixed with 5% formaldehyde, and transported to the Biology Laboratory of Wolkite University for further analysis. In the laboratory, the macroinvertebrates were sorted and identified to the family level under a stereo microscope using keys by Edmondson (1959), Gooderham and Tysrlin (2002), and Bouchard (2004). The identified macroinvetrebrates were counted and the diversity, number of individuals and distribution patterns were determined for specific metrics and correlated with various physicochemical variables. The various physicochemical parameters which were responsible for the diversity and abundance of the benthic macroinvertebrates were also measured using standardized method.

3.4. Biological metrics

Benthic macroinvertebrate metrics measure different components of the community structure and have different ranges of sensitivity to stress. Therefore, it is recommendable to use several metrics because an integrated approach provides more assurance of a valid assessment Klemm *et al.,* 1990. In the present study, the biotic Hilsenhoff Family-level Biotic Index (HFBI), Total number of taxa (Family level Richness), Percentage Dominant taxa (% DT), Shannon Diversity Index (SDI), Simpson's Index of Diversity and Evenness were applied.

Hilsenhoff Family-level Biotic Index (H-FBI)

This biotic index is calculated by multiplying the number of individuals of each family by an assigned tolerance value, summing these products and dividing by the total number of individuals Bouchard 2004.

Each family is given a score between 0-10 depending on its tolerance to low dissolved oxygen level and has only been evaluated for organic pollutants (Bode *et al.,* 1996; Bouchard 2004). Those taxa that is least tolerant to pollution H-FBI Mason 1996. High HFBI values are indicative of organic pollution, while low value indicative of clear-water conditions. The Hilsenhoff Family-level Biotic Index (H-FBI) is calculated as (Sekhar, 2022).

$$HFBI = \sum \frac{\mathrm{Xi} * \mathrm{ti}}{\mathrm{n}} \qquad \text{Where,}$$

Xi: is a few individuals within the taxon, ti: is the tolerance value of a taxon, and n: is the total number of organisms in the sample.

According to Hilsenhoff 1983, HFBI values of 0-3.75 indicate excellent water quality while 3.76-4.25, 4.26-5, 5.01-5.75, 5.76- 6.5, 6.51-7.25, and 7.26-10 indicate very good, good, fair, poor and very poor water quality, respectively.

Taxa/Family Level Richness

This metric is the measure of community's diversity, number of different families found in samples of each site. Reductions in community diversity have been positively associated with various forms of environmental pollution, including nutrient richness increases with increasing water quality, habitat diversity, and habitat suitability Barbour *et al.,* 1996. Therefore, the total number of unique taxa in study area was obtained by simple counting.

Simpson’s Dominance Index

Dominance index is the percent contribution of the most numerous families, and measures community balance, or evenness of the distribution of individual families of the sample. A community dominated by relatively few species would indicate environmental stress. A high percent contribution by a single taxon indicates community imbalance.

It can be calculated as (Talukdar, 2017). $Pi = \frac{Ni}{N}$ Where:

- ✓ Pi: is dominant index
- ✓ Ni: is the number of individuals in a given area and
- ✓ N: is the total number of individuals of all species.

The index uses abundance of the numerically dominant family relative to the rest of the population as an indication of community balance, which means dominant taxa greater than 35% indicates poor water quality, between 25% and 35% indicates fair water quality, and less than 25% indicates good water quality.

Shannon Diversity Index (SDI)

The SDI is a diversity index that combines taxa richness and community balance (evenness) to characterize species diversity in a community. The combination of abundance and richness in SDI is designed to indicate the state of the macroinvertebrates’ communities (Rosenberg and Resh 1993). A community with only a few taxa or with a few abundant taxa has low SDI, while a community exhibits high diversity (high SDI) if many taxa with equal or nearly equal numbers are present. A high SDI indicates good benthic habitat and non-impacted water quality.

Benthic macroinvertebrate dynamics of Lakes Arekit will also examined using the Shannon index as (Beisel and Moreteau, 1997).

$$H' = -\sum_{i=1}^{S} \mathrm{Pi} * \ln(\mathrm{Pi})$$

Where:

- ✓ H': is the Shannon-Weaver diversity index,
- ✓ Pi: is the abundance of species i,
- ✓ S: is the total number of individuals in the community and
- ✓ ln: is the natural log

3.5. Data Analysis

3.5.1. Physicochemical parameters and Nutrients

Variations in physicochemical parameters on a macroinvertebrates index among sites were analyzed using Analysis of Variance (ANOVA) and their mean was separated by using Turkey's HSD analysis.

3.5.2. Diversity of Macroinvertebrate analysis

Macroinvertebrates were identified mainly to family level using Binocular dissecting microscope and identification keys such as; (Bouchard, 2004; Oscoz *et al.*, 2011; Baring, 2015). The composition and structure of macroinveretebrates were evaluated by different taxonomic diversity indices.

3.5.3. Relationships between physicochemical variables and macroinvertebrates

Relationships between physicochemical variables and macroinvertebrates distribution was evaluated by spearman correlation that compares physicochemical variables with macroinveretebrate distribution.

CHAPTER 4. RESULTS

4.1. Temporal variation in physicochemical parameters

From (Table 2), the temporal variation in physicochemical parameters of Lake Arekit reflects the intricate interplay between environmental factors and aquatic health.

Table 2: Temporal variation in physicochemical parameters of Lake Arekit.

	Ph	**Temp°C**	EC**(mS cm^{-1})**	DO**(mg L^{-1})**	Turb(NTU)	TDS
Feb	7.3 ± 0.23^{a}	20.5 ± 0.23^{ac}	84.9 ± 2.4^{a}	4.6 ± 0.16^{a}	30.08 ± 13.4^{a}	0.03 ± 0.00^{a}
Mar	7.3 ± 0.25^{a}	20.4 ± 0.05^{ab}	82.0 ± 1.5^{a}	4.0 ± 1.05^{a}	16.14 ± 7.25^{a}	0.02 ± 0.01^{ac}
Apr	7.7 ± 0.52^{a}	20.3 ± 0.04^{b}	192.2 ± 0.3^{a}	4.7 ± 1.06^{a}	26.40 ± 17.62^{a}	0.03 ± 0.00^{b}
May	7.8 ± 0.78^{a}	20.6 ± 0.15^{c}	94.6 ± 12.6^{a}	4.1 ± 0.12^{a}	24.35 ± 9.12^{a}	0.01 ± 0.01^{c}
Mean	7.5±0.52	20.5±0.19	113.4±1.52	4.3±0.78	24.24±13.03	0.02±0.01

4.2. Temporal variation in Nutrient parameters

The data in (Table 3) indicates varying concentrations of nitrogen compounds (NO3-N, NO2-N, and NH3-N), silica (SiO2), total phosphorus (TP), and soluble reactive phosphorus (SRP) in Arekit and the analysis of temporal variation in nutrient variables within the Lake reveals significant fluctuations across the months from February to May.

Table 3: Temporal variation in Nutrient parameters of Lake Arekit.

	NO_3-N **(µg L^{-1})**	**NO_2-N** **(µg L^{-1})**	NH_3-N **(µg L^{-1})**	SiO_2 **(mg L^{-1})**	TP **(mg L^{-1})**	SRP **(mg L^{-1})**
Feb	86.4±17.00^{a}	60.09±37.38^{a}	46.65±20.2^{a}	7.12±2.19^{a}	117.46±26.41^{a}	40.35±14.20^{a}
Mar	78.7±13.49^{a}	30.88±21.80^{a}	63.95±8.66^{b}	3.17±0.97^{b}	101.02±14.94^{a}	68.57±21.59^{a}
Apr	80.50±15.9^{a}	35.17±18.65^{a}	39.50±5.53^{a}	4.13±0.69^{b}	140.32±46.65^{ab}	104.80±24.67^{b}
May	76.12±17.4^{a}	39.30±18.45^{a}	66.17±13.0^{b}	7.07±1.71^{a}	134.80±6.16^{a}	62.57±26.23^{a}
Mean	80.43±15.80	41.36±26.73	54.07±16.98	5.37±2.29	123.40±31.01	69.07±31.63

4.3. Spatial variation in physicochemical parameters

The spatial variation of physicochemical parameters in Lake Arekit (Table 4) reveals important insights into the ecological conditions of different sites within the Lake, namely the macrophyte, shore, and opens water areas. Each section of the Lake demonstrates distinct characteristics that can influence aquatic life and overall water quality.

Table 4: Spatial variation in physicochemical parameters of Lake Arekit.

	pH	Temp **°C**	EC**(mS cm^{-1})**	DO**(mg L^{-1})**	Turb (NTU)	TDS
Macrophyte	7.34±0.19^{a}	20.48±0.16^{a}	82.19±1.12^{a}	4.2±0.58^{a}	22.84±17.47^{a}	0.02±0.01^{a}
Shore	7.84±0.80^{b}	20.45±0.25^{a}	164.56±263.51^{a}	4.7±0.62^{a}	25.88±13.69^{a}	0.03±0.02^{a}
Open	7.38±0.20^{ab}	20.48±0.17^{a}	93.55±11.51^{a}	4.1±0.99^{a}	24.03±6.57^{a}	0.03±0.01^{a}
Mean	7.52±0.52	20.47±0.19	113.43±152.42	4.3±0.78	24.25±13.04	0.03±0.01

4.4. Spatial variation in Nutrient parameters

The spatial variation in nutrient variables within Lake Arekit highlights the distinct ecological dynamics present in different sites of the Lake, namely the macrophyte, shore, and opens water areas. The analysis of nutrient concentrations, including nitrate nitrogen (NO3-N), nitrite nitrogen (NO2-N), ammonia nitrogen (NH3-N), silica (SiO2), total phosphorus (TP), and soluble reactive phosphorus (SRP), reveals important insights into the nutrient cycling and potential productivity of each site.

Table 5: Spatial variation in Nutrient parameters of Lake Arekit.

	NO_3-N **(μg L^{-1})**	**NO_2-N**	NH_3-N**(μg L^{-1})**	SiO_2**(mg L^{-1})**	TP**(mg L^{-1})**	SRP**(mg L^{-1})**
Macrophyte	84.98±13.67 ^{a}	35.93±24.71 ^{a}	53.00±8.57 ^{a}	5.35±2.60 ^{a}	131.63±17.63 ^{a}	87.30±22.61^{a}
Shore	74.21±15.07 ^{a}	36.29±14.25 ^{a}	52.76±28.01 ^{a}	5.95±2.50 ^{a}	106.80±15.37 ^{a}	51.30±21.59^{b}
Open	82.12±17.68 ^{a}	51.88±35.77 ^{a}	56.45±7.06 ^{a}	4.83±1.75 ^{a}	131.78±45.40 ^{a}	68.63±38.7^{ab}
Mean	80.44±15.80	41.37±26.74	54.07±16.98	5.38±2.30	123.40±31.01	69.08±31.63

4.5. Diversity of Macroinverebrates

From (Table 6) the Notonectidae family was observed high (39.19%) from the all sites followed by Coenagrionidae family (22.74%) found in high percentage at all sites. Chironomidae takes the third rank (20.64%) at all site while family Hirunidae was observed (5.80%) at all site and family Gerridae was (1.61%) at all sites. Family Betidae was observed at all site of the present study (0.96%)

Table 6: Number of benthic macroinvertebrates collected from the study sites (Lake Arekit) during this study and pollution tolerance value for each taxon given by Bouchard (Bonada *et al.*, 2006).

Taxa	**Sampling site**					
	Open	**Shore**	**Macrophytes**	**Total**	**Tolerance value**	**H-FBI**
Notonectidae	70	15	158	243	9	3.53
Coenagronidae	43	13	141	197	9	2.86
Chironomidae	28	5	95	128	8	1.65
Hirunidae	8	2	26	36	10	0.58
Gerridae	1	0	9	10	6	0.097
Baetidae	1	1	4	6	4	0.039
Total individual	151	36	433	620		
Number of taxa	6	5	6			

4.5.1. Diversity indices of Macroinverebrates

Based on the provided diversity indices for macroinvertebrates in Lake Arekit (Table 7) across three different sites (macrophyte, shore, and open water).

Table 7: Diversity indices for macroinvertebrates in Lake Arekit

	Macrophyte	Shore	Open
Taxa_S	6	5	6
Individuals	433	36	151
Dominance_D	0.2914	0.3272	0.3333
Simpson_1-D	0.7086	0.6728	0.6667
Shannon_H	1.359	1.267	1.249
Evenness_e^H/S	0.6485	0.71	0.5809

Taxa Richness (Taxa_S)

Macrophyte and Open Water sites have the highest taxa richness with 6 taxa, indicating a slightly higher diversity of macroinvertebrate species compared to the shore site, which has 5 taxa. This suggests that the macrophyte and open water areas support a more diverse range of macroinvertebrate species.

Simpson's Dominance Index (Simpson_1-D)

The macrophyte area has the highest Simpson's Index of Diversity, indicating greater diversity. This is followed by the shore and open water areas, which have similar but slightly lower values.

Shannon Index (Shannon_H)

The macrophyte area has the highest Shannon Index, followed by the shore and open water areas. This index accounts for both the number of species and the evenness of their distribution.

4.6. Relationships between physicochemical variables and macroinvertebrates distribution

From the relationships between physicochemical variables and macroinvertebrates distribution through **axis loadings (Figure 2)** it indicates how strongly each variable contributes to the principal component, in which positive or negative loadings show the direction of influence.

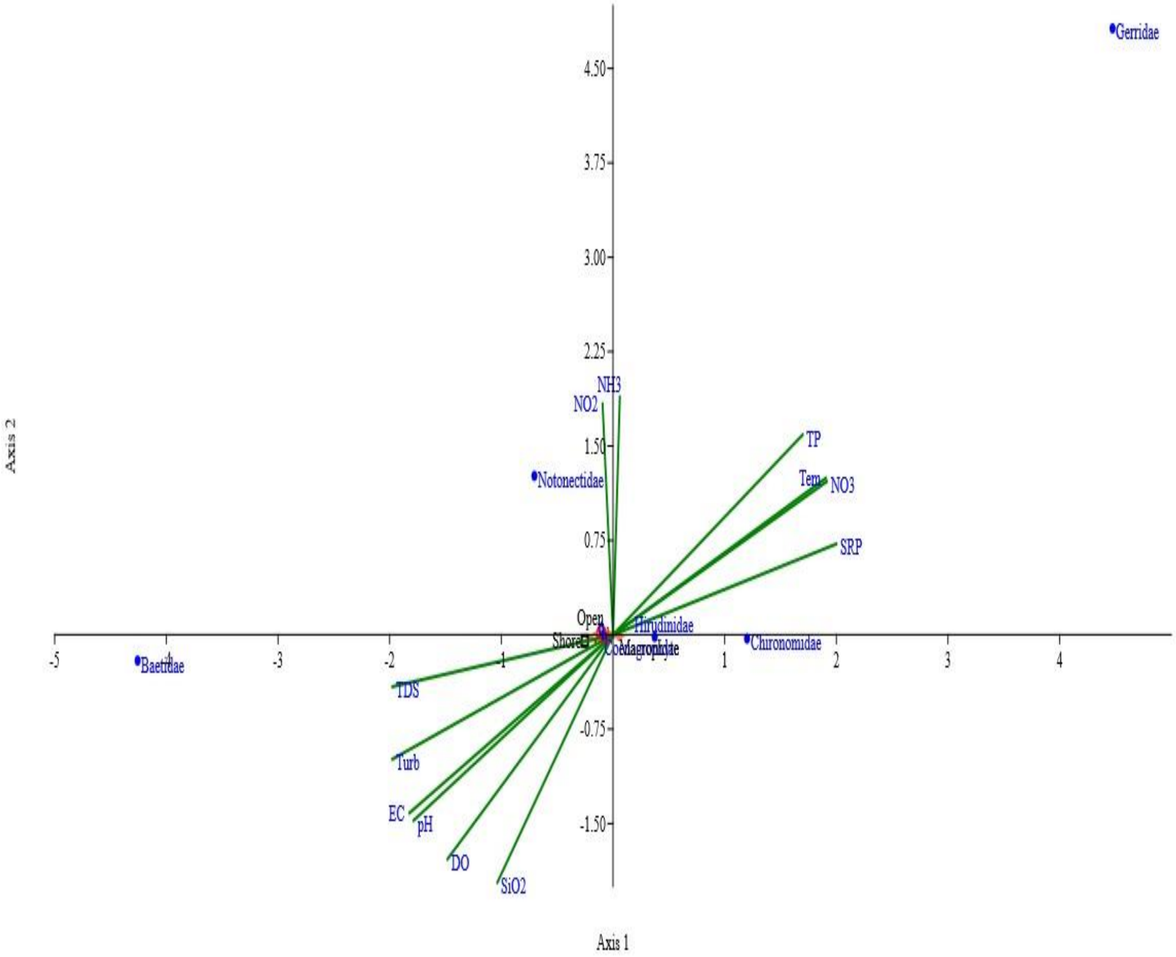

Figure 2. Relationships between physicochemical variables and macroinvertebrates distribution

CHAPTER 5. DISCUSSION

5.1. Temporal variations in physicochemical parameters

The data presented in (Table 2) indicates a consistent pH level across the months of February to May, averaging 7.5 ± 0.52. This slightly alkaline condition is generally favorable for aquatic life, promoting a balance in nutrient availability and biological activity. The observed pH range is consistent with findings from Soszka *et al.* (2016), who noted that pH levels in temperate freshwater systems often range from slightly acidic to slightly alkaline. Slightly alkaline conditions (pH ~7.5) are generally favorable for aquatic life, supporting a stable environment for nutrient availability and biological activity.

In other hand, the temperature readings remained relatively stable, with an average of 20.5 ± 0.19 °C, and the little fluctuations observed (from 20.3 °C in April to 20.6 °C in May) are typical for temperate aquatic systems, suggesting minimal thermal stratification during this period. Such stable temperatures are conducive to the metabolic processes of Macroinvertebrates, thus supporting biodiversity. The recorded stability in temperature aligns with the findings of Sokolova and Lannig, (2008) who reported that stable temperatures in temperate aquatic systems typically support metabolic processes and biodiversity. The minor fluctuations are typical in temperate regions, suggesting minimal thermal stratification, which is consistent with Cowley *et al.* (2022) who noted that stable temperatures contribute to a balanced aquatic ecosystem by preventing extreme changes that can stress aquatic organisms.

Electrical conductivity (EC) exhibited notable variability, especially with a peak of 192.2 ± 0.3 mS cm^{-1} recorded in April. This may be attributed to increased ion concentrations due to seasonal runoff or biological activity within the Lake, highlighting the potential impact of external factors on water chemistry. Conversely, a lower EC of 82.0 ± 1.5 mS cm^{-1} was observed in March, indicating a shift in ionic composition, possibly influenced by precipitation or dilution effects. The variability in EC observed, particularly high in April, is consistent with research by Sarma, *et al.* (2010) which highlighted that EC can fluctuate significantly due to factors such as seasonal runoff, precipitation, and biological activity while the lower EC in March could be due to dilution effects from precipitation or other sources. Studies by Shin *et al.* (2014) have shown similar patterns, where seasonal variations in EC reflect changes in water chemistry and external influences on ionic composition.

Dissolved oxygen (DO) levels showed a relatively narrow range, averaging 4.3 ± 0.78 mg L^{-1}. This level is critical for sustaining aquatic life, although the lower values in March (4.0 ± 1.05 mg L^{-1}) could indicate a temporary reduction in biological activity or increased organic matter decomposition. The stability of DO levels suggests a balanced ecosystem capable of supporting diverse Macroinvertebrates. Kengne *et al.* (2022), noted that DO level are critical for sustaining aquatic life, particularly for macroinvertebrates, which have specific oxygen requirements.

Turbidity measurements varied significantly, with a maximum of 30.08 ± 13.4 in February and a decrease to 16.14 ± 7.25 in March. This trend may reflect monthly changes in sediment suspension or algal blooms, which can influence light penetration and, consequently, photosynthetic activity in the Lake.

Similarly, Mason, (2022) discussed how increased turbidity can reduce light penetration, affecting photosynthetic activity and aquatic plant health and thus reduction in turbidity from February to March in this study could be beneficial for aquatic plants, promoting better light conditions and photosynthesis.

Total dissolved solids (TDS) showed minimal variation, averaging 0.02 ± 0.01, which suggests a stable water quality with limited fluctuations in the concentration of dissolved substances. Chapman and Kim, (2005) and Harrison and Whitfield, (2006) emphasized that stable TDS levels suggest a consistent water quality with limited fluctuations in the concentration of dissolved substances and the data in this study reinforces this view, indicating a stable aquatic environment.

5.2. Temporal variation in Nutrient parameters

The nitrate nitrogen (NO3-N) in (Table 3) recorded the highest in February with mean concentration at 86.4 μg L^{-1}, and decreased slightly in March to 78.7 μg L^{-1} and further to 80.50 μg L^{-1} in April. In May, the concentration reduced to 76.12 μg L^{-1}, suggesting a declining trend over the observed months.

This decrement could be attributed to several environmental factors, including potential biological uptake by phytoplankton during the spring bloom, which typically occurs in late April or early May, leading to reduced nutrient availability in the water column. Reynolds, (2012) highlighted that spring blooms of phytoplankton often lead to increased biological uptake of nutrients, including nitrate nitrogen in which as phytoplankton populations grow, they absorb nitrate from the water column, reducing its concentration.

Nitrite nitrogen (NO2-N) exhibited more variability, with a peak in February at 60.09 µg L^{-1}, followed by a significant drop to 30.88 µg L^{-1} in March, and a slight increase in April with a value of 35.17 µg L^{-1}. The variability could indicate dynamic biochemical processes, particularly nitrification and de-nitrification, which are influenced by temperature and microbial activity. The low levels observed in March may also indicate a period of reduced biological activity or dilution effects resulting from precipitation. Seitzinger *et al.* (2006) discussed how denitrification, the microbial reduction of nitrate to nitrogen gases, can influence nitrite levels. This process might explain the lower levels observed in March if denitrification was enhanced or if there was significant uptake of nitrite by aquatic plants or microorganisms.

Ammonia nitrogen (NH3-N) concentrations showed fluctuations with the highest value recorded in March at 63.95 µg L-^{1,} followed by a decrease to 39.50 µg L^{-1} in April, and a slight increase to 66.17 µg L^{-1} in May. The peak in March may suggest an influx of organic matter decomposition or possibly runoff, which can often be prevalent in early spring. Seitzinger *et al.* (2006) noted that the decomposition of organic material, which can be more pronounced in early spring, is a significant source of ammonia.

The peak observed could thus be linked to seasonal processes and increased nutrient input from runoff. Subsequent decrease could imply uptake by aquatic plants or assimilation by microorganisms, while the rise in May could indicate a shift in nutrient dynamics as temperature increases and biological activity intensifies. Silica (SiO2) showed relative stability throughout the months, with values ranging from 3.17 mg L^{-1} in March to 7.12 mg L^{-1} in February, suggesting that silica concentrations were less impacted by monthly changes compared to nitrogen compounds.

This stability indicates that the silica cycle may be less influenced by direct anthropogenic inputs or biological uptake in the short term. Soszka *et al.* (2016), and Harrison and Whitfield, (2006) observed that silica concentrations in freshwater systems can show relative stability compared to more reactive nutrients like nitrogen and phosphorus and Silica tends to be less subject to rapid changes because it is typically less influenced by short-term biological uptake and decomposition processes.

Total phosphorus (TP) concentrations varied significantly, high in April at 140.32 mg L^{-1}, and decreasing slightly to 134.80 mg L^{-1} in May. The high concentration in April may correlate with increased runoff or the re-suspension of sediments as water temperatures rise, stimulating biological activity and nutrient release from sediments. Whitehouse, (2010) noted that spring runoff is a significant source of phosphorus inputs into freshwater systems and this influx can temporarily elevate TP concentrations, especially if there is substantial agricultural or urban runoff. Soluble reactive phosphorus (SRP) also displayed a similar pattern, with its highest concentration observed in April at 104.80 mg $L^{-1,}$ and a decrease in May to 62.57 mg L^{-1}. The elevated SRP levels during April may contribute to algal blooms, which can have detrimental effects on water quality and aquatic life. Reynolds, (2006) explained that high SRP levels can lead to eutrophication and algal blooms.

5.3. Spatial variation in physicochemical parameters

The pH levels in (Table 4) indicate a range from 7.34 in the macrophyte site to 7.84 in the shore area. The macrophyte region shows a significantly lower pH compared to the shore area, which may be attributed to the higher productivity and organic matter decomposition occurring in the macrophyte site. Increased biological activity can lead to localized changes in pH due to the release of organic acids, whereas the shore area, being influenced by terrestrial runoff and less dense vegetation, may maintain a more stable and slightly alkaline pH. Similar to this, Harrison and Whitfield, (2006) suggest that shore areas are less affected by localized changes in pH and tend to maintain a more alkaline pH due to dilution effects and buffering from surrounding soils and sediments.

Temperature measurements across the sites were relatively consistent, with values ranging from 20.45 °C in the shore site to 20.48 °C in both the macrophyte and open water areas. This temperature suggests homogeneity in thermal conditions, likely due to the shallow nature of Lake Arekit and its exposure to sunlight, which can lead to uniform heating across different sites. Harrison and Whitfield, (2006) highlight that shallow lakes are exposed to sunlight more uniformly, leading to less thermal stratification and more homogeneous temperature distributions. Electrical conductivity (EC) exhibited notable variation, particularly with the shore area recording a substantially higher mean value of 164.56 mS cm^{-1} compared to the macrophyte and open water regions, which had values of 82.19 mS cm^{-1} and 93.55 mS cm^{-1}, respectively. This increased conductivity in the shore area may indicate higher concentrations of dissolved salts and minerals, often a result of runoffs that carry nutrients and pollutants from surrounding land.

Conley, (2002) discussed how storm water runoff and surface water drainage can increase the concentrations of dissolved ions in lake water, contributing to higher EC measurements in areas with significant runoff influence. Dissolved oxygen (DO) levels were relatively stable across the sites, with values ranging from 4.1 mg L^{-1} in the open water to 4.7 mg L^{-1} in the shore region. These levels are indicative of a well-oxygenated environment, which is crucial for the survival of aerobic Macroinvertebrates. The slight increase in DO in the shore area may be due to wave action and increased contact with the atmosphere, while the macrophyte area appears to have a lower DO concentration, possibly due to higher rates of respiration associated with dense plant biomass. Rynolds, (2006) discussed on how temporal and spatial variability in DO levels can be influenced by factors such as plant growth cycles, water temperature, and seasonal changes in which the current observations reflect the idea that DO levels are not static and can vary depending on environmental and biological factors, which is consistent with the findings in the literature.

Turbidity levels varied across the different sites, with the shore site showing the highest mean value of 25.88 NTU compared to the macrophyte (22.84 NTU) and open water (24.03 NTU) regions. Increased turbidity in the shore site may result from sediment resuspension due to wave action or runoff, which can negatively affect light penetration and, consequently, photosynthetic activity in submerged plants. Soszka *et al.* (2016) and Harrison and Whitfield, (2006) suggest that shore areas often experience higher turbidity due to wave action and increased sediment resuspension and waves can disturb the sediment on the lake bed, causing particles to become suspended in the water column and increasing turbidity.

Total dissolved solids (TDS) showed minimal variation among the sites, with values consistently low across all sampling locations, suggesting that the Lake's water quality remains relatively unaffected by excessive mineral concentrations. These low TDS levels are favorable for many Macroinvertebrates, indicating that the Lake has not yet reached a state of significant pollution or eutrophication. Soszka *et al.* (2016), noted that low TDS levels generally reflect good water quality, with minimal influence from dissolved minerals and pollutants and consistent low TDS values are indicative of a lake that has not been significantly impacted by industrial or agricultural activities that could contribute to higher mineral concentrations.

5.4. Spatial variation in Nutrient parameters

From (table 5) the recorded nitrate nitrogen (NO3-N) in the macrophyte area exhibited the highest concentration at 84.98 µg L^{-1}, compared to 74.21 µg L^{-1} in the shore site and 82.12 µg L^{-1} in the open water. The elevated levels of NO3-N in the macrophyte site may be linked to both biological activity and sediment interactions. The dense vegetation likely facilitates nutrient uptake and retention, while also contributing to localized nitrogen cycling through processes such as nitrification. The higher NO3-N levels in the Macrophyte area of Lake Arekit align with findings from Lake Victoria, Thiemer, (2022), where macrophyte zones often show elevated nitrate levels that is consistent with the role of macrophytes in nutrient cycling and decomposition. Nitrite nitrogen (NO2-N) concentrations were relatively consistent across the sites, with the macrophyte area showing 35.93 µg $L^{-1,}$ the shore at 36.29 µg $L^{-1,}$ and the open area at a higher level of 51.88 µg L^{-1}. The higher nitrite levels in the open water may suggest a different dynamics, possibly influenced by the decomposition of organic matter or a higher rate of nitrification occurring in that region.

These nitrite concentrations, while lower overall than nitrate, indicate active nitrogen cycling processes occurring in all areas. The elevated NO2-N in the Open area of Lake Arekit is consistent with the studies from Lake Erie (Knights, 2020) and Lake Geneva (Hellweger *et al.*, 2022) where open water areas often exhibit higher nitrite concentrations due to reduced biological activity and nutrient dynamics.

Ammonia nitrogen (NH3-N) concentrations varied slightly, with the open water site recording the highest mean at 56.45 µg L^{-1}, while the macrophyte and shore sites had similar values of 53.00 µg L^{-1} and 52.76 µg L$^{-1,}$ respectively. The higher ammonia levels in the open water could be a result of organic matter decomposition and less immediate uptake by vegetation compared to the macrophyte site, where plant uptake might limit. The slightly higher NH3-N in the Open area of Lake Arekit aligns with trends observed in Lake Huron (Chomicki *et al.,* 2022) and Lake Michigan (Miller *et al*., 2022), where open water areas sometimes exhibit higher ammonia levels due to reduced biological uptake.

The Shore area having the highest SiO2 concentration in Lake Arekit may be due to reduced biological uptake compared to the Macrophyte area. The higher SiO2 concentration in the Shore area of Lake Arekit is consistent with findings from Lake Michigan (Miller *et al.,* 2022) and Lake Ontario (Cohen *et al.,* 2024) where silica levels are often higher in shore areas due to less biological uptake and physical mixing.

The high TP in both the Macrophyte and Open areas of Lake Arekit could indicate nutrient loading and accumulation in these zones this is consistent with observations from Lake Victoria (Denny *et al.,* 2018) and Lake Erie (Baker *et al.,* 2019) where such areas often exhibit elevated phosphorus levels due to high nutrient inputs and accumulation and the elevated SRP in the Macrophyte area of Lake Arekit suggests a high concentration of reactive phosphorus, potentially related to biological processes and nutrient recycling. This is in line with Lake Ontario (Cohen *et al.,* 2016) and Lake Geneva (Hellweger, 2022), where high SRP levels are associated with high biological activity and nutrient sources.

5.5. Diversity of Macroinverebrates

5.5.1. Diversity indices of Macroinverebrates

Hilsenhoff Family-level Biotic Index (H-FBI)

Notonectidae (3.53) family (Table 6) is moderately tolerant to pollution. They are generally found in a range of water quality conditions but can indicate moderate levels of pollution. Coenagrionidae (2.86) family is relatively sensitive to pollution, indicating better water quality. Chironomidae (1.65**)** are quite tolerant of pollution, often found in polluted or degraded environments. Hirunidae (0.58) family is very sensitive to pollution, indicating good water quality when present in higher numbers.

Gerridae (0.097) is very sensitive to pollution and typically found in clean environments. Batidae (0.039) is with very low tolerance to pollution, indicating very high water quality if present in significant numbers. Lake Michigan Miller *et al.* (2022) macroinvertebrate communities with a high proportion of pollution-tolerant taxa like Chironomidae reflect areas with higher nutrient loading and poorer water quality.

Taxa Richness (Taxa_S)

The finding that both the macrophyte and open water areas have the highest taxa richness, with 6 taxa each, compared to the shore area which supports 5 taxa, provides insights into the biodiversity and ecological dynamics of Lake Arekit. Cornacchia *et al. (*2018) reported that aquatic plants enhance habitat complexity and, consequently, biodiversity and found that areas with dense macrophyte growth support greater macroinvertebrate diversity due to the availability of diverse microhabitats and stable environmental conditions.

Simpson's Dominance Index (Simpson_1-D)

The macrophyte area has the highest Simpson's Index of Diversity, indicating greater diversity. This is followed by the shore and open water areas, which have similar but slightly lower values. The observation that the macrophyte area has the highest Simpson's Index of Diversity, indicating greater diversity, with the shore and open water areas showing slightly lower values, provides important insights into the biodiversity and ecological dynamics of Lake Arekit. Dodds *et al.* (2009) suggest that open water environments with high nutrient levels and less complex habitats may exhibit lower biodiversity compared to more vegetated or complex areas. This aligns with the observed lower Simpson's Index in the open water area of Lake Arekit.

Shannon Index (Shannon_H)

The observation showed that the macrophyte area has the highest Shannon Index, followed by the shore and open water areas, provides valuable information about the biodiversity and community structure in Lake Arekit. Pérez *et al.* (2014) have noted that aquatic plant-dominated habitats generally exhibit higher Shannon Index values due to their ability to support diverse and stable communities. Cobb *et al.* (2014) found that shoreline habitats often have moderate diversity due to these stressors also Moss *et al.* (2006) and Dodds *et al.* (2009) have highlighted that simpler open water environments often exhibit lower Shannon Index values.

5.6. Relationships between physicochemical variables and macroinvertebrates distribution

From the Figure 2 Notonectidae shows a negative loading on Axis 1 and a positive loading on Axis 2, indicating it is associated with high values on Axis 2 and low values on Axis 1. A negative loading on Axis 1 indicates that the taxon is associated with lower values of TDS, Turbidity, EC, pH, DO and SiO_2.

However, the taxon is associated with high values of NH_3, NO_2, NO3, TP, Temperature and SRP. Taylor, (2011) show that taxa associated with lower nutrient levels are often found in low-nutrient conditions because they are adapted to lower organic pollution and have evolved to exploit resources in such settings.

Gerridae shows high positive loadings on both Axis 1 and Axis 2 (Figure 2), high positive loading on Axis 1 suggests that it is positively associated with TDS, Turbidity, EC, pH, DO and SiO_2, Gerridae is more prevalent in conditions where this factor is high. Similarly, Gerridae has high positive loading on Axis 2 indicates that it is also positively associated with NH_3, NO_2, NO3, TP, Temperature and SRP. This suggests that Gerridae is found in areas with higher values of the environmental factor(s) represented by this axis as well. This is similar with Sierp *et al*. (2009) show that Gerridae, can thrive in nutrient-rich environments. High nutrient concentrations can lead to increased primary productivity, which supports a greater abundance of prey and thus benefits predatory insects like Gerridae.

A high negative loading on Axis 1 indicates that Baetidae is strongly associated with lower values values of TDS, Turbidity, EC, pH, DO and SiO_2, Baetidae is more prevalent in conditions where this factor is low. A smaller negative loading on Axis 2 suggests that Baetidae is also associated with lower values of NH_3, NO_2, NO3, TP, Temperature and SRP, but the association is weaker compared to Axis 1. This indicates that while Baetidae prefers conditions with lower values of the factor represented by Axis 2, the influence is less pronounced. Similarly Boehrer and Schultze (2008) found that Baetidae, often thrive in low-nutrient or oligotrophic environments. Their presence is commonly associated with clear, less nutrient-enriched waters, which corresponds to a high negative loading on a nutrient-related axis (Axis 1).

Chironomidae's moderate positive loading on Axis 1 indicates that this taxon is positively associated with higher values of TDS, Turbidity, EC, pH, DO and SiO_2. This suggests that Chironomidae are more prevalent or abundant in conditions where the factors represented by Axis 1 are elevated. The near-zero loading on Axis 2 implies that Chironomidae's distribution is less influenced NH_3, NO_2, NO3, TP, Temperature and SRP on Axis 2. This means that while Chironomidae may be present across a range of conditions represented by Axis 2, their abundance is more strongly related to the conditions represented by Axis 1. Clements and Carlisle, (2003) observed that Chironomidae are often tolerant of a wide range of nutrient levels, including high nutrients which can indicate organic enrichment.

CHAPTER 6. CONCLUSION AND RECOMMENDATION

6.1. Conclusion

The study on Lake Arekit reveals significant temporal and spatial variations in both physicochemical and nutrient parameters, which have notable implications for water quality and aquatic life.

Monthly fluctuations observed in Physicochemical Parameters such as pH, temperature, electrical conductivity, dissolved oxygen, turbidity, and total dissolved solids indicating seasonal changes that influence water quality. For instance, increased temperatures in May resulted in reduced dissolved oxygen levels, which can impact aquatic organisms. Variations in nutrient Parameters like nitrate nitrogen, nitrite nitrogen, ammonia nitrogen, silica, total phosphorus, and soluble reactive phosphorus were noted, suggesting dynamic nutrient loading and potential eutrophication trends. Elevated phosphorus levels, particularly in May, could contribute to algal blooms and decreased water quality.

Physicochemical Parameters differences across macrophyte areas, shore areas, and open water highlighted the influence of habitat characteristics on water quality. For example, higher turbidity and total dissolved solids in macrophyte areas suggest the impact of aquatic vegetation on sediment and nutrient accumulation. Spatial variation in nutrient concentrations across different lake sites indicates diverse nutrient sources and their effects on local water chemistry. The higher nutrient levels in certain areas may exacerbate localized eutrophication and impact aquatic ecosystems differently across the lake.

The diversity and abundance of macroinvertebrates varied across sites, reflecting the influence of physicochemical conditions and nutrient availability. Macroinvertebrate communities showed varying tolerance to pollution, with certain taxa thriving in more nutrient-rich or turbid conditions, while others were more sensitive. Overall, the study highlights the complex interplay between temporal and spatial factors affecting water quality in Lake Arekit and underscores the importance of continuous monitoring to manage and protect aquatic ecosystems effectively

6.2 Recommendation

In order to improve the physical and chemical status as well as the biological integrity of the Lake, this study proposes that management of nonpoint sources of pollution, loss of natural riparian habitat, and occasional untreated point inflow are necessary. Water and riparian quality are predicted to improve as a result of management efforts to increase habitat complexity, which focused on preserving the morphological status of the littoral zone, weighed various shoreline modification or erosion protection strategies against their biological impacts. Furthermore, if there are plans to manipulate the Lake's water level, past knowledge of dose-effect connections pertaining to the amplitude and time of fluctuation must be utilized. The simple solution to nutrient enrichment and domestic cattle trampling is to fence out.

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APPENDICES

Appendix 1. Physical features of study area.

Appendix 2. Sample collection and Identification.

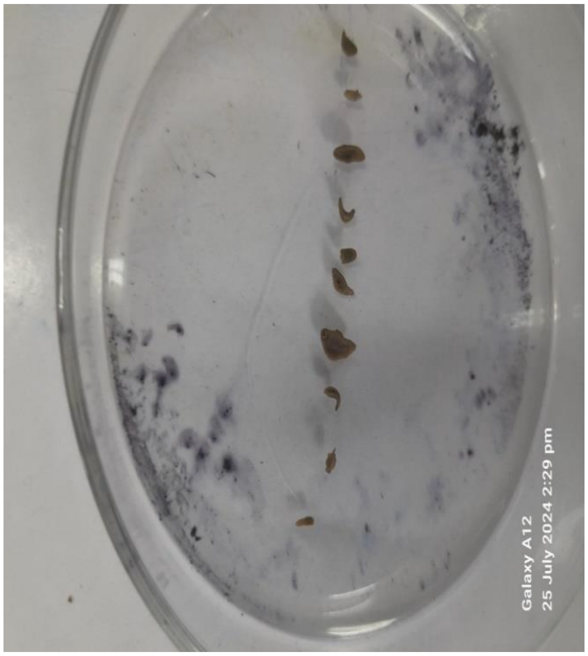
Galaxy A12
25 July 2024 11:17 am

Appendix 3. Result of the study

A group of Taxa

Coenagrionidae

Garridae



Hirunidae

Notonectidae

Chironomidae