International Journal of
BONOROWO WETLANDS

| Intl J Bonorowo Wetl | vol. 12 | no. 1 | June 2022 | ISSN 2775-8052 | E-ISSN 2775-8044 |

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International Journal of
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Published semiannually **PRINTED IN INDONESIA**

ISSN 2088-110X
E-ISSN 2088-2475

9 772088 110230

International Journal of
BONOROWO WETLANDS

| Intl J Bonorowo Wetl | vol. 12 | no. 1 | June 2022 |

ONLINE
http://smujo.id/bw

p-ISSN: 2775-8052, **e-ISSN:** 2775-8044

PUBLISHER
Society for Indonesian Biodiversity

CO-PUBLISHER
Institut Pertanian Bogor, Bogor, Indonesia
Nusantara Institut of Biodiversity, Universitas Sebelas Maret, Surakarta, Indonesia

OFFICE ADDRESS
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Saharjo BH, Nurhayati AD. 2006. Domination and composition structure change at hemic peat natural regeneration following burning; a case study in Pelalawan, Riau Province. Biodiversitas 7: 154-158. DOI: 10.13057/biodiv/d070213.

The usage of "et al." in long author lists will also be accepted:

Smith J, Jones M Jr, Houghton L et al. 1999. Future of health insurance. N Engl J Med 965: 325-329. DOI: 10.10007/s002149800025.

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Chapter in the book:

Webb CO, Cannon CH, Davies SJ. 2008. Ecological organization, biogeography, and the phylogenetic structure of rainforest tree communities. In: Carson W, Schnitzer S (eds.). Tropical Forest Community Ecology. Wiley-Blackwell, New York.

Abstract:

Assaeed AM. 2007. Seed production and dispersal of *Rhazya stricta*. 50th annual symposium of the International Association for Vegetation Science, Swansea, UK, 23-27 July 2007.

Proceeding:

Alikodra HS. 2000. Biodiversity for development of local autonomous government. In: Setyawan AD, Sutarno (eds.). Toward Mount Lawu National Park; Proceeding of National Seminary and Workshop on Biodiversity Conservation to Protect and Save Germplasm in Java Island. Universitas Sebelas Maret, Surakarta, 17-20 July 2000. [Indonesian]

Thesis, Dissertation:

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Information from the internet:

Balagadde FK, Song H, Ozaki J, Collins CH, Barnet M, Arnold FH, Quake SR, You L. 2008. A synthetic *Escherichia coli* predator-prey ecosystem. Mol Syst Biol 4: 187. DOI: 10.1038/msb.2008.24. www.molecularsystembiology.com.

INTL J BONOROWO WETLANDS
Volume 12, Number 1, June 2022
Pages: 1-11

P-ISSN: 2775-8052
E-ISSN: 2775-8044
DOI: 10.13057/bonorowo/w120101

Structure and composition of molluscs (bivalves and gastropods) in the mangrove ecosystem of Pacitan District, East Java, Indonesia

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Manuscript received: 27 December 2021. Revision accepted: 14 February 2022.

Abstract. *Wiraatmaja MF, Hasanah R, Dwirani NM, Pratiwi AS, Riani FE, Hasnaningtyas S, Nugroho GD, Setyawan AD. 2022. Structure and composition molluscs (bivalves and gastropods) in the mangrove ecosystem of Pacitan District, East Java, Indonesia. Intl J Bonorowo Wetlands 12: 1-11.* The mangrove ecosystem is a habitat for several species of molluscs. Therefore, mollusc's existence has an important role in the sustainability of the mangrove ecosystem. This research was conducted to determine the structure and composition of molluscs (gastropods and bivalves) in the mangrove ecosystem of Pacitan, East Java, Indonesia. The mollusc sampling was carried out at three mangrove ecosystems: Teleng Ria, Grindulu, and Siwil. Each sampling research location made a plot of 10 x 10 m. Then, the collected individual and mollusk species were analyzed using the Shanon-Weiner biodiversity index, species density, Margalef species richness index, evenness index, and Simpson dominance index. The molluscs found were 25 species, consisting of 17 gastropod species with 13 families. The gastropod families with the most species are Littorinidae (3), Ellobidae (2), and Potamididae (2). Meanwhile, there are eight species of bivalves consisting of five families. Then, Mytilidae (2), Ostreidae (2), and Veneridae (2) are the families with the most number of species in bivalves. The total molluscs obtained had a density of 36.9 individuals/100 m^2. The mollusc biodiversity index was 2.14 (medium), mollusc species richness reached 4.1 (medium), the evenness of species in this research reached 0.66 (quite even), and the species dominance index was recorded at 0.34 (low), which means that there are no dominant species in the research location. Then, each research location has a different species and density species of molluscs. That is caused by differences in the type of substrate (such as sandy, muddy, or rocky) and abiotic factors found in the research location (such as temperature, pH, and salinity).

Keywords: Bivalves, gastropods, mangrove ecosystem, mollusc diversity, Pacitan

INTRODUCTION

Indonesia has the largest island group in the world, with a very wide sea. It has 17,508 islands and a coastline of 81,000 km^2, the second-longest in the world after Canada (Wiryawan et al. 1999). Based on this, Indonesia has one of the largest and highest biodiversity in the world. Moreover, it has many diverse coastal ecosystems, such as coral reefs, extensive seagrass beds, and mangrove forests. However, not all coastal areas are covered with mangroves, and this is due to several requirements or environmental factors that control the formation of these forests, such as coastal physiography and climate.

Mangrove ecosystems are ecosystems located in coastal areas, and this area is affected by the tides of seawater so that the bottom is always flooded. Therefore, mangrove ecosystems have very important benefits in supporting living things from a physical, ecological, and economic perspective (Li et al. 2015; Alvareza and Leilani 2020). The benefits of mangrove ecosystems for humans are disaster mitigation, such as wave absorbers, and coastal protection from abrasion, tidal waves, and tsunami; this is because they are located on the border and close to the sea (Harahab and Setiawan 2017). In addition to its benefits in preventing natural disasters, another benefit of the mangrove ecosystem is as a natural tourist attraction because it is a cool place even though it is on the beach (Rahmila and Halim 2018).

In addition to its benefits for humans, the mangrove ecosystem can also function as a good habitat for various species of animals, especially animals that live in water. For example, mangrove ecosystems can be natural places for breeding and foraging fish, crabs, shrimp, and other organisms (Igulu et al. 2014; Shing et al. 2014; Vermeiren et al. 2015; Onyena and Sam 2020; Irwansyah et al. al. 2021).

In addition to the animals mentioned above, the mangrove ecosystem is also a habitat for molluscs.

Molluscs are the second-largest invertebrate group in the world, most of which members live in water (Saputra et al. 2020). Molluscs have the two largest members of the class, namely bivalves, and gastropods (Irma and Sofyatuddin 2012; Marshall et al. 2015). Both have different body shapes and shell sizes. This shell modification has the function of helping to distinguish the two classes (Dolorosa and Gallon 2014). Gastropods have a single-threaded shell characteristic, while bivalves have two dorsally interlocked shells (Nur'aini 2012). Gastropod habitats are found on various sand-mud slopes, and this is because gastropods are infauna animals, which give a striking reaction to the size of the texture of the seafloor, while bivalves have different characteristics way of life from gastropods (Dibyowati 2009). The way of life of bivalves is digging, immersing, and gluing themselves using an adherent device to the substrate (Ulmaula et al. 2016).

Bivalves and gastropods have many important values in human life because they can be used as food. In addition, the presence of molluscs, such as bivalves and gastropods, in an ecosystem can be used as a reference to assess the ecological quality of the ecosystem or as a bioindicator of environmental health (El-Sorogy et al. 2013; Sharma et al. 2013). If there are bivalves and gastropods with various species, the quality of the environment and water is still sustainable. In addition, the important role of bivalves and gastropods concerning biotic components in the mangrove ecosystem, apart from being detritus, plays a role in the process of litter decomposition and neutralizes organic matter that is herbivores and detrivores. That plays an important role in the food chain for ecosystem stability because it can support the life of other higher trophic animals (Pawar 2012; Suresh 2012; Irma and Sofyatuddin 2012). The existence and distribution of bivalves and gastropods are strongly influenced by pressure and changes from abiotic and biotic factors, such as environmental conditions, food sources, predation, competition, vegetation, and human activities (Akhrianti et al. 2014). Geographically, bivalves and gastropods are widely distributed on all continents except Antarctica, but patterns of diversity and distribution differ across regions: some areas represent high mollusc diversity (Neubauer et al. 2015), whereas others are faunistically poor, for example, Greenland (Vinarski et al. 2017).

Pacitan District, East Java, Indonesia, has a sustainable mangrove ecosystem with white sand and muddy and rocky substrate as a habitat for several species of mangrove plants and animal organisms (Irwansyah et al. 2021). The research about mangrove trees species in the Pacitan mangrove ecosystem is dominated by *Rhizophora mucronata, R. stylosa, Avicennia alba, Sonneratia alba, Nypa fruticans, A. marina* (Setyawan et al. 2002; Sholiqin et al. 2021). The mangrove vegetation creates habitat and provides suitable food for various living fauna. However, until now, the existence of molluscs, especially bivalves, and gastropods, both in terms of diversity and ecological aspects in the Pacitan mangrove ecosystem, has not been widely carried out by researchers. Whereas knowing the existence of molluscs in an ecosystem can potentially meet human economic needs, become a regulator of mangrove ecosystems, and other benefits for humans and nature.

The limited information about molluscs in Pacitan encourages researchers to determine the structure and composition of molluscs (bivalves and gastropods) in the Pacitan mangrove ecosystem. Furthermore, the information obtained from knowing the diversity of molluscs in this area is expected to be useful for planning the management of the coastal area of Pacitan District and extracting the potential of molluscs in the mangrove ecosystem of Pacitan District, East Java, Indonesia.

MATERIALS AND METHODS

Study site

The sampling was carried out on November 2021 and is located on the mangrove ecosystem of Pacitan District, East Java, Indonesia. Pacitan District is at coordinates between 7º55'-8º17'S and 110º55'-111º25'E. In 2021, according to Badan Pusat Statistik Kabupaten Pacitan (2021), the air temperature was 26-29°C, the average humidity was 23-27.5%, the number of rainy days is 179 days with rainfall 841 mm^3. Mangrove areas in Pacitan District are generally not polluted, and the land has the potential to grow mangrove plants (Dinas Kelautan dan Perikanan Kabupaten Pacitan 2014). The sampling consisted of three locations close to the beach: Teleng Ria, Grindulu, and Siwil. Teleng Ria and Grindulu in Pacitan Sub-district and Siwil in Ngadirojo Sub-district, Pacitan District, East Java, Indonesia (Figure 1).

Teleng Ria. The location research is at coordinates 08º13'19.63"S and 111º04'28.82" E. The vegetation mangrove trees of *A. alba* dominate this location. It has sandy and muddy soil characteristics because it is a riverbank area that leads directly to Teleng Ria Beach (Figure 2A). This location is also at the mouth of the river but not too close to Teleng Ria Beach and is a fishing pier and residential area.

Grindulu. This location is near Grindulu Beach at coordinates 08º13'55.75" S and 111º06'21.42"E. Grindulu is a mangrove conservation area still under development. It is evident from the number of mangrove plant seeds planted. Later this area will become ecotourism Watu Mejo Park and a mangrove nursery center in East Java. This location is a sandy and muddy area dominated by vegetation mangrove trees of *R. stylosa* (Figure 2B).

Siwil. The third location in this research is a river estuary close to Siwil Beach (08°15'37.4"S and 111°17'00.1"E) dominated by the vegetation mangrove tree of *S. alba.* This location is a tidal area with sandy, muddy, and rocky substrate (Figure 2C). According to Sholiqin et al. (2021), this location is intended for mangrove restoration after hit a big flood at the end of 2017. Also, it intended prevention for the flood season. This location's estimated area of mangrove trees is the smallest compared to Teleng Ria and Grindulu.

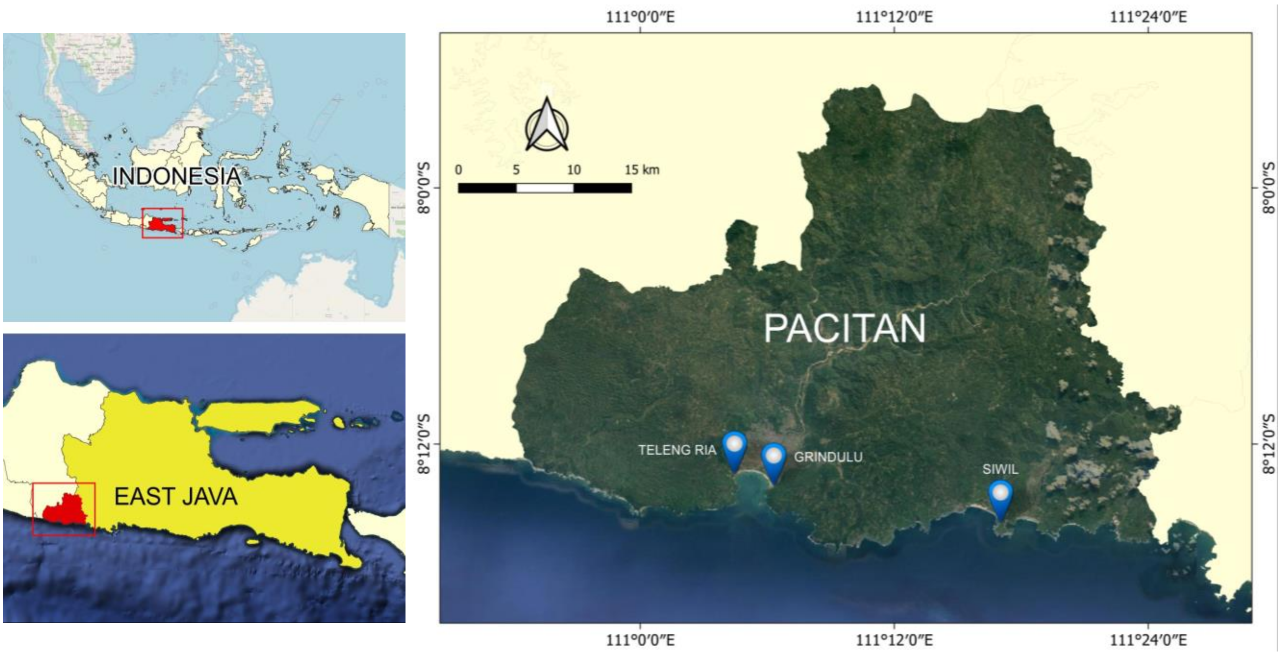

Figure 1. The sampling area in the mangrove ecosystem of Pacitan District, East Java, Indonesia (Location 1: Teleng Ria; location 2: Grindulu; location 3: Siwil)

A **B** **C**

Figure 2. Condition of each mangrove ecosystem of Pacitan District, East Java, Indonesia: A. Teleng Ria; B. Grindulu; C. Siwil

Sampling molluscs

Sampling was conducted by making transect 10 x 10 m^2 plots in each location research. This method is a modified method of the published journal by Islamy and Hasan (2020). The plot selection was based on the condition of the substrate, i.e., sandy, muddy, rocky, etc. The mollusc specimens in the plots were collected by handpicking on the surface of the substrate, and the arboreal was collected from mangrove trees like stems, roots, and leaves, following Kantharajan et al. (2017). After that, the species and number of mollusc in each plot were counted and recorded. Then, abiotic factors measured, including temperature (air, water, and soil), pH (water and soil), and water salinity, were recorded. The dominant species of mangrove trees were also recorded in each location research.

Identification molluscs

The samples that had been collected from the research location were preserved using 70% alcohol, then identified using literature studies by Irma and Sofyatuddin (2012), Dolorosa and Gallon (2014), Kementerian Kelautan dan Perikanan Indonesia (2014), Baderan et al. (2019), and Islamy and Hasan (2020). The species identification process is carried out at the Laboratory of Animal Taxonomy, Faculty of Mathematics and Natural Sciences, Universitas Sebelas Maret, Surakarta, Indonesia. Identification of molluscs was carried out by observing morphological characteristics such as shell shape, shell color, and shell pattern. Once identified, the sample was photographed using a digital camera and stored in a sample bottle.

Data analysis

Data on the number of species and the number of individuals of each species that have been collected will be analyzed using the density formula, Shanon-Weiner diversity index (H'), Margalef species richness index (R), Evenness index (E) and Simpson dominance index (D) which will be described as follows:

Density

$$Density\ (K) = \frac{number\ of\ individuals}{plot\ area}$$

Shanon-Weiner diversity index (H')

$$H' = -\Sigma pi \times \ln pi$$

Where:
H' : Diversity index of Shannon-Wiener
Pi : Number of individuals of the *i* species divided by the total number of individuals

Evenness index (E)

$$E = \frac{H'}{\ln(S)}$$

Where:
E : Evenness index
H' : Shanon-Weiner diversity index
S : Total number of species found

Simpson dominance index (D)

$$D = \sum_{1}^{s} \frac{n_i(n_i - 1)}{N(N-1)}$$

Where:
D : Dominance index
n_i : Number of individuals of the *i* species
N : Total number of individuals

Margalef species richness index (R)

$$R = \frac{s-1}{\ln(N)}$$

Where:
R : Richness index
S : Total number of species found
N : Total number of individuals found

RESULTS AND DISCUSSION

Molluscs at the research location

A total of 25 mollusc species were recorded from three research locations, namely Teleng Ria, Grindulu, and Siwil (Table 1, Figure 3). Gastropods are represented by 17 species (Figure 3A-3Q) with 13 families, namely Achatinidae, Ampullariidae, Batillariidae, Cypraeidae, Ellobiidae, Littorinidae, Naticidae, Neritidae, Lottidae, Pachychilidae, Partulidae, Patellidae, and Potamididae. Meanwhile, eight species of bivalves were recorded (Figure 3R-3Y) from five families: Arcidae, Myidae, Mytilidae, Ostreidae, and Veneridae. In gastropods, the most common families in terms of the number of species are Littorinidae (3), Ellobidae (2), and Potamididae (2), and the others are only one species (Table 1). Members of the family Littorinidae are arboreal and are found attached to the roots, trunks, or leaves of mangrove trees (see Figure 4). Then, Mytilidae (2), Ostreidae (2), and Veneridae (2) are the dominant families among bivalves (Table 1). Of the 17 species, 13 are terrestrial gastropods; *Achatina fulica* (Bowdich, 1822) *Pomacea canaliculata* (Lamarck, 1822); *Batillaria zonalis* (Bruguière, 1792); *Ellobium aurisjudae* (Linnaeus, 1758); *Pythia scarabaeus* (Linnaeus, 1758); *Littoraria angulifera* (Lamarck, 1822); *Littoraria scabra* (Linnaeus, 1758); *Littoraria* sp.; *Neritina turrita* (Gmelin, 1791); *Faunus ater* (Linnaeus, 1758); *Partula* sp.; *Pyrenella cingulata* (Gmelin, 1791); and *Terebralia sulcata* (Born, 1778). Then the other four species are marine gastropods; *Cypraea annulus* (Linnaeus, 1758), *Patelloida* sp., *Patella* sp., and *Polinices* sp. While in bivalves, eight species were found; *Anadara granosa* (Linnaeus, 1758); *Mya* sp.; *Mytilus* sp.; *Perna viridis* (Linnaeus, 1758); *Crassostrea gigas* (Thunberg, 1793); *Ostrea* sp.; *Meretrix* sp.; and *Paphia* sp.

The density of molluscs in the research location

The total density of molluscs found at the three research locations was 36.9 individuals/100 m^2 (ind/100 m^2). Each species density value in each location, such as Teleng Ria with 11.4 ind/100 m^2, Grindulu with 1.6 ind/100 m^2, and the highest in Siwil with 23.9 ind/100 m^2. The species *F. ater* was the species with the most individuals in Teleng Ria with 3.1 ind/100 m^2, while *A. granosa* was the species with the most individuals in Grindulu with 0.8 ind/100 m^2, then with a value of 16.2 ind/100 m^2 by species *P. scarabaeus*, thus making this species the most abundant individual found in Siwil. That makes *P. scarabaeus* the species with the most individuals in this research. Meanwhile, the lowest species number ind/100 m^2 with a value of 0.1 ind/100 m^2 were *Paphia* sp., *B. zonalis*, *E. aurisjudae*, *Polinices* sp., and *T. sulcata*. The density of a species shows the number of individual species with a certain unit area. Table 1 provides detailed information regarding this.

Ecological index in the research location

Shanon-Weiner diversity index (H'). Teleng Ria has the highest diversity index value in this research, 2.02. Meanwhile, Siwil has a diversity index value of 1.2 and is followed by Grindulu as the lowest diversity index value in this research, with 1.04 (Figure 5). According to Krebs (1989), the criteria for the diversity index are; $H'<1.5$, then the species diversity is low; $1.5 <H'<3.5$ means that the species diversity is moderate; and $H'>3.5$, then the species diversity is high. So in this research, Teleng Ria has a moderate diversity index category, followed by Siwil and Grindulu, whose diversity index category is low. However, if three locations are combined, the value reaches 2.14, categorized as moderate. The diversity index (H') describes the state of the population of organisms mathematically to make it easier to analyze the number of individuals of each species in a community.

Evenness index (E). The evenness index determines the evenness of each species in each community encountered. According to Pielou (1977), if the results show a value of 0.00-0.25, it means that the evenness index is uneven; if the result shows a value of 0.26-0.50, it means that the evenness index is less evenly distributed; if the results show a value of 0.51-0.75, it means that evenness index is relatively even; if the result shows a value of 0.76-0.95, it means that the evenness index is almost evenly; then if the result shows a value of 0.96-1.00, it means that the evenness index is evenly distributed. The total evenness index of the three locations shows a value of 0.66 which means quite even. In contrast to the total results of the research locations, Teleng Ria has a value of 0.81 and Grindulu of 0.95, so it is categorized almost evenly. The lowest evenness index value is in the Siwil, which shows a value of 0.45, so it is categorized as less evenly distributed (Figure 5).

Simpson dominance index (D). The species dominance index values from highest to lowest were Teleng Ria, Grindulu, and then Siwil, with each value being 0.47, 0.36, and 0.21. At the same time, the overall value of all locations is 0.34. Simpson's dominance index is a parameter that states the level of centralized dominance (mastery) of a species in a community. Suppose the results show a value of $0<C<0.5$, it means that the species dominance is low; if the result shows a value of $0.5<C<0.75$, it means that the species dominance is moderate; if the results show a value of 0.75-1.00, it means that species dominance is high (Odum 1971). Therefore, based on Odum's (1971) explanation, all the results of this research were categorized as low, meaning there were no dominant species in each location (Figure 5).

Margalef species richness index (R). Species richness is the number of species in a certain area. Species richness index combined with individual abundance/density values for each unit. The highest species richness index value was obtained at the research location of Teleng Ria with a value of 2.38, followed by Siwil at 2.19, and the lowest was Grindulu, which was 0.96. According to Magurran (1998), if the results show a value of $R>5$, it means that species richness is high; if the result shows a value of $3.5<H<5$, it means that the species richness is moderate; if the results show a value of $H<3.5$, it means that species richness is low. Therefore, the three research locations have the same low species richness category. However, in the total calculation of all locations, the value reached 4.1, which can be categorized as medium species richness.

For more details on the comparison of all the results of the ecological index in each research location, see Figure 5.

Environmental (abiotic) factors in the research location

The suitable environmental conditions for molluscs' life will help them play an important role in the mangrove ecosystem. In this research, the parameters of temperature (air, water, and soil), pH (water and soil), and salinity were measured (Table 2). Measurements of all temperatures at the three sampling locations ranged from 28-33°C. The highest temperature was recorded at Siwil. Then, all pH at the three locations ranged from 6-8, with the lowest value (close to acid) being Teleng Ria. The results of temperature and pH measurements at research locations tend to vary. Then, Teleng Ria and Siwil have the same salinity, 5 ppt, while in Grindulu, it is 10 ppt. In general, the three research locations are not too close to the sea so that the salinity value of the water is not too salty.

Table 1. List and density of molluscan species recorded in the mangrove ecosystem of Pacitan District, East Java, Indonesia

Class	Family	Species	Density (ind/100 m^2) in each location			Total
			Teleng Ria	Grindulu	Siwil	
Bivalves	Arcidae	*Anadara granosa* (Linnaeus, 1758)	-	0.8	-	0.8
	Myidae	*Mya* sp.	-	-	3.5	3.5
	Mytilidae	*Mytilus* sp.	-	0.4	-	0.4
		Perna viridis (Linnaeus, 1758)	-	-	0.4	0.4
	Ostreidae	*Crassostrea gigas* (Thunberg, 1793)	2,6	-	-	2.6
		Ostrea sp.	0.2	-	-	0.2
	Veneridae	*Meretrix* sp.	0.2	-	-	0.2
		Paphia sp.	-	-	0.1	0.1
Gastropoda	Achatinidae	*Achatina fulica* (Bowdich, 1822)	0.2	-	0.1	0.3
	Ampullariidae	*Pomacea canaliculata* (Lamarck, 1822)	2	0.4	-	2.4
	Batillariidae	*Batillaria zonalis* (Bruguière, 1792)	0.1	-	-	0.1
	Cypraeidae	*Cypraea annulus* (Linnaeus, 1758)*	-	-	1.1	1.1
	Ellobiidae	*Ellobium aurisjudae* (Linnaeus, 1758)	-	-	0.1	0.1
		Pythia scarabaeus (Linnaeus, 1758)	-	-	16.2	16.2
	Littorinidae	*Littoraria angulifera* (Lamarck, 1822)	0.4	-	-	0.4
		Littoraria scabra (Linnaeus, 1758)	-	-	0.2	0.2
		Littoraria sp.	0.8	-	1	1.8
	Lottiidae	*Patelloida* sp.*	-	-	0.5	0.5
	Naticidae	*Polinices* sp. *	-	-	0.1	0.1
	Neritidae	*Neritina turrita* (Gmelin, 1791)	1.1	-	-	1.1
	Pachychilidae	*Faunus ater* (Linnaeus, 1758)	3.1	-	-	3.1
	Partulidae	*Partula* sp.	0.5	-	-	0.5
	Potamididae	*Pirenella cingulata* (Gmelin, 1791)	0.2	-	-	0.2
		Terebralia sulcata (Born, 1778)	-	-	0.1	0.1
	Patellidae	*Patella* sp.*	-	-	0.5	0.5
Total			11.4	1.6	23.9	36.9

Note: -: not found, *: marine gastropods

Table 2. Environmental (abiotic) factors in the mangrove ecosystem of Pacitan District, East Java, Indonesia

Location of research	Temperature °C			pH		Salinity (ppt)
	Air	Water	Soil	Water	Soil	
Teleng Ria	33	32.6	33	7.6	6	5
Grindulu	32	32.5	33	8	7	10
Siwil	32	28.2	31	7.6	6.1	5

Figure 3. Checklist molluscs in the coastal of Pacitan District, East Java, Indonesia: A. *Achatina fulica* (80 mm), B. *Pomacea canaliculata* (50 mm), C. *Pirenella cingulata* (15 mm), D. *Faunus ater* (15 mm), E. *Batillaria zonalis* (13 mm), F. *Neritina turrita* (19 mm), G. *Littoraria angulifera* (21 mm), H. *Polinices* sp. (37 mm), I. *Pythia scarabaeus* (18 mm), J. *Terebralia sulcata* (22 mm), K. *Littoraria* sp. (15 mm), L. *Partula* sp. (10 mm), M. *Littoraria scabra* (29 mm), N. *Ellobium aurisjudae* (34 mm), O. *Patelloida* sp. (15 mm), P. *Patella* sp. (16 mm), Q. *Cypraea annulus* (20 mm), R. *Anadara granosa* (20 mm), S. *Ostrea* sp. (64 mm), T. *Crassostrea gigas* (89 mm), U. *Mya* sp. (46 mm), V. *Mytilus* sp. (25 mm), W. *Perna viridis* (24 mm), X. *Paphia* sp. (59 mm), Y. *Meretrix* sp. (55 mm)

Figure 4. *Littoraria scabra* (Gastropod) clinging to the leaves of the *Sonneratia alba* tree

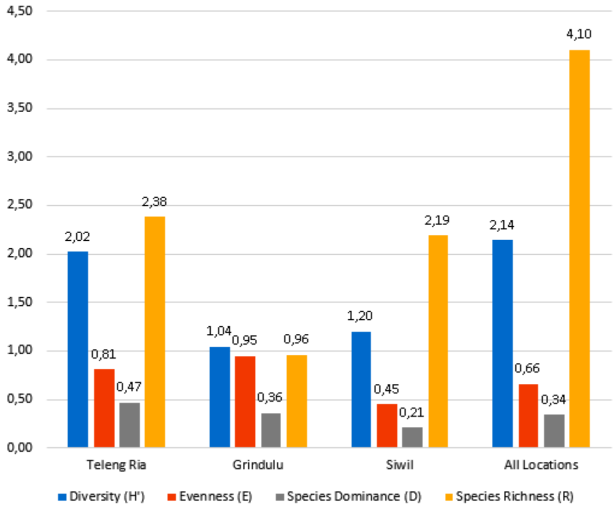

Figure 5. Comparison of ecological index values at each research location

Discussion

Mollusc studies elsewhere also found that the species diversity of gastropods was higher than bivalves. For example, 202 species of gastropods were found in France, Europe, while 39 species of bivalves (Bichain et al. 2019). Then in the Americas, precisely in the Strait of Magellan, Chile, Aldea et al. (2020) reviewed 134 research articles on the diversity of molluscs (gastropods and bivalves) in the area and managed to collect data on 173 gastropod species and 107 bivalves species. Then on the African continent, Morocco to be precise, 26 gastropods and ten bivalves were found (Irikov and Gerdzhikov 2013). In the Australian continent, Murphy (2015) identified 142 species comprising 34 bivalves (24 families) and 108 gastropods (51 families). Finally, some countries in Asia, such as the Philippines, found up to 50 gastropods and 15 bivalves (Dolorosa and Gallon 2014), while in India, 46 gastropods and 14 bivalves were found by Kantharajan et al. (2017). Meanwhile, in another area in Indonesia, Baderan et al. (2019) also found 21 species of gastropods and three species of bivalves. The higher diversity of gastropods compared to bivalves in mangrove ecosystems is due to the ability of gastropods to tolerate better environmental changes and harsh conditions in these ecosystems (Dolorosa and Gallon 2014).

This research revealed that mollusc families, such as Potamididae, Littorinidae, Ellobiidae, and Neritidae, have a high abundance of individuals and are commonly found in mangrove ecosystems near mangrove trees (Table 1). According to Walthew (2012), Baderan et al. (2017), and Yadav et al. (2019), the family mentioned previously can dominate in the mangrove ecosystem because it likes to live in areas affected by tides and muddy areas with mangrove trees vegetation. Meanwhile, the distribution of the abundance of bivalves is generally limited to a very narrow zone of the low tide boundary due to the need for feeding and larval life. Very few species, such as *A. granosa*, *C. gigas*, *Mytilus* sp., *Mya* sp., *Ostrea* sp., and *P. viridis*, can adapt well to the spray zone and changes in environmental conditions in intertidal mangrove areas (Rangan 2010; Khade and Mane 2012; Kabir et al. 2014; Kementerian Kelautan dan Perikanan Indonesia 2014; Samson and Kasale 2020). Therefore, that species was the dominant bivalve found in this research locations (Table 1).

Among the molluscs found in this research, the following species have the potential to be cultivated for food: such, *Meretrix* sp., *A. granosa*, *P. canaliculata*, *P. viridis*, *F. ater*, and *Ostrea* sp. (Lok et al. 2011; Khade and Mane 2012; Sawant and Mohite 2013; Shan et al. 2013; Wang et al. 2013; Pritchard et al. 2015; Soon and Ransangan 2016; Agustina et al. 2019). Therefore, exploring the cultivation potential of this species can help provide a stable source of economic income among local residents. First *P. scarabaeus*, the highest abundance in this research, and most other members of Ellobiidae were considered pollution indicators in mangrove ecosystems (Satheeshkumar and Khan 2012; Dissanayake and Chandrasekara 2014; Kantharajan et al. 2017). Next, *N. turrita* has a familiar shell used for aquarium ornaments because of its unique color (Hristov 2020). Finally, *Achatina fulica*, *P. canaliculata*, and *P. cingulata* are invasive animal species because they become a pest to agricultural land and pond (Karraker and Didgeon 2014; Goldyn et al. 2017; Solanki et al. 2017).

Family Littorinidae has the most species in this research (Table 1). There are three species, including *L. angulifera*, *L. scabra*, and *Littoraria* sp. The Littorinidae family is a facultative gastropod because many individuals and species are found inside and outside the mangrove ecosystem (Susanti et al. 2021). Khade and Mane (2012) and Marshall et al. (2015) suggested that *Littoraria* lives on mangrove trees' trunks, branches, roots, and leaves. Several factors, such as the function of an area as a shelter for molluscs and the species of vegetation mangrove trees, determine the distribution of species. In addition, *P. scarabaeus* and *Partula* sp. are also a gastropod that lives on parts of mangrove trees to breed and find food (Hopper and Smith 1992; O'Rorke et al. 2016; Sischo and Hadfield 2017). This species is very abundant in this research location and lives in mangrove trees, such as A. alba and *S. alba*.

The stability of the mollusc community can be described with analysis of the values of diversity index (H'), evenness (E), dominance (D), and species richness (R). The results of the Shannon-Wiener diversity index in each location showed different values (Figure 5). The mollusc diversity index in Teleng Ria was higher (2.02) than in Siwil (1.2) and Grindulu (1.04). The mollusc diversity index value in Teleng Ria is moderate compared to Siwil and Grindulu, which are low. The same thing happened with Teleng Ria, the diversity index value in general from three locations research is 2.14 (medium). Diversity includes two main things: variations in the number of species and the number of individuals of each species in an area. At each location, the abundance of each species varied in number because several species were much larger than other species, resulting in a low diversity of an ecosystem (Table 1). Suppose the number of species and the variation in the number of individuals of each species is relatively low. In that case, it means an imbalance in the ecosystem caused by disturbance or pressure. Research with a moderate value diversity index was also found by Hasidu et al. (2020) in the mangrove ecosystems of Kolaka District, Southeast Sulawesi, Indonesia, and Yadev et al. (2019) in mangrove ecosystems of Paradeep, east coast of India. Furthermore, Baderan et al. (2019) explain that the moderate diversity index is enough to describe the complexity of the ecosystem.

Teleng Ria has a value of 0.81 and Grindulu of 0.95, so the evenness index is categorized almost evenly. At the same time, Siwil shows the lowest evenness value in this research with 0.45, so it is categorized as less even. Meanwhile, the total evenness index for three locations shows a value of 0.66 which means quite even (Figure 5). The evenness index value is close to 1, indicating the number of individuals each species is almost the same and the stability of the ecosystem. On the other hand, if the evenness index is close to 0, it means a certain species predominate in the ecosystem (Asadi et al. 2018; Baderan et al. 2019; Hasidu et al. 2020).

The dominance index value obtained from each research location is between 0.21-0.47 (Figure 5). The dominance value is used to determine whether certain species dominate an ecosystem. Based on this, the dominance index obtained is mostly close to the value 0, meaning no particular species dominate in the community (Odum and Barrett 2005). Furthermore, the value of the evenness index with the dominance index in this research shows inversely proportional results. The evenness index value is close to 1, and the dominance index is close to 0 (zero). That indicates that the evenness of species is evenly distributed in an ecosystem, and no dominant species exist.

The highest richness index value was obtained at the research location of Teleng Ria with a value of 2.38, followed by Siwil at 2.19, and the lowest was Grindulu, which was 0.96 (Figure 5). However, in the total calculation of all locations, the value reached 4.1, which can be categorized as medium species richness. The species richness index is the simplest measure of biodiversity because it only considers differences in the number of species in a certain area (Magurran 1998). Species richness in an area depends on the health of its habitat or ecosystem. If the habitat or ecosystem can provide food and a breeding ground for a species, the species richness can be high (Aditya et al. 2019).

The numbers of individual and species molluscs in the three research locations have differed. According to Odum and Barrett (2005) and Stagg and Mendelssohn (2012), the type of substrate and mangrove vegetation greatly determines the density and composition of molluscs. The research location in Siwil has the most varied substrate types compared to other locations, such as sandy, rocky, and muddy. Substrate variations play a role in the diversity of mollusc species in Siwil, so in this research, the number of species was the highest compared to Teleng Ria and Grindulu, namely 13 species (Table 1). In contrast to Teleng Ria, the Grindulu and Siwil areas have less mangrove cover (Figure 2). In addition to providing a living habitat for mollusks, mangrove trees are also a source of food and reproduction, especially for terrestrial gastropods. Not surprisingly, coastal areas with mangrove vegetation have a high diversity of terrestrial gastropods. Teleng Ria is where the most terrestrial gastropod species were found nine species (Table 1). Meanwhile, according to Irma and Sofyatuddin (2012), the abundance of marine bivalves does not affect the existing plant vegetation.

Based on the 25 species of molluscs in this research (Table 1), the environmental conditions at the research location were considered optimal for molluscs to survive and reproduce with temperatures between 28-33°C (Table 2). These results are close to temperature measurements in the mangrove area of the Musi River estuary, South Sumatra, Indonesia, which ranges from 28-31.5°C (Hartoni and Agussalim 2013) and in the mangrove area Nusa Lembongan, Bali, Indonesia, which ranges from 27-30°C (Pratiwi and Ernawati 2016). According to Maretta et al. (2019), the optimum temperature for gastropod metabolism ranges from 25-32°C, while bivalves can metabolize optimally at 25-28°C. The differences in temperature characteristics at the research location are influenced by mangrove vegetation cover and measurement time.

The degree of acidity (pH) is important to support the survival of mollusc organisms. That is because pH can affect the type and availability of nutrients and the toxicity of trace elements. The pH conditions at the research location were recorded in 6-8 (Table 2). According to Odum and Barrett (2005), waters and soil with a pH of 6-9 are waters and soils with high fertility because they can encourage the dismantling of organic matter in the waters, and soils into minerals are ideal for mollusc life. Conversely, according to Artiningrum and Anggraini (2019), environmental conditions with a pH below or above the previously mentioned values can interfere with mollusc life.

The salinity level at the research location ranged from 5-10 ppt (Table 2). It is why the marine gastropod and bivalves species that were in low abundance in this research, such as *Paphia* sp. It is estimated that this newcomer species is carried away by currents from the sea due to the effects of floods that often occur at the research location (Sholiqin et al. 2021). Then, according to Mathius et al. (2018) and Saputra et al. (2020), the optimal salinity to support mollusc life ranged from 28-34 ppt. The evaporation rate influences the difference in salinity in locations and freshwater entry into the sea coast from river flows. These research locations are not too close to the sea, so the fresh water from the river is more than seawater intrusion into the estuary. That causes the salinity value of the water to be not too salty and is categorized as brackish water.

Even so, the results of this research still have a higher diversity of bivalves and gastropods than in other regions in Indonesia. For example, research by Irma and Sofyatuddin (2012) in the mangrove ecosystem of Aceh Besar (14 gastropods and five bivalves), then nine gastropods and two bivalves were found in the Mangrove ecosystem at the Kumbe River Estuary, Merauke by Katukdoan et al. (2018) and Manusawai et al. (2020) with eight gastropods and two bivalves in Kaisu Mangrove Forest of Sarmi Regency, Papua. Information on molluscs diversity in the mangrove ecosystem in Indonesia is not optimal, especially in Pacitan. Therefore, considering mollusc diversity is important to support the conservation effort. Therefore, it needs more research about mollusc diversity in the mangrove ecosystem in Pacitan, East Java, Indonesia.

It can be concluded a total of 25 species were found in this research, with gastropods being more species than bivalves. The gastropods were 17 species and 13 families, while eight species and five families were bivalves. Then, the total molluscs obtained had a density of 36.9 individuals/100 m^2. The mollusc diversity index was 2.14 (medium), mollusc species richness reached 4.1 (medium), the evenness of species in this research reached 0.66 (quite even), and the species dominance index was recorded at 0.34 (low), which means that there are no dominant species in the research location. Then, each research location has a different species and density species of molluscs. That is caused by differences in the type of substrate (such as sandy, muddy, or rocky) and abiotic factors found in the research location (such as temperature, pH, and salinity).

REFERENCES

Aditya, Nugroho GD, Jauhar MF, Sunarto. 2018. Keanekaragaman burung diurnal dan potensi burung sebagai objek daya tarik avitourism di Taman Nasional Gunung Merbabu. Pros Sem Nas Masy Biodiv Indon 5: 362-368. DOI: 10.13057/psnmbi/m050139. [Indonesian]

Agustina R, Sarong MA, Yulianda F, Dewi E. 2019. Histological damage at gonad of *Faunus ater* (Gastropod Mollusc) obtained from heavy metal contaminated river. J Ecol Eng 20 (8): 114-119. DOI: 10.12911/22998993/110787.

Akhrianti I, Bengen DG, Setyobudiandi I. 2014. Spatial distribution and habitat preference of Bivalves in the coastal waters of Simpang Pesak Sub District, East Belitung District. J Ilmu dan Teknologi Kelautan Tropis 6 (1): 171-185. DOI: 10.28930/jitkt.v6i1.8639.

Aldea C, Novoa L, Alcaino S, Rosenfeld S. 2020. Diversity of benthic marine molluscs of the Strait of Magellan, Chile (Polyplacophora, Gastropoda, Bivalves): A historical review of natural history. Zookeys 963: 1-36. DOI: 10.3897/zookeys.963.52234.

Alvareza M, Leilani I. 2020. Community structure of the mangrove forest in the tourism area of Pariaman City, West Sumatra. Bioscience 4 (1): 62-72. DOI: 10.24036/0202041108192-0-00.

Asadi MA, Iranawati F, Andini AW. 2018. Ecology of bivalves in the intertidal area of Gili Ketapang Island, East Java, Indonesia. AACL Bioflux 11 (1): 55-56.

Badan Pusat Statistik Kabupaten Pacitan (BPS). 2021. Kabupaten Pacitan dalam angka 2021. BPS Kabupaten Pacitan, Pacitan. https://pacitankab.bps.go.id/. [Indonesian]

Baderan DWK, Hamidun MS, Utina R, Rahim S, Dali R. 2019. The abundance and diversity of Molluscs in mangrove ecosystem at coastal area of North Sulawesi, Indonesia. Biodiversitas 20: 987-993. DOI: 10.13057/biodiv/d200408.

Bichain J-M, Cucherat X, Brulé H, Durr T, Guhring J, Hommay G, Ryelandt J, Umbrecht K. 2019. Liste de référence fonctionnelle et annotée des Mollusques continentaux (Mollusca: Gastropoda & Bivalves) du Grand-Est (France). Naturae 2019 (11): 285-333. DOI: 10.5852/naturae2019a11.

Cappenberg HA. 2008. Beberapa aspek biologi kerang hijau *Perna viridis* Linnaeus 1758. Oseana 33 (1): 33-40.

Dibyowati L. 2009. Keanekaragaman Moluska (Bivalves dan Gastropoda) di Sepanjang Pantai Carita, Padeglang Banten. [Hon. Thesis]. Institut Pertanian Bogor, Bogor. [Indonesian]

Dissanayake N, Chandrasekara U. 2014. Effects of mangrove zonation and the physicochemical parameters of soil on the distribution of macro benthic fauna in Kadolkele mangrove forest, a tropical mangrove forest in Sri Lanka. Adv Ecol 2014: 564056. DOI: 10.1155/2014/564056.

Dinas Kelautan dan Perikanan Kabupaten Pacitan. 2014. Data Kawasan Mangrove Kabupaten Pacitan. https://adoc.pub/dinas-kelautan-dan-perikanan-kabupaten-pacitan-0.html.

Dolorosa RG, Galon FD. 2014. Species richness of bivalves and gastropods in Iwahig River-Estuary, Palawan, the Philippines. Intl J Fish Aquatic Stud 2 (1): 207-215.

El-Sorogy A, Kammar AE, Ziko A, Aly M, Nour H. 2013. Gastropod shells as pollution indicators, Red Sea Coast, Egypt. J Afr Earth Sci 87: 93-99. DOI: 0.1016/j.jafrearsci.2013.08.004.

Goldyn B, Kaczmarek Ł, Roszkowska M, Guayasamín PR, Książkiewicz-Parulska Z, Cerda H. 2017. Urban ecology of invasive giant African snail *Achatina fulica* (Férussac)(Gastropoda: Achatinidae) on its first recorded locations in the Ecuadorian Amazon. Am Malacol Bull 35 (1): 59-64. DOI: 10.4003/006.035.0108.

Harahab N, Setiawan. 2017. Suitability index of mangrove ecotourism in Malang Regency. ECSOFiM: J Econ Soc Fish Mar 4 (2): 153-165. DOI: 10.21776/ub.ecsofim.2017.004.02.05.

Hartoni H, Agussalim A. 2013. Komposisi dan kelimpahan moluska (Gastropoda dan Bivalves) di ekosistem mangrove muara sungai Musi Kabupaten Banyuasin Provinsi Sumatera Selatan. Maspari J 5 (1): 6-15. [Indonesian]

Hasidu LOAF, Jamili, Kharisma GN, Prasetya A, Maharani, Riska, Rudia LOAP, Ibrahim AF, Mubarak AA, Muhasafaat LO, Anzani L. 2020. Diversity of molluscs (bivalvess and gastropods) in degraded mangrove ecosystems of Kolaka District, Southeast Sulawesi, Indonesia. Biodiversitas 21: 5884-5892. DOI: 10.13057/biodiv/d211253.

Hopper DR, Smith BD. 1992. Status of tree snails (Gastropoda: Partulidae) on Guam, with a Resurvey of Locations Studied by H. E. Crampton in 1920. Pac Sci 46 (1): 77-85.

Hristov KK. 2020. Observations on *Neritina turrita* (Gmelin 1791) breeding behaviour in Laboratory Conditions. Ind J Pure Appl Biosci 8 (5): 1-10. DOI: 10.18782/2582-2845.8319.

Igulu MM, Nagelkerken I, Dorenbosch M, Grol MGG, Harborne AR, Kimirei IA, Mumby PJ, Olds AD, Mgaya YD. 2014. Mangrove habitat use by juvenile reef fish: Meta-analysis reveals that tidal regime matters more than biogeographic region. PLoS One 9: e114715. DOI: 10.1371/journal.pone.0114715.

Irikov A, Gerdzhikov G. 2013. Mollusca (terrestrial and marine Gastropods et Bivalves) from Morocco. ZooNotes 50: 1-5.

Irma D, Sofyatuddin K. 2012. Diversity of Gastropods and Bivalves in mangrove ecosystem rehabilitation areas in Aceh Besar and Banda Aceh Districts, Indonesia. Aquac Aquar Conserv Legis 5 (2): 55-59.

Irwansyah RM, Azzahra SIN, Darmastuti SA, Ramadhandi AR, Firdaus O, Daeni F, Safitri N, Fajri OPA, Nugroho GN, Naim DM, Setyawan AD. 2021. Crab diversity and crab potential as support ecotourism in Teleng Ria, Grindulu and Siwil Beach, Pacitan, East Java, Indonesia.

Intl J Bonorowo Wetlands 11: 75-83. DOI: 10.13057/bonorowo/w110204.

Islamy RA, Hasan V. 2020. Checklist of mangrove snails (Mollusca: Gastropoda) in South Coast of Pamekasan, Madura Island, East Java, Indonesia. Biodiversitas 21 (7): 3127-3134. DOI: 10.13057/biodiv/d210733.

Kabir M, Abolfathi M, Hajimoradloo A, Zahedi S, Kathiresan K, Goli S. 2014. Effect of mangroves on distribution, diversity and abundance of molluscs in mangrove ecosystem: A review. AACL-Bioflux 7 (4): 286-300.

Kantharajan G, Pandey PK, Krishnan P, Samuel VD, Bharti VS, Purvaja R. 2017. Molluscan diversity in the mangrove ecosystem of Mumbai, west coast of India. Reg Stud Mar Sci 14 (2017): 102-111. DOI: 10.1016/j.rsma.2017.06.002.

Karraker NE, Dudgeon D. 2014. Invasive apple snails (*Pomacea canaliculata*) are predators of amphibians in South China. Biol Invasions16: 1785-1789. DOI: 10.1007/s10530-014-0640-2.

Katukdoan M, Monika N, Sunarni S. 2018. Molluscs association (Gastropoda and bivalves) in Mangrove Ecosystem of Kumbe Estuary. Agricola 8 (1): 7-23.

Kementerian Kelautan dan Perikanan Indonesia. 2014. Daftar Moluska yang Berpotensi sebagai Spesies Asing Invasif di Indonesia. Kementerian Kelautan dan Perikanan Indonesia, Jakarta. [Indonesian]

Khade SN, Mane UH. 2012. Diversity of bivalves and gastropod, molluscs of some localities from Raigad district, Maharashtra, west coast of India. Recent Res Sci Technol 4 (10): 43-48.

Krebs JC.1989. Ecological Methodology. Wm.C.Brown Publisher, Dubuque.

Li W, Cui L, Zhang M, Wang Y, Zhang Y, Lei Y, Zhao X. 2015. Effect of mangrove restoration on crab burrow density in Luoyangjiang Estuary China. For Ecosyst 2 (1): 1-9. DOI: 10.1186-s40663-015-0046-3.

Lok AFSL, Ang WF, Ng PX, Ng BYQ, Tan SK. 2011. Status and distribution of *Faunus ater* (Linnaeus, 1758) (Mollusca: Cerithioidea) in Singapore. Nat Singap 4: 115-121.

Magurran AE. 1998. Ecological Diversity and Measurement. Crom Helm Limited, London.

Manusawai J, Sinery AS, Jowey RN. 2020. Diversity of Gastropod and Bivalves Molluscs in Kaisu Mangrove Forest of Sarmi Regency, Papua Province, Indonesia. J. Environ. Treat Tech 3 (3): 1176-1181.

Marshall DJ, Baharuddin N, Rezende E, Helmuth B. 2015. Thermal tolerance and climate warming sensitivity in tropical snails. Ecol Evol 5 (24): 5905-5919. DOI: 10.1002/ece3.1785.

Murphy MJ. 2015. A survey of the marine bivalves and gastropods of Shelly Beach, Port Macquarie, on the north coast of New South Wales, Australia. Vic Nat 132 (3): 73-85.

Neubauer TA, Harzhauser M, Georgopoulou E, Kroh A, Mandic O. 2015. Tectonics, climate, and the rise and demise of continental aquatic species richness hotspots. Proc Natl Acad Sci 112 (37): 11478-11483. DOI: 10.1073/pnas.1503992112.

Nur'aini Y. 2012. Keanekaragaman dan Distribusi Bivalves dan Gastropoda (Moluska) di Pesisir Glayem Juntinyuat, Indramayu, Jawa Barat [Dissertation]. Institut Pertanian Bogor, Bogor. [Indonesian]

Odum EP. 1971. Fundamental of Ecologi, 3th ed. Toppan Company, Ltd., WB Saunders Co., Philadelphia.

Odum EP, Barrett GW. 2005. Fundamentals of Ecology. 5th ed. Thomson Brooks/Cole, Belmont, CA.

Onyena AP, Sam K. 2020. A review of the threat of oil exploitation to mangrove ecosystem: Insights from Niger Delta, Nigeria. Glob Ecol Conserv 22: 1-12. DOI: 10.1016/j.gecco.2020.e00961.

O'Rorke R, Holland BS, Cobian GM, Gaughen K, Amend AS. 2016. Dietary preferences of Hawaiian tree snails to inform culture for conservation. Biol Conserv 198: 177-182. DOI: 10.1016/j.biocon.2016.03.022.

Pawar PR. 2012. Molluscan diversity in mangrove ecosystem of Uran (Raigad), Navi Mumbai, Maharashtra, west coast of India. Bull Environ Pharmacol Life Sci 1 (6): 55-59.

Pielou EC. 1977. Mathematical Ecology. Wiley, New York.

Pritchard C, Shanks A, Rimler R, Oates M, Rumrill S. 2015. The Olympia oyster *Ostrea lurida*: recent advances in natural history, ecology, and restoration. J Shellfish Res 34 (2): 259-271. DOI: 10.2983/035.034.0207.

Rahmila YI, Halim MAR. 2018. Mangrove forest development determined for ecotourism in Mangunharjo Village Semarang. E3S Web Conf 73: 04010. DOI: 10.1051/e3sconf/20187304010.

Rangan JK. 2010. Inventarisasi gastropoda di lantai hutan mangrove Desa Rap-Rap, Kabupaten Minahasa Selatan, Sulawesi Utara. J Perikanan dan Kelautan Tropis 6 (1): 63-66. DOI: 10.35800/jpkt.6.1.2010.163.

Samson E, Kasale D. 2020. Keanekaragaman dan kelimpahan bivalves di Perairan Pantai Waemulang Kabupaten Buru Selatan. J Biologi Tropis 20 (1): 78-86. DOI: 10.29303/jbt.v20i1.1681. [Indonesian]

Saputra R, Zulkifli, Nasution S. 2020. Diversity and mollusca distribution patterns (Gastropoda and Bivalves) In the North of Poncan Gadang Island, Sibolga City North Sumatera Province. J Coast Ocean Sci 1 (1): 16-24.

Satheeshkumar P, Khan AB. 2012. Influence of environmental parameters on the distribution and diversity of molluscan composition in Pondicherry mangroves, Southeast Coast of India. Ocean Sci J 47: 61-71. DOI: 10.1007/s12601-012-0006-6.

Sawant PP, Mohite SA. 2013. Length frequency analysis of the great clam, *Meretrix meretrix* along south west coast of Maharashtra, India. Discovery 4 (10): 19- 21. DOI: 10.13005/bbra/1127.

Setyawan AD, Susilowati A, Wiryanto. 2002. Relics habitat of mangrove vegetation in south coast of Java. Biodiversitas 3 (2): 242-256. DOI: 10.13057/biodiv/d030206.

Shan Lv, Zhang Y, Liu H, Hu L, Liu Q, Wei F, Guo Y, Steinmann P, Hu W, Zhou X, Utzinger J. 2013. Phylogenetic evidence for multiple and secondary introductions of invasive snails: *Pomacea* species in the People's Republic of China. Divers Distrib 19 (2): 147-156. DOI: 10.1111/j.1472-4642.2012.00924.x.

Sharma KK, Bangotra K, Saini M. 2013. Diversity and distribution of Mollusca in relation to the physicochemical profile of Gho-Manhasan stream, Jammu (J & K). Intl J Biodivers Conserv 5 (4): 240-249. DOI: 10.5897/IJBC12.127.

Shing YL, Jurgene HP, Dahdouh-Guebas F, McKee K, Jared OB, Stefano C, Karen D, Francois F, Nico K, Cyril M, Irving M, Nibedita M, Sydne R. 2014. Ecological role and services of tropical mangrove ecosystems: A reassessment. Glob Ecol Biogeogr 23 (7): 726-743. DOI: 10.1111/geb.12155.

Sholiqin M, Pramadaningtyas PS, Solikah I, Febriyanti S, Pambudi MD, Mahartika SB, Umam AF, Liza N, Setyawan AD. 2021. Analysis of the diversity and evenness of mangrove ecosystems in the Pacitan Coast, East Java, Indonesia. Intl J Bonorowo Wetlands 11: 84-94. DOI: 10.13057/bonorowo/w110205.

Sischo DR, Hadfield MG. 2021. The tree snail on Rota Island, Northern Mariana Islands, long identified as *Partula gibba* (Partulidae), is a different species. ZooKeys 1037: 105. DOI: 10.3897/zookeys.1037.56303.

Solanki D, Kanejiya J, Gohil B. 2017. Ecological status of *Pirenella cingulata* (Gmelin, 1791)(Gastropod: Potamididae) in mangrove habitat of Ghogha Coast, Gulf of Khambhat, India. Cibtech J Zool 6 (2): 10-16.

Soon TK, Ransangan J. 2016. Feasibility of green mussel, *Perna viridis* farming in Marudu Bay, Malaysia. Aquac Rep 4: 130-135. DOI: 10.1016/j.aqrep.2016.06.006.

Stagg CL, Mendelssohn IA. 2012. *Littoraria irrorata* growth and survival in a sediment-restored salt marsh. Wetlands 32 (4): 643-652. DOI: 0.1007/s13157-012-0297-5.

Suresh M, Arularasan S, Srikumaran N. 2012. Screening on antimicrobial activity of marine gastropods *Babylonia zeylanica* (Bruguière, 1789) and *Harpaco noidalis* (Lamarck, 1822) from Mudasalodai, Southeast Coast of India. Intl J Pharm Pharm Sci 4: 552-556.

Susanti L, Ardiyansayh F, As'ari H. 2021. Keanekaragaman dan pola distribusi Gastropoda mangrove di Teluk Pangpang Blok Jati Papak Taman Nasional Alas Purwo Banyuwangi. Biosense 4 (1): 33-46. DOI: 10.36526/biosense.v4i01.1415. [Indonesian]

Ulmaula Z, Purnawan S, Sarong MA 2016. Keanekaragaman Gastropoda dan Bivalves Berdasarkan Karakteristik Sedimen Daerah Kawasan Pantai Ujung Pancu, Kecamatan Peukan Beda Kabupaten Aceh Besar. J Kelautan dan Perikanan Unsyiah 1 (1): 124-134. [Indonesian]

Vermeiren P, Abrantes K, Sheaves M. 2015. Generalist and specialist feeding crabs maintain discrete trophic niches within and among estuarine locations. Estuar Coasts 38 (6): 2070-2082. DOI: 10.1007/s12237-015-9959-x.

Vinarski MV, Bolotov IN, Schniebs K, Nekhaev IO, Hundsdoerfer A. 2017. Endemics or strangers? The integrative re-appraisal of taxonomy and phylogeny of the Greenland Lymnaeidae (Mollusca: Gastropoda). Comptes Rendus Biologies 340: 541-557. DOI: 10.1016/j.crvi.2017.09.005.

Walthew G. 2012. The distribution of mangrove-associated gastropod snails in Hong Kong. In: Wong YS, Tam NFY (eds.). Asia-Pacific Symposium on Mangrove Ecosystems: Proceedings of the International Conference Held at the Hong Kong University of Science and Technology. Springer, Netherlands.

Wang C, Chai X, Wang H, Tang B, Liu B. 2013. Growth performance of the clam Meretrix meretrix, breeding selection populations cultured in different conditions. Acta Oceanologica Sinica 32 (10): 82-87. DOI: 10.1007/s13131-013-0369-2.

Wiryawan B, Marsjen B, Susanto HA, Mahi AK, Ahmad M, Poepitasari H. 1999. Atlas Sumberdaya Wilayah Pesisir Lampung. Pemda Tk I Lampung- CRMP Lampung, Bandar Lampung. [Indonesian]

Yadav R, Malla PK, Dash D, Bhoi G, Patro S, Mohapatra A. 2019. Diversity of gastropods and bivalves in the mangrove ecosystem of Paradeep, east coast of India: a comparative research with other Indian mangrove ecosystems. Molluscan Res 39 (4): 325-332. DOI: 10.1080/13235818.2019.1644701.

INTL J BONOROWO WETLANDS
Volume 12, Number 1, June 2022
Pages: 12-22

P-ISSN: 2775-8052
E-ISSN: 2775-8044
DOI: 10.13057/bonorowo/w120102

Review:
Causes and impacts of anthropogenic activities on mangrove deforestation and degradation in Indonesia

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Manuscript received: 14 August 2021. Revision accepted: 15 February 2022.

Abstract. *Cahyaningsih AP, Deanova AK, Pristiawati CM, Ulumuddin YI, Kusumaningrum L, Setyawan AD. 2022. Review: Causes and impacts of anthropogenic activities on mangrove deforestation and degradation in Indonesia. Intl J Bonorowo Wetlands 12: 12-22.* Indonesia is a country with the largest extent of mangroves in the world. However, as the human population increases the extent and quality of the mangroves is decreasing. The increasing population is in line with increase in human activities, leading to deforestation and degradation of the mangrove forests. This paper aimed to review anthropogenic activities that cause mangrove deforestation and degradation in Indonesia as well as the impacts of such anthropogenic activities on the mangrove ecosystems, and to formulate the efforts to overcome mangrove deforestation and degradation in the country. Various human activities are the primary drivers of deforestation and degradation of mangroves, including land conversion from mangrove forests into other land uses (e.g., agricultural lands, ponds, infrastructure developments and human settlements), extractive activities (e.g. mining and logging), and household and industrial activities (e.g., plastic debris, heavy metals pollutants). These anthropogenic factors have impacts on reduced mangrove forest area, changes in mangrove composition, uneven species abundance and pollution. In combination with sea level rise caused by global climate change, such activities pose devastating threats to Indonesian mangroves. Efforts to overcome these problems include increasing awareness of communities regarding the conservation of mangrove, conducting mangrove restoration programs and promoting sustainable management of mangrove, for example through ecotourism. On top of that, community participation plays essential role in sustainable management and conservation of mangrove forests in Indonesia.

Keywords: Anthropogenic activity, mangrove degradation, mangrove forest

INTRODUCTION

Mangrove ecosystem is transitional ecosystem between terrestrial and marine ecosystems located along coastlines and river estuaries and is strongly influenced by intertidal conditions. In general, mangroves grow in places with a mud and sand substrate on relatively flat or gently sloping areas, and not in places like beaches with steep and undulating terrain, and strong tides. The flat areas provide space for mangroves to grow so that the mangrove vegetation can expand (Akbar et al. 2015). The muddy and sandy mangrove habitat is also very suitable for the growth and development of several organisms that depend on mangroves (Afriza 2019).

Mangrove ecosystem plays an important role in delivering various ecosystem services. Mangrove forests store a very large carbon stock in aboveground and belowground biomass as well as in the soils in which in total it is five times greater than tropical boreal forests per unit area, implying that mangrove forests have a major influence on global climate change mitigation (Hamden et al. 2014). Mangrove ecosystems also protect the coast from waves, winds, storms and even tsunamis. For marine biotas, mangrove ecosystems serve as habitat that supports their life and are suppliers of organic materials to provide food. Meanwhile, for local communities, mangroves provide materials for household needs such as firewood, charcoal, building materials, foodstuffs, and medicines (Nugroho et al. 2013).

Indonesia is a country with the largest extent of mangrove ecosystem in the world. A large extent of the coastal areas in Indonesia are covered by mangrove forests with various widths, ranging from a few meters to several kilometers from the coastline (Kusmana 2011). However, deforestation and degradation of mangrove forests in Indonesia have been going at an alarming rate and increasing every year. As the consequence, there is continuing decline in the function of mangrove forests with impacts of carbon emissions, biodiversity loss, increase of risk on abrasion and land subsidence, and reduction of community livelihoods (Wahyuni et al. 2014).

Mangrove forest ecosystem is a fragile ecosystem that is very sensitive to environmental changes. However, the easy access characteristic of mangrove ecosystem is prone to exploitation of its resources, leading to deforestation and degradation (Samsumarlin et al. 2015). According to Akbar et al (2017), anthropogenic activities are the major causes of deforestation and degradation of mangrove forests in Indonesia. Anthropogenic activities are human activities that can change landscape structures such as land-use conversion, habitat destruction, invasion of alien species, and overexploitation of biological resources. These activities result in changes in trophic status in mangrove ecosystem (Zulfia and Aisyah 2013). Examples of disturbances caused by human activities are tree cutting of mangroves, conversion of mangrove forests into plantations, settlements, industries, irrigated rice fields, rainfed rice fields, and fish or shrimp ponds (Mappanganro et al. 2018). Such human activities cause the degradation of mangrove ecosystem, damaging ecosystem stability and reducing biodiversity, especially mangrove organisms (Athirah et al. 2013). The decline in mangrove extent and quality is a serious threat to this ecosystem whose people depend on coastal resources. According to Schaduw (2015), if the mangrove forest ecosystem is damaged, its social and economic functions will also be disturbed.

The purpose of this paper is to review anthropogenic activities that damage mangrove ecosystems in Indonesia, to find out the impacts of such anthropogenic activities on the mangrove ecosystems, and to formulate efforts to overcome mangrove deforestation and degradation in the country. We expected that this review provides a strong scientific basis for better management and conservation of mangrove ecosystems in Indonesia.

MANGROVE FOREST ECOSYSTEM

Mangroves are defined as a community of woody plants that grow in intertidal areas between terrestrial and marine ecosystems and are resistant to high salinity levels (Biswas et al. 2012; Biswas and Biswas 2019). The plant community of mangroves forms vegetation called a mangrove forest. Rahim and Baderan (2017) define mangrove forests as a variety of tropical coastal vegetation communities that are dominated by typical species of trees or shrubs that have the ability to grow in salty waters. Mangroves ecosystem has high values on ecological aspects, such as maintaining aquatic productivity, and economic aspects for the people living around mangrove ecosystems (Indrayanti et al. 2015; Widiastuti et al. 2016). Mangrove forest is one of ecosystems in the tropics and subtropics that has the largest carbon content among various ecosystems on the earth, as well as the most important CO_2 sequester in the atmosphere (Wang et al. 2021).

Mangrove ecosystem is among the complex ecosystems that have mutual relationship with the surrounding ecosystems, such as onshore and offshore ecosystems. It is known as one of the most fertile ecosystems because the litterfall of the mangroves acts as source of nutrients to the soil and water (Diarto et al. 2012). Mangrove ecosystem is also considered one of the most valuable and productive coastal ecosystems on the planet (Himes-Cornell et al. 2018). Mangrove vegetation is regulated by several biotic and abiotic factors, such as anthropogenic impacts, nutrients, light, salinity, sedimentation, wind, and tides. Mangroves allocate most of their physiological energy to respond to high saline conditions. The availability of soil nutrients in mangroves is regulated by soil pH, and nutrients such as N, P and K (Cooray et al. 2021).

Mangrove ecosystem is affected by the tides of seawater which is suitable for the habitat of avifauna, including migratory birds. The ebb and flow of seawater in this ecosystem causes large fluctuations in environmental factors, especially temperature and salinity. As such, animals that are able to survive in this ecosystem must have great tolerance to extreme changes in environmental factors. There are around 150 to 250 species of bird species to occur in mangrove ecosystems with 65 species are categorized as endangered and vulnerable to extinction.

In Indonesia, mangrove forests are found in almost all parts of the country, spreading in 30 provinces from Sumatra to Papua. However, the existence of mangrove forests in Indonesia is threatened due to various human activities (e.g., timber cutting, overfishing and land conversion), and natural factors, such as tsunami (Purnobasuki 2011). Of the total 8.6 million hectares of mangrove ecosystems in Indonesia, some 5.9 million hectares (or 68%) have experienced disturbances (Wijaya et al. 2018).

ROLES OF MANGROVE FOREST

Mangrove ecosystems deliver various ecosystem services in the form of supporting services, regulatory services, provision services, and cultural services (Eddy et al. 2014). In terms of supporting and regulatory services, mangrove ecosystems maintain and protect tropical and subtropical marine biodiversity, global biogeochemical processes and climate change (Wang and Gu 2021). Mangrove forest ecosystem has the highest productivity level compared to other coastal ecosystems, and it also serves as habitat for aquatic biota that provides organic matters which is important in the life cycle of various types of shrimp, fish and mollusks. The detailed roles of mangrove forests are elaborated on below.

Mangroves as carbon stocks

Mangroves have the ability to absorb and store a large amount of carbon which is critical for controlling CO_2 levels in the atmosphere (Purnamasari et al. 2021), implying its critical role in reducing carbon emissions for climate change mitigation. The carbon absorbed by mangroves is stored it in stems, roots, leaves and soils. In mangrove ecosystem, soils contribute the majority of carbon stock, accounting for more than 84 percent of total carbon contained in the ecosystem (Trettin et al. 2021). For carbon stored as biomass, the capacity of carbon sequestration of mangrove forests is influenced by its dimensional size where the larger the tree diameter, the greater the storage of biomass and the amount of CO_2 absorbed by mangrove trees (Heriyanto and Subiandono 2012). The height and age of mangrove stand also affect the absorption rate.

Biological function

Mangrove forest consists of unique biotic and abiotic components. The biological elements of mangrove forests are the main component of food for organisms that live around mangroves, such as shrimp, fish, crabs, birds and mammals. Mangroves serve as habitats for living organisms which are used as gathering and hiding places, especially for young fish, shrimp, and other marine biotas. Mangrove forests also provide an excellent site for spawning marine biotas (Kustanti 2011). The canopy of mangrove vegetation is useful as a habitat for various types of arboreal faunas, such as insects, birds, bats, and primates.

The existence of mangrove forests which act as coastal green belt influence the surrounding ecosystems (Pontoh 2011). It is proven that mangrove forest ecosystem can be useful as a protector of seawater intrusion, stabilizer of the coastline, the main point of circulation of nitrogen and sulfur, mud collection and land formation (Samosir and Restu 2017). Mangrove forests are also useful as an organic waste processor by neutralizing toxic substances caused by chemical wastes.

Mangrove root structures can improve the chemical and physical properties of soils through litter decomposition in which the organic matter of the decomposed litter fall can reduce salinity levels. Starting the food chain, mangrove leaves that fall into the water will be decomposed by microorganisms so that the results of this decomposition serve as food for small aquatic animals. The roots of mangroves become a source of nutrition for fish and invertebrates that live in the mangrove ecosystem.

Physical function

According to Utomo et al. (2017) and Ahnanto et al. (2014) mangrove ecosystems have the ability to control seawater intrusion by preventing $CaCO_3$ deposition by its root, holding mud and trapping sediment. This physical protection is beneficial for the surrounding communities from the damage that can be caused by big waves, strong winds, coastal abrasion, tsunamis, mud flood, and sediment traps transported by surface water flows and prevention of intrusion.

Social and economic function

Mangrove ecosystem provides social and economic benefits for coastal communities (Sondakh et al. 2019; Ritohardoyo and Ardi 2011; Gumilar 2012). The economic value of resources can be defined on how it benefits humans in the monetary term (Saprudin and Halidah 2012). The economic value that can be utilized from mangrove forests consists of direct benefits (e.g. wood and fishery products) and indirect benefits (Jumaedi 2016). The direct benefits of mangrove forests are in the form of various forest products to fulfill community needs to improve the economy and social state of communities living around the mangrove forest. According to Riwayati (2014), the indirect benefits of mangroves for human life include nutrient providers, resistance of seawater intrusion and abrasion, and the ability to form islands and stabilize beaches.

The parts of the mangrove tree, including the roots, bark, leaves, tree trunks, and flowers can be beneficial for humans. Mangrove forests can produce wood and non-timber which can be used by the community as fuelwood, foodstuffs, handicrafts, medicines, animal feed, industrial raw materials. The bark of mangrove trees can be used as preservatives and natural medicines as well as building materials. In mangrove ecosystem, it can be found various types of fish, shrimp and crab that have high economic value. The mangrove roots or trunks can be used as protection for boats and ships by tying ropes to these parts.

Some mangrove plants that can be processed into food include Lindur (*Burguiera gymnorhiza*), Pedada (*Sonneratia* spp), Api-api (*Avicennia alba*), Nipah (*Nypa fructicans*) and Jeruju (*Achantus ilicifolius*). Mangroves also contain high calories and carbohydrates, more than some various types of food such as rice, corn, cassava. Some mangroves can also be used as natural coloring agents. Parts of mangrove plants that can be used as dyes are waste or unused plant parts, for example, mangroves in the species *Rhizophora* which can produce black, brown, dark brown, light brown, and pink colors. Another economic function of mangrove forests is as a place for fish and shrimp pond cultivation, a place for making salt fields, and source of materials for high-quality charcoal.

Indirect benefit values of mangrove forests, for example, are the provision of natural food for marine biota, barriers to seawater intrusion into land, and land expansion towards the sea (Ariftia et al. 2014). Mangroves can also be developed for ecotourism and education purposes (Setyawan and Winarno 2006a). The ecotourism activities are aimed to conserve the environment and preserve or improve the welfare of the local community. The ecotourism and education function in mangrove forests can be developed, for example by sightseeing the beauty of flora and fauna, as well as boating around the mangroves.

ANTHROPOGENIC ACTIVITIES THAT DESTROY MANGROVE ECOSYSTEMS

Mangrove ecosystem, which is located between the terrestrial and marine realms, is very sensitive to changes caused by natural processes and anthropogenic activities, such as urbanization and economic development (Wang and Gu 2021). Human activities with various forms of utilization pose threats to cause ongoing changes in mangrove forest resources (Kustanti et al. 2014; Oktavia et al. 2020). Anthropogenic activities that cause the loss of mangrove forests in Indonesia include fisheries, agriculture, plantations, settlements, industry, mining, and aquaculture (Kumar 2012; Sukwika and Putra 2018). On the other hand, human factors that lead to mangrove degradation include illegal logging, unsustainable use, and establishment of shrimp ponds (Basyuni et al. 2012).

The larger the population community living around the mangrove forest area, the greater the rate of mangrove deforestation and degradation (Konom et al. 2019). The existence of coastal communities has interaction with mangrove ecosystem, for example, the utilization of wood extracted from mangrove for building materials (Hendrawan et al. 2018). Since most mangrove areas are under the authority of government (which are often considered open access by some people) and they are located in a sandy coastal area around the estuary which is easily accessible by humans, mangrove forests are prone to be exploited and disturbed (Heriyanto et al. 2020). Various anthropogenic activities that threaten mangroves in Indonesia are detailed below.

Land conversion

Mangrove forests have been converted into various types of land use. From 2002 to 2012, mangrove forests were lost at an average rate of 0.18% per year with a total estimate of more than 100,000 ha of mangroves have been logged over 10 years. In Indonesia, the drivers of mangrove forest conversion include aquaculture development, rapid expansion of rice farming, and oil palm plantations. The conversion of mangrove forests into aquaculture mainly occurs in Kalimantan and Sulawesi (Richards and Friess 2016). This is dilemmatic since aquaculture expansion is largely supported by the government to increase fisheries production in Indonesia (Rimmer et al. 2013). For example, mangrove forest conversion which results in degradation and deforestation due to unsustainable use of resources has occurred in the mangrove forests of Takalar District, South Sulawesi, Indonesia. The decline in mangrove area has reached 66.05% over the 33 year analysis period with the drivers is mainly due to land conversion for aquaculture. Yet, the clearing of mangroves for shrimp and seaweed cultivation in the area provides new job opportunities and generates profits for the community (Malik et al. 2017).

In Java, most of the mangrove habitat has been converted into rice fields, settlements and some river mouths have been converted into ponds, as has happened on the southern coast of the island (Setyawan et al. 2002). For instance, the conversion of land from mangrove forests to ponds occurs in Rembang, Central Java, Indonesia. In this area, many mangrove habitats have been opened to be used as shrimp, milkfish and salt ponds so that ponds dominate along the coast. Regrettably, intensive and massive aquaculture activities have resulted changes in hydrological conditions, edaphytes, and environmental pollution, leading the ponds to become unproductive, abandoned and degraded. Until now, there is no natural mangrove ecosystem left in the area (Setyawan and Winarno 2006b). Another land conversion in Central Java is in the form of rice fields, fish ponds, and salt extraction. In this area, the impacts of anthropogenic activities can be noted from the remained vegetation which shows canopy gaps and vacant land in the area (Setyawan et al. 2008).

In Kalimantan, mangrove conversion is caused by oil palm plantations which are responsible for 70% of coastal areas degradation on the island. Oil palm plantation becomes more common and causes significant environmental concerns in coastal regions. The presence of oil palm industry, which contributes significantly to the destruction of the mangrove ecosystem, produces waste that not only contains heavy metals such as Pb, Cd, Fe, Cu, Cr, Zn, Ni, but can also lower the pH of waters and sediments and affect mangrove vegetation (Kadarsah et al. 2020). This pollution as the excess of oil palm mills occurs in many rivers and coastal estuaries in Kalimantan, affecting the function of the mangrove ecosystem.

Coastal reclamation for infrastructure development and human settlement are other anthropogenic activities that convert mangrove ecosystems. Coastal area is very dynamic and intensively influenced by anthropogenic activities. The initial goal of coastal reclamation is to control flooding in a city. In fact, coastal reclamation can result in changes in the ecosystem around the reclaimed area including the mangrove ecosystem (Puspitasari et al. 2017). Clearing mangrove forests into settlements can occur because the area is considered to be open access, being assumed that they have no owner (Ulumuddin and Setyawan 2017). The high population pressure has implications for the magnitude of land conversion activities, sedimentation, reclamation, and environmental pollution (Setyawan et al. 2003).

Mining and logging activities

Legal and illegal coal mining is one of anthropogenic activities that degraded coastal regions in Indonesia, for example in South Kalimantan. The presence of coal mining operations which are becoming more widespread, also causes disturbances in mangrove ecosystems. In addition to open-pit operation, the presence of special port for coal shipping also contributes significantly to the destruction of the mangrove ecosystem (Kadarsah et al. 2020). On the other hand, Balikpapan City, East Kalimantan is the area with the largest oil mining in Indonesia with mining activities occurring on the coast and offshore. Oil mining activities in the area often result in oil leaks from tankers and oil pipelines causing damage to several mangrove forests in Balikpapan Bay (Anwar et al. 2021).

Logging of mangrove trees is another anthropogenic activity that destroys mangrove ecosystem. Moreover, illegal logging of mangroves creates devastating impacts on mangrove forest ecosystem. Several types of mangrove vegetation are used by community as firewood, charcoal and wood for building materials (Malik et al. 2017). Mangrove logging in Dumai, Riau, Indonesia mostly uses mangrove species including *Rhizophora*, *Xylocarpus*, and *Bruguiera* (Mulyadi and Amin 2016). Several mangrove areas in Central Java also suffer from logging activities (Setyawan et al. 2008). Fairly massive logging activity in the mangrove habitat in Central Java is indicated by the vegetation which is undergoing secondary succession and a large number of vacant lands. The composition and structure of the vegetation where the importance index of young plants is relatively high suggest that the stands do not reach a climax (Setyawan et al. 2005a). The vertical and horizontal vegetation diagram shows that the ecosystem is dominated by young mangrove plants and there are many canopy gaps.

Household and industrial activities

Human activities, such as household and industrial activities, also threaten mangrove ecosystem, for example in Dumai, Riau, Indonesia. Such activities in the coastal area contribute to contamination and pollution of the ecosystem around the mangrove area. The pollution is caused by waste from residences, hotels, and restaurants. Solid, liquid, and gas wastes, as well as organic and inorganic wastes, are all produced, often contained with heavy metals. Potential sources of pollution in coastal areas due to domestic activities are also influenced by the population especially for people who do not have the technology to manage the wastes. The presence of a large population will increase economic activities while producing an increasing amount of trash (Mulyadi and Amin 2016). The garbage dumped by people will be brought by rains, entering the river and ending up in mangrove forest. This situation persisted for years with no consideration to develop waste disposal, causing more waste to be accumulated (Nombere 2019).

Another type of waste that is increasing of concern is plastic debris. Van Bijsterveldtwe et al. (2021) investigated the magnitude of the plastic trash problem in mangroves along Java's north coast of Indonesia. Plastic was prevalent in the field, with 27 plastic pieces per m^2 on average, covering up to 50 percent of the mangrove floor in various sites. Furthermore, the study indicated that plastic was commonly buried in the top layers of sediment, where it becomes immobile and can result in protracted anoxic conditions.

Surveys of anthropogenic waste in mangrove forests have also been conducted, and it is known that big plastic objects are discovered more frequently in mangroves than on beaches. Mangrove serves as a landfill for marine trash and a garbage trap for land-based rubbish. Plastic trash is captured by mangrove forests and may be found in large concentrations on the forest floor and in sediments. The mangrove air roots act as a filter that holds out large plastic debris (Martin et al. 2019). In general, mangroves sustain high sediment accretion rates and efficiently absorb plastic in their sediments. Microplastics with a size of 0.5 mm dominated the mangrove sediments in *Avicennia marina* mangrove forests, and it is assumed that tons of plastics have been buried in the mangrove sediments since the 1930s (Martin et al. 2020). It also showed that mangrove sediments function as long-term plastic reservoirs and that mangroves retain marine plastic debris for a long period of time (do Sul et al. 2013).

IMPACTS OF ANTHROPOGENIC ACTIVITIES ON MANGROVE ECOSYSTEMS

The damage and loss of half of the mangrove area in Indonesia is largely influenced by human anthropogenic activities. The underlying drivers of mangrove deforestation and degradation are increase in population, increase in human needs, types of work, and people's perceptions of mangrove forests (Majid et al. 2016; Hartati and Harudu 2011). By knowing the impact of anthropogenic activities, humans are expected to be able to make efforts to manage the mangrove ecosystem. Several issues regarding the impacts of anthropogenic factors are elaborated on below.

Impact of pollution

Anthropogenic activities cause the increasing levels of pollutants in the atmosphere, particularly atmospheric CO_2 and tropospheric O_3 which are important for forest ecosystems functioning. Increased CO_2 and O_3 levels affect tree metabolism, growth, and chemical composition. Changes in chemical composition may then cascade down through the trophic levels, influencing ecosystem processes (Couture and Lindroth 2013).

Pollution that often occurs in the mangrove ecosystem includes oil pollution, heavy metals, waste from tourists, aquaculture (ponds) that produce liquid waste, agriculture, roads, industry, residential areas and mining. The impacts of pollution on mangrove ecosystem include reduced and damaged mangrove ecosystems, loss of various species of flora and fauna due to unable to adapt to pollution and disconnection of the life cycle of fish and shrimp, which in the long term will disturb and affect the balance of the mangrove ecosystem and coastal ecosystem.

Pollution that often occurs in the waters is pollution in the form of oil spills, such as tanker accidents, oil loading and unloading, port activities, etc. which can result in physical disturbances in mangrove vegetation such as yellowing leaves, falling leaves, mangrove death due to coating of mangrove roots by oil spills, and permanent damage to mangroves which causes the destruction of the nursery ground for marine biota (Prasetyo et al. 2017).

Impact of land conversion

The increasing human population causes an increase in the need for job opportunities. For people living around the coast, the available employments are in the sectors of agriculture, fisheries, fishponds, ports, salt fields, and industries which of course require a large extent of land, implying that mangrove deforestation and land conversion is unavoidable. However, such land conversions do not regard capacity and development designation, causing many negative things, such as threatening the regeneration of fish and shrimp in offshore waters, marine pollution, shallow beaches due to sedimentation and water intrusion (Rusdianti and Sunito 2012).

Anthropogenic activities have different impacts on the degradation of mangrove forests. So far, land conversion activities, such as for agricultural lands and shrimp ponds development, are the main factors of mangrove degradation. These anthropogenic activities result in the reduction of the area of mangrove forests. There is no agreement on the extent of mangrove forests in Indonesia, but land conversion such as the development of coastal agricultural land and shrimp ponds in tidal areas is considered as the main factor behind mangrove deforestation (Rasyid et al. 2016) which accounts for 90% of the reported loss (DasGupta et al. 2013).

Oil mining activities have a very detrimental impact on mangrove forests, such as the case in Balikpapan, East Kalimantan. Oil spills can damage mangrove forests due to oil deposits brought by the ebb and flow of seawater, causing the death of mangroves because of the sensitive nature of their roots exposed to oil. On the other hand, the effects of oil spills take a long time to overcome. Apart from causing the death of mangroves, it can also worsen soil and water conditions, making it difficult to regrow mangroves naturally (Anwar et al. 2021).

In coal mining areas and oil palm plantations, many environmental changes in mangrove forests have occurred as a result of these activities as occurred in South Kalimantan. These changes include pH of the waters becoming more acidic (5.76-6), higher organic matter content in sediments ranging from 0.61-6.59%, high heavy metal Pb content in sediments and waters. This has an impact on the reduction of population of mangrove species due to death although some true mangrove species such as *Avicennia alba*, *Acanthus ebracteatus, Nypa fruticans* and *Rhizophora apiculate* can still survive (Kadarsah et al. 2020).

Timber cutting of mangrove forests can cause changes in mangrove composition which are likely to be replaced by species with low commercial value. Such vegetational changes cause loss on the ecological and economic functions of mangrove forests as a place to find food and a place reproduction of various kinds of marine biotas which have high economic value. In addition, changes in the composition of mangroves cause mangroves to no longer function as absorbers of the waves of seawater properly and mangroves cannot withstand erosion and abrasion of seawater (Hadayatullah and Pujiono 2014). Often, timber cutting of mangroves is done illegally as occurred in East Java's mangrove forests (Rudianto et al. 2020). Mangrove forests are naturally functioned to reduce carbon dioxide concentrations, and because of the illegal logging the average carbon sequestration and storage in East Java's mangrove forests are reduced (Rudianto et al. 2020).

Impact of domestic and industrial activities

Various domestic and industrial activities that produce multiple pollutants, both solid, liquid and gas, affect the growth and life of young and mature mangrove trees. For example, mangrove areas in Dumai City, Riau are impacted by pollutants resulting from industrial activities, such as solid waste, organic and inorganic liquid waste processing, and generator residues. Besides industrial activities, mangrove vegetation is affected by wastewater and solid waste generated by domestic activities (Mulyadi and Amin 2016). According to Budiastuti et al (2016), a decrease in dissolved oxygen content in the waters of the mangrove ecosystem shows a strong indication of pollution. This pollution usually occurs due to the disposal of liquid waste from industries around the mangrove forest ecosystem. The decrease in oxygen levels occurs in an anoxic state so that organic matter undergoes anaerobic decomposition and produces sulfides and ammonia which will be toxic to aquatic biota in the mangrove ecosystem.

Plastic wastes trapped in mangrove forests also have a negative impact. The accumulation of plastic wastes in mangrove areas creates a layer above the mangrove air roots, leading to depressed root growth and the mangrove eventually dies (Martin et al. 2019). According to Kinanti et al (2014), river estuaries in mangrove areas become waste centers so that these estuaries often become high input for organic pollutants. Chemical pollution, particularly metal accumulation and biotransformation, can be a significant factor in reducing mangrove biodiversity (Maiti and Chowdhury 2013) and an indication of altered biogeochemical cycles with a significant decrease in organic carbon in sediments (Carugati et al. 2018).

Damage to mangroves can cause mass mangrove mortality. This is due to a drastic increase in salinity and sedimentation rate, and a decrease in the water level. The underlying driver is human behaviors which cause changes in the nature of rivers due to, for example, disposal of garbage, fishing with bombs and narrowing of the river replaced by human settlements. In fact, mangroves will grow in abundance on beaches close to rivers or river deltas, because the river flows carry mud and sand which are the main media for mangrove growth (Wardhani 2011). In addition, the practice of fishing or shrimp catching that uses spread nets and moving from one place to another will damage and disturb the habitat of fauna and benthic species whose lives are relatively sedentary at the bottom of the water. All such causes will risk the death of mangroves with negative consequences such as flooding, coastal erosion, loss of catches, decreased fishery products, and loss of community income due to lost fishery resources.

Impact of sea level rise

Climate change occurs due to human activity because of greenhouse gas (GHG) emissions mainly CO_2, disrupting the ozone layer in the atmosphere to cause global warming. According to the IPCC through the Special Report on Emission Scenarios (SRES), global climate change can affect sea-level rise and cause flooding, damage to settlements, damage to fresh land supplies from coastal aquifers, and inundation of a coastal area even inland beyond the coastal boundary. Sea level rise is characterized by tidal conditions, tsunamis, storm surges, waves, and global warming (Anggraini et al. 2012).

The rises in sea level will cause a decrease in the level of sediment, limiting landward expansion of mangrove vegetation. This of course will disrupt the distribution of mangroves and threaten the existence of mangroves in the long run. Climate change also triggers the decrease in rainfall, leading to increase in salinity and salt sulfate levels in seawater and resulting in the decline of mangrove growth, reduced biophysical ability of mangrove forests due to several types of plants being unable to adapt to the drastic climate change, and increasing air temperature with consequence on the loss of marine faunas that are unable to adapt, and causes reducing water sources in forest areas (Nandini and Narendra, 2011). In addition, climate change will cause a very long drought, contributing to death in mangrove vegetation and inhibiting mangrove growth (Subardjo and Pribadi 2012).

EFFORTS FOR SUSTAINABLE MANGROVE MANAGEMENT AND CONSERVATION

Mangrove forests have many roles in maintaining the balance of the ecosystem. Deforestation and degradation of mangroves caused by human activities can be resolved by developing strategies of mangrove management that are win-win for both humans and mangroves (Sofuan 2016). The main obstacles to mangrove management and conservation are sectoral perspective when viewing mangrove forests, weak community participation, poverty, and lack of concern for the ecological value of mangroves. These management problems are also compounded by the lack of knowledge about silvicultural techniques, potential uses, and regeneration techniques (Setyawan et al. 2003). Introducing coastal communities to environmental conservation also plays essential role in efforts to overcome anthropogenic disturbances to mangrove forests (Ali et al. 2017). Sustainable mangrove management is an alternative strategy that can be done for mangrove forests. There are various options for sustainable mangrove management, yet the main emphasis should be based on a co-management approach (Tetelepta et al. 2020) as detailed below.

Community participation

Community participation in many cases of natural resource management is fundamental for its sustainability. The involvement of community is essential in the conservation and rehabilitation of mangrove forest, including program preparation, implementation and monitoring so that people feel responsible for its long-term sustainability (Martuti et al. 2018). In the first instance, community awareness is required to not disturb mangroves (Syahroni 2016). Efforts to increase public understanding include in the form of environmental care activities. Approval and commitment from local communities are required for the management of mangrove including efforts to restore it if it is in degraded condition. Such efforts will be more successful if the community is educated to carry out restoration with a desire to do it themselves which can also provide personal benefits (Romañach et al. 2018).

The active participation of the community can help to maintain the diversity of mangroves by maintaining the mangrove habitat properly. As in the case of mangrove areas on the coast of Central Java, the community's concern for the mangroves is very high. In these areas, the government forms groups and allows the mangroves areas to be used by the communities while maintaining sustainability. This community participation is quite successful in preserving the ecosystem and mangrove diversity (Setyawan et al. 2005b).

Mangrove conservation

Mangrove conservation requires joint effort or responsibility between the community and the government (Winata and Yuliana 2016). Mangrove forest conservation efforts can be carried out with three main things, namely the protection of life support systems, preservation of flora and fauna diversity and ecosystems, and sustainable use of biological resources and ecosystems (Yuliani et al. 2018). In mangrove conservation, integrated strategies must be conducted in line with regional development plans (Fitriah et al. 2013). The environmental management process should be carried out by looking more at the local situation and conditions so that the management approach can be adjusted to the local conditions of the area to be managed (Muharuddin 2019).

Efforts to overcome problems related to mangrove conservation require strategic and continuous steps through education and guidance that can understand the community about the importance of preserving the ecosystem. Local wisdom in the form of local knowledge can support management and efforts to overcome mangrove damage due to anthropogenic activities (Sondakh et al. 2019). Three indicators that can be used to support mangrove conservation are explained here. First, attitude; this includes social values, a strong sense of concern for natural resources, preservation and sustainable use of nature, motivation to participate in efforts protection, and participation in protection efforts. Ecological awareness in coastal communities needs to be increased by providing motivation to the community to generate a sense of enthusiasm and be able to change human or individual behavior for the better (Sulastri and Haryadi 2019). Extension also needs to be carried out for efforts in the form of practical actions taken to encourage behavior change in individuals, groups, communities, and communities to know, want to implement, and be able to solve the problems faced (Ambo-Rappe et al. 2020). Second, skill; is needed to solve problems that occur in the conservation of natural resources. Third, participation; is necessary to develop a sense of responsibility for a natural resource conservation problem so that it can take actions that are relevant or in accordance with the actual situation.

Efforts to prevent mangrove degradation can be carried out through several stages of activities, including understanding the ecology of mangrove species and communities, assessing factors that would hinder succession, understanding hydrological suitability and location of planting, knowing the level of community dependence, and understanding the traditions, perceptions and needs of the local community. Analysis of ecosystem services that exist in mangrove ecosystems needs to be done to see the balance of ecosystem services between human use and exploitable mangrove resources (Handayani et al. 2020).

Mangrove restoration

Mangrove restoration is currently promoted in Indonesia as a large extent of mangrove forest is deforested and degraded. This effort is carried out to reforest damaged mangrove areas and minimize seawater intrusion into land and coastal abrasion. When doing mangrove restoration, it is necessary to take a broader approach by integrating coastal area management, by incorporating essential elements such as ecology, socio-economic, and sociocultural so that it can meet the community's needs while maintaining biodiversity at large (Setyawan et al. 2004).

For example, planting mangrove trees has been carried out in Baros Village, Bantul, Yogyakarta. The restored mangroves can grow well and have many ecological, economic, social and tourism benefits for the surrounding community. This program is useful for preventing abrasion to protect agricultural areas around the mangroves. In mangrove restoration, the role of the village government, tour guides and community participation are needed so that mangrove restoration can be successful and provide benefits for environmental and socio-economic sustainability for the community (Djumanto 2020).

Afforestation and reforestation of mangrove areas can involve community in various activities, such as seedling preparation in the nursery, planting, maintenance, and utilization of conservation-based mangrove forests. Mangrove planting can be done in various coastal locations, especially in areas of ex-ponds, former mangrove habitats that have been damaged either as a result of human activity or as a result of the tsunami waves (Suriani and Bahagia 2012). For instance, restoration efforts have been carried out in several locations in the south and north coasts of Central Java, Indonesia by planting mangrove seedlings. Several sites show significant results in improving and enhancing the functions of the mangrove. In this case, the success of mangrove restoration is due to the support and active participation of the along with strong consideration on biotic, abiotic and cultural factors of the local community (Setyawan and Winarno 2006a).

Sustainable mangrove ecotourism

Sustainable ecotourism is gaining popularity nowadays including in mangrove areas. This strategy provides positive benefits for both human and mangrove ecosystem. For example, mangrove ecotourism in East Java shows that mangroves can be used as nature-based tourism and contribute to mangrove conservation. The ecotourism strategy must involve the participation of local communities so that local wisdom can improve the sustainability of the mangrove ecosystem (Hakim et al. 2017). Community awareness programs related to the importance of mangrove forests are very important to be carried out, especially for the communities who live around the mangrove forest (Pattipeilophy 2014). Counseling and training for communities around the mangroves are also necessary in order to increase community understanding on ecotourism-based mangrove forest management and to avoid damage to mangrove forest ecosystems (Nugraha et al. 2015).

In the management of mangrove ecotourism, activities related to enhancing social and economic benefits for the surrounding communities should also be taken as primary objective. This needs to be done to avoid the behavior of the people around the mangrove to not take actions that are not environmentally friendly when exploiting the mangrove forest. Therefore, increasing the social and economic status of communities around the mangroves is very necessary to meet their needs without destroying the mangrove forest ecosystem. This point is important since local people are the main stakeholder in protecting mangrove forests.

CONCLUDING REMARKS

In conclusion, various anthropogenic activities are the drivers of mangrove deforestation and degradation in Indonesia, including domestic activities, land conversion, mining and logging activities. Anthropogenic activities can reduce the extent as well as the function of the mangrove ecosystem. The impacts arising from anthropogenic activities on mangrove forests are the reduction of mangrove forests, changes in mangrove composition, uneven species abundance, mass mangrove mortality, pollution in mangrove areas, rising sea levels, global climate change, and coastal erosion or abrasion. Solutions that can be done to conserve and restore mangrove forests include increasing community awareness regarding the importance of mangrove forests, replanting mangrove forests, and developing community participation in maintaining natural resources. Three main things that can be done in efforts to conserve mangrove forests include protection of life support systems, preservation of flora and fauna diversity as well as ecosystems, and sustainable use of biological resources and ecosystems.

REFERENCES

Afriza D, Effendi I, Siregar YI. 2019. Isolation, identification, and antagonism test heterotrohic bacteria in mangrove plants againts pathogenic bacteria (*Vibrio alginolyticus, Aeromonas hydrophila*, and *Pseudomonas* sp.). Jurnal Perikanan dan Kelautan 24 (1): 61-68. [Indonesian]

Ahnanto, Syahpirudin E, Waskita IP, Novita, Hartati S, Tjala A, Zid M. 2014. Urgensi pelestarian dan rehabilitasi mangrove bagi masyarakat Desa Pantai Mekar Kecamatan Muara Gembong. Wahana Komunikasi dan Informasi Geografi 12 (2): 28-34. DOI: 10.21009/SPATIAL.122.05. [Indonesian]

Akbar N, Baksir A, Tahir I. 2015. Struktur komunitas ekosistem mangrove di kawasan pasir Sidangoli Kabupaten Halmahera Barat, Maluku Utara. Depik Jurnal 4 (3): 132-143. DOI: 10.13170/depik.4.3.3052. [Indonesian]

Akbar N, Marus I, Haji I, Abdullah S, Umalekhoa S, Ibrahim FS, Ahmad M, Ibrahim A, Kahar A, Tahir I. 2017. Struktur komunitas hutan Mangrove di Teluk Dodinga, Kabupaten Halmahera Barat Provinsi Maluku Utara. Jurnal Enggano 2 (1): 78-89. DOI: 10.31186/jenggano.2.1.78-89. [Indonesian]

Ali SH, Polii BJV, Rotinsulu WC. 2017. Kajian lingkungan hidup strategis (KLHS) Kawasan Peruntukan Pariwisata di Kabupaten Bolaang Mongondow. Agro-Sosio Ekonomi 13 (3): 331-340. DOI: 10.35791/agrsosek.13.3A.2017.18546. [Indonesian]

Ambo-rappe R, Gatta R, Mappangara S, Ukkas M, Faizal A. 2020. Peran generasi milenial dalam pelestarian mangrove dan cagar budaya di Desa Sanrobone, Sulawesi Selatan. Jurnal Penyuluhan 16 (2): 213-223. DOI: 10.25015/16202030038. [Indonesian]

Anggraini N, Trisakti B, Soesilo TEB. 2012. Pemanfaatan data satelit untuk analisis potensi genangan dan dampak kerusakan akibat kenaikan muka air laut. Jurnal Penginderaan Jauh 9 (2): 140-151. [Indonesian]

Anhar FP, Hidayat A, Ekayani M. 2019. Analisis nilai manfaat dan kerugian dari pemanfaatan ekosistem mangrove di Pulau Tanakeke, Sulawesi Selatan. Jurnal Sosial Ekonomi Kelautan dan Perikanan 14 (1): 1-12. DOI: 10.15578/jsekp.v14i1.6773. [Indonesian]

Anwar Y, Setyasih I, Ardiansyah, Partini D, Dewi RP, Wibowo YA. 2021. Identification of mangrove forest damage, and effort to conservation in Balikpapan city, East Kalimantan, Indonesia. GeoEco 7 (2): 121-134. DOI: 10.20961/ge.v7i2.46360.

Ariftia RI, Qurniati R, Herwanti S. 2014. Nilai ekonomi total hutan mangrove Desa Margasari Kecamatan Labuhan Maringgai Kabupaten Lampung Timur. Jurnal Sylva Lestari 2 (3): 19-28. DOI: 10.23960/jsl3219-28. [Indonesian]

Athirah A, Asaf R, Ratna E. 2013. faktor lingkungan yang mempengaruhi produktivitas menggunakan aplikasi analisis jalur di tambak bandeng Kabupaten Indramayu, Provinsi Jawa Barat. Jurnal Kelautan Nasional 8 (1): 35-47. DOI: 10.15578/jkn.v8i1.6219. [Indonesian]

Basyuni M, Hamzah, Rahayu S, Siregar UJ. 2012. Pengaruh aktivitas antropogenik terhadap keragaman genetik *Rhizophora mucronata* Lamk. di hutan mangrove Secanggang Sumatera Utara. FORESTA 1 (2): 41-48. [Indonesian]

Biswas P, Biswas SR. 2019. Mangrove Forests: Ecology, Management, and Threats. Life on Land. DOI: 10.1007/978-3-319-71065-5_26-1.

Biswas SR, Khan MSI, Mallik AU. 2012. Invaders' control on post-disturbance succession in coastal mangroves. J Plant Ecol 5: 157-166. DOI: 10.1093/jpe/rtr050.

Budiastuti P, Raharjo M, Dewanti NAY. 2016. Analisis pencemaran logam berat timbal di badan sungai Babon Kecamatan Genuk Semarang. Jurnal Kesehatan Masyarakat 4 (5): 119-125. [Indonesian]

Carugati L, Gatto B, Rastelli E, Martire ML, Coral C, Greco S, Danovaro R. 2018. Impact of mangrove forests degradation on biodiversity and ecosystem functioning. Sci Rep 8: 13298. DOI: 10.1038/s41598-018-31683-0.

Cooray PLIGM, Jayawardana DT, Gunathilake BM, Pupulewatte PGH. 2021. Characteristics of tropical mangrove soils and relationships with forest structural attributes in the northern coast of Sri Lanka. Reg Stud Mar Sci 44: 101741. DOI: 10.1016/j.rsma.2021.101741.

Couture JJ, Lindroth RL. 2013. Impacts of Atmospheric Change on Tree–Arthropod Interactions. Develop Environ Sci 13: 227-248. DOI: 10.1016/B978-0-08-098349-3.00011-6.

Dako FX, Purwanto RH, Farida LRW, Sumardi. 2018. Kerusakan antropogenik di Kawasan Hutan Lindung Mutis Timau dan upaya penanggulangannya di Pulau Timor bagian barat. Journal of Natural Resources and Environmental Managemen. 9 (2): 437-455. [Indonesian]

DasGupta R, Shaw R. 2013. Cumulative impacts of human interventions and climate change on mangrove ecosystems of South and Southeast Asia: An overview. J Ecosyst 2013: 379429. DOI: 10.1155/2013/379429.

Diarto, Hendrarto B, Suryoko S. 2012. Partisipasi masyarakat dalam pengelolaan lingkungan kawasan hutan mangrove Tugurejo di Kota Semarang. Jurnal Ilmu Lingkungan 10 (1): 1-7. DOI: 10.14710/jil.10.1.1-7. [Indonesian]

Djumanto. 2020. Conservation Management of Planted Mangroves Through Evaluating Ecosystem Services in Baros Village Bantul Regency, Indonesia. Mangrove Ecosystem Restoration. DOI: 10.5772/intechopen.93780.

do Sul JAI, Costa MF, Silva-Cavalcanti JS, Araújo MCB. 2013. Plastic debris retention and exportation by a mangrove forest patch. Mar Pollut Bull 78 (1-2): 252-257. DOI: 10.1016/j.marpolbul.2013.11.011.

Eddy S, Mulyana A, Ridho MR, Iskandar I. 2015. Dampak aktivitas antropogenik terhadap degradasi hutan mangrove di Indonesia. Jurnal Lingkungan dan Pembangunan 1 (3): 240-254. [Indonesian]

Efriyeldi, Ahmadryadi, Amin B. 2018. Kondisi morfometrik *Rhixophora apiculata* Pada kawasan dengan aktivitas antropogenik berbeda di pesisir timur Indragiri, Hilir, Sumatera. Asian Journal of Environment, History, and Heritage 2 (1): 113-121. [Indonesian]

Fitriah E, Maryuningsih Y, Chandra E, Mulyani A. 2013. Studi analisis pengelolaan hutan mangrove Kabupaten Cirebon. Jurnal Scientiae Educatia 2 (2): 1-18. [Indonesian]

Gumilar I. 2012. Partisipasi masyarakat pesisir dalam pengelolaan ekosistem hutan mangrove berkelanjutan di Kabupaten Indramayu. Jurnal Akuatik 3 (2): 198-211. [Indonesian]

Hakim L, Siswanto D, Nakagoshi N. 2017. Mangrove conservation in East Java: The ecotourism development perspectives. J Trop Life Sci 7 (3): 277-285. DOI: 10.11594/jtls.07.03.14.

Handayani S, Adrianto L, Bengen DG, Nurjaya IW, Wardiatno Y. 2020. Pemetaan jasa ekosistem mangrove pada wilayah rehabilitasi di Pesisir Sayung, Kabupaten Demak. Jurnal Ilmu Pertanian Indonesia 25 (4): 574-583. DOI: 10.18343/jipi.25.4.574. [Indonesian]

Hartati, Harudu L. 2016. Identifikasi jenis-jenis kerusakan ekosistem hutan mangrove akibat aktivitas manusia di Kelurahan Lowu-Lowu Kecamatan Lea-Lea Kota Baubau. Jurnal Penelitian Pendidikan Geografi 1 (1): 30-44. [Indonesian]

Hendrawan, Goal JL, Susilo. SB 2018. Studi kerapatan dan perubahan tutupan mangrove menggunakan citra satelit di Pulau Sebatik Kalimantan Utara. Jurnal Ilmu dan Teknologi Kelautan Tropis 10 (1): 99-109. DOI: 10.29244/jitkt.v10i1.18595. [Indonesian]

Heriyanto NM, Subiandono E. 2012. Komposisi dan struktur tegakan, biomassa, dan kandungan karbon hutan mangrove di Taman Nasional Alas Purwo. Jurnal Penelitian Hutan dan Konservasi Alam 9 (1): 23-32. DOI: 10.20886/jpkha.2012.9.1.023-032. [Indonesian]

Heriyanto T, Amin B, Rahimah I, Ariani F. 2020. Analisis biomassa dan cadangan karbon pada ekosistem mangrove di kawasan pantai berpasir Desa Kawal Kabupaten Bintan. Jurnal Manajemen Riset dan Teknologi 2 (1): 31-41. DOI: 10.30649/fisheries.v2i2.36. [Indonesian]

Hidayah Z, Suharyo OS. 2018. Analisa perubahan penggunaan lahan wilayah pesisir selat Madura. Jurnal Ilmiah Rekayasa 11 (1): 19-30. DOI: 10.21107/rekayasa.v11i1.4120. [Indonesian]

Himes-Cornell A, Grose SO, Pendleton L. 2018. Mangrove Ecosystem Service Values and Methodological Approaches to Valuation: Where Do We Stand?. Front Mar Sci DOI: 10.3389/fmars.2018.00376.

Indrayanti MD, Fahrudin A, Setiobudiandi I. 2015. Penilaian jasa ekosistem di Teluk Blanakan Kabupaten Subang. Jurnal Ilmu Pertanian Indonesia 20 (2): 91-96. DOI: 10.18343/jipi.20.2.91. [Indonesian]

Jumaedi S. 2016. Nilai manfaat hutan mangrove dan faktor-faktor penyebab konservasi zona sabuk hijau (*Greenbelt*) menjadi tambak di Wilayah Pesisir Kota Singkawang Kalimantan Barat. Sosiohumaniora 18 (3): 227-234. DOI: 10.24198/sosiohumaniora.v18i3.10104. [Indonesian]

Kadarsah A, Salim D, Husain S, Dinata M. 2020. Species density and lead (Pb) pollution in mangrove ecosystems, South Kalimantan. Jurnal Biodjati 5 (1): 70-81. DOI: 10.15575/biodjati.v5i1.7411.

Kepel TL, Ati RNA, Rahayu YP, Adi NS. 2018. Pengaruh alih fungsi kawasan mangrove pada sifat sedimen dan kemampuan penyimpanan Karbon. Jurnal Kelautan Nasional 13 (3): 145-153. DOI: 10.15578/jkn.v13i3.6620. [Indonesian]

Kinanti TE, Rudiyanti S, Purwanti F. 2014. Kualitas perairan sungai bremi kabupaten pekalongan ditinjau dari faktor fisika-kimia sedimen dan kelimpahan hewan makrobentos. Journal of Maquares 3 (1): 160-167. DOI: 10.14710/marj.v3i1.4433. [Indonesian]

Konom NH, Cabuy RL, Wanma AO. 2019. Identifikasi kerusakan areal hutan mangrove akibat aktivitas penduduk di daerah Air Tuba Kabupaten Kaimana. Jurnal Kehutanan Papuasia 5 (3): 153-163. DOI: 10.46703/jurnalpapuasia.Vol5.Iss2.148. [Indonesian]

Kusmana C. 2011. Management of mangrove ecosystem in Indonesia. Jurnal Pengelolaan Sumberdaya Alam dan Lingkungan 1 (2): 152-157. DOI: 10.29244/jpsl.1.2.152. [Indonesian]

Kustanti A. 2011. Manajemen Hutan Mangrove. IPB Press, Bogor. [Indonesian]

Kustanti A, Nugroho B, Nurrochmat DR, Okimoto Y. 2014. Evolusi hak kepemilikan dalam pengelolaan ekosistem hutan mangrove di Lampung mangrove center. Risalah Kebijakan Pertanian dan Lingkungan 1 (3): 143-158. DOI: 10.20957/jkebijakan.v1i3.10291. [Indonesian]

Maiti SK, Chowdhury A. 2013. Effects of anthropogenic pollution on mangrove biodiversity: A review. J Environ Protect 4 (12): 40795. DOI: 10.4236/jep.2013.412163.

Majid I, MMuhdar MHIA, Rohman F, Syamsuri I. 2016. Konservasi hutan mangrove di Pesisir Pantai Kota Ternate Terintegrasi dengan kurikulum sekolah. Jurnal Bioedukasi 4 (2): 488-496. [Indonesian]

Malik A, Mertz O, Fensholt R. 2017. Mangrove forest decline: Consequences for livelihoods and environment in South Sulawesi. Reg Environ Change 17: 157-169. DOI: 10.1007/s10113-016-0989-0.

Mappanganro F, Asbar, Danial. 2018. Inventarisasi kerusakan dan strategi rehabilitas hutan mangrove di Desa Keera Kecamatan Keera Kabupaten Wajo. Jurnal Pendidikan Teknologi Pertanian 4: 1-11. DOI: 10.26858/jptp.v1i0.6227. [Indonesian]

Martin C, Almahasheer H, Duarte CM. 2019. Mangrove forests as traps for marine litter. Environ Pollut 247: 499-508. DOI: 10.1016/j.envpol.2019.01.067.

Martin C, Baalkhuyur F, Valluzzi L, Saderne V, Cusack M, Almahasheer H, Krishnakumar PK, Rabaoui L, Qurban MA, Arias-Ortiz A, Masqué P, Duarte M. 2020. Exponential increase of plastic burial in mangrove sediments as a major plastic sink. Sci Adv 6 (44): eaaz5593. DOI: 10.1126/sciadv.aaz5593.

Martuti NKT, Susilowati SME, Sidiq WABN, Mutiatari DP. 2018. Peran kelompok masyarakat dalam rehabilitasi ekosistem mangrove di Pesisir Kota Semarang. Jurnal Wilayah dan Lingkungan 6 (2): 100-114. DOI: 10.14710/jwl.6.2.100-114. [Indonesian]

Matatula J, Poedjirahajoe E, Pudyatmoko S, Sadono R. 2018. Sebaran spasial kondisi lingkungan hutan mangrove di Pesisir Pantai Kota Kupang. Jurnal Pengelolaan Sumberdaya Alam dan Lingkungan 9 (2): 467-482. DOI: 10.29244/jpsl.9.2.467-482. [Indonesian]

Muharuddin. 2019. Peran dan fungsi dalam penanggulangan kerusakan lingkungan. JUSTISI 5 (2): 97-112. DOI: 10.33506/js.v5i2.544. [Indonesian]

Nandini R, Narendra BH. 2018. Kajian perubahan curah hujan, suhu, dan tipe iklim pada zona ekosistem di Pulau Lombok. Jurnal Analisis Kebijakan Kehutanan 8 (3): 228-244. DOI: 10.20886/jakk.2011.8.3.228-244. [Indonesian]

Nugraha B, Banuwa BI, Widagdo S. 2015. Perencanaan lanskap ekowisata hutan mangrove di Pantai Sari Ringgung Desa Sidodadi Kecamatan Padang Cermin Kabupaten Pesawaran. Jurnal Sylva Lestari 3 (2): 53-66. DOI: 10.23960/jsl2353-66. [Indonesian]

Nugroho RA, Widada S, Pribadi R. 2013. Studi kandungan bahan organik dan mineral (N, P, K, Fe, dan Mg) sedimen di Kawasan Mangrove Desa Bedono, Kecamatan Sayung, Kabupaten Demak. J Mar Res 2 (1): 62-70. DOI: 10.14710/jmr.v2i1.2057. [Indonesian]

Numbere AO. 2019. Municipal Solid Waste Disposal in Mangrove Forest: Environmental Implication and Management Strategies in the Niger Delta, Nigeria. IntechOpen. DOI: 10.5772/intechopen.83809.

Oktavia S, Adi W, Pamungkas A. 2020. Persepsi dan partisipasi pengunjung terhadap permasalahan sampah laut di pantai Temberan dan pantai Pasir Padi. Journal of Tropical Marine Science 3 (1): 11-20. DOI: 10.33019/jour.trop.mar.sci.v3i1.1448. [Indonesian]

Paddiyatu N. 2018. Analisis tingkat kerusakan wilayah pesisir di Kabupaten Mamuju Tengah. Jurnal Ilmu Arsitektur 1 (2): 91-102. DOI: 10.26618/j-linears.v1i2.1814. [Indonesian]

Pontoh O. 2011. Peranan nelayan terhadap rehabilitasi ekosistem hutan bakau (Mangrove). Jurnal Perikanan dan Kelautan Tropis 7 (2): 73-79. DOI: 10.35800/jpkt.7.2.2011.181. [Indonesian]

Prasetyo A, Santoso N, Prasetyo LB. 2017. Kepekaan lingkungan mangrove terhadap tumpahan minyak di Kecamatan Ujung Pangkah, Gresik. Jurnal Penelitian Hutan dan Konservasi Alam 14 (2): 91-98. DOI: 10.20886/jphka.2017.14.2.91-98. [Indonesian]

Purnamasari E, Kamal M, Wicaksono P. 2021. Comparison of vegetation indices for estimating above-ground mangrove carbon stocks using PlanetScope image. Reg Stud Mar Sci 44: 101730. DOI: 10.1016/j.rsma.2021.101730.

Purnobasuki H. 2011. Ancaman terhadap hutan mangrove di indonesia dan langkah strategis pencegahannya. Bulletin PSL Universitas Surabaya 25: 3-6. [Indonesian]

Putra AR, Sudhartomo A, Ramlah S. 2017. Eksploitasi di Suaka Margasatwa Tanjung Santigi Kabupaten Parigi Moutong. Warta Rimba 5 (1): 87-92. [Indonesian]

Rahim S, Baderan DWK. 2017. Hutan Mangrove dan Pemanfaatannya. Penerbit Deepublish, Yogyakarta. [Indonesian]

Rahman, Effendi H, Rusmana I, Yulianda F, Wardiatno Y. 2020. Pengelolaan ekosistem mangrove untuk ruang terbuka hijau sebagai mitigasi gas rumah kaca di kawasan Sungai Tallo Kota Makassar. Jurnal Pengelolaan Sumberdaya Alam dan Lingkungan 10 (2): 320-328. DOI: 10.29244/jpsl.10.2.320-328. [Indonesian]

Rasyid A, Akbar ASM, Nurdin N, Jaya I, Ibrahim. 2016. Impact of human interventions on mangrove ecosystem in spatial perspective. IOP Conf. Series: Earth and Environmental Science 47: 012041. DOI: 10.1088/1755-1315/47/1/012041.

Rimmer MA, Sugama K, Rakhmawati D, Rofiq R, Habgood RH. 2013. A review and SWOT analysis of aquaculture development in Indonesia. Rev Aquac 5 (4): 255-279. DOI: 10.1111/raq.12017.

Ritohardoyo S, Ardi GB. 2011. Arahan kebijakan pengelolaan hutan mangrove: Kasus Pesisir Kecamatan Teluk Pakedai, Kabupaten Kubu Raya, Provinsi Kalimantan Barat. Jurnal Geografi 8 (2): 83-94. [Indonesian]

Riwayati. 2014. Manfaat dan fungsi hutan mangrove bagi kehidupan. Jurnal Keluarga Sehat Sejahtera 12 (24): 17-23. [Indonesian]

Romañach SS, DeAngelis DL, Koh HL, Li Y, Teh, SY Barizan RSR, Zhai L. 2018. Conservation and restoration of mangroves: Global status, perspectives, and prognosis. Ocean Coastal Manage 154: 72-82. DOI: 10.1016/j.ocecoaman.2018.01.009.

Rudianto, Bengen DG, Kurniawan F. 2020. Causes and effects of mangrove ecosystem damage on carbon stocks and absorption in East Java, Indonesia. Sustainability 12: 10319. DOI: 10.3390/su122410319.

Rusdianti K, Sunito S. 2012. Konservasi lahan hutan mangrove serta upaya penduduk lokal dalam merehabilitasi ekosistem Mangrove. Jurnal Sosiologi Pedesaan 6 (1): 1-17. DOI: 10.22500/sodality.v6i1.5815. [Indonesian]

Samosir DD, Restu. 2017. Analisis manfaat hutan mangrove di Desa Tanjung Rejo Kecamatan Percut Sei Tuan Kabupaten Deli Serdang Sumatera Utara. Jurnal Geografi 6 (1): 1-15. DOI: 10.24114/tgeo.v6i1.8344. [Indonesian]

Samsumarlin, Rachman I, Toknok B. 2015. Studi zonasi vegetasi mangrove muara di Desa Umbele Kecamatan Bumi Raya Kabupaten Morowali Sulawesi Tengah. Warta Rimba 3 (2): 148-154. [Indonesian]

Saprudin, Halidah. 2012. Potensi nilai manfaat jasa lingkungan hutan mangrove di Kabupaten Sinjai Sulawesi Selatan. Jurnal Penelitian Hutan dan Konservasi Alam 9 (3): 213-219. DOI: 10.20886/jphka.2012.9.3.213-219. [Indonesian]

Setyawan AD, Susilowati A, Wiryanto. 2002. Habitat reliks vegetasi mangrove di Pantai Selatan Jawa. Biodiversitas 3 (2): 242-256. DOI: 10.13057/biodiv/d030206. [Indonesian]

Setyawan AD, Winarno K, Purnama PC. 2003. Review: Ekosistem mangrove di Jawa: 1. Kondisi terkini. Biodiversitas 4 (2): 133-145. DOI: 10.13057/biodiv/d040211. [Indonesian]

Setyawan AD, Winarno K, Purnama PC. 2004. Review: Ekosistem mangrove di Jawa: 2. Restorasi. Biodiversitas 5 (2): 105-118. DOI: 10.13057/biodiv/d050212. [Indonesian]

Setyawan AD, Winarno K, Indrowuryatno, Wiryanto, Susilowati A. 2005a. Tumbuhan mangrove di Pesisir Jawa Tengah: 2. Komposisi dan struktur vegetasi. Biodiversitas 6 (3): 194-198. DOI: 10.13057/biodiv/d060312. [Indonesian]

Setyawan AD, Winarno K, Indrowuryatno, Wiryanto, Susilowati A. 2005b. Tumbuhan mangrove di Pesisir Jawa Tengah: 1. Keanekaragaman jenis. Biodiversitas 6 (2): 90-94. DOI: 10.13057/biodiv/d060204. [Indonesian]

Setyawan AD, Winarno K. 2006a. Pemanfaatan langsung ekosistem mangrove di Jawa Tengah dan penggunaan lahan di sekitarnya; kerusakan dan upaya restorasinya. Biodiversitas 7 (3): 282-291. DOI: 10.13057/biodiv/d070318. [Indonesian]

Setyawan AD, Winarno K. 2006b. Permasalahan konservasi ekosistem mangrove di Pesisir Kabupaten Rembang, Jawa Tengah. Biodiversitas 7 (2): 159-163. DOI: 10.13057/biodiv/d070214. [Indonesian]

Setyawan AD, Winarno K, Indrowuryatno, Wiryanto, Susilowati A. 2008. Tumbuhan mangrove di Pesisir Jawa Tengah: 3. diagram profil vegetasi. Biodiversitas 9 (4): 315-321. DOI: 10.13057/biodiv/d090416. [Indonesian]

Sofuan A. 2016. Upaya mengatasi kerentanan kawasan mangrove oleh masyarakat Desa Bondo Kecamatan Bangsri Kabupaten Jepara. Jurnal DISPROTEK 7 (1): 5-11. [Indonesian]

Sondakh JM, Suhaeni S, Wasak MP. 2019. Pengelolaan hutan mangrove berbasis kearifan lokal di Desa Tiwoho Kecamatan Wori Kabupaten Minahasa Utara Provinsi Sulawesi Utara. AKULTURASI 7 (1): 1077-1086. DOI: 10.35800/akulturasi.7.1.2019.24398. [Indonesian]

Sondakh VS, Suhaeni S, Lumenta V. 2019. Persepsi masyarakat terhadap pengelolaan hutan mangrove di Desa Tiwoho Kecamatan Wori Kabupaten Minahasa Utara Provinsi Sulawesi Utara. AKULTURASI 7 (1): 1049-1058. DOI: 10.35800/akulturasi.7.1.2019.24395. [Indonesian]

Subardjo P, Pribadi R. 2012. Studi perubahan luasan vegetasi mangrove menggunakan citra landsat TM dan landsat 7 ETM+ tahun 1998-2010 di Pesisir Kabupaten Mimika Papua. Journal of Marine Research 1 (1): 148-145. [Indonesian]

Sudarso J, Wardiatno Y, Setiyanto DD, Anggraitoningsih W. 2013. pengaruh aktivitas antropogenik di Sungai Ciliwung terhadap komunitas larva Trichoptera. Jurnal Manusia dan Lingkungan 20 (1): 68-83. [Indonesian]

Sukwika T, Putra H. 2018. Analisis sedimentasi dan konsentrasi atmosfer pada zona mangrove di Muaragembong, Bekasi. Jurnal Pengambangan Kota 6 (2): 186-195. DOI: 10.14710/jpk.6.2.186-195. [Indonesian]

Sulastri E, Haryadi FT. 2019. Tingkat kesadaran ekologis masyarakat kampung laut Kabupaten Cilacap Jawa Tengah. Jurnal Kawistara 9 (2): 78-90. DOI: 10.22146/kawistara.31484. [Indonesian]

Suriani M, Bahagia. 2012. Kajian pelaksanaan program rehabilitasi hutan mangrove pasca tsunami di Kecamatan Baitussalam Kabupaten Aceh Besar. Jurnal Geografi 4 (1): 99-110. [Indonesian]

Suzana BOL, Timban J, Kaunang R, Ahmad F. 2011. Valuasi ekonomi sumberdaya hutan mangrove di Desa Palaes Kecamatan Likupang Barat Kabupaten Minahasa Utara. ASE 7 (2): 29-38. DOI: 10.35791/agrsosek.7.2.2011.89. [Indonesian]

Syahroni A. 2016. Dinamika adaptif masyarakat Wonorejo terkait ekowisata mangrove Wonorejo Kelurahan Wonorejo, Kecamatan Rungkut, Kota Surabaya. AutoUnairdotNet 5 (3): 387-410. [Indonesian]

Syawal MS, Wardiatno Y, Hariyadi S. 2016. Pengaruh aktivitas antropogenik terhadap kualitas air, sedimen dan Moluska di Danau Maninjau, Sumatera Utara. Jurnal Biologi 16 (1): 1-44. DOI: 10.29303/jbt.v16i1.210. [Indonesian]

Tetelepta JMS, Loupatty SR, Wawo M. 2020. Sustainable management strategy for mangrove forest of Pelita Jaya Bay and Kotania Bay, Western Seram, Indonesia. Jurnal Triton 16 (2): 53-67. DOI: 10.30598/TRITONvol16issue2page53-67.

Trettin CC, Dai Z, Tang W, Lagomasino D, Thomas N, Lee SK, Simard M, Ebanega MO, Stoval A, Fatoyinbo TE. 2021. Mangrove carbon stocks in Pongara National Park, Gabon. Estuarine Coastal Shelf Sci 259: 107432. DOI: 10.1016/j.ecss.2021.107432.

Ulumuddin YI, Setyawan AD. 2017. Eksplorasi hutan mangrove di Kepulauan Tambelan dan Serasan: Komposisi jenis, peta distribusi hutan, dan potensi ancaman. Pros Sem Nas Masy Biodiv Indon 3 (1): 45-55. DOI: 10.13057/psnmbi/m030109. [Indonesian]

Utomo B, Budiastuti S, Muryani C. 2017. Strategi pengelolaan hutan mangrove di Desa Tanggul Tlare Kecamatan Kedung Kabupaten Jepara. Jurnal Ilmu Lingkungan 15 (2): 117-123. DOI: 10.14710/jil.15.2.117-123. [Indonesian]

van Bijsterveldt CEJ, van Wesenbeeck BK, Ramadhani S,Raven OV, van Gool FE, Pribadi R, Bouma TJ. 2021. Does plastic waste kill mangroves? A field experiment to assess the impact of macro plastics on mangrove growth, stress response and survival. Sci Total Environ 756: 143826. DOI: 10.1016/j.scitotenv.2020.143826.

Wahyuni Y, EPutri EIK, Simanjuntak SMH. 2014. Valuasi total ekonomi hutan mangrove di Kawasan Delta Mahakam Kabupaten Kutai Kartanegara Kalimantan Timur. Jurnal Penelitian Kehutanan Wallacea 3 (1): 1-12. DOI: 10.18330/jwallacea.2014.vol3iss1pp1-12. [Indonesian]

Wang G, Singh M, Wang J, Xiao L, Guan D. 2021. Effects of marine pollution, climate, and tidal range on biomass and sediment organic carbon in Chinese mangrove forests. Catena 202: 105270. DOI: 10.1016/j.catena.2021.105270.

Wang Y-S, Gu J-D. 2021. Ecological responses, adaptation and mechanisms of mangrove wetland ecosystem to global climate change and anthropogenic activities. Intl Biodeterioration Biodegradation 162: 105248. DOI: 10.1016/j.ibiod.2021.105248.

Wardhani MK. 2011. Kawasan konservasi mangrove: Suatu potensi ekowisata. Jurnal Kelautan 4 (1): 60-79. [Indonesian]

Widiastuti MMM, Ruata NN, Arifin T. 2016. Valuasi ekonomi ekosistem mangrove di Wilayah Pesisir Kabupaten Merauke. Jurnal Sosek 11 (2): 147-159. DOI: 10.15578/jsekp.v11i2.3856. [Indonesian]

Wijaya MS, Aryaguna PA, Rudiastuti AW, Rahmayani W, Widiastuti R, Hartini S. 2018. Penentuan prioritas pembaharuan peta mangrove Indonesia menggunakan model forest canopy density. Majalah Ilmiah Globe 20 (2): 99-106. DOI: 10.24895/MIG.2018.20-2.858. [Indonesian]

Winata A, Yuliana E. 2016. Tingkat keberhasilan penanaman pohon mangrove (Kasus: Pesisir Pulau Untung Jawa Kepulauan Seribu). Jurnal Matematika, Sains, dan Teknologi 17 (1): 29-39. [Indonesian]

Yuliani E, Liesnoor D, Aji A. 2018. Pelaksanaan Pendidikan konservasi untuk pelestarian hutan mangrove pada kelompok peduli lingkungan Pantai Selatan (Kpl Pansela) Desa Ayah Tahun 2016. Edu Geography 6 (1): 8-15. [Indonesian]

Zulfia N, Aisyah. 2013. Status trofik perairan rawa pening ditinjau dari kandungan unsur hara (NO_3 dan PO_4) serta Klorofil a. Bawal 5 (3): 189-199. [Indonesian]

INTL J BONOROWO WETLANDS
Volume 12, Number 1, June 2022
Pages: 23-32

P-ISSN: 2775-8052
E-ISSN: 2775-8044
DOI: 10.13057/bonorowo/w120103

Benthic infaunal spatial biodiversity, coexistence, and availability for shorebird communities in the Jakarta Coastal Wetlands, Indonesia

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Manuscript received: 4 May 2022. Revision accepted: 8 June 2022.

Abstract. *Wibowo AA, Basukriadi A, Nurdin E, Nasution NS. 2022. Benthic infaunal spatial biodiversity, coexistence, and availability for shorebird communities in the Jakarta Coastal Wetlands, Indonesia. Intl J Bonorowo Wetlands 12: 23-32.* Wetlands on the Jakarta Coast, Indonesia, is one of the important habitats for shorebirds. In the wetland ecosystem, benthic infauna is an important food source for the shorebird community. This study aims to assess the benthic infaunal spatial biodiversity and its availability for the shorebird community in the wetlands covering an ecotourism area and the west and east parts of a protected mangrove forest. The bird diversity was assessed using visual encounter surveys and 10 x 10 m plots for mangroves. The K-means clustering method measured the benthic infaunal coexistence and availability for shorebirds. Based on the result, benthic infauna was available as a food resource for shorebirds in the following order: Oligochaeta > Polychaeta > Gastropod > Crustacea, with the density of oligochaetes reaching 376.66 inds/100 m^2. The Shannon-Wiener biodiversity index (H’) ranges for benthic infauna were 1.306-1.573. Gastropods were available to Ciconiidae and Anhingidae; oligochaetes and polychaetes were available to Anatidae, Scolopacidae, and Phalacrocoracidae. In contrast, crustaceans were not available to any shorebird species. As confirmed in this study, shorebird species were associated with the presence of benthic infaunal communities and it recommends conserving the wetlands to ensure the availability of benthic infauna.

Keywords: Biodiversity, K-means, mangrove, Oligochaeta, spatial

Abbreviations: GIS: Geographical Information System, GPS: Global Positioning System, SAGA: System for Automated Geoscientific Analyses

INTRODUCTION

Coastal ecosystems, with their wetland areas, are vital to shorebirds (Rija et al. 2015; Amarasekara et al. 2021) and also benthic infaunal communities (Rabalais and Baustia 2020). The Ramsar Convention on Wetlands defines wetlands as areas of marsh, fen, peatland, or water, whether natural or man-made, permanent or temporary, containing water (Gaget et al. 2020). Wetlands are among the world's most productive ecosystems, as well as in Indonesia. Furthermore, for thousands of years, those wetlands have provided food, drinking water, building materials, and many other services to human populations. Wetlands also play an important role in preserving global biodiversity, partly through their high production, which supports food chains, and partly by providing habitat for especially suited plant and animal species, including benthic infauna and shorebirds.

Wetlands in Indonesia are characterized by vast mangrove cover. Indonesia has 3 million ha or 23% of the world’s mangrove area (Murdiyarso et al. 2015). Those wetlands and mangroves in Indonesia are important habitats for shorebird communities. For example, in the wetlands of Panjang Island Coast, Jepara, Central Java, there were 27 shorebird species from 15 families, with the biodiversity index of the shorebird species ranging from 1.15 to 2.20 (Utami et al. 2017). At Maron Beach, Prasetyo and Wulandari (2021) reported that there were 42 bird species from 20 families, with a Shannon-Wiener biodiversity index (H’) of 2.915. Meanwhile, in the wetlands of Tegal Coast, Central Java, there were 37 bird species from 18 families, with H’ values that ranged from 2.22 to 2.37 (Isworo and Oetari 2020).

Benthic infauna includes molluscs, polychaetes, oligochaetes, and crustaceans (Rabalais and Baustia 2020). Wetlands in Indonesia, particularly those on Java Island Coast, were also important habitats for benthic infauna (Andriyono et al. 2016). Then, Sahidin et al. (2014) reported a total of 5458 individuals of benthic infauna with a density ranging from 177 inds/m^2 to 634 inds/m^2 in Tangerang Coast, Banten. In the mangrove ecosystems on the Pacitan Coast, East Java, benthic infaunal diversity represented by molluscs has H 'values of 2.14, with 17 gastropod species belonging to 13 families (Wiraatmaja et al. 2022). For polychaetes and crustaceans, Katili and Utina (2019) reported that mangrove *Rhizophora* sp. contained more polychaetes and crustaceans than other mangrove species. In the Musi River wetland, the range of Polychaeta density was 1.054-2.831 inds/m^2 (Sari et al. 2022).

The presence of benthic infauna is very important as a food resource for shorebird communities. The distribution of shorebird communities in the wetlands is influenced by the distribution and availability of benthic infauna, including crustaceans, polychaetes, and oligochaetes (Pérez-Vargas et al. 2016). In the same way that the other coasts on Java Island have a wetland, Jakarta also has a wetland. Despite the abundance of studies on the distribution and abundance of shorebird and benthic infaunal communities along the Jakarta Coast, interactions, the spatial distributions of shorebirds, and the spatial availability of their prey, that is, the benthic infaunal communities were rarely investigated. This study is becoming more important recently since there has been a massive change in wetlands on the Jakarta Coast that has led to the reduction of wetland ecosystems (Sasongko et al. 2014; Sofian et al. 2019), followed by the decline of the benthic infaunal communities and shorebird communities. Here, this study aims to assess the benthic infaunal spatial biodiversity and its availability for the shorebird community in the wetlands of the Jakarta Coast, Indonesia.

MATERIALS AND METHODS

Study area

The study areas included 3 sampling locations selected based on the presence of mangroves in the wetlands of Jakarta Coast, Indonesia (Table 1), including an ecotourism park with latitude coordinates of 6.099^0-6.116^0 South and a longitude of 106.728^0-106.736^0, and the east and west parts of the Angke Kapuk protected forest with latitude coordinates of 6.099^0-6.116^0 South and a longitude of 106.736^0-106.769^0 (Figure 1). Fish ponds in the west bordered the ecotourism park. In contrast, the western parts of the Angke Kapuk protected Jakarta Bay, bordered by forest in the north and fish ponds in the west. The eastern parts of the Angke Kapuk protected forest are bordered by Jakarta Bay in the north and the Angke River in the east. Settlements in the south bordered all of the study areas. Regarding the study area's hydrological conditions, all of the study areas were wetlands that were permanently influenced by the tide and inundated by water.

Procedures

Shorebird survey

The shorebird survey in the wetlands of Jakarta Coast, Indonesia, was conducted for two months, from July to August 2021 with 3 replications for each location. The survey techniques included audiovisual encounter surveys and multiple surveys through random visits involving 2 observers (Buda and Budka 2019). The survey was conducted during various periods of the day using direct observations supported by binoculars and unaided eyes. Based on the bird activities, the survey was conducted from 05.30-7.00 am and continued at 04.00-06.15 pm with at least 2-3 sampling hours. The shorebird species richness of three distinct study areas, including an ecotourism park, west and east parts of Angke Kapuk protected forest, was recorded. The identification of birds was done using a bird identification book and field guide (MacKinnon and Phillipps 1993). The presence of birds is then tabulated into a Geographical Information System (GIS) to be mapped into mangrove and land cover thematic layers. The abundance of shorebirds was calculated as the number of individuals seen within a 100 m x 100 m plot and denoted as inds/Ha (Obunga et al. 2022).

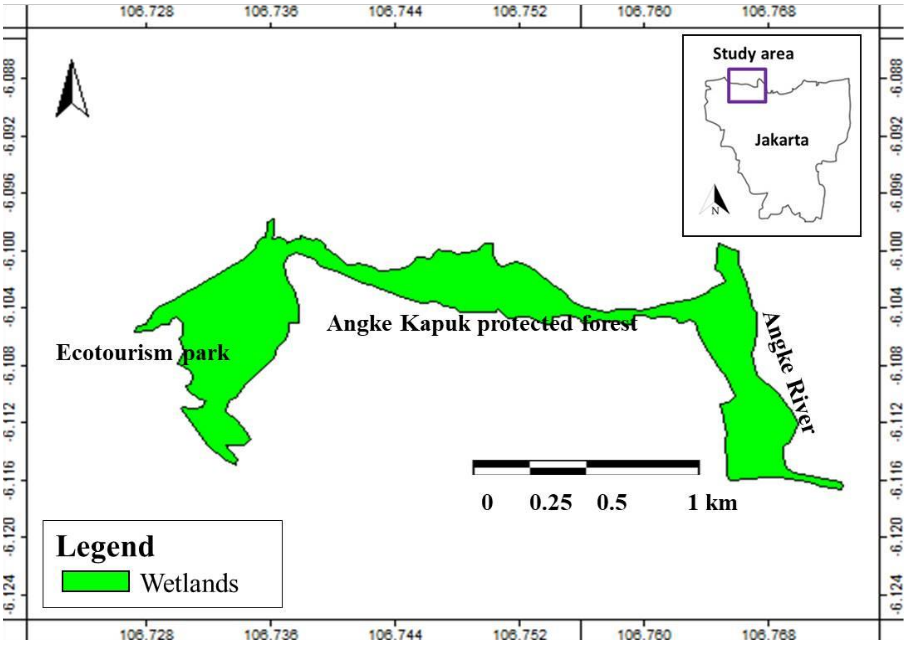

Figure 1. A map of the study area shows three study areas located in an ecotourism park and the west and east parts of the Angke protected forest in the wetlands of Jakarta Coast, Indonesia

Table 1. Locations and descriptions of the study area in the Jakarta Coast, Indonesia

Study area	South latitude	East longitude
Ecotourism park, bordered by fish ponds in the west	6.09^0 S - 6.116^0 S	106.728^0 E - 106.736^0 E
West parts of Angke Kapuk protected forest, bordered by Jakarta Bay in the north and fish ponds in the west	6.09^0 S - 6.116^0 S	106.736^0 E - 106.748^0 E
East parts of Angke Kapuk protected forest, bordered by Jakarta Bay in the north and Angke River in the east	6.09^0 S - 6.116^0 S	106.748^0 E - 106.769^0 E

Benthic infaunal survey

The benthic infaunal survey was conducted using the purposive random sampling method following Basyuni et al. (2018) by making observation plots with the size of each plot being 10 x 10 m or equal to 100 m^2. Each individual benthic infauna was collected from the substrate at a depth of 10 cm considering the maximum length of shorebird bill (Pérez-Vargas et al. 2016) using a 0.1 m^2 Van Veen Grab and sieved using a mesh sieve sized 0.5 mm to separate the benthic infauna from the sediment (Sahidin and Wardiatno 2016). The collected benthic infauna was then identified and its coordinates were recorded using a Garmin Etrex Global Positioning System (GPS) handheld device at the observation plot. The benthic infauna was identified using a benthic infauna identification book and field guide (Sahidin 2020). The presence of benthic infauna was then tabulated into a Geographical Information System (GIS) to be mapped into mangrove and land cover thematic layers.

Shorebird and benthic infaunal biodiversity index Shannon Wiener (H')

The biodiversity of shorebird and benthic infaunal communities was assessed using the biodiversity index Shannon-Wiener (H') with the following equation: $H' = \Sigma[Pi \ln (Pi)]$, where Pi is the proportion of the individuals i shorebird and the benthic infauna in total individuals. The H' range is from 0 (low diversity) to > 1 (high diversity).

Mangrove survey

The mangrove survey was conducted using the purposive random sampling method following Hutabarat (2009) and Sofian et al. (2012) by making observation plots with the size of each plot being 10 x 10 m. Each individual mangrove tree was identified and its coordinates were recorded using a Global Positioning System (GPS) handheld device at the observation plot. Mangrove species identification was conducted using a mangrove identification book and field guide (Chapman 2016; Hirsch 2016; Tomlinson 2016). The recorded mangrove species are then tabulated for further mangrove and land cover mapping.

Mangrove, land cover, shorebird, and benthic infaunal spatial analysis

Mangrove, land cover, shorebird, and benthic infaunal spatial analysis in the wetlands of Jakarta Coast was performed using remote sensing and GIS analyses using SAGA (System for Automated Geoscientific Analyses) GIS version 2.1.2 (Alevizos 2016) following Orimoloye et al. (2019), Philiani et al. (2016), and Sukojo and Arindi (2019). Landsat 8 satellite imagery of Jakarta Coast was retrieved and classified using supervised classification to determine the land cover types. The categories for land cover classifications were mangroves, water bodies, fish ponds, and settlements. The result of mangrove and land cover mapping is a thematic layer. The coordinates of shorebird and benthic infauna are then overlaid with the mangrove and land cover mapping for further analysis and interpretation.

Data analysis

K-means clustering

The availability and coexistence of benthic infaunal for shorebird communities were assessed and measured statistically using K-means clustering (D'Andrea et al. 2019). Suppose the benthic infaunal and shorebird communities were clustered together in one cluster. In that case, it indicates that the benthic infaunal and shorebird communities were overlapping, and the shorebird could utilize the benthic infauna as their food resources. Cluster methods follow current methods to determine the concentrations and hotspots of the benthic infaunal and shorebird communities in 3 studied areas (Zhao et al. 2019). The input data were the abundances of benthic infauna and shorebirds in each study area and were presented as points in the GIS interface. Cluster analysis was conducted using an extension of GIS and the cluster calculation was based on the K-means method. This method uses an algorithm that assigns each point to the cluster whose center, or known centroid is nearest. The center is the average of all the points in the cluster, and the coordinates of the points are the arithmetic mean for each dimension separately over all the points in the cluster. The determination of the centroid, or cluster point, was as follows:

$$z_1=(x_1+y_1)/2,\ z_2= (x_2+y_2)/2,\ z_3=(x_3+y_3)/2$$

Where: z = centroid, x = coordinate in axis x, y = coordinate in axis y

x^2 analysis

The goodness of fit (x^2) statistical test was performed to assess the different significance of benthic infauna among study areas. The significance level was $p < 0.05$.

Correlation analysis using Akaike Information Criterion (AIC)

Correlation analysis based on Akaike Information Criterion (AIC) was used to assess the correlation and dependence of shorebird species with benthic infaunal communities. The AIC was developed using linear regression with straight-line fit equations of $y_i = b_0 + b_1x_i + \varepsilon_i$. The ε_i represents the residuals from the straight line fit. If the ε_i is considered to be i.i.d. (independent and identically distributed) Gaussian with zero mean, the model contains three parameters: b_0, b_1, and the Gaussian distributions' variance. As a result, we should use $k = 3$ when calculating the AIC value of this model. In general, the variance of the residuals' distributions should be counted as one of the parameters in any least squares model using i.i.d. Gaussian residuals. The measured parameters included in AIC, AICc, ΔAIC, AIC weight, AIC cum., and Loglikehood values. To build the model, shorebird density variables correlating with benthic infaunal density variables were included in the analysis to develop the model and assess the dependence of both variables. The best model was selected based on the model that has the lowest AIC values.

RESULTS AND DISCUSSION

Mangrove and land covers

Land covers along with their mangrove covers on the study area in the wetlands of Jakarta Coast can be seen in Figure 2. The land cover consists of mangrove cover and is followed by combinations of water bodies and fish ponds. Mangrove cover was very dense in the west and east parts of the Angke Kapuk protected forest and less in the ecotourism park. In contrast, water bodies and fish pond combinations were more common in the ecotourism parks and fewer in the protected forest. The presence of water bodies in the eastern parts of the protected forest was due to the presence of the Angke River, located on the east side of the protected forest. A common settlement was found on the protected forest's southern border and the ecotourism parks' east side. The mangroves in the study area were dominated by *Rhizophora* sp. The density of mangrove *Rhizophora* sp. ranged from 3807.2 trees/ha to 1399.5 trees/ha.

Benthic infauna and shorebird communities associations with the wetlands

The benthic infauna in the wetlands consisted of individuals belonging to polychaetes, oligochaetes, gastropods, and crustaceans. The benthic infaunal species include *Melanoides* sp., *Pomacea* sp., *Littorina* sp. for gastropods, *Tubifex* sp. for oligochaetes, and *Nereis* sp. for polychaetes. The ranges of H' for benthic infauna were 1.306-1.573. The density of benthic infauna was significantly different ($p < 0.05$) in each study area ($x^2 = 464.848$, $p < 0.001$), indicating influences and effects of study area locations and land covers. Among benthic infauna, oligochaetes have the highest density (376.66 inds/100 m^2), followed by polychaetes, and crustaceans have the lowest density (3.333 inds/100 m^2). According to the study area, the protected forests have the highest benthic infaunal density with a range of 108.442 inds/100 m^2 to 263.441 inds/100 m^2 and the lowest benthic infauna density was found in an ecotourism park with a value of 81.77 inds/100 m^2. Despite the lowest benthic infaunal density in the ecotourism park, this study area has more benthic infauna species compared to other study areas due to the presence of crustaceans (Figure 3). The high density of benthic infauna in the west and east parts of the Angke Kapuk protected forest was due to the dominance of oligochaetes, followed by polychaetes. In this study area, oligochaetes were present more than 50% of the time, as observed in the wetlands of the Angke Kapuk protected forest (Figure 4, Table 2).

This finding is comparable with other studies. The land covered by the protected forest in the wetlands in this study is dominated by the mangrove forests that produce mangrove litter. In the mangrove ecosystems, mangrove litter is the potential food source for benthic infauna. The benthic infaunal communities were associated with the decomposition of leaves in a mangrove forest. Based on the previous study, benthic infauna was present from the fifth day of *Rhizophora*'s leaf decomposition times (De Oliveira et al. 2011). Besides the availability of the mangrove litter as a potential food resource for benthic infauna, the protected forest is located adjacent to the sea. According to Metcalfe and Galsby (2007), the highest abundance was recorded in the soft, unconsolidated substrates of the seaward assemblage, with diversity and abundance decreasing progressively in the landward assemblages. The decline of benthic infaunal species, in particular oligochaetes and polychaetes, toward land was also observed in this study. The oligochaetes, followed by the polychaetes, were declining in the ecotourism park, which is landward and has fewer mangroves covered.

In contrast to benthic infauna that belongs to oligochaetes and polychaetes, in this study, benthic infaunal groups that belong to gastropods had a wider spatial distribution and tended to be contrasted with another benthic infauna that belongs to oligochaetes and polychaetes. In general, gastropods were more available in the ecotourism park and the eastern parts of the protected forest. On the contrary, the distributions of gastropods were limited in the areas characterized by a dense mangrove forest. In fact, gastropods preferred a wetland with the dominance of a water body combined with a water pond. The distribution of benthic infauna in brackish water areas is restricted by factors such as elevation, salt content, substratum, food, and respiration systems (Liu et al. 2014). Gastropod is a benthic infauna with a respiration system that depends on the gills (Koopman et al. 2016). It means that the gastropod requires an environment with stagnant water, also known as a tidal flat, to allow the exchange of oxygen from water between gills and water. Then the water bodies and fish ponds in the ecotourism park and eastern parts of the protected forest were suitable for gastropods since those areas resemble tidal flats due to inundations.

Shorebirds in the wetlands consisted of individuals belonging to Anatidae, Scolopacidae, Ciconiidae, Phalacrocoracidae, and Anhingidae (Figure 3) and density (Table 3). The shorebird species include Sunda teal (*Anas gibberifrons* S. Muller, 1842) for Anatidae, Common sandpiper (*Actitis hypoleucos* Linnaeus, 1758), Wood sandpiper (*Tringa glareola* Linnaeus, 1758) for

Scolopacidae, the Milky stork (*Mycteria cinerea* Raffles, 1822) for Ciconiidae, Little black Cormorant (*Phalacrocorax sulcirostris* Brandt, 1837) for Phalacrocoracidae, and the Oriental darter (*Anhinga melanogaster* Pennant, 1769) for Anhingidae. The ranges of H' for benthic infauna were 1.000-2.322. West parts of the protected forest have more shorebirds compared to east parts and ecotourism park. Anatidae, Scolopacidae, and Phalacrocoracidae are widely distributed in the protected forests and are limited to ecotourism parks. In contrast, Ciconiidae and Anhingidae were widely distributed in ecotourism parks and limited in protected forests (Figure 5, Table 3). The distributions of shorebirds were related to the habitat availability for nesting and food resources in the wetland ecosystems in the particular protected forests. Diverse mangrove species and muddy substrates characterize those wetlands. Dense mangrove covers, as seen in the protected forest, provide suitable habitats for coastal bird species by providing nesting sites for shorebirds who inhabit this forest. This finding is in agreement with previous studies. The Anatidae (Safe'i et al. 2021), Phalacrocoracidae (Da Silva et al. 2018), Scolopacidae (Desmawati et al. 2017; Sholihah 2017), and Anhingidae (Narayanan et al. 2012) known to have preferences for occupying the mangrove covers used as a nest and perch located on the mangrove forest's edge.

Benthic infauna and shorebird community coexistence

Figure 6 shows the coexistences and availabilities of benthic infauna to shorebird community based on K-means clustering and AIC values (Table 4) to show the correlation. Based on clustering analysis, there are 3 clusters. The first cluster consists of Ciconiidae and Anhingidae clustered together with gastropods with AIC values of 10.49-13.63. The second cluster consists of Anatidae, Scolopacidae, and Phalacrocoracidae clustered together with oligochaetes and polychaetes with AIC values of -4.36-10.49 indicating significant dependencies and correlations between the shorebird and benthic infauna. Finally, the third cluster only consists of crustaceans. From this cluster, gastropod was available to Ciconiidae and Anhingidae. Meanwhile, oligochaetes and polychaetes were available to Anatidae, Scolopacidae, and Phalacrocoracidae. In contrast, crustacea were not available to any shorebird communities.

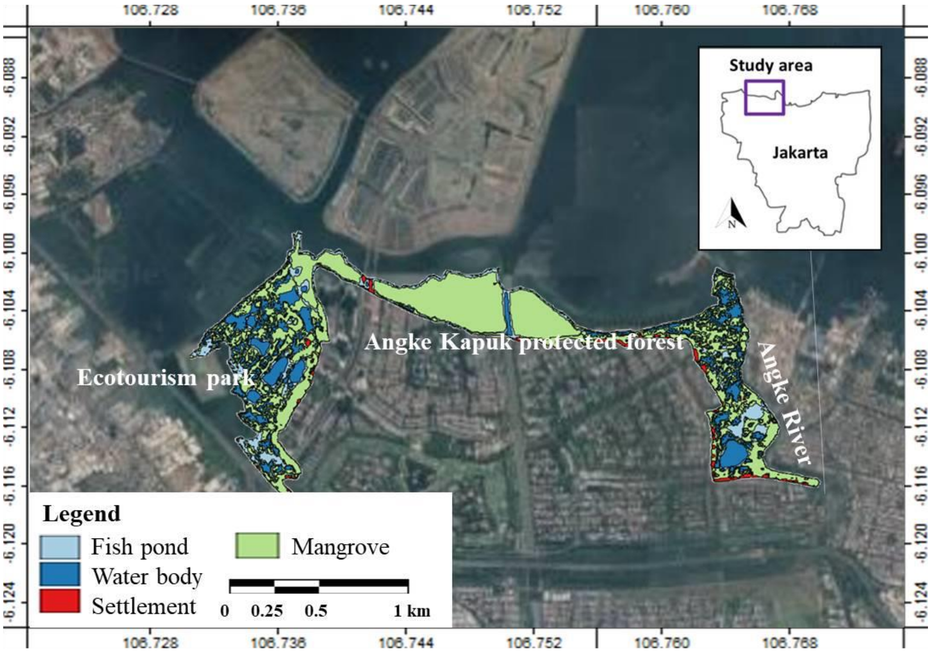

Figure 2. A map of the study area shows mangrove and land covers in the wetlands of the Jakarta Coast, Indonesia

Table 2. Average number and density (inds/100 m^2) of benthic infaunal species presences in each location (Jakarta Coast, Indonesia)

Benthic infaunal presences	Epk	Wpf	Epf	Total	Average (95%CI)
Polychaeta	150	400	150	700	233.33 (70.3, 396)
Oligochaeta	30	900	200	1130	376.66 (-145, 899)
Gastropod	200	180	200	580	193.33 (180, 206)
Crustacea	10	0	0	10	3.33 (-3.2, 9.86)

Note: Epk (Ecotourism park), Wpf (West parts of Angke Kapuk protected forest), and Epf (East parts of Angke Kapuk protected forest)

Table 3. Average number and density (inds/Ha) of shorebird species presences in each location and compared to other studies

Family	Species	Epk	Wpf	Epf	Total	Average (95%CI)	Other studies
Anatidae	*Anas gibberifrons*	1.72	4.68	3.12	9.52	3.17 (1.49, 4.85)	2.81 (Sugiarti et al. 2019)
Scolopacidae	*Actitis hypoleucos*	16.11	43.84	29.23	89.18	29.73 (14.0, 45.4)	0.22 (Martins et al. 2019)
	Tringa glareola	1.34	2.93	1.62	5.89	1.96 (1.00, 2.92)	1.65 (Fajrin et al. 2019)
Ciconiidae	*Mycteria cinerea*	1.78	2.17	1.43	5.38	1.79 (1.37, 2.21)	1.0 (Ronny et al. 2017)
Anhingidae	*Anhinga melanogaster*	2.37	3.56	1.30	7.23	2.41 (1.14, 3.68)	3.46 (Narayanan et al. 2012)
Phalacrocoracidae	*Phalacrocorax sulcirostris*	2.58	7.02	4.68	14.28	4.76 (2.25, 7.27)	

Note: Epk (Ecotourism park), Wpf (West parts of Angke Kapuk protected forest), and Epf (East parts of Angke Kapuk protected forest)

Table 4. Correlation and dependence models of shorebird species with benthic infaunal communities based on Akaike Information Criterion (AIC) values

Variables	Family	AICc	ΔAIC	AIC weight	AIC cum.	Log likehood
Anas gibberifrons ~ benthic infaunal species	Anatidae	10.49^{b}	14.85	0	1	-14.24
Actitis hypoleucos ~ benthic infaunal species	Scolopacidae	10.49^{b}	14.85	0	1	-14.25
Tringa glareola ~ benthic infaunal species	Scolopacidae	$-4.36^{a,b}$	0.00	1	1	-6.82
Mycteria cinerea ~ benthic infaunal species	Ciconiidae	13.63	17.99	0	1	-15.81
Anhinga melanogaster ~ benthic infaunal species	Anhingidae	10.49^{b}	14.85	0	1	-14.24
Phalacrocorax sulcirostris ~ benthic infaunal species	Phalacrocoracidae	10.49^{b}	14.85	0	1	-14.24

Note: ^{a}best model, ^{b}significant correlation

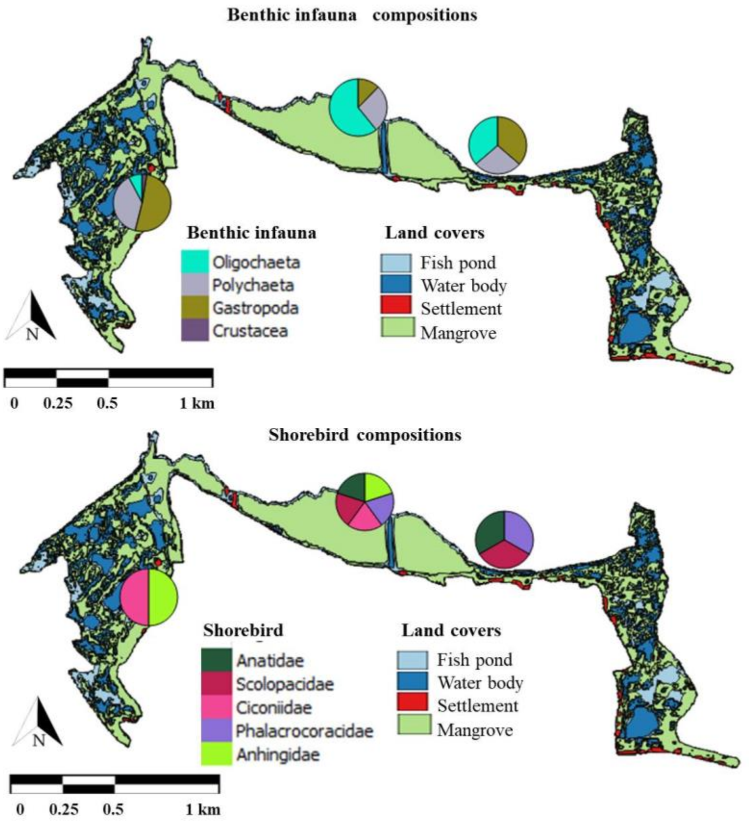

Figure 3. A map of the study area showing benthic infaunal and shorebird compositions in mangrove and land covers in the wetlands of Jakarta Coast, Indonesia

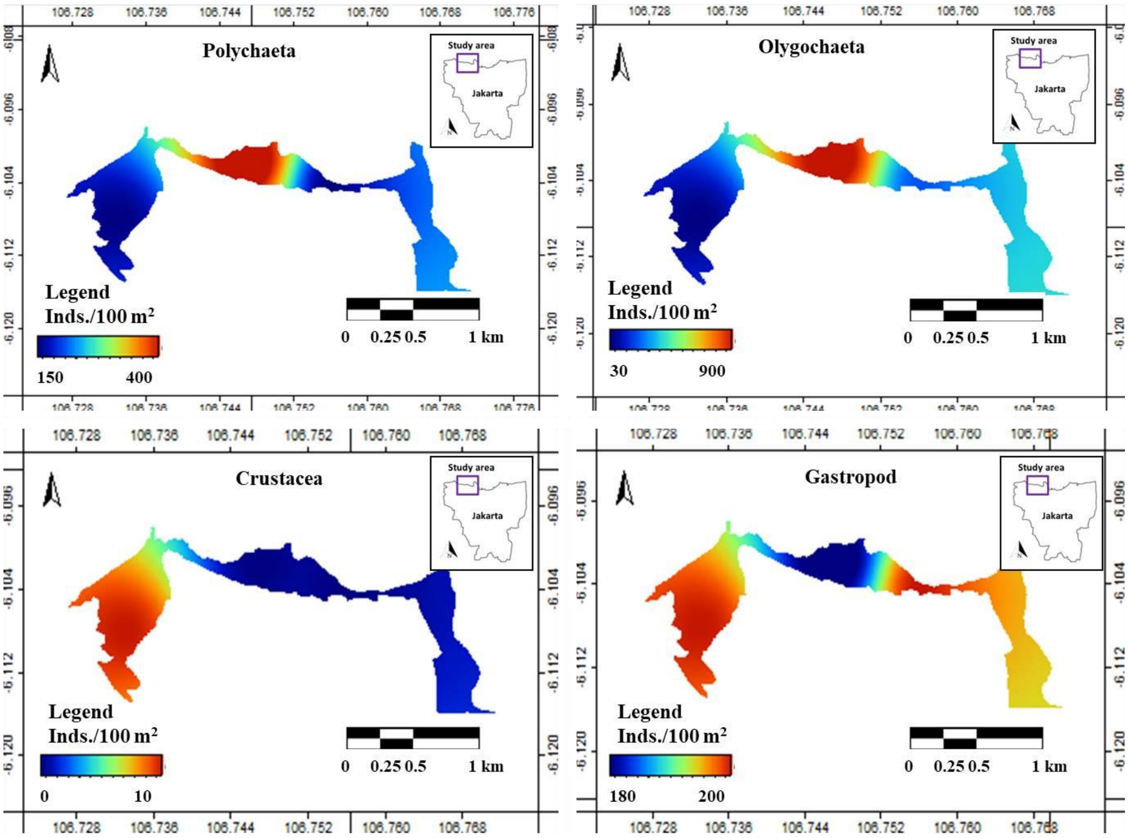

Figure 4. A map of the study area showing the density (inds/100 m^2) of benthic infaunal species presences in mangrove and land covers in the wetlands of Jakarta Coast, Indonesia

Discussion

This study identified significant relationships between shorebird and benthic infaunal communities, as reported before (Mouronval et al. 2007). The relationship was based on and influenced by the benthic infaunal availability, considering the benthic infauna is a food source for shorebirds. This prey availability explains the varied spatial distributions of shorebirds in the wetlands of the Jakarta Coast. The presence of shorebirds indicated the benthic infaunal availability since the presence of shorebirds followed the presence of benthic infaunal species. The shorebird species of Anatidae, Scolopacidae and Phalacrocoracidae were widely distributed in the west and east parts of the wetlands on the Jakarta Coast due to the availability of benthic infauna. In those parts of the wetlands, the west contains oligochaetes and polychaetes, and the east contains gastropods. In contrast, those shorebirds were not distributed to the ecotourism park despite the availability of gastropods. Shorebird distributions following the distributions of oligochaetes and polychaetes were in agreement with previous studies. In the wetlands of Aconcagua Estuary, visual foraging and tactile shorebirds, including American oystercatcher (*Haematopus palliatus* Temminck, 1820), Hudsonian godwit (*Limosa haemastica* Linnaeus, 1758), Lesser yellowlegs (*Tringa flavipes* Gmelin, 1789), Greater yellowlegs (*Tringa melanoleuca* Gmelin, 1789), Whimbrel (*Numenius phaeopus* Linnaeus, 1758), and Black-necked stilt (*Himantopus mexicanus* Statius Muller, 1776) foraged in wetlands with a high abundance of nereid polychaetes and oligochaetes, such that all of that benthic infauna were registered as being in the diet of those shorebirds (Pérez-Vargas et al. 2016).

The strong association between benthic infauna as prey and feeding shorebirds as predators is predictable. Shorebirds exhibit great dietary flexibility and combine specific prey selection and the opportunity to select the most abundant prey locally available. In this study, oligochaetes followed by polychaetes were very abundant compared to other benthic infaunal communities. Despite the abundance of gastropods in ecotourism parks and the east parts of protected forests, Anatidae, Scolopacidae, and Phalacrocoracidae abundances were low in those study areas. Low abundances of certain shorebird communities in some locations are related to the distance to the roost as a determinant factor (Bakker et al. 2021). Foraging patterns of shorebirds can be varied amongst wetlands in terms of the distance traveled between roosting and foraging sites. This explains the low abundance of Anatidae, Scolopacidae, and Phalacrocoracidae in the ecotourism park area since the distance between this area and the protected forest was more than 1 km.

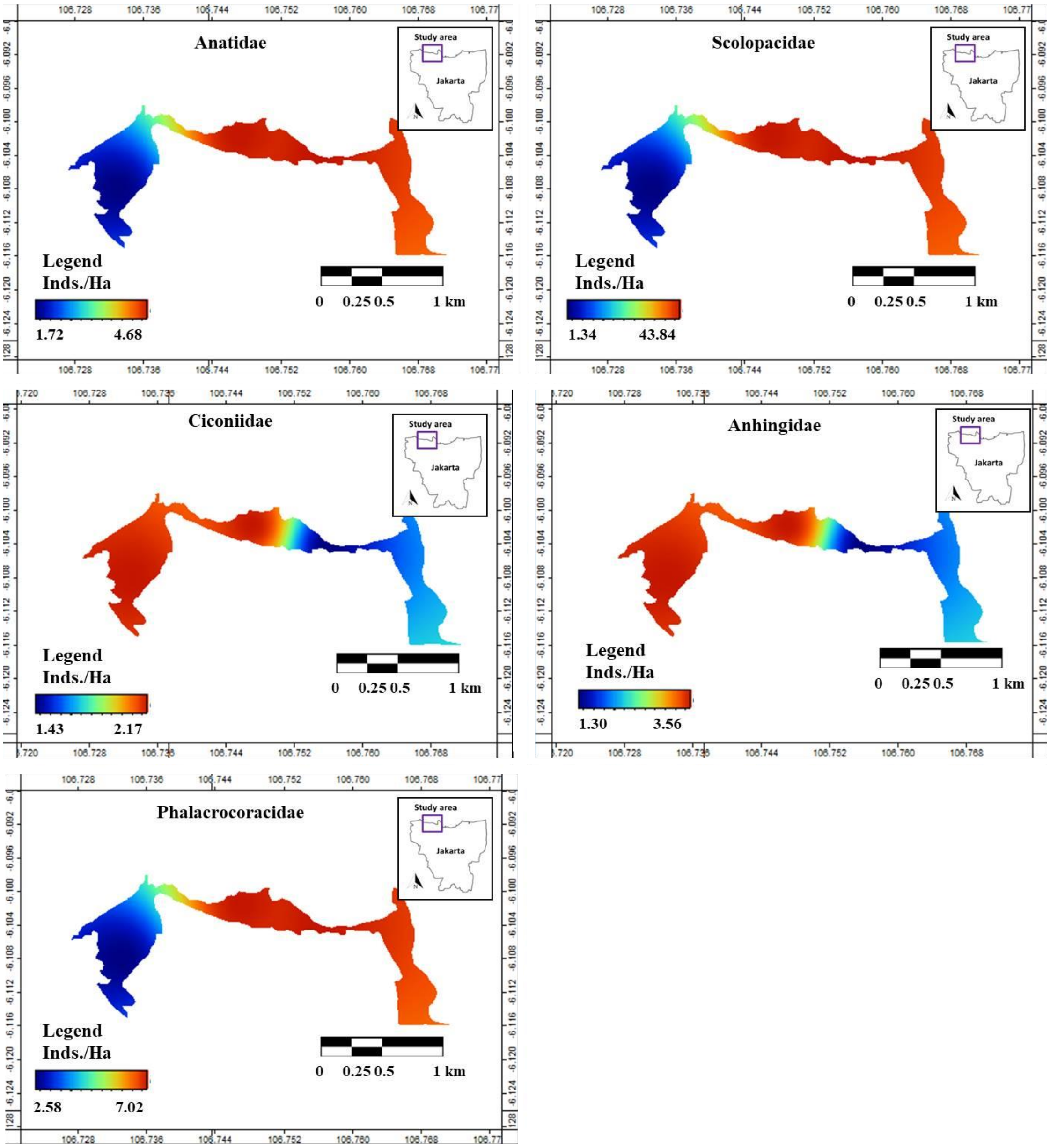

Figure 5. A map of the study area showing the density (inds/Ha) of shorebird species presences in mangrove and land covers in the wetlands of Jakarta Coast, Indonesia

The presence and abundances of shorebird communities belonging to the families Ciconiidae and Anhingidae in the ecotourism park were more closely related to their morphology (Alonso et al. 2011) which suited the conditions of the ecotourism park. Because of this adaptation, these shorebirds can forage in flooded ecosystems. In this study, an ecotourism park is characterized by the dominance of water bodies and fish ponds (Mulyani et al. 2021). Then, morphological characteristics of Ciconiidae and Anhingidae were adapted to the environmental conditions of ecotourism park area, and this may explain the presence of these shorebirds in this particular study area, besides the abundance of gastropods.

It can be concluded that based on the spatial analysis, K-Means cluster, and AIC analyses, the coexistence of shorebird communities in the wetlands of the Jakarta Coast depends on the availability of benthic infauna as prey, and the prey availability is associated with the quality of wetlands. Therefore, wetlands dominated by dense mangroves will lead to an abundance of benthic infauna and influence the presence and distribution of shorebirds. At present, oligochaetes followed by polychaetes are considered the most important food resources for shorebirds considering that this benthic infauna is available in large amounts compared to gastropods and crustaceans. Based on the benthic abundances and aiming to conserve shorebirds, it is recommended to conserve the mangrove forest, considering this forest is an important resource for benthic infaunal communities in the wetlands of the Jakarta Coast, Indonesia.

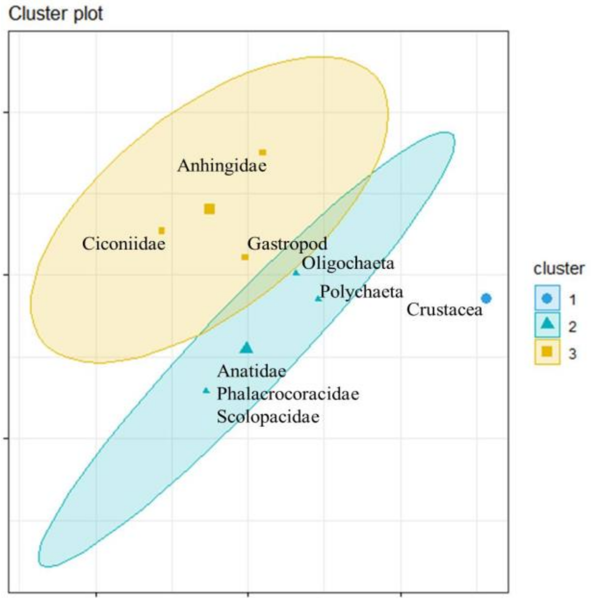

Figure 6. Availability and coexistence of benthic infaunal (Crustacea, Gastropod, Oligochaeta, Polychaeta) with shorebird (Anatidae, Scolopacidae, Ciconiidae, Phalacrocoracidae, Anhingidae) communities in the wetlands of Jakarta Coast based on K-means clustering analysis

REFERENCES

Alevizos E. 2016. Introduction to SAGA GIS. DOI: 10.13140/RG.2.2.31487.87206.

Alonso J, Alonso J, Carrascal L. 2011. Habitat selection by foraging white storks, *Ciconia ciconia*, during the breeding season. Canad J Zool 69: 1957-1962. DOI: 10.1139/z91-270.

Amarasekara EAKK, Jayasiri HB, Amarasiri C. 2021. Avifaunal diversity in urban coastal wetland of Colombo Sri Lanka. Open Access Libr 8: 1-10. DOI: 10.4236/oalib.1107256.

Andriyono S, Nindarwi DD, Kenconojatia H, Budia DS, Azhar MH. 2016. Dominance and diversity of seagrass and macrozoobenthos on transition season in Bama Beach, Baluran National Park, Situbondo. J Ilmiah Perikanan dan Kelautan 8 (1): 36. DOI: 10.20473/jipk.v8i1.11191.

Bakker W, Ens B, Dokter A, van der Kolk H, Rappoldt K, van de Pol M, Troost K. 2021. Connecting foraging and roosting areas reveals how food stocks explain shorebird numbers. Estuar Coast Shelf Sci 259: 107458. DOI: 10.1016/j.ecss.2021.107458.

Basyuni M, Gultom K, Fitri A, Susetya EA, Wati R, Slamet B, Sulistiyono N, Yusriani E, Balke T, Bunting P. 2018. Diversity and habitat characteristics of macrozoobenthos in the mangrove forest of Lubuk Kertang Village, North Sumatra, Indonesia. Biodiversitas 19: 311-317. DOI: 10.13057/biodiv/d190142.

Buda K, Budka M. 2019. Bird species detection by an observer and an autonomous sound recorder in two different environments: Forest and farmland. PLoS One 14: e0211970. DOI: 10.1371/journal.pone.0211970.

Chapman VJ. 2016. Coastal Vegetation. Elsevier, New York.

Da Silva TL, Cabral RBG, Ferreira I. 2018. Behavior and seasonal abundance of neotropic cormorant *Nannopterum brasilianus* (Gmelin, 1789) in southeastern, Brazil. Rev Bras Ornitol 26: 219-226. DOI: 10.1007/BF03544434.

D'Andrea R, Riolo M, Ostling AM. 2019. Generalizing clusters of similar species as a signature of coexistence under competition. PLoS Comput Biol 15 (1): e1006688. DOI: /10.1371/journal.pcbi.1006688.

De Oliveira A, Rizzo A, Da Conceicao E, Couto G. 2011. Benthic macrofauna associated with decomposition of leaves in a mangrove forest in Ilhé us, State of Bahia, Brazil. J Mar Biol Assoc U.K 92: 1-9. DOI: 10.1017/S0025315411001482.

Desmawati I, Trisnawati I, Kurnia O, Hamdani A, Fahmi M, Fahmi M. 2017. Distribution study on migratory bird (Scolopacidae: *Numenius*) in Surabaya, Indonesia: Estimating the effect of habitat and climate change. AIP Conf Proc 1854 (1): 020008. DOI: 10.1063/1.4985399.

Fajrin A, Halang B, Mahrudin. 2019. Species and density of trinil birds (*Tringa* sp.) at the Village Area of Sungai Rasau, Tanah Laut Regency as a handout material in animal ecological enrichment. Prosiding Seminar Nasional Lingkungan Lahan Basah 4 (3): 516-522.

Gaget E, Le Viol I, Pavón-Jordán D, Cazalis V, Kerbiriou C, Jiguet F, Popoff N, Dami L, Mondain-Monval J, Defos Du Rau P, Abdou W, Bozic L, Dakki M, EncarnaçãoV, Erciyas Yavuz K, Etayeb K, Molina B, Petkov N, Uzunova D, Galewski T. 2020. Assessing the effectiveness of the Ramsar Convention in preserving wintering waterbirds in the Mediterranean. Biol Conserv 243: 108485. DOI: 10.1016/j.biocon.2020.108485.

Hirsch P. 2016. Routledge Handbook of the Environment in Southeast Asia. Taylor & Francis, London. DOI: 10.4324/9781315474892.

Hutabarat AA, Yulianda F, Fahrudin A, Harteti Kusharjani S. 2009. Pengelolaan Pesisir dan Laut Secara Terpadu Bogor. PUSDIKLAT Kehutanan.Departemen Kehutanan RI SECEM-Korea International Cooperation Agency. [Indonesian]

Isworo S, Oetari PS. 2020. Mangrove vegetation and bird communities around Tegal Port, Central Java, Indonesia. Biodiversitas 21: 1551-1560. DOI: 10.13057/biodiv/d210436.

Katili AS, Utina R. 2019. Composition and abundance of Crustacea and Polychaeta in mangrove stands at Bulalo Kwandang District North Gorontalo Regency. Jambura Edu Biosfer J 1 (1): 32-40. DOI: 10.34312/jebj.v1i1.2044. [Indonesian]

Koopman K, Collas F, Van der Velde G, Verberk W. 2016. Oxygen can limit heat tolerance in freshwater gastropods: differences between gill and lung breathers. Hydrobiologia 763: 301-312. DOI: 10.1007/s10750-015-2386-y.

Liu W, Tanimura A, Imai T, Kanaya G, Niiyama T, Maegawa S, Kohzu A, Kimura T, Toyohara H. 2014. Distribution of gastropods in a tidal flat in association with digestive enzyme activities. Plankton Benthos Res 9: 156-167. DOI: 10.3800/pbr.9.156.

MacKinnon JR, Phillipps K. 1993. A Field Guide to the Birds of Borneo, Sumatra, Java, and Bali, the Greater Sunda Islands. Oxford University Press, UK.

Martins CO, Zakaria M, Olaniyi E, Angela UO. 2019. Population density of avian species in a man-made wetland of Peninsular Malaysia. IOP Conf Ser: Earth Environ Sci 269. DOI: 10.1088/1755-1315/269/1/012030.

Metcalfe K, Glasby C. 2007. Diversity of Polychaeta (Annelida) and other worm taxa in mangrove habitats of Darwin Harbour, northern Australia. J Sea Res 59: 708-82. DOI: 10.1016/j.seares.2007.06.002.

Mouronval J, Guillemain M, Canny A, Poirier F. 2007. Diet of non-breeding wildfowl Anatidae and Coot *Fulica atra* on The Perthois Gravel Pits, Northeast France. Widlfowl 57 (57): 68-97.

Mulyani Y, Putera A, Perwitasari-Farajallah D, Lhota S, Herliansyah R, Sodikin. 2021. Waterbird foraging habitat selection in Balikpapan Bay: water depth and patch area as important factors. Hayati 28: 312-324. DOI: 10.4308/hjb.28.4.312-324.

Murdiyarso D, Purbopuspito J, Kauffman J, Warren M, Sasmito S, Donato D, Manuri S, Krisnawati H, Taberima S, Kurnianto S. 2015. The potential of Indonesian mangrove forests for global climate change mitigation. Nat Clim Chang 5: 1089-1092. DOI: 10.1038/nclimate2734.

Narayanan S. Sreekumar B, Vijayan L, Thomas AP. 2012. Status of the nests of near-threatened oriental darter *Anhinga melanogaster* at Kumarakom Heronry. Natl Acad Sci Lett 35: 99-101. DOI: 10.1007/s40009-012-0027-9.

Obunga G, Siljander M, Maghenda M, Pellikka PKE. 2022. Habitat suitability modelling to improve conservation status of two critically endangered endemic Afromontane forest bird species in Taita Hills, Kenya. J Nat Conserv 65: 126111. DOI: 10.1016/j.jnc.2021.126111.

Orimoloye I, Mazinyo S, Kalumba A, Nel W, Ibraheem A, Ololade O. 2019. Wetland shift monitoring using remote sensing and GIS techniques: Landscape dynamics and its implications on Isimangaliso Wetland Park, South Africa. Earth Sci Inform 12: 553-563. DOI: 10.1007/s12145-019-00400-4.

Pérez-Vargas AD, Bernal M, Delgadillo CS, González-Navarro EF, Landaeta MF. 2016. Benthic food distribution as a predictor of the spatial distribution for shorebirds in a wetland of Central Chile. Revista De Biologia Marina Y Oceanografia 51: 147-159. DOI: 10.4067/S0718-19572016000100014.

Philiani I, Saputra L, Harvianto L. Muzaki AA. 2016. Pemetaan vegetasi hutan mangrove menggunakan metode Normalized Difference Vegetation Index (NDVI) di Desa Arakan, Minahasa Selatan, Sulawesi Utara. SOIJST 1 (2): 211-222. DOI: 10.31219/osf.io/c8k6j. [Indonesian]

Prasetyo E, Wulandari R. 2021. Richness, diversity, and conservation status of bird species in Maron Beach, Semarang, Indonesia. Quagga: J Pendidikan dan Biologi 13: 95. DOI: 10.25134/quagga.v13i1.3664. [Indonesian]

Rabalais NN, Baustian MM. 2020. Historical shifts in benthic infaunal diversity in the Northern Gulf of Mexico since the appearance of seasonally severe hypoxia. Diversity 12 (2): 49. DOI: 10.3390/d12020049.

Rija A, Mgelwa A, Modest R, Hassan S. 2015. Composition and functional diversity in bird communities in a protected humid coastal savanna. Adv Zool 2015: 1-7. DOI: 10.1155/2015/864219.

Ronny A, Gunawan H, Yoza D. 2017. Penentuan tingkat kepadatan dan sebaran populasi Bangau Bluwok (*Mycteria cinerea*) menggunakan drone di Pulau Basu, Indragiri Hilir. J Riau Biologia 2 (2): 81-89. [Indonesian]

Safe'i R, Sari RN, Iswandaru D, Latumahina FS, Taskirawati I, Kaskoyo H. 2021. Biodiversity and site quality as indicators of mangrove forest health Pasir Sakti, Indonesia. Ann Romanian Soc Cell Biol 25 (2): 4400-4410.

Sahidin A, Setyobudiandi I, Wardiatno Y. 2014. Struktur komunitas makrozoobentos di perairan pesisir Tangerang, Banten. Depik 3 (3): 226-233. DOI: 10.13170/depik.3.3.2150. [Indonesian]

Sahidin A, Wardiatno Y. 2016. Spatial distribution of Polychaeta at Tangerang Coastal Water, Banten Province. J Perikanan dan Kelautan 6: 83. DOI: 10.33512/jpk.v6i2.1102. [Indonesian]

Sahidin A. 2020. Buku Fauna Bentik; Studi Kasus Keanekaragaman Bentik di Perairan Tangerang Banten. Graha Ilmu, Yogyakarta. [Indonesian]

Sari WN, Melki, Putri WAE. 2022. Biodiversitas Polychaeta di perairan muara Sungai Musi, Desa Sungsang Sumatera Selatan. Maspari J 14 (1): 49-61. DOI: 10.36706/maspari.v14i1.16701. [Indonesian]

Sasongko DA, Kusmana C, Ramadan H. 2014. Management strategy of Angke Kapuk protected forest. J Pengelolaan Sumberdaya Alam dan Lingkungan 4 (1): 35-42. DOI: 10.19081/jpsl.2014.4.1.35. [Indonesian]

Sholihah R. 2017. Behavior and ecological study of marine birds in Alas Purwo National Park Southern Cost—Indonesia. KnE Life Sciences: 2017: 177-185. DOI 10.18502/kls.v3i4.702.

Sofian A, Harahab N, Marsoedi. 2012. Kondisi dan manfaat langsung ekosistem mangrove Desa Penunggul Kecamatan Nguling Kabupaten Pasuruan. J Biol El-Hayah 2 (2): 56-63. DOI: 10.18860/elha.v2i2.2208. [Indonesian]

Sofian A, Kusmana C, Fauzi A, Rusdiana O. 2019. Evaluasi kondisi ekosistem mangrove Angke Kapuk Teluk Jakarta dan konsekuensinya terhadap jasa ekosistem. J Kelautan Nasional 15: 1-12. DOI: 10.15578/jkn.v15i1.7722. [Indonesian]

Sugiarti S, Kaspul, Mahrudin. 2019. Kerapatan populasi itik benjut (*Anas gibberifrons*) di Desa Sungai Rasau, Kabupaten Tanah Laut sebagai bahan handout pengayaan mata kuliah Ekologi Hewan. Prosiding Seminar Nasional Lingkungan Lahan Basah 4 (3): 598-602. [Indonesian]

Sukojo BM, Arindi YN. 2019. Analisa perubahan kerapatan mangrove berdasarkan nilai Normalized Difference Vegetation Index menggunakan Citra Landsat 8 (Studi Kasus: Pesisir Utara Surabaya). Geoid 14 (2): 1-5. DOI: 10.12962/j24423998.v14i2.3874. [Indonesian]

Tomlinson PB. 2016. The Botany of Mangroves. 2nd ed. Cambridge University Press, Cambridge, UK.

Utami S, Anggoro S, Soeprobowati T. 2017. Bird Species biodiversity in coastal area of Panjang Island, Jepara, Central Java. Adv Sci Lett 23: 2498-2500. DOI: 10.1166/asl.2017.8655.

Wiraatmaja MF, Hasanah R, Dwirani NM, Pratiwi AS, Riani FE, Hasnaningtyas S, Nugroho GD, Setyawan AD. 2022. Structure and composition molluscs (bivalves and gastropods) in mangrove ecosystem of Pacitan District, East Java, Indonesia. Intl J Bonorowo Wetlands 12: 1-11. DOI: 10.13057/bonorowo/w120101.

Zhao Q, Basher Z, Costello M. 2019. Mapping near surface global marine ecosystems through cluster analysis of environmental data. Ecol Res 35: 1-16. DOI: 10.1111/1440-1703.12060.

INTL J BONOROWO WETLANDS
Volume 12, Number 1, June 2022
Pages: 33-40

P-ISSN: 2775-8052
E-ISSN: 2775-8044
DOI: 10.13057/bonorowo/w120104

Relationship between physicochemical parameters and the abundance of zooplankton in Lake Mweru-Wantipa, Zambia

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Manuscript received: 21 April 2022. Revision accepted: 19 June 2022.

Abstract. *Anamunda A, Lamtane HA. 2022. Relationship between physicochemical parameters and the abundance of zooplankton in Lake Mweru-Wantipa, Zambia. Intl J Bonorowo Wetlands 12: 33-40.* This study aimed to assess the abundance of zooplankton and their relationship with physicochemical parameters within Lake Mweru-Wantipa, Zambia. A National Park borders the lake on one side and settlements on the other, so four sampling stations were selected, two on each side of the lake. Five physicochemical parameters, including temperature, turbidity, salinity, pH, and dissolved oxygen, were measured monthly using portable instruments concurrently with a collection of duplet zooplankton samples at each sampling point at depths between 0.1m and 0.5m. All physicochemical parameters were significantly different between the two sites ($p<0.05$) apart from temperature and pH. A total of 13 genera of zooplankton were identified, belonging to four groups, namely, rotifers, cladoceran, copepods, and ostracods. The cladoceran had the highest number of species (6), followed by copepods (4) in both sites. However, the copepods had the largest contribution in abundance in both sites. The diversity was high in settlement areas, but the National Park had higher species richness. There was a significant difference in species diversity between the two sites ($t=3.96$; $p=0.001$). The most abundant group was the cyclopoid on both sides of the lake, followed by the *Moina* on the settlement site and the *Daphnia* sp. on the National Park site. The densities of *Molina*, *Simocephalus*, *Ceriodaphinia,* and *Cypris* were significantly different between the two sites ($p<0.05$). Except for copepods, all groups were significantly different between the two sites ($p<0.05$). Generally, the total zooplankton density was not significantly different between the two sites ($t=0.73$; $p=0.06$). The results showed that the zooplankton abundance was influenced by turbidity in settlement areas and pH in National park areas.

Keywords: Cladoceran, Mweru-Wantipa, physicochemical, zooplankton

INTRODUCTION

Zooplankton consists of macro and microscopic animals and is an important food item for other aquatic animals in the higher trophic levels (Takarina et al. 2019). Zooplankton is an important water quality indicator due to their shorter life spans and different tolerance levels towards physicochemical parameters (Gajbhiye 2002). Research has shown that zooplankton species have different tolerance limits toward physicochemical parameters. Balakrishna et al. (2013) reported changes in zooplankton species densities affected by changes in physicochemical parameters across different seasons. According to Waikato Environmental Technical Report (2008) in New Zealand, the presence of rotifers can be used to grade the eutrophic status of the lakes.

Understanding zooplankton's variability patterns, both temporally and spatially, provides information on the processes affecting them. Physicochemical parameters have been reported as one of the sources of variation in species composition, abundance, diversity, and distribution of zooplankton. For example, Imaobong (2013) reported in Nigerian lakes, levels of eutrophication determined zooplankton species abundance and distribution. Variations in seasonal abundance and diversity because of changes in physicochemical parameters were also reported by Kedar et al. (2008). Similar studies have been conducted on the relationship between zooplankton and physicochemical parameters in India and Nigeria (Goswami and Mankodi 2012).

Lake Mweru-Wantipa is a small lake found in Zambia. The lake is swampy and muddy; it has no water outlet, although a few small streams flow into it. The lake's boundary lies entirely within the Mweru-Wantipa National Park. Over the past decade, encroachments have been tolerated and have increased to a level where many permanent settlements on the eastern sides of the lake are now established. The lake is the main fish supplier to the residents of the Nsama and Kaputa Districts. Apart from fishing, the settlers diversified into other activities such as agriculture, logging, and charcoal production within the immediate catchment of the lake. Despite these stressors developing around it, the lake has received very little attention from researchers. Lake Mweru-Wantipa has displayed a series of fluctuations in water levels in the past, which variations in rainfall levels have not explained. The lake has also been known to have dried out almost completely in 1916 (Brelsford 1954). The only known study on Lake Mweru-Wantipa was on the biology and exploitation of small pelagic fishes by Mubamba (1989). There is limited published information on the zooplankton abundance and water quality of Lake Mweru-Wantipa.

The two districts of Nsama and Kaputa, that share Lake Mweru-Wantipa, are boarded by Nsumbu and Mweru-

Wantipa National Parks. As Tsetse fly infestations make the rearing of larger livestock impossible, Lake Mweru-Wantipa is the main supplier of animal protein, in the form of fish, to these districts. However, the growing population around the lake has led to overfishing and increasingly large areas close to the lake being opened for agricultural purposes, leading to deforestation in the lake's catchment area. These activities have increased siltation in the lake, as evidenced by the poor transparency of the lake itself. Studies on the relationship between zooplankton and physicochemical parameters have been done in nearby lakes (Nkotagu and Athuman 2007), but there is limited information on the zooplankton of Mweru-Wantipa.

This study aimed to determine the relationship between the physicochemical parameters and zooplankton abundance in Mweru-Wantipa Lake, Zambia. A zooplankton study and its relationship to physicochemical parameters will provide insight into the current limnological status of Lake Mweru-Wantipa, which has never been done. That will be valuable baseline information for researchers and government agencies interested in the management of the lake.

MATERIALS AND METHODS

Description of the study area

The research was conducted in Lake Mweru-Wantipa, in the Northern Province of Zambia. The lake is located between coordinates 8°10'S to 9°10'S and 29°00'E to 30°00'E. The lake's size is 73 km long and 43 km wide, with an average depth of two meters (Frame Survey Report 2004).

Though the lake has several inlets that bring in water, it has no outlet. The western side is boarded by a National park (Figure 1). The lake is shared by two districts, namely Nsama and Kaputa, whose main sources of livelihood are fishing and agriculture. The area falls within an agroecological zone (III) characterized by a humid subtropical climate with the warmest temperature of about 32°C in October and the coolest temperatures of about 5°C in July (Chabala et al. 2013). The districts normally experience dry and wet seasons; the dry season normally starts in May and runs through October, while the wet season starts from November to April. The region receives an average rainfall of 1000-1500 mm and has strong acidic soils with low nutrients caused by high rainfall (Paul 2008).

Sampling design

Four sampling stations were chosen across the lake's midsection, two on the western side closer to the National Park and two on the eastern side closer to the human settlements. The sampling points were identified and marked using a geographical positioning system (GPS).

Table 1 shows the coordinates of the sampling stations and the depths of the water columns, respectively. The water samples were collected in the first week of the three sampling months, in the morning between 08hrs and 12hrs. The physicochemical parameters were measured at sampling sites concurrently with a collection of zooplankton water samples below the water surface. The first sampling occurred in the first week of November 2013, and the last was during the first week of January 2014.

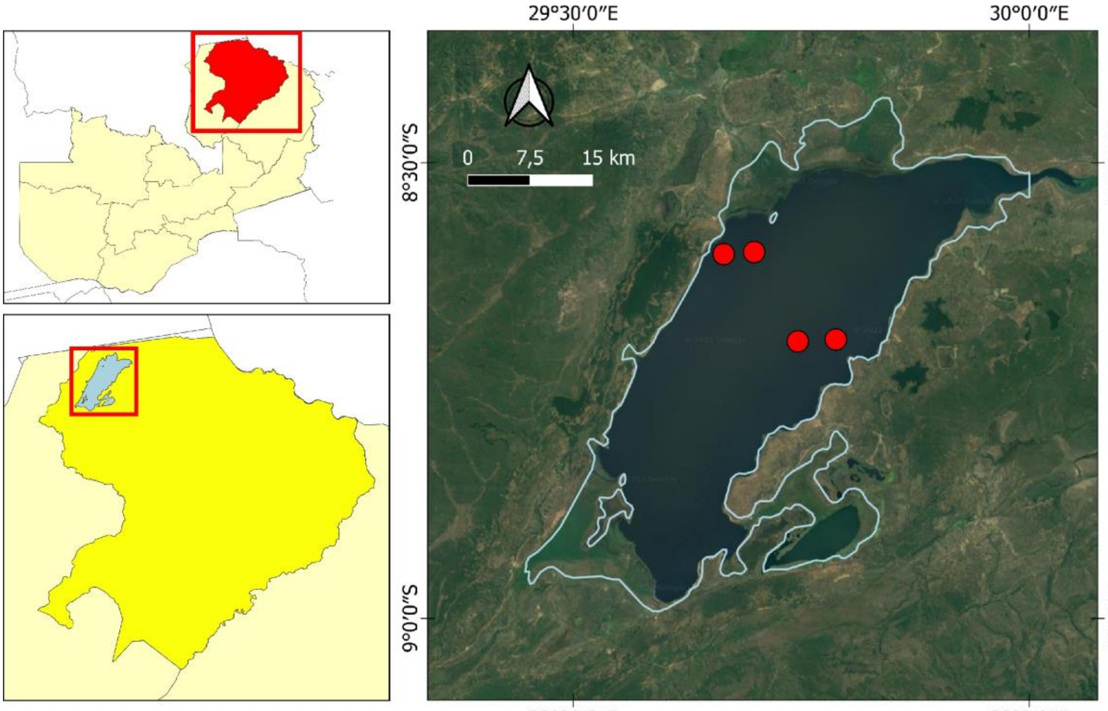

Figure 1. The map of Lake Mweru-Wantipa, Zambia, and sampling points

Table 1. Zooplankton sampling points on the Lake Mweru-Wantipa, Zambia

	National	**Park**	**Settlement**	**Areas**
Sampling stations	1	2	3	4
GPS coordinates	08°36'0" S, 029°40'04.1" E	08°37'54.3" S, 029°42'03.7" E	08°40'19.8" S, 029°44'37.3" E	08°41'23.0" S, 029°47'03.3" E
Water depth	0.64m	0.95m	2.0m	2.3m

Data collection

Zooplankton sampling

A graduated 10 L bucket with a mouth diameter of 20 cm was used to sample zooplankton below the water surface at depths between 0.1 and 0.5 m. Each time, one bucket scope was taken vertically and filtered through a 100 µm zooplankton net. Samples were collected in duplicates. The zooplankton samples were stored in 100ml plastic bottles with 4% formaldehyde. The bottles were kept in cooler boxes and then transported to the laboratory of the Department of Fisheries in Mpulungu, Zambia.

Water quality parameters

Five physicochemical parameters were measured during sampling using portable electronic instruments. Dissolved oxygen (DO) and salinity were measured using YSI Model 54 ABP, model 54 ARC, and Salinometer HI 8033, respectively. Turbidity was measured with a Hach turbidity meter model 2100A with precision + Nephelometric Turbidity Units (NTU). pH was measured using a pH meter (WTH 323), while the temperature was measured using a Hanna temperature probe (HI 9143).

Zooplankton laboratory analysis

A Labovert FS microscope with a magnification of x40 was used to examine the zooplankton. Zooplankton was identified up to a generic level using identification guides by Utzugi and Mazingaliwa (2002). Standard Operating Procedure for Zooplankton Analysis guidelines by the United States Environmental Protection Agency (USEPA 2010) was used to analyze the zooplankton in the laboratory. Samples were thoroughly mixed, and a 1 mL subsample was withdrawn with a pipette. One ml subsamples that had fewer organisms were discarded until consistently high zooplankton numbers were achieved in the subsamples. A zooplankton counting chamber was used for counting the identified zooplankton species.

Data processing and analysis

A Student *t*-test was used to determine the difference between the means of the zooplankton density and physicochemical parameters on the two sides of the lake. The biodiversity index used to determine the species diversity of zooplankton was Shannon-Weaver (1949) in Spellerberg and Fedor (2003). The Hutcheson (1970) *t*-test was used to determine the difference between the diversities between the two sides of the lake. Pearson's correlation (r) analysis was used to investigate whether there was a relationship between physicochemical parameters to zooplankton abundance. Finally, a multiple regression analysis was used to investigate the cause and effect between the zooplankton abundance physicochemical parameters. The regression model used was $Y_i = \beta_o + \beta_1X_1 + \beta_2X_2 + \beta_3X_3 + \beta_4X_4 + \beta_5X_5 + e_i$

Species diversity

Species diversity was calculated using the Shannon-Weaver Index (1949). The following formula was used;

$H' = -\sum (Pi*In (Pi))$ where $Pi = ni/N$ In = the natural log

Pi = Proportion of total sample belonging to the i^{th} specie ni = total number of individuals in a specie

N = total number of individuals

E = *H'*/*H*max where *H*max = In(S) measures the species evenness

(S) = the total number of distinct individual peaks within a profile (Species richness)

E = measures the species evenness

*H*max = measures the maximum evenness the community can have; the closer this is to one, it means the community is optimally even.

*E*exp *H* = Effective Number of Species

Density estimation

The population density was calculated from known densities using the following formula by Tonapi (1980).

Density = (n) (v) / V Where;

Density = Total no. of organization/liter of water filtered

n = Average number of organisms in a 1 mL plankton sample v = Volume of concentrate plankton sample (ml)

V = Volume of total water filtered through (L)

RESULTS AND DISCUSSION

Physicochemical parameters

Temperature

There was a slight difference in temperature between the two sampling sites (Table 2). However, there was no significant difference in water temperature between the study sites (t=0.586; *p*=0.559).

Turbidity

The mean and range of turbidity recorded from the settlement side and National Park are presented in Table 2. The highest turbidity value of 2.7 NTU was recorded on the settlement side, while the lowest value of 0.4 NTU was recorded on the National Park side. The difference in turbidity between the two sides of the lake was significant (t=-5.61; *p*=0.001).

Water salinity

The lowest (3.5 ppt) and highest (4.6 ppt) salinity were recorded from the settlement and National park sides, respectively. There was a significant difference in salinity between National Park and settlement sides (t=12.569; p=0.001).

pH

The highest pH value (9.9) during the study was recorded from both sides of the lake, while the lowest (9.2) was on the settlement side (Table 2). There was no significant difference in pH values between the two sites (t=-2.91; p=0.06).

Dissolved oxygen

The lowest and highest dissolved oxygen concentration was 1.2 mg/L and 12.1 mg/L measured from the National Park and settlement side of the lake, respectively (Table 2). The oxygen concentrations on the settlement side were significantly higher than on the National Park side (t=-3.66; p=0.001).

Zooplankton species composition and abundance

The present study recorded 13 zooplankton genera from four major groups, cladoceran, copepods, ostracods, and rotifers were recorded during the present study (Table 3). The National Park side of the lake had the highest number of genera compared to the settlement side. On both sides of the lake, the cladoceran had the highest number of individual groups, followed by copepods. Rotifers were absent from the settlement side of the lake. Table 4 shows that the cyclopoids had the highest relative abundance on both sides of the lake, followed by *Moina* on the settlement sides and *Daphnia* on the National Park side. The *Moina* ranked third on the National Park side. The lowest relative abundance was recorded as rotifers.

Species diversity

Shannon Weaver's diversity index indicated a higher diversity of zooplankton on the settlement side. A Hutcheson t-test showed a significant difference in zooplankton species diversity (t=3.96; p=0.001) between the two study sites. The species richness was higher on the National Park sites compared to the settlement sites. The species evenness and the effective number of species were greater in the settlements (Table 5).

The density of zooplankton in Mweru-Wantipa Lake, Zambia

Except for *Moina, Simocephalus, Ceriodaphnia,* and *Cypris*, all other species' densities were not significantly different between the study sites (Table 5). The copepods had the highest densities on both sampling sites, followed by cladoceran. The least density was rotifers recorded from both sites (Figure 2). T-tests showed that the densities of all the groups were significantly different between the sites except copepods (p=0.78). Generally, the settlement had higher zooplankton densities than National Park (Figure 3). However, there was no significant difference in total zooplankton density between the two sites (t=-0.729; p=0.06).

Relationships between zooplankton and physicochemical parameters

Table 6 shows the relationships between the zooplankton and physicochemical parameters. Temperature, salinity, and pH showed a negative relationship with zooplankton abundance. Generally, except for turbidity and pH, there was no significant correlation between zooplankton abundance and physicochemical parameters. Significant relationships existed between total zooplankton and turbidity on the settlement and pH of the National park (Tables 8 and 9).

Table 2. Summary of water quality parameters in Lake Mweru-Wantipa, Zambia

Parameters	National Park		Settlement areas	
	Mean	Range	Mean	Range
Depth (m)	0.8 ± 0.2	0.6-0.1	2.1± 6	2.0-2.3
Temp. (°C)	27.8 ± 3.7^a	19.8-31.4	27.4 ± 0.79^a	26.5-28.8
Turbidity (NTU)	1.4 ± 0.6^a	0.4-2.2	1.95 ± 0.5^b	1.4-2.7
Salinity (ppt)	4.1 ± 0.3^a	3.9-4.6	3.6 ± 0.2^b	3.5-4.0
pH	9.5 ± 0.2^a	9.2-9.9	9.6 ± 0.3^a	9.2-9.9
DO (mg/L)	5.6 ± 3.1^a	1.2-9.4	7.6 ± 2.6^b	4.4-12.1

Note: Means within the same row with different superscript letters differ significantly at p<0.05, DO = dissolved oxygen

Table 3. Zooplankton groups composition in the Lake Mweru-wingtip, Zambia

Taxonomic group	Total no. of taxa		Percentage composition	
	NP	ST	NP	ST
Cladocera	6	5	46	56
Copepoda	4	3	31	33
Rotifer	2	0	15.4	0
Ostracoda	1	1	7.6	11
Total	13	9	100	100

Note: NP = National park, ST = settlement areas

Table 4. Zooplankton composition in percentages of Lake Mweru-Wantipa in Zambia

Taxa	Genus	Settlements	National park
Cladocera	*Moina*	20	12.1
	Daphnia	11.1	10.3
	Simocephalus	2.5	0.4
	Ceriodaphnia	2.5	1
	Crystallina	1.6	1.2
	Chydorus	0	0.2
Copepoda	Cyclopoid	38.8	46.6
	Calanoid	11.8	14
	Ergasilus	4.7	4.3
	Angusilus	0	0.2
Rotifer	*Branchionus*	0	1
	Conochilus	0	0.2
Ostracoda	*Cypris*	6.8	8.4

Table 5. Diversity, species richness, and evenness of zooplankton in Lake Mweru-Wantipa, Zambia

Diversity index parameter	**Sampling stations**	
	National park	**Settlement**
Species diversity (*H*)	1.50	1.76
Species richness	13	9
Species evenness (*E*)	0.58	0.76
Effective number of species	5	6

Note: *H*'= is the Shannon Weaver diversity index

Table 6. The density of zooplankton species in the Lake Mweru-Wantipa, Zambia

Taxa	**Genus**	**Settlement**	**National park**
Cladocera	*Moina*	46.7±16.86^{a}	20.67±15.63^{b}
	Daphnia	25.4±5.55^{a}	28.8±3.51^{a}
	Simocephalus	6.25±6.25^{a}	1.67±1.64^{b}
	Ceriodaphinia	5.84±3.67^{a}	2.08±2.1^{b}
	Crystallina	3.75±2.45^{a}	2.5±2.49^{a}
	Chydorus	-	0.5±0.42
Copepod	Cyclopoid	90.4±12.16^{a}	88.75±21.18^{a}
	Calanoid	27.5±12.06^{a}	27.5±7.1^{a}
	Ergasilus	10.84±5.65^{a}	10±4.08^{a}
	Angusilus	-	0.5±0.42
Rotifer	*Branchionus*	-	2.08±1.63
	Conochilus	-	0.4±0.42
Ostracod	*Cypris*	20.9±7.19^{a}	17.16±7.56^{b}

Note: Means within the same row with different superscript letters differ significantly at p<0.0

Table 7. The Pearson coefficient of correlation of total zooplankton and physicochemical parameters in Lake Mweru-Wantipa, Zambia

Particulars	**Co-efficient of correlation (r)**	**P value**	**Remarks**
Turbidity	0.380	0.000	S
Temperature	-0.046	0.63	NS
pH	-0.327	0.000	S
Dissolved oxygen	0.151	0.115	NS
Salinity	-0.149	0.121	NS

Note: NS: Not Significant, S: Significant

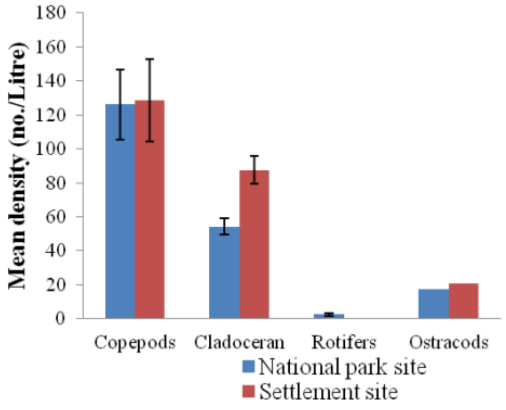

Figure 2. Mean densities of major zooplankton groups from studied sites in Lake Mweru-Wantipa, Zambia

Table 8. Regression analysis for zooplankton and physicochemical parameters on the settlement area in Lake Mweru-Wantipa, Zambia

Predictor	**Coefficients**	**P-value**	**Remark**
Constant	-77.7	0.47	S
Turbidity	11.86	0.04	S
Temperature	0.37	0.81	NS
pH	6.08	0.49	NS
Dissolved oxygen	-0.33	0.51	NS
Salinity	-0.40	0.96	NS

Note: S = Significant, NS = Significant

Table 9. Regression analysis for zooplankton and physicochemical parameters on the National park area in Lake Mweru-Wantipa, Zambia

Predictor	**Coefficient**	**P-value**	**Remark**
Constant	200.70	0.06	NS
Turbidity	2.70	0.64	NS
Temperature	0.89	0.47	NS
pH	-23.59	0.04	S
Dissolved oxygen	0.61	0.57	NS
Salinity	0.08	0.99	NS

Note: S = Significant, NS = Not Significant

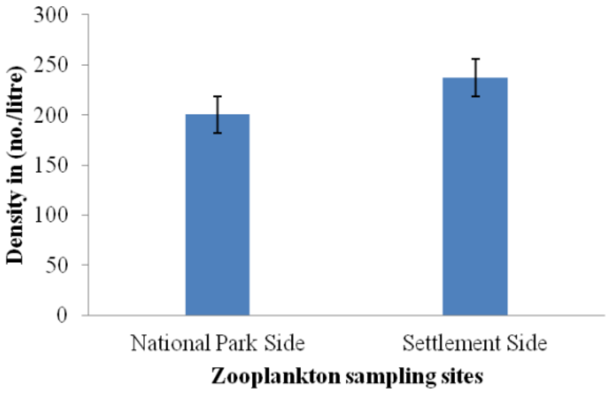

Figure 3. The density of zooplankton in sampling sites in Lake Mweru-Wantipa, Zambia

Discussion

Physicochemical parameters

Lake Mweru-Wantipa is a shallow and small lake bordered by a National Park on the western side, while the lake's eastern, southern, and northern sides are settlement areas. The major economic activities in the settlement areas are agriculture and fishing. The fisheries frame survey report of Lake Mweru-Wantipa by the Zambia Central Statistical Office (Frame Survey Report 2004) showed that there had been an immense increase in the number of fishing villages around the lake. Improper agriculture practices and deforestation lead to land degradation and alteration of the physicochemical parameters of the lake. Yorke and Margai (2007) reported that population growth and developmental activities along water bodies in many sub-Saharan countries have been responsible for negative changes in water quality parameters.

The marginal temperature differences in water between the two studied sides of the lake were probably due to the lake's shallowness (Table 2). The depth of water bodies can potentially affect variations in physicochemical parameters, including temperature and dissolved oxygen. Shallow waters generally warm easier and quicker compared to deeper lakes. Stefanidis and Papastergiadou (2012) studied Greek lakes and reported that variations in some water quality parameters could be due to morphometric measurements of water bodies. The homogeneity in temperature in shallow lakes has been achieved through regular mixing and stirring. This phenomenon has been recorded in other shallow lakes, including Edward and George in Uganda (Otim 2005).

The higher turbidity recorded from the settlement side of the lake is probably due to improper agriculture practices and deforestation. Agricultural activities along the lake's catchment lead to erosion and hence high turbidity in the lake from the runoff. High turbidity has been reported in the Birim river basin in Ghana (Ansa and Asante 2000) and Lake Victoria (Scheren et al. 2001) due to improper agricultural activities. Conversion of land to agriculture has been reported as one of the drivers of the deterioration of aquatic systems (Yorke and Magai 2007; Rucha et al. 2011) in India and Ghana. Removing vegetative cover means loose soil is washed into aquatic systems, creating turbid and eutrophic conditions.

Relatively lower water depth on the National Park side contributed to high salinity (Table 2). Shallow areas are generally more prone to drying up due to evaporation, and the salts become concentrated as the water drops. Saravanakumar et al. (2007) reported that changes in salinity could be due to the loss of water through evaporation and rainfall in India. Surprisingly, salinity levels recorded in the present study (Table 2) were like those in brackish waters reported along the estuaries in Sri Lanka (Gammanpila 2010). However, salinity levels below 5.0 ppt are within the freshwater range (USEPA 1986), as lakes are considered saline when salinity is above 3 ppt.

The stirring and mixing of the lake and its small size could also be attributed to the uniformity in pH values across the lake. Small size, low depth, and wind have been reported as parameters promoting water mixing (Omundi et al. 2014) in the small water bodies of Kenya. The present findings on pH values (Table 2) were like those obtained by Otim (2005) in the Nile basin in Uganda. However, the mean pH values obtained in the present study (Table 2) were slightly above the optimum aquatic range of 6.5 to 9.0 (USEPA 1986). The higher pH values could probably be explained, in part, because of higher primary production. The green color was observed on the entire surface of the lake during the sampling period. Tucker and Dabramo (2008) state that waters with high algae content result in intense photosynthesis during the day; thereby, carbon dioxide is used up in the process resulting in high pH values. A similar finding was reported by Sarvala et al. (1999) in Lake Tanganyika. In contrast, lower pH values (6.3 to 6.9) were reported in Lake Bangweulu, Zambia (Kolding 2011). Higher pH levels (>10) are harmful as they increase ammonia toxicity to fish and other organisms in aquatic systems (Rossana 2013).

Significant differences in dissolved oxygen between the study sites might have been due to biotic and abiotic factors in the lake. Occasional short-duration winds that swept the water surfaces were observed several times a day during the sampling period. These could lead to spatial differences within shorter distances. The differences in water depth could contribute to differences in dissolved oxygen levels since photosynthesis is one of the major sources of dissolved oxygen. The National Environmental Monitoring Standards (NEMS 2013) reported that dissolved oxygen is negatively affected by salinity; thus, the lower oxygen levels in the National Park site could have also been due to the significantly higher salinity observed in the present study (Table 2). The present findings are like those obtained in Lake Tanganyika by Sarvala et al. (1999). The dissolved oxygen levels recorded in the present study (Table 2) were within the natural background level of 5.0 to 7.0 mg/L, which supports aquatic life.

Zooplankton species composition and diversity

The cladoceran dominated species number in the present study, followed by copepods, which concurs with the findings of Abubakar (2013) in Nguru Lake, Nigeria. Ghidini et al. (2009) conducted a similar study in Brazil and concluded that since most cladoceran species are herbivorous and phytoplankton feeders, they can develop in many freshwater environments. Kishe-Machumu et al. (2008) reported that cladoceran is more vulnerable to predation owing to their large size, conspicuous eyes, and mode of movement, making them more attractive prey and easy targets for capture. The low transparency of Lake Mweru-Wantipa could have led to less predation, thus allowing cladoceran to flourish.

Lake Victoria has also reported the dominance of copepods among zooplankton in freshwater (Ajounu et al. 2011; Ngupula 2013). In the present study, the cyclopoids (Table 6) showed higher relative abundance than other zooplankton. Similar findings have been reported by Silva (1998) in Chilean inland waters, respectively. Cyclopoids can survive in most freshwater habitats in the Neotropics. Their feeding behavior can explain this; they are grasping feeders that generally eat more of a variety of food compared to other zooplankton (Irvine and Waya 1992). At the same time, calanoids are limited by their selective and herbivorous nature. Due to their benthic nature, ostracods are not commonly captured in zooplankton studies (Martens et al. 2008).

Contrary to the present study (Table 6), Devaraju (2015) found four taxa of ostracods of 14 taxa in a major tropical lake within the Mandya District, Karnataka. Rotifers constituted the lowest contribution among all zooplankton in the present study (Table 4). In contrast, rotifers were among the dominant taxa in Lake Victoria Basin (Waya and Chande 2004). Rotifers appear to be protected from predation owing to their diminutive size, which offers low caloric value as prey besides being not easily visible to the predators.

The Shannon Weaver diversity index showed a higher diversity of zooplankton on the settlement side, and the Hutcheson *t*-test showed a significant difference between the two sides. The diversity difference could be due to the high species evenness on the settlement side (Table 5). Also, higher salinity in the National Park site could have been responsible for the low diversity of zooplankton. Nielsen et al. (2003) reported that zooplankton diversity was reduced at salinities between 1 to 5 ppt. Low species richness on the settlement side (Table 5) could reflect the high turbidity compared to National Park areas. Ghidini et al. (2009) made a similar observation in a study of eutrophic shallow reservoirs in Brazil. The species richness in the present study (Table 5) was generally poor compared to other studies conducted elsewhere (Ezekiel et al. 2011).

Zooplankton density

Differences in zooplankton densities were observed between the two studied sites (Table 6). These could have been due to the differences in physicochemical parameters and depth between the two sites (Table 2). The lower densities of rotifers on the National Park side and absence in settlement areas compared to other groups could be attributed to higher turbidity in the lake, which was significantly high in settlement areas, and to salinity and pH, which were significantly higher in settlement areas. These factors are well known for limiting the abundance and diversity of zooplankton (Harris and Vinobaba 2012). Contrary to the present findings (Table 6), Gammanpila (2010) reported higher proportions of rotifers (11% to 37%) compared to other groups of zooplankton in a Sri Lankan lagoon. However, similar results of zooplankton total densities in studied sites have been reported by Gammanpila (2010) in Sri Lanka. The higher copepod density compared to other zooplankton has been reported by Ekwu, and Sikoki (2005) found crustacean compositions of 74% in a lower estuary in Nigeria. High densities of cyclopoids in freshwater have been reported by Ngupula (2013), who found that cyclopoids dominated by 73%. In contrast to the present findings, the highest contribution of *Moina* was reported by Abubakar (2013) in Nguru Lake, as crustaceans were 65%, of which 41% were cladocerans dominated by *Moina*.

Relationships between zooplankton, abundance, and physicochemical parameters

The regression analysis revealed a positive and significant relationship between turbidity and zooplankton on the settlement side (Tables 7 and 8). This significant positive relationship may have been contributed by a higher significant abundance of *Moina*, *Ceriodaphnia*, and *Cypris* in the settlement areas (Table 6), which also had significantly high turbidity (Table 2), which could be partly explained by the reduction of the ability of the visual predators to see prey (zooplankton) in turbid waters, thus allowing larger zooplankton to flourish. That was also observed in small lakes in the Lake Victoria basin; low transparency makes these lakes support a high abundance of zooplankton than Lake Victoria with higher transparency (Ngupula 2013). However, in the present study (Table 7), an opposite relationship was observed between zooplankton and pH. Those suggest that high pH contributes to the lower densities of zooplankton. However, contrary to the present findings, Beenamma and Sadanand (2011) reported a positive relationship between zooplankton and pH in India. In addition, Tenner et al. (2005) reported that pH values ranging from 6 to 8.5 are associated with medium productivity.

In conclusion, turbidity, salinity, and dissolved oxygen significantly differed between the two studied sites. The settlement recorded a higher diversity index than the National Park site. Apart from *Moina*, *Simocephalus*, *Ceriodaphnia*, and *Cypris*, the density of all species was not significantly different between the two studied sites. Turbidity and pH were significantly positively and negatively correlated with zooplankton abundance, respectively.

REFERENCES

Abubakar MM. 2013. Biodiversity and abundance of fish and plankton of Nguru Lake, Northeastern Nigeria. J Biol Agric Healthcare 5: 2224-3208.

Ajounu N, Ukaonu SU, Oluwajoba EO, Mbawuike BE, Williams AB, Myade EF. 2011. The abundance and distribution of plankton species in the Bony Estuary; Nigeria. Agric Biol J North Am 2 (6): 1032-1037. DOI: 10.5251/abjna.2011.2.6.1032.1037.

Ansa-Asare OD, Asante KA. 2000. The water quality of Birim River in Southeast Ghana. West Afr J Appl Ecol 1: 23-33. DOI: 10.4314/wajae.v1i1.40567.

Balakrishna D, Reddy TR, Reddy KV, Samatha D. 2013. Physicochemical parameters and plankton diversity of Ghanpur Lake, Warangal, A.P. India. Intl J Zool Res 3 (1): 44-48.

Beenamma J, Sadanand M. 2011. Monthly changes in the abundance and biomass of zooplankton and water quality parameters in Kukkarahalli in lake of Mysore, India. J Environ Biol 32: 551-557.

Brelsford WV. 1954. The problem of Mweru Wantipa. Northern Prov J 2 (2): 4.

Chabala LM, Kuntashala E, Kaluba P. 2013. Characterization of temporal changes in rainfall, temperature, flooding hazard and dry spells over Zambia. Univers J Agric Res 1 (4): 134-144. DOI: 10.13189/ujar.2013.010403.

Devaraju TM. 2015. Studies on crustacean zooplanktons in major lakes of Mandya, District Karnatala. J Intl Acad Res Multidiscip 3 (5): 343-351.

Ekwu AO, Sikoki FD. 2005. Species composition and distribution of zooplankton in lower cross river estuary, Nigeria. Afr J Appl Zool Environ Biol 7: 5-10. DOI: 10.4314/ajazeb.v7i1.41137.

Ezekiel EN, Ogamba EN, Abowei JFN. 2011. The zooplankton species composition and abundance in Sombreiro River, Niger Delta, Nigeria. Asian J Agric Sci 3 (3): 200-204.

Frame Survey Report. 2004. Lake Mweru-Wantipa Pilot Fishery Survey. Zambia Central Statistical Office, Lusaka, Zambia.

Gajbhiye SN. 2002. Zooplankton study methods, importance, and significant observations. In: Quadros (eds). Proceedings of the National Seminar on Creeks, Estuaries and Mangroves. Thane, 28 November to 30 November. [India]

Gammanpila M. 2010. Hydrography, nutrients and abundance and distribution of zooplankton in Negambo Lagoon, Sri Lanka. Sri Lanka J Aquat Sci 15: 13-24. DOI: 10.4038/sljas.v15i0.5447.

Ghidini AR, Moacyr SJ, Gilmar PN, Lineu DB. 2009. Distribution of planktonic cladocerans crustacean: Branchiopoda) of a shallow eutrophic reservoir Panama State, Brazil). Pan-Am J Aquat Sci 43 (3): 294-305.

Goswami AP, Mankodi PC. 2012. Study on zooplankton of Fresh Water Reservoir Nyari-II Rajkot District, Gujarat, India. ISCA J Biol Sci 1 (1): 30-34.

Harris JM, Vinobaba P. 2012. Impact of water quality on species composition and seasonal fluctuation of plankton of Batticaloa Lagoon, Sri Lanka. Jl Ecosyst Ecograph 2 (4): 6.

Hutcheson T. 1970. A test for comparing diversities based on the Shannon formula. J Theor Biol 29: 151-154. DOI: 10.1016/0022-5193(70)90124-4.

Imaobong E. 2013. Effect of physicochemical parameters on zooplankton species and density of a tropical rainforest in Niger Delta, Nigeria using canonical analysis. Intl J Eng Sci 2 (4): 13-21.

Irvine K, Waya R. 1992. Predatory behavior of the cyclopoid copepod mesocyclops aquatorialis aquatorialis in Lake Malawi, a deep tropical lake. Verhandlungen Intl Verein Limnologie 25: 877-881. DOI: 10.1080/03680770.1992.11900271.

Kedar GT, Patil GP, Yeole SM. 2008. Effect of physicochemical factors on the seasonal abundance of zooplankton population in Rishi Lake. Proceedings of Taal, the 12th World Lake Conference 88-91.

Kishe-Machumu MA, Witte F, Wanink J. 2008. Dietary Shift in benthivorous cichlids after the ecological changes in Lake Victoria. Anim Biol 58 (4): 407-417. DOI: 10.1163/157075608X383700.

Kolding J. 2011. A Brief Review of the Bangweulu Fishery Complex. Report Prepared for the Bangweulu Wetlands Project, Department of Biology, University of Bergen, Norway.

Martens K, Schon I, Claude M, David HJ. 2008. Global diversity of Ostracods (Ostracoda, Crustacea) in freshwater. Fresh Water Animal Assessment. Hydrobiologia 595: 185-193. DOI: 10.1007/s10750-007-9245-4.

Mubamba R. 1989. The biology and exploitation of small pelagic fishes in Zambia. Department of Fisheries, PO Box 350100, Chilanga, Zambia. Symposium on Biology, Stock Assessment and Exploitation of Small Pelagic Species in the African Lakes Region in Bujumbura, Burundi from 25 to 28 November 1992.

NEMS. 2013. Dissolved Oxygen Recording, Measurement, Processing and Achieving on Dissolved Oxygen. National Environmental Monitoring Standards New Zealand data version 1: 48

Ngupula GW. 2013. How does increased eutrophication and pollution in Lake Victoria waters impacts zooplankton? J Environ Ecol 4 (2): 151-164. DOI: 10.5296/jee.v4i2.4397.

Nielsen DL, Brock MA, Crossle K, Harris K, Healey M, Jarosinki I. 2003. The effect of salinity on aquatic plant germination and zooplankton hatching from two wetland sediments. Freshw Biol 48: 2214-2223. DOI: 10.1046/j.1365-2427.2003.01146.x.

Nkotagu HH, Athuman CB. 2007. The limnology of the Lake Tanganyika sub-catchment. Geology Department, University of Dar es salaam. Tanzania J Sci 33: 19-26. DOI: 10.4314/tjs.v33i1.44272.

Omundi R, Kimbenya, Nyameveya, Ouma H, Machua SK, Ogari Z. 2014. Recent limnological changes and their implication on fisheries in Lake Baringo, Kenya. J Ecol Nat Environ 6 (5): 154-163. DOI: 10.5897/JENE2014.0438.

Otim M. 2005. Baseline Study of the Status of Water Quality Monitoring in Uganda. Nile Basin Initiative and Nile Transboundary Environment Action Project-Nile Basin National Water Quality Monitoring Baseline Study Report for Uganda.

Paul BS. 2008. Profile of Zambia's Small Holder. Where and Who are the Potential Beneficiaries of Agricultural Commercialization. World Bank Report no. 36573-ZM.62

Rossana S. 2013. Understanding Water Quality Parameters to Better Manage Your Pond. Department of Extension Animal Sciences and Natural Resources, New Mexico State University, Guide no.104: 4pp.

Rucha RJ, Warthe M, Shared D, Ritesh V, Tapan C. 2011. Monitoring changes in land use land cover of Yamuna riverbed in Delhi: A multi-temporal analysis. Intl J Remote Sens 32 (24): 9547-9558. DOI: 10.1080/01431161.2011.565377.

Saravanakumar A, Rajikumar M, Serebiah SJ, Thivakaran GA. 2007. Abundance and seasonal variations of zooplankton in the arid zone mangrove of Gulf Kachchh-Gujarat, west coast of India. Pak J Biol Sci 10 (20): 3525-3532. DOI: 10.3923/pjbs.2007.3525.3532.

Sarvala J, Salomen K, Javinen M, Aro E, Huttula T, Kotrilainea P, Kurk H, Langenberg V, Maunini P, Peltonen A, Plisnier PD, Vourinen I, Molsa H, Lindqvist OV. 1999. Trophic structure of Lake Tanganyika: Carbon flows in the pelagic food web. Hydrobiologia 407: 45-58. DOI: 10.1007/978-94-017-1622-2_15.

Scheren PAGM, Mirambo V, Lemmens AMC, Katima JHY, Jansee FJJG. 2001. Assessment of pollution sources and socio-economic circumstances related to the eutrophication of Lake Victoria. Paper Presented at LVEMP Conference, Kisumu, Kenya.

Silva VM. 1998. Diversity and distribution of the free living freshwater cyclopodia (Copepod: Crustacea) in Neotropics. Braz J Biol 68: 1099-1106. DOI: 10.1590/S1519-69842008000500016.

Spellerberg IF, Fedor PJ. 2003. A tribute to claude Shannon (1916-2001) and a plea for more rigorous use of species richness, species diversity and the Shannon Wiever Index. Glob Ecol Biogeogr 12: 177-179. DOI: 10.1046/j.1466-822X.2003.00015.x.

Stefanidis K, Papastergiadou E. 2012. Relationships between Lake Morphometry, water quality and aquatic macrophytes in Greek lakes. Fresenius Environ Bull 21: 3018-3926.

Takarina ND, Nurliansyah W, Wardhana W. 2019. Relationship between environmental parameters and the plankton community of the Batuhideung Fishing Grounds, Pandeglang, Banten, Indonesia. Biodiversitas 20: 171-180. DOI: 10.13057/biodiv/d200120.

Tenner CC, Cragga RJ, Sukias JP, Park JBK. 2005. Comparison of maturation ponds and constructed wetlands as the final stage of an advanced pond system. Water Sci Technol 51 (12): 307-314. DOI: 10.2166/wst.2005.0489.

Tonapi GT. 1980. Freshwater Animals of India an Ecological Approach. Oxford and IBH. Publishing Company New Delhi, India.

Tucker CS, Dabramo LR. 2008. Managing pH in Freshwater Ponds. South Regional Aquaculture Centres (SRAC) Publication no.4604: 6pp.

USEPA. 1986. Quality criteria for water, office of water regulations and standards, Washington, DC EPA/440/5-86-001: 477pp.

USEPA. 2010. Standard Operating Procedures for Zooplankton Analysis. Sampling and Analytical Procedures. Revision No.3: 20pp.

Utzugi K, Mazingaliwa K. 2002. Field guide to Zambian Fishes, Planktons, and Aquaculture. Japan International Cooperation Agency, Japan.

Waikato Evironmental Technical Report. 2008. Zooplankton Composition and Water Quality Assessment of Seventeen Lakes Using Rotifer Community Composition. Environmental Waikato-Regional Council, New Zealand.

Waya RK, Chande AI. 2004. Species composition and biomass estimate of zooplankton in some water bodies within lake Victoria Basin. Tanzania J Sci 30 (11): 43-52. DOI: 10.4314/tjs.v30i1.18386.

Yorke C, Magai FM. 2007. Monitoring land use change in the Densu river basin Ghana using GIS and Remote Sensing Methods. Afr Geogr Rev 85-93. DOI: 10.1080/19376812.2007.9756203.

5INTL J BONOROWO WETLANDS
Volume 12, Number 1, June 2022
Pages: 41-54

P-ISSN: 2775-8052
E-ISSN: 2775-8044
DOI: 10.13057/bonorowo/w120105

The biological characteristics of the Nile tilapia (*Oreochromis niloticus*) in the Weija Reservoir, Ghana

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Manuscript received: 19 March 2022. Revision accepted: 27 June 2022.

Abstract. *Kpelly DK, Blay Jr.J, Yankson K. 2022. The biological characteristics of the Nile tilapia (*Oreochromis niloticus*) in the Weija Reservoir, Ghana. Intl J Bonorowo Wetlands 12: 41-54.* Studies on the biological characteristics of the Nile tilapia (*Oreochromis niloticus* L.) were undertaken in a manufactured lake, the Weija Reservoir, in Ghana from September 2007 to August 2008. From commercial landings, a sum of 657 specimens of *O. niloticus* was examined, of which 357 were males and 300 were females (sex ratio of 1: 0.84; $P < 0.05$). GSI was highest in March and May 2008 and low in December 2007 and August 2008, which suggested that *O. niloticus* spawned throughout the year, but major spawning occurred in March. The length at first sexual maturity (L_{50}) was 16.33 cm for males and 13.49 cm for females. Observations on the frequency distribution of ovum size measured between 1.2 to 3.1 mm showed two distinct peaks that were not completely separated, which were probably shedding eggs in batches, indicating protracted spawning in the population. Fecundity ranged from 563 to 1542 ova for fish of total length 15.6-21.5 cm with a mean of 851 ± 13.2 eggs. The relationships between fecundity and body weight (BW) and fecundity (F) and total length (TL) were: $F = 0.519\ BW + 785.4$ and $F = 12.36\ TL + 628.5$. Furthermore, based on studies on monthly fluctuations in the GSI and the occurrence of the ripe gonad, the major reproductive activity of *O. niloticus* in the reservoir coincided with months with increasing total alkalinity, reduced water temperature, and dissolved oxygen. The regression coefficient of 3.1 for the length-weight relationship (for both females and males) was not significantly different from the expected value of ($P > 0.05$), indicating isometric growth of the population. Therefore, collection of the species as seed for stocking or brood stock or by fish farmers is recommended in March-April. Moreover, appropriate management policies with periodic studies of the fishery and the limnology of the reservoir are essential to sustain fishery production.

Keywords: Body weight, fecundity, *Oreochromis niloticus,* Weija Reservoir

INTRODUCTION

Fish supplies over 50% of the total animal protein consumed in developing countries, a little below this value in developed countries among the first natural resources to be exploited by man (FAO 2000). It is the most important source of animal protein in Ghana, contributing about 60% of the daily animal protein intake (Frimpong et al. 1992). Among various protein sources, fish stands out as the most important concerning food security because its low price is very competitive compared to the price of other high-quality protein sources such as milk, meat, and eggs (FAO 2000). Moreover, it is the only source of high-quality protein whose shelf life can be readily enhanced through low-cost sustainable technologies such as smoking, drying, and salting.

Tilapias are among the most important warm-water fishes used for aquaculture production. The culture of Tilapia is practiced in over 80 countries in the tropical and subtropical regions globally, with a total production of 478,641 metric tonnes and an average compound growth rate of about 12% yearly since 1986 (FAO 1997; Lovshin 1997). Regarding Balarin and Hatton (1979), Tilapia's culture can provide a welcome solution to the insufficient protein supply in numerous developing countries. Furthermore, from simple waste-fed fish to intensively stocked and managed culture systems, Tilapias are widely known as one of the most important groups of cultured purposes finfish in a wide range of freshwater culture systems (Pullin 1985; Fitzsimmons 2000). Moreover, attempts to grow them in full-strength seawater have also been made in some developed countries. However, the research in full-strength seawater (32 ppt) observed extremely poor hatching success in ova spawned (Watanabe et al. 1992).

Tilapias belong to the family Cichlidae with three well-known genera: *Tilapia, Oreochromis,* and *Sarotherodon*. The species are classified mainly according to some extent, their feeding habits, biogeography, morphology, and to differences in their mode of reproduction. (Lowe-McConnel 1958). Members of the genus *Tilapia* (for example, *Tilapia zillii*) are substrate spawners. Both parents aerate the brood, protect, guard, and help move the clutch to different nest sites. The fry at first feeding is 4-6 mm long (TL) and shows feeble swimming ability, resulting in relatively low fry survival (Lowe-McConnel 1958). The genus *Sarotherodon* shows biparental or paternal care. In *S. melanotheron,* for example, both parents stay close to each other, and their eggs and fry are brooded in the oral cavity of the male, with the females sometimes helping when the male fails to pick up all the eggs. The fry is between 7-9 mm (TL) at first feeding, with well-developed fins for swimming, giving them a high fry survival (Lowe-McConnel 1958; Pauly 1976). In the genus Oreochromis, they are regarded as maternal brooders due to the female is solely involved in the brood care by orally incubating the fry until they reach the free-swimming stage (Lowe-McConnel 1958; Pauly 1976). The female was also observed to leave the nest to orally brood her clutch in safety among the submerged rocks and vegetation with an extended caring period during which fry seeks shelter in the buccal cavity of the female. This genus includes *Oreochromis niloticus* (L.), commonly called Nile tilapia.

The *O. niloticus* and its hybrids are the most important cultured fish species, particularly the subspecies *O. niloticus niloticus*, which is becoming an increasingly important food fish globally (Pullin et al. 1991; Garibaldi 1996). The *O. niloticus* is the major species farmed in Ghana and, according to FAO (2005), constitutes over 80 % of aquaculture production in the country. It occurs in several rivers and natural as well as manufactured lakes. The yields are almost fixed at their maximum from many capture fisheries (FAO 1997), while the global human population continues to increase rapidly (FAO 2005), and to meet the increased demand for food fish, aquaculture production should increase by 50 million metric tonnes by 2050 (Tacon 2001).

Many tilapias' biology in natural systems is well documented (Fryer and Iles 1972; Pauly 1976; Siddiqui 1977; Trewevas 1983; Silva 1985; Blay and Asabere-Ameyaw 1993; Njiru et al. 2006; Olurin and Aderibigbe 2006). However, despite the predominance of *O. niloticus* in the Weija Reservoir, the key characteristics of the species' biology in reproduction and its growth have not been fully observed. Therefore, this research seeks to present important scientific data by examining aspects of the biology of *O. niloticus* in the Weija Reservoir, a manufactured lake, that are relevant to its culture.

MATERIALS AND METHODS

Study area

The Weija Reservoir, also known as Lake Naye Densu, is part of the 116 km long River Densu located about 17 km west of Accra (5° 33' and 5° 36' N; 0° 20' and 0° 23' W), which takes its source from the Atewa Atwiredu mountain range in the Eastern Region of Ghana. (Figure 1). It was created to replace an earlier dam that collapsed in 1968. The Weija Dam was constructed in 1978 by Ghana Water Company Limited (GWCL), intending to provide potable water for domestic consumption, while fisheries development and irrigation serve secondary purposes. The reservoir is 2.2 km wide, 14 km long, and has a total surface area of 38 km^2 with a mean depth of 5 m (Vanden-Bossche and Bernacsek 1990). The catchment area covers 2,460 km^2, and the reservoir irrigates about 220 ha of land (Gordon 2006). The average water temperature is 27^{o}C, and rainfall in the area is seasonal, with an annual average of 65.5 mm and peaks in June and September, while the dry period is between December and March (Asante et al. 2006). The vegetation is of the coastal savannah type around the lake, mainly characterized by grasses, shrubs, and some trees dominated by *Cassia* sp. The main economic activities in the catchment area are crop farming and fishing. The crops cultivated are cassava, sugar cane, maize, and vegetables.

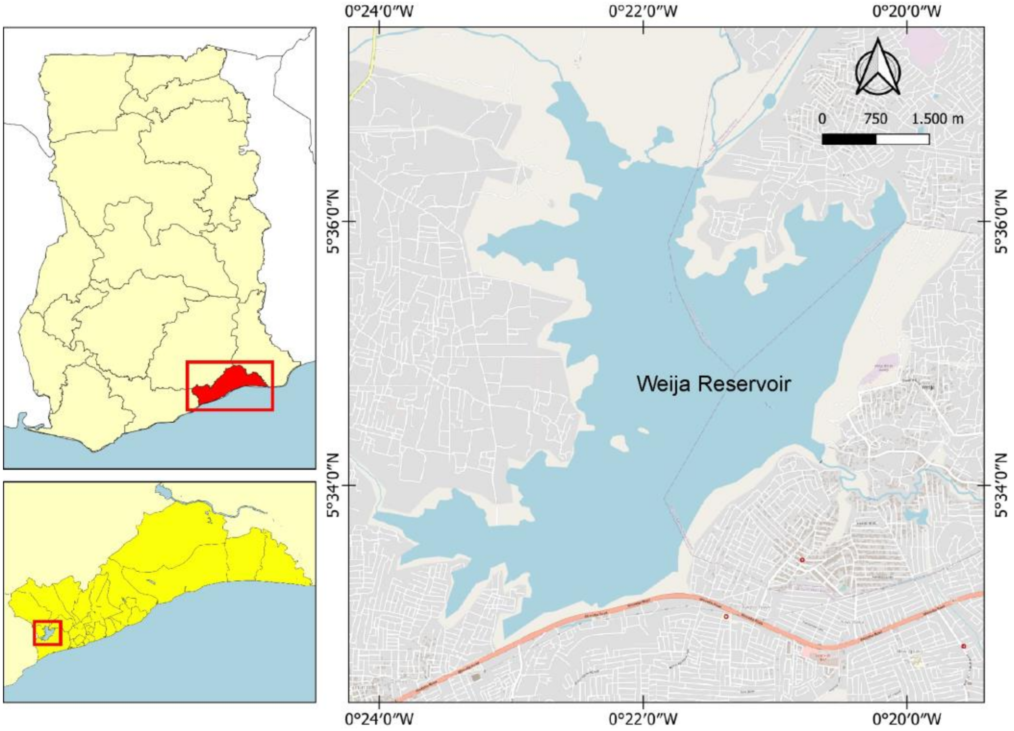

Figure 1. Map of Weija, Ghana area showing the reservoir (Source: Ghana Survey Department 2008)

Fish sampling and data collection

Fish samples were obtained in the middle of each month from September 2007 to August 2008 from commercial landings at the New Galilea landing site of the Weija Reservoir (Figure 1). The fish were caught with cast nets (20.0 mm) and gill nets (large mesh size 50-70 mm, knot to knot; medium mesh size 30-40 mm; small mesh size 10-25, and overall net size about 20 m long). The samples transported to the laboratory for routine studies (at least 40 specimens) were kept on ice in a plastic container.

The total length (TL) and standard length (SL) were taken to the nearest 1.0 mm using a fish measuring board in the laboratory. The total length and the standard length from the tip of the snout to the base of the caudal fin were measured from the tip of the snout to the end of the caudal fin. The fish body weight (BW) was determined to the nearest 0.01g using an electronic top loading balance, and the fish were blotted dry with a towel. Each fish was dissected to determine its sex, and the gonads were removed and weighed using an electronic top-loading balance to the nearest 0.01g.

Determination of length frequency distribution and length-weight relations

The data obtained on the standard length (SL), total length (TL), and body weight (BW), were analyzed to investigate the monthly length-frequency distribution and length-weight relationships to show the recruitment of the fish and their growth pattern.

Determination of condition factor (K)

The measurement data obtained of body weights (BW) and standard lengths (SL) were used to calculate each fish factor (K) condition using the formula:

$$K = \frac{BW}{SL^3} \times 100 \quad \text{(Tesch, 1971)}$$

The mean monthly condition factor of female and male fish was plotted to illustrate the fluctuations in the relative fish's 'fatness' or 'well-being' during the study period.

Determination of visceral fat index

After dissection, each fish's visceral fat amount content was observed and estimated using a three-point scale, according to Kwei (1970), and recorded as follows: 1: viscera with very low-fat volume; 2: viscera with moderate fat volume; 3: viscera with high-fat volume.

The data on the fat index of the male and females were analyzed using Minitab (version 15) statistical software, obtaining their mean monthly fat index and standard errors. Then, these were plotted using Microsoft Excel (version Office 2007) to observe the fish 'fatness' fluctuations, indicating their condition.

Determination of the sex ratio

After sexing the fish, the monthly sex ratio of male to female fish was determined using the data obtained. In addition, a Chi-square goodness of fit test was also performed to observe any difference in the sex ratio of 1:1 on the monthly samples from the *O. niloticus* population.

Gonadal staging

Based on their appearance, the gonads of the fish were examined and staged, according to Witte and Van Densen (1995). The three-point scale was used to stage ovaries: Stage 1- Immature: small haline ovaries with very small whitish ova; Stage 2- Developing: ovaries with yellow ova or few yellowish-white; Stage 3- Ripe: gravid ovaries with large yolky ova.

Testes were classified as either ripe or immature due to difficulties describing intermediate developmental stages. Testes of immature fish were thin and very transparent, while ripe testes were swollen with milt and creamy or reddish, which flows out easily when the testes are cut. The monthly fluctuations of the ripe gonads in the fish were obtained from the percentage of ripe males and females in each monthly sample that was determined and then plotted.

Determination of gonado-somatic index (GSI)

The gonado-somatic index (GSI) of females and males was calculated as

$$GSI = \frac{GW}{BW - GW} \times 100 \quad \text{(Htun-Han, 1978)}$$

Where, GW is gonad weight (g), and BW is body weight (g).

Next, to obtain the mean monthly GSI of the fish and their standard errors, the data were analyzed using Minitab (version 15) statistical software. In addition, the mean monthly GSI of both male and female fish were plotted to determine the changes in the reproductive activities of the fish.

Determination of fecundity

The fecundity was determined by the whole count method (Bagenal and Braum 1978). First, all ripe ovaries were cut into pieces and preserved for about four weeks in sample bottles containing Gilson's fixative. That was done to ensure the hardening of the eggs and permit infiltration of the ovarian tissues by the fixative. Next, each bottle was periodically agitated vigorously to facilitate the separation of the eggs from ovarian tissues (Bagenal and Braum 1978). Then, the content was poured into a petri dish, and eggs were separated from the ovarian tissue using a dissecting pin and a pair of forceps before counting. Finally, all ripe eggs in an ovary were counted, and using analysis of variance, the relationship between fecundity, standard length (SL), and body weight (BW) were established.

Determination of length at first maturity (L_{50})

The data on the fish's total length (TL) and gonadal stages were used to determine the length at first maturity (L_{50}). In addition, the percentage of males and females considered mature for the selected class size was used to determine the maturity percentage. Finally, these values were used to plot graphs for males and females using the statistical software Origin Professional (version 6.0) to estimate the length at which 50% of the population is sexually mature.

Measurement of ovum diameter

A sum of four ripe female ovaries diameters selected at random were used. After determining the fecundity, the ova were measured using a dissecting microscope with a stage micrometer on their longest axis to the nearest 0.1 mm and recorded. Next, in the diameter class at 0.2 mm intervals, the data for each pair of ovaries selected were sorted and grouped. Next, using Microsoft Excel (version Office 2007), the data was used to plot the ovum diameter frequency distribution to predict the fish's spawning frequency.

Hydrographic factors

Data on the monthly mean on some hydrographic factors, namely, dissolved oxygen of the surface water of the reservoir, temperature, and total alkalinity, were obtained from the study period from the Ghana Water Company Limited (GWCL)/AVRL at Weija (see Appendix 6). Next, to show their range and monthly fluctuations during the study period, the data were plotted using Microsoft Excel (version Office 2007) to show any possible relation between them and the growth or reproductive activities of the fish.

RESULTS AND DISCUSSION

Monthly length-frequency distribution

The monthly length-frequency distributions of *O. niloticus* in the Weija Reservoir from September 2007 to August 2008 are presented in Figure 2. Most of the fish sampled ranged from 14 to 20 cm TL. Apart from November 2007, where juveniles were encountered, other monthly samples were skewed towards individuals measuring 12 or 13cm (TL). Males were relatively larger than females, as shown in the modal classes of 16.0 –16.9 cm and 19.0 – 19.9 cm for female and male fish, respectively. The distribution size was bimodal in July 2008, with the modal groups as 14.0-14.9 and 20.0-20.9 cm TL. In September, October, November, and December 2007, they presented unimodal distribution with their modal class within 17.0-17.9, 18.0-18.9, 12.0-12.9, and 13.0-13.9 cm, respectively. Furthermore, in January, February, March, April, May, June, and August also presented unimodal distribution with their modal classes in the 11.0-11.9, 15.0-15.9, 16.0-16.9, 15.0-15.9, 12.0-12.9, 16.0-16.9 and 19.0-19.9 cm respectively. In Juli, bi-modal size distribution was observed with the modal classes in the 14.0-14.9 cm and 20.0- 20.9 cm size groups.

Length-frequency distribution of female and male

Figure 3 shows the length-frequency distribution for females and males. The females and males distributions were unimodal, with the modal classes within the 16.0-16.9 cm and 19.0-19.9 cm groups, respectively. The males were also larger than the females, apart from the fact that more males (N= 357) were encountered. The smallest fish size obtained was a female in the 7.0-7.9 cm group, and the largest female obtained belonged to the 30.0-30.9 cm group. The smallest male was within the 8.0-8.9 cm group, with the largest in the 33.0-33.9 cm group.

Overall length-frequency distribution

The overall size-frequency distribution of *O. niloticus* in the Weija Reservoir during the study period is shown in Figure 4, with the modal size within 17 cm TL; thus, the distribution is unimodal. A total of 657 specimens with a total length range of 7.0-33.3 cm were sampled.

Length-weight relationship

Figure 5 is shown the scatter plots of the length-weight relation of the population of *O. niloticus* in the Weija Reservoir during the study period and the curve of this relationship. In all the 657 specimens sampled, the total length (TL) ranged between 7.0 and 33.3 cm, and weighed between 17.85 g and 700.00 g. The relationship between the standard length (SL) and body weight (BW) revealed was exponential, with the equation describing this relationship as $BW = 0.0317\ SL^{3.0932}$ (Figure 5). In addition, Microsoft Excel (version Office 2007) generated a student t-test which revealed that $b = 3.0932$ was not significantly higher ($P > 0.05$) than the expected 3.0 (see Appendix 2).

Condition factor

Figure 6 shows the condition factor (K) fluctuations of females and males of O. niloticus. The mean monthly condition factor ranged from 3.76 ± 0.06 to 4.96 ± 0.125 for the males and from 3.56 ± 0.191 to 4.31 ± 0.131 for the females. During the study period, the condition factor for both sexes seemed to follow a similar trend, except for May 2008, with the males being relatively higher. The fish's best condition was in October 2007, declining in November 2007 and not varying significantly for the rest of the study period in 2008 except in May 2008.

Analysis of visceral fat index

The female and male *O. niloticus* monthly fluctuations in the visceral fat index in the Weija Reservoir are presented in Figure 7. During the study period, the female visceral index ranged between 1.17 ± 0.09 and 1.87 ± 0.15, that is increased from 1.57 ± 0.16 in September 2007, reaching a peak of 1.87 ± 0.15 in November 2007. The females' fluctuations in the visceral fat index were generally irregular, with the lowest value of 1.17 ± 0.10 observed in April 2008. On the other hand, the visceral fat of males decreased from 1.65 ± 0.11 in September to 1.42 ± 0.10 in December 2007, which reaching a peak of 2.69 ± 0.15 in March 2008, and then dropped sharply till May 2008, where the lowest value was observed at 1.23 ±. The visceral fat of males then increased steadily from June, reaching 1.82 ± in August 2008, with two major peaks observed in March and August 2008. Generally, males' monthly visceral fat index was higher than the females except for November 2007 and June 2008.

Sex ratio

Table 1 shows the monthly sex ratio of *O. niloticus;* from a total of 657 Nile tilapia examined, 357 were males, and 300 were females, giving a sex ratio of 1:0.84 overall. Except for January, March, and June 2008, where the females outnumbered the males, the males generally were more in all the monthly samples. However, the sex ratios for the monthly samples of April, July, and August 2008 differed significantly from the expected 1:1 in favor of the males.

Fecundity

The absolute fecundity from the Weija Reservoir of 94 individuals of *O. niloticus* was observed. The fecundity ranged from 563 ova for fish weighing 108 g and measuring 15.6 cm TL to 1,542 ova for fish weighing 200 g and measuring 21.5 cm TL. The mean fecundity of examined fish was 850.8 ± 13.2. Figure 8 presents the relationship between fecundity (F) and total length (TL), and the relationship between fecundity and body weight (BW) is presented in Figure 9.

The equations describe the relationships:

$F= 12.36TL + 628.5$ ($r = 0.40$)

$F= 0.519BW + 785.4$ ($r = 0.37$)

The correlation value was weak between fecundity and body weight ($r = 0.38$) and fecundity and total length ($r = 0.4$).

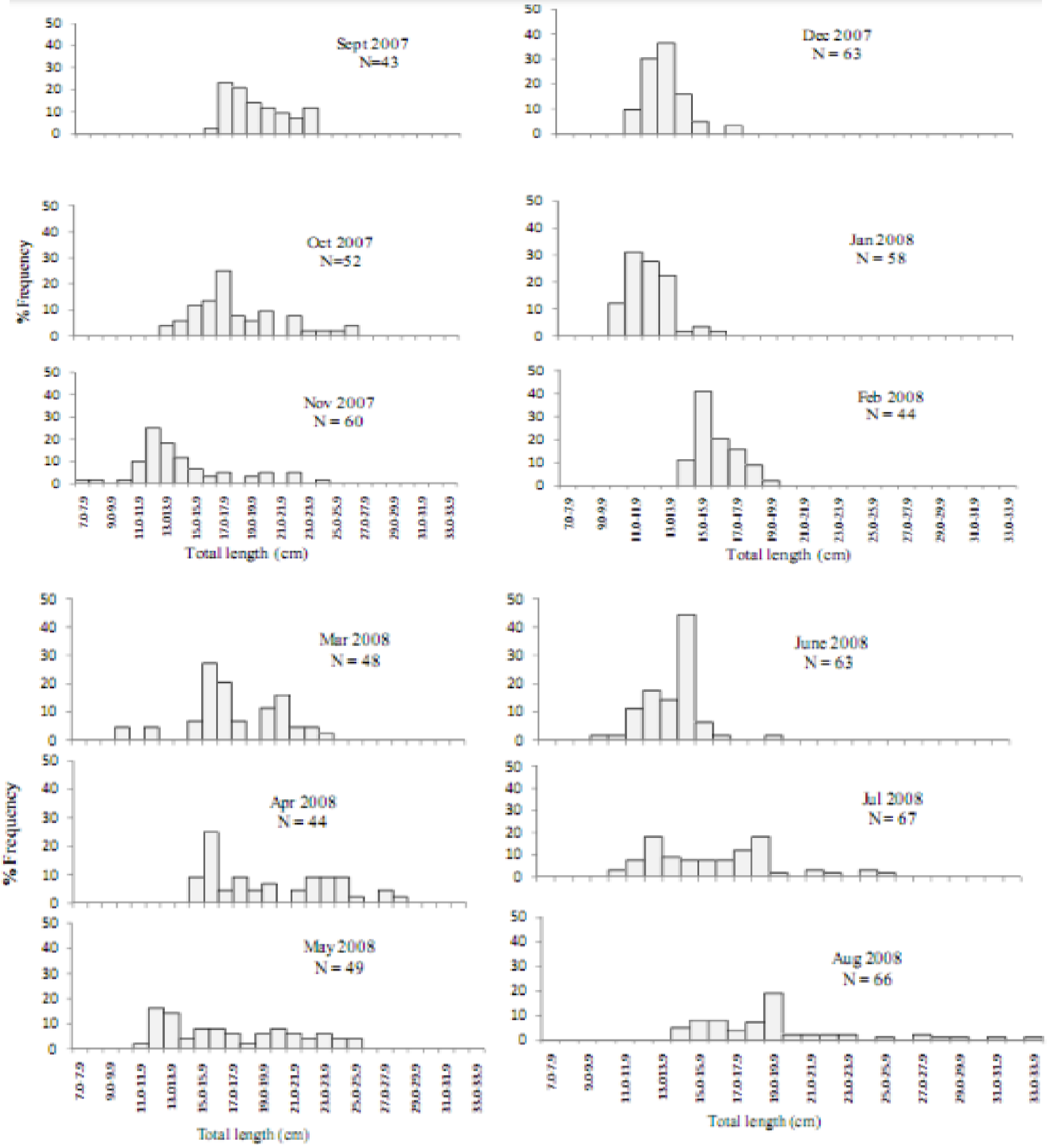

Figure 2. The monthly length-frequency distribution of *O. niloticus* caught in the Weija Reservoir, Ghana

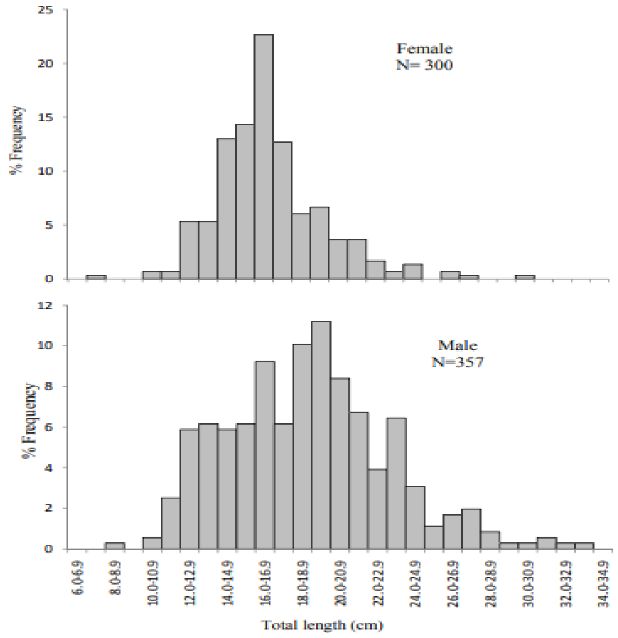

Figure 3. The female and male *O. niloticus* length-frequency distribution caught in the Weija Reservoir, Ghana, from September 2007 to August 2008 (N=total number of fish)

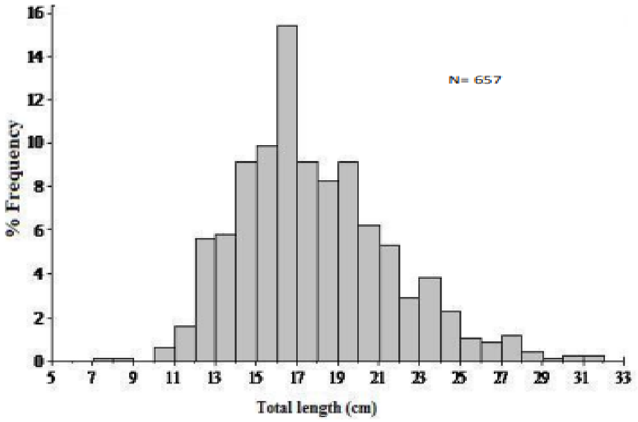

Figure 4. The overall *O. niloticus* length-frequency distribution caught in the Weija Reservoir from September 2007 to August 2008 (N= number of fish)

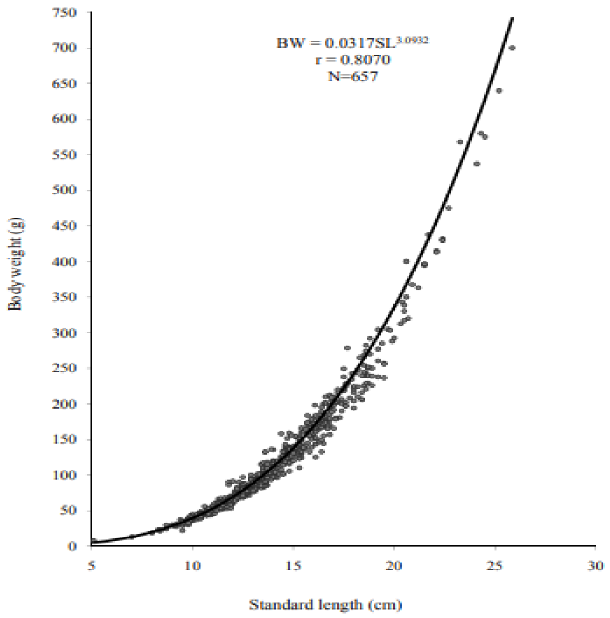

Figure 5. Length-weight regression of the *O. niloticus* population in the Weija Reservoir, Ghana, from September 2007 to August 2008

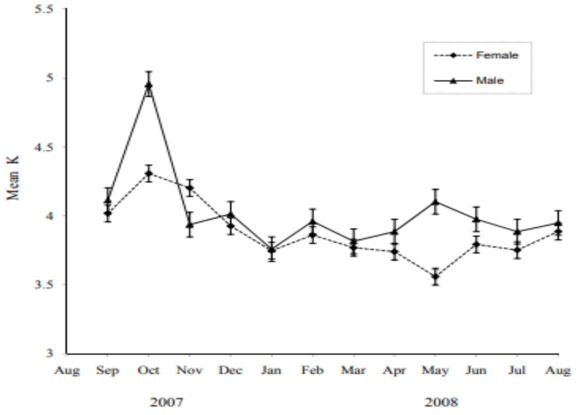

Figure 6. The fluctuations of *O. niloticus* Condition Factor (K) in the Weija Reservoir, Ghana (vertical bars = standard error)

Length at first maturity (L_{50})

Figure 10 is presented the fitted logistic curves for length at first maturity of both male and female *O. niloticus* in the Weija Reservoir. The majority of the fish obtained were sexually mature. The fish attains maturity beyond 10.00 cm TL. Females larger than 13.49 cm TL have a 50% and more chance of being sexually mature. Males with a size greater than 16.33 cm TL also have a more than 50% chance of being sexually mature.

Fish of a total length greater than 18.00 cm are expected to be fully matured. Males mature larger (L_{50} = 16.33 cm) than females (L_{50} = 13.49 cm).

Fluctuations in ripe gonad representation

Figure 11 is presented the monthly occurrence of ripe gonads in the samples. The percentage of ripe testes ranged between 7% and 100%, and that of the ovaries was between 15% and 53%. The percentage of *O. niloticus* ripe testes peaks occurred in September 2007, February, and July 2008, while ovaries occurred in September 2007 and March 2008. The lowest value for the percentage of ripe ovaries occurred in June 2008, and the testes were recorded in November 2007.

Figure 12 represents the relationship between monthly variation in ripe gonads representation and some hydrographic factors (dissolved oxygen, temperature, and total alkalinity) of the Weija Reservoir. The water's dissolved oxygen concentration decreased from 11.2 mg/L in September 2007 to 7.8 mg/L in January 2008, then increased to 9.7 mg/L in March, and finally decreased to 6.6 mg/L in June 2008. The oxygen content after that increased steadily to 7.6 mg/L in August. From September to October 2007, the water temperature remained at 27.5ºC, then increased to 29ºC in November and dropped to 26.7ºC in January 2008. in February 2008, a sharp increase in the water temperature was observed, reaching the highest value of 30.5ºC, then this value dropped sharply to a minimum of 23ºC the following month. The water temperature again increased steadily to 27.8ºC in June and then declined slightly to 26.8ºC in August.

In October 2007, the minimal total alkalinity value of 98 mg/L was recorded, then increased through November, December 2007, and January 2008, and a maximum of 128 mg/L was reached in February. There was a slight drop in the water's total alkalinity in March and April 2008, followed by a sharp drop to a value of 100 mg/L in May. The value increased sharply through June and July until 115 mg/L was reached in August 2007. The periods of marked changes in the hydrographic factors (dissolved oxygen, temperature, and total alkalinity) in the reservoir arrived to coincide with the month of the highest ripe ovaries for *O. niloticus* (Figure 12). The highest percentage of ripe testes was recorded in February 2008; however, it seemed to be a phase ahead of the months with the marked changes in hydrographic factors.

Fluctuations in the Gonado-Somatic Index (GSI)

The male and female *O. niloticus* monthly variations of the GSI in the Weija Reservoir are presented in Figure 13. The GSI for females was generally higher than that of males due to heavier ovaries than the testes. In females, GSI values dropped from 2.60 in September to 1.73 in December 2007, after which it increased marginally from January to February 2008, then on March 2008 with a sharp increase in GSI to a peak of 3.63, and generally remained low for the rest of the study period except a sharp decrease followed that in April (2.07). On the other hand, the GSI values for males increased from 0.29 in September to 0.39 in November 2007, followed by a drop to 0.25 in December 2007. Generally, GSI increased from January, reaching a peak of 0.69 in March 2008, followed by another drop to 0.43 in April 2008.

The maximum GSI value for males was 0.78, observed in June 2008. However, the GSI of both males and females declined from October to December 2007. Then the GSI increased from January to March 2008, with the major peaks observed for both sexes. The trend followed a sharp decline the following month, then decreased gradually during the remaining months.

Figure 14 shows the relationship between some hydrographic factors of the Weija Reservoir and monthly variation in the gonado-somatic index (GSI). The periods of marked changes in the hydrographic factors (dissolved oxygen, temperature, and total alkalinity) observed in March 2008 also seem to coincide with the month with the highest GSI values for male and female *O. niloticus* in the reservoir.

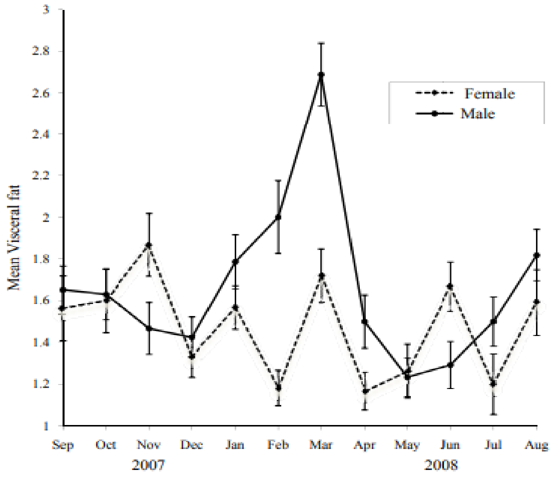

Figure 7. The fluctuations in the visceral fat index of *O. niloticus* in the Weija Reservoir, Ghana (vertical bars = standard error)

Table 1. Sex ratio of *Oreochromis niloticus* in the Weija Reservoir, Ghana

Year	Month	Males	Females	Sex ratio ♂:♀	χ^2	p (0.05)
2007	Sep	25	18	1:0.7	1.14	NS
2007	Oct	27	25	1:0.9	0.077	NS
2007	Nov	30	30	1:1	0.0	NS
2007	Dec	33	30	1:0.9	0.78	NS
2008	Jan	28	30	1:1.07	0.31	NS
2008	Feb	22	22	1:1	0.0	NS
2008	Mar	16	32	1:2	5.33	S
2008	Apr	26	18	1:0.7	1.45	NS
2008	May	30	19	1:0.6	2.47	NS
2008	Jun	24	39	1:1.6	3.57	NS
2008	Jul	52	15	1:0.3	20.43	S
2008	Aug	44	22	1:0.05	7.33	S
	Total	**357**	**300**	**1:0.84**	**4.95**	**S**

Note: S = Significant; NS = Not significant. A Chi-square test was examined to indicate that the overall difference from the 1:1 ratio was significant ($P < 0.05$) and in favor of the males ($\chi^2 = 4.95$)

Ovum diameter frequency distribution

Figure 15 illustrates the frequency distributions of the ovum diameter of four ripe ovaries of *O. niloticus.* for fish between 14.4 and 21.0 cm TL, the ovum diameter ranged between 1.2 and 3.1 mm. Two distinct peaks were observed in each ovary studied, not completely separated from each other. The modal sizes were: 1.4 mm and 2.0 mm for fish of a total length of 15.4 cm, 1.6 mm and 2.2 mm for fish of a total length of 16.8 cm, 1.6 mm and 2.2 mm for fish of a total length of 14.6 cm and 1.8 mm and 2.4 mm for fish of total length of 21.0 cm. The batches of ova to be spawned are represented by the peak of the modal sizes

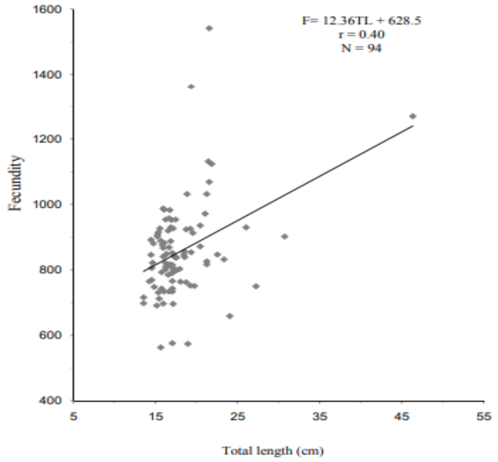

Figure 8. The relationship between fecundity and total length of *O. nolíticus* in the Weija Reservoir, Ghana

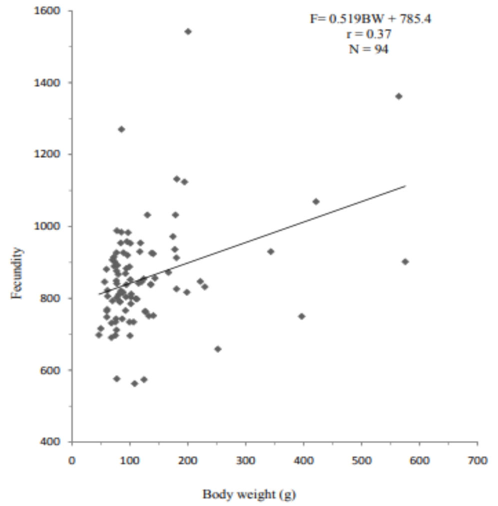

Figure 9. The relationship between fecundity and body weight of *O. niloticus* in the Weija Reservoir, Ghana

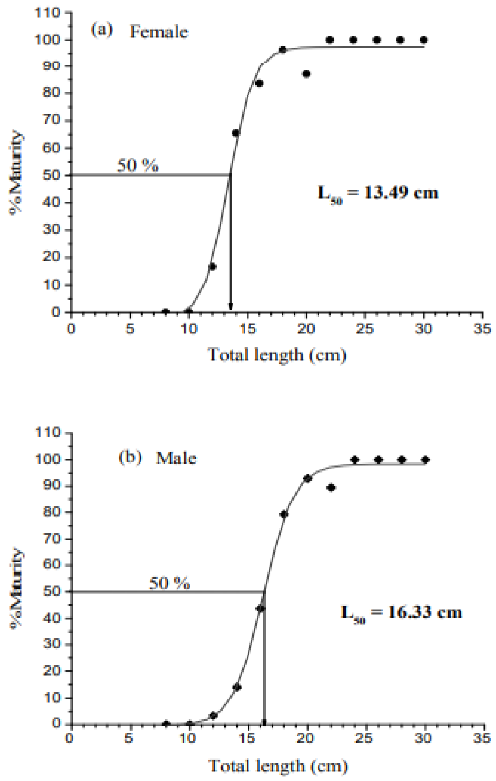

Figure 10. The length at first maturity (L_{50}) of (a) female and (b) male *O. niloticus* in the Weija Reservoir, Ghana (L_{50} = 13.49 cm and 16.33 cm for females and males respectively)

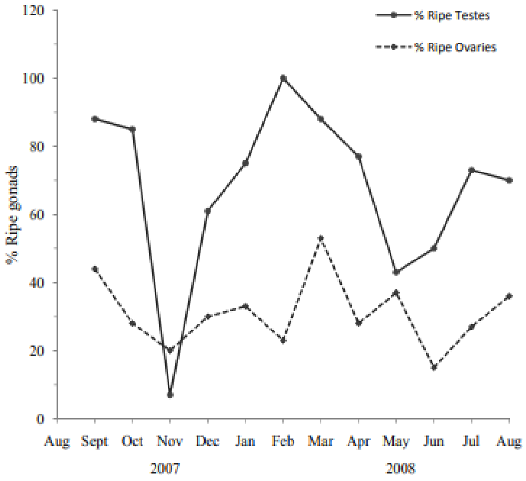

Figure 11. Monthly fluctuations in ripe gonads of *O. niloticus* in the Weija Reservoir, Ghana

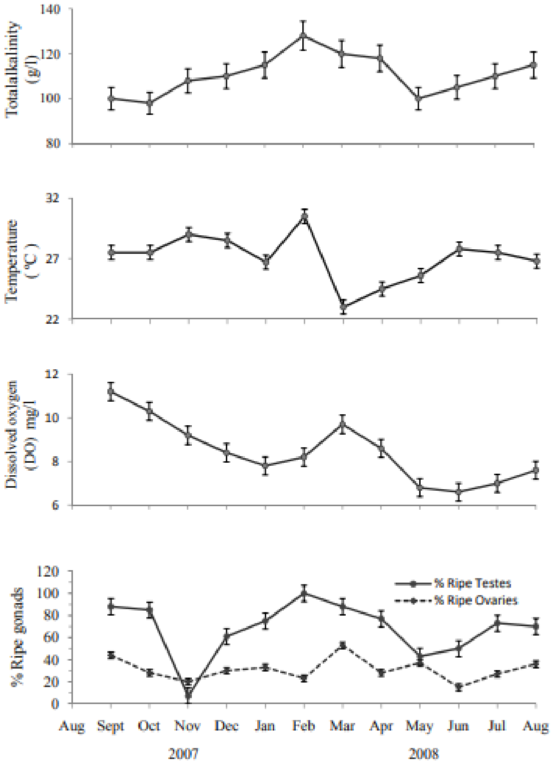

Figure 12. The relation between *O. niloticus* fluctuations in the ripe gonads of and some hydrographic factors (From: GWCL/AVRL-Weija, 2008) of the Weija Reservoir, Ghana (vertical bars = standard error)

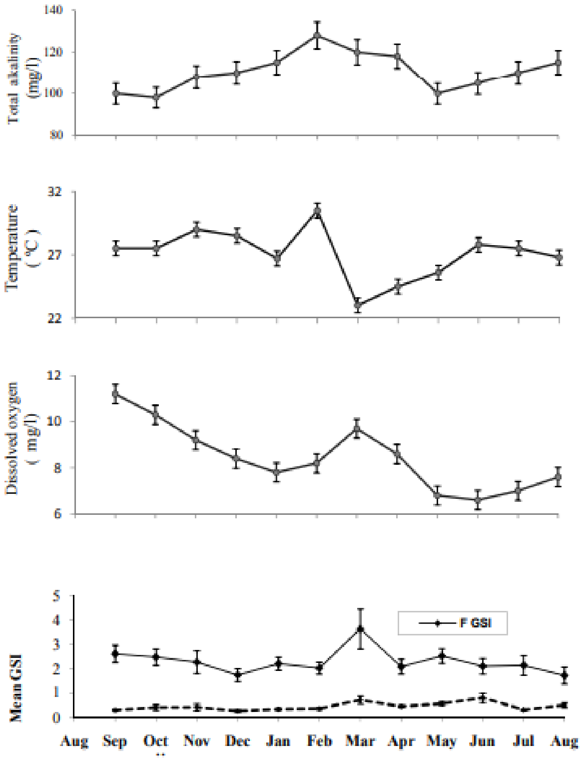

Figure 14. The relation between *O. niloticus* fluctuations in the gonado-somatic index (GSI) of females and males and some hydrographic factors (From GWCL/AVRL-Weija 2008) of the Weija Reservoir, Ghana (vertical bars = standard error)

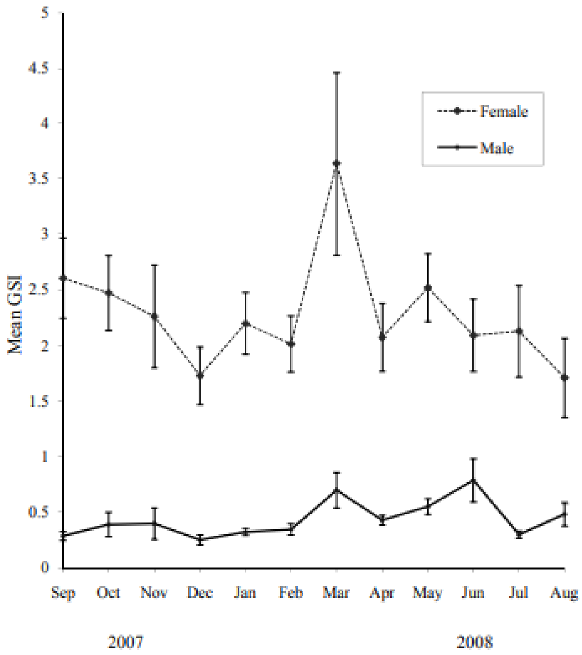

Figure 13. The fluctuations in the Gonado-Somatic Index (GSI) of female and male *O. niloticus* in the Weija Reservoir, Ghana (vertical bars = standard error)

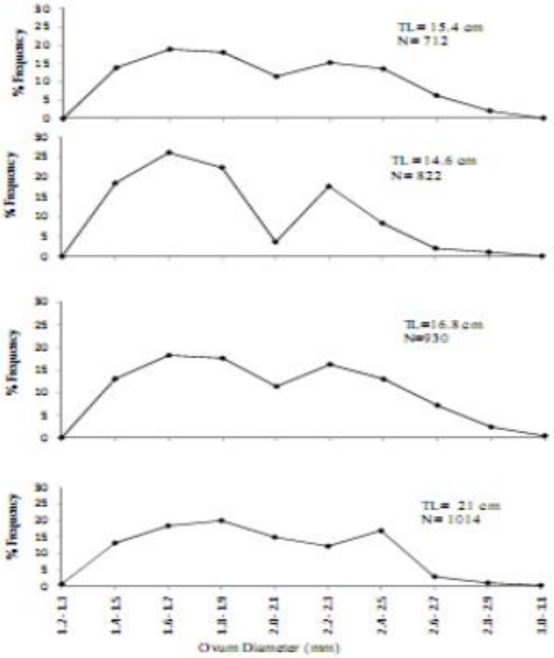

Figure 15. The frequency distribution of *O. niloticus* ovum diameter of four ripe ovaries in the Weija Reservoir, Ghana (N = total number of ova)

Discussion

The range of sizes of *O. niloticus* from the Weija Reservoir was observed at 7.0 to 33.3 cm TL, and this observation could be because of the fishermen's fishing method and the gear they used. Bwanika et al. (2004) reported size ranges of 4.9 to 29.5 cm and 5.0 to 26.0 cm TL of *O. niloticus* in Lake Nyamusingiri and Lake Nyamusingiri, respectively, in Uganda. The present study's wider size range may be because of better water environmental conditions and genetic factors, allowing the Weija population to grow relatively larger than those in the Ugandan lakes.

The *O. niloticus* length-weight relationship was exponential, as described by the relation BW = $0.0317SL^{3.0932}$ for the length-weight relationship of *O. niloticus* in the Weija Reservoir. That is a common phenomenon in fisheries (Pauly, 1993; King, 1996; Kariman and Hanan, 2008). The relationship on the curvilinear between the length and weight of the species is common among fishes (King 1996). the regression coefficient is 3.0 for an ideal fish that shows isometric growth. (Allen 1978; Bagenal and Tesch 1978; Dalzell 1987), and allometric growth exhibited in populations in which the exponent differs significantly from 3.0. Similarly, Olurin and Aderibigbe (2006) have been reported for pond-reared juvenile *O. niloticus* in Nigeria with a regression coefficient of 3.09, indicating isometric growth in the species. Njiru et al. (2006) observed positive allometric growth in both males and females of the species in Lake Victoria, Kenya. Their regression coefficient reported for the length-weight relationship was 3.22 and 3.32 for females and males, respectively. Barnes (2007) also reported fish ponds in the Sunyani district of Ghana to have negative allometric growth of reared Nile tilapia. The variation in the exponent of length-weight relationships within the same species could be due to different stages in the ontogenetic development, sex differences, and the differences in a geographical location with the associated environmental conditions. (Tudorancea et al. 1988; Kraljevic et al. 1996).

Furthermore, the condition factor studies describe a fish's health and general well-being as related to its environment; it represents how plumpy or healthy the fishes are (Reynold 1968). in the Weija Reservoir, the males of the *O. niloticus* population investigated were generally heavier than the females. However, some researchers have also reported the same similarity (Stone 1980; Behrends 1983; Eknath et al. 1993; Bentsen et al. 1998). Therefore, as suggested by Pagan (1970) and Tave (1980) in the present study, the difference in the growth of the sexes could be attributed to genetic differences between the males and the females observed. Therefore, the observed increase in the condition of Nile tilapia could be attributed to the development of gonad materials just before the breeding seasons. Moreover, the present study also exposes that the periods of better condition in the fish (October 2007 and February 2008) corresponded with months with reduced reproductive activities (see Figure 6). Lowe-McConnell (1958) also observed the similarity trend in *O. niloticus* in Lake Turkana, as did Moriarty and Moriarty (1973) in Lake George.

The high condition observed on factor value for males in October 2007 might be due to larger matured fish moving into the deeper part of the water after spawning from their spawning arena in the shallow areas, where the fishermen's net easily catches them spawning to avoid competition for food. The changing conditions of fishes could be used to interpret various biological features such as fatness, reproductive activities, food availability, and environmental health (Le Cren 1951; Dadzie et al. 2000).

Studies on visceral fish fat have also been used to explain the well-being of the fish (El-Sayed and Teshima 1991; Hanley 1991; Webster and Lim 2002). Most fish accumulate fat around the visceral organs during reduced reproductive activities and periods of intense feeding (Kwei 1966; Welcomme 1967). The main function of this fat is to store high-energy molecules, which could be utilized during periods of less-feeding activities as they channel their energy into parental care activities and spawning. Fat is digested and serves as a much better source of energy for protein sparing and metabolized with greater relative ease than carbohydrates (Welcomme 1967; El-Sayed 2006). During long periods, fat also provides energy for immediate physiological needs with higher energetic demands. In *O. niloticus*, the excess lipid is deposited mainly as visceral fat during months when there is a high food intake (Welcomme 1967; El-Sayed 2006; Njiru et al. 2006). The species can utilize lipid reserves for energy needs, especially in the females who fast when orally brooding their eggs and young ones (Love 1957). That is evident in the graph of the female fat index (see Figure 7), having several peaks in November 2007, January, March, June, and August 2008, while males have three peaks in September 2007, March, and August 2008. In March 2008, the highest value of 2.69 for males observed may be due to increased feeding. In April 2008, the sharp drop in visceral fat could be attributed to the protracted reproductive activities during the major breeding season in March 2008. Males were observed to have more visceral fat than females since they diverted less energy into reproductive activities (Welcomme 1967; El-Sayed 2006; Njiru et al. 2006). That is in agreement with the present study (see Figure 7), which probably could be a linkage between fat accumulation and spawning. That could be seen in the monthly fluctuation in the visceral fat index and GSI of the females, with peaks of these graphs coinciding in September 2007, January, and March 2008. Reproductive activities probably cause the fluctuation as the fish accumulate fat and become 'lean' after spawning because the females eat very little or no food when they are mouth-brooding their eggs and young and therefore utilize their fat reserves. During these periods, the increase in visceral fat after spawning indicates intense feeding.

The observed overall sex ratio in favor of males of 1:0.84 for the *O. niloticus* population caught in the Weija Reservoir during the study period was significantly different from the expected 1:1 (see Table 1), which indicates that males than females were relatively higher in number. Njiru et al. (2006) presented a similar observation for the species in Lake Victoria. However, other researchers (Gómez-Márquez et al. 2003; Peña-Mendoza et al. 2005) revealed contradictory observations where females were more in the populations. Fishery workers such as Barioller et al. (1995) associated the occurrence of unbalanced sex ratios with environmental influences such as temperature. The sex ratio variation may be because of the maternal brooding characteristic of the species. That is because once the fertilization of the eggs is completed, males, possibly due to differential sexual migration, leave the spawning areas for the feeding grounds located in the shallow part of the lake, where they are captured. At the same time, females go into submerged vegetation and rocky areas to carry out the oral brooding and protection of offspring and to avoid predators (including fishermen) (Peña-Mendoza et al. 2005; Offem et al. 2007). The movement of spawning *O. niloticus* from foraging to breeding grounds was also reported by Lowe-McConnel (1958) and Rinne and Wanjala (1982). During this time and fishing operations, differential migration of sexes could occur, resulting in more males than females being caught. Balirwa (1998) also observed that different habitats might favor one sex over the other.

In the present study, the cause of the skewed sex ratio in favor of the males might be genetic factors because the species shows differential growth between the sexes. That is supported by the observation of Fryer and Iles (1972) that, in African lakes, it is common for males to dominate because they generally exhibit faster growth than females causing the selection of males against the smaller females, which means that they obtain the size at which they are caught more quickly. In the present study, males grow to a relatively larger size as represented by the length at first maturity and length-frequency distribution of female and male *O. niloticus* (L_{50} = 13.49 cm for females and 16.33 cm for males, respectively, and modal class length TL= 16.0-16.9 cm for females and 19.0-19.9 cm for males, respectively). Furthermore, to some extent, the influence of environmental factors cannot be completely set aside. In the major breeding season on March 2008 observed that females dominated in the monthly sample, which could be because of bias method of fishing employed by the fishermen as they made their catches or set their nets near the breeding grounds in the submerged vegetation and rocky areas targeting the females carrying out oral brooding of their fertilized eggs or fry, that is less actively.

Studies on the fish's total number of ripe ova in the ovary (fecundity) are essential to estimate the reproductive potential of the species in the wild. In the present study, the fecundity of *O. niloticus* ranged between 563 to 1,542 ova for fish with a body weight of 108 g to 200 g and a total length of 15.6 cm to 21.5 cm. This fecundity is low compared to that reported elsewhere. Lowe-McConnel (1955) reported on the fecundity of the species, ranging from 340 to 3,706 eggs in various East African river waters. The fecundity in the range of 864 to 6,316, with an average of 2,141 eggs in the species in the Nyanza gulf of Lake Victoria, Kenya, was also reported by Lung'ayia (1992) (see Appendix 5). The dissimilar reports made by some researchers that there is a strong correlation between fecundity and body weight and fecundity and the total length (Lowe-McConnel 1955; Lung'ayi, 1994; Njiru et al. 2006; Kariman and Hanan 2008), the present study showed weak correlations between fecundity and body weight (r = 0.38), and fecundity and length (r = 0.40). That could be because the size range observed in the present study is smaller than those reported by other researchers (see Appendix 5). Njiru et al. (2006) observed a decreasing trend in the fecundity of Lake Victoria of *O. niloticus* due to overfishing. The higher fecundity could probably be attained if the population in the Weija Reservoir could grow larger by reducing the fishing pressure on the species.

The length at first maturity (L_{50}) of the fish population is a function of their size (Siddiqui et al. 1997; Mehanna 2007). That may be influenced by environmental factors such as abundance and seasonal food availability, predation, temperature (especially in temperate regions), photoperiods, and the locality. Siddiqui et al. (1997) stated that fish size is influenced by the feeding level, which affects their growth. Moreover, at first maturity (L_{50}), the length is also an essential management parameter that can be used to monitor whether enough juveniles in an exploited population mature and spawn (Mehanna 2007). In this study, the estimated L50 for female and male O. niloticus was 13.49 cm and 16.33, respectively. The male growth superiority might be because of genetic factors. Fryer and Iles (1972) stated that genetic factors give males a competitive advantage enabling them to have higher survival rates and larger sizes at sexual maturity. The L_{50} values for Coatetelco Lake in Mexico population seemed to be relatively lower than those of the Weija, where the length at maturity was 11.7 cm for females and 12.0 cm for males, respectively (Gómez-Márquez et al. 2003) and Bontanga reservoir (near Tamale in Ghana's Northern region) with an L_{50} value of 9.1 cm for both females and males (Kwarfo-Apegyah 2010). That is because environmental factors such as the Weija and hydrographic factors of the species in the Weija Reservoir are better than those of the Bontanga reservoir and Coatetelco Lake, where the species are stressed. Payne and Collinson (1983), however, studied the *O. niloticus* population in Lake Manzalah, Egypt, and examined L_{50} values of 16.3 cm for females and 17.4 cm for males, respectively, with the higher L_{50} values than those of the present study. According to Ofori-Danson (1999), the population of species in the Yeji area (Stratum VII) of the Volta Lake, Ghana, examined an L_{50} value of 19.73 cm SL. Ochumba and Manyala (1992) observed the *O. niloticus* population in the Sondumiru River, Kenya, even had higher L_{50} values of 31.5 and 27.5 cm for females and males. According to Balirwa (1998), Lake Kyoga also had L_{50} values of 26.0 cm and 23.0 cm for females and males, respectively, which follows with Eknath et al. (1993) they reported on wild strains of *O. niloticus* from Egypt and East Africa grew better than those from West Africa (Senegal and Ghana). On Lake Manzalah and Lake Coatetelco, the females *O. niloticus* in Lake Victoria appear to become sexually mature at a relatively larger size than the males, contrary to what has been observed in the Weija reservoir (Table 2).

Table 2. The maturity length of some populations of *O. niloticus*

Waterbody	Location	L_{50} of females (cm)	L_{50} of males (cm)	Source
Weija Reservoir	Ghana	13.49	16.33	Present study
Volta Lake at Stratum VII (the Yeji sector)	Ghana	19.73		Ofori-Danson (1999)
Bontanga reservoir (near Tamale)	Ghana	9.1		Kwarfo- Apegyah (2010)
Lake Coatetelco	Mexico	11.7	12.0	Gómez- Márquez *et al.* (2003)
Lake Manzalah	Egypt	16.3	17.4	Payne and Collinson (1983)
Lake Kyoga	Uganda	26.0	23.0	Balirwa (1998)
River Sondumiru	Kenya	27.5	31.5	Ochumba and Manyala (1992)

This observation can explain no specific reason, and the major contributing factors are probably genetic causes and better environmental conditions. The relatively small maturation sizes of *O. niloticus* in Bontanga reservoir and Coatetelco Lake, than those in Weija, Lake Kyoga, Sondumiru River, and Manzalah, may be indicating a relatively higher level of stunting of the population in Bontanga reservoir and Lake Coatetelco. The phenomenon of 'dwarfing' or 'stunting' in the tilapia population is well known and is one of the major problems in tilapia culture (Fryer and Iles 1969; Lorenzen 2000). Smaller size at maturity may serve as a strategy to maximize reproduction in a topographically restricted habitat, a response to intensive fishing, a response to a high level of competition, or unfavorable environmental conditions such as pollution. This observation follows the findings from Lowe-McConnell (1982) in several East African river waters. She suggested that *O. niloticus* will breed when younger and smaller in small water bodies such as crater lagoons, ponds, and lakes with stressful conditions and delayed maturation when inhabiting large lakes.

Analysis of fluctuations in percentage mature gonads revealed that peaks of percentage mature testes (see Figure 11) appeared to be a phase ahead of that of GSI. In contrast, that of mature ova coincided with GSI. Following the observations by El-Sayed (2006), Welcomme (1985), and Trewevas (1983), this is a strategy in Tilapia of the genera *Oreochromis* to get the males to increase the success of fertilization and get ready for spawning.

GSI is the percentage ratio of the gonad and body weight, and fish with ripe ovaries have higher GSI values. Therefore, the spawning activities of the fish are indicated by the fluctuation of GSI values; during major spawning periods, the value is high, and after spawning, the low values occur. Moreover, because the gonads of the females are generally heavier than the males, the GSI value for the females is higher than that of the males. In the present study, an indication of breeding throughout the years shown by the female *O. niloticus* population exhibited peaks in September 2007 and January, May, and July 2008. While for the females, the major reproductive activities of the males occurred from March to April 2008.

From the hydrographic factors of the water data obtained from GWCL/AVRL at Weija, based on analysis of GSI and occurrence in ripe gonads, it appeared the major spawning period of the *O. niloticus* population coincided with months with lower water temperature (below 24°C), dissolved oxygen and increasing concentrations of total alkalinity, which possibly corresponds to the major rainy period (see Figs. 12 & 14). This finding agrees that most wild cichlids respond to changes in these hydrographic factors, which serve as indicators that trigger their reproductive activities (Welcomme 1985; El-Sayed 2006). From this study, the O. niloticus population in the Weija Reservoir appears to synchronize their major breeding season with periods of favorable environmental conditions for rearing their young juveniles.

The fish species' frequency distribution of ovum diameter can be used to predict the spawning frequency of the fish. The female ovaries contain oocytes of different developmental stages in multiple-spawning fish species. While fish's oocyte distribution with short and definite spawning periods exhibits distinct ovum sizes, those of intermittent spawners do not show any distinct modes (Hickling and Rutenberg 1936). In the current study, the distribution shows two peaks that are not completely separated (Figure 14), indicating that *O. niloticus* in the Weija Reservoir might exhibit protracted spawning, which was possibly shedding the eggs in batches. Moreover, if suitable ambient environmental conditions are attained, the *O niloticus* can spawn several times a year (Welcomme 1985; El-Sayed 2006). Blay (1981) examined Sarotherodon galilaeus, a species-related tilapia, in restricted spawning activity in a small concrete pond. Therefore, the ability to spawn more than once may be a strategy to reduce the risk of wiping out a reproductive run in a season. Welcomme (1967) also reported the range of ova diameters within the range of 1.2 mm to 3.1 mm. Residual eggs that progress into atresia in post-spawned ovaries of the species have been reported (Babiker and Ibrahim 1979; El-Sayed 2006). After spawning, the tilapia ovaries regenerate immediately and rapidly and are recruited in as little as one week (Coward and Bromage, 2000).

In conclusion, the length-weight relationship for *O. niloticus* in the Weija Reservoir is described by the relation $BW = 0.0317\ SL^{3.0932}$ which shows that the Nile tilapia population in the Weija Reservoir is growing isometrically ($r = 3.09$). Examinations on the condition factor show that, generally, the males are plumpier than the females. The fish obtain their best condition in the months with reduced reproductive activities in February and October.

The Chi-square test result revealed that males of the Nile tilapia population in the Weija Reservoir significantly outnumber the females with a sex ratio of 1: 0.84 (P <

0.05). The GSI result and gonadal stages show that the species spawned around the year, and March was the major breeding period. In addition, the frequency distribution of ovum diameter exhibits protracted spawning shedding eggs in batches showing on the females *O. niloticus*. There is also a link between fat accumulation and spawning in females. The fish build up fat reserves before spawning and become lean. Moreover, because males divert less energy into reproductive activities, they have more visceral fat than females. The fish's major breeding period also coincided with total alkalinity concentrations of the reservoir, water temperature slightly below 24ºC, and increased dissolved oxygen.

The fish attains maturity beyond 10.00 cm TL. Fish of a total length greater than 18.00 cm are expected to be fully matured. Females mature at a shorter size (L_{50} = 13.49 cm) than males (L_{50}= 16.33 cm). In the Weija Reservoir, the fecundity of *O. niloticus* ranged between 563 to 1,542 eggs, corresponding to the fish's total length of 15.6-21.5 cm and weight of 108-200 g. There is a weak correlation between fecundity and body weight (r = 0.38) and fecundity and total length (r = 0.40).

REFERENCES

Allen KR. 1978. Some observation on the biology of the trout (*Salmo trutta*) in Windermare. J Anim Ecol 7: 333-349. DOI: 10.2307/1168.

Asante KA, Quarcoopome T, Amevenku FYK. 2006. Water quality of the Weija Reservoir after 28 years of impoundment. West Afr J Appl Ecol 13: 1-8. DOI: 10.4314/wajae.v13i1.40588.

Babiker MM, Ibrahim H. 1979. Studies on the biology of reproduction in the cichlid *Tilapia nilotica* (L): gonadal maturation and fecundity. J Fish Biol 14: 437-448. DOI: 10.1111/j.1095-8649.1979.tb03541.x.

Bagenal TB, Braum E. 1978. Eggs and early life history. In: Bagenal TB (Eds). Methods of Assessment of Fish Production in Fresh Water. Blackwell Scientific Publications, Oxford, London, Edinburgh, Melbourne.

Bagenal TB, Tesch FW. 1978. Age and growth. In: Bagenal TB (Eds). Methods for Assessment of Fish Production in Freshwaters 3rd ed. IBP Handbook, Blackwell, Oxford.

Balarin JD, Hatton JP. 1979. Tilapia-A Guide to their Biology and Culture in Africa. University of Stirling, Scotland.

Balirwa JS. 1998. Lake Victoria Wetlands and the Ecology of the Nile Tilapia, *Oreochromis niloticus* (L) [PhD Thesis]. University of Wageningen, A.A. Balkema, Rotterdam.

Barioller JF, Chourrout D, Fostier A, Jalabert B. 1995. Temperature and sex chromosomes govern sex ratios of the mouth brooding cichlid fish *Oreochromis niloticus*. J Exp Zool 271: 1-8.

Barnes F. 2007. Observations on some Environmental Factors and Growth Characteristics of the Nile Tilapia (*Oreochromis niloticus* L.) in Fish Ponds in Sunyani District, Ghana. [M.Phil Thesis]. University of Cape Coast, Ghana.

Behrends LL. 1983. Evaluation of Hatchery Techniques for Intraspecific and Interspecific Seed Production in Four Species of Tilapia. [Doctorial Dissertation]. Auburn University, Alabama, USA.

Bentsen HB, Eknath AE, Palada-de Vera MS, Danting JC, Bolivar HL, Reyes RA, Dionisio EE, Longalong FM, Circa AV, Taymen MM, Gjerde B. 1998. Genetic improvement of farmed tilapias: Growth performances in a complete diallel cross experiment with eight strains of *Oreochromis niloticus*. Aquaculture 160: 145-173. DOI: 10.1016/S0044-8486(97)00230-5.

Blay JJnr, Asabere-Ameyaw A. 1993. Assessment of the fishery of a stunted population of the cichlid, *Sarotherodon melanotheron* (Ruppel), in a closed lagoon in Ghana. J Appl Ichthyol 9: 1-11. DOI: 10.1111/j.1439-0426.1993.tb00382.x.

Blay JJnr. 1981. Fecundity and spawning frequency of *Sarotherodon galilaeus* in a concrete pond. Aquaculture 25: 95-99. DOI: 10.1016/0044-8486(81)90103-4.

Bwanika GN, Makanga B, Kizito Y, Chapman LJ, Balirwa J. 2004. Observation on the biology of Nile tilapia, *Oreochromis niloticus* L. In two Ugandan crater lakes. Afr J Ecol 42 (1): 93-101. DOI: 10.1111/j.1365-2028.2004.00468.x.

Coward K, Bromage NR. 2000. Reproductive physiology of female tilapia broodstock. Springer, Netherlands. Rev Fish Biol Fish 10: 1-25. DOI: 10.1023/A:1008942318272.

Dadzie S, Abou-Seedo F, Manyala JO. 2000. Length weight relationship and condition factor of Pampus argentus (Euphrasen 1788) in Kuwait waters. Kuwait J Sci Eng 26: 123-136.

Dalzell P. 1987. Notes on the biology of *Spratelloides leuvisi* (Wongratana, 1983) a recently described species of sprat from Pupa New Guinea waters. J Fish Biol 30: 691-700. DOI: 10.1111/j.1095-8649.1987.tb05798.x.

Eknath AE, Tayamen MM, Palada-de Vera MS, Danting JC, Reyes RA, Dionisio EE, Capili JB, Bolivar TA, Abella AV, Circa HB, Bentsen B, Gjerde T, Gjedrem, Pullin RSV. 1993. Genetic improvement of farmed tilapias: the growth performance of eight strains of *Oreochromis niloticus* tested in different farm environments. Aquaculture 111: 171-188. DOI: 10.1016/B978-0-444-81527-9.50021-X.

El-Sayed AM, Teshima SI. 1991. Tilapia nutrition in aquaculture. Rev Aquat Sci 5: 247-265. DOI: 10.1079/9780851990149.0000.

El-Sayed AM. 2006. Tilapia Culture. CABI Pub. Series Oxfordshire, UK.

FAO 2000. Fishery Statistics. FAO, Rome.

FAO 2005. Report of the FAO-Worldfish Centre On Small-Scale Agriculture. Revisiting the Aquaculture Target Group Paradigm. Limbé, Cameroon.

FAO 2006. National aquaculture sector overview Ghana. In: Fisheries Global Information System (FIGIS). FAO, Rome.

FAO. 1997. Aquaculture Development. FAO Technical Guidelines for Responsible Fisheries No. 5. FAO, Rome.

Fitzsimmons K. 2000. Tilapia: The most important aquaculture species of the 21st century. In: Fitzsimmons K, Fillho JC (Eds). Tilapia Aquaculture in the 21st Century Proceedings of the Fifth international Symposium on Tilapia in aquaculture 1: 320.

Frimpong MO, Attipoe FYK, Paddy JN. 1992. Development of Improved Culture Systems for Increased Pond Production. Pond Culture (Ghana). IAB/IDRC-3.

Fryer G, Iles TD. 1969. Alternative routes to evolutionary success as exhibited by African cichlid fishes of the Genus *Tilapia* and the species flocks of the Great Lakes. Evolution 23: 359-369. DOI: 10.1111/j.1558-5646.1969.tb03520.x.

Fryer G, Iles TD. 1972. The Cichlid Fishes of the Great Lakes of Africa: Their Biology and Evolution. Oliver and Boyd, Edinburgh, Scotland.

Garibaldi L. 1996. List of animal species used in aquaculture. FAO Fisheries Circular. In: Abban EK, Casal CMV, Dugan P, Falk TM. (Eds.) 2004. Biodiversity, Management and Utilization of West African Fishes. WordFish Centre Conference Proceedings, Penang Malasia.

Ghana Survey Department 2008. Map of Weija dam area showing the reservoir. Government of Ghana, Accra.

Gómez-Márquez JL, Peña-Mendoza B, Salgado-Ugarte IH, Guzmán-Arroyo M. 2003. Reproductive aspects of *Oreochromis niloticus* (Perciformes: Cichlidae) at Coatetelco lake, Morelos, Mexico. Revista de Biologia Trop 51: 221-228.

Gordon C. 2006. Background Paper for the Multi-Stakeholder Consultation Process for Dams' Development in Ghana. Volta Basin Research Project. University of Ghana, Ghana.

Hanley F. 1991. Effects of feeding supplementary diets containing varying levels of lipid on growth, food conversion, and body composition of Nile tilapia *Oreochromis niloticus* (L.). Aquaculture 93: 323-334. DOI: 10.1016/0044-8486(91)90224-U.

Hickling CF, Rutenberg E. 1936. The ovary as an indicator of spawning period in fishes. J Mar Biol Assoc UK 21: 311-317. DOI: 10.1017/S0025315400011322.

Kariman ASS, Hanan MMS. 2008. Investigations on some aspects of reproductive biology in *Oreochromis niloticus* (Linnaeus, 1757) Inhabited Abu-zabal Lake, Egypt. Glob Vet 2 (6): 351-359.

King RP. 1996. Length-weight relationship of Nigerian coastal water fishes. NAGA-ICLARM 19: 53-58.

Kraljevic M, Dulcuic J, Cetinic P, Pallaoro A. 1996. Age, growth and mortality of the striped sea bream, *Lithognathus mormyrus* L. in the northern Adriatic. Fish Res 28: 361-370. DOI: 10.1016/S0165-7836(96)00518-8.

Kwarfo-Apegyah K. 2010. Stock Assessment of Major Fish Species in Bontanga Reservoir, Northern Ghana, for Sustainable Management. [Thesis]. University of Ghana, Legon.

Kwei EA. 1966. The fat cycle in the sardine (*Sardinella aurita* C & V) in Ghana waters. Proceedings of the Symposium on the Oceanography and Fisheries Resources of Tropical Atlantic. UNESCO/FAO/OAU.

Kwei EA. 1970. The migration and biology of the Spanish mackerel, *Scomber japonicas* (Houttyn). Ghana J Sci 11 (2): 75-86.

Le Cren ED. 1951. The length weight relationship and season cycle in gonad weight and condition in Perch (*Perca fluviatilis*). J Anim Ecol 20: 179-219. DOI: 10.2307/1540.

Lorenzen K. 2000. Population dynamics and management. In: Beveridge MCM, Mcandrew BJ (Eds). Tilapias: Biology and Exploitation. Kluwer Academic Publishers, Dordrecht, the Netherlands. DOI: 10.1007/978-94-011-4008-9_6.

Love R. 1957. The biochemical composition of fish. In: The physiology of fishes. Academic Press Inc., New York. DOI: 10.1016/B978-1-4832-2817-4.50016-0.

Lovshin LL. 1997. Worldwide tilapia culture. In: Anais do I Workshop International de Aquicultura, Sao Paulo, Brazil.

Lowe-McConnell RH. 1982. Tilapias in fish communities. Proceedings of the International Conference on the Biology and Culture of Tilapias 1: 83-113.

Lowe-McCornnell RH. 1955. The fecundity of tilapia species. East Afr Agric J 11: 45-52. DOI: 10.1080/03670074.1955.11665004.

Lowe-McCornnell RH. 1958. Observations on the biology of *Tilapia nilotica* L in East African water. Revue de la Zoologie et de Botanique Africaines 57: 129-170.

Lung'aiya HBO. 1994. Some aspects of the reproductive biology of the Nile tilapia, *Oreochromis niloticus* (L.) in the Nyanza Gulf of Lake Victoria, Kenya. In: Okemwa E, Wakwabi E, Getabu A (Eds). Recent trends of research on Lake Victoria Fisheries. ICIPE Science Press, Kenya.

Lung'ayia HBO. 1992. Some Aspects of the Reproductive Biology of the Nile Tilapia *Oreochromis niioticas* (L) in the Nyanza Gulf of Lake Victoria, Kenya. Kenya Marine and Fisheries Research Institute (KMFRI), Kisumu.

Njiru M, Ojuok JE, Okeyo-Owuor JB, Muchiri M, Ntiba MJ, Cowx IG. 2006. Some biological aspects and life history strategies of Nile tilapia *Oreochromis niloticus* (L.) in Lake Victoria, Kenya. Afr J Ecol 44: 30-37. DOI: 10.1111/j.1365-2028.2006.00610.x.

Ochumba PBO, Manyala JP. 1992. Distribution of fishes along the Sondumiru River of Lake Victoria, Kenya with special reference to upstream migration biology and yield. Aquacult Fish Manag 23: 701-719. DOI: 10.1111/j.1365-2109.1992.tb00813.x.

Offem BO, Akegbejo-Samsons Y, Omoniyi IT. 2007. Biological assessment of *Oreochromis niloticus* (Pisces: Cichlidae; Linne, 1958) in a tropical floodplain river. Afr J Biotechnol 6: 1966-1971. DOI: 10.5897/AJB2007.000-2300.

Ofori-Danson PK. 1999. Stock Assessment of the five Major Commercial Fish Species in the Yeji Area (Stratum VII) of the Volta Lake. [Phd Thesis]. University of Ghana, Legon.

Olurin KB, Aderibigbe OA. 2006. Length-weight relationship and condition factor of pond reared juvenile *Oreochromis niloticus.* World J Zool 1 (2): 82-85.

Pagan FA. 1970. Cage Culture of the Cichlid Fish *Tilapia aurea* (Steindachner). [Doctoral Dissertation]. Auburn University, Alabama, USA.

Pauly D. 1976. The biology, fishery and potential for aquaculture of *Tilapia melanotheron* in a small West African lagoon. Aquaculture 7: 33-49. DOI: 10.1016/0044-8486(76)90030-2.

Pauly D. 1993. Some Simple Methods for the Assessment of Tropical Fish Stocks. FAO Fish Tech. Pap. No. 234. FAO, Rome.

Payne AI, Collinson RI. 1983. A comparison of the biological characteristics of *Sarotherodon niloticus* (L) with those of *S. aureus* (Steindachner) and other *Tilapia* of the Delta and lower Nile. Aquaculture 30: 335-351. DOI: 10.1016/0044-8486(83)90174-6.

Peña-Mendoza B, Gómez-Márquez JL, Salgado-Ugarte IH, Ramirez-Noguera D. 2005. Reproductive biology of *Oreochromis niloticus* (Perciformes : Cichlidae) at Emiliano Zapata dam, Morelos, Mexico. Intl J Revista de Biologia Trop 53: 515-522. DOI: 10.15517/rbt.v53i3-4.14666.

Pullin RSV, Eknath AE, Gjedrum T, Tayamen MM, Macaranas JM, Abella TA. 1991. The Genetic Improvement of Farmed *Tilapias* (GIFT) project. The story so far. NAGA. ICLARM Quarterly 14 (2): 7-9.

Pullin RSV. 1985. Tilapias: Everyman's fish. Biologist 32 (2): 84-92.

Reynold TD. 1968. The biology of the clupeids in the New Volta. In: Man-Made Lakes. The Accra Symposium. Ghana University Press, Accra.

Rinne JN, Wanjala B. 1982. Observations on movement patterns of *Tilapia* spp. in Nyanza Gulf, Lake Victoria, East Africa. J Fish Biol 20: 317-322. DOI: 10.1111/j.1095-8649.1982.tb04713.x.

Siddiqui AD. 1977. Reproductive biology, length-weight relationship and relative condition of *Tilapia leucostica* (Trewavas) in Lake Navaishe Kenya. J Fish Biol 10: 251-260. DOI: 10.1111/j.1095-8649.1977.tb05130.x.

Siddiqui AQ, Al-Harbi AH, Al-Hafedh YS. 1997. Effects of food supply on size at first maturity, fecundity and growth of hybrid *Tilapia*, *Oreochromis niloticus* (L) × *Oreochromis aureus* (Steindachner), in outdoor concrete tanks in Saudia Arabia. Aquat Res 28: 341-349. DOI: 10.1111/j.1365-2109.1997.tb01051.x.

Silva SS. 1985. Body condition and nutritional ecology of *Oreochromis mossambicus* (Pisces, Cichlidae) populations of man-made lakes in Sri-Lanka. J Fish Biol 27: 621-633. DOI: 10.1111/j.1095-8649.1985.tb03207.x.

Stone NM. 1980. Growth of Male and Female *Tilapia nilotica* in Ponds and Cages. [Masters Thesis]. Auburn University, Alabama, USA.

Tacon AGJ. 2001. Increasing the contribution of aquaculture for food security and poverty alleviation. In: Subasinghe RP, Bueno P, Phillips MJ, Hough C, McGladdery SE (Eds). Aquaculture in the Third Millennium. Technical Proceedings of the Conference on Aquaculture in the Third Millennium, Bangkok, Thailand, 67–77 pp.

Tave D. 1980. Genetic and breeding of *Tilapia*: A review. In: Pullin RSV, Bhukaswan T, Tonguthai K, Maclean JL (Eds.) The second international symposium on Tilapia in aquaculture. ICLARM conference proceedings 15. Department of Fisheries, Bangkok, Thailand, and International Centre for Living Aquatic Resources Management. Manila, Philippines.

Trewevas E. 1983. Tilapiine Fishes of the Genera *Sarotherodon*, *Oreochromis* and *Danakilia*. British Museum of Natural History, Publ. Num. 878. Comstock Publishing Associates, Ithaca, New York. DOI: 10.5962/bhl.title.123198.

Tudorancea C, Fernando CH, Paggi JC. 1988. Food and feeding ecology of *Oreochromis niloticus* (L.) juveniles in Lake Awassa (Ethiopia). Arch Hydrobiol 79: 267-289.

Vanden-Bossche JP, Bernacsek GM. 1990. Source book for the inland fishery resources of Africa, Vol 1. CIFA Technical Paper No. 18/1. 240.

Watanabe WO, Wicklund RI, Olla BL. 1992. Saltwater culture of Florida Red *Tilapia*: A summary of research at the Caribbean Research Centre. In: Al-thobaiti SA, Al-Hinty HM, Siddiqui AQ, Hussain G (Eds). Aquaculture-Technology and Investment Opportunities. Proceedings of the First International Symposium on Aquaculture Technology and Investment Opportunities. Ministry of Agriculture and Water, King Abdulaziz City for Science and Technology and the Riyadh Chambers of Commerce and Industry, Riyadh, Saudi Arabia.

Webster CD, Lim CE. 2002. Nutrient Requirements and Feeding of Finfish for Aquaculture. CABI Publishing Series, Wallingford, UK. DOI: 10.1079/9780851995199.0000.

Welcomme RL. 1967. Observations on the biology of introduced species of Tilapia in Lake Victoria. Rev Zool Bot Afr 76: 249-279.

Welcomme RL. 1985. River fisheries. FAO Tech. Pap. 262: 330 pp. In: Wootton RJ. Fish Ecology-(Tertiary Level Biology). Chapman and Hall, New York, USA.

Witte F, Van Densen WLT. 1995. Fish Stocks and Fisheries of Lake Victoria. A Handbook for Field Observations. Cardigan, Britain, Samara Publishing, Cardigan UK.