OPEN ACCESS

February 2023
ISSN 1996-0786
DOI: 10.5897/AJEST
www.academicjournals.org

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Vol. 17(2), pp. 45-50, February 2023
DOI: 10.5897/AJEST2021.3024
Article Number: 863121370533
ISSN: 1996-0786

http://www.academicjournals.org/AJEST

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African Journal of Environmental Science and Technology

Full Length Research Paper

Pollution assessment of light rare earth metals in sediments of the Ogun River

Diayi M. A.[1], Gbadebo A. M.[1*], Arowolo T. A.[1] and Awomeso J. A.[2]

[1]Department of Environmental Management and Toxicology, Federal University of Agriculture, Abeokuta, P.M.B. 2240, Abeokuta 110001, Nigeria.
[2]Department of Water Resources Management and Agrometeorology, Federal University of Agriculture, Abeokuta 110001, Nigeria.

Received 19 May, 2021; Accepted 13 September, 2021

This research was aimed at determining the concentrations of light earth metals in sediments of the Ogun River as well as their pollution and contamination status. Sediment samples collected using a graduated hand held sediment grab were stored in polythene bags and transported to the laboratory in an ice chested cooler. At the laboratory, the sediment samples were air dried, pulverized and sieved. Induced Couple Plasma Mass Spectrometry (ICP/MS) was used to determine the concentrations of light rare earth metals in the samples. Results obtained showed that the Cs values obtained which ranged from 21.10 to 121.40 mg kg^{-1} were higher than the Average Shale Value (ASV) of 5.00 mg kg^{-1}. Similarly, La ranged from 47.60 to 56.20 mg kg^{-1}; a value which exceeded the ASV value of 43.00 mg kg^{-1}. Values obtained were subjected to Geochemical Pollution Intensity (Igeo) and Contamination Factor (CF) analyses where all Cs values were seen to be very strongly contaminated while Igeo values for Cs which ranged from 3.20 to 4.00 showed highly polluted status. Cs is of great concern to sediment quality in areas surveyed. Likewise, light rare earth metals from Arakanga New Scheme (dam area) and Lafenwa (residential area) should be closely monitored to avoid adverse effects on aquatic or human lives due to high values observed.

Key words: Light rare earth metals, sediment, river, concentration, contamination.

INTRODUCTION

Sediments can be defined as loose sand, clay, silt and other soil particles that are deposited at the bottom of water bodies or accumulated at depositional sites. Their emergence can likely be from bedrocks, erosion, deposition of plants and animals or even soil (Andem et al., 2015). Through these processes as well as anthropogenic factors like industrial emissions and indiscriminate dumping of refuse, metals can be released into water bodies. These metals when in aquatic systems settle as bottom sediments. Metals can be considered as serious pollutants of the aquatic ecosystem due to their environmental persistence, toxicity and ability to be incorporated into food chains (Mashiatullah et al., 2011). Metals have toxic properties which results in deleterious

*Corresponding author. E-mail: dmanslem06@yahoo.com. Tel: +234-8063517259.

health effects on humans and ecosystem (Mahfuza et al., 2012).

Rare earth metals have been utilized in a number of industrial, medical and agricultural or zootechnical applications due to their specific properties such as high electrical conductivity and high luster capabilities. Fertilizers which are enriched with rare earth metals are released into cultivated lands from where they can find their way into water bodies and eventually, sediments due to run-off (Zhuang et al., 2017). Additionally, rare earth metals in water bodies like rivers may be due to mining activities.

Rare earth elements in water and sediments enter into humans through multiple exposure pathways majorly via food ingestion. The environmental impact of rare earth elements mining has been associated with bioaccumulation (Thomsen, 2016). Continuous exposure to low levels of rare earth elements on human health have been raising concerns because they tend to accumulate in the blood, brain and bone after entering the body (Zaichick et al., 2011). Long term exposure to rare earth elements could lead to health issues like changes in the brain and bone. It could also result in decreased Intelligence Quotient (IQ) and memory loss when children are exposed to them (Zhuang et al., 2017).

Additional implications of rare earth elements extraction and refining activities which make the activity a major environmental concern is the fact that strong acids are used at various stages of ore processing and refining. This results in the release of acidic effluents which can affect downstream water bodies (USEPA, 2012).

The Ogun River basically serves as means of water supply (for drinking and domestic purposes) to the people of Ogun State, South West Nigeria, with an estimated population of 5,217,716 (Wikipedia, 2021). Fishes from this river are also consumed by the residents and others from neighboring states. Since rare earth metals settle in bottom river sediments and can be released into water bodies which have the potential to increase concentrations of the metals in the environment, it is important to assess their levels in the Ogun River sediment so that information generated can serve as a basis for monitoring. Additionally, information on this area of study/research is limited therefore the study was embarked on to provide significant contribution to knowledge on rare earth metal levels (in the Ogun River sediments) which would be useful in policy/decision making on areas identified as culpable sources of release of these metals into the water body.

MATERIALS AND METHODS

Sampling area

The sampling locations include: Ikere-gorge, Iseyin, Olokemeji, Ofiki, Opeki, Iganagan, Oyan, Lafenwa, Arakanga Old Scheme, Arakanga New Scheme, Mokoloki, Oke-Oko and Kara (Figure 1). Using a calibrated sediment grab, sediment samples were collected from 13 locations along the Ogun River course namely: Ikere-gorge, Iseyin, Olokemeji, Ofiki, Opeki, Iganagan, Oyan, Lafenwa, Arakanga Old Scheme, Arakanga New Scheme, Mokoloki, Oke-Oko and Kara between January and March, 2013. Samples were placed in polythene bags after which they were transported to the laboratory via ice chested coolers. The sediment samples were then sun dried to remove moisture, sieved using 2 mm sieve and pulverized in order to obtain finer particles. Three sediments samples were collected per location for analysis. Sediment metals were determined using the ICP/MS technique.

Determination of metal concentration

A 0.5 g sample (sediment) was digested in aqua regia at 90°C in a controlled microprocessor digestion block for 2 h. Digested samples were diluted and analyzed by Perkin Elmer Sciex Elan 9000 ICP/MS. Based on the ICP/MS technique, sediment samples were introduced into argon plasma as aerosol droplets. The plasma dried the aerosol, dissociated the molecules, and then removed an electron from the components, thereby forming singly-charged ions, which were directed into a mass filtering device known as the mass spectrometer. A quadrupole mass spectrometer which rapidly scans the mass range was then used. At any given time, only one mass-to-charge ratio was allowed to pass through the mass spectrometer from the entrance to the exit. Upon exiting the mass spectrometer, ions stroke the first dynode of an electron multiplier, which served as a detector. The impact of the ions released a cascade of electrons, which were amplified until they became a measureable pulse. The software compared the intensities of the measured pulses to those from standards, which made up the calibration curve, to determine the concentration of the element. The ICP/MS analysis was carried out at Actlab, Canada.

Data generated were subjected to Analysis of Variance (ANOVA) while various means obtained were separated by Duncan Multiple Range Test using Statistical Package for the Social Sciences version 20. Other analysis carried out using generated data includes: Geochemical Pollution Index analysis was determined using Muller (1979) and mathematically expressed as:

$$\mathrm{Igeo} = \log_2\left(\frac{\mathrm{Cn}}{1.5\mathrm{Bn}}\right) \tag{1}$$

where Cn represents heavy metal concentration in sediment of the study area, Bn is the geochemical background value of element of interest in average shale (background value) and 1.5 is the background matrix correction factor due to lithogenic effects. This index is classified into various classes for easy identification of sediment pollution status. If the value obtained falls within 0-1, the pollution intensity is termed unpolluted while 1-2 is classified as moderately polluted to unpolluted, 2-3 is moderately unpolluted, 3-4 is moderately to highly polluted, 4-5 is highly polluted while 5 and above is very highly polluted.

Contamination factor analysis was determined using the formula:

$$\mathrm{CF} = \frac{(\mathrm{Cheavy\ metal})}{(\mathrm{Cbackground})} \tag{2}$$

CF stands for the Contamination Factor while $C_{\text{heavy metal}}$ stands for the concentration of the heavy metal in a location and $C_{\text{background}}$ stands for the background concentration (in this case, the world shale value of the element). Based on this, contamination levels were classified on their intensities on a scale ranging from 1-6, where 0= none, 1= none to medium, 2 = moderate, 3 = moderate to strong, 4 = strongly polluted, 5 = strong to very strong, 6 = very strong.

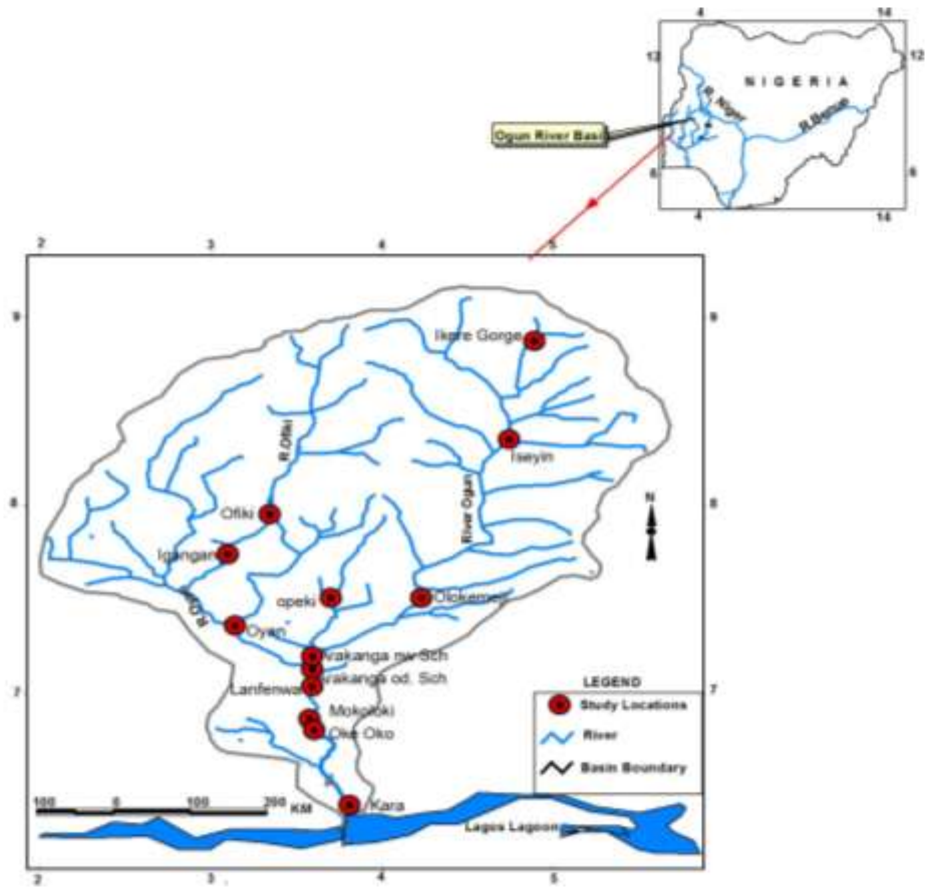

Figure 1. Map of the Ogun River.
Source: Author

Quality assurance

Three samples were taken per location for analysis while glasswares used were soaked in a 1% HCl solution after which they were washed using Alconnox. The wares were then rinsed 5-6 times with distilled water and oven dried at 160°C for proper sterilization. A series of standard reference materials used at the Activation Laboratory for sediment metal concentration analysis include: GXR-1 Meas, GXR-1 Cert, GXR-2 Meas, DH-1a Meas, DH-1a Cert, GXR- 4 Meas, GXR-4 Cert, GXR-6 Meas, GXR-6 Cert, SAR-M (U.S.G.S) Meas and SAR-M (U.S.G.S) Cert, while TILL-1 Meas, TILL-1 Cert, TILL-2 Meas and TILL-2.

RESULTS

Values obtained for Lanthanum (La) analysis showed that the highest concentration of the metal was obtained at Arakanga New Scheme (56.20 mg kg^{-1}), while the least value of 9.00 mg kg^{-1} was obtained at Igangan (Table 1). When compared with the average shale value standard of 43.00 mg kg^{-1}, La values from Lafenwa (47.60 mg kg^{-1}), Ofiki (48.60 mg kg^{-1}), and Arakanga New Scheme (56.20 mg kg^{-1}) exceeded the Average Shale Value (ASV). When subjected to the geochemical pollution intensity (Igeo) analysis, all values obtained for La ranged from -2.80 to -0.20 (Table 2). Contamination Factor (CF) values obtained from Ikere, Iseyin, Olokemeji, Opeki, Igangan, Arakanga Old Scheme, Mokoloki, Oke-Oko and Kara were less than 1 while Lafenwa, Ofiki and Arakanga New Scheme had values that fell between 1 and 2 (Table 3).

The Caesium (Cs) analysis carried out showed a range of 21.10 mg kg^{-1} (Iganagan) to 121.40 mg kg^{-1} (Arakanga New Scheme) (Table 2) which exceeded the ASV standard of 5.00 mg kg^{-1} (Table 1). Igeo values ranged from 1.50 (Igangan) to 4.0 (Arakanga New Scheme) while CF values ranged from 4.20 to 24.30 (Table 3). All Praseodynium (Pr) values obtained were lower than the ASV standard of 9.80 mg kg^{-1} except for those from Ofiki (11.90 mg kg^{-1}), Oyan (10.10 mg kg^{-1}), Lafenwa (12.00 mg kg^{-1}) and Arakanga New Scheme (15.30 mg kg^{-1}) (Table 1).

Neodymuim (Nd) values obtained ranged from 7.10 mg kg^{-1} (Iganagan) to 54.20 mg kg^{-1} Arakanga New Scheme (Table 1). Values which exceeded the ASV standard value of 33.00 mg kg^{-1} were obtained from Ofiki (41.40 mg kg^{-1}), Oyan (35.20 mg kg^{-1}), Lafenwa (42.70 mg kg^{-1}), Arakanga Old Scheme (33.80 mg kg^{-1}), Mokoloki (33.70 mg kg^{-}1) and Arakanga New Scheme (54.20 mg kg^{-1}) (Table 1).

Table 1. Mean and standard deviation of light rare earth element from the Ogun River .

Location	La	Cs	Pr	Nd	Sm	Eu	Gd
	mg kg^{-1}						
Ikere	20.40 ± 12.00	52.30 ± 14.40	4.80 ± 2.95	17.40 ± 10.85	3.20± 2.10	0.40 ± 0.30	2.80 ± 1.80
Iseyin	15.00 ± 12.80	55.10 ± 11.30	3.50 ± 3.10	12.50 ± 11.20	2.20 ± 2.20	0.30 ± 0.20	1.90 ± 1.60
Olokemeji	14.20 ± 2.90	32.70 ± 7.20	3.20 ± 0.60	11.60 ± 2.30	2.00 ± 0.40	0.30 ± 0.10	1.60 ± 0.30
Opeki	36.00 ± 1.40	99.70 ± 1.10	8.70 ± 0.30	30.20 ± 1.20	4.00 ± 0.20	0.50 ± 0.10	2.80 ± 0.15
Ofiki	48.60 ± 7.50	100.00 ± 12.50	11.90 ± 1.80	41.40 ± 6.00	6.20 ± 0.85	0.50 ± 0.10	4.30 ± 0.60
Igangan	9.00 ± 1.80	21.10 ± 3.80	2.00 ± 0.40	7.10 ± 1.69	1.00 ± 0.20	0.20 ± 0.00	0.80 ± 0.15
Oyan	42.50 ± 11.30	100.00 ± 25.40	10.10 ± 3.00	35.20± 10.5	5.50 ± 1.60	0.40 ± 0.10	3.60 ± 0.90
Lafenwa	47.60 ± 8.10	105.10 ± 17.85	12.00 ± 2.10	42.70 ± 7.60	6.70 ± 1.05	0.60 ± 0.10	4.70 ± 0.75
Arakanga Old Scheme	40.50 ± 9.30	83.10 ± 15.20	9.60 ± 2.30	33.80 ± 7.90	5.30 ± 1.00	0.60 ± 0.10	3.80 ± 0.55
Arakanga New Scheme	56.20 ± 27.60	121.40 ± 60.95	15.30 ± 8.15	54.20 ± 29.60	8.10 ± 4.30	0.40 ± 0.20	5.10 ± 2.70
Mokoloki	38.40 ± 8.80	79.30 ± 9.80	9.40 ± 2.30	33.70 ± 8.60	5.60 ±1.50	1.00 ± 0.35	4.4 ± 1.25
Oke Oko	32.70 ± 1.90	69.90 ± 4.30	7.90 ± 0.50	27.8 ± 1.95	4.60 ± 0.25	0.60 ± 0.10	3.40 ± 0.25
Kara	19.90 ± 2.30	27.40 ± 20.00	4.80 ± 0.60	16.7 ± 2.20	2.70 ±0.30	0.20 ± 0.10	2.00 ± 0.25
ASV	43.00	5.00	9.80	33.00	6.20	1.20	5.10
ECSQG	NS	NS	NS	NS	NS	NS	NS

ASV- Average Shale Value, ECSQGS – Environment Canada Sediment Quality Guideline Standard, NS – Not Detected.
Source: Author

Table 2. Geochemical pollution index values (Igeo) of light rare earth element in sediments samples from the Ogun River.

Location	La	Cs	Pr	Nd	Sm	Eu	Gd
Ikere	-1.70	2.80	-1.60	-1.50	-1.50	-2.20	-1.70
Iseyin	-2.12	2.90	-2.11	-2.00	-2.10	-2.60	-2.30
Olokemeji	-2.20	2.10	-2.30	-2.10	-2.20	-2.60	-2.50
Opeki	-0.80	3.70	-0.70	-0.70	-1.20	-1.83	-1.70
Ofiki	-0.42	3.70	-0.30	-0.25	-0.60	-1.83	-1.10
Igangan	-2.80	1.50	-2.80	-2.80	-3.20	-3.20	-3.50
Oyan	-0.60	3.70	-0.50	-0.49	-0.80	-2.20	-1.10
Lafenwa	-0.40	3.80	-0.30	-0.10	-0.50	-1.60	-1.00
Arakanga Old Scheme	-0.70	3.47	-0.60	-0.60	-8.80	-1.60	-1.30
Arakanga New Scheme	-0.20	4.00	0.50	0.10	-0.20	-2.20	-0.80
Mokoloki	-0.70	3.40	-0.60	-0.60	-0.70	-0.80	-1.10
Oke Oko	-1.00	3.20	-1.60	-0.25	-1.00	-1.60	-1.40
Kara	-1.70	1.90	-0.90	-1.60	-1.80	-3.20	-2.20
Average Shale Value	43.00	5.00	9.80	33.00	6.20	1.20	5.1

Source: Author

All Europium (Eu) and Gadolinium (Gd) values were below the ASV standards of 1.20 and 5.1, respectively except for Gd value of 5.1 from Arakanga New Scheme which was of same value with the standard (Table 1).

Pr values obtained from the research ranged from 2.00 to 15.30 mg kg^{-1} (Table 1), while Igeo values ranged from -2.80 (Igangan) to 0.50 (Arakanga New Scheme) (Table 2) with CF values ranging from 0.20 (Igangan) to 1.60 (Arakanga New Scheme) (Table 3). Similarly, Nd values ranged from 7.10 to 54.20 mg kg^{-1} with the least value from Igangan and highest from Arakanga New Scheme (Table 1). Igeo and CF values for Nd ranged from -2.80 to 0.10 (Table 2) and 0.20 to 1.60 (Table 3), respectively. Igangan and Arakanga New Scheme had the least and highest values, respectively for Igeo and CF, respectively. All Eu and Gd values obtained were not higher than the ASV standards of 1.20 and 5.10, respectively.

DISCUSSION

With La values from Lafenwa (47.60 mg kg^{-1}), Ofiki (48.60 mg kg^{-1}), and Arakanga New Scheme (56.20 mg

Table 3. Contamination factor values of light rare earth element in sediments samples from the Ogun River.

Location	La	Cs	Pr	Nd	Sm	Eu	Gd
Ikere	0.50	10.50	0.50	0.50	0.50	0.30	0.55
Iseyin	0.35	11.00	0.40	0.40	0.35	0.25	0.40
Olokemeji	0.30	6.50	0.30	0.35	0.30	0.25	0.30
Opeki	0.80	19.90	0.90	0.90	0.65	0.40	0.55
Ofiki	1.10	20.00	1.20	1.30	1.00	0.40	0.84
Igangan	0.20	4.20	0.20	0.20	0.10	0.20	0.20
Oyan	1.00	20.00	1.00	1.10	0.90	0.30	0.70
Lafenwa	1.10	21.00	1.20	1.30	1.10	0.50	0.90
Arakanga Old Scheme	0.90	16.60	1.00	1.00	0.90	0.50	0.75
Arakanga New Scheme	1.30	24.30	1.60	1.60	1.30	0.30	1.00
Mokoloki	0.90	15.90	1.00	1.00	0.90	0.80	0.90
Oke Oko	0.80	14.00	0.80	0.80	0.70	0.50	0.70
Kara	0.50	5.50	0.50	0.50	0.40	0.20	0.40
Average Shale Value	43.00	5.00	9.80	33.00	6.20	1.20	5.1

Source: Author

kg^{-1}) exceeding the 43.00 mg kg^{-1} average shale value standard the metal is seen to be a threat to sediment quality in those locations. The Igeo results obtained (-2.80 to -0.20) from all locations were classified as unpolluted since values obtained were below 0. Similarly, Ikere, Iseyin, Olokemeji, Opeki, Igangan, Arakanga Old Scheme, Mokoloki, Oke-Oko and Kara were seen not to be contaminated based on the CF values obtained which were less than 1 while Lafenwa, Ofiki and Arakanga New Scheme were seen to have medium contamination since their values fell between 1 and 2 when subjected to Contamination Factor analysis. Lanthanum accumulates in different organisms and thus may pose a hazard to species belonging to higher levels of the food chain (Henning et al., 2016). La values obtained from Gora River which ranged from 5.00 to 21.00 mg kg^{-1} were lower than the ASV standard of 43.00 mg kg^{-1}. Similarly, research by Dilioha and Onwualu-John (2016) showed that all La values ranging from 8.30 to 38.00 mg kg^{-1} were observed to be lower than the ASV standard.

According to Das et al. (1988), Lanthanum (La), is bioavailable in its trivalent form (La^{3+}) and has a high risk of biological effects while Dave et al. (1991) asserted that La^{3+} could compete for binding sites in biological systems with Ca^{2+} thereby inhibiting channels for calcium within cell membranes thus affecting the work of cells and tissues. Hua et al. (2017) also reported that La^{3+} has toxic effects on gills and livers of rare minnows. Additionally, high affinity of La to phosphate was observed by Herrmann et al. (2016) where insoluble lanthanum phosphate complexes were formed. This could have adverse effect on algal growth as a result of phosphate limitation. Similarly, high La concentrations inhibited growth of *Tetrahymena shanghaiensis,* a ciliate which grazes on planktonic bacteria while low La concentrations stimulated growth (Herrmann et al., 2016).

Caesium values obtained were seen to be at least 4 to 20 times higher than the ASV standard of 5.00 mg kg^{-1}. This presents a very high risk to biota in the aquatic environment. Igangan and Kara Igeo values (1.50 and 1.90) were observed to be unpolluted to moderately polluted, while values from Olokemeji (2.10), Ikere (2.80) and Iseyin (2.90) showed moderate pollution. Oke-Oko (3.20), Mokoloki (3.40), Opeki (3.70), Ofiki (3.70), Oyan (3.70), Lafenwa (3.80) and Arakanga New Scheme (4.00) were observed to be highly polluted. Similarly, all Cs values obtained showed very strong contamination when subjected to Contamination Factor analysis. Ren et al. (2007) reported that increased CE concentrations resulted in increased bioaccumulation intracellular and extracellular processes, which in turn lead to a decrease in lichen viability. Animals which are exposed to extremely high doses of cesium showed changes in behavior which are manifested in form of increased or decreased activity (www.lenntech.com).

Praseodymium (Pr) values from Oyan (10.10 mg kg^{-1}), Ofiki (11.90 mg kg^{-1}), Lafenwa (12.00 mg kg^{-1}), and Arakanga New Scheme (15.30 mg kg^{-1}) are a threat to sediment quality since they exceeded the ASV standard of 9.80 mg kg^{-1}. Igeo values obtained which ranged from -2.80 (Igangan) to 0.5 (Arakanga New Scheme) showed unpolluted status for all locations while CF analysis showed medium contamination for Ofiki (1.2), Lafenwa (1.2) and Arakanga New Scheme (1.6). Values obtained at the Southern Benue Trough ranging from 2.18 to 8.54 were observed to be lower than the ASV standard (Dilioha and Onwualu-John, 2016).

Effects of Pr on *Spirodela polyrhiza* at a concentration range of 0 to 60 µM for 20 days, in which it was observed that significant increase in cell death occurred with a reduction in photosystem II activity (Xu et al., 2016). This is indicative of the presence of Pr-inducing oxidative stress. Similarly, significant increase in cell death was observed in Pr-treated plants. Praseodymium can be a threat to the liver when it accumulates in the human body (www.lenntech.com).

Mokoloki (33.80 mg kg^{-1}), Ofiki (41.40 mg kg^{-1}), Lafenwa (42.70 mg kg^{-1}) and Arakanga New Scheme (54.20 mg kg^{-1}) Nd values which exceeded the 33.0 mg kg^{-1} ASV standard were seen as threats to sediment quality. Igeo values obtained ranged from -2.80 (Igangan) to 0.10 (Arakanga New Scheme) which indicates unpolluted status while CF values which ranged from 1.00 (Arakanga Old Scheme and Mokoloki) to 1.60 (Arakanga New Scheme) were observed to be moderately contaminated. According to Lenntech (2018), Nd causes damage to cell membranes of aquatic organisms which could have several negative influences on reproduction and the functions of the nervous system. Nd values from the Southern Benue Trough which ranged from 8.00-31.50 were found to be lower than the ASV standard of 33.00 mg kg^{-1} (Dilioha and Onwualu-John, 2016).

All Igeo values for Eu and Gd were observed to have shown unpolluted status. Similarly, all CF values for Eu and Gd were observed to be uncontaminated except for that of Arakanga Ne Scheme (1.00) which showed moderate pollution. Eu values obtained from the Southern Benue Trough which ranged from 1.29 to 1.46 were found to be higher than the ASV standard of 1.20 mg kg^{-1} (Dilioha and Onwualu-John, 2016). This showed that Eu values are of great concern to sediment quality in that area. Similarly, Gd values obtained from River Gora, Minna, Niger State, which ranged from 8 to 26 mg kg^{-1} were observed to be higher than the ASV standard of 5.1 mg kg^{-1} (Obaje et al., 2015). This poses a threat to sediment quality as well as aquatic organisms. Wilde et al. (2002) stated that Gd can be toxic to selected microorganisms such as algae and bacteria at very high concentrations (average 80,000 mg/L, maximum 259,000 mg/L).

Conclusion

Eu and Gd sediment metals were of no threat to sediment quality due to the unpolluted status observed while Cs should also be monitored closely bearing in mind that values obtained from all locations were 4 to 20 times higher than the ASV standard. This showed high Cs contamination status in all locations. Arakanga New Scheme and Lafenwa sediments are a concern to sediment quality in view of metals values obtained which exceeded ASV standards in these locations except for Eu and Gd values. Metals assessments in these locations are of high importance in order to avoid adverse effect occurrences on human and aquatic lives since dam for public water supply is situated at Arakanga while Lafenwa has high human residential areas that depend on the river water for various purposes.

CONFLICT OF INTERESTS

The authors have not declared any conflict of interests.

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Vol. 17(2), pp. 51-62, February 2023
DOI: 10.5897/AJEST2021.3034
Article Number: 8D9209870535
ISSN: 1996-0786

http://www.academicjournals.org/AJEST

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African Journal of Environmental Science and Technology

Review

Benthic macroinvertebrates in the biomonitoring of a Nigerian coastal water

Joseph A. Nkwoji

Department of Marine Sciences, Faculty of Science, University of Lagos, Lagos, Nigeria.

Received 15 June, 2021; Accepted 3 August, 2021

Most coastal waters are exposed to high influx of pollutants due to the obvious elevated human activities. In order to adequately evaluate the extent of toxicity of contaminants in the ecosystem, and their synergistic effects, marine ecologists prefer biomonitoring to chemical approach. The benthic macroinvertebrates, due to their sedentary mode of life and residence at the sediment-water microcosm, are regarded as the most veritable tool for biomonitoring. This is because these organisms are impacted by the interstitial forces of the sediment-water interface in the marine ecosystem and also serve as the main channel for the interchange of biomass. This review focuses on the biomonitoring status and prospects of a coastal lagoon with reference to the benthic macroinvertebrates. The levels of biomonitoring activities which are suborganismal, organismal, population, community and ecosystem levels are critically analyzed. However, most of the molecular and biochemical assays, and biomarkers used in biomonitoring studies at the suborganismal and organismal levels of biological development are being outsourced. This poses great challenges to holistic biomonitoring programs in the vast Nigerian coastal ecosystem.

Key words: Biomonitoring, coastal waters, Lagos lagoon, bioindicators, benthic macroinvertebrates.

INTRODUCTION

The anthropogenic stress exerted the coastal waters has resulted in pollution hazards at an alarming rate in such aquatic ecosystems. For some obvious reasons, coastal environments are choices for localization of industries, high human population concentration, developmental projects and agricultural and fishing activities. The combination of all these generate variant pollutants that are directly or indirectly released into the nearby coastal waters as the last sink (Nkwoji et al., 2020). Monitoring according to Cullen (1990) involves the process of repetitively observing one or more elements of the environment, for defined purposes and according to

E-mail: jnkwoji@unilag.edu.ng.

prearranged schedules in space and time using comparable methods for environmental sensing and data collection. Environmental monitoring is both systematic and repetitive and this differentiates it from environmental survey.

Biomonitoring is defined as the systematic use of living organisms or their responses to determine the condition or changes in the environment (Rosenberg, 1998; Gerhardt, 1999; Oertel and Salanki, 2003). It involves the direct measurements of pollution impacts on organisms of interest as opposed to the use of abiotic chemical surrogates. One advantage of this is its capability to integrate and also to measure subtle variables caused by minor and intermittent contaminants. Biological assessment of the outcomes of the fluctuating factors that drive any ecosystem would seem more sensible than attempting to measure these varying driving factors and then to estimate how they might affect biological production' (Cullen, 1990).

According to Schöne and Krause (2016) and Prabhakaran et al. (2017), when chemicals are directly employed to assess the level of contamination in water and sediment, the extent of toxicity of contaminants and their synergistic effects in the ecosystem would not be evaluated. This is because some toxicants which are introduced into the aquatic environments in levels that are undetectable through chemical means could still be bioavailable to the biota for uptake and subsequently bioaccumulated in their tissues despite the low concentration (Schöne and Krause, 2016). According to Li et al. (2010), there is an urgent need for more holistic and methodological approaches to evaluate the actual state of these ecosystems and to monitor their rate of alterations.

In a complex natural ecosystem such as lagoons, the use of chemical means to evaluate the characteristics of pollutants is often difficult. It is therefore, important that the effects of such pollution should be studied in relation to biological systems (Parmar et al., 2016; Cerveny et al., 2016). According to Masese et al. (2013), the aquatic communities are better indicators of anthropogenic stressors in their environment at different levels because they integrate the different variations. This underscores the need for biomonitoring in aquatic ecosystem management and conservation.

According to Zhou et al. (2008), biomonitoring exhibits obvious advantages over the routine chemical monitoring in the following ways: It reveals both the subtle biological changes of organisms affected by exogenous chemicals, which is usually missed by the conventional chemical analysis, and the integrated effects of the complex pollutants on the organisms in the environment. When the organisms are exposed to pollutants, they respond rapidly and this sensitivity is often harnessed. Some pollutants below the detection limits by the instrumental analytical techniques could still be biomonitored due to the occurrence of the chronic toxicities of the pollutants in the organisms under long-term exposure.

Biomonitoring gives a complete evaluation of the exact impacts of the pollutants by considering both the potential impacts and the actual joint toxicities of the toxicants on the environment. This is possible due to the fact that biological responses could get stimulated at chemical concentration levels below the conventional analytical detection limits, and may persist even at the end of the chemical exposure (Zuykov et al., 2013). According to Prabhakaran et al. (2017), the success of any biomontoring project would depend on the right choices of the biomonitors. While any of such terms as bioindicators, biological monitors and sentinel organisms may be used to mean the biota employed as biomonitors (Hellawell, 2012), it is important to state here that, while bioindicators show an ecological effect either by their presence or absence, ecological monitors are species that indicate the actual degree of ecological alterations by the behavioural, as well as their physiological and biochemical responses (Tsygankov et al., 2017; Müller and Müller, 2018).

The definition of biomonitoring by Zhou et al. (2008) as "a scientific technique for assessing environment including human exposures to natural and synthetic chemicals, based on sampling and analysis of an individual organism's tissues and fluids", could be disputed. This is because the definition which emphasizes the chemical analysis of tissues and fluids of organisms is consistent with biomarker which is more of a biomonitor than a bioindicator, and therefore does not represent the general definition of biomonitoring. The technique takes advantage of the fact that when organisms are exposed to certain chemicals, they leave markers that reflect the extent and duration of the exposure. These markers could be the chemical itself, components of the chemical or other biological variations in the biota resulting from the effects of the chemicals on the organism (Zhou et al., 2008).

BIOMONITORING TECHNIQUES

The multi dynamic interactions in the aquatic ecosystem by its biotic and abiotic components make it a complex one. This complexity includes the bioassessment of some stress factors which often have synergistic, additive or antagonistic effects in the aquatic environment (Solimini et al., 2009). In order for a more holistic assessment, biomonitoring should be addressed at the different levels of biological organization; suborganismal, organismal, population, community and ecosystem. This however, is not always the case as many biomonitoring programs tend to be restricted to a few levels of biological organization, thereby limiting the potential spectrum of measurable of cause-effect responses to different anthropogenic impacts (Cortes et al., 2016). In this review, different biomonitoring techniques, based on the specific aim, have been adopted.

Bioaccumulation

When an organism absorbs a toxic substance at a rate greater than that at which the substance is eliminated, biomonitoring occur. According to Waykar et al. (2011), this results from a dynamic equilibrium between exposure from the outside environment and uptake, excretion, storage, and degradation within an organism. The processes of uptake, storage and elimination of toxicants are involved during bioaccumulation. Understanding of the dynamic process of bioaccumulation is a critical consideration in the regulation of chemicals such as aquatic metals (Zhou et al., 2008).

The mechanism and quantity of toxicant's accumulation in the tissues and organs of biota in the aquatic ecosystem is key to assessing the adverse effects of those toxicants on the ecosystem. For example, Liu et al. (2016) stated that the organochloride pesticide contamination in Lake Chaohu in China could be determined by the correlation between the OCP contamination in the aquatic biota with that in the water/or suspended matter. By interacting with the environment, aquatic biota accumulates pollutants by directly picking them up from their surroundings as well as through the food chain at the various trophic levels. The extent of accumulation of pollutants by the aquatic biota will determine the bioavailability in the water and sediment as well as along the trophic levels.

The accumulation of these toxicants constitutes threat to the ecosystem, its biogeochemical processes, and risks to human health (Prabhakaran et al., 2017). In the case of Polycyclic Aromatic Hydrocarbons (PAHs), their level of accumulation in marine biota is dependent on the length of exposure and concentration of the toxicants in that environment as well as extent of the mobilization (Meador et al., 1995). Aquatic biota that accumulates PAHs during short-term acute incidences (Prabhakaran et al., 2017) may have the opportunity of system clean up when returned to pristine condition. If however the exposure is chronic and continuous, the ability to eliminate the toxicant may not be possible. According to Prabhakaran et al. (2017), fishes, oligochaetes and crustaceans easily metabolise PAHs while molluscs and bivalves are the least in the ability to metabolise PAH. The implication is that consumers of molluscs have high risk of the shellfish toxicity. In most bivalves, the gills are the main sites for metal accumulation. This is because the bivalves are mostly filter feeders and in the process of feeding, the first contact point is the gills. Also, the gills have large surface area and are lined with mucus, thus making them better accumulator of such metals as cadmium, lead, and zinc in the feeding ambient water than the other soft body tissues (Zhou et al., 2008).

Biochemical alteration

Biomontoring adopted using biochemical alteration involves the bioassessment at the suborganismal level of the biological organization. The tools employed at this level are the biomarkers. Biomarkers are measured at the suborganismal level to identify the biological effects of some toxicants in the environment at an early stage for effective quality assessment of that environment (McCarthy and Shugart, 1990).

According to Kumar et al. (2017), biomarkers are linker between environmental contamination (cause) and its effects in respect to changes in biological systems. Prabhakaran et al. (2017) stated that alteration of the biochemical defense mechanism is the first typical response to any toxic assault by xenobiotics. The protective nature of antioxidative enzymes makes them effective biomarker for identifying pollutants in the environment at an early stage. The measurement of this excitation, according to Schlenik et al. (2008), can serve as sensitive indicators of an altered cell function. This is because according to Capela et al. (2016), biochemical responses are triggered in aquatic organisms even at a low concentration of the toxicants. Moore et al. (2004) has proposed that biomarkers should be related to the different functions of the organisms and that different levels of the biological scale of the organisms studied in order "for better understanding of the mechanisms underlying the effect of the stressors" (Lavarías et al., 2016) on the aquatic environment.

Biomarkers such as metallothionein and cytotoxicological responses such as genotoxicity, lysosomal alterations, immunocompetence and gencholinesterase activities (Zhou et al., 2008) have currently been developed. There is need however, to take some precautions at the sensitivity of these biomarkers at each developmental stage of the organisms. Some special protein could be purified and harnessed to serve as biomarker for some metal exposure (Zhou et al., 2008). Biomarkers should be well selected for specificity on particular pollutants as well as organisms and areas of interest.

MORPHOLOGICAL AND BEHAVIOURAL OBSERVATIONS

Observing the effects of toxicants on the morphology and behaviour of organisms is the most direct form of bioassessment. In bioassay analysis, the sublethal effects of toxicants on aquatic biota can easily be visually observed through the changes in their physical appearance and behavioural pattern. Behavioural ecotoxicology deals with morphological and behavioural observation of organisms in relation to environmental quality (Prabhakaran et al., 2017). Behavioural changes in aquatic biota in response to chemical stress in their environment are among the early and most sensitive observable parameters, and they incorporate both the biochemical alterations at the sub-organismal level and changes at the whole organism level (Hartmann et al., 2016).

The study by Zhou et al. (2008) on the masculinization phenomena exhibited in prosobranch gastropod, *Rapana venosa* is a typical example of morphological observation in biomonitoring. The imposex was as a result of organotin pollution that the gastropod was continuously exposed to. In imposex-affected species, the entire female genital system is conserved but superimposed by male organs such as penis and/or vas deferens (Zhou et al., 2008). This could result to infertility in the female species thereby affecting the population. According to the study, imposex occurred in some snails in southeast China indicating the feasibility of the biomonitoring technique based on imposex investigation in the assessment of organotin contamination caused by frequent marine traffic (Zhou et al., 2008).

In biomonitoring, the sublethal effect of chemical exposures in the aquatic ecosystem could be bioassessed through some behavioural patterns of the organisms. The organism exhibits such behaviours like avoidance, feeding depression and valve closure behaviour (Zhou et al., 2008).

A study carried by Lopes et al. (2004) showed that *Daphnia longispina* avoid certain concentration of copper and further revealed that avoidance to copper was much more sensitive than lethality by copper in the organisms. Lopes et al. (2004) therefore, recommended that the avoidance assays be used as a complementary tool, for ecological risk assessments and effluent biomonitoring since such assays can provide cost-effective and ecologically relevant information.

A research conducted by Zhou et al. (2008) indicated that certain concentration of cadmium and zinc at a sublethal level would result in the significant reductions of the feeding rate of the Decapod, *Atyaephyra desmarestii* and Amphipoda, *Echinogammarus meridionalis*. They therefore, posited that chronic feeding assays be used in biomonitoring studies because they are rapid, cheap and effective. According to Liao et al. (2007) and Zhou et al. (2008), the valve closure behaviour in the freshwater clam, *Corbicula fluminea* in response to copper concentration, could be used in the bioassessment of the aquatic heavy metals.

POPULATION AND COMMUNITY LEVEL APPROACH

Biomonitoring approach at the level of population and community is also very critical. At this level of biomonitoring, the bioindicators are mostly employed for the assessment. The population and community level approaches in aquatic biomonitoring are based on the understanding of the influence of environmental factors on patterns of distribution, abundance and species diversity of aquatic communities (Prabhakaran et al., 2017). The community structure can be described by computing the richness and diversity indices and used to determine the impact of any toxicants and the general health of the ecosystem (Hering et al., 2006). The relevant indices are discussed later in this paper.

Researches by De Castro-Catala et al. (2015), Chiu et al. (2016), and Hasenbein et al. (2016) indicate that biomonitoring approach at the population and community level of an ecosystem is very necessary in the bioassessment of its aquatic biota and general health. However, according to Prabhakaran et al. (2017), emphasis on this biomonitoring approach cannot be considered as an all-purpose concept to determine all aspects of biodiversity loss. This is because in a situation where an abrupt change in environmental conditions has caused the loss or total elimination of certain species, the community could be replaced by ecologically similar ones capable of occupying the new niche. It is important to note that population and community approach of biomonitoring can only capture the loss of species, and this might not be reflected in the overall species diversity assessment (Feld et al., 2014).

Due to its masculinisation properties on some molluscs, organotin pollution, because of the imposex response to tributyltin compound, are commonly used in biomonitoring for populations of some gastropods (Axiak et al., 2003). The organotin may cause infertility in the female gastropod thereby obstructing reproduction and drastically affecting their population. Researches have shown the water quality of any natural aquatic ecosystem could be evaluated by the population dynamics of a particular species (Crespo et al., 2015). According to Ajao and Fagade (1990a), loss in biodiversity is the most obvious effect of pollution. Alterations in population and community level indicate disturbances in the normal balance in the studied ecosystem but may also be an indication that the population of a peticular ecosystem is advanced (Zhou et al., 2008).

BIOMONITORING OF COASTAL WATERS: BENTHIC MACROINVERTEBRATES AS SUITABLE BIOINDICATORS

Zhou et al. (2008), in the review, “Biomonitoring: An appealing tool for assessment of metal pollution in the aquatic ecosystem”, posited that “suitable bioindicators usually give great help to the biomonitoring” and that a perfect bioindicator is expected to have the following characters: (1) it can accumulate high levels of pollutants without death; (2) it lives in a sessile style, thus definitely representing the local pollution; (3) it has enough abundance and wide distribution for the repetitious sampling and comparison; (4) its life is long enough for the comparison between various ages; (5) it can afford suitable target tissue or cell for the further research at microcosmic level; (6) easy sampling and easy raising in the laboratory; (7) it keeps alive in water; (8) it occupy the important position in food chain; (9) well dose-effect relationship can be observed in it.”

According to Markert et al. (2003), bioindicators are organisms or communities of organisms whose content of certain elements or chemical (organic) compounds and/or whose morphological, histological or cellular structure, metabolic-biochemical processes, behaviour or population structure(s), including changes in these parameters, supply information on the quality of the environment or the nature of environment changes. It then implies that a bioindicator should be able to provide enough information to ascertain the quality of part or all of an environment. Additionally, “an ‘ideal’ indicator should have the characteristics as follows: (a) taxonomic soundness (easy to be recognized by nonspecialist); (b) wide or cosmopolitan distribution; (c) low mobility (local indication); (d) well-known ecological characteristics; (e) Numerical abundance; (f) suitability for laboratory experiments; (g) high sensitivity to environmental stressors; (h) high ability for quantification and standardization” (Li et al., 2010). All these qualities are inherent in the benthic macroinvertebrates, and thus make them most suitable bioindicators for pollution assessment in the coastal waters.

Macroinvertebrates are organisms without backbones that can be seen without the aid of a microscope. As a result of their habitat choice, macroinvertebrates are often referred to as benthos, a term that refers collectively to organisms that lives on, in or near the bottom. These organisms reside in aquatic systems for long enough to reflect chronic effects of pollutants (Reboredo-Fernández et al., 2014), and thus, data on these organisms are used individually or in combination with other environmental characteristics, to assess the extent of environmental impairment (Huh, 2019) and the general health of an aquatic ecosystem.

The benthic macroinvertebrates constitute the key components of aquatic food webs, linking organic matter and nutrient resources with higher trophic levels (Kiljunen et al., 2020). Their sedentary lifestyle makes them “representative of site-specific ecological conditions” (Nkwoji et al., 2016). With the sensitive life stage and relatively long lifespan, the benthic macroinvertebrates have the ability to integrate the effects of short-term environmental variations. Besides, these assemblages are made up of many species among which there is a wide range of trophic levels and pollution tolerances (Muzón et al., 2019), thereby providing vital information for interpreting cumulative effects. According to Kiljunen et al. (2020), the community structure of the assemblages frequently changes in response to environmental disturbances in predictable ways and this has formed “the basis for development of biocriteria to evaluate anthropogenic influences” (Karrouch et al., 2017).

DIVERSITY INDICES

Diversity indices are mathematical expressions which use three components of community structure; namely, richness (number of species present), evenness (uniformity in the distribution of individuals among the species) and abundance (total number of organisms present), to describe the response of a community to the quality of its environment (Metcalfe-Smith, 1994). They are efficient in describing responses of a community of organisms to variation (Ghosh and Biswas, 2015) in the environment.

The underlying assumption in the diversity approach is that undisturbed environments will be characterized by a high diversity or richness, an even distribution of individuals among the species, and moderate to high counts of individuals (e.g., Shannon-Wiener Index, Simpson Index and Margalef Index) (Metcalfe, 1989). However, it is not in all cases that this assumption has proved to be true. High diversity does not necessarily imply unpolluted water, and low diversity on the other hand, does not necessarily indicate pollution.

Shannon index

This index is more popular among other diversity indices. It is referred to as Shannon's diversity index, the Shannon-Wiener index, the Shannon-Weaver index and the Shannon entropy (Niklaus et al., 2001; Sax, 2002). The index was first introduced by Claude E. Shannon to quantify the entropy (uncertainly or information content) in strings of text. The concept is that the more diverse letters there are, and the more equal their proportional abundances in string of interest, and the more difficult it would be to correctly predict which letter will be the next one in the string. The Shannon entropy was therefore, introduced to quantify the uncertainly (entropy or degree of surprise) (Shannon, 1948) associated with this prediction (Shannon, 1948; Tandon et al., 2007). Applied in ecology, the Shannon entropy quantifies the uncertainty in predicting the species identity of an individual that is taken at random from the dataset (Sarma and Das, 2015). The Shannon diversity index (H) is applied in community structure studies to characterize their species diversity. Shannon diversity index (H) is not a diversity in itself, but is an index used as a determinant of diversity. It is expressed in the following equation:

$$\mathrm{H} = \sum_{i=1}^{s} Pi(lnPi) \qquad (1)$$

where H = Shannon index of diversity, Pi = fraction of the entire population made up of species i, that is, pi is the proportion (n/N) of individuals of one particular species found (n) divided by the total number of individuals found (N), S = Numbers of species encountered, ln = natural logarithm, and ∑= sum from species 1 to species S (Shannon, 1948).

Generally, the Shannon index ranges between 1.5 and 3.5 in most ecological studies, and could hardly exceed 4. The index increases as both the richness and the evenness of the community increase and *vice versa*. Shannon index is the preferred of all other indices by ecologists because it incorporates both components of biodiversity. However, this could be seen as both strength and weakness. It is a strength because it provides a simple, synthetic summary, but it is also a weakness because it makes it difficult to compare two communities that have much difference in richness (Chao and Shen, 2003).

Simpson's index

Simpson's index of diversity is used to calculate the diversity that incorporates both the number of species that are present and the relative abundance of each species in the community (Gorelick, 2006). It is based on probability of any two individuals drawn at random from an infinitely large community belonging to the same species (Ardura and Planes, 2017). The index is useful especially when researchers are dealing with very large quantities of data such that the level of diversity within that data becomes difficult to ascertain by merely reading from the table of results.

The Simpson's index gives relatively little weight to the rare taxa and more weight to the common taxa, as it weighs towards the abundance of the most common taxon (Ghosh and Biswas, 2015). The value, *D* is a measure of dominance, so as *D* increases, diversity (in the sense of evenness) would decreases. This index is therefore, usually reported as its complement 1-D (Hurlbert, 1971). Since *D* takes on values between zero and one and approaches one in the limit of a monoculture, (1-D) provides an intuitive proportional measure of diversity that is much less sensitive to species richness (Magurran and McGill, 2011).

$$D = 1 - \frac{\Sigma n(n-1)}{N(N-1)} \qquad (2)$$

The value of *D* ranges between 0 and 1, where 1 represents complete diversity and 0 represents complete uniformity (Whittaker, 1972). The value of *D* in a location is better appreciated when compared with that from a spatially different location.

Margalef's index

Margalef's index *d* (Margalef, 1958) has been used by researchers to estimate the community structure. The index measures species richness and it is highly sensitive to sample size although it tries to compensate for sampling effects (Magurran and McGill 2011). It is a measure "of the variety of species in a given community and calculated based on emerged species and the individuals of the emerged species" (Gamito, 2010). The index is calculated as the species number (S) minus 1 divided by the logarithm of the total number of individuals (N)' (Dong et al., 2020) as expressed in the following equation:

$$d = \frac{(S-1)}{\ln N} \qquad (3)$$

Unlike Shannon and Simpson indices which may use proportional values or densities, absolute values or the

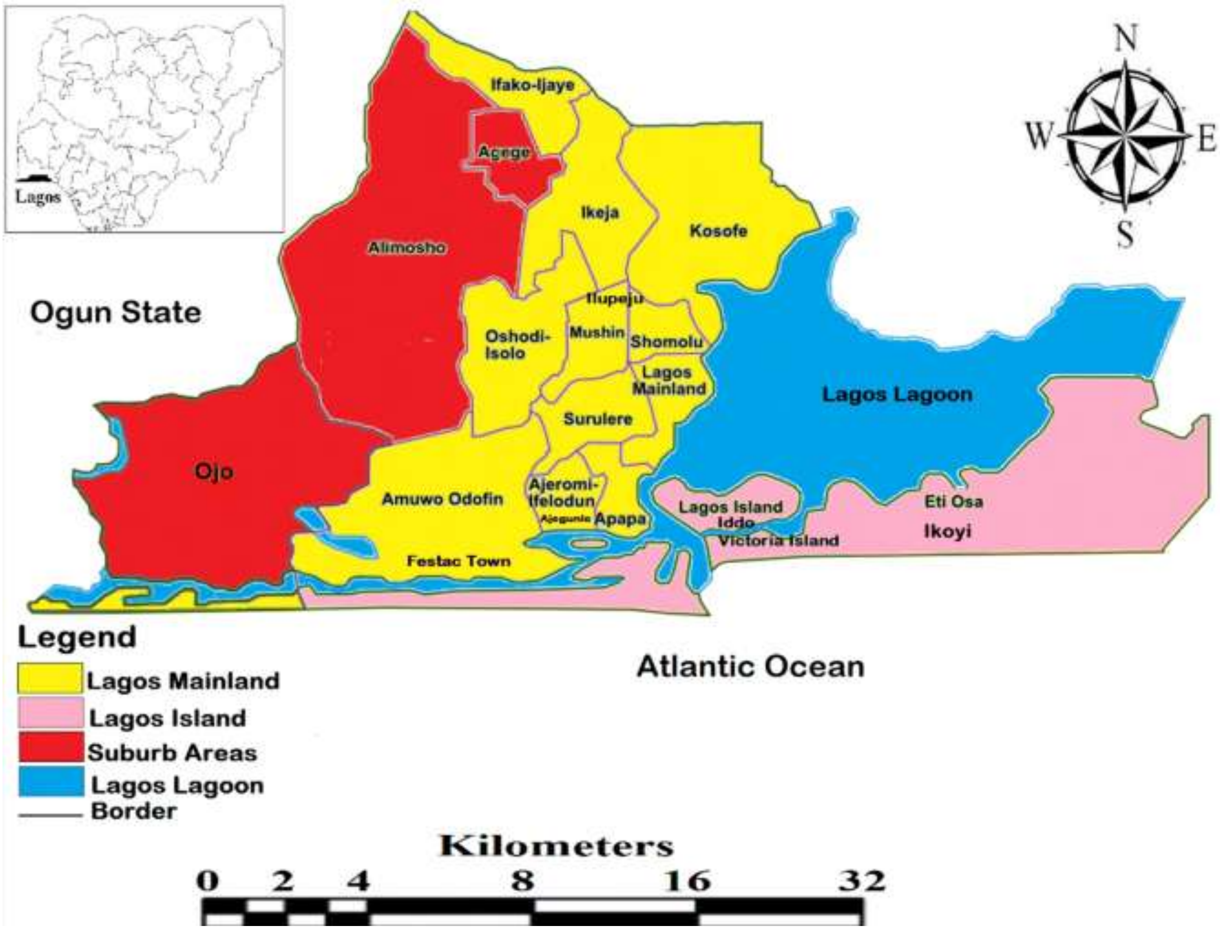

Figure 1. Map of Lagos State, Nigeria showing the Lagos lagoon.
Source: Author

total numbers must be used while calculating Margalef index. The sole use of the index by researchers to determine the health of an aquatic ecosystem is highly discouraged because species richness alone does not reflect the true health status of the environment as it depends largely on the size of the sample.

BIOMONITORING OF THE LAGOS LAGOON: STATUS AND PROSPECTS

The strategic location of the Lagos lagoon (Figure 1), coupled with its economic and ecological relevance, made it one of the most widely researched coastal lagoons in the world. Its link with the Atlantic ocean, the concentration of industries, and high human population around the lagoon have caused the lagoon to be much impacted with pollutants from maritime, industrial and domestic sources respectively. There is need for regular biomonitoring of the lagoon so as to maintain its ecosystem functions and reduce health hazards on the human population.

The first published report on the ecology of Lagos lagoon was by Webb (1958). In this report, no mention was made about the living resources of the lagoon as the study was basically on the geomorphology and depositional features of the lagoon system. Many subsequent studies on Lagos lagoon have focused on heavy metals and other associated toxicants in water and sediment (Okoye, 1991; Okoye et al., 1991; Olatunji and Abimbola, 2010; Lawson, 2011; Alani et al., 2013; Benson et al., 2014; Elijah and Isa, 2015; Olayinka et al., 2016; Olafisoye et al., 2016; Bawa-Allah et al., 2018) without due recourse to the biota of the lagoon.

Table 1. Biomonitoring studies of Lagos lagoon at the suborganismal and organismal levels of biological organization

Research aim	Sentinel organisms	Stressors/Biomarker	References
To determine pesticide contamination in fish muscles	Croaker fish	Organochloride pesticide (OCP)	Williams (2013)
Investigate impacts of heavy metals on blood parameters and enzymatic activities in some fishes	*Chrysichthys nigrodigitatus* and *Pythonichthys macrurus*	Cd, Cr, Cu, Fe, Pb, Zn	Ayoola and Dansu (2014)
To investigate the oxidative stress in fish in sawdust and wood waste pollututed areas	*Pomadys jubelini*	Superoxide dismutase (SOD) and lipid peroxidation (MDA)	Ekaete (2014)
To measure trace metal and biochemical markers in *Callinectes amnicola*	*Callinectes amnicola*	Zn, Pb, Cd and Cu, SOD, GPx, CAT, reduced glutathione and malondialdehyde	Olakolu and Chukwuka (2014)
To assess the toxicity of leachates from dump sites in the Lagos lagoon on brackish water shrimp	*Palaemonetes africanus*	Dumpsite leachate	Amaeze and Abel-Obi (2015)
To investigate the impact of BTEX on macrobenthic community structure	*Nais eliguis* and *Heteromastus filiformis*	benzene, toluene, ethylbenzene, and xylene (BTEX)	Doherty and Otitoloju (2016)
To assess the effects of PAHs on the embryo of zebra fish the from the Lagos lagoon	*Danio rerio*	PAHs	Sogbanmu et al. (2016)
To assess TOC and Heavy Metals Bioaccumulation in *Sarotherodon Melanotheron*	*Sarotherodon Melanotheron*	Cd, Cr, Cu, Fe, Pb, Zn	Ayoola (2017)
To evaluate antioxidant and oxidative stress responses in Mangrove Crab inhabiting contaminated Lagos lagoon mudflats	*Sesarma huzardii*	CAT, SOD, GSH, MDA, TBARS	Usese et al. (2018)

Source: Author

Don-Pedro et al. (2004) however, monitored the trends of heavy metal concentration of the lagoon with reference to the bioaccumulation of the metals in the body tissues of benthic fauna, *Typanotonus fuscatus* and *Clibanarius africanus*. More recent studies on the biomonitoring of Lagos lagoon include “the use of aquatic macrophytes to monitor the distribution of heavy metals” (Adesuyi et al., 2018) and *Palaemonetes africanus* to monitor the toxicity of dumpsite leachate (Amaeze and Abel-Obi, 2015).

The earliest and robust biomonitoring survey of the Lagos lagoon that used the community structure of the benthic macroinvertebrates as bioindicators (Nkwoji et al., 2020) was conducted by Ajao and Fagade (1990a). In this study however, more emphasis was placed on the sediment, the habitat of the benthos rather than the benthos themselves.

The characteristics of the sediments were analysed in details and the benthic macroinvertebrates studies were conducted only to the extent it related to the sediment composition and type. Moreover, no biomonitoring was conducted at the suborganismal and organismal levels and the identification of the sentinel organisms was purely morphological. Some of the biomonitoring studies of the Lagos lagoon at both the suborganismal and organismal levels are shown in Table 1 while the studies at population, community and ecosystem levels are

Table 2. Biomonitoring studies of Lagos lagoon at the population, community and ecosystem levels.

Research aim	Indicator organisms/group	Source of pollution	References
To obtain information on the distribution, habitats and communities in the Lagos lagoon in relation to environmental factors	*Capitella capitata*, *Nereis* spp. and *Polydora*	Heavy metals, Petroleum hydrocarbons, Organic Pollutants	Ajao and Fagade (1990a)
To identify the effects of organic pollution on the distribution of benthic macroinvertebrates	*Capitella capitata*	Organic pollutants	Ajao and Fagade (1990b)
To identify the effects of habitat modification on the composition and abundance of macrofauna	Benthic macrofauna	Dredging and sand mining	Brown and Ajao (2004)
To study on the impact of Land based pollution on the macrobenthic community	Benthic macrofauna	Industrial and domestic wastes	Chukwu and Nwankwo (2005)
To identify the impacts of two different organic pollutants on the macrobenthic fauna	Benthic macrofauna	Organic wastes	Ogunwenmo et al. (2005)
To determine the effects of domestic sewage on the community structure of benthic macroibvertebrates	Benthic macroinvertebrates	Domestic sewage	Edokpayi and Nkwoji (2007)
To identify the impact of various pollutants on fish species diversity in the lagoon	Fish species	Chemical wastes, domestic sewage, solid waste.	Amaeze et al. (2012)
To study on the Impacts of Organic Pollution on the Community Structure of Benthic Macrofauna	Benthic macrofauna	Organic pollutants	Nkwoji (2017)
To identify the impacts of sediment mining on the macrozoobenthos community	Macrozoobenthos	Sediment mining	Nkwoji and Awodeyi (2018)

Source: Author

shown in Table 2.

CONCLUSION AND FUTURE DIRECTION

Biological monitoring of the Lagos lagoon would serve as the reference for other Nigerian coastal waters. However, most studies on biomonitoring of the coastal water have centered on the use of bioindicators, especially the benthic macroinvertebrates as sentinel organisms. Unfortunately, modern biomonitoring techniques that incorporate the different levels in biological organization are yet to be developed in Nigeria. The molecular and biochemical assays, as well as the biomarkers used for assessment at the suborganismal level of biological development are being sourced outside the country. Their cost, availability and accessibility pose great challenges and constitute serious impediments to biomonitoring programs. Incidentally, biomonitoring at this level, because of their early warning signal features, is key to identifying the xenobiotics in our coastal ecosystem and help to forestall any serious future damage to the ecosystem. Biomonitoring at the organismal level also has its challenges in Nigeria. There is erratic public power supply and this is the bane of experimental procedures. For instance, the bioassay setups need constant power for aeration and general conditioning of the experimental environment to assume a near natural ecosystem and for acclimatization of the test organisms. Private power generators pose the problem of emitting carbon fumes and gases that add pollutants to the bioassay setup thereby distorting the experiment. There is the need for concerted effort by the government and non-governmental bodies to fund researches on biomonitoring of Nigerian coastal waters and Lagos lagoon in particular. Infractural deficit should be tackled and green and clean energy provided. These are necessary for laboratory experimental setups required for biomonitoring studies.

CONFLICT OF INTERESTS

The author has not declared any conflict interests.

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