# Ocean Life

| Ocean Life | vol. 4 | no. 2 | December 2020 |
| E-ISSN: 2580-4529 |

*Fiddler Crab* photo by Michael

# Ocean Life

| Ocean Life | vol. 4 | no. 2 | December 2020| E-ISSN: 2580-4529|



**Published semiannually** **PRINTED IN INDONESIA**

E-ISSN: 2580-4529

9 772580 452084

# Ocean Life

| Ocean Life | vol. 4 | no. 2 | December 2020 |

**ONLINE**
http://smujo.id/ol

**e-ISSN**
2580-4529

**PUBLISHER**
Society for Indonesian Biodiversity

**CO-PUBLISHER**
Universitas Papua, Manokwari, Indonesia

**OFFICE ADDRESS**
Research Center for Pacific Marine Resources, Institute for Research and Community, Universitas Papua.
Old Rectorat Complex Block III No. 7-8, Jl. Gunung Salju, Amban, Manokwari 98314, Papua Barat, Indonesia
Tel./Fax.: +62-986-212156/211455, email: ol@smujo.id, joceanlife@unipa.ac.id, joceanlife@gmail.com

**PERIOD OF ISSUANCE**
June, December



**Society for Indonesian Biodiversity**

**Universitas Papua, Manokwari, Indonesia**

Published by Smujo International for The Society for Indonesian Biodiversity and Universitas Papua Manokwari, Indonesia

## GUIDANCE FOR AUTHORS

**Aims and Scope:** ***Ocean Life*** (abbreviated as **Ocean Life**) encourages submission of manuscripts dealing with all aspects of maritime and marine resources in estuaries, coastal zones, continental shelf, the seas and oceans, including marine biodiversity and fisheries resources, biochemistry, physiology, behaviour, and genetics of marine life, socio-economic and cultural aspects, conservation and management, as well as biogeochemistry, marine pollution, and climate change.

**Article types:** The journal seeks original full-length research papers, reviews, and short communication. Manuscript of original research should be written in no more than 8,000 words (including tables and picture), or proportional with articles in this publication number. Review articles will be accommodated, while, short communication should be written at least 2,000 words, except for pre-study.

**Submission:** The journal only accepts online submission through system or email to the editors at **joceanlife@gmail.com** (until the end of 2017). Submitted manuscripts should be the original works of the author(s). The manuscript must be accompanied by a cover letter containing the article title, the first name and last name of all the authors, a paragraph describing the claimed novelty of the findings versus current knowledge. Submission of a manuscript implies that the submitted work has not been published before (except as part of a thesis or report, or abstract); and is not being considered for publication elsewhere. When a manuscript written by a group, all authors should read and approve the final version of the submitted manuscript and its revision; and agree the submission of manuscripts for this journal. All authors should have made substantial contributions to the concept and design of the research, acquisition of the data and its analysis; drafting of the manuscript and correcting of the revision. All authors must be responsible for the quality, accuracy, and ethics of the work.

**Acceptance:** The only articles written in English (U.S. English) are accepted for publication. Manuscripts will be reviewed by editors and invited reviewers (double blind review) according to their disciplines. Authors will generally be notified of acceptance, rejection, or need for revision within 1 to 2 months of receipt. The manuscript is rejected if the content does not in line with the journal scope, does not meet the standard quality, inappropriate format, complicated grammar, dishonesty (i.e. plagiarism, duplicate publications, fabrication of data, citations manipulation, etc.), or ignoring correspondence in three months. The primary criteria for publication are scientific quality and biodiversity significance. **Uncorrected proofs** will be sent to the corresponding author as *.doc* or *.rtf* files for checking and correcting of typographical errors. To avoid delay in publication, corrected proofs should be returned in 7 days. The accepted papers will be published online in a chronological order at any time, but printed in April and October.

**Ethics:** Author(s) must obedient to the law and/or ethics in treating the object of research and pay attention to the legality of material sources and intellectual property rights.

**Copyright:** If and when the manuscript is accepted for publication, the author(s) still hold the copyright and retain publishing rights without restrictions. Authors or others are allowed to multiply article as long as not for commercial purposes. For the new invention, authors are suggested to manage its patent before published.

**Open access:** The journal is committed to free-open access that does not charge readers or their institutions for access. Readers are entitled to read, download, copy, distribute, print, search, or link to the full texts of articles, as long as not for commercial purposes. The license type is CC-BY-NC-SA.

**A charge:** The journal is committed to free of charge for submission and publication of non-institutional funded research (waiver).

**Reprints:** The sample journal reprint is only available by special request. Additional copies may be purchased when ordering by sending back the uncorrected proofs by email.

**Manuscript preparation:** Manuscript is typed on A4 (210x297 $mm^2$) paper size, **in a single column**, single space, 10-point (10 pt) Times New Roman font. The margin text is 3 cm from the top, 2 cm from the bottom, and 1.8 cm from the left and right. Smaller lettering size can be applied in presenting table and figure (9 pt). Word processing program or additional software can be used, however, it must be PC compatible and Microsoft Word based (*.doc* or *.rtf*; **not *.docx***). **Scientific names** of species (incl. subspecies, variety, etc.) should be written in italic, except for italic sentence. Scientific name (genera, species, author), and cultivar or strain should be mentioned completely for the first time mentioning it in the body text, especially for taxonomic manuscripts. Name of genera can be shortened after first mentioning, except generating confusion. Name of the author can be eliminated after first mentioning. For example, *Rhizopus oryzae* L. UICC 524, hereinafter can be written as *R. oryzae* UICC 524. Using trivial name should be avoided, otherwise generating confusion. **Biochemical and chemical nomenclature** should follow the order of the IUPAC - IUB. For DNA sequence, it is better used Courier New font. Symbols of standard chemical and abbreviation of chemistry name can be applied for common and clear used, for example, completely written butilic hydroxyl toluene (BHT) to be BHT herein after. **Metric measurement** use IS denomination, usage other system should follow the value of equivalent with the denomination of IS first mentioning. Abbreviations set of, like g, mg, mL, etc. do not follow by dot. Minus index ($m^{-2}$, $L^{-1}$, $h^{-1}$) suggested to be used, except in things like "per-plant" or "per-plot". **Equation of mathematics** does not always can be written down in one column with text, in that case can be written separately. **Number** one to ten are expressed with words, except if it relates to measurement, while values above them written in number, except in early sentence. The fraction should be expressed in decimal. In the text, it should be used "%" rather than "percent". Avoid expressing ideas with complicated sentence and verbiage, and used efficient and effective sentence.

**Title** of the article should be written in compact, clear, and informative sentence, preferably not more than 20 words. Name of author(s) should be completely written. **Name and institution** address should also be completely written with street name and number (location), postal code, telephone number, facsimile number, and email address. Manuscript written by a group, author for correspondence along with address is required. First page of the manuscript is used for writing above information.

**Abstract** should not be more than 200 words. **Keywords** is about five words, covering scientific and local name (if any), research theme, and special methods which used; and sorted from A to Z. All important **abbreviations** must be defined at their first mention. **Running title** is about five words. **Introduction** is about 400-600 words, covering the background and aims of the research. **Materials and Methods** should emphasize on the procedures and data analysis. **Results and Discussion** should be written as a series of connecting sentences, however, for manuscript with long discussion should be divided into subtitles. Thorough discussion represents the causal effect mainly explains for why and how the results of the research were taken place, and do not only re-express the mentioned results in the form of sentences. **Concluding** sentence should be given at the end of the discussion. **Acknowledgments** are expressed in a brief; all sources of institutional, private and corporate financial support for the work must be fully acknowledged, and any potential conflicts of interest are noted.

**Figures and Tables** of maximum of three pages should be clearly presented. Title of a picture is written down below the picture, while title of a table is written above the table. Colored figures can only be accepted if the information in the manuscript can lose without those images; chart is preferred to use black and white images. Author could consign any picture or photo for the front cover, although it does not print in the manuscript. All images property of others should be mentioned source. **There is no appendix**, all data or data analysis are incorporated into Results and Discussions. For broad data, it can be displayed on the website as a supplement.

**References:** Author-year citations are required. In the text give the authors name followed by the year of publication and arrange from oldest to newest and from A to Z. In citing an article written by two authors, both of them should be mentioned, however, for three and more authors only the first author is mentioned followed by et al., for example: Saharjo and Nurhayati (2006) or (Boonkerd 2003a, b, c; Sugiyarto 2004; El-Bana and Nijs 2005; Balagadde et al. 2008; Webb et al. 2008). Extent citation as shown with word "*cit*" should be avoided. Reference to unpublished data and personal communication should not appear in the list but should be cited in the text only (e.g., Rifai MA 2007, pers. com. (personal communication); Setyawan AD 2007, unpublished data). In the reference list, the references should be listed in an alphabetical order (better, if only 20 for research papers). Names of journals should be abbreviated. Always use the standard abbreviation of a journal's name according to the **ISSN List of Title Word Abbreviations** (www.issn.org/2-22661-LTWA-online.php). The following examples are for guidance.

**Journal:**

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.

**Book:**

Rai MK, Carpinella C. 2006. Naturally Occurring Bioactive Compounds. Elsevier, Amsterdam.

**Chapter in 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:**

Sugiyarto. 2004. Soil Macro-invertebrates Diversity and Inter-Cropping Plants Productivity in Agroforestry System based on Sengon. [Dissertation]. Universitas Brawijaya, Malang. [Indonesian]

**Information from 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. www.molecularsystemsbiology.com. DOI:10.1038/msb.2008.24

THIS PAGE INTENTIONALLY LEFT BLANK

**OCEAN LIFE**
Volume 4, Number 2, December 2020 E-ISSN: 2580-4529
Pages: 51-62 DOI: 10.13057/oceanlife/o040201

# Physicochemical factors influencing zonation patterns, niche width and tolerances of dominant mangroves in southern Oriental Mindoro, Philippines

**AARON FROILAN M. RAGANAS[1,♥], DAMASA B. MAGCALE-MACANDOG[2]**
[1]Department of Biological Sciences and Environmental Studies, College of Arts and Sciences, University of the Philippines Mindanao, Mintal, Davao City 8022, Philippines. Tel.: +63-82-293-0302, Fax.: +63-82-293-0312, ♥email: amraganas@up.edu.ph
[2]Institute of Biological Sciences, College of Arts and Sciences/School of Environmental Science and Management, University of the Philippines Los Banos, College 4031, Laguna, Philippines

Manuscript received: 9 June 2020. Revision accepted: 29 August 2020.

**Abstract.** *Raganas AFM, Magcale-Macandog DBM. 2020. Physicochemical factors influencing zonation patterns, niche width, and tolerances of dominant mangroves in southern Oriental Mindoro, Philippines. Ocean Life 4: 51-62.* Physicochemical factors strongly influence mangroves' spatial patterns and structural complexity. In this regard, we aimed to contribute to filling this information gap in the six mangrove ecosystems on the southern coast of Oriental Mindoro, Philippines. In each of the six mangrove ecosystems, the dominant mangrove species were identified in four mangrove ecotypes - seaward, riverine, middle, and landward - using a stratified random sampling method for vegetation survey. These ecozones also obtained physiological parameters of water, air, and soil. Results of the Principal Component Analysis revealed that temperature and water salinity are the factors that show a strong influence on the spatial distribution of the dominant mangrove species. Canonical Correspondence Analysis revealed that *Avicennia marina, Sonneratia alba,* and *Rhizophora apiculata* are species associated with a highly saline environment. At the same time, *Xylocarpus granatum, Ceriops decandra, Avicennia rumphiana,* and *Rhizophora mucronata* are species associated with low to an optimum saline environments. Most of these dominant species preferred ecotypes with low to optimum salinity levels, as revealed by their individual niche width and tolerances. The different adaptations and dominance of these mangrove species provide insights into identifying appropriate species that could be used as planting materials for the rehabilitation endeavors of the respective mangrove ecosystem.

**Keywords:** Dominant mangrove species, ecotypes, physicochemical factors, stratified random sampling, zonation patterns

**Abbreviations:** CA: Correspondence Analysis, CCA: Canonical Correspondence Analysis, CI-III: Dendrogram clusters, cm: centimeter, DO: Dissolved Oxygen, lux: the amount of illumination, LZ: Landward Zone, MZ: Middle Zone, PAST: Paleontological Statistics, PCA: Principal Component Analysis, pH: degree of acidity and basicity of soil and water, ppm: parts per million, psu: practical salinity unit, RH: Relative humidity, RZ: Riverine Zone, SA: Marine water category classified as "protected marine waters," SZ: Seaward zone

## INTRODUCTION

In many environmental settings, the physicochemical factors are among the most important parameters that regulate the structural characteristics of a plant community. In the mangrove environment, physicochemical factors greatly influence the ecosystem's structural development and productivity (Das et al., 2019). The physiological tolerance of different mangrove species to waterlogging, salinity, sulfides, nutrients, sedimentation, soil texture, nutrients, and redox potential has been linked with their structural and distribution patterns (Cardona and Botero 1998; Sherman et al. 1998; Das et al. 2019). The development of each mangrove species is influenced by the physicochemical characteristics of soil, which may compromise their growth and structure (Perera et al., 2013; Harahap et al., 2015; Bomfim et al., 2018). Perhaps, the soils and mangrove vegetation have a strong interaction with each other, resulting in the formation process of both the soil and the characteristic of the growing mangrove plants (Bomfim et al., 2018).

Among the aforementioned physicochemical factors, salinity is considered the limiting factor that has a critical role in the establishment and productivity of mangroves, aside from the influence inflicted by humans and other biotic factors (Ball 2002; Feller et al. 2010; Kodikara et al. 2018). The variations in water salinity and the corresponding ability of mangrove species to adapt to saline conditions significantly contribute to their growth and distribution patterns (Bomfim et al., 2018). Mangroves growing in habitats with lower salinity are likely to grow more rapidly than those living in the highly saline habitat (Perera et al., 2013). The differences in the mangrove environment can result in a particular species' dominance, leading to their habitat differentiation.

Water quality is also among the parameters that provide basic scientific information in understanding the physical and chemical influences in the mangrove environment (Mariappan et al., 2016). For example, the patterns of tidal inundation in different local settings influence the mangrove soil characteristics controlling species zonation in the mangrove ecosystem (Joshi and Ghose 2003;

Chandarasekara and Dissanayake 2014; Bomfim et al. 2018). Furthermore, the increase in temperature due to climate change can also cause stress to mangrove seedlings (Lovelock et al., 2009; Gillis et al., 2019). Hence, mangrove forests depend on seedling survival for expansion and maintenance (Gillis et al., 2019). With those conditions, it is apparent that mangrove species distribution is governed by the complexity of the mangrove environment conditions (Joshi and Ghose 2003; Van Tang et al. 2020).

In the Philippines, studying physicochemical factors and their influence on mangrove distribution patterns is scarce. Such a study has not yet been conducted in Oriental Mindoro province. From this perspective, the present study was carried out to investigate the ecology and spatial aspects of various dominant mangrove species in six mangrove areas in southern Oriental Mindoro, Philippines. Specifically, this study aimed to determine the physicochemical factors influencing the mangrove zonation patterns and their individual niche width and salinity tolerances. Knowledge from this study will guide the local environment sectors in understanding the mangrove ecosystem complexity in the province, which will fill the information gap regarding physicochemical influences on the spatial distribution patterns of the dominant mangroves on the southern coast of Oriental Mindoro, Philippines.

## MATERIALS AND METHODS

### Study sites

The study sites are located in the southern district of Oriental Mindoro (Figure 1), consisting of six mangrove areas in the municipalities of Gloria, Bansud, Bongabong, Roxas, Mansalay, and Bulalacao (from 12° 53'N and 121° 29'E to 12° 19'N and 121° 21'E). The coastal bays in these municipalities are commonly used for recreational activities such as bathing, swimming, and diving. In addition, several marine reserves and protected areas are also located on the coast of these municipalities. According to the Department of Environment and Natural Resources Administrative Order 2016-08 (DENR-AO 2016-08) water classification standard, two of the coastal bays in the province are formally classified as Class SA. One is Bulalacao bay, which was one of the sampling sites. This water classification category is considered protected marine waters designated for national or local parks. The other coastal areas are classified as fishery water class I, suitable for shellfish harvesting for direct human consumption (SOCOM 2015).

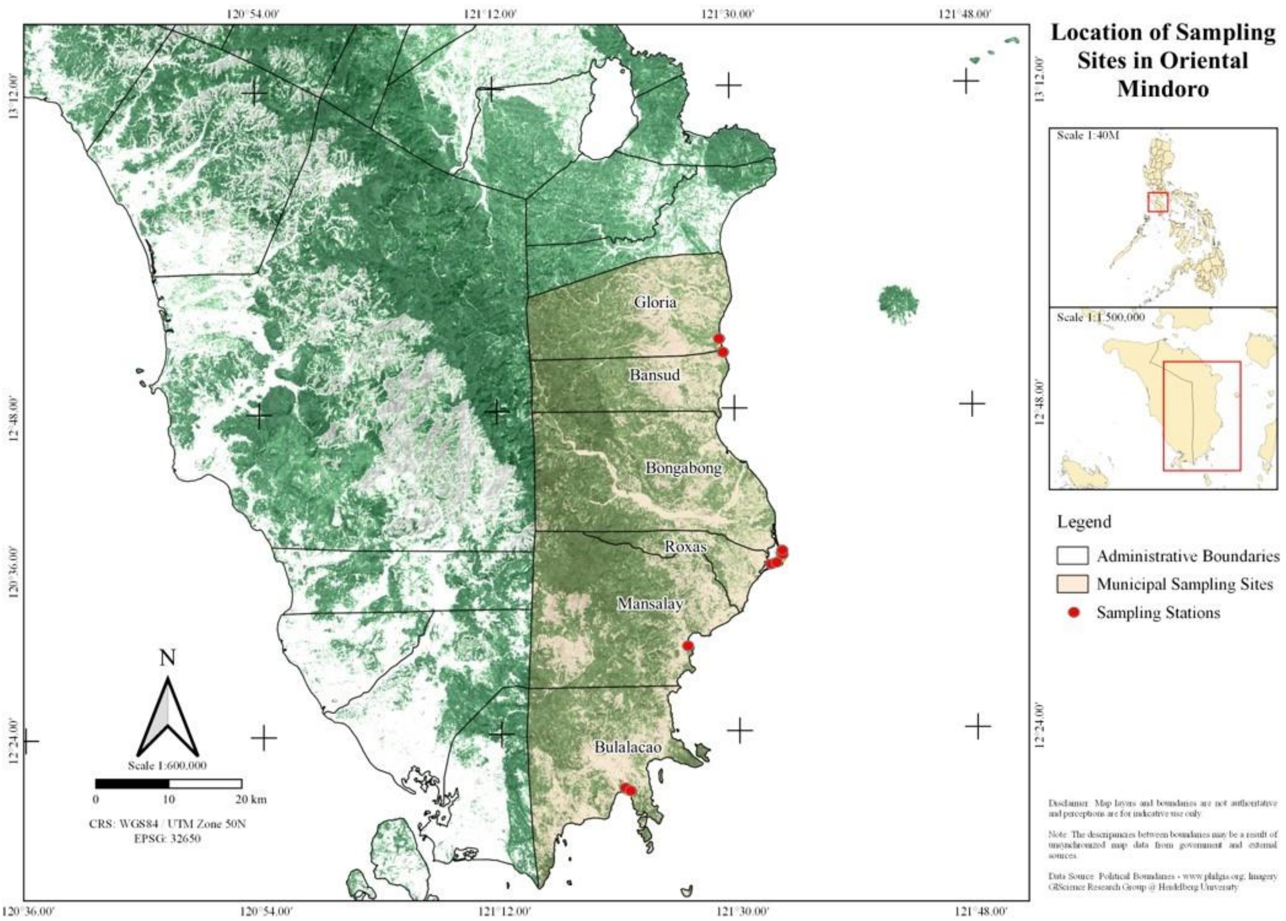



**Figure 1.** Location of sampling sites in the southern Oriental Mindoro, Philippines. Map generated using QGIS v. 3.3.10.

In terms of climate, the study areas have two climatic types: the type 1 climate (from western Mansalay to Bulalacao), with pronounced seasons, dry from November to April and wet during the rest of the year; and Type 3 climate (from Gloria to the eastern part of Mansalay), with seasons, not very pronounced, relatively dry from November to April, and wet during the rest of the year (SOCOM 2015). However, the whole province is under climate type 3, according to Basconcillo et al. (2016), with a short dry season from December to February and a wet season the rest of the months. The average annual rainfall in the province is 2,285 mm, with a mean of 177 mm during rainy days. The minimum amount of rainfall is 65 mm in February, while the maximum is 325 mm in October. Average annual temperatures are at a maximum of 33.4°C recorded in October, while the minimum is 19.2°C in December (SOCOM 2015).

The study sites were selected based on the zonal ecotypes observed in the mangrove ecosystem, such as seaward, middle, landward, and riverine zones. The classification was based on the following features: seaward zone- situated at the intertidal zone where mangroves are daily submerged to seawater; middle zone- situated at the transition zone between the seaward and the landward zones, where a combination of species from both zones was observed. Landward is the zone inland from the middle zone's boundary, and the riverine zone is situated along the river banks.

**Sampling procedures**

A stratified random sampling method was employed to determine the dominant mangrove species across each mangrove ecosystem's ecotypes-seaward, middle, landward, and riverine zones. Depending on the mangrove stand's size and geomorphology, five plots were established at each zone, parallel or perpendicular. Each plot measuring 10 x 10 $m^2$ was laid within a 100-meter transect line with 20-meter intervals using Gareth's (1991) method. First, all the mangrove species found dominant in each ecotype were noted and identified up to species level using the field guide to Philippine mangroves by Primavera (2004) and with the help of the local field guides. Next, the dominant mangrove species was identified through a vegetation survey, considering the species with the highest importance value (Raganas et al. 2019, unpublished data). Finally, these dominant mangrove species were identified to determine their distribution and zonation patterns across ecotypes in all mangrove areas. The result of this analysis was presented through a cluster dendrogram using the Jaccard similarity index (presence/absence data).

**Physicochemical parameters**

The physicochemical parameter data were taken from the five sampling plots established in each mangrove ecotype. A Geographic Positioning System (GPS) navigation device was used to obtain coordinates from all sampling plots. YSI Multi-parameter Professional Pro equipment collected water quality data such as water pH, temperature, dissolved oxygen, and conductivity. Soil and air temperatures were obtained using a conventional thermometer while slinging a psychrometer for relative humidity. A soil pH meter was used to determine the acidity and basicity of the mangrove soil. Sediment depths were measured through an improvised bamboo stick and then measured using a tape measure (cm). Finally, a light meter was used to obtain the light intensity in the mangrove forest's open and shaded canopies. All these physicochemical measurements were obtained three (3) times a day per ecotype; in the morning (7:00-9:00), noon (11:00-12:00), and afternoon (3:00-4:00) for two consecutive days. Data were collected from the third week of October to the end of November 2018. The readings from all the measurements were then computed to get the average. A total of 11 physicochemical parameters were considered in the study.

**Statistical analyses**

Parametric One-Sample t-Test was performed to test significant differences among the average values of physicochemical parameters obtained from all study sites. This parametric test was used after log-transformed data and obtained a normalized dataset. Various multivariate statistical tools were also performed, such as the Principal Component Analysis (PCA), Canonical Correspondence Analysis (CCA), and Correspondence Analysis (CA) in the analyses of data. Principal Component Analysis was used to identify dataset variables with maximum variances. It gives information on high-importance variables in the data set and removes redundant or less important variables. It is an ordination diagram consisting of points and lines (vectors) representing the dependent and independent variables. The 11 physicochemical components chosen in the study were used for this analysis to determine their influence on the six mangrove ecosystems. These variables were grouped into two categories: the non-water components, including air and soil temperatures, soil pH, relative humidity, light intensity (open and shade), and sediment depths (cm); and the water quality components, which include water salinity, water pH, water temperature and dissolved oxygen. The water salinity value (psu) was derived from the water conductivity data to determine the total concentration of salts suspended in the sea and riverine waters. These components were analyzed in the Paleontological Statistics (PAST) package software version 4.02 (Hammer et al. 2001) using a correlation matrix (normalized variance-covariance) since variables are of different units. The eigenvalues of each environmental component were compared to the significant Jacliffe cut-off score of 0.7. Components with eigenvalues higher than the cut-off value were considered significant. In contrast, components with eigenvalues below the cut-off score were considered insignificant and were excluded from the final analysis.

On the other hand, CCA is also an ordination method used to determine the association between dominant mangrove species and physicochemical components. The significant physicochemical components determined from the PCA and the abundance data of dominant mangrove species were used to generate the model for CCA analysis. The model explains the relationship between the dominant mangrove species and the highly important physicochemical components determined by the PCA. The model presented includes points representing the species and vectors (lines) representing the highly influential physicochemical components. The axes with the highest accounted variances best represented the model's data.

Meanwhile, the niche width and tolerances of the dominant mangrove species were determined using Correspondence Analysis (CA). This analysis determined the habitat preference and tolerances of the dominant mangrove species concerning salinity. All the results from both CCA and CA were also generated using PAST software version 4.02 and presented through bi-plots.

## RESULTS AND DISCUSSION

### Mangrove species dominance across ecotypes in the study sites

Seven mangrove species were found dominant across ecotypes in the six mangrove areas. The dominant mangrove species include the *Avicennia marina* (Forssk.) Vierh., *Avicennia rumphiana* Hallier f., *Ceriops decandra* (Griff.) W.Theob., *Rhizophora apiculata* Blume, *Rhizophora mucronata* Lam., *Sonneratia alba* Sm. and *Xylocarpus granatum* J.Koenig (Figure 2). The seaward zone in the mangrove stands of Gloria, Bansud, and Bongabong were not considered due to the absence of mangroves in the zone. Dendrogram revealed that four dominant mangrove species, namely A. marina, A. rumphiana, R. apiculata, and R. mucronata, can dominate most or all of the ecotypes. Meanwhile, *C. decanda* and *X. granatum* can dominate the inland ecotypes (middle and landward zones). In contrast, species *S. alba* dominates in the riverine zone only, specifically in the mangrove stand of Bansud. Common dominant species in most mangrove sites were *A. marina, A. rumphiana,* and *R. apiculata*. Among these mangrove areas, the mangrove stand in Gloria has a unique dominant species, as depicted in the dendrogram, separating from the two major clusters (CI and CII). The separation is mainly attributed to species *X. granatum*, which co-dominated the *R. mucronata* in the landward zone of the mangrove stand. No clear species zonation patterns were observed in most of these mangrove areas because one or two particular species can dominate in most ecotypes or the entire mangrove stand.

### Physicochemical factors across mangrove sites

*Non-water components*

Table 1 presents the physicochemical data obtained from six mangrove areas. Amongst all mangrove areas, Gloria (30.3˚C) had the highest average air temperature, while Mansalay (27.4 ˚C) had the lowest. Regarding ecotypes, the middle zone (31.5˚C) of Bansud had the highest average air temperature, while Mansalay also recorded the lowest, particularly in the seaward zone (25.3˚C). A similar trend was observed for the average soil temperature, with Gloria having the highest (29.7˚C), particularly in the seaward zone (31.0˚C), while Mansalay (26.8 ˚C) had the lowest, particularly in a riverine zone (25.5˚C). The lower air and soil temperatures recorded in Mansalay may be attributed to the rainy weather during data collection.

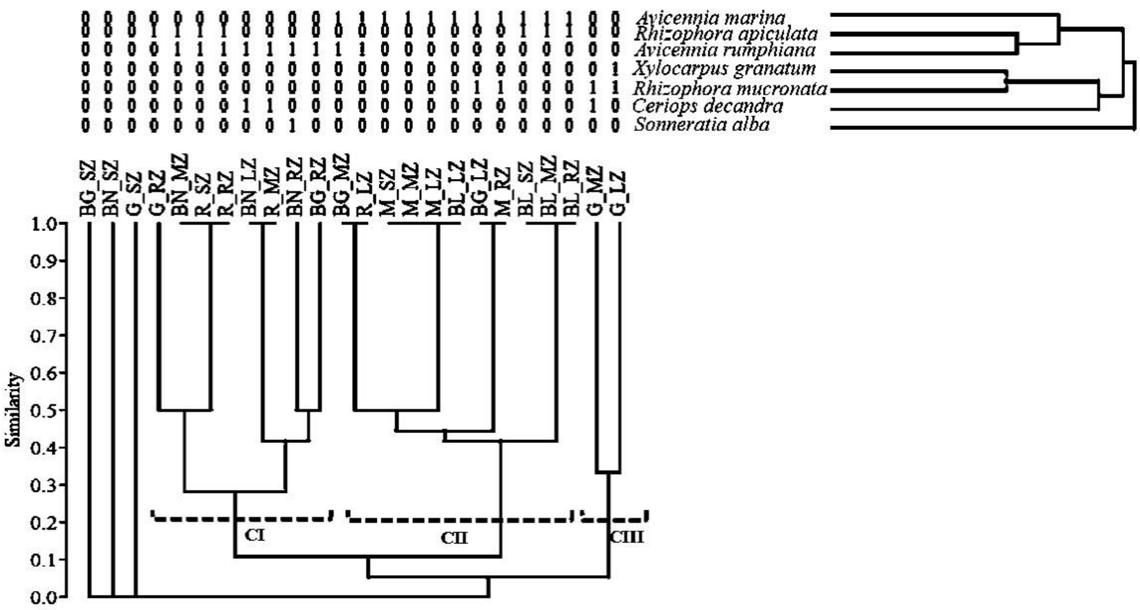



**Figure 2.** The Jaccard similarity index shows mangrove species dominance across ecotypes in six mangrove areas. Site codes: Gloria (G); Bansud (BN); Bongabong (BG); Roxas (R); Mansalay (M); Bulalacao (BL). Ecotypes: Seaward Zone (SZ); Middle Zone (MZ); Landward Zone (LZ); Riverine Zone (RZ). CI-III means clusters. Graph modified from Raganas et al. (2020)

Regarding relative humidity (RH), the mangrove stand in Roxas had the highest average RH with 91.3 %, while Bulalacao had the lowest with 87.5 %. A huge disparity was observed in the light interception in all mangrove sites' open and closed canopies. Results show that light intensity in the open area was significantly higher than in the canopy because the canopy intercepts the light from reaching the ground. Among the mangrove areas, the highest average light intensity was recorded in Roxas (12430.8 lux), particularly in the landward zone (15414.7 lux), while the lowest was recorded in Mansalay (3713.2 lux), particularly in the middle zone (1330.3 lux)

The light interception under mangrove canopies in all study sites also observed a similar trend. However, statistical analysis revealed that light intensities in the open canopies across mangrove sites were significantly different, with Mansalay and Bulalacao being significantly lower. That was attributed to the rainy and gloomy weather during the survey in these mangrove sites at the specific time of the day. On the other hand, Roxas showed significantly higher light intensity among sites, attributed to the highest intensity obtained from the riverine zone.

Regarding soil pH, results show that most mangrove soils were slightly acidic to slightly alkaline. The average soil pH ranges from 6.5 to 7.2, a condition favorable for the growth of mangrove plants. The slightly alkaline soil in Bongabong was attributed to the high pH in the seaward zone (9.3). The average sediment depths of the mangrove soils in all study sites show that the mangrove stand in Bulalacao had the highest sediment depth (75.1 cm), while Bongabong showed the lowest (16.3 cm). The highest sediment deposited in Bulalacao, particularly in the seaward zone (103.4 cm), was attributed to the various networks of interconnecting rivers located within the mangrove stand. Sediments from the upland areas were possibly carried out through these river channels during the rainy season and deposited in the seaward zone.

Meanwhile, the slightly elevated topography of the riverbank in the mangrove stand of Bongabong was assumed to be the result of the lower sediment deposition in the mangrove area. Statistical analysis revealed that the sediments in the mangrove stands of Mansalay and Bulalacao were significantly deeper than the other mangrove stands. The deepest sediment deposition was recorded in the riverine zones of most mangrove sites except in Bulalacao.

*Water components*

The highest average water salinity across study sites was recorded in Gloria (16.9 psu), while the lowest was recorded in Bansud (7.3 psu). Regarding ecotypes, the salinity level between sea and riverine waters across mangrove sites showed some degree of disparities. As observed, sea waters had a higher salinity level than riverine waters because sea waters have higher salt concentrations than riverine waters. The seaward zone of Gloria (22.1 psu) recorded the highest salinity level, while Bansud (10.7 psu) recorded the lowest. Among riverine zones, the highest salinity was recorded in Gloria (11.6 psu), while the lowest was in Mansalay (2.1 psu).

For the average water temperature across study sites, Bansud (28.9˚C) recorded the highest, while Mansalay (26.1˚C) recorded the lowest. Across ecotypes, the seaward zone in Bansud (29.5 ºC) recorded the highest, while Mansalay (25.6 ºC) recorded the lowest. For the riverine zones, the highest was recorded in Bongabong (28.8 ºC), while the lowest was recorded in Mansalay (26.5 ºC). Results further revealed that the average water temperature in the seaward zones of the mangrove stands in Gloria, Bansud and Roxas were a bit higher compared with the riverine zones. That contrasts with the results of the mangrove stands in Bongabong, Mansalay, and Bulalacao, where riverine zones have higher average water temperatures. The high water temperature recorded in the riverine zones of the latter mangrove areas was attributed to the sunny weather during the data gathering.

The seaward and riverine zones across mangrove sites show little variation regarding water pH. As observed, the water pH in the seaward zones was higher compared with the riverine zones. But the highest water pH was recorded in Bongabong (8.3), while the lowest was in Mansalay (7.8). Bongabong and Roxas recorded the highest water pH (8.6) among the seaward zones, while Mansalay (8.2) recorded the lowest. For the riverine zones, the highest pH was recorded in Bansud (8.0), while the lowest was recorded in Mansalay and Bulalacao (7.4). Moreover, both zones have alkaline water values above neutral (pH>7). The high pH level in the seaward zone was also attributed to the higher salt concentrations in the seawater.

For the dissolved oxygen (DO) present in water, results show that riverine waters had a higher presence of DO than sea waters. Across sites, the highest DO was recorded in Gloria (6.0 ppm), while the lowest was in Bulalacao (4.3 ppm). Bansud recorded the highest DO (5.7 ppm) among the seaward zones, while Bulalacao recorded the lowest (3.9 ppm). Next, among the riverine zones, the highest DO was recorded in Gloria (7.4 ppm), while the lowest was recorded in Bulalacao (4.6 ppm). The average DO present in the sea, and riverine waters across mangrove sites are favorable for maintaining aquatic life, which is above 4 ppm, except in the seaward zone of Bulalacao, which is quite below the threshold.

The specific conditions influenced all the variations in the physicochemical data across study sites during the time of the survey, zonation, and location. For example, rainy and gloomy weather was experienced in other study sites, especially in Mansalay and Bulalacao, under climate type 3 (Basconcillo et al. 2016) with the rainy season during sampling months.

**Physicochemical influences across ecotypes**

Principal component analysis revealed six out of 11 physicochemical parameters significantly influenced the distribution and dominance of mangrove species in different ecotypes across mangrove sites. Of the 11 physicochemical parameters tested, four non-water and two water components were found significant (Tables 2 and 3). The non-water components (Table 2) with eigenvalues greater than the Jacliffe cut-off score (0.7) are air and soil temperatures, relative humidity, and light intensity (open space) with eigenvalues of 2.81, 1.26, 1.10, and 1.03; and accounted variances of 40.17%, 17.95%, 15.67%, and 14.77%, respectively. For water components (Table 3), PCA shows that water temperature and salinity had a significant influence, as indicated by their eigenvalues of 2.08 and 1.20, with accounted variances of 51.88% and 29.98%, respectively. The results were also presented in scree plots (Figure 3), showing the downward curve of eigenvalues (largest to smallest) contributed by the 11 physicochemical parameters tested for the analysis.

The ordination diagram (Figure 4) depicts the grouping of mangrove ecotypes as influenced by the six highly correlated physicochemical components. The distribution of ecotypes is greatly influenced by temperature and salinity. Upper axes are ecotypes with higher salinity levels and more open areas, as indicated by the increasing light intensity, water, soil, and air temperatures. These ecotypes are the seaward and riverine environs of the mangrove forests in the study sites. Meanwhile, the lower axes are ecotypes with low salinity levels with relatively cooler temperatures (high relative humidity) represented by the middle and landward zones of the mangrove forests. These ecotypes are not regularly inundated by seawater, hence situated inland and have relatively intact forest canopies.

**Table 1.** Physicochemical data across ecotypes in six mangrove ecosystems in southern Oriental Mindoro, Philippines

| | | Non-water components | | | | | | | Water components | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Site** | **Ecotype** | **Air temp (˚C)** | **Soil temp (˚C)** | **RH (%)** | **Light open (lux)** | **Light shade (lux)** | **Soil pH** | **Sediment depth (cm)** | **Water salinity (psu)** | **Water temp (˚C)** | **Water pH** | **DO (ppm)** |
| Gloria | Seaward | 29.0$^{a}$ | 31.0$^{a}$ | 86$^{a}$ | 10364$^{c}$ | 1511$^{a}$ | 8.4$^{a}$ | 0 | 22.1$^{a}$ | 28.2$^{a}$ | 8.5$^{a}$ | 4.6$^{a}$ |
| | Middle | 30.5$^{a}$ | 29.0$^{a}$ | 93$^{a}$ | 9472$^{b}$ | 1223$^{a}$ | 5.6$^{a}$ | 12$^{a}$ | 0 | 0 | 0 | 0 |
| | Landward | 30.5$^{a}$ | 28.5$^{a}$ | 93$^{a}$ | 8627.3$^{a}$ | 1287.3$^{a}$ | 5.3$^{a}$ | 14.6$^{a}$ | 0 | 0 | 0 | 0 |
| | Riverine | 31.3$^{a}$ | 30.2$^{a}$ | 86$^{a}$ | 9327.7$^{b}$ | 1430.7$^{a}$ | 7.1$^{a}$ | 66$^{b}$ | 11.6$^{a}$ | 27.7$^{a}$ | 7.6$^{a}$ | 7.4$^{a}$ |
| | Average | 30.3$^{A}$ | 29.7$^{A}$ | 89.5$^{A}$ | 9447.8$^{D}$ | 1363.0$^{A}$ | 6.6$^{A}$ | 23.2$^{A}$ | 16.9$^{A}$ | 27.9$^{A}$ | 8.1$^{A}$ | 6.0$^{A}$ |
| Bansud | Seaward | 29.6$^{a}$ | 29.3$^{a}$ | 93$^{a}$ | 9844.7$^{c}$ | 2536.7$^{c}$ | 8.0$^{a}$ | 0 | 10.7$^{a}$ | 29.5$^{a}$ | 8.3$^{a}$ | 5.7$^{a}$ |
| | Middle | 31.5$^{a}$ | 28.4$^{a}$ | 86$^{a}$ | 9233.7$^{c}$ | 2295.7$^{c}$ | 6.5$^{a}$ | 34$^{a}$ | 0 | 0 | 0 | 0 |
| | Landward | 28.3$^{a}$ | 28.1$^{a}$ | 93$^{a}$ | 8079.3$^{b}$ | 863$^{a}$ | 6.8$^{a}$ | 9$^{a}$ | 0 | 0 | 0 | 0 |
| | Riverine | 29.8$^{a}$ | 28.7$^{a}$ | 86$^{a}$ | 7617.7$^{a}$ | 1577.7$^{b}$ | 6.4$^{a}$ | 90$^{b}$ | 3.9$^{a}$ | 28.3$^{a}$ | 8.0$^{a}$ | 6.1$^{a}$ |
| | Average | 29.8$^{A}$ | 28.6$^{A}$ | 89.5$^{A}$ | 8693.9$^{C}$ | 1818.3$^{A}$ | 6.9$^{A}$ | 33.3$^{AB}$ | 7.3$^{A}$ | 28.9$^{A}$ | 8.2$^{A}$ | 5.9$^{A}$ |
| Bongabong | Seaward | 29.5$^{a}$ | 28.9$^{a}$ | 86$^{a}$ | 10113$^{c}$ | 1165$^{a}$ | 9.3$^{a}$ | 0 | 16.1$^{a}$ | 28$^{a}$ | 8.6$^{a}$ | 4$^{a}$ |
| | Middle | 28.0$^{a}$ | 28.5$^{a}$ | 93$^{a}$ | 9045$^{b}$ | 1083$^{a}$ | 6.3$^{a}$ | 18$^{a}$ | 0 | 0 | 0 | 0 |
| | Landward | 28.0$^{a}$ | 28.6$^{a}$ | 93$^{a}$ | 8036.3$^{a}$ | 1882.3$^{a}$ | 6.4$^{a}$ | 16$^{a}$ | 0 | 0 | 0 | 0 |
| | Riverine | 29.5$^{a}$ | 27.0$^{a}$ | 86$^{a}$ | 10473$^{c}$ | 1452$^{a}$ | 6.6$^{a}$ | 31$^{a}$ | 3$^{a}$ | 28.8$^{a}$ | 7.9$^{a}$ | 5.3$^{a}$ |
| | Average | 28.8$^{A}$ | 28.3$^{A}$ | 89.5$^{A}$ | 9416.8$^{D}$ | 1395.6$^{A}$ | 7.2$^{A}$ | 16.3$^{A}$ | 9.6$^{A}$ | 28.4$^{A}$ | 8.3$^{A}$ | 4.7$^{A}$ |
| Roxas | Seaward | 27.5$^{a}$ | 27.9$^{a}$ | 93$^{a}$ | 10710.3$^{b}$ | 2905.7$^{b}$ | 8.0$^{a}$ | 63$^{b}$ | 12.6$^{a}$ | 28.3$^{a}$ | 8.6$^{a}$ | 5.5$^{a}$ |
| | Middle | 29.3$^{a}$ | 27.5$^{a}$ | 93$^{a}$ | 8468$^{a}$ | 1541$^{a}$ | 6.5$^{a}$ | 13$^{a}$ | 0 | 0 | 0 | 0 |
| | Landward | 30.2$^{a}$ | 29.7$^{a}$ | 93$^{a}$ | 15414.7$^{c}$ | 2136$^{b}$ | 5.2$^{a}$ | 11$^{a}$ | 0 | 0 | 0 | 0 |
| | Riverine | 28.3$^{a}$ | 28.2$^{a}$ | 86$^{a}$ | 15130.3$^{c}$ | 4145.7$^{c}$ | 6.3$^{a}$ | 43$^{b}$ | 6.3$^{a}$ | 27.5$^{a}$ | 7.8$^{a}$ | 5$^{a}$ |
| | Average | 28.8$^{A}$ | 28.3$^{A}$ | 91.3$^{A}$ | 12430.8$^{E}$ | 2682.1$^{B}$ | 6.5$^{A}$ | 32.5$^{AB}$ | 9.5$^{A}$ | 27.9$^{A}$ | 8.1$^{A}$ | 5.3$^{A}$ |
| Mansalay | Seaward | 25.3$^{a}$ | 25.9$^{a}$ | 92$^{a}$ | 4260.3$^{c}$ | 953.3$^{b}$ | 7.9$^{a}$ | 56$^{b}$ | 20.3$^{a}$ | 25.6$^{a}$ | 8.2$^{a}$ | 4$^{a}$ |
| | Middle | 27.9$^{a}$ | 28.5$^{a}$ | 93$^{a}$ | 1330.3$^{a}$ | 516$^{a}$ | 6.5$^{a}$ | 24$^{a}$ | 0 | 0 | 0 | 0 |
| | Landward | 28.3$^{a}$ | 27.3$^{a}$ | 86$^{a}$ | 5900$^{d}$ | 1644.3$^{c}$ | 6.7$^{a}$ | 41$^{b}$ | 0 | 0 | 0 | 0 |
| | Riverine | 28.2$^{a}$ | 25.5$^{a}$ | 86$^{a}$ | 3362.3$^{b}$ | 933.7$^{b}$ | 6.6$^{a}$ | 128$^{c}$ | 2.1$^{a}$ | 26.5$^{a}$ | 7.4$^{a}$ | 5.3$^{a}$ |
| | Average | 27.4$^{A}$ | 26.8$^{A}$ | 89.3$^{A}$ | 3713.2$^{A}$ | 1011.8$^{A}$ | 6.9$^{A}$ | 62.3$^{B}$ | 11.2$^{A}$ | 26.1$^{A}$ | 7.8$^{A}$ | 4.7$^{A}$ |
| Bulalacao | Seaward | 28.5$^{a}$ | 27.0$^{a}$ | 86$^{a}$ | 6985.3$^{b}$ | 1259.3$^{b}$ | 7.7$^{a}$ | 103.4$^{c}$ | 17$^{a}$ | 27.9$^{a}$ | 8.5$^{a}$ | 3.9$^{a}$ |
| | Middle | 25.5$^{a}$ | 26.0$^{a}$ | 85$^{a}$ | 6647$^{b}$ | 1247.3$^{b}$ | 6.3$^{a}$ | 88$^{bc}$ | 0 | 0 | 0 | 0 |
| | Landward | 28.8$^{a}$ | 27.5$^{a}$ | 93$^{a}$ | 3031.3$^{a}$ | 880.7$^{a}$ | 6.3$^{a}$ | 47$^{a}$ | 0 | 0 | 0 | 0 |
| | Riverine | 29.2$^{a}$ | 27.0$^{a}$ | 86$^{a}$ | 6158.7$^{b}$ | 1187.7$^{b}$ | 6.5$^{a}$ | 62$^{ab}$ | 3$^{a}$ | 28.3$^{a}$ | 7.4$^{a}$ | 4.6$^{a}$ |
| | Average | 28.0$^{A}$ | 26.9$^{A}$ | 87.5$^{A}$ | 5705.6$^{B}$ | 1143.8$^{A}$ | 6.7$^{A}$ | 75.1$^{B}$ | 10$^{A}$ | 28.1$^{A}$ | 8.0$^{A}$ | 4.3$^{A}$ |

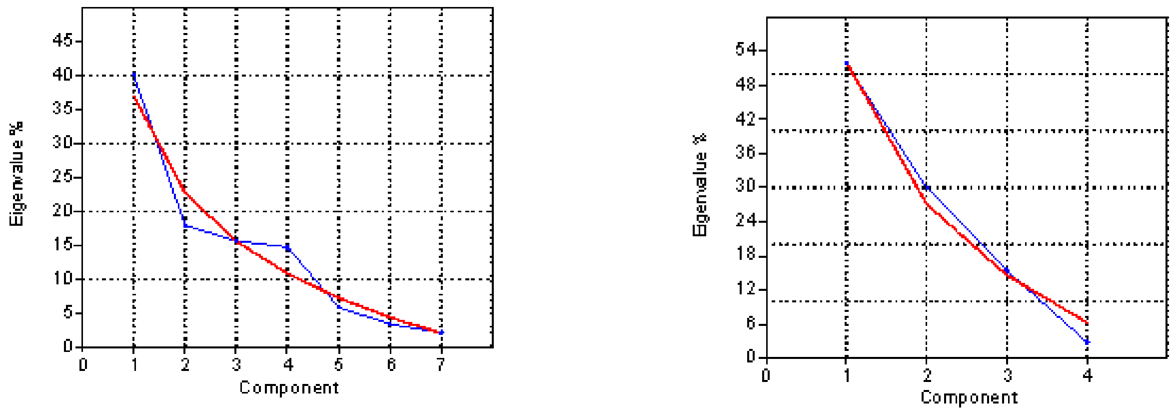
Note: RH (Relative humidity); lux (amount of illumination); cm (centimeter); psu (practical salinity unit); DO (Dissolved Oxygen); ppm (parts per million). The values presented are the average measurements of each physicochemical component. Superscript letters indicate significant differences ($p= \leq 0.05$) at a 95% confidence level using the One Sample t-Test. Uppercase letters indicate comparisons of physicochemical average means across sites, while lowercase letters indicate comparisons across ecotypes per site

**Table 2.** Eigenvalues and accounted variances of non-water components used in the PCA analysis based on the Jacliffe significant cut-off score of 0.7, at 95% bootstrapped confidence intervals

| Principal components | Eigenvalue | % Variance |
|---|---|---|
| Air temperature | 2.81 | 40.17 |
| Soil temperature | 1.26 | 17.95 |
| Relative humidity | 1.10 | 15.67 |
| Light intensity (open) | 1.03 | 14.77 |
| Light intensity (shade) | 0.41 | 5.88 |
| Soil pH | 0.24 | 3.41 |
| Sediment depth | 0.15 | 2.16 |

**Table 3.** Eigenvalues and accounted variances of water components used in the PCA analysis based on the Jacliffe significant cut-off score of 0.7, at 95% bootstrapped confidence intervals

| Principal components | Eigenvalue | % Variance |
|---|---|---|
| Water temperature | 2.08 | 51.88 |
| Water salinity | 1.20 | 29.98 |
| Water ph | 0.62 | 15.40 |
| Dissolved oxygen | 0.11 | 2.74 |




**A** **B**

**Figure 3.** Scree plots of non-water (A) and water (B) components with their eigenvalues (red) and accounted for %variances (blue)

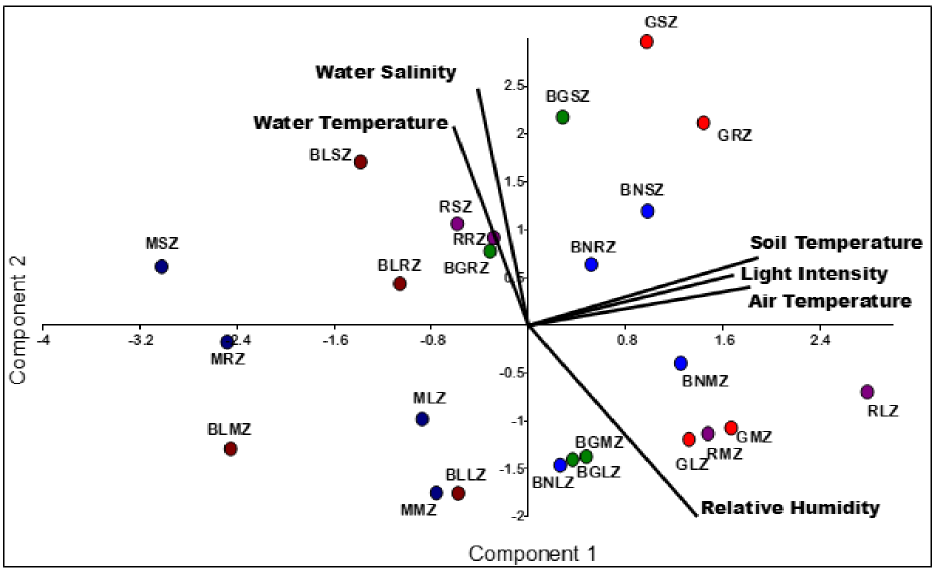



**Figure 4.** Bi-plot showed the significantly influential physicochemical components across ecotypes in all mangrove sites. Site codes: Gloria (G); Bansud (BN); Bongabong (BG); Roxas (R); Mansalay (M); Bulalacao (BL); Ecotypes: Seaward (SZ); Midzone (MZ); Landward (LZ); Riverine (RZ)

The diagram further shows mangrove areas with the presence and absence of mangroves along the seaward zone. Mangrove areas without mangroves along the shore are situated at the right-hand axes, while those having mangroves along the shore are situated at the left-hand axes. The absence of mangroves along the shore was observed in Gloria, Bansud, and Bongabong, which was also portrayed in the cluster dendrogram (Figure 2).

### Association between dominant mangroves and influential physicochemical components

Figure 5 presents the association between the dominant mangrove species and the significantly influential physicochemical components in six mangrove sites. Axes 1 and 2 (Table 4) were used to plot the CCA model since they showed higher accounted variances (52.59%; 31.81%). The diagram depicts the distribution of various dominant mangrove species as influenced by the highly influential physicochemical components determined by the PCA. Upper axes are mangrove ecotypes near the sea, as indicated by the increasing salinity level, while lower axes indicate habitats away from the sea. The diagram suggests that the dominant mangrove species closer to the sea are *A. marina*, *R. apiculata*, and *S. alba*. These species are considered with high tolerance to salinity and thus, can be found thriving in a highly saline ecotype. The species *A. rumphiana* and *R. mucronata* are adapted to moderately saline ecotype, while *C. decandra* and *X. granatum* are adapted to ecotypes with lower salinity, such as in the transition zone to the inland parts of the mangrove forest. Diagram further suggests that species *R. mucronata*, *C. decandra*, and *X. granatum* are the species found at ecotypes with high relative humidity, indicating an association to dense canopy cover. The shrub species *C. decandra* was mostly encountered under the canopies of taller mangrove trees, hence the species associated with these ecotypes. Species *A. rumphiana* was associated with zones with high light intensities and air and soil temperatures, indicating mangrove areas with open canopies. *Sonneratia alba*, on the other hand, is considered a generalist species situated near the central axis. It means that this species can be associated with any of these conditions of the ecotypes.

### Niche width and tolerances of dominant mangroves

Bi-plot (Figure 6) presents the niche width and position of the dominant mangrove species regarding their salinity tolerance. The diagram shows that most mangrove species prefer habitats away from the sea. Species *A. marina*, *S. alba*, and *R. apiculata* most likely preferred waterlogged and highly saline habitats such as in the seaward and riverine zones (right-hand axes). For instance, *A. marina* can extend its niche towards the seaward zone and tolerate much higher salinity. However, the said species was also dominant in other ecotypes, as observed in the mangrove stands of Bongabong, Mansalay, and Bulalacao (Figure 1). Species *S. alba* preferred estuarine habitat, but the species was also thriving, ranging from the seaward up to the landward zones in some mangrove areas. A similar pattern was also observed for *R. apiculata*, where the species can be found in any ecotypes but in a more preferred riverine habitat.

**Table 4.** Eigenvalues and accounted variances of the dominant mangrove species and significant physicochemical components computed for Canonical Correspondence Analysis

| Axis | Eigenvalue | %Variance |
|---|---|---|
| 1 | 0.22 | 52.59 |
| 2 | 0.14 | 31.81 |
| 3 | 0.05 | 10.99 |
| 4 | 0.02 | 4.51 |
| 5 | 0.00041 | 0.10 |
| 6 | 2.71E-08 | 6.37E-06 |

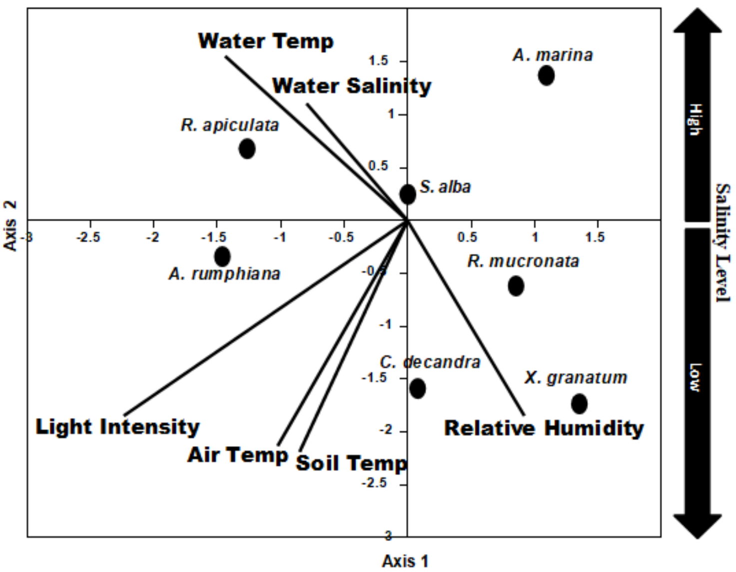



**Figure 5.** Bi-plot showing the association between dominant mangrove species and significantly influential physicochemical components

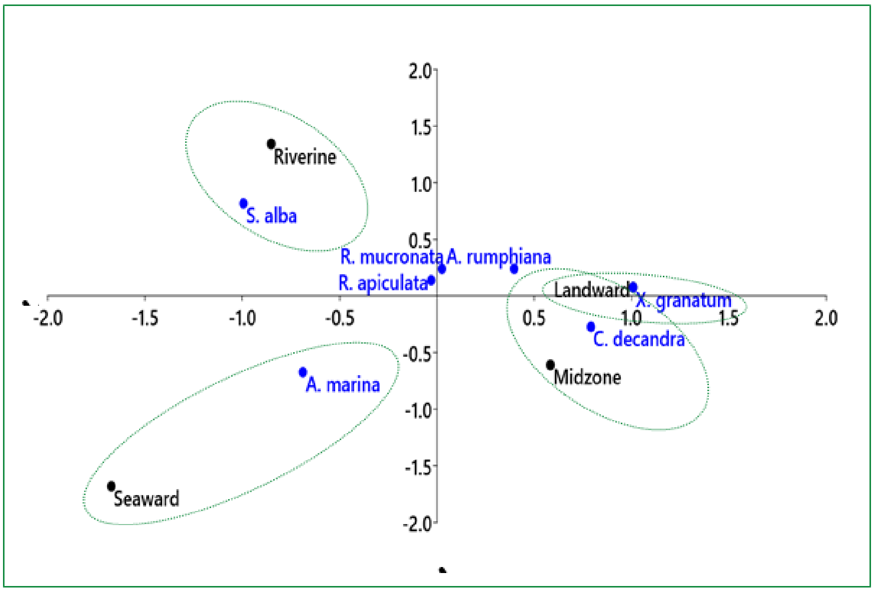



**Figure 6.** Niche width and tolerances of dominant mangrove species across ecotypes

Meanwhile, species *R. mucronata*, *A. rumphiana*, *C. decandra*, and *X. granatum* are the species that prefer inland ecotypes (left-hand axes). Species *X. granatum* preferred landward habitat, hence considered a back mangrove species. *Ceriops decandra* has niche width ranging from middle to landward zones of the mangrove forests. Moreover, species *R. apiculata*, *R. mucronata*, and *A. rumphiana* are situated near the central axis, which means these species can be found in all ecotypes. However, the dominance of *R. mucronata* in a particular ecotype was attributed to the rehabilitation activities where the species was used as planting material in most mangrove sites. Generally, these three species, together with *A. marina*, can extend their niche from seaward up to the landward zones (see Figure 2), thus considered species with wider niche width and salinity tolerances.

## Discussion

The average measurements of the physicochemical parameters related to climates such as temperature, relative humidity, and light intensity depending on the weather conditions during the data collection. At any site, the air temperature is higher than the soil temperature. The canopy cover may keep the soil temperature cooler than the air above it (von Arx et al., 2012). The higher soil temperature than the air temperature recorded in the seaward zone of Gloria is attributed to the sandy texture of the soil in this zone. Sandy soil has a rougher texture, low water holding capacity, and low moisture content, thus absorbing more heat when the temperature rises, making the soil warm. Perhaps, we presumed that it is one of the reasons for the absence of mangroves along the shore in the mangrove stand. The lowest temperatures recorded in Mansalay are attributed to the rainy weather during the study. The slightly acidic to slightly neutral soil pH (6.5 to 7.2) in all mangrove sites is favorable for the growth of dominant mangroves (Mustapha et al., 2016). The salinity level (<25 psu) in all mangrove areas is also favorable for the growth of mangrove seedlings, but the range of tolerance may depend upon the adaptability of each dominant species (Chen and Ye, 2014; Siddique et al., 2017). The higher salinity level recorded in the seaward zones is attributed to the higher salt concentration in sea waters than in the riverine waters. The temperature difference may affect salinity levels which may also have a cascading effect on the pH and DO in the sea and riverine waters (Wilde 2006). The variations in the physicochemical gradients across ecotypes are also influenced by the forest canopy characteristics in respective mangrove stands and the weather condition during the survey.

On the other hand, the six significant components identified by the PCA are highly correlated variables. Most of these variables are influenced by the temperature. For instance, light is important for the growth of plants (Hatfield and Prueger 2015). The variation in the amount of light affects the temperature of the surroundings, such as the air, water, and soil. In particular, too much soil exposure to light, especially in areas with open canopy, can result in higher soil temperature. Mangrove soils can keep the salinity level high, especially when the soil loses moisture due to high temperatures (Ward et al., 2016). The spatial distribution of the dominant mangrove species leading to their dominance in a particular zone or the entire mangrove forest zones can be linked to the influence of temperature in the surroundings. Some studies reported that temperature is one limiting factor in the mangrove environment (Bomfim et al., 2018; Gillis et al., 2019). High temperatures can affect mangroves, especially during the establishment of seedlings. It can inhibit the rooting of mangrove seedlings, thereby reducing their ability to stabilize in the soil (Gillis et al., 2019). Perhaps, there is a critical period for mangrove seedlings where they need to develop root structures to establish in soil because waves and water currents can wash them away (Wang et al., 2018; Gillis et al., 2019). Salinity also affects the distribution and productivity of mangrove plants (Chen and Ye 2014).

Some studies reported that salinity determines the survival and growth performances of mangrove seedlings and is an indicator of their establishment and development (Hoppe-Speer et al. 2011; Chen and Ye 2014; Mariappan et al. 2016), which could be one of the reasons for the zonation and spatial patterns of the dominant mangroves in the study areas. Since high salinity can affect the seedling establishment, this could lead to habitat partitioning among mangrove species, favoring those suitable for the condition. Eventually, this could resolute in the displacement of other species leading to their dominance in a particular ecotype. Several studies have already been conducted regarding the growth performances of mangrove species under different salinity regimes. Studies by Hoppe-Speer et al. (2011), Mahmood et al. (2014a), and ChenandYe (2014) found that some mangrove species have maximum growth performances in minimum salinity, ranging from 0 to 10 psu. However, an increase in salinity can decrease the biomass growth of the mangrove seedlings (ChenandYe 2014; Mahmood et al. 2014b; Kodikara et al. 2018). It was reported that salinity concentration above 25 psu is lethal to mangrove plants (Chen and Ye 2014; Siddique et al. 2017). Since our results for salinity concentrations are below the 25 psu threshold, this might be why other dominant species considered with low to medium salt tolerance can thrive in the seaward zone.

Some mangrove areas have no specific zonation patterns typically described for various mangrove species. We observed that one or two particular species could dominate a mangrove stand, as Feller et al. (2010) described. In our findings, species considered with medium salinity tolerance, such as *A. rumphiana* and *R. mucronata,* can also thrive close to or along the seaward zone. However, the dominance of *R. mucronata* in some ecotypes of the mangrove areas was attributed to the rehabilitation activities where the species was used as planting material. The dominance of *A. rumphiana* along the seaward zone in Roxas might be due to the zone's topography and the sampling site's location near the estuarine, where seawater and freshwater mixed, favoring the growth of the species.

On the contrary, species with high tolerance to salinity, such as *A. marina,* can also dominate inlands in some mangrove areas. The absence of distinct zonation patterns could also be attributed to various factors, including species population dynamics, physiological adaptation, and physical conditions of the mangrove stand (Naskar 2004; Feller et al. 2010). These three aspects could explain the meager degree of zonation patterns observed in the present study. In terms of the adaptability of dominant mangrove species, our findings support the results from other studies conducted, stating that *S. alba, R. apiculata,* and *A. marina* are the species that grow in ecotypes with medium to higher salinities such as in estuarine and seaward zones (Reef and Lovelock 2015). However, these species can also be dominant in low saline habitats such as in the middle and landward zones of the mangrove stand. This scenario is not new since it was observed worldwide across mangrove areas. That explains why species distribution patterns in some mangrove environments are difficult to identify because some show no distinct zonation pattern (Bunt 1996; Schmiegelow et al. 2014; Eswaran et al. 2017). According to Schmiegelow et al. (2014), even with great competition for the same resources among mangrove species, niche partitioning does not necessarily determine the composition of species in the plant community. Coexistence can happen among mangrove species; hence, facilitation can lead to their coexistence over a particular habitat.

In the Philippine setting, the result of the present study somehow agrees with the adaptations of various mangrove species to certain levels of salinity, as reported by Primavera et al. in 2008 and 2011. Accordingly, the mangrove species considered with high salinity tolerances are *A. marina, S. alba,* and *R. apiculata,* hence, portrayed in the CCA and niche tolerance results (Figures 5 & 6). On the other hand, the dominant mangrove species with low to optimum salinity tolerances are *X. granatum, C. decandra, A. rumphiana,* and *R. mucronata*. The varying ecophysiological adaptations of these mangrove species to different salinity levels somehow imply suitable conditions where they can grow best. However, we cannot deny that zonation patterns in other mangrove ecosystems are difficult to delineate since some species can dominate an entire stand.

The specialized rooting systems, salt-secreting glands, and reproductive strategies of various mangrove species are among their adaptive mechanisms toward salinity. The stilt or prop roots, common to *Rhizophora* species, are believed to be more adapted to regularly flooded habitats. The specialized respiratory areal roots (pneumatophores) of *Avicennia* and *Sonneratia* are more adapted to highly saline habitats (e. g. seaward). While the knee root system such as in *Ceriops* and buttress root type of *Xylocarpus* are more adapted to inland habitats (Warming 1883; Feller et al. 2010; Srikanth et al. 2015). The salt-exclusion mechanisms of the dominant mangrove species are also their advantage for adapting to a highly saline habitat. For example, the ultra-filtration mechanism of *R. apiculata* and *R. mucronata* in their roots enables them to survive in the seaward through selective absorption of ions, maintaining low salinity concentration in their body (Noor et al. 2015). Other species such as *A. marina* and *S. alba* have developed secretory structures on their leaves and roots that could enable them to secrete excess salts (Krishnamurthy et al. 2017). These characteristics confer the survival advantage of these mangrove species in a saline environment. With regards to the mangrove reproductive strategies such as vivipary, cryptovivipary, and vegetative propagation described by Bhosale and Mulik (1991) and Feller et al. (2010), viviparous species are said to be more advantageous when growing on the sea borders as their propagules can easily establish after being detached from the mother tree. All these ecological adaptations of dominant mangrove species are essential in understanding their survival advantages under different mangrove environment conditions. That clarifies the individual species distribution patterns, all influenced by the physicochemical factors prevailing in the mangrove environment. Competition appears when the conditions become limiting, wherein those highly adapted to it could out-compete others and become dominant.

Overall, the dominance of mangrove species over a particular ecotype suggests favorable conditions for the species. Temperature and salinity strongly influenced the spatial distribution patterns of these dominant mangroves in the study areas. As revealed in the CCA and niche tolerance results, species A. marina, S. alba, and R. apliculata are tolerant to highly saline environments. Other mangrove species can also thrive close to the sea, except for *X. granatum*, most prefer low to optimum saline habitats (inland and riverine). The dominance of one or two species in most or entire mangrove stands makes it difficult to delineate zonation patterns in these mangrove areas.

Moreover, all the multivariate tests applied in the analyses have provided results useful in portraying significant conclusions in this study. Even though this study utilized only limited environmental parameters in the analyses, the results are somehow comparable with other similar studies. With this, we recommend further studies especially prolonged observation of the physicochemical factors in these mangrove areas. Since we only obtained all the data in a very limited time. Other physicochemical and biotic factors such as the enigmatic effects of hydrologic systems, soil nutrient contents, soil salinity, soil oxygen conditions, heavy metals, and microorganism complexes should also be considered to understand the complexity and dynamics in these mangrove ecosystems fully. Nevertheless, the dominance of mangrove species in a particular ecotype provides insights into what species could be used for future rehabilitation undertakings in these mangrove ecosystems.

## ACKNOWLEDGEMENTS

The authors would like to thank the Asia Pacific Network (APN) through the APNIS Project (Plausible alternative futures of island mangroves in the Asia-Pacific: Scenario-based analysis and quantification of mangrove ecosystem services in coastal hazard mitigation and climate change adaptation (CRRP2018-03MY-Hashimoto) for their financial support in the conduct of this study. Due recognition is also expressed to the provincial and municipal agriculture offices of Oriental Mindoro for their assistance during the conduct of the study. The authors declare no conflict of interest in the publication of this study.

## REFERENCES


Ball MC. 2002. Interactive effects of salinity and irradiance on growth: implications for mangrove forest structure along salinity gradients. Trees-Struct Funct 16: 126-139.

Basconcillo J, Lucero A, Solis A, Sandoval Jr BE, Koizumi T, Kanamaru, H. 2016. Statistically downscaled projected changes in seasonal mean temperature and rainfall in Cagayan Valley, Philippines. J Meteor Soc Japan 2 (94A): 151-164.

Bhosale LJ, Mulik NG. 1991. Endangered mangrove areas of Maharashtra. Proceedings of the Symposium on Significance of Mangroves, Pune, India.

Bomfim MR, Santos JA, Costa OV, et al. 2018. Morphology, physical and chemical characteristics of mangrove soil under riverine and marine influence: A case study on Subaé River Basin, Bahia, Brazil, Mangrove Ecosystem Ecology and Function, Sahadev Sharma, IntechOpen, London, United Kingdom.

Bunt JS. 1996. Mangrove zonation: an examination of data from seventeen riverine estuaries in tropical Australia. Ann Bot 78: 333-341.

Cardona P, Botero L. 1998. Soil characteristics and vegetation structure in a heavily deteriorated mangrove forest in the Caribbean coast of Colombia. Biotropica 30: 24-34.

Chandarasekara W, Dissanayake N. 2014. Effects of mangrove zonation and the physicochemical parameters of soil on the distribution of macrobenthic fauna in Kadolkele mangrove forest, a tropical mangrove forest in Sri Lanka. Adv Ecol 2014: 1-13.

Chen Y, Ye Y. 2014. Effects of salinity and nutrient addition on mangrove *Excoecaria agallocha*. PloS One 9 (4): e93337. DOI: 10.1371/journal.pone.0093337.

Das L, Patel R, Salvi H, Kamboj RD. 2019. Assessment of natural regeneration of mangrove with reference to edaphic factors and water in Southern Gulf of Kachchh, Gujarat, India. Heliyon 5 (8): e02250. DOI: 10.1016/j.heliyon.2019.e02250.

Department of Environment and Natural Resources (DENR). 2016. Water quality guidelines and general effluent standards. DENR-AO 2016-08. http://water.emb.gov.ph/wp-content/uploads/2016/06/DAO-2016-08-WQG-and-GES.pdf

Eswaran Y, Dharanirajan K, Subramanian J, Saravanan, Balasubramaniam J. 2017. Distribution and zonation pattern of mangrove forest in Shoal Bay Creek, Andaman Islands, India. Indian J Mar Sci 46: 597-604.

Feller I, Lovelock C, Berger U, McKee K, Joye S, Ball M. 2010. Biocomplexity in mangrove ecosystems. Ann Rev Mar Sci 2: 395-417.

Gareth W. 1991. Techniques and fieldwork in ecology. Collins Educational Publishers, Hammersmith, LondonGillis LG, Hortua DAS, Zimmer M, Jennerjahn TC, Herbeck LS. 2019. Interactive effects of temperature and nutrients on mangrove seedling growth and implications for establishment. Mar Environ Res 151: 104750.

Hammer O, Harper D, Ryan P. 2001. PAST: Paleontological Statistics Software Package for Education and Data Analysis. Palaeontol Electron 4: 1-9. Version 4.02 https://folk.uio.no/ohammer/past/

Harahap N, Lestariadi RA, Soeprijanto A. 2015. The effect of soil quality on the survival rate of mangrove vegetation. J Eng Appl Sci 10 (7): 154-156.

Hatfield JL, Prueger JH. 2015. Temperature extremes: Effect on plant growth and development. Weather Clim Extrem 10: 4-10.

Hoppe-Speer SCL, Adams JB, Rajkaran A, Bailey D. 2011. The response of the red mangrove *Rhizophora mucronata* Lam. to salinity and inundation in South Africa. Aquat Bot 95: 71-76.

Joshi H, Ghose M. 2003. Forest structure and species distribution along soil salinity and pH gradient in mangrove swamps of the Sundarbans. Trop Ecol 44 (2): 195-204.

Kodikara KAS, Jayatissa LP, Huxham M, Dahdouhguebas F, Koedam N. 2018. The effects of salinity on growth and survival of mangrove seedlings change with age. Acta Bot Bras 32: 37-46.

Krishnamurthy P, Mohanty B, Wijaya E, Lee D, Lim T, Lin Q, Xu J, Loh C, Kumar P. 2017. Transcriptomics analysis of salt stress tolerance in the roots of the mangrove *Avicennia officinalis*. Sci Rep 7: 10031. DOI: 10.1038/s41598-017-10730-2.

Lovelock C, Ball M, Martin K, Feller I. 2009. Nutrient enrichment increases mortality of mangroves. PLoS One 4 (5): e5600. DOI: 10.1371/journal.pone.0005600.

Mahmood H, Saha S, Serajis S, Siddique MRH, Abdullah SMR. 2014b. Salinity influence on germination of four important mangrove species of the Sundarbans, Bangladesh. Agric For 60 (2): 125-135.

Mahmood H, Saha S, Siddique MRH, Hasan MN. 2014a. Salinity stress on growth, nutrients and carbon distribution in seedlings parts of *Heritiera fomes*. Intl J Energy Environ Eng 1 (4): 71-77.

Mariappan N, Ethirajan V, Hari Nivas, A. 2016. A study of water quality status of Mangrove Vegetation in Pichavaram Estuary. J Agric Ecol Res Int 5: 1-11.

Mustapha A, Gandaseca S, Rosli N, Hamzah A, Tindit A, Nyangon L. 2016. Soil pH and Carbon at Different Depth in Three Zones of Mangrove Forest in Sarawak, Malaysia. Malays For 79: 164-173.

Naskar K. 2004. Manual of Indian Mangroves, New Delhi: Daya Publishing House, India
Noor T, Batool N, Mazhar R, Ilyas N. 2015. Effects of Siltation, Temperature and Salinity on Mangrove Plants. Eur Acad Res 2 (11): 14172-14179.
Perera KAR, Amarasinghe MD, Somaratna S. 2013. Vegetation structure and species distribution of mangroves along a soil salinity gradient in a micro tidal estuary on the north-western coast of Sri Lanka. Am J Sci 1: 7-15.
Primavera JH, Esteban JMA. 2008. A review of mangrove rehabilitation in the Philippines: success, failures, and future prospects. Wetland Ecol Manag 16 (5): 345-358.
Primavera JH, Rollon RN, Samson MS. 2011. The pressing challenges of mangrove rehabilitation: Pond reversion and coastal protection. In: Reference Module in Earth Systems and Environmental Sciences. Treatise on Estuarine and Coastal Science. Elsevier, Amsterdam
Primavera JH, Sabada RS, Lebata MJHL, Altamirano JP. 2004. Handbook of Mangroves in the Philippines-Panay. SEAFDEC Aquaculture Department, Iloilo, Philippines
Raganas AFM, Hadsall AS, Pampolina NM, Hotes S, Magcale-Macandog DB. 2020. Regeneration capacity and threats to mangrove areas on the southern coast of Oriental Mindoro, Philippines: Implications to mangrove ecosystem rehabilitation. Biodiversitas 21 (8): 3625-3636.
Raganas AFM, Magcale-Macandog DB, Hadsall AS, Pampolina NM, Hotes S. 2019. Regeneration capacity of mangrove ecosystems on the southern coast of Oriental Mindoro, Philippines: Implication to future mangrove rehabilitation. [Dissertation]. University of the Philippines, Los Banos. [Philippines]
Reef R, Lovelock CE. 2014. Regulation of water balance in mangroves. Ann Bot 115(3): 385-395.
Schmiegelow J, Marcos M, Gianesella S, Maria F. 2014. Absence of Zonation in a Mangrove Forest in Southeastern Brazil. Braz J Oceanogr 62 (2): 117-131.
Sherman RE, Fahey TJ, Howarth RW. 1998. Soil-plant interactions in a neotropical mangrove forest: Iron, phosphorus and sulfur dynamics. Oecologia 115: 553-563.
Siddique M, Raqibul H, Saha S, Salekin S, Hossain M. 2017. Salinity strongly drives the survival, growth, leaf demography, and nutrient partitioning in seedlings of *Xylocarpus granatum* J. König. Iforest 10: 851-856.
Srikanth S, Lum S, Chen Z. 2015. Mangrove root: adaptations and ecological importance. Trees 30: 451-465.
State of the Coasts of Oriental Mindoro (SOCOM) 2015. The provincial government of Oriental Mindoro. [Philippines]
Van Tang T, Rene ER, Binh TN, Behera SK, Phong NT. 2020. Mangroves diversity and erosion mitigation performance in a low salinity soil area: case study of Vinh City, Vietnam. Wetland Ecol Manag 28: 163-176.
von Arx G, Dobbertin M, Rebetez Martine. 2012. Spatio-temporal effects of forest canopy on understory microclimate in along-term experiment in Switzerland. Agric For Meteorol 166-167: 144-155.
Wang W, Li X, Wang M. 2018. Propagule dispersal determines mangrove zonation at intertidal and estuarine scales. Forests 10 (3): 245.
Ward RD, Friess DA, Day RH, MacKenzie RA. 2016. Impacts of climate change on mangrove ecosystems: a region by region overview. Ecosyst Health Sustain 2 (4): e01211. DOI: 10.1002/ehs2.1211
Warming. 1883. Tropiche Fragment II. *Rhizophora mangle* L. Bot Jahrb 4: 519-548.
Wilde F. 2006. Chapter A6. Section 6.1. Temperature: Techniques of Water-Resources Investigations. US Geol Surv 2: 22. http://water.usgs.gov/owq/FieldManual/Chapter6/6.1_ver2.pdf

**OCEAN LIFE**
Volume 4, Number 2, December 2020
Pages: 63-73
E-ISSN: 2580-4529
DOI: 10.13057/oceanlife/o040202

# Impact of salt pond industry on the changes of mangrove ecosystem in Kupang Bay, Timor Island, Indonesia

**ELSSI M. KAPITAN[1], I GUSTI BAGUS ARJANA[2], PRIYO SANTOSO[3,♥]**

[1]Program of Environmental Science, Graduate Program, Universitas Nusa Cendana. Jl. Adi Sucipto Penfui, Kupang 85001, East Nusa Tenggara, Indonesia
[2]Faculty of Teacher Training and Education Science, Universitas Nusa Cendana. Jl. Adi Sucipto Penfui, Kupang 85001, East Nusa Tenggara, Indonesia
[3]Faculty of Fisheries and Marine Science, Universitas Nusa Cendana. Jl. Adi Sucipto Penfui, Kupang 85001, East Nusa Tenggara, Indonesia.
[♥]email: priyosantoso@staf.undana.ac.id

Manuscript received: 1 July 2020. Revision accepted: xxx October 2020.

**Abstract.** *Kapitan EM, Arjana IGB, Santoso P. 2020. Impact of salt pond industry on the changes of mangrove ecosystem in Kupang Bay, Timor Island, Indonesia. Biodiversitas 21: 63-73.* The study of the salt pond industry's effect on changes in Kupang Bay's mangrove ecosystems was carried out between July and September 2018. This study aims to examine the effect of the salt pond area and salt pond production on the dominance and diversity of the mangrove population. The study was conducted by survey method through direct observation in the field and community interviews at three research stations. Data on mangrove vegetation were collected using the line transect method. The study found 9 (nine) species from 4 (four) families. Tree density is relatively rare, while seedling levels are categorized as high. The highest frequency types for tree level are saplings, and seedlings, namely *Ceriops tagal*. The regression analysis showed that the area of salt ponds and the production of salt ponds did not significantly affect the dominance and diversity of the mangrove population ($p>0.05$). However, there was a positive relationship between salt pond area and salt pond production with the dominance of the mangrove population. In comparison, both salt pond area and salt pond production with diverse mangrove populations showed a negative relationship. These results indicate that an increase in the salt pond area and salt production will increase the dominance of the mangrove population but decrease its diversity. Finally, this shows that the influence of the salt pond area and salt pond production has not significantly affected the dominance and diversity of mangrove populations in the current conditions.

**Keywords:** Diversity, dominance, ecosystems, mangroves, production, salt pond

## INTRODUCTION

East Nusa Tenggara (NTT) is an archipelago Province consisting of 566 islands, with a coastline of ± 5,700 km long and a sea area of ± 200,000 km² outside the waters of the Indonesian Exclusive Economic Zone (IEEZ). This province has a wealth of marine resources and fisheries resources that are the potential for increasing community income. Therefore, the potential of coastal and marine resources in NTT for regional economic contributions needs to be explored and pursued at an optimal while maintaining the carrying capacity of the coastal and marine environment for the improvement of the community's economy and NTT's regional income.

One of the marine and fisheries resources utilized by the NTT community, especially residents around the coastal area, is the mangrove ecosystem. Kupang Bay has a long coastline of 17,578 m with a total area estimated at 1,106 ha, making it an area with mangrove forests that coastal communities can utilize. Coastal communities in the Kupang Bay have long used mangrove ecosystems as a source of livelihood in the form of fishing, shrimp, crabs, and shellfish (Santoso et al. 2015). The existing activities are not limited to hunting or fishing activities but also cultivation activities that utilize the suitability of mangrove conditions for milkfish ponds and salt. In addition, this ecosystem plays an ecologically important role as a spawning ground, nursery area, and feeding ground for a number of species in Kupang Bay.

Kupang Bay is an inseparable coastal area of the sea waters of Kupang Regency, where it has an area of marine waters reaching 4,086.33 km² with a coastline length of 551.61 km. This region has a mangrove population that stores a wealth of marine resources (fish and other marine products), which are quite large but have not been managed optimally to improve the regional economy (BPS-Statistics of Kupang Regency, 2020). Kupang Bay in NTT Province is also designated as one of the Marine Nature Tourism Parks (TWAL) in Indonesia with an area of 50,000 ha. This status is specifically regulated since 1990 concerning the conservation of natural resources and its ecosystem, where its function and role is set as conservation areas and the implementation of natural marine tourism. Although mangrove ecosystems are included in the recoverable resources, if the function or conversion is carried out on a large scale and continuously without any sustainability considerations, the ability of the ecosystem to recover itself is not only hampered but also does not take place ideally, because of the weight of the changes. Damage to the mangrove ecosystem has a large impact on ecological, economic, and social. Mangrove degradation is usually caused by conversions for settlements, conversions for ponds (both salt ponds and fish ponds), timber extraction, and pollution (Hakim et al. 2017; Santoso et al. 2015).

The area of mangrove ecosystem in Kupang Regency, according to the statistics data 2008 is 11,352.92 Ha, and in 2019 the remaining mangrove forest area is 3,266.10 Ha. This condition shows that from 2008 to 2019 there was a decline in the area of mangrove forests in Kupang Regency, which was an area of ± 8,126.82 ha, and possibly already converted into ponds and other uses (BPS-Statistics of Kupang Regency, 2020). The degradation of mangrove ecosystems in Indonesia has been going on for a long time and threatens the preservation of the world's coastal and marine biological resources, because Indonesia has the largest mangrove ecosystem with the highest biodiversity in the world. Mangrove ecosystems in the Kupang Bay TWAL also have environmental degradation due to uncontrolled mangroves utilization. Human activities such as the making of brackishwater ponds for fish or salt pond, deforestation, and environmental pollution, are the main causes of mangrove degradation. In addition, mangrove degradation can also be caused by reclamation, sedimentation, mining, and natural factors such as storms or tsunamis (Kusmana and Sukwika 2018; Kerry et al. 2017).

The development of salt ponds around Kupang Bay, whose location is in the mangrove area, began in 2009 since the government launched a national salt self-sufficiency program. This region has the potential to develop salt ponds of 39.2%, with an existing area of 7,700 hectares and potential production of 870,000 tons/year. The extensification of salt land in Bipolo Village, Sulamu District, Kupang Regency by PT. Garam that has previously been carried out in 2016 was 55 ha become 318 ha in 2020. Its supporting physical superiority is having a high saltwater content due to the ideal climate and abundant sunshine that tend to be even hot and dry, making it a suitable area for salt processing. Of course, this is an attraction for residents or the public and the government and the private sector in an effort to produce salt both conventional and with Geomembrane Filter (TuF) Technology. Through the PUGAR (Program of the People's Salt Industry Development) the Governor of East Nusa Tenggara wants to realize the salt self-sufficiency in Kupang Regency. This program has been started in 2011 until now which is based in the Merdeka Sub-District, East Kupang District, Bipolo Village, and Oeteta Village, of Sulamu District.

Deforestation of mangrove for development of salt pond industries will certainly disrupt the existence of the surrounding mangrove ecosystem. Various impacts will be sensed both by biota that interacts in the mangrove area itself and by people who depend on their lives by utilizing life around the mangrove area. Based on the description above, it is necessary to conduct a study to determine the effect of salt pond industry on changes in mangrove ecosystems. This study aims to examine the effect of the total area of salt pond and production of salt on the dominance and diversity of mangrove species in Kupang Bay.

## MATERIALS AND METHODS

### Study area

The study was carried out between July and September 2018 in the Kupang Bay area on 3 (three) research stations, namely in Merdeka Sub-District (East Kupang Sub-District), Bipolo Village and Oeteta Village (Sulamu Sub-District). The three villages are located on the Kupang Bay, West Timor, Indonesia. The location is the mangrove ecosystem area near the business activities of salt ponds (Figure 1).

This study used a quantitative approach to survey methods. The data collection techniques in the survey method were sampling techniques for collecting data on mangrove ecological condition and interview techniques with questionnaire guidelines for collecting respondents' data. The population in this study was dominantly salt farmers in the Kupang Bay, which were concentrated in 3 (three) locations, i.e. Merdeka Village (East Kupang Sub-District), Bipolo and Oeteta Villages (Sulamu Sub-District). Variables studied included mangrove ecological condition, total area of salt ponds and production of salt.

### Procedures

Collection of the mangrove ecological data was conducted by the line transect method. The stages of data collection were as follows: The location determined from the observation of mangroves represents the study area, and can also represent each zone of mangrove in the study area. The sampling steps were as follows: At each location, a conceptual observation transect was determined based on the representation of the location of the study at each observation station. After that, line transects from the sea to the land are determined (perpendicularly from the sea to the land as long as in the mangrove ecosystem). In each zone, mangrove vegetation used transect lines, with sample plots located along the transect line, randomly placing plots in the form of squares measuring 10 x 10 m2 for tree levels, size 5 x 5 m2 for sapling level, and size 2 x 2 m2 for seedling level as much as 3 (three) sample plots with plot distance on each transect of 50 m. In each predetermined plot, the determination of each type of mangrove plant is present, calculate the number of individuals per species, and measure the circle of the stems of each mangrove tree at breast height, about 1.3 m.

The method used in data collection of production and the total area of salt pond industry were observation and interview. Retrieval of the respondents' data was carried out with the assistance of questionnaires by direct interview with each respondent in the research area. Determination of the sample is done by a purposive sampling method, namely the selection of respondents intentionally with certain sample requirements. The requirements of the respondents selected in this study were salt farmers who owned farmland and communities as cultivators of salt ponds.

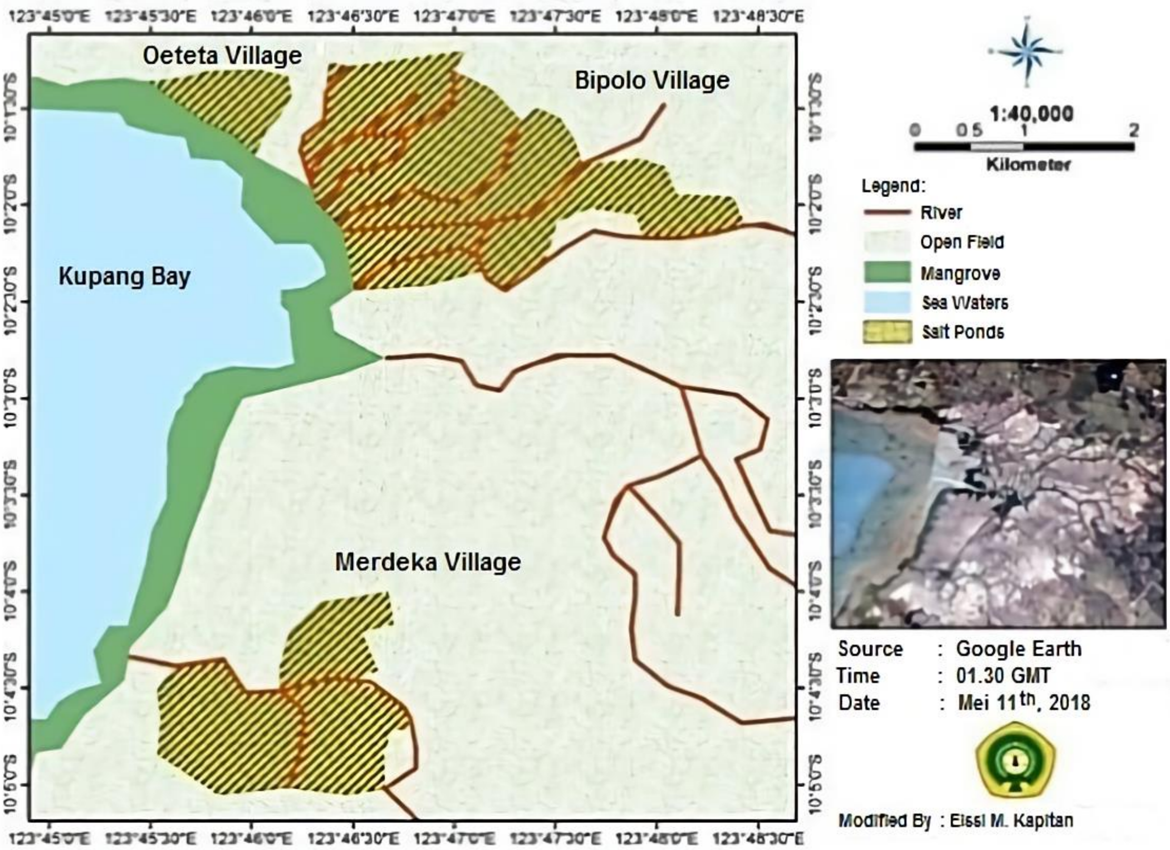



**Figure 1.** Location of salt ponds area in Kupang Bay (Timor Island), East Nusa Tenggara, Indonesia

**Data analysis**

Analysis of the mangrove ecological condition includes diversity index, dominance index, dominance value, relative density, relative frequency, relative closure and importance value index, with the following formulas:

*Diversity index (H')*

The Shannon-Wienner diversity index is used to determine species diversity at each growth rate (Odum 1971) with the following formula:

$$H' = -\sum pi \ln pi \text{ atau } H' = -\sum_{i=1}^{s} [\frac{ni}{N}] \ln [\frac{ni}{N}]$$

where:
H' = diversity index
Pi = ni/N
ni = Total of individual of the $i^{th}$ species
N = Total number of the whole individual

Categories of the Shannon-Wienner diversity index (H') are referred to:

H'<1 = Low diversity
1 <H'≤3 = Moderate diversity
H' >3 = High diversity

*Dominance index (C)*

Dominance index is used to determine the extent to which a biota group dominates another group. Significant dominance will lead to unstable and depressed communities. Dominance index is calculated based on the formula of the index of dominance of Simpson (Odum 1971) as follows:

$$C = \sum (ni/N)^2$$

Where:
C = Dominance Index
ni = Total of individual of the ith species
N = Total number of the whole individual

Category of dominance index:
C = 0 < C < 0,5 = Low Dominance
C = 0,5< C < 0,75 = Moderate Dominance
C = 0,75< C <1 = High Dominance

Dominance index is used to determine the extent to which a biota group dominates another group. Significant dominance will lead to unstable and depressed communities. Dominance index is calculated based on the greater dominance index value (C), the greater the tendency for certain species to dominate.

*Dominance*

The dominance of certain species can be determined by using the Simpson Dominance Index as follows:

$$\mathbf{D} = \sum \left[\frac{(\mathbf{ni}-\mathbf{1})}{(\mathbf{N}-\mathbf{1})}\right]^2$$

Where:
D : Dominance
ni : Total of individual of the $i^{th}$ species
N : Total number of the whole individual

*Density (Di)*

Mangrove species density values can be calculated using the formula:

$$\mathrm{Di} = \frac{\mathbf{ni}}{\mathbf{A}}$$

Where:
Di : density of the $i^{th}$ species
Ni : total number of individu of the $i^{th}$ species
A : total area sampling/plot

*Relative density (RDi)*

The relative density value of mangrove species can be calculated using the formula:

$$\mathrm{RDi} = \frac{\mathbf{ni}}{\sum \mathbf{n}} \times 100\ \%$$

Where:
RDi : relative density of the $i^{th}$ species
ni : total number of the $i^{th}$ species
∑n : total stand of all species

*Frequency species (Fi)*

Mangrove species frequency values can be calculated using the formula:

$$\mathbf{Fi} = \frac{\mathbf{pi}}{\sum \mathbf{p}}$$

Where:
Fi : frequency of the $i^{th}$ species
pi : total sampling/plot where found the $i^{th}$ species
∑p : total sampling/plot of the whole species

*Relative frequency (FRi)*

The relative frequency value of mangrove species can be calculated using the formula:

$$\mathrm{RFi} = \frac{\mathbf{Fi}}{\sum \mathbf{F}} \times 100\ \%$$

Where:
RFi : relative frequency of the $i^{th}$ species
Fi : frequency of the $i^{th}$ species
∑F : total number of sampling plots

*Important value index (IVI)*

The importance index of mangrove species can be calculated using the formula:

$IVI_{Trees} = RD_i + RF_i + RC_i$
$IVI_{Saplings}$ : $RD_i + RF_i + RC_i$
$IVI_{Seedlings}$ : $RD_i + RF_i$

The importance of a mangrove species on the sapling level ranges from 0-300 and seedlings level 0-200. This Important Value provides an overview of the influence or role of a mangrove plant species in mangrove communities.

*Analysis statistic*

The research data were analyzed by analysis of variance (ANOVA) using software SPSS 24. Whereas the correlation between diversity and dominance of mangroves with the total area of salt ponds and salt production was analysis with the formula as follows:

$$r^2 = \frac{\sum(Yi-\bar{Y})^2 - \sum(Yi-\bar{Y}i)^2}{\sum((Yi-\bar{Y})^2}$$

Categories of the correlation are:
r = 0.07-1.00 = high correlation
r = 0.40-0.69 = moderate correlation
r = 0.20-0.39 = low correlation
r = < 0.20 = no correlation

While the analysis of relationship between diversity and dominance of mangroves with the area of salt ponds and salt pond production capacity have used linear regression test.

## RESULTS AND DISCUSSION

### General overview of location dan inhabitant

The research location is the coastal area stretching of the coast of Kupang Bay i.e. Merdeka Village (East Kupang Sub-District), Oeteta, and Bipolo Villages (Sulamu Sub-District). These three villages have a region that directly boundaries of the Kupang Bay. Most of the land surface conditions are sloping (<15°) and situated at a height of 26-500m from mean sea level (msl). The climatic conditions are dry climate influenced by monsoons, with a short rainy season between December and April. The average air temperature ranges between 24 and 31°C, with an average rainfall reaching 2,000 mm per year.

The resident in Merdeka, Oeteta and Bipolo Villages varies considerably according to the number of households, number of population, type of work and level of education. The population in Merdeka Village is higher (3,469 people) compared to the Oeteta Village (1,368 people) and Bipolo Village (844 people), although the area is smaller (10.50 $km^2$) than the two villages above namely Oeteta 42.34 $km^2$ and Bipolo 41.47 $km^2$ with a density of kilometers reaching 330 people (BPS-Statistics of Kupang Regency 2020).

The livelihoods of the inhabitant in these three regions are still dominated by the agricultural sector, especially food crops that are characterized by dryland farming, which is as much as 67.09% or 1,763 people. Whereas fishermen are only 11.23% or 295 people, while 21.68% other livelihoods or around 570 people. Based on livelihoods, the population in these three regions basically has an agrarian culture, even though these three regions are coastal villages.

Salt farmers who are currently actively operating salt farms in Merdeka Village are currently 47 people, in Oeteta Village there are 14 people and in Bipolo Village there are 13 people, with a composition of 100% are male, with a relatively good level of education. The average level of formal education of respondents is 23.7% of elementary school graduates, 18.6% of junior high school graduates, 32.2% of high school graduates, 3.4% of graduates are undergraduate and 22.1% do not graduate from school.

Being a salt farmer is the main job of all respondents at this time where they use their time to operate in salt fields for 6-8 hours every day during the dry season while during the rainy season most respondents do activities as farmers and some of them carry out activities as fishermen. Salt farmers in managing salt fields in the three regions apply the salting process using conventional techniques and intensification techniques. Conventional techniques use a crystallization table without using a pedestal, so that the resulting salt is rather gray due to the possibility of mixing dust and soil. While the intensification technique is to use a Geo-membrane media as a base from the crystallization table, so that the salt produced is a pure white color because it is not contaminated by soil or mud because it is insulated by a Geo-membrane. The difference between these two methods of salt production is that the quantity and quality of this salt are also different. This can be distinguished in plain view from the shape and color.

Salt production in these three regions varies based on land area and the techniques applied by each salt farmer. In addition, weather conditions also influence salt production. The total area of salt ponds and the amount of salt production in the three regions i.e. Merdeka Village has the total area of 56 ha (38%) with the amount of salt production reaching 754,800 tons (33%), Oeteta Village has the total area of 64.28 ha (44%) with production of salt reached 1,380,300 tons (61%), while Bipolo village has the total area of 26.4 ha (18%) with salt production of 138,760 tons (6%).

The business activities of salt ponds in 3 (three) locations actually existed for a long time and naturally carried out in groups for the common interest in accordance with their work, namely the people's salt farm business, because generally more than one person is usually employed up to 4 people. Public salt farmers in the three regions currently process salt depending on the size of the salt ponds they have to work on and on average 0.5-10 ha/person (family head). However, the existence of a group of salt farmers was officially formed in 2011 when the program of PUGAR (Program of the People's Salt Industry Development) was launched from the Ministry of Marine Affairs and Fisheries Indonesia.

During that year salt production gradually increased because the PUGAR provided technological innovations in the process of making salt starting from irrigation channels, pumping, production processes to salt storage systems. Between 2014 and 2015, an innovative filter technology was implemented. This thread is expected to improve the evaporation process of water to reduce the time of making the salt, while the filter is expected to filter out the impurities that attach to the salt can be deposited on the filter. With the salt pond intensification system by the government, salt farmers are also provided with a membrane that is placed on the crystallization table so that the salt produced looks pure white.

The raw materials of salt produced are usually sold to companies that use salt raw materials or to salt cooking communities from the area around the location of salt or from outside the area such as from Oebelo Village, as well as from outside the Kupang Regency area. The selling price of the raw materials of salt varies greatly. During the dry season where the amount of production increases, the price is 500 to 600 IDR/kg, while in the rainy season when the salt does not produce the price is 2,500 IDR/kg. Usually, these salt entrepreneurs if in the dry season with a low selling price they sell their salt products only for sudden needs such as the needs of school children or sick children or for their daily needs and to buy sacks as a place to store salt raw materials while the rest are raw materials This salt is collected for sale during the rainy season. The salt business is only carried out during the dry season, while during the rainy season these farmers carry out activities in agriculture to cultivate rice or secondary crops and some return to search for products in the sea such as catching Shrimp, crabs, fish and other marine products as a source their food and the rest are sold around their neighborhood.

**Ecological condition of mangrove**

Data collection of Ecological Condition of Mangrove in Kupang Bay was carried out on 3 (three) observation stations, namely: Merdeka Village (S: 10º05'14.74" and E: 123º45'05.53"), Oeteta Village (S: 10º01'41.51" and E: 123º45'26.10"), and Bipolo village (S: 10º02'27,64" and E: 123º46'15.06"). Based on the results of the identification of mangroves in the three observation locations, it shows that overall there are 9 (nine) mangrove species from 4 (four) families found in 3 (three) observation locations: Merdeka Village= 4 (four) species of mangroves: *Rhizophora mucronata*, *Ceriops tagal*, *Avicennia alba*, and *Sonneratia alba*. Oeteta Village= 7 (seven) mangrove species: *Rhizophora apiculata*, *Rhizophora stylosa*, *Ceriops tagal*, *Ceripos decandra*, *Avicennia alba*, *Sonneratia alba*, and *Sonneratia caseolaris*. While in Bipolo Village, 9 (nine) species were found: *Rhizophora apiculata*, *Rhizophora stylosa*, *Rhizophora mucronata*, *Ceriops tagal*, *Ceriops decandra*, *Avicennia alba*, *Sonneratia alba*, *Sonneratia caseolaris*, and *Luminitzera racemosa*. All species originating from 4 families: Rhizophoraceae, Avicenniaceae, Combretaceae and Soneratiaceae.

The dominance index (C) values at each research station ranged between 0.03 and 0.84. The dominance index value obtained is categorized as low to high. Bipolo village has a relatively low dominance index value, which ranges between 0.03 and 0.23. For Oeteta village, it has a moderate dominance index value, ranging between 0.2 and 0.33 while the Merdeka village has a high dominance index value ranging between 0.51 and 0.84 (Figure 2).

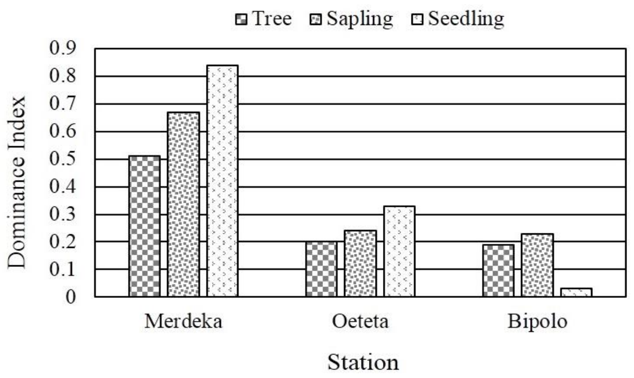



**Figure 2.** Dominance index of mangrove

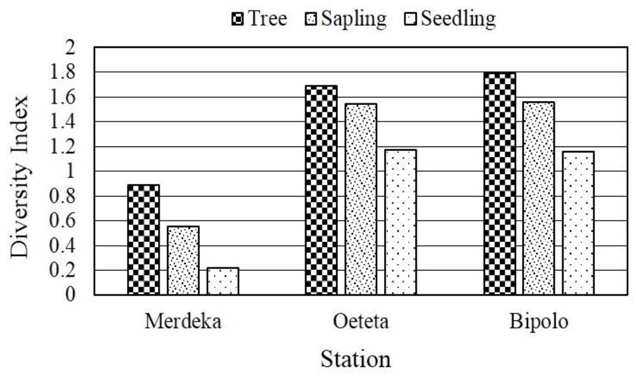



**Figure 3.** Diversity index of mangrove

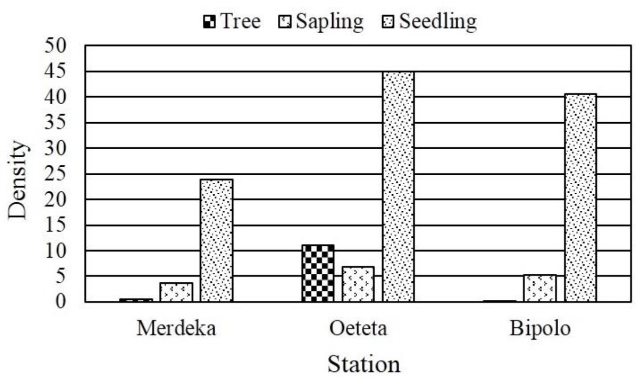



**Figure 4.** Density of mangrove

The range of the diversity index (H') value for each research station, namely the Merdeka Village, ranges from 0.22-0.89, this value is relatively low because of the value of H '<1. For Desa Oeteta the value of the diversity index (H ') ranges from 1.17-1.69, this value is classified as moderate because the value of H' 1 <H '<3, while for the Village of Bipolo the range of H' is between 1.16-1.79 and this value is also classified as moderate. This is in accordance with the criteria for the diversity index value (H ') according to Odum (1971) that if the value H' <1 is categorized as low, if the diversity index value 1 <H '<3 is categorized as medium, whereas if the value of H'> 3 is categorized as high (Figure 3).

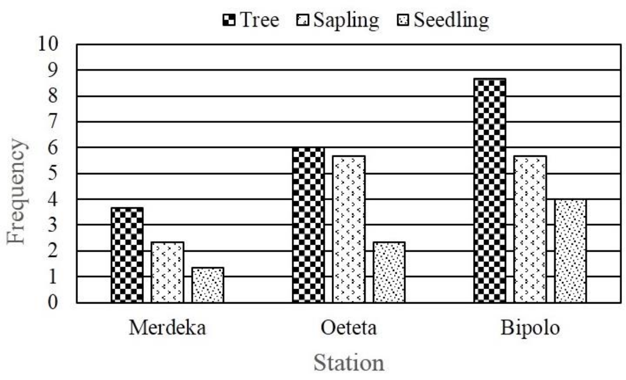



**Figure 5.** Frequency of mangrove

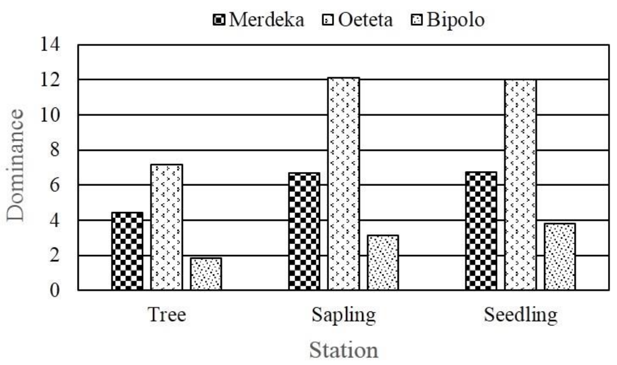



**Figure 6.** Dominance of mangrove

The density of the mangrove species studied consisted of several levels, namely the level of trees, saplings, and seedlings. From the overall research station, the density type at the tree level is less than the saplings and seedlings (Figure 4).

The highest frequency of mangrove species is at station 3 (Bipolo Village), where the nine of mangrove species are found in this region and have high-frequency values both for the tree, sapling, and seedling categories. This shows that the level of presence of mangrove species is evenly distributed. The lowest frequency of mangrove species is at station 1 (Merdeka Village), where only 4 mangrove species are found. This condition indicated that the distribution of mangrove species in this region is uneven (Figure 5).

The high dominance value of species for both the tree, sapling, and seedling categories is the *Ceriops tagal* (Figure 6). This type has a high dominance value because the characteristics of the research location are in accordance with the characteristics desired by the mangrove species. Besides that, the condition of the substrate which is generally sludge containing organic material is very suitable for the growth of its species so that this type of mangrove is spread evenly at each observation station.

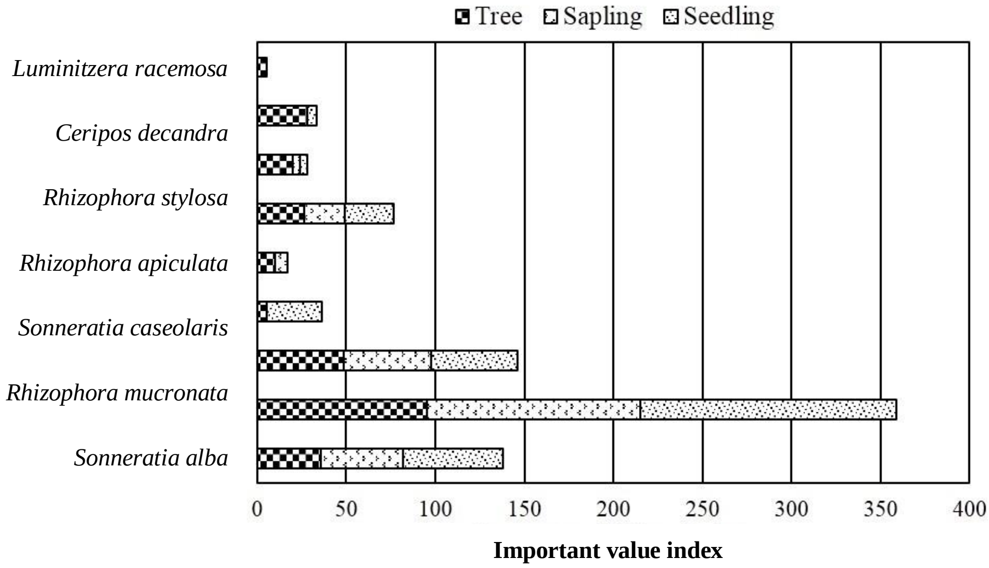



**Figure 7.** Important value index of mangrove

All of stations, there are 3 (three) categories of vegetation found that had the highest IVI and were spread both at the level of trees, saplings, and seedlings, namely *Ceriops tagal, Avicennia alba,* and *Sonneratia alba.* These three species play quite an important role in the Kupang Bay coastal environment. IVI shows the representation of mangrove species that play a role in ecosystems with a range of values between 0-300 (Figure 7). The high value of IVI shows that a species has better adaptability, competition, and reproductive capacity compared to other plants in a particular environment.

**Effect both of total area and production of salt ponds on dominance of mangrove**

The ANOVA test on the data on land total area to the dominance of mangroves showed that the land area did not significantly influence the dominance of mangrove species ($p$>0.05). There was a positive relationship between the two variables (Y = 0.152 + 0.003X). The coefficient is positive, so the implication that there is a positive relationship between land area and dominance, the more extensive the area of salt ponds, the higher the dominance. The results of the correlation analysis showed that there was no significant correlation between the area of salt ponds and the dominance of mangroves. However, the direction of the relationship was positive (r = +0.316). It would be said that the land area is positively related but not significant to the dominance of mangroves.

The ANOVA test results showed that the production of salt ponds did not significantly influence the dominance of mangrove species ($p$>0.05), but there was a positive relationship between the two variables (Y = 0.296 + 8.105E-6X). The coefficient correlation is positive (r =+0.925). It would be said that there is a positive relationship between salt pond production and dominance, the higher the production of salt ponds, the higher the dominance of certain mangrove seedlings stage. The results of the correlation analysis showed that there was no significant correlation between the production of salt ponds with mangrove dominance where the significant value was 0.947>α (0.05), but the direction of the relationship was positive (r = +0.026). It can be said that salt pond production is positively related but not significant to mangrove dominance.

Based on the results of the observation, the location that has the highest mangrove dominance is in the Merdeka Village, both for the tree category (C = 0.51), saplings (C = 0.67), and seedlings (C = 0.84), while those with dominance the lowest type of mangrove was Bipolo Village, both for the tree category (C = 0.19), saplings (C = 0.23), and seedlings (C = 0.03).

Land area and high salt pond production are found in the Merdeka Village (56 Ha, production of 754,800 tons/year) and Oeteta Village (64.28 Ha, production of 1,380,300 tons/year), while the lowest is Bipolo Village (26.4 Ha, production 138,760 tons/year). The high dominance of mangrove species is also found in these two locations (Merdeka and Oeteta Villages) while the lowest dominance is in Bipolo Village. This may occur because the substrate conditions support mangrove growth and the presence of growth mangroves are more adaptive to environmental changes.

**Effect of total area and production of salt ponds on mangrove species diversity**

The ANOVA test results, the land area data on diversity shows that the land area does not significantly influence the diversity of mangrove species (P> 0.05), and there is a negative relationship between the two variables (Y = 1,930-0,010X). The coefficient is negative, so it can be said that there is a negative relationship between the area of the land and the diversity of mangrove species, where the more the area of salt ponds, the lower the diversity of mangrove species.

The results of the correlation analysis showed that there was no significant correlation between the area of salt ponds and the diversity of mangroves where the significant value was 0.334> α (0.05), but the direction of the relationship was negative (r =-0.365). It can be said that the land area is negatively and not significantly related to the diversity of mangrove species.

However, the Result of ANOVA test showed that salt pond production did not significantly influence the diversity of mangrove species (P> 0.05), and there was a negative relationship between the two variables (Y = 1.507-6.551E-5X). The coefficient is negative, so it can be said that there is a negative relationship between the production of salt ponds and the diversity of mangrove species, where the higher the production of salt ponds the lower the diversity of mangrove species.

The results of the correlation analysis showed that there was no significant correlation between the production of salt ponds with the diversity of mangrove species where the significant value was 0.843> α (0.05), and the direction of the relationship was negative (r =-0.077). It can be said that salt pond production is negatively and not significantly related to the diversity of mangrove species.

Based on observations, the location that has the highest mangrove diversity index is in Bipolo Village (H' = 1.79) for the tree category while the one with the lowest mangrove species diversity index is Merdeka (H' = 0.89). For the puppies category that has the highest diversity index in the Oeteta Village (H '= 1.69). While those with the lowest index of diversity of mangrove species are in Merdeka Village (H '= 0.55). For the seedling category which has the highest diversity index in the Oeteta Village (H '= 1.17) while the lowest mangrove species diversity index is in the Merdeka Village (H' = 0.22).

## Discussion

In the Merdeka Village, the value of the diversity index is low because there are species that dominate this region, which is of the type of *Ceriops tagal* which dominate other species in this region. Oeteta and Bipolo villages have a diversity index value in the moderate category. This is presumably due to the condition of the substrate and a considerable distance from the sea. A community is said to have lower species diversity if the community is composed of a few species and only a few species dominate.

Factors that determine changes in species diversity in one ecosystem, namely time, space heterogeneity, competition, predation, environmental stability, and productivity. During the geological period, there will be changes in environmental conditions which result in many individuals who cannot maintain their lives, but there are also groups of individuals who are able to survive continuously for a relatively long time as a result of the evolutionary process (Utina et al. 2019).

Through the analysis of dominance index, it can be seen that very few species dominate the place because of its moderate dominance, which causes a moderate level of diversity, because if the dominance is high the diversity is low, and vice versa. Furthermore, the dominance index is high, then dominance is centralized in one species. But if the dominance index value is low, then dominance is centralized in some species (Lignon et al. 2011; Hakim et al. 2017; Fredrik et al. 2019).

Mangrove species that have the highest density are in the seedling category, while the lowest density is in the tree category. One of the factors that influence the low value of tree species density is the root condition which is classified as large so that the growth of the mangrove becomes less optimal, thus the low density of species in the tree category causes incoming sunlight to illuminate the mangrove forest land. This makes the seedlings and saplings grow effectively (Sambu et al. 2014). High density in the seedling category is due to the low density of the tree so that the sun needed by the seedlings is not blocked by the tree, so it supports the growth of the seedlings. Seedling regeneration in mangrove forests is an important part of the process of secondary succession, then the growth of species of natural mangrove seedlings has a close relationship with the availability of the parent trees. The high value of the density of this species is also influenced by the closing value of sapling species which are still relatively small with a diameter of <10 cm. This factor supports the optimal growth of mangrove species (Hakim et al. 2017; Fredrik et al. 2019).

The species with the highest density and has the most influence in the three study sites is the *Ceriops tagal* of all categories. This condition is caused because this species is a type of mangrove whose growth is tolerant of environmental conditions, especially to the condition of the substrate which likes the clay substrate and grows well in areas that are inundated by high tides and tides as well as a very widespread of seeds and occurs throughout the year (Kusmana and Sukwika 2018; Utina et al. 2019). *Ceriops tagal* is a characteristic of the development of the final stages of coastal forests, as well as the initial stages in the transition to terrestrial vegetation. This species likes clay substrates and the flowering occurs throughout the year (Awn et al. 2016).

The distribution of species of a mangrove community can be determined by calculating how much the frequency value or the level of presence of the species, high-frequency values indicate that the species has an even distribution and is often found in a forest area. Vice versa if the frequency value is low then the distribution in a forest area is not evenly distributed (Krebs 2009; Kerry et al. 2017).

*Ceriops tagal* is a pioneering plant or a pioneer and includes species that have seeds that can germinate while still on the parent are very supportive of the widespread process of other species. The dependence of pioneer plant species on soil types is shown by the *Rhizophora* genus which is a common characteristic for muddy soils mixed with organic matter (Awn et al. 2016; Utina et al. 2019). While the specific life cycle of the mangrove species (*Rhizophora* sp.) with seeds that can germinate while still in the parent plant is very supportive of the broad distribution process of this species in the mangrove ecosystem (Kerry et al. 2017). These species are spread evenly in almost all mangrove ecosystem areas in Merdeka Village, Oeteta Village, and Bipolo Village. This indicates that these species are the most adaptive.

Species with the highest importance show species mastery value in a community and are able to take advantage of environmental conditions so that they can grow better than other species (Lignon et al. 2011). Species with the lowest dominance value are depressed, unable to develop and adapt so that growth is unstable. The importance of a species can be used as an indication that the species are considered dominant by having a higher relative density, relative frequency, and relative dominance value compared to other species (Sambu et al. 2014).

Important Value Index (IVI) is a quantitative parameter that can be used to express the level of species dominance in mangrove communities. Furthermore, the IVI value reflects the existence of the role (dominance) and structure of mangrove vegetation in a location. The dominant species in a plant community will have a high importance value index, so the most dominant species of course has the greatest importance value index (Krebs 2009).

Factors that determine changes in species diversity in one ecosystem, namely time, space heterogeneity, competition, predation, environmental stability, and productivity. During the geological period, there will be changes in environmental conditions that result in many individuals who cannot maintain their lives, but there are also groups of individuals who are able to survive continuously for a relatively long time as a result of the evolutionary process. (Lignon et al. 2011; Utina et al. 2019).

Through the analysis of dominance index, it can be seen that very few species dominate the place because of its moderate dominance, which causes a moderate level of diversity, because if the dominance is high the diversity is low, and vice versa. Furthermore, the dominance index is high, then dominance is centralized in one species. Whereas if the dominance index value is low, then dominance is centralized in some species (Sambu et al. 2014; Fredrik et al. 2019).

Dominance is a description of the level of mastery of species of sample plots, so that the dominance value of a type can give an idea of the level of mastery of the species in a particular area (Kerry et al. 2017). Determination of the effect of the salt pond area on the dominance of mangrove species and predict whether experiencing outbreaks or decreases, a simple linear regression analysis was carried out.

The dominance of mangrove species in an area is influenced by environmental factors such as humidity, temperature, and the inability of a species to survive or lose the competition, such as the struggle for nutrients, sunlight, and growing space with other species that greatly affect the growth and diameter tree trunk. The dominant type has large productivity and the existence of the dominant species in the research location is an indicator that the community is in suitable habitat and supports its growth. In the type of mangrove that has a low dominance due to the high utilization of mangrove species, habitat or substrate that are not suitable, there is interaction between species or the inability of the type of mangrove to adapt to environmental conditions (Lignon et al. 2011; Hakim et al. 2017; Kusmana and Sukwika 2018).

The high diversity index value in Bipolo Village is due to the low land area and production of salt ponds so that the distribution of species is evenly distributed and no species dominates the area compared to Oeteta and Merdeka Villages where the area of land and production of salt ponds causes high species diversity due to land use or new land clearing which causes a decrease in species diversity in the region.

Diversity is a characteristic community that is related to the number of species or species richness and species abundance as community compilers. Community diversity is characterized by the many species of organisms that formed the community. Conversely, if the diversity of an environment is low, then the environment is susceptible to interference. So, in a community where high diversity will occur species interactions involving energy transfer, predation, competition, and more complex niches (Krebs 2009; Kerry et al. 2017).

Diversity index characterizes the level of community-based on its biology and community stability, namely the ability of a community to keep it stable even though there are disruptions to its components. In principle, the index value is higher, meaning that communities in the waters are increasingly diverse and are not dominated by one or more of the existing taxon. Diversity is an indicator of stability or stability of a growth environment. High diversity shows that a community has high complexity. Old and stable communities will have high species diversity. Whereas a community that is developing at the level of succession has a lower number of species than a community that has reached a climax (Hakim et al. 2017; Kusmana and Sukwika 2018; Utina et al. 2019).

The industry of salt ponds on the coast of Kupang Bay, especially around mangrove areas, indirectly affects the surrounding mangrove ecosystems and can reduce the quality of seawater around the mangrove area but for now, the area of land and the production of salt ponds have not had a significant influence on dominance or the index diversity of mangrove species because there is not too much land cultivated for salt pond activities.

This study found that salt pond area and salt pond production were positively correlated with the dominance of mangrove species, although the effect was not significant. On the other hand, salt pond area and salt pond production have a negative correlation with mangrove species diversity, although only salt pond area has a significant effect on mangrove species diversity. This shows that the impact of the development of the salt pond industry on the mangrove ecosystem in the short term can be detected from a decrease in mangrove species diversity, and in the long term it can also be detected from an increase in the dominance of mangrove species. This phenomenon can be applied in monitoring mangrove ecosystems related to the development of the salt pond industry, in an effort to conserve mangrove ecosystems in Kupang Bay.

## ACKNOWLEDGEMENTS

The authors sincerely thank PT. Garam and Salt Farmer for their help and cooperation in this study. Thanks also to Markus Olla, for his help during observation in Kupang Bay, West Timor, Indonesia.

## REFERENCES

Awn MSM, Yulianda F, Yonvitner. 2016. Characteristics and above-ground biomass of mangrove species in Enggano Island, Bengkulu Sumatra, Indonesia. Intl J Adv Eng Manag Sci 2 (7): 1084-1091.

BPS-Statistics of Kupang Regency. 2020. Kupang Regency in Figures, 2020. BPS-Statistics of Kupang Regency, Kupang. [Indonesian]

Fredrik D, Santoso P, Alayubi A. 2019. Composition and structure of mangrove community on sapling and seedling levels in Coastal of Dolulolong, Lembata, Indonesia. Intl J Biosci 15 (4): 153-160. DOI: 10.12692/ijb/15.4.153-160.

Hakim L, Siswanto D, Nobukazu Nakagoshi. 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.

Kerry RG, Das G, Patra JK. 2017. Biodiversity and conservation of mangrove ecosystem around the World. J Biodivers Conserv 1 (1): 9-9.

Krebs CJ. 2009. Ecology, The Experimental Analysis of Distribution and Abundance. Haper and Row Publ, New York.

Kusmana C, Sukwika T. 2018. Coastal community preference on the utilization of mangrove ecosystem and channelbar in Indramayu, Indonesia. AACL Bioflux 11 (3): 905-918.

Lignon CM, Coelho CJ, Almeida R, Menghini PR, Schaeffer NY, Cintrón G, Dahdouh GF. 2011. Characterisation of Mangrove Forest Species in View of Conservation and Management: a review of Mangals at the Cananéia region, São Paulo State, Brazil. Intl J Coast Res 5 (7): 349-353.

Odum EP. 1971. Fundamentals of Ecology. 3rd ed. WB. Saunder Company Ltd., Philadelphia.

Sambu AH, Rahmi, Khaeriyah A. 2014. Analysis of characteristics of and use-value of mangrove ecosystem (case study in Samataring and Tongketongke Sub-Districts, Sinjai Regency). J Environ Ecol 5 (2): 222-233. DOI:10.5296/jee.v5i2.6826.

Santoso P, Marsoedi, Maftuch, Susilo E. 2015. Strategy of blood cockle aquaculture development for conservation and welfare in Sub-district of Central Kupang, West Timor, Indonesia. J Biodivers Environ Sci 7 (6): 1-9.

Utina R, Katili SA, Lapolo N, Dangkua T. 2019. The composition of mangrove species in coastal area of Banggai District, Central Sulawesi, Indonesia. Biodiversitas 20 (3): 840-846. DOI: 10.13057/biodiv/d200330.

**OCEAN LIFE**
Volume 4, Number 2, December 2020
Pages: 74-81

E-ISSN: 2580-4529
DOI: 10.13057/oceanlife/o040203

# A contribution to understanding blue carbon sequestration and forest structure in mangroves of different ages in a small island (Mauritius)

**DEVESEE RAMDHUN♥, CHANDANI APPADOO♥♥**
Department of Biosciences and Ocean Studies, Faculty of Science, University of Mauritius. Réduit 80837, Republic of Mauritius.
Tel.: +230-59240270, ♥email: devesee.ramdhun@gmail.com; ♥♥chandani@uom.ac.mu

Manuscript received: 9 October 2020. Revision accepted: 21 December 2020.

**Abstract.** *Ramdhun D, Appadoo C. 2020. A contribution to understanding blue carbon sequestration and forest structure in mangroves of different ages in a small island (Mauritius). Ocean Life 4: 74-81.* Mangrove biomes, a blue carbon coastal ecosystem, are considered the most carbon-rich forest in the tropics. Still, their ecosystem services have a knowledge gap, especially on small islands. The purpose of this study was to evaluate the carbon stock in sediment and aboveground biomass of three different ages of *Rhizophora mucronata* forests along the coastline of Mauritius. Forest structure was assessed, and diameter measurements at breast height were used to calculate biomass using an allometric equation. Corer samples were collected at each site and sectioned into subsamples of 0-10 cm, 10-20 cm, and 20-30 cm to analyze soil bulk density and soil organic carbon (SOC). The highest SOC stock (470.08 t $ha^{-1}$) was recorded at Mahebourg and the lowest (331.33 t $ha^{-1}$) in the young stand at Le Morne. The total organic carbon contents varied from 931.50±17.06 t $ha^{-1}$ (at Mahebourg) to 350.76±4.058 t $ha^{-1}$ (young forest at Le Morne). There was no significant difference in soil carbon density at different soil depths studied at Le Morne (n=81, p=0.430 (old forest), p=0.875 (young forest)); however, a different scenario was observed at Mahebourg (n=81, p=0.027). This study is the first to report on the potential of carbon storage at these sites in Mauritius and add to the knowledge of the ecosystem services of this ecosystem.

**Keywords:** Above ground biomass, blue carbon, carbon stocks, coastal ecosystem, *Rhizophora mucronata*

## INTRODUCTION

Mangrove forests, seagrass meadows, and saltmarshes are blue carbon ecosystems, sequestering organic carbon in their sediments and biomass to mitigate climate change (Wu et al., 2018; Cuellar-Martinez et al., 2019). Although the terrestrial plants have greater area and function as carbon sinks, the coastal carbon sinks have larger carbon sequestration capacity (DelVecchia et al., 2014). The mangrove biomes growing profusely in the paleotropical and neotropical regions' sea-land interface are considered the richest blue carbon forest, blooming in an anoxic and saline environment (Martin et al. 2019). These halophytic plants with valuable services such as buffering zones for heavy waves and winds, preventing coastal erosion and stabilizing sediments, nutrient filters, and shelters for floras and faunas are subjected to anthropogenic threats (Appadoo 2003).

The intertidal buttress once occupied 75% of the tropical coastal zone worldwide (Dahdouh-Guebas and Koedam 2008), but the population has decreased steeply (Thorhaug et al. 2018). The wealthiest carbon forests can sequester five times more carbon than other forests in both the above and belowground biomass including in their soils (Cui et al. 2018), thus contributing to the global carbon budget (Yang et al. 2014). Carbon sequestration is when a mangrove ecosystem captures carbon from the atmosphere and stores them in its leaves, branches, roots, and soils (Wu et al., 2018). The anoxic soil prevents the organic carbon from oxidizing to other forms, thus providing long-term carbon storage (Luo and Gu 2016).

Nowadays, these evergreen forests do not even represent 1% of the tropical forests because of human disturbances (Penaranda et al. 2019), and they are still exceedingly threatened (Wang et al. 2019); thus, conservation is a necessity. Moreover, it is a serious challenge in this century as carbon emission has exceeded its threshold, and scientists and policymakers are initializing new blue carbon ideas to counter carbon emissions (Ahmed et al. 2017). Globally, deforestation is the main reason for the steep decline in mangrove population, with unsustainable use of resources leading to species extinction and hurricanes and strong cyclones destroying aboveground biomass and reducing the carbon-storing capacity (Ranjan 2019).

Studies on mangrove blue carbon storage and sequestration quantification in islands are important due to the lack of concurrent data on mangrove population, carbon concentration, and quantity of carbon according to depth. These studies also raise awareness about the importance of carbon ecosystems. Blue carbon storage is studied on islands because of direct oceanic swell from sides, and each part of the islands is affected by different amounts of wave energy because of coral reefs. The objective of this study is to assess the status of the organic carbon content in soil and aboveground biomass (AGB) in mangroves of different ages in three mangrove forests in Mauritius. It also aims to estimate the total carbon stocks and explore variations in belowground carbon.

## MATERIALS AND METHODS

### Study site

The study was conducted from October 2019 to March 2020, and sampling was done during October and November 2019 in old forests at Mahebourg (S 20°24'08.95", E 057°42'26.94") and at Le Morne (S 20°27'34.91", E 057°20'16.51") and in a young forest at Le Morne (S 20°27'45.87", E 057°20'26.03") in Mauritius (Figure 1). Le Morne, found on the south coastline, has a temperature range between 17°C to 30°C, and the rainfall ranges from 54 mm to 245 mm (Climate-Data.Org 2020a). The forest is about 2.4 hectares, and consists of a large patch of old *Rhizophora mucronata* stands but mostly of young plants of *Rhizophora mucronata* and *Bruguiera gymnorrhiza.* The study sites at Le Morne consist only of *Rhizophora mucronata* but a forest of different ages (Figure 2). The young plants at Bassin Leon, Le Morne were planted by Association pour la Developpement Durable (ADD) in 2008 and 2009. The study site in Mahebourg (Figure 3) is located on the southeast coastline of Mauritius, with temperatures ranging from 17°C to 29.7°C and precipitation ranging from 61 mm to 255 mm (Climate-Data.Org 2020b), and the old forest is about 0.3 hectare and characterized by *Rhizophora mucronata* of different height categories. Diurnal tides influence all studied regions and the mean salinity in Mahebourg was 26.11 ± 0.74 ppt and that for Le Morne in the old forest was 34.30 ± 0.37 ppt and 31.00 ± 0.23 ppt for the young forest.

### Tree sampling and measurement procedures

The study was conducted in an area of 0.0225 ha at each site in different inundation zones, including the seaward, middle ward, and landward zone (Yang et al. 2014). A transect line of 25 m was laid parallel to the shore on each zone, and three quadrats of 5m x 5m were set up in each transect, leaving a space of 5 m between each quadrat. The heights of plants were measured with a graduated telescopic rod (Betts 2006). The diameter at breast height of the mangrove was measured with a measuring tape following the method of Penaranda et al. (2019). Mangrove plants were categorized into seedlings, saplings, and adult trees (Clarisse et al., 2016).

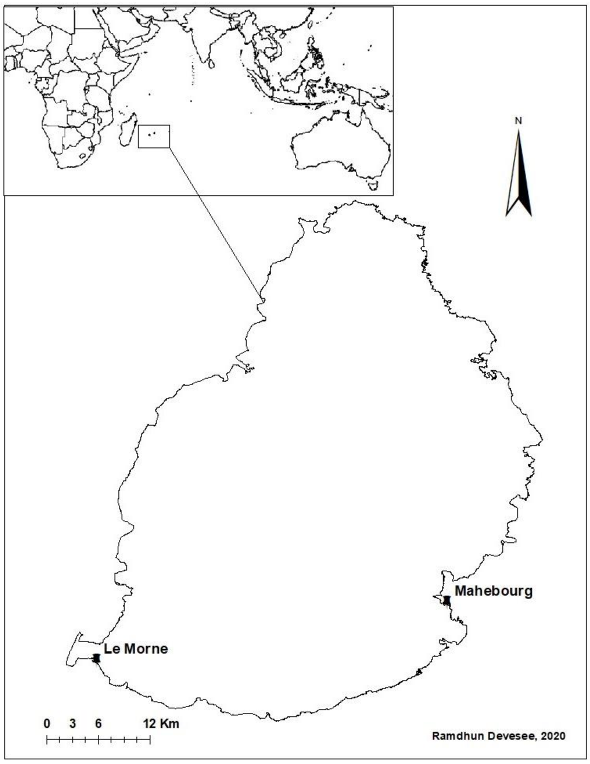



**Figure 1.** Map of Indian Ocean and map of Mauritius where the pin symbols represent the different studied regions

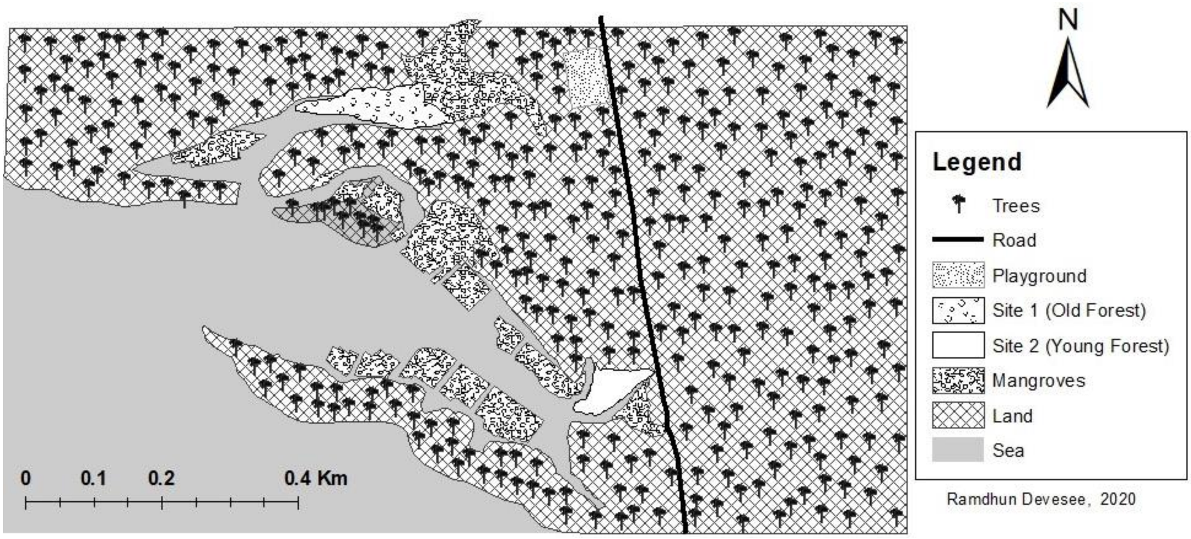



**Figure 2**. The study sites at Le Morne include an old forest (Site 1) and a young forest (Site 2), refer to as study area 1 and study area 2 in legend as well

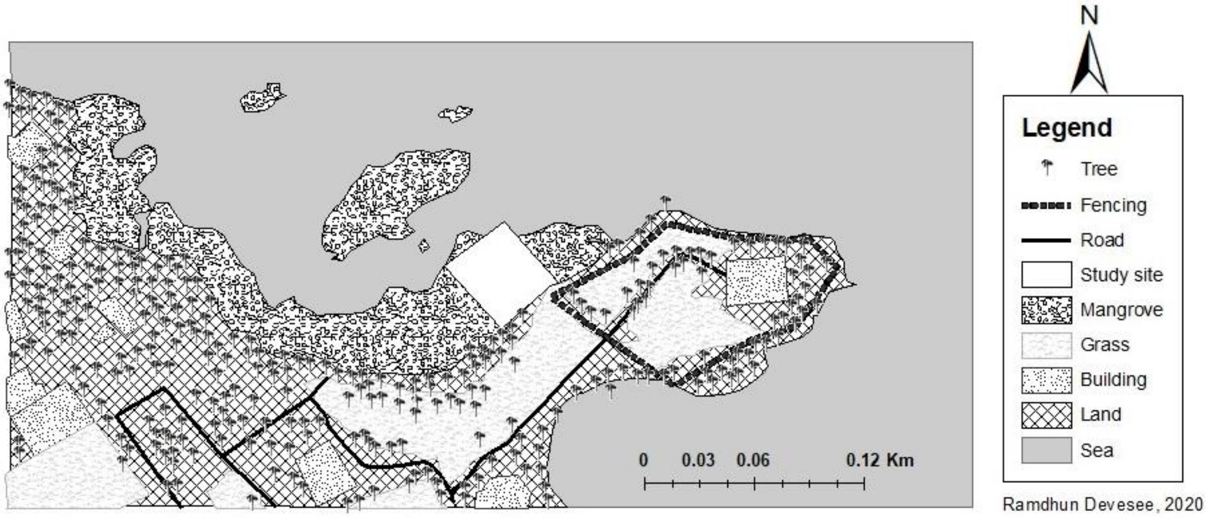



**Figure 3.** Mahebourg (Old forest) study site

**Sediment samplings**

For this study, 27 cores were collected at each site. The sediment samples were collected during low tide with a polyvinyl chloride hand corer having a length and diameter of 30 cm and 10 cm, respectively (Shaltout et al. 2019) and using the technique outlined by Howard et al. (2014a). Triplicates were undertaken in every quadrat. Each core was immediately sectioned with a knife according to 0-10, 10-20, and 20-30 cm depths, stored in labeled zip-lock bags, and kept at 4°C, preserving the organic matter until further analysis (Shaltout et al., 2019).

**Sample analysis**

The sediment samples were first air-dried on aluminum foil for two days, and each sample was weighed on a petri dish using an electronic balance. It was then oven-dried at 60°C until a constant mass was obtained. Next, the dried samples were crushed using a mortar and pestle to a constant size for homogenization, and they were left in a desiccator for further analysis (Cui et al., 2018). Next, organic carbon was analyzed using the Loss on Ignition method, where 5 g of each crushed sample was transferred into a ceramic crucible, weighed, and put in a muffle furnace at 450°C for 4-8 hours (Heiri et al. 2001).

**Data analysis and statistical analyses**

The data were entered in a spreadsheet, and the following equations were used to compute different parameters.

Diameter at breast height (DBH) = C/ π (C= Circumference and π = 3.14)

Allometric equation (Abib and Appadoo 2012)

Log AGB = a Log DBH + b (a= 2.383 and b = -0.799 are regression coefficients)

% LOI = {(difference in mass after ignition (g))/ dry mass before ignition(g)} *100

Percentage of organic carbon= 0.415* % LOI + 2.89

Before any statistical analysis, the normality test was carried out to determine whether the data collected were normal. Next, the IBM SPSS Statistics Software was used for hypothesis testing where the Shapiro-Wilk Test showed that the data was not normal; thus non-parametric test should be used with 0.05 significance. Finally, the non-parametric Kruskal-Wallis test was used to test the hypothesis between soil carbon density, soil organic carbon, and depths.

## RESULTS AND DISCUSSION

**Mangrove forest structure and aboveground biomass**

Forest structures are represented in Table 1. The tree density in the two old forests and one young forest varied significantly: Mahebourg's forest had more trees than Le Morne. Mahebourg had the tallest tree, while the shortest was found in the young forest at Le Mome. The old forest at Le Morne had the broadest DBH range, meaning that its ecosystem contained the thickest trees among all the sites, and the young forest's sapling had the shortest range. The relative density at le Morne was 99.38% in the old, 100% in the young, and 99.16% in Mahebourg. In the old forests, biomass ranged from 461.88 t $ha^{-1}$ from 591 plants at Mahebourg to 261.42 t $ha^{-1}$ from 479 plants at Le Morne, whereas biomass of only 19.42 t $ha^{-1}$ from 360 plants was obtained in the young forest at Le Morne. At Le Morne, the maximum frequency was between the DBH range 2<x<4 cm (n=175) in the old forest, 0<x<2 cm (n= 180) in the young forest, and at Mahebourg, it was 6<x<8 cm (n= 260).

**Percentage of organic carbon in the soil in different inundation zones**

The mean organic carbon (OC) percentage in the seaward, middle and landward soil at Le Morne was 9.27 ± 0.03%, 9.80 ± 0.26%, and 9.86 ± 0.47%, respectively, in the old forest and 9.90 ± 0.16%, 9.74 ± 0.14% and 10.55 ± 0.16 respectively in the young forest and Mahebourg had a mean OC percentage of 13.71 ± 0.62%, 13.21 ± 0.25% and 13.20 ± 0.34% in the seaward, middle and landward respectively

**Table 1.** Structural features and aboveground biomass of the mangrove forests per 225 m² sample area

| Site | Forest structure | | | | | Aboveground biomass (t ha⁻¹) |
|---|---|---|---|---|---|---|
| | Height range (m) | Dbh range (cm) | Seedlings density (per hectares) | Saplings density (per hectares) | Adult density (per hectares) | |
| Le Morne (Old forest) | 0.40 - 5.00 | 1.11- 15.60 | $4.67 \times 10^4$ | $1.16 \times 10^4$ | $9.73 \times 10^3$ | 261.42 |
| Le Morne (Young Forest) | 0.20 - 3.50 | 0.95 – 5.09 | $1.03 \times 10^4$ | $1.51 \times 10^4$ | $9.33 \times 10^2$ | 19.42 |
| Mahebourg (old forest) | 0.30 - 6.90 | 2.23 – 9.55 | $9.0 \times 10^4$ | $6.9 \times 10^3$ | $1.94 \times 10^4$ | 461.88 |

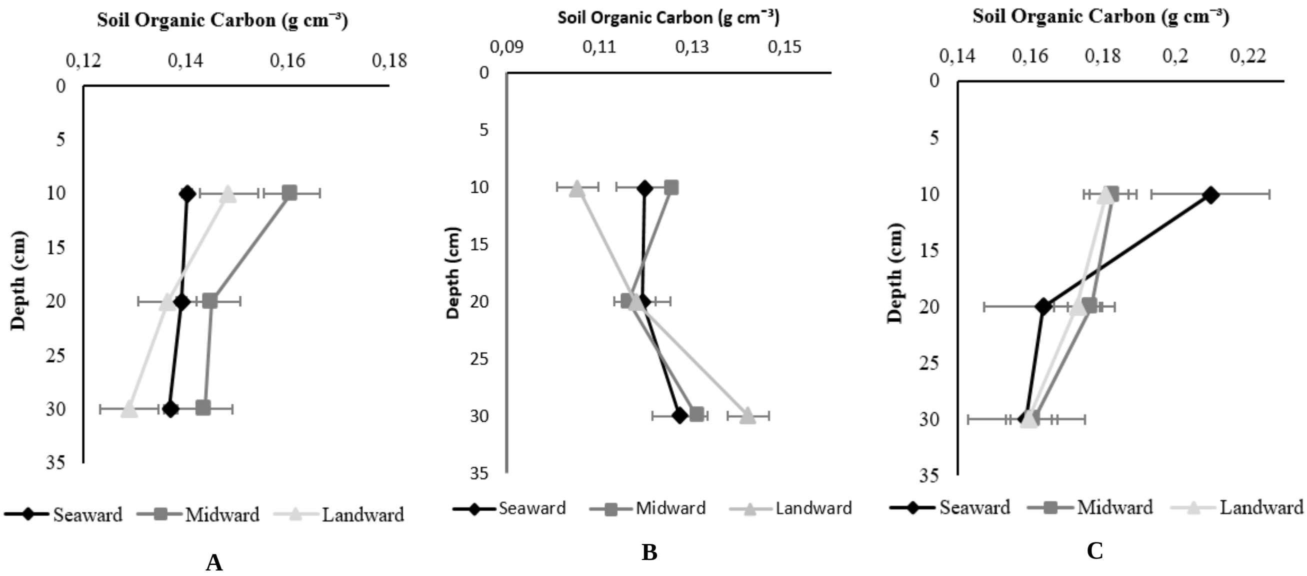



**Figure 4.** Distribution of soil organic carbon [g cm⁻³] in the topsoil of 30 cm in the *Rhizophora mucronata* stands along the coastline of Mauritius in different inundation zones at Le Morne and Mahebourg where the horizontal bars indicate the standard error of the means [n = 81]. A. Le Morne (Old forest); B. Le Morne (Young forest); C. Mahebourg (Old forest)

## Soil organic carbon content

Figure 4 represents the soil organic carbon contents at Le Morne and Mahebourg. The greatest soil organic carbon (SOC) stock value was observed in the mangrove forest at Mahebourg, with a mean total of 1.567 ± 0.005 g cm⁻³ in the 30 cm topsoil. At Le Morne, a mean total of 1.271 ± 0.003 g cm⁻³ and 1.104 ± 0.004 g cm⁻³ in the 30 cm topsoil in the old forest and young forest, respectively, was found. The difference between SOC and depths was significant at Mahebourg (Kruskal-Wallis Test, n= 81, p= 0.027) but not at Le Morne (Kruskal-Wallis Test, n= 81, p= 0.177 (old forest), p= 0.875 (young forest)).

## Soil carbon density

Soil carbon density (SCD) for 81 subsamples on each site for the upper 30 cm was determined at Le Morne and Mahebourg (Figure 5). SCD was not significantly different in each depth in the old forest (Kruskal-Wallis Test, n=81, p= 0.430) and the young forest (Kruskal-Wallis Test, n= 81, p= 0.875) at Le Morne. The variations of SCD in the old forest represent a slight decrease from 0.0140 g cm⁻³ at a depth of 0-10 cm to 0.0137 g cm⁻³ at a depth of 20-30 cm in the seaward zone, a decrease from 0.0161 g cm⁻³ at depth 0-10 cm to 0.0144 g cm⁻³ at depth 20-30 cm in the middle zone and a decrease from 0.0149 g cm⁻³ at the depth 0-10 cm to 0.0129 g cm⁻³ at the depth 20-30 cm in the landward zone. The variations in SCD in the young forest show that there was a decrease from 0.0120 g cm⁻³ at a depth of 0-10 cm to 0.0105 g cm⁻³ at a depth of 20-30 cm in the seaward zone, a slight decrease from 0.0119 g cm⁻³ at the depth 0-10 cm to 0.0118 g cm⁻³ at the depth 20-30 cm in the middle zone and an increase from 0.0127 g cm⁻³ at the depth 0-10 cm to 0.0142 g cm⁻³ at the depth 20-30 cm in the landward zone.

SCD significantly differed between 0-30 cm at Mahebourg (Kruskal-Wallis Test, n=81, p= 0.027). The variations at Mahebourg show that there was a decrease in all zones. In the seaward zone, SCD decreased from 0.0210 g cm⁻³ at a depth of 0-10 cm to 0.0159 g cm⁻³ at a depth of 20-30 cm. In the middle zone, SCD decreased from 0.018 g cm⁻³ at a depth of 0-10 cm to 0.0161 g cm⁻³ at a depth of 20-30 cm, and in the landward zone, SCD decreased from 0.0181 g cm⁻³ at the depth 0-10 cm to 0.0160 g cm⁻³ at the depth 20-30 cm.

## Total carbon storage

Organic carbon is sequestered in both biomass and soil in different quantities, with the soil pool as a greater contribution to total carbon storage. Table 2 represents the organic carbon in the aboveground biomass and soil, which are then computed to give the total organic carbon in the ecosystem. At Le Morne, in the old forest, AGB was highest in the middle zone (142.42 t ha$^{-1}$), and lowest in the seaward zone (13.41 t ha$^{-1}$), and in the young forest, AGB was highest in the seaward zone and lowest in landward accounting for 8.27 t ha$^{-1}$ and 4.19 t ha$^{-1}$ respectively. For the old forest at Le Morne. SOC was highest in the middle zone (135.00 t ha$^{-1}$) and lowest in the landward zone (124.21 t ha$^{-1}$), and for the young forest, SOC was highest in landward and lowest in seaward, with recorded values of 120.12 t ha$^{-1}$ and 105.19 t ha$^{-1}$ respectively. At Mahebourg, AGB was highest in the middle zone and lowest in landward, having 199.84 t ha$^{-1}$ and 89.50 t ha$^{-1}$, respectively, and SOC was highest in seaward and lowest in landward with recorded values of 159.71 t ha$^{-1}$ and 154.08 t ha$^{-1}$ respectively. The organic carbon between AGB and SOC was not significantly different at Le Morne (Kruskal-Walis Test, n=81, $p = 0.202$ (old forest), $p= 0.670$ (young forest)). Still, it was significantly different at Mahebourg (Kruskal-Walis Test, n=81, $p = 0.027$).

**Carbon storage in Mauritian mangrove forests of different ages compared to other mangrove forests**

*Total organic carbon in mangrove forests*

Total carbon stock consisted of the carbon sequestered in soil and biomass. The mean carbon storage for this study (642.56 t ha$^{-1}$) falls within the global range of 55-1376 t ha$^{-1}$ (Howard et al. 2014b). The mean carbon stock obtained during this study was higher compared to the mean value obtained from a mangrove stand in Kerala, India consisting of eight different species (153.64 t ha$^{-1}$: Vinod et al. 2019) and in the Farasan Islands in Saudi Arabia, consisting of *Rhizophora mucronata* forest having 108 t ha$^{-1}$ of organic carbon (Eid et al. 2019). Compared to other countries with different species, the mangrove *Avicennia marina* in Qatar had a total carbon stock of 45.70 t ha$^{-1}$ which was very low compared to the Mauritian mangrove forest (Chatting et al. 2020). In a study in Berau and Segara Anakan Lagoon in Central Java, each site had a mean total organic carbon of 615 t ha$^{-1}$ and 298 t ha$^{-1}$, respectively (Kusumaningtyas et al. 2018). The OC varied spatiotemporally because of tree density, forest age, soil texture, primary productivity, geographical and morphological setting, species composition, and regional climate (Eid et al., 2019). Organic carbon in the old forests was higher because of more productivity and autochthonous sources such as underground roots, dropped litter, and other locally produced mangrove materials.

*Soil total organic carbon in mangrove forests*

In this study, the soil carbon pool in the old forest at Le Morne accounted for 60% of the total blue carbon and 95 % for the young forest, whereas the biome at Mahebourg accounted for 50% only. Other studies confirmed that mangrove soils have the capacity to sequester 50 to >90% of the total carbon storage of mangrove ecosystems (Johnson et al., 2019). The mean SOC stock for this study was 395.04 ± 66.3 t ha$^{-1}$ which was greater than the global mean value (361 t ha$^{-1}$: Sanderman et al. 2018). The soil of the old forests holds larger stores of blue carbon than young forests, plausibly because the latter had small supplies of allochthonous material, small carbon accumulation time, low productivity, and an increase in the rate of sediment respiration. The high level of SOC was due to autochthonous matter as the old forest had high biomass levels, which contributed to more productivity. The pedogenetic layers were enriched in autochthonous organic matter, which increases with forest age and seagrass leaves. Also, a blue carbon ecosystem was washed into the bushes by tidal flux, stuck between roots which decomposed, adding to the SOC (Xue et al. 2009).

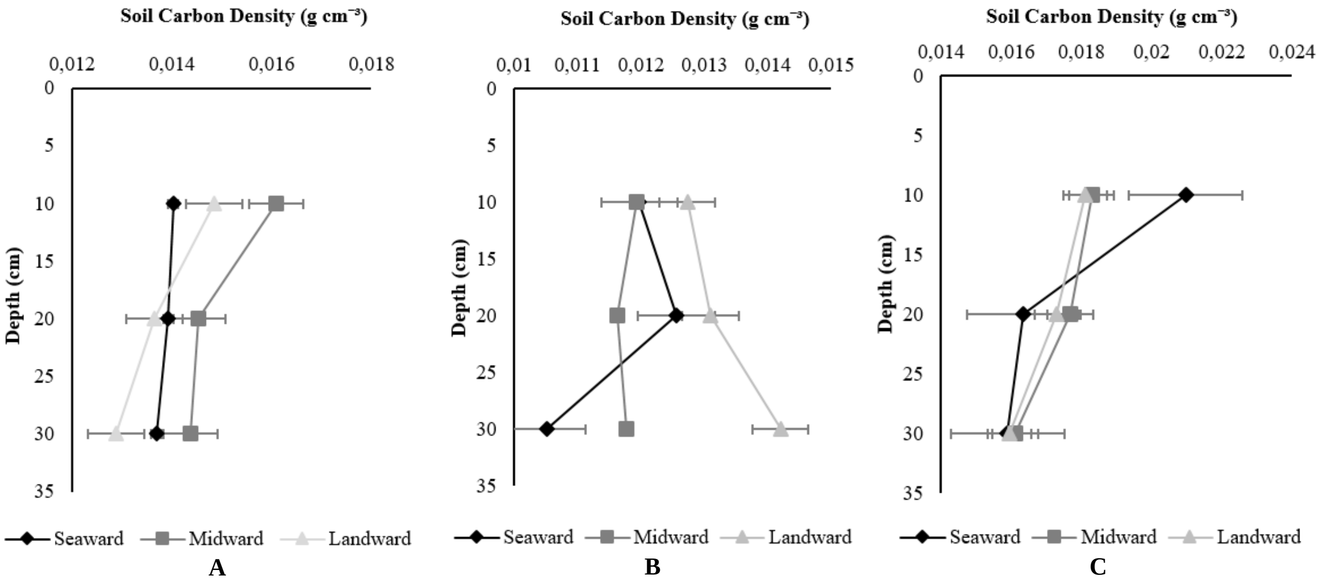



**Figure 5**. Distribution of soil carbon density [g cm$^{-3}$] concerning soil depth [cm] in the *Rhizophora mucronata* forest at Le Morne and Mahebourg along the coastline of south of Mauritius. Horizontal bars represent the standard error of the means [n=81]. A. Le Morne (Old forest); B. Le Morne (Young forest); C. Mahebourg (Old forest)

**Table 2**. Blue organic carbon (± SD) in the aboveground and belowground, and the total carbon storage is the sum of all the carbon pools in each site

| Studied sites | Aboveground (t ha⁻¹) | Belowground (t ha⁻¹) | Total carbon (t ha⁻¹) |
|---|---|---|---|
| | Using allometric equation | Soil (0-30 cm) | |
| Site 1 (Old forest at Le Morne) | 258.94 ± 11.95 | 383.70 ± 0.14 | 642.64 ± 5.62 |
| Site 2 (Young forest at Le Morne) | 19.43 ± 2.66 | 331.33 ± 0.92 | 350.76 ± 4.06 |
| Site 3 (Old forest at Mahebourg) | 461.88 ± 53.44 | 470.08 ± 0.85 | 931.50 ± 17.06 |

*Carbon sequestration in AGB in forests of different ages*

The results showed that AGB first asymptotes with the age of the forest. Salum et al. (2020) studied Guaras Island in Brazil to get an estimate of 246.90 t ha⁻¹ of aboveground biomass in the forest of *Avicennia germinus* and *Rhizophora* species using the LiDAR method. Kirui et al. (2006) showed a *Rhizophora mucronata* mangrove forest in Gazi Bay, Kenya, had a biomass of 452.02 t ha⁻¹, where the plants had a mean diameter of 11.62 cm. The difference in biomass was because of diameter, environmental conditions, geographical location, and different sampling methods; for the Kenyan study, the harvested method was used, whereas the non-destructive method was used for this study. The biomass of mangroves in different parts of the world varies due to climatic conditions, species and topography of the surveyed region, geomorphology, history of the forest structure and its age, tide variations, and edaphic factors (Kamruzzaman et al. 2017). Another reason could be the distance of the forest from the equator, as the production of mangroves depends on latitude and longitude (Saenger and Snedaker 1993). The biomass of the old forest was greater than the young forest as the former had taller trees with greater canopy acting as an indicator of the status of the stand and also had larger DBH classes with greater circumference than the young forest resulting in greater biomass.

**Carbon storage varies with different forest ages, structures, and environmental factors.**

Variability in carbon storage can be determined using variables such as forest age, structure, tree size, and density (Johnson et al., 2019). Taller trees have the capacity to store more carbon in their soil than the shrub mangroves, and without structural variations, the estimation of carbon storage would be difficult. An old forest could have greater carbon storage due to years of accumulation of allochthonous and autochthonous carbon in their carbon pools (Johnson et al., 2019). Coastal geomorphological characteristics influence mangrove distribution, and their heights decrease as latitudes increase, temperature and precipitation decrease, and high salinity (Vinh et al. 2019). Wave action, rainfall, and ample freshwater input are important factors that help determine the study site's forest structure. These factors affect erosion control, aridity, salinity, nutrient inputs, and soil quality (Lacerda 2002).

*Structure and distribution of mangrove forests*

Mauritian mangrove forest structures differ from other countries as Mauritius is a remote island, and zonation patterns are influenced by the tidal influence (Ball 1980). Seedling density was highest in forests nearer to the sea due to the hydrochory process, and most of them were observed in canopy gaps due to the requirements for growth, such as light and space (Putz and Chan 1986). The old forests being closer to the oceans, seedlings were distributed all over the intertidal forest because of dispersion and establishment success of propagules with currents and tides. The eco-geomorphic conditions determine the mangrove's diversity pattern, and terrestrial runoff maintains the eco-physiological condition of Mahebourg's forest as it is closer to the land, preventing salinity stress and supplying nutrients to support forest development (Chen and Twilley 1999). The young forest had lower seedling density because it was found far from the sea actions that could propagate. The seedling population was used as an indicator of the reproductive status of the studied sites.

*Variability of soil organic carbon in different inundation zones*

Mangrove wetlands have the potential to sequester blue carbon in their soil, and most OC in this study was found in the soil. However, the quantity of SOC fluctuated widely due to environmental variables and their interactions (Chen et al., 2020). For example, in Mauritius at Le Morne, the old forest had the highest SOC (135.00 t ha⁻¹) in the middle zone because of the accumulation of many years' leaf litters, belowground dead fine roots, and micro and macro algae colonizing the forest floor which was added to the total SOC. Lowest SOC was observed in the landward zone (124.21 t ha⁻¹) because many young plants resulted in a small supply of autochthonous matter and little freshwater inflow. However, in the young forest, the landward zone had the highest SOC (120.12 t ha⁻¹). The seaward zone had the lowest SOC (105.19 t ha⁻¹), as the landward zone had OC from allochthonous sources of terrestrial origin and fluvial inputs and less hydrological flushing, which helped the accumulation of organic carbon-rich litter on the soil surface (Sasmito et al. 2020).

Moreover, in Mahebourg, the highest SOC was observed in the seaward zone (159.70 t ha⁻¹) and the lowest in the landward zone (154.08 t ha⁻¹) since the substratum consisted of a tidally submerged suboxic layer that supported anaerobic decomposition. The seaward zone had the highest OC because of autogenous changes in the soil surface and larger inundation times. That facilitates prolonged anaerobic conditions in the soil, resulting in organic matter retention. In addition, the hydroperiod decreased soil bulk density which caused changes in porosity, permeability, and ventilation of the mangrove soil, giving rise to OC (Chen et al., 2020). Still, the landward zone had less adult density, little productivity, and more artificial stressors.

*Variability of soil bulk density in different soil depths*

Soil bulk density (SBD) was used to describe soil's compaction and water permeability. Soil carbon density was an important factor for blue carbon storage, where higher SBD had little liquid phase space resulting in lower soil moisture content (Xiong et al. 2018). SBD affected soil porosity, permeability, ventilation, and OC storage, and a high SBD resulted in low OC accumulation (Chen et al., 2020). The variations of SCD were effects of physical and chemical factors such as pH, soil bulk density, soil types, and minerals (Eid and Shaltout 2015). Similar results were reported by Chen et al. (2020) in Zengying, China, that SCD decreased with increasing depth, and this could be because of soil moisture and the number of fine roots decreasing with increasing depth. The SCD also changed due to the interplay between variables; decomposition, leaching, hydrologic regimes, biological cycling, soil erosion, and weathering minerals (Eid et al., 2019). However, SCD was less on topsoil in the seaward and landward zone in the young forest because of the influence of tidal inundation on the erosion of accumulated matter on the floor, less productivity, and human stressors as the forest were near the main road.

It can be concluded that the structural attributes, including tree density, height, and diameter at breast height, were considered in the present study as key factors in evaluating the variability of carbon storage of mangrove forests of different ages. The carbon content from aboveground biomass and soil was highest at Mahebourg, with values of 461.88 t $ha^{-1}$ and 470.08 t $ha^{-1}$, respectively. Most of the results showed that the surface layers of the soil stored the highest amount of blue carbon. The forest with taller trees holds more blue carbon than the forest with shrub mangroves. The former benefited from autochthonous and allochthonous carbon and had a high biomass level, which contributed to more productivity and eventually increased litter, thus increasing the soil organic matter. This study is an important contribution to understanding the mangrove ecosystem as stores of blue carbon and reducing atmospheric carbon in small islands. It is imperative to protect these ecosystems to ensure that their roles of sequestrating blue carbon and other benefits they provide.

## ACKNOWLEDGEMENTS

The authors are thankful to the Ministry of Blue Economy, Marine Resources, Fisheries, and Shipping for granting us permission to collect samples. We also acknowledge the logistical support of the University of Mauritius. Moreover, part of the work has been accepted for presentation as a poster at the University of Mauritius Research Week 2020.

## REFERENCES

Abib S, Appadoo C. 2012. A pilot study for the estimation of aboveground biomass and litter production in *Rhizophora mucronata* dominated mangrove ecosystems in the island of Mauritius. J Coast Dev 16 (1): 40-49.

Ahmed N, Cheung WWL, Thompson S, Glaser M. 2017. Solutions to blue carbon emissions: Shrimp cultivation, mangrove deforestation and climate change in coastal Bangladesh. Mar Policy 82: 68-75. DOI: 10.1016/j.marpol.2017.05.007

Appadoo C. 2003. Status of mangroves in Mauritius. J Coast Dev 7 (1): 1-3.

Ball MC. 1980. Patterns of secondary succession in a mangrove forest of Southern Florida. Oecologia 44 (2): 226-235. DOI: 10.1007/BF00572684

Betts T. 2006. An Assessment of Mangrove Cover and Forest Structure in Las Perlas, Panama. Thesis [Dissertation]. Heriot-Watt University, Edinburgh. [Scottish]

Chatting M, LeVay L, Walton M, Skov MW, Kennedy H, Wilson S, Al-Maslamani I. 2020. Mangrove carbon stocks and biomass partitioning in an extreme environment. Estuar Coast Shelf Sci 244: 106940. DOI: 10.1016/j.ecss.2020.106940

Chen J, Huang Y, Chen G, Ye Y. 2020. Effects of simulated sea-level rise on stocks and sources of soil organic carbon in Kandelia Obovata mangrove forests. For Ecol Manag 460: 117898. DOI: 10.1016/j.foreco.2020.117898

Chen R, Twilley RR. 1999. Patterns of mangrove forest structure and soil nutrient dynamics along the shark river estuary, Florida. Estuaries 22: 955-970.

Clarisse JO, Appadoo C, Boojhawon R. 2016. Spatial distribution of *Rhizophora mucronata* and *Bruguiera gymnorrhiza* mangroves using kriging technique: A contribution to understanding forest structure in southeast coast of Mauritius (Indian Ocean). Univ Mauritius Res J 22: 458-483.

Climate-Data.Org, 2020b. Mahebourg Climate. Mauritius. https://en.climate-data.org/africa/mauritius/grand-port/mahebourg-25187/#:~:text=Mahebourg%20climate%20summary&text=The%20average%20annual%20temperature%20in,mm%20%7C%2071.4%20inch%20per%20year. [Accessed 15 December 2020].

Climate-Data.Org, 2020a. Le Morne Climate. Mauritius. https://en.climate-data.org/africa/mauritius/black-river/le-morne-773895/#:~:text=Le%20Morne%20climate%20summary&text=The%20K%C3%B6ppen%2DGeiger%20climate%20classification,is%201629%20mm%20%7C%2064.1%20inch. [Accessed 15 December 2020].

Cuellar-Martinez T, Ruiz-Fernández AC, Sanchez-Cabeza JA, Libia-Hascibe PLH, Sandoval-gil J. 2019. Relevance of carbon burial and storage in two contrasting blue carbon ecosystems of a north-east pacific coastal lagoon. Sci Total Environ 675: 581-593. DOI: 10.1016/j.scitotenv.2019.03.388

Cui X, Liang J, Lu w, Chen H, Liu F, Lin G, Xu F, Luo Y, Lin G. 2018. Stronger ecosystem carbon sequestration potential of mangrove wetlands with respect to terrestrial forests in subtropical China. Agric For Meteorol 249: 71-80. DOI: 10.1016/j.agrformet.2017.11.019

Dahdouh-Guebas F, Koedam N. 2008. Long-term retrospection on mangrove development using transdisciplinary approaches: A review. Aquat Bot 89: 80-92. DOI: 10.1016/j.aquabot.2008.03.012

Delvecchia AG, Bruno JF, Benninger L, Alperin M, Banerjee O, Morales IDD. 2014. Organic carbon inventories in natural and restored Ecuadorian mangrove forests. PeerJ 2: e388. DOI: 10.7717/peerj.388

Eid EM, Khedher KM, Ayed H, Arshad M, Moatamed A, Mouldi A. 2019. Evaluation of carbon stock in the sediment of two mangrove species, *Avicennia marina* and *Rhizophora mucronata*, growing in the Farasan islands, Saudi Arabia. Oceano 227: 1-14. DOI: 10.1016/j.oceano.2019.12.001

Eid EM, Shaltout KH. 2015. Distribution of soil organic carbon in the mangrove *Avicennia marina* (Forssk) Vierh. Along the Egyptian Red Sea Coast. Reg Stud Mar Sci 3: 76-82. DOI: 10.1016/j.rsma.2015.05.006

Heiri O, Lotter A, Lemcke G. 2001. Loss on ignition as a method for estimating organic and carbonate content in sediments: reproducibility and comparability of results. J Paleolimnol 25 (1): 101-110. DOI: 10.1023/A:1008119611481

Howard J, Hoyt S, Isensee K, Pidgeon E, Telszewski M. 2014a. Coastal blue carbon methods for assessing carbon stocks and emissions factors in mangroves, tidal salt marshes, and seagrasses meadows. In: Fourqurean J, Johnson B, Kauffman JB, Kennedy H, Lovelock C, Saintilan N, Alongi DM, Cifuentes M, Copertino M, Crooks S, Duarte C, Fortes M, Howard J, Hutahaean A, Kairo J, Marba N, Murdiyarso D, Pidgeon E, Ralph P, Serrano O (eds). Field Sampling of Vegetative Carbon Pools in Coastal Ecosystems. Conservation International, Intergovernmental Oceanographic Commission of UNESCO, International Union for Conservation of Nature. Arlington, Virginia, USA.

Howard J, Hoyt S, Isensee K, Pidgeon E, Telszewski M. 2014b. Coastal blue carbon methods for assessing carbon stocks and emissions factors in mangroves, tidal salt marshes, and seagrasses meadows. In: Howard J, Hoyt S, Isensee K, Telszewski M, Pidgeon E, Telszewski m, Crooks S, Emmer I, Herr D, Hoyt S, Laffoley D, Quesada M, Valdes JL, Wagey T (eds). Why Measure Carbon Stocks. Conservation International, Intergovernmental Oceanographic Commission of UNESCO, International Union for Conservation of Nature. Arlington, Virginia, USA.

Johnson JL, Raw J, Adams JB. 2019. First report on carbon storage in a warm-temperate mangrove forest in South Africa. Estuar Coast Shelf Sci 235: 106566. DOI: 10.1016/j.ecss.2019.106566.

Kamruzzaman M, Ahmed S, Osawa A. 2017. Biomass and net primary productivity of mangrove communities along the Oligohaline zone of Sundarbans, Bangladesh. For Ecosyst 4: 16. DOI: 10.1186/s40663-017-0104-0.

Kirui B, Kairo JG, Karachi M. 2006. Allometric equations for estimating above ground biomass of *Rhizophora mucronata* Lamk. (Rhizophoraceae) mangroves at Gazi Bay, Kenya. Western Indian Ocean J Mar Sci 5 (1): 27-34. DOI: 10.4314/wiojms.v5i1.28496.

Kusumaningtyas MA, Hutahaean AA, Fischer HW, Pérez-Mayo M, Pittauer D, Jennerjahn TC. 2018. Variability in the organic carbon stocks, sources, and accumulation rates of Indonesian mangrove ecosystems. Estuar Coast Shelf Sci 218: 310-323. DOI: 10.1016/j.ecss.2018.12.007

Lacerda LD. 2002. Mangrove ecosystem: Function and management. In: Lacerda LD, Conde JE, Kjerfve B, Alvarez-Leon R, Alarcon C, Polania J (eds). American Mangroves. Springer, Verlag Berlin Heidelberg, New York.

Luo L, Gu JD. 2016. Alteration of extracellular enzyme activity and microbial abundance by biochar addition: Implication for carbon sequestration in subtropical mangrove sediment. J Environ Manage 182: 29-36. DOI: 10.1016/j.jenvman.2016.07.040

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.

Penaranda MLP, Kintz JRC, Pena EJ. 2019. Carbon stocks in mangrove forests of the Colombian pacific. Estuar Coast Shelf Sci 227: 106299. DOI: 10.1016/j.ecss.2019.106299.

Putz FE, Chan HT. 1986. Tree growth, dynamics, and productivity in a mature mangrove forest in Malaysia. For Ecol Manag 17: 211-230. DOI: 10.1016/0378-1127(86)90113-1.

Ranjan R. 2019. Optimal mangrove restoration through community engagement on coastal lands facing climatic risks: The case of Sundarbans region in India. Land Use Policy 81: 736-749. DOI: 10.1016/j.landusepol.2018.11.047.

Saenger P, Snedaker SC. 1993. Pantropical trends in mangrove aboveground biomass and annual litterfall. Oecologia 96 (3): 293-299. DOI: 10.1007/BF00317496.

Salum RB, Souza-Filho PWM, Simard M, Silva CA, Fernandes MEB, Cougo MF, Rogers K. 2020. Improving mangrove aboveground biomass estimates using LiDAR. Estuar Coast Shelf Sci 236: 106585. DOI: 10.1016/j.ecss.2020.106585.

Sanderman J, Heng T, Fiske G, Solvik K, Adame MF, Benson L, Bukoski JJ, Carnel P, Cifuentes-Jara M, Daniel-Donato D, Duncan C, Eid EM, Ermgassen PZ, Lewis CJE, Macreadie PI, Glass L, Gress S, Jardine SL, Jones TG, Nsombo EN, Rahman M, Sanders CJ, Spalding M, Landis E. 2018. A global map of mangrove forest soil carbon at 30m spatial resolution. Environ Res Lett 13: 055002. DOI: 10.1088/1748-9326/aabe1c

Sasmito SD, Kuzyakov Y, Lubis AA, Murdiyarso D, Hutley LB, Bachri S, Friess DA, Martius C, Borchard N. 2020. Organic carbon burial and sources in soils of coastal mudflat and mangrove ecosystems. Catena 187: 104414. DOI: 10.1016/j.catena.2019.104414.

Shaltout KH, Ahmed MT, Alrumman SA, Ahmed DA, Eid EM. 2019. Evaluation of the carbon sequestration capacity of arid mangroves along nutrient availability and salinity gradients along the red sea coastline of Saudi Arabia. Oceanologia 227: 1-14. DOI: 10.1016/j.oceano.2019.08.002

Thorhaug AL, Poulos HM, Portillo JL, Barr J, Lara-Dominguez AL, Ku TC, Berlyn GP. 2018. Gulf of Mexico estuarine blue carbon stock, extent and flux: Mangroves, marshes, and seagrasses: A North American hotspot. Sci Total Environ 653: 1253-1261. DOI: 10.1016/j.scitotenv.2018.10.011.

Vinh TV, Marchand C, Linh TVK, Vinh DD, Allenbach M. 2019. Allometric models to estimate aboveground biomass and carbon stocks in *Rhizophora apiculata* tropical managed mangrove forests (Southern Viet Nam). For Ecol Manag 434: 131-141. DOI: 10.1016/j.foreco.2018.12.017.

Vinod K, Asokan PK, Zacharia PU, Ansar CP, Vijayan G, Anasukoya A, Koya VAK, Nikhiljith M. 2019. Assessment of biomass and carbon stocks in mangroves of Thalassery estuarine wetland of Kerala, South West Coast of India. J Coast Res 86: 209-217. DOI: 10.2112/SI86-031.1

Wang L, Jia M, Yin D, Tian J. 2019. A review of remote sensing for mangrove forests: 1956-2018. Remote Sens Environ 231: 111223. DOI: 10.1016/j.rse.2019.111223

Weather atlas, 2019. Monthly weather forecast and climate Le Morne, Mauritius. Mauritius. https://www.weather-atlas.com/en/mauritius/le-morne-climate#rainfall.

Wu J, Chen B, Mao J, Feng Z. 2018. Spatiotemporal evolution of carbon sequestration vulnerability and its relationship with urbanization in China's coastal zone. Sci Total Environ 645: 692-701. DOI: 10.1016/j.scitotenv.2018.07.086

Xiong Y, Liao B, Proffitt E, Guan W, Sun Y, Wang F, Liu X. 2018. Soil carbon storage in mangroves is primarily controlled by soil properties: A study at Dongzhai Bay, China. Sci Total Environ 619-620: 1226-1235. DOI: 10.1016/j.scitotenv.2017.11.187

Xue B, Yan C, Lu H, Bai Y. 2009. Mangrove-derived organic carbon in sediment from Zhangjiang estuary (China) mangrove wetland. J Coast Res 25: 949-956. DOI: 10.2112/08-1047.1

Yang J, Gao J, Liu B, Zhang W. 2014. Sediment deposits and organic carbon sequestration along mangrove coasts of the Leizhou Peninsula, Southern China. Estuar Coast Shelf Sci 136: 3-10. DOI: 10.1016/j.ecss.2013.11.020.

**OCEAN LIFE**
Volume 4, Number 2, December 2020 E-ISSN: 2580-4529
Pages: 82-93 DOI: 10.13057/oceanlife/o040204

# Impact of nutrients on phytoplankton productivity on coastal marine waters of Mtwapa, Mida, and Kilifi Creeks, Kenya

**TUNJE MWAMUYE POLE[1], MWAKIO P. TOLE[2,♥], SAMUEL C.J. OTOR[2]**
[1]Department of Environmental Sciences, Pwani University. Kilifi, Kenya
[2]Department of Environmental Science, Kenyatta University. P.O. Box 43844-00100, GPO Nairobi, Kenya. Tel.: +254-20-810901,
♥email: tole.mwakio@ku.ac.ke

Manuscript received: 2 October 2020. Revision accepted: 24 December 2020.

**Abstract.** *Pole TM, Tole MP, Otor SCJ. 2020. Impact of nutrients on phytoplankton productivity on coastal marine waters of Mtwapa, Mida, and Kilifi Creeks, Kenya. Ocean Life 4: 82-93.* Some of the major sources of coastal and marine pollution affecting coastal conditions vary worldwide. The objective of this study was to determine the levels of three nutrients ($PO_{4}$-, $NO_{3}$-, and $NH_3$) and whether they are within acceptable limits, their relationship with ongoing human activities, and marine primary productivity. Furthermore, to achieve these objectives, marine water samples were sampled and analyzed from identified point and area pollution sources in Mtwapa, Mida, and Kilifi Creeks, Kilifi County, Kenya. A quantitative one-way ANOVA was used to determine the variations between treatments. The Spearman correlation test was computed between nutrients and carbon and nutrients and rainfall. These studies revealed that the levels of nutrients fluctuated throughout the entire study period along the three creeks and the sampling stations. Moreover, the levels of nitrates were within the oligotrophic range in all three creeks. Mtwapa Creek recorded the highest levels of nitrates at $\alpha = 0.05$ from June to November 2011. The phosphate levels in the three creek waters were not significantly different throughout the sampling period. The ammonia levels in Mtwapa were in the higher mesotrophic levels of up to 0.009 mg/L, while Kilifi and Mida were within the oligotrophic levels. Alongside Mtwapa Creek, there was a positive correlation coefficient between phosphate and carbon fixation ($r = 0.869$) in the outer creek; on the other hand, the correlation coefficient for nitrates and ammonia was negative ($r = -0.624$ and -0.295). All the nutrients had a negative correlation coefficient with rainfall in outer Mtwapa Creek ($r = -0.76$, -0.37, and -0.336 for nitrates, phosphates, and ammonia, respectively. Kilifi was positively correlated with carbon fixation and all three nutrients in the inner creek. There was a positive correlation between nutrients and rainfall in all the sampling stations. At Mida, there were positive correlations between rainfall and phosphates in all six stations ($r = 0.78$, 0.3, 0.22, 0.78, 0.23, respectively). In Mtwapa, the high levels of nutrients in the outer creek stations and the negative correlation coefficient between the nutrients and rainfall suggests that runoff did not contribute to increased levels of nutrients but rather the waster roadside canal and sewage outfall from urban and tourist development contributed to increased levels. The higher levels of nutrients in the inner creek waters of Kilifi can be attributed to the destruction of vegetation close to the creek for farming upstream of the creek. Mida Creek had the lowest recorded nutrient levels apart from phosphates in all the sampling stations. These phosphate levels most likely were contributed by wastewater from some tourist establishments, especially at the Temple point hotel, which crept to the shore waters. Furthermore, to reduce the impacts of land-based human activities, it is recommended to analyze existing land-based activities that negatively impact coastal marine ecosystems and livelihoods to design mitigation measures.

**Keywords:** Coastal marine waters, Kilifi Creeks, Mida Creeks, Mtwapa Creek, nutrients, phytoplankton productivity, pollution

## INTRODUCTION

The contamination of marine environments by urban developments, the tourism industry, and animal and crop farming is increasing worldwide. This water quality degradation affects various aquatic environments, including coastal waters, wetlands, estuaries, mangroves, rivers, and reefs, creating a serious health risk for aquatic life, humans, and wildlife (Webb and Gome-Gomez 2009; Bayan et al. 2016). Over 60 million people inhabit the WIO region's coastal areas, and most depend on exploiting the marine and coastal natural resources for their livelihoods (UNEP 2009). This region has had a rapidly growing population in recent years, which exerts pressure on the marine environment, causing degradation of critical coastal habitat, pollution, and changes in freshwater flow and sediment loads from rivers draining into the Western Indian Ocean. Many efforts have been made, and one initiative, such as the WIO-LAB project, addressed land-based activities in the West Indian Ocean (UNEP 2009). Formulating a Strategic Action Plan (SAP) to address the challenges of increased coastal water pollution was one of the outputs of the WIO-LAB project, among others

The algae often support the marine food web in the ocean. Their photosynthetic activity tends to be greatest near the coastline, where primary producers bring nitrogen, phosphorous carbon, and other nutrients from the land and fertilize. However, ocean currents transport nutrients and phytoplankton far from shore and contribute to the distribution of biological productivity (Hogarrth 1998). Apart from primary production, secondary productivity is a measure of zooplanktonic productivity. The marine zooplankton, useful in secondary productivity, belong to the mollusks, Arthropoda, phyla protozoa, and the crustaceans' largest group.

The marine pollution caused along shorelines with urban development includes discharges from sewage outfalls, which potentially cause environmental impacts on surrounding aquatic ecosystems. The testing with Direct Toxicity Assessment (DTA) or Whole Effluent Toxicity (WET) is an integral part of the regulatory framework in many countries to assess and manage leachates, effluents, and contaminated ambient waters in freshwater and marine environments. The DTA can serve as an early warning for implementing management actions and also provide a direct measure of the bioavailability and toxicity of mixtures whose chemical composition is unknown. Those common regulatory microbial indicators of fecal pollution include *Enterococcus* spp and *Escherichia coli*. The acceptable fecal indicator levels were determined during epidemiological studies (USEPA 2000).

Runoff from anthropogenic land use is a significant source of pollution impacting rivers, coastal streams, and marine waters. Agricultural production is a leading source of diffuse, non-point sources of pollution impacting waterways and coastal ecosystems. Agricultural management practices can result in depositing nutrients, sediments, pesticides, and microbial contaminants into waterways (Stuart 2010).

Due to their proximity to urban areas, Creeks are the most vulnerable marine ecosystem, and areas with agricultural practices have the potential to be polluted by agricultural fertilizers, sewerage effluents, and sediments. Those are among the most heavily polluted areas globally, with the world's population living along creeks and estuaries about 60%. In addition, 22 of the world's largest cities, including New York City, are located on an estuary (SCECAP 2010). The contamination problem of uncontrolled runoff from urban and agricultural areas, coastal and offshore environments and aquatic organisms by pathogenic bacteria manifests along the Kenyan coastal marine waters. Regardless of their vulnerability to the direct impacts of anthropogenic and natural disturbances, there have been few efforts to continuously monitor, determine, and understand the nature and extent of the impacts of land-based pollution sources. Studies on the impact of sewage discharge into coastal marine waters (Okuku et al. 2010) and the nutrient enrichment characterization in estuaries by the Kenya Marine and Research Institute (KEMFRI 2013) revealed that Mtwapa Creek was on higher mesotrophic levels than Tudor and Makupa creeks. As a result, there was an increased nutrient trend down the estuarine systems. However, there is no available information on similar nutrient studies along Kilifi and Mida Creeks. Therefore, this study evaluated whether the nutrient distribution and its impact on phytoplankton productivity have changed over time, forming the basis for suggesting further recommendations.

This study aims (i) to determine the levels of Nitrate ($NO_{3-}$), Ammonia ($NH_3$), and Phosphates ($PO_{4-}$) in the three creeks, i.e., Mtwapa, Mida, and Kilifi Creeks, Kilifi County, Kenya; (ii) to examine the spatial variations along the three creeks in the nutrient loads; (iii) to classify nutrient concentrations levels of the creek water following to standard water quality criteria; (iv) to determine the correlation between the net primary productivity levels and nutrient levels along the three creeks.

## MATERIALS AND METHODS

### Description of the study sites

The study sites were three creeks along the coastal shoreline of Kilifi County, Coast Province of Kenya (Figure 1). Mtwapa Creek, which marks the boundary between Kilifi and Mombasa counties, lies between 3°45'0 S and 3°47'0 S to 39°42" 0 E and 39°48" 0 E. Mida Creek is situated at the border of Kilifi and Malindi counties at between 3°20'0" S and 3°25'0" S and 39°55'0" E and 39°58'0" E. Kilifi Creek in the middle of Kilifi town is located between 3°35'0" S and 3°38'0" S, and 39°43'0" E and 39°52'0" E. The region experiences an average rainfall of about 900 mm to 1,100 mm annually, with mean temperatures between 25°C to 30°C.

Seven sampling sites were selected along Mtwapa Creek: i.e., Moorings floating restaurant (S39), Shimo-La-Tewa waste water drainage outlet (S38), Drainage outlet from Kadongo landing site (S27), Maize farms opposite Kadongo (S24), Mtwapa town (S36), Shoreline near Kwetu Training Center (S29), and the Control site in the open ocean facing the mouth of the creek (SC). These sites constitute the creek point and diffuse sources of nutrient input.

Seven sampling sites along Kilifi Creek were also identified and marked using a GPS instrument. These sites were diffuse sources, i.e., Mnarani Club beach (K01), Boat Yard beach (K02), Maringoni valley shoreline (K05), Maya Mangrove conservation shorelines (K06), Sea Horse shoreline (K03), Maringoni farms shorelines (K04), and the control samples obtained from the open ocean facing the mouth of the creek.

Seven sampling stations along Mida Creek were also identified and marked with the GPS instrument. These also constitute diffuse nutrient sources entering the creek. They include the Temple point shoreline (md1), Mida Creek Mangrove project conservation shoreline (md3), Sudi Island picnic site (md4), Captain Andy and Hemmingway's Boatyard (md2), Shoreline opposite Captain Andy (md5), Safari Blue beach (md6) and the control at the open ocean facing the mouth of the creek.

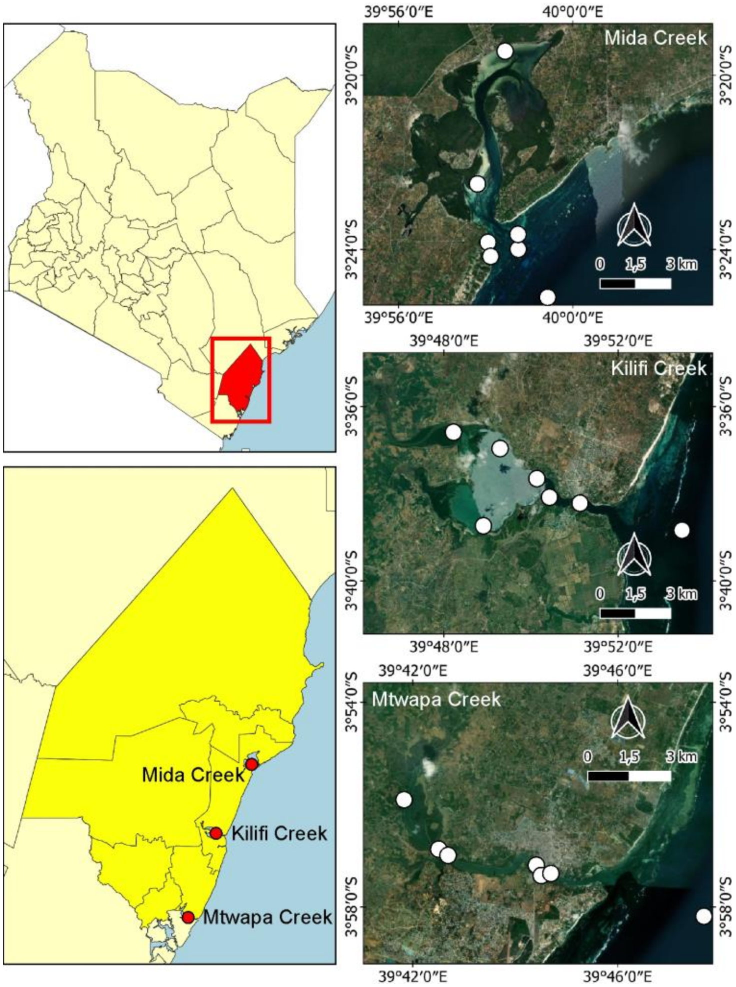



**Figure 1.** Sampling sites (solid dots) along Mtwapa, Kilifi, and Mida Creeks in Kenya

The sampling design was purposive, with sampling sites confined at identified locations with proximity to the identified sources of pollution within a 10 m distance from the shoreline. The sampling sites targeted areas close to urban developments, conservation areas, tourist hotel establishments, the Shimo La Tewa Maximum prison establishment, and the agricultural areas upstream of the creeks, representing both the sources area and point of effluents into the creeks. The sampling stations were marked using a GPS instrument, and grid references were saved for the entire study period. Sampling was done along the three creeks each month for six months, from June 2011 to November 2011. This period could allow a temporal comparison of the dry season from June to August 2011 and the short rain-wet seasons from October and November 2011. Furthermore, from the seven stations, triplicate surface water (0.5 m) samples were obtained, including the open water control in each creek. The water samples for determining nutrients were fixed on-site with Mercury Chloride (HgCl). Simultaneously, the water sample for instant carbon determination was fixed with a mixture of 0.4 mL of Manganese Sulphate ($MnSO_4$) and 0.4 mL of Potassium Iodide (KI).

**Determination of nutrient levels in the creek waters**

The water samples were collected in pre-cleaned polypropylene sample bottles that were autoclaved at 121°C 12 Lbs/sq In for 20 min. Nitrates, Phosphates, and Ammonia were determined using standard colorimetric procedures (Ascorbic Acid-Molybdate and Cadmium reduction methods), ) at the Kenya Marine and Fisheries Research Institute (KEMFRI) laboratories in Mombasa, which produced colored complexes that were measured photometrically according to APHA (1995).

*Determination of levels of nitrates ($NO_3^-$)*

Concentrated ammonium chloride was prepared by dissolving 125 g of analytical reagent-grade ammonium chloride in 500 mL of distilled water and then stored in a glass or plastic bottle. Next, a dilute ammonium chloride solution was prepared by diluting 12.5 mL of conc. Ammonium chloride solution to 500 mL with distilled water, then stored in a glass or plastic bottle.

The sulfanilamide solution was prepared by dissolving 5 g of sulfanilamide in a mixture of 50 mL of hydrochloric acid concentrated (sp.Gr.1.18) and around 300 mL of distilled water and diluted with distilled water to 500 ml.

N-(-1-naphthyl)-ethylenediamine dihydrochloride solution was prepared by dissolving 0.5 g of dihydrochloride in 500 mL of distilled water. This solution was then stored in a dark bottle.

Cadmium-Copper fillings reactivation was done by removing fillings from the column, washing with 5% v/v hydrochloric acid, and next with distilled water until the pH of the decanted solution was >5. Next, the fillings were reactivated with 2% w/v copper sulphate until the blue color left the solution. Next, a small glass wool plµg was placed at the bottom of the reduction column, and then the column was filled with dilute ammonium chloride solution. Finally, a slurry of the cadmium–copper fillings was poured in, and the column was gently packed to a height of about 30 cm. The fillings were kept on not to be dried out during the procedure.

Stock standard A (400 µgat N/L) was prepared by dissolving 0.51 g of analytical grade potassium nitrate ($KNO_3$) in a 500 mL volumetric flask and made to volume using distilled water. Stock standard B (20 µgat N/L) was prepared by taking 4 mL of solution A and diluting it in a volumetric flask with distilled water to make 100 mL of solution.

The determination of nitrates in the sample was done by measuring 50 mL of water sample in a volumetric flask, adding 1.0 mL of concentrated ammonium chloride, and mixing. Next, 5 mL of this solution was poured on top of the column and allowed to drain off. Finally,15 mL of the residue was again poured into the column and allowed to drain off. The remaining 30 mL was poured into the column, and 25 mL was collected into the drain Erlenmeyer flask and put under the collection tube. The remaining solution was left to drain, and 100 mL of dilute ammonium chloride was added to rinse the column. The processes were repeated for each sample until all the samples were passed through the column.

To the 25 mL samples collected, 0.5 mL of sulfanilamide solution was added from an automatic pipette, mixed, and allowed to react for 2 minutes. Still, in less than 10 minutes, 0.5 L of the naphthalene-diamine solution was added and immediately mixed.

The absorbance was measured after 1 hour using a UV/Vis spectrophotometer at a 543 nm wavelength following APHA (1995).

*Determination of levels of phosphates ($PO_4^-$)*

The sulphuric acid solution was prepared with 100 mL of analytical-grade concentrated sulphuric acid (sp.gr = 1.82) added to 900 mL of distilled water, which was left to cool and stored in a glass bottle.

The ascorbic acid solution was prepared with 27 g of ascorbic acid dissolved in 50 mL of distilled deionized water and frozen-stored in a plastic bottle. Ammonium molybdate reagent was prepared with 15 g of analytical grade ammonium paramolybdate dissolved in 500 mL of distilled deionized water and stored in a glass bottle. A potassium antimony-tartrate solution was prepared by dissolving 0.34 g of potassium antimony tartrate in 250 mL of distilled deionized water. A mixed reagent was prepared by mixing 100 mL ammonium molybdate, 250 mL sulphuric acid solution, 100 mL ascorbic acid, and 50 mL Potassium antimony tartrate. This reagent was mixed as required and was not stored. This quantity prepared was approximately adequate for 100 samples.

Stock A (1000 µgat P/L) was prepared by dissolving 0.136 g of Anhydrous potassium dihydrogen phosphate in 900 mL of distilled deionized water in 1 mL of chloroform topped with 1 liter of distilled deionized water. Stock B (40 µgat P/L) was prepared by diluting 4 mL of stock standard A in a volumetric flask to 100 mL with deionized distilled water.

The following serial dilutions were prepared from the stock B as follows:

| mL of stock B in 100 mL Volumetric flask | Concentration (µgat N/L) |
|---|---|
| 0.2 | 0.1 |
| 2.0 | 1.0 |
| 4.0 | 2.0 |
| 6.0 | 3.0 |
| 8.0 | 4.0 |
| 10.0 | 5.0 |

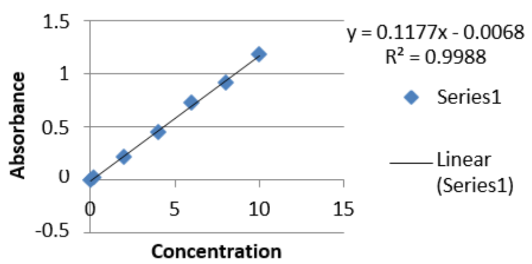



**Figure 2.** Calibration curve for $NO_3^-$

Working standards were prepared by using the following serial dilutions of stock B:

| mL of stock B in 100 mL | Concentration (µgat P/L) |
|---|---|
| 0.2 | 0.08 |
| 2.0 | 0.8 |
| 4.0 | 1.6 |
| 6.0 | 2.4 |
| 8.0 | 3.2 |
| 10.0 | 4.0 |

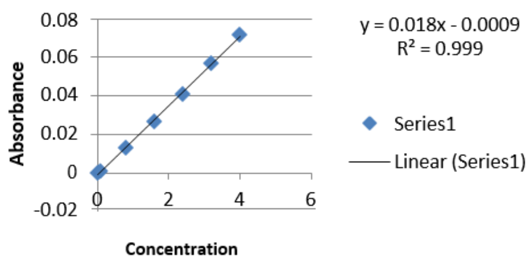



**Figure 3.** Calibration curve for $PO_4^-$

The procedure on samples to determining phosphates was by measuring 20 mL of water sample in a volumetric flask and adding 2 mL of mixed reagent using a micropipette. The samples were left to stand for between 30 minutes to two 2 hours, and then the absorbance reading was taken at a wavelength of 885 nm using a UV/Vis spectrophotometer

**The model for the spectrophotometer (Beer-Lambert Law)**

$\%T = (P/P_o) \times 100\%$
$A = -\mathrm{Log}\,(\%T/100)$
$A = \mathcal{E}bc$

Where;
T : Transmittance
A : Absorbance
b : Solution path length
$\mathcal{E}$ : Molar absorptivity
c : Molar concentration

**Determination of levels of ammonia ($NH_3$)**

Reagent 1 was prepared by dissolving 17.5 g of phenol and 0.2 g of sodium nitroprusside in 400 mL distilled deionized water and stored to 500 mL volumes in an amber bottle to prevent direct light in the refrigerator. Reagent II was prepared by dissolving 140 g sodium citrate and 11 g sodium hydroxide in 22 mL of sodium hypochlorite and 400 mL of distilled deionized water and added into 500 mL volumes with distilled deionized water.

Stock solution A (500 µgat $NH_3$/L) was prepared by dissolving 0.3310 g of ammonium sulphate in 900 mL distilled deionized water and 1 mL chloroform and added to a volume of 1 liter and stored in a refrigerator. Stock Solution B (100 µgat $NH_3$/L) was prepared by transferring 2 mL of stock solution A to a 100 mL volumetric flask and adding to the volume with distilled water

The working standards were prepared by making serial dilutions from stock solution B as follows:

| mL of stock B in 100 mL | Concentration (µgat$NH_3$/L) |
|---|---|
| 1.0 | 1.0 |
| 2.0 | 2.0 |
| 4.0 | 4.0 |
| 6.0 | 6.0 |
| 8.0 | 8.0 |
| 10.0 | 10.0 |

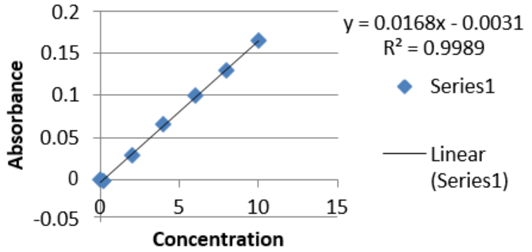



**Figure 4.** Calibration curve for $NH_3$

Ammonia was determined by measuring 35 mL of water sample in a volumetric flask, adding to reagent I, and shaking well, adding 1 mL of reagent II and then shaking well again. The reaction takes about 6 hours to complete. The absorbance reading is taken between 6 to 24 hours using a UV/vis double beam spectrophotometer at a wavelength of 630 nm. Next, 1 mL of reagents I and II were added to 35 mL of working standards and shaken well, and absorbance readings were taken at 630 nm wavelength to prepare the standard curve.

Calculation
Using the curve relationship
$Y = MX + C$
Where,
Y : the absorbance
M : the gradient of the curve
C : the Y-intercept
X : the concentration

On the making X the subject of the formula, absorbance readings were converted to concentration values
$X = Y - C/M$

*Determination of carbon levels in the creek waters*

Ex-situ primary productivity was measured using the Winkler method following APHA (1995) to determine the photosynthetic activity and the carbon assimilation per unit of time.

**The Winkler method procedure**

The creek water samples were to be analyzed (avoiding bubbles) by filling 50 ml of Winkler bottles and adding 0.4 mL $MnSO_4$ for every 50 mL sample. Then, 0.4 mL alkaline KI was added to every 50 mL water sample. The bottles were closed tight and shaken well. The precipitate formed was left to settle for half an hour. In this way, bottles could be preserved for 1-2 days. Finally, 0.2 mL of $H_2SO_4$ was added, bottles were shaken until all precipitate was dissolved, and titration was done within half an hour.

**Reactions**

$2MnSO_4 + 4NaOH \rightarrow 2Na_2SO_4 + 2Mn(OH)_2$ $Mn(OH)_2$ uses the free $O_2$ in the water to form $MnO(OH)_2$
$2Mn(OH)_2 + O_2 \rightarrow 2MnO(OH)_2$
In the presence of KI, the addition of a strong acid (H2SO4) liberates $I_2$
$2MnO(OH)_2 + 4H_2SO_4 + 4KI \rightarrow 2MnSO_4 + 2K_2SO4 + 6H_2O + 2I_2$

The amount of liberated $I_2$ is equal to the dissolved $O_2$ in the water: 2 molecules of $I_2$ are formed for each molecule of $O_2$.

Next, $I_2$ colored the solution brown, and $I_2$ was titrated with a standardized solution of $Na_2S_2O_3$
$I_2 + 2Na_2S_2O_3 \rightarrow 2NaI + Na_2S_4O_6$

Finally, the end of titration was made visible by adding a bit of starch which colored the $I_2$ blue.

**Titration**

50 mL was poured into Erlenmeyer flasks after shaking bottles. $Na_2S_2O_3$ (diluted 0.01 N solution) drop by drop till the color is light yellow. Next, with a spatula, a small amount of starch was added, which changed the color to dark blue. The drops of $Na_2S_2O_3$ were carefully added until the water became colorless.

Calculation of $O_2$ content

$O_2 mgL^{-1}$ is calculated as follows:

$$\frac{\text{Norm } Na_2S_2O_3 \times 0.25 \times 32 \times 1000 \times \text{Vol } Na_2S_2O_3 \text{ added}}{\text{Volume titrated}}$$

$$= \frac{0.1 \times 0.25 \times 32 \times 1000 \times \text{Vol. } Na_2S_2O_3 \text{ added ( mL)}}{50 \text{ mL}}$$

$$= \text{Vol. } Na_2S_2O_3 \text{ added ( mL)} \times 1.6$$

To express final results in mg $C.I^{-1}$ instead of $O_2.I^{-1}$:

$$\frac{\text{mg } O_2}{L} \times \frac{MWC}{MWO2} = \frac{\text{mg } O_2}{L} \times \frac{12}{32} = \frac{\text{mg } O_2}{L} \times 0.375$$

To convert to mL $L^{-1}$: 1 mole of a gas has a volume of 22.4 L at 1 atm and 4°C temperature

$$\frac{\text{mg } O_2}{L} \times \frac{1}{32} \times 22.4L = \frac{\text{mg}O_2}{L} \times \frac{0.7}{L} = LO_2$$

**Data analysis**

After the data were tested for normality to determine the variations between treatments, the spatial water quality patterns resulting from the different parameters from the identified sampling sites among the three creeks were analyzed quantitatively using ANOVA. Dissolved Oxygen (converted into values of carbon, mgC/L), ammonia (µgat $NH_3$), Phosphates (µgat P/L), and nitrates (µgat N/L) were determined. ANOVA was computed using the Statistical Analysis System (SAS). The rainfall data were also recorded during the entire study period. Moreover, because rainfall data for the six months did not reveal a normal distribution pattern, a correlation analysis between nutrient loads and monthly mean rainfall was performed using Spearman's correlation test. Finally, a comparison with threshold levels for various parameters was made following the criteria of standard water quality variables by Siokou-Frangou and Pangou (2000).

## RESULTS AND DISCUSSION

**Levels and variations of nitrates**

The levels of nitrates ($NO_3^-$) for Mtwapa, Kilifi, and Mida Creeks are presented in Tables 1, 2, and 3. These levels are for the period between June 2011 to November 2011 of the different sampling sites presented in the columns to the left of the table.

Along Kilifi Creek, station K02 (Kilifi Boatyard beach) had high levels of nitrates throughout the research period. In contrast, stations K05 and K06 recorded low levels in June but increased from August to November 2011. The $NO_3^-$ concentration in the two stations also positively correlated with rainfall and was found further inland from the creek mouth. That indicated that runoff water from the adjacent farms could have contributed to the increase in the $NO_3^-$ levels. The open water as the control samples (K0C) had low levels of nitrates throughout the sampling period, which also happened in the same situation as in Mtwapa Creek. However, the Mnarani Club beach area as Station K01 had a significantly high level of nitrates compared with the control in October 2011 when the monthly rainfall means was highest, suggesting that the increase in the levels of nitrates contributed by the runoff water composition (Barnes and Hughes 1988).

All sampling stations, except the open water control station (SC) along Mtwapa Creek between June 2011 and November 2011, had no significant difference in the monthly nitrate levels at $\alpha = 0.05$. The open waters as the control station (SC) had the lowest nitrates levels throughout the sampling period from June to August 2011. The results, however, reveal that in August 2011, there was an increase in nitrate levels in all the stations. This increase can be attributed to the waste discharged increase from urban development around the outer creek. All the sampling stations negatively correlated the mean rainfall amounts and nitrates. Therefore the increased levels in August cannot be attributed to runoff or stormwater from the neighborhood. These findings confirm that the amounts of runoff waters do not significantly influence the open ocean waters into the creek waters or human disturbances along the creeks (Kaiser et al. 2005). Those findings proved that agricultural and industrial pollutants and rivers or runoff water convey terrestrially derived materials loaded with sewage, while further offshore, the direct human influences are, so far, limited. Moreover, the decreased influence of human activities with distance from the coast is related to physical limitations inflicted by depth and wave height and the logistics of getting there (Kaiser et al. 2005).

**Table 1.** Levels of $NO_3^-$ (µgat N/L), Mtwapa Creek, Kenya, June to November 2011

| Site | Jun | Jul | Aug | Sep | Oct | Nov |
|---|---|---|---|---|---|---|
| S24 | $0.704^{b}$ | $0.849^{b}$ | $0.994^{b}$ | $0.820^{b}$ | $0.646^{b}$ | $0.675^{b}$ |
| S27 | $0.749^{b}$ | $0.845^{b}$ | $0.940^{b}$ | $0.856^{b}$ | $0.772^{b}$ | $0.760^{b}$ |
| S29 | $0.584^{b}$ | $0.852^{b}$ | $1.120^{b}$ | $0.932^{b}$ | $0.744^{b}$ | $0.664^{b}$ |
| S36 | $0.615^{b}$ | $0.885^{b}$ | $1.154^{b}$ | $0.882^{b}$ | $0.610^{b}$ | $0.612^{b}$ |
| S38 | $0.698^{b}$ | $0.739^{b}$ | $0.780^{ab}$ | $0.699^{b}$ | $0.618^{b}$ | $0.658^{b}$ |
| S39 | $0.553^{b}$ | $0.690^{b}$ | $0.826^{b}$ | $0.771^{b}$ | $0.715^{b}$ | $0.633^{b}$ |
| SC. | $0.244^{a}$ | $0.261^{a}$ | $0.245^{a}$ | $0.271^{a}$ | $0.255^{a}$ | $0.240^{a}$ |

Note: The Mean values in each column with the same letter are not significantly different at $\alpha = 0.05$

**Table 2.** Levels of $NO_3^-$ (µgat N/L), for Kilifi Creek, Kenya, June to November 2011

| Site | Jun | Jul | Aug | Sep | Oct | Nov |
|---|---|---|---|---|---|---|
| K01 | $0.339^{ab}$ | $0.322^{a}$ | $0.304^{c}$ | $0.517^{bc}$ | $0.729^{a}$ | $0.534^{ab}$ |
| K02 | $0.419^{a}$ | $0.410^{ab}$ | $0.402^{bc}$ | $0.561^{bc}$ | $.720^{ab}$ | $0.570^{a}$ |
| K03 | $0.353^{ab}$ | $0.444^{ab}$ | $0.535^{b}$ | $0.456^{cd}$ | $0.376^{bc}$ | $0.365^{bc}$ |
| K04 | $0.296^{bc}$ | $0.331^{bc}$ | $0.364^{bc}$ | $0.530^{bc}$ | $0.695^{ab}$ | $0.496^{ab}$ |
| K05 | $0.287^{bc}$ | $0.547^{a}$ | $0.806^{a}$ | $0.846^{a}$ | $0.886^{a}$ | $0.587^{a}$ |
| K06 | $0.233^{bc}$ | $0.519^{a}$ | $0.806^{a}$ | $0.740^{ab}$ | $0.880^{a}$ | $0.557^{a}$ |
| K0C | $0.233^{c}$ | $0.265^{c}$ | $0.265^{c}$ | $0.265^{d}$ | $0.265^{c}$ | $0.265^{c}$ |

Note: The Mean values in each column with the same letter are not significantly different at $\alpha = 0.05$

**Table 3.** Levels of $NO_3^-$ (µgat N/L) for Mida Creek, Kenya, June to November 2011

| Site | Jun | Jul | Aug | Sep | Oct | Nov |
|---|---|---|---|---|---|---|
| md1 | $0.418^{b}$ | $0.381^{b}$ | $0.344^{a}$ | $0.356^{a}$ | $0.367^{a}$ | $0.393^{b}$ |
| md2 | $2.395^{a}$ | $1.329^{a}$ | $0.262^{a}$ | $0.329^{a}$ | $0.396^{a}$ | $1.396^{a}$ |
| md3 | $0.615^{b}$ | $0.429^{b}$ | $0.242^{a}$ | $0.268^{a}$ | $0.290^{a}$ | $0.453^{b}$ |
| md4 | $1.344^{ab}$ | $0.789^{ab}$ | $0.233^{a}$ | $0.258^{a}$ | $0.282^{a}$ | $0.813^{ab}$ |
| md5 | $0.823^{ab}$ | $0.553^{ab}$ | $0.282^{a}$ | $0.348^{a}$ | $0.413^{a}$ | $0.618^{ab}$ |
| md6 | $0.720^{b}$ | $0.484^{b}$ | $0.248^{a}$ | $0.245^{a}$ | $0.242^{a}$ | $0.481^{b}$ |
| mdC | $0.256^{b}$ | $0.256^{b}$ | $0.256^{a}$ | $0.256^{a}$ | $0.256^{a}$ | $0.256^{b}$ |

Note: The Mean values in each column with the same letter are not significantly different at $\alpha = 0.05$

Along Mida Creek, the nitrates levels were significantly higher at station Md2 (Captain Andy and Hemmingway's boat yard), while Md3 and Md6 had low levels with nitrates significant difference in June and July 2011, but after that, with no significant difference in nitrate levels between August and October 2011. The nitrate levels in Mida Creek did not correlate with rainfall, suggesting that runoff water did not influence nitrate levels. In addition, being a conservation area, this creek's presence of mangrove bushes and their ability to stabilize nutrients and filter off nitrates may have minimized this effect (Hogarrth 1998).

*Levels and variations in phosphates*

The Phosphates ($PO_{4-}$) levels for Mtwapa, Kilifi, and Mida Creeks are presented in Tables 4, 5, and 6. These levels are for the period between June to November 2011 for the different sampling sites on the column to the left of the table. The measurement units are µgatoms of P per liter of the water samples.

The phosphates levels along Mtwapa Creek were highest at station S38, which shows significant differences from all the other stations, i.e., S24, S27, S29, S39, and SC, in June. Next, in July, stations S38 and S39 had significant differences in phosphate levels; finally, they were shown in station S38 in November 2011. There were no significant differences in the stations' levels between August and October. The highest phosphate levels were recorded in the outer creek stations (S38 and S39), with more urban and business developments compared to the inner creek. That confirms findings that wastewater and runoff from urban development contribute to inputs of phosphates and other pollutants into marine waters close to urban and tourist development (Bizsel and Uslu 2000). On the other hand, the inner creek stations' findings reflected the lack of major pollutant sources.

The phosphate levels also strongly correlate with mean monthly rainfall amounts for stations S27, S36, S29, and S39, which means that runoff water did not wash phosphates from land but rather human wastes near these stations. On the other hand, the open water control station did not correlate with $PO_4^-$ and rainfall. Station S38 (discharge point of sewage and wastewater from Shimo-La-Tewa GK prison) had the highest amounts of phosphates (an important ingredient of detergents) in June, indicating that wastewater was being released into the creek from time to time from the prison. However, runoff water from rainfall does not seem to have affected the levels of phosphates for all stations except S24 which is further inland along the creek with no urban developments. That suggests that runoff water from the neighboring farms washed some phosphates into the creek.

Along Kilifi Creek, phosphate levels were generally higher at stations K04 and K05. Still, they were not significantly different from the control, in which all the other stations recorded low levels throughout the research period. These stations were also considered close and adjacent to maize farms in the Maringoni area (K04) and Maringoni valley (K05). These results suggest that phosphate levels may have been influenced by farming activities. That confirms the claim that agricultural practices contribute nutrients, sediments, and biological contaminants to waterways (Stuart 2010). In addition, the strong positive correlations between phosphates and rainfall for all stations except the open ocean water, as the control indicates that phosphates from land entered the creek by rainwater.

**Table 4.** Levels of $PO_4^-$ (µgat P/L), for Mtwapa Creek, Kenya, June to November 2011

| Site | Jun | Jul | Aug | Sep | Oct | Nov |
|---|---|---|---|---|---|---|
| S24 | $0.555^{a}$ | $0.704^{a}$ | $0.852^{a}$ | $0.870^{a}$ | $0.889^{a}$ | $0.722^{a}$ |
| S27 | $1.019^{a}$ | $1.009^{abc}$ | $1.000^{a}$ | $0.935^{a}$ | $0.870^{a}$ | $0.945^{a}$ |
| S29 | $1.444^{ab}$ | $1.222^{abc}$ | $1.000^{a}$ | $0.843^{a}$ | $0.685^{a}$ | $1.065^{ab}$ |
| S36 | $0.908^{a}$ | $0.917^{a}$ | $0.926^{a}$ | $0.889^{a}$ | $0.851^{a}$ | $0.880^{a}$ |
| S38 | $2.241^{b}$ | $1.565^{c}$ | $0.889^{a}$ | $0.889^{a}$ | $0.889^{a}$ | $1.565^{b}$ |
| S39 | $1.611^{ab}$ | $1.306^{c}$ | $1.000^{a}$ | $0.907^{a}$ | $0.814^{a}$ | $1.213^{ab}$ |
| SC. | $0.770^{a}$ | $0.960^{a}$ | $0.871^{a}$ | $0.895^{a}$ | $o.888^{a}$ | $0.876^{a}$ |

Note: The Mean values in each column with the same letter are not significantly different at $\alpha = 0.05$

**Table 5.** Levels of $PO_4^-$ (µgat P/L) for Kilifi Creek, Kenya, June to November 2011

| Site | Jun | Jul | Aug | Sep | Oct | Nov |
|---|---|---|---|---|---|---|
| K01 | $0.629^{c}$ | $0.602^{e}$ | $0.574^{c}$ | $0.815^{d}$ | $1.055^{a}$ | $0.842^{c}$ |
| K02 | $0.777^{bc}$ | $0.685^{de}$ | $0.592^{c}$ | $0.843^{cd}$ | $1.092^{a}$ | $0.935^{c}$ |
| K03 | $0.555^{c}$ | $0.602^{e}$ | $0.648^{c}$ | $0.815^{d}$ | $0.981^{a}$ | $0.785^{c}$ |
| K04 | $1.240^{a}$ | $1.139^{b}$ | $1.037^{b}$ | $1.074^{bc}$ | $1.111^{a}$ | $1.176^{ab}$ |
| K05 | $1.111^{ab}$ | $1.047^{bc}$ | $0.981^{b}$ | $1.111^{b}$ | $1.240^{a}$ | $1.176^{ab}$ |
| K06 | $0.796^{bc}$ | $0.870^{cd}$ | $0.944^{b}$ | $1.157^{b}$ | $1.370^{a}$ | $1.083^{bc}$ |
| K0C | $1.469^{a}$ | $1.462^{a}$ | $1.462^{a}$ | $1.412^{a}$ | $1.370^{a}$ | $1.412^{a}$ |

Note: The Mean values in each column with the same letter are not significantly different at $\alpha = 0.05$

**Table 6.** Levels of $PO_4^-$ (µgat P/L), for Mida Creek, Kenya, June to November 2011

| Site | Jun | Jul | Aug | Sep | Oct | Nov |
|---|---|---|---|---|---|---|
| md1 | $0.685^{b}$ | $0.592^{d}$ | $0.500^{e}$ | $0.731^{c}$ | $0.963^{b}$ | $0.824^{b}$ |
| md2 | $0.852^{b}$ | $0.722^{cd}$ | $0.592^{de}$ | $0.676^{c}$ | $0.759^{b}$ | $0.805^{b}$ |
| md3 | $1.648^{a}$ | $1.296^{b}$ | $0.944^{de}$ | $1.130^{b}$ | $1.314^{ab}$ | $1.481^{a}$ |
| md4 | $0.778^{b}$ | $0.796^{cd}$ | $0.814^{bc}$ | $0.926^{bc}$ | $1.037^{ab}$ | $0.908^{b}$ |
| md5 | $0.722^{b}$ | $0.739^{cd}$ | $0.722^{cd}$ | $0.815^{bc}$ | $0.907^{b}$ | $0.815^{b}$ |
| md6 | $0.944^{b}$ | $0.806^{c}$ | $0.667^{cd}$ | $0.741^{c}$ | $0.815^{b}$ | $0.880^{b}$ |
| mdC | $1.648^{a}$ | $1.648^{a}$ | $1.648^{a}$ | $1.648^{a}$ | $1.648^{a}$ | $1.648^{a}$ |

Note: The Mean values in each column with the same letter are not significantly different at $\alpha = 0.05$

Station Md3, adjacent to the sandy beach and tourist entertainment area, generally had high but fluctuating phosphates levels. At the same time, the control station (MdC) showed high levels in the open water but was comparable with Md3 throughout the sampling period. The mangrove forest cover density was very low, and wave action was high, suggesting that land-based activities had influenced the phosphate levels.

*Levels and variations in ammonia*

The ammonia (NH3) levels for Mtwapa, Kilifi, and Mida Creeks are shown in Tables 7, 8, and 9. These levels are for the different sampling sites presented in the column to the left for the period from June to November 2011. The units of measurement are µgatoms of $NH_3$ per liter of the water samples.

Along Mtwapa Creek, Station S38 (the discharge point for sewage and wastewater from Shimo-La-Tewa G.K prison) had extremely high ammonium levels in June, July, and November 2011, while the levels were not significantly different in August, September, and October 2011.

**Table 7.** Levels of $NH_3$ (µgat $NH_3$/L) for Mtwapa Creek, Kenya, water samples, June to November 2011

| Site | Jun | Jul | Aug | Sep | Oct | Nov |
|---|---|---|---|---|---|---|
| S24 | 1.063$^{a}$ | 1.219$^{ab}$ | 1.375$^{a}$ | 1.802$^{a}$ | 2.229$^{b}$ | 1.646$^{a}$ |
| S27 | 0.584$^{a}$ | 1.146$^{ab}$ | 1.709$^{a}$ | 2.615$^{ab}$ | 3.521$^{b}$ | 2.052$^{a}$ |
| S29 | 0.688$^{a}$ | 1.000$^{a}$ | 1.313$^{a}$ | 1.844$^{a}$ | 2.375$^{b}$ | 1.532$^{a}$ |
| S36 | 0.729$^{a}$ | 1.104$^{ab}$ | 1.479$^{a}$ | 1.938$^{a}$ | 2.396$^{b}$ | 1.563$^{a}$ |
| S38 | 23.065$^{b}$ | 12.428$^{c}$ | 1.792$^{a}$ | 2.375$^{ab}$ | 2.959$^{b}$ | 13.012$^{b}$ |
| S39 | 1.771$^{a}$ | 1.532$^{ab}$ | 1.292$^{a}$ | 1.761$^{a}$ | 2.229$^{b}$ | 2.000$^{a}$ |
| SC | 1.123$^{a}$ | 1.286$^{b}$ | 1.258$^{a}$ | 1.511$^{a}$ | 1.938$^{a}$ | 1.130$^{a}$ |

Note: The Mean values in each column with the same letter are not significantly different at $\alpha = 0.05$

**Table 8.** Levels of $NH_3$ (µgat $NH_3$/L) for Kilifi Creek, Kenya, water samples, June to November 2011

| Site | Jun | Jul | Aug | Sep | Oct | Nov |
|---|---|---|---|---|---|---|
| K01 | 1.000$^{b}$ | 1.094$^{c}$ | 1.187$^{c}$ | 2.000$^{a}$ | 2.813$^{a}$ | 1.906$^{b}$ |
| K02 | 1.250$^{ab}$ | 1.417$^{abc}$ | 1.583$^{bc}$ | 2.375$^{a}$ | 3.167$^{a}$ | 3.875$^{a}$ |
| K03 | 0.979$^{b}$ | 1.146$^{bc}$ | 1.313$^{bc}$ | 1.729$^{a}$ | 2.146$^{a}$ | 1.563$^{b}$ |
| K04 | 1.021$^{b}$ | 1.427$^{abc}$ | 1.833$^{abc}$ | 2.510$^{a}$ | 3.187$^{a}$ | 2.104$^{ab}$ |
| K05 | 1.292$^{ab}$ | 1.990$^{a}$ | 2.687$^{a}$ | 2.250$^{a}$ | 1.813$^{a}$ | 1.552$^{b}$ |
| K06 | 0.917$^{b}$ | 1.510$^{abc}$ | 2.104$^{bc}$ | 1.906$^{a}$ | 1.708$^{a}$ | 1.312$^{b}$ |
| K0C | 1.833$^{a}$ | 1.094$^{c}$ | 1.833$^{abc}$ | 1.833$^{a}$ | 1.833$^{a}$ | 1.833$^{b}$ |

Note: The Mean values in each column with the same letter are not significantly different at $\alpha = 0.05$

**Table 9.** Levels of $NH_3$ (µgat $NH_3$/L) for Mida Creek, Kenya, water samples, June to November 2011

| Site | Jun | Jul | Aug | Sep | Oct | Nov |
|---|---|---|---|---|---|---|
| md1 | 1.000$^{b}$ | 2.209$^{a}$ | 2.208$^{ab}$ | 2.542$^{a}$ | 2.875$^{a}$ | 1.938$^{a}$ |
| md2 | 1.500$^{b}$ | 1.521$^{a}$ | 1.542$^{b}$ | 1.386$^{a}$ | 1.229$^{a}$ | 1.365$^{a}$ |
| md3 | 2.292$^{ab}$ | 2.135$^{a}$ | 1.979$^{b}$ | 1.844$^{a}$ | 1.709$^{a}$ | 2.000$^{a}$ |
| md4 | 3.583$^{a}$ | 2.761$^{a}$ | 1.937$^{b}$ | 1.844$^{a}$ | 1.750$^{a}$ | 2.667$^{a}$ |
| md5 | 1.021$^{b}$ | 2.156$^{a}$ | 3.291$^{a}$ | 2.594$^{a}$ | 1.896$^{a}$ | 1.458$^{a}$ |
| md6 | 3.250$^{a}$ | 2.573$^{a}$ | 1.896$^{b}$ | 1.948$^{a}$ | 2.000$^{a}$ | 2.625$^{a}$ |
| mdC | 1.542$^{b}$ | 1.542$^{a}$ | 1.542$^{b}$ | 1.542$^{a}$ | 1.542$^{a}$ | 1.542$^{a}$ |

Note: The Mean values in each column with the same letter are not significantly different at $\alpha = 0.05$

The presence of high and sporadic ammonia levels are evident that untreated sewage and wastewater were discharged into the creek from time to time from the GK prison. Station S38's ammonia levels were comparable with the other stations in August and October. Furthermore, this suggests some form of wastewater treatment during certain times within the prison activity. A high tourist season still goes on in October, which can be associated with generating comparably high amounts of ammonia-containing wastes, which could have contributed to the observed high ammonia levels in all samples except S38. The open ocean water control (SC) had consistently moderate ammonia levels comparable with the other stations throughout the sampling period, indicating insignificant levels of biological contaminants offshore.

There were spatial and temporal variations in ammonia levels along Kilifi Creek. The levels were high at station K02 (Boatyard beach) from September to November, while station K05 (Maringoni valley) showed high ammonia levels between July and September 2011. The high ammonia levels in the stations could be attributed to agricultural sources and tourism. The other stations showed no significant difference in ammonia levels due to limited nitrification or less important pollutant sources (Bizsel and Uslu, 2000). The correlation values between ammonia and rainfall were strongly positive for stations K01, K3, K04, and K02. There was no correlation between rainfall and $NH_3$ for station K06 and a slightly positive correlation at the open water control station.

Stations Md3, Md4, and Md6 along Mida Creek had a significantly high ammonia level between the June and July sampling period. These stations are adjacent to the Mida beach, Sudi Island picnic site, and Safari Blue beach, respectively. Therefore, these high levels could be attributed to the tourist activities along these beaches, except for station Md1, all the other stations negatively correlated with rainfall. Moreover, the ammonia levels for station Md1 (Temple point shoreline) and Md5 (Shoreline opposite Captain Andy) were initially quite low. Then, from July, the levels arose and became comparable with stations Md3, Md4, Md6, and the control. That was a low season for tourist activities, suggesting insignificant levels of waste discharged into the creek.

## Effect of nutrient loads on primary productivity

The primary productivity results for Mtwapa, Kilifi, and Mida Creeks calculated as a difference between photosynthesis and respiration using the Winkler method are shown in Tables 10, 11, and 12. The measurement units are milligrams of carbon per liter of water sample per hour. These results are for the period from June to November 2011.

The carbon assimilation levels along Mtwapa Creek were slightly high at stations S39, S38, and S36 around the sampling period, while lowest in the control sample. Station S29 also recorded low carbon values from June to November 2011. These results supposed more phytoplanktonic activities within the outer creek near Mtwapa town than the inner creek. In addition, these sites' levels of nutrients (phosphates) showed a strong positive correlation with carbon. They were generally higher in all these three stations than in the inner stations, although the other nutrients showed a negative correlation with carbon. These findings reveal that the higher fertility levels of the outer creek waters influenced the rate of photosynthetic activity among the phytoplanktonic cells in addition to the decomposition and decay of these cells (Kaiser et al. 2005).

On the other hand, the open ocean waters revealed close to zero correlation between carbon and nutrients. That could be attributed to either dilution of the creek water or low biological activity as it mixes with the water from the open ocean. However, other chemical and physical properties of the water, including dissolved oxygen and turbidity, influence the photosynthesis rate.

Carbon assimilation levels for K01 and K02 stations were highly significant from June to August 2011, and there was no significant difference from September to November. The correlation coefficient at station K01 was positively strong with ammonia, whereas the correlation coefficient was almost zero at K02. These results confirm that higher rates of photosynthetic activity (Boat yard and Mnarani club beaches) are associated with higher nitrates and phosphates levels between the two months (Tew et al. 2006). However, the average does not show a clear relationship between nitrates and phosphates for the entire period. On the other hand, throughout the sampling period, ammonia maintained a clear positive correlation. The other stations did not record significantly different amounts. Moreover, throughout the research period, due to the lack of major pollutant sources and biological processes in the outer ocean, the control had consistently low carbon assimilation rates (Bizsel and Uslu 2000).

**Table 10.** Rate of carbon assimilation (mgC/L), levels for Mtwapa Creek, Kenya, water samples, June to November 2011

| Site | Jun | Jul | Aug | Sep | Oct | Nov |
|---|---|---|---|---|---|---|
| S24 | $0.367^{ab}$ | $0.243^{abc}$ | $0.120^{ab}$ | $0.860^{d}$ | $1.600^{c}$ | $0.983^{c}$ |
| S27 | $0.360^{ab}$ | $0.230^{a}$ | $0.103^{a}$ | $0.230^{b}$ | $0.360^{b}$ | $0.360^{b}$ |
| S29 | $0.136^{a}$ | $0.140^{a}$ | $0.150^{b}$ | $0.150^{a}$ | $0.150^{a}$ | $0.140^{a}$ |
| S36 | $0.330^{ab}$ | $0.330^{bc}$ | $0.330^{c}$ | $0.323^{c}$ | $0.320^{b}$ | $0.323^{b}$ |
| S38 | $0.340^{ab}$ | $0.353^{bc}$ | $0.370^{c}$ | $0.383^{c}$ | $0.400^{b}$ | $0.370^{b}$ |
| S39 | $0.450^{b}$ | $0.393^{c}$ | $0.343^{c}$ | $0.343^{c}$ | $0.343^{b}$ | $0.393^{b}$ |
| SC. | $0.137^{ab}$ | $0.130^{a}$ | $0.170^{a}$ | $0.132^{a}$ | $0.140^{a}$ | $0.141^{a}$ |

Note: The Mean values in each column with the same letter are not significantly different at α = 0.05

**Table 11.** Rate of carbon assimilation (mg C/L), levels for Kilifi Creek, Kenya, water samples

| Site | Jun | Jul | Aug | Sep | Oct | Nov |
|---|---|---|---|---|---|---|
| K01 | $0.340^{b}$ | $0.286^{a}$ | $0.240^{a}$ | $0.203^{a}$ | $0.170^{b}$ | $0.253^{b}$ |
| K02 | $0.550^{a}$ | $0.326^{a}$ | $0.110^{cd}$ | $0.123^{b}$ | $0.140^{bc}$ | $0.340^{a}$ |
| K03 | $0.083^{c}$ | $0.110^{c}$ | $0.140^{bc}$ | $0.190^{a}$ | $0.240^{a}$ | $0.160^{c}$ |
| K04 | $0.270^{b}$ | $0.190^{b}$ | $0.110^{cd}$ | $0.180^{a}$ | $0.260^{a}$ | $0.260^{b}$ |
| K05 | $0.100^{c}$ | $0.123^{c}$ | $0.150^{b}$ | $0.150^{b}$ | $0.150^{bc}$ | $0.123^{c}$ |
| K06 | $0.130^{c}$ | $0.110^{c}$ | $0.100^{d}$ | $0.130^{b}$ | $0.160^{bc}$ | $0.140^{c}$ |
| K0C | $0.130^{c}$ | $0.123^{c}$ | $0.130^{bcd}$ | $0.126^{b}$ | $0.136^{c}$ | $0.126^{c}$ |

Note: The Mean values in each column with the same letter are not significantly different at α = 0.05

**Table 12.** Rate of carbon assimilation (mg C/L), levels for Mida Creek, Kenya, water samples, June to November 2011

| Site | Jun | Jul | Aug | Sep | Oct | Nov |
|---|---|---|---|---|---|---|
| md1 | $0.026^{c}$ | $0.120^{c}$ | $0.220^{c}$ | $0.320^{a}$ | $0.420^{a}$ | $0.220^{a}$ |
| md2 | $0.016^{c}$ | $0.086^{d}$ | $0.160^{d}$ | $0.196^{c}$ | $0.240^{c}$ | $0.126^{de}$ |
| md3 | $0.080^{c}$ | $0.080^{d}$ | $0.080^{e}$ | $0.100^{d}$ | $0.120^{d}$ | $0.100^{e}$ |
| md4 | $0.026^{c}$ | $0.156^{b}$ | $0.290^{b}$ | $0.276^{b}$ | $0.273^{bc}$ | $0.146^{cd}$ |
| md5 | $0.050^{bc}$ | $0.196^{a}$ | $0.350^{a}$ | $0.323^{a}$ | $0.300^{b}$ | $0.170^{bc}$ |
| md6 | $0.090^{a}$ | $0.203^{a}$ | $0.320^{ab}$ | $0.306^{ab}$ | $0.300^{b}$ | $0.193^{ab}$ |
| mdC | $0.033^{c}$ | $0.032^{e}$ | $0.033^{f}$ | $0.032^{e}$ | $0.033^{e}$ | $0.033^{f}$ |

Note: The Mean values in each column with the same letter are not significantly different at α = 0.05

Along Mida Creek, station Md1 (Temple point) had highly significant levels of carbon assimilation from August to November 2011. Station Md5 recorded highly relative levels of carbon fixation between July and September, which relates to the higher levels of ammonia and phosphate, which also showed a positive correlation coefficient during the same period. These results can be attributed to high input rates of nitrogenous organic wastes from anthropogenic sources. The control station recorded low levels of carbon assimilation throughout the research period. The correlation coefficient between carbon and ammonia and carbon and nitrates was negative, while between carbon and phosphates, there was a slightly positive correlation.

*Correlation between mean nutrient levels and carbon assimilation*

The correlation coefficients were calculated using Spearman's correlation analysis, and the correlation relationships for the three creek creeks are shown in Tables 13, 14, and 15.

Along Mtwapa Creek, the element which mostly limits primary production in the ocean, Nitrate nitrogen, showed a negative correlation in all stations except S29 and S36. Although Nitrites and Ammonium ions can be taken up, nitrates are algae's primary source of nitrogen utilized (Kaiser et al. 2005).

Phosphates along Mtwapa Creek showed unclear trends concerning productivity and nutrient levels. The correlations varied from negative at stations S29 and S38 to positive at S39 and S36. Phosphorous is the second limiting nutrient in marine systems and happens in several forms. It is also present in various organic compounds, which can be broken down by enzymes located in many algal species' membranes. Within the outer stations, including S36, S38, and S39, ammonia negatively correlated with carbon assimilation. In the oligotrophic deep ocean sites, the values are normally very low, below 0.1 $\mu molL^{-1}$, due to be little input of fresh nitrates into the upper mixed layer from below. However, coastal upwelling in coastal sites can result in high input of nitrates.

Spearman's correlation analysis was used to manage the correlation between nutrients and carbon. Nitrate nitrogen showed a negatively strong correlation within the outer creek station of Mnarani Club beach (K01) and a negatively moderate correlation along the Boat Yard beach

(K02). However, the inner creek stations showed positive correlations, particularly strong near the agricultural area of Maringoni valley. The outer ocean control station revealed a near-zero correlation.

The correlation between phosphates and carbon along Kilifi Creek was closely related to nitrates, with a negatively strong relationship within the outer creek stations and a positively strong relationship within the inner creeks. The correlation results for nitrates and phosphorous indicate that land-based activities within the inner creek could have contributed to the high levels of carbon fixation coupled with the other inner creek waters' physical and chemical characteristics. In addition, other characteristics, such as turbidity and dissolved oxygen, affect photosynthetic activity.

The correlation between carbon and ammonia showed a strongly positive trend in both the outer and inner creek waters. Only two stations where the correlations were slightly negative, at K02 and K06. That reveals if other factors were to be assumed constant, that ammonia nitrogen in all stations played a role in the photosynthetic activities of phytoplanktons.

**Table 13.** Correlation r values between mean levels of nutrients and carbon assimilation for Mtwapa Creek, Kenya, June to November 2011

| Station | $NO_3^-$ | $PO_4^-$ | $NH_3$ |
|---|---|---|---|
| S24 | -0.742* | 0.451 | 0.914* |
| S27 | -0.994* | -0.393 | 0.1337 |
| S29 | 0.7105* | -0.886* | 0.761* |
| S36 | 0.495 | 0.947* | -0.831* |
| S38 | -0.506 | -0.857* | -0.831* |
| S39 | -0.624 | 0.869* | -0.295 |
| SC | -0.517 | -0.186 | -0.141 |

Note: R values with * are significant

**Table 14.** Correlation r values between mean levels of nutrients and carbon assimilation, Kilifi Creek, Kenya, June to November 2011

| Station | $NO_3^-$ | $PO_4^-$ | $NH_3$ |
|---|---|---|---|
| K01 | -0.763* | -0.743* | 0.86* |
| K02 | -0.4033 | -0.088 | -0.269 |
| K03 | -0.0367 | 0.983* | 0.996* |
| K04 | 0.2915 | 0.817* | 0.0759 |
| K05 | 0.9912* | 0.0028 | 0.783* |
| K06 | 0.143 | 0.736* | -0.289 |
| K0C | -0.161 | -0.477 | 0.586 |

Note: R values with * are significant

**Table 15.** Correlation r values between mean levels of nutrients and carbon assimilation, Mida Creek, Kenya, June to November 2011

| Station | $NO_3^-$ | $PO_4^-$ | $NH_3$ |
|---|---|---|---|
| Md1 | -0.665 | 0.601 | 0.897* |
| Md2 | -0.913* | -0.498 | -0.689 |
| Md3 | -0.393 | 0.0298 | -0.844* |
| Md4 | -0.999* | 0.558 | -0.988* |
| Md5 | -0.996* | 0.356 | 0.878* |
| Md6 | -0.995* | -0.885* | -0.999* |
| MdC | -0.433 | 0.305 | -0.542 |

Note: R values with * are significant

Along Mida Creek, Nitrates correlation analysis revealed a completely different trend using the same analysis model. There was a negatively strong correlation between nitrate nitrogen and carbon in all sampling stations except the outer ocean control and Md3, which had a positively slight correlation throughout the sampling period. These results indicate that the carbon assimilation rate was influenced by other marine water chemicals, nutrient amounts, and biological and physical factors along Mida Creek, which might have had a strong influence.

Ammonia and phosphates showed a positively strong correlation at the Temple Point shoreline (Md1) and the sampling station opposite captain Andy (Md5). The rest of the stations revealed a negative correlation between ammonia and phosphates. However, Sudi Island (Md4) positively correlated with phosphates and carbon.

*Correlation between mean monthly rainfall amounts and nutrient loads*

Correlation between mean monthly rainfall amounts and nutrient loads are shown in Tables 16, 17, and 18 for the three creek creeks. The tables show a negative correlation between rainfall and nitrate levels for stations S24, S27, S29, S36, and S38. There is almost no correlation between rainfall and nitrates in the open water control station and station S39.

The phosphate levels also have a strong correlation with mean monthly rainfall amounts for stations S27, S36, S29, and S39. Station S24, however, phosphate still shows some positive correlation with rainfall. On the other hand, the open water control station does not correlate with $PO_4^-$ and rainfall. Moreover, for all stations except S38, the ammonia level has a strong correlation with the monthly mean rainfall amounts. Only station S38 had a negative correlation between $NH_3$ and rainfall.

Along Kilifi Creek, there was a positively strong correlation between $NO_3^-$ and rainfall for stations K01, K02, and K04. Correlation values were low for stations K05 and K06. K0C had almost zero correlation, and station K03 recorded a negative correlation value. There was a positively strong correlation between rainfall and $PO_4^-$ for stations K01, K02, K03, K05, and K06. Conversely, the open water station control had a negatively strong correlation between rainfall and $PO_4^-$ while station K04 had almost no correlation.

Correlation values between rainfall and ammonia were positively strong for stations K01, K3, K04, and K02. Conversely, there was a slight positive correlation at the open water control station and no correlation between rainfall and $NH_3$ for station K06. Moreover, for all the stations, the correlation values between nitrates and rainfall were all close to zero to slightly positive.

There was a positively strong correlation between rainfall and $PO_4^-$ at stations Md1, Md4 and Md5. There was a slightly positive correlation between the rainfall and $PO_4^-$ for stations Md2, Md3, and Md6. A weak negative correlation showed on the open water station control, implying that runoff water contributed to increased phosphates in the creek water.

**Table 16.** Correlation r values between rainfall and $NO_3^-$, $PO_4^-$ and $NH_3$, Mtwapa Creek, Kenya, June to November 2011

| Station | $NO_3^-$ | $PO_4^-$ | $NH_3$ |
|---|---|---|---|
| S24 | -0.585 | 0.45 | 0.81* |
| S27 | -0.399 | -0.843* | 0.768* |
| S29 | -0.273 | -0.66 | 0.773* |
| S36 | -0.486 | -0.85* | 0.736* |
| S38 | -0.768* | -0.374 | -0.336 |
| S39 | 0.032 | -0.537 | 0.766* |
| SC | 0.0743 | 0.004 | 0.877* |

Note: R values with * are significant

**Table 17.** Correlation r values between rainfall and $NO_3^-$, $PO_4^-$ and $NH_3$, Kilifi Creek, Kenya, June to November 2011

| Station | $NO_3^-$ | $PO_4^-$ | $NH_3$ |
|---|---|---|---|
| K01 | 0.863* | 0.865* | 0.83* |
| K02 | 0.86* | 0.85* | 0.549 |
| K03 | -0.48 | 0.79* | 0.778* |
| K04 | 0.818* | 0.018 | 0.736* |
| K05 | 0.392 | 0.814* | -0.27 |
| K06 | 0.44 | 0.789* | 0.01 |
| K0C | 0.152 | -0.813* | 0.268 |

Note: R values with * are significant

**Table 18.** Correlation r values between rainfall and $NO_3^-$, $PO_4^-$ and $NH_3$, Mida Creek, Kenya, June to November 2011

| Station | $NO_3^-$ | $PO_4^-$ | $NH_3$ |
|---|---|---|---|
| Md1 | 0.0315 | 0.784* | -0.474 |
| Md2 | 0.167 | 0.32 | -0.787* |
| Md3 | -0.12 | 0.22 | -0.54 |
| Md4 | -0.18 | 0.735* | -0.28 |
| Md5 | -0.027 | 0.789* | -0.299 |
| Md6 | -0.219 | 0.23 | -0.157 |
| MdC | 0.069 | -0.33 | 0.14 |

Note: R values with * are significant

The study found that primary productivity along Mtwapa Creek was higher within the outer creek, and the levels exposed a positive correlation (S39 and S36 had 0.869 and 0.947, respectively) with phosphates. However, there was a negative correlation between these rainfall amounts and nutrient levels. Therefore, this suggests that this fertility level resulted from land-based human developments, including businesses and tourist hotels close to the outer creek stations. The roadside drainage channels from the neighboring Mtwapa town could be the most likely source of these phosphates due to wastewater collected from business establishments. The null hypothesis is therefore rejected. The inland station positively correlated with rainfall, suggesting that runoff water from the neighboring cleared farm fields was the most likely source of the nutrients. The control station, which was marine water samples from the open ocean, had a very close to zero correlation coefficient between nutrients and rainfall, implying there was very little influence on its water fertility from land-based anthropogenic activities. The null hypothesis is therefore rejected.

Along Kilifi Creek, at the Sea Horse shoreline (0.983), Maringoni valley (0.817), and Maringoni area (0.736), the correlation coefficient results were positive between carbon and the three nutrients. The correlation coefficient between the nutrients and rainfall was also positive among all the creek stations, indicating the most likely source of these nutrients in the creek could be runoff water from the surrounding area. The most important human activity, allowing the nutrients from land to enter the creek directly, maybe the opening up of vegetation, including mangrove destruction within areas close to the creek.

In Mida Creek, in five sampling stations, the analysis results show a positive relationship between rainfall and nutrient levels and slightly close to zero at the outer ocean control station. However, only phosphates showed a positive correlation (0.601, 0.498, 0.0296, 0.558, 0.356, 0.885, and 0.305) with carbon assimilation. Phosphates and ammonia had a positive relation (0.601 and 0.897, respectively) with carbon at the Temple point shoreline. These results suggest that some level of land-based nutrients are washed into the creek, although Mida Creek is a conservation area under the strict rules of Kenya Wildlife Service (KWS) and the presence of the Marine Park. The highest nutrient levels, especially phosphates, recorded at the Temple Point Hotel suggest that some contaminated wastewater from this establishment could have contributed to the increased levels. The monthly averages in nutrient levels in all three creeks fluctuated throughout the study period.

Mtwapa Creek consistently recorded higher levels of nitrates within the oligotrophic level throughout the period, even within acceptable levels (Siokou-Frangou and Pangou 2000). The nutrient level along Kilifi and Mida Creeks kept fluctuating and did not give a clear, consistent trend throughout the study period. The outer ocean control for nitrate in Mtwapa, Kilifi, and Mida were 0.00024, 0.000256, and 0.00023 mg/L, respectively. The Kilifi, Mida Creeks, and the controls nitrate levels were all less than 0.00067, which is within the oligotrophic levels of less than 0.0087 mg/L. Therefore, in all three creeks, the phosphate levels were between the higher oligotrophic ranges of 0.002 mg/L. According to the trophic classification scheme, these levels are within acceptable limits.

The higher mesotrophic level sampling was station S38 (Shimo-La-Tewa GK Prison), while all the other stations were in the oligotrophic levels below 0.008 mg/L; those levels are above acceptable limits according to the trophic classification (Siokou-Frangou and Pangou. 2000). However, the ammonia levels along Kilifi and Mida Creeks were within the oligotrophic range of less than 0.008 mg/L. Furthermore, for the entire study period, the distance of the sampling sites from the identified sources of nutrients was fixed at 10 meters from the shoreline along the creek. Changing the distance of the sampling sites from the potential sources of nutrients would have allowed a proximity analysis, making the results more valid.

## REFERENCES

American Public Health Association (APHA). 1995. Standard Methods for Examination of Water and Waste Water, 18th Ed. APHA, Washington DC.

Barnes RSK, Hughes RN. 1988. An Introduction to Marine Ecology. 2nd Edition. Blackwell science Ltd Feldgasse 13, A-1238, Wein, Austria.

Bayan IE, Yulianda F, Setyobudiandi I. 2016. Degradation analysis of mangrove ecological function as macrozoobenthos habitat and its management in the Angke Kapuk Coastal Area, Jakarta. Bonorowo Wetlands 6: 1-11. DOI: 10.13057/bonorowo/w060101.

Bizsel N, Uslu O. 2000. Phosphates, nitrigen and iron enrichment in the polluted Izmir Bay, Aegean Sea. Mar Chem J 71: 317-324.

Hogarrth PJ. 1998. The Biology of Mangroves. In: Crawley MJ, Little C, Southwood TRE, Ulfstrand S (eds). Oxford Ox2 6DP, Great Clarendon Street, New York.

Kaiser MJ, Attril MJ, Jennings S, Thomas ND, Barnes DKA, Brierly SA, Polunin NVC, Raffaelli DG, Williams PJ. 2005. Marine Ecology: Processes, Systems and Impacts. Oxford University press Inc., New York.

KEMFRI. 2013. Characterization of nutrient enrichment in the estuaries and related systems in Kenya coast. Environ Resour J 2 (6): 181-190.

Okuku EO, Ohowa B, Mwangi SN, Munga D, Kiteresi LL, Wanjeri VO, Okumu S, Kilonzo J. 2010. Sewage pollution in coastal waters of Mombasa City: A norm Rather than an exception. Environ Resour J 5 (4): 865-874.

SCECAP. 2010. Estuarine Condition Monitoring and Assessments. Marine Resources Division, South Carolina. Department of Natural Resources. 217Fort Johnson Road, Chaarleston SC29412.

Siokou-Frango I, Pagou K. 2000. Assessment of the Trophic Conditions and Ecological Status in the Inner Saronikos Gulf. Technical Report for the Ministry ofEnvironment, Planning and Public Works, NCMR, Athens.

Stuart D. 2010. Coastal ecosystems and agricultural landuse: New challenges on California's central coast. Coast Manag 38 (1): 42-64. DOI: 10.1080/08920750903363190.

Tew HS, Meng P-J, Glover DC, Wang J-T, Leu M-Y, Chen C-C. 2006. Characterising and predicting algal blooms in a subtropical coastal lagoon. Mar Freshw Res 5 (4): 276-285.

UNEP/Nairobi Convention Secretariat 2009. Strategic Action Programme for the Protection of the Coastal and Marine Environment of the Western Indian Ocean from Land-basedSources and Activities, Nairobi, Kenya.

United States Environmental Protection Agency (USEPA). 2000. Water Quality Standards for Coastal and Great Lakes Recreation Waters; Final Rule (40 CFR part 131). www.epa.gov/EPA-WATER/2004 (accessed August 2010)

Webb MTR, Gomez-Gomez F. 2009. Synoptic Survey of Water Quality and Bottom Sediment, San Juan Bay Estuary System, Puerto Rico. Water Resources Investigation Report 97-4144. DOI: 10.3133/wri974144.

**OCEAN LIFE**
Volume 3, Number 2, December 2020
Pages: 94-101

E-ISSN: 2580-4529
DOI: 10.13057/oceanlife/o010205

# Concentration of heavy metal in sediment, water, and fish from Ankobra and Tano River Estuaries, Ghana

**GILBERT KOFI AWUAH, JOHN ARKO MENSAH, JULIUS N. FOBIL♥**
Department of Biological, Environmental & Occupational Health Sciences, University of Ghana. Legon, Accra, Ghana. Tel: +233-302213850, ♥email: jfobil@ug.edu.gh

Manuscript received: 21 October 2020. Revision accepted: 30 December 2020.

**Abstract.** *Awuah GK, Mensah JA, Fobil JN. 2020. Concentration of heavy metal in sediment, water, and fish from Ankobra and Tano River Estuaries, Ghana. Ocean Life 3: 94-101.* The heavy metals arsenic (As), mercury (Hg), and lead (Pb) are all poisonous and found in nature. The increase in their concentrations, however, is a result of human activities that pose a danger to both aquatic life and humans. The researchers set out to measure the levels of arsenic (As), hexavalent chromium (Hg), and lead (Pb) in sediment, water, and fish from the Ankobra and Tano Rivers in Ghana's Western Region. That was thought to be heavily polluted due to human activity, particularly artisanal gold mining. To assess the prevalence of As, Hg, and Pb in fish and the aquatic media (water and sediment) in the Tano of Jaway Wharf - Ellenda and Ankobra River of Sanwoma basins, an analytical cross-sectional study was carried out. The Atomic Absorption Spectrophotometer (AAS) Pinnacle 900T was used to determine the levels of As, Hg, and Pb in the tissues (muscles) of "Ekpoke" *Ctenopoma kingsleyae,* "Adwene" *Clarias gariepinus,* "Nzerma" *Pomadasys jubelini,* "Bile" *Parachanna obscura,* "Ekpoke" *Sarotherodon galilaues,* "Ekpoke" *Sarotherodon melanotheron,* and "Senzeke" *Parapenaeus longirostris* (n=53), water samples (n=30), and sediment samples (n=25) (Perkin Elmer, USA). STATA version 13 and the Excel spreadsheet from Microsoft version 10 were used for the statistical Analysis (StataCorps LP, Chicago, USA). Hg, As, and Pb levels were the highest in all the analyzed locations. In general, Hg (2.09 ± 1.29 mg/kg) and As (2.80 ± 1.52 mg/kg) were found in the highest amounts in fish, next by sediment (1.40 ± 1.78 mg/kg) and (0.61 ± 0.40 mg/kg), and water (0.09 ± 0.36 mg/L) and (0.06 ± 0.05 mg/L), respectively. Sediment (1.46 ± 4.26 mg/kg), water (0.14 ± 0.10 mg/L), and fish (0.11 ± 0.11 mg/kg) all had higher Pb concentrations than the other two. The total heavy metal variability in soil, water, and fish was not different (p=0.828, 0.570, and 0.978, respectively). Concentrations of As, Hg, and Pb in river water, sediment, and fish did not differ significantly between the Ankobra and Tano Rivers. Tano and Ankobra have As and Hg concentrations higher than those considered safe by the World Health Organization (WHO) for use in freshwater habitats.

**Keywords:** Arsenic, fish, lead, mercury, water

## INTRODUCTION

There is widespread awareness about Heavy Metal (HM) exposure. It is the most pervasive environmental threat the mining industry poses worldwide. Water pollution endangers aquatic ecosystems, threatens food supplies, and harms human health (Akan et al. 2012; Ida 2012; Taylor 2012; Abualtayef et al. 2014; Retnaningdyah et al. 2014; Vikas and Dwarakish 2015; Krupnova et al. 2018; Laibu et al. 2018; Primadiani et al. 2018). Weathering of rocks and the subsequent conveyance of surface runoff and wind could naturally release heavy metals into aquatic systems. However, contamination of the environment, notably the aquatic environment, has grown due to human activities. The chemical composition of bedrock has been unlocked into water habitats due to the excavation of soil and minerals and the removal of enormous swaths of land for mining operations (Wuana and Okieimen 2011). Mining operations, such as rock crushing, may discharge heavy metals inside rocks into the surrounding environment. For instance, mining for gold (Au) often necessitates using poisonous chemicals like cyanide and mercury, which may wreak havoc on the ecosystem. In addition, the mining industry faces a significant environmental concern in the form of the waste produced after extraction (often referred to as tailing damp), which contains these harmful compounds and affects most countries and all mining sectors globally (Cobbina et al. 2015). Heavy metals from tailing damps are sometimes released into aquatic systems due to dike failure or the release of excess water from the tailing damp, both of which can happen accidentally or on purpose (Cobbina et al. 2015).

For people living in rural poverty, especially in Africa, Asia, and South America, rivers and streams are sometimes the only reliable supplies of drinkable water and household needs (Rijsberman 2006). However, introducing extra dissolved anions and cations causes water chemistry shifts, negatively impacting aquatic ecosystems. If aquatic organisms cannot endure the conditions or adapt physiologically, they will either be relocated or exterminated. Disruption of trophic levels and the consequent failure of aquatic ecosystems are the results. Because of this, communities dependent on the aquatic ecology within mining areas face food insecurity and health issues that can lead to malnutrition and extreme poverty (Falkenmark and Lundqvist 1998).

Mining is a major contributor to the seven-fold rise in heavy metal concentrations in the environment that has resulted from a confluence of human activity and climatic shifts. For example, researchers in Ghana have linked the continuous release of mercury (Hg), lead (Pb), arsenic (As), and cadmium (Cd), among other harmful and toxic metals, into aquatic habitats with gold mining activities (mineral exploitation, disposal of tailings and waste waters, ore transportation, smelting, refining) (Cobbina et al. 2015).

Increases in urbanization, agricultural runoff carrying fertilizers and pesticides, leaching from landfills, industrial and sewage effluents, and other human activities are all potential sources of heavy metal pollution of water and sediments (Akan et al. 2012; Sulardiono et al. 2018; Mwatsahu et al. 2020). Due to their inert nature, heavy metals remain in the aquatic environment indefinitely. They build up to toxic levels in the tissues of sedentary creatures like clams and are primarily found in sediments or are digested by microbes (Length 2011). Heavy metal accumulations in aquatic creatures often rise from lower to higher trophic levels. Proteins, minerals, vitamins, and omega-3 fatty oils can all be found in fish, making it an important part of a healthy diet for illness prevention and general well-being (Basim and Khoshnood 2013). Yet, fish may contribute to heavy metal pollution because of their natural bioaccumulation potential. Fish can accumulate heavy metals in their liver, gills, and bones, where they can remain for extended periods before being passed up the food chain to other animals, including humans (Asante et al. 2014).

Fish heavy metal contamination is related to pollution, diet, and sediments in the water. Because of their dual nature as pollutant "carriers" and "sinks," sediments help shape heavy metal contamination patterns in aquatic systems (Li 2014). However, the accumulation in fish varies on the mode of uptake, the kind of heavy metal, and the fish species. Therefore, due to these factors, fish is sometimes employed as a biomonitor to evaluate the environment's bioaccumulation and biomagnification of heavy metals (Begum et al. 2009).

Heavy metal pollution in fish increases the risk for various acute and chronic health problems (Anim-Gyampo et al. 2013). For example, evidence is mounting that toxicants including mercury (Hg), lead (Pb), arsenic (As), and cadmium (Cd) cause cognitive impairments, especially in children, as well as various cancers (with a disproportionately high rate of incidence in the upper gastrointestinal tract). In addition, toxicants could also weaken immune systems, malnutrition-related disabilities, fetal development retardation, kidney issues, reproductive disorders, skin lesions, vascular diseases, impaired psychological faculties, and endocrinological abnormalities (Cobbina et al. 2015).

While items like paints and agrochemicals are widely used and are known to contain heavy metals, regulations and control on the use of heavy metals are not adequately implemented in most developing nations. Despite being a member of the "International Convention on the Control of Trans border Movements of Hazardous Waste and their Disposal," or the "Basic Convention," Ghana has not yet enacted strict legislation governing the usage and disposal of hazardous waste (Dogbevi 2009). There are rules in place to restrict the use of trace metals, such as the Food and Drug Law (PNDC Law 305 B). However, they have not been properly applied. The Environmental Protection Agency (EPA) of Ghana established maximum contaminant levels (MCLs) for water sources as follows: 0.1 mg/mL for lead, ten micrograms per milliliter (mg/L) for zinc, ten micrograms per milliliter (mg/L) for iron, and 0.1 mg/mL for cadmium (Bannerman et al. 2003). Consequently, the purpose of this study was to evaluate the levels of arsenic, mercury, and lead in sediment, water, and fish from the Ankobra and Tano Rivers in the Western Region of Ghana. Those study locations where human activities, particularly artisanal gold mining, are thought to have significantly polluted the drainage systems.

## MATERIALS AND METHODS

### Study area

The research used two locations in the Ellembelle and Jomoro Districts of Western Ghana (Sanwoma and Jaway Wharf-Ellenda) (Figure 1).

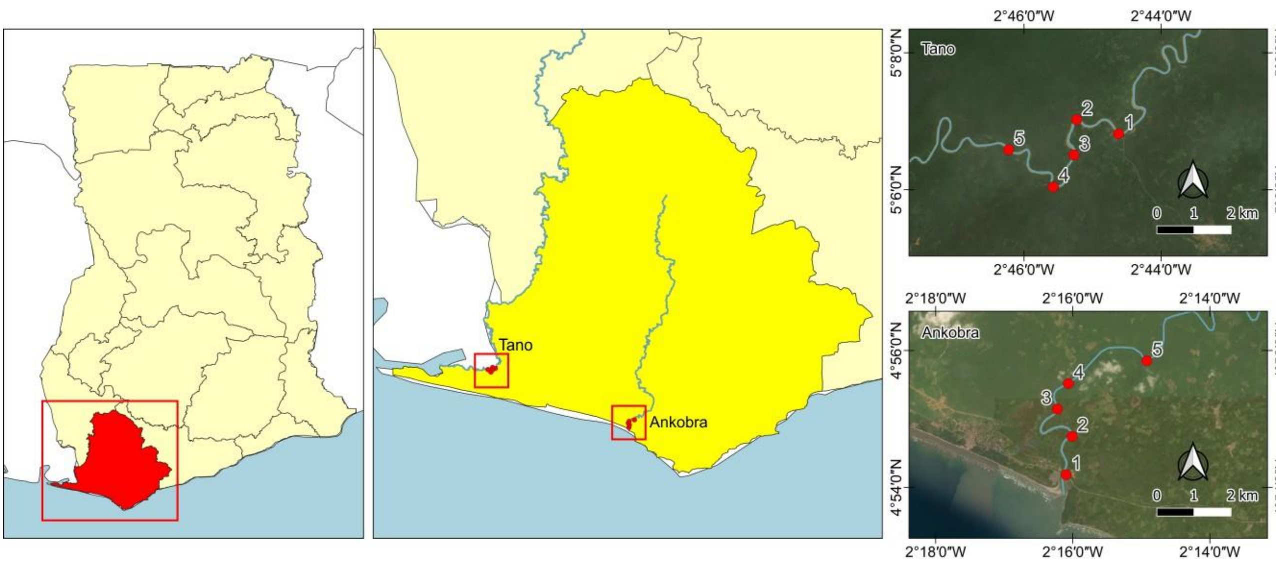



**Figure 1.** A map showing Tano and Ankobra Rivers Basins, Ghana

**Figure 2.** Ankobra River at Sanwoma (*left*) and Tano River at Ellenda Jaway (*right*), Ghana

**Table 1.** GPS coordinates

| Point | GPS Coordinates | | | |
|---|---|---|---|---|
| | Ankobra | | Tano | |
| | Latitude | Longitude | Latitude | Longitude |
| 1 | 4° 54' 11.14" N | 2° 16' 5.91" W | 5° 6' 49.08" N | 2° 44' 37.24" W |
| 2 | 4° 54' 44.71" N | 2° 16' 0.58" W | 5° 7' 01.41" N | 2° 45' 14.02" W |
| 3 | 4° 55' 8.72" N | 2° 16' 13.58" W | 5° 6' 30.43" N | 2° 45' 16.30" W |
| 4 | 4° 55' 31.00" N | 2° 16' 3.83" W | 5° 6' 2.56" N | 2° 45' 34.40" W |
| 5 | 4° 55' 51.00" N | 2° 14' 55.19" W | 5° 6' 35.01" N | 2° 46' 13.55" W |

*Sanwoma*

The Ankobra River drains and forms the eastern border of Sanwoma. The Gulf of Guinea forms its southern border, while Asanta and Nkroful form its western and northern ones. As an estuary of the River Ankobra, fishing has long been the main source of income for the region. Additionally, there is some farming, albeit on a much lesser scale. The study region along the Ankobra River ran from 4° 54' 11.14" N, 2° 16' 5.91" W, to 4° 55' 51.00" N, 2° 14' 55.19" W in geographical coordinates.

*Jaway Wharf-Ellenda*

Jaway Wharf - Ellenda is a border town in the Tano River basin in the Jomoro District. People from around Ghana and Ivory Coast come here to trade items in the bustling markets. The Nzema communities of the Western area and other regions of the country rely heavily on tilapia caught in the Tano River, a key source of income for many locals. Tano River region coordinates ranged from 5°6’49.08” N, 2°44’37.24”W to 5°6’35.01”N, 2°46’13.55” W.

## Variables of interest

Concentrations of heavy metals in fish, water, and sediments were the independent variables of interest.

## Fish sample collection

Both freshwater and dark-water fish species were represented in the collection. Fifty-three (53) fish samples were gathered overnight in a trap set by local fishermen and analyzed for arsenic (As), mercury (Hg), and lead (Pb).

In the Ankobra and Tano Rivers, sediments were taken from five different spots (Figure 2). Fifteen sediment samples were taken from various points along the Ankobra River, with three replicates taken from each location (Figure 2). Additionally, for ten samples from the Tano River, two replicate sediment samples were obtained from each of the five sites or points across the river (Figure 2). Totally twenty-five (25) sediment samples were taken from the two rivers.

There were five sites on the Ankobra River at Sanwoma and on the Tano River at Jaway Wharf - Ellenda, where water samples were taken. There were a total of 15 water samples taken from each river, with three replicates taken at each of five locations throughout the river's length. Fifty-three (53) fish, thirty (30) water, and twenty-five (25) sediment samples were taken from the two rivers and studied.

## Sampling method

A simple random sampling technique was employed to collect fresh fish directly from fishermen at the various landing sites of the study areas. Six fishermen were contracted to set up fish traps randomly overnight within the lower, middle, and upper sections of Rivers Ankrobra and Tano, respectively. Sediments were collected from each location using Ekman grab onboard a local canoe. Water samples were also collected from the sub-surface of the same area in the two rivers.

## Data collection techniques and tools

As soon as the fishermen brought in their catches, they were cleaned and divided into species. The sorted samples were placed in plastic ziplock bags, labeled, and transported to the lab in a refrigerator with ice packs for further treatment and Analysis. The substrate was collected in labeled plastic bags and stored in a fridge with ice packs. A pre-cleaned acid-washed 250 mL sampling vial was used to collect the water samples. In addition, 1.5 mL of concentrated analytical-grade nitric acid was added to the water samples to acidify them (Apha 2008). Gamine eTrex GPS units were used to geo-reference the sample sites (Garmin Ltd, Schaffhausen, Switzerland). Table 1 shows the recorded GPS coordinates of the numerous sampling stations along the Ankobra and Tano Rivers.

**Laboratory procedures**

Thawed and identified using the FAO fish identification guide, the fish specimens were studied in the Department of Marine and Fisheries Sciences lab. Six fish families (Clariidae, Anabantidae, Cichlidae, Channidae, Haemulidae, Paenaediae) were represented in the Tano River, with a total of six fish species, including 'Ekpoke' *Ctenopoma kingsleyae* (Gunther, 1896); 'Ekpoke' *Sarotherodon galilaues (*Linnaeus, 1758*)*, 'Ekpoke' *Sarotherodon melanotheron* (Ruppel, *1852)*; 'Adwene' *Clarias gariepinus* (Burchell, 1822) and 'Bile' *Parachanna obscura* (Gunther, 1861).

Four distinct species were discovered in the Ankobra River. Because it is an estuary, the Ankobra was home to both freshwater and marine species. Among the freshwater animals were the "Adwene" *C. gariepinus (*Burchell, 1822) and "Bile" *P. obscura (*Gunther, 1861). However, 'Senzeke' *Parapenaeus longirostris* (Lucas, 1846) *and* 'Nzerma' *Pomadasys jubelini* (Cuvier, 1830) were the saltwater fish of choice. Analyses of mercury (Hg), arsenic (As), and lead (Pb) concentrations in fish (muscle), water, and soil sediments were performed at the Ecological Laboratory, Institute of Environment and Sanitation (ISA), the University of Ghana using an Atomic Absorption Spectrophotometer Pinnacle 900T (Perkin Elmer, USA).

*Fish sample preparation*

Fish samples were cleaned with de-ionized water to remove any remaining contaminants before having their muscles extracted and prepared for heavy metal analysis (Anim et al. 2011). (i) A macro-Kjeldahl digestion flask containing 5 g of fish was weighed, and then 20 mL of strong nitric acid and 20 mL of distilled water were added. By boiling the mixture, the volume was decreased to 20 mL; (ii) after the mixture had cooled, 10 mL of concentrated sulphuric acid was added and heated; and (iii) the mixture was again boiled after the addition of the concentrated acid. Next, nitric acid was added until the fish tissues were digested; (iv) after cooling, 10 mL of saturated ammonium oxalate solution was added and heated until abundant white fumes were created; the oxalate treatment helped remove the yellow coloration caused by nitro compounds such as lipids; (v) the solution was corrected to the 100 mL mark by adding distilled water; (vi) the solution was aspirated for heavy metals analysis using an Atomic Absorption Spectrophotometer Pinnacle 900T.

Nitric acid was added to distilled water and ammonium oxalate to make a blank solution for quality control. In addition to hydrogen peroxide, common oxidants include nitric acid, sulfuric acid, perchloric acid, and hydrochloric acid. As mentioned earlier, mixtures of two or more of the chemicals were suggested because of their combined benefits. Using a combination of oxidizing agents can potentially speed up the process greatly. Since less than 350°C is needed for oxidation, nitric acid can be utilized inexpensively. Moreover, there is less risk of element loss due to volatilization, and the process does not need constant monitoring.

*Sediment sample preparation*

Sediment samples were wet digested with a solution of nitric and perchloric acid, and then mercury, lead, and arsenic were aspirated using an atomic absorption spectrophotometer (Perkin Elmer Pinnacle 900T). (i) Sediment samples weighing between 0.2 and 1.0 g were weighed into a 125 mL Erlenmeyer flask after being oven-dried at 60°C to a constant; (ii) Under a fume hood, 10 mL of a ternary mixture was added (20 mL $HClO_4$; 500 mL $HNO_3$; 50 mL $H_2SO_4$); (iii) The materials were combined, then heated slowly on a hot plate at low to medium heat, all while being monitored by a perchloric acid fume hood; (iv) The sediment samples were digested until thick white fumes (sulfuric acid fumes) were produced; (v) After vigorously heating the mixture (between medium and high heat for half a minute), the liquid was cooled; (vi) 40 to 50 mL of distilled water were added, heated for thirty seconds, and then cooled; (vii) After filtering the resultant mixture into a 100 mL Pyrex volumetric flask, distilled water was added to get the volume up to the desired mark. Mercury, lead, and arsenic concentrations were measured in the stored solution.

*Water sample preparation*

Atomic absorption spectrometer Pinnacle 900T (Perkin Elmer, USA) was used to aspirate and analyze the acidified water samples for Hg directly, As, and Pb concentrations after they were mixed and agitated.

**Quality control**

Stringent quality control procedures were used to guarantee the reliability of the findings. After every ten samples were evaluated, a blank or duplicate was reanalyzed to verify precision and accuracy. All reagents and standard stock solutions were of analytical-reagent grade, and the instruments were calibrated using standard solutions of the various metals (Bortey-Sam et al. 2015).

**Data processing and Analysis**

The collected information was organized and analyzed using Microsoft Excel version 13 (Microsoft Corporation, Redmond, Washington, USA). The data in an Excel file was then transferred to STATA version 13 (StataCorp LP, Chicago, USA) for further statistical Analysis.

The data were summarized using means, standard deviations, ranges, p-values, and percentages. The data was tabulated and shown in bar charts and tables when necessary. The Kruskal-Wallis test was used to compare the concentrations of the samples. Parametric tests like the T-test and the Analysis of Variance (ANOVA) could not be utilized due to the nonparametric character or distribution of the data. Hence this test was employed instead. However, it resulted in highly skewed statistics. An in-text summary of the qualitative information was created. For a fair comparison with WHO-recommended values of heavy metals in fish, water, and sediments, mg/L concentrations were converted to mg/kg and mg/L.

**Ethical considerations/issues**

Before beginning the study, the researchers presented their plan to the Ghana Health Service's Ethics and Research Committee for approval. In addition, chiefs, fishermen, and assemblypersons from Sanwoma and Jaway Wharf-Ellenda were consulted for approval.

## RESULTS AND DISCUSSION

**The concentration of heavy metals in sediment**

Sediment from the Tano and Ankobra Rivers was analyzed for the average quantities of arsenic, mercury, and lead, and the results are shown in Table 2. Tano had heavy metal concentrations of Pb (2.75 ± 6.68 mg/kg) > Hg (1.09 ± 0.73 mg/kg ) > As (0.47 ± 0.29 mg/kg), while Ankobra had heavy metal concentrations of Hg (1.59 ± 2.21 mg/kg) > As (0.71 ± 0.46 mg/kg) >Pb (0.59 ± 0.30 mg/kg). Ankobra sediments had greater mercury concentrations (1.59 ± 2.21 mg/kg) than Tano sediments (1.09 ± 0.73 mg/kg). However, the difference was not statistically significant (p=0.828). Ankobra samples contained 0.71 0.46 mg/kg of arsenic, while Tano samples contained 0.47 ± 0.29 mg/kg. Again, no appreciable variations in water level were observed between the two rivers.

**The concentration of heavy metals in water**

The mean concentration of As, Hg, and Pb in water samples from the Tano and Ankobra Rivers are shown in Table 2. The mean concentration of heavy metal in water from Tano was in the order; Pb (0.11 ± 0.07 mg/L) > As (0.07 mg/L) > Hg (0.01 ± 0.01 mg/L), and Hg (0.15 ± 0.47 mg/L) > Pb (0.15 ± 0.11 mg/L) > As (0.06 ± 0.06 mg/L) in the Ankobra River. Higher levels of Hg were measured in water from the Ankobra (0.15 ± 0.47 mg/L) than in Tano River (0.01 ± 0.01 mg/L), although the difference was not statistically significant (p=0.680). Low but detectable levels of As were measured in water samples from Tano (0.07 mg/L) and Ankobra (0.06 ± 0.06 mg/L), respectively. The mean concentration of Pb in water samples from Ankobra (0.15 ± 0.11 mg/L) was slightly higher than that of Tano River (0.11 ± 0.07 mg/L), suggesting that Pb pollution in Ankobra water could be of much more concern than the levels determined in the Tano River. However, the two rivers had no significant difference (p=0.680).

The average levels of arsenic, mercury, and lead found in river water samples from the Tano and Ankobra are listed in Table 2. Tano's water had a mean heavy metal value of 0.11 0.07 mg/L for lead, 0.07 0.01 mg/L for arsenic, and 0.01 0.01 mg/L for mercury, whereas the Ankobra River's heavy metal concentrations were 0.15 0.47 0.11 0.06 mg/L for mercury, arsenic, and lead, respectively. Despite the lack of statistical significance (p=0.680), Hg concentrations in Ankobra water were higher (0.15 0.47 mg/L) than those in Tano River water (0.01 0.01 mg/L). Tano and Ankobra water samples contained measurable quantities of As (0.07 mg/L and 0.06 0.06 mg/L, respectively). Because the mean concentration of Pb in Ankobra water samples was 0.15 0.11 mg/L, and the mean concentration of Pb in Tano River water samples was 0.11 0.07 mg/L, the contamination of Pb in Ankobra water may be more of a problem than the levels determined in the Tano River. Still, comparing the two rivers, there was no discernible change (p=0.680).

**The concentration of heavy metals in fish**

Overall, the mean concentration of Hg, As, and Pb in the Tano and Ankobra Rivers were; (2.72 ± 1.36 mg/kg), (3.36 ± 1.52 mg/kg) and (0.12 ± 0.10 mg/kg) and (1.45 ± 1.21 mg/kg), (2.15 ± 1.54 mg/kg), and (0.10 mg/kg) respectively as shown in Table 2. Although fish from Tano had relatively higher heavy metal levels, there was no statistical difference in heavy metal levels between the two rivers.

**The concentration of heavy metals in different fish species**

The concentration of Hg in the different fish species from the Tano was; "Ekpoke" *S. melanotheron* (3.71 ± 2.01 mg/kg) > "Bile" *P. obscura* (2.96 ± 0.80 mg/kg) > "Adwene" C. *gariepinus* (2.66 ± 1.02 mg/kg) > "Ekpoke" *C. kingsleyae* (2.51 ± 1.28 mg/kg) > "Ekpoke" *S. galilaues* (1.73 ± 1.14 mg/kg) whereas As was in the order; "Ekpoke" *S. galilaues* (3.86 ± 1.29 mg/kg) > *"Bile "P. obscura* (3.54 ± 1.68 mg/kg)> *"Adwene" C. gariepinus* (3.48 ± 1.78 mg/kg) > "Ekpoke" *S. melanotheron* (2.86 ± 1.97 mg/kg) > "Ekpoke" *C. kingsleyae* (2.72 ± 1.33 mg/kg) as shown in Table 2.

There was no statistically significant difference between the variability of mercury and arsenic levels across fish species (p=0.524 and 0.8250, respectively). Hg levels in Ankobra River fish varied as follows: "Senzeke" *P. longirostris* (2.32 ± 1.95 mg/kg) > "Nzerma" *P. jubelini* (1.24 ± 0.48 mg/kg) > "Adwene" *C. gariepinus* (1.08 ± 0.69 mg/kg)> "Bile" *P. obscura* (0.98 ± 0.86 mg/kg) and that of As was in the order of "Adwene" *C. gariepinus* (2.22 ± 2.22 mg/kg) > "Senzeke" *P. longirostris* (2.07 ± 1.04 mg/kg) > "Bile" *P. obscura* (2.71 ± 1.38 mg/kg) > "Nzerma" *P. jubelini* (1.73 ± 1.70 mg/kg). There was no statistically significant variation in Hg levels between fish species in the Ankobra River (p=0.7200). As demonstrated in Table 2, As levels were higher than Hg levels for three of the four fish species sampled from the Ankobra, except for the "Senzeke," *P. longirostris*.

**Comparative analysis of fish heavy metal concentration from fresh and brackish water in the Ankobra Estuary**

Table 3 shows the concentrations of Hg (1.78. ± 1.21 mg/kg) and (1.03 ± 0.77 mg/kg), and As (1.90 ± 1.36 mg/kg) and (2.47 ± 1.81 mg/kg) in brackish and freshwater fishes from the Ankobra Estuary. The levels of Hg in brackish water were higher than those in freshwater, but the levels of As in freshwater were higher. On the other hand, neither estuary water type showed significantly different heavy metal concentrations (p= 0.338 and 0.621, respectively).

**Comparative Analysis of heavy metal concentration in sediment, water, and fish from the Tano and Ankobra Rivers**

Table 4 compares the heavy metal concentrations in the sediment, water, and fish of the Tano and Ankobra Rivers.

**Table 2.** Mean concentration, standard deviation, and p-values of mercury, arsenic, and lead in sediment, water, and fish from the Tano and Ankobra Rivers, Ghana

| Source | Mercury concentration | | | Arsenic concentration | | | Lead concentration | | |
|---|---|---|---|---|---|---|---|---|---|
| | Mean | SD | p-value | Mean | SD | p-value | Mean | SD | p-value |
| **Sediment (mg/kg)** | | | | | | | | | |
| Tano | 1.09 | 0.73 | 0.828 | 0.47 | 0.29 | 0.570 | 2.75 | 6.68 | 0.978 |
| Ankobra | 1.59 | 2.21 | | 0.71 | 0.46 | | 0.59 | 0.30 | |
| **Water (mg/L)** | | | | | | | | | |
| Tano | 0.01 | 0.01 | 0.680 | 0.07 | * | 0.527 | 0.11 | 0.07 | 0.610 |
| Ankobra | 0.15 | 0.47 | | 0.06 | 0.06 | | 0.15 | 0.11 | |
| **Fish (mg/kg)** | | | | | | | | | |
| ***Tano*** | | | | | | | | | |
| *Ctenopoma kingsleyae* | 2.51 | 1.28 | | 2.72 | 1.33 | 0.825 | 0.1194 | 0.1014 | |
| *Sarotherodon galilaeus* | 1.73 | 1.14 | | 3.86 | 1.29 | | ND | ND | |
| *Clarias gariepinus* | 2.66 | 1.02 | 0.524 | 3.48 | 1.78 | | ND | ND | NA |
| *Parachanna obscura* | 2.96 | 0.80 | | 3.54 | 1.68 | | ND | ND | |
| *Sarotherodon melanotheron* | 3.71 | 2.01 | | 2.86 | 1.97 | | ND | ND | |
| All species | 2.72 | 1.36 | | 3.36 | 1.52 | | 0.1196 | 0.1014 | |
| ***Ankobra*** | | | | | | | | | |
| *Pomadasys jubelini* | 1.24 | 0.48 | | 1.73 | 1.70 | 0.749 | ND | ND | |
| *Parapenaeus longirostris* | 2.32 | 1.95 | | 2.07 | 1.04 | | ND | ND | |
| *Clarias gariepinus* | 1.08 | 0.69 | 0.720 | 2.22 | 2.22 | | ND | ND | NA |
| *Parachanna obscura* | 0.98 | 0.86 | | 2.71 | 1.38 | | 0.1024 | * | |
| All species | 1.45 | 1.21 | | 2.15 | 1.54 | | 0.1024 | * | |

Note: * Only one value recorded

**Table 3.** Shows the average levels of mercury and arsenic found in brackish and freshwater fish caught in the Ankobra River Estuary (Ghana), together with their standard deviations and p-values

| Source | Mercury concentration (mg/kg) | | | Arsenic concentration (mg/kg) | | |
|---|---|---|---|---|---|---|
| | Mean | SD | p-value | Mean | SD | p-value |
| Ankobra | | | | | | |
| Blackish | 1.78 | 1.21 | 0.338 | 1.90 | 1.36 | 0.621 |
| Freshwater | 1.03 | 0.77 | | 2.47 | 1.81 | |

**Table 4.** Shows the average concentrations, standard deviations, and p-values of mercury, arsenic, and lead in sediment, water, and fish from the Tano and Ankobra Rivers, Ghana

| Source | Mercury concentration | | | Arsenic concentration | | | Lead concentration | | |
|---|---|---|---|---|---|---|---|---|---|
| | Mean | SD | p-value | Mean | SD | p-value | Mean | SD | p-value |
| Tano and Ankobra | | | 0.828 | | | 0.570 | | | 0.978 |
| Soil (mg/kg) | 1.40 | 1.78 | | 0.61 | 0.40 | | 1.46 | 4.26 | |
| Water (mg/L) | 0.09 | 0.36 | | 0.06 | 0.05 | | 0.14 | 0.10 | |
| Fish (mg/kg) | 2.09 | 1.29 | | 2.80 | 1.52 | | 0.11 | 0.11 | |

The mean concentrations of Hg and As, overall, were highest in fish (2.09 ± 1.29 mg/kg) and (2.80 ± 1.52 mg/kg), next by sediment (1.40 ± 1.78 mg/kg) and (0.61 ± 0.40 mg/kg), and water (0.09 ± 0.36 mg/L) and (0.06 ± 0.05mg/L) respectively. On the other hand, the highest lead levels were in sediment (1.46 ± 4.26 mg/kg), next by water (0.14 ± 0.10 mg/L) and then fish (0.11 ± 0.11 mg/kg), respectively. However, the variability of overall heavy metal levels in sediment, water, and fish was not significantly different (p=0.828, 0.570, and 0.978, respectively).

In terms of mean concentrations, Hg and As were found to be most abundant in fish (2.09 ± 1.29 mg/kg) and (2.80 ± 1.52 mg/kg), then in sediment (1.40 ± 1.78 mg/kg) and (0.61 ± 0.40 mg/kg), and finally in water (0.09 ± 0.36 mg/L) and (0.06 ± 0.05mg/L). The sediment (1.46 ± 4.26 mg/kg) had the highest lead content, followed by the water (0.14 ± 0.10 mg/L) and the fish (0.11 ± 0.11 mg/kg). There was no statistically significant difference, however, between the sediment, water, and fish in terms of overall heavy metal variability (p=0.828, 0.570, and 0.978, respectively).

## Discussion

This research aimed to quantify arsenic, mercury, and lead concentrations in sediment, fish, and water samples collected from the Ankobra and Tano River basins in Ghana's Western Region. Mining has occurred in the region between the Tano and Ankobra basins at least as far back as the 15th century, as reported by Akabzaa and Darimani (2001); as a result, industrial effluent, including heavy metal, is likely to flow into the streams that feed the Tano and Ankobra River. Comparisons were made between the heavy metal concentrations in aquatic medium and fish and previous findings, as well as the USA EPA, WHO, and US FDA standards and recommendations.

Generally speaking, the mean levels of mercury and arsenic were found to be at their highest in fish; (2.09 ± 1.29 mg/kg), (and 2.80 ± 1.52 mg/kg), respectively. It was followed by sediment; (1.40 ± 1.78 mg/kg), (0.61 ± 0.40 mg/kg), respectively, with water recording the lowest concentrations of mercury and arsenic; 0.09 ± 0.36 mg/L), (0.06 ± 0.05 mg/L), respectively. On the other hand, the lead concentration was highest in the sediments (1.46 ± 4.26 mg/kg), while it was about the same in the water (0.14 ± 0.10 mg/L) and the fish (0.11 ± 0.11 mg/kg) (Table 4).

Consistent with the notion that living things (biota) are exceptional at storing trace elements (Anim-Gyampo et al. 2013). Since fishes are known to bioaccumulate harmful metals, the higher levels of As and Hg detected in fish tissues, compared to sediment and water, are not surprising (Voegborlo and Adimado 2010). The fact that Pb has a higher affinity for sediment may explain why it is found in higher concentrations in sediment than in fish and water (Abouelnasr 2009). Tano appears to be more polluted than the Ankobra River, as evidenced by the greater levels of As and Hg in Tano fish than in Ankobra fish (Table 2). These results are in line with those found by Adjei-Kyereme et al. (2015), who discovered increased heavy metal concentrations in the Tano River and concluded that the presence of so many tributaries to the Ankobra River could be to blame for the apparent lack of a heavy metal concentration gradient between the two rivers. Both the As and Hg concentrations were higher than what is considered safe by the Australian and British governments (recommended level: 1 mg/kg; Neff, 1997; Edmonds and Francesconi, 1993) and the United States Food and Drug Administration (1.0 mg/kg mercury for fish and shellfish).

As levels in fish tissues were found to be highest in the "Bile" *P. obscura*, followed by the "Adwene" *C. gariepinus*, the "Senzeke" *P. longirostris*, and finally the "Nzerma" *P. jubelini*, among the various fish species found in Ankobra. *P. longirostris* and *P. jubelini* are diadromous species and thus do migrate between estuarine and marine environments. That is also reflected in increased As levels from the "brackish" water environment towards the freshwater environment. Voegborlo and Adimado (2010) also observed low concentrations of heavy metal in the marine environment off the coast of Ghana. Therefore, these findings are consistent with theirs. On the other hand, *P. obscura* and *C. gariepinus* are strictly freshwater species. Accordingly, it indicates that As contamination is higher in the Ankobra's freshwater section than in its brackish water section. The Hg concentrations in fish tissues, on the other hand, are highly variable and without any discernible trends. Being a demersal fish species (feeding on detritus), *P. longirostris* may have a high Hg concentration because of its relationship with sediment, which may contain high amounts of mercury (Johnston and Battram 1993).

As before, the arsenic (As) levels in fish tissues were frequently higher than the Hg values observed in Table 2 from the Tano River. A possible explanation is that fish absorb As from water and sediment more efficiently than Hg (Wayah and Gadima 2015). As a result, the As and Hg concentrations in the Tano River were significantly higher than the federal guidelines for safe exposure.

Table 2 shows that the levels of As (0.06 ± 0.06 µg/L), Hg (0.15 ± 0.47 µg/L), and Pb (0.15 ± 0.11 µg/L) in Ankobra's water were greater than those found in Tano with As (0.07 µg/L), Hg (0.01 ± 0.01 µg/L) and Pb (0.11 ± 0.07 µg/L). One possible cause is the influx of highly contaminated effluent from gold mining activities, especially artisanal (Galamsey) gold mining, into the Ankobra River basin via its many tributaries. In addition, the contaminated Hg levels were significantly higher than the acceptable threshold for both drinking water (2 µg/L) (WHO 2004) and pristine freshwater ecosystem (0.005 µg/L) (ATSDR, 1997). meanwhile, As (arsenic) concentrations were far higher than the safe limit of 50 g/L set by the World Health Organization (IPCS/WHO 1992).

Pb concentrations in sediments were found to be greater in Tano (2.75 ± 6.68 mg/kg) than in Ankobra 0.59 ± 0.30 mg/kg), although Hg and As concentrations were just slightly higher in Ankobra (Table 2). But the average levels of Hg, As, and Pb in the sediments of the two rivers were not significantly different from one another (p=0.828, 0.570, and 0.978, respectively). Adjei-Kyereme et al. (2015) found the same for Hg in the Tano and the Ankobra.

Since many of the fish used for human consumption in Ghana are taken in the Ankobra and Tano Rivers, the results of this study showing Hg, As, and Pb levels beyond the acceptable threshold from both study locations are somewhat worrying. Extreme neurotoxic and genotoxic consequences have been linked to the high amounts of Hg in fish (Hibbeln et al. 2007). In addition to causing symptoms like chest pain, shortness of breath, coughing up blood, paresthesias, and numbness in the hands and feet, a high mercury intake can slow down a baby's brain development. In addition, eating mercury-tainted foods can negatively affect the digestive system and lead to renal damage (Järup 2003). Increased toxicity from Hg has been linked to various negative outcomes for fish, including stunted growth and development, aberrant behavior, blood chemistry changes, decreased oxygen exchange, and even mortality (Folmar 1993).

A higher risk of skin cancer and other skin diseases, including hyperkeratosis and pigmentation alterations, has also been related to As contamination in food consumed at more than acceptable levels. A recent review by the World Health Organization (2004) found that exposure to arsenic in drinking water is causally linked to lung, kidney, bladder, and skin cancers, the latter of which is preceded by easily detectable precancerous lesions. These findings raise the possibility that exposure to environmental arsenic contributes to the development of Buruli ulcers (Gyasi et al. 2012). In addition, cancers of organs other than the lungs, bladder, kidneys, and skin are supported by data, as are hypertension and cardiovascular disease (Järup 2003).

Acute lead poisoning symptoms include headache, irritability, abdominal discomfort, and other symptoms connected to the nervous system, and they have been linked to exposure to greater Pb levels. Lead encephalopathy can cause severe symptoms such as acute psychosis, confusion, lowered consciousness, insomnia, and irritability. In addition, affected children may have trouble paying attention in class and displaying appropriate behavior. In mild cases, anemia due to disruption in hemoglobin production is the most noticeable symptom of lead poisoning.

Overall, the study found that the Ankobra and Tano Rivers, known to flow through several industrial and artisanal mining regions, contain heavy metal loads above the World Health Organization's (WHO) recommenced values for sediment, water, and fish. The mean levels of As, Hg, and Pb in the Tano River were higher than those in the Ankobra River. Because of this, Tano River sediment contains more metals than Ankobra River sediment. Since these heavy metals are regarded as toxic and can cause poor health of fish and other aquatic species, the health of aquatic life in these rivers is at risk. Communities' health that relies on these water sources for drinking, bathing, and other purposes are similarly threatened.

## REFERENCES


Abouelnasr DM. 2010. The relationship between soil particle size and lead concentration. Proc Ann Intl Conf Soils Sediments Water Energ 14: 8.

Abualtayef MT, Abd Rabou AN, Abu Foul AA, Ghabayen SM, Elsinwar HM. 2014. Microbial water quality of coastal recreational water in the Gaza Strip, Palestine. Nusantara Biosci 6: 26-32. DOI: 10.13057/nusbiosci/n060105.

Adjei-Kyereme Y, Donkor AK, Golow AA, Yeboah PO, Pwamang PJ. 2015. Mercury concentrations in water and sediments in rivers impacted by artisanal gold mining in the Asutifi District, Ghana. Res J Chem Environ Sci 3: 40-48.

Akabzaa T, Darimani A. 2001. Impact of mining sector investment in Ghana: A study of the Tarkwa mining region. Third World Network.

Akan JC, Mohmoud S, Yikala BS, Ogugbuaja VO. 2012. Bioaccumulation of some heavy metals in fish samples from River Benue in Vinikilang, Adamawa State, Nigeria. Am J Anal Chem 3 (11): 727-736. DOI: 10.4236/ajac.2012.311097.

Anim AK, Ahialey EK, Duodu GO, Ackah M, Bentil NO. 2011. Accumulation profile of heavy metals in fish samples from Nsawam, along the Densu River, Ghana. Res J Environ Earth Sci 3 (1): 56-60.

Anim-Gyampo M, Kumi M, Zango M. 2013. Heavy metals Concentrations in some selected Fish Species in Tono Irrigation Reservoir in Navrongo, Ghana. J Environ Earth Sci 3 (1): 109-120.

Apha W. 2008. AWWA, 1998. Standard Methods for the Examination of Water and Wastewater, 20th Ed. American Public Health Association, Washington, DC.

Asante F, Agbeko E, Addae G, Quainoo AK. 2014. Bioaccumulation of heavy metals in Water, Sediments and Tissues of Some Selected Fishes from the Red Volta, Nangodi in the Upper East Region of Ghana. Brit J Appl Sci Technol 4 (4): 594-603. DOI: 10.9734/BJAST/2014/5389.

Bannerman W, Potin-Gautier M, Amouroux D, Tellier S, Rambaud A, Babut M, Beinhoff C. 2003. Mercury and arsenic in the gold mining regions of the Ankobra River basin in Ghana. J de Phys IV France 2003: 107-110. DOI: 10.1051/jp4:20030255.

Basim Y, Khoshnood Z. 2013. Target hazard quotient evaluation of cadmium and lead in fish from Caspian Sea. Toxicol Ind Health 32 (2): 215-220. DOI: 10.1177/0748233713498451.

Begum A, Harikrishna S, Khan I. 2009. Analysis of heavy metals in water, sediments and fish samples of Madivala Lakes of Bangalore, Karnataka. Intl J Chemtech Res 1 (2): 245-249.

Bortey-Sam N, Nakayama S, Akoto O, Ikenaka Y, Fobil J, Baidoo E, Ishizuka M. 2015. Accumulation of heavy metals and Metalloid in Foodstuffs from Agricultural Soils around Tarkwa Area in Ghana, and Associated Human Health Risks. Intl J Environ Res Public Health 12 (8): 8811-8827. DOI: 10.3390/ijerph120808811.

Cobbina S, Duwiejuah A, Quansah R, Obiri S, Bakobie N. 2015. Comparative Assessment of heavy metals in Drinking Water Sources in Two Small-Scale Mining Communities in Northern Ghana. Intl J Environ Res Public Health 12 (9): 10620-10634. DOI: 10.3390/ijerph120910620.

Dogbevi EK. 2009. Cover Story: Africa-Living with E-waste. Gov News 29 (3): 22.

Falkenmark M, Lundqvist J. 1998. Towards water security: Political determination and human adaptation crucial. Nat Resour Forum 22 (1): 37-51. DOI: 10.1111/j.1477-8947.1998.tb00708.x.

Folmar LC. 1993. Effects of chemical contaminants on blood chemistry of teleost fish: A bibliography and synopsis of selected effects. Environ Toxicol Chem 12 (2): 337-375. DOI: 10.1002/etc.5620120216.

Gyasi SF, Awuah E, Larbi J, Koffuor GA, Osei O. 2012. Clinical, hematological and histopathological responses to arsenic toxicity in ICR mice using arsenic levels synonymous to buruli ulcer endemic communities in the Amansie West District of Ghana. Eur J Exp Biol 2 (3): 683-689.

Hibbeln JR, Davis JM, Steer C, Emmett P, Rogers I, Williams C, Golding J. 2007. Maternal seafood consumption in pregnancy and neuro-developmental outcomes in childhood (ALSPAC study): An observational cohort study. Lancet 369: 578-585. DOI: 10.1016/s0140-6736(07)60277-3.

Ida CJ. 2012. Heavy metals in Suchindramkulam (a Lentic Water Body) of Kanyakumari District, Tamil Nadu, India. J Trop Exp Biol 8 (3-4): 141-145.

IPCS/WHO. 1992. International Programme on Chemical Safety/World Health Organization. Diethylhexylphthalate. Environ Health Crit 131: 1-141.

Järup L. 2003. Hazards of heavy metal contamination. Brit Med Bull 68: 167-182. DOI: 10.1093/bmb/ldg032.

Johnston IA, Battram J. 1993. Feeding energetics and metabolism in demersal fish species from Antarctic, temperate and tropical environments. Mar Biol 115: 7-14. DOI: 10.1007/BF00349380.

Krupnova TG, Mashkova IV, Kostryukova AM, Egorov NO, Gavrilkina SV. 2018. Bioconcentration of heavy metals in aquatic macrophytes of South Urals region lakes. Biodiversitas 19: 296-302. DOI: 10.13057/biodiv/d190140.

Laibu PK, Maingi J, Kebira A. 2018. Determination of bacterial composition, heavy metal contamination and physicochemical parameters of fish pond water in Abothuguchi Central, Meru County, Kenya. Bioteknologi 15: 70-83. DOI: 10.13057/biofar/c150103.

Length F. 2011. Bioaccumulation of some heavy metals in tilapia fish relevant to their concentration in water and sediment of Wadi Hanifah, Saudi Arabia. Afr J Biotechnol 10 (13): 2541-2547. DOI: 10.5897/AJB10.1772.

Li J. 2014. Risk assessment of heavy metals in surface sediments from the Yanghe River, China. Intl J Environ Res Public Health 11 (12): 12441-12453. DOI: 10.3390/ijerph111212441.

Mwatsahu SH, Wanjau R, Tole M, Munga D. 2020. Heavy metal contamination in water, sediments, and fauna of selected areas along the Kenyan coastline. Ocean Life 4: 37-47. DOI: 10.13057/oceanlife/o040105.

Primadiani D, Iskandar J, Sunardi. 2018. Local knowledge of Karangwangi People of Cianjur District, West Java, Indonesia on water pollution of the Cikawung River. Asian J Ethnobiol 1: 9-14. DOI: 10.13057/asianjethnobiol/y010102.

Retnaningdyah C, Arisoesilaningsih E, Samino S. 2017. Use of local Hydromacrophytes as phytoremediation agent in pond to improve irrigation water quality evaluated by Diatom Biotic Indices. Biodiversitas 18: 1596-1602. DOI: 10.13057/biodiv/d180440.

Rijsberman FR. 2006. Water scarcity: Fact or fiction? Agric Water Manag 80 (1): 5-22. DOI: 10.1016/j.agwat.2005.07.001.

Sulardiono B, A'in C, Muskananfola MR. 2018. Profiles of water quality at Menjangan Besar Island, Karimunjawa, Central Java Province, Indonesia. Biodiversitas 19: 2308-2315. DOI: 10.13057/biodiv/d190639.

Taylor JR. 2012. Global heavy metal Pollution (AMD/ARD) Impacts, 3101.

Vikas M, Dwarakish GS. 2015. Coastal pollution: A review. Aquat Proc 4: 381-388. DOI: 10.1016/j.aqpro.2015.02.051.

Voegborlo RB, Adimado AA. 2010. Total mercury distribution in different fish species representing different trophic levels from the Atlantic Coast of Ghana. J Sci Technol 30 (1): 1-9. DOI: 10.4314/just.v30i1.53933.

Wayah SB, Gadima BS. 2015. Bioaccumulation of Mercury, Arsenic and Copper in two freshwater fish species from River Kaduna (Nigeria). Toxicol Sci 4 (6): 337-341.

WHO. 2004. Arsenic in drinking-water. IARC Monographs on the Evaluation of Carcinogenic Risks to Humans 84: 41-267. DOI: 10.1016/j.kjms.2011.05.002.

Wuana RA, Okieimen FE. 2011. Heavy metals in contaminated soils: A review of sources, chemistry, risks and best available strategies for remediation. Isrn Ecol 2011: 402647 DOI: 10.5402/2011/402647.