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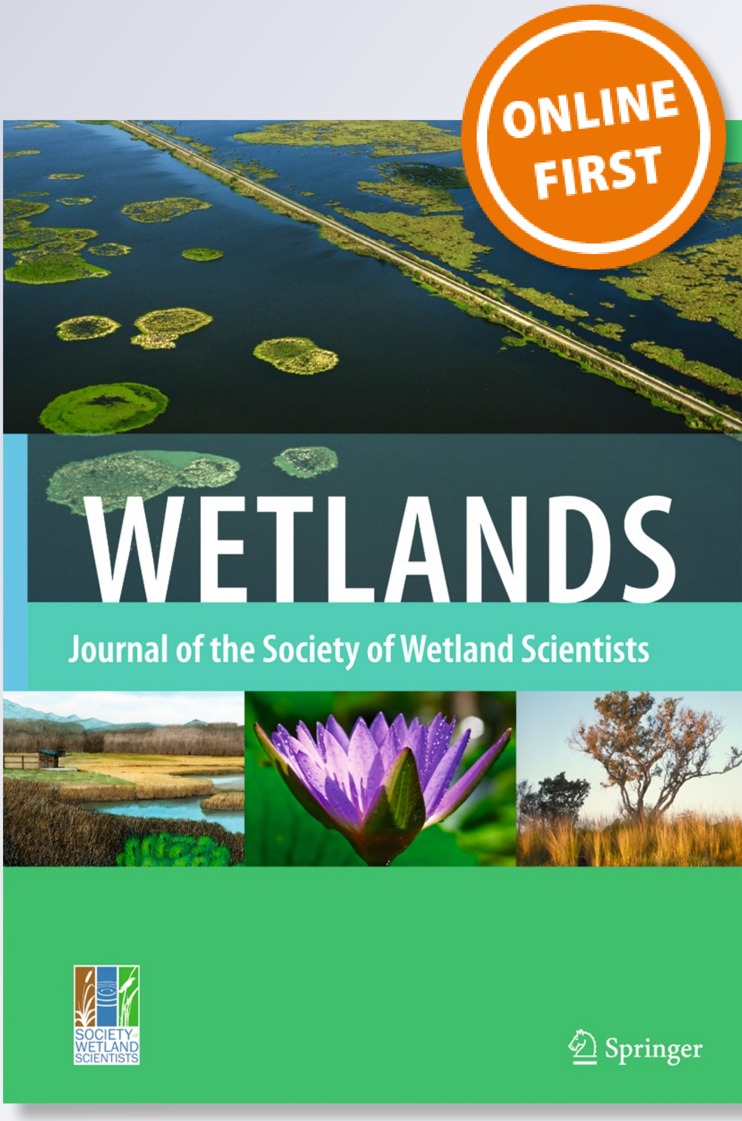

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ORIGINAL RESEARCH

Comparative Assessment of Mangrove Biomass and Fish Assemblages in an Urban and Rural Mangrove Wetlands in Ghana

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Abstract Mangrove wetlands are one of the most important ecosystems on earth, providing habitats for both marine and terrestrial organisms as well as supporting essential human services. However, high dependence of humans on these systems is leading to significant transformation of mangrove wetlands and reduction in their ecosystem services including fisheries. The objectives of this study were to estimate the biomass of two mangrove wetlands in Ghana within urban and rural contexts and determine the fish fauna assemblages as part of baseline setting. The study used the structural parameters of mangrove species and allometry to estimate the biomass of both forest systems. Fish community structure were determined based on ecological surveys. The findings show that the standing biomass of the mangrove forests were significantly higher ($p < 0.01$) in the rural wetland (394.49 t/ha) compared to the urban wetland (126.29 t/ha). Fish fauna assemblages, referring to species richness and diversity were higher in the urban wetland at 4.21 and 2.64 respectively compared to the rural wetland at 3.46 and 2.09 respectively. This paper concludes that a well-developed mangrove system with high mangrove biomass may not necessarily imply high fish species richness and diversity.

Keywords Human impacts · Mangrove conservation · Biomass · Allometry · Ghana

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Introduction

Mangroves are structurally and functionally unique habitats restricted to intertidal and adjacent communities in tropical and subtropical regions worldwide (Tomlinson 1986; Nagelkerken et al. 2008). Mangrove tree species possess unique adaptations including stilt roots and pneumatophores which allow exchange of gases for their root tissues (Kauffman and Donato 2012). These aerial root systems of mangrove trees also extend into the intertidal and subtidal areas where they provide stability to the otherwise soft sediment environment (Ellison and Farnsworth 1992). Indeed, mangroves are important habitats for many terrestrial and marine fauna including several important commercial fish species (Aheto et al. 2014). Within the water column, the mangrove roots are overgrown by epibionts such as tunicates, sponges, algae, and bivalves with spaces between roots serving as shelter and food for motile fauna such as prawns, crabs and fishes (Nagelkerken et al. 2008). Mangrove trees and canopy provide important habitat for a wide range of species including birds, insects, mammals and reptiles (Nagelkerken et al. 2008). Though restricted to coastlines, mangroves are one of the most productive coastal ecosystems that support a wide range of goods and services (Field et al. 1998). Humans have traditionally relied on mangrove products such as finfish, shellfish, firewood, timber, tannins, dyes and medicinal products for their basic subsistence (Robertson 1988; Barbosa et al. 2001; Aheto 2011) that has possibly contributed to the high degradation of mangrove ecosystems worldwide. It is noteworthy that among the major terrestrial land covers, the carbon storage potential of mangroves is highest, yet mangrove ecosystem services are among the least investigated (Kauffman and Donato 2012).

Estimating the biomass of mangrove tree species for example is an essential component for determining the carbon storage potential of mangrove ecosystems (Kairo et al. 2008; Aheto et al. 2011), also relevant in estimating mangrove forest status (Soares and Schaeffer-Novelli 2005), essential for assessing the yield of commercial products from forests, and for the development of sound silvicultural practices (Kairo et al. 2009). Above ground biomass of mangrove forests is usually estimated indirectly from measurements of stem diameter at a height of 1.3 m above ground (Kirui et al. 2006; Kairo et al. 2009), using allometric relationships derived from diameter at breast height alone, or in combination with height, as the independent variables and the dry weights of different parts of the tree as dependent variables (Kirui et al. 2006; Komiyama et al. 2008; Aheto et al. 2011). The allometric biomass estimation method uses mathematical regression models and avoids disturbances of the ecosystem under investigation, which is an important prerequisite in many experimental settings (Gehring et al. 2008). Globally, seventy true mangrove species have been recorded (Spalding et al. 1997), with mangrove forests estimated to occupy about 14 million ha of land worldwide (Giri et al. 2011; Kauffman and Donato 2012). Even though West Africa has fewer true mangrove species (eight) than East Africa (nine), there is more extensive mangrove coverage along the West African coast due to the extensive riverine systems not present in the east (Shumway 1999). Mangroves cover an approximate area of 137 km^2 of Ghana's land area which represent about 0.5 % of total mangrove area cover in Africa (United Nations Environment Programme, UNEP 2007). Unfortunately, mangrove cover in Ghana continues to decline since 1980 due to overexploitation and conversion of mangrove areas and saltmarshes into saltpans, sugarcane production, clearing for building, fish processing and construction and only 1.5 % of total cover occur in protected areas (United Nations Environment Programme, UNEP 2007).

Though the existence of six mangrove species, *Laguncularia racemosa*, *Avicennia germinans*, *Rhizophora harrisonii*, *Rhizophora racemosa*, *Acrostichum aureum* and *Cornocarpus erectus*, has been documented (United Nations Environment Programme, UNEP 2007), the occurrence of a seventh species, i.e. *Rhizophora mangle*, has been highly disputed. For instance, whilst United Nations Environment Programme, UNEP (2007) disregards *R. mangle* as occurring in Ghana, Ellison et al. (2015) identifies *R. mangle* as occurring in Ghana and a species of least concern in the IUCN Red List. In this study, samples of known *R. mangle* are taken through systematic identification procedures to determine the existence of this species in Ghana.

The study was carried out to identify mangrove species present in an urban and rural wetlands in Ghana and provide ecological information in relation to species population parameters and the biomass of mangroves in both ecosystems. The study further determines the hydrographic conditions prevalent in the two wetlands and their relationship with fish fauna assemblages within each system.

Study Area

Two coastal wetlands were selected for this study; the Nyan, considered a rural wetland due to its location with comparatively low human disturbance to the Whin, classified as an urban wetland due to its proximity to urban development. The Whin river estuary is located on the westernmost section of the Sekondi-Takoradi Metropolis whilst the Nyan river estuary is located in the Ahanta West District; both in the Western Region of Ghana. The study areas, Whin and Nyan wetlands, are located at 4°52′44.61″ N; 1°46′29.57″ W and 4°47′52.21″ N; 2°08′28.36″ W respectively. Both wetlands are fed with riverine water systems flowing downstream with saline water systems flowing upstream to the highest high tide level. At high tides, sea water intrudes the Nyan and Whin wetland systems supplying them with saline water. During low tides, the two wetlands are fed with fresh water and nutrients from land. The wetlands open into the sea all year round due to a direct connection of the Nyan and Whin rivers to the sea.

The geology of the Whin estuary area is predominantly sandstone and grits while that of the Nyan estuarine area is made up of a granite complex (Ghana Minerals Commission, 2011). The Whin River estuary has a rocky shore on the eastern side with the other side made up of a sandy beach stretch with coconut trees, mangroves and other coastal vegetation. Both sides of the mouth of the Nyan estuary are made up of sandy beaches with dunes occupied by mangrove forests that stretch inland.

Materials and Methods

Mangrove Forest Structure Assessment and Above-Ground Biomass Determination

A 2005 orthophotograph obtained from the Ghana Survey Department was used to map out the study area (Fig. 1) prior to field data collection. Field surveys were carried out over a seven month period, from December 2010 to June 2011. Using ArcGIS, one hectare (10,000 m^2) plot from the mouth of each estuary were selected for the survey of mangroves. Six different transects were laid at 20 m intervals perpendicular to the general direction of the shoreline. On each transect, five 5 m x 5 m quadrats were established at 15 m intervals for sampling. The coordinates of all sampling points were fed into a Garmin GPS Etrex 20 and was used to identify specific survey points on the field.

A preliminary survey was conducted to identify all resident mangrove species at both study sites. Samples of mangrove trees, including flowers, stems, leaves, fruits, silt roots and propagules were collected and taken to the Herbarium of the Department of Environmental Science at the University of Cape Coast for identification. Samples were identified using species identification manuals. Samples were also compared

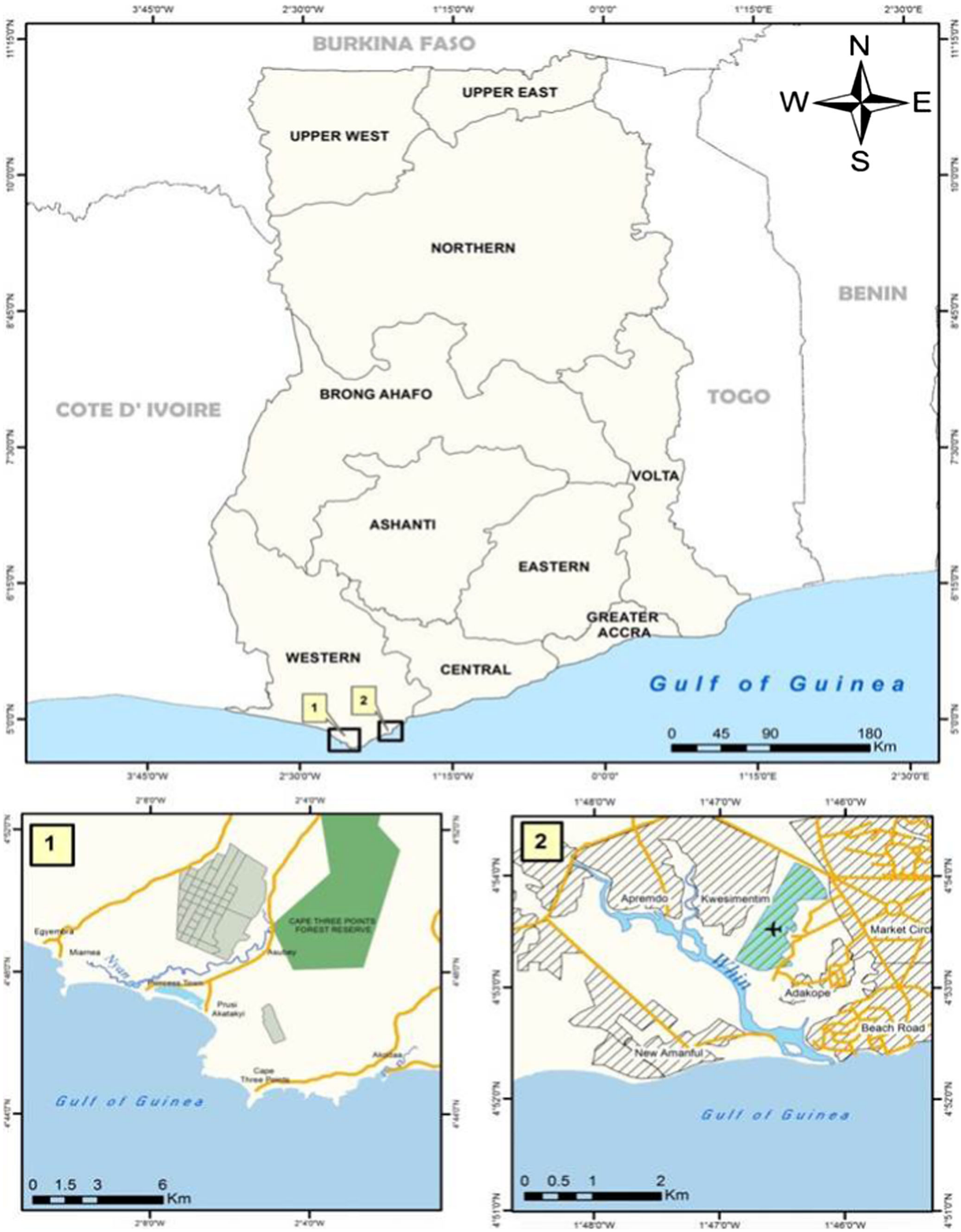

Fig. 1 Map showing Nyan estuary (1) and the Whin estuary (2)

with already identified specimen from other jurisdictions to verify species.

During the actual field surveys, measurements of height and diameter at breast height (DBH) were done for all three encountered species in all quadrats. Ten trees each of the three mangrove species were harvested randomly at both study sites from the ground level using a handsaw. Heights of the trees of each species were taken using a 10 m calibrated pole. The diameter at breast height (DBH) of the different species was determined using vernier callipers. The mean tree height and the DBH, the basal area, density of trees, importance value and the diversity index per species were calculated.

The above ground parts were separated into woody parts (stem and branches), leaves, and fruits. In the case of *Rhizophora* species, their prop roots were included in the analysis. The total harvested fresh weight of each component was measured in the field and representative sub-samples were measured and oven dried to a constant weight at 85 °C in order to calculate fresh-dry weight ratio. Each sampled individual tree was then described by its structural parameters and its partitions (above-ground roots, leaves, branches, stems and fruits) as well as the total biomass values.

Assessment of Fish Assemblages and Hydrographic Conditions

Surveys of fish species present in the wetlands and hydrographic conditions were conducted for seven months, from February – August, 2011. Surveys were done at low tides in both wetlands, using GHAPOHA (2011) tide tables. A 7 m long x 1.5 m deep pole-seine net with stretched mesh size of 5 mm was used for the fish survey; by dragging nets from the mouth of the estuary to an average distance upstream of about 300 m over a 30 min period. Collected fish samples were kept in iced containers and transported immediately to the laboratory for sorting and identification. Each individual fish was identified using the identification keys of Dankwa et al. (1999) and Paugy et al. (2003).

Water quality data, including temperature, salinity, pH, dissolved oxygen and turbidity, were collected once monthly from the two wetlands over eleven months. Each estuary was divided into three sections (mouth, middle and upper) to allow for increased hydrographic data accuracy. Using a YSI (model 63) multi-parameter probe, 3 replicate hydrographic data were collected from each section.

Data Analysis

Mangrove Biomass

Simple correlations (Pearson product moment correlation) were derived between tree structural variables and dry weights. The statistical significance of the regression equation was assessed by the coefficient of determination (R^2) and single classification ANOVA, at $p < 0.01$ level. The allometric eqs. - were used to estimate the above-ground biomass values to a selected number of tree individuals in the sample quadrats. Total plot biomass was obtained from summation of all biomass values of trees in the 30 plots sampled at each site. Biomass values per species were then expressed in dry weight tonnes per hectare (t/ha).

The density of individual tree species (Di), their basal area (BA) and importance values (I_V) were calculated according to Cintron and Schaeffer-Novelli (1984) equations as follows:

Density:

$$\mathrm{Di} = \frac{\mathrm{n}i}{\mathrm{A}}$$

Where, Di = Density of species i; ni = Total number of species i and

A = Area of sampling area (25 m^2)

Basal Area of Trees (BA):

$$\mathrm{BA} = \pi \mathrm{r}^2$$

$$\mathrm{r} = \mathrm{DBH}/2 \text{and} \pi = 3.142$$

Importance Values (I_V):

The I_V of a species was obtained by adding the values of relative frequency, relative density and relative dominance according to the formulas used to obtain these values are below.

$$\text{Frequency of species}\, i = \frac{\text{Number of individuals of species}\, i}{\text{Total Number of individual of all species}}$$

$$\text{Relative frequency}(R_f) = \frac{\text{Frequency of species}\, i}{\Sigma\,\text{Frequency of all species}} \times 100$$

$$\text{Relative density}(R_d) = \frac{\text{Number of individuals of species}\, i}{\Sigma\,\text{Number of individuals of all species}} \times 100$$

$$\text{Relative dominace}(RD) = \frac{\text{BA of species}\, i}{\Sigma\,\text{BA of all species}} \times 100$$

Fish Community Structure

Different indices were used to identify fish community structure: Species richness was determined using Margalef index (d) given as:

$$d = (S\text{–}1)/(\mathrm{In}\,N) \tag{7}$$

where S is the total number of species in the sample and N is the total number of individuals in the sample. Diversity within the two wetlands were ascertained by the Shannon-Weiner index (H′), given as:

$$H' = \left[-\sum_{t=1}^{s} Pi(lnPi)\right] \quad (8)$$

where S is the number of species in the community and Pi is the proportion belonging to species I in the community. Evenness component of diversity was calculated from Pielou's index given as:

$$J' = H'/H_{max} \text{ where } H_{max} = \ln S \quad (9)$$

The degree of similarity between fish communities in the two wetlands was determined by Sorensen's similarity index:

$$C_s = \left[\frac{2j}{(a+b)}\right] \quad (10)$$

where j is the number of species common to the two estuaries, a and b are the number of species occurring in either of the wetlands.

In addition, a non-metric multidimensional scaling (NMDS) was used to determine distance matrix between fish assemblages and water quality conditions within the urban and rural wetlands. Water quality data were normalized using the expression [x-mean]/stdev and an NMDS similarity matrix developed using Bray–Curtis similarity index. A similarity percentages-species contributions analysis (SIMPER) was also used to determine the contribution of different species within the assemblages, using the Bray–Curtis similarity/distance measure. An ANOSIM was done to test the significance of the observed similarities.

Results

Structural Parameters of Mangrove Species

Mean height of *R. mangle* was significantly higher ($p < 0.01$) at the rural wetland compared to those at the urban wetland. The heights of *Laguncularia racemosa* and *Avicennia germinans* were higher at the urban wetland than those at the rural wetland (Table 1). The overall mean height of mangrove species found at the rural mangrove forest varied significantly from those found at the urban mangrove forest ($p < 0.01$).

Within the rural mangrove system, heights of trees varied significantly among the three mangrove species (ANOVA, $F_{2, 1188} = 67.48$, $p < 0.01$) while no significant differences were observed in the height of tree species at the urban mangrove system (ANOVA, $F_{2, 841} = 1.349$, $p > 0.01$). Between sites, mean height of *Laguncularia racemosa* and *Avicennia germinans* varied significantly, $p < 0.01$ and $p < 0.01$ respectively, while no significant difference was observed in the mean heights of *R. mangle* ($t = 0.8017$, df = 367, $p > 0.01$).

There was also a significant difference in the overall means of DBH of all mangrove species at both the rural and urban wetland ($t = 9.788$, df = 2034, $p < 0.01$). No significant difference existed in mean DBH of *R. mangle* ($t = 0.7042$, df = 367, $p > 0.01$) between the two wetlands, while for *Laguncularia racemosa* and *Avicennia germinans*, mean differences were observed to be statistically significant between the wetlands ($t = 8.371$, df = 718, $p < 0.01$ and $t = 17.36$, df = 1145, $p < 0.01$ respectively). Within sites, mean DBH varied significantly among species at the rural wetland (ANOVA, $F_{2, 1189} = 63.81$, $p < 0.01$) whereas at the urban wetland, mean DBH among mangrove species were not significantly different (ANOVA, $F_{2, 841} = 1.13$, $p > 0.01$). Mean DBH values for individual species were relatively larger at the urban wetland except for *Rhizophora mangle* which was larger at the rural wetland (Table 2).

Allometric Equations

There was a strong positive correlation between the height and diameter at breast height of all mangrove species encountered at both the rural and urban wetland (Fig. 2). The regression equations used for predicting the total above-ground biomass per unit area for all mangrove species at the two sites are presented in Table 3. The Pearson product moment correlation coefficient (R^2) between height and diameter at breast height ranged from 0.7–0.91, and were statistically significant at $p < 0.01$. The regression constants (a and b), and the coefficient of determination (R^2) are based on the independent variables including diameter at breast height (D), Height (H) and product of D and H (DH). The best estimate of biomass for all three mangrove species at both sites was obtained when D was substituted with DH, i.e. the product of diameter at breast height and height, which also resulted in the highest R^2 values for all mangrove species.

Standing Biomass

Biomass estimates for all the mangrove species found at the rural (Nyan) and urban (Whin) wetlands are presented in Table 4. The standing biomass of *Rhizophora mangle* dominated the rural wetland and was about three times the highest standing biomass species at the urban wetland (*Avicennia germinans*). The standing biomass of all corresponding mangrove species was higher at the rural wetland (Table 4).

Wetland Water Conditions

There were significant differences in all the means of the physico-chemical parameters between the rural and the urban wetland (Table 5). The means of temperature, pH and salinity

Table 1 The mean height (mean ± S.E) of mangrove species at the studied rural and urban wetlands in the Western Region of Ghana (n = 30)

Species	Rural		Urban	
	Number of replicates	Mean height (cm)	Number of replicates	Mean height (cm)
Rhizophora mangle	243	3.04 ± 0.17	135	2.84 ± 0.19
**Laguncularia racemosa*	360	2.48 ± 0.06	159	3.04 ± 0.11
**Avicennia germinans*	597	1.68 ± 0.05	550	3.13 ± 0.10

*- denotes significant difference between sites

were higher at the urban wetland, while the rural wetland recorded a higher mean dissolved oxygen value (Table 5).

Fish Community Structure

In all, 28 species of shellfishes and finfishes from 18 families were encountered in both estuaries, comprising of freshwater, brackish and marine species (Table 6). Twenty seven species were encountered in the Whin wetland, made up of 23 finfish species and 4 shellfish species. In the Nyan wetland, 21 species were encountered made up of 17 finfish species and 4 shellfish species. Sixteen out of the total 23 finfish species recorded in the study were common to both estuaries. Seven out of the remaining 8 finfish species were encountered only at the Whin wetland while the remaining one finfish species was found at Nyan. All four shellfish species were also common to both estuaries. Three main fish species dominated the two wetlands: the urban wetland was dominated by *Sarotherodon melanotheron* (30.8 %) and *Sardinella aurita* (10.9 %) with the Nyan Community also being dominated by *Sarotherodon melanotheron* (49.4 %) and *Liza falcipinnis* (6.5 %) which was the third dominant in Whin (7.9 %). Compositions of the remaining species ranged between 0.2 % and 5.6 % within the two communities.

Sampled *Sarotherodon melanotheron* had total length range of 4.5–3.7 cm and 4.7–15.3 cm at the urban and rural wetlands respectively. *Liza falcipinnis* had total length range of 8–21.3 cm at urban and 3.4–18.2 cm at the rural wetland (Table 6).

The urban wetland had higher richness, diversity and evenness values compared to the rural wetland, with the similarity index value of 0.83 (Table 7).

Table 2 Mean diameter at breast height (cm ± S.E) of mangrove species at the studied rural and urban wetlands

Species	Rural (cm ± S.E)	Urban (cm ± S.E)
Rhizophora mangle	3.04 ± 0.15	2.84 ± 0.16
Laguncularia racemosa	2.52 ± 0.08	3.04 ± 0.11
Avicennia germinans	1.27 ± 0.07	3.13 ± 0.09

Overall, there was an average dissimilarity of 44.56 % between the species composition within the urban and rural wetland (Table 8). Six species, *Sarotherodon melanotheron*, *Sardinella aurita*, *Liza falcipinis*, *Sardinella maderensis*, *Liza dumerilii* and *Lutjanus goreensis*, were most responsible for distinguishing between the urban and rural wetland. These six species combined contributed to 53.52 % of the differences between the urban and rural wetland. The other 22 species account for the remaining 46.48 % differences. The fish assemblages were found to be significantly different between the urban and rural wetland (ANOSIM, $p < 0.01$, $R = 0.5155$).

The dissimilarity matrix on the two dimensional scatter plot shows clear dissimilarities in the urban and rural wetland fish species assemblages in response to prevailing aquatic environmental conditions (Fig. 3). It is noted that the relative tie or cluster of salinity, temperature, pH, and turbidity suggest that these factors are major confounding reasons for the observed fish species assemblages especially in the urban wetland.

Discussion

Rhizophora mangle Species Controversy and Verification in Ghana

There has been an on-going controversy amongst coastal researchers about the occurrence of *Rhizophora mangle species* in Ghana since the early 2000s. Mangrove species listed as occurring in Ghana by FAO (2007) and UNEP (2007) did not include *R mangle*. However, according to Ellison et al. (2015), *Rhizophora mangle* occurs in Ghana and is classified as a species of least concern in the IUCN Red List. Other local studies including Aheto et al. (2011) have also identified *Rhizophora mangle* as occurring in Ghana. The other *Rhizophora spp.* that are known to occur in Ghana are the *RHIZOPHORA racemosa* and *Rhizophora harrisonii*. Cornejo (2013) and Ceron-Souza et al. (2014) have identified *Rhizophora harrisonii* as a natural hybrid produced by ongoing hybridization and introgression between *Rhizophora mangle* and *Rhizophora racemosa* when they occur in sympatry. Thus, giving strong indications that once *Rhizophora*

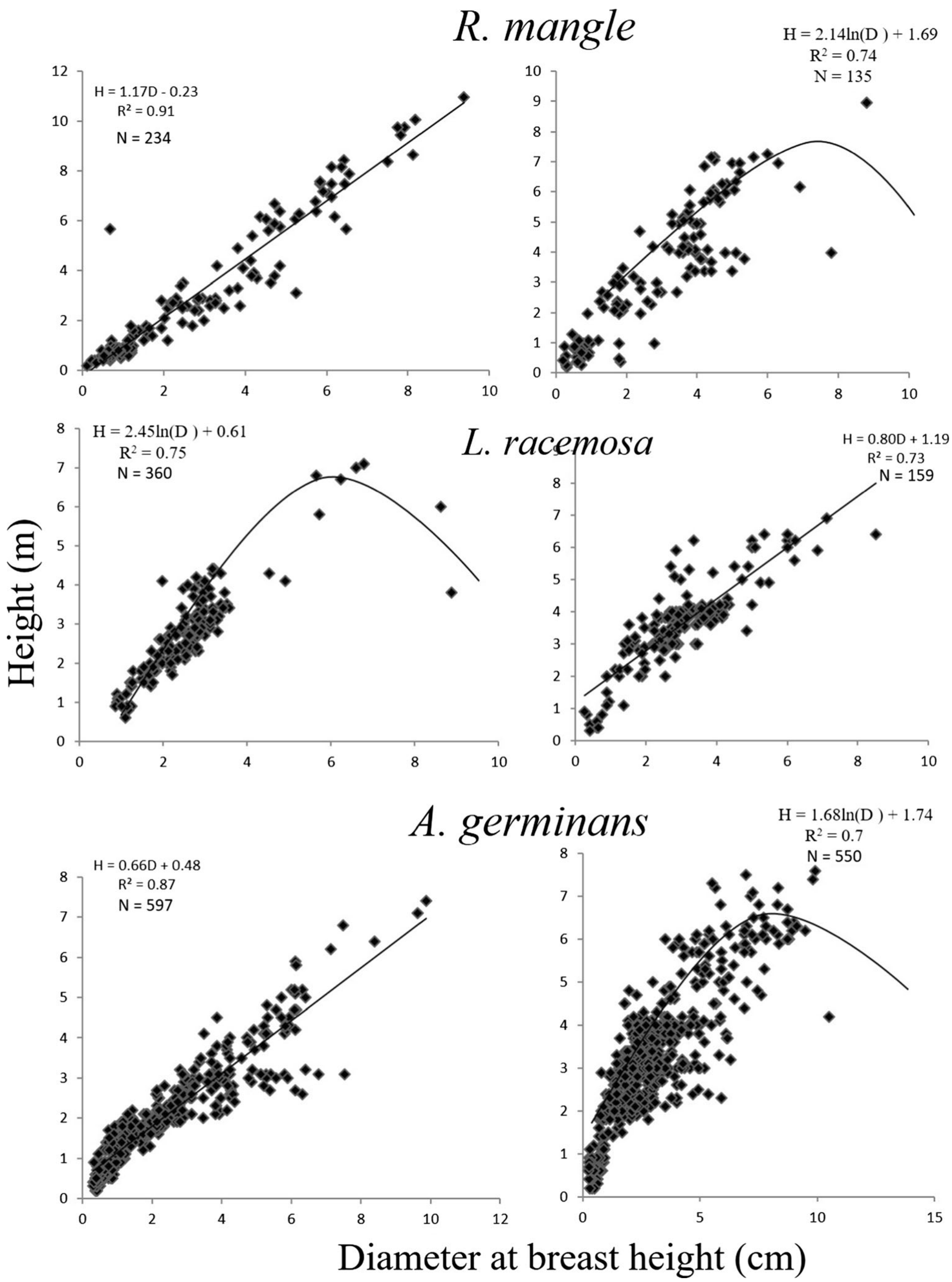

Fig. 2 Mangrove tree population species measured at the rural and urban wetlands depicted in the left and right charts respectively

racemosa and *Rhizophora harrisonii* are known to occur in Ghana, *Rhizophora mangle* will most likely be present.

During this study, three mangrove species, *Rhizophora mangle*, *Avicennia germinans* and *Laguncularia racemosa*, were identified to occur at both wetlands. Collected mangrove samples were taken through systematic identification procedures to confirm the *Rhizophora mangle* species occurence. *Rhizophora mangle* has mostly 0–3 inflorescence joints, are distinguished from *Rhizophora racemosa* and *R. harrisonii*, which have 3–8 inflorescence joints. For *Rhizophora mangle*,

Table 3 Regression equations developed during this study for estimating the biomass of mangrove species at the rural and an urban wetland in Ghana

Species	Rural	Urban
Rhizophora mangle	$y = 924.3e^{0.001\ (DH)}$	$y = -0.001(DH)^2 + 4.922(DH) + 158.3$
Laguncularia racemosa	$y = 287.9e^{0.001\ (DH)}$	$y = 2.707(DH) + 76.90$
Avicennia germinans	$y = 349.6e^{0.001\ (DH)}$	$y = 182.4(DH)^{0.441}$

Table 4 The relative values, importance value and standing biomass of mangrove tree species at the studied Rural and Urban wetland

Species	Relative Values (%)						Importance value Iv		Standing biomass (t/ha)	
	Frequency		Density		Dominance					
	Rural	Urban	Rural	Urban	Rural	Urban	Rural	Urban	Rural	Urban
Rhizophora. mangle	19.65	16	19.65	16.00	46.27	30.65	85.56	62.66	172.55	44.44
Laguncularia. racemosa	30.23	19	30.23	18.84	30.71	31.22	91.16	69.07	96.07	21.63
Avicennia. germinans	50.13	65	50.13	65.19	23.02	38.12	123.27	168.31	125.87	60.22
Total			100.0	100.0	100.0	100.0	299.99	300.04	394.49	126.29

Rhizophora harrisonii, and *Rhizophora racemosa*, mature buds and flowers are located at 1–2, 3–5, and 7–9 nodes down from the apical shoot, respectively.

The hybrid character of *Rhizophora harrisonii* is shown where it has characters intermediate between *Rhizophora racemosa* and *Rhizophora mangle*. Also, the leaves of *Rhizophora mangle* are opposite, simple, bright green, obovate, leathery with a curved surface. The margins revolute, with obtuse blunt apex, and a minute lip folded under (Duke and Allen 2006). Flowers are borne in axillary clusters, which have been characterized as simple cymes or as a modified dichasium. Mature buds and flowers are located at 1–2 nodes down from the apical shoot (Fig. 4). The calyx is typically waxy yellow to creamy white and green at maturity, with four lobes (Allen 1998). Buds elongate to ovate, green when immature to lighter colours as they mature. Dimensions of the buds are 1–2 cm long and about 0.5 cm wide (Duke and Allen 2006). The petals, usually four, are lanceolate to linear, creamy white, with woolly to sparsely hairy margins. The petals are about 12 mm long and 4 mm wide. Stamens number eight, pale yellow, to golden brown at maturity (Allen 1998). Style is pale green, filiform and 0.5–4 mm above ovary base; it is 1.5–3 mm wide, dichotomous tip, pale yellow. Bracts and bracteoles are distinct. Peduncle is 3–4 cm long, ~0.3 cm wide (Duke and Allen 2006). Hypocotyls are narrowly cylindrical, elongate, green, smooth with irregular small brown lenticels, distal half wider, distal tip pointed (Irvine 1961). *Rhizophora mangle* samples collected from the field, exhibited these characteristics and hence were indeed confirmed as *Rhizophora mangle.* On the basis of this identification, the occurrence of *Rhizophora mangle* in Ghana is confirmed contrary to earlier opposing opinions.

Table 5 Water column hydrographic conditions in the studied rural and urban wetlands in the Western region of Ghana

Parameters	Mean (± S.E)	
	Rural	Urban
*Temperature (°C)	21.15 ± 0.18	28.13 ± 0.39
*pH	6.76 ± 0.06	7.23 ± 0.07
Salinity (PSU)	26 ± 0.57	28.25 ± 0.57
*Dissolved Oxygen (mg/L)	5.21 ± 0.12	4.63 ± 0.16
*Turbidity (NTU)	29.27 ± 1.18	41.67 ± 0.95

*denotes significant difference between sites

Mangrove Forest Attributes and Biomass in the Urban and Rural Wetlands

The assessment of forest structure is considered very important for obtaining accurate information about the status of mangrove forests, assessing the yield of commercially important mangrove timber products and modelling their carbon sequestration potential (Soares and Schaeffer-Novelli 2005). *Avicennia germinans* was the principal dominant species found in both the rural and urban wetlands, as it recorded the highest importance value at both sites. Diameter at breast height and height of resident trees are useful measures when undertaking such forest status assessments (Gehring et al. 2008).

In the present study, the mean diameters at breast height for all species were 2.26 cm ± 0.05 and 3.06 cm ± 0.064 at the rural and urban wetland respectively. Mean heights of all mangrove species were 2.19 m ± 0.047 and 3.34 m ± 0.052 at the rural and urban wetland respectively. These parameters recorded suggest that the mangrove forests are of low structural development according to the classification of Pellengrini et al. (2009).

A similar assessment of a mangrove forest conducted by Aheto et al. (2011) at the Kakum River estuary in the Central Region of Ghana also identified a low structural development of that forest, suggesting that low structural developments of mangrove forest observed in the present study may not be limited to the Nyan and Whin wetlands alone. Despite the low structural development of mangrove stands at the current study sites, they were found to have strong correlation coefficients of height and diameter at breast heights of the trees that were sampled.

In this study, we used linear transformations and power curves with diameter at breast height as the independent variables to estimate the standing biomass for all three identified mangrove species at the two study areas. The best estimate of

Table 6 Occurrence, classification and size range of fish communities in the studied rural and urban wetlands in Ghana

Categories of species	Family	Species	Rural		Urban	
			No.	Length range (cm)	No.	Length range (cm)
Truly estuarine that spend their entire lives in estuary	Cichlidae	*Sarotherodon melanotheron*	159	4.7–15.3	147	4.5–13.7
	Grapsidae	*Goniopsis cruentata*	12	4.1–7.1	5	4.7–6.8
	Gobiidae	*Periopthalmus barbarus*	14	6.8–17.4	17	5.9–15.8
	Ocypodidae	*Uca tangeri*	14	4.5–6.2	10	3.2–5.7
Estuarine-marine species that use estuaries primarily as nursing and spawning grounds, spending adult life at sea and returning occasionally to estuaries	Mugilidae	*Liza falcipinis*	21	3.4–18.2	38	8–21.3
		Liza dumerilii	15	6.1–18.7	27	6.8–17.3
		Mugil cephalus	-	-	12	6.2–12.4
		Mugil curema	4	9.1–11.6	14	8.3–13.7
		Mugil bananensis	-	-	9	7.8–15.1
	Portunidae	*Callinectes amnicola*	11	3.8–9.2	9	4.6–8.7
	Clupeidae	*Sardinella aurita*	7	4.5–5.2	52	3.9–10.4
		Sardinella maderensis	-	-	22	4.6–6.9
	Acanthuridae	*Acanthurus monroviae*	-	-	1	6.8
	Penaeidae	*Penaeus notialis*	4	7.6–8.2	2	7.1–7.9
	Gerreidae	*Eucinostomus melanopterus*	5	9.2–11.8	12	8–12.3
		Gerres melanopterus	13	6.1–9.3	21	5.2–8.6
	Haemulidae	*Plectorhynhus mediterraneus*	2	4.1–4.3	8	4.2–5.1
	Carangidae	*Selene dorsalis*	12	7.5–13.1	6	9.5–10.4
		Caranx hippos	5	7.8–12.4	13	7.2–11.8
	Lutjanidae	*Lutjanus goreensis*	3	8.1–12.6	19	7.6–13.1
	Bothidae	*Scyacium micrurum*	1	6.8	7	6.3–7
		Citharichthys stampflii	-	-	5	6.9–8.3
	Serranidae	*Epinephelus aeneus*	1	12.6	3	9.6–15.3
	Gobiidae	*Bathygobius soporator*	-	-	3	8.7–11.2
Freshwater species that occasionally enter brackish water	Cichlidae	*Oreochromis niloticus*	6	5.7–9.7	-	-
	Eleotridae	*Eleotris senegalensis*	10	9.6–11.4	7	8.2–9.7
	Gecarcinidae	*Cardiosoma armatum*	3	5.6–7.4	7	6–7.9
Occasional visitors occurring irregularly (ornamental)	Labridae	*Thalassoma pavo*	-	-	2	12.4–15.7

total above ground biomass (R^2 range of 0.70–0.91 for all three species at both sites) was obtained by using the product of height and diameter at breast height as the independent variables.

The results support the findings of other studies that indicate that regression models developed for estimating biomass of natural mangrove stands in the Indo-Pacific and Asia are reliable. Regression models for mangrove are known to vary between species (Clough 1992) and hence the difference in models for mangrove species in this study is not unusual. It has been observed that even within the same species depending on conditions such as difference in localities, on site-specific factors such as tree density, location on the ground, whether it is a monoculture or mixed forest, and management practices, regression models for mangrove species may vary (Christensen 1978; Woodroffe 1985). The estimation of standing biomass is known to provide an indication of carbon allocation to plant tissues: an indispensable information when making local and regional carbon sequestration accounting (Kairo et al. 2008; Aheto et al. 2011). Such in situ estimation of biomass is also known to give a better and more accurate result than using satellite data (Hirata et al. 1998). Even though many studies have been carried out on biomass of upland forest and plantations in Africa (Brown 1997), biomass estimation of mangrove forests have rarely been done compared to other continents.

In this study, the above ground biomass of *Rhizophora mangle*, *L. racemosa* and *A. germinans* was estimated to be 44.44 t/ha, 21.63 t/ha and 60.22 t/ha respectively for the mangrove stands at the urban wetland. The rural wetland had estimates of 172.55 t/ha, 96.07 t/ha and 125.87 t/ha for *Rhizophora mangle*, *L. racemosa* and *A. germinans*

Table 7 Species richness, diversity, evenness and similarity in the studied rural and urban wetlands in the Western Region of Ghana

	Margalef index (d)	Shannon-Wiener (H′)	Pielou's evenness (J')	Sorensen's similarity (Cs)
Rural	3.46	2.09	0.67	0.83
Urban	4.21	2.64	0.8	

respectively. Overall, plant biomass was higher at the rural wetland probably because of the low exploitation and degradation over the past few decades compared to the high level of mangrove degradation within the urban wetland.

Fish Assemblage Structures in Urban Versus Rural Wetlands

The main indicator species responsible for the differences between the two wetlands are *Sarotherodon melanotheron*, *Sardinella aurita*, *Liza falcipinis*, *Sardinella maderensis*, *Liza dumerilii* and *Lutjanus goreensis*, since they account for more than 53 % of the observed differences between the urban and the rural wetland.

The urban wetland had a higher diversity of fish species and richness (d = 4.21, H′ = 2.64) as compared to the rural wetland (d = 3.46, H′ = 2.09). Low fish diversity is confirmed to be a good indicator of a possibly stressed ecosystem while the higher the fish diversity the more stable the fish community (Leveque 1995). At the urban wetland, surveys indicated less dependence on fish products by local residents who mostly had other sources of livelihoods. However, due to the location and the low economic levels of residents at the rural wetlands, we observed a high dependence of residents on fishing for their livelihoods. It is possible that high fishing pressure on the rural wetland contributed to their lower fish species diversity. Impacts of artisanal fishing on fish diversity have been reported by Tomlin and Kyle (1998).

Both the rural and urban wetlands recorded a high diversity of fish fauna dominated by marine species which are known to occasionally use coastal wetlands as nursery and spawning grounds (Nagelkerken et al. 2008). Truly estuarine species

Table 8 Similarity percentages-species contributions of fish assemblages within the studied Rural and Urban wetland in Ghana

Species	Average dissimilarity	Contribution %	Cumulative %	Mean abundance (Rural)	Mean abundance (Urban)
Sarotherodon melanotheron	7.164	16.08	16.08	22.7	21
Sardinella aurita	5.772	12.95	29.03	1	7.43
Liza falcipinis	3.723	8.354	37.38	3	5.43
Sardinella maderensis	2.731	6.128	43.51	0	3.14
Liza dumerilii	2.405	5.397	48.91	2.14	3.86
Lutjanus goreensis	2.055	4.612	53.52	0.429	2.71
Gerres melanopterus	1.729	3.881	57.4	1.86	3
Caranx hippos	1.663	3.732	61.13	0.714	2
Mugil cephalus	1.538	3.451	64.58	0	1.71
Goniopsis cruentata	1.454	3.262	67.84	2.29	0.714
Eucinostomus melanopterus	1.373	3.082	70.93	0.714	1.71
Mugil curema	1.365	3.062	73.99	0.571	2
Periopthalmus barbarus	1.258	2.823	76.81	2	2.43
Selene dorsalis	1.239	2.781	79.59	1.71	0.857
Mugil bananensis	1.21	2.715	82.31	0	1.29
Eleotris senegalensis	1.195	2.682	84.99	1.43	1
Plectorhynhus mediterraneus	0.8562	1.921	86.91	0.286	1.14
Cardiosoma armatum	0.8514	1.91	88.82	0.429	1
Uca tangeri	0.8271	1.856	90.68	2	1.43
Scyacium micrurum	0.7933	1.78	92.46	0.143	1
Oreochromis niloticus	0.6514	1.462	93.92	0.857	0
Callinectes amnicola	0.6005	1.348	95.27	1.57	1.29
Citharichthys stampflii	0.5482	1.23	96.5	0	0.714
Penaeus notialis	0.5453	1.224	97.72	0.571	0.286
Epinephelus aeneus	0.3593	0.8062	98.53	0.143	0.429
Bathygobius soporator	0.3314	0.7436	99.27	0	0.429
Thalassoma pavo	0.2168	0.4865	99.76	0	0.286
Acanthurus monroviae	0.1084	0.2433	100	0	0.143

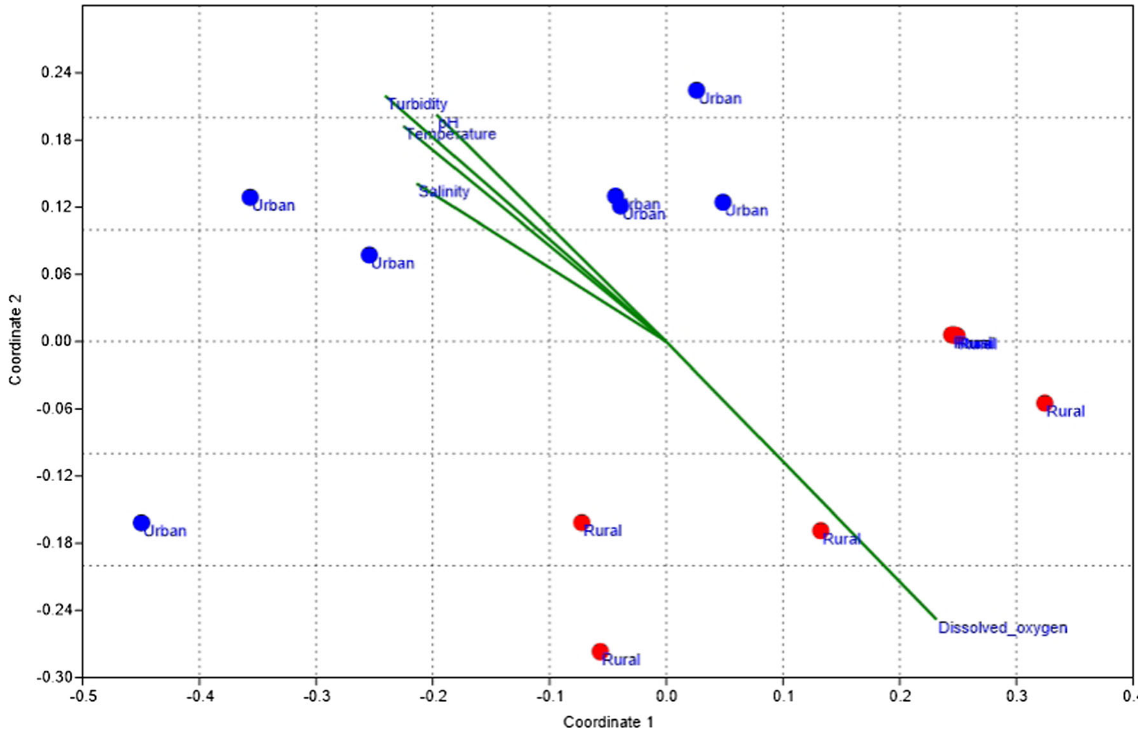

Fig. 3 Non-metric multidimensional scaling plot showing differences in fish assemblages and water quality conditions in the studied urban and rural wetlands Stress =0.096

recorded included *Periophthalmus barbarus*, *Sarotherodon melanotheron* and *Uca tangeri*. Only one freshwater species, *Eleotris senegalensis*, was found at the urban wetland (Whin) confirming the observation made by Okyere et al. (2011) in their assessment of the same wetland. However, an additional freshwater species, *Oreochromis niloticus*, was found at the rural wetland (Nyan). Ornamental fish species namely *Thalassoma pavo* was also identified in both wetlands.

Despite the high diversity of fish species encountered, the size ranges of the most common marine fishes such as *Liza falcipinnis*, *Eucinostomus melanopterus*, *Liza dumerilii*, *Mugil curema* (Mugilidae), *Lutjanus goreensis* (Lutjanidae), *Sardinella maderensis* (Clupeidae) were generally smaller than reported normal maturity sizes (Schneider 1990; Paugy et al. 2003), and were identified to be mostly juvenile fishes. This suggests that these wetlands are nursery and feeding grounds for these commercially important marine fish species. These findings corroborate with studies published by Okyere et al. (2012) and Aheto et al. (2014) for a different mangrove wetland in Ghana.

Implication of Water Quality Conditions

Salinity is a major factor determining the composition, diversity and abundance of flora and fauna in coastal water bodies in Ghana (Twilley and Chen 1998; Gordon 2000; Lamptey and Armah 2008). Changes in salinity are normally controlled by climate, hydrology, rainfall, topography and tidal flooding. Very high levels of salinity is known to reduce mangrove biomass (Naidoo 1990) and could denature terminal buds in

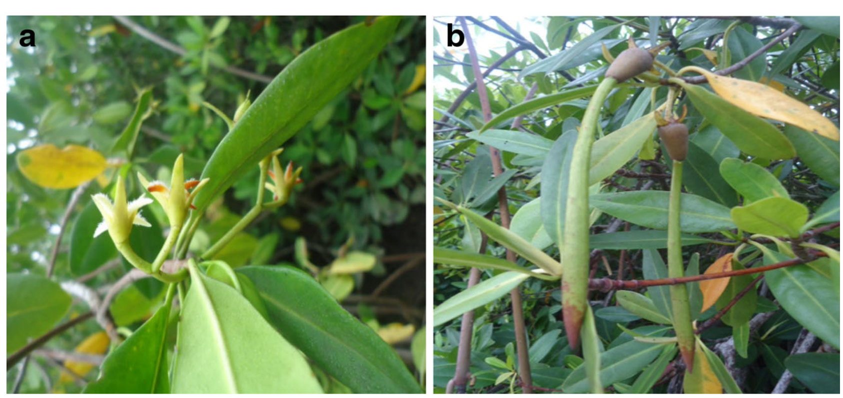

Fig. 4 *Rhizophora mangle*: **a** downwardly curved whitish petals with bell-shaped, leathery, persistent, pale yellow sepals; **b** propagule showing elongated hypocotyl with distinctive brown distal ending

Rhizophora mangle seedlings (Koch and Snedaker 1997). However, low salinity, associated with long periods of flooding, contributes to mangrove degradation through reduced cell turgidity and decreased respiration. The mean values recorded for rural and urban wetlands were 26 ± 0.57 ‰ and 28.25 ± 0.57 ‰ respectively, which may have accounted for the lower biomass in the urban environment.

Dissolved oxygen (DO) is one of the most important abiotic parameters influencing the life of organisms in the coastal environment. Normally high dissolved oxygen is encountered in unpolluted areas, while in polluted areas, levels of DO is very low. Depletion of DO is a critical manifestation of pollution (Lester 1975). DO levels in the rural and urban systems were above 5 mg/l, the threshold required to support aquatic life (Hynes 1970). This observation suggests that as far as DO is concerned, the estuaries are suitable for fisheries. Higher dissolved oxygen concentration observed might be due to the cumulative effect of wind velocity coupled with heavy rainfall and the resultant freshwater mixing (Das et al. 1997; Saravanakumar et al. 2007). Within the urban environment, mean DO was lower compared to the rural environment. This may be attributed to high organic matter content in the urban wetland.

A pH range of 6.5 to 8.5 is generally suitable for growth of aquatic organisms (Pillay 2004). Generally, fluctuations in pH values is attributed to factors like removal of CO_2 by photosynthesis through bicarbonate degradation, dilution of seawater by freshwater influx, reduction of salinity and temperature and decomposition of organic matter (Upadhyay 1988; Rajasegar 2003). The pH of urban wetland was estimated at 7.23 ± 0.07 whereas that of the rural wetland was 6.67 ± 0.06. The urban wetland was more turbid than the rural wetland which could be linked to more intense human activities around the urban area.

High temperature within the urban system could be attributed to the influence of higher sedimentation due to impact of human activities such as upstream sand winning. The impacts of such temperatures should be of concern because aquatic organisms are dependent on certain temperature ranges for optimal growth and survival. Temperature affects many other parameters in the water, including the amount of dissolved oxygen available, the types of plants and animals present, and the susceptibility of organisms to parasites, pollution and disease (Govindasamy et al. 2000). Generally, surface water temperature is influenced by the intensity of solar radiation, evaporation, insolation, freshwater influx and cooling and mix up with ebb and flow from adjoining neritic waters (Govindasamy et al. 2000). Mangrove leaves are sensitive to high temperature and their photosynthetic capacity gets reduced, falling to zero at leaf temperatures of 38 °C − 40 °C, as against the optimum leaf temperature for photosynthesis which is 28 °C − 32 °C (Clough et al. 1982; Andrews et al. 1984) even though that was not investigated, the temperatures recorded for both systems may not have adverse effect on the mangroves.

Conclusions

From this study, three mangrove species that commonly occur along the West African coastline were encountered but in relatively degraded state. Contrary to some earlier reports, the occurrence of *Rhizophora mangle* can also be confirm through the findings of this study.

Clearly, destructive human uses of mangrove wetland resources may have wider ecological and economic implications, including impacts on fishery and other ecosystem roles played by mangroves. It is clear from the study that a well-developed mangrove system with high mangrove biomass may not necessarily lead to or imply high fish population assemblages. Other context specific factors including impacts of fishing pressure must be keenly monitored in rural coastal environments to determine overall ecological health of mangrove wetland systems in those places.

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