

Qualitative and Quantitative Characterization of Mangrove Vegetation Structure and Dynamics in a Peri-urban Setting of Douala (Cameroon): An Approach Using Air-Borne Imagery

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Abstract Qualitative and quantitative characterization of mangrove vegetation structure and dynamics is required for assessment of coastal habitat vulnerability. Changes in mangrove forests around Douala, Cameroon, have been documented using aerial photography between 1974 and 2009. The distribution pattern of tree species was also assessed in 2009 following the point-centered quarter method (PCQM+) protocol. Pristine mangroves observed in 1974 had been disturbed markedly in 2003 and 2009. Some of the pre-existing mangroves were entirely replaced by settlements, road, and crops (maize, bean, banana, oil palm, green vegetables, and sugar cane plantations). From 1974 to 2003, 39.86 % of mangrove forests have disappeared; the net loss of 22.10 % occurred between 2003 and 2009 alone. Mangrove forest area had decreased 53.16 % around Douala over a 35-year period from 1974 to 2009 concurrent with a substantial increase of settlements (60 %), roads (233.33 %), agriculture areas (16 %), non-mangrove areas (193.33 %), and open water (152.94 %). Field survey showed that almost one third of the quadrants in the remaining mangrove forest were empty. The disrupted

mangrove forest has an overall mean height, absolute density, and basal area of 19.80 m, 158 trees ha⁻¹, and 110.44 m² ha⁻¹, respectively. In comparison with scientific literature on mangrove degradation, this puts the mangroves around Douala at the top of the “peri-urban mangrove degradation” list. In addition, beyond listing of mangrove plants on the Red List of Threatened Species which will seldom lead to widely distributed species being listed, we call for the creation of a Red List of Locally Threatened Ecosystems, which in contrast is likely to list mangroves as an ecosystem under critical risk of (local) extinction in many countries around the globe, in particular, peri-urban sites.

Keywords Peri-urban mangrove · Aerial photography · Remote sensing · Forest loss · Wouri estuary · Douala (Cameroon)

Introduction

Mangrove forests are experiencing long-term and severe decline (Valiela et al. 2001; Alongi 2002). The rate of deforestation is high in many developing countries (Duke et al. 2007; Giri et al. 2007), possibly higher than in any other type of tropical forests (Rakotomavo and Fromard 2010). The causes of such losses include not only natural disasters (Dahdouh-Guebas and Koedam 2006a; Mukherjee et al. 2010) and sea-level rise (Di Nitto et al. 2008; Shearman 2010), but also coastal land-use development leading to losses due to over-harvesting (Walters et al. 2008; Nfotabong-Atheull et al. 2009), expanded agriculture (Hossain et al. 2009), and conversion into shrimp farming ponds (Guimarães et al. 2010), to name but a few.

Although moderate natural expansion of mangrove has been reported in certain parts of the world (McKee 2004;

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Chimner et al. 2006; Eslami-Andargoli et al. 2009; Everitt et al. 2010), this phenomenon is relatively rare in Africa (FAO 2007) where mangrove deforestation and degradation still appears to be dominant (Dahdouh-Guebas et al. 2004; UNEP 2007; Mmom and Arokoyu 2010; Rakotomavo and Fromard 2010). It has been estimated that Cameroon lost approximately 22,000 ha of mangroves between 1980 and 2005 (Food and Agriculture Organization (FAO) 2007; United Nations Environment Programme (UNEP) 2007), although the estimate is based on figures taken from several reports without a detailed explanation of the underlying data or the precision of the methods, such as changes in statistical design or new data used in the interpretation (Grainger 2008).

According to the World Mangrove Atlas (Spalding et al. 1997) and the World Atlas of Mangroves (Spalding et al. 2010), almost 53,216 ha of the Cameroon's mangrove forests have been lost over the last 13 years. This estimation may likely have considerable margin errors because of differences in reporting or assessment techniques. For example, Cameroon's phytogeographical map of scale 1:500,000 was used in the World Mangrove Atlas (Spalding et al. 1997) whereas Landsat (Thematic Mapper 5 and 7) images with spatial resolution of 30 m were employed in the World Atlas of Mangroves (Spalding et al. 2010). Mangrove areas are not mapped at very high level of resolution in either case (i.e., 1 m pixel⁻¹ or less). As a result, some smaller stands of mangrove like those located in the Southern part of the Cameroon's coastline may have been missed (Dahdouh-Guebas 2011). Better, more standardized, estimates of ecosystem loss are needed to provide the foundation for even better conservation policy (Friess and Webb 2011). In particular, temporal and spatial changes of mangrove forests available in Cameroon should be critically reassessed. In the mean time, however, the online Mangrove Reference Database and Herbarium (Massó i Alemán et al. 2010), which includes data from Spalding et al. (2010) and Giri et al. (2011) as well as more than 10,000 records of mangrove distribution, is the most complete database available (Appeltans et al. 2012).

Disturbances in mangroves have been attributed to a combination of factors such as the ever-increasing human pressure on coastal lands, sea-level rise, absence of adequate legislation regarding mangrove protection, and pollution in the peri-urban setting of Douala (Cameroon) (Din et al. 2008; Nfotabong-Atheull et al. 2009, 2011; Ellison and Zouh 2012). To discriminate among such disparate stressors, successive field sampling to document vegetation dynamics on a monthly, seasonal, and annual is required, and employs field monitoring techniques adapted to the muddy substrate and entangling root systems in mangroves. As an alternative or as a combinatory method, the local ecological knowledge and experience of local people can be used for reconstructing changes that have occurred in their surrounding environment

(Kovacs 2000; Dahdouh-Guebas et al. 2005b; Nfotabong-Atheull et al. 2011; Satyanarayana et al. 2012), but their perception of changes should be verified, to the extent possible, with remotely sensed quantitative data. These techniques provide a timely, cost-effective means to obtain reliable information over less accessible areas such as mangroves (Dahdouh-Guebas et al. 2000; Verheyden et al. 2002; Krause et al. 2004; Satapathy et al. 2007; Dahdouh-Guebas and Koedam 2008; Everitt et al. 2010; Shearman 2010; Heumann 2011).

In this paper, we used three diachronic panchromatic aerial photographs coupled with extensive ground surveys to accurately map and estimate mangrove vegetation changes around Douala, Cameroon. By doing so, we also characterized the mangrove vegetation structure. This peri-urban area was particularly interesting because of the influence of anthropogenic activities on mangrove vegetation structure and dynamics.

Material and Methods

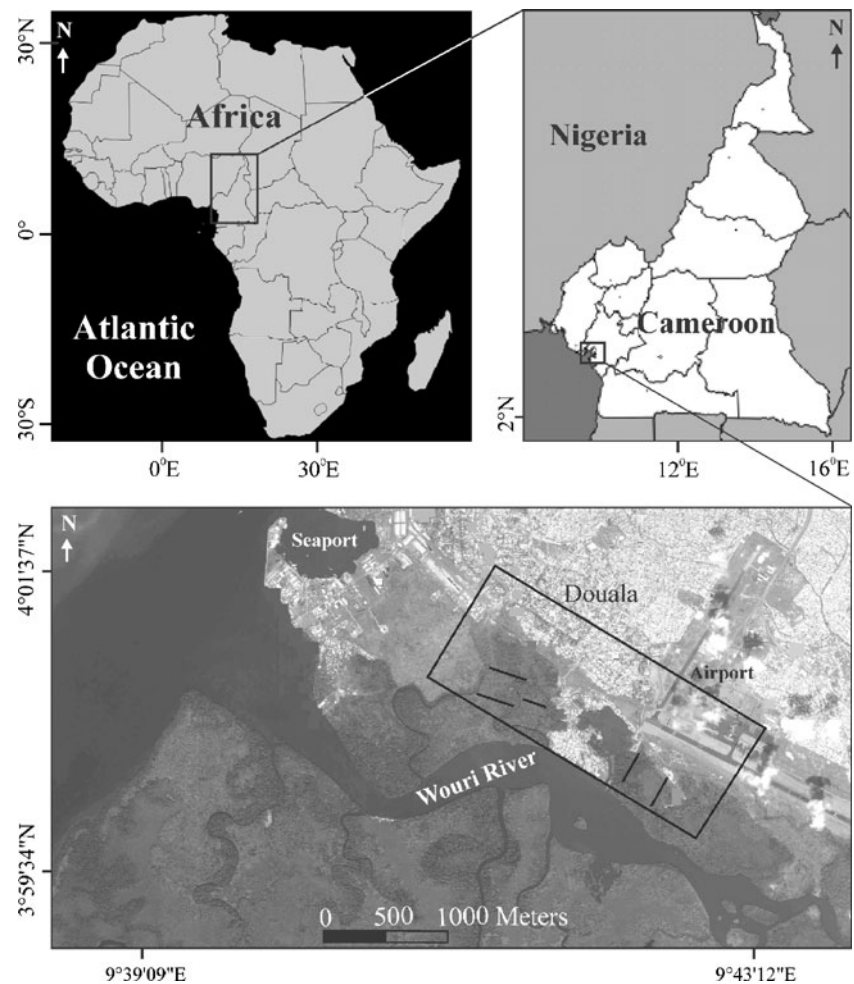
Site Description

The study site is located near the major city of Douala, Cameroon, adjacent to the airport, seaport, and several settlements (Fig. 1). It covers approximately 5.8 km², representing about 0.53 % of the mangrove forest in the Cameroon estuary and about 0.15 % of the total mangrove area in Cameroon. The region is characterized by 9 months of precipitation, and the mean temperature is 26.5 °C. The tides are semi-diurnal with amplitudes ranging from 1.35 to 3 m. Douala is the first most populous city in Cameroon, with a population that grew rapidly from 1,352,833 inhabitants in 1987 to 2,510,263 people in 2005 (Cameroon National Institute of Statistics 2009). The majority of coastal communities are dependent on the surrounding mangroves for both subsistence and commercial uses (Din et al. 2008; Nfotabong-Atheull et al. 2009, 2011). Here, mangroves are under severe pressure from housing development, agriculture, sand quarrying, road construction, lumber harvesting, and sea-port maintenance and expansion.

Field Data Collection

The point-centered quarter method (PCQM+) (Dahdouh-Guebas and Koedam 2006b) in combination with aerial photographs has been used to produce accurate statistics of species assemblages (Verheyden et al. 2002) and describe the vegetation quantitatively (Dahdouh-Guebas et al. 2004). It has also been combined with space-borne imageries for mangrove species mapping and stand delineation and labeling (Neukermans et al. 2008), producing land-use/cover classification and Normalized Differential Vegetation Index

Fig. 1 Map showing the location of the study area and transect lines



mapping (Satyanarayana et al. 2011a) and testing the predictions made 10 years ago concerning the evolution of a mangrove forest (Satyanarayana et al. 2011b). In the present combinatory investigation, mangrove vegetation cover and structure were assessed through aerial photographs and ground-truth observations.

Sampling of the vegetation was carried out between January and February 2009 following the (PCQM+) protocol. The first sampling point near the water edge corresponded to the center of four 10 m×10 m quadrants. The distance between each sampling point and the nearest tree in each quadrant was measured following a clockwise direction. We used, however, a limit of 10 m in which a tree needed to be found, otherwise we classified the quadrant as “empty” (Dahdouh-Guebas and Koedam 2006b). Only trees with a stem diameter larger than 2.5 cm were identified and sampled (cf. Kairo et al. 2002). The diameter and height of each sampled tree were measured using a measuring tape and a hypsometer, respectively. The same methodological procedure was applied for five different transects established perpendicular to the main creeks. Additional field checks (visual observation) were made between

March and July 2010 to characterize mangrove changes qualitatively.

Calculation of Structural Attributes

We used the formula proposed by Warde and Petranka (1981) and also developed by Dahdouh-Guebas and Koedam (2006b) and Mitchell (2007).

$$\text{Absolute density (number of trees ha}^{-1}\text{)} : \bar{\lambda}_c = \frac{(4n - n_0)^2}{\left(\sum_{m=1}^{4n-n_0} R_m\right)^2} \times CF$$

where n is the number of sampling point, $4n$ is the number of sampling quadrants, n_0 is the number of empty quadrants, and R_m is the distance from tree m to its corresponding sampling point. A correction factor was used when calculating the absolute density of the whole mangrove forest to account for empty quadrants (within the mangrove forest) during our field campaign. As a result, an absolute density calculated from only those quarters containing trees will overestimate the true density. Different values of CF corresponding to the

proportion of empty quadrants ($n_0/4n$) were assigned from Table 1 of Warde and Petranka (1981). For each species, k , the absolute density ($\bar{\lambda}_k$), basal area, absolute frequency,

relative density, relative frequency relative basal area, and importance value of each species as well as the complexity index of the residual mangrove forests were calculated as:

$$\text{Absolute density (number of trees ha}^{-1}\text{)} : \bar{\lambda}_k = \frac{\text{Number of quadrants with species } k}{4n} \times \bar{\lambda}_c$$

$$\text{Absolute basal area (m}^2\text{ha}^{-1}\text{)} : = \frac{\pi \times \text{diameter}^2}{4}$$

$$\text{Absolute frequency (\%)} := \frac{\text{Number of } n \text{ with species } k}{\text{Total number of } n} \times 100$$

$$\text{Relative density (\%)} := \frac{\text{Number of quadrants with species } k}{4n} \times 100$$

$$\text{Relative basal area or relative dominance (\%)} := \frac{\text{Total basal area of species } k}{\text{Total basal area of all species}} \times 100$$

$$\text{Relative frequency (\%)} : = \frac{\text{Absolute frequency of species } k}{\text{Total frequency of all species}} \times 100$$

$$\text{Importance value (unitless)} : = \text{Relative density} + \text{Relative frequency} + \text{Relative basal area}$$

$$\text{Complexity index or C.I. (unitless)} = \text{Number of species} \times \text{Mean height} \times \text{Mean density} \times \text{mean basal area} \times 10^{-5}$$

Air-Borne Imagery Analysis

Panchromatic aerial photographs were obtained from the Cameroon's National Center of Cartography and the Douala Urban Council. These photos were acquired for 1974, 2003, and 2009 at a nominal scale of 1:10,000, 1:8,000 and 1:12,000, respectively. An airplane equipped with a camera (focal length of 152.81 mm) was used to obtain the 1974, 2003, and 2009 photographs along east–west flight lines at an altitude of about 3,400, 3,200, and 3,820 m, respectively. The overlap between two neighbouring aerial photos ranged from 20 to 25 %. The aerial photographs were taken at low tide in 1974 and 2009 whereas the 2003 aerial photograph was obtained at high tide.

The aerial photographs were scanned at 600 dots per inch with a pixel resolution of 0.45, 0.5, and 0.60 m for 1974, 2003, and 2009, respectively. Different pixel resolutions were used to help overcome differences in scale between the three sets of aerial photographs. The images were imported into ArcGIS (ESRI Inc. version 9.3), assembled in a mosaic and georeferenced to UTM Zone 32N WGS 84 projection using topographic maps obtained from the Cameroon National Center of Cartography and the Douala Urban Council. A digital map of scale 1:60,000 were used for the rectification of the 1974 photograph. Digital copies of the 1:40,000 topographic maps provided by the Douala Urban Council were used to rectify the 2003 and 2009 photographs. A minimum of 12 ground control points present in both the topographic map and aerial photograph and a second-order polynomial transform were used to georectify the digital aerial photographs. The root mean square error for all geo-registered images was less than 0.3 m.

We found it to be impossible to accurately map the areal extent of different species of *Rhizophora* because, contrary to some space-borne imagery (Dahdouh-Guebas et al. 2005a),

they were not visually distinguishable on the black and white photographs. Because *Rhizophora* spp. has been described as the most abundant mangrove species in the Cameroon estuary (Van Campo and Bengo 2004; Din et al. 2008; Nfotabong-Atheull et al. 2009), we judged it relevant to combine it with all little represented species (i.e., *Rhizophora racemosa* Meyer, *Rhizophora harrisonii* L., *Avicennia germinans* (L.) Stearn, *Laguncularia racemosa* (L.) Gaertn. f., *Conocarpus erectus* L., and the mangrove-associate *Guibourtia demensei* (Harms)) growing in our study area into the common label “mangrove” (see Chimner et al. 2006).

Land features were differentiated based on brightness (light, medium, dark) of objects on aerial photographs; mangroves had light grey to dark grey tones whereas settlements as well as roads and sand quarries had white tonal signatures. Agricultural lands and other vegetation types (grassland, fallow land, ornamental woody species, and terrestrial trees) have medium-grey tone; bare soil has a whitish-gray tone. Runway (airport area) had a darker color whereas water displayed a grey appearance. As a matter of fact, anthropogenic land cover classes could also be well-distinguished based on other image attributes such as shape (cf. Dahdouh-Guebas et al. 2006).

The mangrove cover as well as open water areas and other types of land uses were mapped through on-screen digitization. During high tide, the disturbed mangrove stands on the 2003 aerial photograph appeared as open water. At first glance, one may think that these disrupted and inundated stands had already started to rejuvenate. In such case, the juvenile or young vegetation should likely appear on the 2009 aerial photograph that was taken during low tide. Instead, after a visual analysis of this aerial photograph, we noticed that the disrupted mangrove stands appeared as bare soil. As it was impossible to manually map the boundaries between water and bare soil on the 2003

aerial photograph, we digitized the disrupted mangrove stands with and without water as a single class called “water.”

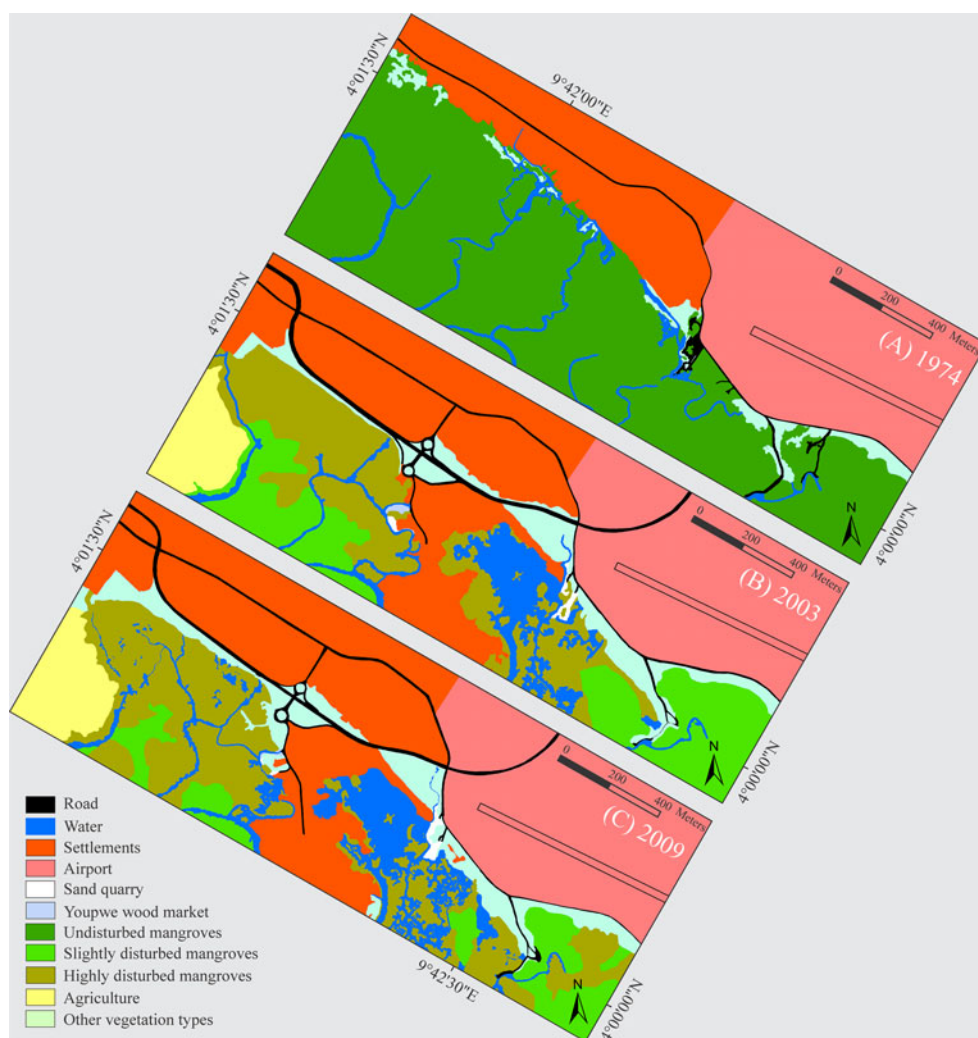
Mangroves habitats were delineated manually by digitizing the outer edge of the canopy based on visual contrasts and other image attributes (*loc. cit.*), referenced field data and information from Nfotabong-Atheull et al. (2011). A total of 12 ground truth points were positioned on the field. With regard to the ground reference data, random points placed across each vegetation type were geo-registered (using a hand held GPS (Global Positioning System Garmin etrex Vista HCx GPS)), and those features were projected over the vegetation map of 2009. Each remotely sensed vegetation type was then checked using sample points and field observations. Overall accuracy of commission/omission and Kappa index of agreement for the land/use cover map of 2009 were calculated. Accuracy assessment for the maps of 1974 and 2003 were unfortunately not possible because no field data were available. In some cases, boundaries between mangroves and other terrestrial forest types remain somewhat uncertain because the stands were inaccessible and could not be mapped by GPS in the field. There was not sufficient information to quantify any associated errors.

The main issues in delineating mangrove forest occurred when separating mangrove from non-mangrove vegetation in the 1974 and 2003 images. This was achieved by examining the position of the vegetation in the estuary (i.e., was it in a potential internal area of the estuary, or in a place near the water edge) and using slight differences in vertical vegetation structure (mangrove versus agriculture, grassland and fallow land) and also in grey tone (mangrove versus agriculture and other vegetation types) (cf. Benfield et al. 2005).

Delineation of agricultural land on the 2003 aerial photograph was partly assisted by the fact that the area had undergone land-use activities as it contrasted with the pristine mangrove stands (in 1974) and displayed similar brightness like what observed on the 2009 photograph.

Mangrove forest types were classified based on the crown shape, crown size, the spatial distribution of artificial gaps, and degree of crown aggregation. We characterized mangroves with closed canopy as “undisturbed forests”; mangroves that displayed sparse, smaller gaps we characterized as “less disturbed” and those that exhibited frequent, large gaps as “much disrupted.” After digitizing all aforesaid

Fig. 2 Land use/cover map around Douala (Cameroon) at **a** 1974, **b** 2003, and **c** 2009



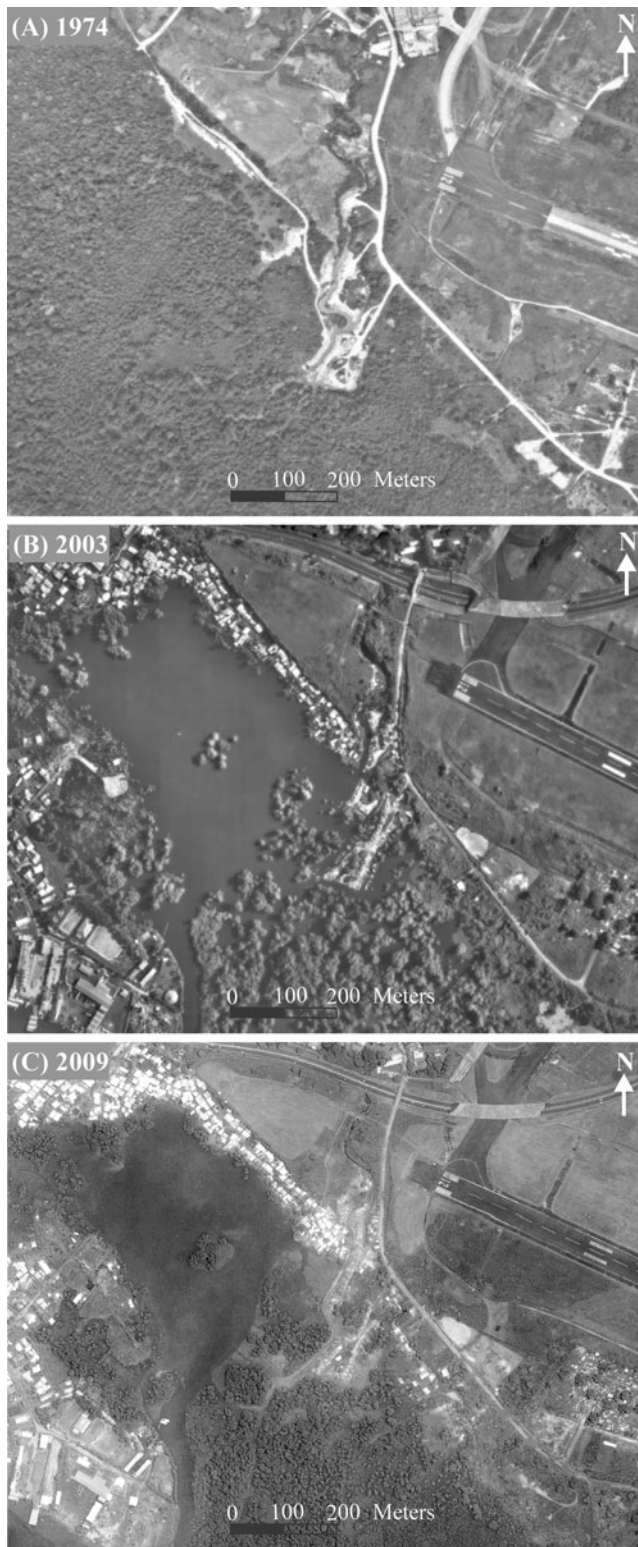


Fig. 3 Three time series photographs of land use/cover patterns illustrating mangrove vegetation structure dynamics in **a** 1974, **b** 2003, and **c** 2009. Notice that the 1974 and 2003 photographs were obtained during high tide whereas the acquisition period of the 2009 photograph coincided with the low tide. The newly large open water area occurring in the 2003 and 2009 photographs (**b** and **c**) is used by the local inhabitants for sand and gravel extraction

land features, we calculated the rate of changes in mangrove cover from 1974 to 2009 using the ArcGIS intersect function. The annual means of mangrove loss between the two sampling periods (1974–2003 and 2003–2009) were compared using the Student's *t* test.

Results

Qualitative Changes in Mangrove Vegetation

Over a span of 35 years, extensive pristine mangrove, with closed canopy, observed in the 1974 photograph appeared as small sparse tree patches (strongly disturbed mangroves) or forest with few artificial canopy gaps (less disturbed mangroves) in the 2003 and 2009 imagery (Figs. 2 and 3). Some clear-felled mangrove areas were occupied by settlements or covered by the brackish water during the high tide.

Generally, settlements are located in the intertidal zone at higher elevation where inundation is less frequent (dyke establishment often modified the water flow). Other noticeable causes of qualitative change in mangrove structure were also masked by conversion to agricultural ponds in some places (Fig. 4). Other peri-urban mangrove forests have been cleared by bulldozers to make roadways. Mixed vegetation was present on all three time series photographs (Fig. 2).

In 2009, the introduced mangrove palm *Nypa fruticans* Thunb. Wurm. was found growing in large open areas and along the muddy river front. This species was also found in the field under the canopy of less degraded mangrove forests, places where it had not been possible to accurately delineate its distribution on the airborne imagery. Mixed vegetative stands consisting of *Hibiscus tiliaceus* L., *Acrostichum aureum* L., *Ormocarpum verrucosum* P. Beauv., *Phoenix reclinata* Jacq., *Drepanocarpus lunatus* G. F. Meyer, *Raphia palmapinus* (Gaertn.) Hutch., *Cynometra mannii* Oliv., *Pandanus candelabrum* P. Beauv., and *Dalbergia ecastaphyllum* (L.) Taub., all of which are mangrove associates, were sometimes observed along the less muddy shorefronts.

Quantitative Changes in Mangrove Vegetation

The overall accuracy of the land use/cover map of 2009 was above 90 % with an overall Kappa index of 0.87. A trend of decrease in mangrove area and increase in human settlements was observed between 1974 and 2009 in our study site. In 1974, mangrove forests covered 51.89 % (3.01 km²) of the study site, whereas in 2003 it was 31.20 % (1.81 km²), indicating a decrease of 39.86 % over the 29-year period (Table 1).

In the 2009 photograph, mangroves represented 24.29 % (1.41 km²) of the study site, a decrease of 22.10 % to that of 2003. For the overall 35-year period from 1974 to 2009, the decrease in mangrove cover was of about 53.16 %.



Fig. 4 Conversion of mangrove stands into agriculture areas around Douala (Cameroon). On the *left side* of the photo, one could observe a remnant tree of *Rhizophora* spp. Behind this tree, there are two farmers working within a plantation of maize, bean, banana, oil palm, green vegetables, and sugar cane. The level of salinity is moderate because of

the landward position of plantations and the high rate of water dilution throughout the year (at least 9 months of wet season). These plants might likely be affected by prolonged exposure to extreme acidity and toxicity due to accumulation of ions such as Fe^{2+} , S^{2-} , Mn^{2+} , etc

Between 2003 and 2009, the coverage of less disturbed mangrove forests decreased from 0.80 to 0.29 km² representing a decrease of 63.75 % over the 6-year period (Table 2). In contrast, the rate of change in large disturbed mangrove extend after 2003 ranged from 55.80 % to 79.43 %, amounting to an increase of 10.89 % between 2003 and 2009. In 1974, settlements made up 18.11 % (1.05 km²) of the study site, whereas in 2009 they extended to 28.94 % (1.68 km²) with an increase of 60 % over the 35-year period. The increase in areas occupied by water, road, agricultural land, sand quarry, wood market, and other vegetation types (mainly composed of *Annona glabra* L., *D. lunatus*, *Paspalum vaginatum* Sw., *Anthocleista vogelii* Planch., *Sesuvium portulacastrum* L., and *Alchornea cordifolia* Müll. Arg.) is shown in Table 1. The unbalanced sampling intervals may bias the comparison of mangrove loss between the two sampling periods (1974–2003 and 2003–2009). However, there was no significant difference between study intervals before and after 2003 in terms of

annual means (1.37 % and 3.68 %, respectively) of mangrove loss ($t=-0.44$; d.f.=1; $p>0.05$).

Current State of the Mangrove Forest

During the field survey in 2009, 30.31 % of the total sampled quadrants were found without mangrove trees. We recorded five native true mangrove species (*R. racemosa*, *R. harrisonii*, *A. germinans*, *L. racemosa*, and *C. erectus*), along with one mangrove associated species (*G. demensei*) and one introduced mangrove species (*N. fruticans*) in the study area. The mangrove forest had a mean height, absolute density, and basal area of 19.80 m, 158 trees ha⁻¹ and 110.44 m² ha⁻¹, respectively. The disrupted ecosystem was more complex than other sectors (C.I.=20.78) in terms of the number of species growing in the area. The structural characteristics of each mangrove species is summarized in Table 3. Based on the importance value (I.V.), *R. racemosa* was most abundant followed by *A. germinans*, *R.*

Table 1 Land use/cover changes derived from 1974, 2003, and 2009 aerial photographs of Douala (Cameroon) peri-urban setting

Land use/cover	Area at 1974		Area at 2003		Area at 2009		Change 1974–2003 (%)	Change 2003–2009 (%)	Total change from 1974 to 2009 (%)
	(km ²)	(%)	(km ²)	(%)	(km ²)	(%)			
Mangroves	3.01	51.89	1.81	31.20	1.41	24.29	-39.86	-22.1	-53.16
Settlements	1.05	18.11	1.57	27.06	1.68	28.94	49.52	7	60
Agriculture	–	–	0.25	4.31	0.29	4.99	–	16	–
Other vegetation types	0.15	2.59	0.34	5.86	0.43	7.41	126.66	29.41	193.33
Water	0.17	2.93	0.33	5.69	0.43	7.41	94.11	30.3	152.94
Road	0.06	1.03	0.15	2.58	0.2	3.44	150	33.33	233.33
Sand quarry	–	–	0.01	0.17	0.01	0.17	–	0	–
Youpwe wood market	–	–	0.002	0.03	0.006	0.10	–	66.67	–
Airport	1.36	23	1.34	23	1.35	23	-1.47	0	0

Table 2 Area and rate of changes of different mangrove types between 2003 and 2009 in a peri-urban setting of Douala (Cameroon)

Land use/cover	Area at 2003		Area at 2009		Change 2003–2009 (%)
	(km ²)	(%)	(km ²)	(%)	
Mangroves					
Slightly disturbed	0.80	44.20	0.29	20.57	−63.75
Highly disturbed	1.01	55.80	1.12	79.43	10.89

harrisonii, and *G. demensei* (Table 3). *G. demensei* was often found on elevated ground behind a population of true mangrove species.

During field inspections between March and July 2010, significant changes in mangrove vegetation structure were observed in the study area. The large disturbed mangrove forests found in 2009, which were easily accessible by foot, had been progressively clear-cut from the landward margin toward the main water channel (Fig. 5). The unvegetated areas were subsequently developed for housing.

In the sites only accessible by boat, the lower stratum of the highly disturbed stands consisted of juvenile *Rhizophora* propagules that grew among cut stumps, branches, and trunks lying on forest floor. Young *Rhizophora* trees were found growing in some old, large, disrupted mangrove stands. Slightly disturbed mangrove sites were composed either of monospecific stands of mature *Rhizophora* spp. or of mixed population of *Rhizophora* spp. and *A. germinans*. The upper stratum of these mixed formations was composed of *A. germinans* (adult trees) whereas the middle stratum was made up of youngest individuals of *Rhizophora* spp. and *A. germinans*. In such stands, juvenile trees were less frequent.

Discussion

Decline in Mangrove Vegetation Cover

The present study has been carried out in an area where several anthropogenic factors (e.g., over-harvesting, anarchic

urbanization, agriculture, industrial pollution, sand extraction) are responsible for a decrease in the mangrove cover. In fact, we have documented the effect of each anthropogenic land use in the reduction of mangrove cover around Douala both qualitatively and quantitatively. Although our study has been carried out on a discrete location, these results are similar to those reported for other peri-urban setting (e.g., Mohamed et al. 2009; Guimarães et al. 2010; Nfotabong-Atheull et al. 2011). To ensure that future generations enjoy the ecosystem services provided by these valuable natural ecosystems, there is an urgent need to detect, protect, restore degraded tidal forests (Duke et al. 2007), and preserve remaining natural mangrove systems.

Despite mangrove forests in Cameroon being legally protected since 1996 (Frame-law n°96/012 relative to management of environment in Cameroon), there was an effective loss of 53.16 % of mangrove forests in the peri-urban settings of Douala in the 35-year period ending in 2009 and possibly persisting beyond. The measured rate of mangrove decrease is twice as high as that estimated for tidal forest degradation at Mngazana estuary, Eastern Cape, South Africa (Rajkaran and Adams 2010), which amounted to ~21 % (i.e., 32 ha) of mangrove loss since 1982 or to a decline of about 1 ha/year. Although both Cameroon and the Mngazana estuary are close to the heavily populated coastal zones and easily accessible, the mangrove forests of the Mngazana estuary were only stressed by harvesting. Other peri-urban mangrove forests are also characterized by a decrease in vegetation cover. In the state of Nayarit, north-west Mexico, for instance, mangrove forests were reduced

Table 3 Structural characteristics of mangrove forests in a peri-urban setting of Douala (Cameroon)

Species	Number of individuals	Mean height (m)	Mean diameter (cm)	De (trees ha ⁻¹)	BA (m ² ha ⁻¹)	F (%)	De _r (%)	BA _r (%)	F _r (%)	I.V.	Rank	C.I
<i>R. racemosa</i>	134	18.46±8.56	20.92±16.16	95.89	49.31	81.25	42.19	44.65	48.51	135.35	1	20.78
<i>A. germinans</i>	54	25.99±9.59	38.46±19.75	38.35	55.38	43.75	16.87	50.15	26.12	93.14	2	
<i>R. harrisonii</i>	21	16.05±7.23	15.98±8.55	14.91	3.91	26.25	6.56	3.54	15.67	25.77	3	
<i>G. demensei</i>	6	16.18±5.90	14.12±11.40	4.26	0.9	7.50	1.87	0.81	4.48	7.16	4	
<i>L. racemosa</i>	4	17.76±3.77	17.70±7.01	2.84	0.77	5.00	1.25	0.70	2.99	4.94	5	
<i>C. erectus</i>	3	12.31±4.33	9.31±4.68	2.13	0.17	3.75	0.94	0.15	2.23	3.32	6	

Note that *G. demensei* is not a true mangrove but a mangrove associate plant

De absolute density, BA basal area, F absolute frequency, De_r relative density, BA_r relative basal area, F_r relative frequency, I.V. importance value, C.I. Complexity Index

Fig. 5 A view of a clear-cut mangrove stand in the peri-urban setting of Douala (Cameroon). The large disturbed *Rhizophora* spp. stand is clear-felled for wood collection and housing



approximately 13 % over a 35-year period from 1970 to 2005 (Ruiz-Luna et al. 2010) due to both natural and manmade causes. Similarly, Mohamed (2008) reported a 12.5 % decline in closed canopy mangrove between 1969 and 1992 in the Tudor creek, Kenya, and attributed this loss to both episodic natural disturbances and recurrent human impacts. Besides the deteriorating status of the peri-urban forests, mangroves adjacent to the village are also degrading. Dahdouh-Guebas et al. (2004) observed a reduction in total mangrove vegetation area (51.42 % of forest loss between 1972 and 1992) around the Gazi village, Kenya, and related this change to over-harvesting of preferred mangrove species and occasional natural hazards (i.e., the El-Niño rains of 1997).

The primary threats to all mangrove species are habitat destruction and removal of mangrove areas for conversion to aquaculture, agriculture, urban and coastal development, and over-harvesting for wood products (Hossain et al. 2009; Polidoro et al. 2010; Guimarães et al. 2010; Spalding et al. 2010; Nfotabong-Atheull et al. 2011). Of these, the contribution of unplanned urbanization to the reduction of mangrove cover was dominant in our study area. Comparison between the 1974, 2003, and 2009 photographs clearly revealed the encroachment of human settlements into mangrove ecosystems.

Mangrove forest regression is often the result of heavy pressure from human exploitation (Rakotomavo and Fromard 2010) or changes in open-water hydraulic conditions (Dahdouh-Guebas et al. 2005b) and intertidal habitat (Cahoon et al. 2003; Rajkaran et al. 2010). For instance, in the Wouri estuary, mangrove ecosystems are traditionally harvested for the production of firewood, charcoal, boats, fish-traps, timber, and poles for houses (Din et al. 2008; Nfotabong-Atheull et al. 2009). This may explain the lack of significant differences in the mean annual rate of mangrove loss between the two sampling periods. Though trees of *R. racemosa* are commonly harvested, because of their frequency they turn out to be the most important species within the disrupted mangrove forests (Table 3). The relatively low abundance of *Rhizophora* trees that exceeded 20 cm in diameter may be related to their collection for commercial

purposes (Nfotabong-Atheull et al. 2009). This is not true for *A. germinans* trees which currently showed large basal areas (small number of trees with stem diameter greater than 30 cm) in the residual mangrove forests. In contrast to earlier findings (Din et al. 2002), seeds and seedlings of *A. germinans* were scarce around the adult trees (pers. obs.). The relatively closed canopy might be unfavourable for the germination and growth of the heliophilous *Avicennia* propagules (Patterson et al. 1993; McKee 1995; Fromard et al. 2004; Pickens and Hester 2011). In the basin, ghost forest of Mangrove Bight, Guanaja, Honduras, the paucity of seedling recruitment and survival coupled with the high rate of peat collapse, lowest redox potential, highest concentration of sulphides, and high interstitial salinity after a storm have been suggested as limiting the recovery rates of damaged forests (McKee and McGinnis 2003; Cahoon et al. 2003; Krauss et al. 2008).

Mangrove Health Disturbances

The negative impact of plant harvesting on the health of mangrove forests is illustrated by the presence of numerous small tree patches surrounded by water bodies (Fig. 2b, c). As a consequence of tree harvesting, some of the water channels which had been covered by tree canopies in the 2003 had become visible in 2009. In contrast, some parts of the largely disturbed mangrove forests in 2003 had rejuvenated by 2009. Young individuals of *Rhizophora* spp. were predominant in such areas. Although there was a relatively slight increase in the width of water channels, we assume that the aforesaid phenomenon could have led to an overestimation of the rate of water increase over the 35-year period.

The improvements to the city's transport infrastructure also appeared to be a factor contributing to mangrove encroachment in the supra-littoral zones at the landward edge of our study site. Indeed, road construction often led to an ultimately modification of the surrounding habitat which become favorable to the establishment of non-mangrove species (Dahdouh-Guebas et al. 2011; Nfotabong-Atheull et al. 2011; Satyanarayana et al. 2011a, b). Moreover, in

the high intertidal zone, mangroves are cleared for development of agriculture. Such anthropogenic land use in the estuarine zones is the likely cause of reductions in seawater supply.

Although none of the mangrove species present in Cameroon have been listed as threatened (Polidoro et al. 2010), populations are at risk from habitat loss. The regression of mangrove forest is exacerbated by the masses of migrants recently arriving from villages. Because many peri-urban mangrove areas around Douala are accessible by foot, migrants have been easily able to clear mangrove forests for wood collection and housing. The large removal of vegetation might lead to rapid surface elevation loss (peat collapse) as dead root materials are decomposed and soil is eroded/oxidized (Cahoon et al. 2003). The Cameroon estuary has already experienced daily discharge of untreated domestic wastes, dyke construction, landfill, trenching, and sand extraction (Nfotabong-Atheull et al. 2011). All such disruptions can affect the hydrodynamic processes, leading to a reduction of commercial fish and shellfish species that breed, spawn, hatch, or develop in the mangrove ecosystems (Barbier 2003; Mumby et al. 2004). With a rise in sea level, the habitat of the remaining mangrove species will be furthermore disrupted (Gilman et al. 2008), and species zones will suffer mortality since their movement inland would be blocked by coastal development (Polidoro et al. 2010).

Conclusion

The present study confirms the potential of air-borne imagery for qualitative and quantitative characterization of mangrove vegetation dynamics. These kinds of studies are of great concern especially in the current context of vulnerability of coastal ecosystems to the changes associated with anthropogenic land uses and global warming. Our results clearly revealed a considerable decrease of mangrove cover in the vicinity of Douala in Cameroon, with unseen rates of degradation. This coastal ecosystem is continuously cleared for the purpose of other land-use activities such as settlements, road construction, sand quarry, and agriculture. The disruption of mangrove habitat could lead to the loss of individual mangrove species and associated ecosystem services (Polidoro et al. 2010). It is therefore necessary to create awareness amongst the locales as well as enforce the local existing conservation rules. However, beyond listing of mangrove plants on the Red List of Threatened Species (*loc. cit.*), which will seldomly lead to widely distributed species being listed, we call for the creation of a Red List of Locally Threatened Ecosystems, which in contrast is likely to list mangroves as an ecosystem under critical risk of (local) extinction in many countries around the globe.

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