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Mangroves in peril: unprecedented degradation rates of peri-urban mangroves in Kenya

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Abstract

Marine ecosystems are experiencing unprecedentedly high degradation rates than any other ecosystem on the planet, which in some instances are up to four times that of rainforests. Mangrove ecosystems have especially been impacted by compounded anthropogenic pressures leading to significant cover reductions of between 35 and 50 % (equivalent to 1–2 % loss pa) for the last half century. The main objective of this study was to test the hypothesis that peri-urban mangroves suffering from compounded and intense pressures may be experiencing higher degradation rates than the global mean (and/or national mean for Kenya) using Mombasa mangroves (comprising of Tudor and Mwache creeks) as a case study. Stratified sampling was used to sample along 22 and 10 belt transects in Mwache and Tudor respectively, set to capture stand heterogeneity in terms of species composition and structure in addition to perceived human pressure gradients using proximity to human habitations as a proxy. We acquired SPOT (HRV/ HRVIR/ HRS) imageries of April 1994, May 2000 and January 2009 and a vector mangrove map of 1992 at a scale of 1 : 50 000 for cover change and species composition analysis. Results from image classification of the 2009 image had 80.23 % overall accuracy and Cohen's Kappa of 0.77, thus proving satisfactory for use in this context. Structural data indicate that complexity index (CI) which captures stand structural development was higher in Mwache at 1.80 compared to Tudor at 1.71. From cover change data, Tudor had lost 86.9 % of the forest between 1992 and 2009, compared to Mwache at 45.4 % representing very high hitherto undocumented degradation rates of 5.1 and 2.7 % pa, respectively. These unprecedentedly high degradation rates, which far exceed not only the national mean (for Kenya of 0.7 % pa) but the global mean as well, strongly suggest that these mangroves are highly threatened due to compounded pressures. Strengthening of governance regimes through enforcement and compliance to halt illegal wood extraction, improvement of land-use practices upstream to reduce soil erosion, restoration in areas where natural regeneration has been impaired, provision of alternative energy sources/building materials and a complete moratorium on

wood extraction especially in Tudor creek to allow recovery are some of the suggested management interventions.

1 Introduction

Marine ecosystems are experiencing unprecedentedly high degradation rates than any other ecosystem on the planet. In some instances, they are up to four times that of rainforests. Currently between 2–7 % of these ecosystems are lost annually, a seven-fold increase compared to only half a century ago (Nelleman et al., 2009). Mangrove ecosystems have especially been impacted by anthropogenic pressures like unsustainable wood harvesting, sewage discharge, dredging, conversion for agriculture, land development and unplanned development leading to significant cover reductions of between 35–50 % for the last half century (Alongi, 2002; Duke et al., 2007; Giri et al., 2011). Consequences of this widespread degradation include biomass loss and increased carbon emissions (Donato et al., 2011), alteration of forest structure, and change in species composition (Kairo et al., 2002; Bosire et al., 2003, 2006), reduced fisheries production and aggravated coastal erosion (Rönnbäck, 1999; Nageikerken et al., 2002; Alongi, 2008; Zhang et al., 2012) among others. As human activities continue to expand in coastal environments with escalating world population, the likelihood of increasing magnitude of such impacts is foreseeable.

Additionally, climate change impacts have further compounded pressure on mangrove forests. They include sea level rise, flooding, erosion and sedimentation, fluctuating precipitation and temperature regimes, and associated phenomena like hurricanes and cyclones among others (McLeod and Salim, 2006; IPCC, 2007; Gilman et al., 2008; Bosire, 2010; Bosire et al., 2012).

Extensive mangrove die-back, that resulted from flooding and massive sedimentation following the Indian Ocean Dipole (IOD) events in 1997/98 and 2006 mangrove forests along the Kenyan coast, practically epitomized the impacts of disasters associated with climate change (Kitheka et al., 2002, Wieczkowski, 2009). IOD, described as unique to the Indian Ocean, is a distinct coupled oscillation in ocean-atmosphere interactions, controlling sea surface temperature (SST) anomalies (Saji et al., 1999, 2006). This phenomenon occurs in phase or out phase with El-Nino Southern Oscillation (ENSO) (Kayanne et al., 2006; Marchant et al., 2006; Pillai and Mohankumar, 2010) and causes warmer than normal SSTs in the western basins but cooler SSTs in the eastern basins.

Although the concomitant impacts of climate-related phenomena on mangrove forests might vary in different localities (Ellison and Stoddart, 1991; Twilley, 1998; Allen et al., 2001), the risk of more costly climate-driven ecological feedbacks to mangrove forests with changing climate is probable; and this necessitates studies oriented at understanding mangrove vulnerability and resilience to climate-driven disturbances at local, regional and global scale. Hitherto, the long-term impacts of climate related disturbances on mangrove ecosystems still remain unclear.

Remote sensing has been identified as an effective tool to study otherwise difficult-to reach and difficult-to-penetrate mangroves along coastal areas. Landsat and SPOT imageries have been used for visual interpretation (Gang and Agatsiva, 1992), determining vegetation index (Blasco et al., 1986; Chaudhury, 1990; Jensen et al., 1991), classification (Aschbacher et al., 1995), and band rationing (Long and Skewes, 1994) of all types of mangrove vegetation. Remote sensing applications have been applied mainly for mangrove inventory, mapping, and change detection. Landsat and SPOT data, as well as high spatial resolution airborne multispectral and SIR-C radar data, were also used for management purposes in a number of countries (Gang and Agatsiva, 1992; Gao, 1998; Green et al., 1998; Kairo et al., 2002). Nevertheless, remote sensing techniques applied to mangrove vegetation are still not as common as for terrestrial systems, particularly along the east coast of Africa (Blasco et al., 1994; Dale et al., 1996).

Although the global mean annual cover loss of mangroves has been estimated at 1–2 % (Alongi, 2002; Duke et al., 2007; Giri et al., 2011), it is highly probable that this global estimate masks degradation rates in some locations, which may be higher than previously perceived. Recent assessment in Kenya revealed that the country's mangroves have experienced 20 % loss over a period of 25 yr (1985–2010) representing an annual loss of 0.74 % (Kirui et al., 2013). This study suggested that the Kenyan mangroves are falling below the global mean in terms of annual cover loss. The risk of using such averaged measures, however, is a misleading perception of the situation. Any complacence in management might be inadequate if the average loss is benign while some areas within the country may be experiencing much higher degradation rates, which have hitherto not been documented. Consequently, the study presented addresses explicitly vegetation structure, natural regeneration and spatio-temporal dynamics of the most peri-urban mangroves in the country with a hypothesis that they are experiencing much higher degradation rates due to perceived compounded pressures.

2 Materials and methods

2.1 Site description

The study was undertaken in two heavily impacted mangrove forests of Tudor and Mwache creeks (Fig. 1), Mombasa in the coastal province of Kenya. Tudor creek (4°02′ S, 39°40′ E), located at the northwest of Mombasa island, extends some 10–15 km inland with two main seasonal rivers, Kombeni and Tsalu, draining over 45 000 and 10 000 ha, respectively. The creek is characterized by a 20 m mean depth single narrow sinuous inlet that widens inland to a central 5 m depth basin, covering an area of 637 ha and 2235 ha at low and high water spring tides, respectively (Mohammed et al., 2008); and an average tidal range of 0.6 and 4.5 m, in the respective neap and spring tides. Within the creek is a mangrove forest, extending over an area of 1,641 ha, principally composed of *Rhizophora mucronata*, *Avicennia marina* and *Sonneratia alba* and has no display of distinct species zonation along tidal gradient. The forest is covered by sediments that are predominantly made up of mud, and sand in some parts (Mohamed et al., 2008).

Mwache Creek (4°3.01′ S and 39.06°38.06′ E) is located 20 km Northwest of Mombasa island (Fig. 1). The total area of the wetland is approximately 1500 ha with about 70 % of the surface area being covered with mangroves comprising of both basin and riverine mangroves. The dominant mangrove species in Mwache are: *Avicennia marina*, *Rhizophora mucronata*, *Ceriops tagal* and *Sonneratia alba* (Kitheka et al., 2002). These species display a zonation pattern typical of mangroves in Eastern Africa. The creek receives freshwater from Mwache River, which is seasonal and thus there is usually no flow during the dry season especially between December and March, and July and September. The rate of sediment production within Mwache River basin reaches a high of 3000 tons per year due to poor land-use activities upstream, high rainfall intensity during the rainy season and steep land gradient (Kitheka et al., 2002; Bosire et al., 2006).

Characterizing the climate of both creeks is the influence of semi-annual passage of the inter-tropical convergence zone (ITCZ) and the monsoons in two distinct seasons. The Northern Easterly Monsoon (NEM) manifests between December and March, and the Southern Easterly Monsoon (SEM) is experienced between May and October. The mean annual rainfall averaged at 1038 mm, with peaks in May and June; and the mean annual temperatures are 23.9 and 28.5 °C, for the rainy and dry seasons, respectively (Obura, 2001; Mohamed et al., 2008).

2.2 Sampling methods

2.2.1 Vegetation structure and species composition

Data on mangroves structure and species composition were acquired using stratified sampling technique. Sampling transects perpendicular to the shoreline were identified prior to field campaigns using unsupervised SPOT images of 2009. The locations of the different transects were determined based on observed vegetation classes, canopy cover and length of intertidal area so as to capture different plant assemblages as representatives for the whole forest. Vegetation sampling were carried out using standard

10 m × 10 m quadrats, that were laid 100 m away from each other perpendicularly along the transect lines in the forest. Stratified sampling was used to sample along 22 and 10 belt transects in Mwache and Tudor, respectively, set to capture stand heterogeneity in terms of species composition and structure in addition to perceived human pressure gradients using proximity (or otherwise)to human habitations as a proxy.

Within each quadrat, tree height, stem diameter and crown diameter for all the trees greater than 2.5 cm diameter were determined. Tree height was measured using a *Suunto* hypsometer, while DBH was measured using forest calipers. Consequently, information on the composition, diversity, structural parameters and community indices (Basal Area, Stem Density, Complexity index, Importance Value Index) were computed, together with diameter size class distribution and height profile, to describe the structure and composition of the forest.

$$\mathrm{BA} = \frac{\pi DBH^2}{4}\,\mathrm{cm}^2, \quad \text{where} \quad \pi = 3.142 \tag{1}$$

Importance value index, describing the structural role of individual tree species in the habitat, was calculated following Husch et al. (2003):

$$\mathrm{IV}_j = \text{Relative Density} + \text{Relative Dominance} + \text{Relative Frequency} \tag{2}$$

$$\text{Relative Density} = 100 \times \left(\frac{d_j}{D}\right) \tag{2.1}$$

$$\text{Relative Dominance} = 100 \times \left(\frac{x_j}{X}\right) \tag{2.2}$$

$$\text{Relative Frequency} = 100 \times \left(\frac{n_j}{N}\right) \tag{2.3}$$

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where IV_j = importance value of j species; and d_j = number of individuals of the j species present in sample population (density), D = total number of individual in sample population ($D = \Sigma d_j$); and x_j = sum of basal area for j species (dominance), X = total of basal area across all species ($X = \Sigma x_j$); and n_j = number of sampling units where j species is present (occurrence), N = total number of sampling units.

Importance value (IV) of each species was calculated by summing its relative density, relative frequency and relative dominance so as to get relative contribution of each species to the overall stand structure. Stand complexity index (CI) was calculated according to Holdridge et al. (1971). This index is used to illustrate how complex or structurally developed a stand is and is derived from combining all the measured stand structural attributes (stem density (number of stems/0.1 ha × 10^{-3} in a 0.1 ha plot), D_{130} calculated into basal area ($m^2\ 0.1\ ha^{-1}$), height (m) and number of a species.

$$CI = s \times d \times \bar{h} \times BA \times 10^{-5} \quad (3)$$

where s = number of species; d = stand density; $\bar{h}$ = mean height; and BA = basal area

2.2.2 Natural regeneration

Data on the composition and distribution pattern of natural regeneration was obtained using the method of Linear Regeneration Sampling (Sukardjo, 1987; FAO, 1994; Kairo et al., 2002), which was used to sample all juveniles in 5 × 5 m^2 subplots (within the main 10 × 10 m^2 quadrats). According to Stoddard and Stoddard (1987) occurrence of all trees of different species with diameter less than 2.5 cm, classified as juveniles was recorded and grouped according to their regeneration classes based on height. Seedlings < 40 cm were classified as regeneration class I (RCI). Saplings between 40 and 150 cm height were classified as RCII and RCIII was for all small trees with heights > 150 cm but < 2.5 cm DBH.

2.2.3 Mangrove cover change detection

Image processing

We acquired SPOT (HRV/HRVIR/HRS) imageries of April, 1994, May, 2000 and January, 2009 and a vector mangrove map of 1992 at a scale of 1:50000 from a mapping project done by Food and Agriculture Organization (FAO, 1992). This map was an interpretation from aerial photograhps. The map provided a validated baseline from which to detect future changes (Kirui et al., 2013). Both 1994 and 2000 images had a ground resolution of 20 m while the 2009 image had a ground resolution of 5 m pan-sharpened to 2.5 m resolution. Image classifications were done on the composites of the three SPOT images with each containing the three primary bands (B1 (Green); B2 (Red) and B3 (NIR). All image processing was done using ENVI + IDL 4.72 while ArcGIS 9.3 was used to create final maps and to compute statistics. All the images were registered to WGS 84 UTM Zone 37S projection. We adopted Nearest Neighbour re-sampling method (Reddy and Roy, 2008; Ardli and Wolff, 2009) for geometric correction of the 2009 image, with a resultant Root Mean Square Error (RMSE) of half pixel which is approximately 1.25 m. This was an acceptable error for subsequent image analysis. A pre-registered Landsat image of the study area obtained in 2003 was used as the source for ground control points for correcting the 2009 image. Consequently, a feature-based image-image registration was applied to both 1994 and 2000 images as the warp images and 2009 as the base image in order to align the two images with the base image. Four reference points identified from the 2009 image were used as control points for subsequent registration of the 1994 and 2009 images. The resultant RMSE errors were 1 (20 m) and 1.5 (30 m) pixels respectively. Since change detection was entirely based on comparative post-classification direct analysis between the total mangrove cover for the four time steps, these errors were deemed sufficient in this context. This comparison involved converting the resultant raster classes into vector in order to compare with the 1992 baseline map. Before classifications, dark object subtraction, using atmospheric correction method (Song et al., 2001), was later done on the three images to remove effects of the different atmospheric conditions on the reflectance for the three images taken at different temporal resolutions.

Mangrove forest mapping and cover change analysis

ISODATA and K-Means unsupervised classification methods were separately done on the 2009 image prior to fieldwork. These classifications were set to retrieve 26 different spectral classes for comparison of best result yielding method. K-Means method was found most suitable for field campaign as it clearly delineated major mangrove zones. These were later grouped into 9 broad informational classes after close expert knowledge examination. This helped identify regions of interest (ROI), collection of ground control points (GCPs) and delineation of training sites for supervised classifications. Rigorous field campaigns were done at a cross section of main mangrove species aggregation area. Ground control points were collected using a Garmin GPS 76 in UTM coordinates. This model had between 5–10 m positional accuracy. To minimize errors resulting from the GPS accuracy, we ensured that collected GCPs were within a 10m radius of the same land cover type.

We identified 8 main classes representing the four main mangrove species in the area (*Rhizophora mucronata*, *Avicennia marina*, *Ceriops tagal* and *Sonneratia alba*), mud, sand, water and terrestrial areas to map mangroves to species level using the 2009 high resolution image. Due to their coarse ground resolution, the 1994 and 2000 images were not suitable to map mangroves to species level hence the images were only classified to two broad categories; mangrove areas and non-mangrove areas. Training sites were later digitized by overlaying the GCPs on the three images and creating polygons representing the identified classes. Before the classifications, we examined the spectral separability between all pairs of training ROIs using the transformed divergence separability index (Richards and Xiuping, 1999). Values of this index range from 0–2, with 2 indicating 100 % separation. Maximum Likelihood classification method was later done on the three images, confusion matrices calculated to obtain producer's and user's accuracies and the subsequent overall classification accuracy.

We only performed accuracy assessments on the 2009 image since the GCPs were collected in 2011 and no significant change was notable in the period between these two years. We used the 1992 vector map as the baseline year to calculate species loss/gain till 2009 and overall mangrove cover change between 1994, 2000 and 2009. Maps on cover change were used to display the variation on mangrove vegetation areal extent based on maps for 1992 and 2009.

2.2.4 Statistical analysis

All variables were subjected to normality test using Kolmogrov-smirnov test and subsequently, structural parameters–mean height, mean DBH, stand density, and BA were subjected to analysis of variance for mean comparison among sub-sites within each creek and Turkey's HSD multiple comparison test was adopted for mean separation, if the main effects were significant. Statistical differences in these structural variables between Mwache and Tudor creeks were examined using Welch t-test. All statistical analyses were conducted using R 2.14.1 environment for statistical computing (R Development Core Team, 2011).

3 Results and discussion

3.1 Stand structural characteristics and spatial variability

The structural attributes describing the mangrove vegetation in Tudor and Mwache creeks are summarized in Table 1. In Mwache, five species were encountered in both the adult canopy and juveniles though not in the same spatial variability, whereas in Tudor, five species were encountered but four species were represented at both the adult and juvenile stages. *Bruguierria gymnorrhiza* occurred only at juvenile stages and no juveniles of *S. alba* were encountered in Tudor creek contrary to what was observed in Mwache creek.

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Mangrove vegetation in Tudor and Mwache creeks differed significantly in mean tree diameter, DBH ($t = 9.42$, $p < 0.001$) and mean height ($t = 12.75$, $p < 0.001$). In Tudor, mean tree DBH varied significantly among sub-sites within creek ($F = 8.489$, $p < 0.001$), ranging from 3.29 to 7.75 cm and mean height also varied significantly within the creek ($F = 9.975$, $p < 0.001$) with values ranging from 3.29 to 5.45 m, with the maximum recorded DBH and height being 60.50 cm and 15.00 m respectively. In Mwache, the mean DBH range was higher than in Tudor ranging from 6.40 cm to 12.95 m and significantly different ($F = 23.26$, $p < 0.001$) among sub-sites within the creek; and the mean tree height also differed significantly ($F = 22.2$, $p < 0.001$). The maximum DBH and height recorded in Mwache was 53.0 cm and 15.00 m respectively. The graphical comparison of mean height and mean DBH of adult trees encountered across the creeks is presented in Fig. 2. The pattern is similar for both mean height and mean DBH with no significant difference, except that they were both higher in the Islands zone.

Diameter distribution followed mostly the inverse-J shaped which is typical of naturally regenerating forests, but with a slight deviation as some size classes are devoid of any individuals especially in Tudor creek. It is apparent that Tudor creek is vigorously regenerating with most of the stems being < 5cm. Overall, trees of diameter class 5.0–7.0 were not overexploited compared to those of diameter class 7.0–9.0 which were more utilized in Mwache creek (Fig. 3). The trees in the higher size classes were low in both creeks as expected of natural/uneven-aged stands.

Based on the IV presented in Fig. 4, *R. mucronata* and *A. marina* were the dominant species in Tudor creek whereas *R. mucronata* and *S. alba* are the dominant species in Mwache creek. Also, there exists some spatial variability in species dominance across different locations with the dominance of *S. alba* in the islands being noteworthy as compared to other locations.

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3.2 Regeneration patterns of juvenile mangrove species in Tudor Creek

Juveniles had a varying distribution pattern across the study area depending on site. Most of the juveniles were found landward compared to the seaward sites. *Rhizophora mucronata*, *A. marina, C. tagal* appeared to be rejuvenating in most parts of the creeks (Table 2). The highest regeneration occurred for *A. marina* (9200 juveniles ha^{-1}) and *R. mucronata* (4190 juveniles ha^{-1}) in Tudor, while *R. mucronata* (7016 juveniles ha^{-1}) and *C. tagal* (1025 juveniles ha^{-1}) in Mwache, represented 83 and 12 % respectively of all juveniles encountered. The least was for *B. gymnorrhiza* in both sites representing less than 1 % of the whole creek's juveniles. The juveniles for *S. alba* were scanty and were only represented as RCIII in Mwache but were entirely absent in Tudor despite their presence in the adult canopy.

3.3 Mangrove extent and cover change

The overall classification accuracy for the 2009 image was 80.23 % and Cohen's Kappa of 0.77 showing satisfactory results for its use in this context (Table 3. Change in areal extent of mangrove forest in Tudor and Mwache creek is summarized in the matrix provided in Table 4. It depicts mangrove loss with subsequent years. In 2009 the forest cover had reduced to 215.3 ha for Tudor and 1016.9 for Mwache from a cover of 1641.3 and 1861.4 ha in 1992, respectively. This was a loss of 1425.0 and 844.5 ha of mangrove cover from 1992 (Fig. 7) representing 86.5 and 45.4 % less cover, respectively. The highest rate of cover loss was between 2000 and 2009, which was −73.68 and −20.04 % for Tudor and Mwache creeks respectively. Change in the area covered by individual species is provided in Table 4 and Fig. 6. In Tudor creek, four species were observed in 1992 and five for Mwache creek but they had reduced to four in 2009 with no complete loss of any species in the former. The most affected species was *X. granatum* which had a cover of 13.11 ha in 1992 but was not observed in 2009.

Rhizophora mucronata and *C. tagal* had also suffered drastic losses in both creeks (Table 5). *Avicennia marina* reduced in cover by 40.5 % in Tudor creek contrary to its

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increase by 115.6 % in Mwache creek. However, *S*. *alba* was largely increased in cover by 1199 % in Mwache compared to 137.4 % in Tudor creek.

3.4 Discussion

This study purposed to investigate the hypothesis that peri-urban mangroves are experiencing higher degradation rates far exceeding the global mean of 1–2 % pa commonly reported in literature using the Mombasa mangroves in Kenya as a case study. Additionally, a recent study by Kirui et al. (2013) had indicated that Kenyan mangroves had lost only 20 % cover over a period of 25 yr representing a 0.74 % annual loss. We sought to investigate whether peri-urban mangroves are experiencing much higher degradation rate than this national average as well, which could have significant management implications. Emch and Peterson (2006) described changes in mangrove forests as multidimensional, resulting from biotic, geomorphic, and anthropogenic influences. Although it is difficult to isolate the singular effect of each factor in a complex mangrove system, it is speculated that massive degradation in the study areas can be linked to mangrove response to man-induced stressors in combination with indirect impacts of climate change and variability.

Analyses of forest structure and temporal changes in mangrove cover strongly suggest that the mangrove forests in Tudor and Mwache creeks have been severely degraded. These mangrove forests being in a peri-urban setting, have suffered significant loss in cover change and altered forest structure. From the size class distribution figures, we can deduce selective wood harvesting in Mwache compared to indiscriminate harvesting in Tudor creek. This has left gaps in the forest, which explains the high regeneration especially for *R. mucronata*, which is shade intolerant (Ellison and Farnworth, 1993), in Tudor creek compared to Mwache where mature stands were encountered in some parts of the creek. The size class distribution in Mwache obeys the expected reversed J pattern typical of uneven aged regenerating natural stands, while the pattern in Tudor is indicative of unusually high extractive pressure which targets all classes, suggesting that while the drivers of change among the two creeks may be similar, the scale and magnitude is substantially different. The insignificant difference in complexity indices between mangrove forests in both creeks (1.80 and 1.71) for Mwache and Tudor creeks, respectively) however showed no conspicuous variability in structural complexity of the two creeks. From the size class distribution for instance, Tudor creek is portrayed as an overly degraded and young forest compared to Mwache creek. In addition, during the field study in Tudor forest, we observed lots of illicit distillers for local brew business, which wholly depends on mangroves as a source of fuel-wood. These human modifications within the coastal zone will reduce the resilience of these ecosystems, making them more vulnerable to environmental pressures like climate change (Ellison and Farnsworth, 1996; Kitheka et al., 2002; Mc-kee et al., 2007; Lovelock and Ellison, 2007; Bosire, 2010). Interestingly and on a more positive note, natural regeneration in the creeks was substantially higher and vigorous than the minimum recommended of 2500 seedlings ha^{-1} (FAO, 1994), for successful forest re-stocking thus suggesting that natural recovery may be possible if current anthropogenic pressures are moderated.

Tudor and Mwache creek mangroves experienced a cover loss of 86.9 and 45.4 % respectively over this 17 yr period with the highest loss occurring between 2000 and 2009. This loss was higher in Tudor (−73.68 %) compared to (−20.04 %) in Mwache creek attributed to indiscriminate and uncontrolled harvesting, pollution from industrial and domestic sewage discharge and aggravated siltation, among other anthropogenic factors which are prominent in this forest (Mohamed et al, 2008). It is also within this period that the IOD events of 1997/98 and 2006 occurred causing massive destruction of these forests. This concurs with other studies done on mangrove cover in Kenya (Bosire et al., 2008; Bosire, 2010; Kirui et al., 2013). Perhaps, this suggests that mangrove dieback following these IOD events could be a major driver for this cover change. Increase in population over the years attributed in part to migration of people to Mombasa city from other parts of the country in search of employment or business opportunities (GOK, 1999) has also led to loss of biotic integrity and threatens biodiversity since use of mangroves for firewood and building poles is typical of all coastal areas

(Dahdouh-Guebas et al., 2000). The structural attributes reported here are much lower than those of mangrove stands in other parts of the country, a fact attributable to the inordinate pressure experienced by the peri-urban mangroves in this study. For instance mangroves of the northern part of Kenya in Lamu (stand density of 2075–2142 stems ha^{-1}, basal area of 24.5–46.97 m^2 ha^{-1}, and canopy height of 16–26.5 m, (Kairo et al. 2001) are much more developed than the mangroves of the current study area. The mangroves of the southern coast of Kenya are also more developed with stem densities ranging from 1573 to 1839 stems ha^{-1}; mean height 6–7.4 m and basal area 9.7–13 m^2 ha^{-1} (unpublished data). The south and north of Kenya mangroves are distant from Mombasa and thus under less extractive pressure as these areas are less populated.

Sonneratia alba which is a pioneer species has thrived in both creeks with its coverage increasing by more than 100 % over the period under consideration. This species is adapted to long periods of inundation and was thus not impacted significantly during the IOD event. Continued sedimentation while a major threat to the mangroves within the study area in general, has led to creation of suitable conditions for pioneer species, hence establishment of new islands (Fig. 5(a). *Avicennia marina* coverage has also increased in the creeks owing to its tolerance to a wide range of environmental conditions (Wells, 1982; Clarke, 1995; Dadouh-Guebas et al., 2004; Huisman et al., 2009; Wang'ondu et al., 2010). However, *R*. *mucronata* and *C tagal* normally most preferred by the locals for their quality and diverse uses, have dwindled significantly over the years. The stumps observed in the current study were both old and recent cuttings largely belonging to these two species. Severely reduced density of standards (parent trees) may compromise propagule production and thus limit natural regeneration (Bosire et al., 2003, 2008). *Avicennia marina* has overtaken *R. mucronata* in terms of cover in Mwache creek over the years raising questions whether in the long-term there may be species shifts in the canopy; but this is unlikely since *R. mucronata* comprises 83 % of the juvenile density and thus likely to play a major role in future forest re-stocking still almost guaranteeing continued dominance of *R. mucronata* in the adult stratum.

An annual cover loss of 5.1 % yr^{-1}, estimated in Tudor mangrove forest is distinctively higher than that of Mwache creek (2.7 % yr^{-1}). These losses were significantly higher compared to the average of 0.7 % pa recently estimated for Kenyan Mangrove forests (Kirui et al., 2013) and that of 1–2 % global degradation rate of mangrove forests (Giri et al., 2011). These unprecedentedly high degradation rates, which far exceed not only the national mean but the global mean as well, strongly suggest that these mangroves are highly threatened due the compounded pressures already discussed. For instance in Tudor creek, only 215.3 ha of mangroves are remaining from a cover of 1642.3 ha in a span of less than 20 yr. Most of the studies that have been conducted previously on mangrove cover change are majorly at country or global level. The current study has narrowed mangrove cover loss to a specific impacted zone which makes it easier for forest managers to allocate resources based on the acquired data of high resolution at species level degradation and rejuvenation. This provides a baseline on what species may be used as candidates for restoration before their extinction and those that can be used to improve on the forest cover on bare sites based on their suitability to colonize degraded areas or withstand different and harsh environmental conditions as "smart species". Strengthening of governance regimes through enforcement and compliance to halt illegal wood extraction, improvement of land-use practices upstream to reduce soil erosion, restoration in areas where natural regeneration has been impaired, provision of alternative energy sources/building materials and a complete moratorium on wood extraction especially in Tudor creek to allow recovery are some of the suggested management interventions.

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Table 1. Structural characteristics of Tudor and Mwache mangroves.

Summary of structural attributes of mangrove vegetation in Mwache Creek						
Parameters	Mwakuzimu	Ngare	Mashazani	Maguzoni	Islands	All sub-sites
Number of Species	5	4	4	3	1	5
Mean DBH (cm)	9.90 ± 0.37^a	12.95 ± 0.57^b	8.82 ± 0.40^a	6.40 ± 0.46^c	10.07 ± 0.46^a	$6. \pm 1.064$
Mean Height (m)	5.5 ± 1.3^a	6.4 ± 0.8^a	4.8 ± 0.4^a	4.6 ± 0.7^a	6.2 ± 0.6^a	5.3 ± 2.4
Stand Density (stems ha^{-1})	1840 ± 22^a	1448 ± 18^a	84 ± 9^b	2633 ± 44^a	2000	1701 ± 105
BA ($m^2\ ha^{-1}$)	4.2 ± 2.1^b	6.9 ± 3.5^b	0.2 ± 0.1^c	5.1 ± 2.1^b	19.3^a	4.0 ± 0.2
Complexity Index (CI)	2.13	2.56	0.003	1.85	2.39	1.80

Summary of structural attributes of mangrove vegetation in Tudor Creek					
Parameters	Mikindani	Islands	Jomvu	Kijiwe	All sub-sites
Number of Species	3	3	5	3	5
Mean DBH (cm)	5.04 ± 0.53^a	7.67 ± 0.38^b	7.75 ± 2.77^b	4.43 ± 0.66^a	6.35 ± 0.1
Mean Height (m)	3.43 ± 0.48^a	4.43 ± 0.27^b	5.45 ± 1.12^b	3.29 ± 0.01^a	4.31 ± 0.42
Stand Density (stems ha^{-1})	1567 ± 188^a	1446 ± 97^a	891 ± 245^b	1313 ± 113^{ab}	1304 ± 118
BA ($m^2\ ha^{-1}$)	5.90 ± 0.63^a	11.74 ± 0.93^a	10.11 ± 3.20^a	5.84 ± 3.78^a	8.39 ± 1.61
Complexity Index (CI)	0.95	2.26	1.52	0.76	1.71

[*] Values are mean ± standard error. Same superscript letter notation in each row shows no significance difference between sub-sites.

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Table 2. Juvenile density in forests within the study area.

Sites	Species	Regeneration Class RCI	RCII	RCIII	Total (ha^{-1})
Mwache	*A. marina*	285 ± 161(95)	11 ± 8(4)	4 ± 4(1)	300(4)
	B. gymnorrhiza	16 ± 16(73)	4 ± 4(20)	1 ± 1(7)	22(< 1)
	C. tagal	501 ± 273(49)	355 ± 169(35)	168 ± 83(16)	1025(12)
	R. mucronata	2503 ± 479(36)	2279 ± 445(32)	2234 ± 495(32)	7016(83)
	S. alba	0(0)	22 ± 22(30)	51 ± 30(70)	72(1)
	Total (ha^{-1})	3306 (39)	2672 (32)	2459 (29)	8436(100)
Tudor	*A. marina*	8876 ± 701(96)	107 ± 05(1)	217 ± 16(2)	9200(66)
	B. gymnorrhiza	0	0	34(100)	34(< 1)
	C. tagal	146 ± 33(27)	207 ± 47(39)	183 ± 07(34)	537(4)
	R. mucronata	646 ± 04(15)	1080 ± 08(26)	2463 ± 11(59)	4190(30)
	Total (ha^{-1})	9668 (69)	1395 (10)	2898 (21)	13 961(100)

[a] Values are mean ± standard error.
[b] The values in parenthesis in a row represent the percentage of the total juveniles of a species in the different regeneration classes.

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Table 3. Classification accuracy for the 2009 image based on the different classes delineated.

Class	Producer's accuracy	User's accuracy
Sonneratia alba	78.09	86.67
Ceriops tagal	78.1	72.89
Avicennia marina	79.56	67.73
Rhizophora mucronata	87.5	94.94
Sand/sandy beaches	73.37	96.82
Water	93.34	99.99
Mud	58.08	76.66
Open mangrove areas	93.83	100
Overall accuracy	80.23 %	
K coeff.	0.77	

Table 4. Mangrove cover change over the years.

Site	Variables	Time series			
		1992	1994	2000	2009
Mwache	Areal extent (ha^{-1})	1861.4	1536.2	1271.9	1016.9
	Cover loss (ha^{-1})	–	−325.2	−589.6	−844.5
	Percentage cover change against 1992 (%)	–	−17.5	−31.7	−45.4
Tudor	Areal extent (ha^{-1})	1641.3	1281.4	818.1	215.3
	Cover loss (ha^{-1})	–	−359.9	−823.2	−1426.1
	Percentage cover change against 1992 (%)	–	−21.95	−50.2	−86.9

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Table 5. Species cover change in 1992 and their shift in 2009.

Site	Species	Year		
		1992	2009	% change
Mwache	*Avicennia marina*	171.6	370.1	+115.6
	Ceriops tagal	685.5	192.1	−72.0
	Rhizophora mucronata	978.3	287.4	−70.6
	Sonneratia alba	12.9	167.4	+1199.0
	Xylocarpus granatum	13.1	0	−100
Tudor	*Avicennia marina*	110.8	65.9	−40.5
	Ceriops tagal	252.4	38.1	−84.9
	Rhizophora mucronata	1244.2	30.6	−97.5
	Sonneratia alba	33.9	80.6	+137.4

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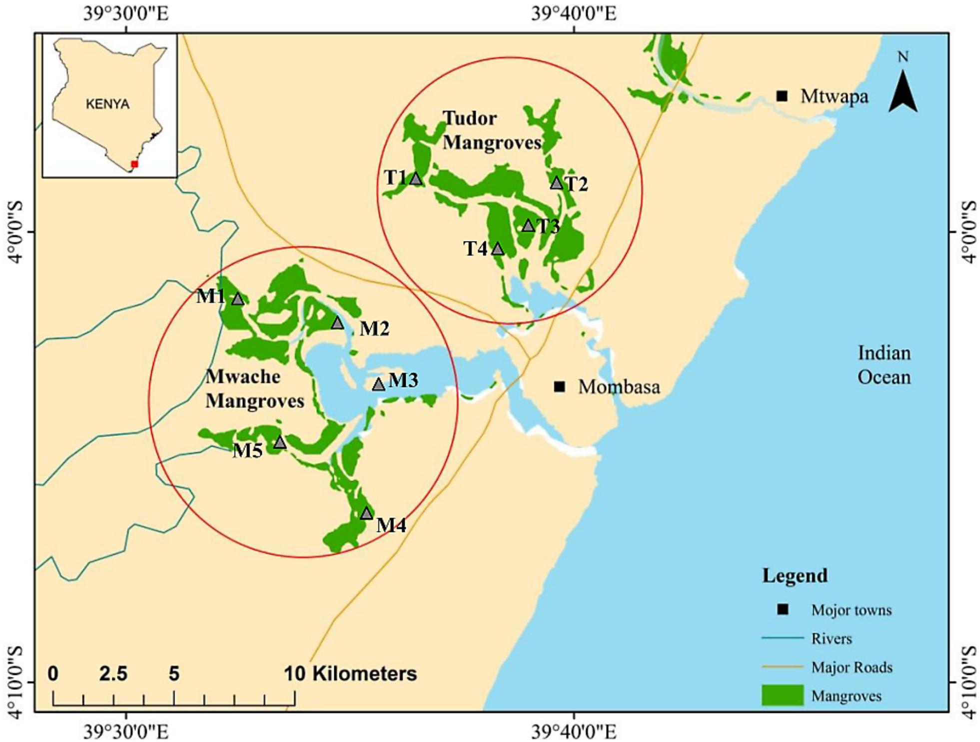

Fig. 1. Map of the study area showing the Mombasa mangroves (Mwache and Tudor Creeks): M1, M2, M3, M4 and M5 represent Mashazani, Ngare, KPA, Mwakuzimu and Maguzoni respectively (in Mwache Creek), whereas T1, T2, T3 and T4 represent Jomvu, Kijiwe, Island and Mikindani, respectively (in Tudor Creek).

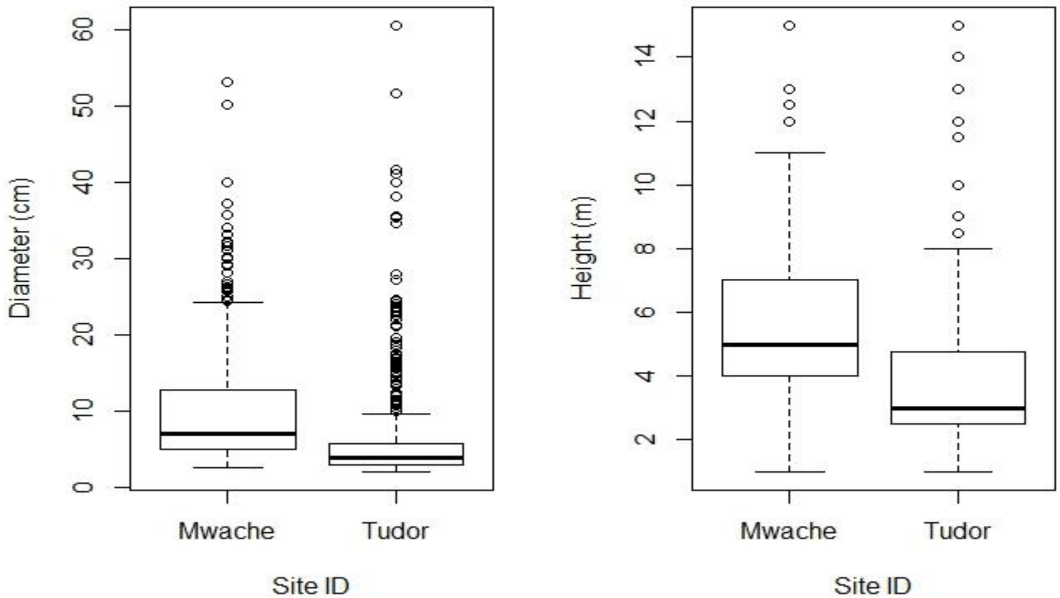

Fig. 2. Diameter and height distribution in Tudor and Mwache creek.

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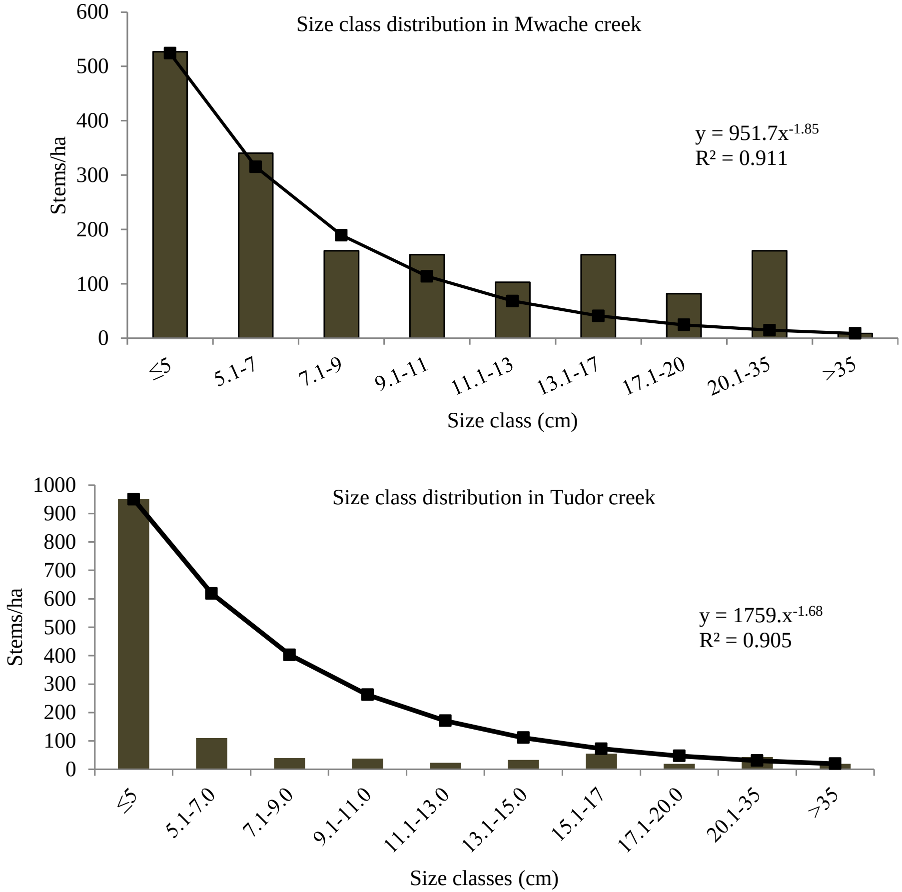

Fig. 3. Size class distribution.

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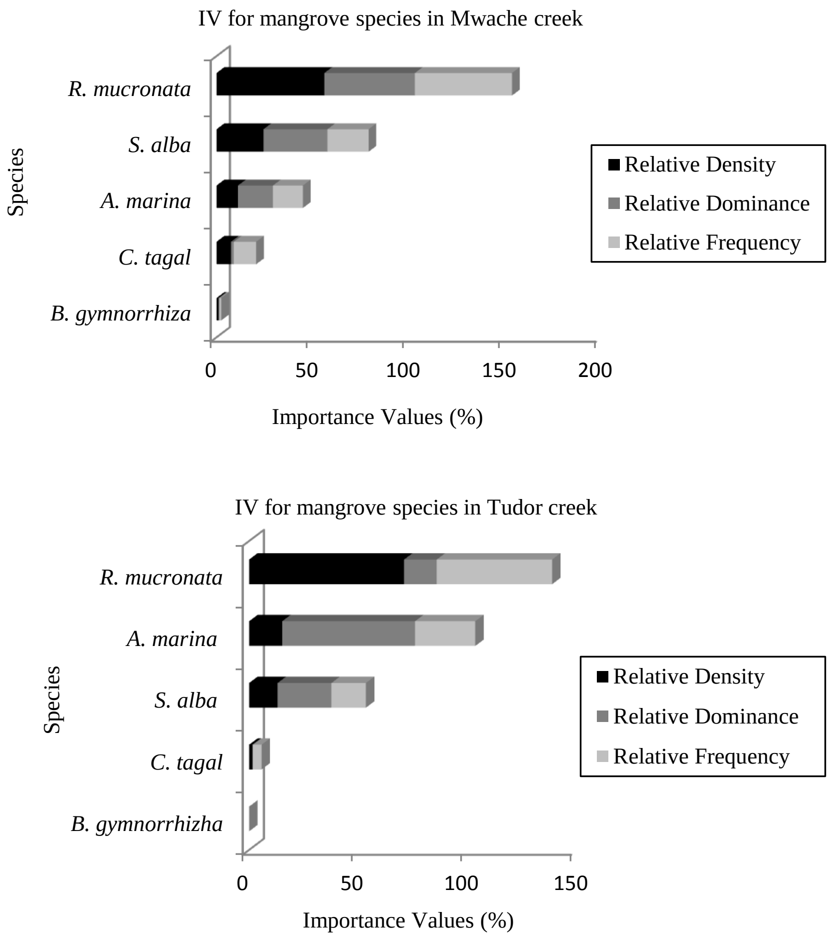

Fig. 4. Importance Values for the various species in Mwache and Tudor creeks.

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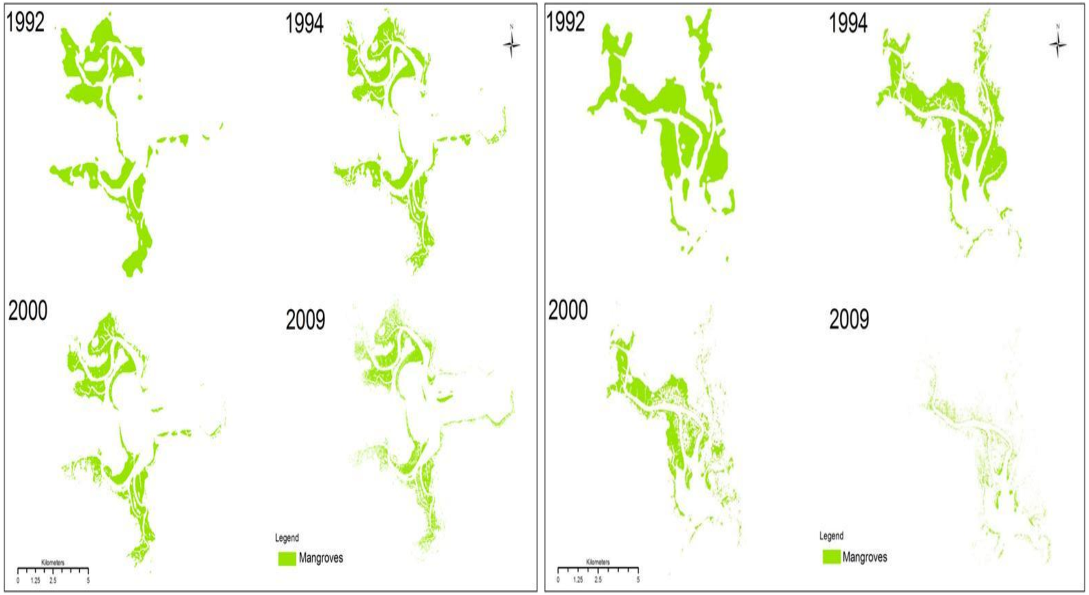

Fig. 5. Forest cover change from 1992 to 2009 in Mwache and Tudor creek, respectively.

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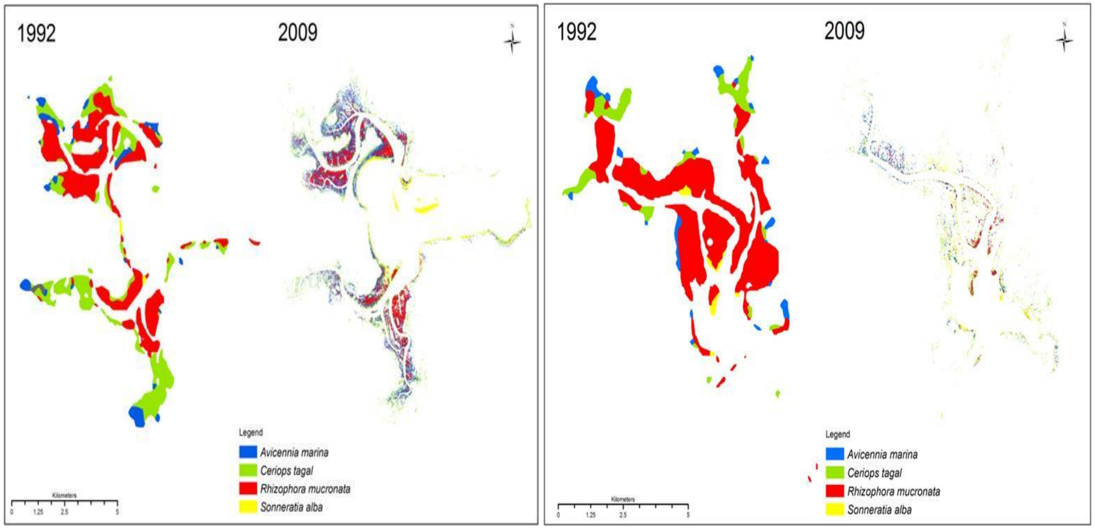

Fig. 6. Species shift between 1992 and 2009 in Mwache and Tudor creeks, respectively.

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