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Effects of Urban Wastewater Loading on Macro- and Meio-infauna Assemblages in Subtropical and Equatorial East African Mangroves

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Abstract—Benthic infaunal community studies are a useful tool to determine the health of an ecosystem. In this study, an unbalanced ACI (After Control/Impact) design was used to compare the infaunal community structure in a contaminated peri-urban mangrove swamp with two nearby pristine mangroves of similar ecological traits in Kenya and Mozambique. Densities of Oligochaeta decreased significantly in the macro-infaunal communities of contaminated mangroves in both countries. While crustaceans increased and molluscs decreased in the Mozambican peri-urban mangrove, a clear decrease of polychaetes was observed in the Kenyan equivalent. Meiofauna similarly responded in opposite ways, increasing in the Mozambican peri-urban mangrove and decreasing in the Kenyan equivalent. Diversity indices calculated using major meiofaunal groups were lower at contaminated sites. Differences observed between the peri-urban and pristine mangroves may be attributable to domestic sewage-derived contaminants in the former, but differences in mangrove structure and abiotic factors might also play an important role. Differences between the countries may relate to the fact that the peri-urban mangroves are contaminated differently. While in Mozambique, the sewage is discharged diffusely from the peri-urban zone and spreads into the mangrove through the creek, in Kenya it is discharged directly from the landward zone, making it more localized and concentrated.

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INTRODUCTION

In considering a polluted environment, levels of contamination need to be correlated with ecosystem health (Chapman, 2007b). Biomonitoring and bio-indicators show promise in observing the impact of external factors on ecosystems and their progression over a long period, or in differentiating between a polluted and non-polluted sites (Underwood, 1992a, 1994b). Both macro- and meiofauna are good biological candidates to assess the quality of a aquatic ecosystem and both exhibit repeatable patterns of response to environmental stressors (Chapman, 2007a).

Meiofauna have the advantage of being closely associated with the sediment matrix, thus changes in interstitial chemistry quickly lead to changes in meiofaunal abundance and diversity (Kennedy & Jacoby, 1999). Short generation times and asynchronous reproduction in the majority of meiofaunal species expose all stages in the life cycle to the pollutant, which results in a rapid response by the community to pollution (Coull & Chandler, 1992). Meiofauna may also persist in the absence of macrofauna and, in such cases, the characteristic composition of the residual assemblage may help to diagnose the type of pollutant (Kennedy & Jacoby, 1999). Macrofauna, on the other hand, have longer life expectancies, allowing the integration of environmental conditions over longer periods, as well as the accumulation of contaminants that are transferred to higher levels in the food web (Markert *et al.*, 2004).

Both communities have been successfully used to assess anthropogenic contamination in situ in river basins (Beier & Traunspurger, 2001; Chambers *et al.*, 2006; Heininger *et al.*, 2007; Saunders *et al.*, 2007), intertidal zones (Bigot *et al.*, 2006; Frouin, 2000; Rossi & Underwood, 2002; Solis-Weiss *et al.*, 2004; Sutherland *et al.*, 2007; Wear & Tanner, 2007; Yu *et al.,* 1997), the ocean floor (La Rosa *et al.*, 2001), and in experimental mesocosms subjected to specific contaminants (Austen & McEvoy, 1997; Austen *et al.*, 1994; Gyedu-Ababio & Baird, 2006). Most studies have focused on heavy metals. Sewage effluents can affect the benthos in a variety of ways by changing salinity regimes and introducing contaminants (such as metals, pesticides and hydrocarbons). Also, they increase detritus and nutrient availability that cause eutrophication and anoxic events, severely affecting the structure and dynamics of biological communities (Diaz & Rosenberg, 2008; Pearson & Rosenberg, 1978; Wu, 2002).

East African peri-urban coastal areas receive large quantities of untreated sewage, which is typically discharged into creeks lined by mangrove forests (Kruitwagen, *et al.*, 2008; Mohammed, 2002). Preliminary results from field trials on peri-urban mangrove systems (compared with pristine areas) showed that sewage loading did not produce harmful effects on the higher plant communities (Wong *et al.*, 1997), while its role on the benthic macrofauna community structure was found to be insignificant (Yu *et al*., 1997) or it severely affected faunal diversity and/or biomass (Cannicci *et al.,* 2009; Machiwa & Hallberg, 1995; Penha-Lopes *et al.*, 2009b, 2010c). Studies on the influence of anthropogenic sewage on mangrove-associated meiofauna are nonexistent. However, it is known that changes in community structure and dynamics, as well as behaviour, can lead to a significant decrease in the sediment turnover potential and consequently a loss of mangrove ecosystem functioning (Bartolini *et al.,* 2009; Bartolini *et al.*, 2010; Coleman & Williams, 2002; Penha-Lopes, *et al.*, 2009a, 2010a).

In order to save time and resources, the effect of environmental or anthropogenic disturbance on meio- and macrofauna communities can be detected by multivariate analysis at the higher taxonomic levels of these biota, from family to phylum (Chapman, 1998; Gesteira *et al.*, 2003; Kennedy & Jacoby, 1999; Savage *et al.*, 2001; Thorne & Williams, 1997). This is particularly relevant in countries where limited taxonomic expertise or resources are available, placing greater emphasis on cost-effective techniques such as taxonomic minimalism (Thorne & Williams, 1997), which is also currently the case in most mangrove meiofaunal studies (Nagelkerken *et al.*, 2008).

The present study formed part of the PUMPSEA Project (funded by European Commission: FP6, INCO-CT2004-510863), the global aim of which was to demonstrate the ecological and economical ecosystem services provided by mangroves in filtering discharged wastewater, and thereby limiting coastal sewage pollution, thus improving coastal zone management within East Africa. The present study aimed at investigating differences in infaunal structure, at high taxonomic level, between peri-urban mangroves known to be impacted by sewage dumping and non-urban sites where wastewater dumping was not evident, using basic (simple and inexpensive) metrics applicable in developing countries. The study was thus intended to provide information for the development of biological indices to monitor and manage mangrove environments. The reliability of infauna as bioindicators of domestic sewage pollution was not its goal, needing this to be addressed in further studies.

MATERIALS AND METHODS

Study area and sampling design

Appropriately replicated Before/After sampling designs are perhaps the most reliable methods for detecting an environmental impact (Underwood, 1992b, 1994a). However, in the absence of such data, it has nevertheless been possible to assess differences between potentially disturbed and control locations after the ‘‘disturbance’’ with appropriate spatial and temporal replication (Chapman *et al.*, 1995). For these reasons, we followed a unbalanced ACI (After Control Impact) design, comparing the ecological features recorded in a peri-urban mangrove swamp with those of two nearby, pristine mangroves of similar ecological traits in two countries, Kenya and Mozambique.

The mangrove areas, sampling design and site selection have already been extensively described in previous publications (Cannicci, *et al.,* 2009; Penha-Lopes *et al*, 2009b). Three sampling sites were chosen on the southern Kenya coast (Figure 1), Mikindani (impacted site), Gazi Bay and Shirazi creek (pristine sites). The Mikindani mangrove system is located within the Tudor Creek which surrounds the city of Mombasa. This mangrove has been exposed to sewage for more than a decade, being primarily affected by sewage from the Mikindani residential estate, but also partially by Mombasa city sewage (Mohammed, 2008). The sewage is discharged directly into the landward zone of the nearby mangrove forest. The mangroves are subsequently dosed with sewage every tidal cycle and the loading reduces exponentially with distance from the source (PUMPSEA, 2008).

In Mozambique, sampling was conducted in three different mangrove ecosystems, a peri-urban mangrove, Costa do Sol, and two pristine mangroves, Saco and Ponta Rasa at Inhaca Island (Figure 1). Costa do Sol mangrove is located within the north region of Maputo city. The seasonal river, Quinhenganine, discharges into the swamp after passing through the city. The mangrove is bordered by a residential area, and has been receiving domestic sewage, aquaculture residues and solid waste from various sources over the last decades (PUMPSEA, 2008). Costa do Sol has higher concentrations of nutrients, especially nitrites and nitrates, relative to the Saco and Ponta Rasa mangroves (PUMPSEA, 2008).

At each site, a stratified random sampling approach was used, since these mangroves manifest clear zonation in their vegetation, characterising different ecological features colonised by different faunal assemblages (Hogarth, 2007; Macnae, 1968). The zones considered were: a landward sandy belt dominated by *Avicennia marina* trees, representing the zone only flooded at spring tides (hereafter called the *Avicennia* belt); and a seaward, muddy belt dominated by *Rhizophora mucronata* trees, flooded twice a day by high tides (from now on called the *Rhizophora* belt). At the Costa do Sol site (southern Mozambique) only a wide *A. marina*-dominated area is present and we concentrated our efforts on this zone, comparing it with the corresponding landward *Avicennia* belts at the Mozambican control sites.

Sample collection and processing

At each sampling site, two *A. marina* and two *R. mucronata* study areas were selected, 50 m apart. In each area, two sub-areas were delimited, separated by more than 5 m. In each sub-area, three replicates (within a diameter of 1 m) were sampled for meiofauna (5 cm Ө corer to 20 cm depth) and three for macrofauna (15 cm Ө corer to 20 cm depth), preserved in 10% formalin. Although three replicates were sampled and analyzed, data were pooled and treated statistically as a single replicate to cover the minimum required sampling area. Two temporal replicates were sampled in a dry season sampling campaign during the months of July and August 2005 in Mozambique in October 2005 at the Kenyan locations. A second campaign followed in the wet season, in February 2006 in Mozambique and in April 2006 in Kenya. Meiobenthos were extracted from the sediment samples by retention on a 63 µm sieve after washing the sample with tap water through a 500 µm sieve to remove coarse root material and macrobenthos. The material retained on the fine sieve was thoroughly mixed with Ludox HS40® (density 1.15 g.cm^{-3}). The filtrate was washed on the 63 µm sieve with tap water (Burgess, 2001) and the material retained preserved in 4% formalin to which a small amount of Rose Bengal stain was added to aid counting. Macro- and meiobenthos were counted and identified to phylum, order or class.

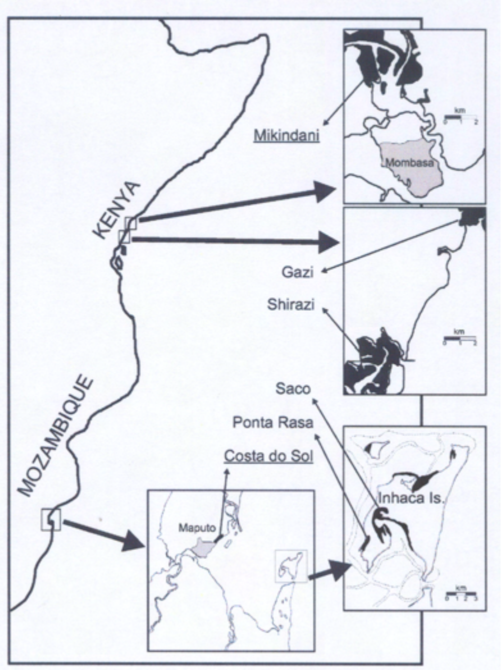

Figure 1. Map of sampling sites in Kenya and Mozambique.

Data analysis

The data were analysed using univariate and non-parametric multivariate techniques in the PRIMER (Plymouth Routines In Multivariate Ecological Research) package (Clarke & Gorley, 2006). For meiofauna groups, generic diversity was assessed by using the Shannon-Wiener diversity index (H'), Pielou's evenness index (R') and Margalef's species richness index (d). The nematode to copepod ratio was also calculated by dividing the number of nematodes in a sample by the number of copepods. These indices were statistically evaluated using two-way ANOVA. Non-metric multi-dimensional scaling (NMDS) was used to represent the multivariate dimensions of the community structure, based on lower triangular similarity matrices constructed using the Bray-Curtis similarity measure of square root (macrofauna) and fourth root (meiofauna) transformed data (Clarke, 1993; Clarke & Green, 1988). For macrofauna, a 3- or 4-way ANOVA was used to test for differences in each of the major groups between sites within Kenya, Mozambique and both countries. Distance-based permutational multivariate analysis of variance (PERMANOVA) (Anderson, 2001; McArdle & Anderson, 2001) was performed to test for the null hypothesis of no differences in meiofaunal assemblages between sites over time (Anderson & Ter Braak, 2003). A SNK test (Post-hoc test) was used in conjunction with ANOVA to determine which specific group pairs were statistically different from each other when significant differences between samples were observed. Species contributing to dissimilarity between sites were investigated using similarity percentage procedure (SIMPER) (Clarke, 1993). For all analyses, ANOVA assumptions were tested for homogeneity of variance (tested with Cochran C) and normal distribution (tested with Shapiro-Wilks).

RESULTS

Macrofauna

Four major macrofaunal groups were recorded: Polychaeta, Oligochaeta, Mollusca and Crustacea. In the *Avicennia* belt in both countries, macrofaunal densities were similar between contaminated and pristine mangroves (Table 1 - A), except for polychaetes, which occurred in very low abundance at Mikindani, and crustaceans that were absent from the Saco and Costa do Sol mangroves during the wet season (Figure 2). Oligochaetes and molluscan assemblages were highly variable at small spatial scales.

In the Kenyan contaminated mangrove, Mikindani, there was a clear decrease in the abundance of oligochaetes in both tree zones, and polychaetes in the *Avicennia* belt, compared to the pristine mangroves (Figure 2, Table 1 - B and SNK tests). MDS plots on the Kenyan data showed that Mikindani was separated from pristine locations in both tree belt and season (although more strongly in the wet season), while both pristine locations were barely separable (Figure 3). SIMPER analysis between sites corroborated these results, with Oligochaeta contributing to clear differences between sites in both belts (dissimilarity 36-45%), more clearly elucidating the separation of the peri-urban mangroves from those pristine.

In Mozambique, there was a clear decrease in molluscs and oligochaetes at Costa do Sol and, while most faunal groups decreased during the wet season, polychaetes increased in density (Figure 2, Table 1 - C and SNK tests). The MDS plots clearly separated all three sampling sites (Figure 3). SIMPER analysis showed that Mollusca and Crustacea (together contributing nearly 66%) were the groups responsible for the separation of the Costa do Sol mangrove (peri-urban) from the Saco (dissimilarity 81%), while Mollusca (49%) were the group that contributed most towards dissimilarities (39%) between the Costa do Sol and Ponta Rasa mangroves.

Meiofauna

Ten major taxa were recorded: Copepoda, Foraminifera, Halacaroidea, Insecta, Mollusca, Nematoda, Oligochaeta, Polychaeta, Tardigrada and Turbellaria. Nematodes were the dominant group (43-98%) in all samples with copepods usually the second most abundant taxon. Halacaroidea and Turbellaria were also relatively abundant in many samples (Figure 4). In the *Avicennia* belt, significant interactions (see Table 2 - A) were found between seasons and sites in both countries, with a significant decrease of the most abundant meiofauna groups in the Kenyan contaminated mangrove, while the opposite was true in Mozambique (Figure 4).

In Kenya, significant interactions were found between Season x Site and Belt x Site (Table 2 - B). In the *Rhizophora* belt in the non-urban mangroves, nematodes were more abundant and Halacaroidea and Turbellaria decreased significantly (Figure 4) when compared with Avicennia belt (SNK tests). Both of these meiofaunal groups were almost absent in the peri-urban mangrove and significantly lower in density when compared with pristine mangroves (SKN tests).

While MDS plots (Figure 5) for combined Kenyan *Avicennia* and *Rhizophora* belts manifested no significant differences between assemblages, data for the *Avicennia* belt at Mikindani separated out, although with some overlap, from both pristine mangroves, particularly in the wet season. The MDS plots for the *Rhizophora* belt more or less overlapped in the wet season but were well separated in the dry season (Figure 5).

In Mozambique, MDS plots for the *Avicennia* belts revealed that all the mangroves were clearly separated (Figure 5). PERMANOVA results corroborated the MDS plots and there were significant differences between the sites (Table 2). SNK post-hoc tests revealed significant differences between all three mangrove meiofaunal communities. Meiofaunal composition differed between Saco and Ponta Rasa (dissimilarity = 68%), mainly due to the contribution of nematodes and Foraminifera. Differences in abundance in nematodes and halacaroids were responsible for nearly half of the dissimilarity between the Saco and Ponta Rasa compared to Costa do Sol (dissimilarity = 76%). Copepods, nematodes and polychaetes were the groups that contributed nearly half for the dissimilarity between the Saco and Costa do Sol (dissimilarity = 38%).

In Mozambique, the Costa do Sol (peri-urban) mangrove had similar diversity indices in terms of its major meiofaunal groups to both pristine mangroves (Table 3), except for a much higher nematode/copepod ratio during the wet season. In Kenya, Margalef's index was significantly lower at Mikindani in both tree belts, particularly in the wet season, compared to both pristine mangroves (Table 3). The Shannon-Wiener diversity index obtained at Mikindani was significantly different in the the *Rhizophora* belt from the other two mangroves during the wet season. The nematode to copepod ratio in Kenya was similar in both the contaminated and pristine mangroves, except in the dry season when the Gazi ratios were significantly different in the *Avicennia* belt at Mikindani compared to both other mangroves in the *Rhizophora* belt (Table 3).

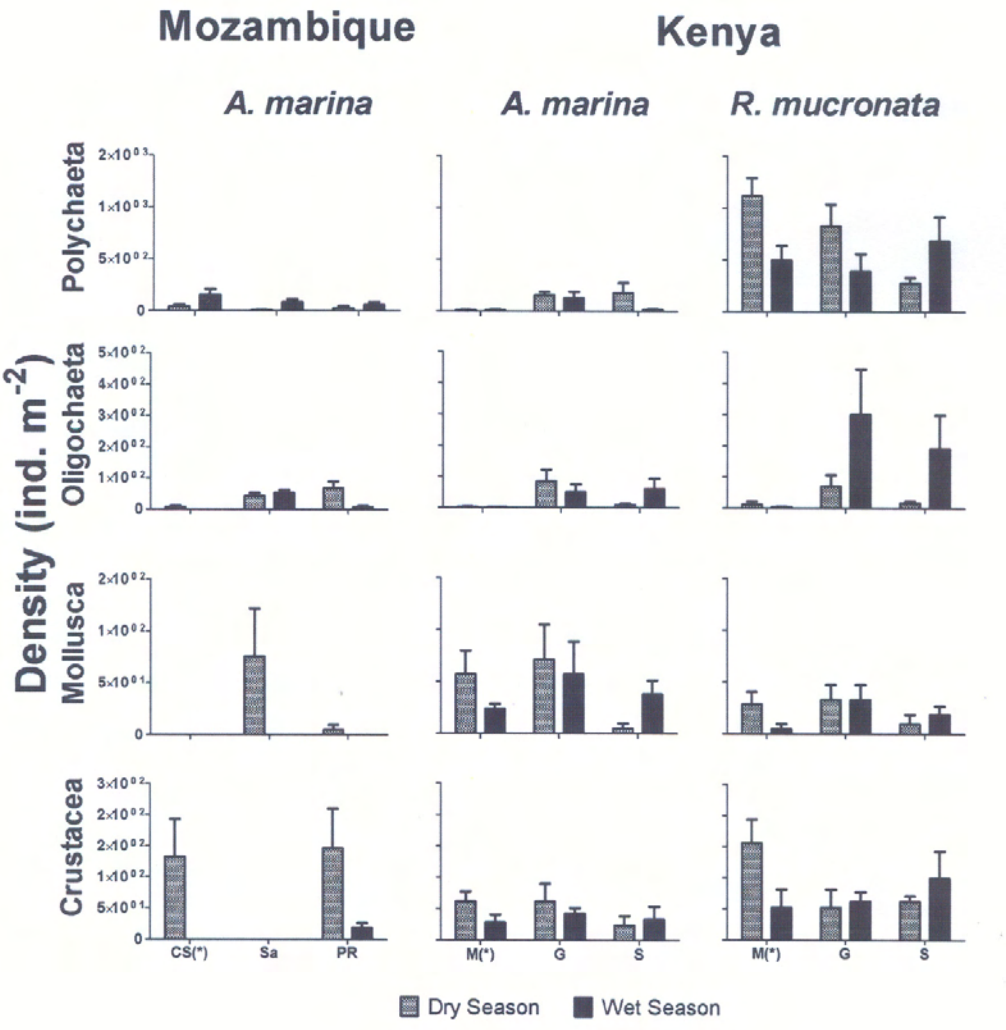

Figure 2. Average (±SE) major macrofaunal group abundance (m^{-2}) in contaminated and pristine mangrove sites in Kenya and Mozambique in the dry and wet season.

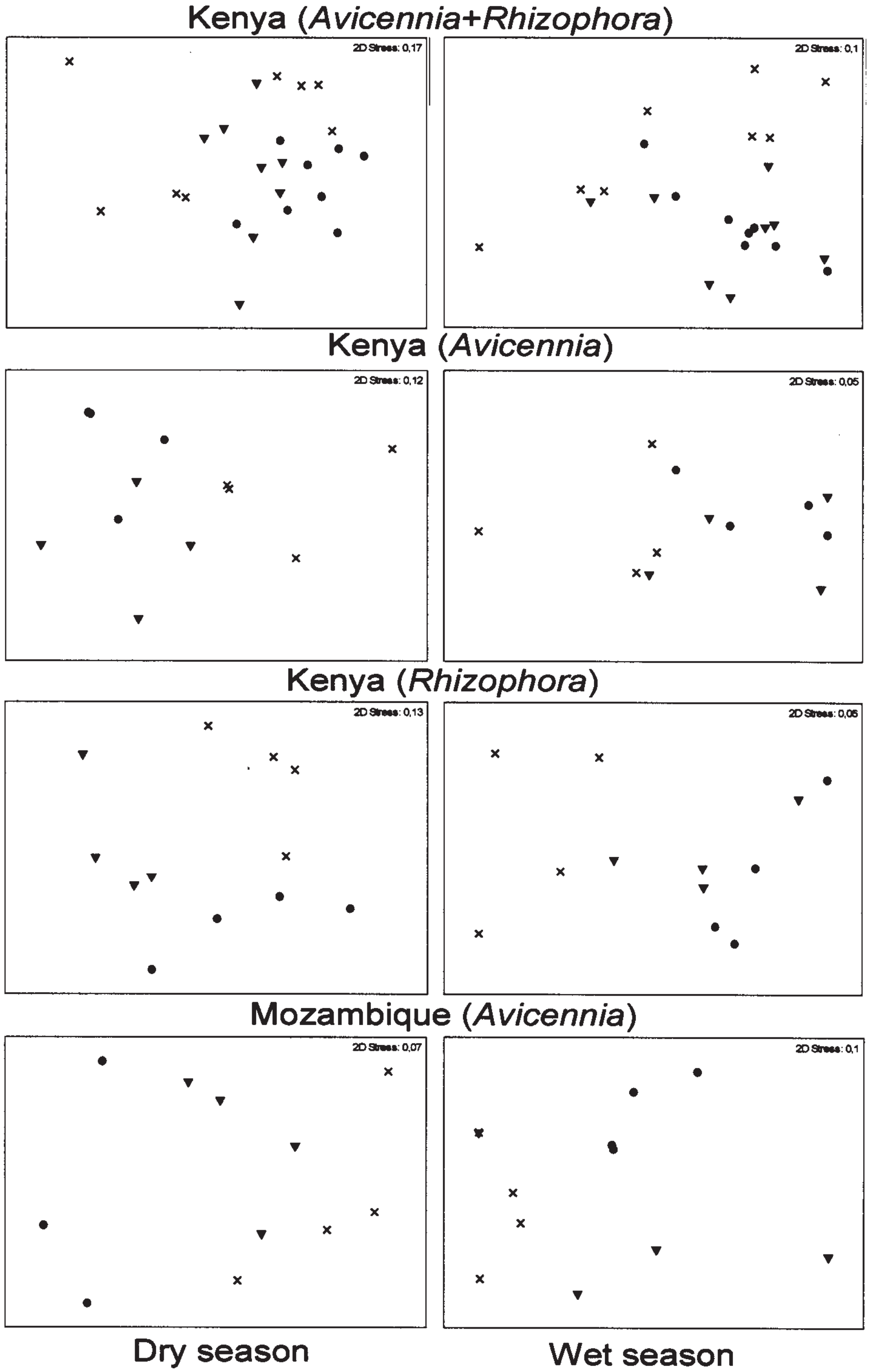

Figure 3. Non-metric multidimensional scaling (NMDS) ordination of root-transformed macro infaunal abundance data for three sampling sites in *Avicennia* and *Rhizophora* belts in Kenya: Mikindani (x), Gazi (●), Shirazi (▼) and Mozambique: Costa do Sol (x), Saco (●), Ponta Rasa (▼).

Table 1. Results of 3- or 4-way ANOVA on macro infaunal density, log (x+1) transformed data for the three mangrove sampling sites: A - in both countries (*Avicennia* belt); B - Kenya (*Avicennia* and *Rhizophora* belt) and C - Mozambique (*Avicennia* belt). The factors used in the analysis were: Season (fixed and orthogonal), Country (fixed and orthogonal), Belt (random and orthogonal), Site (Country) (random and nested in country), Site (random and orthogonal) and/or Area (Site) (random and nested in Site). Degrees of freedom (df), mean squares (MS) and F-ratio value (F).

		Crustacea		Polychaeta		Oligochaeta		Mollusca	
	d.f.	MS	F	MS	F	MS	F	MS	F
A - Kenya and Mozambique									
Season	1	1,62	4,68	0,13	0,48	0,04	0,04	0,16	0,65
Country	1	0,10	0,17	0,12	0,10	9,84	2,65	1,49	5,56
Site	4	0,58	157,81[b]	1,17	9,25[b]	3,71	3,18	0,27	1,09
Area	6	0,00	0,02	0,13	0,61	1,17	4,52[b]	0,25	3,38[a]
Season x Country	1	0,78	2,26	3,38	12,07[a]	0,29	0,30	0,17	0,69
Season x Site	4	0,35	14,07[b]	0,28	0,84	0,98	3,52	0,25	1,74
Season x Area	6	0,02	0,12	0,33	1,59	0,28	1,08	0,15	1,99
Residual	24	0,21		0,21		0,26		0,07	
Total	47								
B - Kenya									
Season	1	0,17	0,34	1,60	8,61	3,59	1,17	0,01	0,02
Belt	1	1,03	4,74	33,03	17,20	5,09	12,92	0,44	11,16
Site	2	0,09	1,00	0,59	1,36	18,28	27,53[a]	0,37	4,21
Area	3	0,09	0,49	0,44	2,24	0,66	1,06	0,09	0,96
Season x Belt	1	0,00	0,01	0,04	0,02	2,39	1,96	0,01	0,22
Season x Site	2	0,52	6,27	0,19	0,39	3,06	2,03	0,32	4,05
Season x Area	3	0,08	0,44	0,48	2,44	1,51	2,40	0,08	0,87
Belt x Site	2	0,22	2,15	1,92	12,06[a]	0,39	0,20	0,04	0,23
Belt x Area	3	0,10	0,54	0,16	0,82	2,00	3,18[a]	0,17	1,86
Season x Belt x Site	2	0,12	5,75	1,82	7,15	1,22	7,76	0,04	0,34
Season x Belt x Area	3	0,02	0,11	0,25	1,30	0,16	0,25	0,12	1,28
Residual	24	0,19		0,20		0,63		0,09	
Total	47								
C - Mozambique									
Season	1	0,34	1,37	2,43	27,48[a]	2,33	3,84	0,27	0,86
Site	2	0,25	0,89	0,31	4,09	1,02	274,01[b]	0,64	16,94[a]
Area	3	0,28	30,56[b]	0,08	0,36	0,00	0,02	0,04	1,04
Season x Site	2	0,25	0,89	0,09	0,26	0,61	26,73[a]	0,31	10,97[a]
Season x Area	3	0,28	30,56[b]	0,34	1,61	0,02	0,09	0,03	0,78
Residual	12	0,01		0,21		0,24		0,04	
Total	23								

(a = $p < 0.05$; b = $p < 0.01$)

DISCUSSION AND CONCLUSION

Pollution events can exert different effects on exposed biota, generating changes in their community structure and ecosystem function (Parker, *et al.*, 1999). Infaunal species are considered to be key organisms for the detection of ecosystem changes, including those caused by human activities and contamination, due to their short life span and low mobility (Warwick & Clarke, 1993; Chapman, 2007). However, infaunal assemblages are influenced not only by the

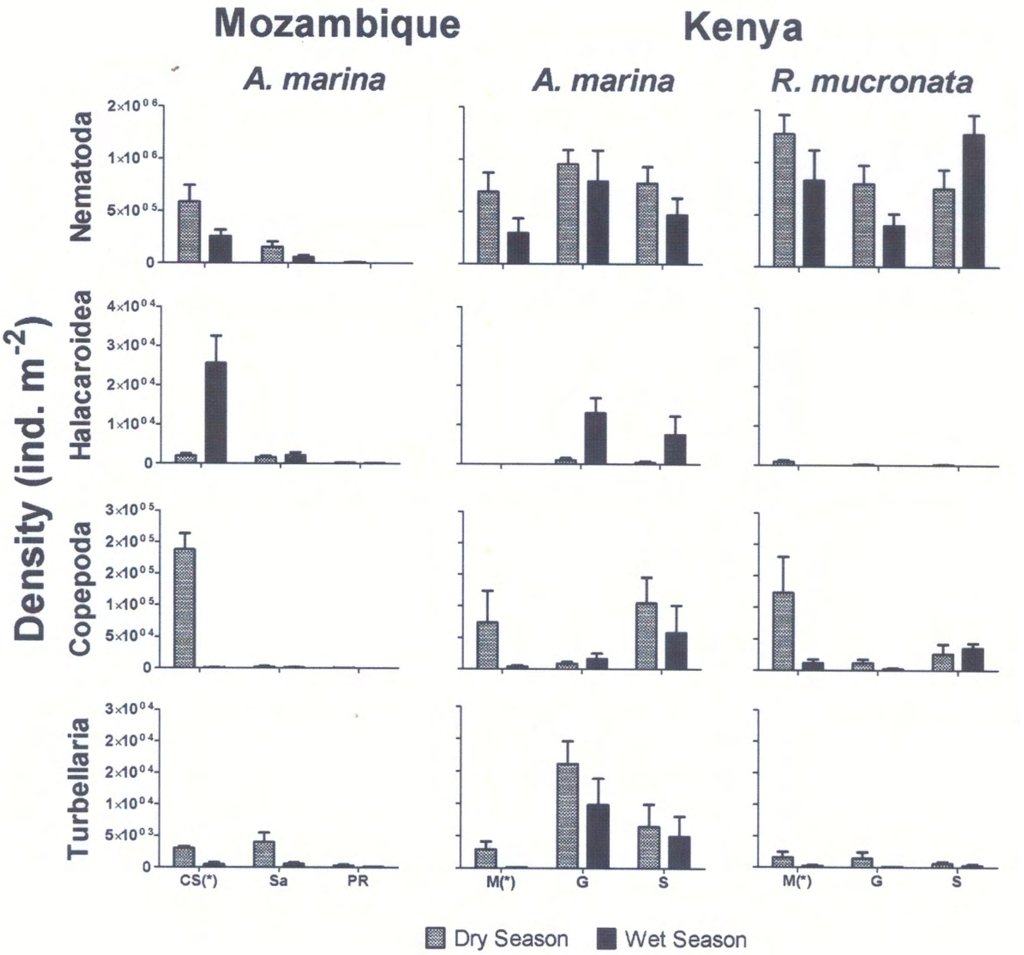

Figure 4. Average (±SE) major meiofaunal group abundance (m^{-2}) in contaminated and pristine mangrove sites in Kenya and Mozambique in the dry and wet season.

bioavailability and toxicity of contaminants (Markert *et al.*, 2004), but also several other factors, such as hydrology, sediment characteristics and abiotic conditions (e.g., temperature and salinity) (Alongi, 1987b; Coull, 1999; Lee, 2008). The abundance of infauna in soft sediments can be patchy at a range of spatial scales, from meters up to several kilometres (Chapman & Tolhurst, 2004; 2007; Morrisey *et al.*, 1992) and these communities also undergo strong temporal changes linked to natural and unpredictable fluctuations in ecological factors in the systems they inhabit (Underwood & Chapman, 1996).

Anthropogenic stress tends to exacerbate this spatial and temporal patchiness by increasing variability in the abundance of taxa or changing the taxonomic composition at affected sites (Warwick & Clarke, 1993). For this reason, nested sampling designs incorporating various spatial and temporal scales must be used to disentangle human impacts from natural variations in infaunal communities, (Cannicci *et.,* 2009; Warwick & Clarke, 1993). Although the present study yielded high spatial variability between samples, there was a clear distinction between communities inhabiting pristine and contaminated mangroves in both Kenya and Mozambique.

Nematodes, together with copepods and turbellarians, as well as larger organisms such as polychaetes and oligochaetes, comprised more than 90% of the infauna, a typical observation in most mangrove areas, in densities within the range recorded in previous studies (e.g., Alongi, 1987a; Fondo & Martens, 1998; Nagelkerken *et al.*, 2008). However, the meiofauna responded in opposite ways to anthropogenic contamination, increasing in Mozambican and decreasing in Kenyan peri-urban mangroves.

Within the macrofauna, only oligochaetes manifested a decrease under contamination conditions in both countries, confirming their sensitivity and usefulness as bioindicators of

Table 2. PERMANOVA on Bray-Curtis distances for abundance of major meiofaunal groups recorded at the three mangrove sampling sites: A - in both countries (*Avicennia* belt); B - Kenya (*Avicennia* and *Rhizophora* belt) and C - Mozambique (*Avicennia* belt). The factors used in the analysis were: Season (fixed and orthogonal), Country (fixed and orthogonal), Belt (random and orthogonal), Site (Country) (random and nested in country), Site (random and orthogonal) and/or Area (Site) (random and nested in Site). Degrees of freedom (df), sum of squares (SS), mean squares (MS) and F-ratio value (F).

Source	df	SS	MS	F	P
A - Kenya and Mozambique					
Season	1	3057,22	3057,22	2,01	0,13
Country	1	16668,02	16668,02	2,81	0,05
Site(Country)	4	23711,77	5927,94	11,44[b]	0,00
Area(Site)	6	3108,06	518,01	1,03	0,44
Season x Country	1	966,28	966,28	0,64	0,67
Season x Site	4	6073,49	1518,37	2,44[a]	0,02
Season x Area	6	3725,55	620,93	1,23	0,23
Residual	24	12098,73	504,11		
Total	47	69409,13			
B – Kenya					
Season	1	2204,75	2204,75	2,42	0,14
Belt	1	2411,14	2411,14	2,40	0,14
Site	2	1375,53	687,77	1,37	0,31
Area(Site)	3	1508,28	502,76	2,71[b]	0,01
Season x Belt	1	562,64	562,64	1,01	0,45
Season x Site	2	1819,37	909,69	3,49[a]	0,03
Season x Area	3	780,79	260,26	1,41	0,19
Belt x Site	2	2006,55	1003,28	6,42[b]	0,00
Belt x Area	3	468,41	156,14	0,84	0,58
Season x Belt x Site	2	1109,39	554,70	2,05	0,14
Season x Belt x Area	3	810,03	270,01	1,46	0,17
Residual	24	4438,78	184,95		
Total	47	19495,68			
C - Mozambique					
Season	1	2267,61	2267,61	1,03	0,43
Site	2	20912,57	10456,28	19,54[b]	0,00
Area(Site)	3	1605,25	535,08	0,72	0,70
Season x Site	2	4418,25	2209,12	2,41	0,10
Season x Area	3	2754,61	918,20	1,23	0,29
Residual	12	8942,77	745,23		
Total	23	40901,05			

(a = $p < 0.05$; b = $p < 0.01$)

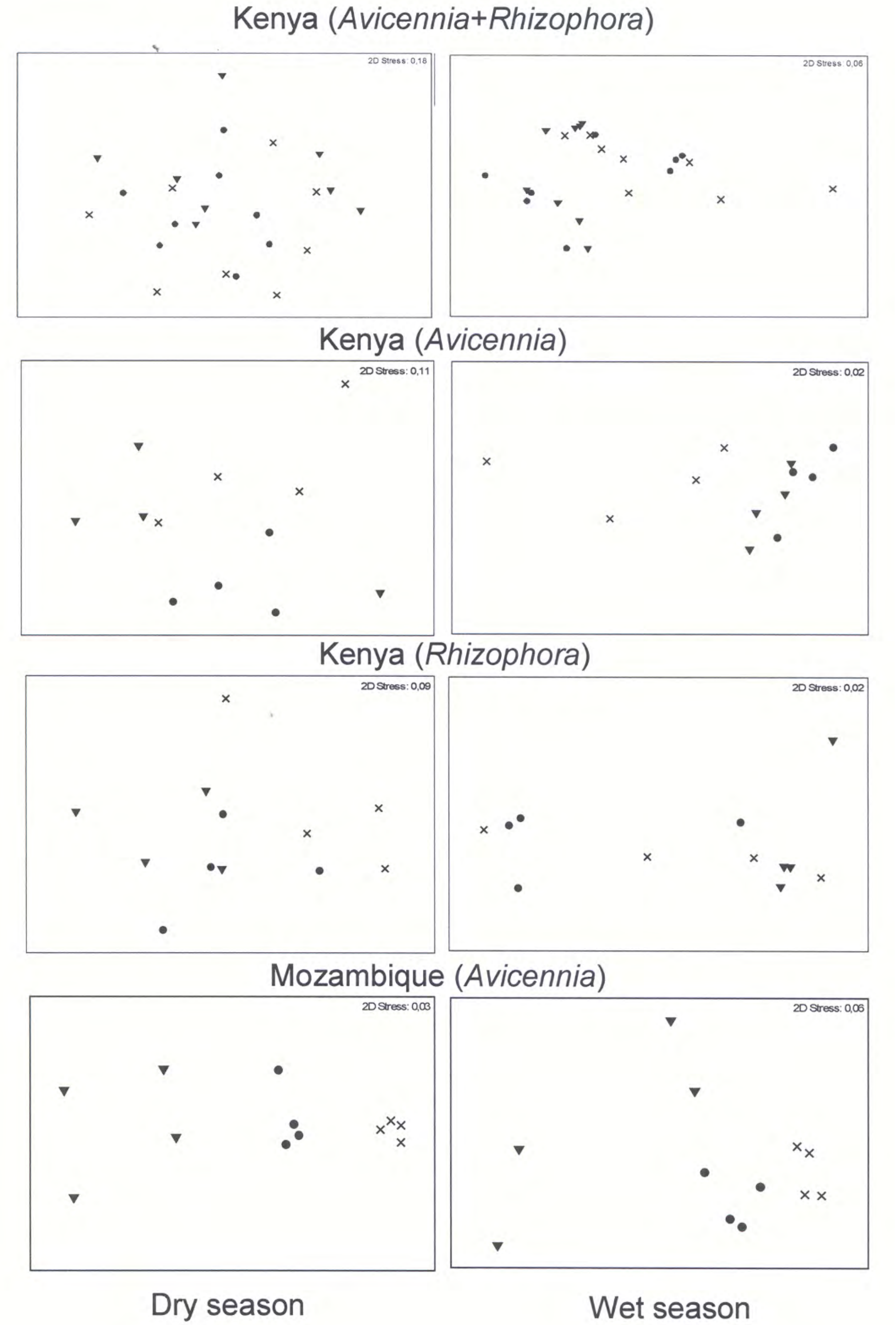

Figure 5 - Non-metric multidimensional scaling (NMDS) ordination of root-transformed meiofaunal abundance data for three sampling sites in *Avicennia* and *Rhizophora* belts in Kenya: Mikindani (x), Gazi (●), Shirazi (▼) and Mozambique: Costa do Sol (x), Saco (●), Ponta Rasa (▼).

organic pollution (Chapman, 2001; Lin & Yo, 2008; Markert *et al.*, 2004). In Mozambique, there was a significant decrease in molluscs and an increase in crustaceans and most meiofaunal groups (such as nematodes, halacaroideans and copepods) in the peri-urban mangrove, probably because of an increase in sediment organic matter, microphytobenthos and bacteria. A parallel study on macro-epifauna, such as fiddler and sesarmid crabs, also indicated that the peri-urban systems were richer, in diversity, than the mangroves unaffected by urban sewage; these are known to feed on benthic microalgae and bacteria or the substratum and leaf litter, respectively (Cannicci *et al.* 2009). Cannicci *et al.* (2009) recorded the complete eradication of certain mollusc species (e.g., *Terebralia palustris*) from the contaminated mangroves which was probably caused by their low tolerance to this sort of pollution. In Kenya, the decrease in Oligochaeta, Polychaeta, Halacaroidea and Turbellaria, largely in the *Avicennia* belt in the peri-urban mangrove, was probably indicative of higher organic loadings and other anthropogenic contaminants known to affect these communities (e.g., Danovaro *et al.*, 1995; Gillet *et al.*, 2008). The fact that this primarily affected the *Avicennia* belt was probably attributable to the landward dumping of sewage at Mikindani, before it flowed through the *Rhizophora* belt. Since both the

Table 3. Comparison of average (±SE) richness (d), evenness (J') and diversity (H') indices and the nematode/copepod ratio in meiofaunal communities at contaminated and pristine locations in the dry and wet season in Kenya and Mozambique (superscript letters indicate significant differences ($p < 0.05$) between season for the same site; superscript numbers indicate significant differences ($p < 0.05$) between sites for the same season).

Kenya *A. marina*	Season	Margalef's index (d)	Pielou's index (J')	Shannon index (H')	Nematod/Copepod Ratio
Mikindani	Dry	0.69±0.07	0.19±0.06	0.33±0.10	177±82^{a}
	Wet	0.60±0.04^{a}	0.16±0.05	0.21±0.05	288±106
Shirazi	Dry	0.86±0.08	0.16±0.03	0.31±0.06	301±239
	Wet	0.86±0.06^{b}	0.14±0.03	0.32±0.05	410±130
Gazi	Dry	0.76±0.08	0.13±0.02	0.25±0.03	881±295^{b}
	Wet	0.98±0.06^{b}	0.14±0.03	0.30±0.05	410±131
Kenya *R. mucronata*					
Mikindani	Dry	1.05±0.12$^{(1)}$	0.15±0.03	0.31±0.05$^{(1)}$	151±60^{a}
	Wet	0.61±0.05$^{(2)a}$	0.07±0.01	0.12±0.02$^{(2)a}$	445±47
Shirazi	Dry	1.00±0.08	0.11±0.02	0.23±0.04	189±94^{a}
	Wet	0.81 ±0.07^{b}	0.10±0.01	0.20±0.03^{b}	283±113
Gazi	Dry	0.94±0.04	0.12±0.01	0.25±0.04	638±86^{b}
	Wet	0.99±0.04^{b}	0.12±0.02	0.25±0.04^{b}	467±87
Mozambique *A. marina*					
Costa do Sol	Dry	1.03±0.02	0.35±0.03	0.75±0.06	3±1$^{(1)}$
	Wet	1.03±0.07	0.29±0.05	0.58±0.09	291±26$^{(2)a}$
Saco	Dry	0.91±0.08	0.42±0.07	0.76±0.14	102±53
	Wet	1.05±0.12	0.29±0.05	0.52±0.10	85±43^{b}
Ponta Rasa	Dry	1.47±0.34	0.54±0.14	0.77±0.18	10±5
	Wet	4.39±3.24	0.65±0.22	0.97±0.40	1±0^{b}

soil and vegetation in the *Avicennia* belt can efficiently absorb an overload of nutrients (Tam & Wong, 1995; Wong *et al.,* 1997), the landward *Avicennia* belt at Mikindani is probably acting as the first phytoremediation system, mitigating the effect of dumping before it reaches the seaward *Rhizophora* belt of the forest. This was also observed by Cannicci *et al.* (2009) who found that macro-epibenthic fauna were more severely affected in landward (*Avicennia*) mangroves.

In environmental monitoring, multivariate techniques have traditionally required detailed taxonomic data, but this is no longer the case. For a variety of benthic organisms (macrobenthos, meiobenthos and reef corals), aggregation trials suggest that little information is lost when assessing perturbation using higher taxonomic data alone (Warwick, 1988). This is based on the assumption that anthropogenic stresses modify community structure at a higher taxonomic level than that caused by natural environmental variation. Taxonomic minimalism thus facilitates discrimination between human and natural perturbation. In the present study, MDS analysis clearly differentiated between the meio- and macrofaunal groups in peri-urban mangroves and control sites in both countries, regardless of mangrove zone or season. Nevertheless, the low taxonomic resolution may not be sensitive enough to distinguish changes in functional groups within the same taxa, as well as changes in animal biomass or behaviour, which are known to have significant effects on ecosystem function.

Diversity indices have been widely used in many ecosystems (Kennedy & Jacoby, 1999; Markert *et al.*, 2004), including mangrove forests (e.g., Bosire *et al.*, 2004), as ecological indicators. The diversity values obtained in this study were significantly lower than those obtained in other studies due to the lone use of higher taxonomic levels. Nevertheless, reduced diversity indices were obtained for contaminated sites. In Kenya, the Margalef and Shannon-Wiener indices at Mikindani were indicative of lower diversity than the pristine mangroves, although the nematode/copepod ratio were suggestive of lower contamination stress (Raffaelli & Mason, 1981). Contrastingly, at Costa do Sol, the latter index was only indicative of severe contamination during the wet season, when rain possibly washes contaminants from higher ground into the mangrove creek.

Very little is known about species interactions in meio- and macrobenthos, and much is yet to be learnt of processes that shape community structure and function in complex and productive mangrove environments (Lee, 2008), especially when subjected to intensive anthropogenic stress. In studies reviewed by Oláfsson (2003), it is apparent that macrofaunal activity (predation, physical disturbance, competition for food and the creation of biogenic structures) significantly affects meiobenthic community structure, diversity and dynamics in soft substrata. In this way, the overall effects of sewage contamination on the infaunal community are more complex than the direct effects of the contaminants on individual organisms.

Mangrove infaunal species also serve as important food sources for resident and mobile nekton that enter the mangrove at high tide, including commercially important fish and crustacea, as well as several bird species that feed in mangroves (Lee, 2008; Nagelkerken *et al.*, 2008). These fauna play an important role in nutrient cycling and organic matter decomposition (such as litter), by feeding directly on detritus, performing bioturbation activities (sediment reworking and bio-irrigaton) and constructing biogenic structures that stimulate microbial activity (Cannicci *et al.,* 2008; Kristensen, 2000; Schrijvers *et al.*, 1995). This contributes to biogeochemical cycling and ecological function within mangrove sediments (Kristensen, 2008; Lee, 2008), and consequently to mangrove nutrient and carbon dynamics (Penha-Lopes *et al.,* 2010; Kristensen *et al.*, 2010), influencing the natural potential of mangroves for pollution buffering and sewage treatment.

The results obtained for the peri-urban and pristine mangroves examined in this study could thus be differentiated using univariate, multivariate and diversity analyses, although the meio- and macrofaunal communities responded differently in each country. While the sites chosen were deemed typical regarding their biotic and abiotic variables, it is known that mangrove ecosystems are patchy and dynamic; this may have lead to variation in these variables, both spatial and temporal. This may account for some differences between the Mozambican and Kenyan mangroves, as well as between some of the systems within each country, but the clear distinction between the infaunal communities (using diversity and multivariate indexes) at peri-urban and pristine sites was indicative of perturbation, discernible at a high taxonomic level. Furthermore, the abundance of oligochaetes decreased significantly in both contaminated mangroves, rendering them potential indicators of pollution in East African mangroves. This suggests that they alone may serve this purpose in East Africa but further research is needed to confirm this.

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