Vrije Universiteit Brussel

UNIVERSITÉ LIBRE DE BRUXELLES, UNIVERSITÉ D'EUROPE

ULB

Biologie

Laboratory of Plant Science and
Nature Management

Are Peri-urban Mangrove Forests Viable?

Effects of Sewage Pollution and Wood Exploitation on the Structure and Development of the Mangroves of Mombasa (Kenya)

Thesis submitted in Fulfilment of the Requirements for the Degree of Doctor of Philosophy in Science of the Vrije Universiteit Brussel and Université Libre de Bruxelles

Mohamed Omar Said MOHAMED

VUB Promotor: Prof. Dr. Nico KOEDAM
ULB Promotor: Prof. Dr. Farid DAHDOUH-GUEBAS
Kenyan Promotor: Dr. James G. KAIRO

Brussels November 2008

KENYA WILDLIFE SERVICE

VLIR UOS

Are Peri-urban Mangrove Forests Viable?

Effects of Sewage Pollution and Wood Exploitation on the Structure and Development of the Mangroves of Mombasa (Kenya)

Thesis submitted in Fulfilment of the Requirements for the Degree of Doctor of Philosophy in Science of the Vrije Universiteit Brussel and Université Libre de Bruxelles

Mohamed Omar Said MOHAMED
Mombasa Field Research Station
Coast Conservation Area
Kenya Wildlife Service
P.O. Box 82144 – 80100 Mombasa,
Kenya

VUB - Promotor
Prof. Dr. Nico KOEDAM
Laboratory of Plant Biology and Nature Management,
Mangrove Management Group,
Vrije Universiteit Brussel,
Pleinlaan 2, B-1050 Brussel,
Belgium

ULB - Promotor
Prof. Dr. Farid DAHDOUH-GUEBAS
Complexité et Dynamique des Systémes Tropicaux,
Département de Biologie des Organismes,
Université Libre de Bruxelles – ULB,
Campus du Solbosch, CP 169, Avenue Franklin D. Roosevelt 50,
B-1050 Bruxelles,
Belgium

Kenyan Promotor
Dr. James Gitundu KAIRO
Mangrove System Information Service
Gazi Bay Mangrove Research Centre
Kenyan Marine and Fisheries Research Institute
P.O. Box 81651 - 80100 Mombasa,
Kenya

Dedication

To my entire family
wife Mima,
daughter Khadija,
mom Roshan,
brothers Abdulmajeed & Hashim,
& sister Fatma
for their patience, encouragement & support all through the study

"To do science is to search for repeated patterns,
not simply to accumulate facts..."
(R.H. MacArthur 1972)

ACKNOWLEDGEMENT

I express my utmost joy and appreciation, first, for the completion of this work, which has taken me a good four years. During that period I gained tremendous experience in diverse fields that essentially puts me in a good position for the future responsibility at the Kenya Wildlife Service. I am highly appreciative to all the support and guidance I got from my promoter Prof. Dr. Nico Koedam, Co-promoter Prof. Dr. Farid Dahdouh-Guebas and Local promoter Dr. James Gitundu Kairo. I am also highly indebted to the Flemish Interuniversity Council (VLIR) for a fruitful support for a masters degree in Ecological Marine Management, and for this study. All this would not have been possible were it not for the concerted efforts of the mentioned parties.

I acknowledge the entire APNA team for their help and valuable suggestions and sharing of valuable scientific ideas, especially during the weekly meetings. During the writing period of this thesis, I shared an office with an amazing scientist, Elizabeth Roberts, from whom I also got to learn some art – painting, an experience I take back home! Caroline, Ronny, Katrien, Diana, Griet, Nele, Faidra, Perine, Tim and Rauff. Elizabeth Roberts, Fleur van Nedervelde and Marilia Rodrigues Da Cunha contributed to this work by translating the preface into Dutch and French. I also take cognisance of the staff at the International Relations Mobility Office, Jacqueline COUDER and her colleagues for making all the four years workable, and especially my stay in Belgium comfortable.

Kenyan students at VUB who brightened up my stay: Josephine Jambuya, Cosmus Munga, Judith Okello, Elisha Mrabu, Lillian Nduku, Ahmed Mohamed, Margaret Mendy, Faith Githui, Kapere Joshua, Paul Obade and Bigeyo Kuboja. I must also recognise the hospitality of Prof. Farid, who warmly invited me to spend time with his family, Fatma and Yasmin – a lovely family. Back in Kenya, there are also those who “sunk with me in the lush mangrove mud”, capsized into the sewage laden waters of Tudor creek and had a car crashed into a ditch during field sampling campaigns - Ogwoka, Masoud and Obinga – many thanks. The Kenya Wildlife Service was pivotal to the completion of this work, through fieldwork support. I am particularly grateful to Ben Wale Kavu and Dr. Richard Bagine for their support. I also appreciate the advice received from Dr. Jared Bosire, Jacob Ochiewo and Steve Mwangi.

I thank my entire family for always being there for me, knowing that they always held me in their thoughts and being there for me whenever I needed them. During this study, my grandmother was particularly helpful in getting support from a number of villages, in particular Jomvu and Maunguja, her passing on in 2006 was a painful loss. According to her, our family links to mangroves dates back to our great grandfather who was a successful mangrove trader in the late nineteenth century and early twentieth century. At least I got to prove that from elderly villagers during this study. Within the course of my studies in Belgium, in 2002, I also lost my caring father, a big loss that has forever changed our family life. I am grateful to all the support I have received from 'baba', and only sad he is not here to share my achievements.

My four years of study were also marked with moments of joy. In August 2007, I got married to a beautiful and loving wife, Mima. This was followed with a new arrival in our life, Khadija, our first daughter, named after my grandmother. I have only seen her on photo, and I really long for the day I will hold her in my arms. I know this time of absence from my family was particularly hard, and I really appreciate the patience and endurance of my wife. HOPE TO JOIN THEM SOON……..

PREFACE

Acknowledging the increasingly prominent urban character of ecosystems globally, mangroves being no exception, and possible impediments to the viability of these ecosystems (i.e. the inherent capacity or ability to grow, develop or recover after disturbances), we adapt a system's approach to establish the viability of the peri-urban mangrove of Tudor creek in Mombasa, Kenya. Three important aspects of the peri-urban mangroves are assessed. These include (i) structural aspects (vegetation structure and regeneration), (ii) functional aspects (productivity) and (iii) human aspects (socioeconomics).

Chapter 1 and 2 introduce the study, outlining the objectives and the study area. The chapters lay down an overview of the mangrove trees physiological adaptations and the ecological attributes that make the mangrove ecosystems unique and highly adaptable to a harsh and dynamic intertidal environment. The chapters further outline the extent and status of mangroves in Kenya, their socioeconomic importance, and the legislation that governs their management and conservation.

Chapters 3 and 6 describe the structural attributes and regeneration status of the peri-urban mangroves of Tudor creek. Based on species importance values the dominant mangrove species were *Rhizophora mucronata* Lam. (Rhizophoraceae) and *Avicennia marina* (Forssk.) Vierh. (Acanthaceae)[♣]. *Lumnitzera racemosa* Willd. (Combretaceae), reported in an earlier floristic survey, was not encountered. Tree density varied between 1,264 trees ha^{-1} and 1,301 trees ha^{-1}, which are within the range of values reported for similar forests globally. However, the size-class structure showed the numerical dominance of small trees over larger trees. The spatial distribution pattern of adults and juveniles varied greatly between sites had a close to uniform pattern (Morisita's Index $I_\delta << 1$) for adult trees, but a tendency for clustered distribution ($I_\delta >> 1$) for juveniles. This pattern of distribution is not expected for a healthy forest. The distribution of regenerating seedlings was mainly impacted by canopy gap sizes. These chapters shows that unmanaged but exploited peri-urban mangroves are structurally degraded, having enlarged canopy gaps, characterised by spatial and temporal heterogeneity in edaphic conditions that influences regeneration. This enlarged gaps and edaphic heterogeneity imposes longer periods for canopy

---

♣ Nomenclature according to Angiosperm Phylogeny Group, 2003

♣ Nomenclatuur volgens de *Angiosperm Phylogeny Group*, 2003 closure by lowering regeneration and promoting mortality of seedlings. Larger gaps (> $60m^2$) had lower regeneration levels and intermediate gaps (20-$50m^2$) had adequate regeneration. The occurrence of *R. mucronata* seedlings and saplings in the understorey under all cover types and inundation confers advantages to this species under the current disturbance regime. Disturbances include sewage pollution, unregulated harvesting and siltation. The current status of the forest is reminiscent of a recovery phase, a multiphase species succession stage, after a major disturbance event, accompanied by recurrent anthropogenic pressure. This study shows that species composition and thus recovery of a mangrove forest after disturbance depends in part on the balance between subsequent large-scale natural and recurrent small-scale human disturbances.

Chapter 4 assesses the human dependence on the peri-urban mangrove. Through questionnaires and field surveys, the study demonstrates the challenges of questionnaire surveys targeting respondents involved in exploitation of sensitive resources. This potentially limits if not inhibit information flow. Firewood is the ubiquitous form of mangrove wood utilisation, exploited at both subsistence and commercial scales. Forest assessments indicate the lack of preferred or specific harvesting sites, with *R. mucronata* being the most harvested, probably due to its distribution range and ease of access. Sewage pollution was viewed with mixed feelings. Many appreciate the nutrient enrichment of the sewage rather than the filtration capacity of mangroves, resulting in the usage of sewage for irrigating small plots of subsistence farms. The study shows that resource exploitation is intense in an urban setting due to an economic drive and an increasing demand. In the rural setting on the other hand, utilisation included both subsistence and commercial charcoal production. This has promoted more efficient, destructive and unsustainable exploitation levels. Associated benefits of these activities grossly under-value the ecosystem goods and services in addition to degrading the ecosystem. Our observations, consistent with other studies, shows that management of mangroves for wood extraction in urban areas may not be a viable and/or sustainable option, as it conflicts with the essential '***filtration***' and '***habitat provisioning***' functions and services of mangrove ecosystems. These functions and services are increasingly important in a "diminishing" urban environment. This arises out of the lack of adequate alternatives and conflicting interests in growing urban areas. It is recommended that '*adaptive*' and '*participatory management'* based on multiple uses and users, with specific legislative, education and institutional interventions. Integrating local ecological knowledge, may further expedite the formulation of sustainable management plans for peri-urban mangroves.

Chapter 5 presents insights on the productivity of an under-valued, over-exploited and sewage polluted peri-urban mangrove through litter fall studies on three common mangrove species, *R. mucronata*, *A. marina* and *S. alba.* The study covers a period of two years. The mean annual litter fall was estimated at 12 ± 3 t $ha^{-1}yr^{-1}$ for the whole stand, which is within the range of values reported for similar forests occupying the same latitudinal range. Litter fall, in both content and quantity was highly seasonal, with high rates occurring in the dry North Easterly Monsoon (NEM) season, January-April (ca. 5 ± 1 g DW m^{-2} day^{-1}) and lower rates in the cool and wet South Easterly Monsoon (SEM) season, June-October (ca. 3 ± 0.5 g DW m^{-2} day^{-1}). *R. mucronata* recorded the highest annual rate of 15 ± 3 t $ha^{-1}yr^{-1}$. No significant differences in litter fall rates were observed between *A. marina* and *S. alba,* (11 ± 3 and 10 ± 5 t $ha^{-1}yr^{-1}$ respectively). Sewage exposure levels did not appear to influence litter production rates despite higher nutrient levels in completely exposed sites. $\delta^{15}N$ varied with species; *A. marina* (6.48 ± 0.03‰) and *S. alba* (6.76 ± 0.24‰) having higher composition than *R. mucronata* (3.88 ± 0.64‰). The leaf C:N ratio strongly correlated with elevated leaf $\delta^{15}N$ signature. Higher C:N ratio for *R. mucronata* corresponding with lower leaf $\delta^{15}N$ (3.88 ± 0.64‰) signature, and lower C:N ratio for *A. marina* and *S. alba* (6.48 ± 0.03‰ and 6.76 ± 0.24‰ respectively) corresponding with higher $\delta^{15}N$ signature. This reflects species specific response to raw sewage exposure. This implies the forest has a more open N cycle, favouring $\delta^{15}N$ accumulation within the system. This study shows that sewage exposure does not necessarily translate into elevated productivity in mangroves, but may alter litter nitrate content depending on species, possibly altering the decay of litter and nutrient cycling.

Chapter 7 presents a synthesis linking the findings to possible implications on the general status of the mangrove ecosystem. The major disturbances observed for the peri-urban mangroves of Tudor creek include (i) domestic sewage pollution, (ii) recurrent unregulated harvesting; and (iii) recurrent annual siltation during the rainy season. Our observations indicate that

(i) Raw domestic sewage pollution may not be harmful to the mangrove vegetation, but may affect edaphic conditions through nutrients enrichment. Sewage pollution effects, though not qualitatively proven in our study, enhance growth of mangrove trees. This is due to increase in amounts of nutrients available for biomass formation, observed as leaf nitrates resorption efficiencies. However, the raw domestic sewage is reported to alter the general healthy decomposing aerobic-anaerobic mangrove system to a complete anaerobic system. This tends to lower efficiencies in nutrient cycling, and cause accumulation of nutrients in the sediments. Observations within the same site and other East African mangroves (under the PUMPSEA project) indicate negative effects of sewage on the sediment cyanobacterial diversity, with an increase in microalgal abundance. Furthermore, within Mikindani (a sewage impacted site in Tudor creek), high rates of sediment $\sum CO_2$ production indicate a system under stress due to the presence of easily degradable organic matter.

(ii) Un-regulated harvesting creates and enlarges canopy gaps, lowering availability of seed bearing trees, altering species composition and stem size distribution due to its selective nature, and lowers regeneration under the enlarged canopy gaps. This strongly lowers recovery rates after major disturbances.

(iii) Siltation stands out as a major cause of degradation. Siltation is extrinsic in nature, a result of poor land use practices. This probably makes it a major issue of concern due to its impact on regeneration. A major siltation event, associated with the 1997-1998 ENSO, is widely identified as a cause of enlarged canopy gaps. Little recovery has occurred 10 years after the event due to recurrent anthropogenic pressure.

The combined effects of these factors have important implications on growth, productivity and recovery of the mangrove ecosystem. The effects include shifts or changes in mangrove tree species distribution. This has lowered the system *functional diversity* and *response diversity*, and therefore ecosystem resilience - viability of the ecosystem. It is recommended that *integrated adaptive management*, based on sound knowledge of the system is the recommended approach. The participation of stakeholders (government institutions, the private sector and local communities) is crucial for managing peri-urban mangroves for sustainability. Not intervening may only result in a worst case scenario. Especially with the current global financial crisis, more locals will turn to '*cheap*' mangrove firewood.

VOORWOORD

We moeten het steeds uitgesprokener stedelijk karakter van vele ecosystemen wereldwijd - mangroven zijn hierop geen uitzondering – en de mogelijk gepaard gaande belemmeringen voor de levensvatbaarheid van deze ecosystemen in een stedelijke omgeving erkennen. Met levensvatbaarheid bedoelen we de inherente capaciteit of het vermogen om te groeien, te expanderen, te ontwikkelen of zich te herstellen na verstoringen. We hebben gewerkt met een aanpak gericht op het hele systeem om de levensvatbaarheid van het mangrovebos aan de rand van de stad in de Tudorkreek van Mombasa, Kenia te staven. Drie belangrijke aspecten van mangroven die zich naast de stad bevinden worden behandeld. Deze zijn (i) structurele aspecten (vegetatiestructuur en regeneratie), (ii) functionele aspecten (productiviteit) en (iii) menselijke aspecten (socio-economie).

Hoofdstuk 1 en 2 leiden de studie in door de doelstellingen en het studiegebied te schetsen. De hoofdstukken geven een overzicht van de fysiologische aanpassingen van mangrovebomen en de ecologische kenmerken die mangrove-ecosystemen uniek maken en ervoor zorgen dat ze sterk aangepast zijn aan de harde en dynamisch omgeving van het intergetijdengebied. Verder behandelen de hoofdstukken de grootte en de status van de mangroven in Kenia, hun socio-economisch belang en de wetgeving die hun beheer en behoud bepaalt.

Hoofdstuk 3 en 6 beschrijven de structurele eigenschappen en de regeneratiestatus van de mangroven aan de rand van de stad in de Tudorkreek. Op basis van de belangrijkheidswaarden van de soorten waren *Rhizophora mucronata* Lam. (Rhizophoraceae) en *Avicennia marina* (Forssk.) Vierh. (Acanthaceae)[♣] de dominante mangrovesoorten. *Lumnitzera racemosa* Willd., gerapporteerd in een vroeger floristisch onderzoek, werd niet waargenomen. De boomdensiteit varieerde tussen 1264 en 1301 bomen per hectare. Dit is binnen de range van waarden die gerapporteerd worden voor gelijkaardige bossen wereldwijd. Nochtans vertoonde de structuur van de grootte, opgedeeld in klassen, een numerieke dominantie van kleine ten opzichte van grotere bomen. Het ruimtelijk distributiepatroon van volwassen en juveniele bomen varieerde sterk tussen sites en vertoonde een bijna uniform patroon (Morisita's Index I_{δ} << 1) voor volwassen bomen, maar een tendens tot

[♣] Nomenclatuur volgens de *Angiosperm Phylogeny Group,* 2003 clusterdistributie ($I_\delta >> 1$) voor juveniele bomen. Dit distributiepatroon verwachten we niet voor een gezond bos. De distributie van regenererende zaailingen was hoofdzakelijk beïnvloed door de grootte van de openingen in het bladerdek. Deze hoofdstukken tonen aan dat onbeheerde maar geëxploiteerde mangrovebossen in de buurt van de stad structureel gedegradeerd zijn door grote openingen in hun bladerdek, ruimtelijk en temporeel heterogeen in edafische condities. Deze heterogeniteit beïnvloedt de regeneratie. De vergrote openingen in het bladerdek en de edafische heterogeniteit maken dat er langere periodes nodig zijn voor het toegroeien van het bladerdek omdat de regeneratie erdoor vertraagd en sterfte van de zaailingen erdoor bevorderd wordt. Grote openingen (> 60 m^2) hadden een lager regeneratieniveau en middelgrote openingen (20-50 m^2) hadden een adequate regeneratie. Het voorkomen van zaailingen en jonge boompjes van *R. mucronata* in de onderlaag bij alle types van begroeiing en inundatie bevoordeelt deze soort onder het huidige verstoringregime (vervuiling door afvalwater, ongereguleerde kap en dichtslibbing). De huidige status van het bos doet denken aan een herstelfase, *i.e.* een stadium in een gefaseerde successie van soorten die begint na een belangrijke verstoring, en dit gekoppeld aan steeds terugkerende antropogene druk. De studie toont aan dat de soortensamenstelling en dus het herstel van een mangrovebos na verstoring gedeeltelijk afhangt van de balans tussen opeenvolgende natuurlijke verstoringen op grote en terugkomende menselijke verstoringen op kleine schaal.

Hoofdstuk 4 behandelt de menselijke afhankelijkheid van de mangrove nabij de stad. Via vragenlijsten en veldonderzoek toont de studie de uitdagingen van het voeren van onderzoek via enquêtes waarvan de beoogde ondervraagden betrokken zijn bij exploitatie van kwetsbare hulpbronnen. Dit beperkt mogelijk de informatiestroom of verhindert die zelfs. Mangrovehout wordt alomtegenwoordig als brandhout gebruikt en exploitatie gebeurt zowel voor eigen gebruik als op commerciële schaal. We stellen vast dat er in het bos geen favoriete of specifieke kapplaatsen zijn. *R. mucronata* is de meest gekapte soort, waarschijnlijk door zijn verspreiding en bereikbaarheid. Vervuiling door afvalwater werd verschillend beoordeeld. Vele verkiezen eerder de nutriëntverrijking van de afvalwatervervuiling dan de capaciteit van mangroven om water te filteren, waardoor afvalwater gebruikt wordt voor het irrigeren van kleine percelen van boerderijen voor eigen gebruik. De studie toont aan dat de exploitatie van hulpbronnen intens is in een stedelijke setting als gevolg van economische impulsen en een steeds stijgende vraag. In de naburige landelijk setting daarentegen werd mangrovehout gebruikt voor houtskoolproductie voor zowel eigen gebruik als voor verkoop. De beperkte waaier aan bestaansmiddelen in de landelijke setting, gekoppeld aan armoede en de nood aan goedkope energie voor huishoudens, noodzaakt de afhankelijkheid van de hulpbronnen uit de mangrove. Dit heeft geleid tot efficiëntere, destructieve en niet-duurzame exploitatieniveaus. De voordelen die aan deze activiteiten gekoppeld zijn onderwaarderen de producten en diensten van het ecosysteem sterk en dit bovenop de degradatie van het ecosysteem. Onze observaties, die consistent zijn met andere studies, tonen aan dat management van de mangroven voor houtgebruik in stedelijke gebieden misschien geen levensvatbare en/of duurzame optie is gezien het in conflict is met de essentiële functies en diensten van mangrove-ecosystemen, namelijk ***filtratie'*** en ***'habitatvoorziening'***. Deze functies en diensten zijn zeer belangrijk zijn in een 'afnemende' stedelijke omgeving. Dit komt voort uit het gebrek aan adequate alternatieven en tegenstrijdige belangen in groeiende stedelijke gebieden. Het is aanbevolen dat *'aangepast beheer'* en *'groepsmanagement'* gebruikt zou worden om de vorming van duurzame managementplannen voor mangroven aan de rand van de stad te bevorderen. Dit soort management is gebaseerd op de vele gebruikstoepassingen en gebruikers, heeft specifieke legislatieve, onderwijskundige en institutionele interventies, en integreert locale ecologische kennis.

Hoofdstuk 5 toont de inzichten in de productiviteit van een ondergewaardeerde, overgeëxploiteerde en met afvalwater vervuilde mangrove aan de rand van de stad en dit aan de hand van studies van de bladerval van drie algemene mangrovesoorten, *R. mucronata, A. marina* en *S. alba.* De studie beslaat een periode van twee jaar. De gemiddelde jaarlijkse bladerval werd geschat op 12 ± 3 t ha^{-1} yr^{-1} voor het hele bestand. Dit is binnen de range van waarden die gerapporteerd werden voor gelijkaardige bossen in dezelfde latitudinale gordel. Bladerval, in zowel inhoud als aantal, was sterk seizoensgebonden, met hoge warden in het droge Noordoostelijke Moessonseizoen - januari-april (ca. 5 ± 1 g DW m^{-2} dag^{-1}) - en met lagere waarden in het koude en natte Zuidoostelijke Moessonseizoen – juni-oktober (ca. 3 ± 0,5 g DW m^{-2} dag^{-1}). De hoogste jaarlijkse waarde (15,34 ± 3,34 t ha^{-1} yr^{-1}) werd waargenomen voor *R. mucronata*. Er waren geen significant verschillen tussen *A. marina* en *S. alba* (11,44 ± 2,90 en 9,69 ± 5,26 t ha^{-1} yr^{-1} respectievelijk). Het gehalte van blootstelling aan afvalwater leek geen invloed te hebben op de productie van afgevallen bladeren ondanks de hogere nutriëntgehalten in de volledig blootgestelde sites. $\delta^{15}N$ varieerde afhankelijk van de soort: *A. marina* (6,48 ± 0,03 ‰) en *S. alba* (6,76 ± 0,24 ‰) hadden een hogere samenstelling dan *R. mucronata* (3,88 ± 0,64 ‰). Dit houdt in dat het bos een meer open stikstofcyclus heeft die de accumulatie van $\delta^{15}N$ in het systeem bevoordeelt. Deze studie toont aan dat blootstelling aan afvalwater niet noodzakelijk vertaalt wordt in verhoogde productiviteit in mangroven, maar dat de totale stikstofinhoud van bladeren kan veranderen zodat de afbraak van bladeren en de nutriëntcyclus mogelijk ook veranderen.

Hoofdstuk 7 geeft een synthese die de onderzoeksresultaten linkt met mogelijke implicaties voor de algemene status van het mangrove-ecosysteem. De voornaamste verstoringen die geobserveerd werden voor de mangroven aan de rand van de stad in de Tudorkreek zijn (i) vervuiling door huishoudelijk afvalwater, (ii) herhaaldelijke ongereguleerde kap, en (iii) herhaaldelijke jaarlijkse dichtslibbing gedurende het regenseizoen. Onze observaties tonen aan dat

(i) Vervuiling door ongezuiverd huishoudelijk afvalwater niet schadelijk is voor mangrovebomen maar de wel de edafische condities kan veranderen door nutriëntverrijking. De effecten van vervuiling via afvalwater, hoewel niet kwalitatief aangetoond in onze studie, versnellen de groei van mangrovebomen. Dit is te wijten aan een toename van de hoeveelheid nutriënten beschikbaar voor biomassavorming, gemeten als de efficiëntie in resorptie van bladnitraten. Er werd echter gerapporteerd dat ongezuiverd huishoudelijk afvalwater het algemeen gezond composterende aërobe-anaërobe mangrovesysteem verandert tot een volledig anaëroob systeem. Dit geeft aanleiding tot verlaging van de efficiëntie van het circuleren van nutriënten en zorgt ervoor dat nutriënten in de sedimenten accumuleren. Observaties in dezelfde site en in andere Oostafrikaanse mangroven (in het kader van het PUMPSEA-project) tonen negatieve effecten van afvalwater op de diversiteit van cyanobacteriën in de sedimenten, met een verhoging in de abundantie aan microalgen. Daarenboven wijzen de hoge waarden van ΣCO_2-productie in de sedimenten van Mikindani (een met afvalwater bevuilde site in de Tudorkreek) op een systeem onder stress als gevolg de aanwezigheid van gemakkelijk afbreekbaar organische materiaal.

(ii) Ongereguleerde kap openingen in het bladerdek creëert en ze vergroot. Hierdoor vermindert de beschikbaarheid van zaaddragende bomen en verandert de soortensamenstelling en de verdeling van de stamgrootte, gezien er selectief gekapt wordt. Ongereguleerde kap vermindert ook de regeneratie onder de groter wordende openingen in het bladerdek. Dit doet de mate van herstel na grote verstoringen sterk dalen.

(iii) Dichtslibbing de hoofdoorzaak is van degradatie. Dichtslibbing is van extrinsieke aard als gevolg van de manieren waarop arme grond gebruikt wordt. Dit maakt het een belangrijk aandachtspunt, omwille van impact ervan op de regeneratie. De grote mate van dichtslibbing die geassocieerd wordt met de ENSO in 1997-1998, wordt algemeen beschouwd als een oorzaak van groter geworden openingen in het bladerdek. Als gevolg van voortdurend aanwezige antropogene druk, heeft er maar weinig herstel plaatsgevonden 10 jaar na deze gebeurtenis.

Het gecombineerde effect van deze factoren heeft belangrijke implicaties op de groei, de productiviteit en het herstel van het mangrove-ecosysteem. Verschuivingen of veranderingen in de distributie van mangrovesoorten zijn enkele van de effecten. Dit heeft de *functionele diversiteit* en de *verschillende mogelijkheden tot reactie* van het systeem verlaagt en dus ook de veerkracht - levensvatbaarheid van het ecosysteem. Een *geïntegreerd aanpasbaar management*, gebaseerd op grondige kennis van het systeem, is aangewezen. De bijdrage van de betrokken partijen (gouvernementele instellingen, de private sector en lokale gemeenschappen) is cruciaal voor het beheren van mangroven nabij de stad met het oog op duurzaamheid. Niet zal enkel resulteren in het allerslechtste scenario. Zeker met de huidige wereldwijde financiële crisis zullen meer plaatselijke bewoners gebruik maken van '*goedkoop*' brandhout uit de mangrove.

PRÉFACE

En vue de l'augmentation de l'urbanisation dans les multiples écosystèmes du monde, les mangroves n'étant pas une exception, on observe des obstacles à la viabilité de ces écosystèmes, c.-à-d. leur capacité inhérente de se développer, de s'étendre ou de récupérer après des perturbations sous des conditions urbaines. Nous avons adapté une approche systémique pour établir la viabilité des mangroves périurbaines de la baie de Tudor à Mombasa, Kenya. Trois aspects importants des mangroves périurbaines sont évalués. Cela inclus (i) les aspects structurels (structure de la végétation et la régénération), (ii) aspects fonctionnels (productivité) et (iii) aspects humains (socio-économiques).

Les Chapitres 1 et 2 introduisent l'étude, en exposant brièvement les objectifs et le domaine d'étude. Les chapitres présentent un panorama des adaptations physiologiques des palétuviers et des attributs écologiques qui rendent l'écosystème de la mangrove unique et hautement adaptable à un environnement intertidal rigoureux et dynamique. Les chapitres suivants exposent l'étendue et le statut des mangroves au Kenya, leur importance socio-économique et la législation qui cadre leur gestion et conservation.

Les chapitres 3 et 6 décrivent les caractéristiques structurelles et les étapes de la régénération des palétuviers périurbains de la baie de Tudor. Par ordre d'importance des espèces, les palétuviers les plus souvent rencontrés étaient *Rhizophora mucronata* (Rhizophoraceae) et *Avicennia marina* (Forssk.) Vierh. (Acanthaceae)♣. *Lumnitzera racemosa* Willd. (Combretaceae) n'a pas été rencontrée lors d'un voyage floristique précédent. La densité d'arbre varie entre 1.264 arbres ha^{-1} et 1.301 arbres ha^{-1}, ce qui correspond aux moyennes rapportées pour des forêts comparables dans le monde. Cependant, la structure des classes de taille montre une dominance numérique des palétuviers moins développés par rapport aux plus développés. Le pattern de distribution spatiale des adultes et des juvéniles varie considérablement entre les sites et montre presque un modèle uniforme (l'index de Morisita; $I_\delta << 1$) pour les individus adultes. En revanche, pour les juvéniles on retrouve une tendance de distribution groupée ($I_\delta >> 1$). Ce modèle de distribution n'était pas attendu pour une forêt non altérée. La distribution de la régénération des pousses est principalement

♣ Nomenclature selon l'Angiosperm Phylogeny Group, 2003 dictée par la taille des *gaps*. Ce chapitre montre que les mangroves périurbaines, exploitées et non aménagées, sont structurellement dégradées, ayant des larges *gaps* caractérisés par une hétérogénéité des conditions édaphiques qui influencent la régénération. Quant aux *gaps* les plus larges, on retrouve un taux de régénération plus faible. Les *gaps* élargis et l'hétérogénéité édaphique imposent de plus longues périodes de fermeture des canopées, en diminuant la régénération et promouvant la mortalité des pousses. Les *gaps* plus grands (> $60m^2$) ont des niveaux de régénération plus bas et les *gaps* intermédiaires ($20\text{-}50m^2$) ont une régénération adéquate. L'occurrence de pousses et de juvéniles de *R. mucronata* dominent le niveau inférieur de tous les types de couvertures végétales et d'inondation. Cela confère un avantage à cette espèce sous le régime de perturbation actuel. Les perturbations incluent les eaux usées non traitées, les récoltes non-réglementées et l'envasement. L'état actuel de la forêt est une phase de récupération, un stade de succession d'espèces en multiples phases, suite à un événement majeur de perturbation, accompagné d'une pression anthropogénique récurrente. Cette étude démontre que la composition d'espèces et donc de la récupération de la forêt de mangrove après une perturbation dépend en partie de l'équilibre entre les perturbations naturelles à grande échelle et humaines à moindre échelle.

Le chapitre 4 évalue la dépendance humaine à l'égard de la mangrove périurbaine. Par des questionnaires et des travaux de terrains, on a constaté que l'utilisation des questionnaires est délicate. En effet, ceux-ci ciblent des personnes impliquées dans l'exploitation des ressources sensibles. Ceci limite potentiellement sinon empêche le flux de l'information. Le bois de feu est la forme d'utilisation la plus répandue du bois de mangrove, utilisé dans le cadre de la subsistance et du commerce. Les résultats indiquent qu'il n'y a pas de sites de récolte préférés. *R. mucronata* est l'espèce la plus récoltée, probablement en raison de sa distribution et facilité d'accès. La pollution par les eaux usées a appréciation mixte. Certains l'apprécient pour l'irrigation de leurs cultures de subsistance, malgré la capacité de filtration des mangroves. L'étude montre que l'exploitation des ressources est intense, dans un cadre urbain, dû à une conduite économique et une demande croissante. Par contre, dans le cadre rural, l'utilisation inclus la production de charbon de subsistance ainsi que commercial. La limitation des moyens de vie dans le cadre rural, couplés à la pauvreté et le besoin d'énergie domestique bon marché, créent une dépendance des ressources de mangroves. Ceci a favorisé des niveaux d'exploitation plus importants, destructeurs et non viables. Les gains associés à ces activités qui dégradent l'écosystème, sont beaucoup moins importants que les biens et les services offerts par la mangrove. Nos observations, en accord avec d'autres études, montrent que la gestion de l'extraction du bois des mangroves dans des zones urbaines n'est pas une option viable et/ou durable, puisqu'il entre en conflit avec les fonctions et services essentiels de '***filtration***' et '***d'approvisionnement pour l'habitation***' de la mangrove. Ce phénomène résulte du manque d'alternatives adéquates et des conflits d'intérêts des zones urbaines en expansion. Il est recommandé de mettre en place une gestion '*adaptative*' et '*participative*' fondée sur les utilisations et utilisateurs multiples, ainsi qu'une législation, une éducation et des interventions institutionnelles spécifiques, intégrant les connaissances écologiques locales, afin de faciliter des plans de gestion durable pour les mangroves périurbaines.

Le chapitre 5 présente un aperçu de la productivité de la mangrove périurbaine sous-évaluée, surexploitée et polluée par des eaux usées à travers des études de la litière de trois espèces communes de palétuvier, *R. mucronata*, *A. marina* et *S. alba.* L'étude couvre une période de deux ans. La chute moyenne annuelle de litière a été estimée à 12 ± 3 t $ha^{-1}an^{-1}$ pour la totalité du secteur, ce qui correspond aux valeurs de forêts comparables occupant des latitudes similaires. La composition et la quantité de la litière changent fortement avec les saisons. Les taux élevés (ca. 5 ± 1 g DW m^2 j^{-1}) s'observent au cours de la saison sèche (Janvier-Avril), sous le *North Easterly Monsoon* (NEM).Tandis que les taux inférieurs (ca. $3 \pm 0{,}50$ g DW m^2 j^{-1}) s'observent au cours de la saison humide (Juin-Octobre), sous le *South Easterly Monsoon* (SEM). Il y a des variations significatives entre les espèces. *R. mucronata* a enregistré le taux annuel le plus élevé 15 ± 3 t $ha^{-1}an^{-1}$. Par contre, il n'y a pas de différences significatives entre *A. marina* et *S. alba,* (11 ± 3 et 10 ± 5 t $ha^{-1}an^{-1}$, respectivement). Le niveau de pollution par les eaux usées ne semble pas influencer les taux de production de litière en dépit de la concentration plus élevées des nutriments dans les zones complètement exposées. Le $\delta^{15}N$ a varié entre les espèces, *A. marina* ($6{,}48 \pm 0{,}03$‰) et *S. alba* ($6{,}76 \pm 0{,}24$‰) ayant une composition plus élevée que *R. mucronata* ($3{,}88 \pm 0{,}64$‰). Cela implique que la forêt a un cycle de l'N plus ouvert, favorisant l'accumulation dans le système. Cette étude démontre que l'exposition aux eaux usées ne se traduit pas nécessairement en une productivité élevée des palétuviers, mais peut changer le contenu de nitrogène total des feuilles, probablement en changeant la décomposition de la litière et le cycle des nutriments.

Le **chapitre 7** présente une synthèse liant les résultats aux implications possibles sur le statut général de l'écosystème des mangroves. Les perturbations principales observées pour les mangroves périurbaines de la baie de Tudor incluent (i) la pollution par les eaux usées domestiques, (ii) la récolte récurrente irrégulière et (iii) l'envasement annuel pendant la saison des pluies. Nos observations indiquent que:

(i) la pollution par les eaux usées non traitées n'est pas nocive pour les palétuviers, mais peut affecter les conditions édaphiques par l'enrichissement en éléments nutritifs. Les effets de la pollution des eaux usées, quoique non prouvés qualitativement dans notre étude, augmentent la croissance des palétuviers. Ce par une augmentation des quantités de nutriments disponibles pour la formation de biomasse, observée lors de l'étude de l'efficacité de résorption des nitrates par les feuilles. Cependant, on rapporte que les eaux usées domestiques non traitées changent le système de décomposition aérobique-anaérobique naturel des mangroves vers un système uniquement anaérobique. Ce qui diminue l'efficacité du cycle nutritif et engendre l'accumulation des nutriments dans les sédiments. Les observations sur le même site et sur d'autres mangroves de l'Afrique de l'Est (sous le projet PUMPSEA), indiquent des effets négatifs des eaux usées sur la diversité des cyanobactéries du sédiment et une augmentation de l'abondance des micro-algues. En outre, à Mikindani (site affecté par des eaux usées dans la baie de Tudor), les taux élevés de production de $\sum CO_2$ dans le sédiment indiquent un système sous pression (ou sous stress), suite à la présence de la matière organique facilement dégradable.

(ii) L'exploitation non réglementée va créer et agrandir des gaps. En fonction de la nature sélective de l'exploitation, il y aura une diminution de la disponibilité des arbres produisant des propagules, un changement de la composition des espèces et de la distribution des tailles des arbres. De plus, elle réduit la régénération sous les larges gaps. Cela diminue fortement les taux de rétablissement après d'importantes perturbations.

(iii) L'envasement est la cause majeure de la dégradation. L'envasement est extrinsèque par nature, parce qu'il découle de mauvaises pratiques d'exploitation de la terre. Ce phénomène est probablement le point de concertation majeur, dû à son impact sur la régénération. Un événement d'envasement majeur, lié à l'ENSO de

1997-1998, est identifié comme étant une cause d'agrandissement des *gaps*. Peu de récupération s'est produite lors des dix années qui suivirent l'événement, suite à la pression anthropogène récurrente.

Les effets combinés de ces facteurs ont des implications importantes sur la croissance, la productivité et la récupération de l'écosystème des mangroves. Les effets incluent des variations ou des changements de la distribution des espèces de palétuviers. Ceci a diminué la *diversité fonctionnelle* et *la diversité des réponses* du système et a donc diminué la résilience de l'écosystème - la viabilité de l'écosystème. Il est recommandé qu'une *gestion adaptative intégrée*, fondée sur la connaissance orale du système, soit la meilleure approche. La participation des décideurs (institutions gouvernementales, secteur privé et communautés locales) est cruciale pour la gestion des mangroves périurbaines. Dans l'absence d'intervention, seul le pire scénario peut être envisagé. En particulier avec la crise financière globale actuelle, la population locale se tournera encore plus vers le bois de feu '*bon marché*' de la mangrove.

Table of Contents

GLOSSARY

Disturbance	a temporary change in average environmental conditions that causes a pronounced change in an ecosystem. In the context of this study, it refers to the removal of biomass from the forest
Stress	refers to physical, chemical, and biological constraints on the productivity of species and on the development of ecosystems
DBH	Diameter at breast height (approximately 1.3 meters), also referred to as D130
ENSO	El Niño-Southern Oscillation - commonly referred to as simply El Niño. It is a global coupled ocean-atmosphere phenomenon
Viability	the inherent capacity or ability of an ecosystem to grow, develop or recover after disturbances
Sustainability	*Ecological context* - the ability of an ecosystem to maintain ecological processes, functions, biodiversity and productivity into the future *Social context* - meeting the needs of the present without compromising the ability of future generations to meet their own needs
Resilience	the capacity of a system to absorb disturbance, to undergo change and still retain essentially the same function, structure and feedbacks
Score	a bundle of 20 mangrove poles, or 20 pieces of mangrove firewood
Urban	an urban area is an area with an increased density of human-created structures in comparison to the areas surrounding it. Urban areas may be cities, towns or conurbations
Peri-urban	refers to location in relation urban area i.e. the periphery
Rural	rural areas (also referred to as "the country" or "the countryside") are large and isolated areas of a country, often with low populations.

1. Introduction

1.1 Background

According to estimates, mangroves in Kenya, occupy between 52,980 ha (Doute *et al.*, 1981) and 64,426 ha (Forest Department of Kenya, 1983). For centuries, mangrove wood has been exploited both for subsistence and commercial purposes (Rawlins, 1957; Kokwaro, 1985; Dahdouh-Guebas *et al.*, 2000). In the last 50 years, 50% of these mangroves are estimated to have been lost (FAO, 2005) at a global estimated rate of 1-2% per annum (Valiela *et al.*, 2001; Duke *et al.*, 2007) despite harvesting and export bans imposed by the Government (Luvanda *et al.*, 1997; Abuodha and Kairo, 2001). This decline is characterised by the absence of good quality poles (Dahdouh-Guebas *et al.*, 2000; Kairo *et al.*, 2001), reduced mangrove fishery (Mirera *et al.*, 2007) and enhanced coastal erosion (Abuodha and Kairo, 2001; Dahdouh-Guebas *et al.*, 2004). This dissertation assesses the viability of the peri-urban mangrove forest of Tudor creek. We adapt a system's approach to evaluate the viability of the forest. We assess the vegetation aspects (structure and regeneration status), functional aspects (productivity as litterfall) and human aspects (exploitation patterns). In the framework of this study, viability is defined as the inherent ecosystem's capacity or ability to grow, develop or recover after disturbance. This study will enable us gain insight into the threats, and thus management requirements for long-term sustainability of mangrove goods and services.

1.2 Mangroves

The term 'mangrove' refers to both a type of plant and a type of ecosystem. As ecosystems, mangroves are typical wetland ecosystems distributed worldwide on sheltered, tropical coastlines (Saenger, 1998; Spalding *et al.*, 1997; Ellison *et al.*, 1999). At the global scale, the distribution, structural development and growth of mangrove forests is predominantly controlled by temperature. The winter 20°C seawater isotherm limits the pole-ward extension in each hemisphere. However, prevailing warm currents and a broader tolerance of environmental extremes allow the existence of *Avicennia marina* (Forssk.) Vierh. var. *resinifera* southward to the northern Island of New Zealand and the eastern coast of South Africa (Spalding *et al.*, 1997; Duke *et al.*, 1998). The congener, *A. germinans*, ranges northward to the southern coast of Louisiana (USA). Standing biomass and tree height are at a maximum in the humid tropics and decline progressively with increasing latitude to about 35°N and 38°S, where mangroves are replaced by salt marsh ecosystems (Saenger and Snedaker, 1993). Globally, mangroves are estimated to have occupied 75% of the tropical coasts (Chapman, 1976; Chapman, 1977; Alongi, 2002). Anthropogenic pressures have reduced the global range of these forests to less than 50% of the original total cover (Saenger and Snedaker, 1993; Spalding *et al.*, 1997; Duke *et al.*, 2007). Mangrove cover is estimated to have declined from 18.8 million hectares in 1980 to the current 15.2 million hectares (Spalding *et al.*, 1997; Alongi, 2002; FAO, 2005; Duke *et al.*, 2007).

The aerial coverage of mangroves is small compared to other tropical forests (Lugo, 1980; Lugo *et al.*, 1990), but the ecological roles relative to their area are magnified by their position at the ecotone between terrestrial and marine systems (Alongi, 1990; Othman *et al.*, 2004; Nagelkerken *et al.*, 2008; Kristensen *et al.*, 2008). As intertidal ecosystems, they assure a plethora of essential functions and services to coastal zones and to their plant, animal and human populations. Mangroves ***support essential services*** for all other ***ecosystem services*** including soil formation, photosynthesis, primary production, nutrient cycling and water cycling. They play an important role in ***provisioning*** natural products including proteins, tannins and lumber to local communities, while ***regulating*** benefits obtained from the regulation of ecosystem processes such as biological control, nutrient cycling, air quality regulation, and maintenance of biodiversity for ecosystem function and resilience (Millennium Ecosystem Assessment, 2005; Rönnbäck *et al.*, 2007; Bosire *et al.*, 2008; Cannicci *et al.*, 2008; Kristensen, 2008; Kristensen *et al.*, 2008; Gilman *et al.*, 2008; Nagelkerken *et al.*, 2008; Walters *et al.*, 2008).

Human interactions with mangroves have received much attention in the last two decades, with appreciable recognition of the value of mangrove wetlands, once thought of as “wastelands” (Walters *et al.*, 2008). Initial focus was on status and description of uses by coastal communities (FAO, 1994; Dahdouh-Guebas *et al.*, 2000) and more recently, describing humans as agents of ecological change (Alongi and de Carvalho, 2008; Walters *et al.*, 2008; Ellison, 2008a). It is widely perceived that both wood and non-wood forest products are valuable economic resources to marginalised and impoverished coastal communities (Dahdouh-Guebas *et al.*, 2006;

Alongi and de Carvalho, 2008; Nagelkerken *et al*., 2008; Ellison, 2008a; Walters *et al*., 2008). These human mangrove interactions have ultimately translated into overexploitation, exaggerated by an unprecedented human population growth with a resultant high demand for mangrove resources resulting in '***drastic***' ecosystem degradation (Duke *et al*., 2007; Alongi and de Carvalho, 2008; Ellison, 2008a; Walters *et al*., 2008).

Mangrove degradation over time has been recurrently documented (Sherman *et al*., 2000; Walters, 2005a, b; Hauf *et al*., 2006; López-Hofman *et al*., 2006; Nagelkerken *et al*., 2008; Walters *et al*., 2008; Dahdouh-Guebas and Koedam, 2008; Ellison, 2008a), but unfortunately direct and indirect anthropogenic pressure persists, with an alarming scale of degradation especially in the past four decades (Valiela *et al*., 2000; Duke *et al*., 2007; Walters *et al*., 2008). Impacts of mangrove losses are concomitantly diverse and complex, including altered canopy cover, altered canopy-gap microclimate and dynamics, enhanced erosion/siltation, and altered hydrological and biogeochemical cycles, which ultimately affect forest stand dynamics, regeneration, productivity and flora and fauna assemblages within the ecosystem (Alongi and de Carvalho, 2008; Ellison, 2008a; Nagelkerken *et al*., 2008; Bosire *et al*., 2008), ultimately leading to diminished ecosystem resilience. This inexorable degradation and decline in mangroves has drawn much attention of late due to the scale and the implications on losses to biological richness and diversity coupled with immense economic losses (Primavera, 1997; Abuodha and Kairo, 2001; Barbier and Cox, 2002; Duke *et al*., 2007; Walters *et al*., 2008). It is projected that the prospect of a world deprived of the goods and services offered by mangroves, perhaps within the next 100 years is a reality (Duke *et al*., 2007).

Major anthropogenic threats to mangroves include: diversion of freshwater, deteriorating water quality caused by pollutants and nutrients, over-harvesting for firewood and timber as well as conversion into development activities like agriculture, aquaculture, mining, salt extraction and infrastructure (Terchunian *et al*., 1986; Saifullah *et al*., 1989; Ellison, 1998; Dahdouh-Guebas *et al*., 2000; Abuodha and Kairo, 2001; Valiela *et al*., 2001; Dahdouh-Guebas *et al*., 2004; Benfield *et al*., 2005; Dahdouh-Guebas *et al*., 2005a; Walters, 2005a, b; Duke *et al*., 2007). Climate change (Gilman *et al*., 2007; Gilman *et al*., 2008; Alongi, 2008), with the high population in coastal areas (Walters *et al*., 2008), further compounds the challenges to conservation and management of mangroves and related resources. However, threats and impacts of urbanisation on mangroves have received little attention, with limited studies conducted in peri-urban mangroves, even though degradation and the associated factors are diverse and persistently complex in nature, involving socio-cultural, demographic, technological, economic, policy and institutional challenges (Dahdouh-Guebas *et al*., 2000; Dahdouh-Guebas and Koedam, 2001; Kairo *et al*., 2001; Kairo *et al*., 2002a; Walters, 2003; Dahdouh-Guebas *et al*., 2004; Walters, 2004; Walters, 2005a, b; Duke *et al*., 2007).

1.3 Rationale of the study

Amongst the major services offered by mangroves, the filtration or interception of land-derived pollutants and limiting their dispersal offshore is well documented (Nedwell, 1975; Tam and Wong, 1999; MacFarlane and Burchett, 2002). Several studies have proven that mangrove ecosystems possess capacity as sinks for excess nutrients and other pollutants (e.g., Odum and Heald, 1972; Nedwell, 1975; Silva *et al*., 1990; Morell and Corredor, 1993; Corredor and Morell, 1994; Tam and Wong, 1999; Tam and Yao, 1999). Many studies demonstrate the use of mangroves as constructed wetlands for wastewater treatment (Wong *et al*., 1997; Ye *et al*., 2001; Boonsong *et al*., 2003; Yang *et al*., 2008; Wu *et al*., 2008). These studies consider mangroves' unique characteristics that may make them particularly adapted to sewage exposure. The aerial roots and oxygen translocation systems make these trees highly adapted to growth in hypoxic mud (Tomlinson, 1986). Mangrove swamps appear extremely effective at removing phosphates and nitrogen at rates higher than in conventional constructed reed bed systems (Ye *et al*., 2001; Wu *et al*., 2008). Heavy metal sequestering is also equally efficient (MacFarlane and Burchett, 2002). Nutrient additions typically stimulate ecosystem production (Feller *et al*., 2003b; Holmboe *et al*., 2001). However, the survival, regeneration, productivity and floristic stability of mangroves under pollution pressure have hardly been recorded, and the sustainable threshold of exposure remains largely unknown (UNEP/GPA, 2001).

Mombasa has a high population growth rate of 3-4% per year, with a 3,111 persons/km^2 population density. Despite the rapid increase in population, the infrastructure has not grown concurrently. In particular, sewage treatment facilities have not been upgraded or are not operational resulting in the release of untreated domestic sewage into waters of Kilindini, Port Reitz and Tudor Creeks. The peri-urban mangroves of Mombasa, situated within the creeks, are therefore recipients of sewage-polluted rivers and flash-flood waters and are used for sewage dumping, with possible risk to marine ecosystems and public health. Considering the short coastline of Mombasa with the high concentration of socio-economic activities, there is a greater risk of localized pollution and environmental degradation (UNEP/GPA, 2001). Limited studies have shown that water quality in creeks around Mombasa is poor with faecal coliform exceeding safe limits (Mwaguni and Munga, 1997). Considering that the worldwide threat of domestic wastewater discharge to sustainable coastal development is defined as a key global issue, this study is of importance (UNEP/GPA, 2001).

1.3.1 The PUMPSEA project

Our study complements the 3 year PUMPSEA research project♣ funded by the European Commission (www.pumpsea.icat.fc.ul.pt). The overall objective of the Project was to demonstrate the ecological and economical service that peri-urban mangroves provide by mitigating coastal pollution through sewage filtration, and to offer innovative solutions for the exploitation and management of this quality. The Project examined two innovative ways in which mangrove filtration can be utilized to preclude coastal sewage pollution: (1) facilitating sewage filtration by conserving filtering mangroves and replanting mangroves in deforested areas exposed to sewage (´*strategic reforestation and conservation*'), and (2) using constructed mangrove wetlands for sewage treatment. The Project was developed in peri-urban mangrove areas of Maputo (Mozambique), Dar es Salaam (Tanzania) and Mombasa (Kenya) and comprised socio-economy, condition mapping, biogeochemistry, ecology, modelling, controlled experimentation and experimental optimization of a trial wetland used for secondary treatment of sewage and governance analysis.

Within Mombasa, no detailed vegetation studies have been conducted on the critical mangrove ecosystems, especially assessing the effects of pollution. This study,

♣ Peri-urban Mangrove Forests as Filters and Potential Phytoremediators of Domestic Sewage in East Africa (project no. INCO-CT2004-510863) alongside the PUMPSEA project, aims at gaining more knowledge and understanding on the structural attributes, socio-economics status, productivity and the regenerative stability of the mangroves of Tudor creek under exposure of raw domestic sewage. Ultimately, the study aimed at identifying the degradation that has occurred over time.

1.4 The study

The mangroves of Mombasa, mainly located in Tudor, Port Reitz and Mtwapa creeks, cover an estimated area of 2,490 ha. The forests resemble the fringing mangroves described by Lugo and Snedaker (1974), with strong inward tidal currents during the flood tides which reverse during ebb tides (Kitheka *et al.*, 2003; Nguli, 2006). No structural studies have been done in these forests, and in particular, no inventories, despite the high utilisation by an urban population in addition to the threats of raw sewage pollution.

1.4.1 Overall objective

The urban character of many ecosystems, mangroves being no exception, is becoming increasingly prominent, with possible impediments to ecosystems' viability (i.e. the inherent capacity or ability to grow, develop or recover after disturbances). We set out to establish the viability of the peri-urban mangrove of Tudor creek in Mombasa, Kenya, and determine the impacts of domestic sewage pollution on the structure and development of the mangrove vegetation. The hypothesis which were tested under this broad objective were that:

1. The structure of the peri-urban mangrove of Tudor creek will posses characteristics depicting raw domestic sewage exposure.
2. The exploitation or harvesting practices have been sustainable but the livelihoods of the local communities are impacted by domestic raw sewage pollution into the mangrove ecosystem,
3. Sewage exposure promotes productivity rates via litterfall for the mangroves of Tudor creek,
4. The regeneration and growth within the sewage impacted mangroves will be enhanced due to higher availability of nutrients.

These studies are the first ever for the peri-urban mangrove of Mombasa, and in particular considering domestic sewage pollution. This study further adds to the few studies done on peri-urban mangroves (Holguin *et al.*, 2006). These pioneer studies will generate information that will expedite in the formulation of guidelines towards a working management plan for the peri-urban mangroves, through the understanding of the status, utilisation patterns, productivity and the regeneration status of the forest. This informed management of the peri-urban mangrove will benefit both the communities and the mangrove ecosystem by introducing an element of sustainability in the management regime.

1.4.2 The approach

This study adopts a system's approach to answer the questions raised in section 1.4.1 above. The approach adopted involves a broad outlook on the vegetation, functional and human aspects of the peri-urban mangroves of Tudor creek. The vegetation aspects of the forest constitute the 'structural support' or 'pillar' around which the ecosystem functions and services depend on. The structural attributes of the vegetation, regeneration and resilience is broadly assessed in chapters 3 and 6. The second component, the human aspect, is presented in chapter 4. The chapter assesses the wood exploitation practices and estimates the wood harvest rates and the management of the forest. The third component, the functional aspect, assesses the productivity of the forest, while establish the possible impacts of raw domestic sewage pollution. A conceptual model for the structure of the thesis is shown in figure 1.1.

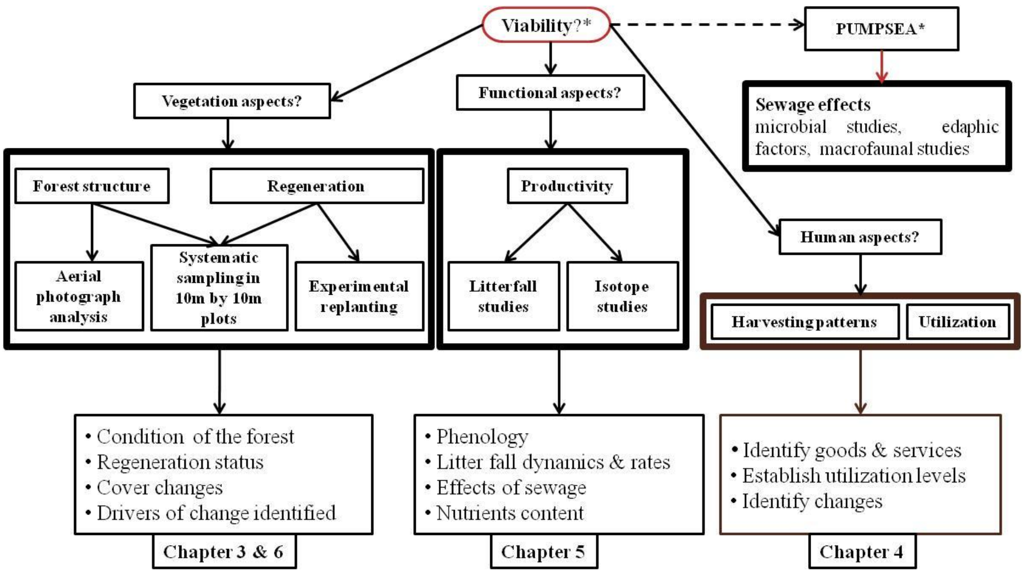

Figure 1.1: A flow chart outlining the structure of the thesis. Emphasis is made on the three main aspects covered, towards assessing the viability of the mangrove ecosystem in peri-urban Tudor creek.

2. Literature review

2.1 The mangrove ecosystem

Mangroves grow at the interface between land and sea where they exist in conditions of high salinity, extreme tides, strong winds, high temperatures and muddy, anaerobic soils. As a result they possess highly developed morphological and physiological adaptations to extreme conditions comparable to no other plant groups (Macnae 1968; Ellison and Stoddart 1991; Krauss *et al*., 2008). Their adaptations to living in a saline intertidal environment (e.g. mechanisms for excluding, translocating and excreting salts) distinguish them from other tropical forests and most mangrove species do not occur in other habitats. They are defined by, and dominated by trees that are essentially terrestrial, but adapted to tidal inundation (Kathiresan and Bingham, 2001).

Mangrove forests are classified according to structural and functional characteristics, geophysical processes and landscape position and are categorized as; over wash islands, riverine, basin, fringe, hammock and scrub forests (Galley *et al*., 1962; Lugo and Snedaker, 1974). A comprehensive ecogeomorphological classification scheme (Woodroffe, 1992; Twilley and Chen, 1998) segregates the classification by geomorphic types (delta, lagoon, delta/lagoon, or estuary mangroves) based upon degree of terrigenous input and position of the mangrove forest relative to this input (Figure 2.1). The classification scheme predicts that forcing functions will act differentially based upon geomorphology and will essentially result in variability in ecosystem dynamics (Krauss *et al*., 2008). These ecological types differ predictably by soil type, salinity, and hydro-period, with structural complexity and productivity increasing from dwarf[♥] to riverine type (Lugo and Snedaker, 1974).

Structurally, mangrove ecosystem are characterised by a general floristic simplicity, with a small number of species occurring in any given mangrove stand compared to other tropical forests. They are dominated by *Avicennia* spp. (Acanthaceae) or various members of the Rhizophoraceae. They consist of tree species occurring in monoculture stands, or mixtures of a few species, and rarely include understorey

[♥] Dwarf mangroves reffers to stunted mangroves mainly due to site conditions such as salinity, nutrient limitation or any other environment or climate factors.

plants with a canopy characterised by limited vertical stratification (Krauss *et al.*, 2008). The mangrove forest floor is often covered with seedlings and saplings of overstorey species (Chapman, 1976; Ball, 1980; Snedaker and Lahmann, 1988), though in some wet areas, species of vines, herbs, ferns, and palms have been listed as occurring in the landward edge, associated with a decrease in soil salinity (Chapman, 1976). The intertidal environment has largely precluded the evolution of: (i) intertidal-halophytic adaptations in shade-tolerant terrestrial or freshwater aquatic species, (ii) true shade-tolerance in intertidal halophytes, or (iii) both characteristics in shade intolerant plants (Snedaker and Lahmann, 1988).

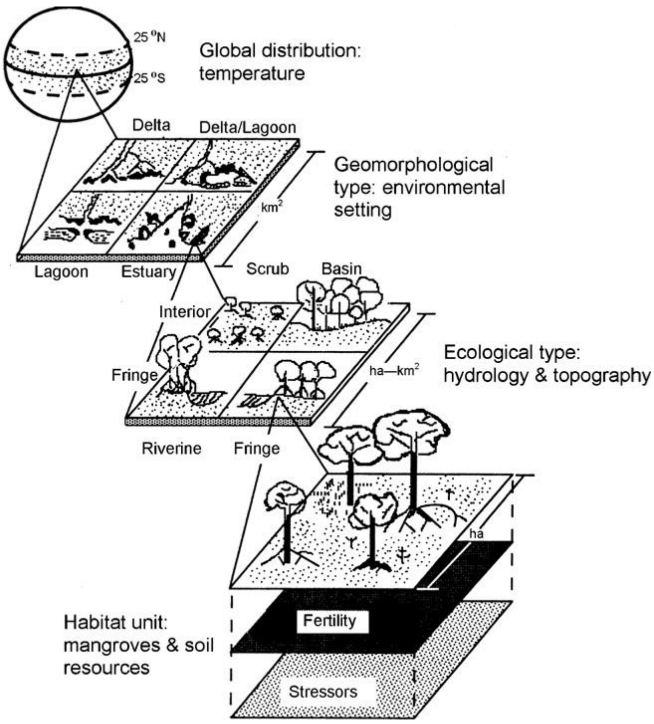

Figure 2.1: Hierarchical classification system (ecogeomorphology) for use among different mangrove ecosystems worldwide, whereby function is based upon geomorphological development, ecological factors, site fertility, salinity gradients, and flood regimes (after Twilley *et al*., 1998 cited in Krauss *et al*., 2008).

Mangrove forest structure is therefore an outcome of geophysical processes and landscape position, with the compounding effects of interactions between species responses to biotic and abiotic stress factors, including disturbances, seed dispersal and competition (Berger *et al.*, 2006). The magnitude and periodicities of forcing functions such as tides, nutrients, hydro-period and stresses such as cyclones, drought, salt accumulation and frost (Lugo *et al.*, 1973; Pool *et al.*, 1977; Cintrón and Schaefer-Novelli, 1984), in addition to processes of gap-phase dynamics, natural disturbance, and forest mosaics (Adams and Levings, 1987; Smith III, 1987a, b;

Duke, 2001), play a major role in influencing mangrove forest community structure. Seed predation by crabs is an important determinant of the forest's species composition and structure (Smith III, 1987a, b; Clarke and Kerrigan, 2002). These factors have been differentially cited to influence the conspicuous segregation of tree species into distinctive bands parallel to the shoreline – zonation (Table 2.1; Macnae, 1968; Chapman, 1976; Tomlinson, 1986).

Table 2.1: Factors affecting zonation of mangroves globally.

Factor	Reference
Geomorphological	Thom, 1967; Woodroffe, 1992
Interspecific competition	Clarke and Hannon, 1971; Ball, 1980
Particle size concentration	Clarke and Hannon, 1971; McKee, 1995
Physiological adaptation to gradients across the intertidal zone	Lugo *et al.*, 1973; Pool *et al.*, 1977; Cintrón and Schaefer-Novelli, 1984
Differential dispersal and predation of propagule	Rabinowitz, 1978; Smith III, 1987a, b; Van Speybroeck, 1992; Dahdouh-Guebas *et al.*, 1999
Interrelatedness of tidal flooding with salinity, fertility, soil saturation, sulphide concentration (redox potential)	Ball, 1988; Ball, 1995; Ball, 2002

From an eco-physiological perspective, species may overlap considerably in their range of tolerances to these factors, and zonation may be absent in some mangroves (Ball, 1988; Bunt, 1996. Krauss *et al*., 2008; Ellison, 2008a). Furthermore, most of the proposed factors that influence zonation are intricately linked to the depth, duration, and frequency of tidal flooding inherent to Watson's (1928) inundation class classification (Table 2.2). Knowledge of zonation and factors influencing it is essential in guiding mangrove restoration through determining the appropriate habitat for preferred species. In Kenya, the typical zonation patterns have been described as *S. alba* growing closest to the low water line, followed by *R. mucronata*, *B. gymnorrhiza*, *C. tagal*, *A. marina*, *L. racemosa* and *X. granatum* respectively (Figure 2.2; Kairo, 1995).

Table 2.2: Watson inundation classification and the related Southeast Asian mangrove species (Source: Watson, 1928).

Inundation class	Description	Flooding frequency (times per month)	Vegetation (species)
1	inundated by all high tides	56–62	None
2	inundated by medium high tides	45–59	*Avicennia* sp., *Sonneratia* sp.
3	inundated by normal high tides	20–45	*Rhizophora* sp., *Ceriops* sp., *Bruguiera* sp.
4	inundated by spring tides,	2–20	*Lumnitzera* sp., *Bruguiera* sp., *Acrostichum aureum*
5	equinoctial tides	Up to 2	*Avicennia* sp., *Lumnitzera* sp.

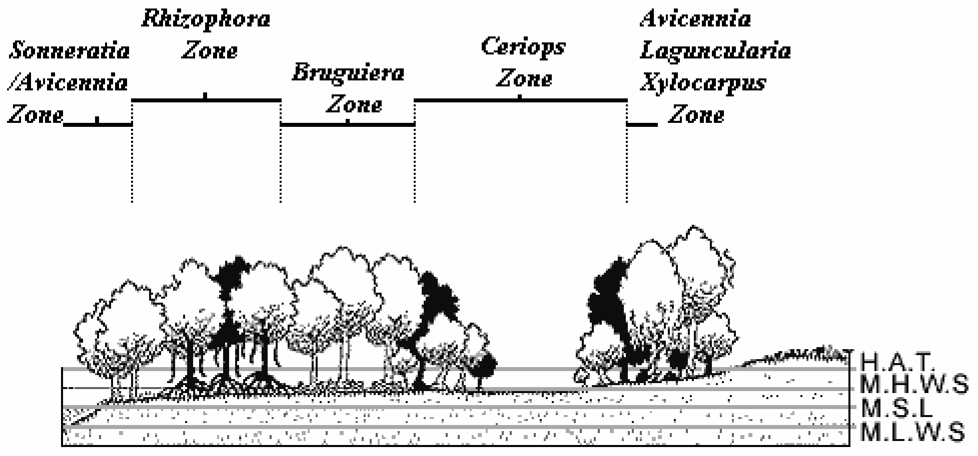

Figure 2.2: The zonation of mangroves in Kenya. Diagram developed based on Kairo (2001).

Description of forest structure includes measures of species composition, diversity, stem height, stem diameter (DBH is usually D_{130} = 130 cm above ground level e.g. Cintrón and Schaeffer-Novelli, 1984; D_{130} *sensu* Brokaw and Thompson, 2000), basal area, tree density, and the age-class distributions (Cintrón and Schaeffer-Novelli, 1984). Maximum D_{130} of 122 cm have been reported for old growth forests in U.S.A (Robertson, 1962; Lugo, 1997) and over 60 cm for mangrove forests in Malaysia, Thailand and Micronesia (Putz and Chan, 1986; Devoe and Cole 1998). The American mangroves had a maximum height of 28 m (Robertson, 1962; Lugo, 1997). A maximum D_{130} of over 80 cm and canopy heights of 35 m has been recorded for mangroves in Kenya (Ferguson, 1993; Kairo, 2001). Spatial distribution patterns of the component species in the forest may also be included. This may reveal insights about the historical and environmental processes, such as regeneration, climate, mortality and competition, which have shaped current structure (Youngblood *et al.*, 2004). Such knowledge can help facilitate the development of silviculture systems and management strategies that meet changing objectives and goals for forested lands (Boyden *et al.*, 2005).

2.2 Adaptations

Temperature: Temperature extremes are important in determining distributional limits of mangroves, but the physiological mechanisms responsible for reduced growth and mortality are not fully understood (Krauss *et al.*, 2008). Mangroves buffer against high temperature damage by negative heliotropism (reorientation of leaves to minimize interception of incident radiation) (Ball *et al*., 1988), increased reflectance properties of the leaf to maintain lower leaf temperature (e.g., hairs on abaxial leaf surfaces of *A. germinans* or the ''silver'' variety of *C. erectus*), dissipation of heat by evaporative cooling during transpiration, and establishment in cool habitats (shade, water). However, high temperatures impacts on root respiration rates of mangrove seedlings, increasing respiration linearly with temperature (20–45°C), and is associated with major changes in root metabolism or membrane integrity near 30°C (McKee, 1996), in addition to causing mortality and decay of rooted seedlings (prior to epicotyl expansion) implying that small changes in soil temperature could have a potentially large effect on growth of mangrove seedlings (McMillan, 1971).

Reproduction: In response to the constraint of frequent tidal and wave action, and substrate instability, mangrove species have adopted strategies to promote successful recruitment (Tomlinson and Cox, 2000; Delgado *et al*., 2001; Thampanya *et al*., 2006). Plants of the family Rhizophoraceae adopt vivipary, where the embryo has no dormancy but grows out of the seed coat and fruit while still attached to the parent plant. The agent of dispersal after detachment (propagule) in this case is a seedling (Juncosa and Tomlinson, 1985). In species of the genus *Avicennia*, the embryo emerges only from the seed coat but does not grow sufficiently to rupture the pericarp. The unit of dispersal being the fruit rather than seedling, a condition referred to as crypto-vivipary (Osborne and Berjak, 1997). Under both these strategies, the agent of dispersal contains stored nutrients, important for rapid rooting in the unstable environment. Other species utilize the classical strategy of producing seeds. Good examples include *Xylocarpus*, where fruits rupture on the parent plant to release angular seeds, and *Sonneratia* which produces fruits laden with seeds. The development of fruits to seedlings in all mangrove species is characterised by an exceptionally high degree of mortality that has necessitated copious production of seedlings, which are not limiting in most mangrove forests (Tomlinson, 1986).

Vivipary is interpreted as a mechanism for either protecting the embryo from high salinity concentrations, or preparing it as a robust seedling for instant immersion in seawater after detachment (Tomlinson and Cox, 2000). For Rhizophoraceae, the most successful mangrove species, vivipary is related to successful establishment strategy in tidally influenced habitats because it produces an elongated seedling that is also capable of long-distance dispersal, which is made possible because of hermetically sealed tissues, inhibiting exchange with ambient fluid.

Root system: Mangroves have developed elaborate rooting systems that firmly anchors them to the substrate, providing anchorage even where substrates are fluid. The root system consists of aerial roots, an adaptation of mangrove species to the environmental stresses of the anoxic intertidal habitat. The root system vary in architecture between species, from tall prop roots in *Rhizophora* to lower pneumatophores in *Avicennia*, *Sonneratia* and *Lumnitzera*, knee roots in *Bruguiera*, and *Xylocarpus mekongensis*, buttresses in *Ceriops* and plank roots in *Heritiera* and *X. granatum* (Tomlinson *et al.*, 1979; Ellison *et al.*, 1999; Kathiresan and Bingham, 2001). The roots of many mangrove species do not penetrate far into the hypoxic substrata, but produce profuse lateral roots for support that extend into the intertidal and subtidal zone, where they become a rare feature: hard substrata in an otherwise soft sediment environment (Ellison and Farnsworth, 1992).

The specialized roots are important sites of gas exchange in hypoxic substrata, with exposed surfaces possessing numerous lenticels (loose, air-breathing aggregations of cells) (Tomlinson, 1986). *Avicennia* possesses lenticel-equipped spongy short pneumatophores (<30 cm) through which oxygen passively diffuses (Ishshalomgordon and Dubinsky, 1992), and under hypoxic and oil-polluted conditions, grow much larger and numerous, increasing the surface area for gas exchange (Saifullah and Elahi, 1992). Oxygen may also pass through non-lenticellular portions of the pneumatophores. Horizontal structures may be important in air exchange, particularly in rapidly growing pneumatophores where the newly formed tip lacks lenticels (Hovenden and Allaway, 1994). Pneumatophores are normally unbranched, but, following the 1991 Gulf War, mangroves in the Arabian Gulf began developing branched pneumatophores and adventitious roots due to oil spill exposure (Boer, 1993).

Salt tolerance: Mangroves are physiologically tolerant to high salinity and possess mechanisms to obtain fresh water despite a low water potential (Ball, 1996). Depending on species, a combination of salt exclusion, salt excretion, and salt accumulation averts the accumulation of salts in tissues. *Rhizophora*, *Bruguiera*, and

Ceriops possess ultra-filters in the root systems which exclude salts while extracting water from the soil. Other genera (e.g., *Avicennia*, *Acanthus*, *and Aegiceras*) take up some salt, but excrete it through specialized salt glands in the leaves (Dschida *et al*., 1992; Fitzgerald *et al*., 1992). Salt excretion is an active process, evidenced by ATPase activity in the plasmalemma of the excretory cells (Drennan *et al*., 1992). Species of *Lumnitzera* and *Excoecaria* accumulate salts in leaf vacuoles and shows a tendency to become succulent. Salt concentrations in the sap may also be reduced by transferring the salts into senescent leaves or by storing them in the bark or the wood (Tomlinson, 1986).

With increasing salinity, some species (e.g. Rhizophoraceae) become conservative in water use and resort to using surface water achieving greater salinity tolerance (Passioura *et al*., 1992). In wet seasons, the fine root biomass increases in response to decreased surface water salinity, directly enhancing uptake of low-salinity water (Lin and Sternberg, 1994). Other species directly regulate salts, but may also accumulate or synthesize other solutes to regulate and maintain osmotic balance (Werner and Stelzer, 1990). *Avicennia marina* accumulates glycine, betaine, asparagine and stachyose (Ashihara *et al*., 1997), while *Sonneratia alba* synthesizes purine nucleotides (Akatsu *et al*., 1996). To facilitate the flow of water from root to leaves the water potential at the leaves is held lower than in the roots (Scholander *et al*., 1964).

As mangrove roots exclude salts during water uptake from soil, soil salts could become concentrated, creating strong osmotic gradients (Passioura *et al*., 1992). This is regulated by the viscous, polymeric substances in the sap, that limits water flow rate and decrease transpiration (Zimmermann *et al*., 1994). Combined with high water-use efficiency, the rate of water uptake is lowered and accumulation of salts in soils is prevented (Ball, 1987; Ball and Pidsley, 1995). These responses are linked to changes in leaf stomatal movement, which may be influenced by factors influencing hydraulic flow through the plant, including vapour pressure deficit and substrata osmotic potential (Naidoo and Von-Willert, 1994). Most species have thick leaves with a thick waxy cuticle on the epidermis and others have hairs and sunken stomata on the underside of leaves to reduce evapo-transpiration.

2.3 Natural regeneration

Mangrove forests are dynamic, ever-growing, and constantly re-establishing and renewing themselves (Duke, 2001). Owing to the multitude of stress factors associated with the tidal zone, and often impacted by terrestrial and river run-off, mangroves possess regenerative properties which are adaptable, progressive, dynamic and mostly successful (Duke, 2001; Bosire *et al*., 2008). They are essentially composed of highly dispersive plants, with buoyant propagules, often utilising vivipary (Tomlinson, 1986; Osborne and Berjak, 1997) and some species that regenerate vegetatively by coppicing, particularly members of the family Acanthaceae (Osborne and Berjak, 1997; Duke, 2001). Seed production is generally abundant with natural regeneration of exploited areas occurring spontaneously provided that sufficient numbers of existing seedlings or new ones survive the consequences of harvesting (Hamilton and Snedaker, 1984). The linear regeneration sampling (LRS) technique provides an overview of the site regeneration potential in terms of seedling abundance, distribution and sizes (FAO, 1994). Seedlings above 40 cm in height are often referred to as "established regeneration", and those below are referred to as "potential regeneration" (FAO, 1994). However, this classification should be adapted to local conditions because in species such as *R. mucronata*, propagules may well exceed 40 cm height without being considered "established regeneration". Effective stocking is assessed by the relative presence, abundance and sizes of all regeneration classes.

The most important site conditions influencing natural regeneration include nature of the substrate, age of swamp, inundation class, salinity and sediments erosion and accretion (Hamilton and Snedaker, 1984; Duke, 2001). Mangrove regeneration is also negatively affected by propagule predation (Dahdouh-Guebas *et al*., 1999; Clarke and Kerrigan, 2002), incomplete removal of the over wood (detrimental to light demanding species) and scouring by drift wood (Duke, 2001). To counter these effects, one of the adopted silviculture practices to improve natural regeneration and diameter growth in mangroves involves thinning to create openings (Devoe and Cole, 1998). Excessive physical damage during logging as well as excess amount of logging debris also threatens regeneration (Duke, 2001). Other hazards include absence of seed bearing trees and weed competition (FAO, 1994).

Studies in Mating forests in Malaysia suggested that for adequate regeneration, the minimum number of parent trees (standards) should be 12 trees ha^{-1} (Tang, 1978). Other studies in the same forest suggested juvenile density of 5,000 to 10,000 ha^{-1} as sufficient for regeneration (UNDP/UNESCO, 1991). Chong (1988) suggested that adequate natural regeneration requires a minimum of 2,500 seedlings ha^{-1}. Based on above figures, most of the previously studied mangrove forests in Kenya can be said to have sufficient natural regeneration (Kairo *et al*., 2001; Bosire *et al*., 2003). An important principle of sustainable forest management is ensuring that an adequate resource of advanced growth natural regeneration survives harvesting operations at the end of each rotation. When species of lower value dominate the composition of the regenerated area, or when regeneration is not sufficiently abundant, natural regeneration cannot be relied upon (Hamilton and Snedaker, 1984) and artificial restoration becomes a necessity (Bosire *et al*., 2008).

2.4 Mangroves and disturbances

Mangroves are prone to frequent disturbances caused by either natural or anthropogenic factors, which alters forest structure and characteristics (Clarke and Kerrigan, 2000; Duke, 2001; Ferwerda *et al*., 2007). Natural disturbances in mangroves arise from events such as windstorms, lightning, frost damage, cyclones, and pathogen out-breaks (Hutchings and Saenger, 1987; Smith, 1996; Sherman *et al*., 2000). Artificial or anthropogenic disturbances vary from selective logging for wood resources (Dahdouh-Guebas *et al*., 2000; Walters 2005a, b; Alongi and de Carvalho, 2008; Walters *et al*., 2008), to total clearance for development purposes (Primavera 2000; Abuodha and Kairo, 2001; Tong *et al*., 2004. Walters *et al*., 2008), and in some areas, die backs as an indirect effect of human pressure (Jupiter *et al*., 2007). This introduces a recurrent and variable stress factor on mangroves, causing degradation, the long-term effects of which are often obscure.

Recovery of mangroves from such variable episodic events may take several decades, and knowledge on the long-term effects of disturbances and dynamics of recovery are either rare or limited. Factors that affect initial re-vegetation of denuded mangroves are diverse, and differ between geographic regions. Investigators have greatly differed on their findings. Some find the effect of salinity on the establishment and survival of propagules to be of major importance (Ye *et al*., 2005), others report the presence of propagules (Sengupta *et al*., 2005) or the predation on propagules (Clarke and Kerrigan, 2002; Bosire *et al*., 2003). But it is widely accepted that a lag in vegetation recovery exists in human impacted areas in contrast to "naturally" disturbed areas. Mangroves affected by hurricanes or lightning recover in shorter periods of around ten years as compared to clear-cut mangroves which take longer than 25 years to recover (Ferwerda *et al*., 2007). This disparity is caused by the absence of juvenile trees and saplings, and probably lack of propagules in human impacted as compared to hurricane impacted areas (Ferwerda *et al*., 2007). The removal of vegetation canopy by hurricanes results in increased growth of saplings and sub-adults replacing dead adult trees fairly quickly (Duke, 2001; Blanchard and Prado, 1995). The amount of vegetation cover left after clearing, however, influences the species composition in the first decades after clearing. This is due to differences in the availability of propagules, predator densities, soil salinities and the availability of micro-sites suitable for establishment (Smith and Duke, 1987; Smith, 1987a, b; Ye *et al*., 2005).

Various studies have been performed on the first stages of regeneration and establishment of mangroves (Sukardjo, 1987; Clarke, 1992; Roth, 1992; Clarke and Allaway, 1993; Blanchard and Prado, 1995). Factors such as fruit dispersal (Clarke and Allaway, 1993; Sengupta *et al*., 2005), predation by crabs (Smith, III, 1987a, b; Osborne and Smith III, 1990; McGuinness, 1997b; Clarke and Kerrigan, 2002; Bosire *et al*., 2005b) and soil properties (Duarte *et al*., 1998) influence the initial stage of colonization. The number of colonising seedlings and subsequent juvenile trees depend on three main factors (i) seed production and dispersion, (ii) their germination and establishment (iii) and their subsequent survival. The proximate presence of well developed mangroves is an essential influence on the rate of seed production, dispersion and the re-establishment of mangroves in cleared areas. Studies on seed dispersal in mangroves indicate short ranges of a few meters for most Rhizophoraceae and a few hundred meters for Acanthaceae (Clarke and Allaway, 1993; Blanchard and Prado, 1995; McGuinness, 1997b).

Difference in recovery rate in later years may reflect a difference in propagule availability between clear-cut and hurricane-affected sites, due to the virtual absence of mangroves in clear-cut areas. However, the circumstances during the first few months or years of the life of individuals may strongly affect the rate at which mangrove species colonize a habitat (McGuinness, 1997a, b). Prevailing conditions during the first years after clearing, favours establishment of *A. marina*, an advantage which diminishes with forest growth. Absence of shade in clearings results in higher soil salinities than under canopy cover (McGuinness, 1997b), particularly in the less frequently inundated areas. Although all mangroves are to a certain extent salt tolerant when fully established, not all propagules are equally resistant (Clough, 1984). *A. marina* is characterized as the most salt tolerant of all the mangrove species (Ball *et al*., 1987), and root initiation and subsequent establishment is hardly hampered by extreme saline conditions (Ye *et al*., 2005).

Even when ample propagules are available, development into maturity may, for several species, be hampered by absence of shade, and subsequent salination of mudflats. Absence of shade and increased salinity favour the settlement of *A. marina* saplings. Under full canopy cover, *A. marina* saplings and juveniles are virtually absent. This indicates that *A. marina* will be replaced by other species in the canopy, showing multi-phase vegetation regeneration in mangrove forests (Ferwerda *et al*., 2007). Predation is also a major factor reducing survival of propagules and differs between mangrove species (Smith III, 1987a; Clarke and Kerrigan, 2002; Lindquist and Carroll, 2004), and geographic regions (Farnsworth and Ellison, 1997a) but occurs globally (Farnsworth and Ellison, 1997b). Globally, *A. marina* seeds suffer most from crab predation. Crab predation is especially high in the mid-tidal range and lower in clearings than under closed canopy (Farnsworth and Ellison, 1997a; Smith, III, 1987a; Osborne and Smith, III, 1990). This is because of the replacement of graspid crabs by the ocypodid crabs in clearings (Osborne and Smith III, 1990). Therefore, Knowledge of the long-term effects of clearing, and the factors involved in the regeneration process can make an essential difference for the success of restoration and management of mangrove areas. With the increased attention for restoration issues (King, 1991; Ellison, 2000a, b), this knowledge becomes even more important.

2.5 Mangroves and potential for sewage treatment

The special adaptations to stressful environments, a high demand for nutrients, rapid growth, high primary productivity, metabolism and turnover, has prompted the use of mangrove ecosystems as natural or constructed wetlands for wastewater treatment (Ye *et al*., 2001; Boonsong *et al*., 2003; Wu *et al*., 2008; Yang *et al*., 2008). Mangroves offer unique advantages for wastewater treatment due to salt tolerance, the perennial nature of the plants and the growth rate and vigour that can be maintained over many years attaining higher vegetation cover and nutrients removal efficiencies (Yang *et al*., 2008). Several studies have established that mangrove ecosystems possess a high capacity as a sink for excess nutrients and other pollutants (e.g., Odum and Heald, 1972; Nedwell, 1975; Silva *et al*., 1990; Morell, 1993; Tam and Wong, 1999; Tam and Yao, 1999). These studies have demonstrated that mangroves can be used as constructed wetlands for wastewater treatment (Wong *et al*., 1997; Ye *et al*., 2001; Boonsong *et al*., 2003). However, a recent observation by Yang *et al*. (2008) showed that mangroves may not be suitable as the sole treatment process for sewage containing high levels of coliform.

Mangrove species possess different efficiencies in the removal of ammonia (NH_4^+), total nitrogen (TN), total phosphates (TP), organic matter and coliform from sewage effluents. Efficiencies ranging between 60 and 90%, with marked seasonal variability, have been reported (Wong *et al*., 1997; Boonsong *et al*., 2003; Yang *et al*., 2008), (Yang *et al*., 2008). The main factors of importance in phosphate removal include precipitation and adsorption (Huett *et al*., 2005). The main mechanism to remove phosphates is physicochemical adsorption. Binding sites available on sediment particles get saturated, eventually lowering phosphates removal (Yang *et al*., 2008). In contrast, wastewater borne nitrogen is distributed in different components of the wetland system, including plant uptake, accumulation in sediments and microbial transformation. The relative importance of these processes is affected by the mangrove species involved, type of wastewater, salinity, tidal regime amongst others (Yang *et al*., 2008; Wu *et al*., 2008). Of the total nitrogen inputs from wastewater, 15-30% is returned to the atmosphere via nitrification-denitrification, the rate being dependent on tidal regime and availability of oxygen in the sediments (Wu *et al*., 2008).

Fertilization experiments on mangrove wetlands indicate that mangroves are either phosphorus (P)-limited (Feller, 1995; Koch, 1997; Koch and Snedaker, 1997) or differentially N- or P-limited across tidal gradients (Boto and Wellington, 1983; Feller *et al*., 2003a, b). The addition of N fertilizer increases photosynthetic electron transport, shoot growth, leaf production, and enhanced photosynthetic CO_2 fixation compared to control and P-fertilized trees (Lovelock and Feller, 2003). However, growth rate may be partly influenced by zonation due to sedimentation and nutrient input variability, as nutrient availability does limit growth (Feller *et al*., 2003b; McKee *et al*., 2002). Few studies have evaluated what types of changes might occur within mangrove ecosystems in response to the ongoing process of eutrophication of the coastal zone, which are often immediately next to oligotrophic, but highly diverse, marine ecosystems (Feller *et al*., 1999; Feller *et al*., 2003a, b). However, patterns of nutrient limitation in mangrove ecosystems are complex. Not all processes respond similarly to the same nutrient, and similar habitats are not limited by the same nutrient when different mangrove forests are compared (Feller *et al*., 2003).

2.6 Impacts of sewage on mangroves

One of the most widely reported effects of sewage pollution involves increased nutrient level in the foliage of mangrove plants (Henley, 1978). Mangroves receiving sewage have been observed to have significantly higher foliage nutrient levels than controls. These foliage nutrient levels were observed to be in equilibrium with those in the soil surrounding the roots. Boto and Wellington (1983) reported increased foliar N (from initial 1.20% to 1.43%) and P (from initial 0.088% to 0.095%) for new leaves of *Rhizophora* from a fertilized site (after ammonium and phosphate enrichment for one year) in Australia. Similarly, Clough *et al*. (1983) reported a foliar N concentration of 2.04% for a mangrove forest which received long-term treated sewage effluent, compared with 1.15% at nearby undisturbed control sites. It is estimated that the increase in plant growth and the nutrient status in plants will lead to the immobilization of a substantial amount of the added nutrients, up to at least 300 Kg N and 30 Kg P per hectare of mangroves annually (Boto, 1992).

Tam and Wong (1997) suggest that mangrove soils have high capacity to retain heavy metals from wastewater, with little uptake by plants, depending on plant age and biomass production. Release into tidal seawater is also minimal. However, they also found higher heavy metal content in roots than in the aerial parts of mangroves. This indicates that roots act as a barrier for metal translocation and protect the sensitive parts of the plant from metal contamination.

On the overall however, no negative effects of sewage on mangrove trees has been reported. However, the effects of sewage pollution on mangrove sediments can be highly variable (PUMPSEA, 2007b; Tam and Wong, 1997). The impacts of wastewater discharge vary significantly from wetland to wetland, depending on the characteristics. Predictions about the impact of wastewater on a specific wetland without first examining its characteristics are impossible (Trattner and Woods, 1989). Within the same type of wetland, different plant species with varied growth rates and physiological adaptation can have varying responses to sewage discharge. In addition, the response of plants to wastewater may also depend on the concentration and composition of the sewage effluent. This is an implication for the system's carrying capacity (Tam *et al*., 1997a, b).

2.7 Stable isotopes studies in mangroves

Mangroves play an important role in sustaining the productivity of inshore and offshore fisheries. There is evidence that mangrove forests are used as feeding areas and nursery grounds, as permanent habitat for some species, and as breeding grounds for some coastal species (Blaber and Milton, 1990; Aksornkoae, 1993; Nagelkerken *et al*., 2000). However, recent studies of food webs in mangrove systems using stable isotope analyses have shown that mangroves do not make a major contribution to shrimp food webs (Kristensen *et al*., 2008; Bouillon *et al*., 2008). But mangroves still play an important role in the carbon balance of tropical coastal ecosystems. They contribute 11% of the total global input of terrestrial carbon into the ocean and 15% of the total carbon accumulating in modern marine sediments (Bouillon *et al*., 2008).

Stable isotope analysis potentially provides indications of the origins and transformations of organic matter (e.g. Newell *et al*., 1995; Bouillon *et al*., 2008) and has proven a useful tool for investigating trophic relationships within food webs (Peterson and Fry, 1987; Bouillon *et al*., 2008). The advantage of this method in food web studies over techniques such as gut content analysis is that it distinguishes assimilated rather than ingested foods, and it can represent a history of assimilation over longer periods (Gearing *et al*., 1991). Isotopes used in these studies include ^{15}N and ^{13}C (Cabana and Rasmussen, 1996; Fry *et al*., 2000; Muzuka and Shunula, 2006; Bouillon *et al*., 2008). $\delta^{13}C$ values can be used to distinguish among photosynthetic pathway types. Typical $\delta^{13}C$ values ranging between -35.1‰ and -21.9‰ have been reported for mangroves (Kristensen *et al.*, 2008; Bouillon *et al.*, 2008). This is the $\delta^{13}C$ range for plants classified as C_3 plants. These $\delta^{13}C$ values may be affected by salinity, nutrient limitation or light limitation (Fry *et al.*, 2000; Bouillon *et al.*, 2008; Medina *et al.*, 2008), factors affecting growth, stomatal conductance and internal carbon dioxide concentrations, potentially influencing carbon isotope fractionation (McKee *et al.*, 2002).

$\delta^{15}N$ on the other hand, has been used as an indicator of anthropogenic impacts or influence (Fry *et al.*, 2000; Bouillon *et al.*, 2008; Medina *et al.*, 2008). The nitrogen pools of animals are typically enriched in $\delta^{15}N$ by 3 – 4‰ relative to their food (Bouillon *et al.*, 2008; Medina *et al.*, 2008). Thus $\delta^{15}N$ studies have been used to define the trophic relationships of organisms (Minagawa and Wada, 1984; Cabana and Rasmussen, 1996; Fry *et al.*, 2000; Muzuka and Shunula, 2006; Bouillon *et al.*, 2008). An assessment of $\delta^{15}N$ signatures of the major sources of nitrogen (atmospheric, fertilizers, soil and sewage) showed that nitrogen derived from sewage stands out as having high $\delta^{15}N$ (modal value of +15‰), compared with the other sources (modal values –5 and +5‰). This shift in $\delta^{15}N$ is attributed not only to the relative trophic level of humans, but also to fractionation occurring during ammonification and subsequent volatilization of nitrogenous waste products (Fry *et al.*, 2000; Bouillon *et al.*, 2008). This results in much greater proportionate losses of the light isotope (Minagawa and Wada, 1984; Cabana and Rasmussen, 1996). Thus, the utility of $\delta^{15}N$ as an indicator of sewage exposure for vegetation is proven.

2.8 Mangroves in Kenya

The Kenyan coastline extends from Kiunga (1° 39’ 10.88” S) in the north to Vanga (4° 40’35.36” S) in the south (ca 575 km; Figure 2.3). Mangroves are a common feature in deltas, creeks, protected bays, islands and river estuaries. Making use of aerial photographs, the Forest Department of Kenya (1983), estimated the total area as 64,426 ha, representing approximately 3.8% of the total forest cover and 0.1% of the total land area. The main difference in cover estimates are due to mapping the vegetation area by Doute *et al*. (1981), against the Forest Department beacons that are set at the high water mark at spring tides. Mangroves are mainly concentrated on the north Kenya coast around the Lamu archipelago (33,500 ha; ca. 67%) and the permanent Tana/Sabaki river estuaries (3,045). Smaller wetlands in the mouths of semi perennial and seasonal coastal rivers occur on the south coast, at Kwale (Shimoni-Vanga, Funzi Bay and Gazi Bay (8,375 ha), Mombasa (Port-Reitz, Tudor and Mtwapa Creeks, 2,490 ha) and Kilifi (Kilifi and Mida Creeks, 5,570 ha) (Figure 2.3).

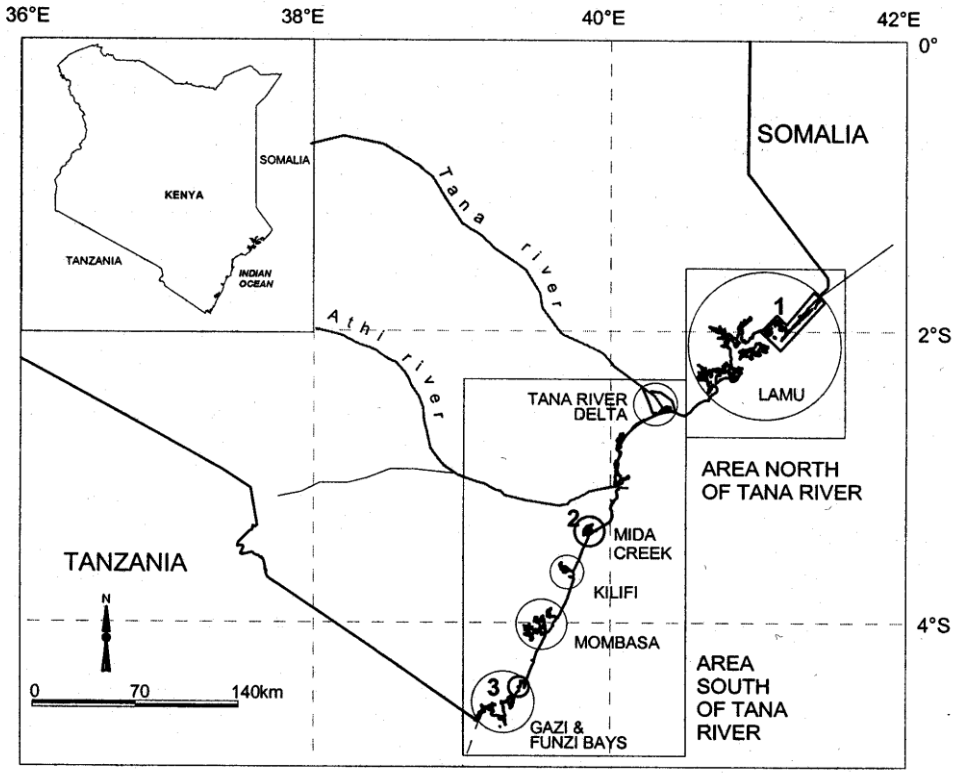

Figure 2.3: The coastline of Kenya showing major mangrove areas, divided into two broad regions north and south of Tana River. The numbers indicate some pilot research areas: 1 – Kiunga Marine National Reserve; 2 – Mida Creek; 3 – Gazi Bay (adapted from Kairo, 2001).

Two types of community, as creek or fringe mangroves occur along the Kenyan coast (Macnae, 1968; Ruwa, 1993; Kairo, 2001). The creek mangrove community is composed of mangrove trees that grow on low lying sedimentary shores in creeks and bays, and usually form well-developed forests that may show species zonation. *R. mucronata* and *C. tagal* are dominant and represented in almost all mangrove formations. The rare species are *Heritiera littoralis* Dryand and *Xylocarpus moluccensis* (Lamk.) Roem.♦ (Macnae, 1968; Kokwaro, 1985; Dahdouh-Guebas *et al*., 2000; Kairo, 2001). A strong zonation of species controlled by the tidal regime, yields the following typical pattern from the sea to land: *Sonneratia alba* J. Smith, *R. mucronata*, *Bruguiera gymnorrhiza* (L) Lam., *Ceriops tagal* (Perr) C. B. Robinson, *Avicennia marina (*Forssk.) Vierh., *Xylocarpus granatum* König, *Lumnitzera*

♦ The distribution of the species *Xylocarpus moluccensis* is out of its known range (Asia), and further verification of the species may be necessary.

racemosa Willd. and *H. littoralis* is common (Kairo, 2001). However, variability can be high leading to shifts in zonation (Gallin *et al*., 1989; Dahdouh-Guebas *et al*., 2002). Detailed botanical descriptions of the species have been covered by Graham (1929), Kokwaro (1985), Gallin *et al*. (1989) and Van Speybroeck (1992).

Studies on mangroves in Kenya have assessed the distribution patterns (Gang and Agatsiva, 1992; Ferguson, 1993), utilization (Kokwaro, 1985; Dahdouh-Guebas, *et al*., 2000; Dahdouh-Guebas *et al*., 2004), community composition (Gallin *et al*., 1989; Van Speybroeck, 1992) and zonation (Kokwaro, 1985; Beeckman *et al*., 1989; Gallin *et al*., 1989; Gang and Agastiva, 1992) and regeneration (Van Speybroeck, 1992; Kairo *et al*., 2002a; Bosire *et al*., 2005a). Some studies have indicated presence of extensive bare lands resulting from indiscriminate cutting of mangroves in Kenya as well as other East African countries (e.g. Semesi, 1998; Kairo, 1992; Ferguson, 1993; Dahdouh-Guebas *et al*., 2000; Kairo, 2001). Recently, there have been several studies on specific ecological aspects of the mangrove ecosystem (Dahdouh-Guebas *et al*., 2004; Obade *et al*., 2004; Bosire *et al*., 2005a, b; Bosire *et al*., 2006; Kairo *et al*., 2008; Kirui *et al*., 2008). Quantitative assessments have been done in Kiunga and Mida creek and functional and structural recovery of restored mangroves in Gazi Bay (Kairo, 2001; Kairo *et al*., 2002a, b; Bosire *et al*., 2003; Bosire *et al*., 2006; Tamoh *et al*., 2008; Kirui *et al*., 2008; Kairo *et al*., 2008; De Nitto *et al*., 2008).

It is estimated that 10,310 ha of Kenyan mangrove forests have already been lost either due to conversion pressure, over-exploitation of resources or pollution effects (Abuodha and Kairo, 2001). The impact of mangrove deforestation in Kenya is reflected in the degradation of forest quality rather than quantity (Abuodha and Kairo, 2001; Kairo, *et al*., 2002a, b). This translates into a loss of economic and ecological values of mangrove forests. The pressure to harvest the market-preferred pole sizes has led to structural changes in mangrove forests at times causing species shifts (Kairo *et al*., 2002a). Climatic changes, especially the 1997-98 ENSO has also contributed to massive sedimentation in mangrove forests leading to extensive destruction (Kitheka *et al*., 2003; Dahdouh-Guebas *et al*., 2004).

2.9 Socio-economics in Mangroves

Mangrove forests have provided a diverse range of goods and services in the form of timber and firewood, finfish and edible crustaceans, bioactive compounds for tanning and medicine and fodder for centuries (FAO, 1994; Bandaranayake, 1998; Hamilton and Snedaker, 1984; Primavera, 1998; Barbier, 2000; Dahdouh-Guebas *et al*., 2000; Nagelkerken *et al*., 2008; Walters *et al*., 2008). Many mangrove species, especially in Southeast Asia, India and Bangladesh, and East Africa, grow large enough to be used extensively as construction and boat-building timber (FAO, 1994; Farnsworth and Ellison, 1997b; Semesi, 1998; Dahdouh-Guebas *et al*., 2000; Kairo *et al*., 2001; Kairo *et al*., 2002a, b). Environmental services of ecological and economic values are diverse, making mangrove wetlands multiple use systems that provide protective, productive and economic benefits worldwide. These include shore erosion protection, buffering against floods and siltation of adjacent sea grass beds and corals, tsunami and storm protection (Wolanski, 1992; Dahdouh-Guebas *et al*., 2005b; Kathiresan and Rajendran, 2005a; Dahdouh-Guebas, 2006), habitats and spawning grounds for fish and endangered fauna like sea turtles as well as feeding habitats and stopovers for numerous migratory birds (Nagelkerken *et al*., 2008), and diverse vertebrates and invertebrates (Kimani *et al*., 1996; Nagelkerken *et al*., 2008; Ellison, 2008a). Mangroves also play an important role in the global carbon cycle (Twilley *et al*., 1986 Kristensen *et al*., 2008; Bouillon *et al*., 2008).

The annual economic value of mangroves estimated by the cost of the products and services they provide has been estimated to be $200,000 - $900,000 per hectare (Barbier, 2000; Barbier and Cox, 2002; Walters *et al*., 2008). However, despite the high utility of mangrove goods and services, it is only within the last 100 years that active management of these forests was initiated (FAO, 1994; Walters *et al*., 2008), first for timber, firewood and pulpwood production and more recently for cultivation of fish, shrimp, and especially the tiger prawn *Penaeus monodon* (Primavera, 1998). However, despite repeated claims that mangrove forests can be managed sustainably, both managed and unmanaged mangroves continue to degrade (Saenger *et al*., 1983; Duke *et al*., 1998; Ellison *et al*., 1999; Duke, *et al*., 2007; Walters *et al*., 2008).

Table 2.2: Mangrove species in Kenya and their uses (Source: Kairo and Dahdouh-Guebas, in press)

Family	Species name	Local Names (Kiswahili)	Uses
Acanthaceae	*A. marina*	Mchu, Mtu, Msingi, Mutsi	timber, firewood, charcoal, insecticides, bed posts, chair legs, table legs, fencing posts, crushing pole, crushing mortar, serving dishes, drums, boat ribs, board games (*bao*)
Rhizophoraceae	*B. gymnorrhiza*	Muia, Mkandaa giza, Msumari, Mkoko jike, Msinzi, Msidi	construction poles, roof supports, firewood, charcoal, boat paddles, oars, handcart handles, axe handles, pounding poles, drums, bee hives
Rhizophoraceae	*C. tagal*	Mkandaa	firewood, construction poles, paddles, oars dyes (incl. tanning compounds), fishing traps, bed posts
Combretaceae	*L. racemosa*	Kikandaa, Mkandaa Dume, Mnamwa	firewood, charcoal, fences
Sonneratiaceae	*S. alba*	Mlilana, Mpia	firewood, charcoal, canoes, boat ribs, paddles, masts, fishing net floats, timber for window and door frames
Rhizophoraceae	*R. mucronata*	Mkoko	Firewood, charcoal, poles, tannin, fence posts, fish traps.
Meliaceae	*X. granatum*	Mkomafi, Mtonga, Mronga	timber for bed construction, window and door frames, firewood, charcoal, ointments, carving
Meliaceae	*X. moluccensis*	Mkomafi dume	firewood, fencing poles
Lythraceae	*Pemphis acidula* Forst	Unknown	firewood, fencing poles
Sterculiaceae	*H. littoralis*	Msikundazi	charcoal, firewood, poles, boat mast

In Kenya, historical records indicate that along with the slave and ivory trade, mangrove poles made up a major regional trade by the 9th Century A.D. (M. Niebuhr, 1792 quoted in Rawlins, 1957), and to date mangroves still have a wide range of uses (Table 2.2). However, due to overexploitation and subsequent degradation of mangrove forests, a ban on charcoal production out of mangrove wood was imposed in 1970 (Luvanda *et al*., 1997), followed by a ban on mangrove exports in 1982. Despite this ban, mangrove deforestation continued in order to satisfy the growing local demand (Dahdouh-Guebas *et al*., 2000; Kairo *et al*., 2002a, b). This is evidenced by the actual average harvest per year from 1983 to 1993 (average 30,000 scores yr-1) which was similar to that for the period 1941-1956 (average 31,700 scores yr-1) (Ferguson, 1993; Kairo, 2001; Figure 2.4). In 1997, the government banned the exploitation of all forests including mangroves but lifted the ban in 2002 to help protect the coastal communities from economic hardships due to their dependence on mangroves.

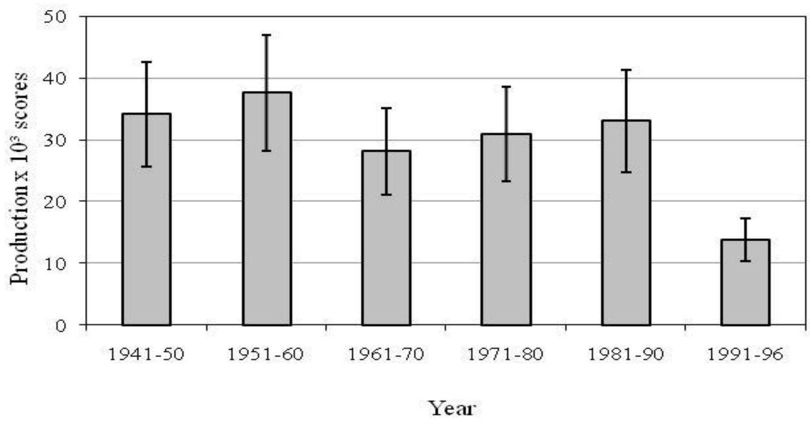

Fig. 2.4: Trends in production of mangrove poles in Lamu. Last complete data set is from 1996. [compiled from Rawlins (1957) and the Forest Records 1950-2000 data available from the Forest Department offices in Lamu and Mombasa]. (1 score = 20 poles).

2.10 The management of mangroves

Mangroves were first gazetted as forests in 1932, and the Forest Department, now re-established as the Kenya Forest Service (KFS) (GoK, 2007) has been responsible for their management. Management of coastal resources and environment in Kenya is governed by several legislation covering different sectors. The National Environmental Action Plan identifies 76 statutes relating to the environment, most of which will apply to the mangrove environment (See Annex 1). Specifically, mangroves are protected under the Forestry Act; as well as Wildlife Conservation and Management Act; when they occur within Marine Protected Areas. Responsibilities to manage mangroves are bestowed on the Kenya Forest Service, either singly, or in partnership with the Kenya Wildlife Service.

2.10.1 Overview of existing legislation

2.10.1.1 Laws and legal status of mangroves

The forest policy in Kenya is contained in Sessional Paper no. 1 of 1968. The legal mandate of Forest Department to manage all forests in the country was given by the Forest Act (Cap 385) of the Laws of Kenya, with its amendments. This Act has been reviewed and re-establishes the Forest Department as Kenya Forest Service. Mangroves have had legal status as government reserve forests since Proclamation

No. 44 of 30th April 1932, and later by Legal Notice No. 174 of 20th May 1964. Under this "Gazette Notification for Mangrove Forests in Kenya" all land between high water and low water marks (ordinary spring tides) in the localities described below, is included:

- On the mainland and islands adjacent to the coast from Kimbo Creek in the South, to the village of Kiunga on the mainland in the north;
- On the banks of Tana river (northern branch) between Kipini and Kao (Kau);
- On the mainland and islands to the coast from the mouth of the northern Kilifi river in the north, to Ras Ngomeni in the south;
- In the following creeks and all branches thereof: Mida (Uyombo), Kilifi (Southern), Takaungu and Mtwapa;
- In all tidal areas lying to the north west, west and south west of the straight line between Ras Kinangone (Flora point) at the entrance of Port Reitz, and Ras Junda (southern most point of subdivision 232 of section II, mainland North at the entrance to Port Tudor, but excluding any portion of the shore of Mombasa island;
- On the mainland and islands adjacent to the coast from Chale Point in the north, to the boundary of the Trust Territory of Tanzania in the south.

The Kenya Wildlife Service (KWS), established in 1990 under the Wildlife Conservation and Management Act, is responsible for protection, conservation and management of wildlife in Kenya. KWS manages marine and terrestrial parks and reserves; some of which contain mangroves. These include the Watamu Marine National Park and Reserve (Mida Creek), the Kiunga Marine National Reserve and the Tana River Primate Reserve.

2.10.1.2 Other Laws Related to mangroves management and conservation

The Fisheries Act (Cap 378 of the Laws of Kenya) provides for the development, management, exploitation, utilization and conservation of fisheries resources in Kenya. Under the Act the Fisheries Department, in collaboration with other appropriate agencies shall promote the development of traditional and industrial fisheries, fish culture and related industries; and may impose measures necessary for the proper management of any fishery. Accordingly, authority to develop mangrove areas for aquaculture must be given by both KFS and Fisheries Department.

Regulations for pollution and its control are spread in several Acts, with different enforcing agencies. These include the *Factory Act*, the *Public Health Act*, *Merchant Shipping Act* and the *Local Government Act*. *The Environment Management Coordination Act* of 2000, which led to the creation of the National Environmental Management Authority (NEMA), consolidates statutes on pollution of the environment, particularly the coastal environment.

International Conventions: Kenya is party to several international conventions including the *Convention of the Law of the Sea*, *Framework Convention on Climate Change*, *Biodiversity Convention*, *Nairobi Convention* (the Convention for the Protection, Management and Development of the Coastal Environment of the Eastern African Region), and the *Ramsar Convention*. These conventions govern the access, use and management of coastal resources that includes mangroves.

Table 2.3: A list of national statutes and international treaties that govern the management of natural ecosystems in Kenya (source: Kairo and Dahdouh-Guebas. In press).

National Statutes	
1. The Public Health Act, Cap.242	
2. The Local Government Act Cap.265	
3. The Trust Land Act, Cap.288	
4. The Land Planning Act, Cap.303	
5. The Agriculture Act, Cap.318	
6. The Wildlife (Conservation And Management) Act, Cap.376	
7. The Tourism Industry Act, Cap.385	
8. The Forests Act, Cap.385	
9. The Tourist Development Corporation Act, Cap.382	
10. The Tana And Athi Rivers Development Authority Act, Cap.443	
11. The Coast Development Authority Act, No.20 Of 1990	
12. The Fisheries Act, No.5 Of 1989	
13. The National Water Conservation Pipeline Corporation Act L/No.270 1988	
14. The Timber Act, Cap.386	
15. The Government Lands Act, Cap.280	
16. The Land Titles Act, Cap.282	
17. The Land Consolidation Act, Cap.283	
18. The Land Adjudication Act, Cap.284	
19. Land (Group Representatives) Act, Cap.287	
International treaties	
Convention on Biodiversity	Ratified 26 Jul 1994
Convention on Wetlands	Entry into force 05 Oct 1990
UN Convention on Climate change	Ratified 30 Aug 1994
Kyoto Protocol on Climate Change	Not Accepted to date
Protection of World Cultural & Heritage Sites	Acceptance 05 Jun 1991
International Trade in Endangered Species	Ratified 13 Dec 1978
UN Convention on the Law of the Seas	Ratified 02 Mar 1989

Although there are many laws that address mangrove conservation and management in Kenya, successful mangrove forest conservation is impeded by: insufficient quantitative information to guide cutting plans, rising human population along the coast (up to 400 households per km^2) (Republic of Kenya, 1999) and lack of adequate trained manpower in mangrove silviculture (Waithaka and Mwathe, 2003), poor administration structures, absence of provisions to specify standard of performance, inadequate incentives, and low level of participation of the communities. Government bureaucracy, outdated policies and meagre budgetary allocation are also to blame (Waithaka and Mwathe, 2003). The enactment of Environmental Management and Co-ordination Act (EMCA) in 1999 and the new Forest Act (2005) which proposes restructuring of the Forest Department is seen as the panacea to the problems dodging forest conservation and management in Kenya. It is hoped that this will promote implementation of sustainable silviculture practices. However, assessing the current status of the remaining forests remains a prerequisite.

2.10.1.3 The local communities perception of the legislation

It is no doubt that Kenya has elaborate laws that address the salient issues on environmental conservation. However, the success and impact of any legislation is linked to the level of understanding of the legislation by the subjects. This may be a factor that has contributed to failures on enforcement, in addition to the poverty of the local communities. Poverty has the effect of necessitating an action to meet a demand, otherwise unaffordable, even if it means infringing on the law. Therefore, whereas a good majority of the population are aware of the legislation, their understanding or working knowledge of it may be poor. Furthermore, most forestry laws are not effectively enforced, with land tenure issues being central to most conservation initiatives.

2.11 Study area

Tudor creek (Figure 2.6) bounds Mombasa Island on the northwest and extends some 10 km inland. The creek has two main seasonal rivers, Kombeni and Tsalu, draining an area of 55,000 ha (45,000 and 10,000 ha respectively) with average freshwater discharge estimated at 0.9 m^3s^{-1} during the long rains (cited in Nguli, 2006). It has a single narrow sinuous inlet with a mean depth of 20m, that broadens out further inland to a central relatively shallow basin (5m) fringed by a well developed mangrove forest mainly composed of *R. mucronata* (Rhizophoraceae), *A. marina* (Acanthaceae) and *S. alba* (Sonneratiaceae). The basin has an area of 637 ha at low water spring tide and 2,235 ha at high water spring tide. Mangrove forests occupy 1,465 ha of the creek. The floristic composition of mangroves of Tudor creek has been described by SPEK (1992). There is no obvious zonation displayed by the dominant mangrove species in Tudor creek. *A. marina* and *L. racemosa* occupy the landward zone, whereas mostly a *C. tagal* and *R. mucronata* mosaic covers the middle zone. Wherever present, *S. alba* occupies the seaward margin, but is replaced by tall *A. marina* and *R. mucronata* along small creeks.

The forest resembles the fringing mangroves described by Lugo and Snedaker (1974), with strong inward tidal current during the flood tides which reverses during ebb tides, attaining maximum tide velocities of 0.6-0.7 ms^{-1} (Nguli, 2006), and the dense, well-developed prop roots that accumulate large stocks of debris, with a spring tidal range of 3.5 m and a neap tidal range of 1.1 – 1.3 m.

The mangroves of Tudor creek are naturally separated into two main tidal creeks, Kombeni and Tsalu, 4.5 and 3 km long respectively, cutting through the mangroves connecting to the upstream rivers (Figure 2.6). In the framework of this study these two tidal creeks were sampled separately and compared. Tsalu, the eastern tidal creek, includes mangroves near the rural villages of Mirarani, Voroni, Junda and Kijiwe. These villages are sparsely populated and lack formal structures and basic amenities such as tarmac access roads, piped water, electricity and sewage and solid waste handling facilities. Kombeni, the western tidal creek, includes the cosmopolitan densely populated townships of Mikindani and the extensive settlements of Jomvu and Miritini. The major Mombasa – Nairobi Highway passes through the area, which is also designated an industrial area, with numerous warehouses that offer storage support services to the Kilindini Harbour, the biggest sea port in East Africa. This area also neighbours the Moi international airport and the Kenya Petroleum Oil Refineries. The creek is bordered by a steep cliff overlooking a tidal flat that extends to the mudflats occupied by the vast mangrove forest (Figure 2.5).

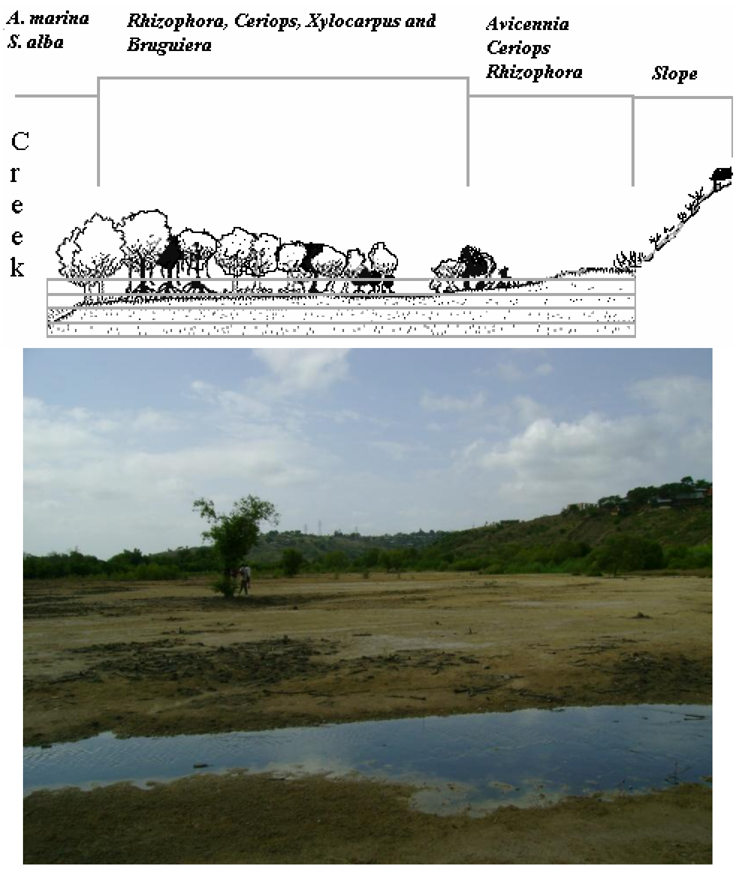

Figure 2.5: Photo with diagram demonstrating the steep slope that surrounds the creek. In the foreground is a channel of sewage flowing into the mangroves at Mikindani (Photo taken in 2007).

The Mikindani mangrove system has been exposed to sewage for more than a decade. The sewage runs through the mangrove forest in canals and is discharged into the Tudor creek waters at a point source. The mangroves of Mikindani are continuously dozed with raw domestic sewage, which is periodically diluted and dispersed into the forest tidal cycle. This creates a pollution gradient, where loading exponentially reduces with distance from the source. About 1,200 kg of nitrogen and 5.5 kg of phosphorus are discharged via sewage into the Mikindani system every day (PUMPSEA, 2007a). This sewage enters via a point source and directly impacts an mangrove area of 300m². The Mikindani township with reportedly 67,000 people is located in the west Mombasa mainland. Sewage from this estate was according to plans intended to be pumped for processing at the Kipevu treatment plant. But due to overloading at the Kipevu plant, sewage from the estate is now discharged directly into the nearby mangrove forest at Mikindani in Tudor creek. Sediments of Tudor creek are predominantly mud and some parts are covered with sand. The land surrounding the creek beyond Mombasa Island is mainly agricultural, largely small-holdings and coconut plantations with rough grazing land further inland.

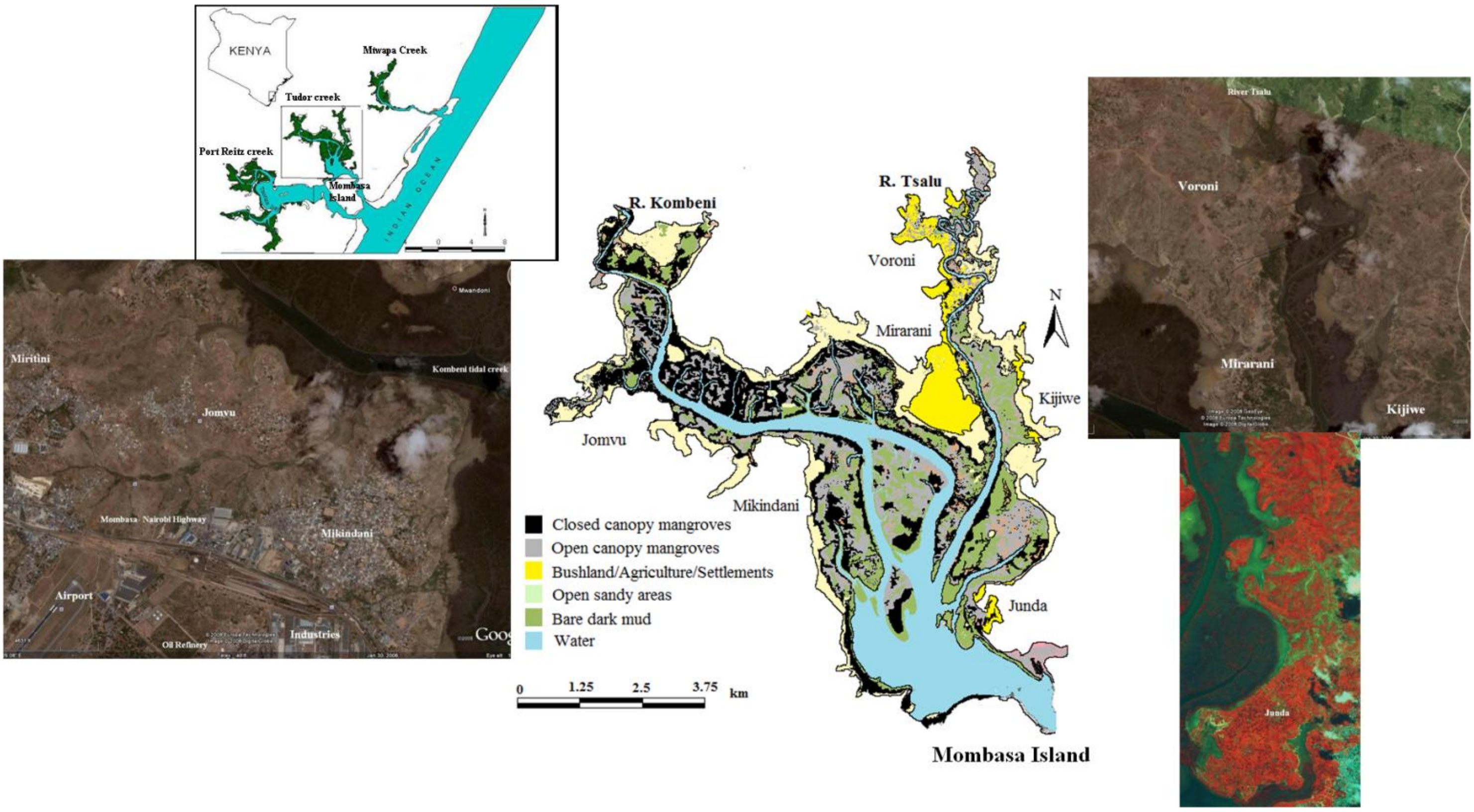

Figure 2.6: Map of Mombasa showing the study area (Tudor creek) in relation to the other creeks. Notice the proximity of settlements to the mangroves and the differences in density between the urban Kombeni and Rural Tsalu tidal creeks (Source, IKONOS image, 2005).

2.11.1 Climate

The climate of Mombasa is influenced by the semi annual passage of the inter-tropical convergence zone (ITCZ) and the monsoons (Figure 2.7). The North Easterly Monsoon (NEM) occurs from December to March, and the South Easterly monsoon from May to October. The mean annual rainfall is 1,038 mm with the months of April, May and June recording the maximum. Average annual temperatures for the two seasons are 23.9°C and 28.5°C respectively (Obura, 2001).

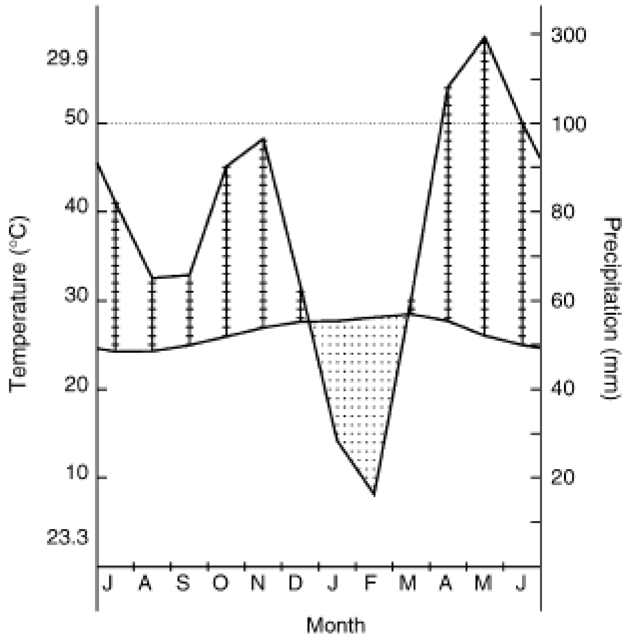

Figure 2.7: Climate diagram for Mombasa (Lieth *et al.*, 1999).

2.11.2 Findings of the PUMPSEA project on the study area

To derive the answers to questions raised in this study, we also looked at the main findings of the PUMPSEA project at Tudor creek. We only present the major findings in relation to sediment conditions and macrofauna. These findings include the sediment microbial and nutrients conditions and the status of the macrofauna. These observations, together with our findings, help to derive conclusions on the viability of the mangroves of Tudor creek.

Sediments: it was observed that the sewage polluted site has a higher benthic metabolism and more reduced sediment conditions. This may have implications for the nutrient removal efficiency because both N and P removal are strongly dependent on oxidized conditions. High rates of ΣCO_2 production were recorded at the polluted sites, indicative of a system under stress due to the presence of easily degradable organic matter. The implication is that the system in the natural setting does not get enough time to stabilize since it is dozed continuously with sewage containing labile organic matter (PUMPSEA, 2007a). It was also observed that sewage pollution tends to have a negative effect on sediment cyanobacteria diversity, while promoting microalgal densities.

Macrofauna: It was established that clear trends of changes in crabs and molluscs assemblages between non peri-urban (Gazi Bay) and peri-urban mangroves exists. Crab assemblages were reported to be more diverse at the Mikindani peri-urban sites than the corresponding non peri-urban sites (Gazi Bay), with a shift in dominance from the Ocypodidae to other Uca spp and Sesarmidae in sewage dumping areas. The increase in biomass of fiddler crabs of the genus Uca was linked to the enhanced nutrients concentration and sediment surface biofilm due to sewage loading, via a likely increase of benthic diatoms and bacteria, which they feed upon (Meziane and Tsuchiya, 2002). However, the nutrients concentration and the purported presence of other pollutants were not sufficient to induce a reduction of either Uca species or populations (Yu *et al*., 1997). The fiddler crabs were observed to occur in high densities under *R. mucronata* stands, while sesarmid crabs were present in high densities under *A. marina* stands (PUMPSEA, 2007c).

On the abundance of mollusc species, *Terebralia palustris,* totally disappeared in sewage polluted areas. *T. palustris* is an important gastropod in East African mangroves that feeds on both mud associated organic matter, as juveniles, and leaf litter, as adult, and compete with sesarmid crabs for food (Pape *et al*., 2008). The bacteria feeding congeneric, *T. sulcata*, increased in density under sewage pollution. Sewage pollution seem to provide more food sources for crabs, but not the case with gastropods, which apparently occupy the same ecological niche with crabs. This indicates the variable physiological tolerance by different species within the ecosystem. Thus these responses do not mean that the sewage polluted systems are healthy ecologically. Such a strong alteration in terms of biomass of a crucial component can lead to an unsustainable alteration of ecosystem functioning and structure (Duke *et al*., 2007; Cannicci *et al*., 2008), eventually causing a collapse of the system itself.

However, the PUMPSEA project is inconclusive on the use of mangroves for sewage treatment. This is mainly because the experimental sewage treatment has not generated enough data on the effects on the ecosystem. What may also be important to establish is how much sewage load can be treated in a given time and area. This is in addition to the stand structure by species composition and stem density.

3: Mangrove forests in a peri-urban setting: the case of Mombasa (Kenya)

Tudor creek mangroves from the rural Tsalu side - a *patchy* distribution of mangroves with wide open spaces

Publication

Mohamed, M. O. S., Neukermans, G., Kairo, J. G., Dahdouh-Guebas, F., and Koedam, N. 2008. Mangrove forests in a peri-urban setting: the case of Mombasa (Kenya). *Wetlands Ecology and Management*, doi: 10.1007/s11273-008-9104-8.

Summary

The structure and regeneration patterns of the peri-urban mangrove vegetation of Mombasa at Tudor creek were studied along belt transects at 2 forest sites of Kombeni and Tsalu. Based on the species importance values, the dominant mangrove species were *R. mucronata* and *A. marina*. *L. racemosa*, reported in an earlier floristic survey, was not encountered. Tree density varied from 1,264 trees ha^{-1} at Kombeni to 1,301 trees ha^{-1} at Tsalu and mean tree height was higher at the former site compared to the latter. The size-class structure at both localities showed the numerical dominance of small trees over larger trees. The spatial distribution pattern of adults and juveniles varied greatly between sites and showed a close to uniform pattern (Morisita's Index $I_\delta << 1$) for adult trees, but a tendency to clustered distribution ($I_\delta >> 1$) for juveniles. The present paper shows that unmanaged but exploited peri-urban mangroves are structurally degraded, with numerous large canopy gaps that are characterised by spatial and temporal site heterogeneity. These gaps influence regeneration, implying longer periods for canopy closure. Diversifying uses of mangrove products and establishing reserves as no-cut zones with regulated harvesting will minimise canopy gap sizes, and promote conservation practices. The proposed management strategy shall boost the ecosystem resilience to both anthropogenic and natural disturbances expected in the peri-urban setting in the long run.

Keywords: anthropogenic, canopy gaps, peri-urban, regeneration

3.1 Introduction

Mangrove swamps are typical wetland ecosystems found in coastal deposits of mud and silt throughout the tropical and subtropical coasts. These ecosystems are currently estimated to cover 15.2 million hectares of the tropical shorelines of the world (FAO, 2003; 2005), representing a decline from 18.8 million hectares of mangroves in 1980 (Alongi, 2002; FAO, 2005; Duke *et al.*, 2007). As intertidal ecosystems, mangrove forests provide essential functions and services to coastal zones and to their plant, animal and human populations (Cannicci *et al.*, 2008; Kristensen *et al.*, 2008; Nagelkerken *et al.*, 2008; Walters *et al.*, 2008). Mangrove degradation over time has been recurrently documented (Dahdouh-Guebas and Koedam, 2008; Ellison, 2008a, b), but direct and indirect anthropogenic pressure persists. Diversion of freshwater flows, deteriorating water quality caused by pollutants and nutrients, over-harvesting for fuel-wood and timber as well as conversion into other uses like agriculture, aquaculture, mining, salt extraction and infrastructure all contribute to the degradation and deforestation of mangrove ecosystems (Saenger *et al.*, 1983; Terchunian *et al.*, 1986; Primavera, 1995; Twilley *et al.*, 1995; Mandura, 1997; Ellison, 1998; Dahdouh-Guebas *et al.*, 2000; Valiela *et al.*, 2001; Abuodha and Kairo, 2001; Kairo *et al.*, 2001; Taylor *et al.*, 2003; Benfield *et al.*, 2005; Dahdouh-Guebas *et al.*, 2004; Dahdouh-Guebas *et al.*, 2005b; Duke *et al.*, 2007). Climate change poses an additional threat (Gilman *et al.*, 2008).

In Kenya, mangroves are well developed in many areas along the coastline, particularly in creeks, bays and estuaries. It is estimated that mangroves cover 52,980 ha in Kenya (Doute *et al.*, 1981), though the KFS (Forest Department of Kenya, 1983) estimates 64,426 ha. The bulk of these forests occur in Lamu (33,500 ha), with smaller forests in Kwale (8,800 ha), Kilifi (6,600 ha) and Mombasa (2000 ha) (Doute *et al.*, 1981). Traditionally, mangroves in Kenya have been used as a source of building poles and firewood (Kokwaro, 1985; Dahdouh-Guebas *et al.*, 2000). During the colonial period (before 1963), mangrove wood products formed major export commodities, and by 1950's, sale of mangrove products were ranked third in the national revenue generation (Rawlins, 1957). Overexploitation led to a presidential ban on mangrove exports in 1982. However, harvesting for the domestic market intensified in many parts of Kenya, leading to further overexploitation and subsequent degradation (Kairo, 2001; Abuodha and Kairo, 2001; Dahdouh-Guebas *et al.*, 2000; Taylor *et al.*, 2003; Dahdouh-Guebas *et al.*, 2006; 2006a). It is estimated that 20% of the total mangrove forests in Kenya have been lost already (Abuodha and Kairo, 2001). This raises concerns about the sustainability of mangrove resource utilisation especially because human migration into coastal zones is high. Coastal zones are currently estimated to provide living space for 60% of the world's population (World Resources Institute, 1996; Sheppard, 2001), and the dependence on mangroves by coastal fisher folk is large (Walters *et al.*, 2008).

The major problem facing the management of mangrove forests in Kenya is the lack of a management plan (Dahdouh-Guebas *et al.*, 2000; Kairo *et al.*, 2002a), with annual quotas for extraction decided on unspecified basis, and compounded with inadequately supervised extraction operations (Kairo, 2001; Ferguson, 1993). Peri-urban mangroves in particular, are under disturbance due to over-harvesting for domestic firewood and industrial energy, human encroachment for housing and pollution (Gang and Agatsiva, 1990; Munga *et al.*, 1993; Rees *et al.*, 1996; Mwaguni and Munga, 1997; Taylor *et al.*, 2003).

Mombasa, a peri-urban island area, is bounded by two main creeks namely Tudor and Port Reitz, has a population of 917,864, an average population density of 3,990 persons per km², and an annual growth rate of 2.5% (Kenya Bureau of Statistics, Coast Office – pers. comm. Mohamed). Kilindini Harbour, East Africa's principal modern deep-water seaport, is located at the entrance of the Port Reitz creek. In the period 1983-93, the port of Mombasa and its adjacent waters experienced 5 tanker accidents spilling a total of 391,680 tonnes of oil (Abuodha and Kairo, 2001; Taylor *et al.*, 2003). A major spill in 1988 destroyed 10 ha of mangroves in Makupa (Abuodha and Kairo, 2001; FAO, 2005), and in 2005, 200 tons of crude oil were spilled, affecting 234 ha of mangroves in Port Reitz creek (Kairo *et al.*, 2005). In addition, the Mombasa municipal waste contributes about 4,369 ton/year of BOD, 3,964 ton/year of suspended solids, 622 ton/year of nitrates and 94 ton/year of phosphates into the creeks in the form of raw sewage (Mwaguni and Munga, 1997; Mwangi *et al.*, 1999). This is in addition to coliform and *Escherichia coli* levels of 1800+ per 100 ml and up to 550 cfu per 100 ml respectively (Mwaguni and Munga, 1997; Mwangi *et al.*, 1999).

Little information exists on the status of peri-urban mangroves globally, and few studies have been conducted on the structure and regeneration status of peri-urban mangroves. Previous mangrove forest structural studies in Kenya were conducted on mangroves of Kiunga, (Kairo *et al*., 2002a), Tana River (Bundotich, 2007), Mida (Kairo *et al*., 2002a); and Gazi Bay (Kairo, 2001). Despite the importance of the peri-urban mangroves of Mombasa, the forest structure and regeneration status is unknown. Knowledge on vegetation structure and regeneration potential of a specific forest is a prerequisite to designing forestry directives like annual allowable cuts and designating specific harvesting areas (FAO, 1994). Ultimately, structural assessment will contribute to the development of sustainable forest management plans (Holdridge *et al*., 1971; FAO, 1994). This study was designed to assess the structural conditions of mangrove forests in Tudor creek and to compare results with published data on other Kenyan mangrove forests in a different setting.

3.2 Materials and methods

3.2.1 Study area

Tudor creek (Figure 3.1) bounds Mombasa Island on the northwest and extends some 10 km inland. The creek has two main seasonal rivers, Kombeni and Tsalu, draining an area of 55,000 ha (45,000 ha and 10,000 ha respectively) with average freshwater discharge estimated at 0.9 m^3s^{-1} during the inter-monsoon long rains (cited in Nguli, 2006). It has a single narrow sinuous inlet with a mean depth of 20m, that broadens out further inland to a central relatively shallow basin (5m) fringed by a well developed mangrove forest mainly composed of *R. mucronata* (Rhizophoraceae), *A. marina* (Acanthaceae) and *S. alba* Sm. (Sonneratiaceae). The basin has an area of 637 ha at low water spring and 2,235 ha at high water spring. Mangrove forests occupy 1,465 ha of the creek. The floristic composition of mangroves of Tudor creek has been described by SPEK (1992). There is no obvious zonation displayed by the dominant mangrove species in Tudor creek. *A. marina* and *L. racemosa* occupy the landward zone, whereas mostly a *C. tagal* and *R. mucronata* mosaic covers the middle zone. Wherever present, *S. alba* occupies the seaward margin, but is replaced by tall *A. marina* and *R. mucronata* along small creeks.

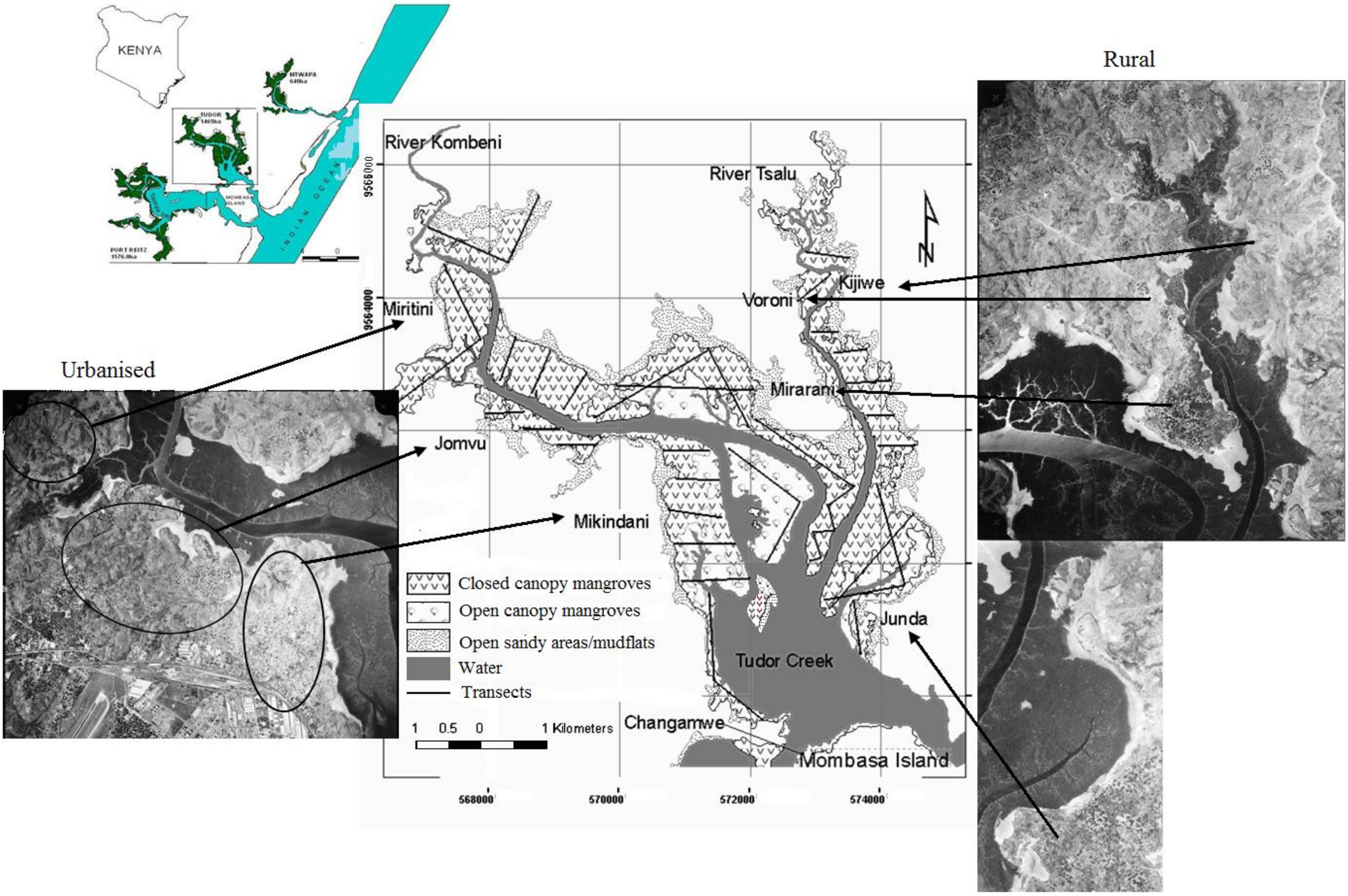

Figure 3.1: Map of Mombasa showing the study area (Tudor creek) and the sampled transects (Source, DRSRS aerial photographs 1992).

The forest resembles the fringing mangroves described by Lugo and Snedaker (1974), with strong inward tidal current during the flood tides which reverses during ebb tides, attaining maximum tide velocities of 0.6-0.7 ms^{-1} (Nguli, 2006), and the dense, well-developed prop roots that accumulate large stocks of debris, with a spring tidal range of 3.5 m and a neap tidal range of 1.1 – 1.3 m.

The mangroves of Tudor creek are separated naturally by two main tidal creeks, Kombeni and Tsalu, 4.5 and 3 km long respectively cutting through the mangroves connecting to the upstream rivers (Figure 3.1). In the framework of this study these two tidal creeks were sampled separately and compared. Tsalu, the eastern tidal creek, includes mangroves near the rural villages of Mirarani, Voroni, Junda and Kijiwe.

Kombeni, the western tidal creek, stretches between the townships of Mikindani, Jomvu and Miritini. The creek is bordered by a steep cliff overlooking a tidal flat that extends to the mudflats occupied by the vast mangrove forest (Figure 3.2).

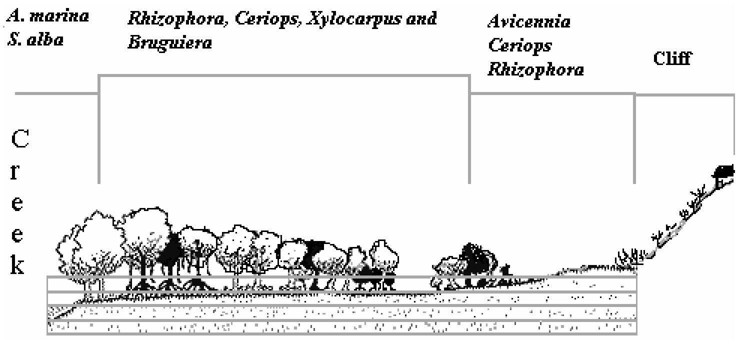

Figure 3.2: Figure illustrating the location of mangroves with respect to the cliff, where settlements and farms are being established

3.2.2 Forest structure and regeneration

A stratified sampling technique was used to sample mangroves of Tudor creek. The location of transect lines were determined by an initial reconnaissance and examination of medium-scale (1:25000) panchromatic aerial photographs of the area. Belt transects of 10 m width were established both perpendicular and parallel to the creek across the entire forest in such a way that they represented as well as possible the general mangrove formation of Tudor creek (Figure 3.1). Vegetation sampling was carried out within 100 m^2 quadrats, established along the transects. A total of 106 and 124 plots were sampled in Kombeni and Tsalu respectively.

Within each quadrat, individual trees with diameter greater than 2.5 cm were identified, counted and position marked. Vegetation measurements included tree height and stem diameter at 130 cm aboveground (D_{130} *sensu* Brokaw and Thompson, 2000, exceptions to this rule are described below). These measurements were used to derive tree basal area, stand density and frequency (Mueller-Dombois, and Ellenberg, 1974; Cintrón and Schaeffer-Novelli, 1984). The ecological importance of each species was calculated by summing its relative density, relative frequency and relative dominance (Cintrón and Schaeffer-Novelli, 1984). The complexity indices (I_c) of the forests were obtained as the product of number of species (s), basal area (m^2/ha) (b), maximum tree height (in meters) (h) and number of stems ha^{-1} (d) x 10^{-5} (Holdridge *et al.*, 1971).

Stems with diameters below 2.5 cm were considered in the category "juveniles" (Kairo *et al*., 2002a, b). Stumps were also counted in each quadrat as an indicator of anthropogenic pressure and tree mortalities.

Tree heights were measured in meters using a Suunto™ hypsometer, whereas stem diameter was measured in centimetres using a forest calliper. For *Rhizophora*, stem diameters were measured 30 cm above the highest prop roots, whereas for *Avicennia*, when the stem forked below 130 cm, individual 'branches' in a clump were treated as separate stems. Pole quality was assessed based on the form of the lead stem and assigned either form 1, 2 or 3. Form 1 stems denote those whose lead stem is straight and therefore excellent for construction but form 2 stems need slight modification to be used for construction. Poles which are unsuitable for construction were assigned form 3 (Kairo, 2001).

Information on the composition and pattern of natural regeneration was obtained using the method of Linear Regeneration Sampling (Sukardjo, 1987). In 5 x 5 m^2 subplots (within the main 10 x 10 m^2 quadrats), occurrence of juveniles of different species was recorded and grouped according to height classes. Seedlings less than 40 cm in height were classified as regeneration class I (RCI). Saplings between 40 and 150 cm height were classified as RCII, while RCIII was for all small trees with heights greater than 1.5 m but less than 3 m. The analysis of spatial pattern of adults and juveniles in the field was carried out inside 10 x 10 m^2 plots along transects. The measure of dispersion used was Morisita's dispersion Index (Morisita, 1959), the application of which is described in Greig-Smith (1983). Morisita's Index (I_δ) is:

$$I_\delta = q\sum_{i=1}^{q} \frac{n(n-1)}{N(N-1)}$$

Where, q is the number of quadrats, n_i is the number of individuals per species in the i^{th} plot, and N is the total number of individuals in all q quadrats. If $I_\delta > 1$, the population is clustered, if $I_\delta = 1$, the population is randomly dispersed and if $I_\delta < 1$, the population is evenly dispersed. Socio-ecological information was obtained primarily from informal interviews conducted with mangrove harvesters, fishermen and Government Forestry Officers.

3.2.3 Data treatment

All data analysis and graphical presentation were obtained with the STATISTICA 8.0 program. One-way ANOVA was performed on stocking densities of different size classes, which we assume as a measure of age. The relative density, dominance and frequency were estimated and the importance values established according to Dahdouh-Guebas and Koedam (2006a).

Stand densities were harmonized using De Liocourt's negative exponential model (Clutter *et al*., 1983). According to the model, the ratio between the numbers of trees in successive diameter classes of uneven-aged stand is roughly constant for a particular forest, but varies from one forest to another. This has been confirmed in a number of uneven-aged forests throughout the world (see e.g. Clutter *et al*., 1983). De Liocourt's model applies particularly in mixed forests where the size classes and recruitment by natural regeneration are continuous. Supposing this constant is q, then the number of trees in successive diameter classes will be represented by a descending geometric sequence of the form aq^n, aq^{n-1}, aq^{n-2}, aq^{n-3},.......aq^3, aq^2, aq^1, a, where a is the number of trees in the largest size class of interest and n is the number of classes.

For such a geometric series, if the logarithm of the frequency in successive classes is plotted against size class, the distribution can be represented as an exponential curve of the form:

$$y = ke^{-ax}$$

where; y is the number of trees in diameter class x; e is the base of natural log (2.718) while k and a are constants. The constants k and a in the equation above vary between forests and with site. The constant k reflects the occurrence of seedling regeneration and tends to be large in forests containing prolific seed-bearing tree species while a determines the relative frequencies of successive diameter classes. A high a is associated with high mortality between classes and is likely to occur in stands comprising light demanding (shade intolerant) tree species (Kairo *et al*., 2002a).

The nature of the future forest was derived from the present forest by fitting exponential models to the size-class structures and comparing the results at a 0.05 significant level. Each class interval was considered to be independent and thus included as within-factor repeated measure variable during the analysis. Regeneration densities and ratios were calculated and the correlation between regeneration densities with canopy gaps analysed using Spearman correlation.

3.3 Results

3.3.1 Floristic composition

The structural characteristics of the mangroves of Tudor creek are given in Table 3.1. Six mangrove species were encountered during the survey. Based on the species' importance values, *R. mucronata* and *A. marina* were the principal species in both sites. *X. granatum* was encountered in Tsalu only, while *L. racemosa* was not encountered despite earlier reports of its occurrence in the landward fringe (SPEK, 1992). *S. alba* was observed to be infested by an unknown insects and/or pathogen. Relative dominance, density, frequency and importance values of these species are shown in Table 3.1. The variation in complexity index between Kombeni and Tsalu is evident, with Kombeni recording a higher index than Tsalu

Table 3.1: Structural attributes of the mangroves of Tudor creek, Kenya

	Species	Height (m)	BA (m^2ha^{-1})	Density (ha^{-1})	Relative Den.	Relative Dom.	Relative Fr.	I.V	Rank	CI
Kombeni	*A. marina*	7.98 ± 2.94	7.07	143.40	11.34	43.43	24.53	79.30	2	4.38
	B. gymnorrhiza	4.09 ± 0.86	0.15	16.98	1.34	0.91	5.66	7.91	5	
	C. tagal	2.62 ± 1.05	0.16	54.72	4.33	1.00	10.38	15.70	4	
	R. mucronata	3.61 ± 2.10	5.91	904.72	71.57	36.31	62.26	170.14	1	
	S. alba	5.22 ± 2.41	2.99	144.34	11.42	18.36	12.26	42.04	3	
Tsalu	*A. marina*	6.23 ± 3.30	5.89	167.74	14.60	60.69	25.00	100.29	2	2.60
	B. gymnorrhiza	3.82 ± 1.72	0.17	29.84	2.60	1.71	8.06	12.37	4	
	C. tagal	2.30 ± 0.55	0.58	175.81	15.30	5.99	29.84	51.13	3	
	R. mucronata	3.00 ± 1.06	2.58	750.00	65.26	26.57	83.87	175.70	1	
	S. alba	5.09 ± 2.09	0.39	20.97	1.82	4.00	3.39	9.21	5	
	X. granatum	3.83 ± 0.68	0.10	4.84	0.42	1.05	1.61	3.08	6	

3.3.2 Stocking density

Table 3.2 gives vegetation inventories for Tudor creek mangroves. There were 1,264 mangrove stems ha^{-1} in Kombeni creek, out of which, 71% were *R. mucronata*, 11% *S. alba* and 11% *A. marina*. While there was 1,301 stems ha^{-1} of in Tsalu creek, of which 58% were *R. mucronata*, 14% *C. tagal* and 11% *X. granatum*. Figure 3.3 shows scattergrams of heights against stem diameters. There were significant differences in height ($F_{(1,\ 2,763)} = 86.765$; $p = 0.0001$) and stem diameter ($F_{(1,\ 2,763)} = 36.727$; $p = 0.0001$) between the tidal creeks. 53% and 58% of the trees in both Kombeni and Tsalu respectively, were in the lower diameter size class (below 6.0 cm). *A. marina* was characterised by low densities of tree in the lower size class (2% ≈ 5%; Table 3.2), and high densities of trees in the medium size class (34% ≈ 50 %). The case of *S. alba* was similar to *A. marina*, being that the effect of selective harvesting has resulted in a relative redistribution of stems to mid-size classes. The observed (bars) and predicted (curve) stem size composition is displayed in figure 3.2. There are significant differences between the observed and predicted size distribution in both sites [Kombeni (χ^2 = 210.8989 df = 5 p < 0.0001); Tsalu (χ^2 = 217.7398 df = 5 p < 0.0001)]. The observed stem size distributions in both sites display selective harvesting, with over-harvesting of stems sizes 6-13 cm and 6-20 cm in Kombeni and Tsalu respectively. The general quality of the standing wood quantity in the two locations was dominated by the crooked tree form (Table 3.3). No significant differences were found in cutting intensity between sites (*Mann-Whitney U Test*, Z = -1.26192, p = 0.21), but stump densities were slightly higher in Tsalu (4,359 stumps ha^{-1}) than in Kombeni (4,167 stumps ha^{-1}).

Table 3.2: Stand table for the mangrove forest of Tudor creek. Values in parentheses indicate percentage of the total stem density per class per species, and the totals.

		D_{130} Class in cm						***Density***
Site	**Species**	**6**	**6.1-9**	**9.1-13**	**13.1-20**	**20.1-35**	**35**	*(Stems ha^{-1})*
Kombeni	*A. marina*	2 *(1.32)*	11 *(8)*	15 *(11)*	33 *(23)*	68 *(47)*	14 *(10)*	**143 *(11)***
	B. gymnorrhiza	8 *(44)*	4 *(22)*	2 *(11)*	4 *(22)*	_	_	**17 *(1)***
	C. tagal	36 *(66)*	16 *(29)*	2 *(4)*	1 *(2)*	_	_	**55 *(4)***
	R. mucronata	607 *(67)*	131 *(15)*	69 *(8)*	63 *(7)*	28 *(3)*	7 *(1)*	**905 *(72)***
	S. alba	24 *(16)*	25 *(17)*	35 *(24)*	38 *(26)*	20 *(14)*	4 *(3)*	**144 *(11)***
	Total	**675 *(53)***	**187 *(15)***	**123 *(10)***	**139 *(11)***	**116 *(9)***	**25 *(2)***	**1,264**
Tsalu	*A. marina*	15 *(9)*	20 *(12)*	24 *(14)*	39 *(23)*	60 *(36)*	10 *(6)*	**168 *(13)***
	B. gymnorrhiza	17 *(57)*	9 *(30)*	2 *(5)*	2 *(5)*	1 *(3)*	_	**30 *(2)***
	C. tagal	123 *(70)*	34 *(19)*	15 *(9)*	4 *(2)*	_	_	**176 *(14)***
	R. mucronata	590 *(79)*	113 *(15)*	23 *(3)*	16 *(2)*	9 *(1)*	_	**750 *(58)***
	S. alba	3 *(15)*	6 *(31)*	5 *(23)*	3 *(15)*	3 *(15)*	_	**21 *(2)***
	X. granatum	2 *(1)*	42 *(33)*	16 *(13)*	30 *(23)*	34 *(26)*	5 *(4)*	**128 *(10)***
	Total	**748 *(58)***	**224 *(17)***	**85 *(7)***	**94 *(7)***	**107 *(8)***	**43 *(3)***	**1,301**

Table 3.3: Tree form distributions in Tudor creek showing the densities per ha and percentages (in brackets) composition per species.

Form	1	2	3
A. marina	-	16 *(11.05)*	124 *(88.67)*
B. gymnorrhiza	2 *(9.80)*	7 *(33.33)*	11 *(56.86)*
C. tagal	7 *(8.61)*	42 *(52.15)*	32 *(39.23)*
R. mucronata	24 *(3.42)*	283 *(40.23)*	396 *(56.35)*
S. alba	-	28 *(40.57)*	40 *(59.43)*
Total	33 *(3.30)*	375 *(37.05)*	603 *(59.66)*

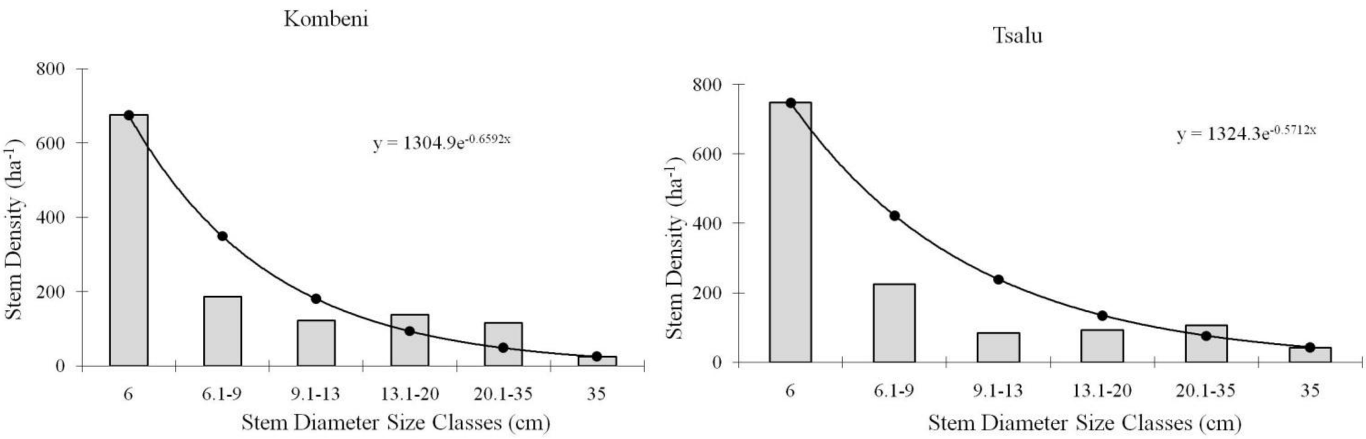

Figure 3.2: Observed (bars) and predicted (curve) size class distribution of mangrove forests of Tudor creek. A high *'k'* value in the stand curve $y = ke^{-ax}$ for Kombeni and Tsalu reflects the occurrence of sporadic natural regeneration in the forest.

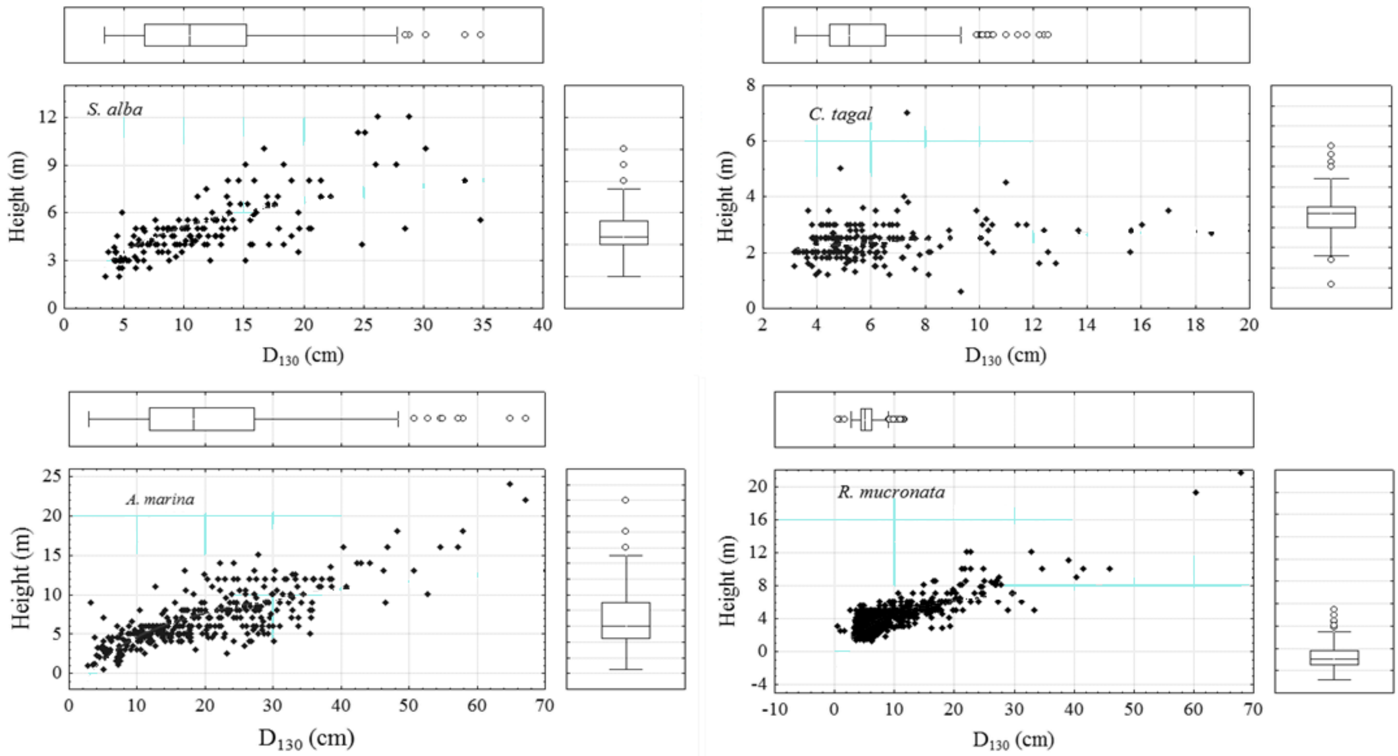

Figure 3.3: Scattergrams of height vs. D_{130} distributions of *R. mucronata*, *A. marina*, *C. tagal* and *S. alba* for Tudor creek mangroves. The box plots display percentile distribution in each case. The extremities of the plot correspond to the maximum and the minimum observations in the data set. The ends of the box are positioned at the 25% and 75% percentiles of the data set.

3.3.3 Regeneration

Significant differences in juvenile densities were observed across the study area. On average the juvenile densities were in the range 21,905 – 33,965 ha^{-1}, with *R. mucronata* representing approximately 45 – 75% of the total juveniles (Table 3.4).

Examination of dispersion pattern showed a tendency towards a random distribution ($I_\delta << 1$) of adult trees, but a clustered distribution for juveniles ($I_\delta > 1$) (Table 3.5). There was a significant negative correlation between open canopy and seedling density per sampled plot (N = 141; Spearman = -0.373086; t (N-2) = -4.74094; p = 0.000005), which together with the proximity to the mother plant and altered environmental gradients in gaps (Clarke, 2004), could have contributed to the clumped distribution of seedlings.

Table 3.4: Juveniles density (saplings ha^{-1}) in Tudor creek. Values in parentheses indicate percentages.

Site	***Species***	***Density ha^{-1}***			***Total Density ha^{-1}***
		RCI 0-40 cm	**RCII** 40.1-150 cm	**RCIII** 150.1-300 cm	
Kombeni	*A. marina*	17,080	37	15	17,132
		(99.70)	(0.22)	(0.09)	(50.46)
	B. gymnorrhiza	60	81	37	177
		(33.90)	(45.76)	(20.90)	(0.52)
	C. tagal	271	411	368	1,051
		(25.78)	(39.11)	(35.01)	(3.10)
	R. mucronata	4,357	4,802	6,432	15,592
		(27.94)	(30.80)	(41.25)	(45.92)
	S. alba	–	–	1	1
					(0.00)
	X. granatum	–	–	–	–
					(0.01)
	Total ha^{-1}	21,768	5,331	6,853	33,953
		(64.11)	(15.70)	(20.18)	
Tsalu	*A. marina*	2,234	78	82	2,395
		(93.28)	(3.26)	(3.42)	(11.09)
	B. gymnorrhiza	318	47	25	390
		(81.54)	(12.05)	(6.41)	(1.81)
	C. tagal	1,251	870	1,158	3,279
		(38.15)	(26.53)	(35.32)	(15.18)
	R. mucronata	3,713	4,066	7,725	15,505
		(23.95)	(26.22)	(49.82)	(71.77)
	S. alba	–	–	–	–
	X. granatum	34	2	–	37
		(91.89)	(5.41)		(0.17)
	Total ha^{-1}	7,550	5,064	8,991	21,605
		(34.95)	(23.44)	(41.62)	

Very few saplings were observed for *S. alba*. The equivalent regeneration ratios i.e. RCI: RCII: RCIII were; 4:1:1 for Kombeni and 1:1:2 for Tsalu. The regeneration ratios are not within the range of effective stocking rate 6:3:1 for saplings described by Chong (1988). However, Tudor creek mangroves can still be considered potentially of good regeneration capacity taking into account the seedling densities.

Table 3.5: Morisita Index in relation to distance from the creek for juvenile *R. mucronata* and *A. marina*. The distribution shows a tendency towards clustering. There is a distinct zonation pattern, with *A. marina* in the landward and seaward sides and *R. mucronata* in the mud flats.

			Distance from creek (m)				
	Site	**Species**	**20**	**50**	**150**	**300**	**700**
Kombeni	Mikindani	*A. marina*	0.0005	-	-	-	-
		R. mucronata	**1.25**	0.52	0.45	-	-
	Jomvu	*A. marina*	**17.4**	-	-	-	-
		R. mucronata	0.01	0.07	0.07	-	-
Tsalu	Junda	*A. marina*	0.066	-	-	-	-
		R. mucronata	**3.32**	**6.41**	**1.62**	0.017	0.016
	Mirarani	*A. marina*	0.0001	**2.60**	-	0.0028	-
		R. mucronata	0.1126	**8.09**	**2.18**	0.056	0.003

3.4 Discussion

3.4.1 Vegetation structure

One notable change includes a decline or loss in *L. racemosa* caused by an observed silted landward fringe. Anthropogenic influences such as indiscriminate and unregulated harvesting, raw domestic sewage pollution and enhanced siltation have had cumulative effects on the structure and regeneration of the forest. As a consequence the forest is characterised by high density of stumps and a dominant crooked tree form. However, the impacts of raw domestic sewage cannot be proven from current observations. Although studies generally indicate sewage enhances mangrove growth rates (Feller *et al*., 2003b; Boonsong *et al*., 2003), with no apparent negative effects (Wong *et al*., 1997; Vaiphasa *et al*., 2007). The complexity index, an indicator of the overall structural development of a forest stand, was low compared to observations along the Kenyan coast (Kairo, 2001; Kairo *et al*., 2002a, b; Bosire *et al*., 2003; Bundotich, 2007). This is an indication of a dominant young vegetation, with low basal area and height.

The stand density, as estimated is lower for large trees, which is to be expected, and diminished for the stem diameter range 6 – 13 cm and 6 – 20 cm for Kombeni and Tsalu respectively. Possibly an implication of higher harvests in a rural setting (Figure 3.2). Theoretically, in an uneven-aged forest there is a normal series of age-gradations, depicted by the reversed J curve [De Liocourt's negative exponential model (Clutter *et al*., 1983)]. This relation in size classes is not observed for Tudor creek, and when put into size-frequency diagrams it is not possible to obtain a simple correlation between size classes and densities (Figure 3.3). It indicates a forest disturbance regime according to direct needs by the people, lacking a consistent harvesting plan, resulting in a haphazard spatial distribution of different size classes, with a highly selective graphical frequency distribution. Assuming that tree size express age, we can use the density curves obtained in this study (Figure 3.2) to predict the composition of the future managed forest. This can be done by harmonizing the irregularities in the stem size distribution by harvesting 'excess' trees in those size-classes observed densities are higher than expected. In the long run, the introduction of multiple uses of mangrove wood will reduce stem density per class.

3.4.2 Regeneration

Natural regeneration was observed all over the creek, with *R. mucronata* seedlings and saplings dominating, while *A. marina* seedlings were abundant, with low density of saplings, implying high mortality of seedlings and/or saplings resulting in low recruitments into successive regeneration classes. *C. tagal*, *B. gymnorrhiza* and *X granatum* had very low regeneration levels, with *S. alba* having particularly low regeneration, with the adults visibly impacted by unknown insects and/or pathogens, and in some areas suffering die backs, an observation also reported in 1992 (SPEK, 1992). Observations indicate regeneration based on the "direct replacement" model, with species replaced by members of the same species as reflected by stand composition. However, establishment and survival is diminished due to site spatial and temporal heterogeneity introduced by canopy gap formation and siltation (Flower and Imbert, 2006).

In this study, we observed high abundance of juveniles in smaller gaps and under canopies than in larger gaps. Clarke and Kerrigan (2000) and Minchinton (2001) reported the impacts of canopy gap size on regeneration. They cited an extremely local dispersal of propagules, coupled with habitat heterogeneity due to disturbance and altered topography as the causes of altered regeneration in large canopy gaps. In Tudor creek, harvesting has enlarged canopy gaps, while siltation has altered edaphic conditions, causing habitat heterogeneity and altered topography, impacting on propagule dispersal, establishment, survival and growth. This scenario is unsustainable, implying inadequate regeneration, altered forest growth and longer times for canopy gap closure as a result of limiting conditions within large canopy gaps (Clarke and Kerrigan, 2000; Clarke, 2004). Dominance of *R. mucronata* can be linked to the larger propagule mass, which enables it to withstand siltation, compared to the smaller seeds of the other species (personal observation). In addition, enhanced predation of dispersed propagules confers advantages to propagules of species that are local canopy members, limiting the range of species available to fill the gaps (Osborne and Smith III, 1990; Clarke and Kerrigan, 2002; Bosire *et al*., 2005b).

3.4.3 Implications

Forest canopy gaps are common in mangroves and usually result after disturbances such as selective harvesting and natural mortality of trees (Duke, 2001). These canopy gaps drive the gap phase regenerative cycles in mangrove forests (Clarke and Kerrigan, 2000; Duke, 2001; Imai *et al*., 2006; López-hoffman *et al*., 2007). The gap creation frequency in Tudor creek is high, stimulating regeneration that approximates selection forest working, favouring *R. mucronata* (FAO, 1994; Ewel *et al*., 1998; Clarke and Kerrigan, 2000; Duke, 2001). However, regenerative turnover might have overwhelmed progress in stand development, causing forest growth reversal resulting in a relatively young forest (Duke, 2001), as older trees have either been selectively harvested or impacted by silt. Siltation, though not systematically studied, causes mortality of both adults and juveniles, through inhibition of gaseous exchange of roots causing root damage, oxygen deficiency and eventually reduced vigour (Ellison, 1998; Thampanya *et al*., 2002).

Gaps are generally characterised by increased light and temperature, high soil evaporation rates with high transpiration rates from trees surrounding the gaps, and high pore-water salinities (Ewel *et al*., 1998; Duke, 2001; Alongi and de Carvalho, 2008). They alter propagule predation by crabs (Osborne and Smith III, 1990; Clarke and Kerrigan, 2002; Bosire *et al*., 2003; Bosire *et al*., 2005b), reduce proportion of reproductive trees (‘propagule limitation’) and diminish soil stability (Ewel *et al*., 1998; Clarke and Kerrigan, 2000; Kairo, 2001). It has been observed that natural gap formation induces less severe physical and chemical changes than gaps formed by human disturbances (Clarke and Kerrigan, 2000; Duke, 2001), resulting in lowered regeneration and development rates under human disturbances (Allen *et al*., 2001; Duke, 2001). In our observation, larger gaps are likely to have more diverse physico-chemical gradients, especially in Mombasa, characterised with a pronounced dry season, manifested by dramatic vegetation responses. This is in contrast to high rainfall areas, lacking distinct wet or dry season, with a buffering effect of freshwater input (Ewel *et al*., 1998; Pinzón *et al*., 2003).

Gaps created by selective harvesting of branches recover over long periods (Ellis and Bell, 2004), with the extent of recovery governed in part, by the regenerative capabilities of the damaged mangrove species and the nature and severity of the causative agent (Snedaker *et al*., 1992; Ellis and Bell, 2004). In contrast to species such as *A. marina*, which hold reserve buds in the stem, mature individuals of the Rhizophoraceae (*Rhizophora* and *Ceriops*) hold reserve buds only in the thin terminal twigs and "conditions severe enough to remove or kill all branches possessing viable reserve buds will potentially eliminate *Rhizophora*" (Hutchings and Saenger, 1987). This implies that specific management principles need to be developed while harvesting different species to ensure that the diversity of the forest is maintained and gap recovery is fast enough to minimise gap sizes.

3.5 Conclusion

The mangroves of Tudor creek, though disturbed, are not irreversibly degraded. However, stand densities and basal areas were lower than for Rhizophoraceae dominated forests along the Kenyan coast (Gazi Bay, 8 – 24 m^2ha^{-1}, 1130 – 2571 stems ha^{-1}; Kairo, 2001; Bosire *et al*., 2003; Mida creek and Ngomeni, 24.05 - 46.97 m^2ha^{-1}, 2075 - 2142 stems ha^{-1}: Kairo *et al*., 2002a, b; Bundotich, 2007). Globally, the values are in the middle of the reported range for similar forests (322 - 2470 stems ha^{-1}; Lugo and Snedaker, 1973; Paijmans and Roller, 1977; Jimenez *et al*., 1985; Smith III, 1988). Within Mombasa, natural disturbances are either relatively small or rare, leaving anthropogenic disturbances the principal threat. Current and future management therefore must focus on regulating the anthropogenic element. Harvesting should be regulated through forest zoning for multiple uses, coupled with a harvesting regime that incorporates replanting and closed periods, ensuring adequate regeneration and forest growth in the long run. This will regulate canopy gap sizes, as intermediate level of gap creation may be optimal for long-term stand stability (Duke, 2001).

To add value to peri-urban mangroves, management for multiple uses as opposed to single (forestry) products is desirable (Rönnbäck, 1999; Barbier, 2000; Ellison, 2008a; Nagelkerken *et al*., 2008). Thus, in addition to meeting the wood demands of local populations, 'environmental' forests as no-cut zones should be established as protection of habitats for migratory birds and other fauna. Introduction of eco-tourism, coupled with an integrated land use plan may regulate pollution and siltation and involve local communities in management. Coupled with an in-depth understanding of the early drivers in mangrove establishment (Krauss *et al*., 2008), growth and productivity of mature forests (Komiyama *et al*., 2008), factors driving mangrove dispersion (Di Nitto *et al*., 2008; Triest, 2008) and spatio-temporal dynamics of mangrove vegetation (Berger *et al*., 2008) this will help in starting and sustaining recovery programmes (Bosire *et al*., 2008). This will serve to boost the ecosystem resilience in the long-run.

4: How sustainable is the utilization of mangrove products in peri-urban Mombasa, Kenya?

A recently cut *A. marina* at Mikindani

Publication

Mohamed, M. O. S., Kairo, J. G., Dahdouh-Guebas, F. and Koedam N. 2008. How Sustainable is the Utilization of Mangrove Products in Peri-Urban Mombasa, Kenya? Ecology and Society, Manuscript #: ES-2008-2617, Version 1.

Summary

This study presents an assessment of a unique exploitation of a peri-urban mangrove. The diversity of respondents, spread over wide areas, some with no apparent links with the ecosystem, coupled with legal and illegal resource exploitation, in the midst of inadequate but punitive enforcement meted on impoverished communities, pose significant challenges in conducting questionnaire surveys. Potentially limiting if not inhibiting information flow. Mangrove wood is exploited at both subsistence and commercial scales, firewood being the most ubiquitous form of use at both urban and rural settings. Forest assessments indicate the lack of preferred or specific harvesting sites. Though harvesting location is mostly dependent on profession. *R. mucronata* is the most harvested mangrove species. The study shows that resource exploitation is intense and "efficient" in an urban setting due to the economic drive and high demand emanating from a high human population. This has given rise to practices that mitigate against traditional norms, promoting destructive unsustainable harvesting. The outcome is the gross under-valuing of ecosystem goods and services in addition to degrading the ecosystem. The management of mangroves for wood extraction in urban areas may not be a viable and/or sustainable option, as it conflicts with functions of mangrove ecosystems, otherwise important in 'resource limited' urban environment. It is recommended that a participatory and *adaptive management*, considering multiple uses and users is the viable way to manage peri-urban mangroves. this will ensure social and ecological resilience in the long-run. However, this may require specific legislative, education and institutional interventions. Integrating local ecological knowledge, may further promote a participatory approach and expedite the formulation of sustainable management plans for peri-urban mangroves.

Keywords: firewood, utilisation, exploitation, mangrove, harvest, peri-urban, sustainable

4.1 Introduction

Mangrove wetlands are multiple use systems that provide protective, productive and economic benefits to human communities worldwide (Dahdouh-Guebas *et al*., 2000; Upadhyay *et al*., 2002; Walters *et al*., 2008). People living in and adjacent to mangrove forests in the tropics exploit these forests for construction materials, firewood and other non-timber products such as fish, shrimps and crabs (Dahdouh-Guebas *et al*., 2000; Walters, 2005a, b; Duke *et al*., 2007; Walters *et al*., 2008). Ecologically, mangroves are spawning grounds for fish, as well as feeding habitats and runways of numerous migratory birds (Nagelkerken *et al*., 2000; Ellison, 2008a; Nagelkerken *et al*., 2008). They also provide habitats for several vertebrates and invertebrates (Farnsworth and Ellison, 1996; Kimani *et al*., 1996; Saintilan *et al*., 2007; Mirera *et al*., 2007; Nagelkerken *et al*., 2008). Mangroves also protect lives and property by shielding against natural catastrophes such as storms and tsunamis as observed in the 2004 Asian tsunami. Areas devoid of mangroves along their coastline were the hardest hit by the 2004 tsunami disaster (Kathiresan and Rajendran, 2005a; Dahdouh-Guebas, 2006; Dahdouh-Guebas and Koedam, 2006b). Mangroves also store high amounts of organic carbon (Bouillon *et al*., 2007a).

The total economic value of mangroves based on marketable and non-marketable ecosystem components has been approximated to be US$ 181 billion (Contanza *et al*., 1997; Barbier, 2000). However, this figure should be interpreted with caution since the valuing system is still relatively new and untested (Bosire, 2006). Globally, mangrove cover is fast disappearing at 1 to 2% of its cover per year mainly due to conversion into agricultural fields, fishponds, saltpans and human settlement (FAO, 1994; Farnsworth and Ellison, 1997b; Duke *et al*., 2007). This rate is greater than or equal to declines in adjacent coral reefs or tropical rainforests (Valiela *et al*., 2001; Alongi, 2002; FAO, 2003). Losses are occurring in almost every country that has mangroves, and rates continue to rise more rapidly in developing countries, where more than 90% of the world's mangroves are located. Mangrove losses during the last quarter century range consistently between 35 and 86% (Duke *et al*., 2007). It is projected that we face the prospect of a world deprived of the services offered by the mangrove ecosystems perhaps within the next 100 years (Duke *et al.,* 2007). But mangroves still remain an important source of wood and other products for many coastal communities (Christensen, 1982; Hamilton and Snedaker, 1984; Diop, 1993; FAO, 1994; Dahdouh-Guebas *et al*., 2000; Dahdouh-Guebas *et al*., 2004; Walters, 2005a), in addition to their broader ecosystem and environmental services.

Historical records indicate that in the 1950's, Kenya was exporting to the middle East an average of 35,451 scores of mangrove poles, equivalent to 709,026 poles per year (Rawlins, 1957). Over-exploitation led to a ban on further export of mangroves in 1982 (Abuodha and Kairo, 2001). This was preceded by a ban on charcoal production from mangroves (Luvanda *et al.*, 1997). However, exploitation of mangroves for local use has continued without proper controls (Luvanda *et al*., 1997; Dahdouh-Guebas *et al.*, 2000). It is estimated that, locally, 70% of the wood requirements by the people living adjacent to mangrove forests are provided by mangroves (Wass, 1995). However, benefits derived from mangroves in Kenya is declining as evidenced by absence of good quality poles (Dahdouh-Guebas *et al*., 2000; Kairo *et al*., 2001; Mohamed *et al*., 2008). Other notable changes include: reduced mangrove fishery (Huxham *et al*., 2004); coastal erosion (Abuodha and Kairo, 2001; Dahdouh-Guebas *et al*., 2004) and deterioration of groundwater quality due to salt water intrusion (UNEP-WCMC, 2006). In urban centres such as Mombasa, the impacts of high populations and pollution from raw domestic sewage have never been established.

This study sets out to evaluate the dependencies on and exploitation of mangrove wood products in a peri-urban setting. We evaluate the disparities between an urbanised setting (Kombeni) and a rural setting (Tsalu), located on opposite sides of a common mangrove resource. We assess the main uses, products and the users' perception on the status of mangroves and impacts of domestic sewage pollution. Other factors assessed include frequency of harvesting, preferred harvesting areas and species. We also look at measures to restore or replant mangroves; and the potential benefits and beneficiaries. The combination of this information with field assessment of cutting intensities in the forest will give an overview of the effects of small scale cutting, and guide the development of an adaptive management plan for the forest.

4.2 Materials and methods

4.2.1 Study area

Mombasa fronts the Indian Ocean in Kenya's coast province and is the second biggest city in Kenya. It covers an area of 230 km^2, and is the smallest district in Kenya with a population of 917,864, ninety three percent of which are classified as urban dwellers (GoK, 2005). East Africa's biggest modern port, the Kilindini Harbour, is located within Mombasa. The ocean surrounding Mombasa Island spreads into the interior forming two creeks namely Port Reitz and Tudor creeks, which are fringed by well developed mangrove forests.

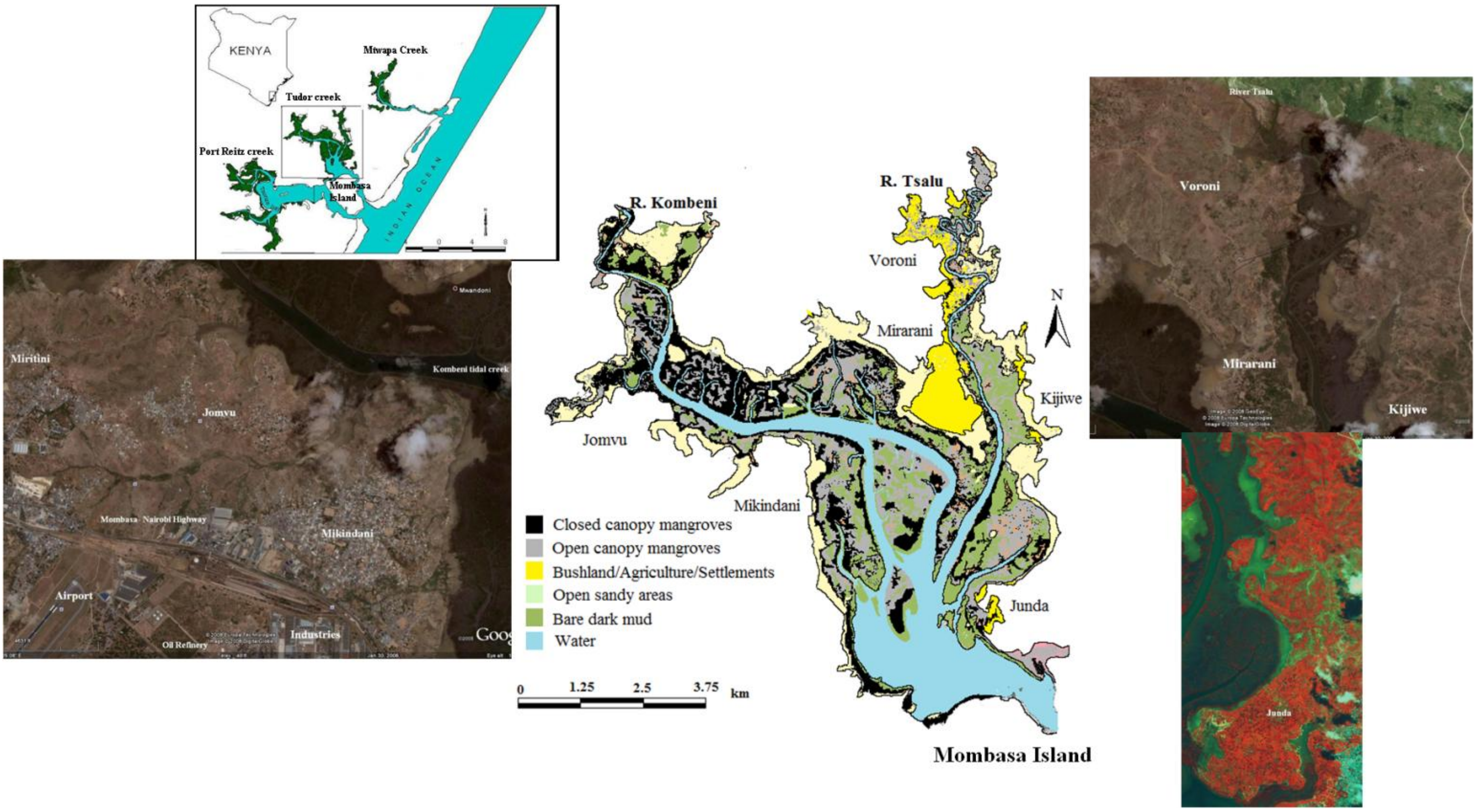

Figure 4.1: Map of Mombasa showing the study area (Tudor creek) in relation to the other creeks Notice the proximity of settlements to the mangroves and the differences in density between the urban Kombeni and Rural Tsalu tidal creeks. The aerial photographs are to scale 1:25,000 km (Source, IKONOS image, 2005).

Tudor creek (Figure 4.1) bounds Mombasa Island on the northwest and extends some 10 km inland. At the entrance into the creek, is Mombasa old Port, situated in the Old Town of Mombasa. It is at this port that mangroves and other items were exported to the Middle East in the nineteenth and early twentieth centuries. Two seasonal rivers, Kombeni and Tsalu flow into the creek. The upper end of the creek is fringed by a well developed mangrove forest mainly of *R. mucronata* and *A. marina*. The creek is narrow and deeper at the entrance but broadens out and becomes shallower further inland. It has a surface area of about 2000 ha at mean sea level, , 80% of which constitutes the wider shallower, upper end (Nguli, 2006).

The mangroves of Tudor creek are naturally separated into two main tidal creeks, Kombeni and Tsalu, 4.5 and 3 km long respectively, cutting through the mangroves connecting to the upstream rivers (Figure 4.1). In the framework of this study, human settlements were classified according to location with respect to the creeks. Tsalu, the eastern tidal creek, includes the rural villages of Mirarani, Voroni, Junda and Kijiwe. These villages are sparsely populated and lack formal infrastructure and basic amenities such as tarmac access roads, piped water, electricity and sewage and solid waste handling facilities. Kombeni, the western tidal creek, includes the cosmopolitan densely populated township of Mikindani and the extensive settlements of Jomvu and Miritini. The major Mombasa – Nairobi Highway passes through the area. The area is also designated as an industrial area, with numerous warehouses that offer storage and support services to the Kilindini Harbour, the biggest sea port in East Africa. This area also neighbours the Moi International Airport and the Kenya Petroleum and Oil Refinery. Settlements in the urban Kombeni include both formal and informal (slums), while the rural Tsalu included old traditional homogenous tribal villages of either Giriyama or Rabai (Mijikenda[♦]) or Jomvu (Swahili[♥]) communities. Land tenure in Mombasa has been cited as a major problem to the alleviation of poverty and a hindrance to the implementation of an ICZM program (UNEP/FAO/PAP/CDA, 2000). This is mainly because land is owned by a few individuals, who rent out land on condition that only temporary structures can be developed. This has caused the establishment of informal settlements, causing significant environmental degradation.

4.2.2 Questionnaire surveys

Initial reconnaissance surveys were done by visiting households and conducting interviews. A total of 30 households were visited, and the questionnaire was revised according to findings from the these surveys. Based on the finding that majority of

[♦] Indigenous coastal inhabitants of Bantu origin comprising 9 tribes sharing a language that varies slightly from one tribe to another. They are either farmers or fishermen.

[♥] Urbanised community that has mixed with other tribes and arabs. The language is adapted as a national language. Their villages are typically diserted as most have migrated to Mombasa Island.

household respondents lacked the basic knowledge on mangroves or would not discuss mangrove exploitation openly, we opted approach people directly in the field. This was also necessitated by the sensitive nature of the subject of interest, coupled with a high urban population in Mikindani and Jomvu (Kombeni). Interviews were conducted in the national language, Swahili. However, most villages in Tsalu were predominantly Giriyama. Therefore, in consultation with elders in each village, a local interpreter was recruited, mainly for translation from the local Giriyama language, help identify respondents directly involved in mangrove exploitation and instil confidence and trust during the interviews. This was important as mangrove harvesting and transportation is a sensitive issue. Mangrove wood harvesting requires a license, a transport permit and a timber statement from KFS. Illegal harvesters are arrested and fined, in addition to confiscation of the timber. The strategy attempted to cover as much area as possible and picking respondents directly linked to specific mangrove resources, permitting a more focused evaluation of specific resource use issues. Respondents included a minimum of at least five representatives from each village around the creek.

From field observations and information from villages, we estimated approximately 300 individuals were actively involved in wood harvest or fishing within Tudor creek. Majority of these people were illegal exploiters of mangroves or mangrove products. In certain instances individuals declined the interviews, as was the case with Mirarani village (Tsalu), particularly because of recent arrests in relation to illegal mangrove harvesting and charcoal production. Some of the punitive measures adopted by the KFS included demolition of houses built using illegally harvested timber, destruction of charcoal by burning with petrol, coupled with arrests and fines. In these villages, group interviews were arranged through the area government appointed administrative chief. In total, two groups of twenty individuals were interviewed. However, information exchange was restricted due to the chief's presence, and interviews focused on general mangrove issues. Additional information on licensing and management was obtained from the KFS, Mombasa office. Excluding the two group interviews, 90 individuals were interviewed.

Questions asked during the interviews included: knowledge on mangrove species in the forest; the frequency, quantity and location of harvesting; the uses of mangrove wood and the preferred species; impacts of sewage pollution; condition of the forest, changes and probable causes; and replanting efforts by locals. In addition to the questionnaires, personal observations were made on the utilisation patterns, and where possible pictures were taken. Interviews were recorded in the field and later transcribed and the texts analysed. Quantifiable information was coded, entered into a database and subjected to quantitative analysis.

4.2.3 Mangrove cutting assessment

Mangrove wood usually has slow decomposition rates especially as a result of the lignocellulose component (Benner and Hodson, 1985). Stem decomposition rates within moist tropical forests has been reported as rapid but variable ranging from 10 years in Panama (Lang and Knight, 1979) to 43 years in Puerto Rico (Odum and Heald 1972). Cuttings were initially assessed in 2005 from 250 100m^2 plots all over the creek, followed by subsequent assessments in 20 plots in 2006 and 2007. Cutting ages were distinguished by the nature of cut surfaces and condition of bark as described by Hauf *et al*. (2006), with modification to suit field conditions. Estimates were also based on knowledge of experienced local mangrove cutters, who were recruited during sampling. The data presented in this study was sampled in January and February 2008 and focused on harvested sites, which were predominantly open canopy. Age was classified as new cut (one month), recent (at least one year), old (one to three years) and very old (more than three years). Even though decomposition may take longer, we limited age estimation to three years because of our observations over the three years period and a traditional custom of harvesting ‘dead trees’ amongst the communities. However, the maximum age of visible stumps may not exceed 10 years. Table 4.1 summarises the age classification of cuttings. Cuttings were particularly easy to assess especially for larger trees which were distinctly cut using chain saws, while most cuttings conspicuously left a distinct stump, cut almost to the base.

Table 4.1: Age estimation criteria for cuttings based on a three year observation in the field.

Cutting age		Cut surface	Bark
New cut	1 month	Freshly cut, brightly coloured surface, with tannins conspicuously present	Bark is largely intact and still moist
Recent cut	at least 1 year	Cut face has ridges with tannins still present	Bark still intact and still firmly attached, but dry
Old cut	1 to 3 years	Smooth cut surface with no tannins	Bark is separating and bent around cut surface
Very old cut	more than 3 years	Bark has separated from cut surface and breaking. Some had started decaying.	Bark is breaking and peeling off, and at times patchy

To sample vegetation data, the quadrat/census method described by Cintrón and Schaeffer-Novelli (1984) was employed. A total of 41 plots were sampled. Each census plot was 400m^2, with corners and boundaries marked using a 100m tape measure. The plot size was selected because of the low density of stems in harvested sites. A stratified sampling approach was used to select representative harvested sites, spread over a wide area attempting to capture variations due to site-specific differences and human influence. Within each quadrat, individual trees with stem diameter greater than 2.5 cm were identified, counted and position marked. Vegetation measurements included tree height and DBH at 130 cm aboveground (D_{130} *sensu* Brokaw and Thompson, 2000). Cuttings were counted, species identified, age estimated, and the diameter measured in each quadrat. All quantitative data were analysed statistically using SPSS (version 13) software.

4.3 Results

4.3.1 Knowledge on mangroves

Most respondents had good knowledge of common species such as *R. mucronata, A. marina, S. alba* and *C. tagal,* compared to rare species such as *B. gymnorhiza, X. granatum* and *L. racemosa* (Table 4.2). Rural dwellers (Tsalu) and older individuals were more knowledgeable and would even distinguish differences within a single species. The older respondents even classified *R. mucronata* as male or female based on bark texture. Most of the respondents had a good appreciation of the mangrove forest (table 4.3). However, majority consider mangroves as mixtures of trees or trees of many important uses or a means of livelihood. Very few, the fishermen, appreciate the flora-fauna interactions.

Table 4.2: Summary on knowledge of species by respondents. Values represent percentage of respondents who could identify the species.

	Common				Less common		
	R. mucronata	*A. marina*	*S. alba*	*C. tagal*	*B. gymnorhiza*	*X. granatum*	*L. racemosa*
Kombeni (N = 46)	100	98	80	61	9	9	7
Tsalu (N = 39)	100	95	72	74	21	10	0

Table 4.3: Summary of understanding of the mangrove ecosystem by respondents. Values represent percentage of respondents. The question was assessing mainly the understanding of the mangrove either as mere trees or as an ecosystem, including flora and fauna. Though an open choice in response was also given.

Comment	**% of respondents**
The vegetation, the wood (House building, firewood, etc)	55.29
The ecosystem, the area	11.76
A forested area distinguished by water	2.35
Source of all income from the sea	1.18

4.3.2 Utilization of mangroves

The uses of mangrove products by priority at the household and village levels is summarised in Table 4.4. Firewood and charcoal were identified as the most important uses at both household and village levels under both urban and rural settings. The utilisation of mangrove firewood as domestic fuel was higher than expected (more than 50%; $\chi^2 = 69.75$, df = 1, $p < 0.05$), with a higher preference for firewood than for charcoal ($\chi^2 = 17.89$, df = 1, $p < 0.05$) (Table 4.4). The use of kerosene was low (less than 30%; $\chi^2 = 21.75$, df = 1, $p < 0.05$). Kerosene was used for lighting of lamps in the evening. Charcoal is produced mainly for commercial purposes, but being illegal, details were secretly guarded by locals, and the trade is shrouded in mystery. This is because the KFS destroys any charcoal kiln and/or charcoal by burning with petrol and arresting the owner.

Table 4.4: Summary of the respondents' preferences for mangrove products in Tudor creek by percentage at household and village levels. Respondents were asked rank uses by importance or preference (priority). Statistical tests are based on the *Chi- square* comparisons between sites.

		1st Priority Firewood	*2nd Priority* Charcoal	*3rd Priority* Construction	*Others* Medicinal
Household	Kombeni*	80	30	15	0
	Tsalu*	72	33	26	10
Village	Kombeni	89	48	41	0
	Tsalu	81	26	26	0

$^{*}p < 0.05$

The national forestry policy identifies firewood and charcoal as major forms of domestic energy in Kenya (GoK, 2007). But the contribution of firewood and charcoal to the country's economy has not been adequately reflected. In addition, the production and transportation of firewood and charcoal is largely unregulated, yet large quantities are consumed annually (GoK, 2007). Use of mangrove poles for construction, also identified as important, has declined in urban Mombasa due to better and more durable alternatives such as coral blocks, cement and steel. However, mangrove poles are still used as scaffolds during construction (SPEK, 1992; Luvanda *et al*., 1997). Mangrove poles used as scaffolds are mainly brought from Lamu due to lack of straight and bigger poles in Mombasa mangrove forests.

Other less common uses for mangrove wood include medicinal, fencing and fishing gear. Foraging and digging for various species of bivalves and gastropods, including the mangrove mud crab, *Scylla serrata*, was also common amongst locals. Perhaps the only non-extractive use of mangroves in this area was the designation of parts of the forest as shrines by some indigenous communities ('kaya'), probably a factor that has contributed to forest conservation for generations. However, this use is rapidly disappearing due to an urban population that is culturally foreign, and does not share common beliefs with the local indigenous communities.

4.3.3 Harvesting patterns

Professions were more diverse in urban Kombeni (businessmen, teachers, drivers, police, fishermen, mangrove cutters, farmers, masons and housewives) than in rural

Tsalu (charcoal burners, mangrove cutters, fishermen, farmers, sand harvesters and housewives). These professions, an indication of alternative means of livelihoods, show limited alternatives in the rural setting. This is because most professions in the rural setting are directly linked to natural resource exploitation. In the urban setting, there is a mixture of both resource exploitation and employment in the formal sector such as law enforcement, education and industry.

These professions influenced harvesting habits, sites and intensity. We coded data on profession and site of harvesting by assigning values♣ and conducting a correlation analysis by the Spearman-R. We observed a significant correlation between profession and site of harvest (Spearman-R = 0.464466, N = 85; t(N-2) = 4.778149; $p < 0.05$). Cutters, fishermen and '*charcoal burners*' harvest deep in the forest or very far away, at times requiring a boat, while housewives, teachers and drivers harvest at the forest edge near home. Reasons for travelling far varied from locating the desirable sizes and forms of trees to avoiding arrests by forest guards. Being peri-urban, traditional rights or village restricted areas for harvesting are no longer recognised, raising a potential conflict between remaining rural villages and urban harvesters.

Harvesting mangrove wood for firewood and charcoal occurs at both subsistence and commercial levels (Figure 4.3). Commercial harvesters are usually licensed. A total of eleven annual licences, two for harvesting mangrove poles and nine for firewood, have been issued in Mombasa. Harvesting is done from Port Reitz, Tudor and Mtwapa creeks (KFS, personal communication) within Mombasa District. Four of these licenses were issued for harvesting in Tudor creek mangroves, though locals around the creek estimate there are slightly more than 50 professional harvesters. Licensed harvesters are allocated quotas ranging between $150m^3$ and $200m^3$ annually. This allocation is based on the evaluation of licence application by a District committee, chaired by the District Commissioner♠.

♣ Each profession was asigned a value between 1 and 10. In cases where a respondent had more than one profession (e.g. fisherman and farmer), an average of the values was used. The harvesting location was also asigned values 1 = forest edge, 2 = mid of forest (walking), 3 = very far (boat). Where more than one site was visited, an average was used.

♠ The official Government Administrator at the District level.

Figure 4.3: Top: (a) women carrying firewood for subsistence use **(b)** un-split commercial firewood for sale at a licensee premises in Jomvu. **Bottom: (c)** A charcoal burning exercise in Tsalu **(d)** and stocking of split mangrove firewood by a dealer in Mombasa (Picture taken in June 2007).

Licensed harvesters never go to the forest, but engage two to three cutters who harvest on demand. A cutter harvests 3 - 5 scores♦ (9 - 15 trees) of firewood per day, earning 50 Kenyan shillings (KShs.) (≈ € 0.50)♣ per score. The harvest is carried from the shore by potters at 30 KShs. (≈ € 0.30) per score. Firewood is then sold at 150 – 170 KShs. (≈ € 1.5 - 1.7) per score, after paying for licence (KShs. 3,000 ≈ € 30), royalties (KShs. 700 ≈ € 7 per m^3) and a transport permit (KShs. 1000 ≈ € 10). Commercial dealers split the firewood billets into smaller pieces depending on diameter. One score can be split to give two scores, which are sold at a retail price of 350 – 370 KShs. (≈ € 3.5 – 3.7) per score, at an average profit of 150 – 170 KShs. (≈ € 1.5 – 1.7) per score. Three firewood dealers estimate average sales of 50 - 60 scores of firewood (≈ 6.9 – 8.3 m^3) per month, but during high seasons (school holidays), 200 scores (≈ 29 m^3 ± 14). Commercial firewood is mainly for sale in Mombasa urban areas and users include the urban dwellers, public schools, a Hindu crematorium, and until recently,

♦ A score is a bundle of 20 pieces of firewood or poles. From field measurements, one score of firewood is equivalent to approximately 0.15 ± 0.07 m^3.

♣ Based on the exchange rate as at 27th March 2008, I euro = 99.241 Kenya Shillings commercial bakeries and brick industries. However, the cost of mangrove firewood is bound to inflate due to revised licence fees and royalties by the KFS at the time of this study. This will hike the retail price of firewood two to threefold, potentially increasing illegal harvesting in the absence of adequate enforcement. Illegally harvested firewood is bound to enjoy market price advantage due to lower costs.

The firewood dealers indicated that they will not opt to sell charcoal. This is because charcoal fetches lower profits than firewood. This means that conversion of mangrove wood to charcoal does not add value to the product, even though it was claimed mangrove charcoal is of superior quality than charcoal from terrestrial trees. On further clarification, the firewood dealers insist that since charcoal is sold by weight, and not quantity like firewood, more quantity of charcoal is required to attain the profit levels of firewood. Charcoal is light in weight compared to firewood. Our observation is that in major community festivals or events such as weddings and funerals, where between 100 and 500 people congregate, the choice of energy source for cooking is mangrove firewood. Preference for mangrove firewood is mainly due to its clean and superior burning properties, burns slowly, strongly and is economical, and has apparently been harvested and used for generations.

Based on estimates of harvesting rates by respondents, harvesting is more intense in rural Tsalu than in urbanised Kombeni (Table 4.5). Estimates indicate significant differences in harvesting of trees and collection of domestic firewood (headload) between the urban and rural settings (Table 4.5). However, the use of canoes in harvesting is popular in both areas. This is because it serves as a good means of avoiding forest guards, who do not have a boat. It is also a convenient way of locating the required tree sizes. Many respondents claimed preference for species while harvesting, but statistical significance was only observed for parts of trees harvested (stems, branches, logs or roots; $\chi^2 = 29.33$, df = 5, $p < 0.05$) (Figure 4.2) and not for species preference ($\chi^2 = 1.33$, df = 5, $p = 0.93$). Although field measurements and respondents indicate *R. mucronata* as the preferred and the most harvested species. *R. mucronata* is also the principal species within the forest (Chapter 3; Mohamed *et al*., 2008). Preference may therefore arise from the wide distribution and ease of access.

Table 4.5: Estimated harvesting rates per week by respondents and the outcome of Mann Whitney U test showing significant differences between sites. One head-load is equivalent to 15 – 20 kg of wood, while one canoe load is equivalent to 30 – 40 poles or 8 – 10 firewood scores.

	Kombeni	*Tsalu*	*Z*	*P*
N	46	39		
Trees	33	332	-2.852	*0.04*
(Vol. (m^3))	(1.22 ± 0.56)	(12.32 ± 5.14)		
Canoe loads	9	5	-0.84	0.933
(Vol. (m^3))	(12.01 ± 5.02)	(6.79 ± 2.79)		
Head-loads	82	160	-3.899	*0.000*
(Wt (Kg))	(1230 – 1640)	(2400 – 3200)		

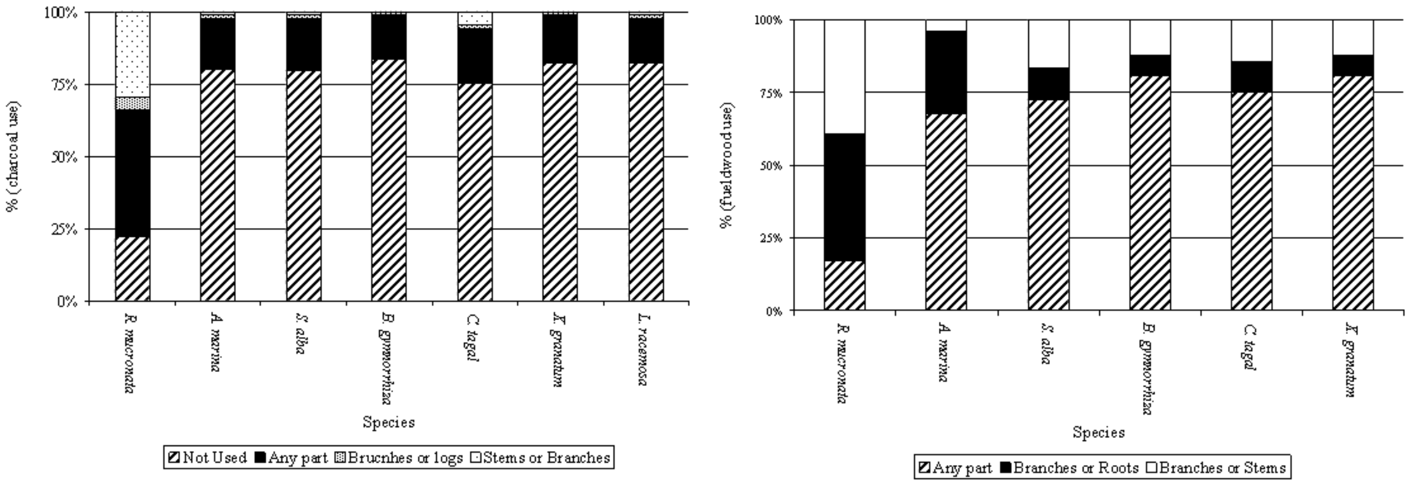

Figure 4.2: Percentage of respondents' utilisation of parts of mangrove species for firewood and charcoal.

4.3.4 Field observations

Field measurements indicate that commercially harvested trees range between 6 – 13 cm diameters (Figure 4.3b). Our field estimates indicate that 3 trees make up one score of commercial firewood (D_{130} 6 – 13cm, average height 5 ± 2m). Firewood for subsistence use at the village is composed of dry logs and branches washed ashore by tides and gathered by women and children, and at times by men while fishing, and there is no preference for a particular species (Figure 4.3a).

Table 4.6: Density per ha, Basal area in m^2/ha for trees and cuttings. Values in brackets represent the percentage of the total basal area.

	Basal area (m²/ha)			
	Kombeni		Tsalu	
Species*	Trees	Cuttings	Trees	Cuttings
A. marina	61	102	16	15
	1.28 (35)	*2.39* (65)	*0.42* (47)	*0.48* (53)
C. tagal	61	108	43	109
	1.74 (61)	*1.09* (39)	*0.22* (23)	*0.74* (77)
R. mucronata	208	256	177	302
	0.42 (18)	*1.93* (82)	*0.65* (31)	*1.43* (69)
Total	330	466^{***}	236	426^{***}
	3.44** (39)	*5.4* (61)	*1.31***(33)	*2.65* (67)

$^{*}p < 0.05$, $^{**}p < 0.05$, $^{***}p = 0.99$

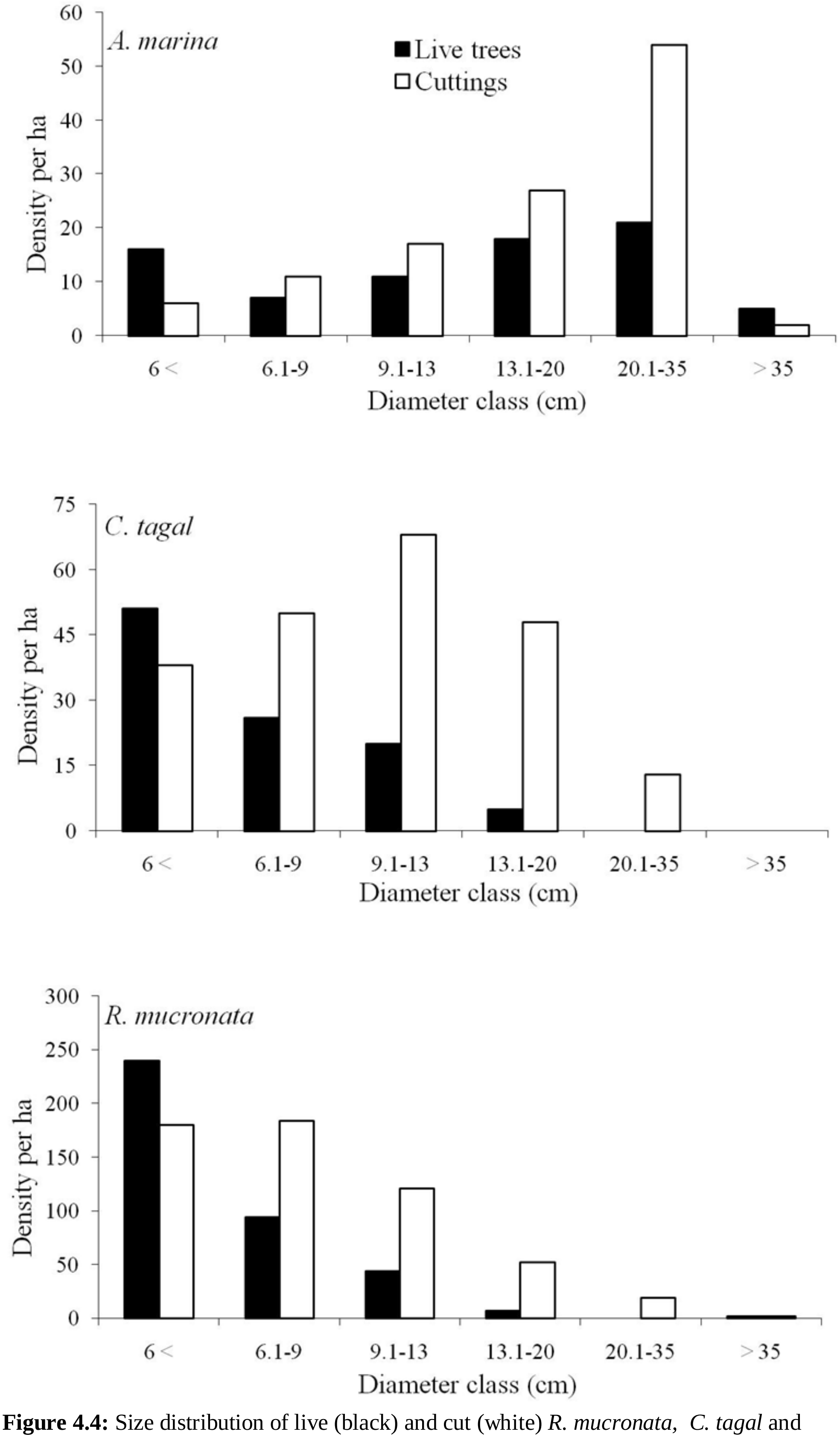

Figure 4.4: Size distribution of live (black) and cut (white) *R. mucronata, C. tagal* and *A. marina* in harvested sites (combined) in Tudor creek.

Live trees and cuttings densities for *R. mucronata*, *C. tagal* and *A. marina* indicate higher densities of cuttings than live trees for all species (Figure 4.4 and Table 4.6). Patterns of stem size selectivity for cuttings differed between *A. marina* and Rhizophoraceae (*R. mucronata* and *C. tagal*). Stem diameters 13 – 35 cm were preferred for *A. marina* compared to 6 – 20 cm for Rhizophoraceae, though harvesting of juveniles and young trees was also common. Selectivity for species and stem sizes is often determined by intended use, and in some if not all cases, presence or distribution of species and stem sizes, Rhizophoraceae being proffered for firewood and charcoal production, while *A. marina* for fencing posts. We observed a narrower range of harvested stem sizes coupled with a conservative harvesting of *A. marina* in Tsalu, compared to Kombeni (Figure 4.4). Probably a reflection of differences in structural complexity and species relative dominance between the two sites, with Kombeni being more structurally complex than Tsalu (Chapter 3; Mohamed *et al.*, 2008).

Field measurements indicate an average of 36 trees per ha were harvested within the past month (New cut), 24 trees per ha (diameter 6.58 ± 2.80 cm) representing *R. mucronata* (Table 4.7). New cuttings averaged 7.7 (± 4.19) cm in diameter compared to 12.84 ± 7.47 cm for older cuttings, indicating diminishing sizes over time for all species. The relatively bigger trees harvested recently is the consequence of more efficient harvesting regimes, targeting large sized *A. marina* mainly in Kombeni (Figure 4.6). Less size specificity for *A. marina* was observed in Tsalu. Trees above 35 cm diameter were scarce in Tsalu, probably exhausted by harvesters who are now turning to the Kombeni side. This may have resulted from charcoal production in Tsalu. As percentage, cuttings basal area represents 61% and 66% of total basal area in Kombeni and Tsalu respectively. This indicates significant contribution of harvesting to canopy gap formation (Table 4.7). However, in all sampled localities, gap sizes surpassed the sampled 20m by 20m plots, raising questions on the causes of the enlarged canopy openings. This is because the harvesting pattern does not involve clear felling.

Consequently harvesting patterns did not reflect the proximity to human settlements and no significant differences in cutting densities between the two locations ($F_{(1, 724)}$ = 0.00028, p = 0.99) (Mohamed *et al.*, 2008). However, significant differences in stem sizes ($F_{(1, 723)} = 82.10$, $p < 0.05$), species ($F_{(2, 721)} = 121.66$, $p < 0.05$), and age ($F_{(4, 720)} = 5.97$, $p < 0.05$) were observed between the two locations (Table 4.6). The basal area for live trees in Tsalu is lower than for Kombeni ($F_{(2, 720)} = 2.89$, $p < 0.05$) and may be associated with harvesting, that has been more intense in the past, leaving a high density of older cuttings (more than three years) (Table 4.6). These field measurements confirms the respondents' estimates of harvest as displayed in table 4.5. This indicates higher harvesting intensities in Tsalu than in Kombeni.

Table 4.7: Density of cuttings/ha in open canopies and the estimated age. Values in brackets represent the mean stem diameter (± standard deviation).

Age	*A. marina*	*C. tagal*	*R. mucronata*	Total
Die back	1	–	–	1
	(16.24 ± 5.40)			(16.24 ± 5.40)
New Cut	1	11	24	36
< 1 month	(20.72 ± 8.53)	(8.67 ± 3.78)	(6.58 ± 2.80)	(7.70 ± 4.19)
Recent cut	20	24	41	148
< 1 year	(20.55 ± 7.98)	(9.85 ± 3.95)	(9.10 ± 4.59)	(12.03 ± 7.21
Old cut	16	39	93	148
1 to 3 Years	(15.47 ± 8.72)	(11.25 ± 4.72)	(7.95 ± 3.57)	(11.32 ± 6.15)
Very old	10	34	126	170
> 3 years	(25.00 ± 6.55)	(10.51 ± 5.38)	(8.97 ± 5.15)	(10.20 ± 6.44)
Total	49	109	284	443
	(19.62 ± 8.53)	(10.44 ± 4.73)	(8.46 ± 4.49)	(10.16 ± 6.21)

4.3.5 Status of the forest and sewage pollution

All respondents agree that the forest is in a state of decline, primarily the outcome of overharvesting, siltation (particularly after the 1997-1998 ENSO event) and domestic sewage pollution (Table 4.8). Respondents were generally concerned on the effects of raw domestic sewage pollution on forest health, the associated fauna and the environment at large. However, 23% of the respondents regard sewage as a resource, which nourishes 'subsistence farms'. This group considered sewage as a 'fertilizer' to the forest, causing nourishment. Particularly during the dry season, locals utilise raw sewage to irrigate small plots of subsistence farms, growing bananas, maize and sugar cane. 64% respondents acknowledge the environmental and health risks associated with domestic sewage pollution and very few (3.53%) were concerned about disease out breaks. This may be a genuine concern as it was recently proven that mangroves do not filter or lower faecal coliform levels in sewage effluents (Yang *et al.*, 2008). Our observations and consequent hypothesis also indicates that filtering of sewage effluent within the Tudor creek mangroves is not efficient as should be, due to enlarged canopy gaps (lowered tree density) as a result of harvesting and silting (PUMPSEA, 2007a). This lowered tree density has been shown to lower filtering efficiency of mangroves (Yang *et al.*, 2008; Wu *et al*., 2008). This may point to the morality of dumping raw domestic sewage into mangroves that are utilized by communities for both wood and food. This raises concerns of disease outbreaks and possible eutrophication in the creek.

Table 4.7: Summary of respondents views on forest status. Respondents were asked to identify changes and their causes. We then asked what indicates the change has occurred. Values are percentage of the respondents.

Cause	*Effect*	*Indicator*	*%*
Over harvesting	Deforestation	Canopy gaps and lack of adult or big trees	51
Sewage Pollution	Decrease in fisheries	Fish kills, decline and mangrove tree mortality	33
El-Niño event of 1997- 1998	Siltation	Mortality of adult trees	26
High human population	Forest decline	low density of trees (more canopy openings)	21
Replanting	Forest regeneration	Domination by smaller trees/young vegetation	13
Heavy annual rains	Die backs	High mortalities of trees	8
Poor farming methods	Siltation	Mortality of both adults and seedlings	2
Climate change	Declining forest	Shrinking forest resources (scarcity)	1

As a result of degradation, replanting has been attempted mostly in the Kombeni side of the creek. However, replanting was cited by some respondents as one of the problems causing degradation. This is because the forest is dominated by young trees (Chapter 3; Mohamed *et al*., 2008). However, an evaluation is not possible due to absence of a monitoring program and well defined objectives for the replanting exercise. We observed that replanting is mostly done in the landward fringe near settlements or villages, where considerable siltation occurs during the rainy season, causing mortality of replanted seedlings and saplings (Figure 4.5). However, the motivation to participate in replanting is economical, even though ownership of the forest seems an issue of conflict. The government is viewed as the owner, but the locals still consider the forest as an important resource for their livelihoods, so they replant with the hope of harvesting in the future, and consider themselves the main beneficiaries.

The level of understanding of the legislation governing forest management for majority of respondents is low ($\chi^2 = 0.576$, df = 1, p = 0.448). However, majority of the respondents felt that the legislation affected them ($\chi^2 = 17.19$, df = 1, $p < 0.05$). This is mainly through arrests, confiscation of harvested wood or timber and in some cases even demolition of houses built using illegally harvested poles. Probably, this has led many respondents to assert that the enforcement of legislation is effective ($\chi^2 = 25.988$, df = 1, $p < 0.05$), despite the poor understanding of the legislation. The effect has been a widening gap between the managers and communities, who treat each other with suspicion or mistrust.

4.4 Discussion

The study presents an overview of the small scale mangrove wood harvesting in a peri-urban setting, whose cumulative effects are shaping the structure of the forest. Mangrove forests in Mombasa constitute the most significant source of wood. Our observations indicate harvesting intensities in Tudor creek are comparable to other studies globally (Walters 2005a, b; Hauf *et al*., 2006; López-Hoffman, *et al*., 2006). As observed in these studies, the impacts of small scale wood harvesting range from shifts in forest size class structure towards substantial reduction in trees above 6 cm D_{130}, a decline in stand basal area and possible species shifts. Walters (2005a, b) presented a quantitative measure on small scale mangrove harvesting in the Philippines. His estimates of cuttings' basal area as a percentage of live trees basal area ranged between 2 and 70%, below the range of values of 156 and 202% observed in this study. A main contrast between Kenya and the Philippines however, is the establishment of mangrove plantations by locals in the later since early 1900s, with well defined benefits and resource ownership (Walters, 2004). In Kenya, the government controls, owns, enacts and enforces any rules regarding the forest and its exploitation. Furthermore, the coastal city of Mombasa has a '*controversial*' land tenure system, with a fair majority of inhabitants designated as squatters (UNEP/FAO/PAP/CDA, 2000).

The size range of harvested trees suggests a general removal of canopy trees, with variable rates on a temporal than spatial scales. Mohamed *et al*. (2008) estimated the stand densities as 1,264 and 1,301 stems ha^{-1} for Kombeni and Tsalu respectively and corresponding 4,167 stumps ha^{-1} and 4,359 stumps ha^{-1}. Based on these estimates and values from the current study, harvesting rates can be estimated to range between 3 – 82% depending on site, higher than estimates made for Kosrae in Micronesia (Hauf *et al*., 2006), Venezuela (López-Hoffman *et al*., 2006) and Timor Leste (Alongi and de Carvalho, 2008). The trend has resulted in a 60-70% loss of trees greater than 6 cm D_{130}, resulting in smaller trees with low basal area and large canopy gaps (López-Hoffman *et al*., 2006; Hauf *et al*., 2006; Alongi and de Carvalho, 2008). However, our study in a peri-urban area shows that the impacts and associated effects are perhaps more bizarre, more spatially and temporally dispersed and potentially more destructive. The human dimension in Mombasa is significant, manifested by high population growth and an escalating demand for natural resources (Centre Bureau of Statistics, Coast Province Offices).

How sustainable are the utilisation practices in Tudor creek? According to the World Commission on Environment and Development (1987), sustainability is defined as "meeting the needs of the present without compromising the ability of future generations to meet their own need". Before answering this question, it is a prerequisite to look at what drives the decline in mangrove health and the underlying causes. Three main causes are usually cited; the lack of choice by users; the will-full overuse or '*cognitive dissonance*' by users (a lack of responsibility and apparent quest for economic gains in the short term); and a misunderstanding by managing based on wrong principles (focusing on unplanned wood exploitation, not considering other functions of mangroves, and lack of local communities' involvement). In Tudor creek, all categories may apply, and more prominently the third category drives the degradation. Bans and restrictions on uses have been instituted in the absence of alternatives, while enforcement is limited, and local communities are seldom involved. Such bans and restrictions through changes in property rights have been shown elsewhere to increase poverty (Reddy and Chakravarty, 1999), promoting illegal and irresponsible harvesting patterns, as well as mitigate against indigenous harvesting regimes.

The urban demand for firewood has created a 'clique' of commercial harvesters, implementing a haphazard harvesting pattern, with technological efficiency. These 'roving bandits' (according to Ellison, 2008a) can persist as they have no connection to local communities and no incentives to harvest sustainably. Given that one score of fuel wood requires 4 trees, which is sold at a retail price of KShs. 350 – 370, and the estimated stand density of between 1200 – 1300 trees per hectare (Chapter 3; Mohamed *et al.*, 2008), the current value of a single mangrove tree can be estimated as KShs. 87.5 – 92.5 (€ 0.9 – 0.95) or KShs. 115,425 ± 2,949.80 per ha (€ 1,163 ± 29.72). These estimates are based on the current mangrove firewood costs. Considering the time for a mangrove tree to attain the desired size (D_{130} = 6 – 13 cm), the habitat function for a diverse fauna (Nagelkerken *et al.*, 2008; Kristenssen *et al.*, 2008), coastal protection and water quality maintenance due to sewage pollution (Ye *et al.*, 2003; Yang *et al.*, 2008; Wu *et al.*, 2008; Walters *et al.*, 2008), exploitation of mangroves for firewood and charcoal grossly undervalues the *use*, *utility* and *importance* values of mangroves, and degrades the ecosystem. The use of mangrove products in this case is best described as "exploitation" or "liquidation" rather than management, sustainable or otherwise (Walters, 2004; Ellison, 2008a).

4.4.1 Restoration

Efforts to 'replant' Tudor creek mangroves demonstrate the appreciation of the values of mangroves and acknowledge the degradation that has taken place. However, efforts are not well informed, lack defined goals or strategy, are uncoordinated and do not consider the causes of degradation or the ecological and socio-economic aspects. Some locals cite replanting, which targets mostly *R. mucronata*, as a cause of degradation. This is because they create mono species stands, and also cause the forest to be dominated by young vegetation. Though we observe that these reasons are farfetched, as replanting is highly localised in the landward side. Replanting in this case is actually a waste of resources and efforts as silting causes significant mortality of the replanted seedlings. Replanting should be an option only when natural recovery is absent (Ong, 1995; Ellison, 2000a, b; Bosire *et al.*, 2008). The stand densities and dominance of a young vegetation in Tudor creek implies that the forest has the capacity to recover spontaneously, giving rise to a higher diversity forest (Ellison, 2000a, b; Mohamed *et al.*, 2008).

Despite replanting efforts, *resource ownership* and *user rights* are a major determinant of participation by locals. A good proportion of communities would not participate in planting campaigns because the forest '*belongs to the Government*', while those who participate, are doing so with the hope of benefits in form of remuneration, or concessions to harvest. The forest is largely viewed as a vital resource for communities, who consider themselves the prime beneficiaries in any replanting effort. The government is viewed as a hindrance to the communities' quest of realising the benefits of the forest resources. However, the levels of exploitation surpass the communities' demands and expectations, due to an enormous urban demand, coupled with a management regime that does not involve local communities.

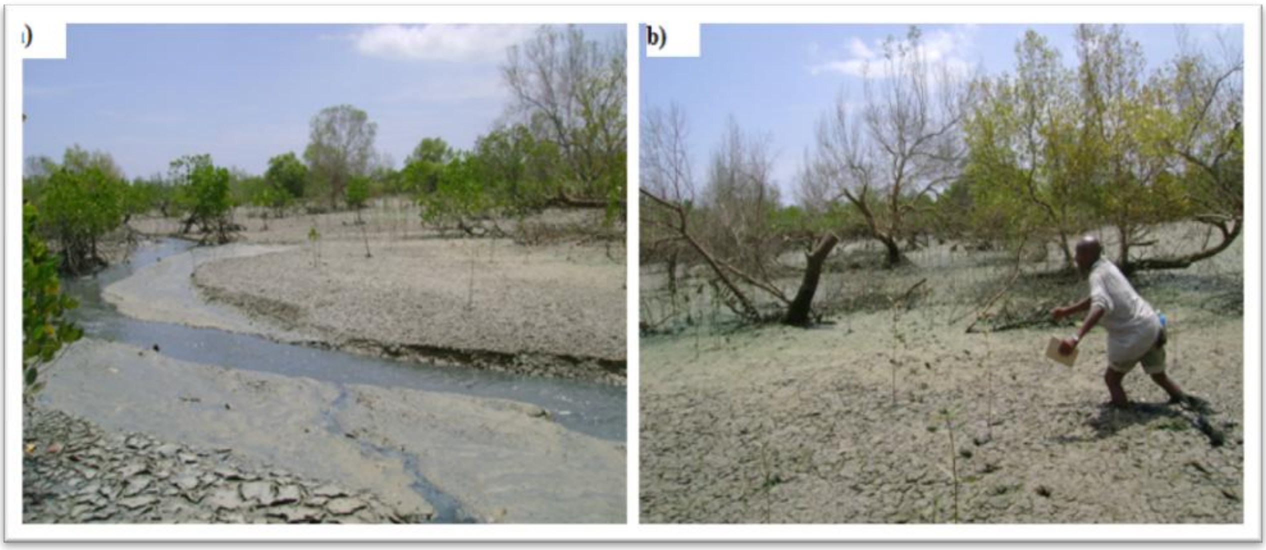

Figure 4.5: Signs of siltation in one of the sampled gaps **(a)** sewage flowing through one of the gaps showing that it is unlikely filtration occurs as a result of enlarged gaps **(b)** a sampling campaign within one of the gaps, notice the *A. marina* trees with pneumatophores covered with sediments, also observe the person walking with legs sinking into the loose sediments (Photos taken in January 2008).

4.4.2 Implications

This study presents a classical example of humans as agents of change in a mangrove forest situated in a peri-urban area. The importance of mangroves as a resource in urban areas is demonstrated, provisioning wood for energy, poles for construction, and marginally, food and servicing water quality in addition to habitat functioning, yet the actual value of these resources still receive little or no recognition from policy makers or mention in the national economic valuation (GoK, 2007). The cumulative impacts of inadequately regulated exploitation and overharvesting imply a diminished mangrove forest and related resources, potentially resulting in significant ecological and economic losses if trends are not reversed. Ecological losses manifest themselves in lowered forest diversity (e.g. loss or reduction in abundance of *L. racemosa*), altered structure and function e.g. enlarged canopy gaps and altered edaphic conditions), and lowered ecosystem stability (lowered regeneration in enlarged gaps), giving rise to a less ecologically resilient system (Chapter 3; Mohamed *et al*., 2008).

Management and valuation of mangroves in Kenya is conspicuously based on a bias towards wood products. There is little or no appreciation of associated services such as faunal habitat and coastal protection. The material poverty of coastal communities, coupled with the local demand for cheap firewood, has resulted in gross overexploitation and under-valuing of mangrove resources. In addition, jurisdictional ambiguity and overlaps have contributed to failures in enforcement of the legislation. The forest is by law managed by the KFS, fisheries is managed by the Fisheries Department, while sewage and waste management is the responsibility of the Municipal Council of Mombasa (MCM). This has resulted in a scenario where 'the right hand does not know what the left hand is doing'. Whereas harvesting licences are issued by the KFS, licences for selling firewood within Mombasa are issued by the MCM. In addition, the dumping of raw domestic sewage into the mangroves is never seen as a management issue. The management of mangroves for wood extraction under an undefined '*silviculture system*' may not be a viable and/or sustainable option in urban areas. This is mainly because such a practice conflicts with the structural traits of mangroves and other functions of the ecosystem. This is especially when there is pollution. The combined effect will include lowered capacity in filtering the sewage effluent, which is influenced by stand density and species composition (Wu *et al*., 2008; Yang *et al*., 2008).

Local ecological knowledge, shown to be critical in natural resource exploitation through complex long-term understandings of the functioning of the local ecosystems, to cultural beliefs and religious views of human environment relations (Walters *et al*., 2008), may be eroded if not lost in peri-urban resource systems. This may occur through intergenerational gaps and barriers and assimilation into contemporary urban cultures. This may potentially evolve into catastrophic utilisation patterns. Our interaction with older community members shows a group with rich and diverse knowledge, experience and understanding of the ecosystem in contrast to the younger generation. This has also been observed in other studies in Kenya (Dahdouh-Guebas *et al*., 2000) and Venezuela (López-Hoffman *et al*., 2006). This local ecological knowledge, '*critically endangered*' in an urban setting, needs to be documented, integrated with scientific research, and can form a valuable tool in understanding and explaining forest changes and dynamics and ultimately, in management planning (Walters *et al*., 2008).

4.5 Conclusion

Management and restoration efforts in Tudor creek must address the human element if they are to succeed at all (Walters, 1997). 'It is as much the human determinants of degradation . . . as the non-human determinants of an ecosystem's structure and functioning that will determine the rate of potential recovery by alternative restoration or rehabilitation pathways at any given site' (Aronson *et al*., 1993). Because human impacts are widespread and intense, successful management depends on (i) curtailment of existing human actions which are to a large extent impeding normal forest rejuvenation and growth; (ii) involving local communities and/or stakeholders in management; and (iii) being an urban area, with an exceptionally high population growth and high demand for resources, alternative means of livelihood and sources of energy to be identified.

Diversifying uses, strict regulations and enforcement and the introduction of non-timber forest products such as nature forests or nature management for eco-tourism may safeguard the long-term survival of peri-urban mangroves and elevate its use and utility values by considering both the flora and fauna in the ecosystem. However, the elusive nature of forest management for multiple uses, a recent development, may pose challenges as little time has elapsed for validation and adoption to address timber and non-timber extraction integration in a peri-urban setting, while experiences from the oldest managed mangroves of Matang and Sundarbans, also indicate that current long-term silviculture management practices involving mangrove trees is not sustainable (Ellison, 2008a; Walters *et al*., 2008) and is characterised by declining yields (Ong, 1995; Walters *et al*., 2008). For success, it may be necessary to train personnel to *adaptively manage* mangroves based on multiple uses, coupled with specific legislative, education and institutional interventions. The documentation and integration of the local ecological knowledge, may further expedite the formulation of sustainable management plans for peri-urban mangroves.

5. Productivity in a peri-urban mangrove: Does sewage exposure have an impact?

Litter trap on *R. mucronata*

Publication

Mohamed O. S. M., Mangion P., Mwangi S., Kairo J. G., Dahdouh-Guebas F., and Koedam N. 2008. Productivity in a peri-urban mangrove: Does sewage exposure have an impact? Submitted to *Hydrobiologia*.

Summary

This study presents insights on the productivity of an under-valued, over-exploited and sewage polluted peri-urban mangrove through litter fall studies on three common mangrove species over a period of two years. These species *R. mucronata*, *A. marina* and *S. alba*. The study site has been exposed to raw sewage for decades, dozed with sewage every tidal cycle, with the loading exponentially reducing with distance from the source. We observed a strong correlation between the leaf C:N ratio and leaf $\delta^{15}N$ signature. Higher C:N ratio for *R. mucronata* corresponding with lower leaf $\delta^{15}N$ (3.88 ± 0.64‰) signature, and lower C:N ration for *A. marina* and *S. alba* (6.48 ± 0.03‰ and 6.76 ± 0.24‰ respectively) corresponding with higher $\delta^{15}N$ signature. This reflects species specific response to raw sewage exposure. This shows that the forest has a more open N cycle, favouring $\delta^{15}N$ accumulation within the system. However, the level of sewage exposure did not appear to impact on litterfall rates. The mean annual litter fall was estimated at 12.16 ± 2.89 t $ha^{-1}yr^{-1}$ for the whole stand. Litter fall, in both content and quantity was highly seasonal, with high rates occurring in the dry North Easterly Monsoon (NEM) season, January-April (ca. 5.10 ± 1.36 g DW m^{-2} day^{-1}) and lower rates in the cool and wet South Easterly Monsoon (SEM) season, June-October (ca. 2.53 ± 0.47 g DW m^{-2} day^{-1}). Productivity varied significantly between species, *R. mucronata* recording the highest annual rate of 15.34 ± 3.34 t $ha^{-1}yr^{-1}$. No significant differences in litter fall was observed between *A. marina* and *S. alba*, (11.44 ± 2.90 and 9.69 ± 5.26 t $ha^{-1}yr^{-1}$ respectively). This study shows that sewage exposure does not necessarily translate into elevated productivity in mangroves, but may alter leaf nitrate content depending on species, possibly altering the decay of litter, affecting nutrient cycling within the system.

Keywords: mangroves, peri-urban, litter fall, sewage, nutrients, $\delta^{15}N$,

5.1 Introduction

Intertidal mangrove forests form some of the most productive tropical communities based on litter fall. They are normally characterized by high productivity, high biomass and litter production compared to terrestrial ecosystems (Twilley *et al.*, 1986; Saenger and Snedaker, 1993; Bouillon *et al.*, 2007b). High rates of primary production and rapid decomposition of the mangrove litter was hypothesized to support a trophic link between the mangrove ecosystem and adjacent habitats, providing an important food and energy source for a variety of aquatic organisms (Odum and Heald, 1972). However, a number of studies using natural tracer techniques (e.g. stable isotopes, fatty acids) have failed to prove such a link (Bouillon *et al*., 2008; Nagelkerken *et al*., 2008). Although faunal communities may have a profound effect on litter dynamics (Lee, 1998), the role of mangrove litter in sustaining epibenthic communities is often fairly limited and decreases when systems are more open with respect to material exchange with adjacent systems (Bouillon *et al.*, 2004a, b). However, litter fall is a commonly measured functional aspect of mangrove forests world-wide. It is used as a proxy measure of net primary production, and dynamically reflects phenological events occurring in the canopy (Twilley, *et al*., 1986; Duke, 1990). Furthermore, litter fall is an important element in the calculation of energy and nutrient fluxes in mangrove ecosystems representing a modest proportion of the carbon fixed by trees (Li, 1997; Wafar *et al.*, 1997; Bouillon *et al.*, 2008).

Although recognized as economically and ecologically important coastal biotopes, resilient to natural catastrophes, mangroves are rapidly disappearing due to anthropogenic impacts such as oil spills, overharvesting due to lumbering, removal for construction of fish and shrimp ponds, pollution and general urban development (Fortes, 1988; Duke *et al*., 2007; Walters *et al*., 2008; Nagelkerken *et al*., 2008). Litter fall studies present a useful means in comparing mangrove communities, the possible export of materials into adjacent subtidal communities, and support of food webs (Mackey and Smail, 1995). The level of litter fall in intertidal forests has been variably attributed to stress (Pool *et al*., 1975), habitat optimization (Saenger and Snedaker, 1993), and tidal flushing (Twilley *et al.*, 1986). The $\delta^{15}N$ leaf signature has been utilised as an indicator of anthropogenic nutrient loading into mangrove marshes and has been recommended as a good biomonitor of N loading (Fry *et al*., 2000; Bouillon *et al*., 2008).

The peri-urban mangroves of Tudor creek, Mombasa, are recipient of raw domestic sewage from the Mikindani Township, though no specific structural attributes have been correlated with sewage exposure (Mohamed *et al*., 2008). This study aims at estimating the annual litter fall levels and trends, hence productivity and phenology of the mangroves at assessing the impacts of sewage exposure, and nutrient uptake through leaf N and $\delta^{15}N$ composition. It is envisaged that this may provide insights into the utility of mangroves in domestic sewage treatment.

5.2 Materials and methods

5.2.1 Study area

The floristic composition of Tudor creek mangrove has been described by Mohamed *et al*. (2008) and SPEK (1992), while the tidal dynamics were described by Nguli (2006). Briefly, Tudor creek has a single narrow sinuous inlet with a mean depth of 20 m, that broadens out further inland to a central relatively shallow basin (5 m) fringed by a well developed mangrove forest mainly composed of *R. mucronata*, *A. marina* and *S. alba*. The basin has an area of 6.37 km² at low water spring and 22.35 km² at high water spring. Mangrove forests occupy 8 km² of the Creek. The forest resembles the fringing mangroves described by Lugo and Snedaker (1974), with strong inward tidal current during the high tides which reverses during ebb tides, attaining maximum tide velocities of 0.6-0.7 ms^{-1} (Nguli, 2006), and the dense, well-developed prop roots that accumulate large stocks of debris, with a spring tidal range of 3.5 m and a neap tidal range of 1.1 – 1.3 m.

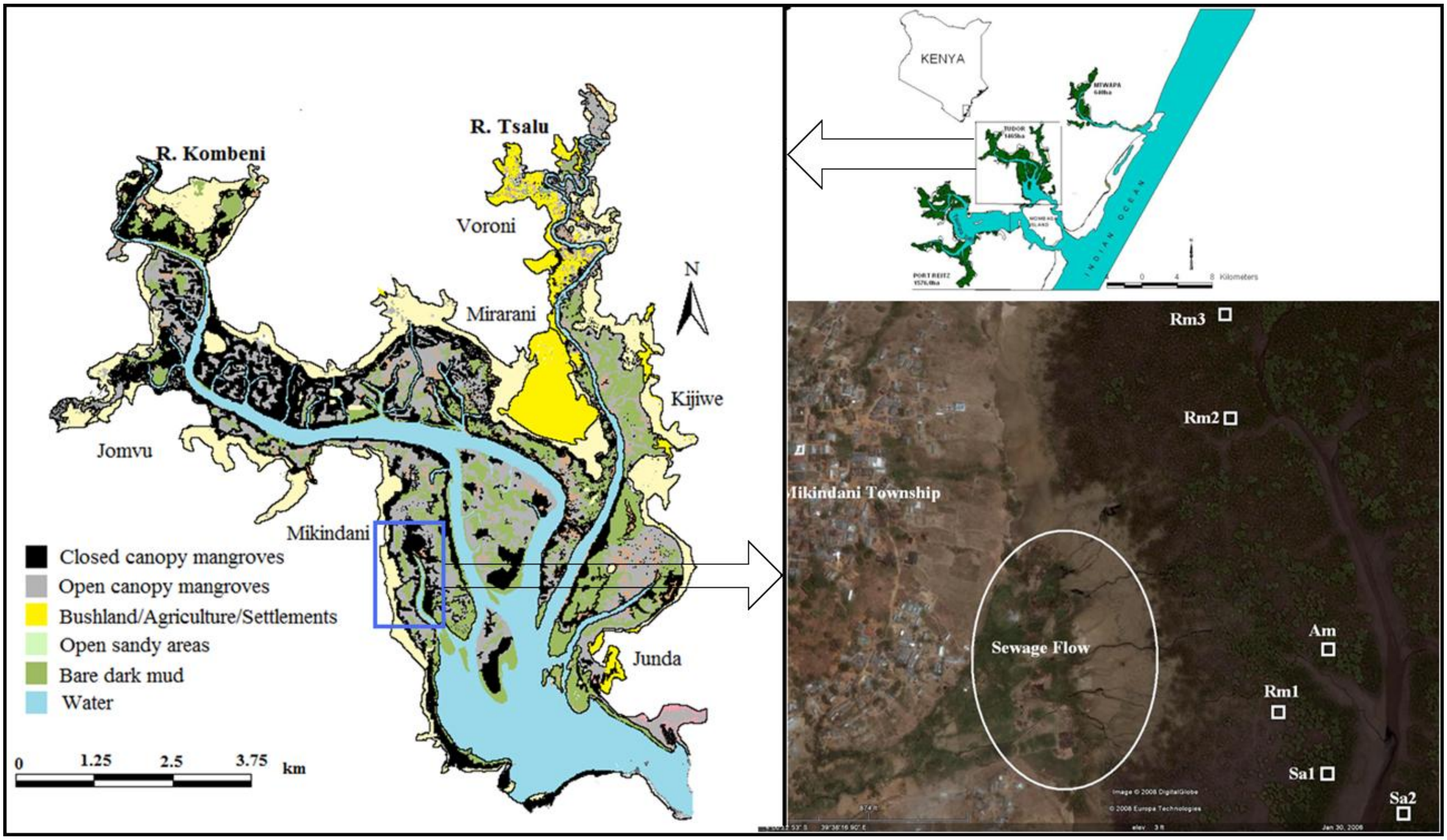

Figure 5.1: Map showing the location of litter traps. *R. mucronata* plots are labelled as Rm1 for direct or full sewage exposure, and Rm2 and Rm3 for partial exposures. *S. alba* is labelled as Sa1 for complete exposure and Sa2 for partial exposure, and Am for *A. marina* (Map scale 1:25000)

The mangroves of Tudor creek are separated naturally by two main tidal Creeks, Kombeni and Tsalu, 4.5 and 3 km long respectively cutting through the mangroves connecting to the upstream rivers. This study focuses on the Mikindani area, a township with about 67,164 people located in the west mainland of Mombasa district. In this area, raw sewage drains directly into the mangroves via an open surface flow system. The sewage runs through the mangrove forest in canals and has been discharged into the Tudor creek waters for more than a decade. The mangroves are dozed with sewage every tidal cycle, with the loading exponentially reducing with distance from the source. It is estimated that about 1,200 kg of nitrogen and 5.5 kg of phosphorus are discharged via sewage into the Mikindani system every day (PUMPSEA, 2007a).

The climate of Mombasa is influenced by the semi annual passage of the inter-tropical convergence zone (ITCZ) and the monsoons. The North Easterly Monsoon (NEM) occurs from December to March, while the South Easterly Monsoon (SEM) from May to October. Most of the rainfall occurs between the monsoons when convention is enhanced. The mean annual rainfall is 1,038 mm with the months of April, May and June recording the maximum. Average annual temperatures for the two seasons are 23.9°C and 28.5°C respectively (Figure 5.2).

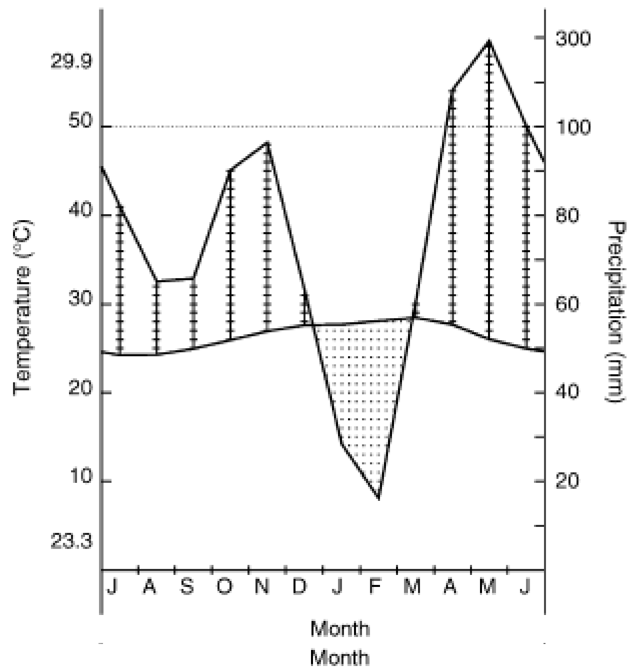

Figure 5.2: Climate of Mombasa (source Lieth *et al*.,1999)

5.2.2 Litter fall

The structural attributes of the Mikindani mangrove stand were assessed along belt transects of 10 m width both perpendicular and parallel to the creek. Based on these surveys, six randomly selected plots measuring 20 m by 20 m were identified according to the sewage exposure gradient. Sites located at the point source were designated as complete exposed and further from the point source as partial exposed (flushed by tidal flooding after dilution of sewage). Litter fall data were obtained from a total of 60 traps (10 traps per plot). Selected sites included three *R. mucronata* plots, two *S. alba* plots, and one *A. marina* plot. The number of plots was based on the rationale that *R. mucronata* was dominant and widely distributed, and like *S. alba*, occurred in both complete sewage exposed and partial exposed sites, while *A. marina* was distributed mainly in complete sewage exposed sites. All plots were inundated daily during flood tides. Litter traps were made of round metal frames of 0.25 m^2 mouth area to which a conical fabric net (mesh size 2 mm) was attached. Traps were positioned above the high water level and emptied monthly. Litter was sorted into: leaves, flowers, wood (twigs, bark and debris), reproductive materials (propagules and fruits). The sorted samples were then oven dried at 60°C for 48 hours and the dry weight recorded.

5.2.3 Leaf $\delta^{15}N$, $\delta^{13}C$ and C:N ratio analysis

Leaf litter samples (brown leaves) collected in April, June, August, and October 2006 were analysed for total carbon (TC), total nitrogen (TN) and $\delta^{15}N$ composition. Additional fresh (green) leaves were also collected directly from the trees, avoiding damaged leaves. To prepare samples for C and N analysis, dried samples were pre-treated with liquid nitrogen to make them brittle and ground to a fine consistency using a pestle and mortar. The carbon and nitrogen content of pre-weighed samples in pre-weighed tin cups were determined with a Thermo Finnigan Elemental Analyser flash 1112 connected to an Isotope Ratio Mass Spectrometer (Thermo Finnigan Delta) by a continuous flow interphase (Finnigan Conflo III). The relative abundance of the heavy and light stable isotopes of N are expressed as $\delta^{15}N$ values; i.e. in relative conventional standard N_2 for nitrogen. $\delta^{15}N$ values are calculated according to the formula:

$$\delta X(\text{in ‰}) = \left[\left(R_{sample} / R_{standard}\right) - 1\right] * 1000$$

where X = ^{15}N and R = $^{15}N/^{14}N$ or X = ^{13}C and R = $^{13}C/^{12}C$. Acetanilide (C = 71.03%, N = 10.36%) were used for standardisation in C:N ratio estimation. Internal sample reference material used for $\delta^{15}N$ and $\delta^{13}C$ was ammonium sulphate (IAEA-N1) and sugar (IAEA-CH-6) respectively.

The leaf resorption efficiency (RE) of N was calculated as:

$$RE = 100(1 - N_{LL}/N_{GL})$$

where N_{LL} is the leaf litter nutrient concentration and N_{GL} is the green leaf nutrient concentration.

5.2.4 Physicochemical parameters

Sediment cores were sampled randomly in twenty plots along four transects laid from the landward to the seaward fringe, two transects under complete sewage exposure and two under partial sewage exposure. In each plot, three sediment cores of diameter 6.4 cm and 30 cm depth were taken by hand, placed on ice (in a cool box) and transported to the laboratory for analysis immediately. The sediment cores were sectioned at 0-5 cm and 9-11 cm and 24-26 cm. Laboratory analysis included porewater analysis. The pore water was extracted with KCl solution under centrifugation (3000x g). The resulting pore water was then filtered through prewashed 0.45 µ Whatman GF/F microfiber filter and analysed for ammonium (NH_4^+), nitrate/nitrite (NO_X^-) and phosphates (PO_4^{3-}). The pore water collecting in the holes after sediment extraction was used to measure salinity with a hand held refractometre. For subsequent nutrient analysis after extraction, the conventional seawater analysis method by Parsons *et al*., (1984) was used. For estimation of organic carbon (%TOM), a pre-weighed sampled was dried to a constant weight at 70°C and then combusted at 450°C for 4 hours (ashing). The weight loss was then estimated as the percentage total organic matter (%TOM).

5.2.5 Data analysis

Litter production was estimated per unit area based on the averages per litter trap per plot and the annual litter fall determined by summing the average masses per month per unit area. Differences in litter production between the 2 years (2005-2006 vs. 2006-2007), between species and among the 12 sampling months (nested within years) were tested by the nested 3 way analysis of variance (ANOVA). Multiple comparisons by the Tukey HSD test were used to assess differences between plots. Data on porewater nutrients were log transformed and analysed by a nested 2-way analysis of variance (factors, exposure, and transects (nested in exposure). The non-parametric Spearman-R was used to establish correlation between litter fall and environmental factors. All analyses and graphical presentation were done using STATISTICA 8.

5.3 Results

5.3.1 Forest composition and physicochemical parameters

The Mikindani mangroves are dominated by young *R. mucronata*, older *A. marina* and *S. alba* (Table 5.1). The forest is being haphazardly exploited for wood, targeting *R. mucronata*, *C. tagal* and *A. marina* under a largely unregulated regime. We did not observe any form of harvesting for *S. alba*. Salinity profiles sampled during both dry and wet seasons displayed significant seasonal variations ($F_{1, 18} = 376.95$, $p < 0.001$), but no significant spatial differences ($F_{2, 15} = 1.116$, $p = 0.38$) during both seasons. Salinity averaged 42‰ (± 3) during the dry season and 29‰ (± 2) during the wet season. The spatial homogeneity of salinity may be the outcome of diurnal flooding, the narrow tidal flat and the flushing time of 13 days that causes complete exchange of waters within the creek (Nguli, 2006).

Table 5.1: Mean density of trees per ha, D_{130} (cm), average height (m), basal area (m²/ha) and volume (m³/ha) for the five species of mangroves at Mikindani in Tudor creek.

	Density (Stems ha^{-1})	Average D_{130} (cm)	Average Ht (m)	Basal Area (m^2 ha^{-1})
A. marina	144	22.06 ±10.61	7.35 ± 2.71	6.77
B. gymnorrhiza	3	17.82 ± 0.00	5.00 ± 0.00	0.04
C. tagal	32	6.36 ± 2.65	3.15 ± 1.48	0.12
R. mucronata	689	8.76 ± 8.57	4.02 ± 2.75	6.52
S. alba	226	12.91 ± 8.96	5.09 ± 2.33	4.17

Statistical tests for sediment pore-water concentrations for NH_4^+, NO_x^- and PO_4^{3-} and total organic matter (%TOM) for complete and partial sewage (status) exposed sites are shown in table 5.2. There were significant differences in NH_4^+ ($F_{1, 116} = 5.93$, $p < 0.02$), NO_x^- ($F_{1, 116} = 7.633$, $p = 0.007$), PO_4^{3-} ($F_{1, 116} = 15.18$, $p < 0.001$) and %TOM ($F_{1, 116} = 64.02$, $p < 0.001$) concentrations between sites (Table 3.2). Completely exposed sites being characterised by higher NH_4^+ and NO_x^- levels, with higher %TOM content. However, multiple comparisons by the Tukey HSD test reveal that PO_4^{3-} levels were significantly different in only one directly sewage impacted transect. Higher nutrients levels were observed in the landward and seaward fringe for completely exposed sites and decreasing nutrients levels from landward to seaward fringe for partially exposed sites and an increase in %TOM from landward to seaward (3.8% ± 3.31 – 7.79% ± 1.11).

Table 5.2: Average (± standard deviation) pore water nutrients concentrations (μM) and the %TOM from 30 cm sediment cores.

Depth	0-5 cm				9-11 cm				24-26 cm			
Parameter	NH_4^+	NO_x^-	PO_4^{3-}	%TOM	NH_4^+	NO_x^-	PO_4^{3-}	%TOM	NH_4^+	NO_x^-	PO_4^{3-}	%TOM
Complete exposure	570 (± 354)	16 (± 11)	2 (± 2)	5 (± 3)	543 (± 271)	22 (± 10)	1 (± 0.9)	5 (± 4)	465 (± 359)	27 (± 14)	2 (± 1)	6 (± 4)
Partial exposure	417 (± 201)	9 (± 5)	2 (± 1)	2 (± 0. 9)	339 (± 153)	12 (± 10)	2 (± 1)	3 (± 2)	305 (± 190)	10 (± 12)	2 (± 1)	4 (± 2)

Table 5.3: A 2-way nested ANOVA showing significant differences in porewater nutrients concentrations between transects and exposure.

	PO_4^{3-}			NO_x^-			NH_4^+			%TOM		
	df	F	p	df	F	p	df	F	p	df	F	p
Transect (Exposure)	2	8.76	***0.0003***	2	1.99	0.14	2	3.37	***0.038***	2	7.59	***0.001***
Exposure	1	15.18	***0.0002***	1	7.63	***0.007***	1	5.93	***0.016***	1	64.02	***< 0.0001***
Error	116			116			116			116		

5.3.2 Leaf $\delta^{15}N$, $\delta^{13}C$ and C:N ratio analysis

Table 5.4: Total organic carbon (TOC) and total nitrogen (TN) (mg/g DW), $\delta^{15}N$, $\delta^{13}C$ and the C:N ratio for senescent leaves (brown leaves) sampled from the three mangrove species. Differences are made between complete and partial sewage exposure.

Species	TN% (mg/g DW)	TOC% (mg/g DW)	$\delta^{15}N$‰	$\delta^{13}C$‰	C:N
A. marina[1]	0.86 ± 0.18 (8.60 ± 1.77)	47 ± 3 (468 ± 30)	6.46 ± 0.59	-27.53 ± 1.76	56 ± 9
R. mucronata[2]	0.39 ± 0.03 (3.85 ± 0.32)	48 ± 2 (476 ± 24)	3.31 ± 0.80	-27.46 ± 0.97	124 ± 12
R. mucronata[1]	0.48 ± 0.11 (4.80 ± 1.09)	48 ± 2 (477 ± 20)	4.73 ± 1.26	-28.37 ± 0.86	103 ± 19
S. alba	1.14 ± 0.34 (11.43 ± 3.42)	44 ± 3 (443 ± 25)	6.62 ± 0.75	-28.18 ± 0.91	42 ± 13
S. alba[1]	1.16 ± 0.41 (11.59 ± 4.07)	44 ± 5 (443 ±48)	7.03 ± 0.64	-28.76 ± 0.61	42 ± 12
R. mucronata[3]	0.33 ± 0.04 (3.35 ± 0.45)	45 ± 44.68 (449 ± 5)	3.45 ± 0.96	- 27.90 ± 0.90	135 ± 8

[1] = complete sewage exposure, [2] = partial sewage exposure, [3] = partial sewage exposure

The total leaf nitrogen and carbon content and the C:N ratio are given in table 5.3. There are significant differences in leaf nitrogen content by plot ($F_{5,\ 47}$ = 14.8876, p < 0.001) and species ($F_{2,\ 50}$ = 38.5027, p < 0.001), but no significant differences in leaf carbon content per plot ($F_{5,\ 47}$ = 0.80, p = 0.56) and species ($F_{2,\ 50}$ = 0.313, p = 0.73). The leaf nitrogen and carbon content did not vary significantly with month ($F_{3,\ 49}$ = 0.33685, p = 0.78) and sewage exposure ($F_{2,\ 50}$ = 1.4357, p = 0.25). Of the three mangrove species, *A. marina* and *S. alba* had higher nitrogen content than *R. mucronata*. However, completely exposed *R. mucronata* displayed higher leaf nitrogen content (4.80 ± 1.09 mg/g DW) than partially exposed (3.35 ± 0.45 mg/g DW). Leaf $\delta^{15}N$ signatures were on average higher than the normal range (-10‰ – +2‰) reported for plants globally (Fry *et al.*, 2000; Muzuka and Shunula, 2006; Bouillon *et al.*, 2008), with higher signatures for complete sewage exposed trees, though not significant ($F_{1, 42}$ = 2.0193, p = 0.16). The $\delta^{13}C$ levels measured were in the same range of values reported for mangrove leaves globally and ranged between -21‰ and -35‰, and is representative of C_3 terrestrial plants (Rao *et al.*, 1994; Muzuka and Shunula, 2006; Bouillon *et al.*, 2008; Kristensen *et al.*, 2008). Slight enrichment in $\delta^{13}C$ (less than 1‰) was generally observed for conspecifics.

Table 5.5: Total organic carbon (TOC) and total nitrogen (TN) (mg/g DW) and the C:N ratio for fresh leaves (green), and the Resorption Efficiency (RE) for the three mangrove species.

Species	Data	Complete	Partial	RE (%)
A. marina	C:N	23.53 ± 3.28	-	
	TN (mg/g dw)	19.77 ± 4.49	-	48 ± 24
	TOC (mg/g dw)	461 ± 109	-	
R. mucronata	C:N	44.29 ± 5.56	44.53 ± 7.08	
	TN (mg/g dw)	9.89 ± 0.07	9.54 ± 0.95	55 ± 23
	TOC (mg/g dw)	438 ± 52	420 ± 33	
S. alba	C:N	25.80 ± 5.20	28.56 ± 3.94	
	TN (mg/g dw)	21.00 ± 9.84	19.02 ± 6.41	53 ± 22
	TOC (mg/g dw)	518 ± 193	531 ± 144	

The C:N ratio reported in this study is comparable to values from earlier studies globally (Twilley *et al.*, 1986; Ellis *et al.*, 2006; Krauss *et al.*, 2008), though lower than rates reported for Gazi Bay (Rao *et al.*, 1994; Slim *et al.*, 1996; Ochieng and Erftemeijer, 2002). The C:N ratio estimated for Gazi Bay mangroves changed from 47.5 ± 21 in fresh leaves to 129 ± 60 in senescent leaves, indicating an approximate 64% resorbed nitrogen by the plants. Similar values have been reported for nutrient deficient arid mangrove in Western Australia (Alongi *et al.*, 2005). In this study, we observed a slight elevation in leaf N content for complete sewage exposed *R. mucronata*, and not for *S. alba*. This implies species specific traits may influence responses to elevated nutrients. This is further strengthened by the strong significant correlation between C:N and $\delta^{15}N$ signature (Figure 5.2), species with higher C:N ratio had corresponding lower $\delta^{15}N$ signatures and vice versa. Our observation corresponds to observations on mangrove N and P fertilization mangroves in Twin Cays, where significantly decreased resorption efficiencies for N and P were reported. Thus nutrient enrichment reduces conservation of essential nutrients resulting in litter that is enriched with nutrients (Wanek *et al*., 2007). However, our findings indicate this response may vary with species.

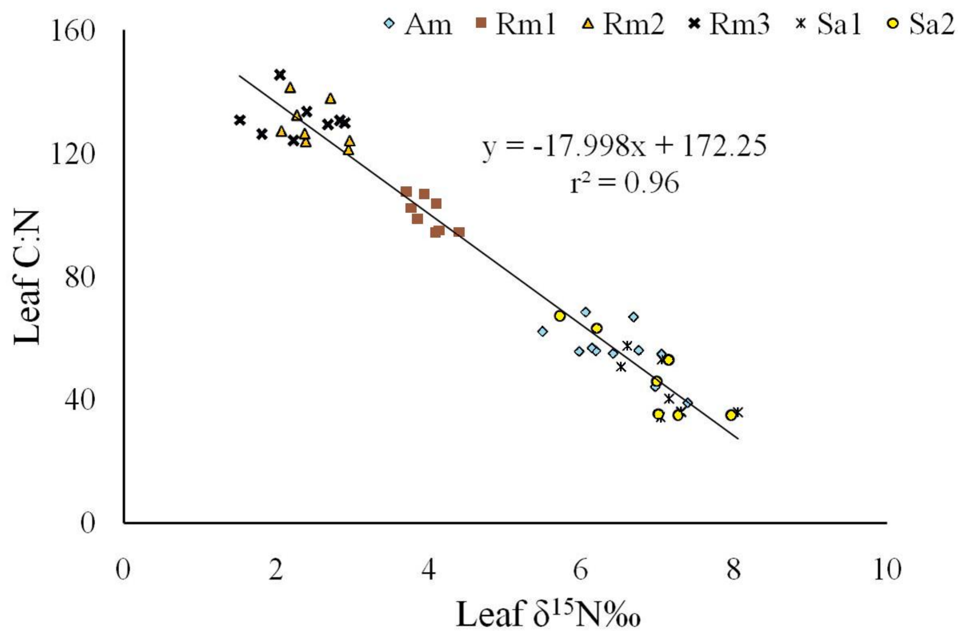

Figure 5.2: Scatter plot of leaf C/N ratio versus leaf δ^{15}N‰ showing a significant correlation ($r^2 = 0.96$, $F = 1231.707$, $p < 0.0001$).

5.3.3 Litterfall

Litter production and fall was observed throughout the year, with significant differences between species (table 5.7 and 5.8), *R. mucronata* recording the highest annual rate (15.34 ± 3.34 t DW $ha^{-1}yr^{-1}$). No significant differences were observed between *A. marina* and *S. alba*, (11.44 ± 2.90 - 9.69 ± 5.26 t DW $ha^{-1}yr^{-1}$ respectively) (Table 5.6 and 5.8). However, *A. marina* recorded higher litter fall than *S. alba*. In the second year of litter collection, a die back of *S. alba* receiving raw sewage occurred after high rainfall between August and December 2006. The die back contributed to the lower litter fall for *S. alba* in the second year. The cause of die back is not obvious, and may be due to abrupt changes in salinity due to sudden input of copious amounts of fresh water, siltation or herbivory by unidentified insects, though this may require a separate study to fully understand the underlying factors. A similar die back of *S. alba* was observed in Gazi Bay (ca. 47 km south; Wang'ondu, University of Nairobi, personal communication).

Table 5.6: Average (± standard deviation) annual litter fall (t DW ha^{-1} yr^{-1}) per species. Values in parenthesis indicate percentage of total litter per species.

	Leaves	**Wood**	**Flowers**	**Reproductive Material**	**Total (t ha^{-1}yr^{-1})**
A. marina	7.63 ± 0.29 (66.64)	0.62 ± 0.14 (5.46)	1.98 ± 2.01 (17.30)	1.21 ± 0.46 (10.60)	11.44 ± 2.90
R. mucronata[2]	13.20 ± 2.29 (73.10)	0.33 ± 0.06 (2.11)	1.42 ± 0.93 (1.68)	4.09 ± 3.45 (9.90)	19.04 ± 0.53
R. mucronata[1]	9.08 ± 0.10 (74.32)	0.45 ± 0.57 (2.61)	0.58 ± 0.08 (6.78)	3.30 ± 0.19 (9.21)	13.42 ± 0.74
S. alba[1]	6.94 ± 6.10 (66.78)	1.66 ± 0.99 (15.97)	0.04 ± 0.06 (0.42)	1.75 ± 0.95 (16.82)	10.4 ± 8.10
S. alba[2]	6.77 ± 4.13 (69.88)	1.73 ± 0.58 (17.84)	0.04 ± 0.04 (0.43)	1.08 ± 0.95 (11.15)	9.69 ± 5.26
R. mucronata[3]	9.17 ± 1.14 (70.73)	0.33 ± 0.10 (2.69)	0.62 ± 0.19 (2.12)	3.12 ± 1.10 (8.97)	13.25 ± 2.53
Average (t ha^{-1}yr^{-1})	8.29 ± 1.94 (68.22)	0.91 ± 0.72 (7.48)	0.97 ± 0.97 (7.94)	1.93 ± 1.36 (15.89)	12.16 ± 2.89

[1] = complete sewage exposure, [2] = partial sewage exposure, [3] = partial sewage exposure

Table 5.7: 3 way nested ANOVA showing significant differences in productivity between months, year and species.

	SS	**df**	**MS**	**F**	**p**
Month (Year)	47.10	22	2.14	16.63	***< 0.001***
Year	0.75	1	0.75	5.83	***0.016***
Species	11.81	2	5.91	45.88	***< 0.001***
Error	186.95	1452	0.13		

Table 5.8: Tukey HSD test showing significance in differences between species and plots for total litter g DW m^{-2} month^{-1}. Figures in bold italics show significant difference at p< 0.05.

	Plot	(1) - 106	(2) - 199	(3) - 142	(4) - 101	(5) - 78	(6) – 170
1	*A. marina*[1]	-	***< 0.001***	0.18	0.99	0.48	***< 0.001***
2	*R. mucronata*[2]	***< 0.001***	-	***0.005***	***< 0.001***	***< 0.001***	0.28
3	*R. mucronata*[1]	0.18	***0.005***	-	0.097	***0.001***	0.35
4	*S. alba*[1]	0.99	***< 0.001***	***0.097***	-	0.69	***< 0.001***
5	*S. alba*[2]	0.48	***< 0.001***	***0.001***	0.69	-	***< 0.001***
6	*R. mucronata*[3]	***< 0.001***	0.28	0.35	***< 0.001***	***< 0.001***	-

[1] = complete sewage exposure, [2] = partial sewage exposure, [3] = partial sewage exposure

Table 5.9: Statistical tests showing significant effects of climatic factors on litter fall.

Litter component	Climate factor	Spearman - R	t(N-2)	*p*
Leaves	Temperature	0.49	8.71	**< *0.001***
	Rain	-0.21	-3.34	**< *0.001***
	Relative Humidity	-0.28	-4.53	**< *0.001***
Wood	Temperature	-0.09	-1.46	0.15
	Rain	0.05	0.84	0.4
	Relative Humidity	0.05	0.84	0.4
Flowers	Temperature	-0.35	-5.69	**< *0.001***
	Rain	0.15	2.4	***0.02***
	Relative Humidity	0.17	2.63	***0.01***
Reproductive Material	Temperature	0.12	1.8	0.07
	Rain	-0.05	-0.71	0.48
	Relative Humidity	-0.12	-1.88	0.06

Litter production and fall displayed distinct seasonal and temporal fluctuations which were repeated between years. The second year was characterised by lower litter fall (Table 5.7 and Figure 5.3) coupled with higher rainfall and temperatures. The seasonal patterns displayed significant correlation with climatic factors such as temperature, rainfall and relative humidity (table 5.9). Leaf fall was significantly correlated with temperature (Spearman-R = 0.49, t(N-2) = 8.71, $p < 0.001$), rainfall (Spearman-R = -0.21, t(N-2) = -3.33671, $p < 0.001$), and relative humidity (Spearman-R = -0.28, t(N-2) = -4.53, $p < 0.001$). High temperatures favour high leaf fall, while rainfall and high relative humidity correlate with low leaf fall. Flowering significantly correlate with temperature (Spearman-R = -0.35, t(N-2) = -5.69, $p < 0.001$), rainfall (Spearman-R = 0.15, t(N-2) = 2.40, $p = 0.017$) and relative humidity (Spearman-R = 0.17, t(N-2) = 2.63, $p = 0.01$).

R. mucronata produced flowers all year round. The main flowering event occur in March and July, and fruiting between July–October. Propagules are produced between March – June. *S. alba* flowered in August – October, with minor flowering events in February–April, fruiting in July–January, and seeds produced biannually in November and April. However, seeds were not produced in November 2006, probably due to observed die back of *S. alba*. *A. marina*, produced flowers between December and April, fruits between February and July and seeds between April and May. Woody materials did not correlate with any climatic factors and did not display any seasonal trends.

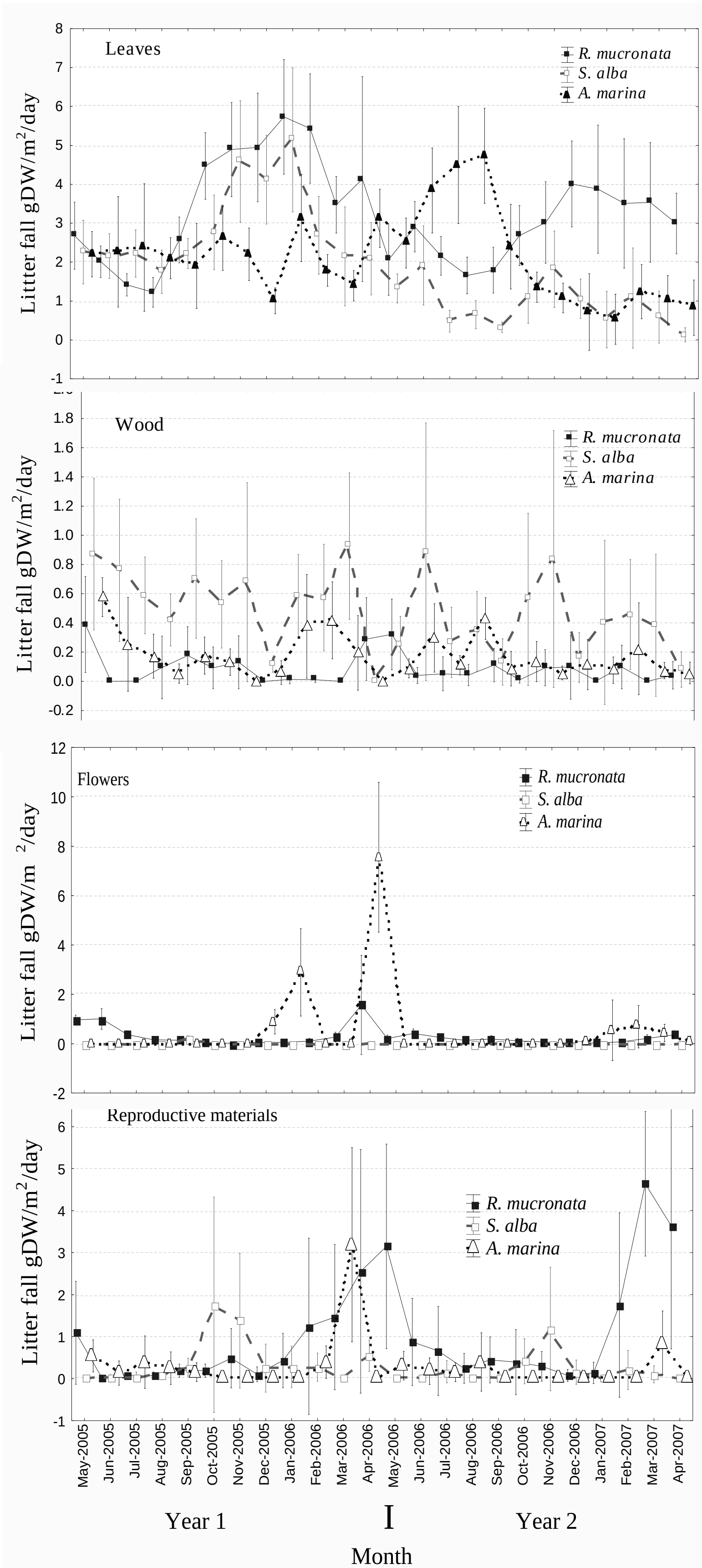

Figure 5.3: Litter fall trends per part/category and species for two years.

The mean annual litter fall was estimated at 12 (± 3) t DW $ha^{-1}yr^{-1}$, with highest litter fall rates in the NEM season, January-April (ca. 5 ± 1 g DW m^{-2} Day^{-1}), a hot period coinciding with the onset of the rainy season and production of reproductive materials, while lowest rates occurred in the cooler and wet SEM season, June-October (ca. 3 ± 0.5 g DW m^{-2} Day^{-1}) (Figure 5.3). Leaves were the most important litter contributors with an annual mean rate of 8 ± 2 t DW $ha^{-1}yr^{-1}$, accounting for 68% of the total litter fall. Reproductive material (fruits and propagules; 2 ± 1 t DW $ha^{-1}yr^{-1}$), wood (0.9 ± 0.7 t DW $ha^{-1}yr^{-1}$) and flowers (0.97 ± 0.97 t DW $ha^{-1}yr^{-1}$) accounted for 16%, 8% and 8% of the mean annual litter respectively (Table 5.6).

5.4 Discussion

The litter production rates for the mangroves of Tudor creek are moderately within the upper global range reported for *R. mucronata*, *A. marina* and *S. alba* in different geographical ranges (Table 5.10). Values in the range 3 t DW ha^{-1} yr^{-1} (Phuket; Chansang and Poovachiranon, 1985) to 16 t DW ha^{-1} yr^{-1} (Malaysia; Sasekumar and Loi, 1983) have been reported for Rhizophoraceae, 8 t DW ha^{-1} yr^{-1} (Australia; Duke *et al*., 1981) to 17 t DW ha^{-1} yr^{-1} (India; Wafar *et al*., 1997) for *S. alba* and 3 t DW ha^{-1} yr^{-1} (Australia; Clarke, 1994) to 16 t DW ha^{-1} yr^{-1} (Australia; Bunt, 1995) for *A. marina*. Consistent with other studies, leaves were the major contributors for litter fall followed by reproductive materials and floral parts. Our observation confirms the reported trend of increasing productivity from the northern and southern hemispheres towards the equatorial zone (Woodroffe, *et al*., 1988; Duke, 1990) and the increased productivity from fringe arid coasts mangroves to fringe humid and riverine mangroves (Lugo *et al*., 1981; Twilley *et al*., 1986; Medina and Francisco, 1997).

Table 5.10: Litter production in mangroves from different parts of the world.

Location	Latitude	Species	Litter fall (t DW ha^{-1} yr^{-1})	Reference
Kenya	4° S	*A. marina*	11	This study
(Tudor creek)		*R. mucronata*	15	
		S. alba	10	
Kenya	4° S	*A. marina*	5	Wang'ondu *et al.*, in press
(Gazi Bay)		*R. mucronata*	11	
		S. alba	10	
Kenya	3° S	*R. mucronata*	16	Kairo, 2001
(Mida Creek)		*C. tagal*	23	
		A. marina	8	
Kenya	4° S	*R. mucronata*	9	Slim *et al.*, 1996
(Gazi Bay)		*C. tagal*	2	
China	22° N	*Aegiceras corniculatum*	12	Tam *et al.*, 1998
(Futian Nature National Reserve)		and *Kandelia candel*		
South Africa	21° S	Mixed forest	5	Steinke and Ward, 1990
(Wavecrest (Transkei))				
India	15° N	*R. mucronata*	11	Wafar *et al*., 1997
(Mandovi-Zuari Estuary)		*S. alba*	17	
New Guinea (Fly River Estuary)	8° S	Mixed forest	8 – 14	Twilley *et al.*, 1992
Malaysia (Matang mangal)	5° N	Mixed forest	4	Gong and Ong, 1990
Australia (Jervis Bay, NSW)	35° S	*A. marina*	3	Clarke, 1994
Australia (Embley River)	12° S	*A. marina*	6	Conacher *et al.*, 1996
Papua New Guinea	9° S	*A. marina*	16	Bunt, 1995
(near Port Moresby)				
Mexico	22° N	Mixed forest	14	Flores-Verdugo *et al.*, 1990
(Teacapan-Ague Brava Lagoon)				

Globally, litter fall from a number of mangrove forests has been observed to be highly variable, a fact mostly attributed to tree height, latitude (Saenger and Snedaker, 1993) and climate (Wafar *et al.*, 1997). Our findings further show that species specific attributes may be determining litter production in addition to latitude and tree height (Woodroffe, 1985), as the shorter *R. mucronata* (height 4.02 ± 2.75 m) was more productive than the taller *A. marina* (height 7.35 ± 2.71 m) and *S. alba* (height 5.09 ± 2.33 m). *R. mucronata* is heavily harvested in Tudor creek, resulting in predominantly young vegetation (Chapter 3; Mohamed *et al.*, 2008), which may partly account for the high litter production, as young mangrove stands have been reported to produce large and high quality litter compared to older established stands (Nga *et al.*, 2005). This may further imply that site conditions in Tudor creek are still conducive for the establishment of a healthy productive mangrove ecosystem.

Litter fall rates in the current study are high compared to other studies, considering that most studies presented data for mangrove trees of 10 m height and above, mostly considering undisturbed sites (Table 5.10). Ellis and Bell (2004) reported the impacts of trimming on litter fall, which does not appear to reveal much impacts as in our study, but we postulate that a decrease in productivity in Tudor creek has occurred through the reduction of closed canopy mangroves and the enlargement of canopy gaps (Chapter 3; Mohamed *et al.*, 2008). The high yields can be explained both in terms of location of the site at lower latitude, specific growing conditions as a result of greater tidal activity and water turn-over within the creek, coupled with the monsoon climate (Wafar *et al*., 1997).

The seasonal variation in litter fall has featured prominently in earlier studies, but the underlying causes remain obscure. As reported by Slim *et al.* (1996), Wafar *et al.* (1997) and Ochieng and Erftemeijer (2002), the maxima in litter fall are associated with the dry NEM months. This is a result of water stress due to low or no rainfall, higher temperatures, characterised by high evapo-transpiration rates leading to higher salinities, complicating metabolism of transpiration and necessitating canopy thinning by leaf loss as reported by Wafar *et al.* (1997), Tam *et al.* (1998) Slim *et al.* (1996) and Eusse and Aide (1999).

5.4.1 Carbon and nitrogen content

The two indicators of mangrove ecosystem N cycling measured in this study – mangrove leaf %N and δ^{15}N‰, were on average higher than reported for unpolluted sites globally (-1.32 – 2.789‰: Fry *et al*., 2000; Costanzo *et al*., 2003; Bouillon *et al*., 2008), and in particular unpolluted sites in Tanzania (Muzuka and Shunula, 2006) and Kenya (Gazi Bay; Marguillier *et al*., 1997). Mangroves of Gazi Bay displayed δ^{15}N‰ values ranging between 0.76 – 2.18 (Marguillier *et al*., 1997). This study also point to species specific traits in response to sewage exposure levels within the site (Alongi *et al*., 2005). Between-site differences in plant isotopic composition may therefore be viewed as an expression of the differences in δ^{15}N levels in the source – the raw domestic sewage effluent (Cabana and Rasmussen, 1996), and/or isotopic fractionation during uptake (Costanzo *et al*., 2003). The tidal flooding within the sites dilutes and distributes the sewage effluent creating a characteristic gradient, resulting in high pore-water NH_4^+ and NO_x^- levels in completely exposed sites compared to partially exposed sites, setting the ecosystem baseline for δ^{15}N. Isotopic fractionation occurs via plant uptake, microbial nitrification-denitrification, soil adsorption and volatilization, as the sewage flows sequentially first through farm lands, then through *R. mucronata* plots, and eventually through *S. alba* plots, where preferential microbial processing and uptake of the lighter ^{14}N results in δ^{15}N enriched effluent (Yoneyama *et al*., 1991; Pennock *et al*., 1996; Costanza *et al*., 2003). The lower leaf δ^{15}N values in *R. mucronata* leaves may arise out of nutrient use efficiency for the species, known to possess leaves with lower nutrient content as an adaptation against herbivory (Alongi *et al*., 2005).

The lack of seasonal variation in mangrove leaf δ^{15}N signatures has also been observed by Costanzo *et al*. (2003). This is because leaf δ^{15}N signatures reflects the mangrove life history and the sediment N source, and is a representation of the long-term nitrogen supply and the edaphic conditions (Fry *et al*., 2000; Costanzo *et al*., 2003). Subsequently, the leaf δ^{15}N levels may reflect the mangrove sediments capacity to retain or immobilize nutrients in wastewater. However, whether mangrove trees contribute to phytoremediation depends on uptake and would require a separate study to establish (Chu *et al*., 1998; Ye *et al*., 2001; Feller *et al*., 2003a, b; Costanzo *et al*., 2003). But from our observation on the strong relation between C:N and δ^{15}N, nutrients uptake may be species specific.

5.5 Conclusion

The effect of sewage exposure on litter production presented no subtle effects on quantity of litter based on exposure regime. Tam *et al.* (1998), observed no differences in litter fall between sewage receiving and control sites. However, whereas in our study we observed high pore-water levels of NH_4^+ and NO_x^- in complete exposed sites, PO_4^{3-} levels were higher in partially exposed sites, a possible variability in limiting nutrients between sites (Feller *et al*., 2003). Mangroves are reported to be either phosphorus (P)-limited (Feller, 1995; Koch and Snedaker, 1997) or differentially N- or P- limited across tidal gradients (Boto and Wellington, 1983; Feller *et al*., 2003a, b; Krauss *et al*., 2008), and in our study, PO_4^{3-} may be more limiting in complete sewage exposed than partial exposed site. Therefore, long term variable exposure to sewage, high nutrients supply with lower PO_4^{3-} levels in completely exposed sites, coupled with siltation effects, may account for the lack of differences in productivity between sites. However, further studies may be necessary to assess the impacts of these multiple factors on productivity in mangroves.

Differences in leaf N content were only observed in *R. mucronata* and not in *S. alba*. This leaf nitrates levels has been observed to vary with sediment nutrients levels (Dittmar *et al*., 2006). Leaf nutritive values, expressed as C:N ratio, have an inverse relationship with litter degradation constant, litter degradation being strongly dependent on availability of nitrogen for microbial decomposers (Dittmar *et al*., 2006; Ellis *et al*., 2006). Therefore leaves enriched in N will potentially accelerate nutrient cycling via litter fall and decomposition in these ecosystems, enhancing organic matter provisioning to mangrove food webs (Wanek *et al*., 2007). This relationship is displayed by the leaf $\delta^{15}N$, which also indicates nutrients dynamics within the Tudor creek mangrove system are affected by inputs from raw domestic sewage and land use patterns by humans.

6. Cover changes and regeneration status of a peri-urban mangrove

R. mucronata sapling covered with silt

Publication

Mohamed, M. O. S., Neukermans, G., Kairo, J. G., Dahdouh-Guebas, F., and Koedam, N. 2008. A glance at the cover changes and regeneration status of a peri-urban mangrove. Submitted to *Forest Ecology and Management*.

Summary

Stability of an ecosystem is determined by its resilience, regenerative capacity and numerous weak trophic links, amongst other natural and human induced factors. The Tudor creek mangroves, a typical peri-urban mangrove, are exposed to both episodic natural and recurrent human disturbances, including decades' long exposure to raw domestic sewage, sporadic unregulated-harvesting and episodic siltation. This study evaluates the regeneration patterns within extended gaps and the understorey. An evaluation on species mix and regeneration patterns is also done. Preliminary analysis of aerial photographs (1969 and 1992) and a satellite image (2005) indicate a 12.5% decline in closed canopy mangrove between 1969 and 1992, and a 55% decline between 1992 and 2005. Distribution of adult trees was variable, with mixed stands and large canopy openings in the mid intertidal range. Species composition of seedlings and saplings did not always reflect the overstorey species composition and varied with gap sizes. Gap sizes range between 10 - 50m² have higher or mostly adequate regeneration, while gaps smaller than 10m² and bigger than 60m² have lower regeneration levels. *R. mucronata* seedlings and saplings occurred in the understorey under all cover types and inundation regime, conferring advantages to this species under the current disturbance regime. This may favour its establishment in relation to other species. *A. marina* and *C. tagal* saplings and seedlings are restricted to the forest edges and gaps. The current status of the forest is reminiscent of a recovery phase, a multiphase succession stage, after a major disturbance event, accompanied by recurrent anthropogenic pressure. This study shows that species composition depends in part on the balance between natural large-scale and recurrent small-scale human disturbances.

Keywords: mangrove, canopy, gap, regeneration, disturbance

6.1 Introduction

The persistent undervaluing and apparent importance of mangroves to humans (Dahdouh-Guebas *et al.*, 2000; Walters, 2000; Rönnbäck *et al.*, 2003; Walters, 2003; Bosire *et al.*, 2004; Dahdouh-Guebas *et al.*, 2004; Walters, 2004; Rönnbäck *et al.*, 2007), wildlife (Nagelkerken *et al.*, 2008), and the global carbon balance (Kristiansen *et al.*, 2008), have been adequately elaborated, and the call for realisation of the extrinsic and intrinsic values of mangroves seems more urgent than ever as human activities continue to threaten the mangrove ecosystems at both local and global scales (Ewel, 2001; Duke *et al.*, 2007; Nagelkerken *et al.*, 2008), with 35% of the world's mangrove forests lost over the last two decades (Valiela *et al.*, 2001), making the prospect of a world deprived of the services offered by mangrove ecosystems, perhaps within the next 100 years, real (Duke *et al.*, 2007). Human induced impacts coupled with a changing global climate calls for an evaluation and understanding of the regenerative characteristics of the mangrove ecosystems. This will facilitate the appreciation of future implications and devise appropriate interventions (Robertson *et al.*, 1991; Kathiresan and Bingham, 2001).

With the high population growth particularly in tropical coastal areas, coupled with rapid urbanisation, the peri-urban context of mangroves, seen as exceptional currently, is fast becoming a recurrent feature for mangroves as observed in Singapore, Hong Kong, Philippines, Tanzania (Daressalam), Mozambique (Maputo), India (Calcutta, Goa), Colombia, Mexico etc….(Lacerda *et al.*, 1993; Kathiresan and Rajendran, 2005b; Tam *et al.*, 1997; Holguin *et al.*, 2001; Holguin *et al.*, 2006; PUMPSEA, 2007a, b; Walters *et al.*, 2008). The cumulative effects of disturbances on mangrove range from reduced adult tree density, reduced canopy height, increased canopy gaps, to altered edaphic conditions (Walters, 2005a, b; López-Hoffman *et al.*, 2006; Alongi and de Carvalho, 2008). Disturbed stands are characterised by few species of widely dispersed, dwarf like trees, with a distinctly bushy appearance. Disturbances that cause nutrients loading, eutrophication or domestic sewage pollution are widely reported to have no apparent negative effects (Wong *et al.*, 1997; Vaiphasa *et al.*, 2007), and impact positively on tree growth (Feller *et al.*, 2003b; Boonsong *et al.*, 2003), but may also enhance herbivory (Feller *et al.*, 1995). Siltation on the other hand, necessary to some extent (Victor *et al.*, 2003; Kitheka *et al.*, 2003), has potentially destructive effects by causing tree mortalities (Ellison, 1998; Thampanya *et al*., 2002), while over-harvesting or clear-felling can potentially lead to species shifts, lowering the intrinsic values (Kairo *et al*., 2002a, b; Dahdouh-Guebas *et al*., 2004) and may lead to stand collapse. However, the apparent confounding effects of these disturbances remain largely unknown and speculative, with no specific studies. Scenarios of such combinations may also be rare or have avoided attention.

The peri-urban mangroves of Tudor creek, Mombasa, have variously been described as structurally stressed (Chapter 3; Mohamed *et al*., 2008), grossly over-exploited, undervalued and poorly managed (Chapter 4; Mohamed *et al*., under review). They present a typical forest impacted by multiple disturbances at variable temporal and spatial scales, with a notable resilience to disturbances, maintaining high productivity (Chapter 5). However, these disturbances can induce changes in forest structure, composition and micro-climate, including tidal regime, seed/propagule availability and dispersal, establishment and survival, in addition to lowering the health and vigour of both seedlings and adult trees and mortalities through suffocation of breathing roots (Ellison, 1998), significantly altering habitat conditions for recruitment and establishment of seedlings (Bosire *et al*., 2003; Walters, 2005a, b; Alongi and de Carvalho, 2008).

In this study, we assess the changes in mangrove canopy cover over time and the regeneration patterns in relation to the current canopy cover and species mix. Aspects of regeneration assessed include the distribution patterns of seedlings and saplings in relation to adult trees, considering the understorey and gaps. The survival and growth performance of replanted seedlings was also done at Mikindani in Tudor creek, considering single and mixed species plots. The outcome of the study will give important information on the recovery potential of the forest and the need for restoration. This will guide on the development of management strategies to ensure ecological sustainability. This study will thus provide useful insights into the resilience of the mangrove forests of Tudor creek.

6.2 Materials and methods

6.2.1 Study area

Tudor creek (Figure 6.1) bounds Mombasa Island on the northwest and extends some 10 km inland. The creek has two main seasonal rivers, Kombeni and Tsalu, draining an area of 550 km^2 (450 and 100 km^2 respectively) with average freshwater discharge estimated at 0.9 m^3s^{-1} during the inter-monsoon long rains (cited in Nguli, 2006). It has a single narrow sinuous inlet with a mean depth of 20 m, that broadens out further inland to a central relatively shallow basin (5m) fringed by a well developed mangrove forest mainly composed of *R. mucronata*, *A. marina* and *S. alba*. The basin has an area of 6.37 km^2 at low water spring and 22.35 km^2 at high water spring. Mangrove forests occupy 8 km^2 of the creek. Detailed structural attributes of the Tudor creek mangroves are described by Mohamed *et al.* (2008; Chapter 3).

The Tudor creek mangrove system has been exposed to raw sewage intensively for more than a decade. The sewage runs through the mangrove forest in canals and is discharged into the Tudor creek waters mainly from Mikindani, Tudor and the Old Town settlements. The mangroves are periodically dozed with sewage every tidal cycle, with the loading exponentially reducing with distance from source (PUMPSEA, 2007a). It is estimated that about 1,200 kg of nitrogen and 5.5 kg of phosphorus are discharged via sewage into the Mikindani system every day. The Mikindani Township with about 67,164 people is located in the west mainland. Sewage from this estate was according to plans intended to be pumped for processing at the Kipevu treatment plant. But due to the overloading at the Kipevu plant, sewage from the township is now discharged directly into the nearby mangrove forest at Mikindani in Tudor creek. Sediments of Tudor creek are predominantly mud and some parts are covered with sand. The land surrounding the creek beyond Mombasa Island is mainly agricultural, largely small-holdings and coconut plantations with rough grazing land further inland, while the immediate slopes bordering the mangrove creek are being intensively cleared of vegetation to create space for informal settlements and subsistence farming.

6.2.2 Canopy change detection

Mapping was done using panchromatic aerial photographs of 1969 and 1992 respectively. It was envisaged that the multi-date data sets will generate sufficient information necessary for detecting changes that have occurred in Tudor creek mangrove forests over the years. The approach was also meant to produce data that could also inform management on status of the forest. The method used was a multi-stage approach starting with; processing and printing of aerial photographs of 1969 (Survey of Kenya) and 1992 (DRSRS) at scale 1:25000; stereoscopic examination; photo interpretation; and maps generation showing the entire Tudor creek with distinct areas under mangrove vegetation and the surrounding land cover classes separately delineated. The map was then digitized and geo-referenced using ARCVIEW 3.1 software. Comparisons in cover classes were made with satellite images (IKONOS, 2005) analysed by ERDAS under the PUMPSEA project (Neukermans, Vrije Universiteit Brussel, unpublished data).

6.2.3 Forest assemblage and regeneration sampling

A stratified sampling technique was used to sample mangroves of Tudor creek. The location of transect lines were determined by an initial reconnaissance and examination of medium-scale (1:25,000) panchromatic aerial photographs of the area. To sample small gap sizes, belt transects of 10 m width were established both perpendicular and parallel to the creeks across the entire forest in such a way that they represented optimally the general mangrove formation of Tudor creek (Figure 6.1). Vegetation sampling was carried out within 10 by 10 m^2 quadrats, established along transects. A total of 196 plots were sampled under dense mangroves, and a total of 41 additional $400m^2$ randomly selected plots were sampled under extended or enlarged canopy gaps.

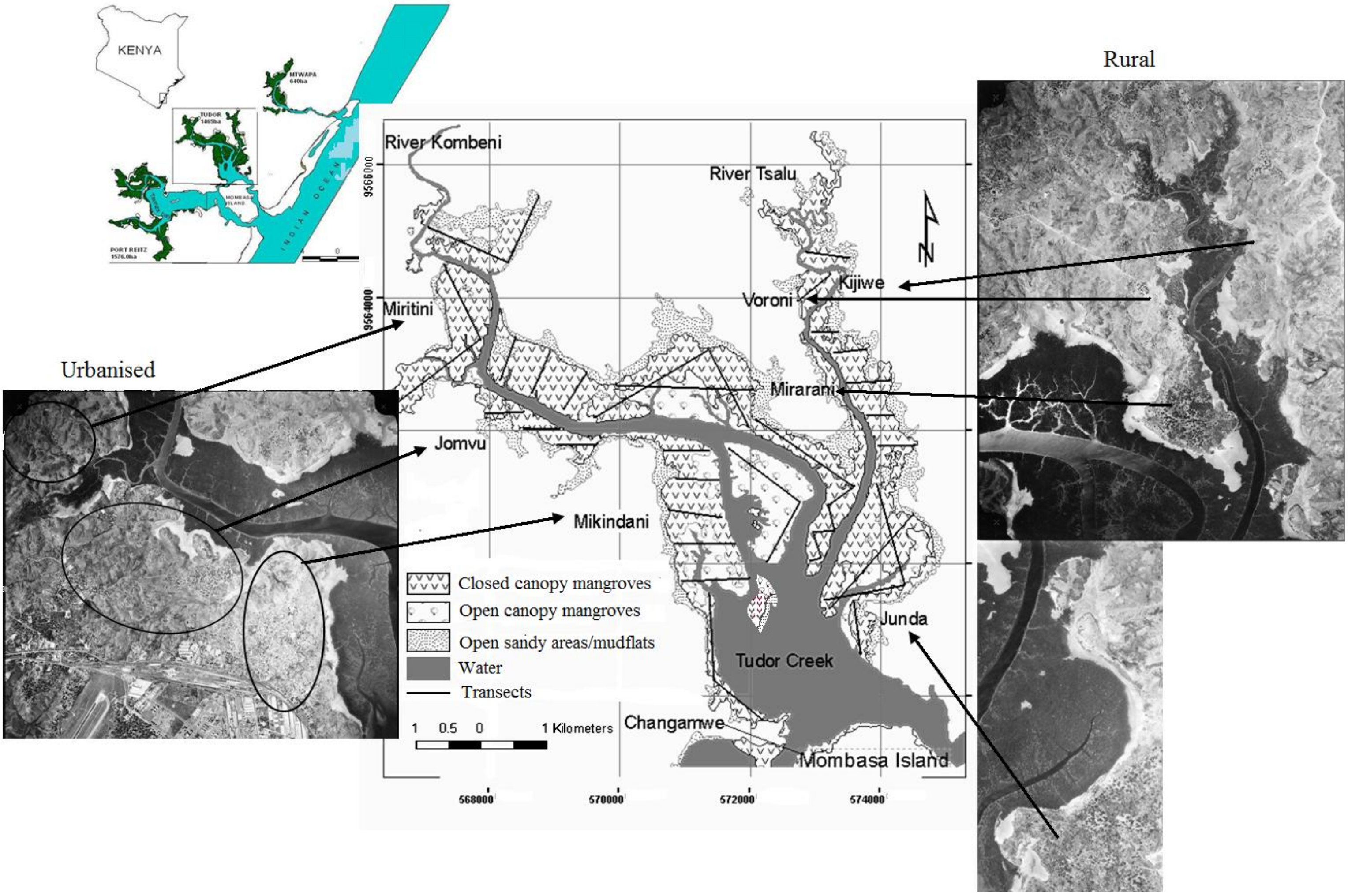

Figure 6.1: Map of Mombasa showing the study area (Tudor creek) in relation to the other creeks Notice the proximity of settlements to the mangroves and the differences in density between the urban Kombeni and Rural Tsalu. The aerial photographs are to scale 1:25,000 km (Source, DRSRS aerial photographs 1992).

In each sampling plot, adult trees species were identified and their height and the D_{130} were measured. This was used to identify species mix or zone within the forest. Information on the composition and pattern of natural regeneration was obtained using the method of Linear Regeneration Sampling (Sukardjo, 1987). In 5 x 5 m^2 subplots (within the main 10 x 10 m^2 quadrats), occurrence of seedlings and saplings of different species was recorded in each plot. Seedlings were identified as below 100 cm in height, and mostly included new recruits normally termed as potential or/and established regeneration, while saplings were young trees, with heights above 100 cm, with D_{130} below 2.5 cm. Careful evaluation was done before assigning classes based on site conditions.

The analysis of spatial pattern of adults and juveniles in the field was carried out inside 10 x 10 m^2 plots along transects. The measure of dispersion used was Morisita's dispersion Index (Morisita, 1959), the application of which is described in Greig-Smith (1983). Morisita's Index (I_δ) is:

$$I_\delta = q\sum_{i=1}^{q} \frac{n(n-1)}{N(N-1)}$$

Where, q is the number of quadrats, n_i is the number of individuals per species in the i^{th} plot, and N is the total number of individuals in all q quadrats. If $I_\delta > 1$, the population is clustered, if $I_\delta = 1$, the population is randomly dispersed and if $I_\delta < 1$, the population is evenly dispersed.

Inundation classes for each plot was estimated according to Watson (1928) and classified as: (1) inundated by all high tides, (2) inundated by medium high tides, (3) inundated by normal high tides, (4) inundated by spring tides, and (5) occasionally inundated by exceptional or equinoctial tides.

6.2.4 Physicochemical parameters

Sediment cores were sampled randomly in twenty plots along four transects laid from the landward to the seaward fringe, two transects in sewage receiving and two in non-sewage receiving sites. In each plot, three sediment cores of diameter 4 cm and 15 cm depth were taken by hand using corers. Laboratory analysis included porewater analysis extracted with KCl. The sediment cores were sectioned at 0-5 cm and 9-11 cm. The pore water was extracted with KCl under centrifugation (3000 x g). The resulting pore water was then filtered through prewashed 0.45 µ Whatman GF/F microfiber filter and analysed for ammonium (NH_4^+), nitrate and nitrite (NO_x^-) and phosphates (PO_4^{3-}). The pore water collecting in the holes after sediment extraction was used to measure salinity with a hand held refractometre. For subsequent nutrient analysis after extraction, the conventional seawater analysis method by Parsons *et al.* (1984) was used. For estimation of organic carbon (%TOM), a pre-weighed sampled was dried to a constant weight at 70°C and then combusted at 450°C for 4 hours (ashing). The weight loss was then estimated as he %TOM.

6.2.5 Growth

Three common species of mangroves were identified for replanting. These include *R. mucronata*, *C. tagal* and *A. marina*. For *R. mucronata* and *C. tagal* viable propagules were collected directly from the plants and stored under the shade for five days before being planted in 10m by 10m plots of mono and mixed species. *A. marina* seeds were collected from the forest floor, under shades, and raised in nurseries. Transplanting of *A. marina* seedlings was done after three months. Survival of the planted seedlings and propagules were monitored regularly for a period of two years. The increment in height, diameter, internodes and leaves were also monitored.

6.2.6 Data analysis

Data was analysed by non-parametric statistics using STATISTICA 8. Graphical representation were used to display distribution patterns of both adults and juveniles across the intertidal zone. Correlation between regeneration and canopy gap sizes, regeneration and cutting densities was done by Spearman R. To assess the occurrence or absence of regeneration by species under different vegetation zone, the *Cochran Q test*, a dichotomous nominal-scale test based on repeated measures as described by Zar (1999) was used, with the vegetation zone as the groups and the species as blocks. Diversity indices, including the Shannon Wiener, species richness and species evenness were also estimated per vegetation zone.

6.3 Results

6.3.1 Forest characteristics

Five cover classes were identified through photo interpretation (Figure 6.2; Table 6.1). In 1969, area under closed canopy mangrove forest in Tudor creek was estimated at 1,742 ha, 1,525 ha in 1992, and 681 ha in 2005. Human settlements, agricultural land occupied only 5.9 ha in 1969 and increased to over 50 ha by 1992 around Mikindani, Jomvu and Mirarani areas. Differences in open sandy areas or mudflats may arise from (i) differences in tide levels at the time of photography and (ii) plant succession brought about by ecological changes such as sedimentation along the coastline or cutting. Changes in terrestrial vegetation (*bushland*) cover with human settlements and agricultural activities over the period indicate a significant increase in demand for wood products and land for farming and settlements. Depletion of *bushlands* will eventually expose the mangrove to further exploitation. Land use changes are also quite notable in the area with more agriculture activities in 1992 compared to 1969. The population of Mombasa has increased from 247,073 in 1969 to 492,024 in 1993. Currently, the population is estimated at 917,864 (GoK, 2005).

Changes between 1969 and 1992 were moderate and may be associated with population growth reflecting a direct increase in demand on mangrove wood. Changes between 1992 and 2005 were pervasive and reflect not only the population growth, but other possible factors. Information from locals indicate massive mortalities of trees during and after the 1997-1998 El-Niño event mainly due to siltation, which gave rise to the numerous open areas within the forest, which little recovery 11 years later.

Table 6.1: Estimated area of cover types in ha for the period 1969, 1992 and 2005.

Class	*1969*	*1992*	*2005*
Closed canopy mangroves	1,741.83	1,525.97	681.99
Open canopy mangroves	203.40	475.16	519.62
Open sandy areas/mudflats	684.62	554.24	692.74
Water	1,498.41	1,273.95	1,057.28
Bare dark mud	0.00	0.00	169.55

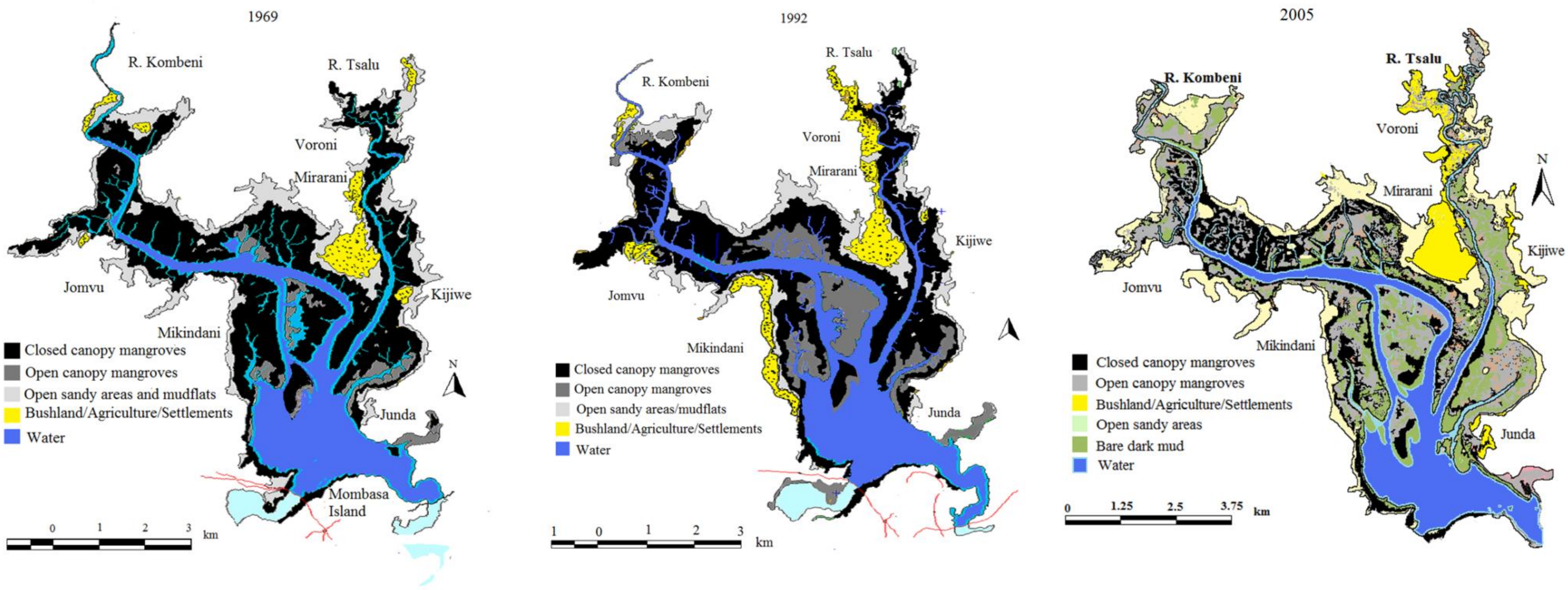

Figure 6.2: Maps displaying the changes in mangrove cover for 1969, 1992 and 2005.

6.3.2 Physicochemical characteristics

The 15 cm sediment cores revealed variations in grain sizes between sites in the landward fringe proper and the intertidal gradient, with the landward fringe having high fractions of larger grain sizes (sandy). The intertidal fringe was almost predominantly composed of fine sediments. However, site conditions along transects did not vary significantly for sediment types (*Kruskal Wallis test*, H (4, N = 200) = 2.004, p = 0.73) for the low and mid tidal fringe. However, we observed that *C. tagal* occupied sites with sandy soils (coarse sediments ≈ 355µm grain size; 75% ± 5% composition) compared to *R. mucronata* which occupied sites with clay to silty sediments (68µm grain size; 64% ± 6% composition).

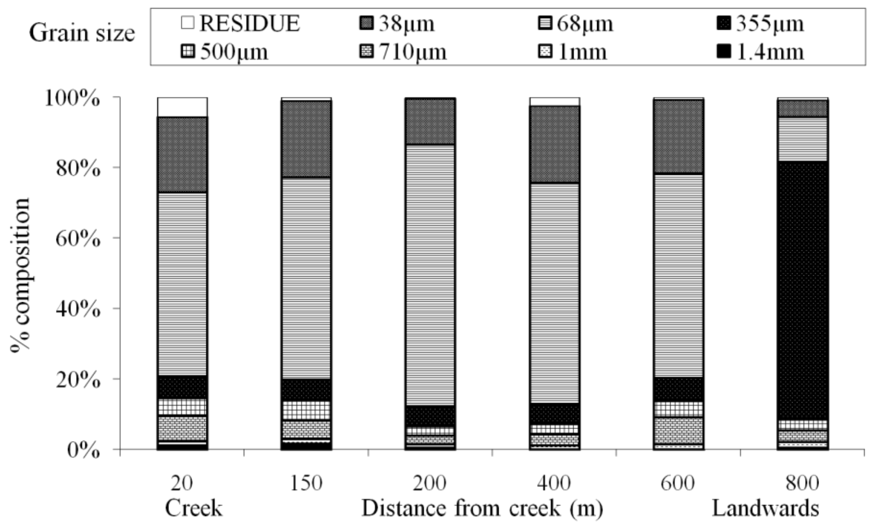

Figure 6.3: Sediment grain size distribution across the intertidal gradient.

The sediment pore-water concentrations for NH_4^+, NO_x^-, PO_4^{3-} and total organic matter (%TOM) for complete and partial sewage exposed sites are shown in table 5.2. There were significant differences in NH_4^+ (*Kruskal-Wallis test*: H (3, N= 124) = 45.36 p < 0.00001), NO_x^- (H (3, N= 124) = 18.48 p = 0.0004), PO_4^{3-} (H (3, N= 124) =29.54084 p < 0.00001) and %TOM (Z = -2.83, p = 0.005) concentrations between sites, with completely exposed sites being characterised by higher NH_4^+ and NO_x^- levels, with higher %TOM content. Higher nutrients levels were observed in the landward and seaward fringe for completely exposed sites and decreasing nutrients levels from landward to seaward fringe for partially exposed sites and an increase in %TOM from landward to seaward (3.8% ± 3.31 – 7.79% ± 1.11). Salinity increased from landwards, attaining values in the range 28‰ – 40‰ in the seaward fringe and 28‰ – 55‰ in the landward fringe during the dry and wet seasons respectively.

6.3.3 Forest composition and structure

The actual density and dispersal of adult trees was highly variable. A total of 16 vegetation zones or associations were observed, depicting large canopy openings, with an obscure zonation pattern. Clustered distribution pattern was common for single species stands and random to uniform dispersion under mixed species (Morisita index; table 6.2). The distribution of species across the intertidal gradient is summarised graphically in figure 6.4. The dominant species, *R. mucronata*, occurred in almost all vegetation zones, *X. granatum* and *B. gymnorrhiza*, seldom occur in pure stands, and were always associated with *R. mucronata*. *C. tagal* is restricted mainly landward and sparsely seaward, where sediments were sandy (coarse) coupled with high elevation. *C. tagal* was characterised by low basal area, D_{130} and height. *S. alba* and *A. marina* were composed of big and tall trees, *S. alba* occupying the seaward fringe, while *A. marina* occurred under mixed stands with *R. mucronata* and *C. tagal*, extending the entire length from landward, mid- to seaward fringes. *A. marina* and *R. mucronata* occupied all zones across the intertidal gradients, unusual for *A. marina*, which was described as displaying a disjunct zonation, occupying only the landward and seaward fringes in Gazi (Ca. 47 km south; Dahdouh-Guebas *et al.*, 2004). However, the presence of channels criss-crossing the intertidal mangrove mudflat also influences species distribution, unlike in Gazi Bay.

The height and D_{130} was defined by species and location across the intertidal gradient, with *A. marina and S. alba* attaining higher heights and bigger D_{130}, while *C. tagal* and *R. mucronata* had relatively lower values, and characteristically lower basal areas. In terms of location across the intertidal gradient, landward or the high tide fringe is dominated by short and smaller trees for all species, with the landward fringe proper being devoid of any vegetation and characteristically occupied by stumps, while the mid tidal range had medium sized trees of each species. The low tide fringe and along edges of channels were occupied by bigger and taller trees of *A. marina*, *S. alba* or *R. mucronata*.

Table 6.2: Density of trees per ha and basal area (in brackets), average D_{130} (± standard deviation), average height (± standard deviation) and the Morisita Index of dispersion (q = 196) by vegetation assemblage.

Vegetation zone	Density/ha (BA m²/ha)	D_{130} (cm)	Height (m)	Frequency (%)	I_δ
A. marina	863 ± 318 (30.00)	19.53 ± 10.26	6.16 ± 2.90	9	***27.44***
C. tagal	867 ± 779 (2.71)	5.39 ± 1.42	1.87 ± 0.27	1	***3.92***
A. marina/	571 ± 229 (10.94)	14.37 ± 9.49	4.19 ± 2.13	1	0.784
C. tagal	557 ± 532 (2.73)	7.91 ± 3.55	1.89 ± 0.39		***2.35***
R. mucronata	1,148 ± 561 (5.31)	6.86 ±4.95	3.48 ± 1.34	28	***88.2***
A. marina /	614 ± 311 (37.18)	24.54 ± 12.27	9.16 ± 3.11	10	***19.01***
R. mucronata	957 ± 911 (4.33)	8.46 ± 5.78	3.93 ± 1.60		***11.76***
A. marina/	200 ± 100 (21.02)	18.67 ± 7.84	5.51 ± 2.00	2	0.80
R. mucronata/	800 ± 200 (0.145)	7.18 ± 3.34	2.44 ± 0.54		***3.90***
C. tagal	600 ± 156 (0.30)	5.05 ± 0.93	2.63 ± 0.75		0.02
A. marina/	400 ± 122 (13.91)	18.62 ± 8.68	6.53 ± 1.87	2	0.80
R. mucronata/	600 ± 346 (4.83)	8.73 ± 7.24	4.62 ± 1.89		0.04
S. alba	533 ± 289 (9.47)	15.49 ± 7.80	5.97 ± 2.52		***4.84***
R. mucronata/	613 ± 421 (2.00)	5.67 ± 2.36	2.53 ± 0.83	23	***65.70***
C. tagal	1,060 ± 638 (3.96)	5.78 ± 3.21	2.83 ± 0.92		***6.21***
R. mucronata/	333 ± 321 (2.96)	6.41 ± 3.19	7.60 ± 11.81	4	0.02
S. alba	1,000 ± 361 (27.36)	14.74 ±12.61	6.88 ± 3.17		***11.90***
S. alba	1,657 ± 1,437 (26.6)	10.66 ± 6.58	4.39 ± 1.44	20	***62.96***
S. alba/	950 ± 71 (18.06)	16.24 ± 6.22	8.00 ± 3.22	0.05	0.16
A. marina	100 ± 141 (4.84)	17.95 ± 3.16	4.00 ± 1.15		0.014

6.3.4 Regeneration patterns

As a general observation, regeneration is higher for the urbanised Kombeni creek, and lower for the rural Tsalu creek. However, regardless of location, regeneration patterns under different vegetation associations or mix and across the intertidal gradient displayed similar distribution patterns. The regeneration patterns are succinctly summarised in table 6.3 and figure 6.5. Regeneration composition by species under different vegetation associations did not differ significantly for both seedlings and saplings (*Cochran's Q test*; χ^2 = 17.667, p = 0.064 and χ^2 = 11.163, p = 0.36 respectively), with species mix within an association not always restricted to overstorey species composition. This observation implies a seed/propagule dispersal and recruitment pattern that promotes colonisation of distant stands by species other than the overstorey species. This has allowed a diverse range of species mix, indicating the presence of suitable conditions for establishment of propagules in most locations.

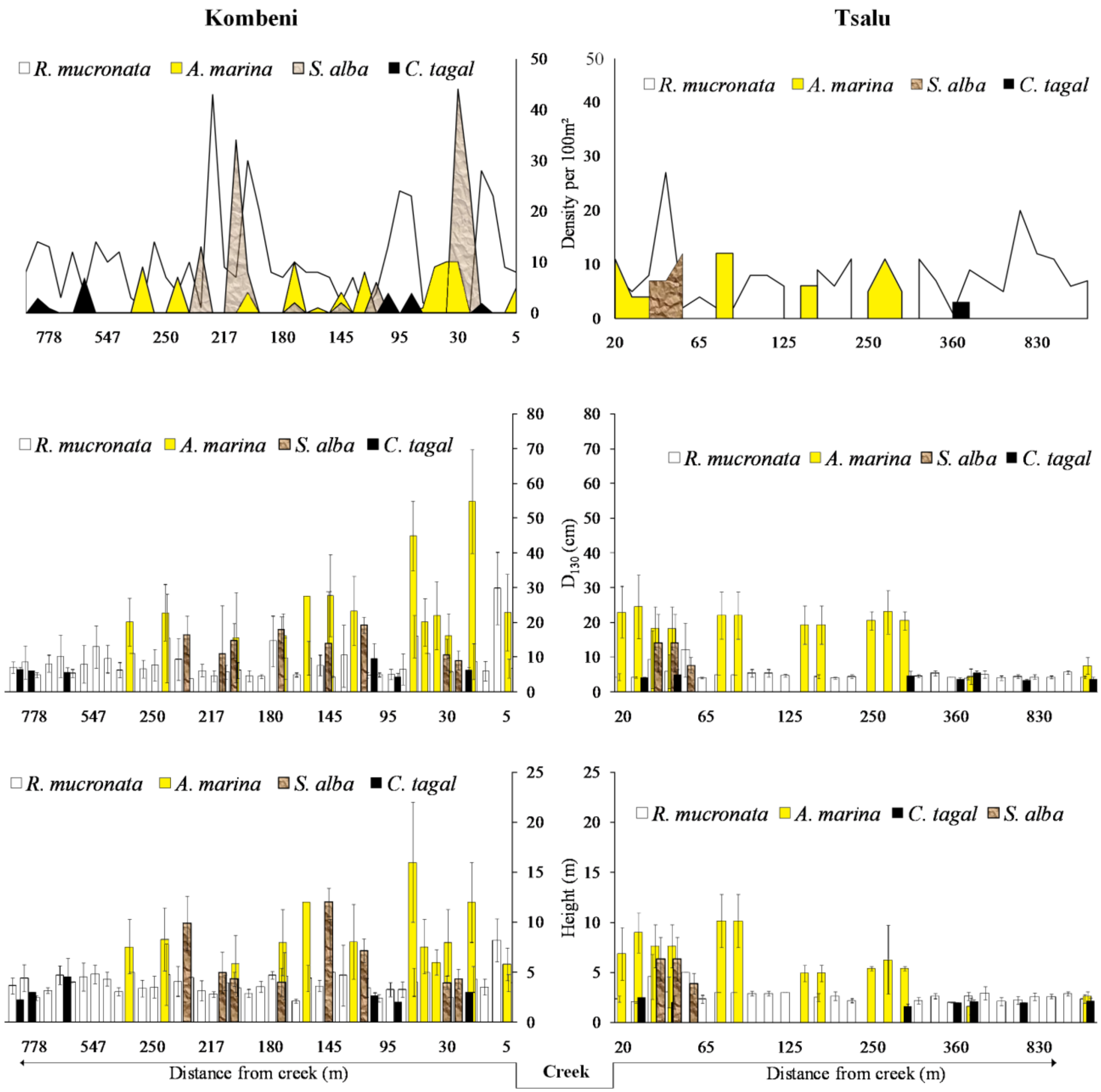

Figure 6.4: The distribution of adult trees per 100m² plots in relation to distance from the creek, with the corresponding height and diameter distributions. Data presented was obtained from five transects of ten plots in each transect.

R. mucronata, the most widely distributed species had high seedling and sapling densities, and like *C. tagal*, saplings were in higher density than seedlings. *R. mucronata* seedlings and saplings occurred in the understorey under all cover types (species mix) and inundation, conferring advantages to this species under the current disturbance regime (Table 6.3). This distribution favours *R. mucronata* establishment in relation to other species, such as *A. marina* and *C. tagal*, which are restricted in distribution. Seedlings of *A. marina* occurred in higher densities than saplings (Figure 6.5), a phenomenon brought about by either high seedling mortality or the effects of '*mast production*', characterised by variable seed production over years, ranging from years of seedlings paucity to years of seedlings abundance. This is an adaptation common in some terrestrial plants to control seed predation or cushioning against adverse conditions (Zlotin and Parmenter, 2008).

Table 6.3: Seedlings and saplings density ha^{-1} in the understorey (Ust.) and gaps. Values in brackets indicate the Morisita Index of dispersion (q = 196).

Vegetation zone		***R. mucronata***		***A. marina***		***C. tagal***	
		Seedling	Saplings	Seedlings	Saplings	Seedlings	Saplings
A. marina	*Ust.*	5 (0.0014)	8 (0.00038)	27 (0.19)	392 (*170.28*)	-	-
	Gap	51	68	495	49	-	-
A. marina/ C. tagal	*Ust.*	2 (-)	12 (0.00033)	-	12 (0.12058)	4 (0.0027)	38 (0.14)
	Gap	-	-	71	17	85	41
C. tagal	*Ust*	-	-	-	-	14 (0.096)	65 (0.60)
	Gap	-	-	-	-	-	-
R. mucronata	*Ust.*	1,378 (*110.57*)	4,303 (*111.27*)	6 (0.0078)	-	7 (0.025)	16 (0.037)
	Gap	1,251	2,263	327	824	37	17
R. mucronata/ A. marina	*Ust.*	188 (*1.51*)	360 (0.665)	784 (*165.42*)	16 (0.27)	3 (0.0099)	22 (0.067)
	Gap	129	322	302	202	-	-
R. mucronata/ A. marina/ C. tagal	*Ust.*	16 (0.015)	45 (0.012)	-	-	27 (0.38)	44 (0.28)
	Gap	176	578	161	-	200	185
R. mucronata/ A. marina/ S. alba	*Ust.*	5 (0.0011)	11 (0.00066)	37 (0.37)	-	-	-
	Gap	-	-	-	-	-	-
R. mucronata / B. gymnorrhiza	*Ust.*	66 (0.254)	356 (0.76)	-	-	-	-
	Gap	-	-	-	-	-	-
R. mucronata/ C. tagal	*Ust.*	153 (*1.35*)	546 (*1.79*)	-	-	555 (*161.89*)	984 (138.68)
	Gap	46	305	15	-	254	417
R. mucronata/ S. alba	*Ust.*	21 (0.026)	44 (0.012)	-	1 (0.00058)	-	-
	Gap	-	-	-	-	-	-

The distribution of seedlings and saplings varied in densities with canopy cover and species composition with location across the intertidal area (Figure 6.5). Regeneration densities correlated significantly with cutting intensity (*Spearman-R* = 0.448 (N = 196), t(N-2) = 6.98, $p < 0.001$), and subsequent canopy cover (*Spearman-R* = 0.228 (N = 196), t(N-2) = 3.26, $p < 0.001$), medium sized gaps (10 – 50% cover ≈ 10 – 50m²), with lower to medium cutting intensities having higher seedling and sapling densities compared to smaller (less than 10% cover ≈ less than 10m²) and larger gap sizes (greater than 50%

cover ≈ greater than 50m²; Figures 6.5 and 6.6). *R. mucronata* and *C. tagal* seedlings occupied small to medium gaps (Figure 6.5), where cutting is low to moderate, while more *A. marina* seedlings and saplings occupied extended gaps.

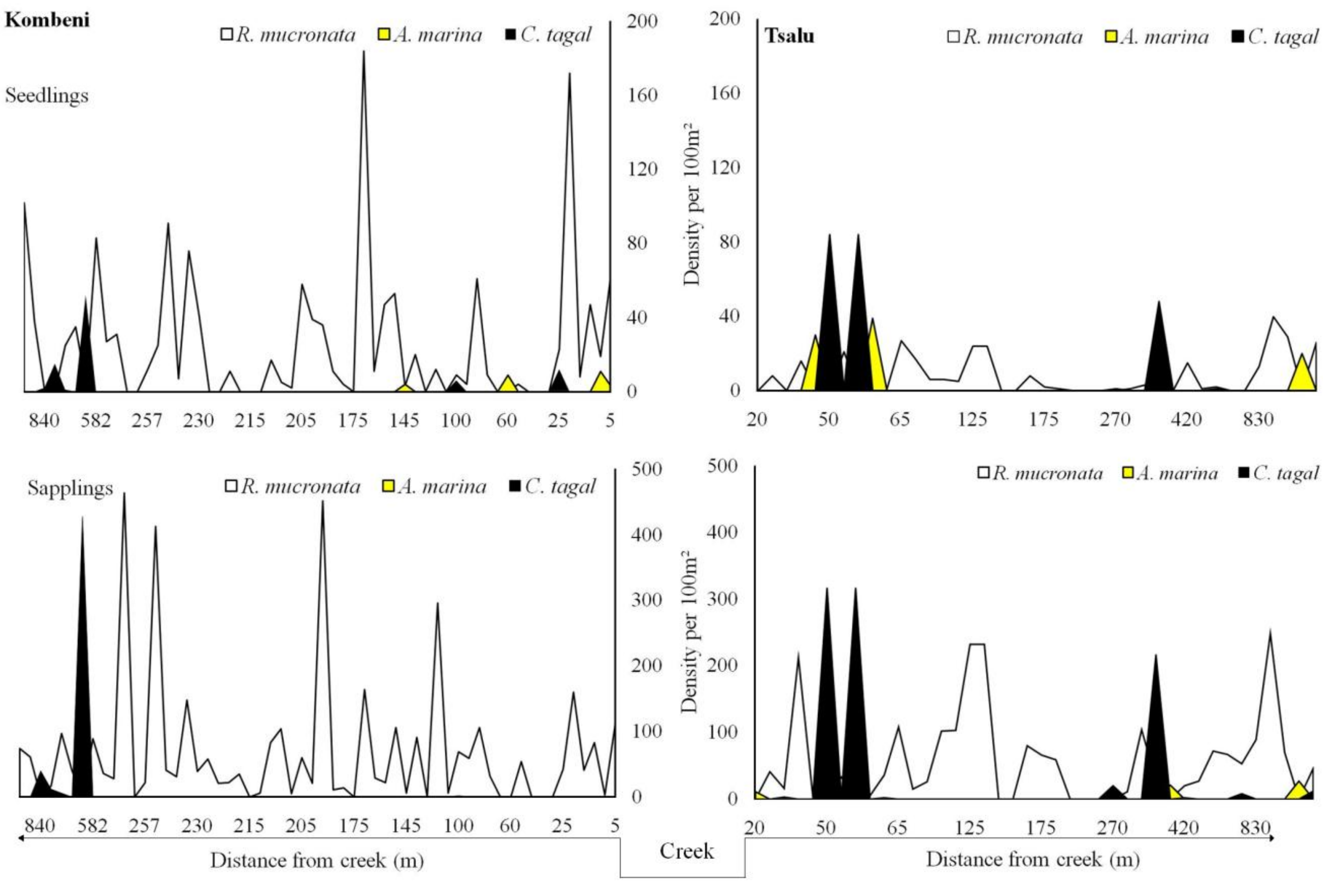

Figure 6.5: The distribution of seedlings and saplings per 100m² plots in relation to distance from the creek. Data presented was obtained from five transects of ten plots in each transect.

Across the intertidal gradient, thus inundation, seedlings and sapling densities and species composition varied (Figure 6.5). *A. marina* and *R. mucronata*, with low densities of *C. tagal* occurred near the creek (the low-tide range). The mid tidal range was dominated by *R. mucronata* and low densities of *C. tagal*. Densities of *C. tagal* seedlings and saplings were higher landwards (the high-tide range), and where elevation was high. The landward fringe proper lacked any form of regeneration for any species. *R. mucronata* seedlings were present regardless of tidal position or vegetation zone. However, as a general observation, *A. marina* seedlings colonised enlarged or extended canopy gaps regardless of vegetation or tidal inundation. Sampled extended or enlarged gaps were mainly composed of low densities or no adult trees, with the presence of colonising seedlings of mainly *A. marina*, *R. mucronata*, *C. tagal*, and only in one gap, *S. alba*.

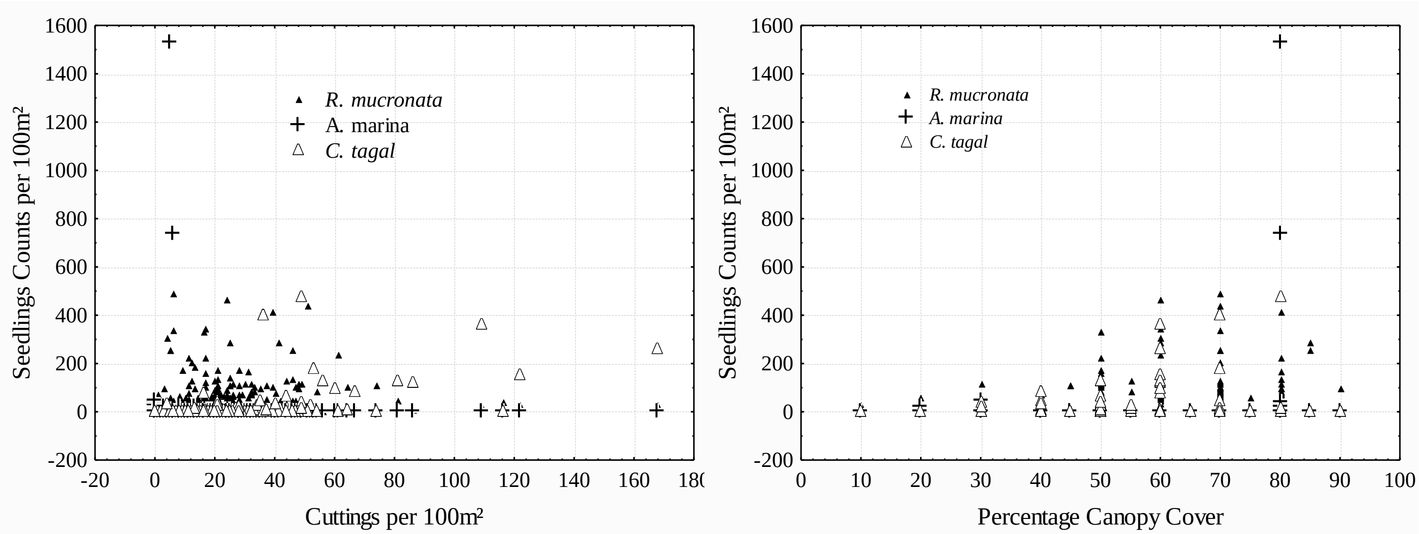

Figure 6.6: Scattergrams showing the distribution of seedlings and saplings by species per 100m² plots in relation to canopy cover (Spearman-R = 0.228 (N = 196), t(N-2) = 3.26, $p < 0.001$) and cutting densities (Spearman-R = 0.448 (N = 196), t(N-2) = 6.98, $p < 0.001$).

6.3.5 Seedling growth and mortality

The mortality rates for replanted plots is summarised in table 6.4 and the growth rates in table 6.5. Mortality was high in the first year (60% – 79% for single species plots, and 87% – 91% for mixed species plots) and lower for the second year (35% – 87% for single species and 38% – 81% for mixed species). There were significant differences in mortalities between species (*Kruskal-Wallis test*: H (28, N = 84) = 57.22, $p < 0.001$), *R. mucronata* recording the highest (79 – 91%) for both single and mixed species plots in the first year and lower rates in the second year (35 – 43%). However, in both years, *C. tagal* and *A. marina* recorded high mortality rates, mostly occurring in the rainy season as a direct impact of siltation. During the rainy season, *A. marina* seedlings stems were either dislodged or broken as a result of run-off loaded with sediments, while *C. tagal* stems were completely buried owing to their characteristic height (Figure 6.7c).

Table 6.4: Summary of replanted seedlings mortalities over a two year period for pure stands and mixed plots.

	Species	Initial counts	after 1 year		after 2 years	
			Counts	% Mortality	Counts	% mortality
Mono species	*A. marina*	800	256	68	33	87
	C. tagal	800	322	60	95	70
	R. mucronata	800	170	79	110	35
Mixed species	*R. mucronata*	800	72	91	41	43
	C. tagal	800	101	87	25	78
	R. mucronata	800	107	87	66	38
	C. tagal	800	105	87	20	81

Table 6.5: Summary of average annual increments (average ± standard deviation) in growth of replanted seedlings. Values in brackets are the relative growth rates (increment as percentage of initial measure).

Species	Diameter (cm)	Height (cm)	Internodes (numbers/yr)	Branches (numbers/yr)
A. marina	0.48 ± 0.15 (57 ± 21)	29 ± 9 (65 ± 8)	10 ± 2	13 ± 5
C. tagal	0.47 ± 0.14 (9.97 ± 7.37)	3.10 ± 2.81 (72 ± 7)	3 ± 1	7 ± 1
R. mucronata	0.96 ± 0.01 (59 ± 17)	24 ± 3 (50 ± 9)	6 ± 2	6 ± 1

During the NE–monsoon season, many seedlings, particularly *A. marina* wilted, probably the impact of high-temperature injury, displaying necrotic lesions, particularly on stems and hypocotyls, followed by subsequent death of the seedling. While the SE – monsoon season was prone to levels of siltation, which in some cases burial of *A. marina* and *C. tagal* seedlings or the entire stems of *R. mucronata* and *C. tagal* occurred (Figure 6.7a and 6.7c). During the two years, no prop roots or pneumatophores were observed forming, while *R. mucronata* produced buds by the second year, which never developed (Figure 6.7b). Diameter and height increments were significantly different between species ($p > 0.05$). *R. mucronata* and *A. marina* were observed to have high growth rates compared to *C. tagal*. However, in terms of relative growth rates, *C. tagal* displayed a high relative height increment. Within the forest, *C. tagal* height ranged between 2m – 3m, and D_{130} between 2.5cm – 10cm, compared to 24.54 ± 12.27cm D_{130} and 6 m – 15m height for *A. marina* and 8.46 ± 5.78 D_{130} and 5 – 12m height for *R. mucronata*, indicating a general slow growth rate for *C. tagal*, which is mostly located in the landward fringe, where inundation is infrequent and tidal heights lower. However, height measurements were confounded by the effects of accreting sediments during the rainy season.

Figure 6.7: **(a)** *R. mucronata* buried by silt **(b)** *R. mucronata* producing flowering buds by the second year **(c)** a buried *C. tagal*, only leaves are visible, observe the crab hole nearby **(d)** measuring growth in a *R. mucronata* plot.

6.4 Discussion

Regeneration, thus recruitment of seedlings, is not restricted to overstorey conspecifics or by vegetation zones/species mix. This implies good seed dispersal ranges, good site conditions within the ranges of species tolerance for establishment, and thus a good capacity for natural regeneration. These are typical traits reflecting the dynamic and resilient nature of the mangrove environment that has withstood disturbances over time, promoting species mixtures (Ball *et al.*, 1988). The patterns of tree regeneration further implies that it is unlikely the current tree species distribution pattern across the intertidal gradient will be maintained, and in the absence of a major disturbance event, *R. mucronata* will be expected to increase in abundance throughout the forest given the current status and prevailing conditions. The species mix and composition in the forests is important in enhancing ecosystem productivity and stability, and confer the forest with a broader niche differentiation, promoting increased total resource use, providing a variety of trophic pathways likely to support richer faunal communities as compared to single species stands (Bosire *et al*., 2006; Bosire *et al*., 2008). This species mix further elevates the perturbation threshold for the mangroves of Tudor creek, increasing its resilience and survival.

The distribution of trees across the intertidal mudflats reflects the general trend of environmental physicochemical parameters reported by studies in other parts of the world (Sherman *et al*., 2000; Ball, 2002; Lovelock and Feller, 2003; Lovelock *et al*., 2003). Mangroves align themselves along edaphic physicochemical and hydrological gradients such as salinity and tidal submergence (Bosire *et al*., 2008; Krauss *et al*., 2008) in addition to nutrients limitations (Lovelock *et al.*, 2003). Landward areas of higher salinities and higher nitrates being characterized by shorter and smaller trees, compared to the taller and bigger seaward fringe trees. However, the proximity to human settlements, coupled with ease of access, the landward fringe may be more exposed to human pressure, resulting in the observed patterns in size distribution.

6.4.1 Forest cover changes

The major factors that seem to fuel the forest cover changes in Tudor creek range from unregulated harvesting, a high population and demand for wood products and siltation (Chapter 3, 4; Mohamed *et al*., 2008). The effect of unregulated harvesting and high wood demands were tackled separately in chapter 4 (Mohamed *et al.*, 2008). Siltation is considered an important factor in mangrove ecology (Hutchings and Saenger, 1987; Ellison, 1998). This is mainly due to their location on low-lying continental coastlines, where sediments supply is regarded as 'essential substrate accretion', and is important in buffering against erosion and the perceived sea level rise (Wolanski and Ridd, 1986; Ellison, 1998; Wolanski and Chappell, 1996). Under normal circumstances, sediments may deposit at an estimate rate of 0.5 cm yr^{-1} – 1.5 cm yr^{-1} allowing mangroves to acclimate (Hutchings and Saenger, 1987; Ellison, 1998). Sediment starvation of this zone as a result of reduced sediment supply may lead to erosion in some susceptible areas (Bird *et al*., 2004. Thampanya *et al*., 2006). However, impacts of sedimentation may be positive or negative. Positive impacts include creation of new areas for colonisation (Thampanya *et al*., 2002), while negative impacts arise out of sudden sedimentation after heavy rains, that lowers the growth and vigour of mangroves if aerial roots are blocked, causing mortality (Terrados *et al*., 1997; Ellison, 1998)

The sedimentation rates that occurred during the single ENSO event are unknown, but studies from the neighbouring Port Reitz creek (Mwache creek), estimates a 1.4m deposition of terrigenous sediments in the middle section of the creek, and 0.2m deposition in the outer edges during the same period, causing significant mortalities of mangroves (Kitheka *et al*., 2003). No recovery has occurred in some of these areas 11 years after the siltation event (Mohamed's personal observation). We speculate that a similar scenario might have occurred in Tudor creek, with variable spatial patterns. This may have caused high mangrove mortalities, resulting in the emergence of the extended open areas. We hypothesize that, these openings altered edaphic conditions, including higher soil salinities and temperatures, particularly in the less frequently inundated areas, altering species distributions. Prevailing conditions after the changes may have favoured the establishment of *A. marina*, characteristically the most salt tolerant of all the mangrove species, identified as a pioneering species (Ball *et al*., 1987; Osborne and Berjak, 1997), producing the highest number of propagules and high success rates in colonising open substrates (Thampanya *et al*., 2006), with root initiation and subsequent establishment hardly hampered by extreme saline conditions (Ye *et al*., 2005).

A. marina stabilised the loose sediments, creating opportunities for other species to colonise, which later out-compete *A. marina* as the canopy closes and sesarmid crabs colonised the understorey. Under full canopy cover, *A. marina* saplings were virtually absent possibly due to light intolerance. It is also possible that crab predation for *A. marina* seedlings intensified as the canopy cover increased, as reported in other mangroves globally, mainly in the mid-tidal ranges (Smith III, 1987a, b; Osborne and Smith III, 1990; Farnsworth and Ellison, 1997a). The PUMPSEA study also reported a higher biomass of sesarmid crabs under *A. marina* cover in Mikindani (Tudor creek site impacted by sewage) than in Gazi Bay and Shirazi mangroves (PUMPSEA, 2007c). Thus *A. marina* is gradually replaced by other species in the canopy as growth creates cover, a typical multi-phase vegetation regeneration characteristic of mangrove forests (Ferwerda *et al*., 2007). This pattern can be observed under enlarged canopy openings, where high densities of *A. marina* and *R. mucronata* seedlings and saplings were present under all vegetation mixes (Table 6.3; Figure 6.8c). It was also under these gaps that the only seedlings and saplings of *S. alba* were encountered in a *R. mucronata* zone in the mid-tidal range (Figure 6.8a), a peculiar observation due to location. Table 6.6 gives a summary of diversity indices for the vegetation zones for adults and juveniles, showing higher species richness under zones with *A. marina*, and greater diversity or species mix for juveniles than for adults under most zones.

6.4.2 Regeneration

With exception of *R. mucronata* saplings (4,303 ha^{-1}), seedling and sapling densities were on average low for all species. The seaward fringe, *S. alba* zone and the landward fringe proper had particularly low regeneration levels. Bosire *et al*. (2006) also observed low regeneration for *S. alba* zone for Gazi Bay mangroves (ca. 47 km south). The variability in regeneration across the intertidal gradient may reflect the diverse factors involved including: the hydrological regime with comparatively stronger mechanical effects of tides and wave currents in the low tidal fringe (*S. alba* zone); accretion and siltation (the landward proper); absence of adequate seed bearing trees (mainly in large gaps and the high tidal fringe including cutting intensity; Figure 6.8b); influence of gaps and gap sizes (lack of root masses to allow for stranding of propagules) (Duke, 2001; Sherman *et al*., 2000; Di Nitto *et al*., 2008; Mohamed *et al*., 2008); and importantly the species adaptation or/and the seedling/propagule predation rates (Dahdouh-Guebas *et al*., 1999; Clarke and Kerrigan, 2002) and dispersal ranges of the species involved (Osborne and Berjak, 1997; Tomlinson and Cox, 2000; Delgado *et al*., 2001). In addition, the stand age is an important factor in determining levels of regeneration, and in particular, recruitment rates of saplings, which increase with stand age (Bosire *et al*., 2006; Bosire *et al*., 2008). This may be a further indication of a recovery phase within Tudor creek, characterised by lower saplings for all species except for *R. mucronata*. However, *R. mucronata* is also quite tolerant to siltation (Thampanya *et al*., 2002) compared to the other species, and possesses a longer propagule, with ample nutrient storage that allows longer survival for the species propagule even in the understorey (McKee, 1995).

Table 6.6: Species diversity indices, evenness and N_{inf} for adult and juveniles under the different vegetation zones.

Zone	Adults				Juveniles			
	Species	J'	H'(log*e*)	N_{inf}	Species	J'	H'(log*e*)	N_{inf}
A. marina	1	****	0	1	4	0.39	0.54	1.22
A. marina/C. Tagal	2	0.97	0.67	1.67	3	0.91	1	2
A. marina/C. tagal/X. Granatum	3	0.83	0.92	1.83	1	****	0	1
A. marina/S. alba	2	0.65	0.45	1.2	0	****	0	****
B. gymnorrhiza	1	****	0	1	2	0.98	0.68	1.69
C. tagal	1	****	0	1	0	****	0	****
R. mucronata	1	****	0	1	4	0.23	0.32	1.09
R. mucronata/A. marina	2	0.99	0.69	1.91	5	0.45	0.72	1.64
R. mucronata/A. marina/C. Tagal	3	0.97	1.06	2.18	2	0.52	0.36	1.13
R. mucronata/A. marina/S. alba	3	0.98	1.08	2.37	3	0.57	0.63	1.26
R. mucronata/B. Gymnorrhiza	3	0.63	0.69	1.35	3	0.42	0.46	1.19
R. mucronata/C. tagal	2	0.91	0.63	1.49	2	0.99	0.69	1.90
R. mucronata/S. alba	2	0.76	0.53	1.29	3	0.58	0.64	1.41
R. mucronata/X. granatum	2	0.95	0.66	1.6	3	0.78	0.86	1.49
S. alba	1	****	0	1	1	****	0	1

Figure 6.8: **(a)** *S. alba* seedlings and saplings under extended gaps **(b)** the landward fringe with snags indicating mortality of adult trees and no regeneration **(c)** High density of *A. marina* seedlings in an extended gap receiving raw sewage **(d)** replanted *A. marina*, observe the variable growth rates, with taller and more bush saplings in the back, also observe the terracing and farming on the cliff in the back, with houses are also built on the cliff.

Low regeneration along the seaward and landward fringes (Figures 6.7, 6.8b, d) has important implications on the forest with the prospects of global climate change and associated sea level rise (Duke, 2001; Gilman *et al.*, 2007; Alongi, 2008; Gilman *et al.*, 2008). Rising sea-level, necessitates that mangrove forests migrate inland with subsequent die off along seaward margins (Duke, 2001; Gilman *et al.*, 2008). In Tudor creek this may not be an option due to the steep cliff that borders the creek on the landward fringe. However, annual sea level rise is estimated at 0.2 cm yr^{-1} (Bates *et al.*, 2008), quite tolerable by mangroves considering established sedimentation rates of between 0.1 - 0.2 cm yr^{-1} (Ellison, 1998; Kitheka *et al.*, 2003). However, even conservative predictions indicate acceleration in the rate of sea level rise, which may potentially result in imminent exhaustion of the replenishment capabilities of seaward fringe stands, which may lead to a general collapse of mangrove stands in relatively exposed locations, once the rate of gap replenishment is exceeded by tree death from sea level encroachment (Duke, 2001). Modelling the impacts of sea level rise in Gazi Bay, Di Nitto *et al*. (2008) predicts that a relatively modest rise in sea level within a time span of 20 years could affect the distribution pattern and the specific proportion of the juvenile vegetation layer, leading to notable regional floristic modifications.

6.5 Conclusion

The species composition of mangrove forests is dependent at least in part on a balance between large scale and small scale disturbances. This potentially determines species mix. Therefore zonation, is not strictly species exclusive within a mangrove forest, and may depend on, or reflect a stage in the stand growth dynamics under a specific disturbance regime. Seed production within the mangrove stand was found to be abundant and natural regeneration in moderately exploited or disturbed sites (small gaps) was generally observed to be good, especially in the absence of persistent stressors mostly from human factors as reported by Hamilton and Snedaker (1984) and Bosire *et al*. (2008). Natural regeneration may confer higher diversity to the stand in terms of species selection and more functionality to the emergent forest in the long run by enforcing and diversifying trophic links through fauna flora interactions, nutrient cycling etc...... (Bosire *et al*., 2008). However, seedling dispersal, establishment and survival may be a major limiting factor slowing recovery in a disturbed mangrove, especially when disturbance is an ongoing process. Under these circumstances, seedlings of more adapted species will establish easily, replacing less tolerant species. Furthermore, the characteristics nature of poorly managed peri-urban ecosystems display a progressively degrading character, lowering diversity, diminishing ecosystem goods and services, limiting, if not inhibit the system's capacity to recover. Thus a lowered resilience for peri-urban mangrove forests.

7. Synthesis
Conclusions and Recommendations

…a resilience approach is about weighing up options, keeping options open, and creating new options when old ones close……………..



….in today's world, that's more important than ever.
(Walker and Salt, 2006)

7.1 An overview of the findings

Mangroves are occupants of a harsh intertidal environment. They are often increasingly proximate to rapidly growing human settlements. Mangroves are subjected to daily tidal fluctuations in temperature, water and salt exposure, and varying degrees of hypoxia, necessitating a high degree of adaptations with relevant tolerance ranges. Our observations in the preceding chapters, on the status and nature of the vegetation structure (chapter 3), socio-economic paradigm (chapter 4), productivity, phenology (chapter 5) and regeneration status (chapter 6) of the peri-urban mangroves of Tudor creek, serve to confirm this assertive, adaptive, and resilience of mangrove ecosystems. These adaptations catalyse responses to diverse and pervasive changes over time, ensuring survival in spatial and temporal scales. However, a limit, '*a breaking point*' or a threshold exists, around which shifts in forest structure and function are imminent (Walker and Salt, 2006). This causes spatial and temporal variability in species composition, regeneration capacity, nutritive quality of productivity material and variable recovery and growth rates after perturbations. The prospects of system collapse is often imminent should the disturbance frequency and intensity persist or multiply over spatial and temporal scales.

The main extrinsic factors impacting Tudor creek mangroves include raw domestic sewage pollution, siltation, and unregulated wood harvesting. These pose a variable spatial and temporal stress factor, and in concert, impact important goods and services offered by mangroves. The main services that are affected include (i) the capacity to filter pollutants (ii) provisioning of a habitat for a diverse fauna, (iii) wood products and proteins to local communities (fish, molluscs and arthropods) (iv) in addition to coastal protection (erosion and accretion of the intertidal area). Mangrove goods and services are evidently of major importance to both emergent and established peri-urban areas. These are exceptionally important where urban waste management is non-existent or inadequate and the energy and economic needs of the 'masses' are wanting. This creates a 'race' towards exploiting a dwindling resource to satisfy an ever growing urban demand. The poverty levels and the 'common' nature of mangrove ecosystems absolves individuals and community from responsibility grossly under-valuing the resource (Walters *et al*., 2008; Bosire *et al*., 2008).

7.2 Productivity and pollution

The mangroves of Tudor creek are still productive despite the state of degradation. Due to raw domestic sewage pollution, the sediment nutrient levels were high. However, low levels of phosphates in the sewage effluents implies a possible phosphate limitation. Total leaf nitrogen levels and the C/N ratios varied with species and with the leaf $\delta^{15}N$ signature reflecting sewage exposure levels. This was more pronounced for *R. mucronata*. Elevated total leaf nitrogen levels may have important nutrients cycling implications in the mangrove ecosystem, affecting litter degradation. However, the complex mangrove ecosystem may not always respond uniformly to nutrients enrichment.

Whereas domestic sewage pollution, serves as an input of nutrients in an otherwise nutrient deficient system, its affects on the important sediment microbial community may not be positive. This means a deficient nutrient recycling within the system. This has been observed as a build up of NH_4^+ in the sediments. This affects both the '*functional'* and '*response*' diversities of the systems, thus lowering the systems resilience.

In agreement with the global recognition of mangroves as highly productive intertidal forests (Wafar *et al*., 1997; Bouillon *et al*., 2008; Dittmar *et al*., 2008), the mangroves of Tudor creek are important in litter production (chapter 5). Litter from mangrove trees (leaves, propagules and twigs) and subsurface root growth constitute significant inputs of detritus material to the mangrove ecosystem (Figure 7.1; Holguin *et al*., 2001). The decomposition of this detritus material contributes significantly to the organic carbon in mangrove sediments (Alongi, 1998), and is reported to form an important base of a detritus food web. Litter fall is estimated to represent about one third of the net primary production in mangroves (Alongi *et al*., 2005), and a significant amount of this litter is usually flushed out as detritus material from the forest floor into bays, lagoons and even into the open sea forming a source of nutrients (Flores-Verdugo *et al*., 1987; Bouillon *et al*., 2008). The magnitude of this exported detritus from mangrove areas depends on geophysical processes: the size of the mangrove ecosystem, frequency and duration of tides, sizes of draining channels, frequency and magnitude of rains, inflow of fresh water and the presence of mangrove crabs (Twilley *et al*., 1997; Holguin *et al*., 2001). The Tudor creek mangroves are productive despite the temporal and spatial degradation that has occurred, fitting the characteristics of mangrove ecosystems that occupy the same latitude.

This study assessed the influence of domestic sewage pollution on the litter fall rates of three common mangrove species, *R. mucronata*, *S. alba* and *A. marina*. However there appears to be no difference in litter fall rates between the fully sewage exposed site and partially exposed. This may not be a peculiar observation as Tam *et al*. (1998) reported a similar observation in Hong Kong. However, our study presents an experimental design that assess responses based on variable field conditions. The set-up constitute a *pseudo-replication* along a pollution gradient, and may not have fully differentiated the impacts of multiple factors such as harvesting, siltation and sewage exposure working in concert. At the tree level, the higher external supply of N relative to P in the sewage effluent, may signal a potential relative deficiency of P in the system, which affects the physiology of individual trees (Lovelock *et al*., 2006) and may impact productivity.

Domestic sewage pollution constitutes an input of nutrients into the mangrove ecosystem, which are recycled by sediment microbial communities, usually via an important and efficient process (Alongi, 1994; Holguin *et al*., 2001). This sediment microbial community structure and function is directly responsible for the well being of the ecosystem (Holguin *et al*., 2001). The alteration of tropical mangrove sediments, either through siltation, canopy-gap creation or sewage pollution (as observed in our study), nearly always causes changes in the densities composition and growth cycles of the microbiota (Findlay *et al*., 1990; Holguin *et al*., 2001). Sewage potentially shifts a healthy decomposing aerobic-anaerobic system of the mangrove to a complete anaerobic system, which is less efficient and slow in recycling nutrients, resulting in the build-up and release of toxic sulphides as was observed by the PUMPSEA project (2007b) in Mikindani. In Mikindani, high rates of ΣCO_2 production at sewage impacted sites were observed. This was interpreted as being indicative of a system under stress due to the presence of easily degradable organic matter, implying that the system in the natural setting does not get enough time to stabilize since it is dozed continuously with sewage containing labile organic matter. Meanwhile, in Tanzania, sewage was reported to have antagonistic effects on cyanobacteria diversity and microalgae abundance, by promoting microalgae abundance, while decreasing cyanobacterial diversity (PUMPSEA, 2007b).

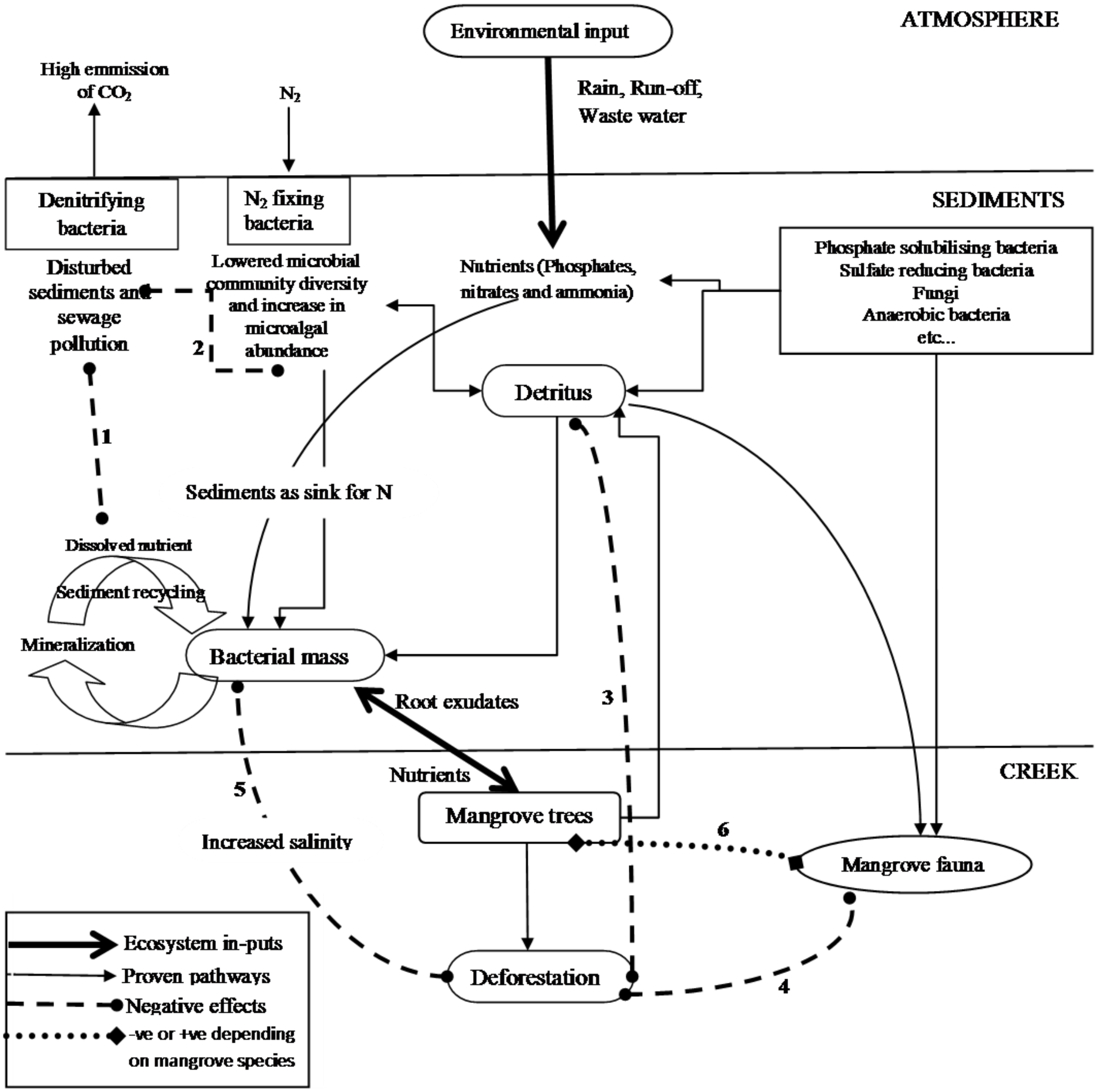

Figure 7.1: A flow chart summarising the major disturbances in Tudor creek. Major effects of the disturbances are the possible inhibition of sediment microbial activity as a result of harvesting (gap creation), siltation and sewage pollution (**1, 2** and **5**). The negative effects of harvesting include the lowered provision of detritus (**3**) and reduction in bacterial activity as a result of high temperatures and salinity (**5**); while the negative effects of domestic sewage pollution include the reduction in cyanobacteria diversity, thus affecting the nitrogen cycle (**2**), resulting in elevated ammonium concentration which may potentially affect mangrove growth. The mangrove fauna composition and distribution on the other hand will be negatively affected by the harvesting activities (gap creation), especially the enlarged gaps (**6**). This will result in lowered degradation of detritus due to lowered burrowing into the sediments (aeration).

In Tudor creek, the influence of sewage on litter fall was observed as increase in total nitrates in leaves. Similar responses were reported by Onuf *et al*. (1977) and Feller *et al*. (2002) for mangroves exposed to sewage and nutrients enrichment. Response variability between species (implicating mechanism of nutrients use efficiency (NUE) by species) and level of sewage exposure has been reported (Lovelock and Feller, 2003; Alongi *et al*., 2005). More exposed sites tend to have higher leaf total nitrates. This presents a functional trait in mangroves that evokes considerable trophic links that directly impacts nutrient cycling. Since litter fall returns a greater fraction of nutrients from the tree canopy to the soil, eutrophication may markedly accelerate nutrient cycling via litter fall and decomposition and enhance organic matter provisioning to mangrove food webs (Wanek *et al*., 2007). However, the compounding effects of the major limiting nutrients, N and P (Feller *et al*., 1999; Feller *et al*., 2003a, b; McKee *et al*., 2002), coupled with the status of the sediment microbial communities and presence of crabs, play an important role in nutrient dynamics and possibly influence vegetation responses such as growth and productivity. Mangrove crabs are important in lowering soil sulphate and ammonium concentrations, probably by aerating the substrate by burrowing, or maceration of leaf litter, making it more amenable to microbial attack (Holguin *et al*., 2001). Further studies are essential for verification of these effects on productivity within Tudor creek. Therefore, in a mangrove ecosystem, mere nutrients enrichment or sewage pollution may not always elicit similar or direct responses. The response may depend on the status of sediment microbial community, the structural condition of the forests and the presence of other disturbances (Figure 7.1). Furthermore, the nutrient load and the ecosystem carrying capacity may vary between mangrove ecosystems, eliciting negative effects when this capacity is exceeded.

We hypothesize that multiple disturbances may trigger multiple synergistic and antagonistic ecosystem responses, reminiscent of natural complex systems. These effects may not always be linear or direct with respect to the inducing agent. These observations imply that the growth, productivity or recovery in these sewage polluted sites will be negatively impacted. This is because the success of structural recovery via growth, and functional recovery via productivity also depends on the preservation of the benthic microbial communities and their geochemical environment (Alongi, 1994; Holguin *et al*., 2001). The elevated availability of organic matter within the Tudor creek mangroves (Mikindani) may have affected the biogeochemical cycle and resulted in the retention of high amounts of ammonium in sediments. This may in turn have affected both tree growth and possibly productivity of mangrove trees in spatial scales.

7.3 Forest status and regeneration

The Forest Structure: The Tudor creek mangrove forest is dominated by *R. mucronata*, with characteristically large canopy gaps which influence regeneration rates. Gap sizes in the range 10-50m² are characterised by optimal regeneration levels compared to larger gaps. The present forest structure reflects the effects of unregulated harvesting based on demand by local people, having a haphazard distribution of tree sizes. This may be viewed as a trait of a fragile system that may easily undergo species shifts and lowered recovery rates.

Natural Regeneration: Regeneration occurred over wide areas, indicating good dispersal ranges, with a pattern not always reminiscent of the adult population. This promotes a diverse species mix, beneficial to the long-term survival of the forest. However, seedling survival remains conspicuously low. Extended or enlarged canopy gaps creates conditions conducive for establishment of the light demanding, salinity tolerant *A. marina* and the gap dependent *R. mucronata*. The forest structure and regeneration patterns reflect a recovery phase after a major disturbance event (1997-98 ENSO), with stagnation as a result of recurrent human pressure.

The vegetation in a mangrove ecosystem constitutes 'the pillar' around which all other ecosystem processes and functions rest. The vegetation drives both the ecological functions, including the biogeochemical cycle (sedimentation, accretion, organic matter production, nutrient production and cycling…. etc.). The proximity of mangroves to human populations, and in our study to a peri-urban area (chapters 3, 4 and 5), introduces a myriad of man induced pressures ranging from pollution, unregulated harvesting to conversion for settlements (Farnsworth and Ellison, 1997b; Kairo *et al*., 2002a; Dahdouh-Guebas *et al*., 2005a), notwithstanding natural pressures such as lightning, hurricanes and sea level rise (climate change) (Clarke and Kerrigan, 2000; Duke, 2001; Gilman *et al*., 2008; Alongi, 2008). These factors cause significant mortalities of mangrove trees in addition to altering site conditions, affecting ecological processes and functions (Ewel *et al*., 1998; Duke, 2001; Bosire *et al*., 2005b; Alongi and de Carvalho, 2008).

To ensure existence, mangroves posses an array of regenerative traits which include both recruitment (reproductive processes) and the sprouting and lateral spreading of surrounding trees (vegetative processes) (Duke, 2001).. The extent to which a mangrove forest recovers, is governed in part, by the regenerative capabilities of the damaged mangrove species and the nature and severity of the causative agent (Hutchings and Saenger, 1987; Snedaker *et al.*, 1992; Duke, 2001; Ellis and Bell, 2004). Thus the species composition of a mangrove forest may be important in determining the regenerative strategy and the recovery time. Secondly, the nature and duration of causative agent defines the nature and extent of degradation (Hutchings and Saenger, 1987; Snedaker *et al*., 1992; Duke, 2001; Ellis and Bell, 2004). Amongst mangrove species, trimming causes mortality in *Rhizophora*, but *Avicennia* and *Laguncularia* recover quickly by releafing, coppicing, and initiating trunk sprouts (Osborne and Berjak, 1997). This has important implications on harvesting practices in relation to stand species composition. Under normal circumstances, and depending on the disturbance intensity, recovery occurs by a combination of the two regenerative processes. Smaller gaps, when occupied by species that *coppice*, will recover in timescales of months by forming new branches (Ellis and Bell, 2004). Larger gaps, however, will require new recruits, which will take 10 years to recover under natural disturbance, and upto 25 years or more under human disturbance (Duke, 2001; Ellis and Bell, 2004). Duke (2001) summarises the gap regenerative cycles, (involving new recruits – "*reproductive processes*") within mangrove forests as passing through recruitment and *growth* stages (figure 7.2). The recovery process is dependent on species specific responses to disturbances, coupled with regenerative strategies, and may vary between mangrove forests depending on species composition. A more diverse forest may be more desirable and stable in the long-run.

The disparity in recovery between natural and man induced gaps are associated with the severity of the physical and chemical changes associated with gaps. More severe changes are associated with human induced gaps (Clarke and Kerrigan, 2000; Clarke, 2004; Duke, 2001; Imai *et al.*, 2006; López-Hoffman *et al.*, 2007). This is mainly due to the selective, widespread and persistent nature of human disturbances as opposed to random and incidental or episodic natural disturbances (Clarke and Kerrigan, 2000; Duke, 2001).

Recovery under human disturbance therefore takes longer time due to lowered regeneration, growth and development rates. Recovery under natural disturbances, occurs over shorter periods due to a viable '*understorey seedling bank*' that survives each disturbance episode coupled with a long period of relative calm or absence of causative agent (Allen *et al.*, 2001; Duke, 2001; Ellis and Bell, 2004).

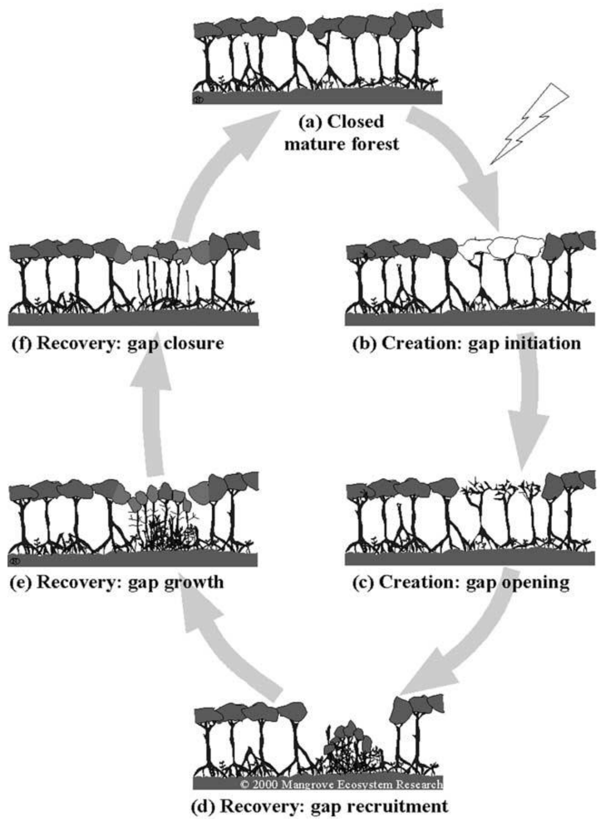

Figure 7.2: A scheme showing six stages of gap creation and recovery observed in common mangrove gaps: mature forest, gap initiation, gap opening, gap recruitment, gap growth and gap closure. The scheme is adapted from Duke (2001) and is based on a *Rhizophora* forest. Disturbances are assumed to arise out of natural disturbances (lightning). It is estimated that the whole cycle can take upto ten years (Ferwerda *et al*., 2007).

Chapters 3, 4 and 6 present findings on the stand structural attributes of the peri-urban mangroves of Tudor creek. Despite the location of Tudor creek mangrove, functional and economic importance, very little is known about the forest. Findings, comparable to other per-urban ecosystems (Moritz-Zimmermann *et al*., 2002), reveal a stressed forest, with poor structural development, low complexity index, and dominant young vegetation. Yet the forest is regenerating and is adaptable to both dynamic natural and anthropogenic pressures. The stem size-frequency depicts a disturbance regime according to direct needs by local people (Chapter 3). Though this is expected under unregulated exploitation, the canopy gap frequencies and sizes cannot be fully explained by harvesting pattern alone (Chapter 4 and 6). Specifically, most enlarged gaps were formed in the period 1992-2005 (Chapter 6). This period is associated with the ENSO event (1997-1998) which induced siltation (Abuodha and Kairo, 2001; Kitheka *et al*., 2003). Due to siltation and recurrent human disturbances, regenerative turnover might have overwhelmed progress in stand development, causing forest growth reversal resulting in a relatively young forest (Duke, 2001), as older trees were either impacted by silt or harvested. Siltation causes mortality of both adults and juveniles, through inhibition of roots gaseous exchange, root damage and oxygen deficiency, resulting in reduced growth and vigour (Ellison, 1998; Thampanya *et al*., 2002). This lowers recovery rates, and sets in a stagnation loop in the recovery phase (recruitment and growth phases; Figure 7.3), accounting for the lack of adequate recovery 10 years after the major disturbance event. The large canopy gaps have introduced altered edaphic conditions, promoting species mix, and altered zonation patterns (Chapter 6).

Recovery by regeneration is influenced by dispersion of propagules, habitat heterogeneity due to disturbance (tree cutting) and altered topography (siltation), characteristic for large canopy gaps (Clarke and Kerrigan, 2000; Minchinton, 2001). In Tudor creek, siltation and harvesting have enlarged canopy gaps (chapter 3 and 4), and in concert with domestic sewage pollution, have altered edaphic conditions, causing habitat heterogeneity, impacting propagule dispersal, establishment, survival and growth, lowering regeneration, altering forest recovery and growth and longer times for canopy gap closure (Clarke and Kerrigan, 2000; Clarke, 2004). Natural regeneration, the major form of gap recovery within Tudor creek, favours *R. mucronata.* On the contrary, *C. tagal*, *S. alba*, *B. gymnorrhiza* and *X granatum* had low regeneration levels. The seedling and sapling distribution patterns are reminiscent of a recovery phase, with the light demanding *A. marina* occupying the large gaps, while the gap dependent *R. mucronata* occupied both the understorey and gaps. The species mix of seedlings and saplings, coupled with the size class distribution of adults may indicate *mast production* amongst mangrove species (chapter 6), though this needs to be verified through long term phenology studies.

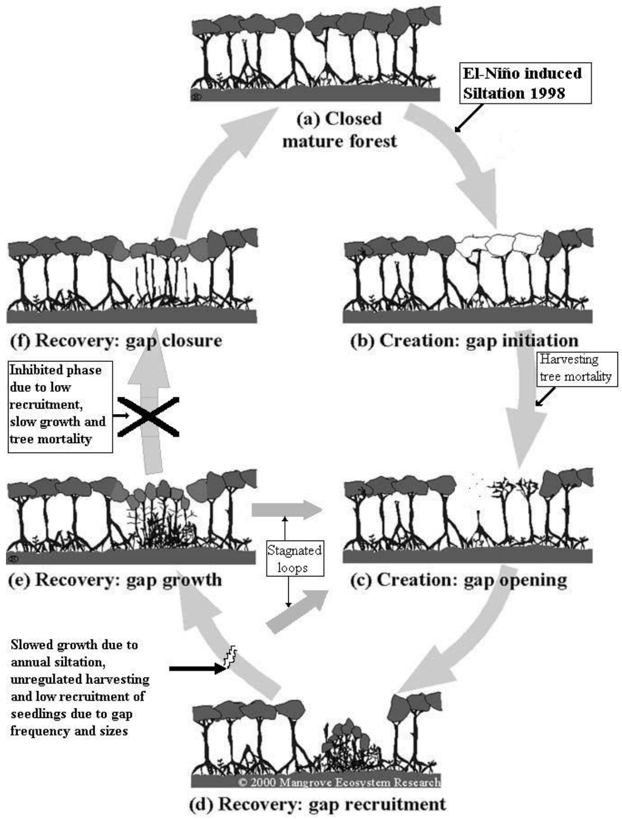

Figure 7.3: A scheme showing the six stages of gap recovery common in mangroves with modified to define the possible scenario for Tudor creek. The major agent of disturbance is identified as the 1997-1998 ENSO event associated with unprecedented siltation into the mangroves and causing major mortalities in many parts of Kenya (Abuodha and Kairo, 2001; Kitheka *et al*., 2003; Dahdouh-Guebas *et al*., 2004), compounded by recurrent unregulated harvesting, and subsequent annual siltation during the rainy season (SE Monsoon) that impedes normal stand recovery, stagnating gap recruitment and gap growth phases. Normal recovery is inhibited by high rates of seedling and sapling mortality either through siltation, harvesting, propagule predation, scouring by drift wood and debris and temperature effects. It is estimated that under human impacted mangroves, recovery may take upto 25 years or longer depending on the agent of disturbance (Ferwerda *et al*., 2007). Lowered recovery rates, thus longer recovery times imply an ecosystem with lowered resilience (Bosire *et al*., 2008).

Siltation may not be a rare event in the Tudor creek mangroves. Observations indicate an approximate loss of 1 ± 0.5m surface soil within the past 50 – 60 years (figure 7.4), mainly due to gradual surface run-off, depositing silt in the mangrove forest. It is our estimate that this rate has been highly variable over the years and has been enhanced in the past two decades as a result of land use changes associated with population growth, mainly involving clearance of terrestrial vegetation for firewood, settlements and agricultural purposes. Most of these activities have no link with the mangrove forest management and are extrinsic in nature. The sudden or abrupt deposition of silt in the mangrove during the annual rainy season buries seedlings of *R. mucronata*, *C. tagal* and *A. marina*, to an approximate depth of 30-40 cm, causing significant mortality (chapter 6). Therefore, annual silting may be a variable spatial inhibitor to natural regeneration, particularly in the landward fringe.

Figure 7.4: Signs of erosion in Jomvu village. The point marked B (Notice the man pointing) is claimed to be the ground level 50-60 years ago when the house was being built. The point marked A is a water pipe that is usually buried more than 1m below the surface. This pipe was installed in the 1960's (Photo taken during study in 2004).

However, stand densities meet the global standards for adequate natural regeneration to occur. This is based on the minimum 12 trees ha^{-1} parent trees (standards) (Tang, 1978), but regeneration by species may be limited by site conditions. Whereas production of propagules was not limiting, factors inhibiting establishment, survival and growth of mangroves were variable spatially. Species composition in a stand is dependent in part on a balance between large-scale and small-scale disturbances and species tolerance range. Lowered recovery rates, longer recovery times, and local species shifts and losses translate into lowered resilience - the ability of the system to effectively recover after a disturbance event. Regulation of harvesting to reduce gap creation frequency and sizes, coupled with appropriate land use practices to minimise siltation may enhance the recovery of Tudor creek mangroves, rather than mere replanting, which is not economically and ecologically viable in ensuring adequate long-term recovery and growth of the forest.

7.4 The utilisation of mangrove goods and services

The ubiquitous form of mangrove wood use is for firewood and charcoal production, which is conducted at both subsistence and commercial scales mainly in an urban setting. The general outlook at the short term financial gains from such a trade vividly undervalues mangrove goods and services in addition to narrowing the long term utility value of the mangrove ecosystem. Dependence on mangrove products is not restricted to impoverished coastal communities, but also the business fraternity in an urban setting, making exploitation technologically effective and efficient, which contravenes principles of sustainability and conservation based on ecosystem resilience. Population growth rates imply that the demand and exploitation of natural resources will increase, thus the need for ‘mutual coercion’ to cost ecosystem goods and services towards better adaptive management for sustainability.

Traditional free access common property resources such as wetlands have of late emerged as sites of special concern. This is especially with human population growth and growing demands for natural resources. This has inevitably resulted in excessive exploitation with subsequent ecosystem degradation, and/or in extreme cases, extinction of species (IUCN, 2001; Duke *et al*., 2007). The ‘traditional common’ resource, where all have access has greatly contributed to the current state of peri-urban ecosystems. The lack of responsibility amongst users, the quest for economic gains in an otherwise competitive urban set-up, coupled with an ineffective management, unfolds a scenario typically defined by Hardin (1968) as ‘the tragedy of the commons’. This tragedy arises out of standalone decision by users to maximise on benefits derived from otherwise limited ecosystem goods and services, while returning nothing to the ecosystem – “*the essence of dramatic tragedy is not unhappiness, but rather it resides in the remorseless working of things”* (Hardin, 1968).

This tragedy, mainly defined by the loss of vital ecosystem goods and service is evident by the increasing number of attempts to restore degraded ecosystems globally (Bosire *et al*., 2003; Abuodha and Kairo, 2001; Ellison, 2000a, b). However, the lack of a market price for mangrove goods, services, aesthetics and existence values, the losses and damage to these wetlands cannot be precisely estimated. Furthermore, most systems used to estimate these goods and services are quite recent and largely untested (Bosire *et al*., 2008; Walters *et al*., 2008). This has caused poor appreciation of mangrove wetlands in relation to other uses, mostly resulting in the gross undervaluing of mangrove resources, with subsequent overexploitation and poor management. In severe cases, justification for mangrove conservation is challenged based on monetary value and has often resulted in clearing for other short term uses such as aquaculture, salt pans etc….(Ewel *et al*., 1998; Abuodha and Kairo, 2001; Alongi, 2002; Wattage and Mardle, 2008).

Mangroves are important to both rural and urban populations, unlike previously reported as being important to impoverished rural communities (Dahdouh-Guebas *et al*., 2006; Alongi and de Carvalho, 2008; Nagelkerken *et al*., 2008; Ellison, 2008a; Walters *et al*., 2008). In urban areas, mangrove dependence takes a commercial dimension, where the major beneficiaries and dependents in the trade have no links with the mangrove ecosystem or are by no means impoverished unlike in a typical rural community, almost similar to what Ellison (2008a) referred to as '*roving banditry'.* In a peri-urban set-up, means of livelihood are more dependent on established industries. In the case of Mombasa, being the biggest sea port in East Africa, serving landlocked countries, the sea-port and transport industries are major employers. The rural community, on the other hand, have no alternative employment, other than farming, natural resources exploitation mainly for subsistence and to supply the urban population with a cheap source of domestic energy - firewood and charcoal. We are unaware of any study that assesses peri-urban mangroves utilisation and related products. With the current observations, the degraded nature of the forest, the poor regeneration status and the levels of mangrove firewood dependence, with an ever increasing demand in an urban setting, it is highly unlikely that the forest will recover if interventions are not enacted in good time. The current practices are unsustainable and impacts negatively on forest structure, ecosystem functions and eventually the communities' livelihood.

The isolated efforts by locals to replant may never translate into recovery, owing to the unplanned nature, absence of defined objectives, selective nature of species and site.

However, natural regeneration levels reveal a recovery potential. Therefore, management and curtailment of local communities' exploitations on the forest seems the best option in managing and mitigating degradation. In this scenario, integrating multiple uses may be contingent on the "*right*" combination of biophysical, social and institutional aspects so that tradeoffs are minimised. Resolving issues related to land tenure system in Mombasa and the implementation of multiple stakeholders dialogue, may be a step towards the establishment of management units geared towards adaptive multiple use systems, designed to meet specific demands at different stakeholder levels. To realise this however, it is essential to set up temporal, spatial and social segregation of timber and non-timber products extraction, which may in turn require specific jurisdictional and institutional framework. The high degree of variation in local perception on the mangroves and related resources need to be harmonised towards sustainability, conservation and resilience thinking.

To counter degradation, mitigate pollution and meet the '*diverse needs*' of an urban population, it may be necessary to ***cost*** the ecosystem goods and services and ***charge users*** based on the mangrove forest management requirements. This means the system needs to change from a "***common resource***" to a "***regulated resource***". Funds raised should then be utilised on *'maintaining the ecosystem structure and function'*. This may include the cost of enforcing the legislation, the manpower and capacity needs, the cost of replanting, the institutional infrastructure, community incentives etc……. This means that the forest needs to be '*institutionalised*' into a management unit (Figure 7.6), where literally each piece of wood, weight of fish, crab or shrimp, unit of domestic sewage filtration etc…. are valued and paid for, for the sake of sustainability. This entails an elaborate plans involving Government departments on one hand (Fisheries Department (FD), Kenya Forest Service (KFS), Kenya Meat Commission (KMC), Municipal Council of Mombasa (MCM), Coast Water and Sewerage Company (CWSC)) and the diverse stakeholders (rural communities, urban dwellers and the wider business fraternity).

Figure 7.5: Other undervalued and over-exploited resources within Tudor creek: a fisherman holding the mud crab, *Scylla serrata* caught from the sewage impacted Mikindani stand and a typical fish landing site within Tudor creek (at 'Bangladesh slum' – sewage impacted). Due to the small sizes of fish caught, weighing is not done and 'small heaps of fish' are used to assign prices (Photos taken in 2005).

7.5 Conclusions

Sewage pollution may not necessarily be harmful to the mangrove vegetation in Tudor creek, and its effects may be variable across spatial scales based on inundation, salinity and sediment types. However, the impacts of sewage pollution may also be influenced by the structural status of the forest, edaphic conditions and the presence of other disturbances. In Tudor creek, it may not be feasible to infer the role of sewage on forest structure based on current observations. This is because sewage pollution, harvesting and siltation have occurred concurrently for decades, at times together with other natural or human induced disturbances. The current forest structure is therefore a consequence of both "natural" disturbances (the 1997-1998 ENSO), recurrent anthropogenic pressure (unregulated harvesting and annual siltation events brought about by annual rains, and enhanced by inappropriate land use practices) and species specific traits (tolerance ranges, regenerative strategy, growth rates……etc). The forest is evidently degraded, with lowered functionality and resilience. Apparently, driven by poverty, locals view sewage pollution differently - a good proportion positively, some negatively, and does not appear to affect way of life, probably an adaptation created out of circumstances, such that all that has been happening or is happening within the society is regarded as the norm, as stated by Hardin (1968) '*morality of an act is a function of the state of the system at the time it is performed*'. However, the current scenario of dumping raw domestic sewage into a resource utilised by locals is morally wrong.

Sewage pollution effects, though not qualitatively proven in our study, enhances growth of mangroves by increasing the amounts of nutrients available for biomass formation as observed in leaf nutrients resorption efficiencies, but does not apparently influence productivity. These observations indicate that plant-nutrients interactions are complex and may be influenced by sediment microbial communities and ***limiting nutrients*** across temporal and spatial scales. Annual 'abrupt' siltation events, extrinsic in nature, with no direct link to forest management, stands out as a major cause of degradation, probably a major concern due to impacts on regeneration by limiting species tolerance ranges and reducing species mix. The erosion of farm lands may also complicate issues, by lowering land fertility, thus productivity and food security in the long run. The current status of the forest and the ongoing unsustainable practices lowers the viability of the mangrove forest. This is evident from the slow recovery 10 years after a major disturbance event, coupled with ongoing disturbances.

Therefore, in an urban context, to ensure viable ecosystems, a deviation from the norm of ‘common resources’ may be the option. The ‘***adaptive management’*** (management of the forest based on current knowledge, while including other extrinsic factors e.g. land use, sewage pollution, harvesting etc… and stakeholders/users) as opposed to ‘***sectorial management’*** (management of each component in isolation e.g. land use, harvesting, fisheries etc…..) is the feasible and practical way to protect urban ecosystems. It may be necessary to cost and impose a fee for ecosystem goods and services to ensure sustainability in conservation and management, while ensuring that the ecosystem remains viable in the long run.

Figure 7.7: Irrigation of subsistence farms with raw domestic sewage at Mikindani.

7.6 Recommendations

7.6.1 "The tragedy of un-managed urban resources?"

This study, like many others (Abuodha and Kairo, 2001; Alongi, 2002; Duke *et al*., 2007; Bosire et al., 2008; Walters et al., 2008), highlights the degradation that has occurred, is occurring and is to occur in global mangrove ecosystems. Additionally, we have presented highlights on the plight of a peri-urban ecosystem, under unique multiple stressors ranging from natural and anthropogenic in terms of cause, to intrinsic and extrinsic in terms of cause origin. Given the 'finite' nature of the world ecosystems, that can support a 'finite' human population, and the exponential population growth, it is unlikely that the causes of degradation will diminish and the condition of the mangrove ecosystem will improve if no interventions are implemented. This will eventually result in significant losses to all direct and indirect dependants on the mangrove ecosystem goods and services. Furthermore, the loss of surface soil and fertility, will threaten the food security of subsistence communities. The worst case scenario may arise out of the global economic crises. With high inflation rates and high cost of essential commodities, more people will resort to a cheaper energy source, firewood, which is already over-exploited.

Addressing the root causes of degradation requires first, lowering the rate of population growth and second lowering the levels of poverty. To promote poverty alleviation and economic development in urban areas, it may be highly beneficial and sustainable in the long run to link such initiatives with environmental protection and biodiversity conservation. Conservation can enhance economic development because un-degraded ecosystems supply valuable goods and services (Walters *et al*., 2008, Kareiva *et al*., 2008). Kaireva *et al*. (2008) has demonstrated that World Bank projects with biodiversity goals were as successful in all development objectives, including poverty reduction and private sector development, as those that focused solely on development. They identified the use of market mechanisms or sustainable finance approaches as a major cause of success in these projects.

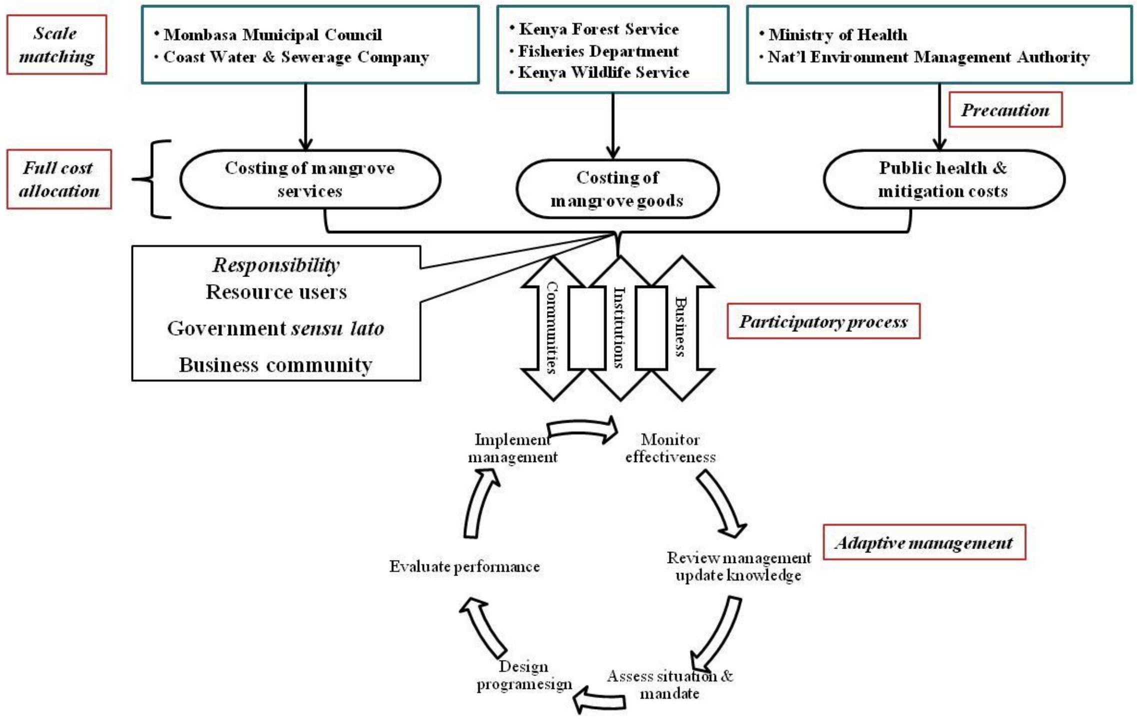

Figure 7.6: A flowchart summarising the main responsibilities and the major stakeholders within the mangrove system of peri-urban Mombasa, with an adaptive management cycle that is to define the relationships of the main parties. The model proposes six principles adopted from Constanza *et al*. (1998). These principles include (i) ***Scale matching*** – aims at broadening the management scope of the resources and ecosystems by appropriate institutions and maximising ecological input; (ii) ***Full cost allocation*** – all internal and external costs and benefits, including social and ecological, of alternative decisions concerning the use of environmental resources should be identified and allocated and the markets adjusted to reflect full costs; (iii) ***Precaution*** – addresses environmental issues arising out of use, with the responsible parties bearing the burden of mitigation; (iv) ***Responsibility*** – attendant responsibility to utilise resources in a sustainable manner, even to bear the cost of resources; (v) ***Participation*** – all stakeholders to be involved in management and decision making; and (vi) ***Adaptive management*** – based on integration of appropriate ecological, social, and economic information with the goal of attaining sociol and ecological sustainability. Funds raised in paying for ecosystem goods and services should be utilized in facilitating the implementation and enforcement of legislation and improving on management towards establishing a resilient urban ecosystem (Adapted from IUCN-WCPA, 2008 with modification). The model proposes the costing and payment for mangrove ecosystem goods and services, with an element of responsibility assigned to both individual and corporate users. This paradigm is based on the hypothesis of the 'tragedy of the commons'. This 'tragedy' requires the realisation that a finite ecosystem can support a finite population. Therefore under high population pressure, "mutual coercion' is an accepted strategy.

By no means does this suggestion imply that conservation and development win-wins are easy to achieve. Globally, and in particular, Kenya, creation of protected areas as 'parks' or/and 'reserves' has been the norm towards conservation (Obura, 2001). Wittemyer *et al*. (2008) reported high rates of human population growth around 306 protected areas in 45 countries across Africa and Latin America. The population growth rates were nearly twice the country averages (the "human honeypot" hypothesis). This was mainly a result of international donor funding to parks and the consequent creation of park-related jobs and services and, disappointingly, is associated with accelerated rates of deforestation, potentially exacerbates the same anthropogenic threats to biodiversity it aims to alleviate. However, establishing protected areas has always been cited as the best option towards boosting ecosystem resilience to both long term and short term disturbances (IUCN-WCPA, 2008). These protected areas offer a spectrum of management strategies ranging from full protection, or no-entry areas, to multiple-use areas which prohibit limited activities (IUCN-WCPA, 2008). Such a move will ensure regulated uses and users of finite resources, imparting sustainability and resilience to the socio-ecological system. This is mainly because the ‘common resource’ is only justifiable under conditions of low population density (Hardin, 1968). Whereas conservation may potentially address poverty, the control of population growth rates can be complex. This is because population growth touches on many traditional beliefs, political, personal issues and traits, and the right to procreate (legal system, personal liberties, morality ..……etc).

7.6.2 Scientific considerations

The recovery phase of the forest of Tudor creek offers an opportunity to follow up on the trends to better understand the phases and dynamics involved and the prospects of managing and restoring degraded peri-urban mangroves. It is therefore recommended to:-

- Build up a database on change dynamics during the recovery phase considering growth rates, standing biomass/volume, phenology, gap formation and recovery rates and species shifts to formulate appropriate management and harvesting plans for the forest.
- Assess conflicting and supporting policies regarding the management of mangrove resources in Kenya, with the view of harmonising the policies and involving local communities, incorporating local ecological knowledge on traditional users and uses management and regulation into management of these urban forests.
- Follow up on the replanted mangroves despite the high mortalities and assess the wood quality in the wake of sewage pollution. This may be important in

establishing the recovery rates and duration, and the selection of species to replant.

- Assess the growth rates of trees at Mikindani and those at Jomvu, Mirarani and Junda through wood anatomy studies to gain a better understanding on domestic sewage pollution effects on tree growth.
- Validate the impacts of sewage pollution on productivity by setting up replicate experiments in different mangrove creeks. This is so as to filter out the effect of site and pollution. Productivity may also vary between forests due to site conditions.

There are no studies that have examined interactions of important growth-limiting factors such as flooding, salinity, nutrients or carbon, temperature, or pollution. This information on multi-stress factors interactions is necessary to make accurate conclusions and predictions on mangrove seedling responses. Whereas we may not be in a position to authoritatively establish the impacts of sewage exposure on the seedling growth with this study, we pose a question: could it be feasible that the raw sewage may have cushioned the seedlings against high temperatures during the dry season, as a possible reason for the mortality of seedlings in un-polluted sites?

7.6.3 Self critic

This work involved extensive and laborious fieldwork, which was also quite costly. Whereas a lot has been learnt about the system, more remains unknown. We had setup a replanting experiment under sewage polluted and control sites. But our experiment was impacted by an unforeseen confounding factor, siltation. Therefore we are not in a position to point out the impacts of pollution on growth or forest structure. Furthermore, the replications used in the productivity studies did not include a control site, but was rather based on pseudo-replication within the same site. This may be viewed positively as representing responses to a single population in a homogenous site, only defined by a pollution gradient. Furthermore, a control site outside Tudor creek, with no pollution may yield differences that are not readily interpreted as pollution effects. For example, litter studies from Gazi bay (ca. 47 km south) yielded values in the same range of values recorded for Tudor creek for *R. mucronata* and *S. alba,* but not for *A. marina.*

Acknowledgements

We express our appreciation to the local communities for their support in the sampling campaigns, and colleagues at KMFRI and KWS Coast Region Research station. The work described in this thesis was supported by a PhD. Scholarship from the Flemish Interuniversity Council (VLIR) in Belgium and the EU funded PUMPSEA project (FP6 – INCO contract no. 510863).

References

Abuodha, P. A. W. and Kairo, J. G. 2001. Human-induced stresses on mangrove swamps along the Kenyan coast. *Hydrobiologia*, 458: 255–265.

Adams, E. S. and Levings, S. C. 1987. Territory size and population limits in mangrove termites. *Journal of Animal Ecology*, 56(3): 1069-1081.

Akatsu, M., Hosoi, Y., Sasamoto, H. and Ashihara, H. 1996. Purine metabolism in cells of a mangrove plant, *Sonneratia alba*, in tissue culture. *Journal of Plant Physiology*, 149(1-2): 133-137.

Aksornkoae, S. 1993. Ecology and management of mangrove forests. IUCN, Bangkok, Thailand.

Allen, J. A., Ewel, K. C. and Jack, J. 2001. Patterns of natural and anthropogenic disturbance of the mangroves on the Pacific island of Kosrae. *Wetlands Ecology and Management*, 9: 279–289.

Alongi, D. M. 1990. Effect of mangrove detrital outwelling on nutrient regeneration and oxygen fluxes in coastal sediments of the Central Great-Barrier-Reef Lagoon. *Estuarine Coastal and Shelf Science*, 31(5): 581-598.

Alongi, D. M., Sasekumar, A., Tirendi, F. and Dixon, P. 1998. The influence of stand age on benthic decomposition and recycling of organic matter in managed mangrove forests of Malaysia. *Journal of Experimental Marine Biology and Ecology*, 225: 197–218.

Alongi, D. M. 2002. Present state and future of the world's mangrove forests. *Environmental Conservation*, 29 (3): 331–349.

Alongi, D. M., Clough, B. F., Robertson A. I. 2005. Nutrient-use efficiency in arid-zone forests of the mangroves *Rhizophora stylosa* and *Avicennia marina*. *Aquatic Botany*, 82: 121–131.

Alongi, D. M. 2008. Mangrove forests: Resilience, protection from tsunamis, and responses to global climate change. *Estuarine, Coastal and Shelf Science*, 76: 1-13.

Alongi, D. M. and de Carvalho, N. A. 2008. The effect of small-scale logging on stand characteristics and soil biogeochemistry in mangrove forests of Timor Leste. *Forest Ecology and Management*, 255(3-4): 1359-1366.

Angiosperm Phylogeny Group. 2003. An update of the Angiosperm Phylogeny Group classification for the orders and families of flowering plants: APG II. *Botanical Journal of the Linnean Society*, 141: 399–436.

Aronson, J., Ovalle, C. and Avendano, J. 1993. Ecological and economic rehabilitation of degraded espinales in the sub-humid Mediterranean climate region of central Chile. *Landscape and Urban Planning*, 24(1-4): 15-21.

Ashihara, H., Adachi, K., Otawa, M., Yasumoto, E., Fukushima, Y., Kato, M., Sano, H., Sasamoto, H. and Baba, S. 1997. Compatible solutes and inorganic ions in the mangrove plant *Avicennia marina* and their effects on the activities of enzymes. Zeitschrift Fur Naturforschung C-a *Journal of Biosciences*, 52(7-8): 433-440.

Ball, M. C. 1980. Patterns of secondary succession in a mangrove forest of Southern Florida. *Oecologia*, 44(2): 226-235.

Ball, M. C., Chow W. S. and Anderson J. M. 1987. Salinity induced potassium deficiency causes loss of functional photosystem II in leaves of the grey mangrove, *Avicennia marina*, through depletion of the atrazine binding polypeptide. *Australian Journal of Plant Physiology*, 14(3): 351-361.

Ball, M. C., Cowan I. R. and Farquhar G. D. 1988. Maintenance of leaf temperature and the optimization of carbon gain in relation to water loss in a tropical mangrove forest. *Australian Journal of Plant Physiology*, 15(1-2): 263-276.

Ball, M. C. and Pidsley S. M. 1995. Growth responses to salinity in relation to distribution of 2 mangrove species, *Sonneratia alba* and *S. lanceolata*, in Northern Australia. *Functional Ecology*, 9(1): 77-85.

Ball, M. C. 2002. Interactive effects of salinity and irradiance on growth: implications for mangrove forest structure along salinity gradients. *Trees-Structure and Function*, 16, 126-139.

Bandaranayake, W. M. 1998. Mycosporines: are they nature's sunscreens? *Natural Product Reports*, 15(2): 159-172.

Barbier, E. B. 2000. Valuing the environment as input: review of applications to mangrove-fishery linkages. *Ecological Economics*, 35(1): 47-61.

Barbier, E. and Cox, M. 2002. Economic and demographic factors affecting mangrove loss in the coastal provinces of Thailand, 1979-1996. *Ambio*, 31(4): 351-357.

Bates, B. C., Kundzewicz, Z. W., Wu, S. and Palutikof, J. P. Eds., 2008: Climate change and water. Technical Paper of the Intergovernmental Panel on Climate Change, IPCC Secretariat, Geneva, 210 pp.

Benfield, S. L., Guzman, H. M., and Mair, J. M. 2005. Temporal mangrove dynamics in relation to coastal development in Pacific Panama. *Journal of Environmental Management*, 76(3): 263-276.

Berger, U., Adams, M., Grimm, V., and Hildenbrandt, H. 2006. Modelling secondary succession of neotropical mangroves: Causes and consequences of growth reduction in pioneer species. *Perspectives in Plant Ecology Evolution and Systematics*, 7(4): 243-252.

Berger, U., Rivera-Monroy, V. H., Doyle, T. W., Dahdouh-Guebas, F., Duke, N., Fontalvo, M., Hildenbrandt, H., Koedam, N., Mehlig, U., Piou, C., and Twilley, R. R., 2008. Advances and limitations of individual-based models to analyze and predict dynamics of mangrove forests: a review. *Aquatic Botany*, 89: 260–274.

Benner, R. and Hodson, R. E. 1985. Microbial degradation of the leachable and lignocellulosic lomponents of leaves and wood from *Rhizophora mangle* in a tropical mangrove swamp. *Marine Ecology-Progress Series*, 23(3): 221-230.

Bird, M. I., Fifield, L. K., Chua, S. and Goh, B. 2004. Calculating sediment compaction for radiocarbon dating of intertidal sediments. *Radiocarbon*, 46(1): 421-435.

Blaber, S. J. M. and Milton, D. A. 1990. Species composition, community structure and zoogeography of fishes of mangrove estuaries in the Solomon-Islands. *Marine Biology*, 105(2): 259-267.

Blanchard, J. and Prado, G. 1995. Natural regeneration of *Rhizophora mangle* in strip clear-cuts in Northwest Ecuador. *Biotropica*, 27(2): 160-167.

Boer, B. 1993. Anomalous pneumatophores and adventitious roots of *Avicennia marina* (Forsk.) Vierh. mangroves 2 Years after the 1991 gulf-war oil-spill in Saudi-Arabia. *Marine Pollution Bulletin*, 27: 207-211.

Boonsong, K., Plyatiratitivorakul, S. and Patanapompalboon, P. 2003. Potential use of mangrove plantation as constructed wetland for municipal wastewater treatment. *Water Science and Technology*, 48 (5): 257-266.

Bosire, J. O., Dahdouh-Guebas, F., Kairo, J. G. and Koedam, N. 2003. Colonization of non-planted mangrove species into restored mangrove stands in Gazi Bay, Kenya. *Aquatic Botany*, 76: 267-279.

Bosire, J. O., Dahdouh-Guebas, F., Kairo, J. G., Cannicci, S. and Koedam, N. 2004. Spatial variations in macrobenthic fauna recolonisation in a tropical mangrove bay. *Biodiversity and Conservation*, 13(6): 1059-1074.

Bosire, J. O., Dahdouh-Guebas, F., Kairo, J. G., Kazungu, J., Dehairs, F., and Koedam, N. 2005a. Litter degradation and CN dynamics in reforested mangrove plantations at Gazi Bay, Kenya. *Biological Conservation*, 126(2): 287-295.

Bosire, J. O., Kairo, J. G., Kazungu, J., Koedam, N. and Dahdouh-Guebas, F. 2005b. Predation on propagules regulates regeneration in a high-density reforested mangrove plantation. *Marine Ecology-Progress Series*, 299: 149-155.

Bosire, J. O., Dahdouh-Guebas, F., Kairo, J. G., Wartel, S., Kazungu, J. and Koedam, N. 2006. Success rates of recruited tree species and their contribution to the structural development of reforested mangrove stands. *Marine Ecology-Progress Series*, 325: 85-91.

Bosire, J. O., Dahdouh-Guebas, F., Walton, M., Crona, B. I., Lewis III, R. R., Field, C., Kairo, J. G. and Koedam, N., 2008. Functionality of restored mangroves: a review. *Aquatic Botany*, 89 :251–259.

Boto, K. G. 1992. Nutrients and mangroves. In Connell, D. W. and Hawker, D. W. (eds), Pollution in Tropical Aquatic Systems. CRC Press Inc., Ann Arbor, London: 129–145.

Boto, K. G. and Wellington, J. T. 1983. Phosphorus and nitrogen nutritional status of a Northern Australian mangrove forest. *Marine Ecology Progress Series*, 11(1): 63-69.

Bouillon, S., Dahdouh-Guebas, F., Rao, A. V. V. S., Koedam, N. and Dehairs, F. 2003. Sources of organic carbon in mangrove sediments: variability and possible ecological implications. *Hydrobiologia*, 495(1-3): 33-39.

Bouillon, S., Moens, T., Koedam, N., Dahdouh-Guebas, F., Baeyens, W. and Dehairs, F. 2004a. Variability in the origin of carbon substrates for bacterial communities in mangrove sediments. *Fems Microbiology Ecology*, 49(2): 171-179.

Bouillon, S., Moens, T., Overmeer, I., Koedam, N. and Dehairs, F. 2004b. Resource utilization patterns of epifauna from mangrove forests with contrasting inputs of local versus imported organic matter. *Marine Ecology-Progress Series*, 278: 77-88.

Bouillon, S., Middelburg, J. J., Dehairs, F., Borges, A. V., Abril, G., Flindt, M. R., Ulomi, S. and Kristensen E. 2007a. Importance of intertidal sediment processes and porewater exchange on the water column biogeochemistry in a pristine mangrove creek (Ras Dege, Tanzania). *Biogeosciences*, 4(3): 311-322.

Bouillon, S., Dehairs, F., Velimirov, B., Abril, G. and Borges, A. V. 2007b. Dynamics of organic and inorganic carbon across contiguous mangrove and seagrass systems (Gazi Bay, Kenya). *Journal of Geophysical Research-Biogeosciences*, 112(G2): -.

Bouillon, S., Connolly, R. M., and Lee, S. Y. 2008. Organic matter exchange and cycling in mangrove ecosystems: Recent insights from stable isotope studies. *Journal of Sea Research*, 59(1-2): 44-58.

Boyden, S., Binkley, D. and Senock, R. 2005. Competition and facilitation between Eucalyptus and nitrogen-fixing Falcataria in relation to soil fertility. *Ecology*, 86(4): 992-1001.

Brokaw, N. and Thompson, J. 2000. The H for DBH. *Forest Ecology and Management*, 129: 89–91.

Bundotich, G. 2007. Structural inventory of mangrove forests in Ngomeni, Kenya. MSc. Thesis, Egerton University.

Bunt, J. S. 1995. Continental scale patterns in mangrove litter fall. *Hydrobiologia*, 295(1-3): 135-140.

Bunt, J. S. 1996. Mangrove zonation: An examination of data from seventeen riverine estuaries in tropical Australia. *Annals of Botany*, 78(3): 333-341.

Cabana, G. and Rasmussen, J. B. 1996. Comparison of aquatic food chains using nitrogen isotopes. *Proceedings of the National Academy of Sciences of the United States of America*, 93(20): 10844-10847.

Cannicci, S., Burrows, D., Fratini, S., Lee, S. Y., Smith III, T. J., Offenberg, J., and Dahdouh-Guebas, F. 2008. Faunistic impact on vegetation structure and ecosystem function in mangrove forests: a review. *Aquatic Botany*, 89: 186–200.

Chansang, H. and Poovachiranon, S. 1985. Fate of mangrove litter in a mangrove forest at Ko-Yao-Yai, Southern Thailand. *Estuaries*, 8(2B): A106-A106.

Chapman, V. J. 1976. Mangrove vegetation. J. Cramer,Vaduz.

Chapman, V. J., 1977. Ecosystems of the World; Wet Coastal Ecosystems. Elsevier Scientific Publishing Company, Amsterdam.

Christensen, B. 1982. Management and utilization of mangroves in Asia and the Pacific, FAO Environment Paper No. 3., Rome: United Nations Food and Agricultural Organisation.

Chong, P. W. 1988. Proposed integrated forest management planning and utilization of mangrove resources in the Terraba-Sierpe Reserve, Costa Rica. TCP. FAO, Rome.

Chu H. Y., Chen, N. C., Yeung, M. C., Tam, N. F. Y. and Wong Y. S. 1998. Tide-tank system simulating mangrove wetland for removal of nutrients and heavy metals from wastewater. *Water Science and Technology*, 38(1): 361-368.

Cintrón, C. and Schaeffer-Novelli, Y. 1984. Methods for studying mangrove structure. In: Snadaker, S.C., Snaedaker, J.G. (Eds). The mangrove ecosystem research methods. UNESCO, Paris, France, pp 91-113.

Clarke, L. D. and Hannon, N. J. 1971. Mangrove swamp and salt marsh communities of Sydney District .4. Significance of species interaction. *Journal of Ecology*, 59(2):535

Clarke, P. J. and Allaway, W. G. 1992. The regeneration niche of the grey mangrove (*Avicennia marina*): Effects of salinity, light and sediment factors on establishment, growth and survival in the field. *Oecologia*, 93: 548–556.

Clarke, P. J. 1992. Predispersal mortality and fecundity in the gray mangrove (*Avicennia marina*) in Southeastern Australia. *Australian Journal of Ecology*, 17(2): 161-168.

Clarke, P. J. 1994. Base-line studies of temperate mangrove growth and reproduction demographic and litter fall measures of leafing and flowering. *Australian Journal of Botany*, 42(1): 37-48.

Clarke, P. J. and Kerrigan, R. A. 2000. Do forest gaps influence the population structure and species composition of mangrove stands in northern Australia? *Biotropica*, 32(4): 642-652.

Clarke, P. J. and Kerrigan, R. A. 2002. The effects of seed predators on the recruitment of mangroves. *Journal of Ecology*, 90(4): 728-736.

Clarke, P. J. 2004. Effects of experimental canopy gaps on mangrove recruitment: lack of habitats partitioning may explain stand dominance. *Journal of Ecology*, 92: 203-213.

Clough, B. F. 1984. Growth and salt balance of the mangroves *Avicennia marina* (forsk.) vierh. and *Rhizophora stylosa* griff. in relation to salinity. *Australian Journal of Plant Physiology*, 11: 419–430.

Clough, B. F., Boto, K. G. and Attiwill, P. M. 1983. Mangrove and sewage: a re-evaluation. In Teas, H. J. (ed.), Biology and Ecology of Mangroves. Tasks for Vegetation Science Series, Vol. 8, Dr W. Junk Publishers, Lancaster: 151–162.

Clutter, J. L., Fortson, J. C., Pienaar, L. V., Brister, G. H. and Bailey, R. L. 1983. Timber management: A quantitative approach. John Wiley and Sons. 333 pp.

Conacher, C. A., Obrien, C., Horrocks J. L. and Kenyon R. K.. 1996. Litter production and accumulation in stressed mangrove communities in the Embley River Estuary, north-eastern Gulf of Carpentaria, Australia. *Marine and Freshwater Research*, 47(5): 737-743.

Corredor, J. E. and Morell, J. M. 1994. Nitrate depuration of secondary sewage effluents in mangrove sediments. *Estuaries*, 17(1B): 295-300.

Costanza, R., d' Arge, R., de Groot., R., Faber, S., Grasso, M., Hannon, B., Limburg, K, Naeem, S., O'Neill, R, Paruello, J., Raskin, R., Sutton, P. and van den Belt, M. 1997. The value of the world's ecosystem services and natural capital. *Nature*, 387: 15.

Costanza, R., Andrade, F., Antunes, P., van den Belt, M., Boersma, D., Boesch, D. F., Catarino, F., Hanna, S., Limburg, K., Low, B., Molitor, M., Pereira, J. G., Rayner, S., Santos, R., Wilson, J. and Young, M. 1998. Principles for sustainable governance of the oceans. *Science*, 281(5374): 198-199.

Costanzo, S. D., O'Donohue, M. J. and Dennison, W. C. 2003. Assessing the seasonal influence of sewage and agricultural nutrient inputs in a subtropical river estuary. *Estuaries*, 26(4A): 857-865.

Dahdouh-Guebas, F., Giuggioli, M., Oluoch, A., Vannini, M., and Cannicci, S. 1999. Feeding habits of non-ocypodid crabs from two mangrove forests in Kenya. *Bulletin of Marine Science*, 64(2): 291-297.

Dahdouh-Guebas, F., Mathenge, C., Kairo, J. G. and Koedam, N. 2000. Utilization of mangrove wood products around Mida Creek (Kenya) amongst subsistence and commercial users. *Economic Botany*, 54(4): 513-527.

Dahdouh-Guebas, F. and Koedam, N. 2001. Are the northernmost mangroves of West Africa viable? a case study in Banc d'Arguin National Park, Mauritania. *Hydrobiologia*, 458: 241-253.

Dahdouh-Guebas, F., Kairo, J. G., Jayatissa, L. P., Cannicci, S. and Koedam N. 2002. An ordination study to view vegetation structure dynamics in disturbed and undisturbed mangrove forests in Kenya and Sri Lanka. *Plant Ecology*, 161(1): 123-135.

Dahdouh-Guebas, F., Van Pottelbergh, I., Kairo, J. G., Cannicci, S., and Koedam, N. 2004. Human-impacted mangroves in Gazi (Kenya): predicting future vegetation based on retrospective remote sensing, social surveys, and distribution of trees. *Marine Ecology Progress Series*, 272: 77–92.

Dahdouh-Guebas, F., Hettiarachchi, S., Lo Seen, D., Batelaan, O., Sooriyarachchi, S., Jayatissa, L.P., and Koedam, N. 2005a. Transitions in ancient inland freshwater resource management in Sri Lanka affect biota and human populations in and around coastal lagoons. *Current Biology*, 15: 579–586.

Dahdouh-Guebas, F., Jayatissa, L. P., Di Nitto, D., Bosire, J. O., Lo Seen D. and Koedam, N. 2005b. How effective were mangroves as a defence against the recent tsunami? *Current Biology*, 15(12): R443-R447.

Dahdouh-Guebas, F. 2006. Mangrove forests and tsunami protection. In: 2006 McGraw-Hill Yearbook of Science and Technology, McGraw-Hill Professional, New York, USA: pp 187-191.

Dahdouh-Guebas, F. and Koedam N. 2006a. Empirical estimate of the reliability of the use of the Point-Centred Quarter Method (PCQM): solutions to ambiguous field situations and description of the PCQM protocol. *Forest Ecology and Management*, 228: 1-18.

Dahdouh-Guebas, F. and Koedam, N. 2006b. Coastal vegetation and the Asian Tsunami. *Science*, 311: 37-38.

Dahdouh-Guebas, F., Vrancken, D., Ravishankar, T., and Koedam, N. 2006. Short-term mangrove browsing by feral water buffalo: conflict between natural resources, wildlife and subsistence interests? *Environmental Conservation*, 33(2): 157-163.

Dahdouh-Guebas, F. and Koedam, N., 2008. Long-term retrospection on mangrove development using transdisciplinary approaches: a review. *Aquatic Botany,* 89: 80–92.

Delgado, P., Hensel, P. F., Jimenez, J. A. and Day, J. W. 2001. The importance of propagule establishment and physical factors in mangrove distributional patterns in a Costa Rican estuary. *Aquatic Botany*, 71(3): 157-178.

Devoe, N. N. and Cole T. G. 1998. Growth and yield in mangrove forests of the Federated States of Micronesia. *Forest Ecology and Management*, 103(1): 33-48.

Di Nitto, D., Daltdouh-Guebas, F., Kairo, J. G., Decleir, H. and Koedam N. 2008. Digital terrain modelling to investigate the effects of sea level rise on mangrove propagule establishment. *Marine Ecology-Progress Series*, 356: 175-188.

Diop, E. S. (Ed). 1993. Conservation and sustainable utilization of mangrove forests in Latin America and African Regions (Part 2: African Region). Mangrove Ecosystem Technical Report 3. International Society for Mangrove Ecosystems and International Tropical Timber Organization, Tokyo.

Dittmar, T., Hertkorn, N., Kattner, G. and Lara, R. J. 2006. Mangroves, a major source of dissolved organic carbon to the oceans. *Global Biogeochemical Cycles*, 20(1): -.

Doute, R. N., Ochanda, N. and Epp, H. 1981. A forest inventory using remote sensing techniques. KREMU, Technological Report, Series No. 10, Nairobi: Kenya Rangeland Monitoring Unit.

Drennan, P. M., Berjak, P. and Pammenter, N. W. 1992. Ion gradients and adenosine-triphosphatase localization in the salt-glands of *Avicennia marina* (Forsskal) Vierh. *South African Journal of Botany*, 58(6): 486-490.

Dschida, W. J., Plattaloia, K. A. and Thomson, W. W. 1992. Epidermal peels of *Avicennia germinans* (L) Stearn - a useful system to study the function of salt-glands. *Annals of Botany*, 70(6): 501-509.

Duarte, C. M., Geertz-Hansen, O., Thampanya, U., Terrados, J., Fortes, M. D., Kamp-Nielsen, L., Borum, J. and Boromthanarath, S. 1998. Relationship between sediment conditions and mangrove *Rhizophora apiculata* seedling growth and nutrient status. *Marine Ecology-Progress Series*, 175: 277-283.

Duke, N. C., Bunt, J. S. and Williams, W. T. 1981. Mangrove litter fall in Northeastern Australia .1. Annual totals by component in selected species. *Australian Journal of Botany*, 29(5): 547-553.

Duke, N. C. 1990. Phenological trends with latitude in the mangrove tree *Avicennia marina*. *Journal of Ecology*, 78(1): 113-133.

Duke, N. C., Ball, M. C. and Ellison, J. C. 1998. Factors influencing biodiversity and distributional gradients in mangroves. *Global Ecology and Biogeography Letters*, 7(1): 27-47.

Duke, N. 2001. Gap creation and regenerative processes driving diversity and structure of mangrove ecosystems. *Wetlands Ecology and Management*, 9: 257–269.

Duke, N. C., Meynecke, J.-O., Dittmann, S., Ellison, A. M., Anger, K., Berger, U., Cannicci, S., Diele, K., Ewel, K. C., Field, C. D., Koedam, N., Lee, S. Y., Marchand, C., Nordhaus, I., and Dahdouh-Guebas, F. 2007. A world without mangroves? *Science*, 317: 41-42.

Ellis, W. L. and Bell, S. S. 2004. Canopy gaps formed by mangrove trimming: an experimental test of impact on litter fall and standing litter stock in Southwest Florida (USA). *Journal of Experimental Marine Biology and Ecology*, 311: 201– 222.

Ellis, W. L., Bowles, J. W., Erickson, A. A., Stafford, N., Bell, S. S. and Thomas, M. 2006. Alteration of the chemical composition of mangrove (*Laguncularia racemosa*) leaf litter fall by freeze damage. *Estuarine Coastal and Shelf Science*, 68(1-2): 363-371.

Ellison, J. C. and Stoddart, D. R. 1991. Mangrove ecosystem collapse during predicted sea-level rise - Holocene analogs and implications. *Journal of Coastal Research*, 7(1): 151-165.

Ellison, A. M. and Farnsworth, E. J. 1992. The ecology of Belizean mangrove-root fouling communities - patterns of epibiont distribution and abundance, and effects on root-growth. *Hydrobiologia*, 247(1-3): 87-98.

Ellison, A. M. 1998. Impacts of sediment burial on mangroves. *Marine Pollution Bulletin*, 37(8-12): 420-426.

Ellison, A. M., Farnsworth, E. J. and Merkt, R. E. 1999. Origins of mangrove ecosystems and the mangrove biodiversity anomaly. *Global Ecology and Biogeography*, 8(2): 95-115.

Ellison, A. M. 2000a. Mangrove restoration: Do we know enough? *Restoration Ecology*, 8: 219–229.

Ellison, A. M. 2000b. Restoration of mangrove ecosystems. *Restoration Ecology*, 8(3): 217-218.

Ellison, A. M. 2008a. Managing mangroves with benthic biodiversity in mind: Moving beyond roving banditry. *Journal of Sea Research*, 59 (1-2): 2-15.

Ellison, J. C. 2008b. Long-term retrospection on mangrove development using sediment cores and pollen analysis: a review. *Aquatic Botany*, 89: 93–104.

Eusse, A. M. and Aide, T. M. 1999. Patterns of litter production across a salinity gradient in a *Pterocarpus officinalis* tropical wetland. *Plant Ecology*, 145(2): 307-315.

Ewel, K. C., Zheng, S., Pinzon, Z. S., Bourgeois, J. A. 1998. Environmental effects of canopy gap formation in high-rainfall mangrove forests'. *BIOTROPICA*, 30(4): 510-518.

Ewel, K. C. 2001. Natural resource management: The need for interdisciplinary collaboration. *Ecosystems* 4(8): 716-722.

FAO. 1994. Mangrove forest management guidelines. FAO Forestry Paper, 117. Rome, 320 pp.

FAO. 2003. New global mangrove estimate. http://www.fao.org/forestry/foris/webview/forestry2/index.jsp%3Fgeold=0%26langid

FAO. 2005. Global forest resource assessment: progress towards sustainable forest management. FAO Forestry Paper, 147. Rome, 348 pp.

Farnsworth, E. J. and Ellison, A. M. 1996. Scale-dependent spatial and temporal variability in biogeography of mangrove root epibiont communities. *Ecological Monographs*, 66(1): 45-66.

Farnsworth, E. J. and Ellison A. M. 1997a. Global patterns of pre-dispersal propagule predation in mangrove forests. *BIOTROPICA*, 29: 318–330.

Farnsworth, E. J. and Ellison A. M. 1997b. The global conservation status of mangroves. *Ambio*, 26(6): 328-334.

Feller, I. C. 1995. Effects of nutrient enrichment on growth and herbivory of dwarf red mangrove (Rhizophora mangle). *Ecological Monographs*, 65(4): 477-505.

Feller, I. C., Whigham, D. F., O'Neill, J. P. and McKee, K. L. 1999. Effects of nutrient enrichment on within-stand cycling in a mangrove forest. *Ecology*, 80(7): 2193-2205.

Feller, I. C., McKee, K. L., Whigham, D. F., and O'Neill, J. P. 2003a. Nitrogen vs. phosphorus limitation across an ecotonal gradient in a mangrove forest. *Biogeochemistry*, 00: 1–31.

Feller, I. C., Whigham, D. F., McKee, K. L. and Lovelock, C. E. 2003b. Nitrogen limitation of growth and nutrient dynamics in a disturbed mangrove forest, Indian River Lagoon, Florida. *Oecologia*, 134(3): 405-414.

Ferguson, W. 1993. A land(scape) ecological survey of the mangrove resource of Kenya (Draft Report). Technical Coperation Project (FAO TCP/KEN/OO51) for the Forestry Department, Ministry of Environment and Natural Resources, Nairobi, Kenya.

Ferwerda, J. G., Ketner, P. and McGuinness, K. A. 2007. Differences in regeneration between hurricane damaged and clear-cut mangrove stands 25 years after clearing. *Hydrobiologia*, 591: 35-45.

Findlay, R. H., Trexler, M. B. and White, D. C. 1990. Response of a benthic microbial community to biotic disturbance. *Marine Ecology Progress Series*, 62(1-2: 135-148.

Fitzgerald, M. A., Orlovich, D. A. and Allaway, W. G. 1992. Evidence that abaxial leaf glands are the sites of salt secretion in leaves of the mangrove *Avicennia marina* (Forsk.) Vierh. *New Phytologist*, 120(1): 1-7.

Flores-Verdugo, F. J., Day, J. W. Jr. and Briseño-Dueñas, R. 1987. Structure, litter fall, decomposition, and detritus dynamics of mangroves in a Mexican coastal lagoon with an ephemeral inlet. *Marine Ecology Progress Series*, 35:83–90.

Flores-Verdugo, F. J., Gonzalez-Farias, F., Ramirez-Flores, O., Amezcua-Linares, F., Yañez-Arancibia, A., Alvarez-Rubio, M. and Day, J. W. Jr. 1990. Mangrove ecology, aquatic primary productivity, and fish community dynamics in the Teacapán-Agua Brava lagoon-estuarine system (Mexican Pacific). *Estuaries*, 13:219–230.

Flower, J. and Imbert, D. 2006. Recovery deficiency following tree mortality in mangroves of two Caribbean islands: field survey and statistical classification. *Wetlands Ecology and Management*, 14: 185-199.

Forest Department of Kenya. 1983. Operational cruise reports on Lamu mangroves. Forest Department, Inventory Section, Nairobi, Kenya: 114 pp.

Fortes, M. D. 1988. Mangrove and seagrass beds of East-Asia - Habitats under stress. *Ambio*, 17(3): 207-213.

Fry, B., Bern, A. L., Ross, M. S. and Meeder, J. F. 2000. $\delta^{15}N$ Studies of nitrogen use by the red mangrove, *Rhizophora mangle* L. in South Florida. *Estuarine, Coastal and Shelf Science*, 50(2):291-296.

Gallin, E., Coppejans, E. and Beeckman, H. 1989. The Mangrove vegetation of Gazi Bay (Kenya). *Bulletin De La Societe Royale De Botanique De Belgique*, 122(2): 197-207.

Gang, P. O. and Agatsiva, J. L. 1992. The current status of mangroves along the Kenyan Coast - a case-study of Mida Creek mangroves based on remote-sensing. *Hydrobiologia*, 247(1-3): 29-36.

Gearing, P. J., Gearing, J. N., Maughan, J. T., Oviatt, C. A. 1991. Isotopic distribution of carbon from sewage sludge and eutrophication in the sediments and food web of estuarine ecosystems. *Environmental Science and Technology*, 25(2): 295-301.

Gilman, E., Ellison, J. and Coleman, R. 2007. Assessment of mangrove response to projected relative sea-level rise and recent historical reconstruction of shoreline position. *Environmental Monitoring and Assessment*, 124(1-3): 105-130.

Gilman, E. L., Ellison, J., Duke, N. C., and Field, C. 2008. Threats to mangroves from climate change and adaptation options: A review *Aquatic Botany*, 89(2): 237-250.

Golley, F., Odum, H. T. and Wilson, R. F. 1962. Structure and metabolism of a Puerto-Rican red mangrove forest in May. *Ecology*, 43(1): 9-&.

Gong, W. K. and Ong, J. E. 1990. Plant biomass and nutrient flux in a managed mangrove forest in Malaysia. *Estuarine Coastal and Shelf Science*, 31(5): 519-530.

GoK, Sessional Paper 1 (1968). A forest policy for Kenya. Sessional Paper No. 1 of 1968, Government Printers, Nairobi 9p.

GoK. 2002. Kenya’s first national communication on climate change. Government of Kenya Nairobi, Kenya.

GoK. 2007. Sessional Paper No. 1 of 2007. Forest Policy.; Ministry of Environment and Natural Resources. Government of Kenya, The Government Printers, Nairobi.

Greig-Smith, P. 1983. Quantitative plant ecology. Blackwell Scientific Publications. 1983, 3rd Edition, 359 pp.

Hamilton, L. S. and Snedaker, S. C. (Eds). 1984. Handbook for mangrove area management. IUCN/UNESCO/UNEP East West Centre, Honolulu, Hawaii.

Hardin, G. 1968. The tragedy of the commons. *Science*, 162: 1243-162.

Hauf, R. D., Ewel, K. C. and Jack, J.. 2006. Tracking human disturbance in mangroves: estimating harvest rates on a Micronesian Island. *Wetlands Ecology and Management*, 14: 95-105.

Henley, D. A. 1978. An investigation of proposed effluent discharge into a tropical mangrove estuary. In Proceeding of International Conference onWater Pollution Control in Developing Countries. September 1978, Thailand: 43–64.

Holdridge, L. R., Grenke, W. C., Hatheway, W. H., Liang, T. and Tosi, J. A. 1971. Forest environment in tropical life zones. Pergamon Press. NY 747 pages.

Holguin, G., Vazquez, P. and Bashan, Y. 2001. The role of sediment microorganisms in the productivity, conservation, and rehabilitation of mangrove ecosystems: an overview. *Biology and Fertility of Soils*, 33(4): 265-278.

Holguin, G., P. Gonzalez-Zamorano, L. E. De-Bashan, R. Mendoza, E. Amador and Y. Bashan. 2006. Mangrove health in an and environment encroached by urban development - a case study. *Science of the Total Environment*, 363(1-3): 260-274.

Holmboe, N., Kristensen, E. and Andersen, F. O. 2001. Anoxic decomposition in sediments from a tropical mangrove forest and the temperate Wadden Sea: Implications of N and P addition experiments. *Estuarine Coastal and Shelf Science*, 53(2): 125-140

Hovenden, M. J. and Allaway, W. G. 1994. Horizontal structures on pneumatophores of *Avicennia marina* (Forsk.) Vierh - a new site of oxygen conductance. *Annals of Botany*, 73(4): 377-383.

Huett, D. O., Morris, S. G., Smith, G. and Hunt, N. 2005. Nitrogen and phosphorus removal from plant nursery runoff in vegetated and unvegetated subsurface flow wetlands. *Water Research*, 39(14):3259-3272

Hutchings, P. A. and Saenger, P. 1987. Ecology of mangroves. University of Queensland Press, St. Lucia, Queensland.

Imai, N., Takyu, M., Nakamura, Y. and Nakamura, T. 2006. Gap formation and regeneration of tropical mangrove forests in Ranong, Thailand. *Plant Ecology*, 186: 37 –46

Ishshalomgordon, N. and Dubinsky, Z. 1992. Ultrastructure of the pneumatophores of the mangrove *Avicennia marina*. *South African Journal of Botany*, 58(5): 358-362.

IUCN. 2001. IUCN Red List Categories and Criteria: Version 3.1. IUCN. Species Survival Commission. IUCN, Gland, Switzerland and Cambridge, UK. ii + 30 pp.

IUCN-WCPA. 2008. Establishing marine protected areas networks – making it happen. Washington D.C.:IUCN-WCPA National Oceanic and Atmospheric Administration and the Nature Conservancy. 118 p.

Jimenez, J. A., Lugo, A. E. and Cintron, G. 1985. Tree mortality in mangrove forests. *BlOTROPlCA*, 17(3): 177-185.

Juncosa, A. M. and Tomlinson, P. B. 1985. Floral development in mangrove Rhizophoraceae. *American Journal of Botany*, 72(6): 818-818.

Jupiter, S. D., Potts, D. C., Phinn, S. R., and Duke, N. C. 2007. Natural and anthropogenic changes to mangrove distributions in the Pioneer River Estuary (QLD, Australia). *Wetlands Ecology and Management*, 15:51–62.

Kairo, J. G. 1995. Artificial regeneration and sustainable yield management of mangrove forests in Gazi Bay, Kenya. MSc. Thesis, University of Nairobi, Botany Department, 116 pages.

Kairo, J. G. 2001. Ecology and restoration of mangrove systems in Kenya. Laboratory of General Botany and Nature Management (APNA). Free University of Brussels, Belgium.

Kairo, J. G., Dahdouh-Guebas, F., Bosire, J. and Koedam, N. 2001. Restoration and management of mangrove systems - a lesson for and from the East African region. *South African Journal of Botany*, 67(3): 383-389.

Kairo, J. G., Dahdouh-Guebas, F., Gwada, P. O., Ochieng, C. and Koedam, N. 2002a. Regeneration status of mangrove forests in Mida Creek, Kenya: A compromised or secured future? *Ambio*, 31(7-8): 562-568.

Kairo, J. G., Kivyatu, B. and Koedam, N. 2002b. Application of remote sensing and GIS in the management of mangrove forests within and adjacent to Kiunga Marine Protected Area, Lamu, Kenya. *Environmental Devevelopment and Sustainability*, 4(2): 153-166.

Kairo, J. G., Bosire, J. and Mohamed, M. O. S. 2005. Assessment of the effects of oil spill on the mangrove forests of Port Reitz, Mombasa. Kenya Marine and Fisheries Research Institute. Mangrove System Information Service, Mombasa.

Kairo, J. G. and Dahdouh-Guebas F. Mangroves of Kenya. In: Principles for a code of conduct for the sustainable management of mangrove ecosystems. Macintosh D. J. and Ashton, E. C. (ed.), International Society of Mangrove Ecosystems (ISME), Okinawa, Japan. (in press).

Kairo, J. G., Lang'at, J. K. S., Dahdouh-Guebas, F., Bosire, J. and Karachi, M. 2008. Structural development and productivity of replanted mangrove plantations in Kenya. *Forest Ecology and Management*, 255(7): 2670-2677.

Kareiva, P., Chang A.,and Marvier M. 2008. Development and conservation goals in World Bank projects. *Science*, 321: 1638-1639.

Kathiresan, K. and Bingham, B. L. 2001. Biology of mangroves and mangrove ecosystems. *Advances in Marine Biology*, Vol 40: 81-251.

Kathiresan, K. and Rajendran, N. 2005a. Coastal mangrove forests mitigated tsunami. *Estuarine, Coastal and Shelf Science*, 65: 601-606.

Kathiresan, K. and Rajendran, N. 2005b. Mangrove ecosystems of the Indian Ocean region. *Indian Journal of Marine Sciences*, 34(1): 104-113.

Kimani, E. N., Mwatha, G. K., Wakwabi, E. O., Ntiba, J. M. and Okoth, B. K. 1996. Fishes of a shallow tropical mangrove estuary, Gazi, Kenya. *Marine and Freshwater Research*, 47(7): 857-868.

King, D. M. 1991. Costing out restoration. *Restoration and Management Notes*, 9: 15–21.

Kirui, B. Y. K., Huxham, M., Kairo J. and Skov, M. 2008. Influence of species richness and environmental context on early survival of replanted mangroves at Gazi Bay, Kenya. *Hydrobiologia*, 603: 171-181.

Kitheka, J. U. 1998. Groundwater outflow and its linkage to coastal circulation in a mangrove-fringed creek in Kenya. *Estuarine Coastal and Shelf Science*, 47(1): 63-75.

Kitheka, J. U., Ongwenyi, G. S. and Mavuti, K. M. 2003. Fluxes and exchange of suspended sediment in tidal inlets draining a degraded mangrove forest in Kenya. *Estuarine Coastal and Shelf Science*, 56(3-4): 655-667.

Koch, M. S. 1997. *Rhizophora mangle* L. seedling development into the sapling stage across resource and stress gradients in subtropical Florida. *BlOTROPlCA*, 29(4): 427-439.

Koch, M. S. and Snedaker, S. C. 1997. Factors influencing *Rhizophora mangle* L. seedling development in Everglades carbonate soils. *Aquatic Botany*, 59(1-2): 87-98.

Kokwaro, J. O. 1985. The distribution and economic importance of the mangrove forests of Kenya. *Journal of East African Natural History and Society*, 75: 1–10.

Komiyama, A., Ong, J. E. and Poungparn, S. 2008. Allometry, biomass and productivity of mangrove forests: a review. *Aquatic Botany*, 89: 128–137.

Krauss, K. W., Lovelock, C. E., McKee, K. L., López-Hoffman, L., Ewe, S. M. and Sousa, W. P. 2008. Environmental drivers in mangrove establishment and early development: a review. *Aquatic Botany*, 89: 105–127.

Kristensen, E. 2008. Mangrove crabs as ecosystem engineers; with emphasis on sediment processes. *Journal of Sea Research*, 59(1-2): 30-43.

Kristensen, E., Bouillon, S., Dittmar, T., and Marchand, C. 2008. Organic carbon dynamics in mangrove ecosystems: a review. *Aquatic Botany*, 89: 201-219.

Lacerda, L. D., C. E. V. Carvalho, K. F. Tanizaki, A. R. C. Ovalle and C. E. Rezende. 1993. The Biogeochemistry and Trace-Metals Distribution of Mangrove Rhizospheres. *Biotropica*, 25(3): 252-257.

Lang, G. E. and Knight, D. H. 1979. Decay rates for boles of tropical trees in Panama. *Biotropica*, 11: 316–317.

Lee, S. Y. 1998. Ecological role of grapsid crabs in mangrove ecosystems: a review. *Marine and Freshwater Research*, 49(4): 335-343.

Li, M. S. 1997. Nutrient dynamics of a Futian mangrove forest in Shenzhen, South China. *Estuarine Coastal and Shelf Science*, 45(4): 463-472.

Lieth H., Berlekamp J., Fuest S. and Riediger S. 1999. Climate diagrams of the world. CD-series: climate and biosphere. Leiden, the Netherlands: Blackhuys Publishers.

Lin, G. H. and Sternberg, L. D. L. 1994. Utilization of surface-water by red rangrove (*Rhizophora mangle* L) - an isotopic study. *Bulletin of Marine Science*, 54(1): 94-102.

Lindquist, E. S. and Carroll, C. R. 2004. Differential and seedling predation by crabs: Impacts tropical coastal forest composition. *Oecologia*, 661–671.

López-Hoffman, L., Monroe, I. E., Narváez, E., Martínez-Ramos, M., and Ackerly, D. D. 2006. Sustainability of mangrove harvesting: how do harvesters' perceptions differ from ecological analysis? *Ecology and Society*, 11(2): 14.

López-Hoffman, L., Ackerly, D. D., Anten, N. P. R., Denoyer, J. L. and Martinez-Ramos, M. 2007. Gap-dependence in mangrove life-history strategies: a consideration of the entire life cycle and patch dynamics. *Journal of Ecology*, 95 (6): 1222-1233.

Lovelock, C. E. and Feller, I. C. 2003. Photosynthetic performance and resource utilization of two mangrove species coexisting in a hypersaline scrub forest. *Oecologia*, 134(4): 455-462.

Lovelock, C. E., Ball, M. C., Choat, B., Engelbrecht, B. M. J., Holbrook, N. M. and Feller, I. C. 2006. Linking physiological processes with mangrove forest structure: phosphorus deficiency limits canopy development, hydraulic conductivity and photosynthetic carbon gain in dwarf Rhizophora mangle. *Plant Cell and Environment*, 29(5): 793-802

Lugo, A. E. and Snedaker, S. C. 1973. The role of mangrove ecosystems: properties of a mangrove forest in south Florida. National Technical Information Service, U.S. Department of Commerce, Springfield, Virginia (Final Report. No. DI-SFEP-74-73).

Lugo, A. E. and Snedaker, S. C. 1974. The ecology of mangroves. *Annual Review of Ecology and Systematics*, 5: 39–64.

Lugo, A. E. 1980. Mangrove ecosystems - successional or steady-state. *BIOTROPICA*, 12: 65-72.

Lugo, A. E., Cintrón, G. and Goenaga, C. 1981 Mangrove ecosystems under stress. In Stress Effects on Natural Ecosystems (Barret, G. W. and Rosenberg, R., eds). Wiley, New York, pp. 129–153.

Lugo, A. E., Wang, D. and Bormann, F. H. 1990. A Comparative analysis of biomass production in 5 tropical tree species. *Forest Ecology and Management*, 31(3): 153-166.

Lugo, A. E.1997. Old-growth mangrove forests in the United States. *Conservation Biology*, 11(1): 11-20.

Luvanda, A. M., Mbuvi, M. T. E. and Wandabwa, A. 1997. The status of mangrove exploitation and trade along the Kenya coastline. A Technical Report, Kenya Forestry Research Institute (KEFRI).

MacArthur, R. H. 1972. Coexistence of species. in J. Benke, ed., Challenging Biological Problems. Oxford. Pp. 253-259

Mackey, A. P. and Smail, G. 1995. Spatial and temporal variation in litter fall of *Avicennia marina* (Forssk.) Vierh in the Brisbane-River, Queensland, Australia. *Aquatic Botany*, 52(1-2): 133-142.

Macnae, W. 1968. A general account of fauna and flora of mangrove swamps and forests in Indo-West-Pacific region. *Advances in Marine Biology*, 6: 73-&.

Mandura, A. S. 1997. A mangrove stand under sewage pollution stress: Red Sea. *Mangrove Salt Marshes*, 1: 255–262.

MacFarlane, G. R., and Burchett, M. D. 2002. Toxicity, growth and accumulation relationships of copper, lead and zinc in the grey mangrove *Avicennia marina* (Forsk.) Vierh. *Marine Environmental Research*, 54(1): 65-84.

McGuinness, K. A. 1997a. Dispersal, establishment and survival of *Ceriops tagal* propagules in a north Australian mangrove forest. *Oecologia*, 109(1): 80-87.

McGuinness, K. A., 1997b. Seed predation in Australian mangrove forest; a test of the dominance predation model. *Journal of Tropical Ecology*, 293–302.

Mckee, K. L.1995. Seedling recruitment patterns in a Belizean mangrove forest - effects of establishment ability and physicochemical factors. *Oecologia*, 101(4): 448-460.

McKee, K. L. 1996. Growth and physiological responses of neotropical mangrove seedlings to root zone hypoxia. *Tree Physiology*, 16(11-12): 883-889.

McKee, K. L., Feller, I. C., Popp, M. and Wanek, W. 2002. Mangrove isotopic (δ^{15}N and δ^{13}C) fractionation across a nitrogen vs. phosphorus limitation gradient. *Ecology*, 83(4): 1065-1075.

Mcmillan, C. 1971. Environmental factors affecting seedling establishment of black mangrove on central texas coast. *Ecology*, 52(5): 927-&.

Macnae, W. 1968. A general account of fauna and flora of mangrove swamps and forests in Indo-West-Pacific region. *Advances in Marine Biology*, 6:73.

Marguillier, S., vanderVelde, G., Dehairs, F., Hemminga, M. A. and Rajagopal, S. 1997. Trophic relationships in an interlinked mangrove-seagrass ecosystem as traced by $\delta^{13}C$ and $\delta^{15}N$. *Marine Ecology-Progress Series*, 151(1-3): 115-121.

Medina, E. and Francisco, M. 1997. Osmolality and $\delta^{13}C$ of leaf tissues of mangrove species from environments of contrasting rainfall and salinity. *Estuarine Coastal and Shelf Science*, 45(3): 337-344.

Meziane, T. and Tsuchiya, M. 2002. Organic matter in a subtropical mangrove-estuary subjected to wastewater discharge: Origin and utilisation by two macrozoobenthic species. *Journal of Sea Research*, 47: 1-11.

Minchinton, T. E. 2001. Canopy and substratum heterogeneity influence recruitment of the mangrove *Avicennia marina*. *Journal of Ecology*, 89: 888-902.

Minagawa, M., and Wada, E. 1984. Stepwise enrichment of N^{-15} along food chains - further evidence and the relation between $\delta^{15}N$ and animal age. *Geochimica Et Cosmochimica Acta*, 48(5): 1135-1140.

Mirera, H. O. D., Kairo, J. G., Kimani, N. E. and Waweru, K. F. 2007. Seasonal variation in fish abundance in mangrove ecosystems: Comparing forested and un-forested habitats. *Bulletin of Marine Science*, 80(3): 929-929.

Mohamed, M. O. S., Neukermans, G., Kairo, J. G., Dahdouh-Guebas, F. and Koedam, N. 2008. Mangrove forests in a peri-urban setting: the case of Mombasa (Kenya). *Wetlands Ecology Management*, DOI 10.1007/s11273-008-9104-8.

Morell, J. M. and Corredor, J. E. 1993. Sediment nitrogen trapping in a mangrove lagoon. *Estuarine Coastal and Shelf Science*, 37(2): 203-212.

Morisita, M. 1959. Measuring of the dispersion of individuals and analysis of the distribution patterns. Mem. Fac. Sci. Kyushu Univ. Ser. E. Biol. 2: 215–235.

Moritz-Zimmermann, A., McGuinness, K. A. and Kuppers, M. 2002. Impacts of urban storm-water drainage channels on a northern Australian mangrove forest. *Trees Structure and Function*, 16(2-3): 195-203.

Mueller-Dombois, D. and Ellenberg, H. 1974. Aims and methods of vegetation ecology. Wiley and Sons Inc. New York, U.S.A.

Munga, D., Yobe, A. C., Owili, M. and Mwaguni, S. M. 1993. Assessment of land based sources of pollution along the Kenyan coast. Report prepared for the WHO, Kenya Marine and Fisheries Research Institute, Kenya.

Muzuka, A. N. N. and Shunula, J. P. 2006. Stable isotope compositions of organic carbon and nitrogen of two mangrove stands along the Tanzanian coastal zone. *Estuarine Coastal and Shelf Science*, 66(3-4): 447-458.

Mwaguni, S. and Munga, D. 1997. Land based sources and activities affecting the quality and uses of the marine coastal and associated freshwater environments along the Kenya coast. Coastal Development Authority, Mombasa.

Mwangi, S., Kirugara, D., Osore, M., Njoya, J., Yobe, A., and Dzeha, T. 1999. Status of marine pollution in Mombasa Marine Park and Reserve and Mtwapa creek. A Technical report; Kenya Marine and Fisheries Research Institute, Government Chemist Department, Kenya Wildlife Service.

Nagelkerken, I., van der Velde, G., Gorissen, M. W., Meijer, G. J., van't Hof, T. and den Hartog, C. 2000. Importance of mangroves, seagrass beds and the shallow coral reef as a nursery for important coral reef fishes, using a visual census technique. *Estuarine Coastal and Shelf Science*, 51(1): 31-44.

Nagelkerken, I., Blaber, S., Bouillon, S., Green, P., Haywood, M., Kirton, L. G., Meynecke, J.-O., Pawlik, J., Penrose, H. M., Sasekumar, A., Somerfield, P. J. 2008. The habitat function of mangroves for terrestrial and marina fauna: a review. *Aquatic Botany*, 89: 155–185.

Naidoo, G. and Von-Willert D. J. 1994. Stomatal oscillations in the mangrove *Avicennia germinans*. *Functional Ecology,* 8(5): 651-657.

Nedwell, D. B. 1975. Inorganic nitrogen-metabolism in a eutrophicated tropical mangrove estuary. *Water Research* 9(2): 221-231.

Newell, R. I. E., Marshall, N., Sasekumar, A. and Chong, V. C. 1995. Relative importance of benthic microalgae, phytoplankton, and mangroves as sources of nutrition for penaeid prawns and other coastal invertebrates from Malaysia. *Marine Biology*, 123(3): 595-606.

Nga, B. T., Tinh, H. Q., Tam, D. T., Scheffer M. and Roijackers, R. 2005. Young mangrove stands produce a large and high quality litter input to aquatic systems. *Wetlands Ecology and Management*, 13: 569–576

Nguli, M. M. 2006. Water exchange and circulation in selected Kenyan creeks. PhD. Dissertation, Earth Science Centre, Goteborg University, C78 2006.

Obura, D. O. 2001. Kenya. *Marine Pollution Bulletin*, 42(12): 1264-1278

Ochieng, C. A., and Erftemeijer, P. L. A. 2002. Phenology, litter fall and nutrient resorption in *Avicennia marina* (Forsk.) Vierh in Gazi Bay, Kenya. *Trees*, 16:167–171.

Odum, W. E. and Heald E. J. 1972. Trophic analyses of an estuarine mangrove community. *Bulletin of Marine Science*, 22(3): 671-&.

Ong, J. E. 1995. The ecology of mangrove conservation and management. *Hydrobiologia*, 295(1-3): 343-351.

Onuf, C. P., Teal, J. M. and Valiela, I. 1977. Interactions of nutrients, plant-growth and herbivory in a mangrove ecosystem. *Ecology*, 58(3): 514-526.

Osborne, D. J. and Berjak, P. 1997. The making of mangroves: the remarkable pioneering role played by seeds of *Avicennia marina*. *Endeavour*, 21(4): 143-147.

Osborne, K. and Smith III, T. J. 1990. Differential predation on mangrove propagules in open and closed canopy forest habitats. *Vegetatio*, 89(1): 1-6.

Othman, J., Bennett, J. and Blamey, R. 2004. Environmental values and resource management options: a choice modelling experience in Malaysia. *Environmental Development Economics*, 9: 803–824.

Paijmans, K. and Rollet, B. 1977. The mangroves of Galley Reach, Papua New Guinea. *Forest Ecology and Management*, 1:119-140.

Pape, E., Muthumbi, A., Kamanu, C. P., Vanreusel, A., 2008. Size-dependent distribution and feeding habits of *Terebralia palustris* in mangrove habitats of Gazi Bay, Kenya. *Estuarine Coastal and Shelf Science*, 76: 797-808.

Passioura, J. B., Ball, M. C. and Knight, J. H. 1992. Mangroves may salinize the soil and in so doing limit their transpiration rate. *Functional Ecology*, 6(4): 476-481.

Pennock, J. R., Velinsky, D. J., Ludlam, J. M., Sharp, J. H. and Fogel, M. L. 1996 Isotopic fractionation of ammonium and nitrate during uptake by *Skeletonema costatum*: Implications for δ^{15}N dynamics under bloom conditions. *Limnology and Oceanography*, 41: 451–459.

Peterson, B. J. and Fry, B. 1987. Stable isotopes in ecosystem studies. *Annual Review of Ecology and Systematics*, 18: 293-320.

Pinzón, Z. S., Ewel, K. C. and Putz, F. E. 2003. Gap formation and forest regeneration in a Micronesian mangrove forest. *Journal of Tropical Ecology*, 19:143-153.

Pool, D. J., Snedaker, S. C. and Lugo, A. E. 1977. Structure of mangrove morests in Florida, Puerto-Rico, Mexico, and Costa-Rica. *BIOTROPICA*, 9(3): 195-212.

Primavera, J. H. 1995. Mangroves and brackish water pond culture in the Philippines. *Hydrobiologia*, 295: 303–309.

Primavera, J. H. 1996. Stable carbon and nitrogen isotope ratios of penaeid juveniles and primary producers in a riverine mangrove in Guimaras, Philippines. *Bulletin of Marine Science*, 58(3): 675-683.

Primavera, J. H. 1997. Socio-economic impacts of shrimp culture. *Aquaculture Research*, 28(10): 815-827.

Primavera, J. H. 1998. Mangroves as nurseries: shrimp populations in mangrove and non-mangrove habitats. *Estuarine Coastal and Shelf Science*, 46(3): 457-464.

Primavera, J. H. 2000. Development and conservation of Philippine mangroves: institutional issues. *Ecological Economics*, 35(1): 91-106.

PUMPSEA. 2003. Peri-urban Mangroves Forests as Filters and Potential Phytoremediators of Domestic Sewage in East Africa. Proposal at Kenya Marine and Fisheries Research Institute.

PUMPSEA. 2007a. D5 – Geographical differentiation of biogeochemical transformations in mangrove compartments exposed to selected sewage doses and compositions. Biological Institute from Southern Denmark University, Report for the Project: Peri-urban mangrove forests as filters and potential phytoremediators of domestic sewage in East Africa project, http://www.pumpsea.icat.fc.ul.pt/restricted.php

PUMPSEA. 2007c. D23 Changes to ecosystem structure, biodiversity, tree productivity and biomass composition caused by sewage, Report for the Project: Peri-urban mangrove forests as filters and potential phytoremediators of domestic sewage in East Africa project, http://www.pumpsea.icat.fc.ul.pt/restricted.php

PUMPSEA. 2007b. D19 – Consequences of sewage exposure on microbial distribution and diversity and the balance between microbial respiration types. Vrije Universiteit Brussel, Report for the Project: Peri-urban mangrove forests as filters and potential phytoremediators of domestic sewage in East Africa project, http://www.pumpsea.icat.fc.ul.pt/restricted.php

Putz, F. E. and Chan, H. T. 1986. Tree growth, dynamics, and productivity in a mature mangrove forest in Malaysia. *Forest Ecology and Management*, 17(2-3): 211-230.

Rabinowitz, D. 1978. Early growth of mangrove seedlings in Panama, and an hypothesis concerning relationship of dispersal and zonation. *Journal of Biogeography*, 5(2): 113-133.

Rao, R. G., Woitchik, A. F., Goeyens, L. van Riet, A., Kazungu, J. and Dehairs, F. 1994. Carbon, nitrogen contents and stable carbon isotope abundance in mangrove leaves from an East African coastal lagoon (Kenya). *Aquatic Botany*, 47(2): 175-183.

Rawlins, P. 1957. The East African mangrove trade. Unpublished typescript in the East African Herbarium, Nairobi: 30 pp.

Reddy, S. R. C. and Chakravarty S. P. 1999. Forest dependence and income distribution in a subsistence economy: evidence from India. *World Development*, 27 (7): 1141-1149.

Rees, J. G., Williams, T. M., Nguli, M. M., Kairu, K. K. and Yobe, A. C. 1996. Contaminant transport and storage in the estuarine creek systems of Mombasa, Kenya. British Geological Survey Technical Report WC/96/42.

Robertson, W. B. 1962. Fire and vegetation in the Everglades. Pages 67–80 in Proceedings of the first annual Tall Timbers fire ecology conference. Tall Timbers Research Station, Tallahassee, Florida.

Robertson, A. I. 1991. Plant animal interactions and the structure and function of mangrove forest ecosystems. *Australian Journal of Ecology*, 16(4): 433-443.

Rönnbäck, P. 1999. The ecological basis for economic value of seafood production supported by mangrove ecosystems. *Ecological Economics*, 29(2): 235-252.

Rönnbäck, P., Troell, M., Zetterstrom, T. and Babu, D. E. 2003. Mangrove dependence and socio-economic concerns in shrimp hatcheries of Andhra Pradesh, India. *Environmental Conservation*, 30(4): 344-352.

Rönnbäck, P., Crona, B. and Ingwall, L. 2007. The return of ecosystem goods and services in replanted mangrove forests: perspectives from local communities in Kenya. *Environmental Conservation*, 34(4): 313-324.

Roth, L. C. 1992. Hurricanes and mangrove regeneration - effects of hurricane Joan, October 1988, on the vegetation of Isla Del Venado, Bluefields, Nicaragua. *BIOTROPICA*, 24(3): 375-384.

Saenger, P., Hegerl, E. J. and Davie, J. D. S. 1983. Global status of mangrove ecosystems. *The Environment*, 3 (3): 88.

Saenger, P. and Snedaker, S. C. 1993. Pantropical trends in mangrove aboveground biomass and annual litter fall. *Oecologia*, 96(3): 293-299.

Saenger, P. 1998. Mangrove vegetation: an evolutionary perspective. *Marine and Freshwater Research*, 49(4): 277-286.

Saifullah, S. M. and Elahi, E. 1992. Pneumatophore density and size in mangroves of Karachi, Pakistan. *Pakistan Journal of Botany*, 24(1): 5-10.

Saifullah, S. M., Khafaji, A. K. and Mandura, A. S. 1989. Litter production in a mangrove stand of the Saudi-Arabian Red-Sea coast. *Aquatic Botany*, 36(1): 79-86.

Saintilan, N., K. Hossain and D. Mazumder. 2007. Linkages between seagrass, mangrove and saltmarsh as fish habitat in the Botany Bay estuary, New South Wales. *Wetlands Ecology and Management*, 15; 277-286.

Sasekumar, A. and Loi, J. J. 1983. Litter production in 3 mangrove forest zones in the Malay Peninsula. *Aquatic Botany*, 17(3-4): 283-290.

Semesi, A. K. 1998. Mangrove management and utilization in Eastern Africa. *Ambio*, 27(8): 620-626.

Sengupta, R., Middleton, B., Yan, C., Zuro M., and Hartman, H. 2005. Landscape characteristics of *Rhizophora mangle* forests and propagule deposition in coastal environments of Florida (USA). *Landscape Ecology*, 20(1): 63-72.

Sheppard, C. 2001. The main issues affecting the coasts of the Indian and western Pacific oceans: A meta-analysis from seas at the millennium. *Marine Pollution Bulletin*, 42 (12): 1199–1207.

Sherman, R. E., Fahey T. J. and Battles J. J. 2000. Small-scale disturbance and regeneration dynamics in a neotropical mangrove forest. *Journal of Ecology*, 88(1): 165-178.

Silva, C. A. R., Lacerda, L. D., and Rezende, C. E. 1990. Metals reservoir in a red mangrove forest. *Biotropica*, 22(4): 339-345.

Sjöling, S., Mohammed, S. M., Lyimo, T. J. and Kyaruzi, J. J. 2005. Benthic bacterial diversity and nutrient processes in mangroves: impact of deforestation. *Estuarine Coastal and Shelf Science*, 63(3): 397-406.

Slim, F. J., Gwada, P. M., Kodjo, M. and Hemminga, M. A. 1996. Biomass and litter fall of *Ceriops tagal* and *Rhizophora mucronata* in the mangrove forest of Gazi Bay, Kenya. *Marine and Freshwater Research*, 47(8): 999-1007.

Smith III, T. J. 1987a. Effects of seed predators level on the distribution of *Avicennia marina* vierh. in tropical, tidal forests. *Estuarine, Coastal Shelf Science*, 25: 43–51.

Smith III, T. J. 1987b. Seed predation in relation to tree dominance and distribution in mangrove forests. *Ecology*, 68(2): 266-273.

Smith III, T. J. and Duke, N. C. 1987. Physical determinants of inter-estuary variation in mangrove species richness around the tropical coastline of Australia. *Journal of Biogeography*, 14(1): 9-19.

Smith III, T. J. 1988. The influence of seed predators on structure and succession in tropical tidal forests. *Proceedings of the Ecological Society of Australia*, 15: 203-211.

Smith, P. T. 1996. Physical and chemical characteristics of sediments from prawn farms and mangrove habitats on the Clarence River, Australia. *Aquaculture*, 146(1-2): 47-83.

Snedaker, S. C., Brown, M. S., Lahmann, E. J., and Araujo, R. J. 1992. Recovery of a mixed-species mangrove forest in South-Florida following canopy removal. *Journal of Coastal Research*, 8(4): 919-925.

Snedaker, S. C. and Lahmann, E. J. 1988. Mangrove understorey absence - a consequence of evolution. *Journal of Tropical Ecology*, 4: 311-314

Spalding, M., Blasco, F., and Field, C. 1997. World mangrove atlas. International Society for Mangrove Ecosystems, Okinawa, Japan.

SPEK. 1992. Mangrove forest of Mombasa. Technical Report by Society for Protection of Environment, Kenya.

Steinke, T. D. and| Ward, C. J. 1990. Litter production by mangroves: III. Wavecrest (Transkei) with predictions for other Transkei Estuaries. *South African Journal of Botany*, 56(5): 514-519.

Stoner, A. W. and Zimmerman, R. J. 1988. Food pathways associated with penaeid shrimps in a mangrove-fringed estuary. *Fishery Bulletin*, 86(3): 543-551.

Sukardjo, S. 1987. Natural regeneration status of commercial mangrove species (*Rhizophora apiculata* and *Bruguiera gymnorrhiza*) in the mangrove forest of Tanjung Bungin, Banyuasin District, South Sumatra. *Forest Ecology and Management*, 20(3-4): 233-252.

Tam, N. F. Y., Wong, Y. S., Lan, C. Y., Wang, L. N. 1998. Litter production and decomposition in a subtropical mangrove swamp receiving wastewater. *Journal of Experimental Marine Biology and Ecology*, 226(1): 1-18.

Tam, N. F. Y. and Wong, Y. 1997. Accumulation and distribution of heavy metals in a simulated mangrove system treated with sewage. *Hydrobiologia* 352: 67–75.

Tam, N. F. Y. and Wong, Y. S. 1999. Mangrove soils in removing pollutants from municipal wastewater of different salinities. *Journal of Environmental Quality*, 28(2): 556-564.

Tam, N. F. Y. and Yao, M. W. Y. 1999. Three digestion methods to determine concentrations of Cu, Zn, Cd, Ni, Pb, Cr, Mn, and Fe in mangrove sediments from Sai Keng, Chek Keng, and Sha Tau Kok, Hong Kong. *Bulletin of Environmental Contamination and Toxicology*, 62(6): 708-716.

Tamooh, F., Huxham, M., Karachi, M., Mencuccini, M., Kairo, J. G. and Kirui, B. 2008. Below-ground root yield and distribution in natural and replanted mangrove forests at Gazi bay, Kenya. *Forest Ecology and Management*, 256(6): 1290-1297.

Tamura, T., Hatano, K. and Suzuki, K. 2006. A new genus of the family Micromonosporaceae, Polymorphospora gen. nov., with description of *Polymorphospora rubra* sp nov. *International Journal of Systematic and Evolutionary Microbiology*, 56: 1959-1964.

Tang, H. T. 1978. Regeneration stocking adequacy standards. *Malaysian Forester*, 41: 176–182.

Taylor, M., Ravilious, C., and Green, E. P. 2003. Mangroves of East Africa. UNEP-WCMC Biodiversity Series No. 13.

Terchunian, A., Klemas, V., Segovia, A., Alvarez, A., Vasconez, B. and Guerrero, L. 1986. Mangrove mapping in Ecuador - the impact of shrimp pond construction. *Environmental Management*, 10(3): 345-350.

Terrados, J., Thampanya, U., Srichai, N., Kheowvongsri, P., Geertz-Hansen, O., Boromthanarath, S., Panapitukkul, N. and Duarte, C. M. 1997. The effect of increased sediment accretion on the survival and growth of Rhizophora apiculata seedlings. *Estuarine Coastal and Shelf Science*, 45(5): 697-701.

Thampanya, U., Vermaat, J. E. and Terrados, J. 2002. The effect of increasing sediment accretion on the seedlings of three common Thai mangrove species. *Aquatic Botany*, 74: 315–325.

Thampanya, U., Vermaat, J. E., Sinsakul, S. and Panapitukkul, N. 2006. Coastal erosion and mangrove progradation of Southern Thailand. *Estuarine Coastal and Shelf Science*, 68(1-2): 75-85.

Thom, B. G. 1967. Mangrove ecology and deltaic geomorphology - Tabasco Mexico *Journal of Ecology*, 55(2): 301-&

Tomlinson, P. B., Primack, R. B. and Bunt, J. S. 1979. Preliminary observations on floral biology in mangrove Rhizophoraceae. *Biotropica*, 11(4): 256-277.

Tomlinson, P. B. 1986. The botany of mangroves. in Cambridge University Press, Cambridge.

Tomlinson, P. B. and Cox, P. A. 2000. Systematic and functional anatomy of seedlings in mangrove Rhizophoraceae: vivipary explained? *Botanical Journal of the Linnean Society*, 134(1-2): 215-231.

Tong, P. H. S., Auda, Y., Populus, J., Aizpuru, M., Al Habshi, A. and Blasco, F. 2004. Assessment from space of mangroves evolution in the Mekong Delta, in relation to extensive shrimp farming. *International Journal of Remote Sensing*, 25(21): 4795-4812.

Trattner, R. B. and Woods, S. J. W. 1989. The use of wetlands systems for municipal wastewater treatment. In Cheremisinoff, P. N. (ed.), Encyclopedia of Environmental Control Technology Vol. 3, Wastewater Treatment Technology. Gulf Publishing Co., Houston TX: 591–622.

Triest, L., 2008. Molecular ecology and biogeography of mangrove trees towards conceptual insights on gene flow and barriers: a review. *Aquatic Botany*, 89: 138–154.

Twilley, R. R., Lugo, A. E. and Pattersonzucca, C. 1986. Litter production and turnover in basin mangrove forests in Southwest Florida. *Ecology*, 67(3): 670-683.

Twilley, R. R., Chen, R. H. and Hargis, T. 1992. Carbon sinks in mangroves and their implications to carbon budget of tropical coastal ecosystems. *Water Air and Soil Pollution*, 64(1-2): 265-288.

Twilley, R. R., Snedaker, S. C., Yañez-Arancibia, A. and Medina, E. 1995. Biodiversity and ecosystem functioning: ecosystem analysis. Mangrove systems. In: Heywood VH, editor. Global biodiversity assessment, UNEP. Cambridge: Cambridge University Press, p. 387–92.

Twilley, R. R., Pozo, M., Garcia, V. H., RiveraMonroy, V. H. and Bodero, R. Z. A. 1997. Litter dynamics in riverine mangrove forests in the Guayas River estuary, Ecuador. *Oecologia*, 111(1): 109-122.

Twilley, R. R. and Chen, R. H. 1998. A water budget and hydrology model of a basin mangrove forest in Rookery Bay, Florida. *Marine and Freshwater Research*, 49(4): 309-323

UNEP/FAO/PAP/CDA. 2000. Progress in Integrated Coastal Management for Sustainable Development of Kenya’s Coast. The Case of Nyali-Bamburi-Shanzu Area. East African Regional Seas Technical Reports Series No. 6: 66 pages.

UNEP/GPA. 2001. Consultative meeting on municipal wastewater. Eastern Africa. Dar es Salaam, Tanzania, June 11-15 2001. UNEP, Eastern Afrian Action Plan/Regional coordinating unit, The Hague.

UNEP-WCMC. 2006. In the front line: shoreline protection and other ecosystem services from mangroves and coral reefs. UNEP-WCMC, Cambridge, UK 33 pp.

Upadhyay, V.P., Ranjan, R. and Singh, J. S. 2002. Human-mangrove conflicts: The way out. *Current Science*, 83(11): 1328-1336.

Vaiphasa, C., de Boer, W. F., Skidmore, A. K., Panit-chart, S., Vaiphasa, T., Bamrongrugsa, N. and Santitamnont, P. 2007. Impact of solid shrimp pond waste materials on mangrove growth and mortality: a case study from Pak Phanang, Thailand. *Hydrobiologia*, 591; 47-57.

Valiela, I., Bowen, J. L. and York, J. K. 2001. Mangrove forests: One of the world's threatened major tropical environments. *Bioscience*, 51(10): 807-815.

Van Speybroeck, D. 1992. Regeneration strategy of mangroves along the Kenya Coast - a first approach. *Hydrobiologia*, 247(1-3): 243-251.

Victor, S., Neth, L., Gobuu, Y., Wolanski, E. and Richmond, R. H. 2003. Sedimentation in mangroves and coral reefs in a wet tropical island, Pohnpei, Micronesia. *Estuarine Coastal and Shelf Science*, 66(3-4): 409-416.

Wafar, S., Untawale, A. G. and Wafar, M. 1997. Litter fall and energy flux in a mangrove ecosystem. *Estuarine Coastal and Shelf Science*, 44(1): 111-124.

Waithaka, J. and Mwathe, K. 2003. Issues Impeding Forest Conservation and Management in Kenya. In Forests and Development: Investing in policy analysis, advocacy and monitoring to resolve forest conservation conflicts in Kenya (eds. D.L Nightingale). Nature Kenya Environmental Legislation and Policy Working Group Conservation Papers. Nature. Kenya, Nairobi.

Walker, B. and Salt, D. 2006. Resilience thinking: Sustaining ecosystems and people in a changing world. Island press, Washington D.C., 174 pages.

Walters, B. B. 1997. Human ecological questions for tropical restoration: experiences from planting native upland trees and mangroves in the Philippines. *Forest Ecology and Management*, 99(1-2): 275-290

Walters, B. B. 2003. People and mangroves in the Philippines: fifty years of coastal environmental change. *Environmental Conservation*, 30(3): 293-303.

Walters, B. B. 2004. Local management of mangrove forests in the Philippines: successful conservation or efficient resource exploitation? *Human Ecology*, 32(2): 177-195.

Walters, B. B. 2005a. Ecological effects of small-scale cutting of Philippine mangrove forests. *Forest Ecology and Management*, 206(1-3): 331-348.

Walters, B. B. 2005b. Patterns of local wood use and cutting of Philippine mangrove forests. *Economic Botany*, 59(1): 66-76.

Walters, B. B., Rönnbäck, P., Kovacs, J., Crona, B., Hussain, S., Badola, R., Primavera, J. H., Barbier, E. B. and Dahdouh-Guebas, F., 2008. Ethnobiology, socio-economics and adaptive management of mangroves: a review. *Aquatic Botany*, 89: 220–236.

Wanek, W., Hofmann, J., and Feller, I. C. 2007. Canopy interactions of rainfall in an off-shore mangrove ecosystem dominated by *Rhizophora mangle* (Belize). *Journal of Hydrology*, 345: 70– 79.

Watson, J. O. 1928. Mangrove forests of the Malay peninsula. *Malayan Forest Records* 6: 1-275.

Wattage, P. and Mardle, S. 2008. Total economic value of wetland conservation in Sri Lanka identifying use and non-use values. *Wetlands Ecology Management*, DOI 10.1007/s11273-007-9073-3

Werner, A. and Stelzer, R. 1990. Physiological responses of the mangrove *Rhizophora mangle* grown in the absence and presence of NaCl. *Plant Cell and Environment*, 13(3): 243-255.

Wittemyer G., Elsen P., Bean W. T., Coleman A., Burton O., and Brashares J. S. 2008. Accelerated human population growth at protected area edges. Science, 321: 123-126.

Wolanski, E. and Ridd, P. 1986. Tidal mixing and trapping in mangrove swamps. *Estuarine Coastal and Shelf Science*, 23(6): 759-771.

Wolanski, E. and Chappell, J. 1996. The response of tropical Australian estuaries to a sea level rise. *Journal of Marine Systems*, 7(2-4): 267-279.

Wolanski, E. 1992. Hydrodynamics of mangrove swamps and their coastal waters. *Hydrobiologia*, 247(1-3): 141-161.

Wong, Y. S., Tam, N. F. Y. and Lan, C. Y. 1997a. Mangrove wetlands as wastewater treatment facility: a field trial. *Hydrobiologia*, 352: 49–59.

Wong, Y., Tam, N. F. Y., Chen, G. and Ma, H. 1997b. Response of *Aegiceras corniculatum* to synthetic sewage under simulated tidal conditions. *Hydrobiologia*, 352: 89–96.

Woodroffe, C. D. 1985. Studies of a mangrove basin, Tuff Crater, New-Zealand .1. Mangrove biomass and production of detritus. *Estuarine Coastal and Shelf Science*, 20(3): 265-280.

Woodroffe, C. D., Bardsley, K. N., Ward P. J. and Hanley J. R. 1988. Production of mangrove litter in a macrotidal embayment, Darwin Harbor, Nt, Australia. *Estuarine Coastal and Shelf Science*, 26(6): 581-598.

Woodroffe, C. 1992. Introduction to World Vegetation - Collinson,As. *Australian Geographer*, 23(1): 88-89.

World Resources Institute. 1996. World Resources 1996 – 97. The World Resources Institute, UNEP, UNDP, World Bank, Oxford University Press, Oxford, p. 365.

Wu, Y., Tam, N. F. Y. and Wong, M. H. 2008. Effects of salinity on treatment of municipal wastewater by constructed mangrove wetland microcosms. *Marine Pollution Bulletin*, 57(6-12): 727-734

Yang, Q., Tam, N. F. Y., Wong, Y. S., Luan, T. G., Su, W. S., Lan, C. Y., Shin, P. K. S. and Cheung, S. G. 2008. Potential use of mangroves as constructed wetland for municipal, sewage treatment in Futian, Shenzhen, China. *Marine Pollution Bulletin*, 57(6-12): 735-743

Ye, Y., Tam, N. F. Y., and Wong, Y. S. 2001. Livestock wastewater treatment by a mangrove pot-cultivation system and the effect of salinity on the nutrient removal efficiency. *Marine Pollution Bulletin*, 42(6): 513-521.

Ye, Y., Tam, N. F. Y., Lu C. Y. and Wong Y. S. 2005. Effects of salinity on germination, seedling growth and physiology of three salt-secreting mangrove species. *Aquatic Botany*, 83(3): 193-205.

Yim, M. W. and Tam, N. F. Y. 1999. Effects of wastewater-borne heavy metals on mangrove plants and soil microbial activities. *Marine Pollution Bulletin*, 39(1-12): 179-186.

Yoneyama, T., Omata, T., Nakata, S. and Yazaki, J. 1991 Fractionation of nitrogen isotopes during the uptake and assimilation of ammonia by plants. *Plant and Cell Physiology*, 32:1211–1217.

Youngblood, A., Max, T. and Coe, K. 2004. Stand structure in eastside old-growth ponderosa pine forests of Oregon and northern California. Forest *Ecology and Management*, 199(1-2): 191-217.

Yu, R. Q., Chen, G. Z., Wong, Y. S., Tam, N. F. Y. and Lan, C. Y. 1997. Benthic macrofauna of the mangrove swamp treated with municipal wastewater. *Hydrobiologia*, 347: 127-137.

Zar, J. H. 1999. Biostatistical analysis. Published by Prentice Hall, Chapter 12, pages 268-270.

Zimmermann, U., Zhu, J. J., Meinzer, F. C., Goldstein, G., Schneider, H., Zimmermann, G., Benkert, R., Thurmer, F., Melcher, P., Webb, D. and Haase, A. 1994. High-molecular-weight organic-compounds in the xylem sap of mangroves - implications for long-distance water transport. *Botanica Acta*, 107(4): 218-229.

Zlotin, R. I. and Parmenter, R. R. 2008. Patterns of mast production in pinyon and juniper woodlands along a precipitation gradient in central New Mexico (Sevilleta National Wildlife Refuge). *Journal of Arid Environments*, 72: 1562– 1572.

ANNEX I

Statutes Relating to the Environment in Kenya

1. The Constitution
2. The Penal Code Cap.63
3. The Chief's Authority Act, Cap.128
4. The Public Health Act, Cap.242
5. The Radiation Protection Act Cap.243
6. The Local Government Act Cap.265
7. The Trust Land Act, Cap.288
8. The Land Planning Act, Cap.303
9. The Mining Act, Cap.306
10. The Petroleum (Exploration And Production) Act, Cap.308
11. The Agriculture Act, Cap.318
12. The Water Act, Cap.372
13. The Wildlife (Conservation And Management) Act, Cap.376
14. The Tourism Industry Act, Cap.385
15. The Forests Act, Cap.385
16. The Merchant Shipping Act, Cap.389
17. The Traffic Act, Cap.403
18. The Tourist Development Corporation Act, Cap.382
19. The Lake Basin Development Authority Act, Cap.442
20. The Kerio Valley Development Authority Cap.441
21. The Tana And Athi Rivers Development Authority Act, Cap.443
22. The Factories Act, Cap.514
23. The Coast Development Authority Act, No.20 Of 1990
24. The Fisheries Act, No.5 Of 1989
25. The Maritime Zones Act, Cap.371
26. The National Water Conservation Pipeline Corporation Act L/No.270 1988
27. Carriage Of Goods By Sea Act, Cap.392
28. The Timber Act, Cap.386
29. The Government Lands Act, Cap.280
30. The Registration Of Titles Act, Cap.281
31. The Land Titles Act, Cap.282
32. The Land Consolidation Act, Cap.283
33. The Land Adjudication Act, Cap.284
34. Registration Of Documents Act, Cap.285
35. Land (Group Representatives) Act, Cap.287
36. Mazrui Land Trusts Act, Cap.288
37. Equitable Mortgages Act, Cap.291
38. Way Leaves Act, Cap.292
39. Distress For Rent Act, Cap.293
40. Land Acquisition Act, Cap.295
41. Rent Restriction Act, Cap.296
42. Survey Act, Cap.299
43. Registered Land Act, Cap.300
44. Landlord And Tenant Act, Cap.301
45. Land Control Act, Cap.302
46. Mortgages (Special)Act, Cap.304
47. Lakes And Rivers Act, Cap.409
48. Grassfires Act, Cap.327
49. Crop Production And Livestock Development Act, Cap.321
50. Local Authorities (Recovery Of Possession Of Property)
51. Antiquities And Monuments Act, Cap.215
52. Occupiers Liability Act, Cap.34
53. Plant Protection Act, Cap.324
54. Fertilizers And Animal Foodstuffs Act, Cap.345
55. Town Planning Act, Cap. 134 (1948)
56. Fire Inquiry Act, Cap.103
57. Wakf Commissioners Act, Cap.109
58. Explosives Act, Cap.115
59. Petroleum Act, Cap.116
60. Housing Act, Cap.117
61. Methylated Spirit Act, Cap.120
62. Malaria Prevention Act, Cap.246
63. Use Of Poisonous Substances Act, Cap.247
64. Food, Drugs And Chemical Substances Act, Cap.254
65. Local Authorities Services Charge Act, Cap.274
66. Continental Shelf Act, Cap.312
67. Suppression Of Noxious Weeds Act, Cap.325
68. Coconut Preservation Act, Cap.332
69. Pests Control Products Act, Cap.346
70. Mineral Oil Act, Cap.307
71. Irrigation Act, Cap.347
72. Territorial Waters Act, Cap.371
73. Ewaso Ng'iro South River Basin Development Authority Act Cap.447
74. Ewaso Ng'iro North River Basin Development Authority Act, Cap.448
75. The Science And Technology Act, Cap.250
76. The National Museums Act, Cap.216.

ANNEX II :
Organizations Participating in Mangrove Conservation and Management in Kenya

Institution	Main Functions
Government Agencies	
Coast Development Authority	Coastal planning and development coordination
Kenya Marine and Fisheries Research Institute	Research on coastal and marine resources
Kenya Forestry Research Institute	Research on national forests
Kenya Wildlife Service	Conservation and management of biodiversity in the protected areas and critical habitats
Forest Department	Forests management in the country
Fisheries Department	Fisheries licensing, monitoring and policing
National Museums of Kenya	Conservation of historical sites. Also carries out research in conservation and management of biodiversity.
Department of Physical Planning	Provides physical plans, but does not execute the plans
National Environmental Management Authority	Coordination of all environmental matters in the country
Local Authorities (Municipalities, Township and County councils)	Approval of structures and delivery of services. Sits in the mangrove licensing committee
Water Department	National planning for both surface and ground water
Tourism Department	Tourism planning and licensing
National Universities (Nairobi, Maseno, Moi, Kenyatta, Egerton, Jomo Kenyatta,)	Research and training
National Oil Spill Response Committee	Oil spill response
Private Sector	
Kenya Association of Hotelkeepers and Caterers	Hotel marketing at national and regional levels. Encourage environmental conservation for better tourism
Mombasa and Coast Tourist Association	Marketing of coastal hotels services. Encourage ecotourism in mangrove areas
Boat Operators Association (in Malindi, Mombasa, and Diani)	Mostly local groups. Organize boat trips to mangroves and participate in planting
NGO	
Wildlife Clubs of Kenya	Youth conservation and education campaigns
East African Natural History Society	Research, Conservation and Education
East African Wildlife Society	Research, Conservation and Education
Society for the Protection of Environment	Community education and awareness
Baobab Trust	Environmental Conservation
Kenya Forestry Working Group	Institutional task force to lobby against conversion of forests to other land uses
Friends of Fort Jesus	Public awareness,
Coastal Forest Conservation Unit	Advisory and coordination for issues on coastal forests
Friends of Mangroves	Mangrove awareness to schools and colleges
Green Belt Movement	Planting of trees
Kenya Wetlands Working Group	Institutional task force for lobbying against reclamation of wetlands
Environmental Trust of Kenya	Conservation of urban environment, including mangroves

ANNEX IV: Questionnaire

Area/village Name: ____________________Code:_________ Interview No.:_________Date:__ __________ GPS code:__________________Picture No. [__________]

GPS Coordinates:__________________°E, __________________°S

Education [ ] Some Primary School education

[ ] Local education, specify ___________________________________

Preferred Language [ ] Swahili English [ ] Other [ ]

1. What do you understand by the term mangrove (mkokoni)?

[ ] The vegetation, the wood [ ] the ecosystem, the area

[ ] How can you explain the meaning of mangrove: [__________________]

2. Can you identify the species? (display pictures and leaf samples of different species)

[ ] *Avicennia marina*- mchu, mtu

[ ] *Bruguiera gymnorrhiza* - muia, mshinzi, mchofi, mkifu, mkoko wimbi

[ ] *Ceriops tagal* - mkandaa, mkoko mtune, mkoko mwekundu

[ ] *Rhizophora mucronata* - mkoko, mkoko mwenye

[ ] *Xylocarpus granatum* - mtonga

[ ] *Sonneratia alba* - mlilana, mpia, mpira

[ ] *Xylocarpus moluccensis* mkomafi

[ ] *Heritiera littoralis* mkungu, msikundazi, mkokoshi

[ ] *Lumnitzera racemosa* - kikandaa, mkaa pwani

3. What are the uses of mangroves within your household? If possible state the alternative to each use.

Use	For	Species	Part used & length or thickness	Alternative	Cost
Fuel wood	Firewood				
	Charcoal				
Timber	House constr.				
	Boat constr.				
	Fencing				
	Paddles				
	Boat masts				
Others	Fishing gear				
Medicinal products	Medicine				
Others	Ointments				
	Dyes Nets				
	Clothes				
Furniture	Chairs, tables, shelves				
	Utensils				
Food for Human					
Food for livestock					
Other uses					

4. Which of the above mentioned uses is predominant within your household and village?

5. Which species of mangroves can you say are most heavily exploited in order of importance and which ones are more difficult to obtain?

6. Where within the forest does the harvesting activity take place?

[ ] Along the forest edge, closest to home [ ] Deep in the heart of the forest
[ ] In a forest region that is far of and requires traveling [ ] other [________]
How do you organize your harvesting?
[ ] Alone [ ] in organized groups
[ ] In ordered consignments [ ] other: [______________________]

Have you ever had a dispute over who could harvest at any location? [Yes] [No]

If yes how is harvesting regulated traditionally? [______________________]

7. Who does the harvesting in the family?
[ ] Self [ ] women [ ] male children
[ ] female children [ ] hired help [ ] other, explain: [_____________]

8. How frequently is the forest visited for harvesting mangroves?
[ ] At least once a month / week [ ] Once a month / week
[ ] Twice a month / week [ ] Once in two months / weeks
[ ] [Other: ____]

9. Can you estimate on average how much is harvested in one visit to the forest?
[ ] Yes On average [__________________] Trees/head loads
[ ] No,
[ ] depends on need but can range between ___-___ trees/head loads per week / fortnight / month / other: [_____________________________]
[ ] Very difficult, Explain [___________________________________]

How would you quantify a head load?
[_____________ Kg] or [_____________________ Number of branches or sticks]

10. Do you buy mangrove poles? [ ] Yes [ ] No
Can you list the products and their prices?
i. [__________________________] @ [_________] KShs per [________________]

11. Do you harvest mangroves for uses other than those within your household? If so for what purposes?
[ ] Yes, for [ ] Trading [ ] to stock in a commercial place [ ] other uses (specify): [_ ____]
Do you know of any form of: -
[ ] Domestic waste water disposal [ ] Solid waste/garbage dumping
[ ] Recreation [ ] other related uses:
For how long has this activity been taking place? [_________________]

12. What main changes do you think have occurred in the forest? What are the causes?
13. Can you identify sites that have been exposed to sewage/wastewater/pollution?

[ ] Yes [ ] No

If yes, which areas? [____]

14. What do you think is the impact of the sewage/wastewater pollution on:

Fisheries

[ ] Good [ ] Bad [Reason: _______________________]

Village aesthetics/hygiene
[ ] Good [ ] Bad [Reason: [_______________]

Water quality in the village
[ ] Good [ ] Bad [Reason:[________________]

The mangrove trees?
[ ] Good [ ] Bad [Reason:_________________________]

15. Have the local people done or are they doing replanting?

[ ] yes [ ] No
[Please specify:____

Do you participate in these local initiatives or plant mangroves and how often?
[ ] Yes [How often: ______________________________]

[ ] No [ ] Are you interested in planting mangroves [ ] Yes [ ] No [Please explain: ______________________________________]

Can you give an overview of the success if you do plant?

[ ] Yes [Explain: ______________________________________]

[ ] No [Explain: ______________________________________]

[ ] Other: [Explain: ____________________________________]

Mention three main economic benefits of replanting/restoration?

Are there particularly vulnerable groups/people that may be impacted by the replanting/restoration? [ ] Yes [ ] No

If yes please name 3:

How would these groups be impacted? [________________________]

16. Are these resources preferentially or strictly restricted/bound to mangroves?

[ ] Yes [ ] No

Please specify by mentioning specific resources:

Do you own a boat or are you employed on a boat?

[ ] Own boat type of boat [________________________]
[ ] Sharing a boat type of boat [________________________]
[ ] Rent a boat price of rent per person__________type of boat___________.
[ ] Other: [______________________________________]

17. Are you affected in any way by the laws/legislation governing the management and exploitation of mangrove forests? [ ] Yes [ ] No

If yes, briefly explain: [____]

18. Do you understand/know the current management policy for mangrove and the marine reserve?

[ ] Yes [ ] No [Other: _________________]

19. Is the current implementation of these policies adequate? If not, suggest what should be done to effectively enforce these policies and regulations??

[ ] Effective [ ] Ineffective [ ] Other [Please explain [____]

20. Age: [______________] Marital status: [_______________]

21. Profession: [_______________________]

22. Other professions in your family

23. Size of household

	< 5	10	20	21-30	31-40	41-50	51-60	> 60
M								
F								

24. For how long have you lived in the area/village? [______________________________]

Where did you live before? [____]

25. Type of House (by interviewer observation)

[ ] Mud and makuti roofed Swahili hut (with mangrove poles)

[ ] Modern Swahili brick house

[ ] Other, Specify: [____]

Access to electricity [ ] Yes [ ] No

Access to piped water [ ] Yes [ ] No

Access to sewage facilities[ ] Yes [ ] No

Domestic fuel for cooking? [____]

26. Family property

[ ] Land (farming) [ ] Livestock [ ] Poultry [ ] Boat [ ] Other

Personal Observations[____

____]