DOCTOR OF PHILOSOPHY

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Productivity and Nutrient Dynamics of *Avicennia Marina* (Forsk.) Vierh. Mangroves grown on the Red Sea coast of Saudi Arabia

By

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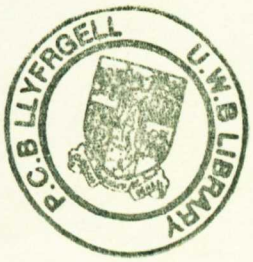

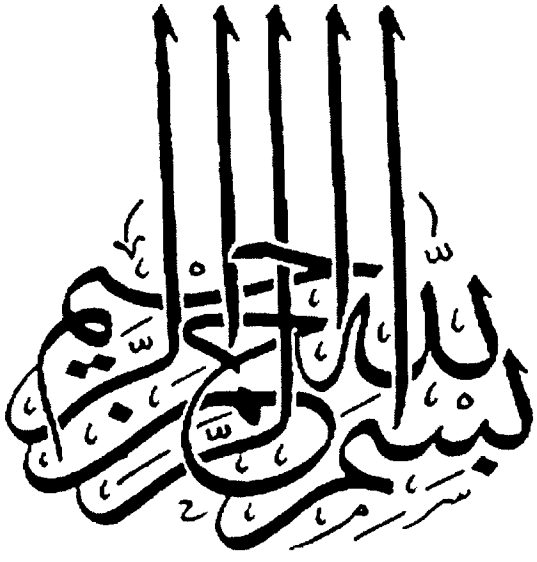

In the name of Allah the Most Gracious
the Most Merciful

ABSTRACT

This study aimed to investigate productivity, nutrient cycling and heavy metal pollution in two *A. marina* stands on the Red Sea coast of Saudi Arabia. The mangrove stands were located at Yanbu region (northern Red Sea) and at Shuaiba region (southern Red Sea). Aboveground biomass production was estimated by developing site specific allometric equations with height and diameter at breast height as biomass predictors. Annual litterfall production was estimated over two years using litter traps and the fate of the fallen litter (*i.e.* accumulation under mangrove stands, removal and export to adjacent waters) was assessed by estimating standing crop leaf litter and monitoring tidal levels and crab activities. Aerial root biomass was estimated by harvesting roots within ground quadrats; fine root biomass was estimated using random coring. Nutrient cycling in the mangrove systems was assessed by litter decomposition and the release of carbon and nitrogen from the decomposing litter. In addition, the importance of mangrove derived carbon as an energy source to the aquatic animals was estimated by ^{13}C stable isotope analysis. The levels and dynamics of eight heavy metal contaminants in mangrove systems were assessed by estimating metal levels in sediment, mangrove components and in the decomposing litter. It was found that aboveground biomass was greater in Shuaiba (18.58 ha^{-1}) than in Yanbu (10.77 t ha^{-1}) (p <0.05); the overall aboveground biomass (14.77 t ha^{-1}) was comparable to estimates reported in other locations at similar extreme environmental conditions. Both aerial and fine root biomass was greater in Shuaiba (23.7 t ha^{-1} and 96.4 t ha^{-1}) than in Yanbu (10.1 t ha^{-1} and 39.1 t ha^{-1} for aerial and fine roots respectively, and overall fine roots estimate (67.8 t ha^{-1}) was comparable to estimates from subtropical and hypersaline regions. Litterfall production was similar in both sites with an overall production of 3.57 t ha^{-1} y^{-1} and litterfall accumulated on the forest floor rather than being exported to adjacent waters owing to low tidal ranges. No significant differences were found in litter decomposition with an overall k value of 0.0076 and half life of 91 days. The levels of carbon and nitrogen at the end of the decomposition period were higher than at the beginning (p <0.05) indicating changes in the leaf chemical composition and microbial activities. However, nitrogen levels in fresh leaves were significantly higher than in senescent leaves indicating nitrogen resorption. The mangrove derived carbon was of moderate importance to a number of crab and fish species; however, this importance was offset by the contribution of other carbon sources to the diet of these animals. Heavy metal pollution in the studied sites was low compared to other contaminated regions, however, heavy metal levels were always higher in the polluted site (Yanbu) than in the minimally exposed site (Shuaiba) indicating the need for monitoring and assessment in other similar sites on the Red Sea coast.

Keywords: *Avicennia marina*, mangrove productivity, Red Sea coast, nutrient cycling, heavy metal pollution

DEDICATION

To

My Parents

My Wife

And my Daughter

ACKNOWLEDGMENT

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Refaat

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ABBREVIATIONS AND ACRONYMS

AGB	Aboveground biomass
ADF	Acid detergent fibres
ADL	Acid detergent lignin
ANOVA	Analysis of variance
BGB	Belowground biomass
B.O.D	Biological oxygen demand
D.O	Dissolved oxygen
C:N	Carbon to nitrogen ratio
Cd	Cadmium
CEC	Electrical conductivity
$CHlO_4$	Perchloric acid
Cr	Chromium
CTAB	Cetyl-trimethylammonium bromide
Cu	Copper
ECE	Electrical Conductivity of Extract
EEAA	Egyptian Environmental Affairs Agency
Eh	Redox potential
FAO	Food and Agriculture Organization of the United Nations
Fe	Iron
GESAMP	Group of Experts on the Scientific Aspects of Marine Pollution
HCl	Hydrochloric acid
ICP-MS	Inductively Coupled Plasma Mass Spectrometer
ISME	International Society for Mangrove Ecosystems
IUCN	The World Conservation Union
LCF	Leaf concentration factor
LCI	Lignified cellulose index
Mn	Manganese
NDF	Nutrient detergent fibres
HNO_3	Nitric acid
MEPA	Meteorology and Environmental Protection Administration
MPA	Marine Protected Area
NCWCD	National Commission for Wildlife Conservation and Development
Ni	Nickel
PCA	Principle component analysis
PERSGA	Regional Organization for the Conservation of the Environment of the Red Sea and Gulf of Aden
ph	Hydrogen-Ion Concentration
Pb	Lead
RCF	Root concentration factor
Ru	Ruthenium

TF	Translocation factor
SENRGY	School of Environment, Natural Resources and Geography
SSM	Standard Survey Methods
UNFCCC	United nations framework on Climate Change
WCMC	World Conservation Monitoring Centre
Zn	Zinc

CHAPTER 1

GENERAL INTRODUCTION

1.1 MANGROVE FORESTS AND THEIR IMPORTANCE

Mangroves are facultative halophytes that dominate the intertidal zones of the world; they flourish in tropical and subtropical zones on estuaries and fringing shorelines. Their unique presence at the boundaries of terrestrial and marine environments renders them an important ecological transition zone. The forest productivity supported by detritus food chains contribute to resident and migratory animals and birds and to the trophic balance in associated ecosystems (Almansi, 1999). Moreover, the litterfall (*i.e.* leaves, flowers, fruits) can be a significant source of energy to marine organisms existing in waters adjacent to mangrove habitat; litterfall can be directly used as feed for aquatic animals such as crabs, shrimps and small fish. Moreover, mangrove forest has been reported to support a nursery ground for many tropical juvenile marine fish and crustaceans (Robertson and Blaber 1992; Bouillon *et al.*, 2002; Sheridan and Hays 2003) and a number of commercial fisheries, such as shrimp farming (Turner, 1977; Saifullah, 1982). In addition, mangrove woods are widely used for many purposes such as shelter and boat building, charcoal, fuel, and tannin extract. Another indirect advantage of mangroves is controlling coastal erosion and contributing to shoreline accretion (Chapman, 1976, 1977).

In Saudi Arabia, mangrove trees (*Avicennia marina* Forsk. Vierh.) are the dominant species on the Red Sea; it has a remarkable visual and biological contrast to the comparatively barren terrain of the surrounding desert. *Avicennia marina* growth is widespread; however, it generally forms a discontinuous and narrow belt along the shoreline of the Red Sea. These forests are interesting because they represent the only forest habitat in the coastal area of the country (Spalding *et al.*, 1997). Mangroves of the region are well known to be the most tolerant plant to severe environmental conditions such as hyper water salinity, minimal fresh water input, extreme high temperature and hot and cold water exchange (Spalding *et al.*,

1997). These forests are a valuable and ecologically significant habitat with many uses to man. For example, mangrove leaves are used as fodder in aquaculture in the southern region of the Red Sea and also as sole feed for desert camels (Almansi, 1999). The wood of *Avicennia marina* is also widely used as fuel (Chapman, 1976).

Information on the Red Sea mangrove ecology is limited although extensive information on the Red Sea biology is available. Most research conducted on mangrove ecosystems has been on the coasts of Sinai where *Avicennia marina* mangroves are stunted and smaller in number; they cling to a thin layer of soil barely covering coral rocks (Sheppard *et al.*, 1992). Sheppard *et al.* (1992) gave an estimated figure of *Avicenna marina* gross productivity in Sinai Peninsula; they estimated mangrove productivity to be 1690 kg O_2 d^{-1} (based on changes in dissolved O_2 concentrations in light and dark bottles) with a relative productivity of 86% compared to other autotrophic communities (benthic macroalgae, microalgae and phytoplankton). However, no experimental measurements on trees and associated biota were done. For the rest of the Red Sea coast of Saudi Arabia, there have been no productivity estimates of mangrove plantations except for a few short term litterfall studies (*e.g.* Saifullah *et al.*, 1989; Khafaji *et al.*, 1991; Mandura 1998) (IUCN/MEPA, 1986; Edwards and Head, 1987; Sheppard *et al.*, 1992). Edwards and Head (1987) hypothesized that mangrove stands in the Red Sea form a major source of high primary productivity in an otherwise barren zone. Moreover, they hypothesized that mangrove stands constitute a nutrient conserving and accumulating ecosystem (evident in the absence of nutrient inputs from rivers and oligotrophic waters of the Red Sea) (Edwards and Head, 1987). Thus, research addressing mangrove productivity and nutrient cycling are important in order to understand the mangroves ecological and environmental significance.

1.1.2 The current status of the mangrove forests

Along semi-arid and arid tropical coasts, the role of mangrove forests is not well understood; a limited number of studies have dealt mainly with physiological tolerances of mangroves to high salinity and intense solar radiation (Cintron *et al.*, 1978; Gordon, 1993; Cheeseman *et al.*, 1997). This is unfortunate considering that vast coastal areas of the dry tropics are inhabited by mangrove forests that are often constrained by low and high seasonal rainfall, high evaporation rates, high soil

salinity, high temperature, low humidity and cloud cover, and intense sunlight (Chapman, 1976). Globally, mangrove forests are undergoing continuous destruction and deterioration for many reasons such as coastal recreation, pollution, population increase, and multiple wood uses.

In the Red Sea, the mangroves are deteriorating mainly due to urbanization adjunct to population increase, oil and sewage pollution, camel overgrazing, and unsustainable cutting for fuel wood, charcoal and animal fodder (Edwards and Head 1987; Abohassan and Osman,1998; PERSGA, 2004; El-Juhany, 2009) In response to this threat, a system plan for Marine Protected Areas (MPA) was formulated by The National Committee of Wildlife Conservation and Development (NCWCD) in 1992 in an attempt to establish protected areas in the coastal zones, in combination with rehabilitation and replantation programs. However, sufficient information of physical environment, floral and faunal communities and the level of disturbance in the mangroves forest were lacking, causing major gaps in the MPA guidelines. Therefore, only a few locations in the southern part of the Red Sea, where stands are bigger and denser are under protection mainly because they support local fisheries and bird nesting. However, most of the other stands along the Red Sea (which might be as significant to local environments as those in the south) are still subject to human (cutting for fuel, charcoal and animal fodder) and animal (overgrazing) destruction. In 2002, The Regional Organization for the Conservation of the Environment of the Red Sea and Gulf of Aden (PERSGA) planned and executed a mangrove survey program to provide an indication of the mangrove's status and suggested guidelines for rehabilitation, conservation and management. Among many recommendations was the need for scientific research in order to implement an integrated management and conservation approach (PERSGA, 2004).

Heavy metal pollution in mangrove systems is another consequence of inappropriate management and conservation practices. It is present mainly in stands with close proximity to urban developments. As a result, the mangroves experience significant direct contamination inputs. Heavy metals are important toxic metals usually discharged to aquatic areas from urban and agricultural runoff, industrial effluents, boating and recreational use of water bodies, chemical spills, sewage treatment plants, leaching from domestic garbage dumps and mining operations (Peters *et al.*, 1997). They can be toxic to plants and animals even in small

concentrations. Mangrove trees are referred to as bio-indicators because of their bioaccumulation ability of heavy metals in their tissues and their role in sediment reactions that influence the mobility of heavy metals (Defew *et al.*, 2005; Machado *et al.*, 2005; MacFarlane and Burchett, 2000). Thus they provide quantitative information on the quality of their environment (Bryan *et al.*, 1985). In addition, the cycling of organic matter through litterfall production, decomposition, and tidal transport may eventually export a fraction of the accumulated heavy metals and therefore expose it to the detritus food chain (Silva *et al.*, 2006). However, little is known about the role of such litter dynamics in the export of heavy metals (Silva *et al.*, 2006). For the Red Sea mangroves, there are only two published reports on pollution disturbed stands. The first one addresses oil pollution impact on mangrove stands in northern Red Sea (Dicks, 1986). The second addresses sewage pollution and its impact on mangrove stands at Jeddah city in the central Red Sea (Mandura, 1997). However, there have been no reported studies assessing the impact of heavy metals pollution on accumulation, partitioning and transport to adjacent marine systems on the Red Sea.

1.2 JUSTIFICATION OF THE STUDY

Mangrove productivity has a significant direct impact on the health and function of the marine ecosystem. In addition, mangrove forests are considered a sink for heavy metal pollution and therefore, they provide quantitative information on their environment. Such information of the Red Sea mangroves is scarce or insufficient. Thus the current investigation will help in assessing the ecological importance of the mangrove stands and in providing site specific ecological information needed to guide in evaluating the status of mangrove habitat and in developing conservation, management and rehabilitation plans.

1.3 SITE DESCRIPTION

Two mangrove locations on the Red Sea coast were selected for the present study, a northern site (Yanbu) in Madinah province located in the industrial city of Yanbu (24° 02' 65" N and 38° 09' 46"E) and a southern site (Shuaiba) in Makkah province located in Shuaiba region (20°46' 2"N and 39° 30' 21"E) (Figure 1.1). These sites were preferably selected as they represent the "soft-bottomed" mangroves

commonly present on the Red Sea, which is characterized by deep sedimentation and fairly well developed mangrove trees (Price *et al.*, 1987; Sheppard *et al.*, 1992).

Shuaiba is an old port lying at about 100 km south of the city of Jeddah, the region comprises two lagoons extending for some 20 km from north to south with the greatest width being 5 km, and each lagoon is connected to the sea through a small channel. Mangrove trees grow on sandy loam sediments reaching approximately 1.8 m depth. Sheltered in the large lagoons, they form a large basin population in the middle of the lagoon with an area of about 2 km^2 (Figure 1.1). The eastern bank of the lagoon is super tidal salt flats traditionally known as *Sabkha* which is characterized by evaporate-carbonate deposits. The flowering season of mangroves extends from April to December. Fisheries resources flourish in the lagoons especially close to the mangroves some of which are commercial such as the *Penaeus* spp. shrimps (Aleem, 1990).

The industrial city of Yanbu is situated at the mouth of the Farah Valley which forms one of the widest deltas along the Red Sea coast and contains the most extensive area of mangrove stands of *Avicennia marina* north of the Tropic of Cancer. The industrial city encompasses an area of approximately 185 km^2 in which mangrove trees grow covering an area of 0.9 km^2 (Figure 1.1). The trees grow on sandy loam sediments covering dead coral surfaces. Sedimentation is generally shallow reaching approximately 60 cm depth. Deeper sedimentation is generally formed toward the shore and gets shallower sea wards where small islands of dead corals are exposed.

The climate of the 2 sites is typical of the hot arid climate of the Red Sea with only a few millimetres of rain annually. In Shuaiba, temperature ranges from 18°C in February to 40°C in July with annual mean temperature of 29°C. The relative humidity is 59 % and mean annual precipitation is 15 mm. In Yanbu, the temperature ranges from 13°C in February to 41°C in August with an average annual temperature of 28°C. The mean annual precipitation is 10 mm and the relative humidity is 48% (Table 1.1), the soil and water physical and chemical characteristics are shown in Table 2.

Figure 1.1 Map of Saudi Arabia showing study locations at Shuaiba and Yanbu.

Table 1.1 Meteorological information in Shuaiba and Yanbu regions, Saudi Arabia

Meteorological parameter	Shuaiba	Yanbu
Total annual rainfall (mm)	15.41	10.16
Maximum temperature (°C)	40.02	41.05
Minimum temperature (°C)	18.16	13.11
Mean annual temp (°C)	28.78	27.89
Relative humidity (%)	58.61	48.07
Wind speed (km h^{-1})	2.64	18.00

Source: Presidency of Meteorology and Environment, Saudi Arabia

Table 1.2 Physical and chemical characteristics of soil and water in mangrove systems on the Red Sea coast, Saudi Arabia

Soil	Soil type	pH	Temp (°C)	ECE (mmhos/cm)	Bulk density (g cm^{3})	D.O (mg l^{-1})	B.O.D (mg l^{-1})
Shuaiba	Sandy loam	7.4	27.5	71.8	1.5	-	-
Yanbu	Sandy loam	7.3	27.6	52.9	2.4	-	-
Water							
Shuaiba	-	7.01	23.4	51.07	-	8.90	3.4
Yanbu	-	7.27	21.05	52.9	-	5.4	-

D.O= Dissolved oxygen; B.O.D= Biological oxygen demand.

1.7 STUDY HYPOTHESES

The overall research hypothesis is that the mangrove stands of the Red Sea play a significant ecological role as a primary producer and/or as an energy source to the aquatic fauna.

The specific hypotheses are:

1. Mangrove biomass and litterfall production is comparable to production in similar arid environments.
2. Mangrove stands constitute a nutrient conserving and accumulating system rather than exporting.
3. Mangrove's detritus significantly contributes to sediment organic matter and in the aquatic food web.
4. Mangrove stands sequester heavy metals and reduce transport to adjacent marine systems.

1.8 STUDY OBJECTIVES AND QUESTIONS

The general objective of the study was to estimate the productivity, nutrient cycling and heavy metal pollution in 2 mangrove stands on the Red Sea coast along with estimating the possible significance of mangrove leaf litter in the aquatic animal food web.

The specific objectives were as follows:

1. Quantitatively estimate the aboveground (via allometric equations) and belowground biomass (via random coring) production of all tree components including stem, branches, leaves, areal and fine roots.
2. Quantitatively estimate the annual litterfall productivity and removal from the system (via crabs) or export (via tidal activity).
3. Estimate nutrient input (Carbon and Nitrogen) via litter decomposition and the concentration of carbon derived from mangrove trees into the aquatic system via measuring the natural $\delta^{13}C$ stable isotope abundance in litterfall, decomposing litter, sediments, and in secondary consumers (crabs).
4. Measure the accumulation and partitioning of heavy metals in mangrove biomasses and transfer rate via the decomposing leaves.

Through these objectives, the current study aimed to answer the following questions:

1. Do mangroves in arid zones constitute a closed system?
2. Do mangroves conserve nutrients in green leaves rather than losing them in litterfall?
3. How does mangrove biomass production compare to estimates in similar arid environments?
4. Does mangrove's detritus contribute to the sediment organic matter and/or food chain?
5. What is the extent and effect of heavy metal pollution in mangrove systems?
6. And finally, are the mangrove systems within the arid zones functioning similarly to those of the Indo-west Pacific and the east African zones?

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CHAPTER 2

LITERATURE REVIEW

2.1 DEFINITION OF MANGROVE SYSTEMS

Mangrove ecosystems commonly refer to the plant community occupying the tropical and subtropical intertidal zone of the world. This is rather simple however, a more specified definition would distinguish between the mangrove trees and mangrove community. The term "mangal" first proposed by MacNae (1968) refers to the mangrove community occupying the intertidal region; this includes mangrove trees and its associated vegetations (mangrove associates). The mangrove trees or what can be referred to as "true mangroves" are trees that are virtually confined to the mangal area while the mangrove associates are those plants that can occur in the mangal or elsewhere (Stafford-Deitsch, 1996). It is noticeable that numerous difficulties in distinguishing between mangroves and mangrove associates have resulted in variable classification of true mangroves and mangrove associates. There are a number of criteria that distinguish true mangroves from their associates: First, mangroves only occupy the intertidal zone and do not extend to terrestrial communities. Second, they can form pure stands on their own. Third, they are adapted to their environment evident in morphological (*i.e.* aerial roots and embryo vivipary) and physiological (*i.e.* salt extrusion) specializations (Tomlinson, 1986). Mangroves are facultative halophytes, they are adapted to live under high salinity and anoxic conditions, however, they can also grow in environments with better conditions. Fresh water flushing provides good conditions for mangrove growth however they are likely to be out-competed by other plants in non intertidal areas. Thus their ability to grow in salty condition pose an advantage of minimizing competition of other plants (Cintron and Shaeffer-Novelli, 1983; Stafford-Deitsch, 1996).

2.2 MANGROVE SPECIES AND DISTRIBUTION

According to the World Atlas of Mangroves (2010) there are 73 mangrove species and hybrids in 28 genera belonging to 20 families (Table 2.1), the majority of which belong to the Avicenniaceae and Rhizophoraceae families (Table 2.2, Hogarth, 2007; Spalding *et al.*, 2010). Mangroves flourish in the tropical zone and their distribution ranges from latitudes of 25°N and 25°S however fewer species (*i.e. A. marina*) extend beyond these limits to latitudes of 28°N in northern Red Sea and 32°S in South Africa (Figure 2.1, Spalding *et al.*, 2010) and therefore, growing under a wide range of climatic conditions. The mangroves are of greater diversity in the Indo-West pacific region however the species number and area cover decrease with latitude suggesting that latitudinal limits are determined by low temperature and lack of frost tolerance (Chapman, 1977). Mangrove growth halts in areas where winter temperatures go below 20°C and the number of species decreases as this limit is approached (Hogarth, 2007).

Although the distribution of mangroves is limited by the climatic conditions, temperature is not the only factor affecting its distribution and growth; there are a series of environmental factors that can determine or modify mangroves areal coverage, these include:

1. Suitable coastal physiography (*e.g.* broad coastal plains with low-lying areas subject to sea water inundation).
2. High tidal range so that sea level reaches low-lying areas.
3. Precipitation exceeding evapotranspiration (best available in estuaries where fresh water results in greater land drainage).
4. River discharges (deltaic systems) determining the existence of riverine mangrove systems.
5. The availability of surface runoffs and ground water in fringe mangrove systems.
6. Shelter and enclosed bays for protection from high energy waves to facilitate seedling establishment and maintenance of mature trees.
7. Sediment availability providing nutrients for tree establishment.

(Cintron and Shaeffer-Novelli, 1983).

Table 2.1 Mangrove species and hybrids of the world

Family	Genus	Number of species
Acanthaceae	Acanthus	2
Arecaceae	Nypa	1
Avicenniaceae	Avicennia	8
Bignoniaceae	Dolichandrone/Tabebuia	2
Bombacaceae	Camptostemon	2
Caesalpinaceae	Cynometra/Mora	2
Combretaceae	Conocarpus/Laguncularia/Lumnitzera	5
Ebenaceae	Diospyros	1
Euphorbiaceae	Excoecaria	2
Lythraceae	Pemphis	1
Meliaceae	Aglaia/Xylocarpus	3
Myrsinaceae	Aegiceras	2
Myrtaceae	Osbornia	1
Pellicieraceae	Pelliciera	1
Plumbaginaceae	Aegialitis	2
Pteridaceae	Acrostichum	4
Rhizophoraceae	Bruguiera/Ceriops/Kandelia/Rhizophora	21
Rubiaceae	Scyphiphora	1
Sonneratiaceae	Sonneratia	9
Sterculiaceae	Heritiera	3

Adapted from Hogarth (2007) and Spalding et al. (2010).

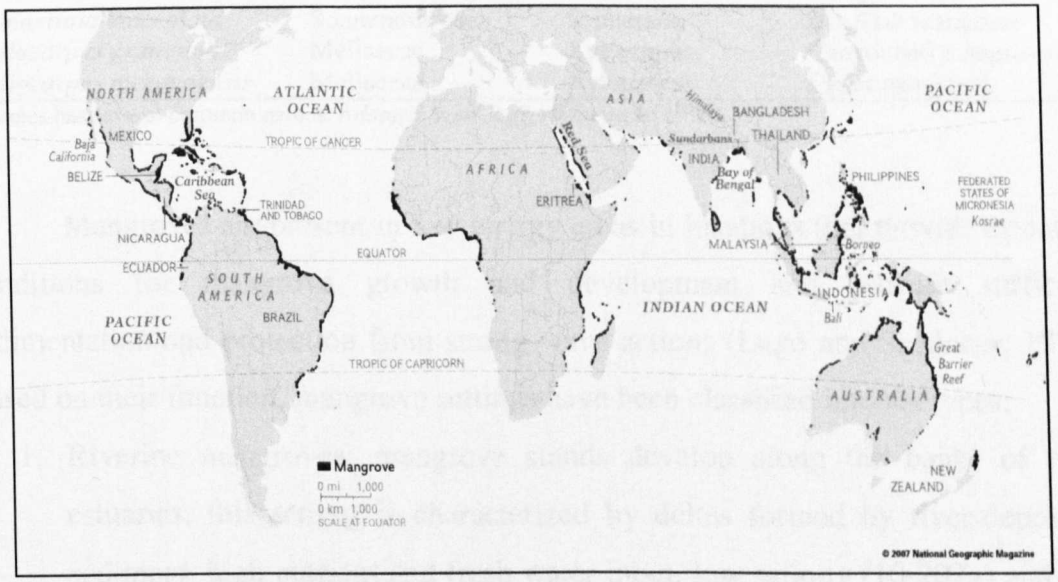

Figure 2.1 Map of global mangrove distribution. *Source: National Geographic Magazine (2007).*

Table 2.2 The taxonomy of common mangrove species found worldwide

Species	Family	Genus	Common name
Aegialitis annulata	Plumbaginaceae	Aegialitis	Club mangrove
Aegiceras corniculatum	Myrsinaceae	Aegiceras	River mangrove
Avicennia alba	Avicenniaceae	Avicennia	*
Avicennia germinans	Avicenniaceae	Avicennia	Black mangrove
Avicennia marina	Avicenniaceae	Avicennia	Gray mangrove
Avicennia nitida	Avicenniaceae	Avicennia	Black mangrove
Avicennia officinalis	Avicenniaceae	Avicennia	*
Bruguiera exaristata	Rhizophoraceae	Bruguiera	Yellow-flowered orange mangrove
Bruguiera gymnorrhiza	Rhizophoraceae	Bruguiera	Large-leafed orange mangrove
Bruguiera parviflora	Rhizophoraceae	Bruguiera	Small-leafed orange mangrove
Camptostemon schultzii	Bombacaceae	Camptostemon	Schultz's mangrove
Ceriops tagal	Rhizophoraceae	Rhizophora	Yellow mangrove
Excoecaria agallocha	Euphorbiaceae	Excoecaria	Blind-your-eye mangrove
Excoecaria ovalis	Euphorbiaceae	Excoecaria	Oval-leafed blind-your-eye mangrove
Laguncularia racemosa	Combretaceae	Laguncularia	White mangrove
Lumnitzera racemosa	Combretaceae	Lumnitzera	Sandy mangrove
Nypa fruticans	Arecaceae	Nypa	Mangrove palm
Rhizophora mangle	Rhizophoraceae	Rhizophora	Red mangrove
Rhizophora mucronata	Rhizophoraceae	Rhizophora	Asiatic Mangrove
Rhizophora stylosa	Rhizophoraceae	Rhizophora	Stilt-rooted mangrove
Sonneratia alba	Sonneratiaceae	Sonneratia	Mangrove apple
Sonneratia lanceolata	Sonneratiaceae	Sonneratia	Brackish mangrove
Xylocarpus granatum	Meliaceae	Xylocarpus	Cannonball mangrove
Xylocarpus mekongensis	Meliaceae	Xylocarpus	Cedar mangrove

*species has several common names. *Adapted from Stafford-Deitsch (1996).*

Mangroves are present in low energy areas in locations that provide optimum conditions for mangrove growth and development and provide sufficient sedimentation and protection from strong wave actions (Lugo and Snedaker, 1974). Based on their function, mangrove settings have been classified into six types:

1. Riverine mangroves: mangrove stands develop along the banks of river estuaries, this setting is characterized by deltas formed by river-deposited sediment, high nutrient and fresh water input, low salinity (10-20‰) and low tidal ranges.
2. Fringe mangroves: stands form a narrow belt along protected shores, high tidal ranges and turbulence, high salinity (reaching 60‰).
3. Basin mangroves: mangroves occur in depressions and lagoons along the coasts and may extend to inland areas where precipitation is collected, slow water flow and salinity range from 30-50‰.

4. Dwarf/Scrub mangroves: this setting type is limited by the shallow sediments and hyper salinity; mangroves are growing under nutrient deficiency conditions. This type of mangrove setting is common in rocky shores and in seasonally dry areas.
5. Overwashed mangroves: this setting is characterized by high tidal level and thus mangroves occur in islands that are frequently inundated with sea water and high organic matter deposition rates.
6. Hammock mangroves: basically a basin mangrove found in the tropical wetlands of southern Florida. It consists of mangrove islands over a mangrove-driven peat filling depressions in the underlying limestone substrate.

(Lugo and Snedaker, 1974; Cintron and Shaeffer-Novelli, 1983, Figure 2.2)

2.3 MANGROVE ECOSYSTEM: A GLOBAL REVIEW AND STATUS

Globally mangroves cover an area of 152,000 km^2 with 40% occurring in Asia. However, this global figure is rapidly decreasing as mangrove forests are lost worldwide. Mangrove area was first investigated in the 1980s with a global area estimate of 187,94 km^2 at the time (FAO, 2007, Table 2.3) Although estimates of the mangrove's global area cover prior to this period is not available, the present satellite and remote sensing imaging technologies allowed prediction of previous mangrove area to be more than 200,000 km^2 of the globe's land (Spalding *et al.*, 2010). The available aerial data showed that the world has lost 19% of its mangroves over the past 25 years. On a regional scale, Asia had the greatest area loss of its mangroves (25%), followed by North and Central America (23%), Africa (14%), South America (11%) and Oceania (9.5%) (Table 2.3). This rapid global decrease in mangrove area reveals the constant degradation and loss of mangrove areal coverage. On the other hand, it is noticeable that the rate of annual loss had dropped gradually from -1.04% in 1980 to -0.66 in 2005. This indicates that the figures of previous annual losses were greater than that initially estimated (-1.04%) due to the fact that mangrove destruction was much more severe and conservation and management programs were very limited. Although the dropping loss rate can indicate global awareness of the significance of the mangrove ecosystems, such loss rate is still seen dangerous considering that mangrove forests grow with restricted extend and that the current

mangrove products (Saenger *et al.*, 1983; Twilley, 1998; Alongi, 2002). Activities related to direct alteration and conversion of mangrove ecosystems is reported to be the most significant factor contributing to the disappearance of mangrove forests (Spalding *et al.*, 2010). Urban mangroves are subjected to a wide range of anthropogenic disturbance including pollution (Duke, 1996; Chindah *et al.*, 2007), clearance of mangrove forest for recreational purposes such as sand beaches (Twilley, 1998), conversion of mangrove areas into lands (*i.e.* industrial, residential, commercial and marina lands) and conversion of mangroves into agricultural and aquacultural lands (Fortes, 1988; Marshall, 1994; Primavera, 1995). The blocking and diversion of fresh waters prevent flushing of the mangroves and results in massive mortality of stands (Hegerl, 1982).

Mangrove products are widely used for many purposes at both local and national scales (FAO, 2007). Major areas for mangrove utilization include fuel, construction timber, fishing related uses, food and drugs, agriculture and household items (Table 2.4). In addition, mangrove ecotourism activities can be a potential sustainable source of income for local people (Taylor *et al.*, 2003) However, the overuse of mangrove products can dramatically reduce mangrove area. In arid and semi arid regions, camel overgrazing and wood cutting have significantly reduced the area of mangrove and resulted in large scale mortality of stands (Mohamed, 1984; Hegazy, 1998; Macintosh and Ashton, 2002).

annual loss figure (-0.66%) is much greater than that of the overall global forest annual loss (-0.22%) (Table 2.3, Spalding *et al.*, 2010).

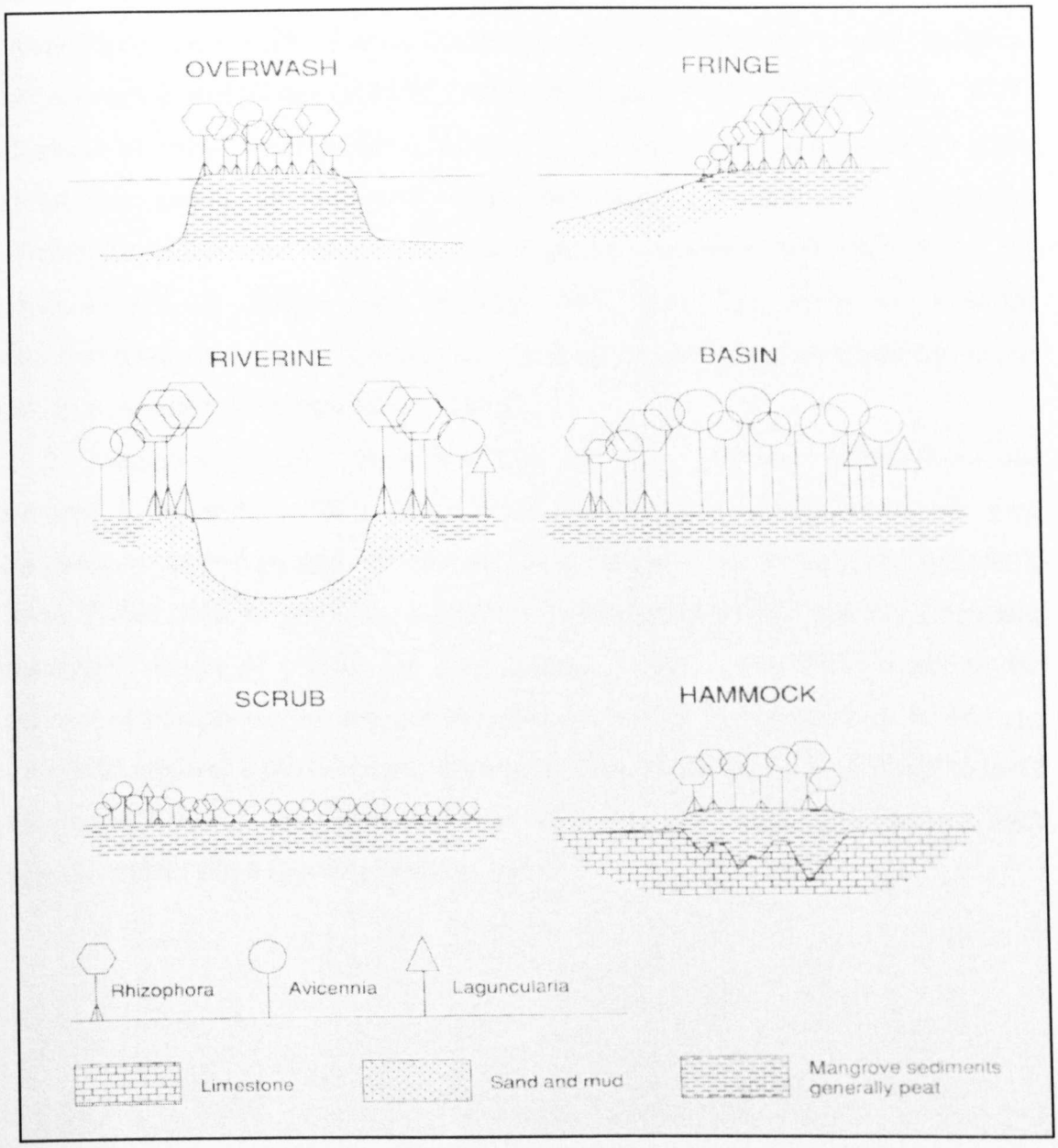

Figure 2.2 Six mangrove setting types found worldwide. *Source: Lugo and Snedaker (1975).*

There are many factors contributing to the disappearance of mangroves worldwide. Global climate changes such as the rise of sea levels may affect mangroves and reduce the intertidal zone and result in the retreat of mangroves (Gilman *et al.*, 2008). However, accretion rates in mangrove forests may be large enough to compensate for the present-day rise in sea level (Ellison 1993; Field 1995). The effect of anthropogenic factors appears to be more damaging and severe than natural factors. These include human alteration/conversion of area and overusing

Table 2.3 The areal extent of the world mangroves (1000 ha) by region (1980-2005)

Region	1980	1990	Annual change (%)	2000	Annual change (%) (1990-2000)	2005	Annual change (%) (2000-2005)
Asia	7769	6741	-1.41	6163	-0.89	5858	-1.01
Africa	3670	3428	-0.68	3218	-0.63	3160	-0.36
North and Central America	2951	2592	-1.29	2352	-0.97	2263	-0.77
South America	2222	2073	-0.69	1996	-0.38	1978	-0.18
Oceania	2181	2090	-0.42	2012	-0.38	1972	-0.39
The World	18794	16925	-1.04	15740	-0.72	15231	-0.66

Source:FAO (2007)

Table 2.4 The different uses and utilizations of mangrove products

Mangrove Uses	Example
Fuel	Fuel wood, Charcoal.
Construction	Timber, Railway sleepers, Mining props, Boat-building, Flooring, Beams and poles.
Fishing	Fishing stacks, Fishing boats, Wood for smoking fish.
Food, drugs and beverages	Sugar, Alcohol, Cooking oil, Vinegar, Tea substitute, Condiments, Sweetmeats (propagules), Vegetables (fruit/leaves), Honey, Wax.
Agriculture	Animal fodder (*i.e.* camel and chicken fodder).
Household items	Hairdressing oil, Tool handles, Rice mortar, Matching sticks.
Paper products	Various types of paper.

Adapted from FAO (2007)

2.4 MANGROVES IN THE INDO-WEST PACIFIC REGION

The Indo-West Pacific region spans the entire Indian ocean from the West including East Africa, the Red Sea, the Arabian Gulf, the Arabian Sea and the Bay of Bengal and Western Pacific Ocean from the East comprising broad mangrove ecosystems on its shores (Figure 2.3). In fact, it contains the largest continuous mangrove area in the world "the Sundarbans" at the bay of Bengal (Agrawala *et al.*, 2003). Mangroves are widely distributed covering wide ranges of environmental conditions. In arid and hyper saline conditions (such as those on the shores of the Red Sea, Pakistan and north western India) mangrove diversity is poor and trees are stunted and discontinuously distributed. Human pressure forms a major danger for the mangrove's existence in the region, the mangrove forests are heavily utilized and mortality and complete loss of some stands were reported (Taylor *et al.*, 2003; Spalding *et al.*, 2010). Except for the Red Sea mangroves, the mangrove ecosystems of the region have been well studied. For example, around 265 papers were published between 1950 and 2000 on the mangroves on the eastern coast of Africa (Taylor *et al.*, 2003), while the mangroves of the Sundarbans were first put under management two centuries ago (Agrawala *et al.*, 2003).

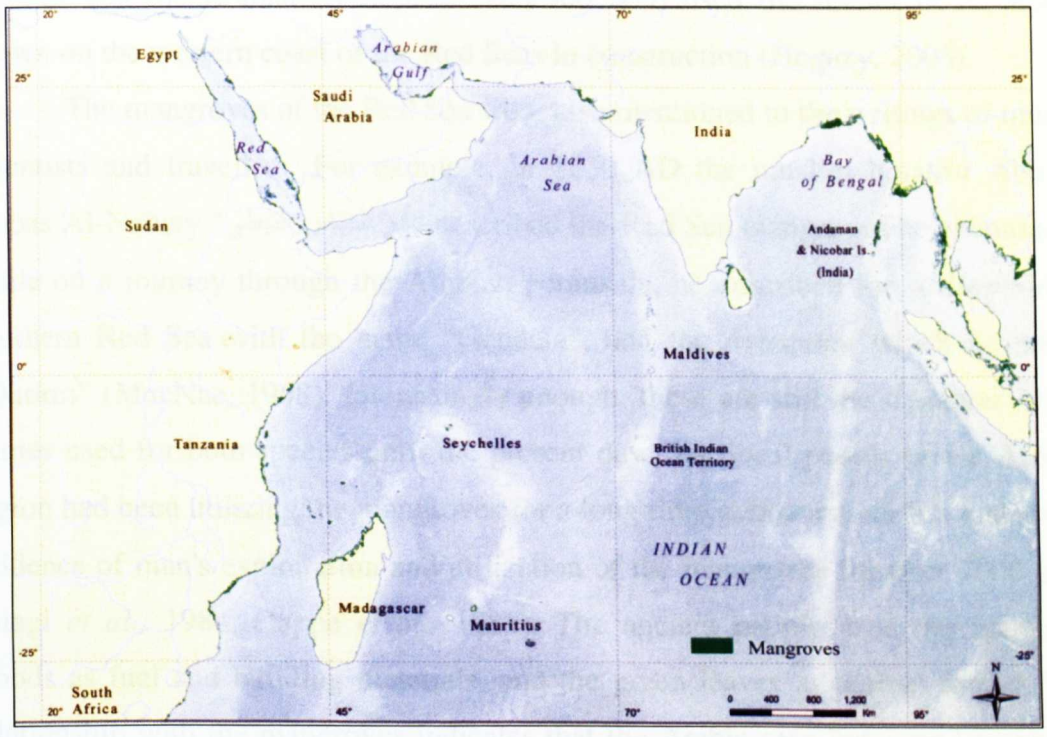

Figure 2.3 Mangrove distribution in the west Indo-Pacific region. *Source: Spalding et al., (2010)*

2.5 MANGROVE FORESTS IN THE ARABIAN PENINSULA

The mangroves of the Arabian Peninsula (Arabian region) have long played a significant role in people's life. The mangroves have been locally utilized as fuel wood, timber, fodder for camels, goats and sheep and for fishing (PERSGA, 2004; FAO, 2005b), they also have an ecological role as a nesting ground for many bird species and source of food or refuge for many aquatic animals (AL-Maslamani, 2006; Kumar *et al.*, 2010).

Not only do the mangroves of the Arabian region have environmental and ecological importance, they also have historical significance. The mangroves of the Arabian region were the first mangroves ever reported in the world's literature by Nearchus and Theopharastus over 2000 years ago (Baker and Dicks, 1982). Theopharastus (350 BC) described the ancient *A. marina* mangroves of the Red Sea in his book titled "Historia Plantarum". It was also described by Pliny (around 77 AD) in his book "Historia Naturalis". The old mangroves of the Arabian region were much more luxurious and widespread than nowadays; Kogo (1984) noted that the extensive mangrove forests of the Arabian Gulf used to be easily observed from a

distance by the Roman sailors in the third century. In addition, excavation evidence showed that the Roman-Byzantines (400-900 AD) used the timber of *A. marina* grown on the western coast of the Red Seas in construction (Hegazy, 2003).

The mangroves of the Red Sea were also mentioned in the writings of muslim scientists and travellers. For example, in 1230 AD the muslim botanist Abu-Al-Abbas Al-Nabaty "أبو العباس النبطي" described the Red Sea mangroves he encountered while on a journey through the Arabian peninsula, he described the *Rhizophora* in southern Red Sea with the name "Gendela", and the *Avicennia* which he named "Quorm" (MacNae, 1968). Interestingly enough, these are still the common Arabic names used for both species until the present day. The local people of the Arabian region had been utilizing the mangroves for a long time in history; studies had shown evidence of man's exploitation and utilization of the mangroves for over 7000 years (Biagi *et al.*, 1984; Coppa *et al.*, 1985). The ancient people used the mangrove woods as fuel and building materials, and the green leaves as animal fodder. This relationship with the mangroves indicates that the Arabic man has long known and depended on the mangroves at very early ages. Such mangrove forests in many parts of the Arabian region have all but disappeared leaving only scattered and fragmented populations on the coasts (Sheppard *et al.*, 1992).

In the Arabian region, mangroves are present on the shores of the Arabian Gulf, the Arabian Sea and the Red Sea with the latter being the most extensive and diverse. The coasts of the Arabian region are characterized by its harsh environment of extreme aridity, hypersalinity, wide temperature extremes and minimal freshwater discharge. These characteristics favor only the most tolerant mangrove species (Edwards and Head, 1987; IUCN/MEPA, 1986; Sheppar *et al.*, 1992). On the Arabian Gulf, *A. marina* is present as a sole species due to its exceptional tolerance to severe environmental conditions (Hutchings and Saenger, 1987). In fact, introduction of new species to the region has made little or no progress (Al-Khayat, 1996; Abohassan and Osman, 1998). Although tolerant to extreme environment, mangrove growth, development and distribution is limited and stands are discontinuous and of patchy occurrence. Along the western coast of the Arabian Gulf, natural mangroves stands are limited to areas covering 30 km^2 in the UAE (Embabi, 1993), 9 km^2 in Qatar (Al-Khayat, 1996), 20 km^2 in Oman (Sheppard *et al.*, 1992) and 35 km^2 in Saudi Arabia (UNFCCC, 2005). These mangroves are growing

in the world largest oil production region which puts them under pressure from pollution and exploitation. The 1991 Gulf war and the associated oil spill have resulted in a devastating environmental catastrophe including massive mortality of the Gulf's mangroves (Böer, 1993; Youssef *et al.*, 2000; FAO, 2005a; Khan and Kumar, 2009). In addition, the development of urban and industrial infrastructure on the coast of the Arabian region has resulted in the deterioration and loss of many mangrove stands and thus the reduction of their area (Khan and Kumar, 2009). In the northern Arabian Gulf, mangroves are absent from the Kuwaiti and Iraqi coasts although recent replanting programs have reported successful establishment of *A. marina* seedlings on the Kuwaiti coast (AboEl-Nil, 2001) and showed positive impacts on the biodiversity, productivity, water quality and soil organic carbon contents (Al-Nafisi *et al.*, 2009).

2.6 MANGROVE FORESTS IN SAUDI ARABIA

In Saudi Arabia, mangrove stands are present on the Arabian Gulf and on the Red Sea coasts. The mangrove area cover was reduced from 210 km^2 in 1980 to 200 km^2 in 1990 and remains unchanged (Figure 2.4). Although the mangrove area was reduced by 0.5% over a decade, reports show no apparent decrease in the following years mainly due to the limited quantitative and reliable information (FAO, 2005b). The World Atlas of Mangroves (2010) viewed an area of 204 km^2 as the most accurate and reliable estimate of mangroves in Saudi Arabia.

The Red Sea holds the most extensive and prevalent mangrove stands in the country, its shores extend for 1932 km between latitude 30°N at the northern end of the Suez Gulf and 13°N at the southern end at Bab Al-Mandab strait with an average width of 280 km (Morcos, 1970, Figure 2.5 and Appendix I). The annual rainfall does not exceed 180 mm and fresh water discharge is very minimal. In addition, the Red Sea is connected to the Indian Ocean only through the narrow strait of Bab Al-Mandab strait (29 km) (Edwards and Head, 1987). These physical characteristics have isolated the Red Sea making it one of the harshest water bodies in the world with average salinity levels of 40‰. The salinity level increases northwards (the shallower region of the Red Sea) and decreases in the deeper waters of the south region. High evaporation rates have kept inshore salinity levels higher than those offshore water throughout the year (Edwards and Head, 1987). The tides in the Red

Sea are semi-diurnal and oscillate around a nodal point of 19°N, spring tides ranges between 0.6-0.9 m at the northern and southern ends of the Red Sea.

The Red Sea also represents the northern limits of mangrove growth in the whole world (Edwards and Head, 1987; Saleh, 2007; Spalding *et al.*, 2010). Mangrove growth extends from latitude 28°'N north at Sinai until Jizan in the south (16° 53'N). The growth and development along the Red Sea is variable, mangroves are poorly developed from its north most limits forming scattered clusters of bushes. Southwards, mangroves form a narrow and discontinuous belt perpendicular to the sea shore. This belt becomes wider and continuous toward the southern part of the Red Sea reaching its climax at Jizan region and Farasan Islands (Abohassan and Osman, 1998; FAO, 2007) The mangroves of the Red Sea grow fringing the shoreline, they also grow in the shallow continental shelves of the coast where they are protected from high energy waves and silt and organic matter are brought by runoff from valleys or 'wades' coming from the 'Sarawat' mountains. The *A. marina* mangroves represent the dominant species on the Red Sea, this species mixes with *Rhizophora mucronata* only in the southern region where the environmental conditions are favorable (IUCN/MEPA, 1986, Plate 2.1).

Similar to other mangrove stands in the Arabian region, the Red Sea mangroves suffer from many anthropogenic and environmental stresses that have resulted in their deterioration. The arid environment offering minimal precipitation, hypersalinity, low sediment and nutrient inputs, poor (sandy) sediment texture and fluctuating temperature can significantly affect the growth and development of the mangroves (IUCN/MEPA, 1986; Edwards and Head, 1987). Moreover, the continuous destruction of the mangroves has significantly affected the stands. Major disturbance includes extensive camel grazing, pollution, oil spills, land modification and conversion, dredging and landfilling (PERSGA, 2004; El-Juhany, 2009).

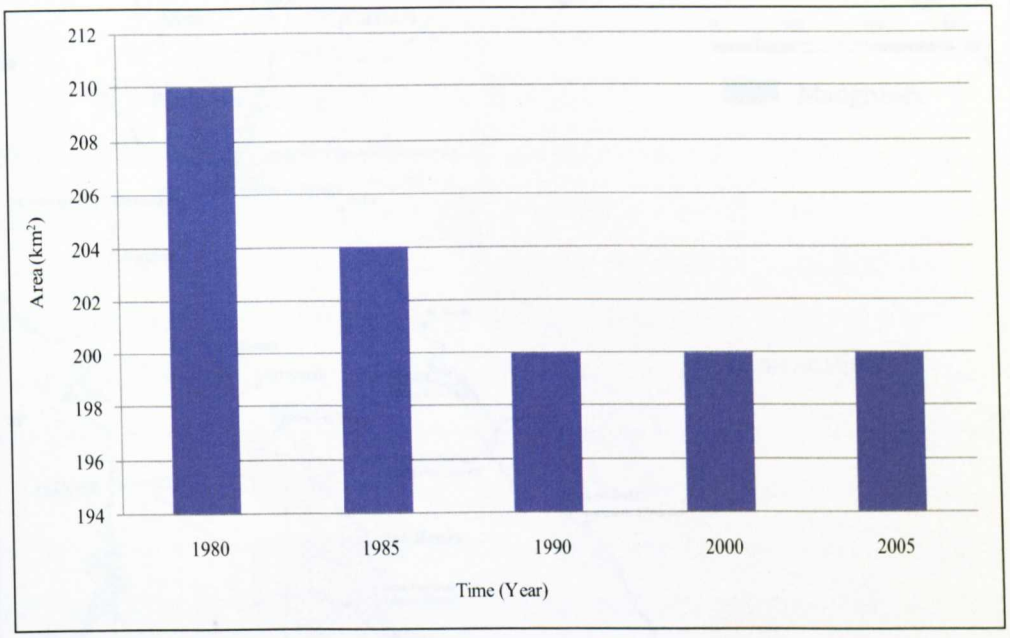

Figure 2.4 Mangrove area change (1980-2005), Saudi Arabia. *Source: FAO, 2005b.*

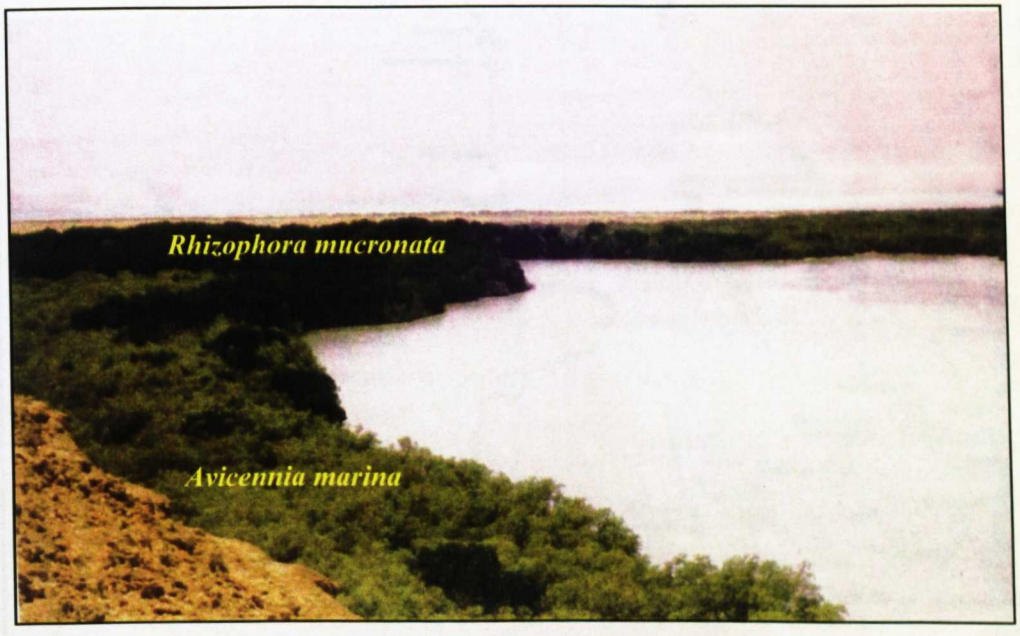

Plate 2.1 *Avicennia marina* and *Rhizophora mucronata* mangroves on the southern shores of the Red Sea, Saudi Arabia. *Source: Chaudhary and Al-Jowaid, (1999).*

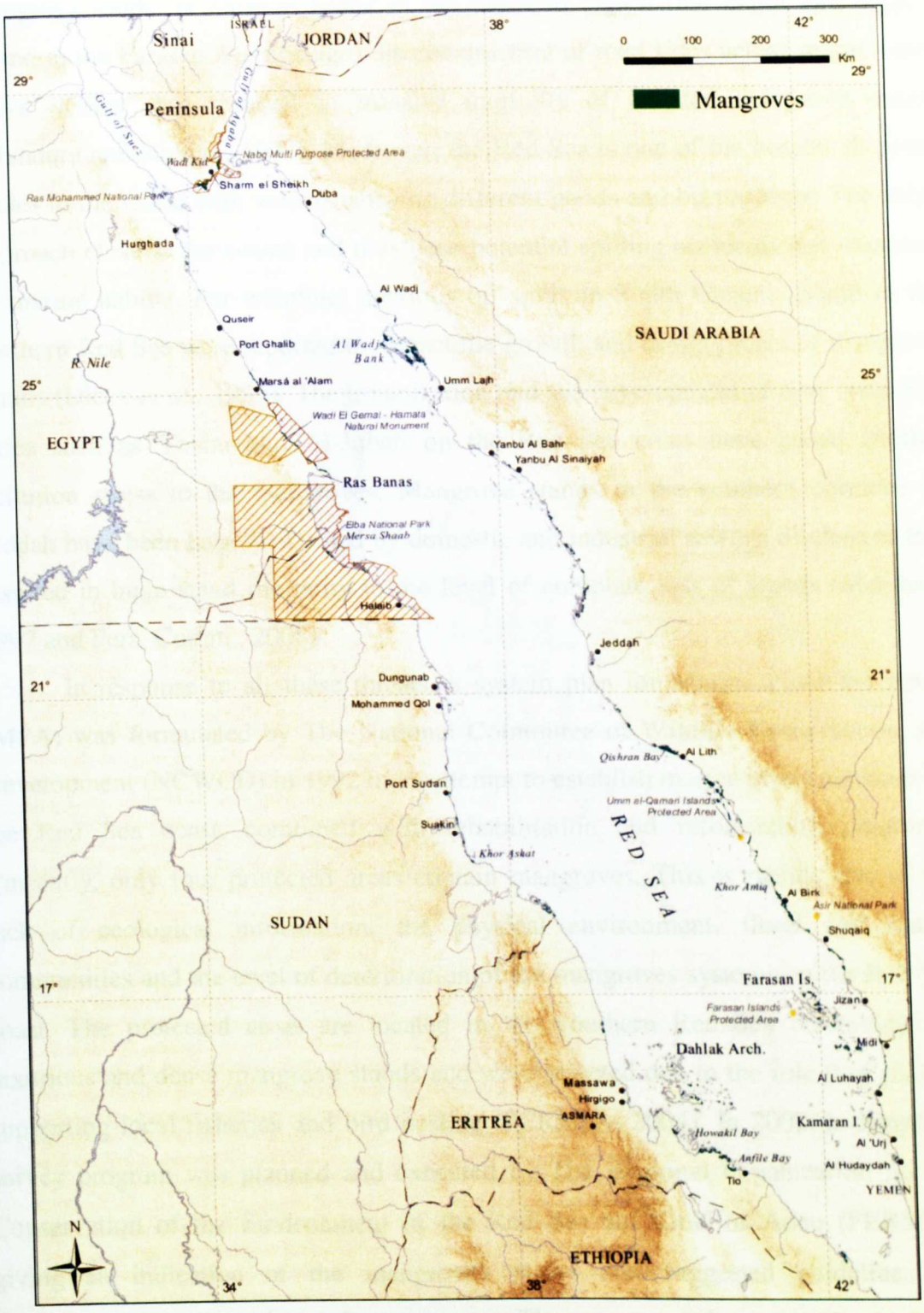

Figure 2.5 Mangrove distribution on the Red Sea coasts. *Source: Spalding et al., (2010).*

Land modification and conversion practices are widespread however; they could potentially be more damaging in the southern region of the Red Sea where the mangroves are much extensive and diverse. Diversion of seawater can diminish

water flushing, nutrient inputs and can cause hypersalinity condition that can kill mangrove sands. In Farasan Island of the southern region (the major and biggest Island in the Farasan Archipelago) the construction of road sides across major water ways "*Khors*" has resulted in massive mortality of isolated mangrove stands (Mandura and Khafaji, 1993). Moreover, the Red Sea is one of the busiest shipping routes in the world with vessels carrying different goods and oil products. The ships approach close to the coasts and thus pose potential spilling accidents and exposure of marine habitat. For example, previous oil spills in South Geisum Island in the northern Red Sea were reported to impact the growth and development of mangrove stands (Dicks *et al.*, 1986). The urbanization and the development of new industrial cities such as Yanbu and Al-Jubail on the Red Sea coast have posed another pollution stress to the mangroves. Mangrove stands in the southern corniche of Jeddah have been largely affected by domestic and industrial sewage discharges that resulted in huge stand mortality to the level of complete loss of stands (Mandura, 1997 and Pers. Comm., 2008)

In response to all these threats, a system plan for Marine Protected Areas (MPA) was formulated by The National Committee of Wildlife Conservation and Development (NCWCD) in 1992 in an attempt to establish marine protected areas on the Red Sea coast, combined with rehabilitation and reforestation programs. Currently, only four protected areas contain mangroves. This is mainly due to the lack of ecological information, the physical environment, floral and faunal communities and the level of deterioration of the mangroves systems on the Red Sea coast. The protected areas are located in the Southern Red Sea containing the luxurious and dense mangrove stands and were selected due to the role they play in supporting local fisheries and bird nesting (PERSGA, 2004). In 2002, a mangrove survey program was planned and executed by The Regional Organization for the Conservation of the Environment of the Red Sea and Gulf of Aden (PERSGA) giving an indication of the mangrove's status and suggested guidelines for rehabilitation, conservation and management. The steps toward conservation include:

1. The development of Standard Survey Methods (SSM) for mangrove habitats.
2. Training regional specialists in these survey methods

3. Surveying the mangrove habitats on the Red Sea to determine the status of mangrove stands.
4. Development of Regional Action Plans (RAP) for conservation of mangroves in the Red Sea.
5. Immediate adoption and implementation of the mangrove RAP.

2.7 MANGROVE BIOMASS PRODUCTION

Compared to other coastal ecosystems, mangroves represent an important source of primary production in coastal regions providing a source of nutrients for the associated biota, supply organic carbon into the sediment, and have a direct impact on the health and function of the marine food web (Saenger *et al.*, 1983; Alongi, 2002). Biomass productivity estimates involve measurements of the amount of living material (*i.e.* leaves, branches, stems and roots) produced by a mangrove community over a specified time. There are three main methods to estimate perennial biomass production (above and belowground) namley tree clearcutting, mean-tree biomass and allometric equations. In the clearcut method, a tree is destructively sampled and thus frequent assessment of biomass increase is not possible. While the mean-tree biomass method requires even aged trees with homogeneous tree size and thus cannot be applied in natural forest (Komiyama *et al.*, 2008). Thus perennial biomass production is commonly measured using allometric equations which estimate the whole or partial weight of a tree from easily measured tree parameters (*i.e* Diameter at Breast Height [DBH], tree height, Basal Diameter [BD]). This method is robust and non-destructive allowing estimation of temporal changes in forest biomass (Brown *et al.*, 1989).

In terrestrial forests, the Aboveground Biomass (AGB) is normally much higher than that of Belowground Biomass (BGB) with AGB:BGB ratios of 4 or higher (Komiyama *et al.*, 2008). In comparison, mangrove trees allocate greater biomass to the roots and AGB:BGB ratio is much less reaching approximately 2 (Tamooh *et al.*, 2008; Komiyama *et al.*, 2008). This great investment in root biomass is not surprising considering the unstable, soft, anoxic, hypersaline and nutrient deficient sediments that mangroves grow on (Hutchings and Saenger, 1987; Ball, 1988; Saintilan, 1997). Greater growth of the root system ensures stabilization and anchoring of the tree (Komiyama *et al.*, 2008). Moreover, the allocation of biomass

into the root system increases also with aridity, light intensity and grazing rates (Aung 1974; Russell 1977; Iwasa and Roughgarden, 1984).

Litterfall production is another common estimate of tree leaf production. Its simplicity and low-cost makes it a common and favourable method for measuring production of a tree (Alongi, 2009). Combining different methods to estimate productivity would aid in obtaining more accurate productivity estimates by accounting for all tree components. In fact, a number of recent biomass investigations have taken such an approach (*i.e.* Coulter *et al.*, 2001; Ross *et al.*, 2001; Sherman *et al.*, 2003). Most investigations of perennial biomass have been of ABG, while BGB were less investigated possibly due to the difficulties associated with root excavation and the difficulties in separating dead and live roots (Clough, 1992). Generally, AGB varies from one region to another depending on species, environmental conditions and latitude (Spalding *et al.*, 2010). Mangroves trees are most developed in the equatorial regions where temperature and rainfall are moderately high with precipitation exceeding evaporation rate. The highest global biomass estimates come from South East Asia with highest species diversity and favourable environmental conditions (Clough, 1992, Table 2.5). In addition, more reliable estimates of biomass can be obtained from managed mangroves of known stand age. For example, Ong *et al.*, (1984) have reported a mean annual AGB increment of 18 t ha^{-1} for ten year old trees in Malaysia.

Table 2.5 Aboveground biomass (AGB) and belowground biomass (BGB) production world wide

Region	Stand Age	Species	AGB ($t\ ha^{-1}$)	BGB ($t\ ha^{-1}$)	Tree height (m)	Basal area ($m^2\ ha^{-1}$)
Australia (33°S)	Primary forest	*Avicennia marina*	144.5	147.3	7	-
Indonesia (Halmahera, 1.1°N)	Primary forest	*Bruguiera gymnorhiza*	407-436	111-181	22-26	36
Japan (Okinawa, 24°N)	Primary forest	B.gymnorhiza	97.6	-	5.5	32.9
Kenya (4.2° S)	Primary forest	*Rhizophora mucronata*	249	-	12	-
Malaysia (Matang, 4.5°N)	>80	*R.apiculata* (dominant)	270-460	-	-	-
South Africa (30°S)	-	*B.gymnorhiza, A. marina*	94.5	-	6	-
French Guiana (4.5°S)	Pioneer stage	*Laguncularia*	31.5	-	3.5	13.7
Kenya (4.2° S)	Primary forest	*Ceriops tagal*	40.1	-	3	-
Puerto Rico (18°N)	Primary forest	*Rhizophora mangle*	62.9	64.4	7.5	-
USA (Florida, 26°N)	-	*R.mangle, A.germinans* (fringe)	56	-	4	13.54

Source: Komiyama et al. (2008)

2.8 MANGROVE ECOSYSTEMS FOOD WEB

The importance of mangrove detritus as energy source for micro and macro fauna is well documented (Day *et al.*, 1981; Odum *et al.*, 1982; Thayer *et al.*, 1987; Hatcher *et al.*, 1989; Robertson and Blaber 1992;. Bouillon *et al.*, 2002). Upon decomposition, mangrove litterfall via detritus pathways make a significant contribution to inshore and estuarine productivity (Fell *et al.*, 1984). The mangrove system provides a rich source of primary productivity supporting a variety of other plants such as algae, and phytoplankton. These primary producers are readily available providing a rich and easy source of nitrogen.

The mangrove system is unique with a wide range of marine and aquatic animals. Complex meofaunal assemblages develop within the mangrove stands feeding on decomposers. Bivalve and gastropod mollusks, crabs and other crustacea (*e.g.* copepods, amphipods, ostracods and shrimps) are abundant as filter feeders, deposit feeders, carnivores and omnivores (IUCN/MEPA, 1986). In addition, the mangrove system is also a food and refuge source for adult and juvenile fishes including commercial species. Adult fishes often migrate from offshore into mangrove systems preying on small invertebrates, while the mangrove roots provide refuge and protection for small fishes from larger predators (Stafford-Deitsch, 1996; Hogarth, 2007). In fact, investigations in Eastern African shores have reported strong dependencies of juvenile commercial fish on mangroves as nursing or refuge habitate (*e.g.* Crona and Rönnbäck, 2007; Mwandya *et al.*, 2009) However, this can also be affected by the condition of the mangrove site (*e.g.* Huxham *et al.*, 2004)

Tree leaf litter is broken down by bacteria and fungi providing access to nutrients by small invertebrates and fish (Dickinson and Pugh, 1974; Osborne, 2000). According to Odum and Head (1975), the main energy flow in the aquatic environment is via mangrove leaf detritus. However, this idea has long been questioned and modified (Odum *et al.*, 1982). The argument was that other marine sources (*e.g.* algae, phytoplankton and seagrass) may be of equal or of greater importance than mangrove detritus (Saenger *et al.*, 1983). For example, mangrove detritus appears to be a significant energy and carbon source for crustaceans and in sediment organic matter in the Indo-Pacific region (*e.g.* Malley 1978; Dahdouh-Guebas *et al.*, 1999; Sheaves and Molony 2000). On the other hand, other studies

suggest that mangroves do not make a major contribution to coastal food webs (Stoner and Zimmerman 1988; Newell *et al.*, 1995).

Tracing energy sources in the aquatic food webs using stable isotope techniques has helped in supporting hypotheses of the significance of some marine sources over others in the aquatic food web. The technique depends on the natural variations of common natural tracers (*e.g.* Carbon, Sulphur and Nitrogen) in different marine sources and thus tracing the isotopic similarity between a source and a consumer (*e.g.* Currin *et al.*, 1995; Fry 2006). Applying this technique in many mangrove systems worldwide has shown that mangrove detritus in some systems represent a significant source of C and N for a variety of micro and macro fauna (*e.g.* Rodelli *et al.*, 1984; Chong *et al.*, 2001) as well as a source of organic carbon in the sediment (*e.g.* Bouillion, 2003). While in other mangrove systems, other sources appeared to be of more significance. For example, Primavera (1996) studied the $\delta^{13}C$ stable isotope contents of juvenile crustaceans and found that plankton and epiphytic algae made a greater contribution to the crustaceans' diet than the mangrove detritus.

The cycling and transport of nutrients in the mangrove ecosystem is driven by physical (*e.g.* tidal ranges, water runoff and rainfall) and biological factors (*e.g.* animal activity, and decomposition rates) that control the rate of import, export and retention of organic matter (Lugo and Snedaker, 1974). In the Indo-Pacific region, the function of the mangrove systems is largely dependent on the abundance of crab species (*e.g.* Robertson and Daniel, 1989; Ashton, 2002). The Sasarmid, Grapsid and Ocypodid crabs are considered major mangrove dwellers and their activity largely affects nutrient cycling in the mangrove system (Alongi, 2002). Mangrove leaves are characterized by thick cuticles and contain high concentrations of cellulose and thus slow microbial decomposition. The processing of litter by crabs can speed the decomposition process by providing access to bacteria and fungi to decompose litter and release nutrients (Hogarth, 2007). Moreover, crabs occasionally retain leaf litter in their burrows playing a role in nutrient accumulation and recycling within the mangrove systems (Alongi, 2002). The effect of crabs is less noticeable in the Eastern Pacific region; the tidal activity is likely to be more influencing factor than those of crab activities. High tidal ranges play a significant role in importing/exporting nutrient from the mangrove systems (Boto and Bunt, 1981; Twilley *et al.*, 1986). High tidal ranges would also affect the rate of allochthonous sedimentation and thus affect the rate of organic matter. Moreover, refractory litter

and organic material can be also washed out of the mangrove system and may become significant energy sources in the adjacent waters (Silva *et al.*, 1998b; Sánchez-Carrillo *et al.*, 2009).

2.9. MANGROVE ECOSYSTEMS AND POLLUTION

The Group of Experts on the Scientific Aspects of Marine Pollution (GESAMP) define marine pollution as "Introduction of man, directly or indirectly, of substances or energy into the marine environment (including estuaries) resulting in such deleterious effects as harm to living resources, hazard to human health, hindrance to marine activities including fishing, impairment of quality for use of sea-water, and reduction of amenities." (GESAMP, 1986). There are many sources of marine pollution including oil spills, industrial and domestic discharges. Generally, pollution from offshore oil exploitation and its related production are widespread and thus pose the major threat for marine life. While terrestrial sewage effluentes are generally discharged from various sources including shipping, dredging, urban sewage discharge, agricultural fertilizer runoffs (Dicks, 1987; Cuong *et al.*, 2005). In Saudi Arabia, sedimentation caused by human activities is a form of pollution that is rarely considered due to the fact that sand and fine grains are natural components of sea beds and shores. However, anthropogenic sedimentation generally happens at unnaturally fast rates posing a threat to the marine environment (Ormond, 1987). This threat becomes greater in shallow and littoral intertidal areas where it may adversely affect the benthic community (Ormond, 1987).

Due to their positions in the intertidal coasts of the world, mangrove ecosystems are very susceptible to marine pollution. The effect of marine pollution on the biota comes through the physical smothering of aerial roots and the presence of toxic substances (Nelson Smith 1984; Clark *et al.*, 1998). The unbroken floating oil that reaches the intertidal region can kill the mangroves by clogging the aerial root lenticels and thus significantly reducing the oxygen supply to the plant. However, excess pollutants are minimized at the root level via allocation (Lacerda *et al.*, 1993) or exclusion (MacFarlane *et al.*, 2007). The concentrations of heavy metals in the plant can reach toxic levels if pollutants exposure is prolonged and in excess (Duke, 1996; Chindah *et al.*, 2007). Organic urban pollutants can carry large bacterial populations involved in breaking down organic matter, which increases the

biological oxygen demand (BOD) in the water and thus increases anoxic conditions (Stafford-Deitsch, 1996)

Mangrove ecosystems can be viewed as an effective barrier to pollutants, especially heavy metals, as the mangrove trees are able to accumulate and tolerate high levels of heavy metal pollution (Thomas and Eong, 1984). This mechanism is largely dependent on the non bio-available form of the metals in the sediment in conjunction with the plants accumulation and exclusion processes (Chiu and Chou, 1991). Moreover, the level of tolerance and metal accumulation is also species specific. Mangroves belonging to the *Avicennia, Rhizophora* and *Kandelia* species tend to have high bioaccumulation compared to other species (Peters *et al.*, 1997). The fine sediment can trap and immobilize heavy metals by complexing metals with organic matter and reducing iron plaque and sulphate via precipitation. In addition, the hypersaline conditions facilitate the formation of metal-chlorine complexes and thus reduce its availability for absorption (MacFarlane *et al.*, 2003; Greger, 2004). Nevertheless, the mangrove ecosystem can also be a source of heavy metals upon disturbance. Anthropogenic disturbance such as clearcutting and dredging can remobilize and export metals to adjacent waters (Riedel and Sanders, 1988). Moreover, the heavy metals might be exported from the mangrove system via litterfall. Leaves containing excess levels of metals are shed and may be exported via tidal activities, or into the food web as detritus upon decomposition (De Laune *et al.*, 1981; Silva *et al.*, 1998a).

In conclusion, research areas addressing mangrove productivity, nutrient cycling and pollution are vitally important for understanding and evaluating the significance of mangrove systems worldwide. And since such research areas are rarely investigated in the Red Sea mangroves, these were addressed in the current study in order to establish baseline information for mangrove systems along the Red Sea coast.

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CHAPTER 3

PERENNIAL BIOMASS PRODUCTION

3.1 PART I-ABOVEGROUND BIOMASS PRODUCTION

3.1.1 Introduction

Accurate estimation of biomass is important for describing the current status of mangrove forests and for predicting the consequences of change (*e.g.* in age-size structure, species composition and disturbance). The use of allometric equations is a reliable non-destructive method for biomass estimation which has been widely used to estimate the biomass of many terrestrial and marine plant species.

The environment of the Red Sea is considered a limiting factor for the development and growth of mangroves; the trees are growing in hyper saline condition reaching 41‰ mainly as a result of low rates of rainfall and high evaporation; it is also characterized by limited nutrient availability evident in the absence of rivers, estuaries and direct influx of water from the Indian Ocean. Mangroves are also exposed to wide ranges of air temperature; shore air temperature is elevated to rates that are sometime higher than desert temperature. In addition the Red Sea mangroves are growing on shallow sedimentation (averaging less that 1 m depth) which limits the growth and development of the trees. These environmental factors have led to the assumption of low productivity of the Red Sea mangroves (litterfall and biomass) compared to the global estimates as productivity. Moreover, estimates of the Red Sea mangrove's productivity are scarce (*i.e.* litterfall) or completely absent (*i.e.* biomass) (IUCN/MEPA, 1986; Edwards and Head, 1987; Sheppard *et al.*, 1992).

The objective of this chapter was to estimate above and belowground biomass of mangrove trees in two mangrove stands in the northern site of Yanbu and the southern site of Shuaiba using allometric relationships between biomass components and tree structural parameters, including diameter at breast height (DBH) and height (Ht). The results obtained from estimating tree characteristics and the models that best describe the biomass data are then discussed.

The hypotheses of this study are:

1. Mangrove biomass can be satisfactorily predicted using height and DBH as combined predictors.
2. Overall mangrove biomass of the Red Sea is low when compared to global estimates.
3. Overall Red Sea mangrove biomass is comparable to mangrove biomass in similar environmental conditions elsewhere.

3.1.2 Study limitation

The study was limited to two locations in the central and northern Red Sea. The sites were selected due to their accessibility for research, logistical considerations, and for previous experience and knowledge of the sites. In addition, they represent the soft bottom mangroves that are largely present on the Red Sea coast. Although researching sites such as those in the southern Red Sea where *A. marina* and *R. mucronata* flourish can be considered more interesting and valuable to the overall Red Sea mangrove estimates, conducting long term research can be difficult, costly and risky. Many of the mangrove stands are in remote areas far from facilities and require costly transportation. In addition, the mangroves are located in regions that are under military control and thus obtaining research permissions is a time consuming and unguaranteed process.

3.1.3 Ethical consideration

It was considered that tree cutting was justified for research purposes, but that minimal destruction could be caused to neighbouring trees and to the environment while doing so. For example, upon felling, a tree was moved into open space to avoid damaging neighbouring trees and to allow component separation and weighing.

3.1.4 Methodology

3.1.4.1 Pilot study and sampling design

A pilot study was conducted prior to developing a sampling scheme in both sites, the sites were initially visited to observe the condition and variations in both sites aided with site maps, GPS and binoculars. In Shuaiba a trend in tree density,

size and tree height was found; trees toward the eastern bank of the lagoon were bigger and denser than those toward the west. Based on these findings, four transects were set in north-south orientation perpendicular to the variation (Plate 3.1a). In each transect, three permanent plots (50×50 m^2) were set at consistent distances along transects with a total of 12 plots. Within these plots, all measurements were taken including those of biomass, litter, soil and tidal monitoring. The location of transects and plots were set using a site map, a Garmin GPSMAP 76S GPS device and a compass. In Yanbu, it was found that trees were more homogenous in growth and density with no visual differences; therefore, 12 plots (50×50 m) were randomly set in the Yanbu site (Plate 3.1b).

3.1.4.2 Stand characteristics and biomass pre-sampling

The height, DBH and density of the stands were estimated in each site from 120 trees randomly sampled for measurement. Tree height was taken from ground level using an elevated ruler with height recorded to the top of the crown. In conjunction, DBH was measured for the same trees using a diameter measuring tape. It should be noted that *Avicennia marina* trees are well known for their multi-stemmed form and irregular growth characteristics (Clough *et al.*, 1997). On average, there are 4 to 6 stems. Thus, special care needs to be taken when measuring DBH of a tree. The following procedure was used for measuring DBH of individual trees in such cases (Snedaker and Snedaker, 1984; English *et al.*, 1997):

1. If a tree forks at or below breast height, each forked stem is measured separately.
2. If a tree forks at or slightly above breast height, DBH at breast height is measured
3. If irregular growth or swelling is present at breast height, DBH is taken just above or below the irregularity.

However, in the current study, trees always forked at levels lower than breast height. Hence, all stems within a tree were measured for their DBH and then summed to attain a DBH value per tree.

Tree population density was estimated by counting each single tree within each of the study quadrats and then expressed on a per hectare basis.

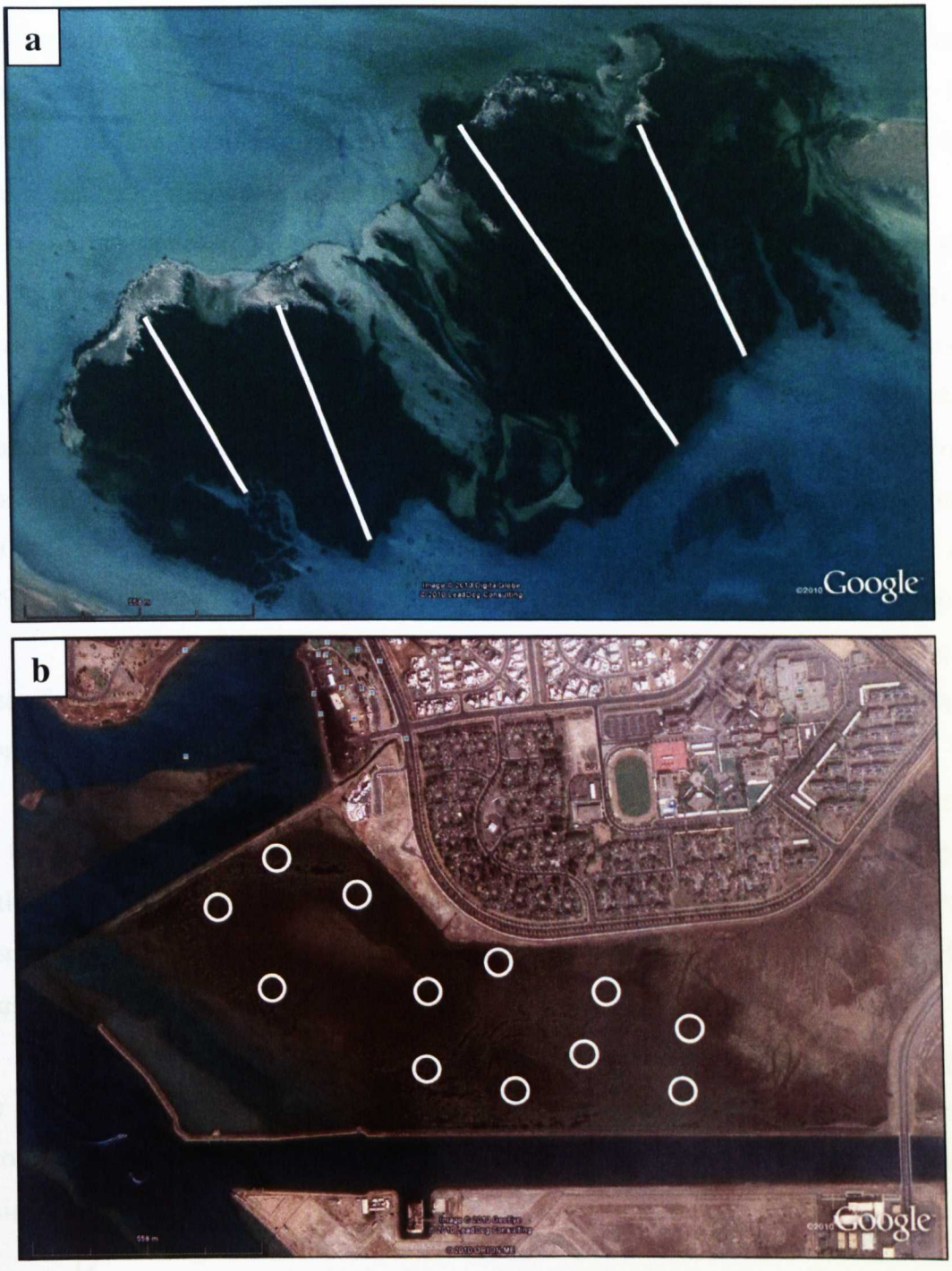

Plate 3.1 Shuaiba (a) and Yanbu (b) sampling schemes.

3.1.4.3 Pre sampling for aboveground biomass estimation

Mangrove aboveground biomass was estimated using allometric equations, this technique allows for the estimation of biomass of a number of harvested trees from easily measurable tree parameters such as DBH, and height (Ht). First the sample size (the number of harvested trees) used to generate the biomass regression was computed; Afterwards, these trees were randomly selected for felling.

A pre-sampling scheme was undertaken to estimate the required sample size necessary for biomass regression estimation. This technique calculates the number of trees to be cut based on the population variance in tree parameters. DBH was used here as an indicator of population variance, DBH measured of all trees with stem diameters greater than 2.5 cm. In conjunction with DBH measurements, tree height was also taken for all measured trees. The required number of trees was determined using Stein's two stage sampling procedure (Hedayat and Sinha, 1991; Steel *et al.*, 1997). This procedure estimates the sample size for the population based on 2 steps:

1. Defining population unit (N). For that, preliminary random sampling of trees from the whole population was undertaken; the number of trees is the single population unit of N.

2. The sample size (n) needed was calculated. When determining n, selection of trees can be either with or without replacement. In cases where a tree is sampled only once (finite population), sampling without replacement is applied. This is normally done in natural forests and open area plots (Kandeel and Abohassan, 2003; Husch *et al.*, 1982).

120 trees from each site were randomly sampled from the population to gain an estimate of the variance. From these estimations, the sample size (n) needed to provide as estimate of aboveground biomass at 0.05 probability level was computed using the equation:

$$n = \frac{1}{\frac{E^2}{t^2 s^2} + \frac{1}{N}}$$

Where: n = sample size, $E^2 = (0.1\text{DBH}\,\bar{x})^2$, s^2 = DBH variance, t = tabulated t value from the t table at 0.05 probability level and N = total tree number in the pre-sampled population (120 tree).

Thus, computed n is the required sample size at the 0.05 probability level. This equation was used according to the following criteria:

1. n was sampled without replacement
2. $\bar{x}$ is the mean of tree DBH
3. Tabulated t value is chosen at infinite degree of freedom =1.96
4. E is the allowable standard error (specified here at ± 10%)

Specific calculation of sample size for Yanbu and Shuaiba sites can be found in Appendix II.

3.1.4.4 Tree sampling for biomass estimation

From the pre-sampling, it was estimated that 16 trees were required for felling from Shuaiba site and 10 trees from Yanbu site. These trees were randomly sampled, measured for DBH (cm) and Ht (m) and then felled. Tree components including stem, branches and leaves were separated and fresh weights determined on site using a heavy duty scale (Plate 3.2 and 3.3). 10 discs from stems and branches ranging in size from 12 to 2.2 cm were taken for moisture content determination which was used to convert tree fresh weight to dry weight. The data from the felled trees from Shuaiba and Yanbu sites can be found in Tables 3.1 and 3.2. In addition, the ranges of height and DBH parameters for the population and sampled trees are shown in Figure 3.1 and 3.2. Leaves from felled trees were used for leaf area estimation using *Image Tool* imaging Software from UTHSCSA (2002). 20 randomly sampled leaves per tree were scanned into a PC and digitally processed to generate the leaf area estimates.

3.1.5 STATISTICAL ANALYSIS

Biomass data were processed and analysed using Excel (2007) and SPSS ver.14 statistical software (2005). Least square regression analysis was used to find the best fit model for the biomass components. Levene's test of equal variance and normal P-P plots were used to test for data normality. Residual homogeneity was confirmed using scatter plots of the residual against predicted values and using residual frequency histograms.

Plate 3.2 Mangrove stem weighing using a heavy duty scale.

Plate 3.3 Mangrove tree felling and component separation process.

Table 3.1 Tree parameters and dry weight biomass (kg) of tree components of 16 harvested trees used to generate regression equations for biomass estimations in Shuaiba, Saudi Arabia

Tree number	DBH (cm)	Ht (m)	Stem (kg)	Branch (kg)	Leaf (kg)	Total (kg)
1	16.37	2.49	3.99	5.64	1.87	11.50
2	9.39	2.59	6.64	7.52	1.87	16.03
3	15.60	3.05	7.53	2.63	3.18	13.34
4	20.22	3.17	10.63	5.64	2.62	18.89
5	6.21	3.81	5.32	1.13	1.49	7.94
6	6.69	3.23	3.99	2.63	1.87	8.49
7	19.58	3.66	16.83	21.99	10.47	49.29
8	3.31	3.05	5.09	3.00	1.68	9.77
9	7.64	3.38	8.86	4.51	1.12	14.49
10	8.60	2.84	5.76	1.88	1.87	9.51
11	12.64	2.79	6.64	3.76	3.36	13.76
12	12.42	3.50	11.96	18.98	7.29	38.23
13	7.00	3.71	5.32	2.82	1.49	9.63
14	10.13	3.71	22.15	15.79	4.11	42.05
15	5.10	2.64	3.99	1.88	1.12	6.99
16	6.21	3.25	4.87	1.13	1.49	7.49

Table 3.2 Tree parameters and dry weight biomass (kg) of tree components of 10 harvested trees used to generate regression equations for biomass estimations in Yanbu, Saudi Arabia

Tree number	DBH (cm)	Ht (m)	Stem	Branch	Leaf	Total
1	4.20	2.74	1.13	7.12	2.21	10.46
2	7.26	2.67	2.06	0.69	0.52	3.27
3	8.28	2.82	7.52	9.28	2.20	19.00
4	5.92	3.17	2.27	3.00	1.19	6.46
5	9.39	2.41	1.54	2.12	1.37	5.03
6	10.16	2.79	2.18	0.91	0.90	3.99
7	5.19	2.84	3.15	7.48	2.59	13.22
8	8.82	2.69	2.90	3.65	2.36	8.91
9	6.40	2.31	1.86	1.48	1.32	4.66
10	9.55	2.74	2.32	2.17	0.97	5.46

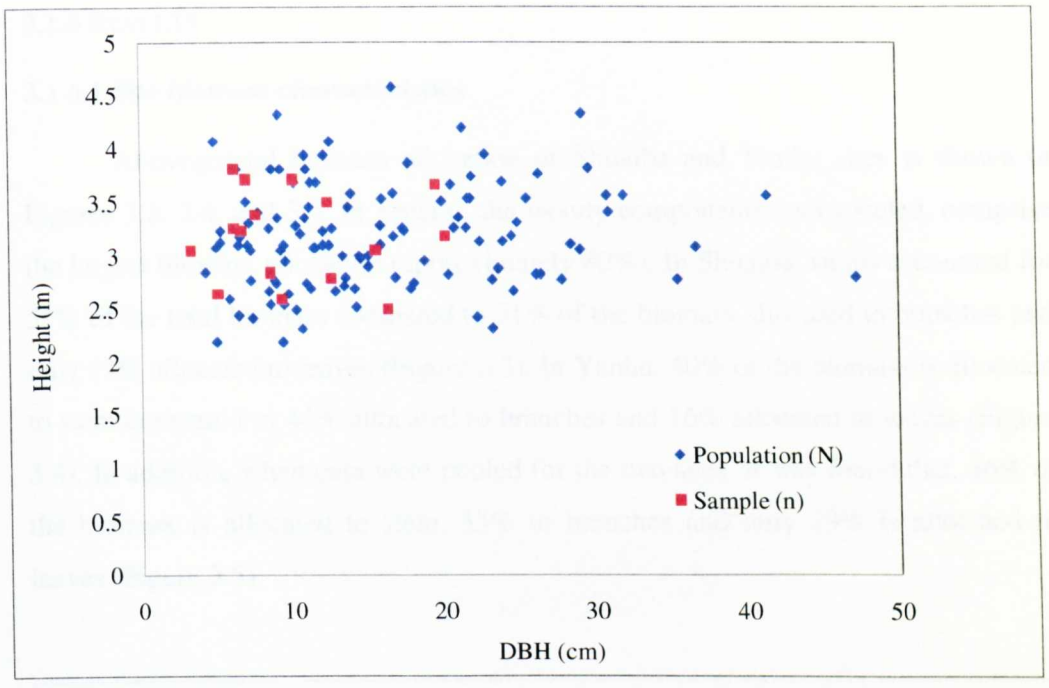

Figure 3.1 The range of height (m) and DBH (cm) for the population (N=120) and sampled trees (n=16) in Shuaiba, Saudi Arabia

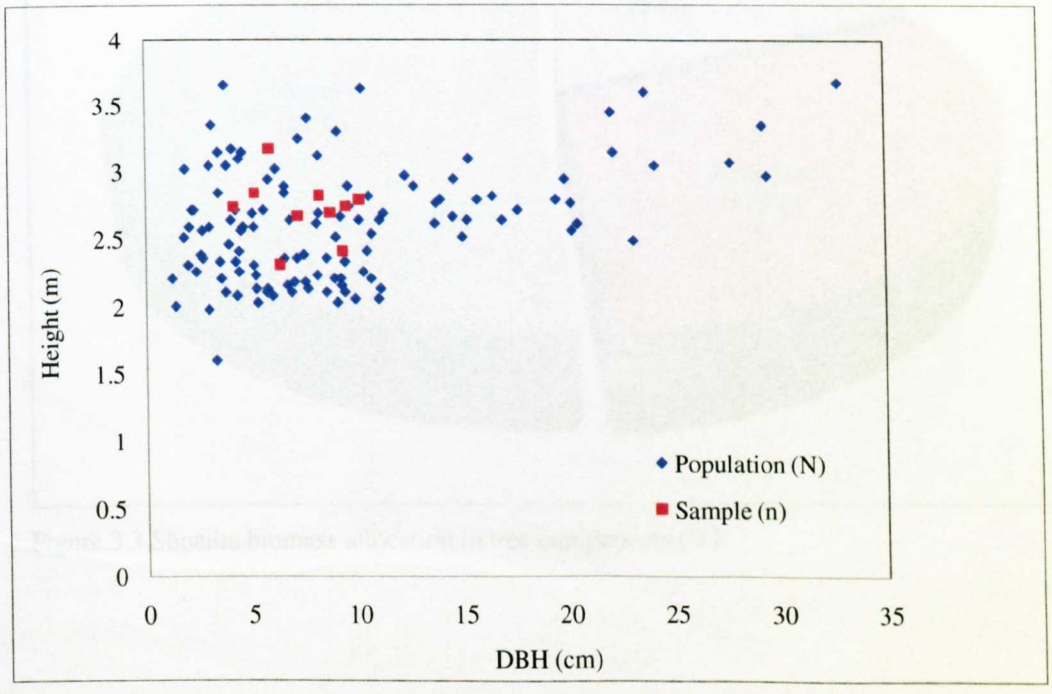

Figure 3.2 The range of height (m) and DBH (cm) for the population (N=120) and sampled trees (n=10) in Yanbu, Saudi Arabia

3.1.6 RESULTS

3.1.6.1 Site biomass characteristics

Aboveground biomass allocation of Shuaiba and Yanbu sites is shown in Figures 3.3, 3.4, and 3.5. In general, the woody components, as expected, comprise the largest biomass allocation (approximately 80%). In Shuaiba, stems accounted for 51% of the total biomass compared to 31% of the biomass allocated to branches and only 17% allocated to leaves (Figure 3.3). In Yanbu, 40% of the biomass is allocated to stem compared to 44% allocated to branches and 16% allocated to leaves (Figure 3.4). In addition, when data were pooled for the two sites, it was found that, 46% of the biomass is allocated to stem, 35% to branches and only 19% is allocated to leaves (Figure 3.5).

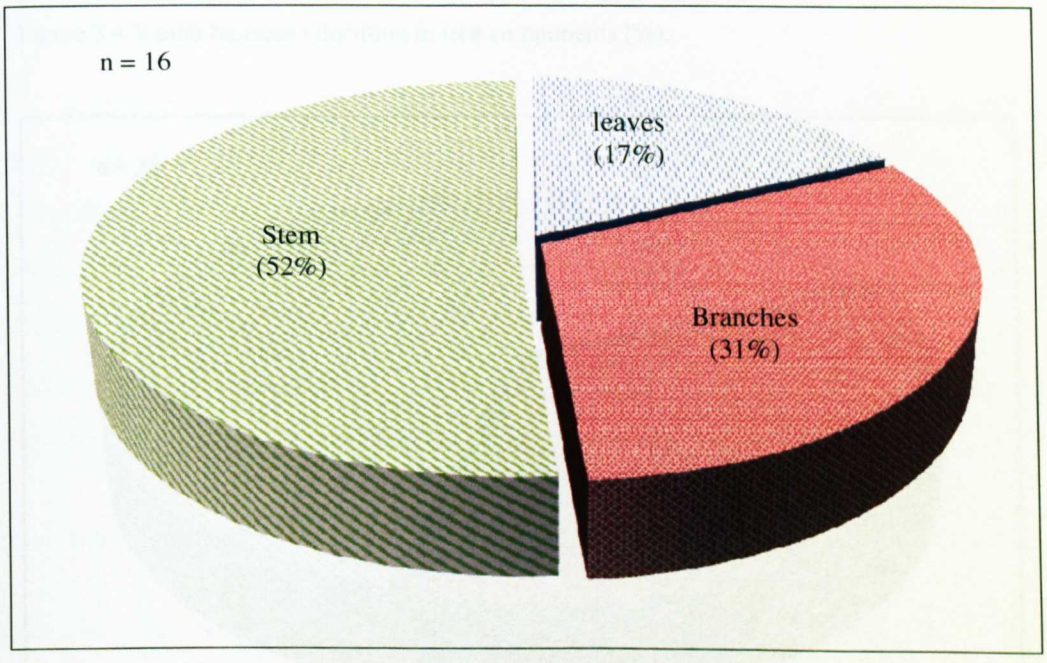

Figure 3.3 Shuaiba biomass allocation in tree components (%).

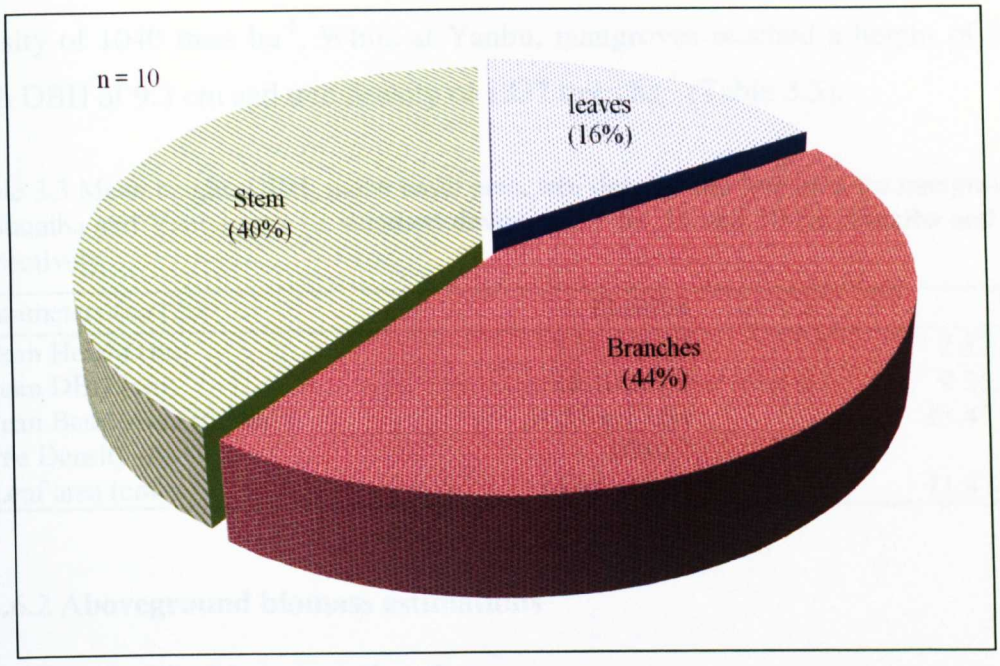

Figure 3.4 Yanbu biomass allocation in tree components (%).

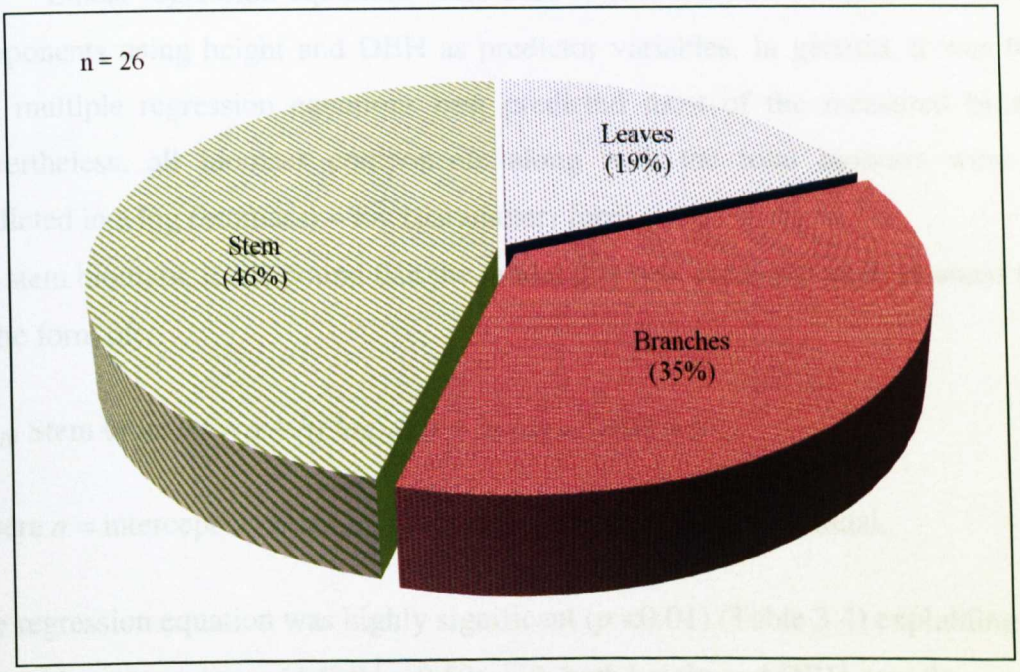

Figure 3.5 Overall biomass allocation in tree components (%).

The tree parameters of Shaiba and Yanbu site are shown in Table 3.3 Mangrove trees in Shuaiba site reached a mean height of 3 m with mean DBH of 16.7 cm and tree density of 1040 trees ha^{-1}. While at Yanbu, mangroves reached a height of 2.57 m with DBH of 9.3 cm and tree density of 1337 trees ha^{-1} (Table 3.3).

Table 3.3 Mean height, DBH, mean basal area, tree density and leaf area for mangrove trees at Shuaiba and Yanbu sites (± standard deviation); * n= 16 and 10 for Shuaiba and Yanbu respectively

Parameters (n=120)	Shuaiba	Yanbu
Mean Height (m)	3.11 ± 0.63	2.57 ± 0.28
Mean DBH (cm)	16.70 ± 3.7	9.26 ± 3.15
Mean Basal Area (m^2 ha^{-1})	31.57 ± 17.54	13.47 ± 8.83
Tree Density (ha^{-1})	1040.7	1337.3
*Leaf area (cm^2)	13.95 ± 2.87	11.47 ± 2.87

3.1.6.2 Aboveground biomass estimations

3.1.6.2.1 Shuaiba biomass estimation

Linear regression equations were used to find the best fit model for biomass components using height and DBH as predictor variables. In general, it was found that multiple regression equations best predicted most of the measured biomass. Nevertheless, all biomass components along with the total biomass were best predicted in a log (biomass) – log (parameter) form.

For stem biomass, it was found that the model that best predicted stem biomass was in the form of:

$$log_{10} \text{ Stem biomass} = a + b_1 \, log_{10} \text{Ht} + b_2 \, log_{10} \text{DBH} + \varepsilon$$

Where a = intercept constant, b = regression coefficients, ε = residual.

The regression equation was highly significant (p <0.01) (Table 3.4) explaining 50% of the biomass variance (Adj R^2 = 0.50) with both height and DBH equally contributing in predicting biomass (p <0.05) (Table 3.5).

Table 3.4 ANOVA and R^2 statistics for Shuaiba stem biomass

Parameter	Sum of Squares	df	Mean Square	F	P probability	R^2	Adj R^2
Regression	2.270	2	1.135	8.542	0.004	0.568	0.501
Residual	1.727	13	0.133				
Total	3.997	15					

Table 3.5. Standardized and unstandardized regression coefficients for Shuaiba stem biomass (± standard deviation)

Parameters	Unstandardized Coefficient (*B*)	Standardized Coefficient (Beta)	*t*	*P* probability
(Constant)	-1.607 ± 3.356	-	-2.696	0.018
log_{10} Height	2.026 ± 0.688	0.538	2.947	0.011
log_{10} DBH	0.552 ± 0.184	0.547	2.998	0.010

Similarly for the branches, biomass was best predicted using the regression model in the form of:

log_{10} Branch biomass = $a + b_1 log_{10} DBH^2 Ht + \varepsilon$

Where a = intercept constant, b = regression coefficients, ε = residual.

The linear regression equation has significantly predicted the branch biomass variation (p <0.05) explaining 32 % of the total variance (Adj R^2 = 0.32; Tables 3.6 and 3.7).

Table 3.6 ANOVA and R^2 statistics for Shuaiba branch biomass

Parameter	Sum of Squares	df	Mean Square	*F*	*P* probability	R^2	Adj R^2
Regression	4.647	1	4.647	8.025	0.013	0.36	0.32
Residual	8.106	14	0.579				
Total	12.753	15					

Table 3.7 Standardized and Unstandardized regression coefficients for Shuaiba branch biomass (± standard deviation)

Parameters	Unstandardized Coefficient (*B*)	Standardized Coefficient (Beta)	*t*	*P* probability
(Constant)	-1.621	-	-1.488	0.159
$DBH^2 Ht$	0.542	0.604	2.833	0.013

For leaf biomass, regression equation and coefficients were in form of:

log_{10} Leaf biomass = $a + b_1 log_{10} Ht + b_2 log_{10} DBH + \varepsilon$

Where a = intercept constant, b = regression coefficients, ε = residual.

The regression equation for leaf biomass was highly significant (p <0.01), and was able to explain 48% of the leaf biomass variations (Adj R^2 = 0.48) (Tables 3.8). A further investigation of the leaf biomass coefficients shows that log DBH variable

made a significantly higher contribution to predicting biomass variance than log Ht at p <0.01 (Table 3.9).

Table 3.8 ANOVA and R^2 statistics for Shuaiba leaf biomass

Parameter	Sum of Squares	df	Mean Square	F	P probability	R^2	Adj R^2
Regression	3.364	2	1.682	8.034	0.005	0.553	0.484
Residual	2.721	13	0.209				
Total	6.085	15					

Table 3.9 Standardized and Unstandardized regression coefficients for Shuaiba leaf biomass (± standard deviation)

Parameters	Unstandardized Coefficient (B)	Standardized Coefficient (Beta)	t	P probability
(Constant)	-2.841	-	-2.056	0.60
log_{10} Ht	1.585	0.341	1.836	0.089
log_{10} DBH	0.838	0.673	3.625	0.003

Finally, all tree components were summed to get an estimate of the total biomass prediction regression (Tables 3.10 and 3.11). The best fit prediction equation was in the form of:

log_{10} total biomass = $a + b_1 log_{10}$ Ht + $b_2 log_{10}$ DBH + ε

Where a = intercept constant, b = regression coefficients, ε = residual.

The regression model for total biomass was highly significant (p <0.01) explaining 47% of the biomass variance (Adj R^2 = 0.47) (Table 3.10). In addition the regression coefficients statistics showed that *log* DBH variable significantly contributed to predicting the total biomass more than height at p <0.01 (Table 3.11).

Table 3.10 ANOVA and R^2 statistics for Shuaiba total biomass

Parameter	Sum of Squares	df	Mean Square	F	P probability	R^2	Adj R^2
Regression	3.166	2	1.583	7.764	0.006	0.544	0.474
Residual	2.651	13	0.204				
Total	5.817	15					

Table 3.11 Standardized and Unstandardized regression coefficients for Shuaiba total biomass (± standard deviation)

Parameters	Unstandardized Coefficient (*B*)	Standardized Coefficient (Beta)	*t*	*P* probability
(Constant)	-1.145	-	-1.019	0.327
log_{10} Ht	1.793	0.394	2.104	0.055
log_{10} DBH	0.777	0.637	3.402	0.005

3.1.6.2.2 Yanbu biomass estimation

In Yanbu, similar to Shuaiba, several linear regression equations were used to find the model that best fitted the biomass data using height and DBH as predictor variables. It was found that none of the tested models significantly predicted any of the biomass components, or the total biomass. All tree components had an r^2 value of less than 0.3, with best r^2 value for stem (0.25) using both height and DBH on log_{10} scale. As such prediction equations for biomass was not obtained, site biomass in Yanbu was calculated using mean biomass values of the sampled trees (10 trees) and using site tree density of 1337.3 tree ha^{-1}.

3.1.6.2.3 Overall biomass estimation

In order to find an overall model that best predicted the overall biomass, biomass components (stem, branches, leaves) of both sites were pooled in one combined data set and several linear and polynomial regression equations were tested to find the best predicting model.

For stem biomass, the best predicting model of the overall was a linear log-log regression equation (Table 3.12) in the form of:

$$log_{10}\text{Stem biomass} = a + b_1\, log_{10}\text{Ht} + b_2\, log_{10}\text{DBH} + \varepsilon$$

Where a = intercept constant, b = regression coefficient, ε = residual.

The log-log regression equation was highly significant (p <0.001) explaining 60% of stem biomass variance (Adj R^2 = 0.59) (Table 3.12). Further examination of parameters coefficients showed that, on a log-log scale, both height and DBH significantly contributed to predicting variance in stem biomass (p <0.01). However, on the standardized coefficient scale, it was found that the height variable predicted more stem variance than DBH (0.61 and 0.44 respectively) (Table 3.13).

Table 3.12 ANOVA and R^2 statistics of the overall stem biomass for Shuaiba and Yanbu sites

Parameter	Sum of Squares	df	Mean Square	*F*	*P* probability	R^2	Adj R^2
Regression	8.504	2	4.252	19.072	0.000	0.624	0.591
Residual	5.128	23	0.223				
Total	13.632	25					

Table 3.13 Standardized and Unstandardized regression coefficients of the overall stem biomass for Shuaiba and Yanbu sites (± standard errors)

Parameters	Unstandardized Coefficient (B)	Standardized Coefficient (Beta)	*t*	*P* probability
(Constant)	-3.550	-	-4.262	0.000
log_{10} Height	3.242	0.615	4.788	0.000
log_{10} DBH	0.725	0.442	3.442	0.002

For branch biomass; the best predicted model for the overall biomass was of a linear regression (Table 3.14) in the form:

Branch biomass= $a + b_1$ Ht + b_2 DBH + ε

Where a = intercept constant, b = regression coefficient, ε = residual.

Table 3.14 ANOVA and R^2 statistics of the overall branch biomass for Shuaiba and Yanbu sites

Parameter	Sum of Squares	df	Mean Square	*F*	*P* probability	R^2	Adj R^2
Regression	265.787	2	132.894	6.005	0.008	0.343	0.286
Residual	509.032	23	22.132				
Total	774.819	25					

The linear regression equation for the overall branch biomass was highly significant (p <0.01) but explained only 29% of the total biomass variance (Adj R^2 = 0.286) (Table 3.14). Further examination of the regression coefficients revealed that both height and DBH significantly contributed to predicting the biomass regression (p >0.05), as height increases by one standard deviation, biomass increases by 0.37 of a standard deviation, while for each one standard deviation increase in DBH, biomass increases by 0.40 of a standard deviation (Table 3.15). Moreover, the unstandardized coefficients showed that for each 1 meter increase in height, biomass increased by 5 kg representing a standard deviation of 2.2, while for each 1 cm increase in DBH there is 0.5 kg increase in biomass over a standard deviation of 0.2.

Table 3.15 Standardized and unstandardized regression coefficients of the overall branch biomass for Shuaiba and Yanbu sites (± standard errors)

Parameters	Unstandardized Coefficient (B)	Standardized Coefficient (Beta)	*t*	*P* probability
(Constant)	-14.092	-	-2.077	0.049
Height	4.900	0.375	2.203	0.038
DBH	0.506	0.404	2.371	0.027

For the overall leaf biomass, the regression model that best predicted the biomass variance was in form of:

Leaf biomass = $a + b_1$ DBH^2 Ht + ε

Where a = intercept constant, b = regression coefficient, ε = residual.

The overall leaf regression equation was highly significant (p >0.001) explaining 45% of the total biomass variance (Adj R^2 = 0.452) (Table 3.16). The regression coefficients statistics shows that the contribution of the DBH^2 Ht variable in explaining the biomass variance was highly significant (p <0.001) (Table 3.17).

Table 3.16 ANOVA and R^2 statistics of the overall leaf biomass for Shuaiba and Yanbu sites

Parameter	Sum of Squares	df	Mean Square	*F*	*P* probability	R^2	Adj R^2
Regression	53.381	1	53.381	21.658	0.0001	0.474	0.452
Residual	59.155	24	2.465				
Total	112.536	25					

Table 3.17 Standardized and Unstandardized regression coefficients of the overall leaf biomass for Shuaiba and Yanbu sites (± standard errors)

Parameters	Unstandardized Coefficient (B)	Standardized Coefficient (Beta)	*t*	*P* probability
(Constant)	1.059	-	2.506	0.019
DBH^2*Ht	0.004	0.689	4.654	0.0001

For the total biomass, it was found that linear regression was best describing the total biomass (Table 3.18) the model was in the form of:

Total biomass = $a + b_1$ Ht + b_2 DBH + ε

Where a = intercept constant, b = regression coefficients, ε = residual.

Table 3.18 ANOVA and R^2 statistics of the overall total biomass for Shuaiba and Yanbu sites

Parameter	Sum of Squares	df	Mean Square	*F*	*P* probability	R^2	Adj R^2
Regression	1791.363	2	895.682	12.451	0.0002	0.52	0.48
Residual	1654.565	23	71.938				
Total	3445.928	25					

The linear regression equation was highly significant (p <0.001) explaining 48% of the biomass variance (Adj R^2 = 0.48) (Table 3.18). Moreover, further examination of the standardized coefficients (Table 3.19) showed that both height and DBH variables significantly contributed in predicting the biomass (p <0.05); both height and DBH had almost equal contribution in predicting the total biomass; As height increases by one standard deviation, biomass increase by 0.49 of a standard deviation while for each one standard deviation increase in DBH biomass increase by 0.47 of a standard deviation (Table 3.19).

Table 3.19 Standardized and Unstandardized regression coefficients of the overall total biomass for Shuaiba and Yanbu sites (± standard errors)

Parameters	Unstandardized Coefficient (*B*)	Standardized Coefficient (Beta)	*t*	*P* probability
(Constant)	-38.299	-	-3.132	0.005
Height	13.483	0.490	3.362	0.003
DBH	1.242	0.470	3.226	0.004

However, examining the unstandardized coefficients showed how different independent variables predict biomass according to their respective units; for height variable, the results showed that for each 1 metre increase in height there is 13 kg increase in biomass representing a standard deviation of 4.0. While for each 1 cm increase in DBH there is 1 kg increase in biomass representing 0.4 of a standard deviation.

After generating the models that best predicted the biomass, site biomass (t ha^{-1}) for each biomass components and for the total biomass was calculated using the current prediction equation, specific biomass data for all sites can be found in Table 3.20. On a t ha^{-1} basis, the biomass values for Shuaiba components were 3.14, 7.27, 8.47 and 18.58 for leaves, branches, stem and total biomass respectively. Where in Yanbu, the biomass values were 2.09, 5.07, 3.6 and 10.67 t ha^{-1} for leaves, branches, stem and total biomass respectively (Table 3.20).

Table 3.20 Dry weight biomass estimation (t ha^{-1}) of mangrove tree components in Shuaiba and Yanbu regions, Saudi Arabia (± standard deviation)

Tree component	Shuaiba	Yanbu	Overall
Leaves	3.14 ± 0.44	2.09 ± 0.96	2.70 ± 1.20
Branches	7.27 ± 0.95	5.07 ± 4.09	5.77± 3.55
Stem	8.47 ± 1.03	3.60 ± 2.40	6.40 ± 3.82
Total	18.58 ± 2.65	10.76 ± 6.64	14.77 ± 9.17

As mentioned earlier, a model that can best predict Yanbu biomass was not achieved. Therefore, mean values of the sampled trees biomass along with site tree density was used to generate Yanbu site biomass values. It is worth noting that an estimate for Shuaiba site using this method was done earlier to have a figure of the difference in mean values. It was found that values obtained from using this method did not differ significantly from the values obtained from the prediction model. The maximum differences between the 2 methods were for the branch biomass (6.56 and 7.22 for mean value of sampled trees and model values respectively). In addition, when biomass of both sites where pooled together, the overall biomass obtained from the regression equation yielded values of 2.70, 5.77, 6.40 and 14.77 t ha^{-1} for leaves, branches, stem and total biomass respectively.

3.1.7 DISCUSSION

3.1.7.1 Site biomass characteristics

As mentioned previously, the multi-stemmed nature of *A. marina*, as described in previous studies, resulted in difficulties in distinguishing between stems and branches and therefore resulted in inaccurate estimations of biomass components (Clough *et al.*, 1997). These characteristics may be increased in the harsh arid and hyper saline conditions of the Red Sea region (Clough *et al.*, 1997). In the current study almost all trees had large multi-stemmed characteristics.

Shuaiba trees had most of the biomass allocated to stem compared to the other components, where in Yanbu, the largest biomass proportions were allocated to branches rather than stem and leaves (42% vs. 37% and 21% for branches and leaves respectively). In Shuaiba, the mangrove trees were larger and vertical in shape making it easier to define stem from branches. In Yanbu, the trees were smaller and

more multi-stemmed. As a result, the trees did not show a clear definition between stems and branches most of the time; in fact, this problem had led previous researchers to conclude that differentiating between stems and branches of a tree in such conditions is difficult and is largely influenced by personal judgment (Clough *et al.*, 1997). This difficulty may have led to measuring error in the current research as a result of including stem-like branches as branch components.

Shuaiba trees were taller and bigger in diameter than Yanbu but the population density was lower. The considerable differences in the tree parameters between sites reflect how biomass of the same species may vary from site to site depending on the environmental conditions and possibly plantation age. Shuaiba mangroves, which are characteristic of basin soft bottom mangroves, grow in well developed sediments reaching a depth of approximately 1.8 m; this allows space for stabilization, vertical growth, and biomass increase. Moreover the basin nature of the Shuaiba mangroves allow for the trees to be spaced and thus minimizing competition and allowing for bigger diameter growth. On the other hand, Yanbu mangroves are fringe growing in a narrow belt parallel to the shore line; they are characterized as hard bottom mangroves because they grow on dead coral beds covered with a shallow sediment layer typically less than 60 cm under mangrove stands. In addition, the trees are borderd by manmade industrial barriers restricting space. This had possibly contributed in limiting vertical tree growth. Moreover, the narrow growth space of the fringe Yanbu mangroves had possibly resulted in the denser tree growth and thus minimized diameter growth.

3.1.7.2 Yanbu biomass estimations

As mentioned earlier, *A. marina* is a multi-stemmed irregular tree, this was more noticeable in Yanbu than in Shuaiba where in the former, trees are shorter, smaller in diameter and denser (Table 3.1). It is worth noting that in the pre-sampling procedure used to derive sample size (n) needed for biomass estimation, a summation of all stems per tree was used to obtain a figure of DBH per tree. Initially, an average value of DBH was used in deriving n. After applying these values in the pre-sampling equation, n for Shuaiba and Yanbu was 13 and 10 respectively. However, after accomplishing fieldwork, it was later decided to use a summation of the stems DBHs as a correct figure instead of the average. Applying the new DBH figures in the pre-sampling equation yielded n values of 16 and 33 for

Shuaiba and Yanbu respectively. This was not a problem in Shuaiba as the total number of harvested trees was 16 (with 3 extra randomly harvested trees). However, cutting extra trees in Yanbu was not possible due to time limitation and other limiting constraints. Thus, the sample size in Yanbu was obtained from the first approach. This error in calculating the sample size may have contributed to the weak relationships between tree parameters and biomass components in Yanbu, a more representative sample size may yield a better relationship between tree parameters and biomass and therefore, obtaining better prediction estimates.

In addition to the multi-stemmed, irregular growth feature, branching of stems starts at very low levels of the tree trunk. And, in some cases, starts below the soil surface making it very difficult to differentiate stems from branches. These characteristics may have caused errors in estimating stem and branch components; similar inaccurate estimates of *A. marina* aboveground biomass using DBH were encountered in the literature (Tam *et al.*, 1995; Clough *et al.*, 1997). Tam *et al.*, (1995) working on stunted, irregular *A. marina* trees in China found no significant relationships between the tree parameters (DBH and height) and any biomass component. A more recent study by Kairo *et al.* (2009) studying aboveground biomass in Gazi Bay, Kenya of several species among which is *A. marina* has also found no simple relationships between DBH and any biomass component. Moreover, it was observed that Yanbu trees in some occasions might have shared close to surface roots with neighbouring trees, this means that both trees share some biomass which might have contributed to the measuring error of total biomass.

Although no simple relationship model describing Yanbu biomass was achieved, similar cases were reported in the literature for *Avicennia* species in which they did not show a straightforward relationship between biomass and tree parameters, or when compared to other species had low coefficient of determinations (Table 3.21).

Table 3.21 Biomass regression data of *Avicennia* mangroves from different resources (* significance)

Source	Species	r^2	Environmental condition	Tree parameter	Location
Kairo *et al.*,(2009)	*A. marina*	0.31	Hot and Humid	*log* DBH	Kenya
Medeiros & Sampaio (2008)	*A. schaueriana*	0.87*	Tropical	DBH^2Ht	Brazil
Saintilan (1997b)	*A. marina*	0.29*	Hyper Saline	DBH^2Ht	Australia
Tam (1995)	*A. marina*	0.20	Humid	DBH^2Ht	China

3.1.7.3 Aboveground biomass regression equations

Stem was the tree component best predicted by allometric equations followed by leaves and branches. In the current study, *A. marina* was noticeably more multi-stemmed and showed more irregular growth characteristics in Yanbu than in Shuaiba where in the former, trees were a lot smaller and denser (Table 3.3) making it very difficult to distinguish between stems and branches. This might have caused error in branch biomass calculation and thus resulted in a lower prediction precision compared to the other components. A similar case was reported in Medeiros and Sampio (2008) on *Avicennia schaueriana* in which biomass variations in woody components occurred; they reported that large variations in one component (branches) resulting in a lower r^2 value might have possibly compensated for biomass of another component (stem) resulting in a higher r^2 value. Another study by Tam *et al.*, (1995), where *A. marina* trees were described as small, irregular in shape and branching at very low levels of the trunk reported r^2 values of 0.39 and 0.46 for stem and branches respectively, and that these low values might be a result of errors caused by these multi-stemming/branching characteristics.

Although easier to estimate than other woody components, leaf biomass was less well predicted by prediction equations than stem, and this has been frequently reported in the literature (Komiyama *et al.*, 2000; Sherman *et al.*, 2003; Ong *et al.*, 2004; Soares and Schaeffer-Novelli, 2005; Smith III and Whelan 2006; Medeiros and Sampio 2008). This might be due to the fact that leaves are susceptible to seasonal variations which cause sampling biomass variations even in the same tree. Moreover, leaves are more vulnerable to environmental conditions such as wind and rain (Robertson and Alongi, 1992) and thus could lead to errors in biomass measurements. In fact, wind action may have caused some leaf loss from a few litter traps in the current study which might have contributed to variations in mean leaf biomass. A further investigation of litterfall production can be found in Chapter 4.

Height and DBH were the common predictors in all regression equations. Both variables are strongly related to the tree biomass and used as acceptable biomass predictor parameters in other tree biomass estimations (Ter-Mikaelian and Korzukhin 1997; Zianis and Mencuccini 2004; Wang, 2006; Zhau *et al.*, 2007). In mangroves, DBH is widely used as an acceptable predictor for biomass and reported to give high prediction precisions either when regressed with height (Mackey, 1993;

Tam *et al.*, 1995; Saintilan, 1997b; Komiyama *et al.*, 2002 and 2005, and Medeiros and Sampaio 2008) or as a sole variable (Clough and Scott, 1989; Ong *et al.*, 2004; Comley and McGuinness, 2005; and Hossain *et al.*, 2008).

It is worth noting that most of the biomass estimations done in mangrove ecosystems were of species other than *A. marina*. Most of the biomass estimations were for species that yield more biomass such as *Rhizophora mangle*. This might be due to the low biomass of *A. marina* tree compared to the other species; moreover, the multi-stemmed and growth irregularity features are other factors that could have made working with *A. marina* less attractive. Thus few biomass studies have considered such multi-stemmed species.

In similar cases where stems are forking close to the surface level, it would be of great interest to use stem diameter (for each stem per tree) just above the stem junction and, if present, the girth of common butts where stems arose from. Clough *et al.*, (1997) attained regression equations for *A. marina* and *Rhizophora stylosa* using stem girth and common butts as biomass predictors, the technique they used involved taking stem girth at 10-15 cm above stem junction and, in case where stems arose from a common butt at height of more than 20 cm above the ground, the butt girth was also recorded. All stems, branches, leaves and total biomass were best predicted using these parameters with high significant correlation (r^2 = 0.97 for total biomass). Another study by Comley and McGuinness (2005) working on *A. marina* and following the same procedure as Clough *et al.*, (1997) also attained similar accuracy (r^2=0.94 for total biomass). To the best of my knowledge, these are the only two published studies that account for the multi-stemming and common butt feature of *A. marina*. This method was not used in the current study due to the fact that such methodology was obtained only after the completion of field work measurements. However, if this method is to be applied in these sites, only girth measurement above stem junction would be desirable since the common butt feature is not present in these study locations. Hence, using girth measurement above stem junction in future estimations could be a good predictor for tree biomass. In addition, crown area/diameter is sometime used, in conjunction with DBH, as other predictor for biomass (Ross *et al.*, 2001, Coronado-Molina, 2004, Soares and Schaeffer-Novelli, 2005). In other studies, allometric equations of different species gave a better prediction when wood specific gravity of each species was considered. Komiyama *et al.*, (2005) reached a common allometric equation using DBH^2 Ht as a biomass

indicator and taking into account wood specific gravity of each species. Similarly, Medeiros and Sampaio (2008) reached a better coefficient of determination for pooled tree components when using specific gravity of each species, such estimate might be applicable in the southern region of the Red Sea where *A. marina* and *Rhizophora mucronata* grow beside each other.

3.1.7.4 Overall Aboveground biomass regression equations

Although reaching a model that can predict Yanbu's biomass was not achieved, pooling biomass data of both sites yielded a model that can generally predict the biomass components and total biomass of the mangrove trees. This regression equation could predict the biomass of all components and for the total with acceptable accuracy. A comparison of biomass estimations of the *Avicennia* species worldwide is shown in Table 3.22.

According to the global estimation range reported for mangrove biomass (6.8 t ha^{-1}- 436 t ha^{-1}) (Saenger and Snedaker, 1993), *A. marina* species fall in the lower half of that wide range. The lowest reported estimation came from New Zealand (6.8 t ha^{-1}) and the highest estimation came from Australia (341 t ha^{-1}). High biomass accumulations occur in tropical humid conditions where temperature and environmental conditions are favourable. In extreme conditions such as arid and temperate environments where temperature, salinity, and nutrient enrichment are limiting factors, few species can thrive and such areas are often mono-specific. Trees growing in such environments need to spend much of their energy production in mechanisms that help cope with the environmental stresses reducing availability for biomass accumulation. Such mechanisms would include physiological adaptations such as salt filtration and extrusion, thick waxy leaf surfaces and morphological adaptations such as aerial and anchoring root systems.

The current study of mangrove systems is of the most extreme environment worldwide, in fact the Red Sea represent the northern growth limits of any mangrove species worldwide (EEAA, 1998; Edwards and Head, 1987; Por *et al.*, 1977) thus *A. marina* species accounts for 90% of mangroves on the Red Sea. The current biomass estimations are comparable to those estimations in extreme environments; the estimation of the Red Sea mangrove of 14.8 t ha^{-1} slightly higher than those reported in the closest region of Gazi Bay, Kenya (11.7 t ha^{-1}) (Kairo *et al.*, 2009), and

sometimes higher than other regions (8.5 t ha^{-1} in China; and 6.8 t ha^{-1} in New Zealand). To the best of my knowledge, this is the first study that provided a quantitative estimation of aboveground biomass in the Red Sea as previous research on mangrove productivity has mainly focused on annual litterfall estimations and tree mensuration (Mandura, 1997, 1998 ; Khafaji *et al.*, 1991; and Saifullah *et al.*, 1989). Therefore, the current biomass estimation can serve as a baseline study for future comparisons.

3.1.8 CONCLUSIONS

The regression equations developed in this study would facilitate future estimation of aboveground mangrove biomass in the Red Sea. It is a valuable practical tool that estimates biomass from easily measured tree parameters. However, applying these equations must have the following considerations:

1. The regression equations are applicable when used within the DBH and height range reported in this study. Extrapolating to trees with wider ranges would yield incorrect biomass estimations. In fact, some studies argue that even the data at the extremes of the range (outliers) should be also avoided in order to avoid calculation errors (Saenger and Snedaker, 1993).
2. Site specific equations (*e.g.* Shuaiba) should be only applicable at the same or similar sites only.
3. The generalized equation can be used if an overall estimation of Red Sea mangrove biomass is desired.
4. Besides using the current predictors it would be of interest to use, when applicable, parameters that were reported to yield good prediction such as girth at base, crown diameter, butt girth, and wood density in the case of multiple species. Thus it is advisable to consider equation modification when necessary.

Table 3.22 Comparisons of aboveground biomass (t ha^{-1}) of *Avicennia* mangrove systems around the world (* mean values)

Source	Species	Leaves (t ha^{-1})	Branches (t ha^{-1})	Stem (t ha^{-1})	Total (t ha^{-1})	DBH range (cm)	Height (m)	Environmetal condition	Region
Current study	*A. marina*	2.70	5.77	6.40	14.77	3.3-20.2	2.5-3.8	Arid	Saudi Arabia
Kairo *et al.*, (2009)	*A. marina*	1.38	4.2	6.1	11.7	> 5cm	-	Hot and humid	Kenya
Medeiros and Sampaio (2008)	*A. schaueriana*	-	-	-	2.76	3.4-10.2	3.1-7.5	Saline	Brazil
Saintilan (1997b)	*A. marina*	-	-	-	56.1	-	0.5-2.0	Hyper saline	Australia
Tam *et al.*,(1995)	*A. marina*	0.62	4.9	2.9	8.5	8.3-14.3	3.1-5.6	Humid	China
Mackey (1993)	*A. marina*	-	-	-	341	-	1.4*	Subtropical	Australia
Woodroffe (1985)	*A. marina*	-	-	-	6.8	-	>1	Temperate	New Zeland
Woodroffe (1985)	*A. marina*	-	-	-	23.7	-	1-2.5	Temperate	New Zeland
Murray (1985)	*A. marina*	-	-	-	21.7	4.4*	4.3*	Humid	Australia
Davie (1984)	*A. marina*	-	-	-	30.0	-	-	Temperate	Australia
Clough and Attiwill (1975)	*A. marina*	-	-	-	86	-	-	Temperate	Australia
Briggs (1977)	*A. marina*	-	-	-	128	58.1*	7.34*	Temperate	Australia

In addition, the developed regression equation would aid in monitoring annual biomass increment as a function of site productivity and health. This is specifically important for sites similar to Shuaiba in the southern region of the Jizan and Farasan archipelago where *A. marina* grows bigger and are mixed with another mangrove species *R. mucronata*. In that region, coastal and industrial development has caused a huge disturbance to the ecosystem that has led to the deterioration of mangrove stands. For example, the construction of a large soil dam in the main Island of Farasan has resulted in the drying off and the significant mortality of a nearby mangrove stand (AL-Wetaid, 2003; Mandura and Khafaji, 1993).

It would be also desirable to investigate the annual biomass production in order to have a figure of production/biomass ratio between different sites. In severe environments, plants tend to spend much of the energy produced from primary production in dealing with the environmental stresses and therefore, little is available for biomass build up. In such harsh conditions of the Red Sea, tree biomass may be low due to various environmental stressors such as high soil salinity, extreme aridity, lack of water exchange, anaerobic conditions, and high hydrogen sulphide concentration in soil (Day *et al.*, 1987). However, this might not be the case for production; generally, mangrove trees are considered highly productive due to the high solar radiation and temperature, a year round growing season, and presumably abundant nutrients in the soil (Day *et al.*, 1987). Thus it is expected that production, biomass and their ratios would be higher in mangrove sites in the southern region of the Jizan and Farasan archipelago than in sites similar to the current study sites due to reduced salinity and high nutrient inputs (as a result of fresh water and nutrient enriched sediment input from close by valleys) and perhaps low other environmental stressors such as anaerobic conditions and hydrogen sulphide concentration (IUCN/MEPA, 1986). Therefore, the developed regression equation will add value to the mangrove biomass estimation in the Red Sea and facilitate comparisons of production/biomass ratio to those of the Indo-Pacific region and also to the global estimates in similar environments.

3.2 PART II-BELOWGROUND BIOMASS PRODUCTION

3.2.1 INTRODUCTION

Belowground is an important yet rarely investigated part of the mangrove tree biomass. The objective of this part of Chapter 3 was to estimate the root biomass of the two mangrove sites including aerial and fine root biomass. First, the methods and instrument used in estimating biomass are addressed followed by the results obtained, discussion and finally conclusion.

3.2.2 METHODOLOGY

3.2.2.1 Aerial root biomass estimation

Aerial root weight and density estimation were done for both Shuaiba and Yanbu sites during the 2008/2009 sampling season. Measurements were done using 1 m^2 quadrats and a portable scale. Quadrats were placed at distances of one, two and three metres away from trees (Plate 3.4); this was the maximum distance as trees are generally less than four metres apart. Quadrats were laid in north-south and east-west orientations to provide a representative sampling (Figure 3.6). A total of 36 trees were used in each site for aerial root estimation, all roots within quadrats were cut at ground level, separated from dead roots, counted and weighted on site. Subsamples of roots were taken for moisture content determination which was later used to derive dry weight.

3.2.2.2 Fine root biomass estimation

Fine root biomass estimation was carried out using random coring. A cylinder core measuring 1.9 cm radius (core area of 0.00113 m^2) was used for the estimate. In each site, core samples were taken at 1 and 2.5 metres away from trees. For each distance, core samples were taken and sectioned by depths into 0-10, 10-20, 20-30, 30-40 and 40-50 cm depths; a total of 24 core samples were taken from each site. Fine roots (<2 mm) from core samples were washed from sediments through a 250 μm sieve, separated from coarse roots and other materials, air dried, oven dried at 70°C for 24 hours, placed in a dessicator until a constant weight was achieved and then reweighed. Fine root biomass was estimated as tonnes per hectare basis using

estimates obtained from cores and core surface area. Further explanation of fine root calculation can be found in Appendix III.

Plate 3.4 Aerial root estimation of *Avicennia marina* mangroves using 1 m^2 quadrats.

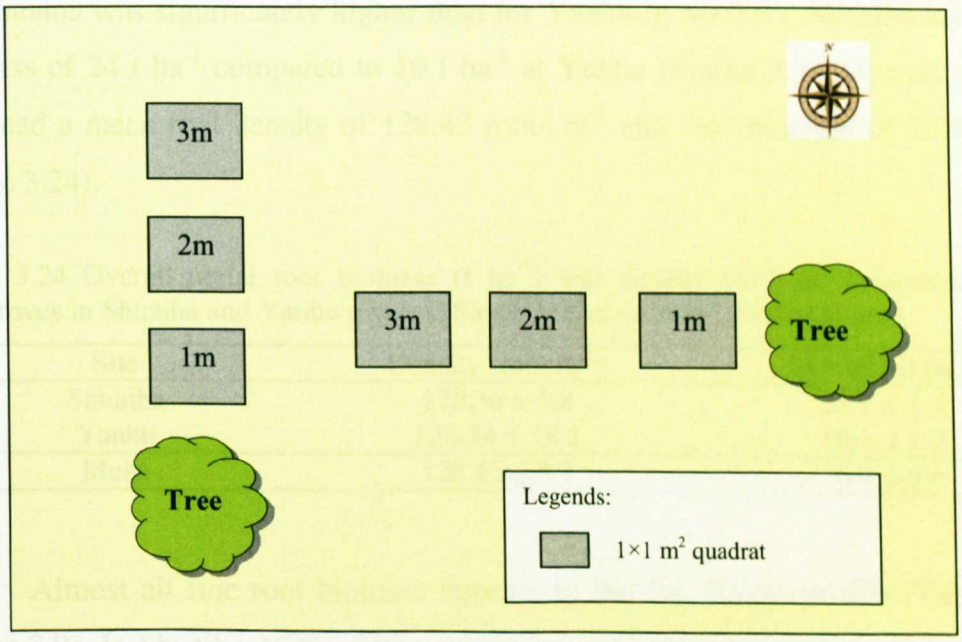

Figure 3.6 Schematic of *Avicennia marina* aerial root sampling method.

3.2.3 RESULTS

For Shuaiba, the number of aerial roots in a square metre was 129, 134, and 122 root m^{-2} at one, two, and three metres away from the trees respectively (Table 3.23). Their respective biomass was 23.5, 25 and 23 t ha^{-1} respectively. For Yanbu, the number of roots in a metre square was 141, 137, and 108 root m^{-2} at one, two, and three metres away from the trees. Their respective biomass was 11.4, 10 and 10 t ha^{-1} (Table 3.23). No significant differences were found in density or biomass at any distance for any site (p >0.05).

Table 3.23 Aerial root density and biomass of *Avicennia marina* mangroves in Shuaiba and Yanbu regions, Saudi Arabia (± standard deviations)

Distance	Shuaiba		Yanbu	
	Density (m^{-2})	Biomass (t ha^{-1})	Density (m^{-2})	Biomass (t ha^{-1})
1 m	128.78 ± 21.41	23.5 ± 4.4	141.1 ± 48.5	11.4 ± 3.5
2 m	133.94 ± 21.23	25.0 ± 4.6	136.8 ± 52.61	10.0 ± 3.7
3m	122.36 ± 17.54	22.6 ± 4.0	107.74 ± 43.27	9.0 ± 2.5

Generally, root density at Shuaiba was 128.36 root m^{-2} not different from those at Yanbu 128.54 (p >0.05) (Table 3.24 and Figure 3.7). However, root biomass for Shuaiba was significantly higher than for Yanbu (p >0.001); Shuaiba had a root biomass of 24 t ha^{-1} compared to 10 t ha^{-1} at Yanbu (Figure 3.8). Overall, the two sites had a mean root density of 128.45 roots m^{-2} and root biomass of 16.95 t ha^{-1} (Table 3.24).

Table 3.24 Overall aerial root biomass (t ha^{-1}) and density (m^{-2}) of *Avicennia marina* mangroves in Shuaiba and Yanbu regions, Saudi Arabia (± standard deviations)

Site	Density (root m^{-2})	Biomass (t ha^{-1})
Shuaiba	128.36 ± 5.8	23.7 ± 1.2
Yanbu	128.54 ± 18.1	10.1 ± 1.2
Mean	128.45 ± 0.1	16.9 ± 9.6

Almost all fine root biomass appears in the top 30 cm profile (Table 3.25, Figure 3.9). In Shuaiba, 97% of fine roots are concentrated in the top 30 cm profile (93.47 t ha^{-1}) with 52% of that is concentrated in the top 10 cm profile. Where in Yanbu, 98% of roots are concentrated in the top 30 cm profile (38.34 t ha^{-1}) However, 83% of that is concentrated in the top 10 cm profile.

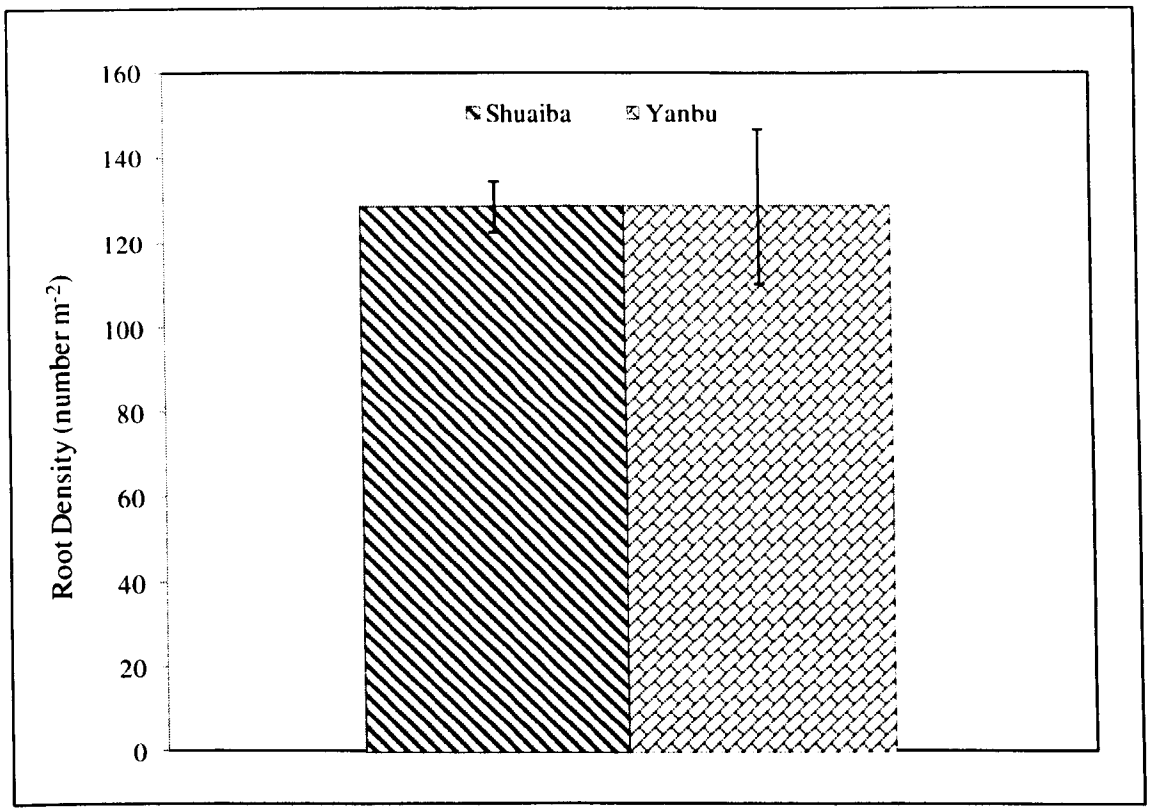

Figure 3.7 Aerial root density (m^{-2}) of *Avicennia marina* mangroves in Shuaiba and Yanbu regions, Saudi Arabia (± standard deviations).

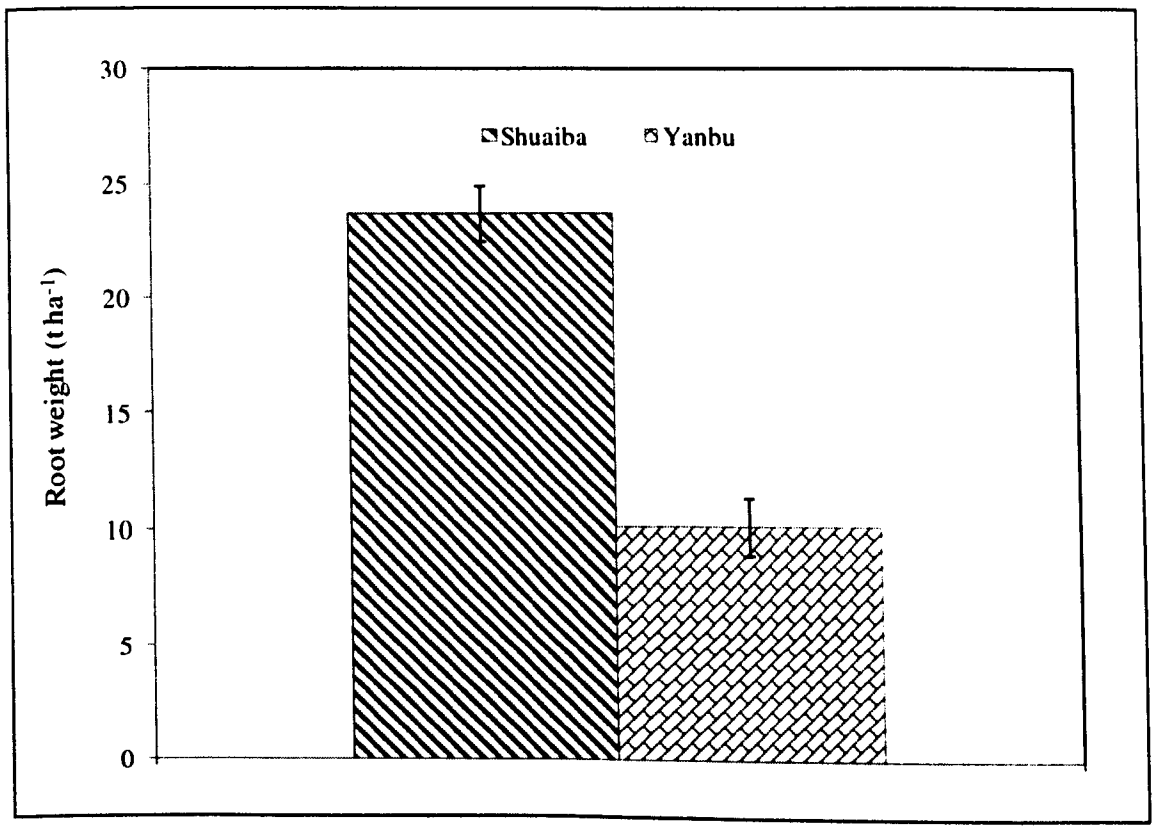

Figure 3.8 Aerial root biomass (t ha^{-1}) of *Avicennia marina* mangroves in Shuaiba and Yanbu regions, Saudi Arabia (± standard deviations).

Table 3.25 Fine root biomass for Shuaiba and Yanbu sites at different depths (± standard deviations)

Depth (cm)	Shuaiba (t ha^{-1})	Yanbu (t ha^{-1})
0–10	48.62 ± 18.26	32.50 ± 18.63
10–20	29.96 ± 14.00	3.41 ± 3.14
20–30	14.89 ± 13.18	2.43 ± 3.85
30–40	1.51 ± 2.54	0.47 ± 0.47
40–50	1.45 ± 3.60	0.30 ± 0.55
Total	96.42	39.12
2-site mean	67.77 ± 40.52	

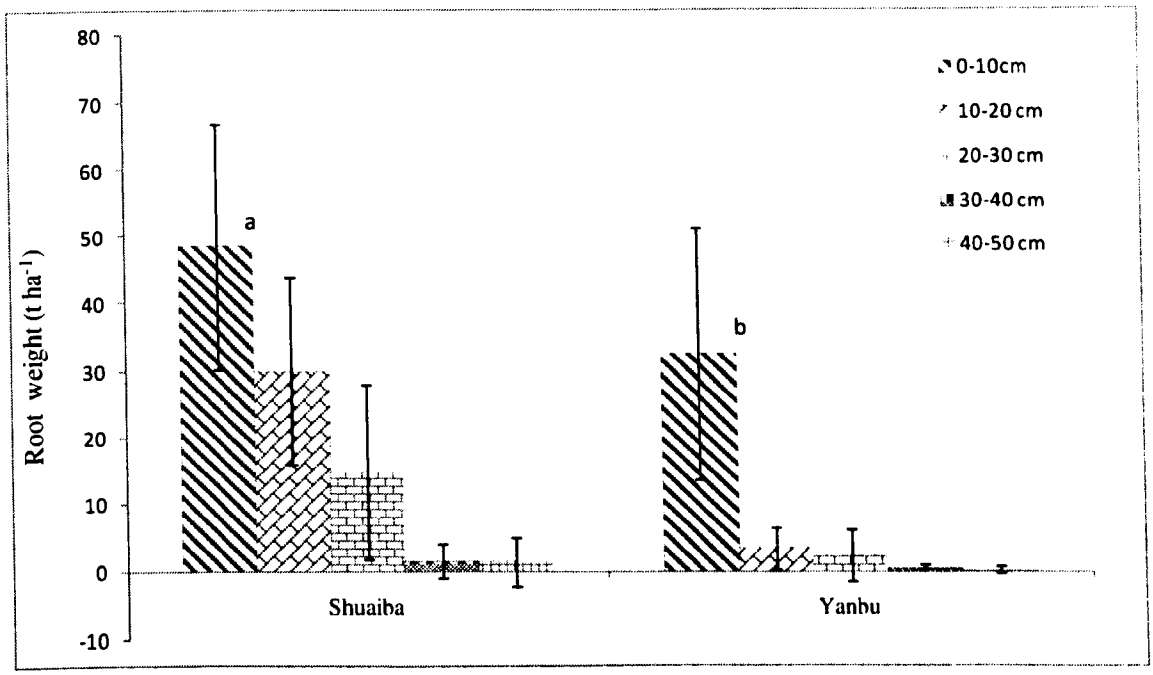

Figure 3.9 Fine root biomass of *Avicennia marina* mangroves in Shuaiba and Yanbu region, Saudi Arabia (error bars are standard deviations; different letters denote significant differences).

When the top 10 cm profile is compared between sites it was found that Shuaiba's fine root biomass was significantly higher than Yanbu's root biomass (p <0.05). In addition, when the total biomass of the two sites were compared, it was found that Shuaiba fine root biomass was higher than Yanbu (96.42 vs. 39.12) (Table 3.25). Moreover, fine root biomass did not differ significantly with distance from tree row (p >0.05), it was found that in Shuaiba, fine root biomass was 88.6 and 99.7 t ha^{-1} at 1 and 2.5 metres away from trees respectively, where in Yanbu, root biomass was 45.7 and 42.7 at 1 and 2.5 metres away from trees respectively.

3.2.4 DISCUSSION

Mangrove belowground biomass estimation is scarce and most of the biomass studies have neglected estimating the belowground part of mangrove trees. This is mainly due to several difficulties associated with quantitative sampling in intertidal habitats such as time consumption, equipment transportation and handling (Clough, 1982; Snedaker and Snedaker, 1984). Belowground biomass estimation in mangrove ecosystem generally follows random coring or systematic coring along transects (Clough and Attiwill, 1975; Briggs 1977; Lichacz *et al.*, 1984; Komiyama *et al.*, 1987). Such techniques are commonly used because of the simple, light, and robust equipment that provide reliable estimates (Snedaker and Snedaker, 1984). Unlike terrestrial forests where fine roots grow at different soil depths along anchoring roots, mangrove fine roots are generally concentrated in the top 100 cm of soil profile. In addition, mangrove fine roots are soft and can be easily penetrated making it even easer for coring (Komiyama *et al.*, 1987). In previous studies, several other methods were used in order to estimate belowground biomass including trench digging around targeted trees along with applying high pressure water in order to expose the root system for belowground estimation (Singer and Hutnik, 1965; Miller and Ng, 1977; Hoffmann and Kummerow, 1978) or mechanically pulling out the roots from the ground (Tamai *et al.*, 1983). These procedures have proven to be time consuming, not completely satisfactory, and may result in loss of significant amount of roots (especially fine roots).

Both Shuaiba and Yanbu have similar aerial root densities. This might be related to the shallow and extensive underground cable root system of *A. marina*; this root system has to be very dense in order to not only stabilize the tree but also, by dispersing tree weight over a large area, to keep the trees from sinking into the mud. Although not different in density, Shuaiba had a higher aerial root biomass than Yanbu. This might be attributed to the substrate, sedimentation and tree density of each site. Shuaiba's mangroves are basin with many in-plantation lakes accompanied by deep sedimentation (reaching approximately 1.8 m depth). This provides space and allow for higher root growth and biomass. On the other hand, Yanbu's mangroves are fringe with higher tree density and shallow sedimentation (reaching approximately 60 cm depth) offering very little for root biomass.

The top 10 cm soil profile contains more than 50% of the fine root biomass. High fine root biomass in top soil profiles is commonly reported in literature. Lauff (1967) found that most of *A. marina* roots are concentrated at the top 30 cm below the ground level. Moreover, Tamooh *et al.*, (2008) working also on *A. marina* in Kenya has found that 65% of fine roots is concentrated in the top 20 cm soil profile. In addition, Komiyama *et al.*, (2000) working on *Ceriops tagal* mangroves, has found few roots present below that same depth. The high fine root biomass in the top 10 cm profile obtained from the current study may be attributed to the mangrove adaptive mechanism for living in soft, saline, and sometimes, hot dry sediments (Briggs, 1977; Komiyama *et al.*, 2008), In addition, the high root biomass in the upper profile may also be attributed to the anoxic environment that halts root growth into deeper soil profiles (Stafford-Deitsch, 1996). The concentrated amount of roots in the top profile would also facilitate efficient uptake of water and nutrients in the sediment layers which are characterized by accumulated organic matter and relatively large amount of available nutrients as in terrestrial forests (Claus and George, 2005)

Estimates of fine root biomass in *A. marina* range globally from 15 t ha^{-1} to 166 t ha^{-1} (Table 3.26). As mentioned earlier, studies of belowground biomass worldwide are limited, thus few data were available for comparison. As most of the studies were from Australia, they are largely dissimilar to the environmental and regional conditions of the current study. Overall, the fine root biomass of the two sites was 67.77 t ha^{-1}, close to the limits reported in subtropical and hyper saline Australian environments (Saintilan 1997a and b) and higher than those reported in Kenya of 41.1 t ha^{-1} (Tamooh *et al.*, 2008). Moreover, the shoot to root ratio obtained from this study was 0.22 which could be one of the smallest reported in the literature. In his review paper on mangrove biomass and productivity, Komiyama *et al* (2008) reported shoot to root ratio of 12 mangrove stands ranging from 1 to 5.

It should be noted that applying allometric equations for belowground biomass was not possible in the current study due to the nature of the root system, which make assigning roots to specific trees impossible, due to the web spreading nature of the *A. marina* roots. Other studies working on similar species have reported estimates of roots at around 2 m radius, this is because it was impossible to trace roots to their final destination (Comley and McGuinness, 2005). This study

partitioned percentage of common stem and common belowground biomass according to relative stem diameter. However, poor relationships between DBH and belowground root biomass were reported in this study owing to limiting root estimate to the 2 m radius around the tree thus underestimating the true belowground biomass. However, applying allometric equations for belowground biomass may be possible in other cases. Such a method may be applicable for *R. mucronata* in the southern region where accessing the anchoring roots is much easier than for *A. marina*, and could thus provide a more accurate estimate of belowground biomass.

Table 3.26 A comparison of *Avicennia marina* fine root biomass (t ha^{-1}) from various sources

Source	Fine root biomass (t ha^{-1})	Environmental condition	Location
Current study	67.77	Arid	Saudi Arabia
Tamooh *et al.*, (2008)	41.4	Tropical	Kenya
Saintilan (1997a)	70.0-166	Subtropical	Australia
Saintilan (1997b)	15-60	Hyper saline	Australia
*Briggs (1977)	153.8	Temperate	Australia
*Mackey (1993)	118.6	Subtropical	Australia
Alongi (2009)	21.2	-	Australia

* Aerial and fine root estimates.

3.2.5 CONCLUSIONS

The current findings are one of the very few belowground estimates done on *A. marina* trees. The investigation showed that *A. marina* belowground biomass was greater than those estimates obtained in East Africa and comparable to estimates obtained in similar environmental conditions. Thus, the current estimation will add a significant value to the regional estimates and to the global estimates of roots in similar environments.

In addition, mangrove roots are an important bio-monitors for heavy metals accumulation and for pollution monitoring, thus an accurate estimation is essential if conducting such research is desirable. For that, a complete estimate of heavy metals in mangrove roots can be found in Chapter 7.

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CHAPTER 4

LITTERFALL PRODUCTION AND REMOVAL

4.1 INTRODUCTION

Litterfall is a useful index of mangrove productivity since it is a major fraction of mangrove's net productivity that supports aquatic animals (Bunt and Boto, 1979). Litterfall production makes a significant contribution to inshore and estuarine productivity; leaves, twigs, reproductive structures and fruits of mangroves fall to the ground providing a primary food source for a wide variety of aquatic animals, such as crabs.

The tidal effect on leaf decomposition, accumulation and export is well documented (Boto and Bunt, 1981; Twilley, 1985; Twilley *et al.*, 1986; Robertson, 1988); High tidal ranges can wash litter off the ground and this results in the export of leaves (and therefore nutrients) from the system. For the Red Sea, the tidal amplitude is one of the smallest in the world; in its northern and southern parts, the amplitude may reach less than a metre and may average 50 centimetres; this gets smaller in the central Red Sea, in fact, a nodal point with zero amplitude at 20° N (in central Red Sea) was reported (Edwards and Head, 1987). Monitoring the tidal ranges in the mangrove systems will aid in estimating the fate of the fallen litter, litterfall in mangrove systems is subjected to tidal activity which normally flushes litter from the mangroves, thus exporting nutrient to adjacent systems (Heald, 1971; Odum, 1971). In the current study the tidal levels will aid in estimating litter removal from forest floor, the patterns of litter decomposition rates (Chapter 5) and in the possible export of nutrients (Chapter 6).

Crab activities within mangrove systems are an important element in mangrove ecology and their feeding habits can largely govern the function and energy flow in mangrove systems of the Indo Pacific region (Robertson and Daniel, 1989). Crab litter-removal and breakdown play a significant part in mangrove nutrient dynamics (Hogarth, 2007). During grazing, crabs break leaf material into smaller fragments, increasing leaf surface area and providing nutrient access to fungi

and bacteria and therefore accelerate decomposition rates (Heald, 1971; Fell, *et al.*, 1975; Odum and Head, 1975; Cundell *et al.*, 1979). Crab species in the families Grapsidae and Sesarmidae are common mangrove-associated species that can play a significant role in litter accumulation and decomposition; in the Red Sea, a number of crab species were reported within the mangrove systems, some of which are mangrove associates (Mandura *et al.*, 1987; Price *et al.*, 1987), However, their role and influence on mangrove litter dynamics was never investigated owing to their limited abundance (IUCN/MEPA, 1986).

Edwards and Head (1987) hypothesized that mangrove stands constitute a nutrient conserving and accumulating ecosystem (evident in the absence of nutrient inputs from rivers and oligotrophic waters of the Red Sea); they also hypothesized that mangrove stands in the Red Sea form a major source of high primary productivity in an otherwise barren zone. On the other hand, it was hypothesized that the Red Sea mangroves do not represent a significant source of primary productivity due to their low litterfall production. The World Conservation Union (IUCN/MEPA, 1986) reported that the annual production of litterfall of the Red Sea mangroves is assumed to be less than the global estimate of 0.5 kg C m^{-2} y^{-1} (based on observations of leaf litter, degree of organic sediment development and the associated biota).

The objectives of this chapter were to estimate annual litter fall production and removal in Shuaiba and Yanbu sites. The factors affecting litterfall accumulation on forest floor including tidal activities and crab removal was also examined.

The hypotheses of this study are:

1. Litterfall production on the Red Sea is less than the global estimates yet comparable to global estimates in similar extreme climatic conditions.
2. Tidal ranges affect litter accumulation and removal from the mangrove systems rather than crab activities.

4.2 MATERIALS AND METHODS

4.2.1 Litterfall estimation

Litterfall production was estimated over a period of two years (June 2007 to May 2009) using the litter trap technique. A 1 m^2 trap with 4 mm^2 pore size was used, and traps were set under the tree canopy at a height higher than the highest tide

to prevent contact with sea water. Five traps were randomly set in each plot and numbered. A total of 60 traps per site were used (Plate 4.1a,b). Litterfall was collected monthly from each site; the litterfall from each trap was collected in paper bags, labelled as to location, dated, numbered and sealed.

Similarly, the standing crop of litter was collected to estimate ground leaf litterfall accumulation and removal. The experiment was initiated in July 2007 and August 2007 for Yanbu and Shuaiba respectively for a period of two years. Unfortunately, the ground litter collections of June 08 and 09 in Yanbu and of July 09 in Shuaiba were accidently lost and thus their corresponding results are not presented. The standing crop estimation method involved collecting leaf litterfall from traps and ground plots at the same collection time, on the assumption that litter in ground plots was subject to removal via water inundation. Therefore the weight difference between leaf litter collected in traps (total litter) and ground leaf litter represent the amount removed per month (Robertson and Daniel, 1989, Twilley *et al.*, 1986). The differences between trap and ground means per quadrat were compared for significant differences and expressed as t ha^{-1}. This method encompasses the following assumption and considerations:

1. Ground plots have the same area (1 m^2) and placed in close proximity (1 m distance) to the litter trap.
2. Ground litter is assumed to be solely removed by tidal activities and not by crab grazing.
3. The same amount of trap litter is assumed to fall on the corresponding ground plots owing to their close proximity.

To set the ground plots, a 1 m^2 area was marked within the vicinity of the litter traps, cleared of aerial roots, debris and other materials (Plate 4.2). In Shuaiba, only two transects (six plots) were selected for standing crop estimation. Within each plot, three ground plots were set with each plot correspond to a litter trap. Similarly in Yanbu, six plots were selected for standing crop estimation with three ground plots in each plot. A total of 18 ground plots per site were set for the standing crop experiment. At each collection time, litterfall was collected from traps and their corresponding ground plots. Ground plot litter was placed in paper bags, labelled to the corresponding trap, dated and sealed to prevent loss. After each collection time, litterfall samples were transferred to the laboratory for analysis, where samples were

Plate 4.1 (a,b) A 1 m^2 litter trap suspended under mangrove trees on the Red Sea coast, Saudi Arabia.

oven dried at 70°C for 24 hours, kept in a dessicator until reaching constant weight and then reweighed using a four decimal digits scale. Moreover, subsamples of litterfall were ground in a Wiley mill for nutrient and leaf component analysis.

Plate 4.2 A 1 m^2 ground plot used for standing crop litterfall estimation in mangrove stands on the Red Sea coast, Saudi Arabia.

4.2.2 Monitoring tidal ranges and crab activities

Tidal information for the Yanbu site was obtained from the Saudi Aramco Tidal Tables (Saudi Aramco Tidal Tables, 2007, 2008 and 2009) in which water height is given in centimetres above the lowest astronomical tide. The tidal information for Shuaiba was obtained from Aramco tidal station at Jeddah city (approximately 90 km north of Shuaiba) which is the closest station that falls within the vicinity of Shuaiba site. In conjunction, *in situ* tidal measurements were made in both Shuaiba and Yanbu sites. To provide a description of the local topography of the mangrove areas with respect to the obtained tidal range, marked timber stakes (Plate 4.3) were used as tidal stations and placed in experimental plots within each

site. In Shuaiba, a numbered stake was inserted into each plot in transect B and C (6 plots) with a total of six stacks per site. In Yanbu one stake per plot was placed at a high tide with a total of five stacks per site. Water levels were recorded at each site and compared with the tabulated monthly tidal ranges (English *et al.*, 1997; LeVay, pers. comm., 2010).

The possible removal of leaf litter by crabs was examined. Prior to the crab removal experiment, qualitative assessments of crab activities and abundance were undertaken in both sites. Gill nets (2.5 cm^2 mesh size) were set perpendicular to major creek mouths by the edge of the mangrove plantations to capture the aquatic animals accessing the mangroves, the nets were laied in the early morning (7 am) and checked for captured animals in the afternoon (3 pm) of the same day, sampling aquatic animals using gill nets were made five times over the study period. In addition, crabs within and around the mangrove plantations were hand captured when found. Crab seasonal occurrences and abundance were also observed by direct contact or by binocular monitoring, samples of captured crabs were taken for species identification and further laboratory analysis.

Based on the crab species found in the mangrove stands, it was decided that an experiment of feeding habits would be necessary. The crab feeding rate on mangrove leaves was assessed using a leaf tethering experiment (Robertson, 1986). In each site, 20 leaf groups (each containing 10 leaves) were tethered along a 2 m nylon twine attached to aerial roots or tidal stakes. Five nylon twines per plot were randomly placed with a total of 20 lines per site. Leaves were monitored over a period of two weeks; the number of eaten, chopped or removed leaf groups from each line was recorded and expressed as index removal per 2 weeks.

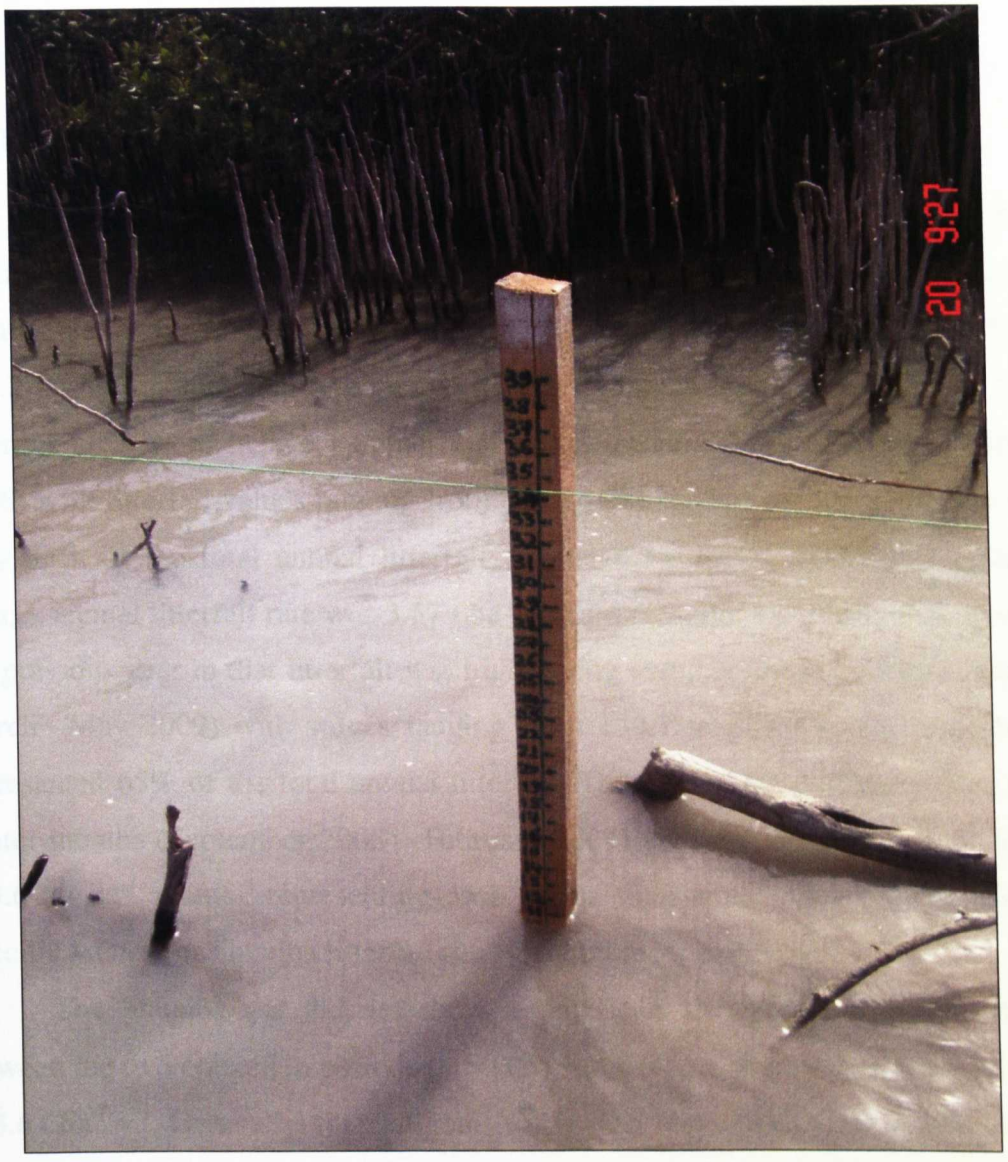

Plate 4.3 A marked timber stake for measuring tidal ranges in mangrove systems, Saudi Arabia.

4.3 STATISTICAL ANALYSIS

Litterfall data were processed and analysed using Excel (2007) and SPSS ver. 14. Homogeneity of variance was confirmed using Levene's test of equal variance and one-way ANOVA test with a $p < 0.05$ significance level was used to test for significance differences in seasonal litterfall rates, differences between mean litterfall rates were compared using Tukey's Honestly Significant Different test (HSD); differences in mean litter fall between sites was tested for significant using

Student T-test; a pair-wise T-test was used to test for significant differences between litterfall and standing crop.

4.4 RESULTS

4.4.1 Shuaiba litterfall

The annual litterfall rate in the 2007/2008 collection period was 3.72 t ha^{-1} y^{-1}. Litterfall rates were significantly greater (p <0.05) during spring and summer months (Jun-July 2007 and March to May 2008) with values ranging from 453.5 to 536.2 kg ha^{-1} $month^{-1}$ and represented 67% of the total annual litterfall. The lowest litterfall rates were in late summer and winter months (August 2007 – February 2008) with litterfall values ranging from 83.9 to 276 kg ha^{-1} $month^{-1}$ and represented only 33% of the total annual litterfall (Figure 4.1). In the 2008/2009 collection period, annual litterfall rate was 3.57 t ha^{-1} y^{-1} and seasonal variability was similar to the previous year in that litterfall was high during summer (June – August 2008 and March –May 2009) with values ranging from 139.7 to 549.9 kg ha^{-1} $month^{-1}$ and represented 65% of the total annual litterfall. The lowest litterfall rates were in the winter months (September 2008 – February 2009) with values ranging from 80.4 to 409.6 kg ha^{-1} $month^{-1}$ representing 35% of the total annual litterfall (Figure 4.1). Specific values for Shuaiba litterfall can be found in Appendix IV.

The Shuaiba site did not show a significant difference in annual litterfall between the two collection periods (p >0.05) and gave an overall annual litterfall rate of 3.6 t ha^{-1} y^{-1}. Generally, litterfall rates followed a bimodal annual cycle, rates were greater in summer, decreasing through winter and started peaking again in the following spring reaching the maximum in summer (Figure 4.2, 4.3). Litterfall rates were significantly greater (p <0.05) during summer (June – August and March – May) with values ranging from 225.9 to 543.1 kg ha^{-1} $month^{-1}$ and were lower during the winter months (September – February) with values ranging from 87.7 to 342.8 (Figure 4.3). Further information on the monthly litterfall averages for Shuaiba can be found in Appendix IV.

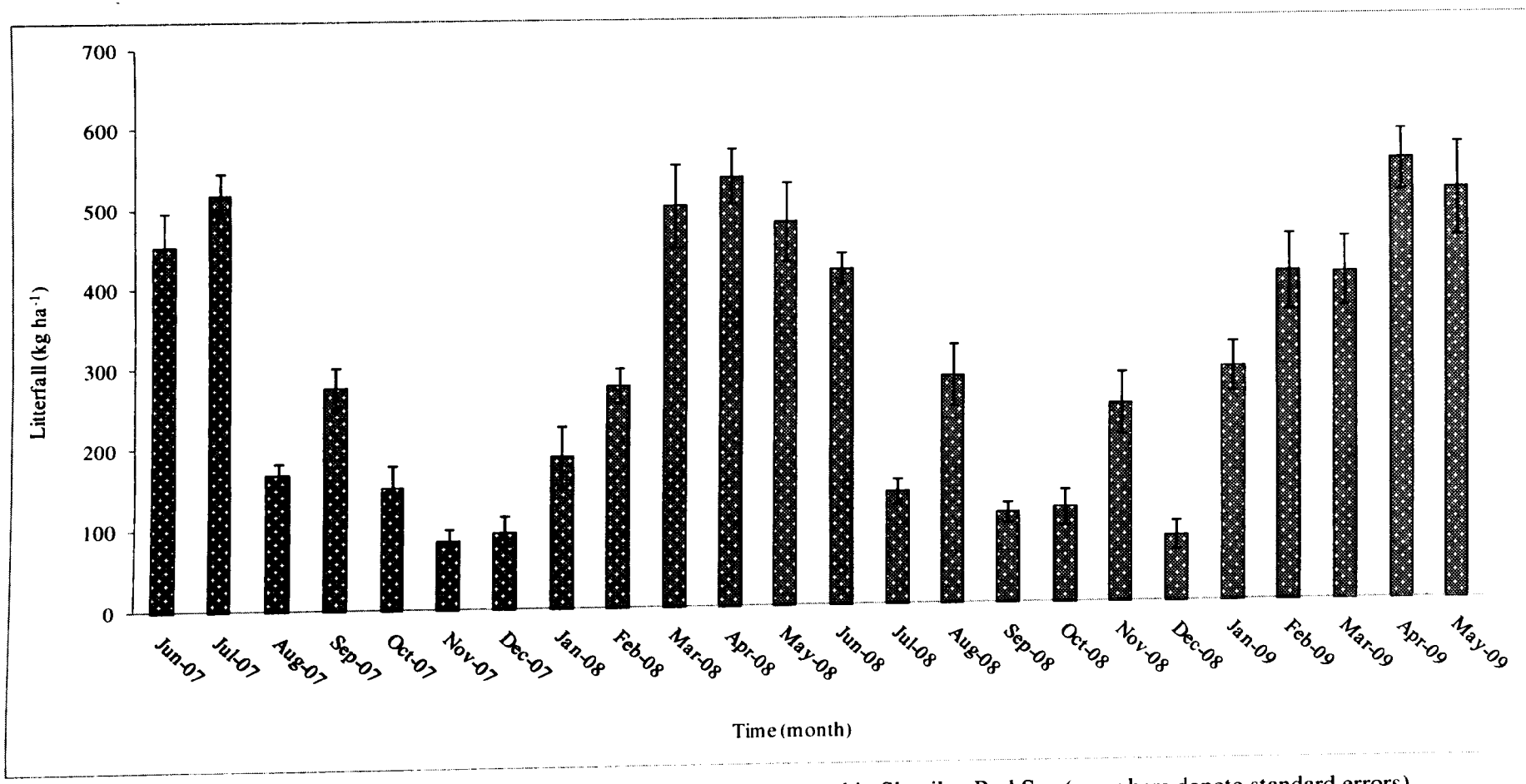

Figure 4.1 Mean monthly litterfall (kg ha^{-1}) for years (2007-2008) in a mangrove stand in Shuaiba, Red Sea (error bars denote standard errors).

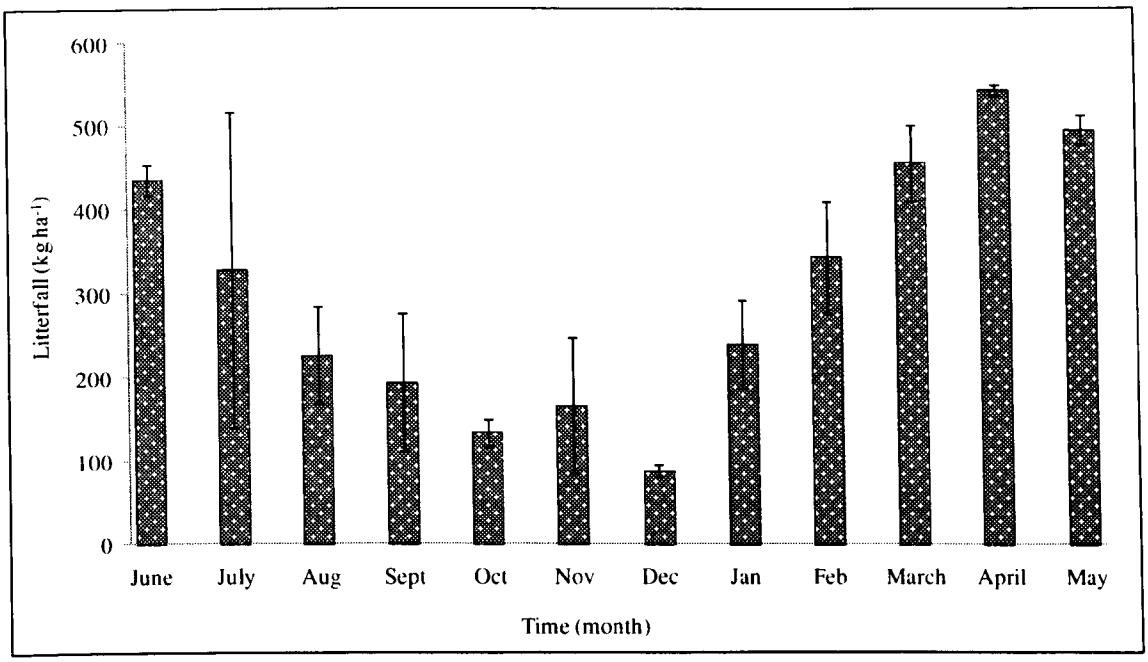

Figure 4.2 Average monthly litterfall (kg ha^{-1}) in a mangrove stand in Shuaiba, Red Sea (error bars denote standard errors).

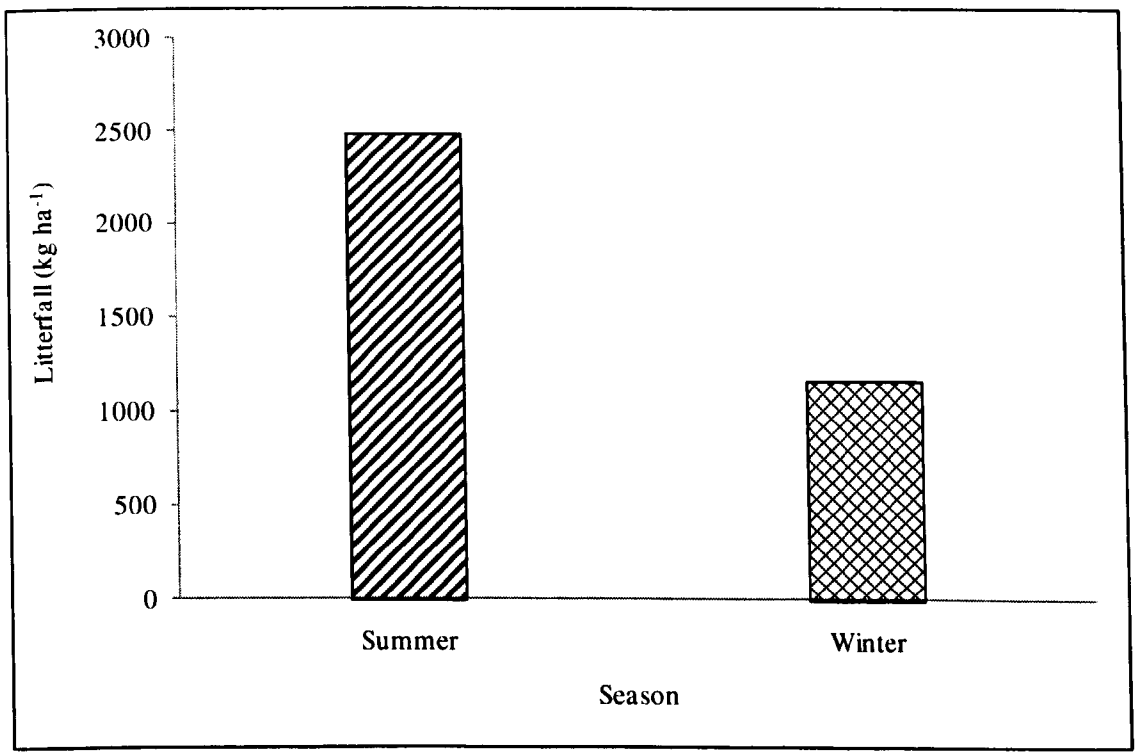

Figure 4.3 Seasonal litterfall (kg ha^{-1}) in a mangrove stand in Shuaiba, Red Sea.

4.4.2 Yanbu litterfall

Litterfall rates in Yanbu were not as seasonally variable as of the Shuaiba site. Rates in the 2007/2008 collection period had an annual rate of 3.54 t ha^{-1} y^{-1} with values ranging from 505.1 to 128.3 kg ha^{-1} $month^{-1}$. Although litterfall rates were not significantly different between seasons, summer rates were generally greater than in winter (60% of total annual litterfall) ranging from 505.1 to 128.3 kg ha^{-1} $month^{-1}$, while winter rates (40% of total annual litterfall) were ranging from 353.5 to 157.4 kg ha^{-1} $month^{-1}$.(Figure 4.4). Similarly, in 2008/2009, summer months represented 61% of the total annual litterfall with values ranging from 557.7 to 155.7. While in winter, litterfall rates ranged from 337.5 to 133.1 kg ha^{-1} $month^{-1}$ and represented 39% of the total annual litterfall (3.48 t ha^{-1} y^{-1}) (Figure 4.4), further information on litterfall monthly averages for Yanbu can be found in appendix V.

Overall, there were no significant differences in annual littefall between the the two collection periods in Yanbu (p >0.05) with an annual litterfall rate of 3.5 t ha^{-1} y^{-1} with values ranging from 539.5 to 142 kg ha^{-1} $month^{-1}$. Similar to Shuaiba, litterfall in Yanbu had a trend of high rates in summer, decreasing toward winter and peaking again in the following summer (Figure 4.5). The summer months represented 60% of the total litterfall with values ranging from 539.5 to 142 kg ha^{-1} $month^{-1}$. Winter litterfall rates represented 40% of the total annual litterfall and values ranged from 345.5 to 148.9 kg ha^{-1} $month^{-1}$ (Figure 4.5, 4.6). In addition, litterfall rates did not differ significantly between Shuaiba and Yanbu sites (Table 4.1)

Table 4.1 Annual litterfall rates (kg ha^{-1} y^{-1}) in mangrove stands in Shuaiba and Yanbu, Saudi Arabia (± standard deviations)

Site	2007/2008	2008/2009	Overall
Shuaiba	3720.6	3569.5	3645.1 ± 106.00
Yanbu	3536.9	3512.9	3509.4 ± 38.81

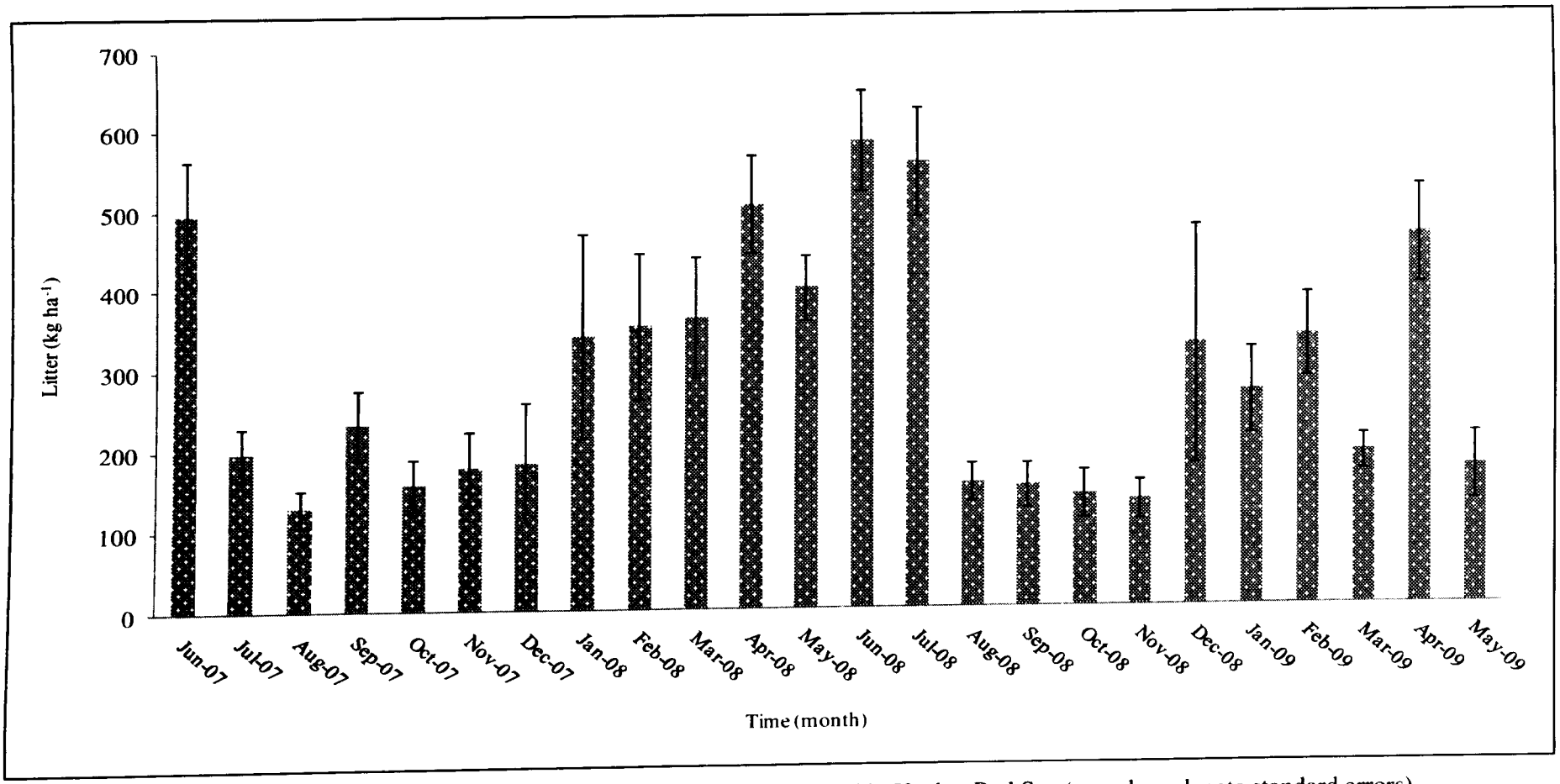

Figure 4.4 Mean monthly litterfall (kg ha^{-1}) for years (2007-2008) in a mangrove stand in Yanbu, Red Sea (error bars denote standard errors).

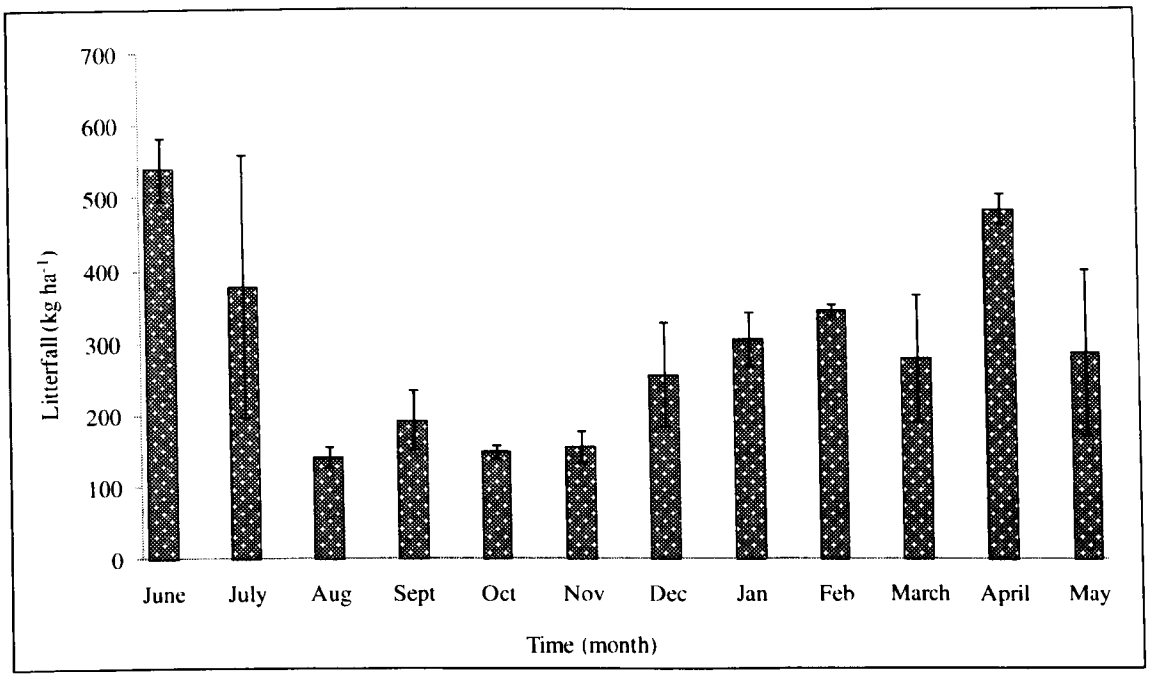

Figure 4.5 Average monthly litterfall (kg ha^{-1}) in a mangrove stand in Yanbu, Red Sea (error bars denote standard errors).

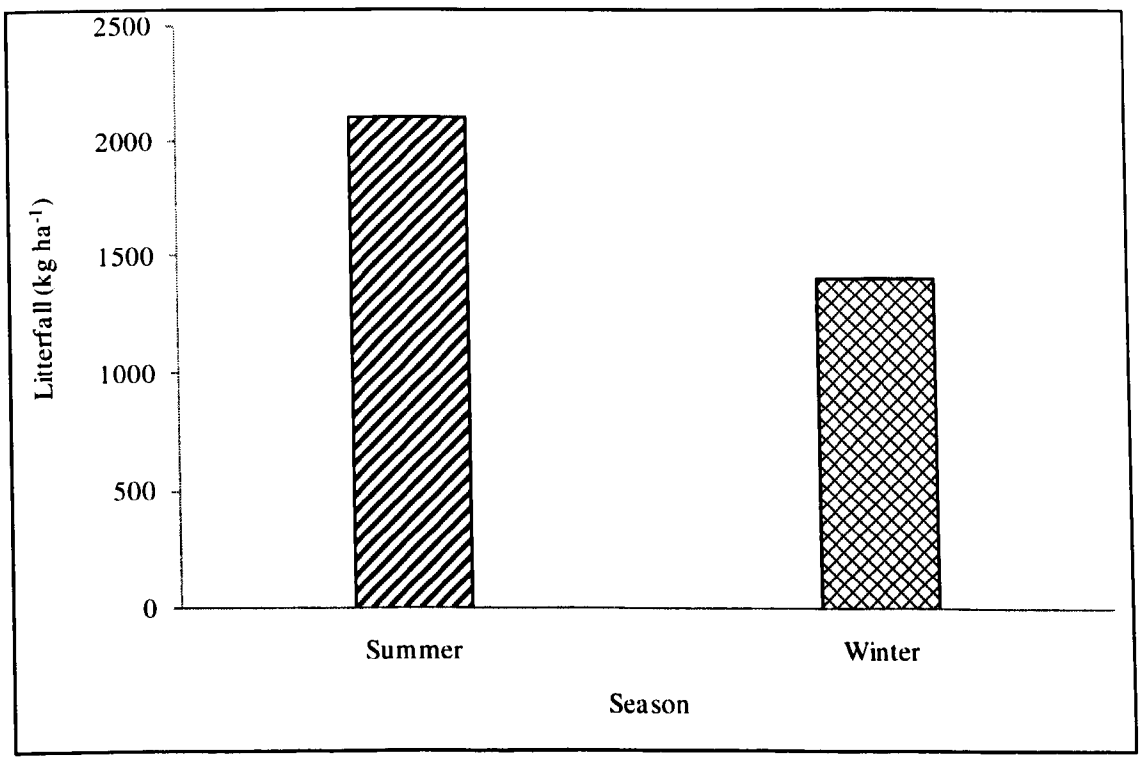

Figure 4.6 Seasonal litterfall (kg ha^{-1}) in a mangrove stand in Yanbu, Red Sea

4.4.3 Shuaiba standing crop litter

Over a period of two years, Shuaiba standing crop litter rate ranged from 18.8 to 440.5 kg ha^{-1} $month^{-1}$ with an overall mean of 86.1 kg ha^{-1} $month^{-1}$ and an overall annual production of 990.2 kg ha^{-1} y^{-1} while litterfall ranged from 15.9 to 532.6 kg ha^{-1} $month^{-1}$ with an overall mean of 247 kg ha^{-1} $month^{-1}$ and an overall annual litterfall of 2840.7 kg ha^{-1} y^{-1} significantly greater than ground litter (p <0.05) (Table 4.2 and Figure 4.7). Significant differences between traps and ground litter was present during winter and spring months (January to June) in which the tidal ranges was highest of the year (Figure 4.7). Furthermore, monthly standing crop litter showed a significant negative correlation with the monthly tidal ranges (spring and neap tides) in which low ground litter corresponded to high tidal levels (Table 4.3 and Figure 4.8). Specific values for Shuaiba litterfall and standing crop can be found in appendix VI.

Table 4.2 Annual litterfall and standing crop (kg ha^{-1} y^{-1}) in a mangrove stand in Shuaiba, Saudi Arabia

Leaf Litter	2007/2008	2008/2009	Overall
Litterfall	2855.6	2825.8	2840.7
Standing crop	1079.4	900.9	990.2

Table 4.3 Correlation of standing crop with the different tidal ranges in a mangrove stand in Shuaiba, Saudi Arabia.

Factor correlation	Correlation coefficient
Ground litter - HHS	-0.45
Ground litter - LLS	-0.42
Ground litter – LHN	-0.50
Ground litter – HLN	-0.40

HHS: highest high spring tide, LLS: lowest low spring tide, LHN: lowest high neap tide, HLN: highest low neap tide

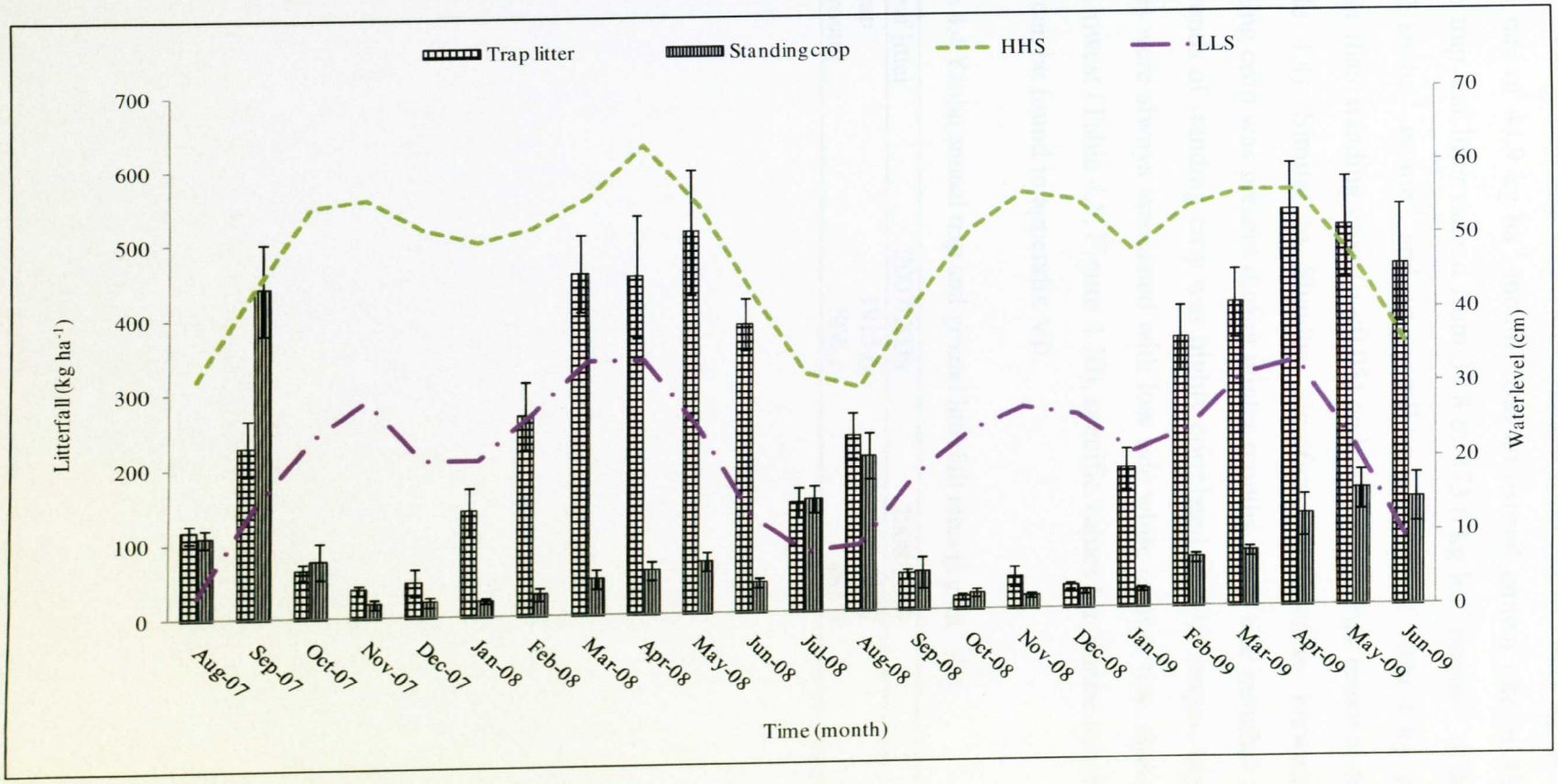

Figure 4.7 Mean monthly litterfall and litter standing crop in a mangrove stand in Shuaiba, Saudi Arabia (HHS: highest high spring tide, LLS: lowest low spring tide, error bars are standard errors).

4.4.4 Yanbu standing crop litter

Yanbu standing crop values ranged from 9.3 to 106.9 kg ha^{-1} $month^{-1}$ with a mean rate of 44.9 kg ha^{-1} $month^{-1}$ and an overall annual rate of 494.7 kg ha^{-1} y^{-1}, while trap leaf litter ranged from 39.8 to 473.6 kg ha^{-1} $month^{-1}$ with a mean rate of 169.2 kg ha^{-1} $month^{-1}$ and an overall annual rate of 1861.4 kg y^{-1}, significantly greater than standing crop (p <0.05) and indicating litter removal from forest floor (Table 4.4). Similar to Shuaiba, significant differences between litterfall and standing crop was present during winter months (high tide months) (Figure 4.9) and the trends of standing crop was highly correlated to tidal ranges, high standing crop values were always associated with low tide while in high tide, standing crop values were lowest (Table 4.5, Figure 4.10), specific values for Yanbu litterfall and standing crop can be found in appendix VII.

Table 4.4 Yanbu annual trap and ground litterfall rates (kg ha^{-1} y^{-1}).

Leaf litter	2007/2008	2008/2009	Overall
Trap	1915.6	1807	1861.4
Ground	506.1	483.3	494.7

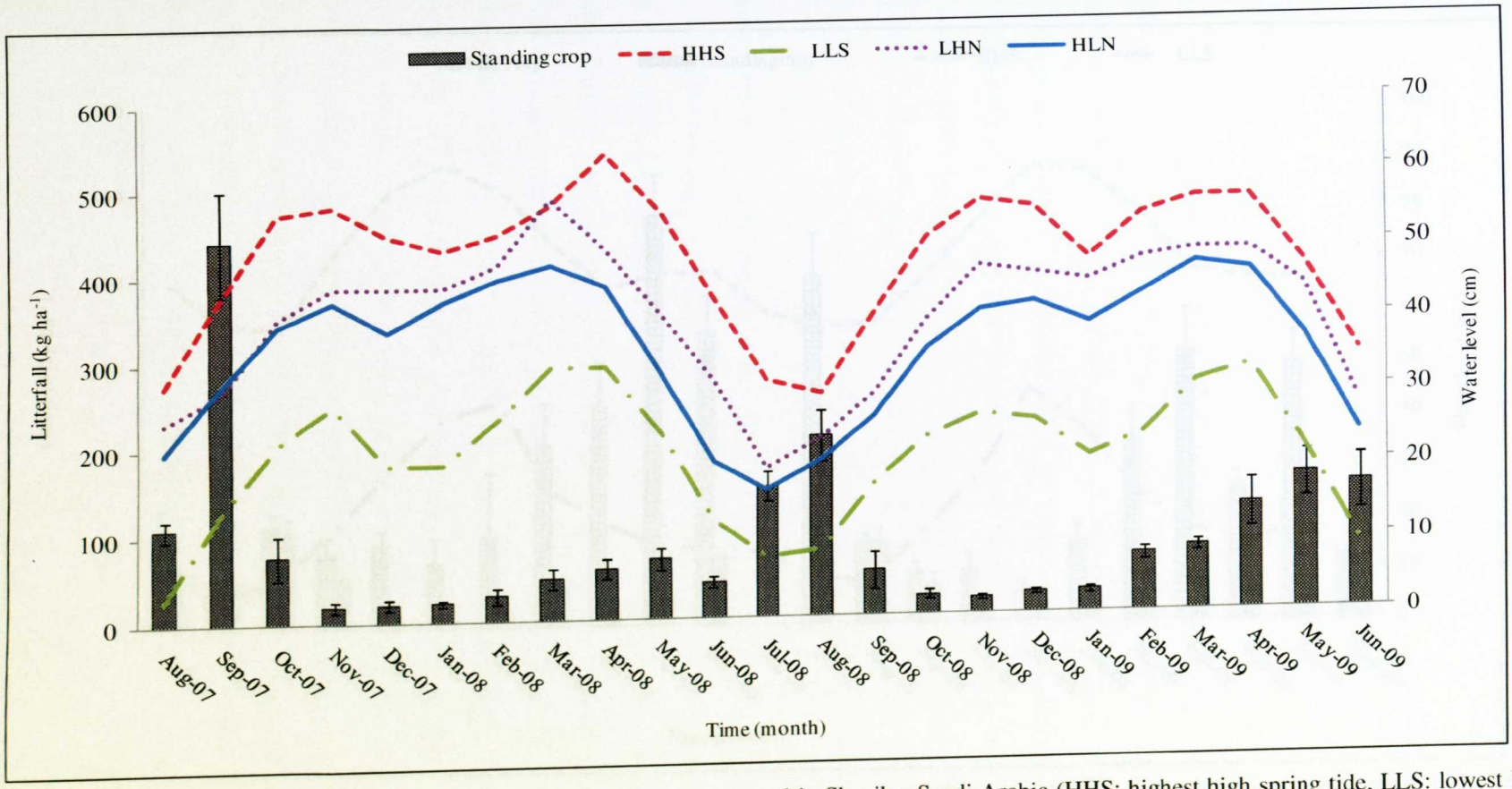

Figure 4.8 Monthly litter standing crop and tidal inundation in a mangrove stand in Shuaiba, Saudi Arabia (HHS: highest high spring tide, LLS: lowest low spring tide, LHN: lowest high neap tide, HLN: highest low neap tide, error bards are standard errors).

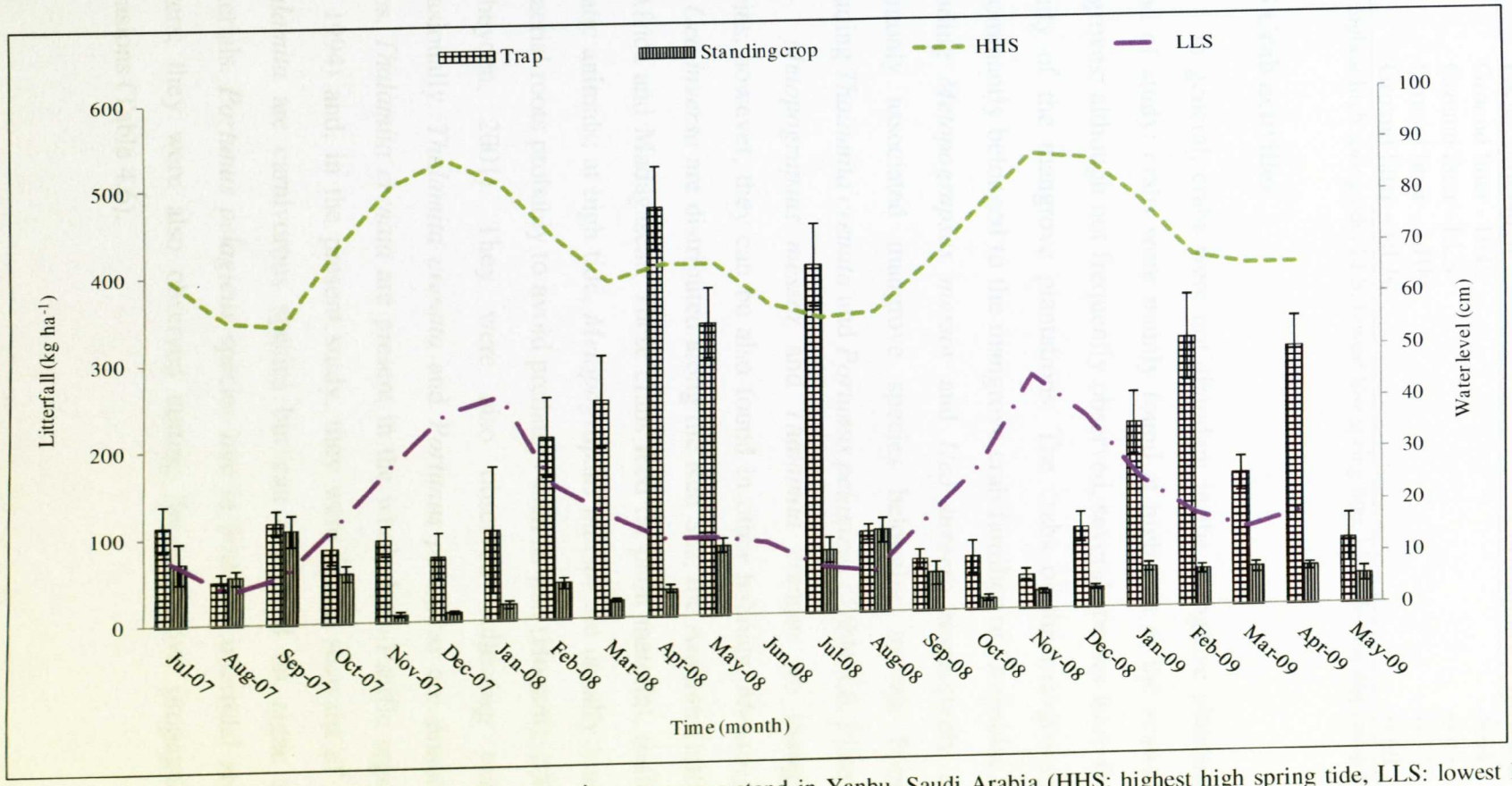

Figure 4.9 Mean monthly litterfall and litter standing crop in a mangrove stand in Yanbu, Saudi Arabia (HHS: highest high spring tide, LLS: lowest low spring tide, error bards are standard errors).

Table 4.5 Correlation of Yanbu standing crop with the different tidal ranges in a mangrove stand in Yanbu, Saudi Arabia.

Factor correlation	Correlation coefficient
Ground litter - HHS	-0.64
Ground litter - LLS	-0.63
Ground litter – LHN	-0.52
Ground litter – HLN	-0.60

HHS: highest high spring tide, LLS: lowest low spring tide, LHN: lowest high neap tide, HLN:

4.4.5 Crab activities

In general, crabs were not abundant in the mangrove plantations during the period of study; crabs were mainly found at high tide at the seaward edge of the mangroves; although not frequently observed, several species were found within the vicinity of the mangrove plantations. The crabs of the mangroves in both sites predominantly belonged to the mangrove crab families of Grapsidae and Ocypdidae including *Metopograpsus messor* and *Uca inversa* respectively and to another commonly associated mangrove species belonging to the Portunidae family including *Thalamita crenata* and *Portunus pelagicus* (Table 4.6, Plate 4.4 and 4.5)

Metopograpsus messor and *Thalamita crenata* are mangrove-associated species, however, they can be also found in other habitats. *Metopograpsus messor* and *Uca inversa* are distributed along the Red Sea, the Arabian Gulf, the East coast of Africa and Madagascar. These crabs feed on plant material, sediment and other aquatic animals; at high tide, *Metopograpsus messor* are usually found climbing on the aerial roots probably to avoid predators such as fish (Hogarth, 2007; Gillikin and Verheyden, 2001). They were also observed collecting mangrove leaves occasionally. *Thalamita crenata* and *Portunus pelagicus* are predatory swimming crabs. *Thalamita crenata* are present in the whole Indo-Pacific region (Cannicci *et al.*, 1994) and, in the present study, they were found seaward of the mangroves. *Thalamita* are carnivorous species but can also feed on algae and other plant materials. *Portunus pelagicus* species live in both the intertidal zone and in open waters; they were also observed cutting the mangrove propagules in different occasions (Table 4.6).

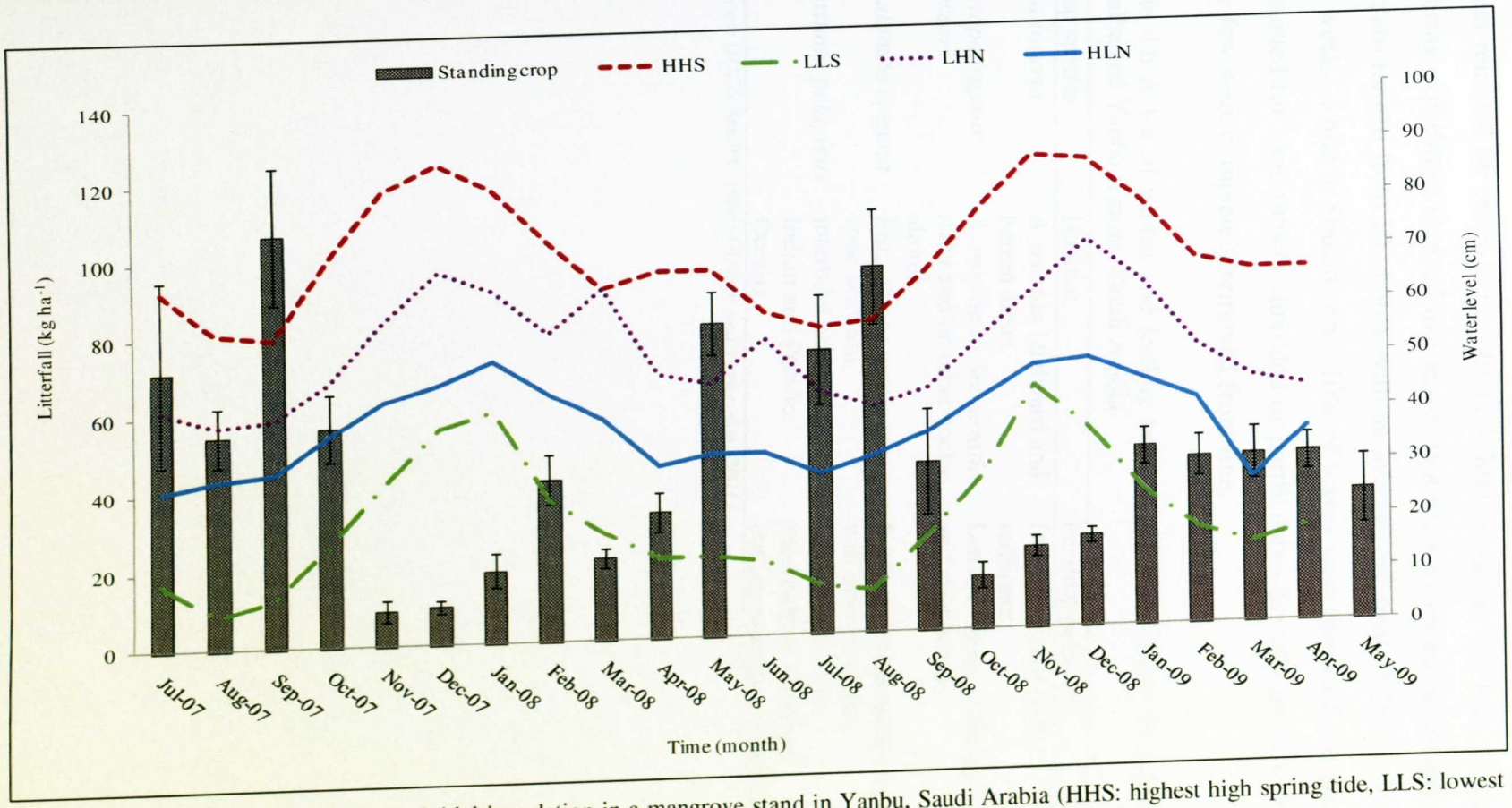

Figure 4.10 Monthly litter standing crop and tidal inundation in a mangrove stand in Yanbu, Saudi Arabia (HHS: highest high spring tide, LLS: lowest low spring tide, LHN: lowest high neap tide, HLN: highest low neap tide, error bards are standard errors).

A further investigation of the crabs feeding behaviour using $\delta^{13}C$ stable isotope can be found in Chapter 6. The leaf tethering experiment estimated the rate of leaf removal by crabs. Generally crab litter removal was low. In Yanbu, the leaf tethering experiment showed that leaves that were partly eaten, chopped or removed by crabs ranged from 10 to 50% with an average removal of 28.7% over a period of two weeks; while in Shuaiba only 10% of leaves were removed, the leaves that were accounted for were mostly shredded or partly eaten but still intact to the twine line, very few were completely removed from line.

Table 4.6 A list of habitat and feeding behavior of crab species in mangrove stands in Shuaiba and Yanbu regions, Saudi Arabia.

Crap species	Habitiat	Feeding behavior	Site found
Uca inversa	*A. marina* landward and barren areas.	Detritus sorted from sediment.	Yanbu
Metopograpsus messor	*A. marina* or *Sonneratia Alba* and on other rocky shores.	Leaves, algae, molluscs and crustaceans.	Shuaiba/ Yanbu
Thalamita crenata	*Rhizophora mucronata* zone seaward.	Primarily carnivorous, but will also eat algae.	Shuaiba
Portunus pelagicus	intertidal estuaries of the Indian and Pacific Oceans.	Crustaceans, fish, macroalgae, observed cutting mangrove seedling.	Shuaiba/ Yanbu

Sources: (IUCN/MEPA, 1986; Gillikin and Verheyden, 2001)

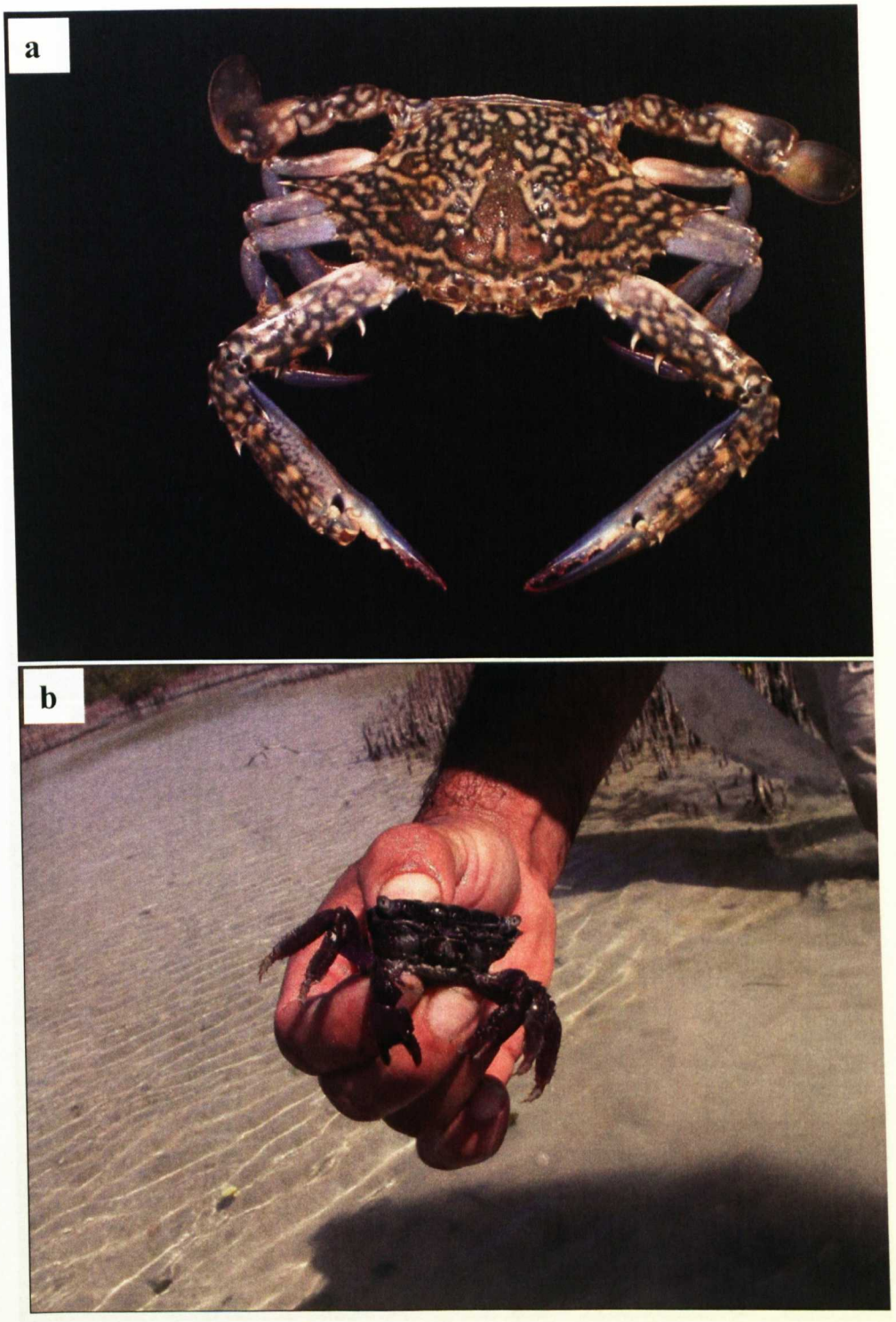

Plate 4.4 Mangrove crabs (a) *Portunus pelagicus,* (b) *Metopograpsus messor* in mangrove stands in Shuaiba and Yanbu, Saudi Arabia.

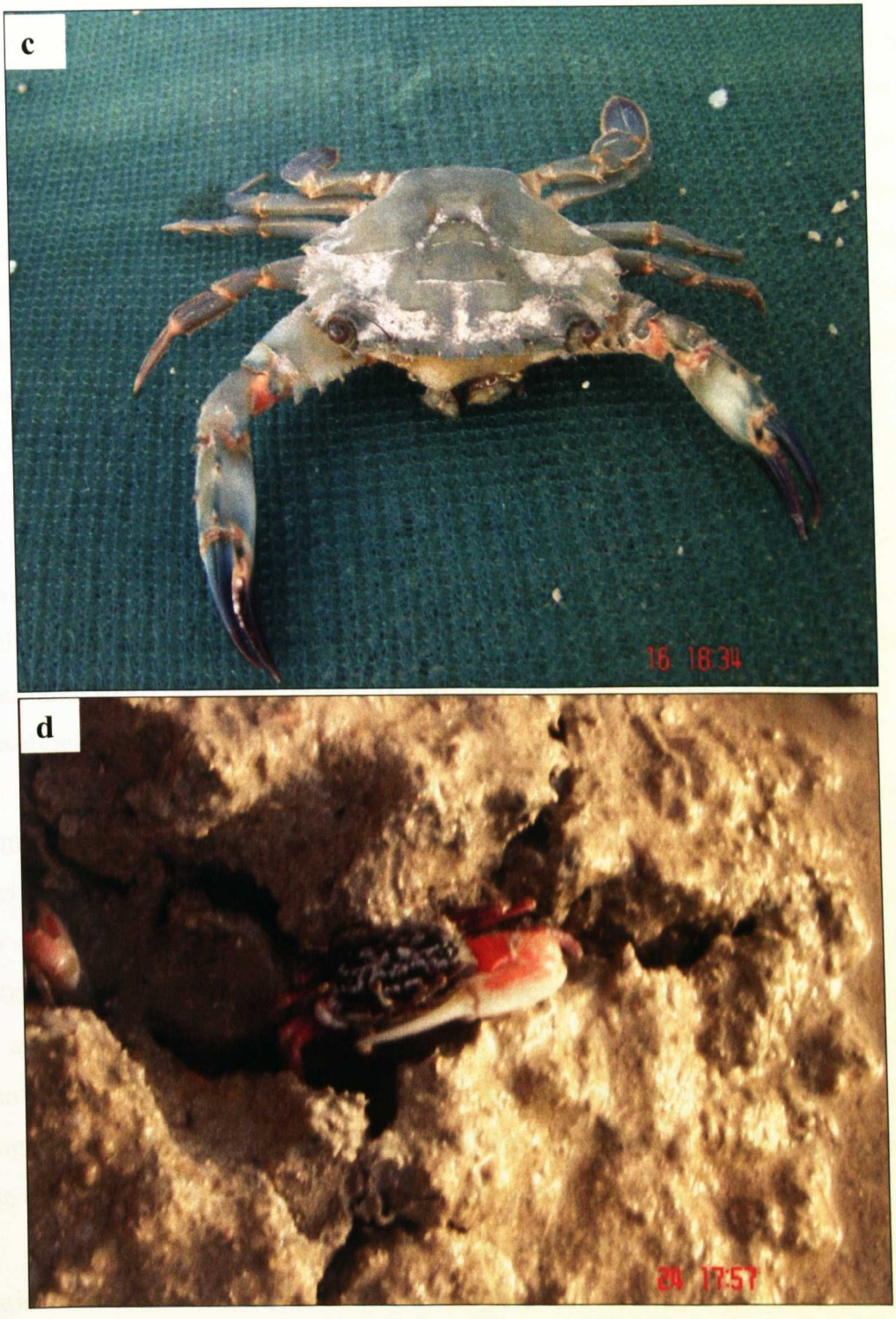

Plate 4.5 Mangrove crabs (c) *Thalamita crenata* and (d) *Uca inversa* in mangrove stands in Shuaiba and Yanbu, Saudi Arabia.

4.4 DISCUSSION

Generally, litterfall rates were always higher in summer than in winter, this seasonal variation was clearer in Shuaiba than in Yanbu. Seasonality in litterfall rates is common and frequently reported in the litterature of mangrove litterfall on the Saudi Arabian Red Sea coast (*e.g.* Saifullah *et al.*, 1989; Mufti, 1990; Khafaji, *et al.*, 1991; Mandura, 1998) and in other mangrove systems around the world (*e.g.* Heald, 1971; Twilley *et al.*, 1986; Day *et al.*, 1987, and Schories *et al.*, 2003). Seasonality in litterfall is attributed to a number of factors including precipitation, relative humidity, wind activity, freshwater discharge, salinity, frequency of tidal flushing and air temperature (Lugo and Snedaker, 1974; Pool *et al.*, 1975; Sasekumar and Loi, 1983). Summer is characterised by high temperature, high solar radiation and by the lowest tidal amplitude. These factors can increase drought and soil salinity; In addition, low frequency of tidal flushing will minimize root oxygenation as well as washing excess salt and toxic sulphide from the top soil. In response to such stressful condition, the trees tend to spend extra energy to maintain the green photosynthetic leaves and shed senescent leaves (Lugo and Snedaker, 1974; Amarasinghe and Balasubramaniam, 1992).

With its high salinity levels (up to 41‰), the Red Sea is considered one of the most saline water bodies in the world (Edwards and Head, 1987). This has a direct effect on the biota in intertidal and subtidal regions, and when accompanied by high evaporation and low tidal inundation in summer can result in hypersalinity (up to 300‰) in coastal lagoons and low water interchange areas. Low water inundation can affect inshore oxygenation and results in complete oxygen depletion during summer (Edwards and Head, 1987). It would be of interest to examine the seasonal changes in soil salinity and redox potential levels in relation to litterfall patterns as these can present a greater seasonal influence than climatic factors.

Litterfall values of the current study were less than estimates of similar locations on the Red Sea (Table 4.7). In central Red Sea, both Shuaiba and Yanbu sites had litterfall estimates that were 34% less than those reported by Mandura (1998) of 5.44 t ha^{-1} in Jeddah City and 58% less than those reported by Saifullah (1989) of 8.34 t ha^{-1} in "Ras Hatba" region. The reported high estimates of the southern Red Sea are not surprising, mangroves of the region are well developed, growing under much favourable climates. Compared to the central and northern

regions, mangroves of the southern region receive sufficient nutrient and freshwater via higher tidal ranges carrying nutrient rich water and through terrestrial runoffs, thus productivity is higher (IUCN/MEPA, 1986; Sheppard *et al.*, 1992). The current litterfall estimate can be considered robust in comparison to other studies on the Red Sea, where the replicated traps used were smaller; approximately 15 traps with sizes ranging from 0.25 to 0.5 cm^2 (*e.g.* Mandura 1998, Khafaji *et al.*, 1991, and Saifullah, 1989). Moreover, in the current study, litterfall was collected over a 2 year period in comparison with 1 year in the previous studies. In addition, considering the time span between the previous and current estimates (10 to 19 years), the obtained results can indicate an overall production deterioration of the mangroves; this can be due to several factors including environmental factors such as changes in overall temperature, salinity levels, low rainfall and anthropogenic factors such as exploitation (*e.g.* fuel wood, animal fodder and over grazing), urbanization, and pollution. Thus a further long term productivity monitoring program that incorporates the effect of climate, the physical and chemical qualities of substrate and water on production (litterfall and biomass) is necessary to gain an accurate figure of the long term productivity status.

Table 4.7 Red Sea litterfall production of *A. marina* mangroves (t ha^{-1} y^{-1}) from various sources.

Source	Litterfall t ha^{-1} y^{-1}	Site	Location
Current study	3.64	Shuaiba	central Red Sea
	3.51	Yanbu	northern Red Sea
Mandura (1998)	5.44	Jeddah	central Red Sea
Khafaji *et al.* (1991)	8.62	Jizan	southern Red Sea
Saifullah (1989)	8.34	Ras Hatba	central Red Sea

Comparing the litterfall of the current study with estimates world wide showed that Red Sea mangroves were similar to other extreme environments (Table 4.8). The litterfall rate of the current study was 3.57 t ha^{-1} higher than the estimate obtained by Clough and Attiwill (1975) of 1.6 t ha^{-1} where mangroves grow close to their southern most growth limits. Moreover, the current estimate was close to those reported in Sri Lanka by Amarasinghe and Balasubraminiam (1992) of 3.6 and those reported by Clough and Attiwill (1982) of 2.0 t ha^{-1}. However, it was lower than those from the temperate regions. The highest estimates came from the tropics (Woodroffe *et al.*,1988) of 12.5 and (Sasekumar and Loi 1983) of 15.4 t ha^{-1} where

mangroves flourish. In addition, the low stand development evident in density and biomass production (Chapter 3) reflected the low litterfall production of the mangroves in the current study.

Table 4.8 Comparisons of annual litterfall production of *A. marina* at different systems around the world.

Source	Litterfall ($t\ ha^{-1}\ y^{-1}$)	Environmental condition	Country
Current study	3.57	Arid	Saudi Arabia
Amarasinghe and Balasubraminiam (1992)	3.64	Dry	Sri Lanka
Woodroffe *et al.*(1988)	12.5	Monsoonal	N. Australia
Steinke and Charles (1984)	7.10	Temperate	South Africa
Sasekumar and Loi (1983)	15.4	Tropical	Malysia
Woodroffe (1982)	5.90	Temperate	New Zealand
Clough and Attiwill (1982)	2.00	-	Australia
Goulter and Attaway (1979)	5.80	Temperate	South Auatralia
Clough and Attiwill (1975)	1.60	Temperate	South .Australia

Litter standing stock measurements in mangrove forests often encounter difficulties in sampling litter from ground plots. Upon falling on ground, leaf litter is often mechanically buried under a thin layer of sediment, this causes difficulties distinguishing them from other materials and therefore preventing accurate sampling. Thus, the standing stock data is likely to be under estimated (Robertson and Daniel, 1989; Schories *et al.*, 2003) there was high variability which masked monthly differences in standing crop litter. Variation among traps is also influenced by non-uniform tidal inundation in which litter is unevenly deposited on the forest floor by high tide forming clusters of litter (Robertson and Daniel, 1989). Although tidal activities removed litter from the forest floor, it is not expected that much of the litter would be exported from the mangroves owing to the low tidal ranges. The highest high tide reached only 63 and 88 cm for Shuaiba and Yanbu respectively. Litter was observed moving within the mangrove forest during high tides however, it was not observed moving outside the forest during a tidal cycle. Thus, it appears that tidal activities only move and "relocate" litter on the forest floor rather than export outside the stand.

Generally the crabs found in the mangrove systems are not mangrove-associated. Crabs can be present in mangrove or any other systems depending on the microclimate preferences/tolerances (Osborne and Smith, 1990; Clarke and Kerrigan, 2002) or food (litter) preference (Ashton, 2002; Hogarth, 2007). Considering the species of crab and their abundance, the crab litter removal is low. In general 19% of tethered leaves were eaten/removed by crabs in both sites; this percentage is considered insignificant comparing to other mangrove systems. Ashton (2002) working on mixed Malaysian mangroves reported a high removal rate of 80% within only one day; an earlier study by Robinson and Daniel (1989) examining crab removal within Australian mixed mangroves (including *A. marina*, *Bruguiera exaristata* and *Ceriops tagal*) found that crabs were responsible for completely removing all tethered leaves in *Bruguiera exaristata* and *Ceriops tagal* stands over a period of 2 weeks (100% removal). The same study however has reported *A. marina* leaf removal of only 33% over the same period much lower than those of the other 2 species.

The findings of the current study suggest that crab removal might be insignificant; this can be attributed to the low abundance and species association to the mangrove system. Most of the removal studies reported key litter-removing species of the sesarmid group (Robertson, 1986; Ashton, 2002; Skov and Hartnoll, 2002; Thongtham *et al.*, 2008), and grapsid group (Werry and Lee, 2005; Imgraben and Dittmann, 2008) which were frequently present within the mangroves and considered to be significant leaf-removing species. However, most of the species found within the current study were less frequent and are of less significance in removing litter such as ocypodid crabs. In fact such crabs were responsible for reducing litter removal rate in some Australian *A. marina* mangrove stands compared to stands with dominated sasarmid population (Robinson and Daniel, 1989). Moreover, the low nutritional value of the *A. marina* leaves would also reduce crab consumption/removal to the minimum. *A. marina* was found to contain high C:N ratio (52:1 at minimum, Chapter 5) which does not meet the nutritional need for the crabs (Fell *et al.*, 1984; Skov and Horton, 2002). In addition, increased lignin content of the senescent leaves even after a period of decomposition (34% at minimum, Chapter 5) would also make them less palatable.

4.5 CONCLUSIONS

With respect to the global estimates of annual litterfall production for mangrove ecosystems, production of the Red Sea *A. marina* mangroves are considered at the lowest end of global estimates. The poor development of the Red Sea mangroves is evident in the low density and biomass estimates compared to the global estimates which reflect the low production of litterfall. A reduction in the annual litterfall production compared to previous estimates may indicate the level of deterioration of the mangroves (at least in the central Red Sea region) and indicate the need for management and conservation programs. Litterfall production is assumed to be even less in the northern Red Sea where mangroves are scarce and less developed. The well developed southern mangroves are assumed to have a much higher litterfall production than those in any other parts of the Red Sea. Reliable estimates of mangrove litterfall production in these regions are needed in order to have an accurate value for litterfall production of the Red Sea mangroves overall. Furthermore, the data obtained from the current study (density and living biomasses, Chapter 3; litterfall production, current chapter) can aid in monitoring biomass and litterfall production in the long run. Moreover, the current litterfall estimate would be valuable in updating the regional litterfall estimates which therefore would enhance global estimations.

The crabs of the mangroves in the current study apparently do not play a major role in leaf processing and decomposition as indicated by abundance observations and tethering experiments. Although small in tidal amplitude, it seems that tide is the major factor removing litter from the mangrove forest floor; owing to the range and duration of the tide. Litter however does not seem to be exported outside the system. Thus, it is likely that the current mangrove systems function in a similar way to the mangroves of the New World (*i.e* Florida and the Caribbean) where tide is the major factor influencing litter removal and export.

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CHAPTER 5

LITTERFALL DECOMPOSITION AND NUTRIENT CYCLING IN MANGROVE SYSTEMS

5.1 INTRODUCTION

Litterfall decomposition of mangrove trees is a significant function in the mangrove ecosystem. The fallen leaves, via detritus pathways, make a significant contribution to inshore and estuarine productivity providing a wide variety of aquatic animals such as molluscs, crabs and fish with a primary source of nutrients. These primary level consumers in turn support an array of secondary consumers, including small fishes and juvenile predators which, when mature, become third level consumers. Thus, high levels of organic matter inputs to the system correspond with a larger and more diverse array of animals supported by a particular ecosystem. In addition, mangrove litter input is believed to represent one third of the net primary productivity in arid zones (Alongi *et al.*, 2005) a fraction of which may resist full degradation and thus be incorporated as organic matter into mangrove sediments or, in cases of export, to the build up of sediment in adjacent ecosystems.

There are many factors that control the rate of litter decomposition such as the physio-chemical condition of surrounding environment, the decomposer's population size and enzymatic capacity, and the availability of substrate resources to microorganisms "resource quality" (Swift *et al.*, 1979). Litter quality is a term used to describe the rate of mineralization of a leaf substance relative to its content of nutrient; it is used to indicate the extent to which litter materials decompose and release nutrients (Anderson and Swift, 1983; Cadisch and Giller, 1997). The chemical composition of litter is generally considered the most significant aspect of litter quality and the most representative (Benner *et al.*, 1986; Cadisch and Giller, 1997). Leaf decomposition goes through different phases in which the chemical composition of litter changes, different types of decomposing microorganisms (primarily bacteria and fungi) selectively break down the different litter components affecting the rate of decomposition and thus the final organic matter input. The C:N

ratio is regarded a good indicator for litter quality and decomposition (Fell *et al.*, 1984); low C:N ratio litter is likely to mineralize C and release N for uptake and thus, such litter is regarded high quality. In addition, litter chemical composition (*i.e* soluble organic matter, hemicelluloses, cellulose and lignin) is an important regulator of decomposition and the relative chemical concentrations over the decomposition process can indicate the overall mass; litter substrate containing high soluble compounds tends to have a higher initial decomposition rate via leaching of soluble carbohydrates. On the other hand, substrates rich in resistant compounds such as lignin tend to have a slower decomposition rate (Dickinson and Pugh, 1974; Swift *et al.*, 1979; Cadisch and Giller, 1997). Although extensive work has been done on mangrove litter decomposition in tropical and subtropical regions, there has been little research on *A. marina* litter decomposition and nutrient cycling in arid and semiarid regions. It was hypothesized that in such an environment with minimal nutrient availability and no inwelling, mangrove trees conserve and recycle nutrients within the ecosystem. Therefore, estimating leaf litter quality change and nutrient content during decomposition phases is crucially important for understanding nutrient cycling within the *A. marina* ecosystems in their region.

The objective of this chapter was to estimate the rate of leaf litter decomposition, the changes in leaf chemical composition including soluble carbohydrates, hemicelluloses, cellulose, lignin and the C and N content of the litter at different decomposition phases.

The hypotheses of this study are:

1. There are no differences in decay rate between Shuaiba and Yanbu sites.
2. *A. marina* leaf litter has low litter quality owing to high lignin content and C:N ratios.
3. Mass loss can be predicted by means of C:N ratio and the concentration of resistant component (lignin) relative to the hemicelluloses and cellulose.
4. As a conservation mechanism, *A. marina* translocate N in fresh leaves before senescence.

5.2 MATERIALS AND METHODS

5.2.1 Litter decomposition estimation

The litter decomposition experiment was initiated in the summer of 2007 using the litterbag technique. In each site, senescent leaves (about to fall from the trees or those just fallen) were handpicked and air dried in the field for a period of four hours; 30 g of the air dried litter were filled into 25 cm^2 mesh bags with 1 mm mesh pore (wide enough to allow entry of small invertebrates, fungi and bacteria). The bags were sealed with staples and tethered along a nylon twine attached to aerial roots to prevent loss and secured into plots (Figure 5.1, 5.2) Litter bags were sampled at nine sampling intervals on days 1, 2, 4, 8, 16, 32, 64, 128 and 256. After placement, a total of 324 and 108 bags were used for the entire study in Shuaiba and Yanbu sites respectively; in Shuaiba, three bags were randomly sampled from each plot at each sampling interval, where in Yanbu, only one bag was sampled from each plot at each sampling interval. The initial litter dry weight (~26 g) was determined after calculating the moisture content of the senescent leaves.

Figure 5.1 25 cm^2 litterbags used for decomposition experiments in mangrove stands on the Red Sea coast, Saudi Arabia.

Figure 5.2 Tethered litterbags secured into sampling plots in mangrove stands on the Red Sea coast, Saudi Arabia.

The sampled bags were externally cleaned of sediment and other materials, labelled as to location, sealed in a large plastic bag and transferred to the laboratory for analysis. In the laboratory, bags were gently cleaned from sedimentation and fine particles under tap water, oven dried at 70°C for 24 hours, placed in desiccator to a constant weight and weighed for weight loss determination. Afterwards, leaf samples were ground in a Wiley mill to pass mesh size of 1 mm, and packed in plastic vials for further nutritional and chemical analysis. For comparison, fresh leaves were randomly picked from trees in different plots in both sites and prepared in the same manner as senescent leaves.

5.2.2. Decomposition constant determination

The weight loss of the decomposing litter was expressed using exponential models; generally, the weight loss of decomposing litter is widely expressed by the single exponential model (Jenny *et al.,* 1949). This model assumes that the relative decomposition rate remain constant along the decomposition process (Olson, 1963). The expression of mass loss by a single constant k provided by the model simplifies modelling of organic carbon in soil, providing an easy way to compare with

constants of other data sets. The proportion of original weight loss is described by the equation:

$$y = W_1 e^{-k_1 t}$$

where y equals the dry mass remaining (%) at time t (days) and k equals the decay constant.

A derivation of the single exponential model is the double exponential model, this model assume that litter can be partitioned into two components, an initial labile, rapidly decaying fraction, and a later resistant, more slowly decaying fraction (Wieder and Lang, 1982). This model has been found to fit decomposition data more efficiently than the single model in many studies (*i.e.* O'Connell, 1987; Robertson, 1988; Ashton *et al.*, 1999). The weight loss in this case is described by the equation:

$$y = W_1 e^{-k_1 t} + W_2 e^{-k_1 t}$$

where y is equal to the dry mass remaining (%) at time t (days); k_1 and k_2 are decay constants for the labile and resistant components respectively and W_1 is the proportion of the labile fraction whilst W_2 is the proportion of the resistant plant material. Half-life of leaf decomposition (time required for accumulated litter to lose half of their dry weight) was calculated following Olson (1963) as followed:

$$t_{(0.5)} = 0.693/k$$

where $t_{(0.5)}$ is the time required for the material to lose half its weight, and k is the decay constant.

5.2.3. Nutritional and chemical composition determination

Leaf litter at different decomposition periods was analyzed for its chemical composition (C and N) and chemical compounds (Leachable carbohydrates, hemicelluloses, cellulose and lignin), these analyses were done on subsamples for each decomposition period except for three which were mislaid (Day 16, Shuaiba; Day 256, both sites). Best fit prediction models were used to estimate missing values. Organic N and C content of leaf litter were analyzed using the combustion method (Carlo-Erba® NA 1500 analyzer, DEVIL, USA), where plant substances are converted into combustion products, Subsamples (5 g) of ground, dried leaf litter were encapsulated in tin foil cones, placed in microplates, wrapped in parafilm and

sent for laboratory analysis where they were combusted at 1020°C in the presence of chromium oxide and silvered cobaltic oxide catalysts to produce purified CO_2 and N_2 gases. Leachable substances, hemicelluloses, cellulose and lignin were analyzed following the methodology of Rowland and Roberts (1994) applied by the ANKOM technology method of fibre analysis (ANKOM, 2008a, b and c). Methods used to determine different litter parameters are listed in Table 5.1.

Table 5.1 Method of determination and place of analysis for the different chemical parameters.

Chemical parameter	Method of determination	Place of analysis
Carbon/Nitrogen	flash combustion technique (Carlo-Erba® NA 1500 analyzer, DEVIL, USA).	Department of Biology, Duke University, Durham, USA.
Hemicelluloses, Cellulose and Lignin	Neutral detergent fibre (NDF) using Filter bag technique (ANKOM^{2000}, 2008a).	Bangor University (SENRGY).
Cellulose and Lignin	Acid detergent fibre (ADF) using Filter bag technique (ANKOM^{2000}, 2008b).	Bangor University (SENRGY).
Lignin	Acid detergent lignin in beakers (ANKOM technology, 2008c).	Bangor University (SENRGY).

5.2.4. Neutral detergent fibre (NDF) and Acid detergent fibre (ADF)

Nutrient detergent fibres are the residues remaining after digestion in a detergent solution, the resulting fibre residues are primarily hemicelluloses, cellulose and lignin. Prior to analysis, water in a water bath was heated to 70°C and the ANKOM machine was rinsed three times to wash any crystallized particles caused by previous solutions. Three sealed filter bags (0.5 – 0.54 g) were weighed and used as blanks. Afterwards, 0.45-0.55 g of dried ground leaf samples were weighed in filter bags, sealed with a heat sealer, labelled using a solvent resistant marker, placed in bag suspension trays (up to 24 samples and three blank bags) and loaded into the analyzer machine. At the start of the NDF extraction, 20 g of Na_2SO_3 and 4 ml of alpha-amylase were manually added to digest the soluble (non-fibre) carbohydrates and a further 8 ml of alpha-amylase diluted in 350 ml of distilled water was added during the two following rinses. At the end of the NDF process, filter bags were manually removed, gently pressed to remove excess water, and covered with acetone in a 250 ml beaker for three to five minutes. Afterwards, bags were removed from

acetone and air dried on a wire screen before being oven dried at 102°C for four hours. Filter bags were then placed in a desiccant pouch until constant weight and reweighed on a four decimal point scale. The %NDF was calculated from:

$$\%NDF = \frac{[W3-(W1xC1)]}{W2} x100$$

Where:

W_1=Bag tare weight

W_2=Sample weight

W_3=Dried weight of bag with fibre after extraction process

C_1=Blank bag correction (Final oven dried weight divided by the original blank bag weight).

The acid detergent fibres are the residues remaining in the filter bags after digestion with H_2SO_4 and Cetyl-trimethylammonium bromide (CTAB); the remaining residues are primarily celluloses and lignin. This method follows the same procedure as of the NDF however, no Na_2SO_3 and alpha-amylase is needed and the ADF solution (20 g CTAB to 1 l 1N H_2SO_4 previously standardized) is used. After the completion of the ADF process, the bags were similarly dried and %ADF was calculated using the same formula as for NDF.

5.2.5 Acid detergent lignin (ADL)

After the ADF determination, dried bags were covered with approximately 250 ml of 72% H_2SO_4 in a 3 l beaker and agitated every 30 minutes for a period of three hours. Afterwards, the H_2SO_4 was poured off and bags were rinsed with tap water until pH was neutral. Bags were then rinsed in acetone for about three minutes, air dried on a wire screen and later oven dried at 102°C for 4 hours; bags were then placed in a desiccant pouch until constant weight and reweighed for lignin determination. Sample and blank bags were then ashed in a muffle furnace at 500°C for a period of 4 hours. Afterwards, the ash was cooled and weighed, the blank bags were used to obtain the bag ash correction using the weight loss on ignition. The %ADL was finally calculated from:

$$ADL = \frac{[W4-(W1xC2)]}{W2} x100$$

Where:

W_1=Bag tare weight

W_2= Sample weight

W_4= Weight of organic matter (loss of weight in ignition of bag and fibre residue)

C_2= Ash corrected blank bag (loss of weight on ignition of bag / original blank bag).

5.2.6 Statistical analysis

Statistical analyses were performed using SPSS ver. 14.0 and Sigmaplot ver. 11.0 (SPSS Inc., Chicago). Data were tested for normality with Levene's test for homogeneity of variance; mean differences between sites were compared using the independent sample t-test; mass loss differences over the decomposition period were assessed using analysis of variance (ANOVA) with Tukey's pair-wise comparison test (p = 0.05, SPSS ver. 14). Non-linear regression was used to find the best model fit, goodness of fit (r^2) and significance of fit (p) of the different decomposition models was explored via least squares regression estimate. Akaike Information Criterion (AIC) was further used to aid in best model selection (Burnham and Anderson, 1998). Principle components analysis (PCA) was employed to assess the relationships between the different chemical components and weight loss (SPSS ver. 14).

5.3 RESULTS

5.3.1 Litter decomposition estimate

After the decomposition period (256 days), there were no significant differences in final mass between Shuaiba and Yanbu sites (p >0.05) (Figure 5.3) with both sites having only 7.5 and 11% of mass remaining for Shuaiba and Yanbu respectively. Shuaiba litter lost 52% of its original mass in the first 64 days, significantly greater than that of Yanbu (44%) (p <0.05, Appendix VIII).

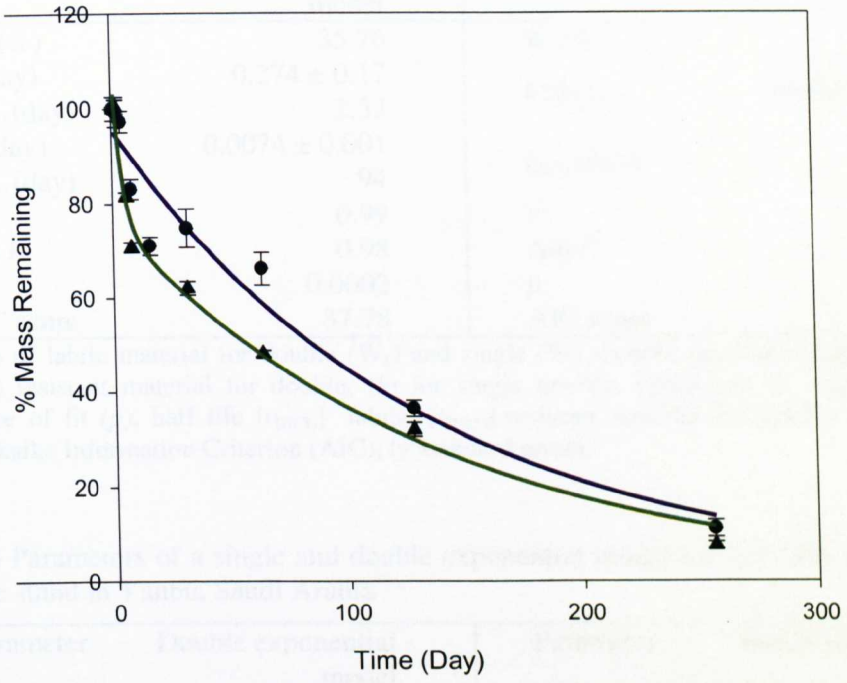

Figure 5.3 Changes of the remaining litter mass over the decomposition period (256 days) in a mangrove stand in Shuaiba (▲) and Yanbu (●), Saudi Arabia (error bars are standard deviations)

For Shuaiba, a double exponential model best fitted the data (r^2 = 0.99). The relatively labile materials constituted a proportion of 35.8% of litter mass with a half life of 2.5 days, the higher proportional mass (resistant materials) had a half life of 94 days (Table 5.2). In Yanbu, both double and single exponential models fitted the data with the same precision (r^2 value of 0.95). However, the standard errors for the double exponential constants were higher which was reflected in the better adjusted r^2 value for the single model (0.94 vs. 0.92 for single and double models respectively). The double exponential model showed that mass loss is consistant with equal decomposition constants for labile and resistant materials and thus a single exponential model was more efficient in describing the data. In addition, the suitability of such a model was further confirmed via the AIC test which gave a smaller (better fit) value for the single model than the double (Table 5.3).

Table 5.2 Parameters of a single and a double exponential model for leaf litter mass loss in a mangrove stand in Shuaiba, Saudi Arabia

Parameter	Double exponential model	Parameter	Single exponential model
W_1 (%)	35.76	W (%)	91.72
k_1(day)	0.274 ± 0.17	k (day)	0.0099 ± 0.002
$t_{1(0.5)}$(day)	2.53		
k_2 (day)	0.0074 ± 0.001	$t_{(0.5)}$(day)	70
$t_{2(0.5)}$(day)	94		
r^2	0.99	r^2	0.94
Adj r^2	0.98	Adj r^2	0.93
p	0.0002	p	<0.0001
AIC score	37.78	AIC score	51.47

Percentage of labile material for double (W_1) and single (W) models, decomposition constants (k_1) labile, (k_2) resistant material for double, (k) for single models, coefficient of determination (r^2), significance of fit (p), half life [$t_{1(0.5)}$] labile, [$t_{2(0.5)}$] resistant material for double, $t_{(0.5)}$ for single models, Akaike Information Criterion (AIC), (± standard error).

Table 5.3 Parameters of a single and double exponential model for leaf litter mass loss in a mangrove stand in Yanbu, Saudi Arabia

Parameter	Double exponential model	Parameter	Single exponential model
W_1 (%)	47.07	W (%)	95.93
k_1(day)	0.0076 ± 61.27	k (day)	0.0076 ± 0.001
k_2 (day)	0.0076 ± 59.10		
$t_{1(0.5)}$(day)	91	$t_{(0.5)}$(day)	91
$t_{2(0.5)}$(day)	91		
r^2	0.95	r^2	0.95
Adj r^2	0.92	Adj r^2	0.94
p	0.0011	p	< 0.0001
AIC score	63.08	AIC score	48.73

Percentage of labile material (W_1), decomposition constants (k_1) labile,(k_2) resistant materials, coefficient of determination (r^2), significance of fit (p), half life [$t_{1(0.5)}$] labile, $t_{2(0.5)}$ resistant materials, Akaike Information Criterion (AIC), (± standard error).

5.3.2 Chemical composition change determination

Although there was a generally increasing trend in C concentration in decomposing leaf litter over the decomposition period (128 days), C concentration appears to increase more at the later phase of decomposition. For Shuaiba, C concentration ranged from 46.24% to 62.32% with significantly higher values at the end phase of decomposition (Day 128, p <0.05) while in Yanbu, C concentration ranged from 46.01% to 56.10% and was significantly increased in day 16 and 128 (p <0.05) (Figure 5.4). The sites did not differ significantly in final C concentration (p >0.05). Similarly, N concentration ranged from 0.48% to 1.21% and from 0.48% to 0.91% for Shuaiba and Yanbu respectively, Values for C and N concentrations over

the decomposition period can be found in Appendix IX and X. N concentration significantly increased at the later phase of decomposition (from day 32 and 16 onwards for Shuaiba and Yanbu sites respectively, p <0.05). Both sites had similar initial concentration. However, by the end of the decomposition period (day 128) Shuaiba had higher N concentrations than Yanbu (p <0.05) (Figure 5.5). Furthermore, when N was determined in fresh and senescent leaves, it was found that both sites always had higher N concentration in fresh leaves than senescent (p < 0.05, Table 5.4), C:N ratios gave values that ranged from 96.64 to 63.44 and from 97.19 to 65.49 for Yanbu and Shuaiba respectively (p <0.05). C:N ratio declined consistently in both sites mainly as a result of the constant increase in N concentration over the decomposition period (Figure 5.6).

Table 5.4 Nitrogen concentrations in fresh and senescent leaves in mangrove stands in Shuaiba and Yanbu regions, Saudi Arabia

Site	Fresh leaves	Senescent leaves	Resorption (%)
Shuaiba	1.19 ± 0.54	0.49 ± 0.01	59
Yanbu	1.34 ± 0.37	0.54 ± 0.05	60

The analysis of litter chemical components showed similar trends between both sites, although initially, Shuaiba had 46% of the soluble carbohydrates lost from litter after one month of decomposition compared to 14% being lost in Yanbu. However, by the end of the decomposition period, the sites did not significantly differ in the final soluble carbohydrates concentration (p >0.05). Hemicelluloses decay appeared constant through the decomposition period with no declining pattern, values ranging from 9.09% to 15.19% in Shuaiba and from 8.44% to 16.64% in Yanbu and predicted to continue with the same pattern until day 256.

Cellulose concentration in Shuaiba was found to decline significantly (from day 8 onwards, p <0.05) contrasting with the increase of lignin concentration. In Yanbu, cellulose was also found to follow a similar trend to lignin, increasing to day 16, decreasing afterwards before increasing again at the end of the decomposition period. Cellulose in Yanbu was found to be significantly higher than Shuaiba at the end of the decomposition period (17.05% vs. 6.29%, p <0.05). The ratio of lignin to lignin and cellulose (lignified cellulose index, LCI) was used to examine the

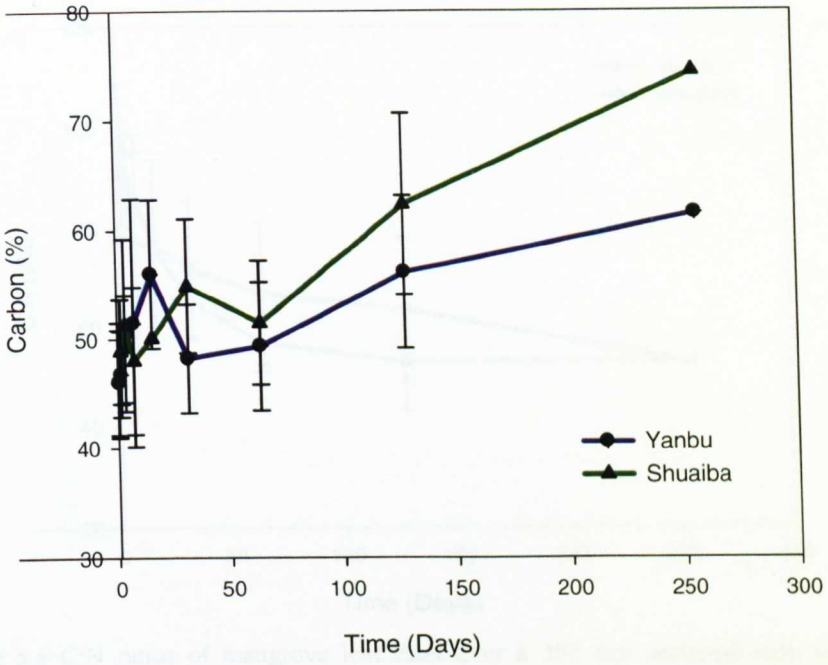

Figure 5.4 Carbon concentrations (%) of mangrove leaf litter over a 256 day decomposition period in mangrove stands in Shuaiba and Yanbu regions, Saudi Arabia (error bars are standard deviations).

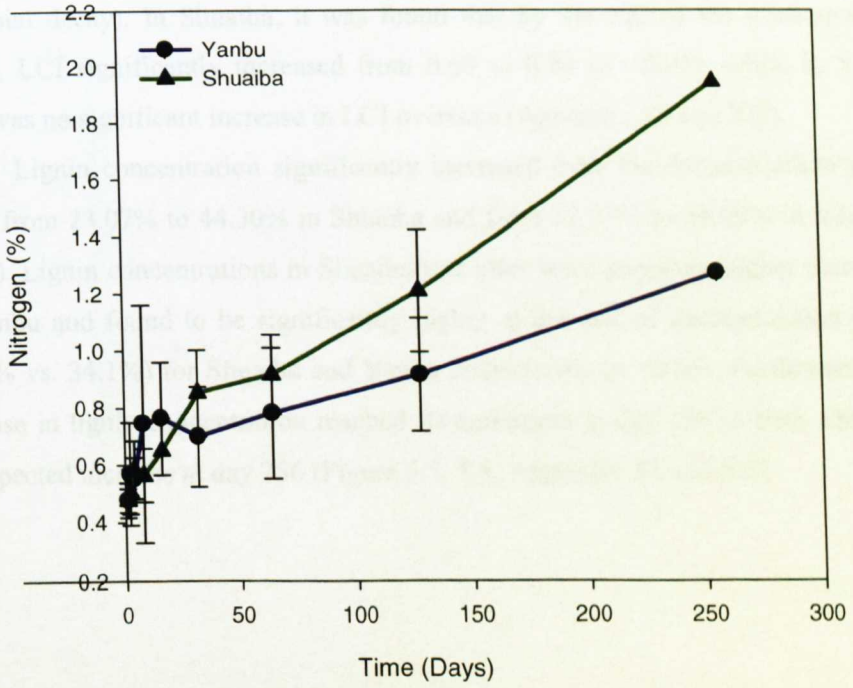

Figure 5.5 Nitrogen concentrations (%) of mangrove leaf litter over a 256 day decomposition period in mangrove stands in Shuaiba and Yanbu regions, Saudi Arabia (error bars are standard deviations).

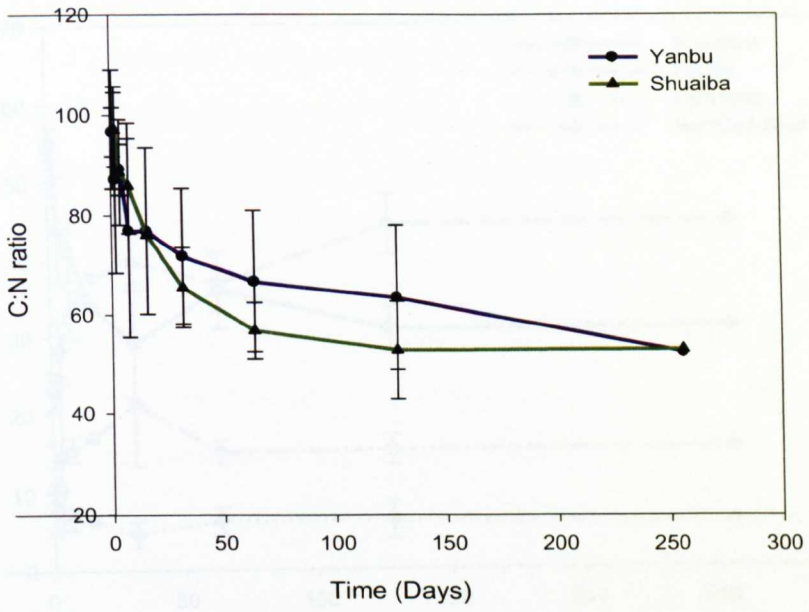

Figure 5.6 C:N ratios of mangrove leaf litter over a 256 day decomposition period in0 mangrove stands in Shuaiba and Yanbu regions, Saudi Arabia (error bars are standard deviations).

relative increase in lignin to cellulose (or the change of litter susceptibility to microbial decay). In Shuaiba, it was found that by the end of the decomposition period, LCI significantly increased from 0.69 to 0.88 (p <0.05), while in Yanbu, there was no significant increase in LCI overtime (Appendix XI and XII).

Lignin concentration significantly increased over the decomposition period rising from 23.07% to 44.30% in Shuaiba and from 18.53% to 34.09% in Yanbu (p <0.05). Lignin concentrations in Shuaiba leaf litter were generally higher than those in Yanbu and found to be significantly higher at the end of decomposition period (44.3% vs. 34.1%) for Shuaiba and Yanbu respectively (p <0.05). Furthermore, the increase in lignin concentration reached its maximum at day 128 in both sites with no expected increase at day 256 (Figure 5.7, 5.8, Appendix XI and XII)

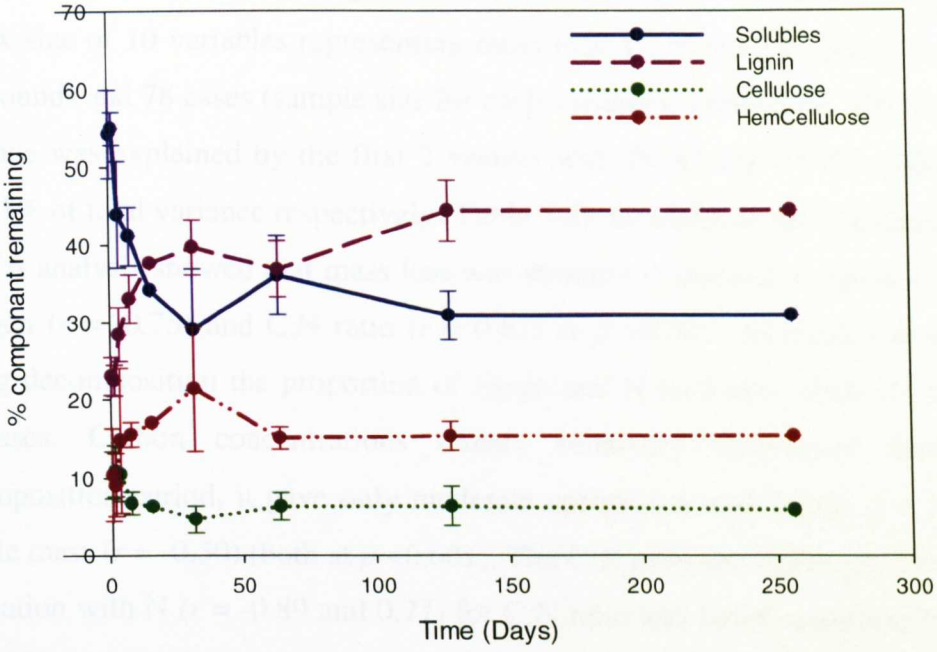

Figure 5.7 Mass remaining of the different chemical components in a mangrove stand in Shuaiba, Saudi Arabia (error bars are standard deviations).

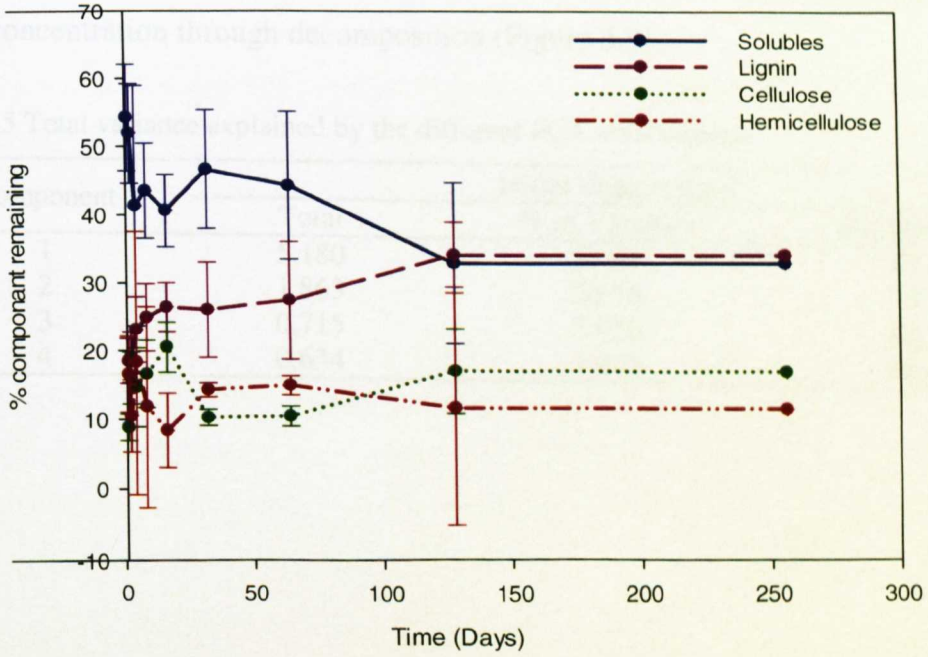

Figure 5.8 Mass remaining of the different chemical components in a mangrove stand in Yanbu, Saudi Arabia (error bars are standard deviations).

Principle component analysis (PCA) was employed to explore the relationships between mass, C and N losses with the different chemical compounds (hemicellulose, cellulose and lignin), C:N, and LCI ratios. The PCA analysis had a matrix size of 10 variables representing mass loss, C, N and the different chemical compounds and 78 cases (sample size for each variable). In Shuaiba, 78% of the total variance was explained by the first 2 vectors with PCA1 and PCA2 explaining 57 and 21% of total variance respectively (Table 5.5). In addition, the correlation matrix of PCA analysis showed that mass loss was strongly correlated to lignin (r = -0.85), nitrogen (r = -0.75) and C:N ratio (r = 0.81) at p <0.001. As mass loss decreased during decomposition the proportion of lignin and N increases while the C:N ratio decreases. Carbon concentrations remain relatively unchanged through the decomposition period, it gave only moderate correlation with lignin (r = 0.54) and soluble mass (r = -0.50) (both at p <0.001). The C:N ratio and lignin gave the highest correlation with N (r = -0.89 and 0.77) for C:N ratio and lignin respectively (both at p <0.001) which is expected due to the steady increase in N immobilization associated with the progressive increase in lignin concentrations. Moreover, the PCA1 showed that degradation of cellulose over the decomposition period is strongly correlated to the increase in the LCI as a result of the steady increase of lignin concentration through decomposition (Figure 5.9).

Table 5.5 Total variance explained by the different PCA components

Component	Initial Eigen values		
	Total	% of Variance	Cumulative %
1	5.180	57.55	57.55
2	1.863	20.70	78.26
3	0.715	7.950	86.21
4	0.634	7.046	93.25

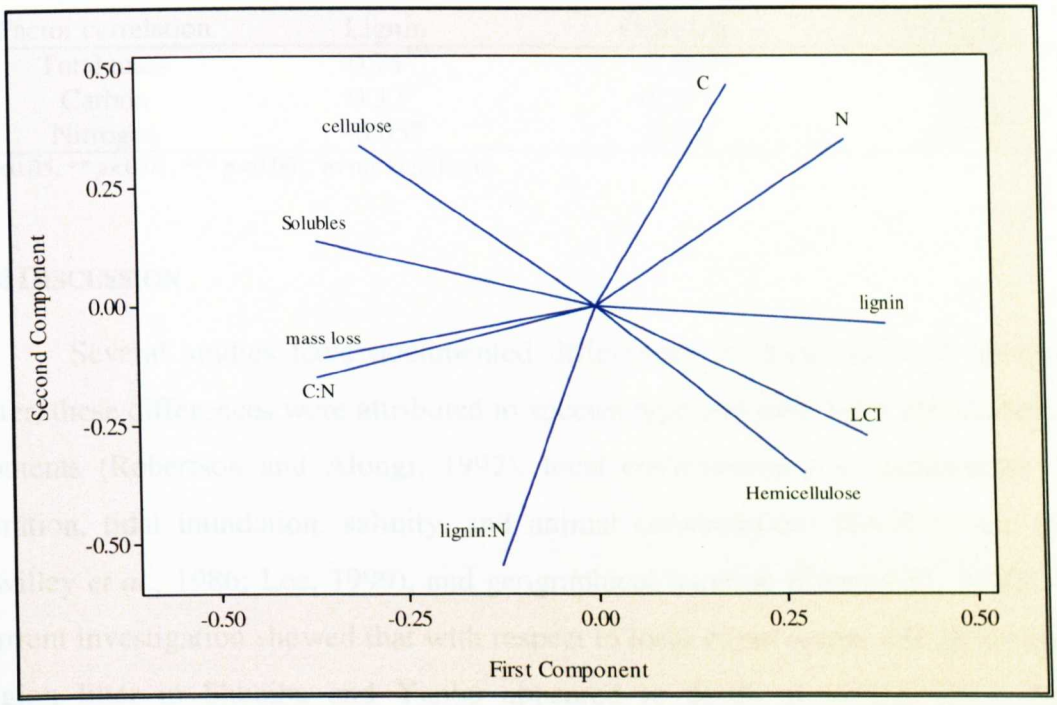

Figure 5.9 PCA analysis of Shuaiba mass, carbon and nitrogen losses with the different chemical compounds.

It was not possible to employ PCA analysis for Yanbu site because data has violated the assumptions of factor analysis, the PCA test assume that the tested data to be suitable for factor analysis (*i.e.* sample size is adequate) and that there are strong correlations between the tested variables. However the KMO test for sampling adequacy for factor analysis showed a value of 0.52 smaller than the required minimal value to meet the assumption (0.60). Nevertheless, the relationships between mass loss and the individual chemical compounds were assessed via partial correlation. Similar to Shuaiba, the total mass loss was highly correlated to lignin increase (r = -0.74, p <0.001), the increase in N concentration over the decomposition period was strongly correlated with the decrease in C:N ratio (r = -0.89, p <0.001) while C did not show a strong correlation with any other chemical compounds, the best correlation was found to be with cellulose (r = 0.36, p <0.01) and lignin (r = 0.32, p <0.01) (Table 5.6).

Table 5.6 Correlation coefficients of total mass, carbon, and nitrogen with lignin, cellulose and C:N ratio, in a mangrove stand in Yanbu, Saudi Arabia

Factor correlation	Lignin	Cellulose	C:N ratio
Total mass	-0.74^{***}	-0.10^{ns}	0.50^{***}
Carbon	0.32^{**}	0.36^{**}	-0.30^{*}
Nitrogen	-0.05^{ns}	0.1^{ns}	-0.89^{***}

*$p<0.05$, ** $p<0.01$, *** $p<0.001$, ns not significant

5.4 DISCUSSION

Several studies have documented differences in decay rates of mangrove litter, these differences were attributed to species type and their litter initial chemical contents (Robertson and Alongi, 1992), local environment (*i.e.* temperature, soil aeration, tidal inundation, salinity, and animal consumption) (Swift *et al.*, 1979; Twilley *et al.*, 1986; Lee, 1999), and geographical location (Tam *et al.*, 1998). The current investigation showed that with respect to local environment and geographical region litter in Shuaiba and Yanbu appeared to decay at similar rates. Direct comparisons of decay need to be treated with caution due to the different model fit but the end result of decomposition and half life was similar in both sites. The sites do not differ significantly in climatic and environmental conditions (Chapter 1) which might contribute to the similarity in decay rates.

A comparison of *A. marina* decay rates from different parts of the world is presented in Table 5.7. It should be noted that single exponential decay rates were used for comparisons although Shuaiba litter decay was best described by a double exponential model. Using the single model facilitates comparison with the published decay constants and half life which are mostly derived from single exponential models. The decay constants of the current study were similar to those obtained in similar environments; in general, decay rates were low in arid and semi arid regions with half life varying from 70 to 91 days. High solar radiation (Austin and Vivanco, 2006), low precipitation (Swift *et al.*, 1979) and low tidal inundation (Mackey and Smail, 1996) are factors that slow decay rates; frequently inundated leaves have higher decay rates than dry or less inundated leaves as moisture promotes leaching and provide a favourable and more stable media for microorganism activity and production (Reice *et al.*, 1984; Robertson and Alongi, 1992; Tam *et al.*, 1990).

Table 5.7 Single exponential decay constants of *A. marina* leaf litter from different geographical locations.

Location (climate)	Half life $t_{(0.5,day)}$	Decay constant (k_{day})	Source
Australia (tropical)	11	0.0630*	Robertson (1988)
China (subtropical)	19	0.0596	Zhou *et al.* (2010)
Africa (warm)	32	0.0213	Steinke and Ward (1987)
Hong Kong (subtropical)	55	0.0126	Tam *et al.* (1990)
South Africa (cold)	58	0.0120	Steinke and Ward (1987)
Australia (subtropical)	70	0.0109	Mackey and Smail (1996)
Saudi Arabia (arid)	70	0.0099	Current Study
Australia (Mediterranean)	80	0.0087	Van der Valk and Attiwill (1984)
Australia (semi arid)	90	0.0077*	Robertson and Alongi (1992)
Saudi Arabia (arid)	91	0.0076	Current Study

*calculated from half life

Nitrogen is generally a scarce nutrient in mangrove ecosystems. In an arid anaerobic system, such as that of the Red Sea mangroves with minimal nutrient inwelling, the availability of N becomes vital in the decomposition, mineralization and incorporation of organic matter into soil processes. N enrichment in decomposing litter is frequently reported in literature (Rice, 1982; Woitchik *et al.*, 1997; Mfiling *et al.*, 2002 and Zhou *et al.*, 2010). Alongi *et al.*, (1992) working on *Bruguiera gymnorrhiza* and *Kandelia candel* mangroves reported that low initial N levels in *Bruguiera gymnorrhiza* were associated with N immobilization while the N enriched *Kandelia candel* favoured N mineralization. The N immobilization of the decomposing litter might be due to incorporation into microbial biomass, and the production of microbial activities such as phenols, small peptides, and amino acids (Fell and Masters, 1980; Rice 1982; Rice and Hanson, 1984; Camilleri and Ribi, 1986). Working under controlled conditions, Benner *et al.*, (1986) reported microbial assimilation and conversion of mineralized leaf nutrients into microbial biomass to be highly efficient (up to 94%), where N concentration increase was found to be strongly correlated with bacterial density (Werry and Lee, 2005), becoming very high in zones where leaves are frequently inundated by water which provide a favourable and stable media for microbial colonization and growth (Camilleri and Ribi, 1986). In addition, microbial N immobilization can be a conservation strategy in such nutrient poor environment which results in the further decrease of the C:N ratio (Melillo *et al.*, 1989). Furthermore, as suggested by prediction models, it is likely that at the end of the decomposition period microbial immobilization still continues and that N concentration will increase further.

Senescent leaves of *A. marina* had 60% less N concentration compared to the fresh leaves indicating nutrient translocation prior to leaf senescence, this agrees with Woodroffe *et al.*, (1988) reporting N translocation to green leaves of *A. marina* by 60%, and Ocheing and Erftemeijer (2002) who reported translocation of *A. marina* N of up to 68%. Mangrove species are known to be efficient in retaining and recycling nutrients as a conservative strategy (Alongi *et al.*, 1992) and with the minimal nutrient inwelling in the Red Sea, it is believed that this becomes vitally important as retaining and recycling are probably the only source of nutrients in such ecosystem (Edwards and Head, 1987) By retaining nutrients, mangroves are able to use the same unit of nutrient to build new leaves and other plant components (Vitousek, 1982).

Initial rapid mass loss occurred in both Shuaiba and Yanbu litter with final soluble mass of approximately 40%, the initial rapid mass loss was mainly due to leaching of the water soluble carbohydrates (including sugar, proteins, and starches) within litter, rather than mediation by microorganisms. It is estimated that up to 50% of mangrove litter mass is in the form of soluble organic matter that are easily leachable to the aquatic system (Hogarth, 2007). Cellulose decomposition was steady throughout the decomposition period, this could be due to the selective bacterial decomposition of cellulose thus slowing down the decay process. Cellulose is a complex carbohydrate polymer that requires a series of extracellular enzymatic hydrolysis to cleave it into smaller units prior to digestion by glucose-metabolizing fungi and bacteria (Sylvia *et al.*, 1999).

Carbon is an element that is present in both soluble and structural carbohydrates, the relatively unchanged C concentration in Shuaiba could be related to the increase in lignin concentration relative to the loss of soluble carbohydrates. The significant increase of C at the end phase of decomposition corresponds with the maximum increase of lignin concentration. This suggests that C enters humic matter in the form of lignin rather than cellulose. The significant increase in the LCI values from initial 0.69 to final 0.88 supporting this assumption and indicating that lignin is the dominant component at later stages of decomposition, Mfiling *et al.*, (2002) has obtained similar C increases attributed to increased lignin concentrations. In Yanbu, the steady C concentration seems to be balanced between cellulose and lignin, and C entering soil seems to be in both lignin and cellulose forms. This can be related to

the lower N concentration compared to Shuaiba which can affect the decay of cellulose. The C:N ratios of *A. marina* were generally higher than those reported in other environments due mainly to low litter N content and resorption level affected by low nutrient input and by N deficiency. High C:N ratio even after 256 days of decomposition, associated with high lignin concentration, slowed down litter mass loss and affected its nutritional status, thus decreasing palatability to aquatic animals (Chapter 4). Generally, organic matter with C:N ratio below 25 and lignin content below 15% is considered high quality (Palm and Sanchez, 1990). Lignin is found to be the best indicator for mass loss and constitutes the major source of C entering mangrove soils; it is thought that lignin-source C utilization for microbial biomass is low, with most C is released as CO_2 or incorporated into soil humus (Sylvia *et al.*, 1999). Lignin is a large complex polymer that only minimal groups of microorganisms able to degrade. The generally high concentration of lignin in *A. marina* leaves probably prolonged half life decay of the litter as the number of microorganisms capable of decaying lignin is minimal.

5.5 CONCLUSIONS

The litter quality of *A. marina* in the extreme Red Sea environment is low, and results in the low rate of decomposition typical of mangrove systems in similar environments. Changes in C:N ratio and lignin concentration can be indicative of mass loss but such indicators can differ between species and locations. *A. marina* flourishes in the southern region of the Red Sea and coexists with *Rhizophora mucronata* in a favourable local environment (*i.e.* milder temperature, nutrient and fresh water inwelling). In addition, the absence of litter processing crabs (Chapter 4) may have contributed to the slow rate of decomposition in the current study even though previous investigations in the southern region reported high faunal diversity. It would be of great interest to investigate the mass loss and litter quality of *A. marina* in such environmental conditions, and between the coexisting *A. marina* and *R. mucronata* of the southern region of the Red Sea coast.

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CHAPTER 6

CARBON FIXATION IN SOIL AND AQUATIC ANIMALS: AN ISOTOPE STUDY

6.1 INTRODUCTION

The transfer of energy through detritus food webs is an important function for an ecosystem's health and sustainability as it governs the residence of animals and birds, enhancing soil organic matter and the trophic balance in associated ecosystems (Almansi, 1999). In mangrove ecosystems, leaf litter detritus has been widely reported as a significant food source that contributes to the stability and occurrence of marine life (Odum and Head, 1972; Odum, 1983) and as a significant source of organic carbon in mangrove sediment (Lallier-Verges *et al.*, 1998; Bouillion, 2003). Litter production can go four possible fates: 1) entering into the detritus food webs via microbial decomposition providing nutrient and energy source for a collection of meiofauna, 2) processed and consumed by grazing crabs, 3) exported to adjacent waters via tidal activities, 4) accumulated within the mangrove system and incorporated as sediment organic matter (Hogarth, 2007).

Tracing carbon sources in detritus-based aquatic food webs using stable isotope ratios is widely used as an important and precise technique in mangrove ecosystems (Fleming *et al.*, 1990; Moran *et al.*, 1991; Rao *et al.*, 1994; France, 1998; and Muzuka and Shunula, 2006) and in other estuarine systems (Haines, 1977 and 1979; Fry and Sherr, 1984; Gearing, 1988). Stable isotope ratios have been also used for assessing the fate of mangrove detritus (accumulation/export) and in expressing the level of significance of such organic matter to offshore communities (Lee, 1995). Generally, carbon isotopic composition provides information on primary energy sources, whereas nitrogen isotopic composition allows discrimination of trophic levels. Gut content analysis is another technique that has been widely used, especially in early studies, to indicate feeding patterns and habitat utilization by consumers. However, it does not truly reflect the diet sources of an animal since ingesting organic materials does not necessarily equate to assimilation (Zieman

al., 1984; Hogarth, 2007). Moreover, identification and quantification of food materials can be extremely difficult in smaller organisms such as crustaceans due to material processing and grinding prior to ingestion (Dall *et al.*, 1990)

While some previous stable isotopes studies in marine systems showed the importance of mangrove derived C in sediment development and in marine animal assimilation including crabs (*e.g.* Rodelli *et al.*, 1984; Newell *et al.*, 1995; Lee, 1998; Boullion *et al.*, 2003; Nordhaus *et al.*, 2005), others have shown that mangrove detritus may not represent a primary source of C and that other sources are suggested to be of greater significance than mangrove detritus (*e.g.* France, 1998; Bouillion *et al*, 2002; Bouillion *et al.*, 2008; Nerot *et al.*, 2009).

The objectives of this chapter were to investigate the sources of C in the mangrove systems of Shuaiba and Yanbu sites, the contribution of these sources in building sediment organic C and whether mangrove derived C is exported to adjacent sediments. In addition, the importance of these potential sources of energy to aquatic animals was investigated. The $\delta^{13}C$ signature of plant components, including leaves, stem, branches, fruits and roots was used to assess the $\delta^{13}C$ signature of aquatic animals including fish and crab species. In conjunction, $\delta^{15}N$ isotope was used to separate the trophic levels.

The null hypothesis of this investigation was that mangrove derived C was not a primary energy source to the aquatic animals, the alternative hypothesis was that mangrove derived C was of primary or moderate importance as an energy source to the aquatic animals.

6.2 STUDY LIMITATIONS

Determining the sources of organic carbon in the mangrove sediments using $\delta^{13}C$ isotope analysis was one of the objectives of the current study. However, obtaining the results for the sediment organic carbon was not achieved. The processing of sample analyses faced several difficulties and obstacles including sample preparation and instrument availability. The sediment samples contained high inorganic carbon which was not sufficiently removed using traditional carbonate extraction methods although three different extraction methods were used to insure carbonate removal, this has required preparing and resending the samples for analysis over a long period of time. Moreover, instrumental availability was also an

obstacle for sample analysis and resulted in significant delay of obtaining results, the sediment and plant samples were sent to different institutions abroad and within the UK depending on availability. Due to these limitations, only total carbon and C:N ratio of the sediments were presented from the current study.

Sampling the aquatic animals for the isotopic investigation was done during the field work period. While some aquatic animals may show seasonal presence in the mangrove systems, seasonal sampling of the animals entering the mangrove systems was not considered in the current study due to time limitation, availability of transportation and man power. Moreover, these constrains also prevented re-sampling of missing or damaged samples.

6.3 Materials and Methods

6.3.1 Sampling $\delta^{13}C$ sources and analysis

Subsamples from the mangrove components including fresh and senescent leaves, wood, and roots were taken from the original samples used for biomass and nutrient cycling studies (Chapter 3 and 5). Fresh leaves were sampled from random trees within each plot, leaves were randomly collected from different parts of the tree crown, packed into paper bags and transferred to the lab. When in the lab, leaves were oven dried at 70°C for 24 hours, ground to powder using a Wily mill and packed into plastic vials for further analysis. Other possible sources of C were collected from within and around the mangrove plantations including seagrass and sponges, these samples were lab processed in the same manner as the mangrove samples. For the analysis, 25 mg of sample were weighed into tinfoil cones, wrapped and ball shaped, inserted into micro plates and sent for analysis.

Surface sedimentation (top 5 cm) was collected from each plot by random coring using a 1.9 cm diameter cylindrical core. Within each plot, three sediment cores were randomly taken and bulked into one sample per plot. In order to assess the relative fixation of mangrove derived C outside the mangrove plantations, soil samples were collected at distances away from the mangroves corresponding with the tidal direction. Replicated sediment samples were taken at 5, 10, 20 and 40 metres away from the mangroves. A total of 20 sediment samples were collected in each site, samples were collected in plastic bags, labelled as to location and transferred to the lab. When in the lab, large debris and shells were removed and the

remaining sediment dried at 105°C for 24 hrs and ground to fine powder (< 250μm). All samples were acidified with dilute HCL prior to analysis to remove carbonate following the methodology described in Kennedy *et al.*, (2005) and Komada *et al.*, (2008). Soil samples (50 mg) were weighed into silver cups and 400 μL of 2M HCL were added to each sample and then oven dried at 60°C until full dryness; this process was repeated until there was no visual effervescence in two consecutive cycles. Upon completion of the acidification process, samples were completely oven dried at 50°C for a period of three hours. Afterwards, samples were encapsulated into the silver cups and sent for analysis.

As previously noted in Chapter 4, the crabs found within the mangrove system were the Grapsidae crabs *Metopograpsus messor*, Ocypdidae crabs *Uca inversa*, and Portunidae crabs *Thalamita crenata* and *Portunus pelagicus*. Upon collection from the field, individual crabs were separately packed into plastic bags, transferred to the lab and stored in a freezer until further analysis. To assess the dietary C sources used by the different crabs, muscle tissues were taken from claws and abdominal parts of the animal; the extracted tissues per animal were pooled and used as composite samples. Samples were dried, ground to powder and packed in plastic vials. Unfortunately, analyzing samples from *Uca inversa* was not possible due to deterioration of the muscle tissue as a result of improper freezing. Thus, no isotopic results of *Uca inversa* are presented in this study. For the remaining species, three samples per species were used for analysis, with a total of 15 samples used in both sites. Fish species were collected from the mangrove sites using the fishing net described in Chapter 4, fishing nets were placed at the mouth of major water channels at early morning and the catch was collected on the same day. Only Shuaiba site yielded fish species while no fish species were captured in Yanbu site. The captured fish belonged to eight species *Oedalechilus labiosus*, *Chanos chanos* (Plate 6.1a and b), *Lutjanus russellii*, *Lutjanus argentimaculatus* (Plate 6.2a and b), *Diplodus noct*, *Siganus rivulatus* (Plate 6.3a and b), *Sphyraena flavicauda*, and *Rastrelliger kanagurta* (Plate 6.4 a and b). The captured fish were collected from nets, inserted into plastic bags and kept on ice until transferred to the lab. When in the lab, fish samples were identified down to species level, and finally frozen until analysis. Fish muscle samples were taken following the EMERGE Protocol for fish sampling (Rosseland *et al.*, 2003), muscle samples were extracted from the area

above the midline between the dorsal and adipose fin, samples were fully dried in an oven, ground to powder and packed in plastic vials for analysis.

For the isotopic analysis, samples were combusted using the Carlo-Erba® NA 1500 elemental analyzer (DEVIL, USA) described in Chapter 5. $\delta^{13}C$ was determined on a ThermoFinnigan (Bremen, Germany) Delta + XL Isotope Ratio Mass Spectrometer and expressed relative to the international standard (PDB) for C isotope analysis:

$$\delta^{13}C_{sample} = \{(^{13}C/^{12}C \text{ sample}) / (^{13}C/^{12}C \text{ standard}) - 1\} \times 1000$$

The fractional contribution of the different food sources to the crabs were examined by applying multisource mixing model using IsoSource computer software. This model examines all the possible contributions of the available sources to the diet. In theory, the isotopic signature of the fish or crab is a mixture of the isotopic signature of all sources in the diet. Thus, all the source combinations that sum to the observed isotopic mixture are considered feasible solutions (Phillips and Gregg, 2003).

The IsoSource mixing model examines all possible source combinations that sum to 100%, this is provided in a range from 0-100% contribution for each source. However, as the tails of the range distribution are sensitive to small number of observations, the 1-99‰ is used instead which is more robust to outliers (Phillips and Gregg, 2003). The possible combinations are calculated using a specified source increment and a mass balance tolerance level. An increment level of 1‰ was used in the current study; this level is reported a convenient level for examining the partitioning among sources (*e.g.* Benstead *et al.*, 2006; Phillips and Gregg, 2003; Sara` *et al.*, 2003). The mass balance tolerance level was determined taking into account that no significant feasible solution is missed during the computation, this is done by determining a tolerance level of more than half the maximum isotopic differences between the sources following the equation:

Tolerance level = 0.5 × increment level (in decimals) × maximum isotopic differences between sources.

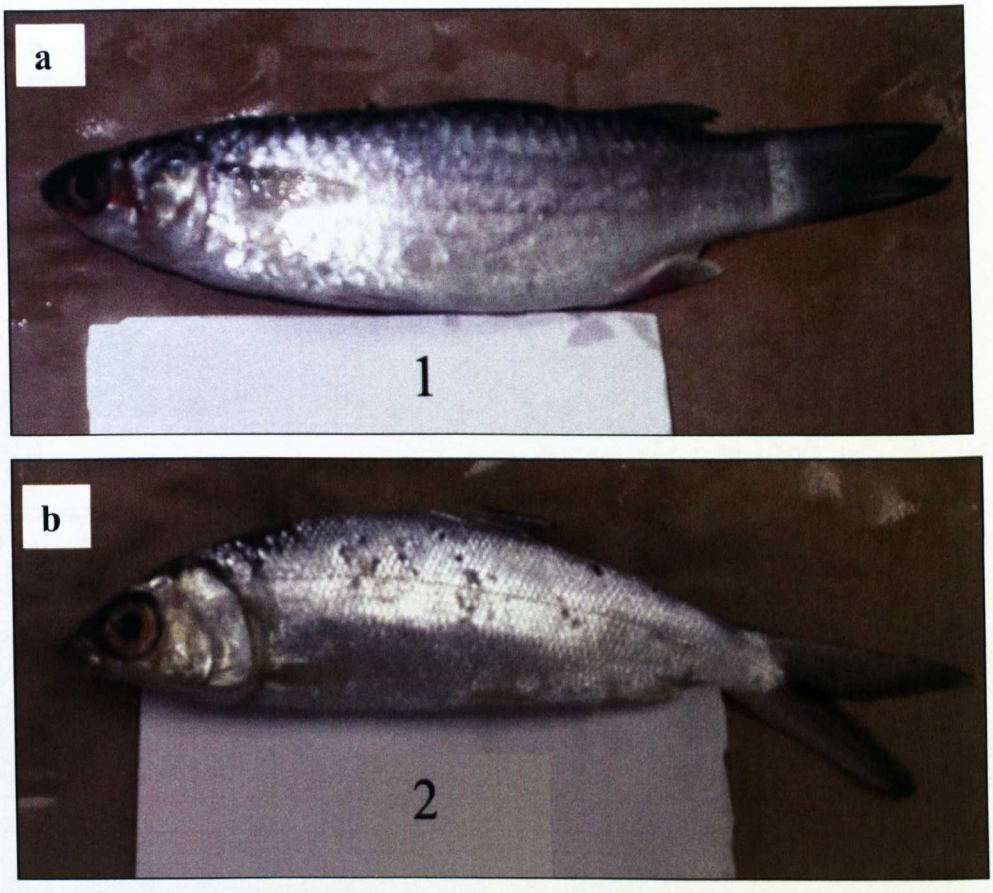

Plate 6.1 Fish species (a) *Oedalechilus labiosus*, (b) *Chanos chanos* in mangrove stands at Shauaiba, Saudi Arabia.

Plate 6.2 Fish species (a) *Lutjanus russellii*, (b) *Lutjanus argentimaculatus* in mangrove stands at Shauaiba, Saudi Arabia.

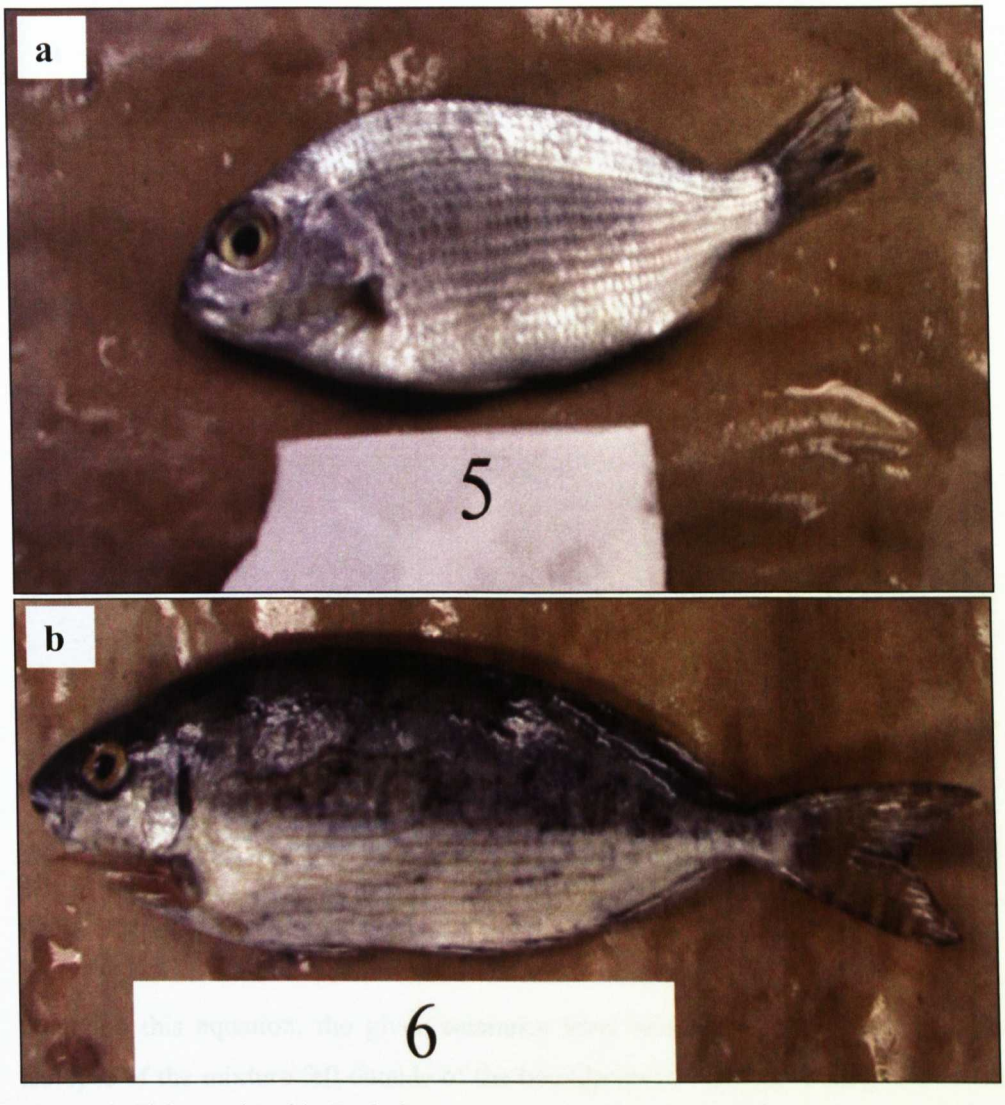

Plate 6.3 Fish species (a) *Diplodus noct*, (b) *Siganus rivulatus* in mangrove stands at Shauaiba, Saudi Arabia.

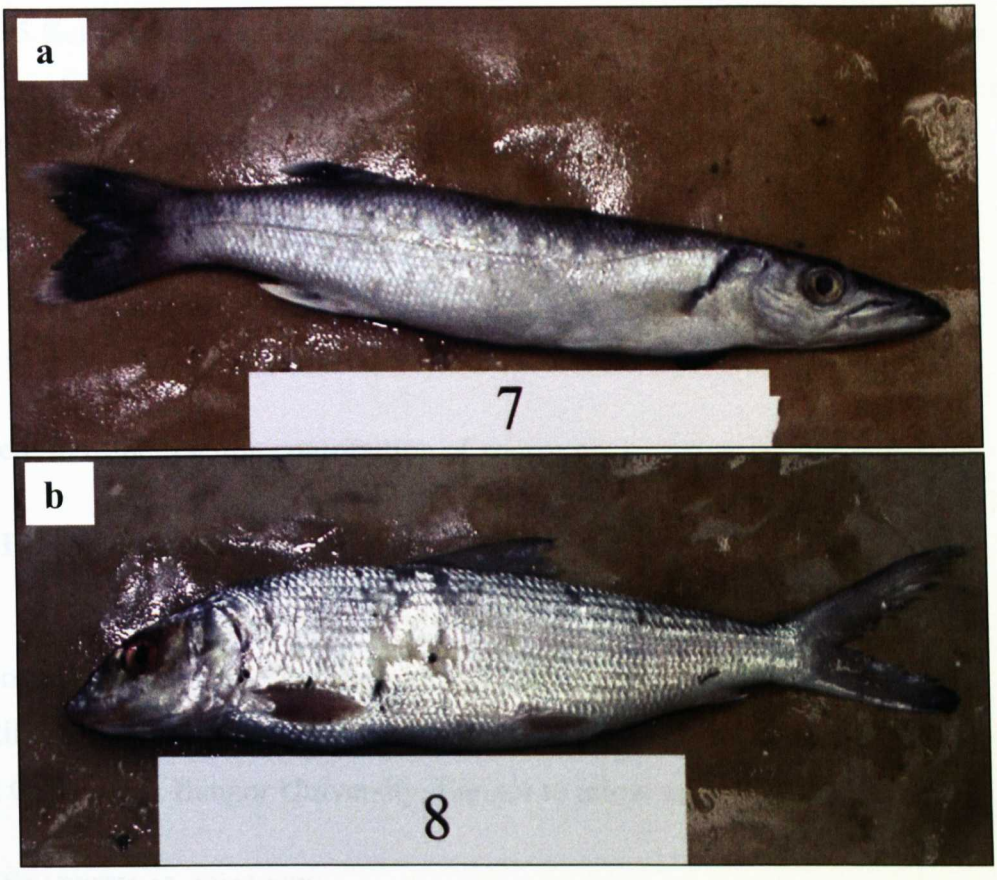

Plate 6.4 Fish species (a) *Sphyraena flavicauda*, (b) *Rastrelliger kanagurta* in mangrove stands at Shauaiba, Saudi Arabia.

Applying this equation, the given tolerance level was 0.1‰. However, when the isotopes of the mixture fell outside of the boundaries of the contributing sources the tolerance level was incrementally increased up to a maximum of 4‰ to allow running of the model by the software.

It should be noted that mixing models often result in undetermined mixtures when many food sources are preset, while measuring multitracers would help resolving such "mixing muddle" (Fry, 2006) therefore, $\delta^{15}N$ was used as a second tracer. The $\delta^{15}N$ is often used to determine the trophic levels (Fry and Sherr 1984; Peterson and Fry 1987). Generally $\delta^{15}N$ is found to be enriched in a consumer's tissue by approximately between 2 and 4 ‰ than in the diet at each trophic level (DeNiro and Epstein 1981; Minagawa and Wada 1984) due to fractionation during digestion and assimilation. A 3.4‰ fractionation level was viewed as appropriate estimate for mussel sample analysis (McCutchan *et al.,* 2003). Thus a 3.4‰

fractionation level was assumed and corrected (subtracted) for consumers prior to analysis in the mixing model. Unlike $\delta^{15}N$, the trophic fractionation of $\delta^{13}C$ is small (< 0.5 ‰) (Peterson and Fry, 1987; McCutchan *et al.*, 2003; Fry, 2006), therefore $\delta^{13}C$ values for the samples were not corrected. The $\delta^{13}C$ and $\delta^{15}N$ fraction contribution of each source was computed following the equations:

$f1\times\delta^{13}C + f2\times\delta^{13}C + f3\times\delta^{13}C + ...fn\times\delta^{13}C$ = observed $\delta^{13}C$ of the sample

and

$f1\times\delta^{15}N + f2\times\delta^{15}N + f3\times\delta^{15}N + ...fn\times\delta^{15}N$ = observed $\delta^{15}N$ of the sample

Where fn is the fraction contribution of source n.

6.4 Ethical consideration

Fish and crab samples were dead prior to muscle tissue sampling by simply exposing them to air. For sample transport to the UK, animal and plant import certificates were obtained from the School of the Environment, Natural Resources and Geography, Bangor University, Bangor to allow sample entry for analysis.

6.5 Statistical analysis

Differences in $\delta^{13}C$ stable isotope values of mangrove leaves including green leaves and senescent leaves at different stages of decomposition were tested using one way analysis of variance (ANOVA) with Tukey's pair-wise comparison test (p = 0.05, SPSS ver. 14). IsoSource mixing model computer software (IsoSource Version 1.3.1) was used to find all feasible solutions of sources contribution in consistence with the isotopic mass.

6.6 Results

6.6.1 Shuaiba $\delta^{13}C$ signatures in plant and animal tissues

In Shuaiba, the various food sources found in the mangroves had a wide range of $\delta^{13}C$ isotopic values. Beside mangrove, the other food sources found within the Shuaiba mangroves were gray sponges, green sponges and seagrass. These sources were not extensive and found growing in small batches, the seagrass were found in low intertidal areas while sponges were growing by the edges of the mangrove stands. Mangrove leaves had a $\delta^{13}C$ signature of -26.4 and -25.8‰ for

green and senescent leaves respectively, and -24.8 and -23.9‰ for aerial and fine roots respectively. No significant differences in $\delta^{13}C$ were found between green and senescent leaves or between aerial and fine roots ($p > 0.05$), The $\delta^{13}C$ signature was -16.9, -25.3 and -5.9‰ for gray sponge, green sponge and seagrass respectively (Table 6.1).

Table 6.1 Stable isotope values ($\delta^{13}C$‰ and $\delta^{15}N$‰ ± SE) of mangrove plant components (leaves, aerial and fine roots), other possible primary producers within the Shuaiba mangrove site

Plant component and primary producer	Sample number (n)	$\delta^{13}C$‰	$\delta^{15}N$‰
A. marina green leaves	7	-26.4 ± 0.26	0.2 ± 0.84
A. marina senescent leaves	12	-25.8 ± 0.08	2.0 ± 0.13
A. marina aerial roots	3	-24.8 ± 0.20	0.1 ± 0.20
A. marina fine roots	4	-23.9 ± 0.90	0.8 ± 0.62
Gray sponges	3	-16.9 ± 0.03	1.0 ± 0.03
Green sponges	3	-25.3 ± 0.00	-2.4 ± 0.03
Seagrass	3	-5.9 ± 0.06	-0.7 ± 0.00

The $\delta^{13}C$ signatures were variable among the aquatic animals. For the crab species, the $\delta^{13}C$ signatures were -21, -11.8 and -13‰ for *Metopograpsus oceanicus*, *Portunus pelagicus*, and *Thalamita crenata* species respectively (Figure 6.1). while it was -12.4, -16.9, -17.5, -18.7, -19.8, -14, -13.3 and -12.2 for *Oedalechilus labiosus*, *Chanos chanos*, *Lutjanus russellii*, *Lutjanus argentimaculatus*, *Diplodus noct*, *Siganus rivulatus*, *Sphyraena flavicauda*, and *Rastrelliger kanagurta* species respectively (Table 6.2). The $\delta^{13}C$ and $\delta^{15}N$ gave a clear separation of the food sources. Although mangrove and green sponge had similar $\delta^{13}C$ signatures, the inclusion of their $\delta^{15}N$ isotope values helped discriminating the two sources in the mixing polygon (Figure 6.1).

Table 6.2 Stable isotope values ($\delta^{13}C$‰ and $\delta^{15}N$‰ ± SE) of fish species (n=3) at Shuaiba mangrove site, Saudi Arabia

Fish species	$\delta^{13}C$‰	$\delta^{15}N$‰
Oedalechilus labiosus	-12.4 ± 0.73	5.3 ± 0.23
Chanos chanos	-16.9 ± 0.42	4.3 ± 0.12
Lutjanus russellii	-17.5 ± 2.65	7.4 ± 0.63
Lutjanus argentimaculatus	-18.7 ± 0.18	7.5 ± 0.09
Diplodus noct	-19.8 ± 0.57	5.7 ± 0.1
Siganus rivulatus	-14 ± 0.52	5.3 ± 0.35
Sphyraena flavicauda	-13.3 ± 0.03	8.1 ± 0.03
Rastrelliger kanagurta	-12.2 ± 0.06	7.5 ± 0.00

Almost all the sampled animals fell outside the mixing polygon suggesting that other food sources were not included in the mixing polygon. Generally, the isotopic signatures of the crab species were closer to the upper border of the mixing polygon than those of the fish species (Figure 6.1). Mangrove detritus ranked first in source contribution for *Metopograpsus oceanicus* with contribution range between 43-74% (at 1-99‰, Figure 6.2), while it ranked third for *Portunus pelagicus*, and *Thalamita crenata* with source contribution between 0-31% and 0-32% for *Portunus* and *Thalamita* species respectively (Figure 6.3 and 6.4). The green sponge was always ranked last for all crab species with a maximum contribution of 6% indicating its negligible importance as a food source.

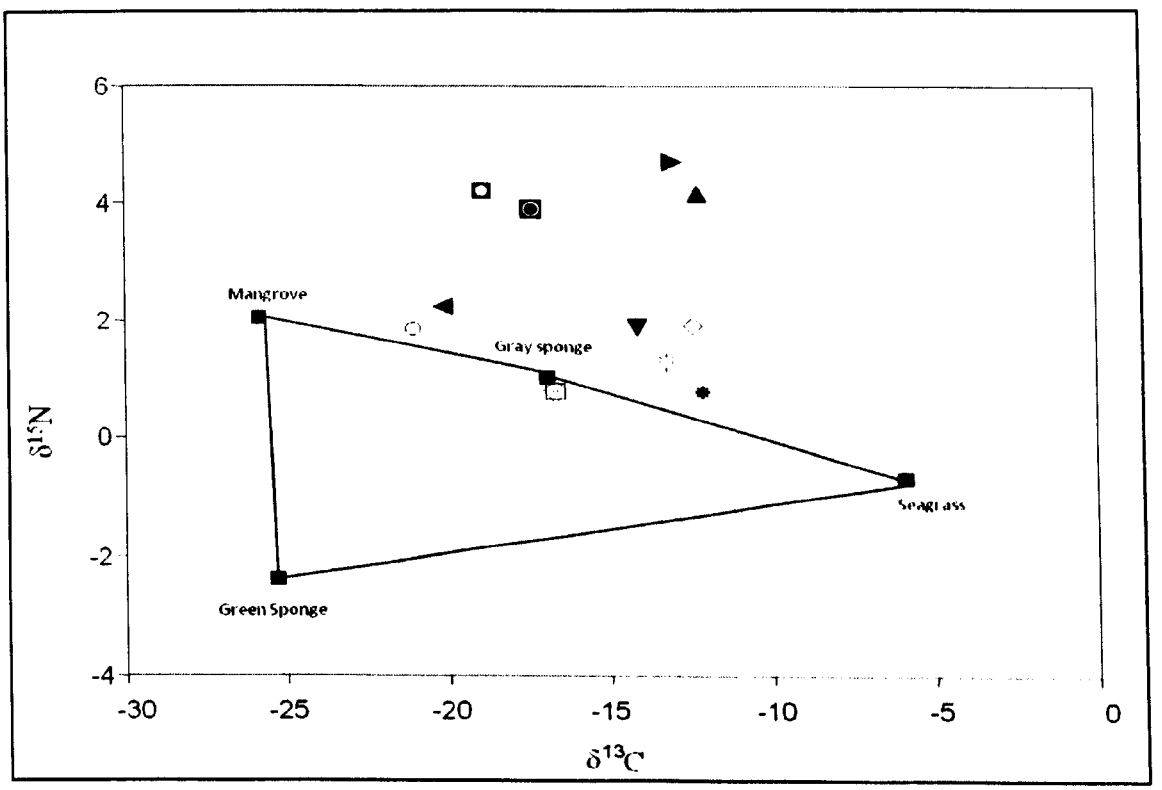

Figure 6.1 Mean $\delta^{13}C$ and $\delta^{15}N$ isotopic signature of food sources and of the crab species (○) *Metopograpsus oceanicus*, (*) *Portunus pelagicus*, (☼) *Thalamita crenata* , and eight fish species (◊) *Oedalechilus labiosus*, (◎) *Chanos chanos*, (■) *Lutjanus russellii*, (◘) *Lutjanus argentimaculatus*, (◄) *Diplodus noct*, (▼) *Siganus rivulatus*, (►) *Sphyraena flavicauda*, (▲) *Rastrelliger kanagurta* in a mangrove system at Shuaiba, Saudi Arabia.

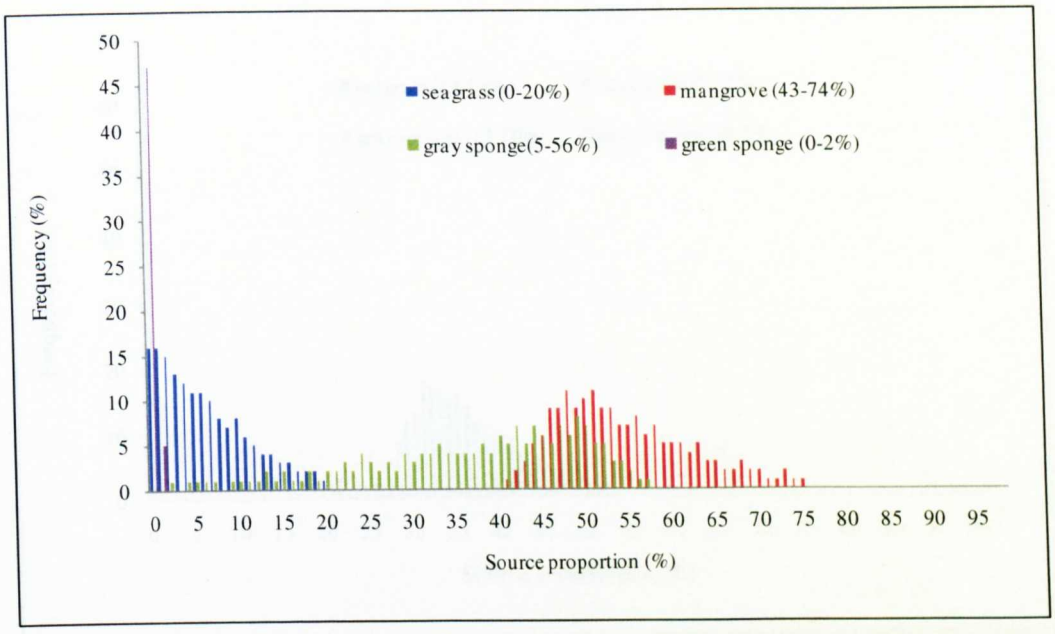

Figure 6.2 The distribution of feasible contributions of food sources to *Metopograpsus oceanicus* crabs in Shuaiba mangroves, Saudi Arabia; values between brackets are the 1-99‰ distribution ranges.

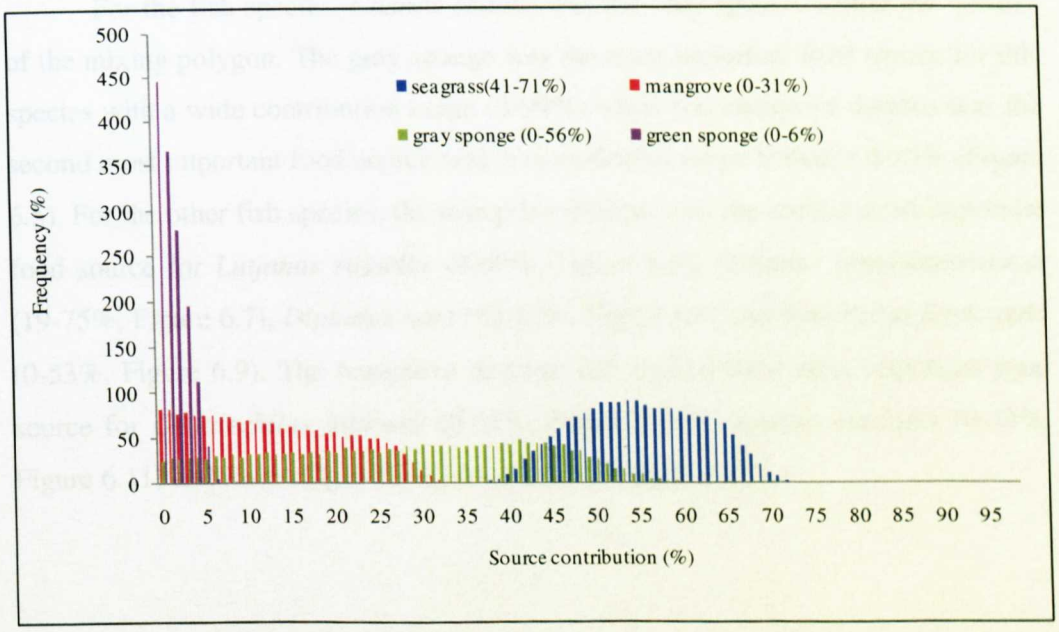

Figure 6.3 The distribution of feasible contributions of food sources to *Portunus pelagicus* crabs in Shuaiba mangroves, Saudi Arabia; values between brackets are the 1-99‰ distribution ranges.

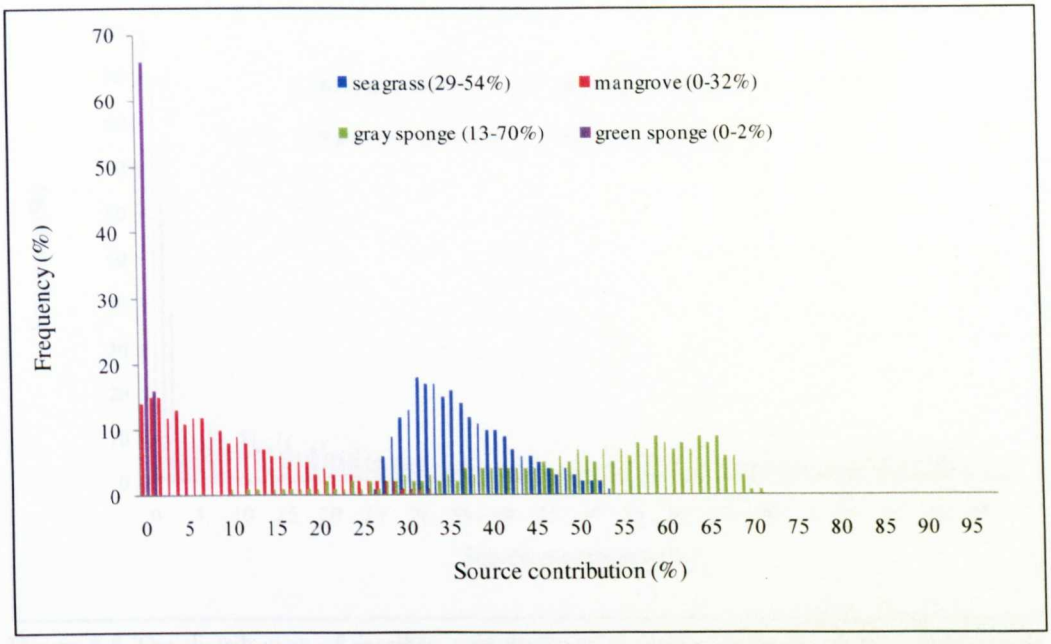

Figure 6.4 The distribution of feasible contributions of food sources to *Thalamita crenata* crabs in Shuaiba mangroves, Saudi Arabia; values between brackets are the 1-99‰ distribution ranges.

For the fish species, *Chanos chanos* was the only species within the borders of the mixing polygon. The gray sponge was the most important food source for this species with a wide contribution range (3-99%) while the mangrove detritus was the second most important food source with a contribution range between 0-53% (Figure 6.5). For the other fish species, the mangrove detritus was the second most important food source for *Lutjanus russellii* (9-69%, Figure 6.6), *Lutjanus argentimaculatus* (19-75%, Figure 6.7), *Diplodus noct* (40-47%, Figure 6.8) and *Sphyraena flavicauda* (0-53%, Figure 6.9). The mangrove detritus also ranked third most important food source for *Oedalechilus labiosus* (0-32%, Figure 6.10), *Siganus rivulatus* (0-44%, Figure 6.11) and *Rastrelliger kanagurta* (0-45%, Figure 6.12).

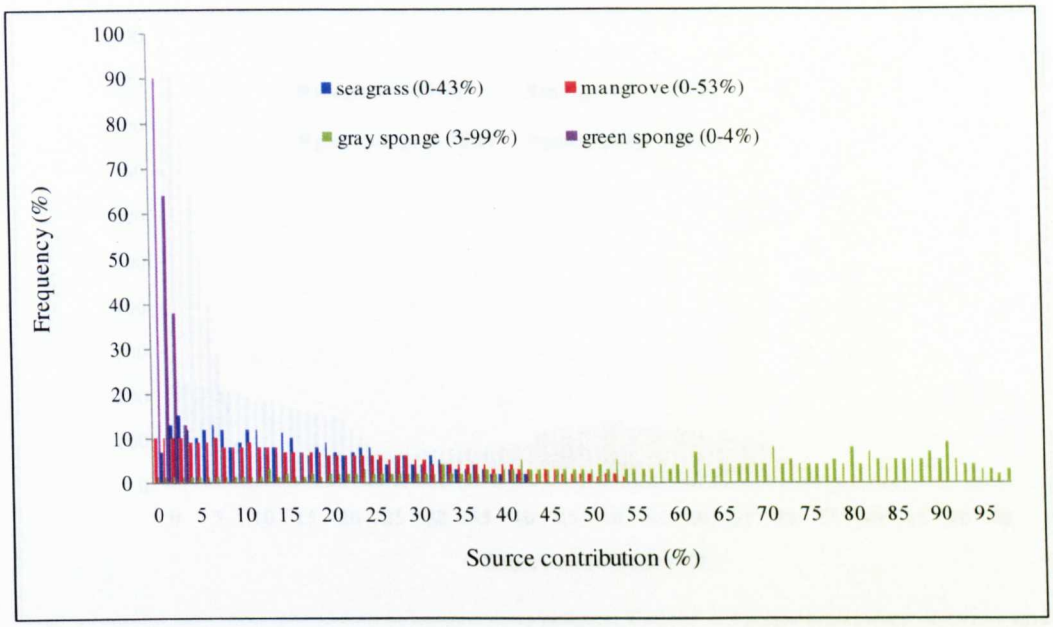

Figure 6.5 The distribution of feasible contributions of food sources to *Chanos chanos* fish in Shuaiba mangroves, Saudi Arabia; values between brackets are the 1-99‰ distribution ranges.

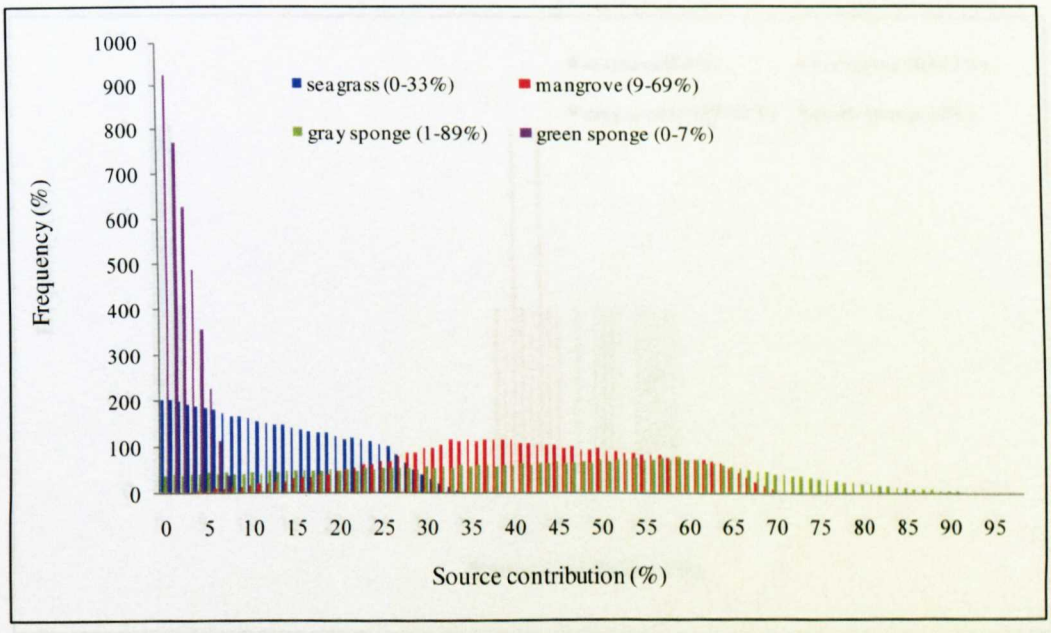

Figure 6.6 The distribution of feasible contributions of food sources to *Lutjanus russellii* fish in Shuaiba mangroves, Saudi Arabia; values between brackets are the 1-99‰ distribution ranges.

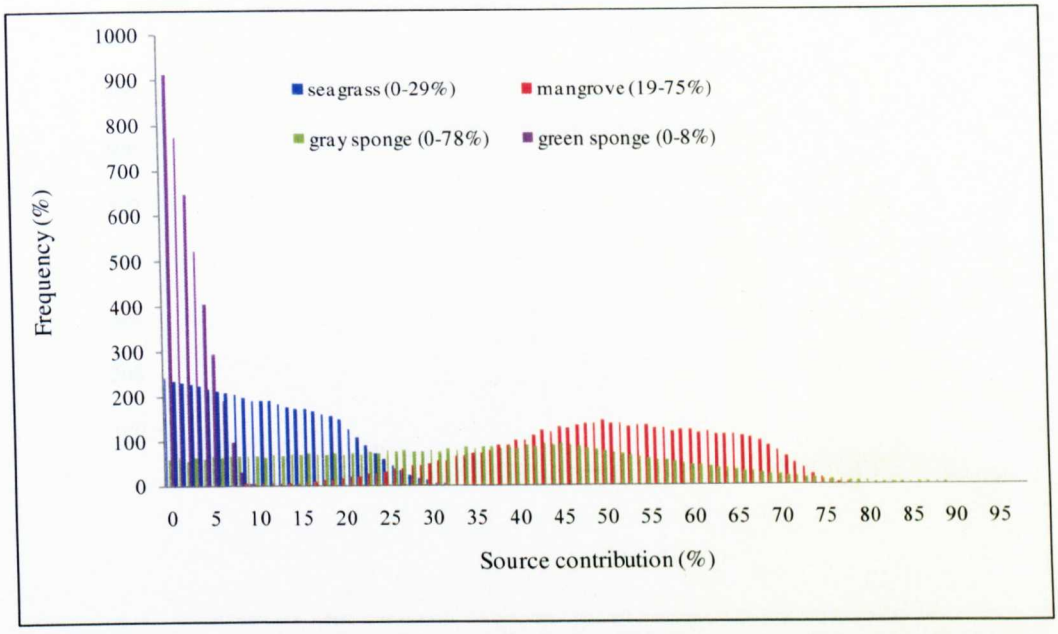

Figure 6.7 The distribution of feasible contributions of food sources to *Lutjanus argentimaculatus* fish in Shuaiba mangroves, Saudi Arabia; values between brackets are the 1-99‰ distribution ranges.

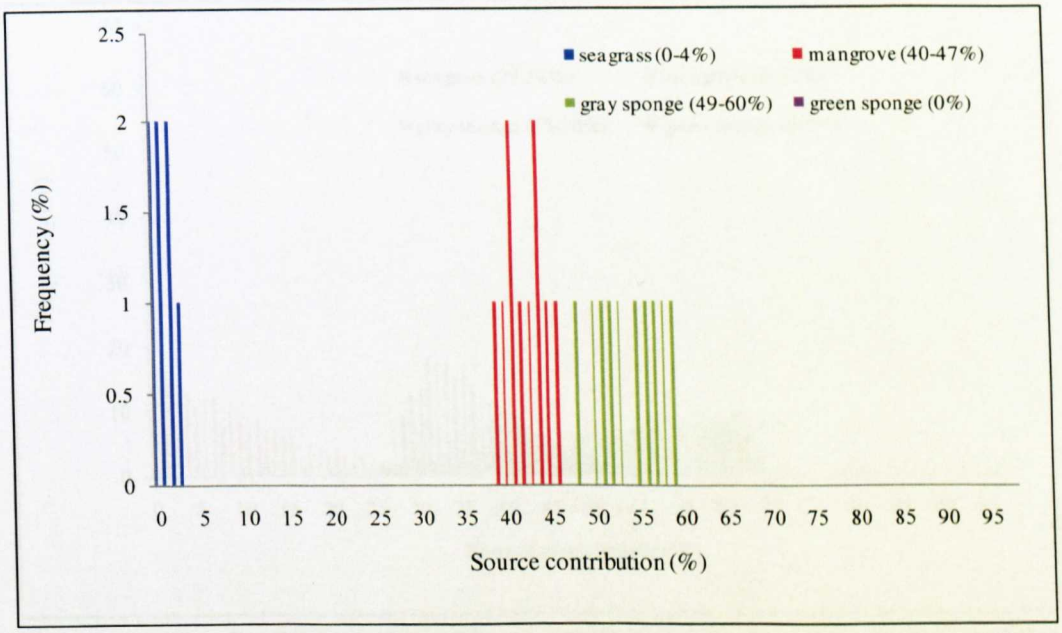

Figure 6.8 The distribution of feasible contributions of food sources to *Diplodus noct* fish in Shuaiba mangroves, Saudi Arabia; values between brackets are the 1-99‰ distribution ranges.

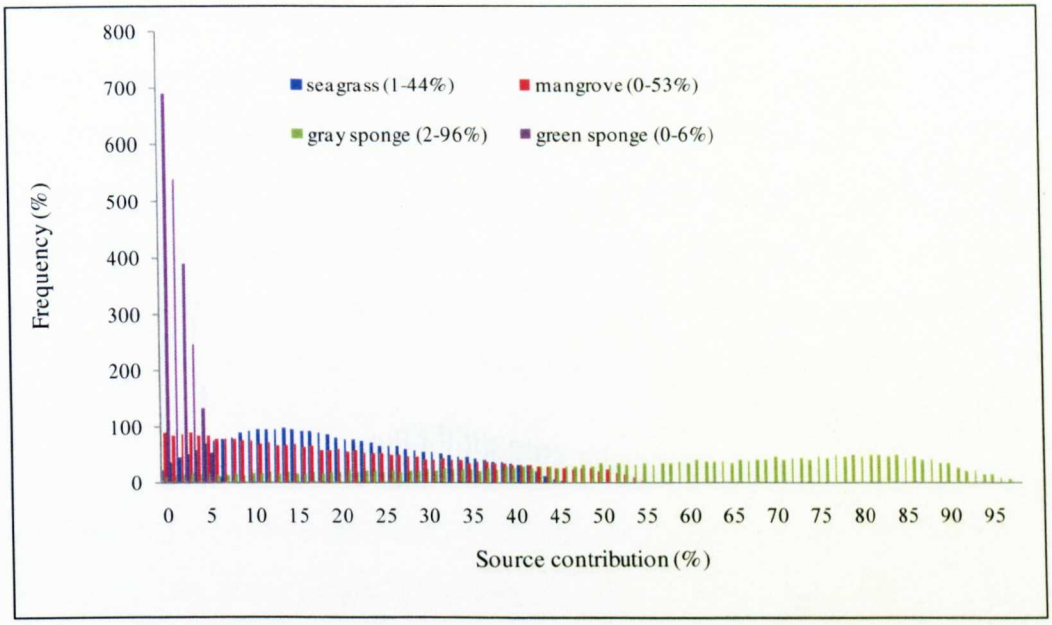

Figure 6.9 The distribution of feasible contributions of food sources to *Sphyraena flavicauda* fish in Shuaiba mangroves, Saudi Arabia; values between brackets are the 1-99‰ distribution ranges.

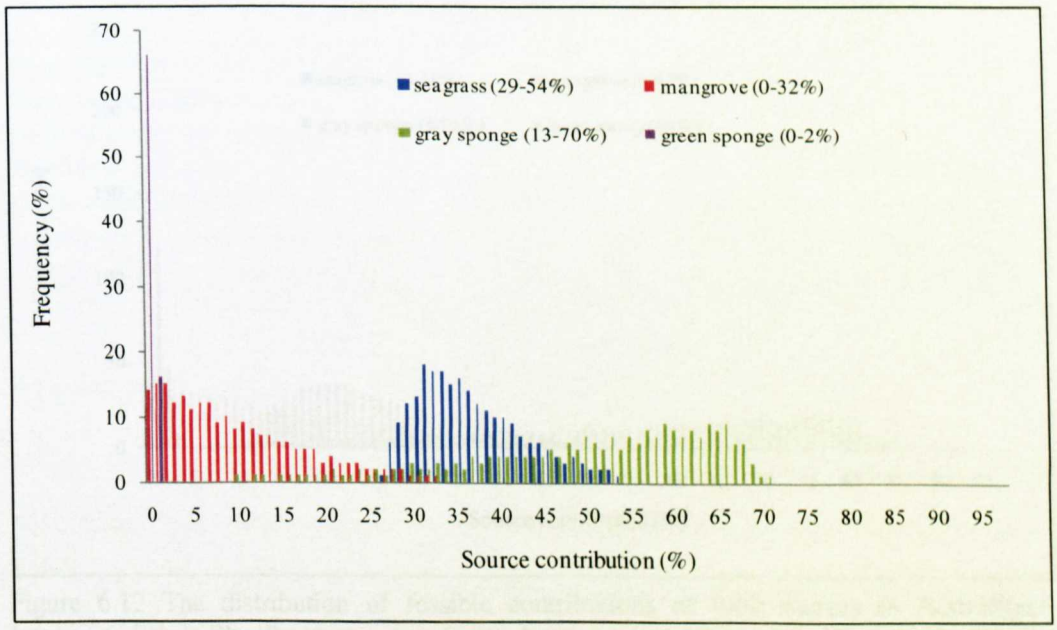

Figure 6.10 The distribution of feasible contributions of food sources to *Oedalechilus labiosus* fish in Shuaiba mangroves, Saudi Arabia; values between brackets are the 1-99‰ distribution ranges.

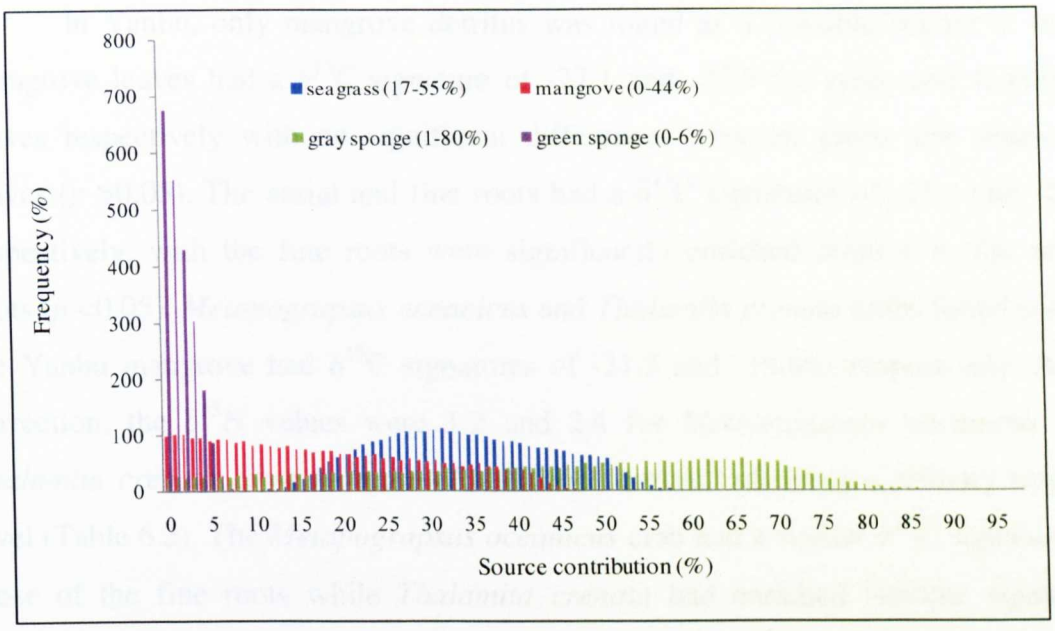

Figure 6.11 The distribution of feasible contributions of food sources to *Siganus rivulatus* fish in Shuaiba mangroves, Saudi Arabia; values between brackets are the 1-99‰ ranges of the distributions.

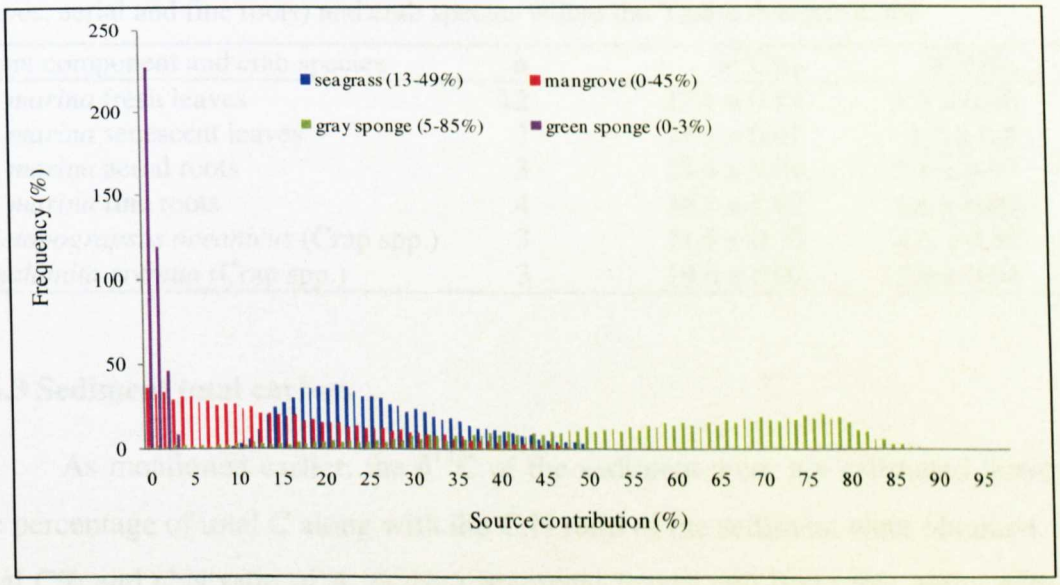

Figure 6.12 The distribution of feasible contributions of food sources to *Rastrelliger kanagurta* fish in Shuaiba mangroves, Saudi Arabia; values between brackets are the 1-99‰ ranges of the distributions.

6.6.2 Yanbu $\delta^{13}C$ signatures in plant and animal tissues

In Yanbu, only mangrove detritus was found as a possible source of food. Mangrove leaves had a $\delta^{13}C$ signature of -27.1 and -27.5 for green and senescent leaves respectively with no significant differences between green and senescent leaves (p >0.05). The aerial and fine roots had a $\delta^{13}C$ signatures of -25.4 and -20.3 respectively, with the fine roots were significantly enriched relative to the aerial roots (p <0.05). *Metopograpsus oceanicus* and *Thalamita crenata* crabs found within the Yanbu mangrove had $\delta^{13}C$ signatures of -21.5 and -19.6‰ respectively. After correction, the $\delta^{15}N$ values were 1.2 and 2.4 for *Metopograpsus oceanicus* and *Thalamita crenata* close to those of the food sources indicating a primary trophic level (Table 6.3). The *Metopograpsus oceanicus* crab had a similar $\delta^{13}C$ signature to those of the fine roots while *Thalamita crenata* had enriched isotopic signature relative to any of the mangrove components suggesting additional unsampled food source for that species.

Table 6.3 Stable isotope values ($\delta^{13}C$‰ and $\delta^{15}N$‰ ± SE) of mangrove plant components (leaves, aerial and fine roots) and crab species within the Yanbu mangrove site

Plant component and crab species	n	$\delta^{13}C$‰	$\delta^{15}N$‰
A. marina fresh leaves	12	27.1 ± 0.13	1.7 ± 0.60
A. marina senescent leaves	7	27.5 ± 0.07	1.7 ± 0.5
A. marina aerial roots	3	25.4 ± 0.04	2.1 ± 0.27
A. marina fine roots	4	20.3 ± 1.62	1.6 ± 0.03
Metopograpsus oceanicus (Crap spp.)	3	21.5 ± 0.70	4.6 ± 0.28
Thalamita crenata (Crap spp.)	3	19.6 ± 0.02	5.8 ± 0.04

6.6.3 Sediment total carbon

As mentioned earlier, the $\delta^{13}C$ of the sediment were not estimated however, the percentage of total C along with the C:N ratio of the sediment were obtained. The total C% and C:N ratio of *A. marina* senescent leaves and fine roots along with the tested C sources were compared to those of the sediment (Table 6.4). The sediment C% were 9.6 and 5.1 for Shuaiba and Yanbu respictivley while C% was 51.9, 50.4, 30.0, 25.2, 19.2, 24.3 and 24.1 for leaves in Shuaiba, leaves in Yanbu, fine roots in Shuaiba, fine roots in Yanbu, gray sponge, green sponge and seagrass (Table 6.4). Moreover, Shuaiba had higher total carbon in sediment than Yanbu (p <0.05)

Table 6.4 Total carbon content (%) and C:N ratio in sediment and other sources of carbon in Shuaiba and Yanbu sites

Material	Shuaiba		Yanbu	
	Total C (%)	C:N ratio	Total C (%)	C:N ratio
Sediment (5 cm depth)	9.6	46.8	5.1	55.9
A. marina senescent leaves	51.9	77.7	50.4	78.6
A. marina fine roots	30.0	22.9	25.2	42.6
Gray sponge	19.2	5.00	-	-
Green sponge	24.3	4.80	-	-
Seagrass	24.1	20.80	-	-

6.7 DISCUSSION

6.7.1 Energy sources in the mangrove system

The $\delta^{13}C$ signatures of *A. marina* were similar to those reported in literature (*e.g.* Rao *et al.*, 1994; Sheaves and Molony, 2000; Melville, 2005; Werry and Lee, 2005; Hogarth, 2007, Table 6.5). Variations in the isotopic signature of the different mangrove plant components have been reported for many mangrove species however, no clear pattern of variation has been observed. For example, Boon *et al.*, (1997) reported no significant differences in $\delta^{13}C$ signature between leaves and woody components of *A. marina* while aerial roots were significantly lighter than leaves. On the other hand Ellison *et al.*, (1996) found that $\delta^{13}C$ signature of *Rhizophora mangle* cable roots were significantly more enriched than the leaves.

Table 6.5 Stable isotope values ($\delta^{13}C‰$) of *A. marina* senescent leaves from various sources

Source	$\delta^{13}C‰$
Current study	-25.8 to -27.5
Hogarth (2007)	-27.1
Melville (2005)	-27.6
Werry and Lee (2005)	-27.4
Rao *et al.* (1994)	-26.2
Sheaves and Molony (2000)	-26.4

The current investigation was aiming to evaluate the contribution of different energy sources within the mangrove system to a variety of aquatic animals, therefore, sources from outside the mangrove system were not considered. The mangrove sediment has been reported to constitute a significant proportion of the diet of crab (*e.g.* Boullion *et al.*, 2002; Skov and Hartnoll, 2002; Boon *et al.*, 2008) and fish species (*e.g.* Kruitwagen *et al.*, 2010). In addition, sediments at lower

intertidal areas has been reported to provide a good substrate for benthic micro and macro algae and bacterial growth making the sediment a rich source of food (Robertson, 1986; Sarpedonti and Sasekumar, 1996; Skov and Hartnoll, 2002). Therefore, not including the sediment as a possible food source in the mixing model may have resulted in excluding a significant food source from the mixing polygon.

Although falling outside the mixing polygon, the crab species were closer to the borders of the polygon than the fish species. The importance of the mangrove detritus to the diet of many crab species belonging to Grapsids has been reported (*e.g.* Robertson, 1986; Kruitwagen *et al.*, 2010). Among the tested sources, the mangrove detritus was most important in the diet of *Metopograpsus* and was slightly less important in eight fish species. *Metopograpsus* are mangrove associated crabs that can feed on plant materials and are found frequently on mangrove ground or clinging onto aerial roots (Chapter 4). In addition, previous investigations on crab diets have revealed moderate contribution of mangrove detritus to the diet of the *Metopograpsus* crabs (Dahdouh-Guebas *et al.*, 1999). Considering that high C:N ratio in leaves decreases its palatability to crabs, it seems that *Metopograpsus* crabs still incorporate a significant amount of mangrove litter in their diet which support field observation (Chapter 4). The feeding habits of Grapsid crabs are still poorly understood and vary greatly from a place to another (Werry and Lee, 2005). Grapsid crabs were reported to directly consume newly fallen litter (Micheli, 1993), or aged litter with lower C:N ratio (*e.g.* Lee, 1993). In environment with few sources of nutrients such as in the Red Sea, it might be possible that crabs are forced to consume or "deal with" the most abundant source (*i.e.* mangrove leaves) if preferable sources were scarce (Thongtham *et al.*, 2008). The importance of mangrove detritus in the diet of *Portunus* and *Thalamita* species was less than *Metopograpsus*. These crab species are swimmer crabs that are present at the edges of the mangrove stands in low intertidal zones (Chapter 4) thus they may be more exposed to feeding on sponge, seagrass and other sources.

The corrected $\delta^{15}N$ values of the different fish species ranged from 1 to 4.7‰ indicating different trophic levels. Species with low $\delta^{15}N$ values indicate herbivorous feeding while species with higher values indicate carnivorous feeding (Nagelkerken, 2008). Generally the non vascular sources were more important in the diet of the aquatic animals than the mangroves as they represent a rich and accessible source of

N. The mangrove leaves containing high levels of lignified compounds and generally high C:N ratios may decrease palatability and may not meet the requirement of the different aquatic animals (Chapter 5). Working on sasarmid crab feeding habits, Skov and Horton (2002) have reported that mangrove detritus with high tannin and C:N ratio was unlikely to meet the N needs of the crabs and that sediments with low C:N ratio was a preferable N source for the crabs.

The mangrove detritus showed moderate contribution to most of the sampled fish, the fact that most of the fish species fell outside the mixing polygon suggests that this contribution is overestimated and that including other sources would have reduced this contribution level. In addition, carnivorous fish species were probably accessing the mangroves to feed on small aquatic animals such as juvenile fishes. As the fish species collected for the current study were adult, their dependency on food sources may not also reflect the importance of these sources. Juvenile aquatic animals including fish and crustaceans tend to shift their diet as they age and migrate off shores evident in the changes in their isotopic signature (Fry 1983; Huxham *et al.*, 2007). Thus an investigation focusing on the contribution of the possible sources in the diet of juvenile fish would clarify the importance of such source in the life cycle of fish species.

Sponges were the most important food source in the mangrove system, unlike seagrass that are found at distances away from the mangroves, sponges normally occur on the bottom of mangrove roots, below water level, but may be able to survive above water level during a tidal cycle (Barnes, 1999). In the current study sponges were found attached to aerial roots only at the edges of the mangrove stands. On average, the sponges had much lower C:N ratio (4.9) than mangrove leaves (78), fine roots (32) or seagrass (21) which may better meet the nutritional requirement of the aquatic animals. Russell-Hunter (1970) has viewed materials with C:N ratios below 17 as adequately meeting the nutritional requirement of the aquatic animals. In general, the sponges found in the mangroves were not extensive and only occupy very small areas below the aerial roots. This might be attributed to the nature of the *Avicenna* roots. Sponges are reported to extensively grow under the long probe roots of *R. mucronata* at the low intertidal zone extending below the lowest low tidal level while the *Avicennia* trees lack this feature. Therefore, sponges are more abundant and diverse in adjacent subtidal habitats afforded by coral reefs, hard-bottom areas

and seagrass beds (Barnes, 1999; Barnes and Bell, 2002). Considering their availability in the mangrove system, it is more likely that the sampled aquatic animals feed on these sources and other sources in the larger subtidal habitat.

Since the remarkable work by Odum and Heald (1975) on the importance of marine plants in estuarine food webs, the role of mangrove detritus as an energy source to the aquatic food web has long been considered significant. However, with the recent investigations employing new techniques such as isotopic analysis it has been thought that the role of the mangrove detritus as a main energy source for aquatic animals may have been overestimated (France, 1988; Meziane *et al.*, 2002; Bouillon *et al.*, 2004). Over the last three decades, several studies examining the importance of mangrove detritus have reported minor or no significant contribution of mangrove detritus into the aquatic food web (*e.g.* Rodelli *et al.*, 1984; Lee, 1995; Loneragan *et al.*, 1997; Macia, 2004; Benstead *et al.*, 2006; Nerot *et al.*, 2009), while few studies have reported utilization of mangrove derived C by shrimps (*e.g.* Chong *et al.*, 2001). It has become clear that the contribution of mangrove detritus in the aquatic animal food webs may be largely dependant on the setting and the geomorphology of the mangrove system. Using $\delta^{13}C$ as an indicator for source contribution in different mangrove systems, Boullion *et al.*, (2004) have reported significant mangrove contribution in systems with low water exchange owing to minimal inputs of aquatic sources. On the other hand, Pineda (2003) showed that consumers in riverine and estuarine systems depend largely on mangrove detritus while mangrove contribution in lagoon and island settings was much more limited owing to the availability of high primary production sources.

Estimating the sources of C in the mangrove sediment and in the adjacent sediments was one of the main objectives of the present study. The C derived from mangrove detritus is frequently reported to be present in the mangrove sediments (*e.g.* Zieman *et al.*, 1984; Fleming *et al.*, 1990), or even extended to adjacent sediments via export (Robertson *et al.*, 1992). As the mangrove derived carbon could not be estimated via isotopic analysis in the current study, the total carbon and C:N ratio were used to gain a picture of the sources of carbon in the sediment. The total carbon content in Shuaiba sediment were higher than those in Yanbu which can be resulted from greater carbon incorporation in the sediment from the multiple available sources. This contribution of the other sources can also be viewed from the

lower C:N ratio in Shuaiba compared to Yanbu indicating the incorporation of the low C:N ratio sources in the sediment total carbon. Generally, the high C:N ratio of sediment correspond to those of the mangrove leaves and indicate greater incorporation of leaves than any other source.

The degree to which mangrove derived C can contribute to the sediment organic C has been largely attributed to the level of water exchange, sites with low tidal levels tend to retain and accumulate mangrove detritus on the forest floor leading to incorporation into sediment organic matter pool, where inwelling of marine sources such as seagrasses may contribute significantly into sediment organic matter in sites with high tidal levels (Lallier-Verges *et al.*, 1998; Boullion *et al.*, 2003, 2004). The former condition broadly applies to the Red Sea mangroves, low tidal levels result in litter accumulation on the forest floor and in slow litter decomposition (Chapter 4 and 5) which can lead to slow build up of a peaty forest floor. The low tidal levels may also limited inputs of external sources into the mangrove sediment. From the current findings of sediment total carbon and C:N ratio, it can be hypothesized that the mangrove detritus may constitute a great proportion of the sediment organic matter; investigation that involves estimates of $\delta^{13}C$ isotope along with the C:N ratios of potential sources and sediments may help supporting this idea. Moreover, although estimating mangrove derived C would be an important element when constructing a C budget for mangrove systems, studies that address the sources of organic C stocks in mangrove sedimentation are rare (Boullion *et al.*, 2003).

6.7.2 Mixing models and limitations

The stable isotope technique has been widely used in ecological studies to trace energy sources in marine food webs, one of the uses of isotopic techniques is to aid in estimating the fractional contribution of the food sources into a sample's signature. Linear mixing models has been frequently used to find the fractional contribution of possible sources into marine aquatic animals. The two source mixing model with a single isotope signature (*e.g.* $\delta^{13}C$) has been used to find the fractional contribution to a sample with the assumption of two major sources contributing into the diet (Balesdent and Mariotti, 1996). A three source model was later developed to account for three major sources utilizing two isotopic elements (*e.g.* $\delta^{13}C$ and $\delta^{15}N$;

Phillips, 2001). However, marine food web normally involve a collection of many sources that can be part of the animals diet (Fry, 2006), which prevents obtaining a feasible solution of source contribution. While no definitive solution exists, some new techniques have been used to cope with multiple sources. One technique involves minimizing the sources to a number that would allow the mixing models to find feasible solutions, this is done by lumping sources with similar characteristics (*i.e.* similar isotopic signature, sources are logically related) into one sample allowing distinctive isotopic signature to be analyzed by the mixing model (Phillips *et al.*, 2005). The problem with this technique is that lumping samples can lead to great variability in the lumped source compared to its original individual sources which can translate into greater uncertainty of the estimate (Phillips *et al.*, 2005).

The IsoSource model technique is an informative technique that can aid in finding feasible solutions for too many sources (more than three) by determining the contribution ranges of each possible source. Since it was proposed by Phillips and Gregg (2003), it has been used in many investigation with promising results (*e.g.* Melvile and Connolly, 2003; Benstead *et al.*, 2006). Despite its usefulness, the technique suffers from some limitations. For example, if more than one source share similar isotopic signature, obtaining a unique solution would be impossible unless other constraints are added to role out one source over another (Fry, 2006). In addition, the assimilation of an element (*e.g.* N) of one source can be variable depending on the element form in the original food material (*e.g.* carbohydrates, lipids, proteins) leading to variable contribution of the sample relative to the other element (*e.g.* C) (Vanderklift and Ponsard, 2003; Boullion *et al.*, 2008). Such inconsistency in elemental assimilation is not accounted for in IsoSource software.

Increasing the tolerance level for some analyses weakens the confidence of the source trophic contribution because samples fall outside the boundaries of the mixing polygon and thus the results obtained of sample contribution are overestimated. Since IsoSource reports all feasible solutions for source contribution, there is no criterion of preferring a unique solution. Incorporating other ecological factors may be necessary; incorporating other techniques can aid in ruling out a unique solution over another. For example, gut content analyses was considered a good technique that can be done in conjunction with isotopic analysis since this allow a direct sampling of animals food not just a potential food source in the system

(Gray *et al.*, 2002). Other techniques including feeding experiments, and behavioral studies were also reported to effectively aid in the interpretation of the stable isotope analysis (Peterson, 1999).

6.8 CONCLUSIONS

The mangrove detritus was found to moderately contribute to the diet of crabs and fish species however, this contribution might be compromised by the exclusion of some food sources from the mixing model. The inclusion of the sediment as a possible food source could add value to the mixing model and may have resulted in capturing all samples within the mixing polygon. In addition, the high trophic level of most of the aquatic animals can infer the missing sources from the model and suggest inclusion of such sources in future investigations.

Further investigations addressing the sources of organic carbon in the mangrove sediment are needed to gain a better picture of the overall importance of mangrove detritus in building sediment organic matter within the mangrove system and in adjacent sediments. Thus aid in estimating the carbon budget in intertidal areas. A full modeling of the food web in mangrove systems on the Red Sea coast is also needed, this would include thorough investigation of all possible sources, the species abundance and seasonality, predatory and herbivory species, and the micro and macrofaunal (*i.e.* aquatic animals and birds) biomass. There is also a need to incorporate multiple techniques to estimate the level of contribution of sources into the sediment organic matter as it might yield a better interpretation of results. Such investigations would add value to the role of mangrove ecosystem as a food, refuge and nursery habitat for the different animals present at the mangrove systems. Moreover, it would aid in setting the management and conservation goals for marine protected areas.

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CHAPTER 7

HEAVY METAL POLLUTION IN MANGROVE SYSTEMS ON THE RED SEA COAST, SAUDI ARABIA

7.1. INTRODUCTION

Coastal areas are largely affected by pollution from different anthropogenic sources including oil, dredging, urban discharges, agricultural and industrial wastewater. Due to their toxicity, bioaccumulation capacity and persistence, heavy metals represent a significant and serious pollution source that is associated with anthropogenic activities in coastal environments (Harbison, 1986; Clark *et al.*, 1998; Tam and Wong, 2000). Heavy metals which are biologically and chemically not degradable are characterised by their high mobility and thus can be transported over long distances to affect adjacent systems (Marchand *et al.*, 2006). As mangrove systems occupy the intertidal areas of coasts, they are extremely exposed to pollution; heavy metal pollutants from various sources such as those from industrial effluence, boating activities and oil spills can reach the low energy mangrove shores, precipitate on sediment and accumulate in plant tissues (Stafford-Deitsch, 1996). In addition, the anaerobic sulphide-rich sediment with its high metal binding capability can trap metals in mangrove sediments, increasing concentration and thus increasing accumulation in sediment and plant tissues (Mackey and Mackay, 1996; MacFarlane *et al.*, 2003).

Mangrove trees are reported to tolerate high levels of heavy metals (Peters *et al.*, 1997; MacFarlane, 2007). *Avicennia* species are thought to exhibit greater tolerance and accumulative properties to numerous metals than other mangrove species (Thomas and Eong, 1984; Peng *et al.*, 1997; MacFarlane and Burchett, 2002). Generally, plants that grow in contaminated environments sequester heavy metals in physiologically inert components (*e.g.* structural tissues) so that the plant's biological activities are not affected (Brooks, 1998). Similarly, mangrove trees that grow on sediments with high levels of heavy metals exclude and regulate uptake of

metals at the root level depending on their importance (*i.e.* essential or non essential metals) (Lacerda *et al.*, 1993; MacFarlane and Burchett, 2002).

The cycling and export of heavy metals is a function of metal concentration in litterfall produced, decomposition, residence time and tidal activities (Silva *et al.*, 2006). High concentrations of heavy metals in the plant can reach toxic levels in which metals with high concentrations are excluded from the plant via translocation to senescent leaves that are about to fall; litterfall containing high levels of excluded heavy metals can in turn release a significant amount of heavy metals through decomposition into the sediment, and when accompanied by high tidal activities, export metals to adjacent systems (Silva *et al.*, 2006). On the other hand, as low metal concentrations are found in leaves compared to other plant components, it is thought that minimal concentrations of heavy metals are exported and that mangrove systems act as a sink for heavy metals with negligible export to adjacent systems (Salt *et al.*, 1995; Silva, 1998; MacFarlane, 2003). However, little research is available on the cycling of heavy metal in mangrove systems through litter production, decomposition and export (Silva *et al.*, 2006).

On the Red Sea coast of Saudi Arabia, anthropogenic pollution is a major cause of the deterioration of many mangrove stands. Urbanisation of native areas on the coast associated with high anthropogenic wastes has resulted in massive deterioration, deformation and decrease of area cover of many mangrove stands within few decades (Dicks, 1986; Mandura and Khafaji, 1993; Mandura, 1997 and El-Juhany, 2009). In response to such threat, the National Committee of Wildlife Conservation and Development (NCWCD) has developed a system plan for Marine Protected Areas (MPA) in an attempt to establish protected areas in the coastal zones, combined with rehabilitation and reforestation programs. However, sufficient information on the environmental quality and the impact of anthropogenic activities on the mangrove stands is lacking causing major gaps in the MPA guidelines. In 2002, The Regional Organization for the Conservation of the Environment of the Red Sea and Gulf of Aden (PERSGA) has planned and executed a mangrove survey programme; giving indication of the mangrove's status and suggested guidelines for rehabilitation, conservation and management. Among many was the need for scientific research in order to implement an integrated management and conservation approach (PERSGA, 2004).

The overall objective of this chapter was to contribute to the MPA environmental database and guidelines by assessing the current environmental condition of mangrove systems at two locations on the Red Sea coast. The results achieved through this investigation is expected to aid in decision making regarding use, management and conservation of mangrove stands and allied resources on the Red Sea coast by assessing the level of heavy metal contamination in an industrially exposed mangrove plantation in Yanbu city and a less exposed stand in Shuaiba.

The specific objectives were:

1. Assessing the heavy metal bioaccumulation in mangrove trees.
2. Assessing the heavy metal concentration in mangrove sediment.
3. Assessing the heavy metal input, release and possible export from the mangrove system through estimates of litterfall production, decomposition, residence time and tidal activities.

The hypotheses of this study are:

1. Mangrove trees prevent uptake of highly concentrated heavy metals by exclusion at the root level.
2. Heavy metal concentrations are higher in more exposed environments.
3. Mangrove systems function as a sink for heavy metals with minimal export through litterfall owing to low metal concentrations and long residence time on the forest floor.

7.2 MATERIAL AND METHODS

7.2.1 Sampling sites:

Two mangrove locations on the Red Sea coast were selected for the present study, a northern site located in the industrial city of Yanbu (24° 02' 65" N and 38° 09' 46"E) and a southern site in Shuaiba region (20°46' 2"N and 39° 30' 21"E). Shuaiba is an old port lying about 100 km south of the city of Jeddah, the region comprises two lagoons extending for some 20 km from north to south with the greatest width being 5 km, and each lagoon is connected to the sea through a small channel. Urbanization in the region is limited to the Shuaiba desalination plant and a small recreational village. Moreover, previous studies had reported the site to be minimally exposed to pollution (Alharbi, 1988). Yanbu is the largest industrial city

on the Red Sea; it accommodates several large refining and petrochemical plants, manufacturing and supporting operations. Because of that, it is considered the cornerstone of the country's industrial development (Hashem, 1998). Yanbu's mangrove stand lie in the centre of the industrial city in a location exposed to multiple pollution sources. Although regulations governing the disposal of treated industrial wastewaters are set, environmental investigation revealed high concentrations of heavy metals in Yanbu soils (Hashem, 1993) and in the coastal ecosystem (Paimpillil *et al.*, 2002) indicating violation of disposal standards.

7.2.2. Sampling procedures

Mangrove tree components including green leaves, branches, and stem were collected from trees used in biomass and nutrient cycling studies (Chapter 3 and 4). Plant components were randomly sampled from trees in different plots, aerial roots were sampled by placing a 1 m^2 quadrat at distances from trees and roots within the quadrats were sampled (Chapter 3). Fine roots were sampled at depths reaching 50 cm using a 1.9 cm radius corer. Random coring was taken at 1 m and 2.5 m distance away from mangrove trees (Chapter 3). A total of three samples per component per site were used for heavy metal determination. The input, release and possible export of heavy metals in the mangrove system were assessed through estimating leaf litterfall production, standing crop litter and litter decomposition. Samples for annual estimate of litterfall in each site were pooled and three subsamples were taken for heavy metal determination. To estimate the heavy metal input through leaf litterfall, the mean concentration of metals in leaf litter were multiplied by the mean leaf litterfall rate (kg ha^{-1} y^{-1}). Metal concentrations in decomposing litter were assessed by taking subsamples of decomposed litter (three subsamples) at each sampling time and analysed for metal concentrations (Chapter 5) with a total of 21 samples per site for the whole decomposition study. Plant metal uptake was assessed via three measurements of root concentration factor (RCF) assessing the metal concentrations in roots relative to sediment metal concentration. Leaf concentration factor (LCF) assessing the metal concentration in leaves relative to the sediment metal concentration and finally metal translocation factor (TF) assessing the transport of root metals to leaves (MacFarlane *et al.*, 2007). The export of heavy metal from the mangrove system is a function of the litter residence time and its metal concentration;

the residence time of litter on the forest floor surface was calculated using the equation:

$$\tau = X_{ss} / L$$

Where τ is residence time, X_{ss} is the annual rate of litter standing crop and L is the annual rate of litterfall input.

The turnover of the heavy metals in each site was estimated using a ratio of the metal in the perennial biomass to the annual input of litter through litterfall (Silva *et al.*, 2007).

Sediment samples were taken from the different plots using random coring (Chapter 6). Sediments were taken at 0-5 and 5-20 cm depths; three replicates were randomly taken from each plot and bulked into one sample per plot. A total of 15 samples per site were used for heavy metal determination. The sediment redox potential (Eh) has a significant impact on metal dynamics in mangrove sediments. Eh measurements have to be taken in field immediately after removing sediment cores from the ground (English *et al.*, 1997). However, such measurements were not done in the current study due to unsuitability of instruments for field measurments.

7.2.3 Sample preparation and laboratory analysis

Plant and soil samples were transported to the laboratory, oven dried at 70°C (plants) and 105°C (soils), ground to powder and packed in plastic vials (Chapter 3, 4 and 6). Samples were wet digested prior to heavy metal analysis with a mixture of concentrated nitric (HNO_3)-perchloric ($CHlO_4$) acids following the method described in Sparks *et al.* (1996). Subsamples (0.2 g) were weighed into 15 ml test tubes in digestion (heating) blocks, 1.6 ml of concentrated HNO_3 were added to each sample followed by 0.4 ml of concentrated $CHLO_4$. Glass stoppers were used to seal the tops of tubes but still allowing for excess fumes to be released and samples were left in acid over night. On the following day, temperature was gradually raised from 100, 150 and 225 °C over a period of 6 hours until full digestion. After digestion, tubes containing sample solutions were topped to 15 ml with distilled water and filtered through acid resistant filter paper.

Samples were then analysed for eight heavy metals including Chromium (Cr), Manganese (Mn), Iron (Fe), Nickel (Ni), Copper (Cu), Zinc (Zn), Cadmium (Cd) and Lead (Pb) using Fisons/VG PlasmaQuad II Inductively Coupled Plasma

Mass Spectrometer (ICP-MS) (Fisons, UK). The ICP-MS is a fast sequential mass analyzer that extracts positively charged ions from argon plasma, the spectrometer is calibrated with a range of multi-element synthetic standards and all samples and standards have a 0.1 μg ml^{-1} internal standard of ruthenium (Ru). Prior to the analysis, the sample solutions were spiked with Ru to give a final concentration of 100 μg l^{-1}, sample solutions were nebulized via a concentric nebulizer and introduced into the argon plasma as a fine aerosol. Metals of interest are ionised in a high temperature 27 MHz plasma, the ions are extracted from this plasma through two Ni cones and passed to the mass analyser where they are separated by a quadropole mass spectrometer based on their mass-to-change ratio, the resulting ions are detected by the detector unit giving mass spectrum of the different ions.

7.2.4. Statistical analysis

Statistical analyses were performed using SPSS ver. 14.0, Homogeneity of data was confirmed using Levene's test for homogeneity of variance; in cases of heterogeneity of variance, data were log-transformed prior to analysis. Mean differences in metal concentrations between soil depths and sites were compared using the Independent Sample t-test, Differences in metal concentration among individual plant components and decomposition periods were assessed using analysis of variance (ANOVA) with Tukey's pair-wise comparison test (p = 0.05, SPSS ver. 14).

7.3 RESULTS

7.3.1 Heavy metal bioaccumulation in mangrove stands

In Shuaiba mangroves, the heavy metal concentrations were variable across the different plant components, the concentration for the different metals (μg g^{-1}) ranged from 0.10 to 29.98 for Cr, 3.47 to 67.27 for Mn, 39.54 to 3913 for Fe, 1.59 to 33.12 for Ni, 3.17 to 40.02 for Cu, 4.23 to 18.27 for Zn, 0.01 to 0.38 for Cd and from 0.38 to 5.01 for Pd (Table 7.1). Generally, the high metal concentrations were always found in fine roots while the leaves accounts for the lowest concentrations. Fine roots had significantly highest concentrations (μg g^{-1}) of Cr (29.98), Mn (67.27), Fe (3913.14), Ni (33.12) Cu (40.02), Zn (18.27), Cd (0.33) and Pb (5.01) compared to the rest of the components (p <0.05). Only Mn and Fe concentrations were

significantly higher in woody components (stem and branches) than in leaves (Mn: 3.47 and 3.67 vs. 11.14, Fe: 271.14 and 198.86 vs. 39.54) for stem and branches against leaves respectively (p <0.05). Leaves had metal concentrations less than half those of the roots, the leaf TF of the essential metals were 0.23 for Zn, 0.10 for Cu, 0.16 for Mn and 0.01 for Fe while the non essential metal ratios were 0.11 for Pb, 0.20 for Ni, 0.03 for Cd and 0.15 for Cr.

In Yanbu, the heavy metal concentrations ($\mu g\ g^{-1}$) ranged from 0.21 to 13 for Cr, 6.36 to 104.81 for Mn, 243.80 to 7911 for Fe, 2.13 to 29.17 for Ni, 5.09 to 31.52 for Cu, 3.17 to 16.77 for Zn, 0.20 to 0.33 for Cd and 0.35 to 7.32 for Pb (Table 7.2). Similar to Shuaiba, higher metal concentrations were found in fine roots compared to the other components; significantly highest fine roots concentrations ($\mu g\ g^{-1}$) were of the metals: Mn (105), Fe (7912), Ni (29), Cu (31) and Pb (7) (p <0.05), while Cr and Zn concentrations were similar in fine and aerial roots but not to the remaining components. Cadmium concentrations were similar in all plant components (p >0.05). The translocation of metals to leaves was highly variable. For some metals, concentrations in leaves account for more that half those of the roots, the TF values were 0.63 and 0.72 for Mn and Ni respectively. However, the concentrations were lower in other metals (0.22 for Cu, 0.05 for Fe, 0.18 for Cr and 0.03 for Pb), while Zn and Cd in leaves had concentrations that were below the detectable limits.

Table 7.1 Heavy metal concentration in $\mu g\ g^{-1}$ (± SEM) (n=3) in *A. marina* tree components in a mangroves stand at Shuaiba region, Saudi Arabia

Components	Mean concentration							
	Cr	Mn	Fe	Ni	Cu	Zn	Cd	Pb
Fine root	29.98 (3.7)^{A}	67.27 (2.7)^{B}	3913.14 (208.2)^{E}	33.12 (1.7)^{H}	40.02 (4.1)^{I}	18.27 (0.8)^{J}	0.33 (0.4)	5.01 (0.08)^{K}
Aerial root	4.35 (0.3)^{a}	4.00 (0.2)^{b}	115.06 (29.3)^{e}	4.07 (1.9)^{h}	3.37 (0.9)^{i}	1.63 (0.7)^{j}	0.07 (0.06)	ND
Stem	ND	3.47 (0.6)^{b,C}	271.14 (42.5)^{e,F}	2.14 (0.5)^{h}	3.17 (1.3)^{i}	ND	0.19 (0.05)	0.38 (0.1)^{k}
Branch	0.10 (0.1)^{a}	3.67 (1.3)^{b,D}	198.86 (43.4)^{e,G}	1.59 (0.3)^{h}*	4.63 (1.2)^{i}	ND	0.38 (0.2)	0.68 (0.2)^{k}
Leaves	4.53 (0.22)^{a}	11.14 (0.5)^{b,c,d}*	39.54 (10.8)^{e,f,g}*	6.74 (3.2)^{h}*	4.17 (1.5)^{i}	4.23 (2.7)^{j}*	0.01 (0.02)	0.57 (0.5)^{k}
TF	0.15	0.16	0.01	0.20	0.10	0.23	0.03	0.11

SEM: standard error of means; ND: below detectable limits; TF: translocation factor; * n=2; values in the same column with the same letters are significantly different from the mean values of the variables with the corresponding capital letters at 0.05 significant level.

Table 7.2 Heavy metal concentration in $\mu g\ g^{-1}$ (± SEM) (n=3) in *A. marina* tree components in a mangroves stand at Yanbu city, Saudi Arabia

Components	Mean concentration							
	Cr	Mn	Fe	Ni	Cu	Zn	Cd	Pb
Fine root	13.00 (0.4)^{A}	104.81 (4.3)^{C}	7911.57 (1252.3)^{G}	29.17 (3.6)^{H}	31.52 (2.4)^{L}	16.77 (0.52)^{M}*	0.33 (0.02)	7.32 (1.8)^{N}
Aerial root	11.94 (3.5)^{B}	29.84 (6.2)^{c,D}	250.42 (84.2)^{g}	3.64 (0.9)^{h,I}*	6.79 (0.6)^{l}	7.79 (2.6)	0.32 (0.2)	0.46 (0.2)^{n}
Stem	0.21 (0.02)^{a,b}*	6.36 (1.7)^{c,d,E}	284.23 (64.7)^{g}	2.13 (0.2)^{h,J}	5.09 (1.8)^{l}	ND	0.19 (0.01)	0.56 (0.2)^{n}
Branch	0.35 (0.1)^{a,b}	6.52 (0.8)^{c,d,F}	243.80 (48.0)^{g}	3.28 (0.3)^{h,K}	5.45 (1.5)^{l}	3.17 (0.9)^{m}*	0.20 (0.01)	0.59 (0.05)^{n}
Leaves	2.37 (0.5)^{a,b}	65.95 (4.1)^{c,d,e,f}	375.39 (40.8)^{g}	21.10 (2.6)^{i,j,k}	6.83 (2.9)^{l}	ND	ND	0.22 (0.2)^{n}
TF	0.18	0.63	0.05	0.72	0.22	-	-	0.03

SEM: standard error of means; ND: below detectable limits; TF: translocation factor; * n=2; values in the same column with the same letters are significantly different from the mean values of the variables with the corresponding capital letters at 0.05 significant level.

7.3.2 Heavy metal accumulation in mangrove sediment

In Shuaiba, heavy metal concentrations did not differ at the 5 and 5-20 cm depths (p >0.05), similarly in Yanbu, metal concentrations were similar at the different sediment depths except for Ni where the metal concentration was higher at the 5 cm depth than at 5-20 cm (Table 7.3). when the sediment of the two sites were compared for concentration differences, it was found that Yanbu site had higher concentrations of Mn, Fe, Cu and Pb metals in sediment than Shuaiba (Table 7.4). Due to significant difference between Ni concentrations at different depths in Yanbu, Ni of the different sites was compared for each depth separately. No significant differences were found between the two sites in Ni concentrations at any depth.

The RCF and LCF are relative measure ratios of metal uptake and metal concentrations in leaves relative to those of the sediments (MacFarlane *et al.*, 2007). In Shuaiba, the RCF of *A. marina* were generally high and root metal concentrations reach up to 9.5 times the concentrations in the sediments. RCF of the essential metals were 6.52, 9.23, 1.24 and 2.16 for Zn, Cu, Mn and Fe respectively, while those for the non-essential metals were 9.45, 1.21, 16.5 and 3.45 for Pb, Ni, Cd and Cr respectively. High RCF values indicate accumulative uptake of metals to rates that are much higher than those of the surrounding environment. The LCF values showed that much less of that sediment concentrations were translocated to leaves, the LCF of the essential metals were 1.51, 1, 0.44, and 0.26 for Zn, Cu, Mn and Fe respectively. While those for the non-essential metals were 1.1, 0.46, 0.50 and 0.34 for Pb, Ni, Cd and Cr respectively (Table 7.5).

Although Yanbu sediment had higher Mn, Fe, Cu, Zn and Pb concentrations than those of Shuaiba, the plant uptake appears to be lower than in Shuaiba. The concentrations of these metals in roots reach up to only 1.9 times the sediment concentrations with RCF values of 0.67, 0.94, 2.31, 1.23 and 1.9 for Mn, Fe, Cu, Zn and Pb respectively. The LCF values for those elements were low and reached only 50% of that of the sediments however, the Ni LCF was the only exceptionally highly concentrated in leaves reaching 85% of that in the sediment (Table 7.6).

Table 7.3 Heavy metal concentration in μg g^{-1} (± SEM) of *A. marina* sediments at different depths in two mangrove stands at Shuaiba and Yanbu regions, Saudi Arabia

Site	Soil Depth (n)	Mean concentration							
		Cr	Mn	Fe	Ni	Cu	Zn	Cd	Pb
Shuaiba	5 cm (7)	8.61 (0.7)	60.73 (6.6)	2025. 90 (499.1)	25.57 (3.4)	4.47 (0.6)	3.53 (1.0)	0.04 (0.03)	0.57 (0.1)
	5-20cm (6)	8.9 (2.4)	61.93 (8.0)	1557.42 (219.6)	29.28 (5.3)	3.79 (0.7)	1.99 (0.5)	0.01 (0.02)	0.50 (0.2)
Yanbu	5 cm (7)	14.86 (3.6)	170.43 (24.5)	8772.57 (5562.9)	32.00 (4.3)^{A}	18.57 (7.0)	16.56 (4.5)	0.32 (0.15)	5.26 (1.3)
	5-20cm (6)	7.9 (0.7)	140.68 (18.4)	7906.17 (3669.1)	18.74 (1.8)^{a}	7.82 (1.1)	10.23 (1.7)	0.03 (0.01)	2.27 (0.23)

SEM: standard error of means; values in the same column with the same letters are significantly different from the mean values of the variables with the corresponding capital letters at 0.05 significant level.

Table 7.4 Heavy metal concentration in $\mu g\ g^{-1}$ (± SEM) of *A. marina* sediments in two mangrove stands at Shuaiba and Yanbu regions, Saudi Arabia

Site	Mean concentration						
	Cr	Mn	Fe	Cu	Zn	Cd	Pb
Shuaiba	8.75 (1.1)	61.33 (4.9)^{A}	1791.66 (284)^{B}	4.13 (0.4)^{C}	2.76 (0.6)^{D}	0.02 (0.02)	0.53 (0.1)^{E}
Yanbu	11.51 (2.1)	150.14 (15.7)^{a}	8825.94 (3307)^{b}	13.97 (4.0)^{c}	13.52 (2.6)^{d}	0.20 (0.1)	3.84 (0.8)^{e}

SEM: standard error of means; values in the same column with the same letters are significantly different from the mean values of the variables with the corresponding capital letters at 0.05 significant level.

Table 7.5 Distribution of heavy metals concentrations in μg g^{-1} (± SEM) in sediment, roots and leaves of *A .marina* mangroves growing in Shuaiba region on the Red Sea coast, Saudi Arabia

Component	Mean concentration							
	Cr	Mn	Fe	Ni	Cu	Zn	Cd	Pb
Sediment	8.74 (1.1)	61.28 (4.9)	1810 (284)	27.42 (1.9)	4.16 (0.4)	2.8 (0.6)	0.02 (0.02)	0.53 (0.1)
Fine roots	29.98 (3.7)	76.00 (8.3)	3913.14 (208.2)	33.12 (1.7)	38.39 (5.6)	18.27 (0.8)	0.33 (0.4)	5.01 (0.1)
Leaves	3.00 (1.5)	27.06 (15.9)	473.18 (433.7)	12.58 (6.1)	4.17 (1.5)	4.23 (2.7)	0.01 (0.02)	0.57 (0.5)
RCF	3.43	1.24	2.16	1.21	9.23	6.52	16.5	9.45
LCF	0.34	0.44	0.26	0.46	1.00	1.51	0.50	1.07

RCF: root concentration factor; LCF: leaf concentration factor.

Table 7.6 Distribution of heavy metals concentrations in $\mu g\ g^{-1}$ (± SEM) in sediment, roots and leaves of *A. marina* mangroves growing in Yanbu region on the Red Sea coast, Saudi Arabia

Component	Mean concentration							
	Cr	Mn	Fe	Ni	Cu	Zn	Cd	Pb
Sediment	11.66 (2.14)	156.7 (15.7)	8373 (3307)	24.85 (7.1)	13.61 (4.0)	13.64 (2.6)	0.18 (0.1)	3.88 (0.80)
Fine roots	13.00 (0.4)	104.81 (4.3)	7911.57 (1252.3)	29.17 (3.6)	31.52 (2.4)	16.77 (0.5)	0.33 (0.02)	7.32 (1.8)
Leaves	2.37 (0.5)	65.95 (4.1)	375.39 (40.8)	21.10 (2.6)	6.83 (2.9)	ND	ND	0.22 (0.2)
RCF	1.11	0.67	0.94	1.17	2.31	1.23	1.83	1.89
LCF	0.20	0.42	0.04	0.85	0.50	-	-	0.06

RCF: root concentration factor; LCF: leaf concentration factor.

7.3.3 Heavy metal input through litterfall in mangrove systems

The heavy metal input to the sediment through litterfall did not show significant differences between the two sites (Table 7.7). Overall the metal concentrations in the litterfall were 2.2 and 3.2 for Cr, 42.3 and 66.2 for Mn, 620 and 581 for Fe, 17.1 and 22 for Ni, 2.9 and 2.9 for Cu, 0.7 and 1.3 for Zn, 0.05 and 0.02 for Cd, 0.4 and 0.6 for Pb for Shuaiba and Yanbu respectively.

Table 7.7 Heavy metal concentrations in $\mu g\ g^{-1}$ (± SEM) (n=2) of *A. marina* litterfall in two mangrove stands at Shuaiba and Yanbu regions, Saudi Arabia

Site	Mean concentration							
	Cr	Mn	Fe	Ni	Cu	Zn	Cd	Pb
Shuaiba	2.21 (0.2)	42.31 (8.9)	619.72 (22)	17.08 (4.2)	2.91 (0.2)	0.73 (0.7)	0.05 (0.05)	0.43 (0.4)
Yanbu	3.17 (0.03)	66.22 (7.9)	581.57 (184)	22.01 (2.9)	2.94 (0.3)	1.33 (2.4)	0.02 (0.03)	0.57 (0.4)

SEM: standard error of means

In Shuaiba, the heavy metal concentrations in the decomposing litter were stable throughout the decomposition period for most of the metals (Figure 7.1, 7.2, 7.3 and 7.4) however, the concentrations for Mn, Zn, and Cu showed a significant increase in the first 64 days (Mn = 51.17 to 49.80 $\mu g\ g^{-1}$; Zn = 2.13 to 25.2 $\mu g\ g^{-1}$; Cu = 3.07 to 11.42 $\mu g\ g^{-1}$) ($p < 0.05$) followed by a decrease until the end of the decomposition period (Mn= 49.80 to 15.50 $\mu g\ g^{-1}$; Zn= 25.26 to 5.14 $\mu g\ g^{-1}$; Cu= 11.42 to 7.81 $\mu g\ g^{-1}$) ($p < 0.05$) (Figure 7.1 and 7.2). When metal concentrations are compared to those of litterfall, it was found that concentrations in the decomposing litter increased for Zn (0.73 to 5.14 $\mu g\ g^{-1}$) and for Cu (2.91 to 7.81 $\mu g\ g^{-1}$) ($p < 0.05$), Mn concentrations did not significantly change between litterfall and decomposing litter (42.31 in litter vs. 15.50 $\mu g\ g^{-1}$ in the decomposing litter) ($p > 0.05$). The estimated residence time in Shuaiba was 127 days and the tidal activities were within the range of 3-63 cm (Chapter 4). The residence time of the litter on the forest floor correspond to the low metal concentration in the litter at that decomposition stage. Thus, the removed litter are likely to contain low metal concentrations.

The patterns of heavy metal concentration were quite different in Yanbu. Initially, the concentrations in the decomposing leaves were constant for the first 64 days (Figure 7.5, 7.6, 7.7 and 7.8); afterwards, the concentrations of Mn, Cu and Pb gradually increased until the end of the decomposition period (Mn = 31.59 to 55.46

$\mu g\ g^{-1}$; Cu = 7.60 to 13.81 $\mu g\ g^{-1}$; Pb = 0.93 to 3.87 $\mu g\ g^{-1}$) (Figure 7.5, 7.6 and 7.7). Similar to Shuaiba, metal concentrations in litter and decomposing litter increased for Cu (2.94 to 13.81 $\mu g\ g^{-1}$) and for Pb (0.57 to 3.87 $\mu g\ g^{-1}$) (p <0.05), while the Mn concentrations stayed relatively similar (66.22 in litter vs. 55.46 in decomposing litter) (p >0.05). The residence time of litter on the forest floor was estimated to be 97 days and tidal activities were within the range of 6-88cm. the residence time of Yanbu litter correspond to the gradual increase of metal concentration in the litter increasing the likelihood of having high metal concentrations in the removed litter.

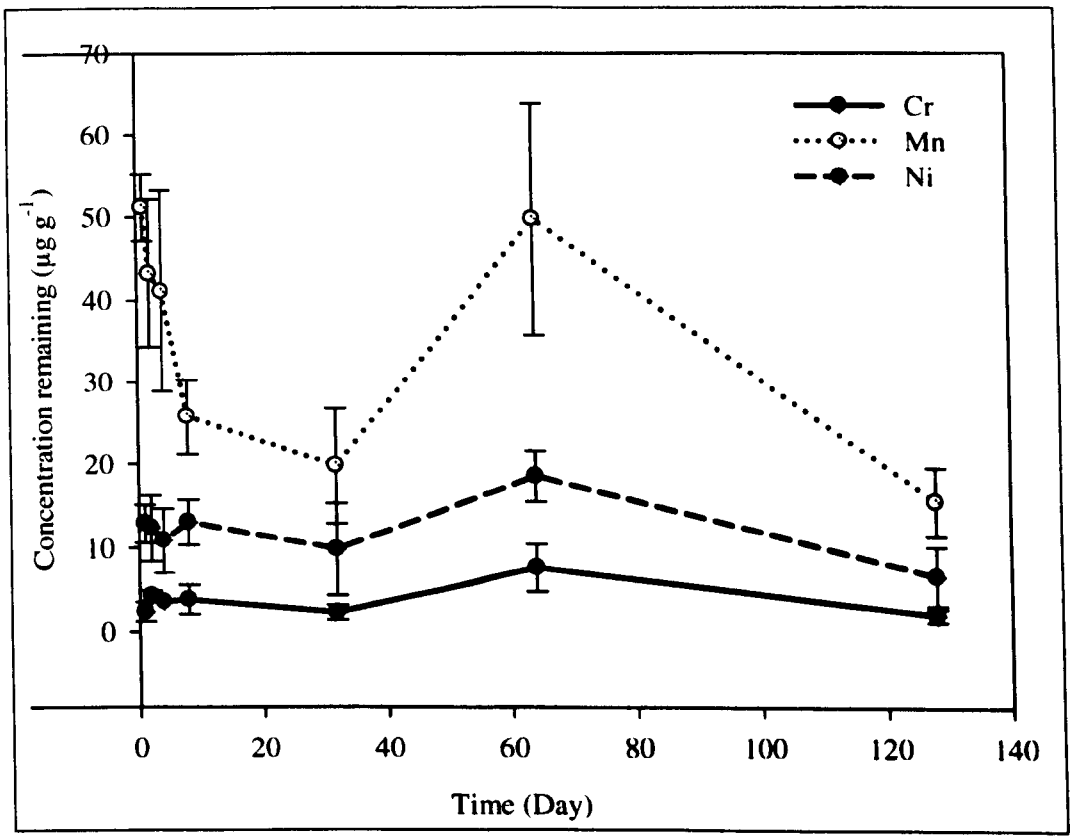

Figure 7.1 Changes in Cr, Mn and Ni metal concentrations ($\mu g\ g^{-1}$) in decomposing litter in a mangrove stand in Shuaiba region, Saudi Arabia (error bars are standard deviations).

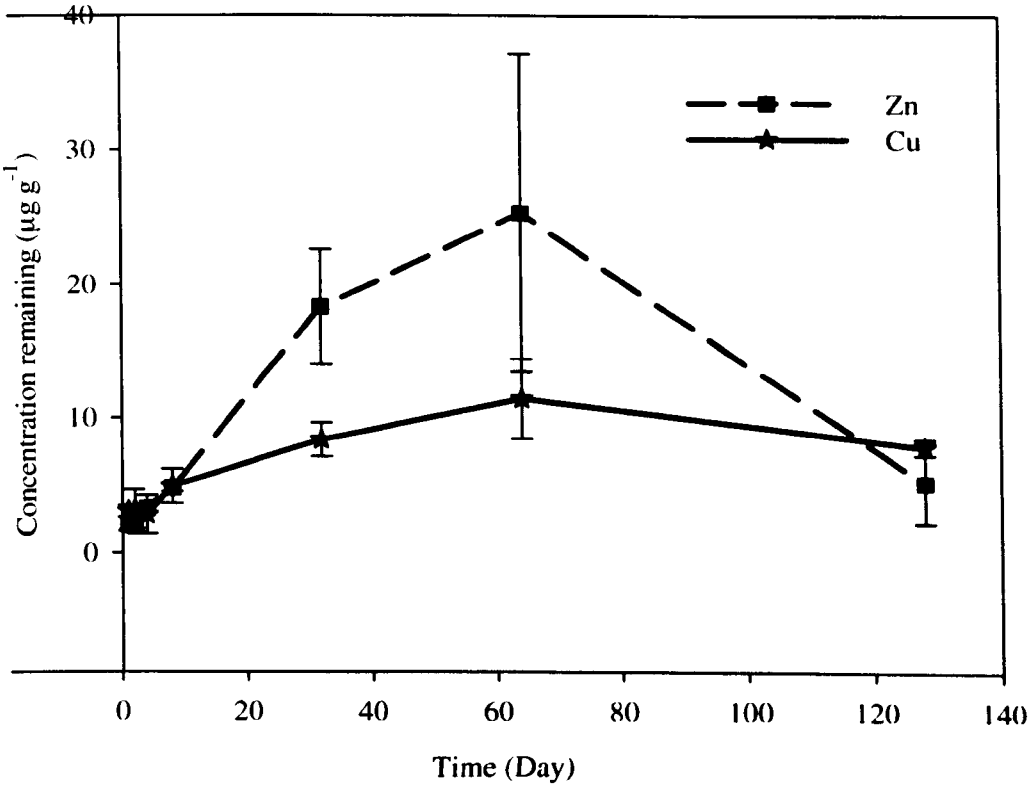

Figure 7.2 Changes in Zn and Cu metal concentrations (µg g^{-1}) in decomposing litter in a mangrove stand in Shuaiba region, Saudi Arabia (error bars are standard deviations).

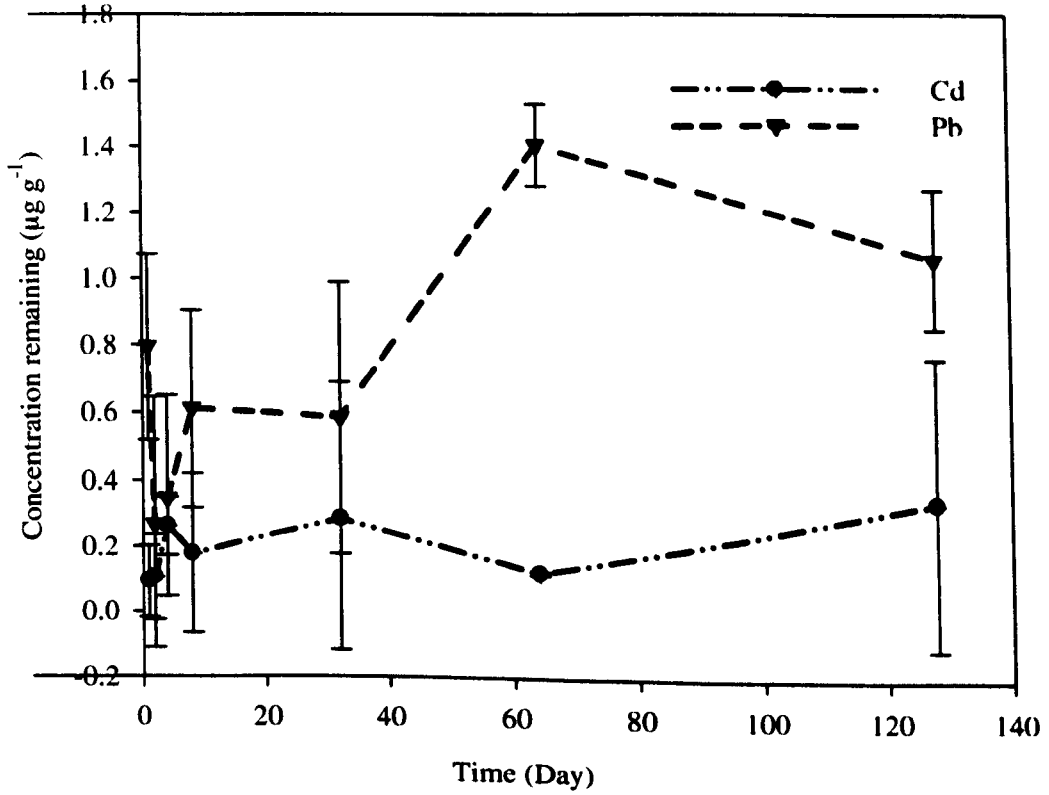

Figure 7.3 Changes in Cd and Pb metal concentrations (µg g^{-1}) in decomposing litter in a mangrove stand in Shuaiba region, Saudi Arabia (error bars are standard deviations).

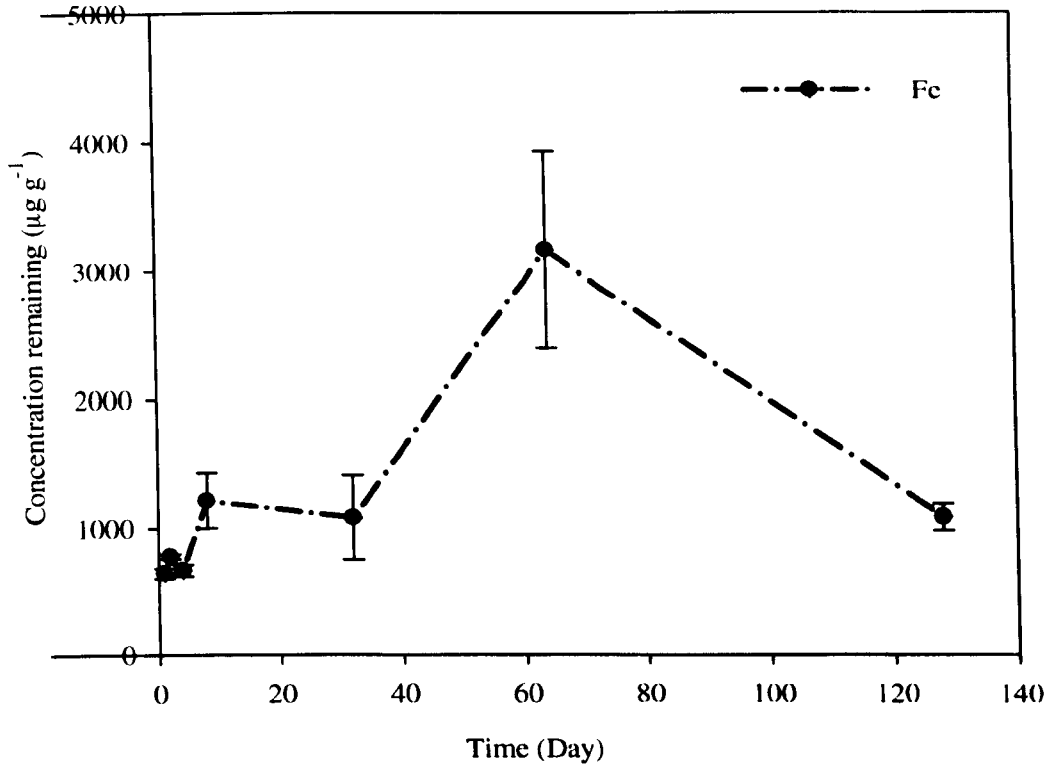

Figure 7.4 Changes in Fe metal concentration (µg g^{-1}) in decomposing litter in a mangrove stand in Shuaiba region, Saudi Arabia (error bars are standard deviations).

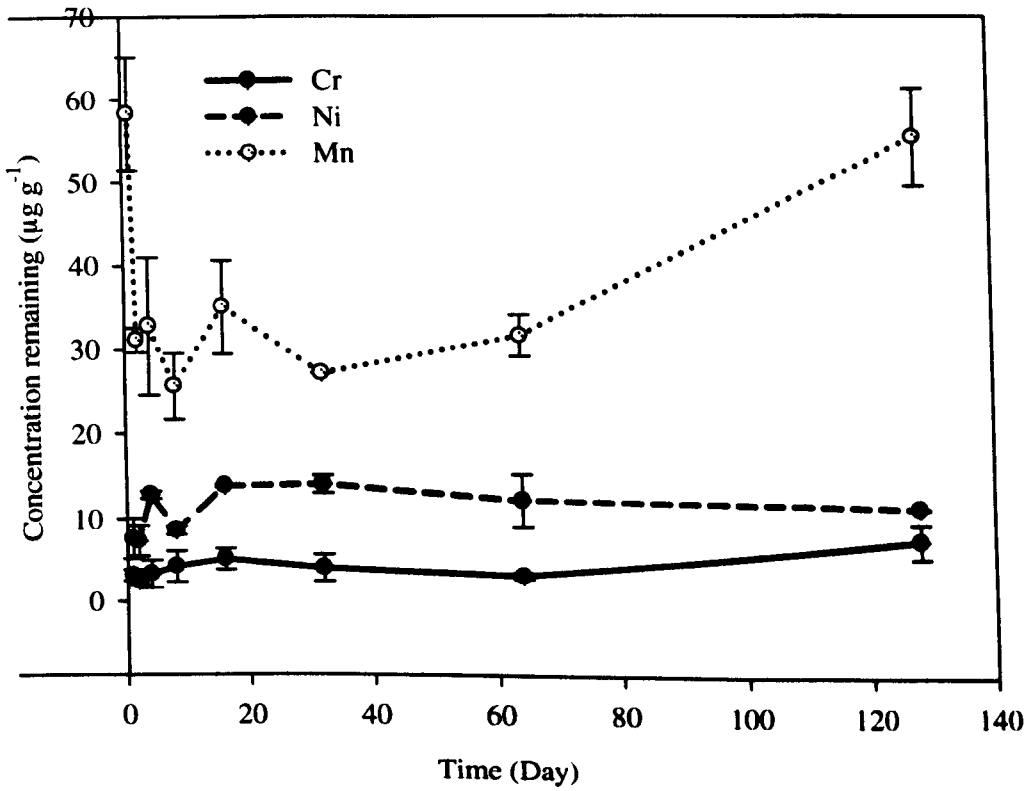

Figure 7.5 Changes in Cr, Ni and Mn metal concentrations (µg g^{-1}) in decomposing litter in a mangrove stand in Yanbu region, Saudi Arabia (error bars are standard deviations).

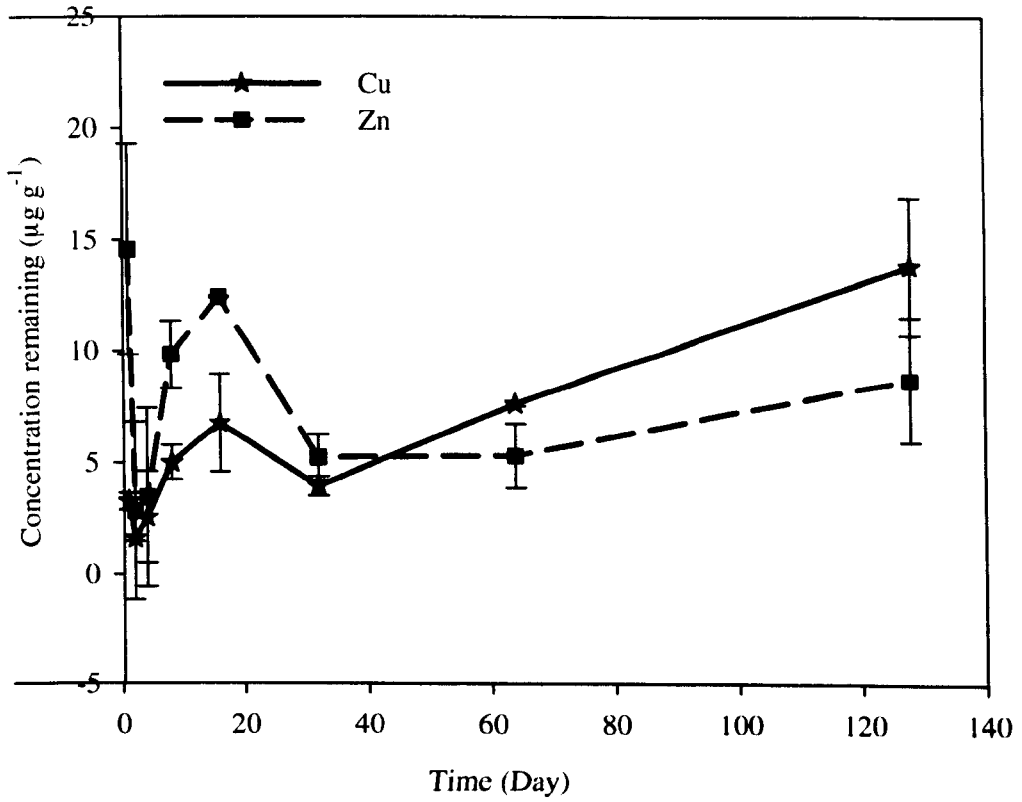

Figure 7.6 Changes in Cu and Zn metal concentrations (µg g^{-1}) in decomposing litter in a mangrove stand in Yanbu region, Saudi Arabia (error bars are standard deviations).

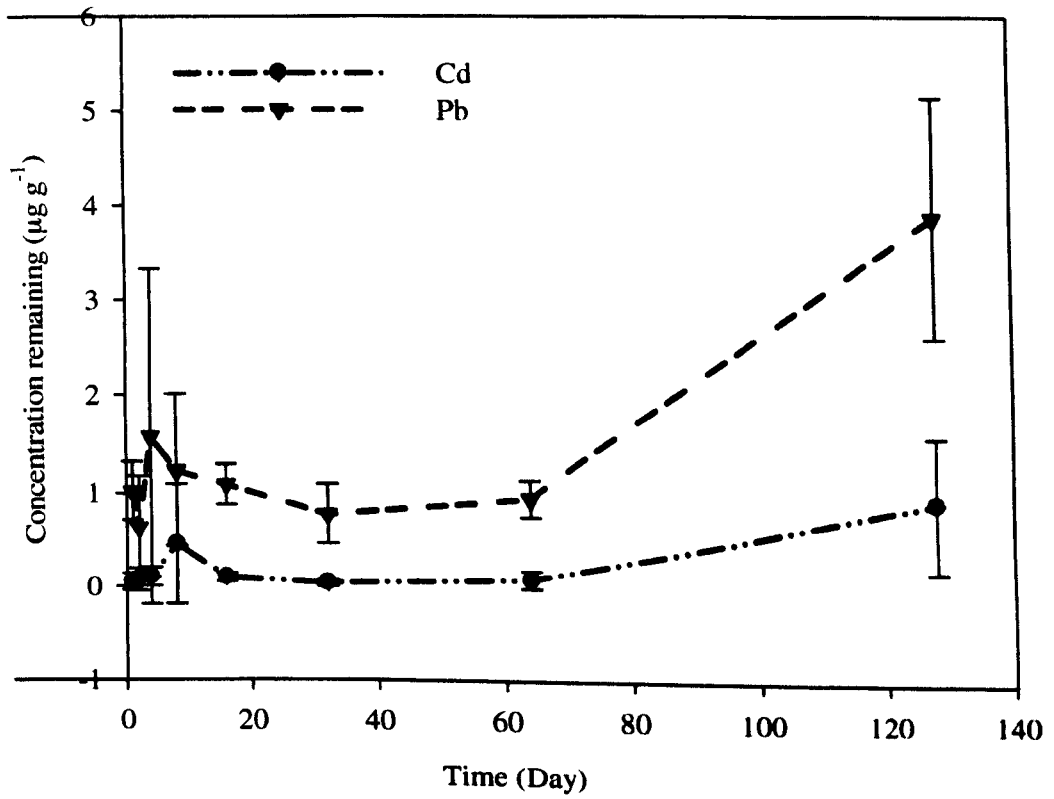

Figure 7.7 Changes in Cd and Pb metal concentrations (µg g^{-1}) in decomposing litter in a mangrove stand in Yanbu region, Saudi Arabia (error bars are standard deviations).

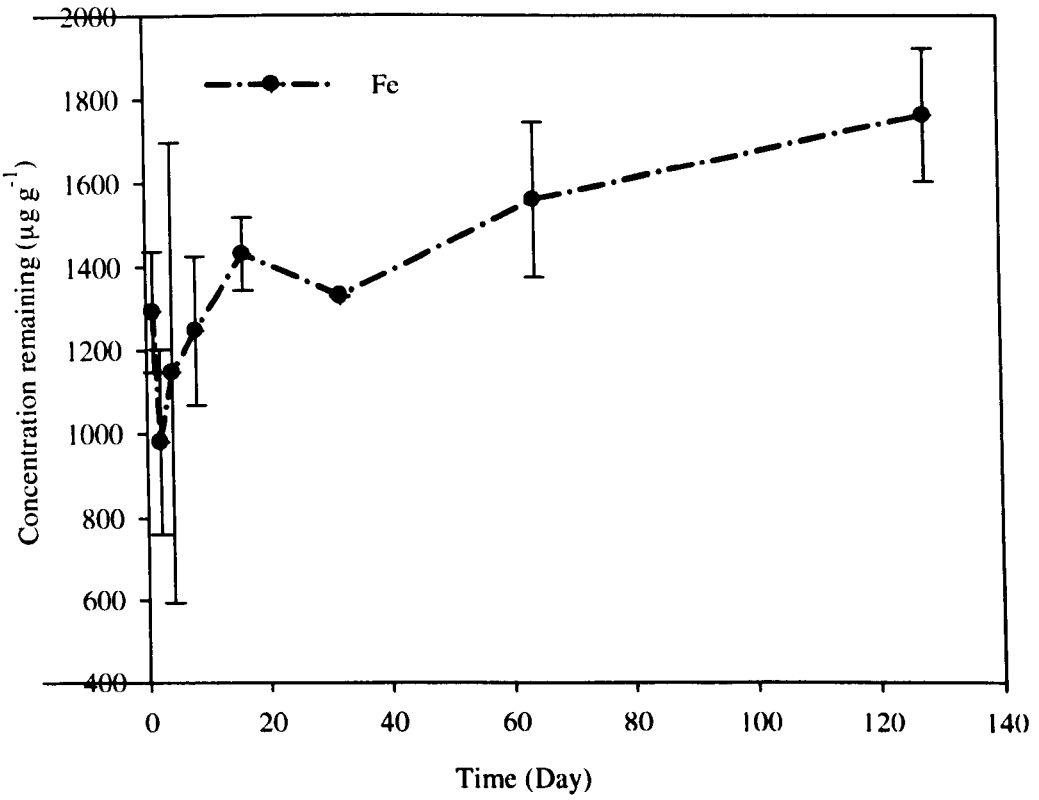

Figure 7.8 Changes in Fe metal concentration (µg g^{-1}) in decomposing litter in a mangrove stand in Yanbu region, Saudi Arabia (error bars are standard deviations).

7.3.4 Heavy metal dynamics in mangrove systems

In Shuaiba mangroves, the heavy metal stocks decreased from root to canopy following the order: fine roots > aerial roots > wood > leaves. Fine root biomass represented the main stock of most of the heavy metals, roots accounted for the main stocks of Cr (2.9), Mn (6.5), Fe (377), Ni (3.2), Cu (3.9), Zn (1.8), and Pb (0.5) kg ha^{-1}, the Cd biomass was low in all tree components with values less than 0.01 kg ha^{-1} for all components (Table 7.8). The heavy metal input through litterfall biomass accounts for up to 1.8% of the metal stocks of the perennial biomass. The stocks of the different metals in the litterfall were: Cr (0.01), Mn (0.12), Fe (1.8), Ni (0.04), Cu (0.008), Zn (0.002) and Pb (0.001) kg ha^{-1} while Cd did not account for any amount in litterfall (Table 7.8).

Similarly in Yanbu, the main stocks of heavy metals were also found in roots while the lowest stocks were found in leaves. The heavy metal stocks in root biomass were: Cr (0.5), Mn (4.1), Fe (309), Ni (1.1), Cu (1.2), Zn (0.7), Cd (0.01) and Pb (0.3) kg ha^{-1} while metal stocks in leaf biomass were Cr (0.005), Fe (0.78),

Ni (0.04), Cu (0.01), and Pb (0.001) kg ha^{-1}, while Zn and Cd did not account for any stocks in leaves. Only for Mn were stocks in leaves higher than those in woody biomass (0.14 vs. 0.05 kg ha^{-1} for leaves and woody biomasses respectively) (Table 7.9). The heavy metal stocks in litterfall biomass accounts for up to 3.3% of the metal stocks in the perennial biomass. The metal stocks of the litterfall were: Cr (0.006), Mn (0.12), Fe (1.1), Ni (0.04), Cu (0.005), Zn (0.002), and Pb (0.001) kg ha^{-1} while Cd did not account for any stocks in litterfall (Table 7.9).

In general the main stocks of metals in Shuaiba were found in sediment followed by the perennial biomass; a minimal amount of metals is transferred to the sediment through litter input and removed from the forest floor (Figure 7.9). Metals stocks in sediments were 306, 2155, 71923, 908, 159, 125, 1.4 and 20.3 kg ha^{-1} for Cr, Mn, Fe, Ni, Cu, Zn, Cd and Pb respectively. The biomass stocks of those metals were 3.0, 6.7, 386, 3.3, 4.0, 1.8, 0.04 and 0.5 kg ha^{-1} respectively. The metal inputs through litterfall accounts for less than 2% of the metal stock in perennial biomass, the inputs of metals were 0.01 (0.2%), 0.12 (1.8%), 1.80 (0.5%), 0.05 (1.5%), 0.01 (0.2%), 0.002 (0.1%), 0.0001 (0.25%) and 0.001 (0.2%) kg ha^{-1} y^{-1} for Cr, Mn, Fe, Ni, Cu, Zn, Cd and Pb respectively. After 127 days (residence time) the amount of metals that are annually removed from the forest floor is negligible (less than 1% of the metal input) while almost all of annual metal input is stored in the sediment. The annual metal stocks of the removed litter were 4×10^{-6}, 3×10^{-5}, 2.1×10^{-3}, 1.2×10^{-5}, 1.5×10^{-5}, 1.0×10^{-6}, 1.0×10^{-7}, and 2×10^{-6} kg ha^{-1} y^{-1} for Cr, Mn, Fe, Ni, Cu, Zn, Cd and Pb respectively (Figure 7.9). Moreover, the turnover of the different heavy metals in Shuaiba mangrove system were 470, 54, 214, 67, 475, 860, 280 and 398 years for Cr, Mn, Fe, Ni, Cu, Zn, Cd and Pb respectively.

In Yanbu, the main stocks of metals in the sediments were 869, 9945, 513220, 1872, 1086, 969, 18.6, and 308 kg ha^{-1} for Cr, Mn, Fe, Ni, Cu, Zn, Cd and Pb respectively. The perennial biomass stocks were 0.6, 4.6, 315.1, 1.2, 1.4, 0.7, 0.02, and 0.3 kg ha^{-1} respectively. The metal input through litterfall accounts for less than 3.5% of the perennial biomass metal stocks with values of 0.01 (0.94%), 0.12 (2.7%), 1.1 (0.34%), 0.04 (3.3%), 0.005 (0.37%), 0.002 (0.27%), 5×10^{-5} (0.25%) and 0.001 (0.33%) kg ha^{-1} y^{-1} of the biomass metal stocks for Cr, Mn, Fe, Ni, Cu, Zn, Cd and Pb respectively (Figure 7.10).

Table 7.8 Biomass and heavy metal stock in kg ha^{-1} (± SEM) of *A. marina* in Shuaiba region, Saudi Arabia

Components	Biomass (kg ha^{-1})	Heavy metals (± SEM)							
		Cr	Mn	Fe	Ni	Cu	Zn	Cd	Pb
Leaf litter	2898	0.006 (0.0)	0.123 (0.02)	1.80 (0.06)	0.05 (0.01)	0.01 (0.0)	0.002(0.0)	0.0(0.0)	0.001(0.0)
Leaves	3140	0.014 (0.0)	0.03 (0.0)	0.12 (0.03)	0.02 (0.01)	0.01 (0.0)	0.01 (0.0)	0.00 (0.0)	0.001 (0.0)
Branch	7270	0.001 (0.0)	0.03 (0.01)	1.45 (0.3)	0.01 (0.0)	0.03 (0.01)	-	0.003 (0.0)	0.005 (0.0)
Stem	8470	-	0.03 (0.0)	2.29 (0.4)	0.02 (0.0)	0.03 (0.01)	-	0.002 (0.0)	0.003 (0.0)
Aerial root	23700	0.103 (0.01)	0.09 (0.0)	2.72 (0.7)	0.10 (0.04)	0.08 (0.02)	0.04 (0.02)	0.002 (0.0)	-
Fine root	96420	2.89 (0.4)	6.49 (0.3)	377.29 (20.0)	3.19 (0.2)	3.86 (0.4)	1.76 (0.1)	0.03 (0.0)	0.48 (0.01)

SEM: standard error of means

Table 7.9 Biomass and heavy metal stock in kg ha^{-1} (± SEM) of *A. marina* in Yanbu region, Saudi Arabia

Components	Biomass (kg ha^{-1})	Heavy metals in kg ha^{-1} (± SEM)							
		Cr	Mn	Fe	Ni	Cu	Zn	Cd	Pb
Leaf litter	1861	0.006 (0.0)	0.12 (0.01)	1.08 (0.3)	0.04 (0.0)	0.005 (0.0)	0.002 (0.0)	0.00 (0.0)	0.001 (0.0)
Leaves	2090	0.005 (0.0)	0.14 (0.01)	0.78 (0.1)	0.04 (0.0)	0.014 (0.01)	-	-	0.001 (0.0)
Branch	5070	0.002 (0.0)	0.03 (0.0)	1.24 (0.2)	0.017 (0.0)	0.028 (0.0)	0.016 (0.0)	0.001 (0.0)	0.003 (0.0)
Stem	3600	0.001 (0.0)	0.023 (0.0)	1.02 (0.2)	0.008 (0.0)	0.018 (0.0)	-	0.001 (0.0)	0.002 (0.0)
Aerial root	10100	0.120 (0.03)	0.30 (0.1)	2.52 (0.8)	0.04 (0.01)	0.068 (0.0)	0.079 (0.0)	0.003 (0.0)	0.005 (0.0)
Fine root	39120	0.51 (0.02)	4.11 (0.2)	309.52 (48.98)	1.14 (0.1)	1.232 (0.1)	0.657 (0.0)	0.013 (0.0)	0.286 (0.1)

SEM: standard error of means

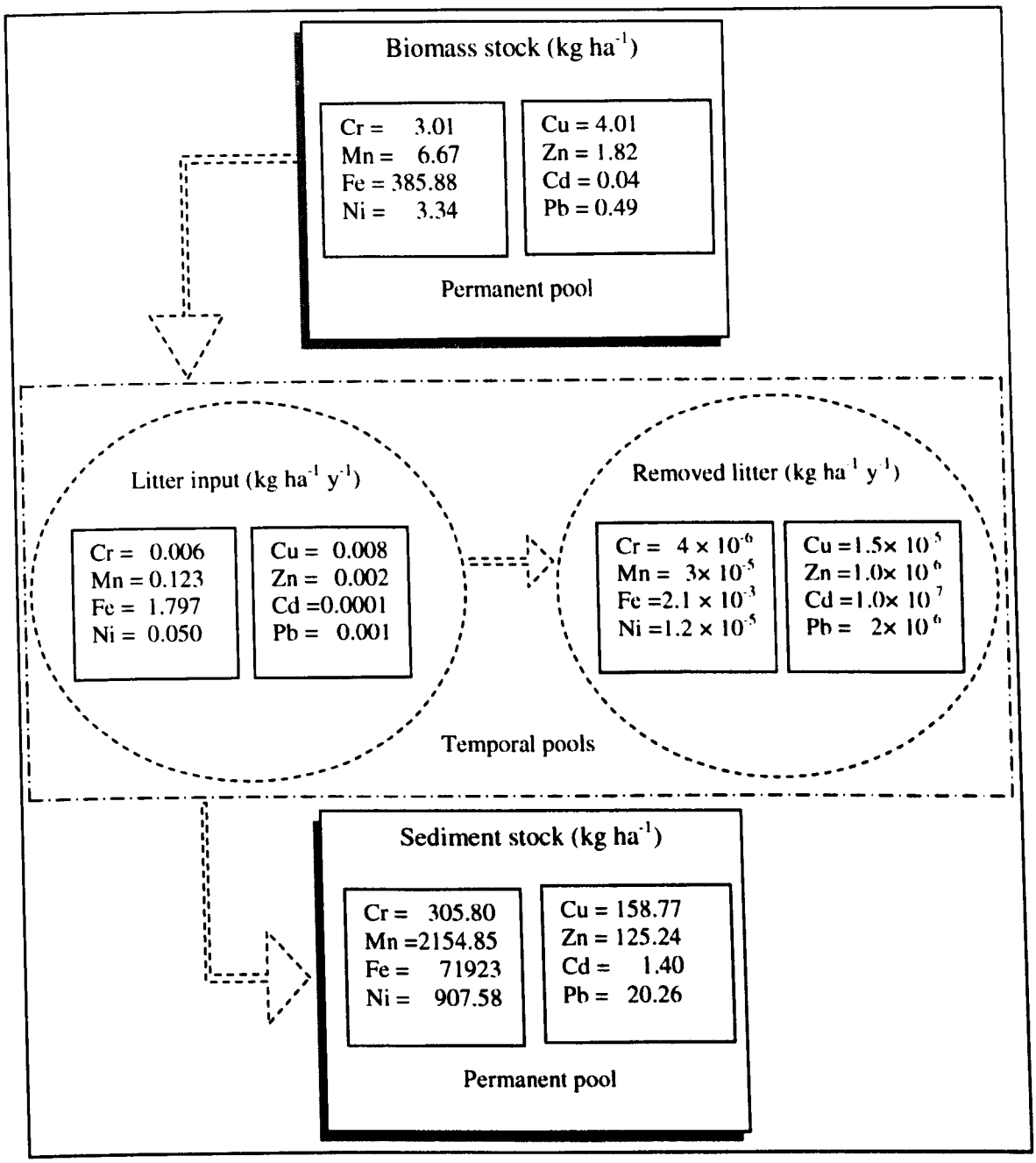

Figure 7.9 Heavy metal dynamics in *A. marina* stands in Shuaiba region, Saudi Arabia.

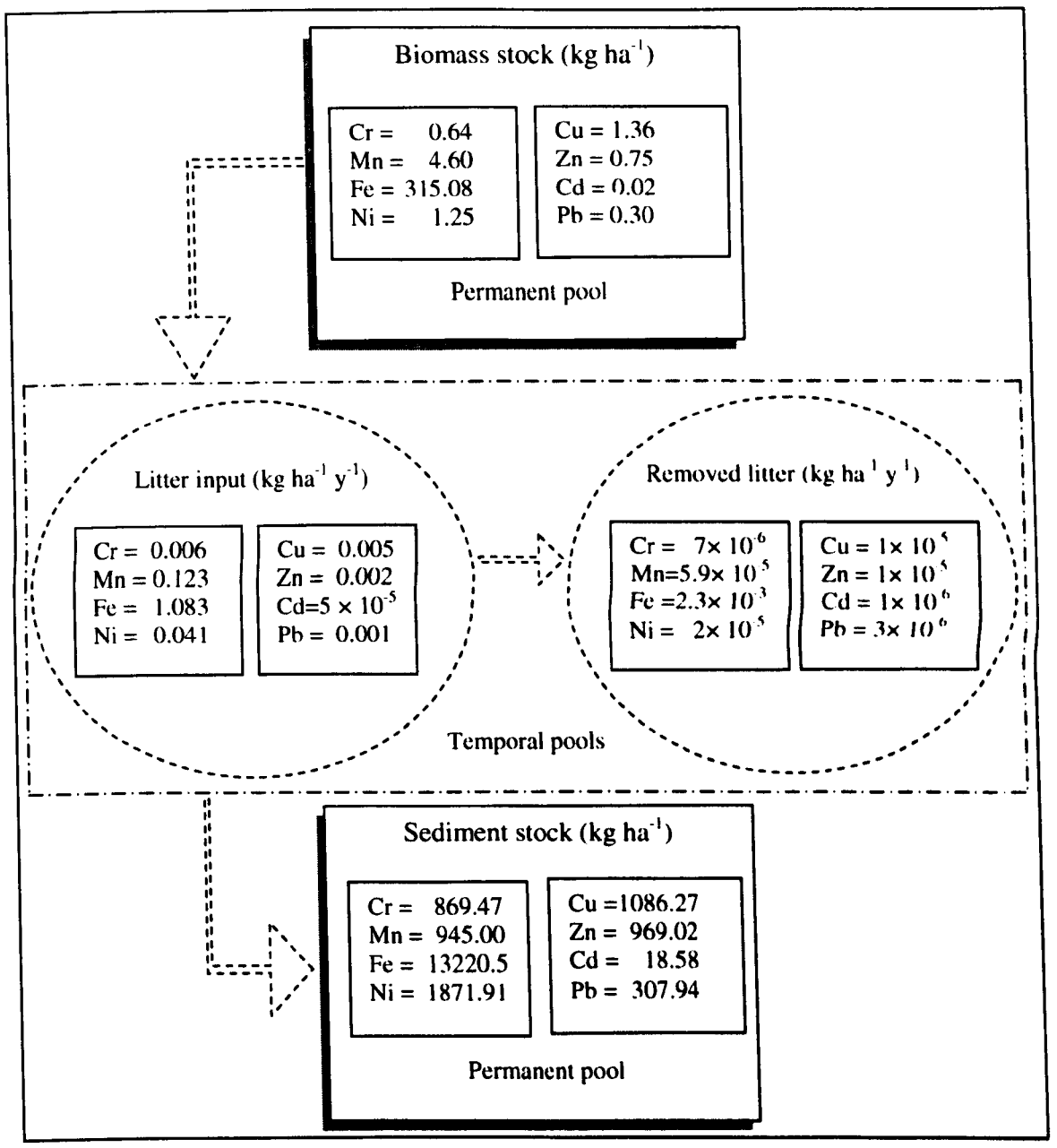

Figure 7.10 Heavy metal dynamics in *A. marina* stands in Yanbu region, Saudi Arabia.

Similar to Shuaiba, almost all the annual input of heavy metals is stored in the sediment and the amount that is removed after 97 days of residence time is negligible (less than 1%); the annual metal stocks in the removed litter were 7×10^{-6}, 5.9×10^{-5}, 2.3×10^{-3}, 2×10^{-5}, 1×10^{-5}, 1×10^{-5}, 1×10^{-6}, 3×10^{-6} kg ha^{-1} y^{-1} for Cr, Mn, Fe, Ni, Cu, Zn, Cd and Pb respectively(Figure 7.12). The turnover of the different heavy metals in Yanbu site were 108, 37, 291, 30, 249, 304, 391 and 278 years for Cr, Mn, Fe, Ni, Cu, Zn, Cd and Pb respectively

7.4 DISCUSSION

7.4.1 Heavy metal accumulation in Shuaiba mangroves system

Mangrove trees have a remarkable ability to tolerate environmental stresses such as pollution which is attributed to the ability of the plant to filter or regulate the uptake of excess amount of metals in the sediment. *Avicennia* species are unique for their root system that comprise cable roots with perpendicular anchoring roots downward and aerial roots extending above sediment level. This root system can efficiently trap clay and silt particles and increase sedimentation. Fine roots extending horizontally from aerial roots and forming a dense medium just below the sediment surface have great efficiency in absorbing surface nutrients and precipitated metals (Harbison, 1986); in fact, previous investigation found strong relationships between metal enrichment of sediment and metal transport rates through fine roots (Wollast, 1982).

In the current findings, root heavy metal concentrations were much higher than those of the sediment which may reflect the continuous accumulation and filtration of metals in roots over a long period of time. The high metal concentration in roots may also indicate high bioavailability of metals in the sediment. In *Avicennia* species, air that is absorbed through aerial root lenticels can be transferred to the rhizosphere, oxidising the anaerobic sediment at the fine root level, reducing sulphide precipitation and lowering stability of iron plaque, thus allowing more metal exchange (Lacerda *et al.*, 1993). In a study of metal accumulation in various sediments, MacFarlane *et al.*, (2003) noted an increase in Zn accumulation in roots in low pH sediments. Similarly, Clark *et al.*, (1998) noted increases in metal exchange under low pH and oxidised (Eh >+100mV) conditions which attract the metal ions to the negative charges of the sediment particles. Although the current sediment pH was neutral (pH = 7.4; Chapter 1) it may not reflect the real pH status since the temporal and seasonal changes in sediment pH were not considered in the current investigation. The sediment Eh is another important parameter that governs metal binding (Guo *et al.*, 1997), which was also not measured here. Periodic and seasonal pH and Eh measurements are needed in conjunction with metal accumulation conditions to accurately establish relationships between these significant chemical characteristics and sediment metal accumulation. In addition, the sediment physical properties also influence metal accumulation. Fine particle

sediment with high proportions of clay and silt tend to accumulate more metals than those with lower ratios (Harbison, 1986). Fine-grain sediments with its high specific surface area can trap heavy metals and act as a substrate for organic matter which in turn can form complexes with heavy metals (Prasad and Strzalka, 2002; Marchand *et al.*, 2006). Tam and Wang (2000) investigated the spatial variation of heavy metals in fine-grain and sandy sediments and found higher metal concentrations in finer-grained sediments than in sandy sediments. In Shuaiba, the clay and silt ratios are low in the sediment (sandy loam) thus the physical characteristics of Shuaiba sediment might have also contributed to the lower metal accumulation.

Although roots had high concentrations of metals, the translocation of metals into leaves was low. *A. marina* trees accumulated most of the absorbed metals at the fine root level and reduced translocation of concentrated metal to leaves to the minimum. Leaf translocation factors of the different metals were less than 0.25 and Zn translocation to leaves was the highest among the rest of metals reflecting its role as an essential micro-nutrient in plant function. Zn is also an important element for carbohydrate and protein metabolism (Kabata-Pendias and Pendias, 1984; Ross, 1994), this is in agreement with a number of studies (*e.g.* Alongi *et al.*, 2003, MacFarlane *et al.*, 2003; Peng *et al.*, 1997; Saifullah *et al.*, 2004 and Zheng and Lin, 1996) who reported metal concentrations in leaves to be equal to, or less than half, that of the roots with Zn having the highest translocation.

Leaf litter in Shuaiba lost 52% of its mass in the first 64 days attributed to leaching of soluble carbohydrates (Chapter 5). Generally, the metal concentration in the litterfall increased followed by a decrease at the end of the decomposition period. The changes in the heavy metal concentrations in the decomposing leaves can be affected by a combination of leaching losses, accumulation via absorption and microbial immobilization which in turn depends on the CEC, pH and Eh conditions (Schierup and Larsen, 1981). The concentrations of Mn, Zn, and Cu increased until day 64 and then decreased by day 128, this increase in concentration may be due to the low tidal levels at the period of sample collection (August) when the tidal levels were at the lowest of the year accompanied by high evaporation in the top soil thus reducing leaching from litter. This period was followed by steady leaching until day 128 (October) when tidal levels where higher. After 127 days (residence time) the concentration of the heavy metals in the removed litter is low indicating that litter

that is removed from the forest floor contained lower metal concentration than those in the litterfall.

The Shuaiba mangrove system stored the largest quantities of heavy metals in the sediment, the relatively long residence time of litter on the forest floor (approximately four months) facilitated annual input and incorporation into the sediment. This long residence time is a result of low and infrequent tidal levels accompanied by low crab activities (Chapter 5). Such long residence time is found in similar basin mangrove systems with low tidal activities (Twilley *et al.*, 1986). Although litter stays long on the forest floor, the annual input of the heavy metals is low (less than 1.9%) suggesting that metal input is likely to be coming from a more enriched, high biomass component (*e.g.* fine roots). It can also indicate that annual input through litterfall is accumulated in the sediment over a long period of time. In addition, the high turnover rate of the different metals can indicate metal sequestration in permanent tree components. Nonetheless, the mangrove trees minimized metal translocation to the upper plant components by storing more than 84% of the accumulated metals in fine roots. After 127 days of residence time, the litterfall that was removed from the mangrove forest floor contained very small amount of heavy metals. Thus the dynamics of heavy metals in the Shuaiba mangroves suggest that the system functions as a sink for heavy metals with negligible export to adjacent systems.

4.7.2 Heavy metal accumulation in Yanbu mangroves system

In Yanbu, it was not surprising to find that sediments contained higher metal concentrations than in Shuaiba. The significantly higher Mn, Fe, Cu, Zn and Pb concentrations in the sediment can indicate higher exposure to these metals from the industrial effluences in Yanbu than in Shuaiba. Generally, the fine roots contained higher metal concentrations than sediment which can also be the result of prolonged root accumulation; low proportions of fine grains may also have contributed to the low metal concentration in the Yanbu sediment. Although the metal concentration in fine roots were higher than those of the sediment, the accumulation in fine roots was lower than in Shuaiba; Yanbu had much lower RCF and LCF ratios compared to Shuaiba which indicates efficient exclusion of metals at the root level.

Except for Ni, the translocation of heavy metals from roots to leaves was generally low with translocation factors of less than 0.65. Nickel concentration was present at levels exceeding the normal concentrations in leaves although Ni concentrations in sediment were lower than frequently reported for contaminated sites (Kabata-Pendias and Pendias, 1984). Nickel is a common oil-related metal that can be present in high concentration in polluted areas and the high Ni concentration in the surface sediment may indicate precipitation of this metal from anthropogenic sources on the sediment surface. Although metal concentration was higher than those in Shuaiba, the concentrations were lower than those previously reported for Yanbu sediments (Hashem, *et al.*, 1993).

The leaf litter in Yanbu lost 44% of its mass in the first 64 days of decomposition (Chapter 5). During decomposition, the concentrations of all heavy metals generally increased to levels higher than those of the freshly fallen leaves with Fe representing the highest concentration. Similar enrichment in decomposing litter of Fe, Mn Cu and Pb metals was previously reported (Larsen and Schierup, 1981; Killingbek *et al.*, 1982). The increase in metal concentrations might be due to the precipitation and absorption of the metal ions, reduced ions under anoxic condition may be liberated from anoxic condition and oxidized (by the influence of the more frequent tidal levels) which then might be absorbed by litter particles (Lacerda *et al.*, 1993 and Lacerda, 1998). In addition, the tidal waters containing organic particles are negatively charged waters that can attract heavy metal cations which in turn can be absorbed by litter via ion-exchange (Silva *et al.*, 2006). The residence time of litter on the forest floor in Yanbu (97 days) was shorter than that at Shuaiba and may be influenced by a higher tidal levels in Yanbu (88 cm) than in Shuaiba (63 cm). In addition, the natural setting of the fringe Yanbu mangroves allow tidal water to inundate the site more frequently than in the Shuaiba basin mangroves which can contribute to the shorter residence time.

Similar to Shuaiba, the sediment constitutes the main metal stock in the system, while the mangrove trees represent the smaller metal stock in the system with less than 3.5% of metal input to the sediment through litterfall. The mangrove trees stored more than 72% of the accumulated metals in fine roots minimizing translocation to the upper plant components. Moreover, the turnover rates of the different metals were as high as those in Shuaiba and indicate metal sequestration in

the mangrove tree components. After 97 days, the leaves that are removed from the forest floor contain higher metal concentration than those of the freshly fallen leaves however, the amount of metals that are annually removed from the forest floor is negligible suggesting that most of the metal input is incorporated into the sediment.

4.7.3 Heavy metal accumulation in Red Sea mangrove systems

Generally, heavy metals in mangrove trees were accumulated most in fine roots, leaves accounts for the least metal concentrations which contributed to the low metal input through litterfall. When metal concentrations in leaves were compared to those reported in the literature (Table 7.10), it was found that the metal concentrations in the leaves of the current study were below those reported thus, there were lower annual input of metals to the sediment. The low metal input through litterfall suggest that a significant amount of metal input to the system came through the decomposition of fine roots which contain higher metal concentration than other plant components. In addition, fine roots are known for their high turnover rate and incorporation into soils, this process is normally slow in anoxic conditions as a result of the slow decomposition and low microbial activities. The slow decomposition of the metal-enriched roots can release a significant amount of highly concentrated metals back into the sediment. Therefore it is of interest to investigate the contribution of metal input through fine roots into the sediment pool. To the best of my knowledge, such an estimate is not present in the published literature possibly due to the difficulties associated with methodological application, labour and time constraints. However, such an estimate would aid in assessing the role of fine roots in the heavy metal dynamics in the mangrove systems. The annual import of metals via tidal activities is out of the scope of the current study however, it could be a major metal input especially when industrial discharges are present. Thus further investigation of the heavy metal import rates are needed in order to fully assess the sources of heavy metal inputs into the sediment.

The litter that is being removed from the forest floor contains negligible amounts of heavy metals, the low tidal ranges in the mangrove systems limit the movement of the litter to within the mangrove system with no expected export (Chapter 4) thus no metal export is expected to occur from the mangrove systems.

The sediment is an important pool in mangrove systems because of its ability to retain heavy metals, and it is frequently reported in literature as an indicative of a system's status and health. The sediment heavy metals of the current study were comparable with or below those reported for mangrove sediments (Table 7.11). When the concentrations compared to those of the same region (*e.g.* Shridah, 1998; Sadiq and Zaidi, 1994; Hashem, 1993) it was also found that concentrations were less than previously reported. In Yanbu, with the exception of Fe, the sediment concentrations were lower than those reported in Hashem (1993), while only Mn, Fe, Ni and Cu had higher concentrations than those reported in Sadiq and Zaidi (1994). From these findings it can be concluded that the mangrove systems in the Shuaiba and Yanbu regions are clean compared to other polluted sites around the world. In Yanbu, several violations of industrial discharges have been reported (*e.g.* Paimpillil *et al.*, 2002; Ahmad *et al.*, 2008). Ahmad *et al.*, (2008) mentioned that many types of pollutants are discharged into coastal areas in excessive concentrations such as hydrocarbon compounds and thus present another form of pollution for Yanbu mangroves. In addition other organic and thermal pollutants were reported to significantly affect mangrove systems in other parts of the Red Sea coast (*e.g.* Mandura 1997; Aleem 1990). Thus investigation of possible contamination, runoff from domestic and industrial sewage from populated areas on the Red Sea coast is recommended in order to assess the possible contamination in the mangrove systems on the Red Sea coast.

7.5. CONCLUSIONS

The Shuaiba mangroves contain relatively lower concentrations of heavy metals than Yanbu due to minimal exposure to anthropogenic sources. Although exposed to pollutants, the heavy metal concentrations at Yanbu were below the concentrations commonly reported for other polluted sites. The current findings suggest that the mangrove systems investigated are not under heavy metal pollution pressure although metals are accumulated within the mangrove systems. However, further investigation of possible anthropogenic contamination in other mangrove stands is crucially important for assessing the environmental health of the individual mangrove systems on the Red Sea coast. In addition, it is equally important to

investigate the sources of pollution in the mangrove systems, violations of industrial discharge regulations and to initiate environmental monitoring programs.

Table 7.10 Comparison of heavy metal concentrations (μg g^{-1}) of leaves in mangrove systems from different publications

Source	Location	Species	Metal concentration (μg g^{-1})							
			Cr	Mn	Fe	Ni	Cu	Zn	Cd	Pb
Current study	Saudi Arabia	*A. marina*	2.4-4.5	11.1-65.9	39.5-375.4	6.7-21.1	4.2-6.8	0-4.23	0-0.01	0.22-0.57
Saifullah *et al.* (2004)	Pakistan	*A. marina*	-	-	309.8	-	-	-	-	-
MacFarlane *et al.* (2003)	Australia	*A. marina*	-	-	-	-	0.14	0.38	-	0.02
Sadiq and Zaidi (1994)	Saudi Arabia	*A. marina*	-	-	-	-	4.4	11	-	6.9
Defew *et al.* (2005)	Panama	*Laguncularia racemosa*	3.7	-	-	-	-	35.8	-	6.2

Table 7.11 Comparison of heavy metal concentrations (μg g^{-1}) in mangrove sediments from different publications

Source	Location	Species	Metal concentration (μg g^{-1})							
			Cr	Mn	Fe	Ni	Cu	Zn	Cd	Pb
Current Study	Saudi Arabia	*A. marina*	8.7-11.7	61.3-156.7	1810-8373	27.4-24.8	4.2-13.6	2.8-13.6	0.02-0.2	0.5-3.9
Awal *et al.* (2009)	Sundarbans-(Bangladesh)	Multi species	15.7	436.8	173890	76.1	10.5	73.6	0.55	19.3
Zöckler and Bunting (2006)	Bangladesh	-	19.5-46.1	-	-	15.9-44.6	6.9-31.6	24.3-76.0	0.01-0.006	8.0-15.7
MacFarlane *et al.* (2003)	Australia	*A. marina*	-	-	-	-	26.8-196.5	133.0-386.1	-	53-177.8
Shridah (1998)	UAE	*A. marina*	10.2-14.1	39.9-111	-	18.0-76.3	6.3-9.2	9.0-13.2	4.7-5.2	20.4-37.3
Sadiq and Zaidi (1994)	Saudi Arabia	*A. marina*	2.4-12.7	2.0-69.3	645-3600	1.9-18.1	0.14-3.8	2.2-17.1	0.2-3.0	6.1-18.8
Hashem (1993)	Yanbu-Saudi Arabia	*A. marina*	-	-	43.1	-	31	13.3	2.3	26.1
IUCN/MEPA (1987)	Pakistan	*A. marina*	1.5-8.6	-	-	-	2.3-14.7	-	0.04-0.15	3.4-15.6

Ranges of the current study are for Shuaiba andYanbu respectively; single values are mean concentrations.

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CHAPTER 8

GENERAL CONCLUSION AND RECOMMENDATIONS

8.1 RESEARCH FINDINGS

The past few decades have witnessed a global leap in knowledge and awareness of the ecological, environmental, economic and social significance of mangrove systems. This has resulted in national and international conservation, management and replantation programs which translated into lowering the degradation and area loss of mangroves worldwide (Spalding *et al.*, 2010). However, the lack of the information on the status of mangrove systems in many parts of the world has left many mangroves stands unprotected, thus leading to further deterioration; such a gap in knowledge has prevented the inclusion of the majority of mangrove stands along the eastern Red Sea coast in national marine protected area programs.

The current investigation intended to provide baseline ecological information of the mangrove systems on the Red Sea coast of Saudi Arabia that would significantly aid in developing conservation, management and rehabilitation plans through providing site specific ecological and environmental information on the status of the mangrove systems. This was done by estimating productivity (standing and litterfall biomass), nutrient cycling and heavy metal pollution of two mangrove stands along the Red Sea coast.

The biomass production (standing and litterfall) of the Red Sea mangroves is low but comparable to estimates in similar arid and temperate regions of the world. Site specific biomass estimations in other locations on the Red Sea (especially those in the southern region) are further required in order to obtain an accurate overall biomass estimation for the Red Sea as a whole. The allometric equations developed for *A. marina* mangroves in the current study will facilitate measurements of mangrove biomass in other locations along the Red Sea coast to obtain a more

accurate overall productivity estimate. *In addition, the allometric* equations will significantly simplify future monitoring of annual biomass production. Moreover, these findings and future monitoring will be useful for national and regional organizations (*i.e.* MEPA, PERSGA) in updating regional mangrove biomass estimates and therefore will be useful for international organizations (*i.e.* ISME, WCMC) in updating global estimates.

The nutrient input through litterfall decomposition is low. The slow litter decomposition may have a long term significance rather than short term, as slow decomposition means that more material will be sequestered in sediment as peat. Such slow decomposition will serve as conserving mechanism and a long term source of nutrients. However, the nutrient cycling of other sites such as in the southern region is probably more dynamic than other parts of the Red Sea and mangroves are likely to have a more significant influence on adjacent systems. In addition, nutrient export to adjacent waters in such system is expected to be higher (Crossland *et al.*, 1987). The current mangrove systems does not appear to contribute to the energy source of the aquatic animals, however, it might be an important refuge and nursery habitat for commercial fish, crustaceans and other large animals (*i.e.* birds, mammals).

Pollution does not seem to be a problem in the sites studied. Although the exposed mangrove system had higher heavy metal accumulation than the minimally exposed system, concentrations were below those reported for polluted mangrove systems in other regions of the world.

The general findings of the current investigation can be summarized as follows:

1. Productivity of the Red Sea mangroves is low compared to global estimates but fall within figures reported for extreme environmental conditions.
2. The DBH and height were the best parameters to describe the aboveground biomass of mangrove trees and are recommended to be used in future estimations.
3. The mangrove systems constitute a closed system with a conserving and recycling mechanism and low export rate.

4. Mangrove detritus was not a major energy source for aquatic animals in the current study.
5. Heavy metal concentrations were lower than globally reported for polluted mangrove stands. However, stands that are exposed to anthropogenic pollutants accumulate higher metal concentrations in their tissues than those with minimal exposure.
6. Heavy metal concentrations were accumulated in roots to levels higher than those of the surrounding sediments and thus the mangrove roots can be employed as bio-indicators in monitoring heavy metal pollution in the aquatic environment.
7. Mangrove systems act as a sink for heavy metals through minimal metal input and export and through sequestration in sediment and fine roots.

8.2 RECOMMENDATIONS AND FUTURE RESEARCH

This research can be considered a baseline for further research on the Red Sea mangroves. Beside the research areas addressed in the current study, there are several other areas that need to be addressed to have an integrated database for mangrove ecosystems on the Red Sea coast of Saudi Arabia, This becomes very important when setting management and conservation plans. Future research objectives should address:

1. Area survey: A complete and updated area survey of mangroves on the Red Sea. This will be an essential baseline in assessing the level of deterioration, development and sustainability of mangroves.
2. Production estimation: Site specific production of mangrove systems can be assessed in conjunction with area survey plans. This would include the annual estimation of litterfall and aboveground biomasses; the allometric equations would facilitate fast and easy estimates of aboveground biomass.
3. Nutrient cycling and foodweb studies: The nutrient cycling of the mangroves differs depending on location and site condition. Thus site or regional-specific nutrient cycling assessment is needed. Studies that address energy sources and trophic interactions would aid in understanding the significance

of mangroves for the existence, biodiversity and breeding of commercial fish and crustaceans.

4. Carbon and nitrous oxide dynamics: the dynamics of green house gases is another legitimate research area that is lacking if not present at all in the region. This would include assessment of above and belowground carbon sequestration in mangrove trees, carbon sequestration in sediment, carbon fluxes in the system, and nitrous oxide emission from the mangrove sediment. Such research will help understanding the role of mangrove systems in reducing green house gas emissions.
5. Pollution in coastal habitats: Mangrove systems close by urban and industrial areas should be periodically monitored for pollution exposure. This can be done by assessing the metal concentration in sediment and mangrove fine roots. In conjunction, assessing the source of pollutants and violation of disposal regulations to sea waters.
6. Socio-economic value of mangrove systems: Studies that aims to survey the rate and form of local utilization of mangroves is encouraged as this utilization varies from region to another. This can help in setting goals for management plans. In addition, ecotourism is another type of projects that can both boost local economies and increase public awareness.
7. Public awareness: Spreading knowledge and understanding of the importance of mangrove ecosystems. One form of increasing public awareness is through the involvement of schools in rehabilitation and replanting projects.
8. Data sharing: Building national, regional and international communication networks with the different organization and institutions that can benefit from the available information.

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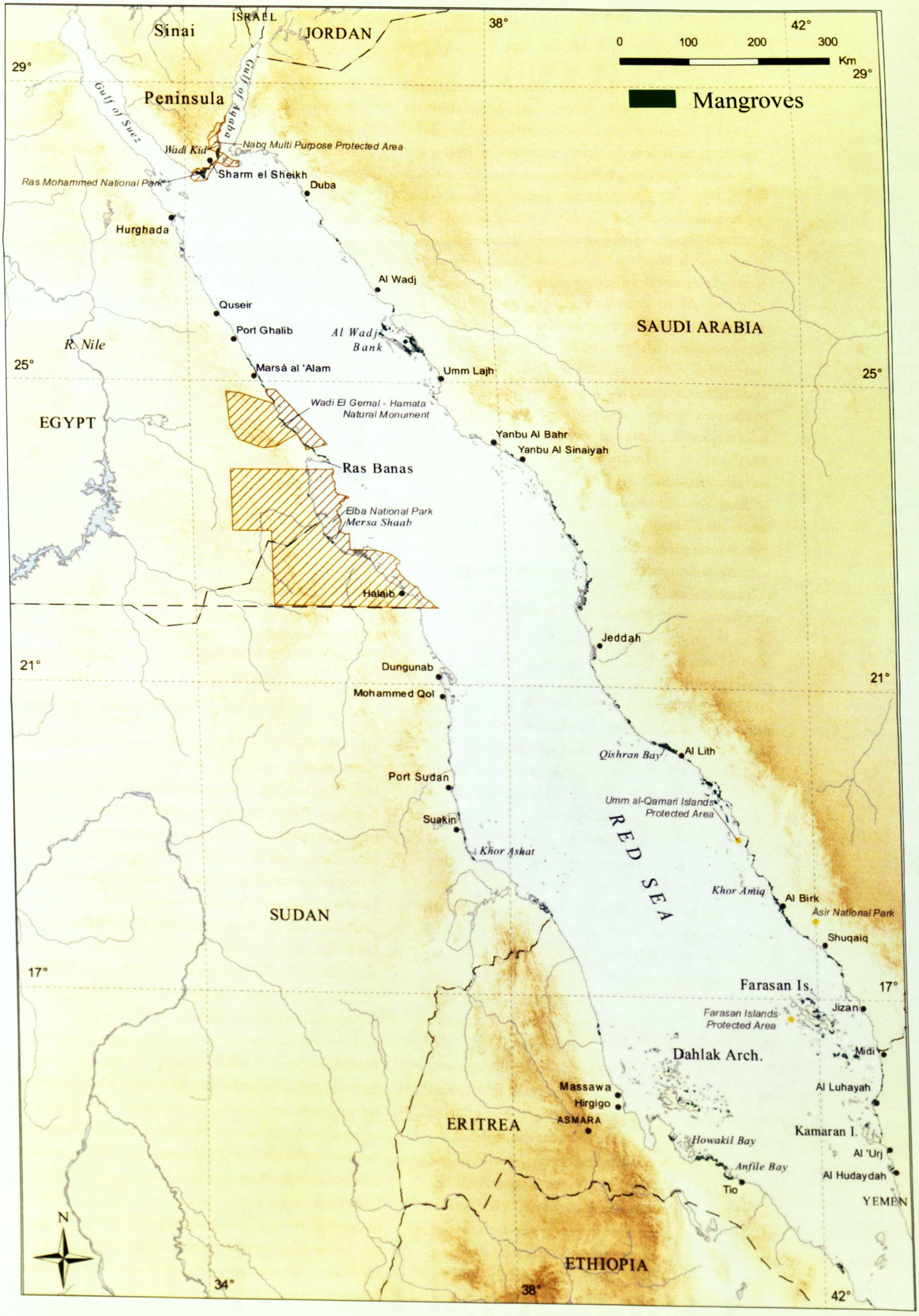
APPENDICES

Appendix I. Mangrove distribution on the Red Sea coasts, *Source: Spalding et al., (2010).*



Appendix II. Aboveground biomass sample size calculation using Stein's two stage sampling procedure:

$$n = \frac{1}{\frac{E^2}{t^2 s^2} + \frac{1}{N}}$$

Where: n = sample size

$E = 0.1 \text{dbh}\,\bar{x}$

S^2 = DBH variance

t = tabulated t value from the t table at 0.05 probability level (at ∞ =1.96)

N = total tree number in the pre-sampled population (120 tree)

Shuaiba site:

$\bar{x}$ = 16.70
S = 3.70
C.V = 22.18
N = 120 tree

Required sampling number (n) = **16 tree**

Yanbu site:

$\bar{x}$ = 3.54
S = 0.582
C.V = 16.47
N = 120 tree

Required sampling number (n) = **10 tree**

Appendix III. Fine root biomass estimation (t ha^{-1}) using soil cores

Core radius = 1.9 cm
Core height = 10 cm
Core surface area = 11.3 cm^2

Shuaiba site:

Fine root weight = 10.89 g $core^{-1}$
⇨ (10.89 /11.3)*10000 = 9637.16 g m^2
⇨ (9637.16 /1000000)*10000 = **96.37 t ha^{-1}**

Yanbu site:

Fine root weight = 4.42 g $core^{-1}$
⇨ (4.42/11.3)*10000 = 3911.5 g m^{-2}
⇨ (3911.5/1000000)*10000 = **39.11 t ha^{-1}**

2 site mean

Fine root weight = 7.66 g $core^{-1}$
⇨ (7.66/11.3)*10000 = 6778.7 g m^{-2}
⇨ (6778.7/1000000)*10000 = **67.78 t ha^{-1}**

Appendix IV. Monthly litterfall rates (kg ha^{-1}) in a mangrove stand in Shuaiba, Saudi Arabia

Month	2007/2008	2008/2009	Overall
June	453.39 ± 87.14	418.29 ± 37.15	435.84 ± 24.81
July	517.56 ± 53.24	139.68 ± 31.35	328.62 ± 267.20
August	168.38 ± 27.40	283.40 ± 77.60	225.95 ± 81.41
September	274.95 ± 53.08	111.64 ± 26.12	193.29 ± 115.48
October	150.19 ± 54.80	117.10 ± 44.65	133.64 ± 23.40
November	83.92 ± 30.80	246.14 ± 77.34	165.03 ± 114.71
December	95.00 ± 37.28	80.41 ± 34.70	87.70 ± 10.31
January	187.36 ± 76.87	291.46 ± 62.79	239.41 ± 73.61
February	276.00 ± 42.42	409.63 ± 96.08	342.81 ± 94.49
March	499.9 ± 103.91	408.92 ± 87.83	454.41 ± 64.33
April	536.19 ± 68.19	549.96 ± 75.05	543.07 ± 9.75
May	477.79 ± 98.01	512.90 ± 117	533.02 ± 78.11
Total	3720.63	3569.55	3682.81 ± 53.55

Appendix V. Monthly litterfall rates (kg ha^{-1}) in a mangrove stand in Yanbu, Saudi Arabia

Month	2007/2008	2008/2009	Overall
June	495 ± 217.01	584 ± 217.40	539.5 ± 62.93
July	196.6 ± 110.92	557.7 ± 232.14	377.15 ± 255.34
August	128.3 ± 84.29	155.7 ± 81.73	142 ± 19.37
September	233 ± 149.46	151.7 ± 86.35	192.35 ± 57.49
October	157.4 ± 104.69	140.4 ± 102.17	148.9 ± 12.02
November	177.72 ± 158.53	133.1 ± 82.25	155.41 ± 31.55
December	184 ± 258.12	327.6 ± 518.70	255.8 ± 101.54
January	341.19 ± 445.00	268.6 ± 189.88	304.90 ± 51.33
February	353.5 ± 292.22	337.5 ± 183.66	345.5 ± 11.31
March	364.62 ± 259.90	190.2 ± 75.73	277.41 ± 123.33
April	505.1 ± 234.22	463.38 ± 216.22	484.24 ± 29.50
May	400.46 ± 137.93	172.12 ± 146.27	286.29 ± 161.46
Total	3536.89 ±132.11	3482.0 ±165.23	3509.44 ± 38.81

Appendix VI. Litterfall and standing crop values (kg ha^{-1}) in a mangrove system in Shuaiba, Saudi Arabia

Month	litterfall	Standing crop
August 07	114.92 ± 42.41	108.33 ± 40.45
September 07	228.24 ± 134.91	440.55 ± 222.28
October 07	62.19 ± 37.42	75.68 ± 84.79
November 07	37.40 ± 24.08	18.85 ± 24.81
December 07	46.93 ± 57.27	20.54 ± 20.90
*January 08	140.40 ± 108.71	21.47 ± 10.32
*February 08	268.25 ± 129.53	28.96 ± 25.91
*March 08	458.89 ± 148.90	47.84 ± 34.48
*April 08	453.53 ± 219.33	58.51 ± 32.01
*May 08	512.10 ± 266.82	68.49 ± 39.58
*June 08	386.72 ± 111.48	39.84 ± 20.25
July 08	146.00 ± 71.29	150.37 ± 62.21
August 08	235.07 ± 109.70	207.38 ± 108.62
September 08	48.28 ± 20.44	50.62 ± 76.23
October 08	15.98 ± 12.83	21.25 ± 19.74
November 08	42.06 ± 43.67	17.08 ± 8.45
December 08	28.93 ± 12.56	22.14 ± 8.29
*January 09	185.92 ± 97.82	23.53 ± 10.94
*February 09	361.01 ± 154.65	64.91 ± 24.73
*March 09	406.62 ± 166.73	73.35 ± 21.67
*April 09	532.61 ± 226.06	121.45 ± 99.25
*May 09	510.33 ± 226.68	155.08 ± 88.87
*June 09	459.02 ± 266.53	144.13 ± 108.18
July 09	114.92 ± 42.41	108.33 ± 40.45
Average	247.02 ± 185.36	86.10 ± 93.96

*denote high tide levels.

Appendix VII. Monthly litterfall and standing crop values (kg ha^{-1}) in a mangrove stand in Yanbu, Saudi Arabia

Month	Litterfall	Standing crop
July 07	112.63 ± 115.22	71.61 ± 104.25
August 07	50.56 ± 38.15	54.82 ± 38.11
September 07	117.18 ± 80.45	106.95 ± 98.99
*October 07	86.08 ± 81.83	56.89 ± 40.08
*November 07	94.48 ± 95.06	9.30 ± 15.43
*December 07	75.73 ± 76.89	10.00 ± 5.26
January 08	104.34 ± 162.69	18.96 ± 9.79
*February 08	211.17 ± 223.70	42.06 ± 31.36
*March 08	251.01 ± 205.68	21.43 ± 9.51
*April 08	473.62 ± 200.40	32.80 ± 21.29
*May 08	338.87 ± 175.60	81.32 ± 34.97
*July 08	403.51 ± 201.05	73.75 ± 60.58
August 08	93.42 ± 48.43	95.01 ± 77.35
September 08	61.33 ± 26.99	43.98 ± 33.19
*October 08	62.09 ± 72.23	13.98 ± 13.08
*November 08	39.86 ± 38.77	21.07 ± 14.10
*December 08	93.58 ± 90.67	23.68 ± 11.89
*January 09	213.83 ± 170.50	46.41 ± 22.25
*February 09	311.77 ± 214.40	43.46 ± 23.03
*March 09	152.73 ± 102.88	43.83 ± 35.77
*April 09	298.99 ± 186.53	44.10 ± 25.89
May 09	75.92 ± 123.09	34.04 ± 38.80
Average	169.21 ± 125.29	44.98 ± 27.11

*denote high tide levels.

Appendix VIII. Percent mass remaining of leaf litter at different decomposition intervals in a mangrove stand in Shuaiba and Yanbu, Saudi Arabia

Day	% Mass remaining	
	Shuaiba	Yanbu
1	100	100
2	100.41 ± 1.46	99.35 ± 3.05
4	81.55 ± 0.94	97.31 ± 2.27
8	70.78 ± 1.03	83.05 ± 2.08
16	68.31*	71.07 ± 1.90
32	62.14 ± 1.47	74.82 ± 4.12
64	48.07 ± 0.65	66.17 ± 3.39
128	31.89 ± 1.42	36.38 ± 1.76
256	7.54 ± 0.71	11.02 ± 1.77

(* predicted values).

Appendix IX. Carbon, nitrogen concentration and C:N ratios over a 256 day decomposition period in a mangrove stand in Shuaiba, Saudi Arabia

Day	Carbon (%)	Nitrogen (%)	C:N
1	46.24 ± 5.32	0.48 ± 0.04	97.20 ± 11.86
2	48.88 ± 4.82	0.51 ± 0.08	97.09 ± 7.40
4	51.32 ± 7.94	0.58 ± 0.10	88.64 ± 10.53
8	48.03 ± 6.81	0.56 ± 0.09	85.90 ± 9.47
16*	50.07	0.65	77.03
32	54.86 ± 6.15	0.85 ± 0.15	65.50 ± 8.07
64	51.47 ± 5.70	0.91 ± 0.14	56.78 ± 5.73
128	62.32 ± 8.28	1.21 ± 0.21	52.82 ± 9.88
256*	74.37	1.94	53.11

(* predicted value)

Appendix X. Carbon, nitrogen concentration and C:N ratios of mangrove leaf litter over a 256 day decomposition period in a mangrove stand in Yanbu, Saudi Arabia.

Day	Carbon (%)	Nitrogen (%)	C:N
1	46.01 ± 4.85	0.48 ± 0.06	96.64 ± 4.86
2	46.79 ± 3.94	0.57 ± 0.18	87.08 ± 18.58
4	49.14 ± 4.93	0.55 ± 0.07	89.19 ± 5.03
8	51.51 ± 11.38	0.74 ± 0.41	76.95 ± 21.49
16	55.98 ± 6.82	0.76 ± 0.20	76.86 ± 16.68
32	48.20 ± 5.01	0.70 ± 0.17	71.72 ± 13.60
64	49.31 ± 5.91	0.78 ± 0.23	66.72 ± 14.26
128	56.10 ± 7.0	0.91 ± 0.2	63.44 ± 14.5
256*	61.43	1.27	52.56

(* predicted value)

Appendix XI. Remaining chemical composition (%) of leaf litter over a 256 decomposition period in a mangrove stand in Shuaiba, Saudi Arabia.

Day	Soluble carbohydrates	Hemicellulose	Cellulose	Lignin	LCI
1	54.34 ± 5.66	10.40 ± 5.93	10.40 ± 2.95	23.07 ± 2.46	0.69 ± 0.08
2	55.05 ± 2.12	9.09 ± 4.04	10.90 ± 3.23	22.51 ± 1.97	0.67 ± 0.07
4	43.99 ± 6.82	14.78 ± 9.32	10.51 ± 3.42	28.49 ± 3.49	0.73 ± 0.07
8	41.22 ± 3.75	15.51 ± 1.59	6.79 ± 1.52	33.11 ± 3.15	0.83 ± 0.04
16*	33.37	17.02	6.51	38.64	0.85
32	29.09 ± 7.90	21.39 ± 8.06	4.69 ± 1.76	39.76 ± 3.56	0.89 ± 0.04
64	36.09 ± 5.07	15.03 ± 1.45	6.36 ±1.78	36.92 ± 3.61	0.85 ± 0.03
128	30.85 ± 3.11	15.18 ± 1.85	6.29 ± 2.52	44.30 ± 3.91	0.88 ± 0.05
256*	30.85	15.19	5.73	44.30	0.88

(* predicted value)

Appendix XII. Remaining chemical composition (%) of leaf litter over a 256 decomposition period in a mangrove stand in Yanbu, Saudi Arabia

Day	Soluble carbohydrates	Hemicellulose	Cellulose	Lignin	LCI
1	54.26 ± 7.75	16.64 ± 5.51	8.89 ± 1.91	18.53 ± 3.29	0.68 ± 0.07
2	51.92 ± 7.31	10.55 ± 5.20	16.06 ± 3.77	19.11 ± 3.72	0.55 ± 0.09
4	41.34 ± 17.65	18.38 ± 19.23	14.70 ± 5.74	23.16 ± 4.83	0.62 ± 0.14
8	43.52 ± 6.91	11.91 ± 14.56	16.64 ± 4.89	24.94 ± 4.96	0.60 ± 0.07
16	40.68 ± 5.28	8.44 ± 5.38	20.57 ± 3.68	26.46 ± 3.60	0.56 ± 0.07
32	46.72 ± 8.67	14.32 ± 1.01	10.32 ± 1.18	26.06 ± 7.00	0.71 ± 0.06
64	44.28 ± 10.82	14.98 ± 1.40	10.50 ± 1.47	27.45 ± 8.67	0.71 ± 0.07
128	32.86 ± 11.80	11.68 ± 16.84	17.05 ± 6.18	34.09 ± 4.73	0.67 ± 0.10
256*	32.92	11.63	17.14	34.09	0.66

(* predicted value)