RESPONSE OF MANGROVES IN SOUTH AFRICA TO ANTHROPOGENIC AND NATURAL IMPACTS

Sabine Clara-Lisa Hoppe-Speer

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Nelson Mandela Metropolitan University

for tomorrow

Department of Botany

Response of mangroves in South Africa to anthropogenic and natural impacts

Sabine Clara-Lisa Hoppe-Speer

Promoter: Prof. Janine Adams

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Declaration

In accordance with Rule G4.6.3, a treatise/dissertation/thesis must be accompanied by a written declaration on the part of the candidate to the effect that it is his/her own work and that it has not previously been submitted for assessment to another University or for another qualification. However, material from publications by the candidate may be embodied in a treatiese/dissertation/thesis.

I, Sabine Clara Lisa Hoppe-Speer (s204032261) hereby declare that this thesis submitted in fulfilment of the requirements for the degree Philosophiae Doctor in Botany is my own work, and that it has not previously been submitted for assessment or completion of any postgraduate qualification to another University or for another qualification.

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Summary

The total mangrove area cover in South Africa is 1631.7 ha, with the largest area cover in a few estuaries in the KwaZulu-Natal Province (1391.1 ha) and the remainder recorded in the Eastern Cape Province with 240.6 ha. This represents 0.05 % of Africa's mangrove area cover and although small adds irreplaceable value to the biodiversity of South Africa. Mangroves are threatened by over-utilization through harvesting for firewood and building materials as well as excessive browsing and trampling by livestock. The objective of this study was to investigate the response of mangroves to different stressors from natural change as well as anthropogenic pressures. This was done by identifying pressures, measuring area cover, population structure and environmental parameters such as sediment characteristics. Mangroves in 17 estuaries along the east coast were investigated. Population structure and the area covered by mangroves in 2011/2012 were compared with data from the same area for 1999. Detailed studies were conducted in St. Lucia Estuary to investigate the response of mangroves to reduced tidal flooding; mangrove expansion at a latitudinal limit in a protected area at Nahoon Estuary was studied and the effect of cattle browsing on mangroves was measured at Nxaxo Estuary.

The St. Lucia Estuary (28°S; 32°E) represented a unique study site as the mouth has been closed to the sea since 2002 and the mangrove habitats have been non-tidal. St. Lucia Estuary is both a Ramsar and World Heritage site and therefore understanding the response of mangroves to changes in the environment is important. In 2010 sediment characteristics and mangrove population structure were measured at four sites which were chosen to represent different salinity and water level conditions. The site fringing the main channel had the highest density of mangrove seedlings and saplings. The dry site had a lower density of mangroves with mostly only tall adult trees and few saplings. Mangrove tree height and density increased at sites with high sediment moisture and low surface sediment salinity. Few seedlings and saplings were found at sites with dry surface sediment and high salinity. Long term data are needed to assess the influence of mouth closure on recruitment and survival of the mangrove forest at St. Lucia Estuary; however this study has shown that sediment characteristics are unfavourable for mangrove growth at sites now characterized by a lack of tidal flooding.

It is not known when exactly the mangroves were planted in Nahoon Estuary (32°S; 27° E), East London, but it is suspected that this was in the early 1970s. *Avicennia marina* (Forrsk.)

Vierh. was planted first, followed a few years later by the planting of *Bruguiera gymnorrhiza* (L.) Lam. and *Rhizophora mucronata* (L.) among the larger *A. marina* trees. Surprisingly the mangrove population appears to be thriving and this study tested the hypothesis that mangroves have expanded and replaced salt marsh over a 33 year period. This study provides important information on mangroves growing at higher latitudes, where they were thought to not occur naturally due to lower annual average temperatures. It further provides insights on future scenarios of possible shifts in vegetation types due to climate change at one of the most southerly distribution sites worldwide. The expansion of mangroves was measured over a 33 year period (1978 - 2011) using past aerial photographs and Esri ArcGIS Desktop 10 software. In addition, field surveys were completed in 2011 to determine the population structure of the present mangrove forest and relate this to environmental conditions. The study showed that mangrove area cover increased linearly at a rate of 0.06 ha^{-1} expanding over a bare mudflat area, while the salt marsh area cover also increased (0.09 ha^{-1}) but was found to be variable over time. The mangrove area is still small (< 2 ha) and at present no competition between mangroves and salt marsh can be deduced. Instead the area has the ability to maintain high biodiversity and biomass. *Avicennia marina* was the dominant mangrove species and had high recruitment (seedling density was 33 822 ± 16 364 ha^{-1}) but only a few *Bruguiera gymnorrhiza* and *Rhizophora mucronata* individuals were found (< 10 adult trees). The site provides opportunities for studies on mangrove / salt marsh interactions in response to a changing climate at the most southern limit of mangrove distribution in Africa. This research has provided the baseline data, permanent quadrats and tagged trees to be used in future long-term monitoring of population growth and sediment characteristics.

At Nxaxo Estuary (32°S; 28°E) the response of mangrove trees (*Avicennia marina*) to cattle browsing and trampling was investigated by using cattle exclusion plots. Exclusion plots were established by fencing in five 25 m^2 quadrats and adjacent to each experimental quadrat a control quadrat (not fenced in, 25 m^2) was set-up. Trees were tagged and measured annually from 2010 to 2012. Sediment salinity, pH, moisture, organic content, compaction as well as sediment particle size was also measured in each quadrat. Sediment characteristics did not vary between control and experimental plots but did show changes between the years. The mangrove trees in the cattle exclusion plots grew exponentially over a period of two years. There was a significant increase in mean plant height (5.41 ± 0.53 cm), crown volume (0.54 ± 0.01 m^3) and crown diameter (7.09 ± 0.60 cm) from 2010 to 2012. Trees in the control plots had significantly lower growth ($p < 0.05$). There was a decrease in plant height (-0.07 ± 0.67cm^1) and only small increases in crown volume (0.14 ±

0.1 m^3) and crown diameter (2.03 ± 2.61 cm). The research showed that browsing on mangroves by cattle stunts growth and causes a shrubby appearance as a result of coppicing. The browsed trees were dwarfed with horizontal spreading of branches and intact foliage close to the ground while the plants in the cattle exclusion plots showed an increase in vertical growth and expansion. In the cattle exclusion plots there was a significantly higher percentage of flowering (67 %) and fruiting (39 %) trees in 2012 compared to the control sites where 34 % of the plants were flowering and 5.4 % of the plants carried immature propagules. Observations in the field also indicated that cattle had trampled a number of seedlings thus influencing mangrove survival. The study concluded that browsing changes the morphological structure of mangrove trees and reduces growth and seedling establishment. This is an additional stress that the mangroves are exposed to in rural areas where cattle are allowed to roam free.

Seventeen permanently open estuaries provide habitat for mangrove forests along the former Transkei coast. This part of the Eastern Cape is mostly undeveloped and difficult to access. Mangrove area cover, species distribution, population structure and health of the mangrove habitat were compared with results from previous studies in 1982 and 1999. The mangrove *Bruguiera gymnorrhiza* had the densest stands and was widely distributed as it was present in 13 of the 17 estuaries. *Avicennia marina* was dominant in those estuaries which had the largest area cover of mangroves and was present in 10 estuaries, while *Rhizophora mucronata* was rare and only present in five estuaries. Anthropogenic and natural impacts were noted within the mangrove habitats in each of these estuaries. Harvesting of mangrove wood, livestock browsing and trampling and footpaths occurred in most of the estuaries (> 70 %). It was observed that browsing on trees resulted in a clear browse-line and browsing on propagules mainly by goats resulted in reduced seedling establishment in most of the estuaries except those in protected areas.

Mangroves had re-established in estuaries where they had been previously lost but mouth closure due to drought and sea storms resulted in the mass die back of mangroves in the Kobonqaba Estuary. There was a total loss of 31.5 ha in mangrove area cover in the last 30 years and this was a total reduction of 10.5 ha (11 %) for every decade. This is high considering that the present total mangrove area cover is only 240.6 ha for all the Transkei estuaries. In this study it was concluded that the anthropogenic impacts such as livestock browsing and trampling as well as harvesting in these estuaries contributed most to the mangrove degradation as these are continuous pressures occurring over long periods and are expected to increase in future with increasing human population. Natural changes such as sea storms occur less frequently but could result in large scale destruction over shorter

periods. Examples of these are mouth closure that result in mangrove mass mortality as well as strong floods which destroy forest by scouring of the banks. However these change impacts would increase with predicted climate change and coupled with anthropogenic impacts, the loss of mangrove could be accelerated. A healthy adult to seedling ratio would ensure a healthy regeneration of the forest. Therefore the rapid losses in area cover and the absence of seedlings in many estuaries (41 %) is concerning. To reduce this degradation some immediate and local remedial measures have been suggested: (1) reduce harvesting especially in the smaller and protected estuaries and have designated planted non-mangrove forested areas used for firewood and building material (2) involve local communities in protecting and possibly restoring mangrove forests where they have been previously lost; (3) restrict livestock to allow mangrove seedling establishment; (4) restrict trampling by having designated paths (5) prevent plastic pollution especially littering by clean-up efforts (6) conserve the estuarine banks by applying better agricultural practices; and (7) make use of alternative income sources for the local people thus increasing the ecotourism value of the forest (seen at Bulungula Estuary). Overall more sustainable practices and conservation plans are urgently needed to protect biodiversity and the ecosystem services that these forests provide.

Keywords: Mangrove degradation; Threats; Anthropogenic pressures; Natural pressures, Climate change; Forest resilience; South Africa

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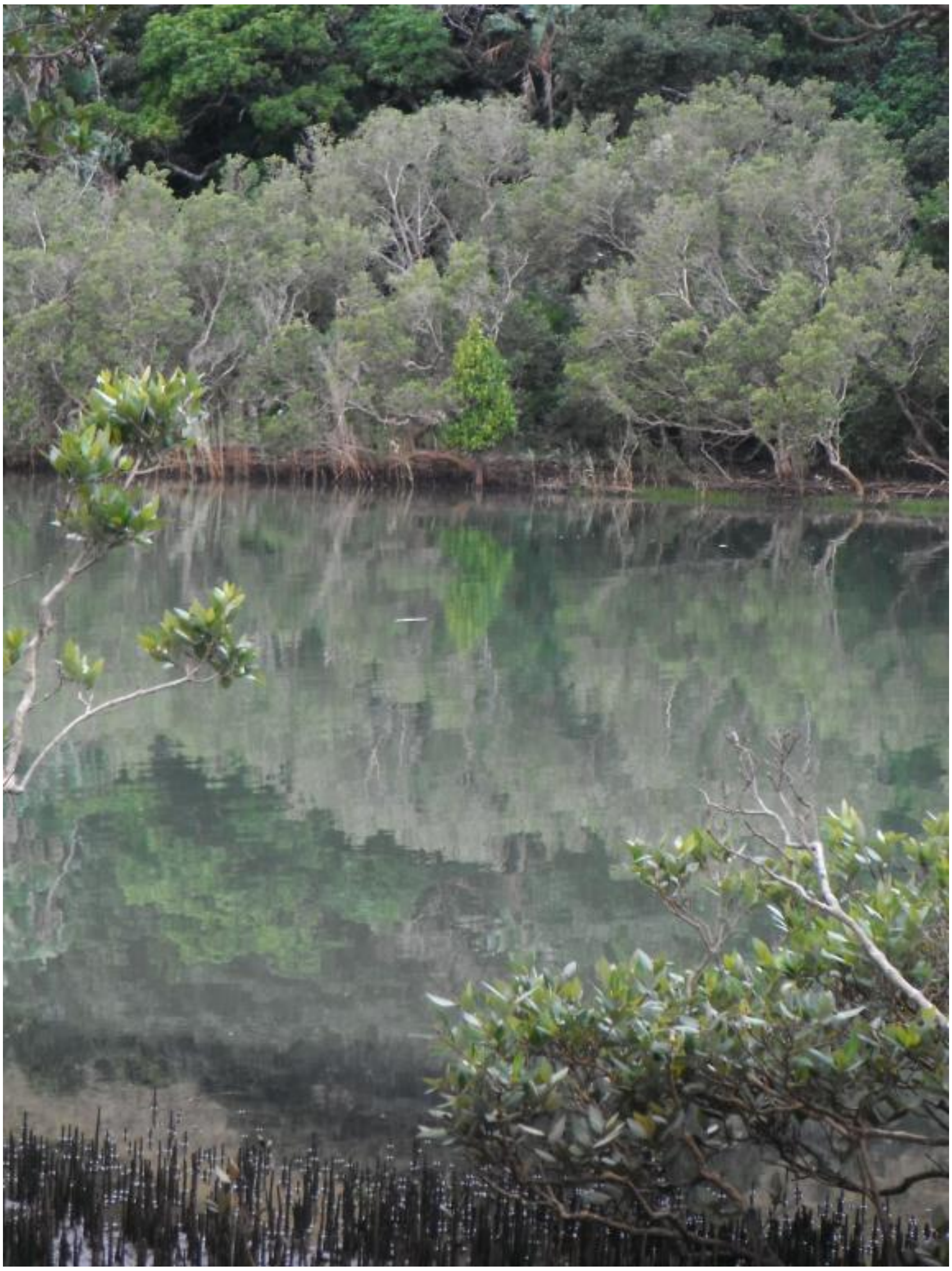
"The Wild Coast, a coastline of endless beauty and smiling faces"



"For most of history, man has had to fight nature to survive; in this century he is beginning to realize that, in order to survive, he must protect it"

- *Jacques-Yves Cousteau* -

Dedication:

I dedicate this thesis to my parents in gratitude for their support in many ways for all these years.

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Chapter 1: General Introduction

1.1 The importance of mangrove ecosystems

Mangrove trees or shrubs, also referred to as mangals, are woody halophytic plants which have a terrestrial origin and are also flowering angiosperms (Tomlinson 1994; Biber 2006). They are typically found in tropical to subtropical regions but also extend to the warm temperate areas (Duke 2006; Morrisey *et al.* 2010). These forests usually grow within intertidal areas from the high water mark to the low water mark in sheltered bays, lagoons or estuaries (Karthiresan and Bingham 2001).

Mangrove forests provide many important environmental services (Barbier *et al.* 2011) such as sediment trapping, nurseries for fish and habitats for other biota (Laegdgard and Johnson 2001; Ellison 2008) and are also important in processes of nutrient cycling (Adame *et al.* 2010a; Adame and Lovelock 2011). These forests provide protection against coastal erosion and the devastating effects of tsunamis and tropical cyclones (Danielsen *et al.* 2005; Galaz *et al.* 2008; Karthiresan and Bingham 2001; Hogarth 1999; Adame *et al.* 2010b). Because they are highly productive habitats, they act as important carbon sinks (Manson *et al.* 2005). They contribute to biodiversity and are the most productive aquatic ecosystems globally (Amosu *et al.* 2012). The high productivity of these forests accounts for dry weight biomass that can be compared to tropical rain forests (Clark *et al.* 2001; Alongi 2009), where their mean net primary production is between 300 to 500 metric tons ha^{-1} year^{-1} (Alongi 2002).

More than two-thirds of the world's population are found to be living around or near coastlines. These coastal regions, especially the low-lying areas are threatened by ever increasing human populations and climate change consequences, with especially devastating effects of sea-level rise and sea storms (Amosu *et al.* 2012). In the tropical and subtropical regions of Africa, many coastal human settlements are directly or indirectly dependent on mangroves and on the environmental services they provide, which have an estimated value between 200 000 to 900 000 USD annually (FOA 2003). However mangrove forests are considered to be the most threatened ecosystems globally (Field *et al.* 1998; Valiela *et al.* 2001), with more than 50 % already lost and between 1 – 2 % of mangrove forest still being lost annually on a global scale (Alongi 2002). This has mostly resulted from over-utilization of these resources (Alongi 2008).

In South Africa six mangrove species have been recognized which are *Avicennia marina* (Forsk.) Vierh., *Bruguiera gymnorrhiza* (L.) Lam., *Rhizophora mucronata* (Lam.), *Ceriops tagal* Perr. C. B. Robinson, *Lumnitzera racemosa* Willd. and *Xylocarpus gratum* König. The last three species occur only in Kosi Bay (27°S; 32°E), which is furthest north east of the country in the province of KwaZulu Natal (Figure 1.1) and the other species are naturally distributed from Kosi Bay down the coast to Kobonqaba Estuary (32°S; 28°E) in the Eastern Cape province. Mangroves have been planted further south at Nahoon Estuary (32°S; 27°E), (Ward and Steinke 1982; Steinke 1999; Taylor *et al.* 2003; Rajkaran 2011, (Chapter 3; Figure 1.1).

South Africa has only 1 634.7 ha of mangroves (Ward and Steinke 1982; Rajkaran and Adams 2011) which represents 0.05 % of the African continent (3 350 813 ha). However these forests are valuable and add to South Africa's rich biodiversity. They are protected by the National Forest Act (84 of 1998) and the Marine Living Resources Act (18 of 1998) that prohibits the harvesting of mangroves (DAFF 2008). The species *Bruguiera gymnorrhiza* and *Rhizophora mucronata* are additionally protected by the Protected Tree List of South Africa (Department of Agriculture, Forestry and Fisheries). Within the Eastern Cape, South Africa as much as 1.04 ha of mangrove forests are still lost per year due to natural and anthropogenic impacts (Adams *et al.* 2004). In KwaZulu-Natal Rajkaran *et al.* (2009) investigated the state of the small estuaries with mangroves. Rajkaran and Adams (2009) reported on the population structure and sediment characteristics of the largest mangrove forests in KwaZulu-Natal. Little recent research has been done on the mangroves in the Eastern Cape Province. At Mngazana Estuary detailed studies of mangrove litter production and the effect of harvesting on the mangroves were completed by Rajkaran and Adams (2007) and Rajkaran *et al.* (2004; 2009). Ward and Steinke (1982) and Adams *et al.* (2004) determined the state and distribution of mangroves of the former Transkei region. One of the objectives of the research presented in this thesis was to repeat these surveys and compare mangrove area cover and species composition to assess changes over time.

The study sites for this research were the St. Lucia Estuary, the Nahoon Estuary, the Nxaxo Estuary and all the 17 estuaries that contained mangroves along the Eastern Cape coast. The St. Lucia Estuary has been chosen for this study because mangroves in this estuary survived even under closed mouth and low water level conditions and there was a need to determine the state of the mangroves. The Nahoon Estuary had mangroves that had been planted and thus provides an opportunity to study mangroves out of its natural distribution. Nxaxo Estuary had mangrove forests that seemed to have been affected by cattle browsing and thus this study site was chosen to determine how mangroves respond to long-term browsing and trampling by livestock. The mangrove-containing estuaries along the former Transkei had been studied by Adams *et al.* (2004) but a detailed study was needed to compare these mangrove forests with previous data and to determine if anthropogenic or natural pressures have affected their state, area cover and distribution. The mangrove forests in South Africa are being lost at an increasing rate and more detailed research is urgently needed for more effective management interventions.

1.1 Background: Mangrove adaptations, distribution and threats to mangroves from climate change and anthropogenic pressures

Mangroves grow where the seawater isotherms in the winter season do not fall below 20 °C (Karthiresan and Bingham 2001; Duke *et al.* 1998) and with fluctuations not more than 5 ºC (Singh and Odaki 2004). Mangroves may tolerate low temperatures for short durations such as observed in Australia, but these trees show defoliation after periods of frost and are thus sensitive to low temperatures. Pickins and Hester (2011) conducted a study on *Avicennia germinans* (L.) L. and found that low temperatures of -6.5 °C for 24 hours increased seedling mortality. For this reason their biogeographical distribution is mostly determined by temperature (Duke 2006). Therefore species diversity is greatly reduced at higher latitudes due to the lower temperatures and often is just represented by only one species such as *Avicennia* (Tomlinson 1994).

With global warming an expected increase in sea surface and ambient temperature could result in mangrove distribution shifts at a global and local scale. Mangroves may extend into the higher latitudes in future, as seen in past global warming events that contributed greatly to the present global mangrove distribution (Ellison 2008). Climate change and temperature increase would also result in sea-level rise which ultimately results in a change in salinity and inundation periods within these forests (Nicholls and Lowe 2004). Mangroves are adapted to environmental conditions such as salinity, aridity, tides, wave action, low pH and anaerobic sediments. These conditions make it impossible for terrestrial trees to grow but even mangroves have limits (Zomlefer *et al.* 2006; Ahmed and Abdel-Hamid 2007). Sea-level rise will bring about changes that will affect the distribution of the mangroves on local and global scales. On local scales, sea-level rise will result in tides being pushed up higher into the estuary causing longer inundation and changes within the intertidal region (Krauss *et al.* 2010). For this reason the sedimentation rates will be important, since this will create new habitats for mangroves to establish, while erosion will result in habitat loss (Anthony 2004; Victor *et al.* 2005; Krauss *et al.* 2010).

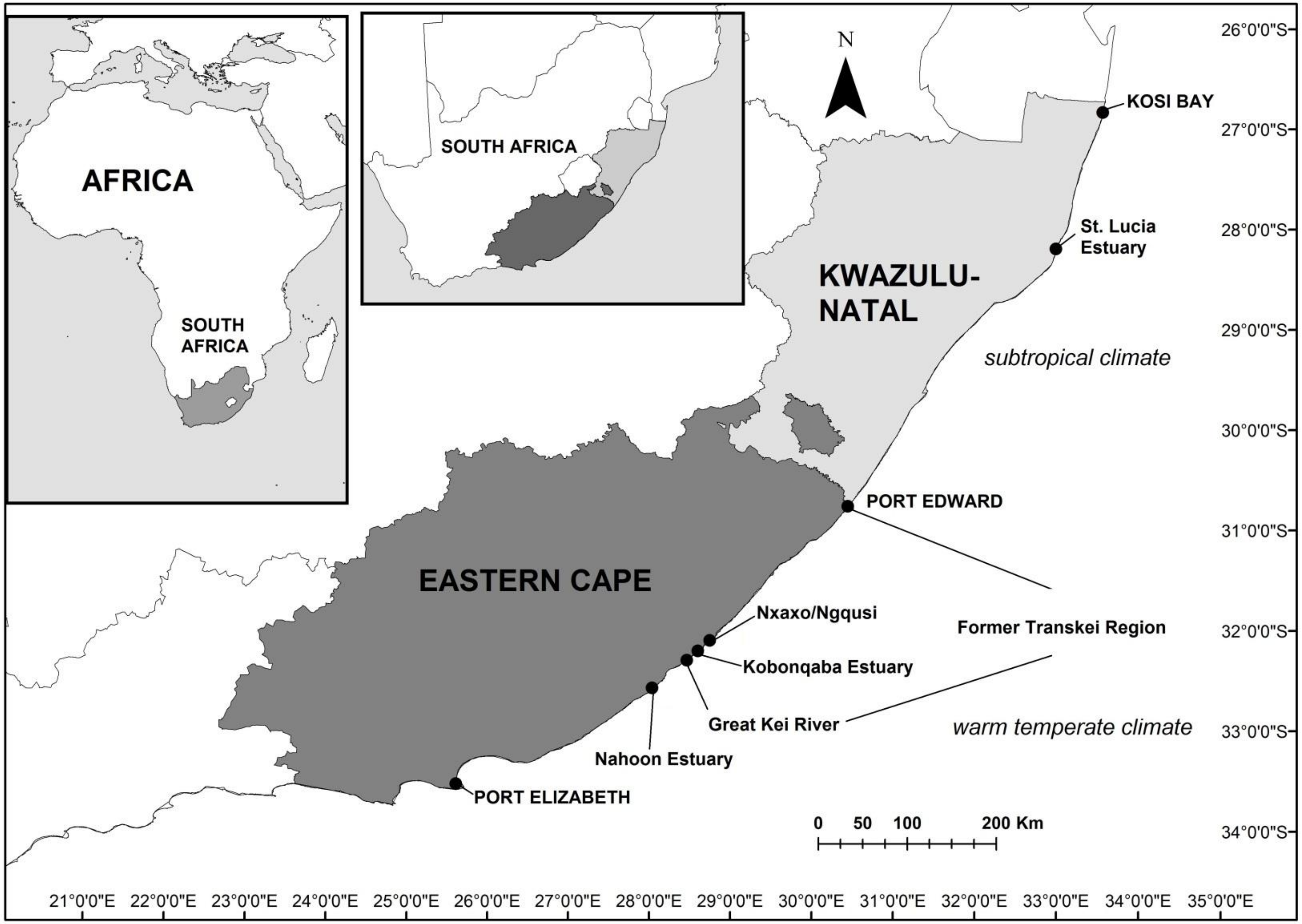

Figure 1.1: The distribution of mangroves in South Africa from the warm temperate regions of the Eastern Cape to the subtropical regions of KwaZulu-Natal.

Previously it was thought that mangroves only exist in high rainfall tropical and subtropical regions, however it is now known that they are also represented in arid environments such as in Egypt and Pakistan, since these plants have adapted to high salinity, which also allows them to survive in dry environments (Singh and Odaki 2004). Different species of mangroves have different tolerance ranges (Hogarth 2007) and their success lies with their adaptation to these environmental conditions. Environmental conditions thus determine mangrove distribution and species composition (Ahmed and Abel-Hamid, 2007; Hogarth 2007; López-Hoffman *et al.* 2007; Krauss *et al.* 2008). The complexity of the abiotic and biotic interactions of the mangrove ecosystem drive the morphological and ecophysiological adaptations of each species (Alongi 2009).

The early life history of mangroves is another important adaptation of these plants which ensures seedling establishment and survival (Clarke *et al.* 2001). Allen and Krauss (2005) mentioned that ocean currents are responsible for propagule distribution into new habitats as well as the concentration of salinity, where higher salinity results in longer times that propagules stay afloat. Propagules of both *Bruguiera* and *Rhizophora* species prefer lower salinity and good light conditions to establish (Allen and Krauss 2005). These propagules will stay afloat for long periods of up to 2 months if environmental conditions are not suitable. With climatic events such as El Niño, ocean currents patterns will change (Allen and Krauss 2005) which may influence propagule dispersal. Mangrove reproduction is highly evolved, where investments into the next generation (propagules) is unusually high (Tomlinson 1994). Mangroves are viviparous, they germinate when still attached to the parent tree, thus ensuring that the new plant has an added advantage for survival. When the propagule is mature it falls from the flower or more collectively known as 'calyx' to the forest floor (Tomlinson 1994), where it is then either transported away by the changing tides or establish under or near the parent tree. Their weight, size and buoyancy will play an important role in dispersal and is also known as 'ecological sorting' (Rabinowitz 1978) or 'tidal sorting' (Delgado *et al.* 2001). For example propagules of *Rhizophora and Bruguiera* species are heavier, with a long cigar shape, also known as 'hypocotyls', compared to the round, lighter and small propagules from *Avicennia* species (Branch and Branch, 1995). The lighter propagules get transported further than heavier ones, which are less likely to be swept out to sea (Delgado *et al.* 2001). The heavier propagules also often fall straight down, roots first into the mud like a spear, where they will establish as new plants but also may stay afloat for up to several months. Establishment is influenced predominantly by salinity and most propagules are not transported a great distance from the parent plants because they are large (Hogarth 1999). This makes the *Avicennia* species very successful pioneer genera where buoyancy and tidal actions as well as currents contribute to their distribution (Delgado *et al.* 2001). This may prove important with changes in sea level and sea currents (Amosu *et al.* 2012). A study presented by Nitto *et al.* (2008) for Gazi Bay, Kenya, suggested that propagules from *Rhizophora mucronata* and *Ceriops tagal* would disperse naturally through "stranding and self-planting" and that mangrove forests would successfully re-generate. However with the current anthropogenic pressures the propagule dispersal would be hampered. They also mentioned that sea-level rise in about 20 years will have effects on the mangrove distribution and structure with an expected increase in distribution on regional scales.

The nature of sediments is critically important in driving ecological processes (Adame *et al.* 2010a; 2010b) since these provide essential nutrients needed for mangrove growth (Mazda and Ikeda 2006; Schwendenmann *et al*. 2006). Mangroves are well adapted to grow in different saline habitats, where they out-compete terrestrial plants. This is because for salinity, especially salinity concentrations close to seawater (35 PSU), plants have to be able to regulate cellular osmosis and decrease the toxic effects of salt by absorbing inorganic ions to counteract the osmotic gradients (Snedaker 1982; Smith and Smith 2001; Mehlig 2006). However an excess of these inorganic ions can also have a toxic effect and thus mangroves have to balance these in order to survive in such harsh environmental conditions (Mehlig 2006). Mangroves that grow in high salinity have reduced growth and productivity as more energy is invested into osmotic regulation than into growth (Clough 1984; Naidoo 1985; Naidoo 1987; Kathiresan *et al.* 1996; Falqueto *et al*. 2008; Li *et al.* 2008; Suarez and Medina 2008; Hoppe-Speer *et al.* 2011).

Different mangrove species have different tolerance ranges (0 to 35 PSU) and for most the most luxurious growth is displayed in salinity concentrations between 5 to 18 PSU (Suarez and Medina 2005) but grow poorly in just freshwater (Pezeshki *et al*. 1989). *Avicenna marina* was found to survive up to 70 PSU in the Arabian Gulf (Dodd *et al.* 1999). It is also known that seedlings seem to be more sensitive to high salinity than adult trees; this is because most trees have access to groundwater (Suarez and Medina 2008) and also the fluctuation in salinity has a more profound negative impact on the plants than continuous high salinity (Kathiresan and Thangam 1990). As seedlings grow into adults they are able to cope with higher salinities. Mangroves are highly adapted to these high salinity environments by maintaining high osmotic potentials in the plant cells, allowing the salinity concentrations to be high in the root tissue, excluding salts from the roots tissues and by excreting salts from secreting glands within the leaves (Aziz and Khan 2001b; Krauss *et al*. 2008). *Avicennia* species are exceptionally adapted to exclude as much as 90 % of the salt from the roots (Branch and Branch, 1995). However *Rhizophora* and *Bruguiera* species will not be able to cope well in prolonged high salinity conditions (Naidoo 1985; Koch and Snedaker 1997). Both latter species are able to withstand moderate salinity similar to seawater and do this by taking up salt by the roots and depositing excess salt within the leaves that are shed in order to rid the trees of the excess salt (Steinke 1999; Li *et al.* 2008).

The species *Avicennia marina* has an optimum salinity range from 5 to 35 PSU (Naidoo 1987) and higher salinities result in reduced growth and even stunting (Naidoo 2006). *Rhizophora mucronata* grows best in 18 PSU (Khan and Aziz 2001a; Hoppe-Speer *et al.* 2011), while *Bruguiera gymnorrhiza* grows well in lower salinities of ≥ 10 PSU but salinities of >35 PSU result in reduced growth and leaf senescence (Naidoo 1990). As salinity increases, the water becomes less available and thus mangroves adapt to these conditions by having a shift towards higher root biomass compared to the leaf biomass thus increasing the root mass to compensate for water uptake (Lopez- Hoffman *et al.* 2006; 2007). Schmitz *et al*. (2006) also suggested that adult *R. mucronata* trees adapt by increasing their vessel density to increase water uptake in high saline environments. A constant supply of freshwater (freshwater runoff from rain and groundwater supply) to dilute the saline waters is essential in sustaining healthy ecosystems (Kitheka 1998; Schwendenmann *et al*. 2006; Mazda and Ikeda 2006; Whitfield and Taylor 2009).

With increasing population there will be an increase in demand on the freshwater resources. This puts additional pressures on the coastal ecosystems especially on estuaries and thus freshwater inflow needs to be continuously monitored as it will ultimately determine the health of the estuaries (Van Niekerk and Turpie 2012). Van Niekerk and Turpie (2012) have also mentioned that these freshwater requirements for estuary health will become increasingly important with climate change and in the events of drought. Freshwater abstraction from estuaries or sea storms can result in the mouth of an estuary closing to the sea. Mangroves mostly grow in tidal habitats and this can therefore have a major influence on their growth and survival. Mouth closures results in back-flooding and with prolonged periods of inundation of 5 months and more in the intertidal regions causes mass mortality of large areas of mangrove forest (Branch and Branch 1995) as seen in Kosi Bay in 1965 (Breen and Hill 1969). This is because for plant metabolism, oxygen is required for respiration and the mangroves will die under prolonged inundation and high water levels (Breen and Hill 1969). Mangroves are adapted to some inundation and waterlogging but not for long periods of time. These special adaptations are by having modified roots also collectively known as ‘aerial roots’, to cope with low oxygen conditions, which are again morphologically different in different species (Mauseth 2003). *Rhizophora* species have the typical ‘stilt or prop roots’, which extend from the main stem in an arch and down into the mud (Branch and Branch 1995). Within the root tissue many air spaces provide for rapid uptake of oxygen into the root and transportation to other parts of the plant (Lovelock *et al*. 2006b). *Bruguiera* species have many "shallow horizontal roots" (Duke 2006) where the roots 'arch' out of the substrate and are known as "knee roots" and *Avicennia* species have large numbers of 'emergent vertical roots' (pneumatophores), which give a needle-like appearance around the tree and may be up to 30 cm long (McKee 2001).

Mangroves are adapted to cope with low oxygen conditions and are able to maintain respiration, which most terrestrial plants cannot in the waterlogged soils (Pezeshki *et al.* 1997). However these adaptations to inundation have limits and again different species of mangroves are adapted to different inundation periods (Luzhen *et al.* 2005). *Bruguiera gymnorrhiza* will tolerate inundation of 0 to 75 cm (Ward 1976), while *R. mucronata* and *A. marina* will die in 50 cm inundation for five months or more (Breen and Hill 1969). In research done by Hoppe-Speer *et al.* (2011) they have reported that *R. mucronata* seedlings in treatments of 24 hr inundation, for 14 weeks, showed stem elongation but significantly reduced biomass allocations compared to the seedlings grown in tidal treatments.

The available nutrients and minerals determine plant growth and these are normally transported to mangrove ecosystems by freshwater run off, which is important in the process of nutrient cycling (Schwendenmann *et al.* 2006). These are again influenced by the tidal cycles as well as the rainfall (Kitheka 1998; Mazda and Ikeda 2006). Mangrove soils are naturally low in nutrients (Reef *et al.* 2010) and nutrient-deficient soils have negative effects on plant growth and productivity. However added nutrients, in the form of fertilizers (Feller *et al.* 2009) or sewage (Mohammed 2009) may also alter the mangrove ecosystem processes and productivity.

Mangroves adapt to waterlogged soils and do this by shifting from a usual aerobic respiration to more anaerobic respiration, but the anaerobic soil prevents the uptake and transport of essential nutrients thus making nutrients such as nitrogen, phosphorus and potassium less available for the plant (Pezeshki *et al.* 1997). Pezeshki *et al.* (1997) mentioned that the most important nutrients for mangroves are phosphorous and nitrogen, which are converted and made available in the form of inorganic nitrate and phosphate. Ammonium, on the other hand, is derived from nitrogen increases in anoxic sediments and can be taken up in this form by the mangroves (Reef *et al.* 2010). Mangrove sediments are usually low in nutrients, the available nutrients are reabsorbed by fast decomposition and this is known as "nutrient-conservation" (Reef *et al.* 2010). Due to this nutrient conservation these forests are highly productive (Alongi 2002; 2009). Microorganisms within the sediment are an important component in this nutrient cycling process as they help in the breakdown of these nutrients (Reef *et al.* 2010). These include bacteria in the sediment that fix atmospheric nitrogen known as ‘nitrogen fixing bacteria’ and toxic ammonia is converted to nitrate by the bacteria. Nutrient conservation processes are also attributed to the abundance of macro and meiofauna, where the “energy efficient plant-soil-microbe relationships” drive the nutrient cycling (Kristensen and Alongi 2006). Macro-organisms such as crabs and shrimps make their burrows in the sediment which results in sediment mixing and in greater sediment oxidation (Ruwa 1990) and thus their activities influence mangrove population structure, distribution and ecological functions (Kristensen and Alongi 2006).

Many communities live in close proximity to the mangroves and are directly and indirectly dependent on these ecosystems. The mangroves are harvested for timber, fuel and building material as well as for charcoal (Mohamed 2008; Rajkaran *et al.* 2009) and for fodder for domestic livestock (Dahdouh-Guebas *et al.* 2006a; Shah *et al.* 2007). Mangrove habitats are converted to settlements, silviculture, aquaculture and agricultural lands (Primavera and Esterban 2008; Primavera *et al.* 2011; Dahdouh-Guebas *et al.* 2006b; Mohamed *et al.* 2009). In South Africa examples of mangrove habitat conversions have occurred in KwaZulu-Natal, where the transformation of wetlands to sugarcane plantations and the construction of bridges for the N2 freeway (Begg 1978; Rajkaran *et al.* 2009) as well as other developments such as residential areas and the construction of Richards Bay Harbour (Rajkaran and Adams 2011) have resulted in extensive mangrove area cover loss. Further south along the former Transkei coast, mangrove area cover has been lost due to excessive wood harvesting observed in many estuaries such as at Mngazana Estuary (Rajkaran *et al.* 2004; Adams *et al.* 2004; Rajkaran and Adams 2009). Catchment activities also affect the estuarine ecosystems and inshore regions due to poor agricultural practices, where large scale erosion of topsoil increase sediment load into rivers, estuaries, salt marsh and mangrove areas and on to corals resulting in much of the coastal habitats being degraded (Victor *et al.* 2005). A typical semidiurnal tidal regime (12.42 hours) is recognised along the South African coastline with little variation in tidal height (Schumann *et al.* 1999). The tidal height can be defined as the “difference between sea-level at high tide and that at low tide” (Schumann *et al.* 1999). In southern Africa, an increase in the frequency and intensity of extreme events such as sea storms, droughts and floods are expected to increase coupled with an expected increase in rainfall along the eastern parts of South Africa, where an increase in floods will increase the sediment loads in the estuaries and thus into mangrove habitats (Engelbrecht *et al.* 2009; Van Niekerk and Turpie 2012).

Sea-level rise will cause a shift in mangrove distribution, species composition and competition as well as recruitment successes or failures at local and global scales (Ye *et al.* 2004; Abel *et al.* 2011). Mangroves are expected to "migrate" landward and up river as a response and adaptation to sea-level rise, but with the increasing human settlements and the "urban squeeze" which are bordering present intertidal settings, competition for space will prevent mangrove expansion into higher elevation areas (Farnsworth and Ellison 1997b; Schleupner 2008). Therefore developments make it impossible for mangroves to colonize new habitats resulting in habitat loss. Habitat loss is also possibly the result of floods, where an increase in rainfall is expected in the northern eastern coastal areas of South Africa, which may cause increased events of extreme floods that scour the banks (Colloty *et al.* 2000). The droughts that cause more frequent estuary mouth closures, and floods coupled with anthropogenic pressures will reduce the resilience for natural regeneration of these mangrove forests (Alongi 2008; Lawrie and Stretch 2011; Van Niekerk and Turpie 2012). Thus adaptive coastal management has become fundamentally important globally and Amosu *et al.* (2012) suggested that in order to conserve these highly productive and biodiverse resources, a "thinking globally and acting locally" approach should be adopted for more effective coastal management.

1.2 Thesis objectives and structure

The main rationale and motivation of this thesis was to investigate the present pressures and impacts on mangrove ecosystems and to establish whether anthropogenic pressures had more profound effects on the state of the mangroves in comparison to natural changes. Therefore the overall hypothesis that was addressed in this study was '**loss of mangrove habitat was due to anthropogenic impacts, which are greater than that caused by natural changes**'. This study focused on the mangroves in the Eastern Cape Province with one study site in the KwaZulu-Natal Province at St. Lucia Estuary (KwaZulu-Natal). Mangrove distribution and population structure was assessed in relation to possible natural and anthropogenic pressures. This thesis consists of five chapters designed to address the research questions. Figure 1.2 illustrates the relationship between the various chapters.

In **Chapter 2**, the study investigated an unusual situation at St. Lucia Estuary, where mangroves had survived under estuary mouth closure and low water levels. This mouth closure had been the result of prolonged drought, water abstraction and the diversion of freshwater from the estuary. The aim was to investigate the state of the mangroves in different habitats (with different salinity and water level conditions) within the same system. The information obtained provides the iSimangaliso Wetland Park (previously known as the

Greater St. Lucia Wetland Park) management with possible insight on the forest degradation and therefore better conservation strategies can be designed and implemented. The key research questions were (1) What effect did closed mouth conditions have on salinity, sediment characteristics and water level and what were their impacts on the present mangrove population structure? (2) How would water level changes influence recruitment and survival of mangrove seedlings? (3) How did other biota (snails and crabs) respond to water level changes within the mangroves and can these be used as indicators for ecosystem health? The results from this study provide management with the necessary information for efficient conservation actions and plans.

In **Chapter 3** the objective was to determine the expansion rate of planted mangroves in a non-native site over the past 33 years. This was to determine the possible effects it had on associated salt marsh habitats as the mangrove population was thriving at Nahoon Estuary (East London) which is beyond the natural latitudinal limit of mangrove distribution in South Africa (Figure 1.1). It is rare and uncommon to plant mangroves in South Africa and especially out of its natural range; therefore this study provided another unusual situation with an opportunity that investigated the expansion of mangroves in a non-native area in South Africa. This was also the first time that a planted forest was investigated in South Africa. The hypothesis tested was that the expansion rate of planted mangroves over 33 years was at the expense of natural salt marsh habitat. This provides information on mangroves growing at higher latitudes. The study provided insights on possible future scenarios of shifts in vegetation types due to climate change (Nicholls *et al.* 2007) at the most southerly mangrove site in South Africa and one of the most southerly distribution sites worldwide.

Chapter 4 investigated the effects of browsing by domestic livestock on mangrove growth at the Nxaxo Estuary. Previous research had shown this to be a major impact in rural areas of the Eastern Cape coast (Steinke 1999; Adams *et al.* 2004). The research questions were: Does browsing and trampling by domestic livestock have an effect on the growth and survival of the mangrove *Avicennia marina*? It was expected that mangroves respond to browsing by having reduced growth and horizontal expansion. It was also expected that livestock browsing would have a negative effect on mangrove recruitment. Such research is lacking in international literature and this would be the first study of this nature in South Africa. Dahdouh-Guebas *et al.* (2006a) investigated short-term mangrove browsing by feral water buffalo in India. Other browsing studies are presented by Spugeon (2002) who reported mangrove browsing impacts from camels in Egypt and Shah *et al.* (2007) reported the impacts that livestock such as cattle, goats and camels have on mangroves in Pakistan.

However these studies focused on the social aspect and the ownership conflicts of mangrove resources, while this study has focused on the quantified impacts that domesticated livestock had on the mangrove forests.

The objectives for **Chapter 5** were to determine the state and distribution of the mangroves in the Eastern Cape Province. The aim was to compare the past distribution by using the information from Ward and Steinke (1982) and Adams *et al.* (2004) and comparing it to present mangrove state, population structure, area cover and mangrove health by assessing associated anthropogenic as well as natural change pressures. At present it is expected that the combination of these threats will increase the pressures on these mangrove areas if there is no intervention or attempts on a local scale to protect these ecosystems from overexploitation. Therefore better management strategies and conservation plans are urgently needed for more sustainable mangrove utilization and to keep in mind the time to allow these forests to regenerate and be more resilient after disturbances from either natural events or anthropogenic activities. The hypotheses that were formulated in achieving the above objectives were: (1) Mangrove cover has decreased since 1999 due to an increase in human development activities (2) Healthy mangrove forests occur in protected areas i.e. the adult to seedling ratio indicates new growth. (3) Harvesting for mangrove wood is the dominant impact in most estuaries.

In **Chapter 6** the findings of the previous chapters were incorporated into the DPSIR (Driving forces-Pressures-State-Impacts-Responses) framework and management recommendations were suggested as well as possible future scenarios if no interventions are made to conserve these ecosystems.

The DPSIR represents the Driving Forces, Pressures, State, Impacts and the Responses and is summarized in Figure 1.3. The DPSIR was developed for 'The Organization for Economic Cooperation and Development (OECD) in the 1980s by using the original PSR (Pressure-State-Response) framework (OECD 2003; UNEP 2006). The DPSIR framework has been used widely in ecological studies including some mangrove-related research such as studies presented by Lin *et al.* (2007); Campuzano *et al.* (2011); Rajkaran (2011) and Wu and Wang (2011). This conceptual framework has been developed as a tool for describing the links or "relationships" between the environment and the stakeholders (UNEP 2006).

The DPSIR Framework (Figure 1.3) is thus a tool which combines the different connections between environment and society. It provides an insight into the various links and can be explained as the following: The driving force or forces are usually the communities (e.g. that live close to an estuary or mangrove forest) or a certain development (e.g. mining or aquaculture or agriculture) that put the natural resources (i.e. in this case mangrove forests) under pressure. These pressures will leave the environment in a certain state (e.g. mangrove degradation) and result in environmental impacts (loss of habitat and biodiversity). The impacts then result in a response from the society or community, which is often in form of legislations and policies (Maxim *et al.* 2009, Rajkaran 2011). Therefore this framework is used to integrate the knowledge obtained and provides some guidelines for management plans and strategies for sustainable resource utilization.

The methodology that was used in Chapters 2 to 5 was similar to the methods used by others (Ellison and Farsworth 1993; Walters 2005; Rajkaran 2011) and included the "quadrat method" for determining the mangrove population characteristics. Sediment characteristics were measured to determine possible stressors for the different habitats and identify those factors influencing seedling establishment. Chapters 3 and 5 included mangrove area cover surveys using ground truthing and mapping. Spot 5 (2010) images and historical aerial photographs as well as the ArcGIS software was used for determining mangrove and salt marsh area cover.

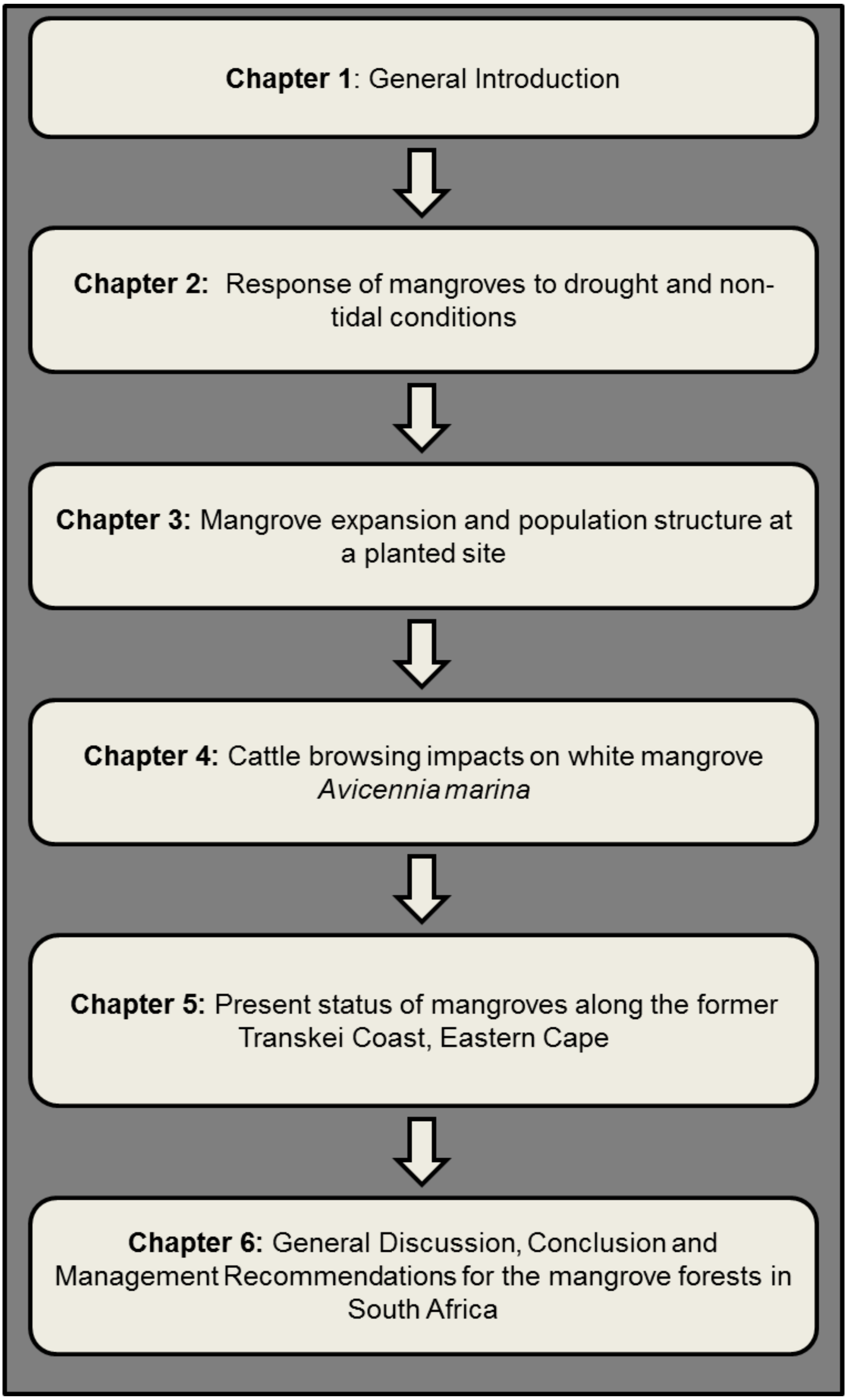

Figure 1.2: A flow diagram of how the different chapters are connected in achieving the overall objectives.

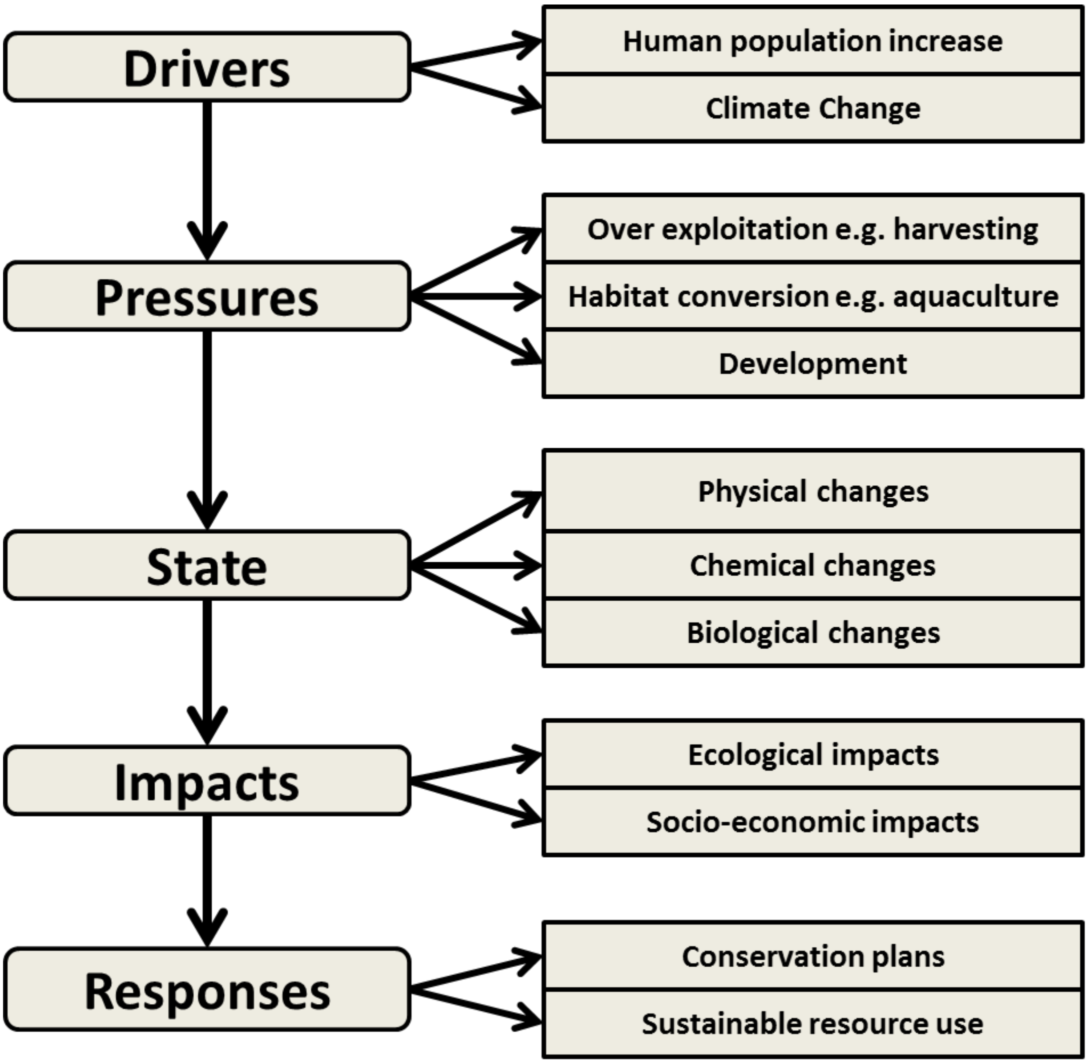

Figure 1.3: The DPSIR Framework adapted from Lin *et al.* (2007).

Chapter 2: Response of mangroves to drought and non-tidal conditions

2.1 Introduction

The St. Lucia system occurs along the subtropical coastline of South Africa. It is one of the largest estuary lake systems in Africa and the oldest protected estuary in the World (Figure 2.1). St. Lucia is both a Ramsar site and a World Heritage site (Taylor 2006). As early as the mid 1940s, scientists recognized the importance of freshwater inflow to the St. Lucia Estuary and predicted that if the natural flow were altered then the mouth would close more frequently and for longer periods of time (Day 1948). Freshwater inflow into the St. Lucia system depended on the Umfolozi River that was linked to, and had a shared mouth with the St. Lucia Estuary. Under closed mouth conditions, freshwater coming from the Umfolozi River was pushed up the Narrows (Figure 2.1) and then into the St. Lucia Lake (Whitfield and Taylor 2009). This no longer occurs as the Umfolozi River was canalized and diverted to the sea with its own separate mouth which was then tidal and in high spring tides would flood the Umfolozi mangrove habitats *(*Taylor pers. comm.). As a result of the drought conditions since 2002 the mouth of St. Lucia Estuary has remained closed. Given that the management practice of artificially keeping the mouth open by dredging has stopped, the mangrove habitats are no longer tidal, which creates an unusual situation as mangroves usually occur from the lower intertidal to the upper intertidal area. In normal conditions when the mouth is open, the lower intertidal area mangroves grow in continuously waterlogged sediments while the upper intertidal area becomes inundated only at high spring tides. Mangroves are therefore typically found between the "mean sea level and mean high water spring tide level" (Steinke 1999). Past studies in South African estuaries have shown that closed mouth conditions can increase water level, resulting in flooding of mangroves and die-back. However this has not been the case in St. Lucia as the water level has been low due to prolonged mouth closure and drought. The main objective of this study was to investigate the effect of closed mouth conditions on the mangrove population structure in the estuary. Within the St. Lucia Estuary, the mangrove species *Avicennia marina* and *Bruguiera gymnorrhiza* fringe the main channel from the mouth of the estuary to the upper part of The Forks, which is 19 km upstream from the mouth in the southern part of the lake (Figure 2.1). They also occur along the lower parts of the Mpate River (Ward and Steinke 1982) and between the Umfolozi and St. Lucia systems. Mangroves in St. Lucia Estuary have a combined area cover of 304.1 ha (Nondoda 2012) and add extensive value to St. Lucia's biodiversity. However with the closed mouth and drought conditions the mangrove forests may be threatened and their future uncertain.

The objective of this study was to investigate whether prolonged closed mouth conditions had influenced the mangrove population structure and density in the St. Lucia Estuary. Key research questions were (1) What effect did closed mouth conditions have on salinity, sediment characteristics, water level and what were their impacts on the present mangrove population structure? (2) How would water level changes influence recruitment and survival of mangrove seedlings? (3) How did other biota (snails and crabs) respond to water level changes within the mangroves and can these be used as indicators for ecosystem functioning? Since mouth closure, mangrove communities have survived what might be considered stressful (low water level) conditions, such that this study provides an unusual opportunity to monitor the extent of changes in mangrove range and population structure in response to changing environmental conditions. This study also intended to provide a better understanding on how mangroves respond to water level fluctuations and allow for predictions on their responses to future natural changes.

2.2 Materials and methods

2.2.1 Study site conditions

The mean water level for St. Lucia Estuary is greater than 2 m above sea level, but when water level drops below this, mouth closure occurs (Lawrie and Stretch 2011). Tides are semi-diurnal with the average neap tidal range at approximately 0.5 m and the spring tidal range at approximately 2 m representing the microtidal range (Schumann *et al.* 1999). Under extended closed mouth conditions the salinity in the Narrows has been lower than in the lake section because of freshwater runoff and groundwater seepage from the local catchment (Lawrie and Stretch 2011*)*. The salinity and water level in the Narrows had fluctuated over the months prior to the study depending on rainfall and river inflows, but water level had remained sufficiently low from July 2008 to November 2010 to prevent the mouth from opening (Figure 2.2). Based on long-term monitoring by Fox and Taylor (2010) it is possible to report that the mean water level was lower than 0.8 m above mean sea level and the average salinity was below 10 PSU at the bridge across the Narrows (Figure 2.2) at the time of this study. This low water level resulted in dry conditions in most of the intertidal areas in the Narrows.

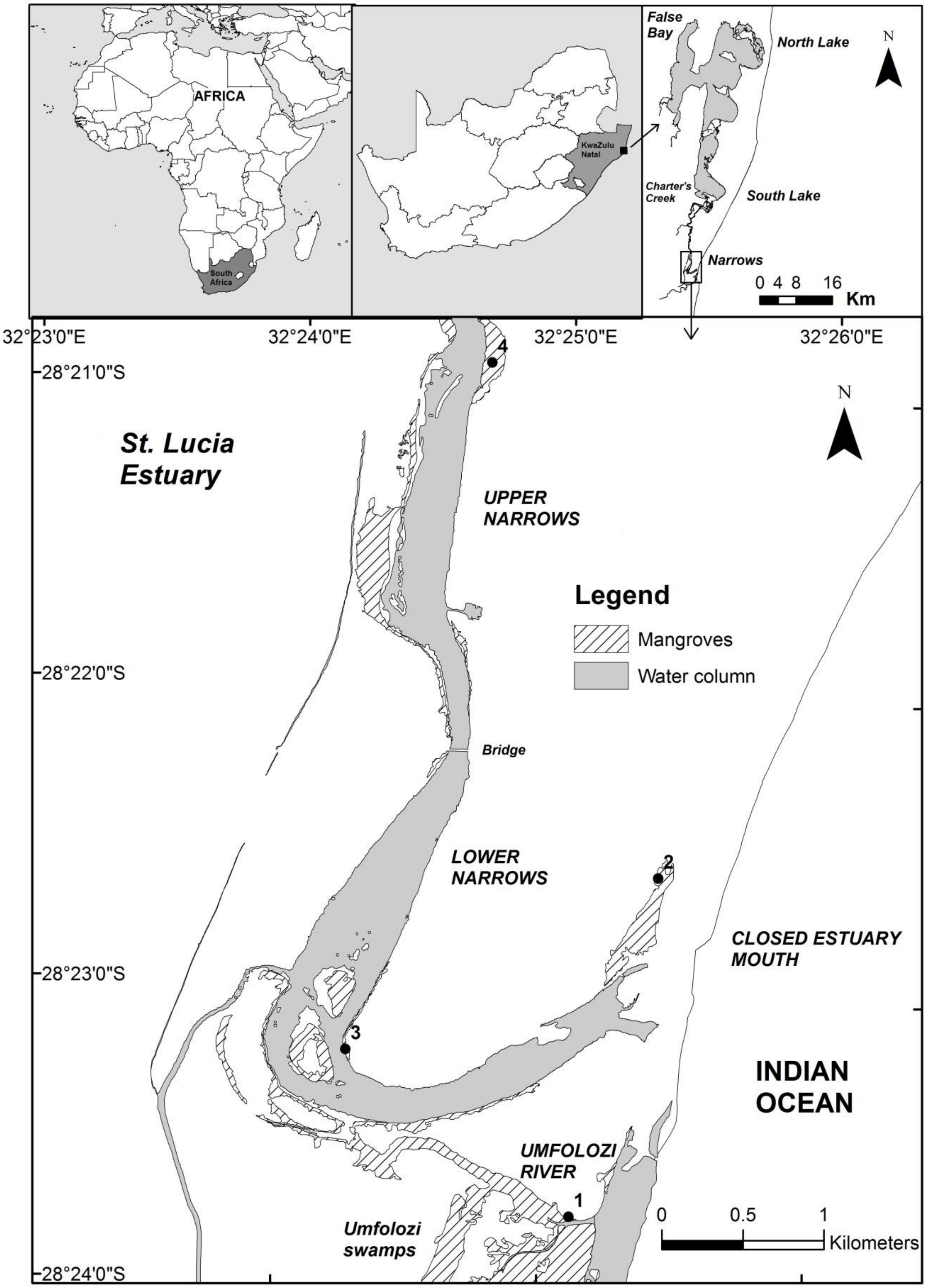

Figure 2.1: A site map of the four sampling localities for mangrove habitats within St. Lucia Estuary.

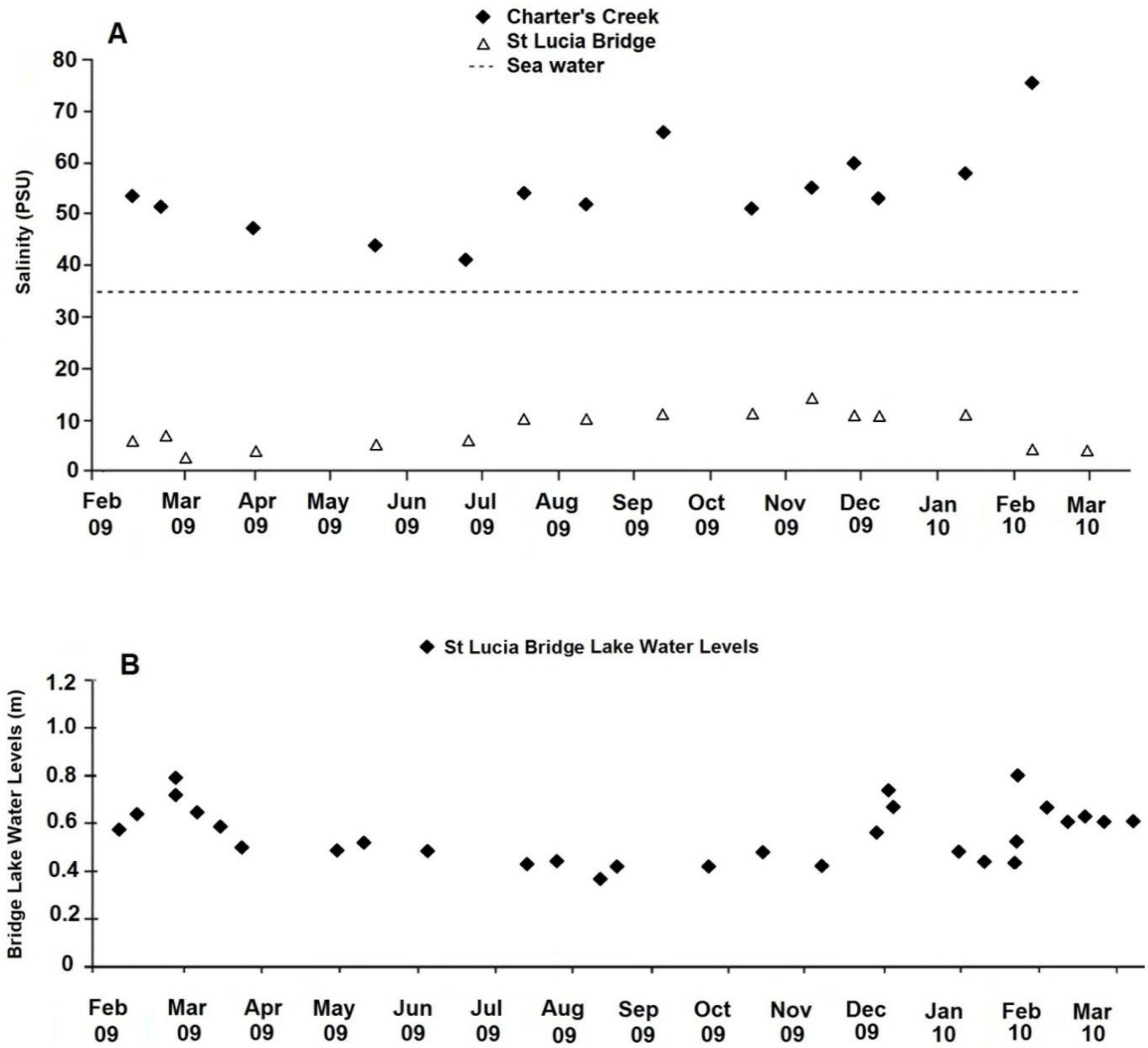

Figure 2.2: Salinity (A) and water level (B) of St. Lucia Estuary (adapted from Fox and Taylor long-term monitoring data for 2009-2010).

2.2.2 Mangrove site descriptions

Mangrove sites were sampled to represent the population structure and therefore quadrates were placed subjectively at the different sites within the St. Lucia Estuary in May 2010, and were chosen to represent areas with varying salinity and moisture regimes (Plate 2.1). Two mangrove species were represented: *Avicennia marina* was the dominant species which was present at all of the sites and *Bruguiera gymnorrhiza*, which was only present at Sites 2 and 4.

Site 1 represented the flooded site for this study (28°23’44.283” S; 32°25’6.585”E; Figure 2.1). Trees at this site established after Cyclone Demoina in 1984 (Taylor pers. comm.). This is the site where an artificial stream was created linking the Umfolozi River with the St. Lucia Estuary via the Narrows. This site would typically be exposed to seawater at spring tide when the Umfolozi mouth is open to the sea. However the estuary mouth had been closed and freshwater was persistent in the system. At the time of sampling the water in this site was fresh (0 PSU) and was flowing strongly due to rainfall the day before. In places water depth within the mangrove forest was between 0.2 to 0.5 m and the soil was waterlogged. Mangroves were represented by dense stands of *A. marina.* The Umfolozi floodplain and upstream has been converted to agricultural fields mainly sugarcane (Grenfell and Ellery 2009).

Site 2 was at a location known as Shark Basin. It represented a freshwater seepage site where dredge soil had been deposited in the past (28°22’6.01”S; 32°25’23.722”E; Figure. 2.1). Mangroves have established here since the 1960s (Taylor pers. comm.). Stands of *A. marina* and *B. gymnorrhiza* were interspersed with stands of *Phragmites australis* (cav.) Trin. Ex Steud. (common reed) and the mangrove fern *Acrostichum aureum* L. (golden leather fern or also known as mangrove fern). Both these species are tolerant of waterlogging and are indicators of freshwater seepage. There were signs that these plants could be out-shading mangrove seedlings.

Site 3 was at a location known as Honeymoon Bend on the main estuary channel. It represented mangroves fringing the main water channel of the Narrows (28°23’12.600”S; 32°24’14.646”E; Figure 2.1). Mangroves have established here since the 1970’s (Taylor pers. comm.). Tall trees of *A. marina* were located along the water’s edge. The sediment was waterlogged at the water’s edge but very dry in more landward positions. Many small mangrove saplings and seedlings were growing in the water under the tall trees.

Site 4 represented a dry site (28°20’54.390”S; 32°24”41.508”E), which had not been tidal for a long time (Figure 2.1). The sediment at the site was compacted and dry. This site also occurred adjacent to the main water channel of the Narrows, but the area sampled was in a landward location. The site was located at the end of a trail towards the estuary from the Crocodile tourist center. Mostly old and tall trees were found, but there were some smaller trees along an old creek. Both species of mangroves *A. marina* and *B. gymnorrhiza* occurred here. Mangroves have established here since the 1970s (Taylor pers. comm.).

Plate 2.1: Site 1 (A and B) dense stands of adult *Avicennia marina* trees, this area was flooded when the Umfolozi came down in flood; Site 2 (C) large adult *Bruguiera gymnorrhiza* trees and *Phragmites australis*; Site 3 (D) mangrove seedlings and sapling were present here mostly in the water behind the large adult *Avicennia marina* trees; Site 4 (E and F) was the dry site with extensive litter fall.

2.2.3 Mangrove characteristics

At each site four replicate quadrats (25 m^2) were marked out to measure population structure based on height classes. Plant height and density were recorded, as was species composition. Height classes were categorized as seedlings (< 50 cm height), saplings (> 50 – 100 cm height), juveniles (101 – 150 cm height) or adults (151- 200, > 200-500, > 500 cm height classes). The general phenology was noted (if plants were flowering, had calyxes or propagules). Five mature and fully expanded leaves were collected from five random trees per quadrat to determine leaf relative water content. The fresh weights of individual leaves from each quadrat were measured after drying the leaves for 48 h in an oven at ± 65 °C. Then the leaf moisture content was calculated as a percentage for the quadrats and the average of these was presented for each site.

Five fully expanded, mature leaves of similar size and thickness of the *Avicennia marina* were chosen for analysis of leaf cation concentrations (Na^+, K^+, Mg^{2+} and Ca^{2+}) following the American standard method EPA3052. A method for cation extraction was adapted from Suarez and Medina (2008). These samples were analyzed by diluting the cell sap 100 times with 1 % (v/v) sub-boiled HNO_3 using a Compudil 3 Autodilutor. Type I ultrapure water was used for all preparations and dilutions. Standard preparations of mixed Na^+, K^+, Mg^{2+} and Ca^{2+} were prepared in 1 % (v/v) sub-boiled HNO_3. The samples were then analyzed for cation concentrations using a Perkin-Elmer Sciex 6100 ICP-MS using a plasma power of 1.1 kW and a nebulizer flow rate of 0.93 L/min.

2.2.4 Sediment characteristics

Sediment samples were collected using a hand-held auger in each quadrat at each site (n = 24). Samples were collected at depths of 0 to 0.05 m ("surface") and 0.5 to 0.55 m ("subsurface") and stored in plastic bags for transportation to the laboratory for analysis. Porewater salinity was measured *in situ* using a hand-held Conductivity/TDS/Temperature meter (CON ID/100/200) following sediment collection when the porewater had filled the holes enough to cover the probe within the hole. Sediment redox potential (ORP) was measured *in situ* using a HANNA redox/pH meter (HANNA Instruments) with a platinum-gold tipped electrode. This was done immediately after sampling by placing the electrode into the sediment collected for laboratory analysis. Analysis in the laboratory was for sediment salinity, moisture and organic content, as well as sediment particle size. For sediment salinity and particle size the samples were air-dried before analysis. Sediment salinity (PSU) was measured using the method of Barnard (1990). The salinity of the solution was measured using a CyberScan, hand-held Conductivity/TDS/Temperature meter (CON ID/100/200). Sediment moisture content (%) was determined according to the methods set out by Black (1965). Sediment organic content (%) was determined according to the methods set out by Briggs (1977) and the hydrometer method was used for sediment particle size of sand, silt and clay fractions (Gee and Bauder 1986). All sediment samples were measured separately and the means were presented graphically for each site.

2.2.5 Faunal abundance

Crab burrows and snails on the trees within each quadrat were counted to determine if there was any relationship between the environmental characteristics and animal abundance. The mangrove snail (*Cerithidea decollate* Linnaeus 1767) lives on the trees within the intertidal area close to the water's edge, and they migrate up and down the trees according to the tides (Branch and Branch 1995). The size of the crab burrows can be used to determine the species distribution of the crabs. *Parasesarma catenatum* Ortmann, is a smaller species of mangrove crab and lives in lower intertidal areas close to the water's edge. A larger species *Neosarmatium meinerti* de Man, lives in the upper intertidal and supratidal areas. This species digs holes up to 2 m deep where it can get to the groundwater table and this enables survival in areas quite far from the shore (Branch *et al*. 2007). The abundance of the crabs was also used to assess the general status of the different mangrove sites.

2.2.6 Statistical analysis

Data were tested for normality using the Kolmogorov – Smirnov test, and those that were not normal were log-transformed, followed by Levene's test for homogeneity of variance on all data. This included leaf cation concentrations, leaf moisture content and the faunal component. The data were then subjected to One-way ANOVA, which compared these variables across sites in conjunction with a Post hoc Tukey HSD test. Two-way ANOVA was used to compare plant density for the different size classes across sites. For those data that were not normal the Kruskal-Wallis Multiple Comparison was used. These included plant height, moisture content, organic content, electrical conductivity, particle size and redox potential. Spearman Rank Order Correlation and Pearson Correlation analysis were used to determine correlation between environmental characteristics and plant height and density. Significance was determined at α=0.05 using Minitab 15 Statistical Software (Minitab Inc., USA).

2.3 Results

2.3.1 Mangrove characteristics

Population structure was compared across sites (Figure 2.3). Site 3 had significantly more seedlings ($H_{(df = 3;\ n = 4)}$ = 7.34, $p < 0.05$) and saplings ($H_{(df = 3;\ n = 4)}$ = 5.09, $p < 0.05$) at 10 600 ± 5093 seedlings ha^{-1} and 4 200 ± 2347 saplings ha^{-1} (adult to seedling ratio was 1.2 : 1), compared to Sites 1 and 4, where seedlings were absent. The lowest sapling density occurred at Site 4 (200 ± 200 ha^{-1}). The freshwater seepage site (Site 2) had significantly more adult trees especially in the height class between 151- 200 cm ($H_{(df = 3;\ n = 4)}$ = 2.06, $p < 0.05$), at 19 600 ± 7200 ha^{-1}, compared to all the other sites (Figure 2.3). Site 1 also had significantly greater density ($H_{(df = 3;\ n = 4)}$ = 8.7, $p < 0.05$) of adult trees taller than 5 m and a total number of adult plants of 23 800 ± 2 560 ha^{-1}, compared to Sites 4 (21 200 ± 1 879 ha^{-1}) and Site 3 (12 400 ± 1 082 ha^{-1}). Tree mortality was high at Site 1 (2 600 ± 840 ha^{-1}) and lower at Site 2 (1 400 ± 945 ha^{-1}). At Site 4 the *Bruguiera gymnorrhiza* were heavily browsed by game and it was suggested that this may have caused some mortality at this site although no browsed individuals were represented in the quadrats. For the different sites *Avicennia marina* was more abundant than *B. gymnorrhiza*, and *A. marina* was growing from the water fringe to the upper spring tidal areas while *B. gymnorrhiza* was found only in small stands mostly at the water's edge (Site 3) and at the freshwater seepage site (Site 2) (Table 2.1).

Table 2.1: Population characteristics for each species (average ± SE) at St Lucia Estuary (*Am* = *Avicennia marina* and *Bg* = *Bruguiera gymnorrhiza*).

Species	Density (number of individuals ha^{-1})			
	Total density	Seedlings	Saplings	Trees
Average	9 367 ± 1 915	3 575 ± 1811	2 175 ± 1088	22 350 ± 3 637
Am	13 026 ± 2 009	4 369 ± 2183	1 457 ± 696	21 285 ± 3961
Bg	1 492 ± 937	50 ± 13	1 800 ± 450	7 450 ± 1863

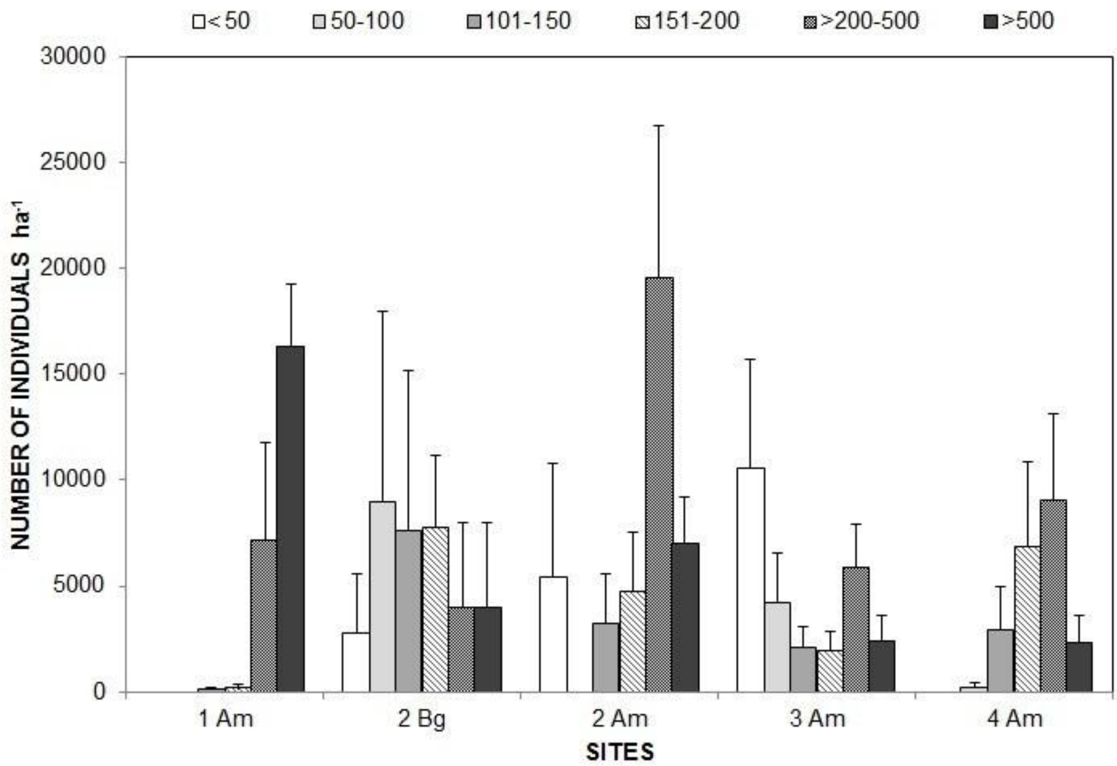

Figure 2.3: Density of mangroves (number individuals ha^{-1}) in different height classes at the different sites at St. Lucia Estuary (Am = *Avicennia marina* and Bg = *Bruguiera gymnorrhiza*) (bars show SE).

Although the leaf moisture content of *A. marina* at all sites was similar and close to 70 %, Site 1 (Figure 2.4) had significantly higher ($F_{(df\ =\ 64;\ n\ =\ 100)}$ = 5.6, $p < 0.05$) leaf moisture content than Site 4.

For the leaf cation concentration it was found that the Na^+ concentration within the leaves at Sites 1 and 3 was significantly higher than those at Sites 4 and 2 ($F_{(df\ =\ 35;\ n\ =\ 36)}$ = 6.45, $p <$ 0.05; Table 2). The Mg^{2+} concentration within the leaves at Site 1 was significantly higher than those at Site 2 ($F_{(df\ =\ 35;\ n\ =\ 36)}$ = 3.88, $p < 0.05$), but no significant differences were found between leaves of Sites 3 and 4. The K^+ concentration within leaves at Site 1 was significantly higher than in leaves at all other sites ($F_{(df\ =\ 35;\ n\ =\ 36)}$ = 9.87, $p < 0.05$). The Ca^{2+} concentration within the leaves showed an opposite trend as concentrations in leaves from Site 2 was significantly higher than that in leaves from Site 1, but no significance was found in the other sites ($F_{(df\ =\ 35;\ n\ =\ 36)}$ = 5.34, $p < 0.05$). Both Na^+ and K^+ were significantly lower in leaves from Site 2 compared to those from all other sites. In contrast, Ca^{2+} was significantly higher in leaves from Site 2 compared to those from all the other sites, while Mg^{2+} was similar in leaves at all sites.

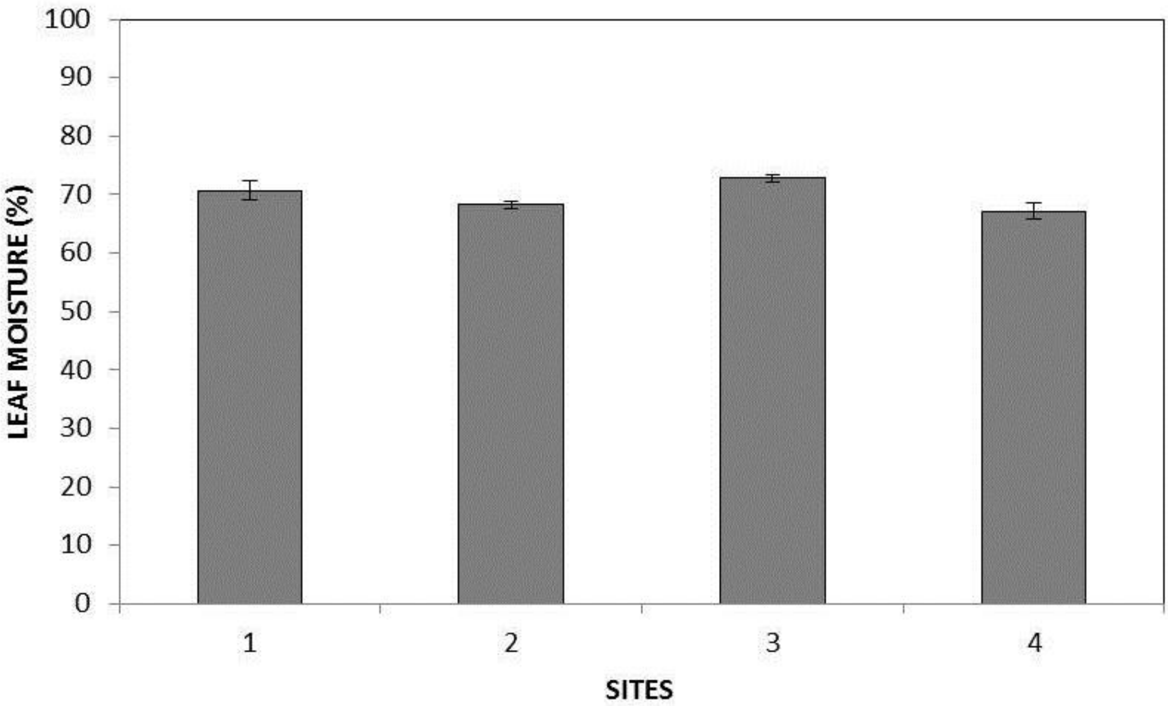

Figure 2.4: Leaf moisture content (%) of *Avicennia marina* at different sites at St. Lucia Estuary (bars show SE).

Table 2.2: Ion concentration (mg g^{-1} dry mass) of fully expanded mature leaves of *Avicennia marina* at different sites at St. Lucia Estuary (sig * and ± SE); ($p < 0.01$; DF= 35, n = 20).

Sites	Na^+	Mg^{2+}	K^+	Ca^{2+}
	F = 6.45	F = 3.8	F = 9.87	F = 5.34
1	45.0 ± 1.4 *	10.8 ± 1.20 *	23.8 ± 0.9 *	5.4 ± 0.2 *
2	24.7 ± 5.0 **	7.1 ± 0.9**	10.2 ± 3.0 **	10.8 ± 1.8 **
3	44.6 ± 1.7 *	7.8 ± 0.7	18.5 ± 0.9 **	7.1 ± 0.4
4	39.3 ± 1.7 **	9.2 ± 0.5	19.0 ± 0.2 **	6.3 ± 0.3

2.3.2 Sediment characteristics

The salinity of the sediment for both surface and sub-surface samples as well as porewater was significantly higher in the Narrows at Sites 3 (34.5 ± 2.1 PSU) and 4 (39.2 ± 1.6 PSU) ($H_{(df = 3; n = 56)}$ = 34.14, $p < 0.05$; Figure 2.5; i), compared to the sites close to the mouth of the estuary at Sites 1 and 2, where there was freshwater seepage. Site 2 had a porewater salinity of 4.9 ± 1.1 PSU and the flooded site, Site 1 had the lowest salinity of 1 PSU.

Sediment moisture at Sites 1 and 2 in both surface and sub-surface samples was significantly higher ($H_{(df = 3; n = 63)}$ = 17.02, $p < 0.05$) compared to that at Sites 3 and 4 (Figure 2.5; ii). At different depths Sites 1, 2 surface and 2 sub-surface sediment samples were significantly higher compared to both Sites 3 and 4 ($H_{(df = 6; n = 56)}$ = 47.19, $p < 0.05$). The surface sediment of Site 4 had significantly lower ($H_{(df = 6; n = 56)}$ = 47.19, $p < 0.05$) soil moisture than the sub-surface sediments of Sites 4 and 3.

Sediment redox potential (Figure 2.5; iii) was significantly lower ($H_{(df = 6; n = 60)}$ = 40.25, $p < 0.05$) in Sites 1 and 2 than Sites 3 and 4. Sediment organic content in Sites 3 and 4 surface samples was found to be significantly higher ($H_{(df = 6; n = 63)}$ = 18.45, $p < 0.05$) compared to Site 2 sub-surface samples (Figure 2.5; iv).

The sand fraction at Sites 3 and 4 was significantly higher ($H_{(df = 3; n = 63)}$ = 19.99, $p < 0.05$) in the surface samples and the silt fraction was significantly lower ($H_{(df = 3; n = 63)}$ = 24.62, $p < 0.05$) in both the surface and sub-surface samples compared to Site 2.

The clay fraction was significantly higher at Site 3 (both depths) and Site 4 (surface sample) ($H_{(df = 6; n = 63)}$ = 29.36, $p < 0.05$) compared to the other sites and depths (Figure 2.6).

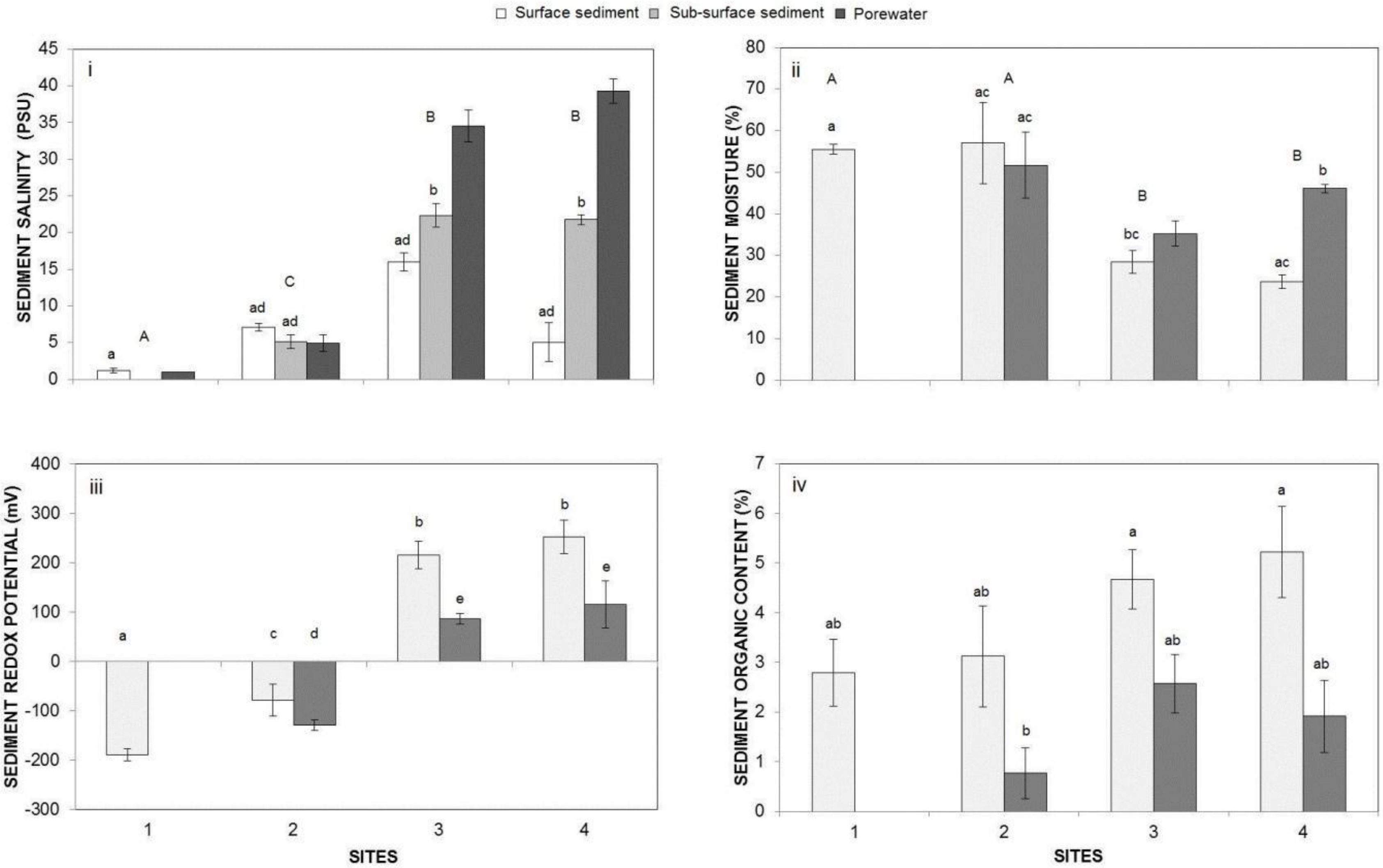

Figure 2.5: (i) Porewater and sediment salinity; (ii) sediment moisture; (iii) sediment redox potential and (iv) sediment organic content at different sites at St. Lucia (capital letters indicate significant differences between sites; small letters indicate significant difference between depths, means with similar superscript letters are not statistically significant and bars show SE).

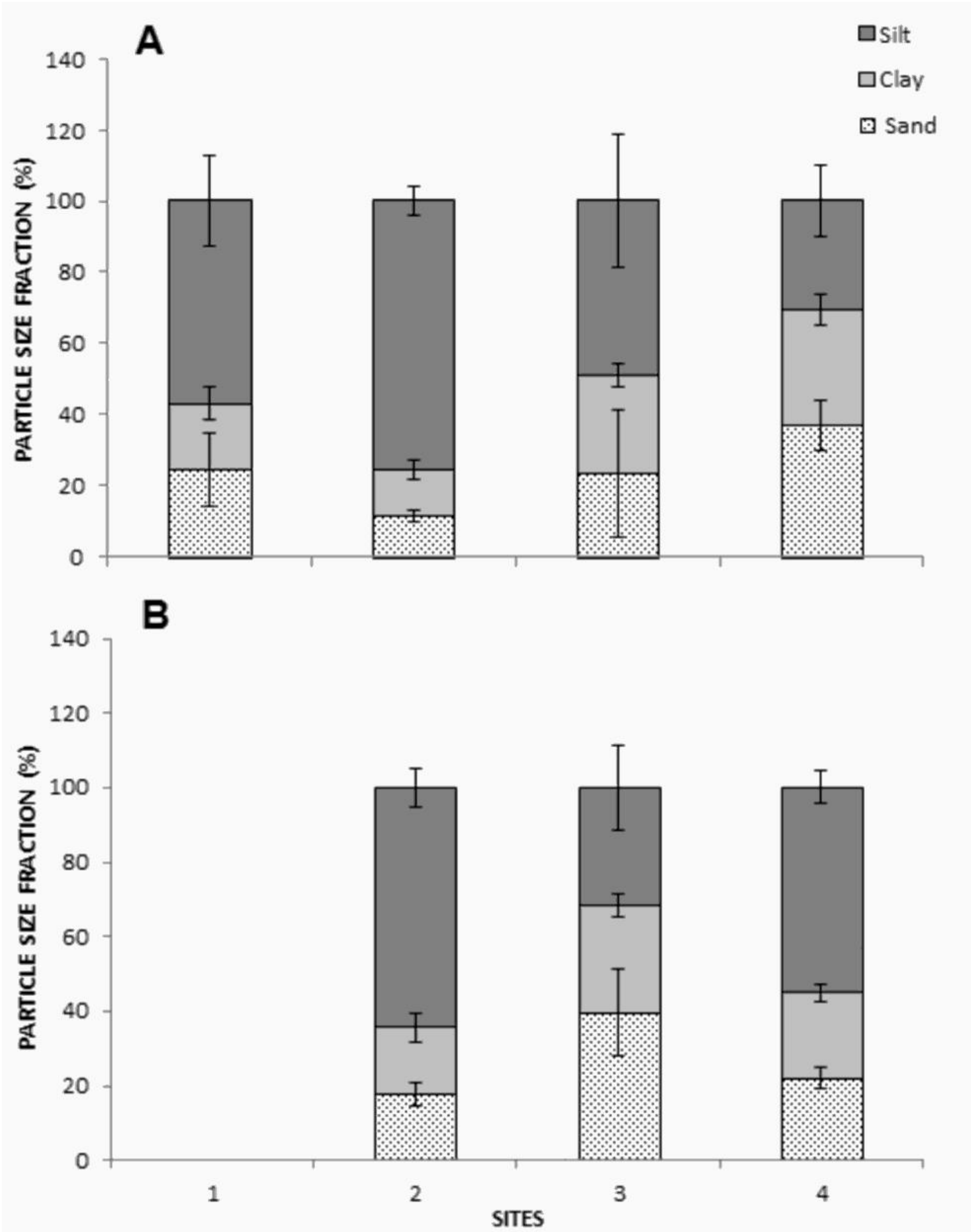

Figure 2.6: Sediment particle size, with A = surface samples and B = sub-surface samples for different sites at St. Lucia Estuary. Site 1 had no sub-surface samples as this site was flooded at the time of sampling (bars show SE).

2.3.3 Fauna abundance

No snails (*Cerithidea decollata*) were present at the dry site (Site 4). The freshwater seepage site (Site 2) had the highest concentration of snails with an average density of 186 900 ± 91 640 ha^{-1}, followed by Site 1 with 148 300 ± 58 750 ha^{-1} and finally by Site 3 with 85 200 ± 21 900 ha^{-1}. The densities of crab burrows (*Neosarmatium meinerti* and *Parasesarma catenatum*) were similar at different sites such that no significant differences were found between sites. Although crabs were present on the trees at Site 1, the number of crab burrows could not be recorded due to the strong river flow and high water level. Site 4 had an average density of 17 900 ± 9 120 burrows ha^{-1}, Site 2 had 15 500 ± 6 690 burrows ha^{-1} and Site 3 had the lowest density of crab burrows with only 14 800 ± 9 500 burrows ha^{-1}.

2.3.4 Correlation analysis

Correlation analysis showed that sediment redox potential had a negative correlation with plant height and density ($R = 0.57$, $p < 0.05$) as did sediment moisture ($R = 0.68$, $p < 0.05$). Positive correlations were found between sediment salinity and the cation Ca^{2+} ($R = 0.65$, $p < 0.05$), but a negative correlation was found between sediment salinity and K^+ ($R = 0.57$, $p < 0.05$). There was a significant positive correlation between snail abundance and surface sediment moisture ($p < 0.05$; $R = 0.72$) for Sites 1-3. No other correlations were found between mangrove population structure and environmental characteristics.

2.4 Discussion

2.4.1 Effects of climate cycles, catchment management and mouth closure

Reduced rainfall due to drought, water abstraction in the catchment area in conjunction with heavy sea storms can cause the mouths of permanently open estuaries to close. This may result in massive diebacks of mangrove forests due to long periods of inundation and the absence of tidal flushing. For example at Kosi Bay in South Africa, mouth closure in 1965 for five months resulted in high water level, submergence of the pneumatophores and mass mortality of the mangroves (Breen and Hill 1969). This study investigated different mangrove habitats in St. Lucia Estuary where the estuary mouth has been predominately closed since 2002 resulting in low water levels and no tidal exchange. Under natural conditions when the Umfolozi River was connected to St. Lucia, then a closed mouth brackish state would have existed during similar drought conditions (Van Niekerk 2004). Mangroves thrived in the system when the mouth area was dredged to keep it artificially open.

There was a period of open mouth conditions between March to August 2007 following Cyclone Gamede, which scoured the mouth area (Whitfield and Taylor 2009). Site 3 and possibly Site 1 would have been tidal during the open mouth condition, but Site 2 has remained at low salinity throughout due to freshwater seepage. Site 4 would have only been flooded at high spring tide, in the upper Narrows. However water levels within the Lake and the upper Narrows dropped after the mouth was breached in March 2007 (0.2 m lake mean water level) and remained low until August 2007 (Fox and Taylor 2010 - long-term monitoring data).

2.4.2 Mangrove population structure and environmental conditions

The objective of this study was to determine if environmental conditions such as sediment moisture, salinity and sediment redox potential had an effect on mangrove population structure under closed mouth conditions. Environmental conditions varied for different sites and mangroves were found to grow in various conditions in a range of salinity from freshwater to saline within St. Lucia Estuary. Surface sediment salinity was found to be low (< 25 PSU) in all the mangrove habitats, while porewater salinity was 39 ± 2 PSU. The low water level conditions have left the sediments within what is normally an intertidal region of the Narrows, exposed and dry (< 30 % surface sediment moisture). In contrast the waterlogged Site 3 and freshwater seepage Site 2 had greater than 55 % surface sediment moisture.

If the system receives predominantly freshwater under closed mouth conditions, it is expected that in areas where mangroves grow higher in the intertidal area, there will be an initial increase in seedling establishment, due to increased sediment moisture followed by species succession and an increase in freshwater tolerant mangrove species such as *Bruguiera gymnorrhiza*. However this scenario might be short-lived if the freshwater conditions prevail as competition with other freshwater mangrove-associated plants such as *Hibiscus tiliaceus* (L.) (sea hibiscus), *Phragmites australis* and *Acrostichum aureum* may result in mangrove seedlings being outcompeted over time. This was seen in studies presented by Dahdouh-Guebas *et al.* (2005b) where *Acrostichum aureum* increased to such an extent that there was a reduction in mangrove area cover in three coastal lagoons in southern Sri Lanka. In a South African example Rajkaran *et al.* (2009) reported that at

Mtamvuna Estuary in the KwaZulu-Natal Province the dense mangrove associates limited mangrove seedling establishment due to shading and prevented seedling dispersal and transport into other habitats as the reed (*P. australis*) stands prevented seedling transport by blocking channels. The movement of propagules by water currents is an important dispersal strategy (Allen and Kraus 2006). In this study the freshwater seepage site, Site 2 had the highest density of adults of *B. gymnorrhiza* (3 500 individuals ha^{-1}) and was growing amongst *P. australis* where possible competition would be expected between the mangroves and its associates.

No seedlings were recorded at Site 4, and there was a low density of saplings (< 500 individuals ha^{-1}) and large trees. This was because no new seedlings established in the dry mid- and upper- intertidal zone as intertidal water exchange had not occurred for years resulting in dry sediment conditions. Sediment moisture is an important factor for successful seedling establishment (Bhat *et al.* 2010). Castaneda-Moya *et al.* (2006) suggested that seedling distribution was determined by tides, conspecific adult trees and high salinity. Proffitt and Travis (2010) stated that the hydroperiod and elevation is an important factor for seedling survival and growth as well as for adaptation to changes in sea level with climate change. Clarke and Myerscough (1993) reported high density of *Avicennia marina* seedlings (15 600 individual ha^{-1}) in the intertidal region which had a normal tidal cycle, in contrast to the upper tidal region that is inundated at spring tides only, where seedling establishment was limited. This may have been due to poor propagule supply, dispersal, possible predation and hydrology (Clarke and Myerscough 1993). No seedlings or saplings were recorded for the flooded Site 1, where only adult trees were found. One of the factors limiting seedling establishment here may have been the strong surface flow from the Umfolozi River. The absence of large numbers of propagules and low sediment organic content at this site also indicates that propagules may have been transported from this site due to strong flow of the Umfolozi River.

Avicennia marina is considered to be a pioneer species among the mangroves, with a tolerance to a wide range of environmental conditions, and it is well adapted to arid environments where it can be found to grow in the driest habitats in the World such as the Arabian Gulf (Dodd *et al.* 1999) and also in Pakistan (Khan and Aziz 2001). Dodd *et al.* (1999) reported that shrub *A. marina* was able to cope with high salinity of up to 70 PSU and a low tidal range (about 1 m), as well as very low annual mean rainfall of less than 100 mm.

Qureshi (1993) found that most mangrove species that survive in salinity greater than 35 PSU show signs of stress through reduced propagule production such as production of immature flowers and producing buds that become senescent before propagules are produced. This again may influence the recruitment in dry habitats.

Castaneda-Moya *et al.* (2006) suggested that dry, saline soils determined mangrove distribution. They also confirmed that seedlings were not found in these dry areas, which would have higher sediment salinity and lower sediment moisture than the lower elevation and that are exposed to tidal flushing. In this study seedlings may have had a similar response, where mouth closure and long periods of drought have resulted in dry sediments at Site 4, due to higher elevations and absence of tidal cycles. The overall total average density of seedlings and saplings was low, indicating a lack of mangrove recruitment in the different habitats.

This study has shown that surface sediment moisture has been a determining factor shaping the mangrove population structure at the different sites, because as sediment moisture increased, plant height and density increased. This is because physico-chemical factors such as salinity (Lopez-Hoffman *et al.* 2006; Jayatissa *et al.* 2008; Ahmed and Abdel-Hamid 2007), inundation (Ye *et al.* 2004; He *et al.* 2007) and sediment redox potential (Pezeshki *et al.* 1997; Krauss *et al.* 2008) affect plant growth and survival.

2.4.3 Leaf tissue cation concentrations and environmental conditions

Sediment salinity and freshwater conditions influence the leaf cation concentrations and leaf moisture within the leaves of mangrove trees. Even with increased salinity plants have to maintain turgor within the plant. To counteract the osmotic gradient plants absorb additional inorganic ions (Smith and Smith 2001). This results in energy expenditure and therefore reduced plant growth, reproduction and survival capacities (Mehlig 2006).

In this study the mangrove leaves at the flooded site (Site 1) had significantly higher Na^+ concentrations within the leaves and the opposite was found for the freshwater seepage site (Site 2) due to spring tide and the Umfolozi mouth being open to the sea. Findings of Aziz and Khan (2001b); Naidoo (2006) and Lugo *et al.* (2007) showed similar trends. Therefore, the lowest leaf concentrations of Na^+ were recorded at Site 2, while Sites 1, 3 and 4 that had higher sediment salinity also had higher leaf Na^+ concentrations. For Site 1 the freshwater runoff and river flow resulted in low sediment salinity at the time of sampling. However high porewater salinity (> 35 - 38 PSU) and high cation concentrations were found in the mangrove leaves indicating previous saline conditions. The cation concentration in leaves would be an indicator of salt stress and could be useful in explaining the osmotic balance within plants (de Larcerda *et al.* 1985). Aziz and Khan (2001b) and Naidoo (2006; 2010) reported that total concentration of cations increased with increasing salinity. Leaf moisture was similar at all sites and it was suggested that plants were adapted to the difference in salinity by using ion accumulation to cope with osmotic stress.

2.4.4 Mangrove structure in relation to biotic interactions

Other factors influencing seedling establishment need to be considered such as crab abundance as these may reduce mangrove recruitment due to predation by mangrove crabs (Nagelkerken *et al.* 2008) and possible browsing by herbivores such as the kudu, which were seen to browse heavily in the mangrove habitats (Adams *et al.* 2012). In this study crab abundance may have had an effect on mangrove recruitment at sites that were at higher elevation, such as Site 4 which had the highest concentration of crab burrows (17 900 ± 9 120 burrows ha^{-1}). Furthermore, this site was devoid of any propagules and seedlings.

Examples of propagule predation by crabs was observed by Dahdouh-Guebas *et al.* (1998) in Kenya and by Farnsworth and Ellison (1997a) in Belize, Central America, where crabs had negatively affected mangrove recruitment in these forests. These aspects were not investigated in this study and no significant relationship existed between sediment moisture and crab burrow abundance. However, a relationship was found for snail abundance as no snails (*Cerithidea decollata*) were present at the dry site (Site 1). The freshwater seepage site (Site 2), had the highest density of snails (186 900 ± 91 640 ha^{-1}), indicating that snail abundance was related to sediment moisture. This sediment moisture may influence the distribution of this gastropod species, where it is usually found foraging at low tides in muddy substrates and migrates up mangrove tree stems at high tide (Steinke 1999).

2.4.5 Anthropogenic impacts and management recommendations

The mangrove trees tolerate the drier conditions of the sediment, which is still in their tolerance range, especially the adult trees. However many interacting environmental and biotic factors may have had an effect on seedling establishment. The recruitment of new individuals plays an important role in maintaining the functionality, sustainability and prevents degradation of a mangrove forest. In general the recruitment and propagule production would determine population structure in the future, and the extended recent drought prevents tidal exchange that is needed for seedling establishment and may result in mangrove forest diebacks. Examples of *A. marina* dieback in non-tidal swamps, examined by Jupiter *et al.* (2007), provide evidence of anthropogenic changes caused by developments that resulted in tidal isolation. At this site there was mass die back of these trees (148 ha^{-1} which was 22 % of the original mangrove area). Freshwater input and tidal exchange are both important for healthy mangrove populations within the St. Lucia Estuary and changes within the system would result in large scale mangrove forest degradation.

For a more effective conservation and management perspective of St. Lucia's mangrove habitats and the system as a whole, an adaptive management plan should be considered (Adams *et al.* 2012). This management plan should take into account these extreme environmental events of droughts and floods, which are predicted to increase in intensity and frequency due to climate change (Nicholls and Lowe 2004). These events may result in increased biodiversity loss as the mangrove habitats may not be able to recover from such extreme events in time (Lawrie and Stretch 2011). Climate change such as sea-level rise, anthropogenic developments and the past as well as possible future management interventions should be monitored on a regular basis to determine any changes within the system as a whole (Adams *et al.* 2012). The management plan would best include the "determination and implantation of the Estuarine Ecological Water Requirements (Reserve)" (Van Niekerk and Turpie 2012; Adams *et al.* 2012). Van Niekerk and Turpie (2012) suggest that the current status of the system can be improved by linking the Umfolozi River to the St. Lucia Estuary, as was naturally the case in the past, and this should improve resilience of the system to extreme environmental events such as droughts. This is necessary as this system provides many important environmental services as well as being of high social and ecological importance.

2.4.6 Conclusion

The study showed that sites with dry surface sediment were characterized by an absence of seedlings or saplings or had low seedling and sapling density. Soil capping due to dry soil conditions may have had an effect on recruitment which would also be influenced by propagule predation. No mangrove recruitment was observed at older stands in the Narrows and therefore it can be predicted that long-term mouth closure would influence survival of the mangrove forest at St. Lucia Estuary. For adult trees, high leaf cation concentration could be used as an indicator of stress, particularly for plants at the highly saline and flooded sites. Long-term data are needed to assess the influence of mouth closure on recruitment and survival of the mangrove forest at St. Lucia Estuary; however this study has shown that sediment characteristics in 2010 were unfavourable for mangrove growth at sites characterized by a lack of tidal flooding. The St. Lucia system, with its unique habitats, has proven to be very variable in the past, but if present conditions prevail it is expected that a shift in vegetation types would occur. Mangroves would be lost in some areas such as the dry sites in the upper intertidal areas of the Narrows and increase in others such as the Umfolozi swamps. For this reason continuous monitoring of the mangroves and their associate habitats will provide more insight on possible future developments.

Chapter 3: Mangrove expansion and population structure at a planted site

3.1 Introduction

Mangrove forests are found in marginal, intertidal areas (Tomlinson 1994) and provide habitat for many biota (such as birds, fish and invertebrate species) which use these as nurseries and breeding grounds (Barnes and Hughes 2004; Branch and Branch 2005). Other mangrove environmental services include high primary production that contributes to the energy transferred via detritus in food chains (Ewel *et al.* 1998). The forests act as buffers against sea storms and floods (Boesch 2002; Nicholls and Lowe 2004), which are expected to increase in frequency and intensity due to climate change (Day *et al.* 2008). The largest mangrove swamps are found in tropical and subtropical areas. The southernmost limit of mangrove forest in South Africa is at Kobonqaba (32°0'S; 28°29'E, which is about 60 km north east from Nahoon Estuary). South African mangrove forests only contribute 0.05 % of Africa's total mangrove (Wilkie and Fortuna 2003), which is a small area, but in spite of this they still add irreplaceable value to South African biodiversity (Adams *et al.* 2004; Emmerson and Ndenze 2007; Rajkaran *et al.* 2009). In South Africa, mangroves are protected by law under the National Forest Act No. 84, (1998) (DAFF 2008).

In many parts of the world, numerous attempts were made to re-establish mangroves in areas previously lost. Studies conducted by Kairo *et al.* (2001); Lewis (2005) and (Primavera *et al.* (2011), are only a few that have provided valuable information on the restoration of mangrove areas. In addition replanting is driven by the fact that mangroves are considered to be "the most threatened forests in the world, with 50 % already lost" (Valiela *et al.* 2001), and the rate of decline increasing worldwide (Dahdouh-Guebas *et al.* 2005a). However in this study mangroves have been planted in an area where they did not occur before and would relate more to studies in Hawaii where mangroves are invaders, introduced in the early 1900s to combat habitat loss of previously damaged coastal forest and open mudflats, thus increasing the biodiversity (Allen 1998; Lugo 1999; Allen and Krauss 2005). Some planted mangroves species such as the *Rhizophora mangle* L. have spread rapidly throughout Hawaii while other species such as *Bruguiera sexangula* (Lour.) Poir. have been found only at the originally planted sites and didn't spread to other areas (Allen and Krauss 2005). Chen *et al.* (2008) investigated the adaptation of introduced mangrove species *Sonneratia apetala* Buch Ham. to its environment and found that it had a competitive advantage over the native species such as *Avicennia marina* because it was better adapted to grow in low light conditions. Lugo (1999) suggested that mangroves are very resilient as long as their optimum environmental conditions are met. Lugo (1999) also mentioned that in some areas it is important to manage mangrove invasions as they caused shifts in vegetation types in introduced areas and Chimner *et al.* (2006) provided evidence that planting mangroves in non-native area such as Hawaii could prove detrimental to existing vegetation as well as the natural functioning of the system. Mangroves blocked creeks and the water became stagnant which resulted in health officials removing the mangroves (Chimner *et al.* 2006).

In South Africa, many mangrove forests occur in rural, unprotected areas and are threatened by overutilization through harvesting for firewood and building materials (Moll *et al.* 1971; Ewel *et al.* 1998; Field 1998; Rajkaran *et al.* 2004; Dahdouh-Guebas *et al.* 2005b). Excessive browsing by livestock and game also adds to the increasing pressures on these natural forests (Dahdouh-Guebas *et al.* 2006a; Shaw *et al.* 2007). Planting of mangrove forests in non-mangrove sites is rare in South Africa but one such example is at Nahoon Estuary. It is not known when exactly the mangroves were planted in Nahoon but it is suspected that this was in the early 1970s. *Avicennia marina* was planted first, followed a few years later by the planting of *Bruguiera gymnorrhiza* and *Rhizophora mucronata* among the larger *A. marina* trees.

The aim of this study was investigate the increase in mangrove cover in the intertidal zone of a planted mangrove forest in the Nahoon Estuary, and if environmental conditions characterised the present population structure as well as if mangroves would eventually replace salt marsh areas. Thus the hypothesis (1) tested was that the expansion rate of planted mangroves over 33 years was at the expense of natural salt marsh habitat. This would provide more information on mangroves growing at higher latitudes, where they were thought to not occur naturally due to lower annual average temperatures. Such a study would provide insights on future scenarios of possible shifts in vegetation types due to climate change (Nicholls *et al.* 2007) at one of the most southerly distribution sites worldwide.

3.2 Materials and methods

3.2.1 Site description

The Nahoon Estuary (32° 59’ 7.38” S; 27° 56’ 59.93” E) (Figure 3.1) runs through the city of East London in the Eastern Cape and falls within the East London Coastal Nature Reserve. The reserve adds value to the local environmental and recreational activities and the mangrove forest falls within the reserve. The climate is warm-temperate with annual temperatures ranging from 13 to 25 °C; with minimum temperatures in winter of 5.3 °C and maximum temperatures of 31.4 °C. Annual precipitation varies between 200 and 600 mm and most rainfall occurs during the spring and summer months (South African Weather Services).

3.2.2 Mangrove and salt marsh area cover

Esri ArcGIS Desktop 10 (2010) software, digital satellite images (Spot 5 2010, 2 m spatial resolution from 2004 to 2010), historical aerial photographs from Surveys and Mapping (1970; 1978; 1989; 1999 and 2004) and ground-truthing data of 2011 were used to establish past and present area cover of mangrove and salt marsh habitats. Salt marsh species included: *Bassia diffusia* (Thunb.) Kuntze; *Sarcoconia tegetaria* S. Steffen, Mucina & G. Kadereit; *Triglochin striata* Ruiz & Pav.; *Stenotaphrum secumdatum* (H. Walter) Kuntze; *Sporobulus virginicus* (L.) Kunth and *Juncus kraussii* Hochst subsp. *kraussii.*

3.2.3 Population structure and sediment characteristics

For the population structure only the species *A. marina* was measured as *B. gymnorrhiza* and *R. mucronata* were too few in number (< 5 and < 10 individual adult trees respectively). The mangroves were sampled within different zones (Z1 – Z3 – Figure 3.2), which were situated from the water’s edge inland, over the mud flat. Zone 1 was closest to the main channel (Figure 3.2) and Zone 3 with the largest trees where the mangroves were initially introduced, was furthest and at the end of an old creek.

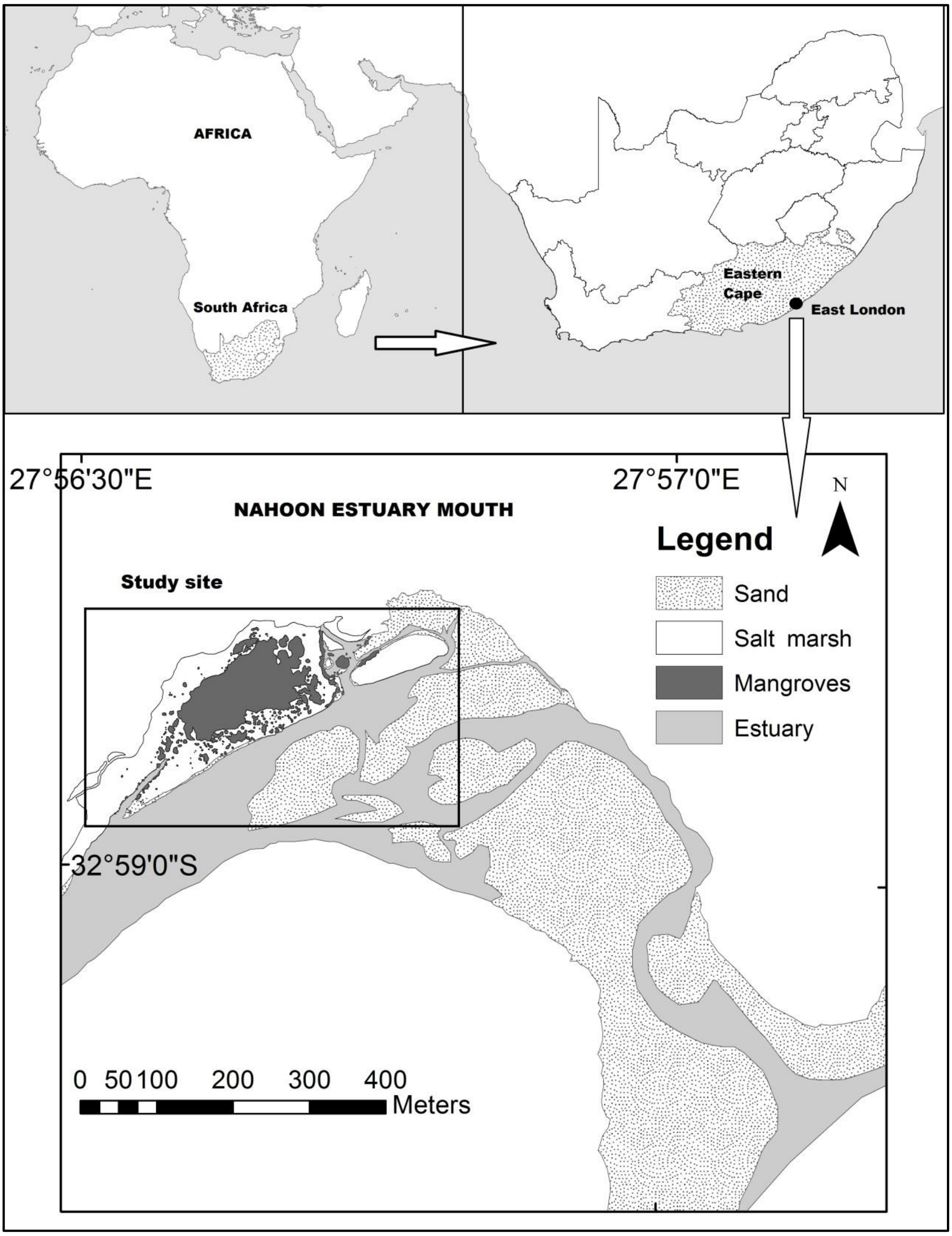

Figure 3.1: Map of South Africa and the location of the mangroves in the lower reaches of the Nahoon Estuary.

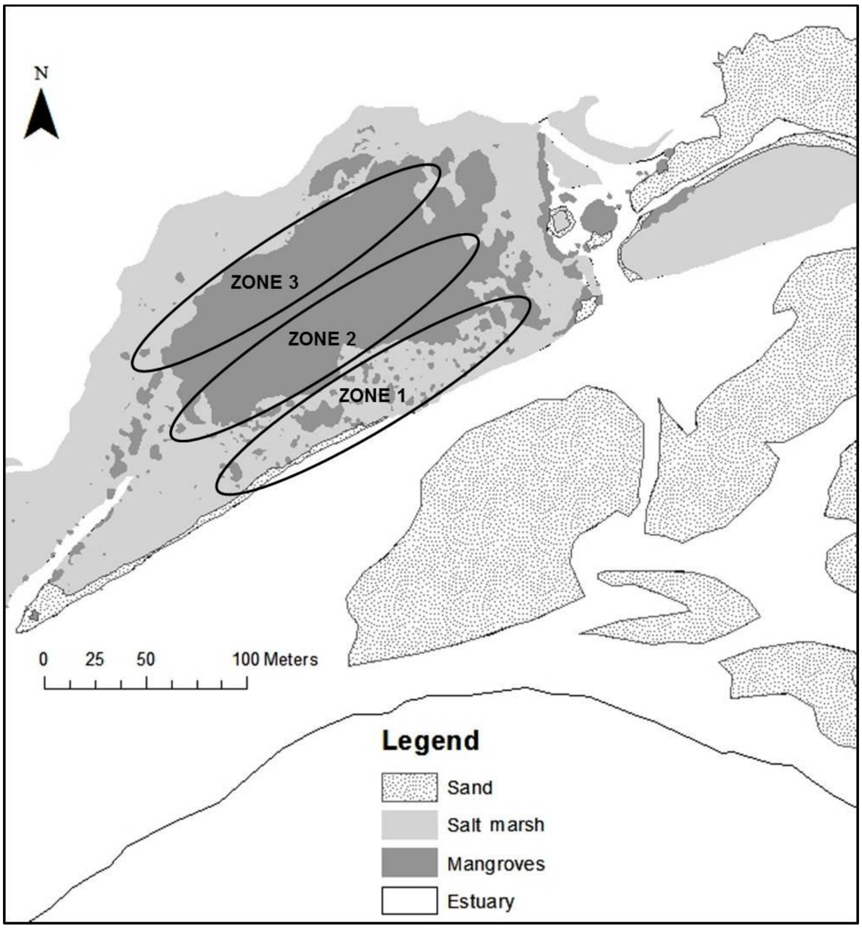

Figure 3.2: Map of the mangrove habitat within the Nahoon Estuary in 2011, indicating the location of the three different zones where the different transects were sampled.

In each zone a total of six 25 m^2 quadrats were measured along each of three transects. In each quadrat, mangrove plant height was recorded using a 1 m pole and metal tape measure. Within each quadrat the number of seedlings (< 50 cm), saplings (> 50 to < 130 cm) and adults (> 130 cm) were recorded. The circumference at breast height (CBH) was recorded for all adult individuals (> 130 cm) using a soft measuring tape and then converted to diameter at breast height (DBH).

In each zone, sediment was collected (n = 27) at 50 cm depth, for further analysis. Analysis in the laboratory included sediment salinity, sediment pH, sediment moisture and organic content as well as sediment particle size. Sediment salinity (PSU) was determined using the method of Barnard (1990), using a CyberScan, hand-held Conductivity/TDS/Temperature meter (CON ID/100/200). Sediment pH was measured using a hand-held HANNA redox/pH meter (HANNA Instruments) and a platinum-gold tipped electrode.

Sediment moisture content (%) was determined according to the method by Black (1965) and sediment organic content (%) according to the method by Briggs (1977), while sediment particle size was determined by the hydrometer method, as set out by Gee and Bauder (1986). Sediment redox potential and porewater salinity was allowed to collect in each hole and measured *in situ* at the different zones (n = 27) using a hand-held HANNA redox/pH meter (HANNA Instruments) with a platinum-gold tipped electrode and a hand-held Conductivity/TDS/Temperature meter (CON ID/100/200) respectively.

3.2.4 Data analysis

All statistical analysis was done using Minitab 15 Statistical Software. (Minitab Inc. USA). The data was first tested for normality. Then Kruskal-Wallis analysis of variance (ANOVA) was used to test differences between zones. One way ANOVA was done on parametric data, following which Tukey post hoc Honestly Significant Difference test was used to separate means. The Student T-Test was used to determine differences in area cover between the years. Pearson Correlation analysis was used to determine correlations between environmental characteristics and mangrove population structure. Pearson Correlation analysis was also used to determine correlations between mangrove and salt marsh area cover. For all analyses, significance was determined at α=0.05.

3.3 Results

3.3.1 Mangrove and salt marsh area cover

All *A. marina* trees appeared healthy and there was good recruitment as shown by the high density of seedlings. Mangrove expansion rate was 0.06 ha per annum (from 0.75 ha to 1.62 ha) while salt marsh cover was variable over the years and expansion rate was 0.09 ha per annum. There was however a steady linear increase in mangrove cover (Table 3.1), but no significant correlation was found between mangrove cover and salt marsh cover ($p > 0.05$) and salt marsh area cover was found to be variable over the years (2.5 ± 0.2 ha) (Table 3.1 and Figure 3.3).

Table 3.1: The changes in area cover (ha) salt marsh and mangrove area cover (ha) for the different years in the Nahoon Estuary

	Area cover (ha)	
Year	Salt marsh	Mangrove
1970	2.86	0
1978	2.04	0.75
1989	1.40	1.16
1999	3.68	1.20
2004	3.60	1.44
2007	2.95	1.51
2011	3.11	1.62
Rate of increase	0.09 ha yr^{-1}	0.06 ha yr^{-1}

3.3.2 Population structure of Avicennia marina

Significant differences in population structure were found between Zone 1 and 3 (Plate 3.1 A and B) for seedlings ($p < 0.05$) and adults ($p < 0.05$). The highest density of seedlings, saplings and adults were found at Zone 3 (Figure 3.4). There were significantly more seedlings (62×10^3 ha^{-1}), followed by saplings (16×10^3 ha^{-1}) and finally adults (9×10^3 ha^{-1}). The numbers of seedlings were highly variable in the different quadrats in Zone 3 as a result of high recruitment near a creek (Figure 3.3) at the back of Zone 3. Tree height (5.8 ± 0.25 m) and DBH (10.6 ± 1.4 cm), (Basal area = 0.74 m^2 ha^{-1}) were also significantly higher in Zone 3 compared to Zone 1 (4.1 ± 0.02 m) (DBH = 6.2 ± 0.8 cm), (Basal area = 1.70 m^2 ha^{-1}). Overall adult to seedling ratio was 1:6, mean height for trees was 5.07 ± 0.49 m, and mean DBH 8.3 ± 1.3 cm.

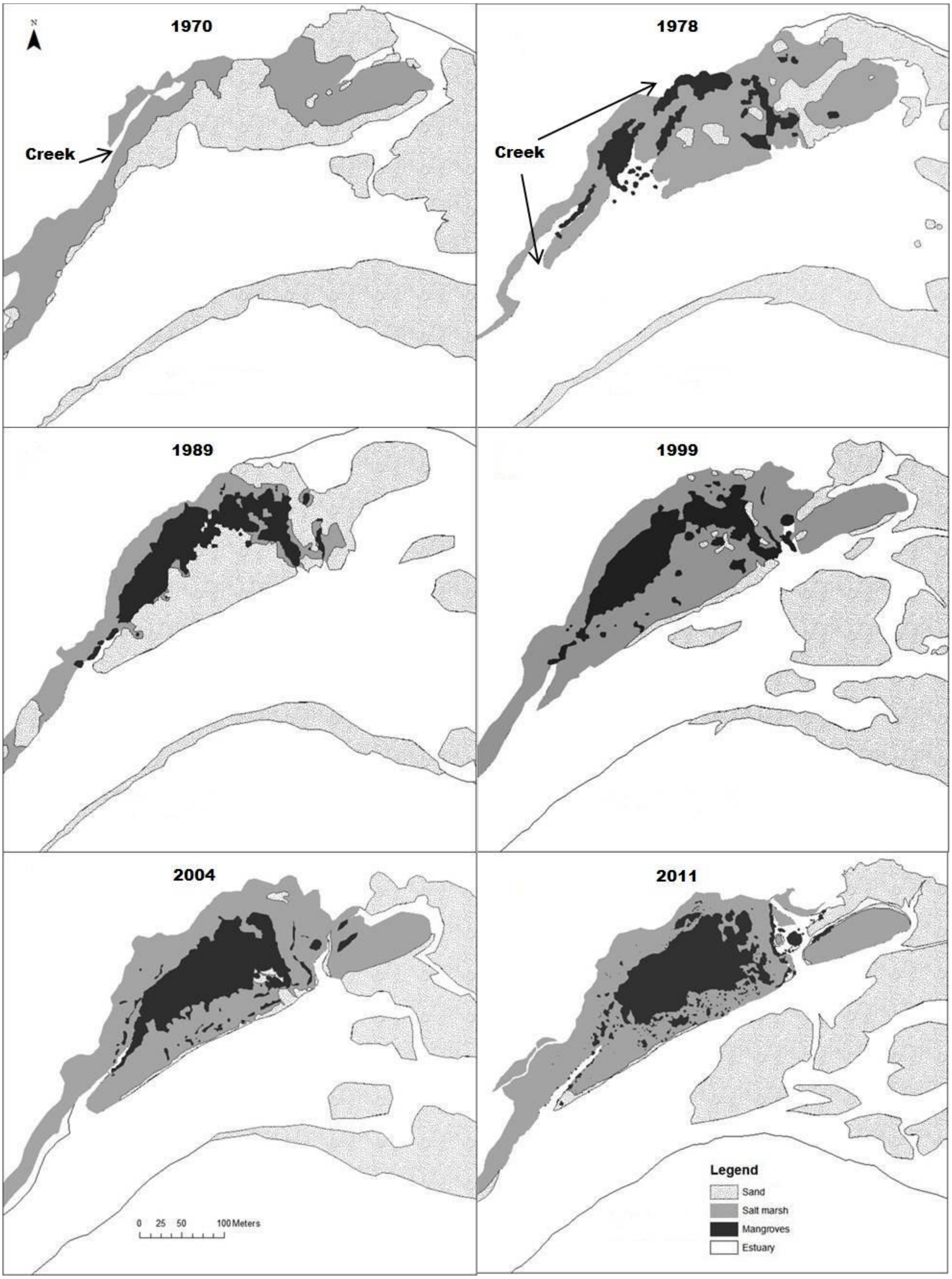

Figure 3.3: The expansion of salt marsh and mangrove forest area cover (ha) for the different years at the Nahoon Estuary.

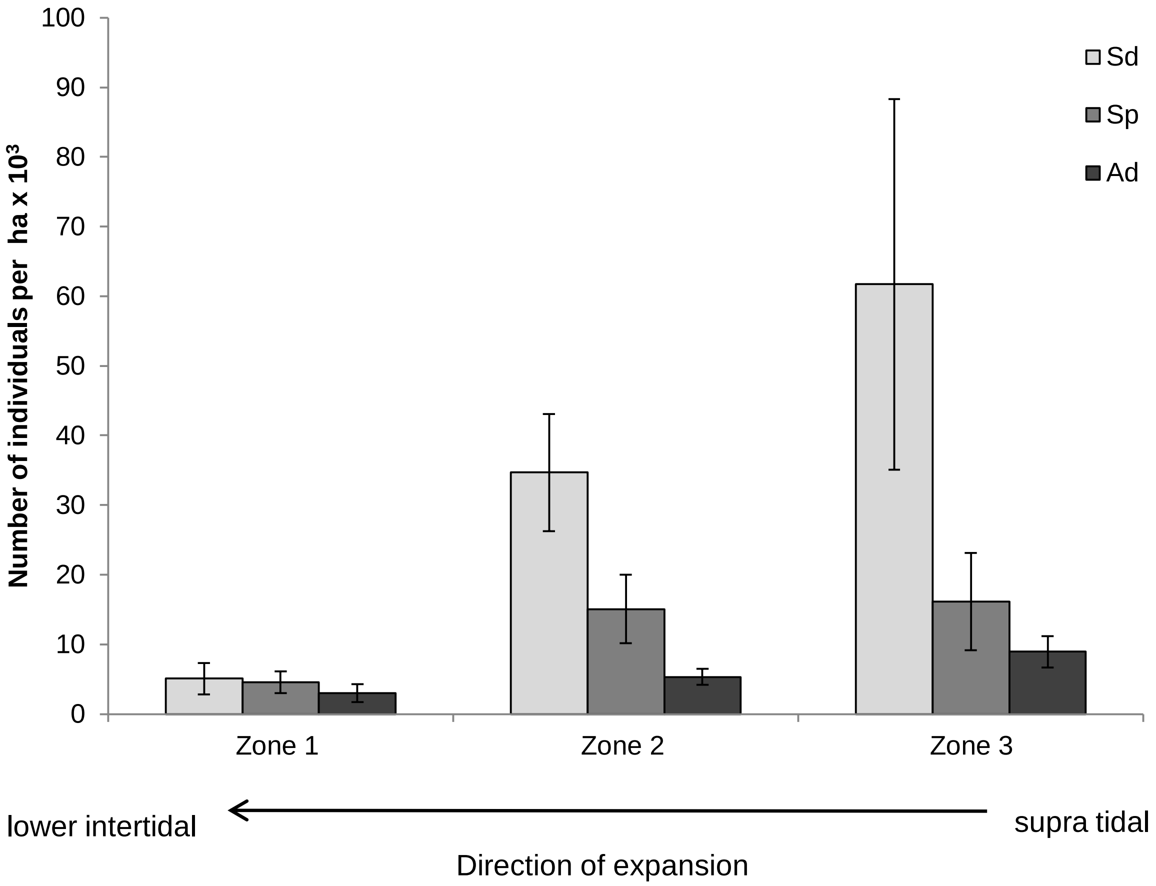

Figure 3.4: The number of individuals for the three zones. Sd = seedlings; Sp = saplings and Ad = adults. Bars = Standard Errors.

Plate 3.1: (A) Zone 1 with shorter and younger *Avicennia marina* trees; (B) Zone 3 behind old creek with old, tall and thick stemmed *A. marina* trees; (C) sedimentation and formation of new mud flats and (D) bank scouring and erosion after strong floods. (Images were taken in March 2011 after an extreme flood event).

3.3.3 Sediment characteristics

The sediment characteristics at the time of sampling were studied to determine potential abiotic drivers of the system (Table 3.2). Porewater depth was significantly shallower ($p < 0.05$) at Zone 1, relative to Zones 2 and 3. Sediment organic content ($p < 0.05$) was significantly lower in Zone 1 compared to Zone 3. With respect to sediment composition the silt content was significantly higher (60 ± 5 %) than the clay (12 ± 1 %) and sand (27 ± 5 %) at all three zones. All other environmental variables did not vary significantly ($p > 0.05$) between the different zones.

Table 3.2: Physical conditions at time of sampling (2011) at Nahoon Estuary. Values represent mean ± SE.

Parameters (n = 27)	Zone 1	Zone 2	Zone 3
Porewater characteristics:			
Depth (cm)	6.7 ± 6.7[a]	28.3 ± 1.3[b]	22.5 ± 6.1[b]
Salinity (PSU)	27 ± 3	31 ± 4	30.2 ± 1
Temperature (°C)	18.2 ± 0.3	18.7 ± 0.1	18.3 ± 0.3
Sediment characteristics:			
Salinity (PSU)	26.9 ± 2.9	31.0 ± 5.6	30.2 ± 0.2
Redox potential (mV)	-106.5 ± 11.1	-49.5 ± 61.3	-187 ± 20.9
pH	7.6 ± 0.1	7.6 ± 0.1	7.7 ± 0.1
Moisture content (%)	26.4 ± 1.8	31.7 ± 2.5	39.3 ± 2.5
Organic content (%)	2.3 ± 0.2[a]	4.6 ± 1.3[b]	4.0 ± 0.2[b]
Particle size:			
Sand (%)	19.4 ± 2.6	34.1 ± 8.0	29.3 ± 3.9
Clay (%)	10.7 ± 1.2	12.5 ± 1.1	13.9 ± 1.7
Silt (%)	69.8 ± 3.4	53.3 ± 8.3	57.0 ± 4.7

3.4 Discussion

Nahoon Estuary is situated outside of the natural distribution limit of mangroves in South Africa, which is 60 km further north of this site at Kobonqaba Estuary (32°0’S, 28°29’E), (Ward and Steinke 1982; Adams *et al.* 2004). Mangroves are therefore not native to this area; however global warming and increases in temperature could encourage mangrove forest expansion and eventually replace the salt marsh area. The dominant mangrove *Avicennia marina* would become a localized invader, as seen in many Australian studies (Morrisey *et al.* 2007; 2010). The objective of this study was to determine the extent to which planted mangroves have expanded over the last 33 years at Nahoon Estuary and if this expansion had resulted in a shift in vegetation types. An understanding of current and historic mangrove and salt marsh area cover, present mangrove population structure, recruitment and phenology as well as sediment characteristics would identify possible stressors and drivers of future mangrove distribution at one of the most southerly mangrove sites in the world.

Analysis of past aerial photographs showed that mangroves only established after the 1970s. The mangrove species *A. marina* was introduced at the beginning of the 1970s when a few specimens were planted within the creek area (Zone 3 in Figure 3.2 and Plate 3.1 B). This is where the tallest trees, with the greatest DBH and basal area were recorded and appeared older compared to the trees in the other zones. Zone 3 was possibly the centre of expansion as larger, taller trees of the species *A. marina* grew within the tolerable environmental limits and produced high quantities of healthy propagules (Allen and Krauss 2005; Cavalcanti *et al.* 2006) with an additional high number of established seedlings. Seedling availibity due to the lack of transportation of propagules over great distances may have had a great affect on seedling and species distribution (Allen and Krauss 2005). From this planted area the mangroves have expanded towards the channel by first colonising the sides of a much wider creek (Figure 3.3 in 1978) and by 2011 the mangroves covered just over half of the available mud bank, where increased sedimentation was apparent (comparing 1970 and 1989 in Figure 3.3) (Plate 3.1 C) in the creek, while the remaining area was colonized by salt marsh. It was also noted that mangrove expansion was mainly seawards which was similar to studies from New Zealand. The most southerly mangroves in South Africa (32°0’S) are comparable in latitude to the most southerly in New Zealand, which represents the most southern mangrove forest sites in the world (Crisp *et al.* 1990). There are many documented studies on mangrove expansion into salt marsh areas and it is a well-known phenomenon in Australia but less common in New Zealand (Morrisey *et al.* 2010). This is because in New Zealand the mangroves usually have a seaward expansion and to a lesser extent a landward expansion as mentioned above. Propagule establishment would be the largest determining factor for mangrove distribution. *Avicennia marina* propagule dispersion is largely influenced by the tides and currents, propagule buoyancy and lifespan (Clark and Myerscough 1991; de Lange and de Lange 1994). In addition those propagules that have been transported by tides and currents are then further influenced by the environmental parameters of the habitat such as temperature, freshwater supply, light, salinity, water-depth, wave energy, pests and predation (Clarke and Allanway 1993; Kathiresan and Bingham 2001).

Studies conducted in the south east of Australia by Saintilan and Williams (2000) and Wilton (2002) showed that the main cause of salt marsh decrease was that mangroves (especially *A. marina*) encroached landward into salt marsh area. This is because salt marsh mainly grows at higher elevations and more landward compared to mangroves (Morrisey *et al.* 2007; 2010). They suggested that the landward expansion replaces the salt marsh area and in south-eastern Australian estuaries there has been as much as 42 % of salt marsh reduction due to mangrove encroachment from the 1940s to the 1990s (Wilton 2002). Coleman (1998) also reported on the landward expansion of mangroves and decrease in salt marsh area in South Australia in 1985-1993. However a seaward expansion of mangroves was also noted in areas that had been previously bare. Rogers *et al.* (2006) suggested that mangrove expansion would be greatest in areas with the greatest surface elevation change, (ranging between -2.6 ± 2.07 to 5.54 ± 2.15 mm yr^{-1} within the mangrove zones and between -0.16 ± 0.94 to 3.25 ± 1.71 mm yr^{-1} within the salt marsh zones) indicating that mangroves can tolerate longer inundation periods and tidal exchange compared to salt marshes. They also mentioned that salt marsh may not keep pace with sea-level rise, which would result in an increase in the frequency of tidal inundation, which again would give the mangrove propagules an advantage over salt marsh establishment. This is because mangrove expansions are mainly dependent on the rate of sea-level rise, sedimentation (Rogers *et al.* 2006) and rainfall (Eslami-Andargoli *et al.* 2009). In New Zealand the more common scenario is seaward expansion (Park 2004) with similar findings to this study. In Park's (2004) study an "edge invasion" was noted, where 5 to 10 m boundary of mangroves have been growing on the edge of the salt marsh area and it was suggested that this boundary was kept "stable" because healthy and thick salt marsh cover prevented the establishment of mangrove propagules. Mangroves were only able to establish in sparse and patchy salt marsh areas due to available open areas. Mangroves also tend to establish close to creeks and inlets as found in this study behind Zone 3 where tidal inundation would have transported propagules for mangrove colonization.

In this study *A. marina* had a high density of seedlings and saplings (Figure 3.4) as well as a high seedling to adult ratio, which is instrumental in its expansion as they may outcompete salt marsh for available space in the future. The other two mangrove species *B. gymnorrhiza* and *R. mucronata* at the time of sampling were represented only by a few individuals and it was argued that these may not impact salt marsh area as their numbers are too low (< 10 adult trees).

One of the most important parameters that limits distribution is salinity (Suarez and Medina 2008). *Avicennia marina* are found to prefer salinity ranges that fall within 5 to 35 PSU (Clough 1984; Naidoo 1987) while *B. gymnorrhiza* grew better at salinity greater or equal to 10 PSU but less than 30 PSU (Ward 1976; Naidoo 1990) and *R. mucronata* prefers to grow in 17.5 PSU (Aziz and Khan 2001b; Khan and Aziz 2001; Hoppe-Speer *et al.* 2011). In this study the salinity range (27 – 30.2 PSU) was within the tolerance range of *A. marina* and was in the upper limits for *B. gymnorrhiza*, but was exceeding the salinity tolerance for *R. mucronata*. Thus *A. marina* was more proliferate in expansion than the two latter mangrove species.

Due to its location close to the permanently open mouth, mangrove and salt marsh plants trap fine sediments, resulting in sediment deposition (Hugarth 1999; Duke 2006; Adame *et al.* 2010b) and thus adding to the area that can be colonized. This may explain the high percentage of silt composition of the sediments. Adame *et al.* (2010b) suggested that the geomorphological setting of mangrove habitats has a profound effect on the spatial distribution of the sediment composition which again will affect other biota. In addition, a dynamic relationship seems to exist between mangroves and salt marsh, with the mangroves responsible for sediment accretion and marsh expansion, and the salt marsh can serve as nursery for mangrove seedlings (Lewis 2009). The constancy in salt marsh cover with concurrent increase in mangrove cover and marsh area can only be indicative of such a dynamic relationship.

Furthermore, estuaries are known to be ‘dynamic and ever changing ecosystems’ (Woodroffe 2000) and mangroves flourish in estuaries that provide a combination of sheltered tidal area, enough muddy sediments and in a high rainfall area (Yulianto *et al.* 2004). However dynamic events such as floods can momentarily alter species composition. These events may flush the estuary, scour estuary banks seen in Plate 3.1 (D) in March 2011 and deposit large amounts of debris. This may result in the temporary die back of salt marsh and mangroves. In addition, salt marsh species which co-habit this area are adapted to unpredictable fluctuations in environmental conditions and are very resilient, with high number of seeds in the sediments which affords them competitive ability (Riddin and Adams 2009).

3.1 Conclusion

In essence, our results indicate that mangroves may extend into even higher latitudes than their existing latitudes with the changing climate and sea-level rise. The results of this study show that at present there is ‘interplay’ between mangroves and salt marsh similar to the studies found in Stevens *et al.* (2006) where salt marsh area will border mangrove settings and the area covered will be co-dependent on climatic conditions and available habitat. However the expansion of mangroves into salt marsh and other extant vegetation types has been inconclusive so far and continuous monitoring may reveal if this non native forest poses a threat to the salt marsh habitat in future. It can therefore be said that the hypothesis which tested if the expansion rate of planted mangroves over 33 years and that these would be at the expense of natural salt marsh habitat, has to be rejected since there had been no evidence supporting this statement in this study.

This planted forest provides a much needed opportunity in long-term, quantitative ecological research as comparative studies to natural forests. This would determine future relationships between mangrove and salt marshes as well as their adaptations or resilience to climate change. The continuous protection of the Nahoon mangrove habitat is thus important. However it is not advisable to plant mangroves in non-native areas because long-term impacts on these habitats are unknown.

Chapter 4: Cattle browsing impacts on white mangrove *Avicennia marina*

4.1 Introduction

The demand for natural resources is continually increasing with human population and coastal developments. In many poverty-stricken areas, people are directly dependent on natural resources (Semesi 1992; Castley and Kerley 1996) and mangrove forests are under ever increasing pressure due to over-utilization. Within the Eastern Cape, South Africa, as much as 1.04 ha of mangrove forests are lost per year due to natural and anthropogenic impacts (Adams *et al.* 2004). Most mangrove forests in the Eastern Cape fall within communal lands where mangrove wood is an important source of fuel (fire wood). Whole trees are felled to make poles used as materials for houses and fences (Rajkaran *et al.* 2009). Examples of extensive harvesting for poles were studied at Mngazana Estuary in South Africa by Rajkaran *et al.* (2004; 2009) who concluded that harvesting was not sustainable and contributed to mangrove degradation in this area.

Mangroves also provide fodder for browsers such as cattle, goats and wild game. These often browse on mangrove leaves and depend on the lush foliage in the dry season, especially during drought years. This is also true in other parts of the world. For example, in Pakistan, mangrove foliage is harvested and used as fodder for camels (Shah *et al.* 2007), and in India (Dahdouh-Guebas *et al.* 2006a). Dahdouh-Guebas *et al.* (2006a) reported that feral water buffalo were responsible for mangrove degradation in some parts of the country. The intensive browsing by water buffalo on mangroves had both environmental and social impacts, as local villagers and conservation officials were in conflict with each other. However, Shah and Kamaruzaman, (2007) provided a few solutions to similar situations. They reported on mangrove conservation programs in Pakistan, where poverty was the main cause of mangrove degradation. They addressed problems of overutilization and the challenges involved with conserving the mangrove forests. These programs involved communities in more sustainable resource management activities such as replanting and restoring degraded mangrove forests. They also implemented conservation plans for existing forests.

One of the challenges at the Nxaxo Estuary is the open access to these forests, where cattle graze freely, and restoration would only be successful if cattle and other browsers were restricted in the replanted areas. This was found by Saifullah *et al.* (2007), where propagules disappeared within the study plots. It was suspected that many had been eaten by herbivores such as cattle, but also by crabs, which influence seedling establishment and studies regarding propagule predation by Dahdouh-Guebas *et al.* (1997); Steele *et al.* (1999) and Delgadon-Sanchez *et al.* (2001) showed that the presence of crabs and snails decreased recruitment in both natural and planted forests. Shah and Kamaruzaman (2007) and Dahdouh-Guebas *et al.* (2006b) conducted studies and ran community programs that involved local populations in mangrove rehabilitation and conservation. A better understanding of environmental, economic and social relationships in relation to mangrove forests was gained, and this knowledge can be integrated and used to develop more sustainable management and conservation plans.

For this research the study area was the former Transkei region of the Eastern Cape, where estuaries are considered to be relatively undisturbed and pristine. However, the long-term effects of cattle browsing have not yet been quantified and this may have an important impact in rural mangrove habitats in South Africa as well as globally.

The objective of this study was therefore to investigate the impact of cattle browsing and trampling on the growth and survival of the mangrove *Avicennia marina.* Hypothesis (1) stated that 'browsed mangroves will have reduced growth; browsed trees would show horizontal but not vertical growth'. Hypothesis (2) stated that 'cattle browsing on the shoots and growth tips of the stunted mangroves will reduce flowering and fruiting production'.

4.2 Materials and Methods

4.2.1 Site description

Figure 4.1 depicts a map of the study site, the Nxaxo Estuary, and also shows the 5 sample regions. This is the southernmost limit for mangroves in their natural habitat within South Africa. The two estuaries, Nxaxo and Ngqusi at Wavecrest (32° 35' S; 28° 31' E), share the same mouth and are considered to be permanently open with large tidal flushing (Harrison *et al.* 1997).

Figure 4.2 shows the rainfall and temperature data for August 2011 to July 2012 for the study area, which falls within a summer rainfall region. Staff members at the nearby Wavecrest Hotel collected rainfall data and this is summarised in Figure 4.2 A. The total rainfall for this period was 683 mm, the lowest annual minimum temperature was 7 °C, the highest annual maximum temperature was 34 °C and the annual mean temperature was 20.1 °C, as can be seen in Figure 4.2 B.

For the closest possible comparison, data from East London, which is located 160 km south of the study site, was used. The average annual rainfall in East London is 930 ± 303 mm (South African Weather Services 2000 – 2012). The total annual rainfall for 2010 and 2011 were 717 and 1 205 mm, respectively. The total rainfall for the first six months of 2012 was ± 500 mm. The annual average maximum temperature in the summer months was 23.4 ± 0.5 °C and the annual average minimum temperatures in the winter months was 14.4 °C (South African Weather Services 2000 – 2012).

4.2.2 Site selection

Five permanent cattle exclusion plots were set up along the Nxaxo Estuary water channel, as shown in Figure 4.1. These were sampled annually, in July, from 2010 to 2012. The study site, in the fringing forest, was adjacent to open grasslands on the east bank. On the opposite (west) bank, a lush coastal dune forest borders the fringing *Avicennia marina* trees. Within all plots, only *A. marina* was represented. Most individuals were found to have stunted growth and were dwarfed (Plate 4.1 B). Mangrove characteristics and environmental parameters were measured to determine whether other environmental factors could have contributed to the stunted growth. The total annual rainfall between 2000 and 2012 is shown in Figure 4.2 for East London. Figure 4.3 shows average rainfall for East London; this was 930 ± 303 mm y^{-1} (South African Weather Service 2012).

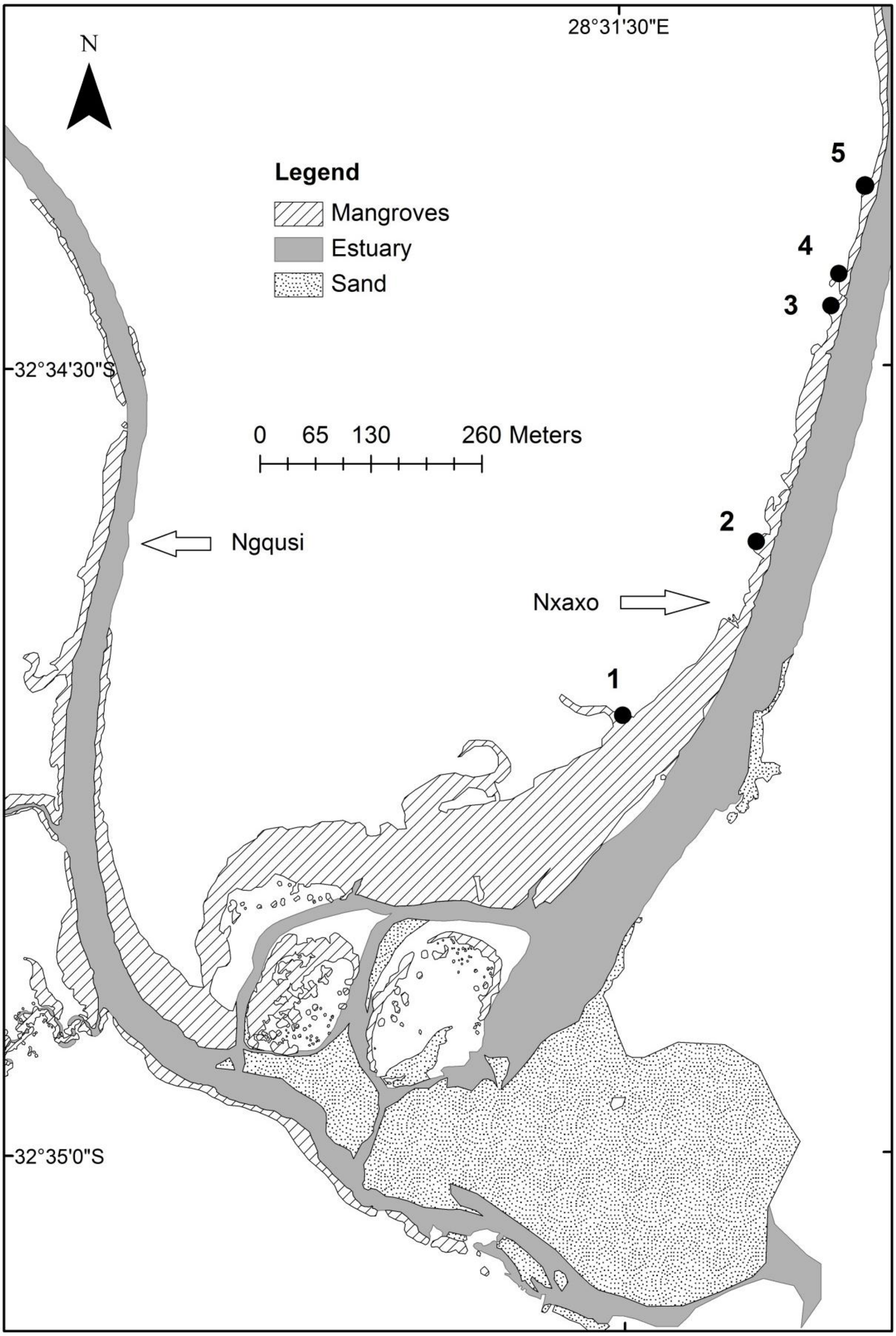

Figure 4.1: The Ngqusi / Nxaxo Estuary and the location of the plots (1 - 5) sampled.

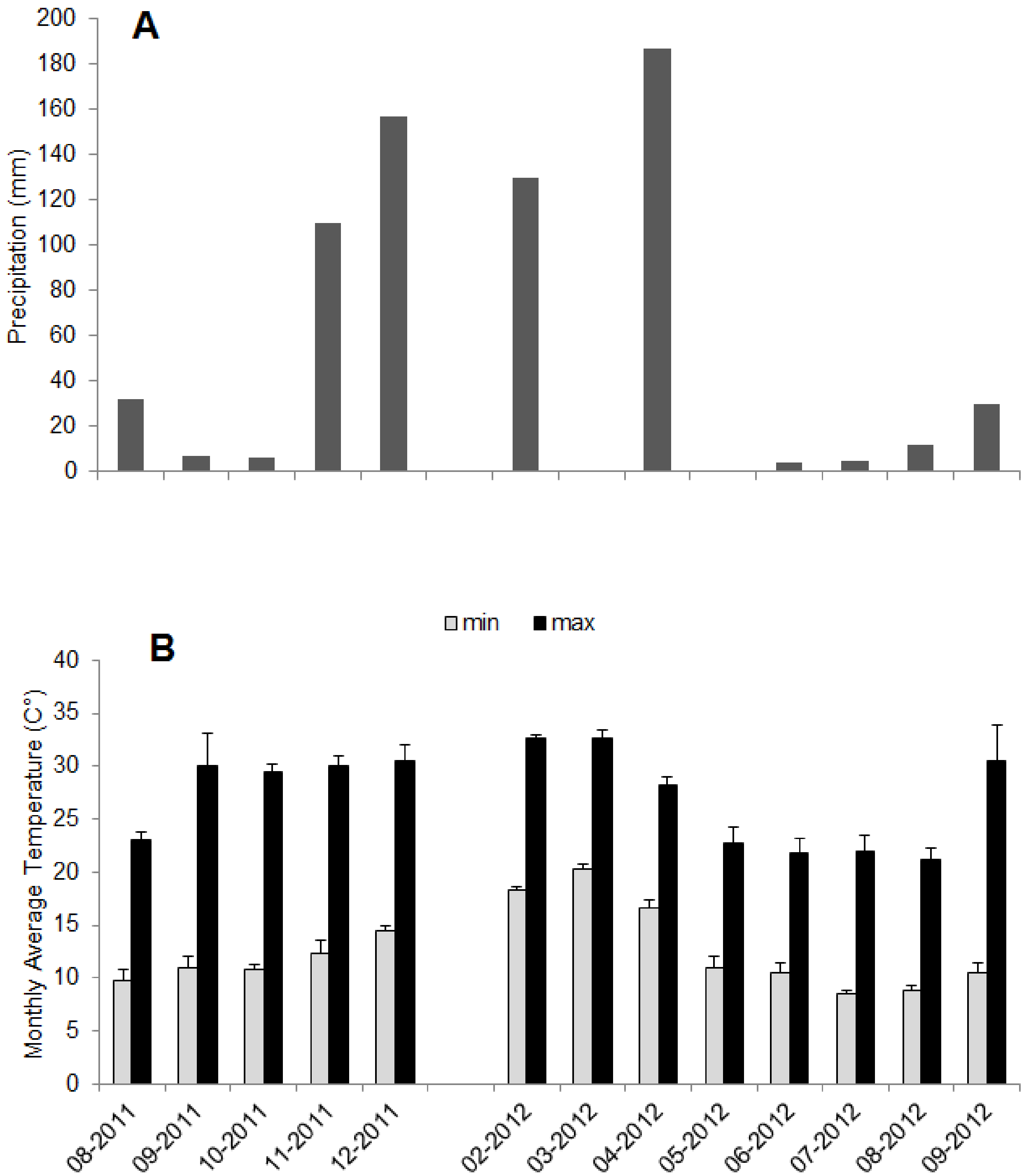

Figure 4.2: (A) The monthly total rainfall and (B) the monthly average minimum and maximum temperatures at Wavecrest from August 2011 to July 2012 (Bars = SE).

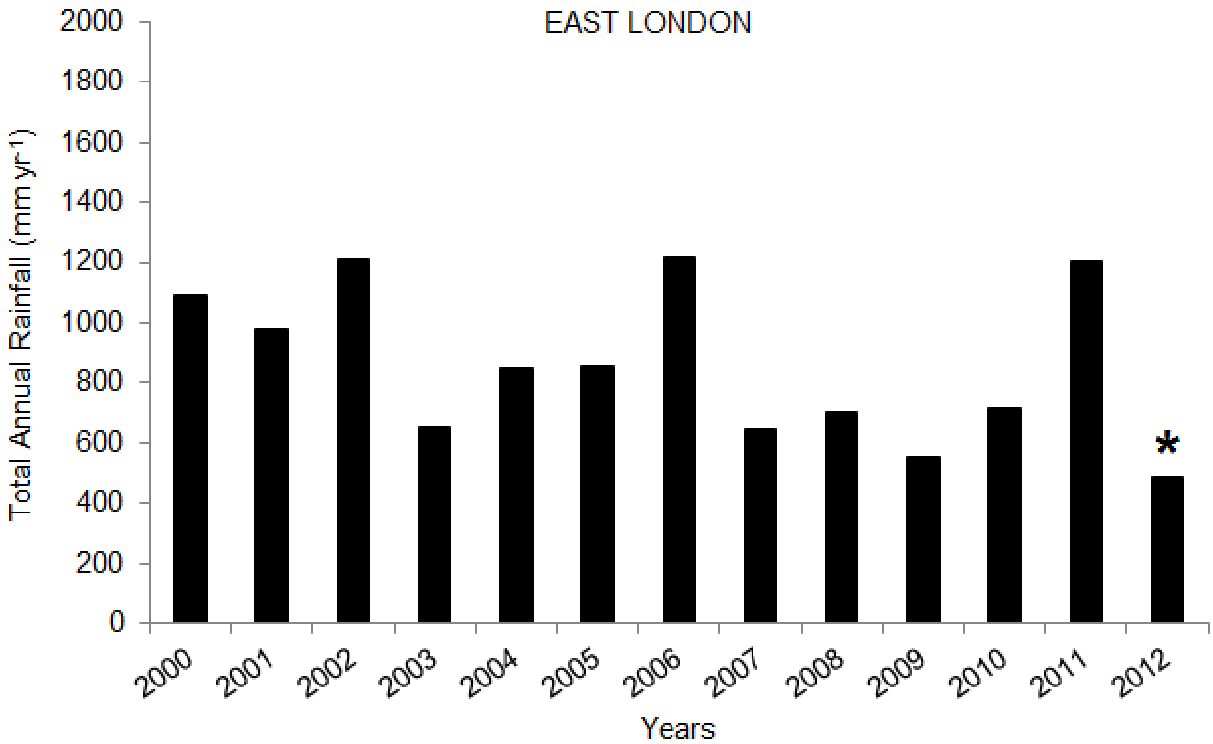

Figure 4.3: The annual total rainfall for East London. (Bars = SE). 2012 rainfall was only for 8 months from January to August 2012 (South African Weather Service, 2012). * Data missing for 6 months.

4.2.3 Mangrove characteristics

The cattle exclusion plots were established along the Nxaxo Estuary by fencing in five 25 m^2 plots as shown in Plates 4.1 A, D and E. Adjacent to each experimental plot, an unfenced control plot of the same size was measured out and marked (Plate 4.1 A), and these served as the area exposed to browsing and trampling (Plate 4.1 B and C).

All plants within the plots were tagged and morphological features and dimensions were measured. Measurements on all plants included measuring tallest heights of three main branches (as the plants were not taller than 1.5 m and displayed the characteristics of creepers), crown depth (measured from the first branching to the tip of the crown), crown circumference and crown diameter. If seedlings were present then seedling height was also measured. In some exclusion plots, cattle had broken the fence and browsed on some of the plants within the plots. Fortunately, these were rare incidences and all plants affected were eliminated from the experiment. Measurements for all plants within each plot were averaged to give the final values for the cattle exclusion and the control plots. In total there were 56 plants for each plot. The plant growth (height, crown diameter and crown circumference) was calculated as the average growth per year in cm.

Figure 4.5 depicts the typical shape of a stunted *A. marina* tree. The crown volume in m^3 can thus be calculated from the diameter, since the morphological growth of the mangroves displayed a typical oval shape. In order to simplify the calculation, volume (V) was approximated by using the formula of a sphere, where r is the radius.

$$V = 4/3\ \pi r^3$$

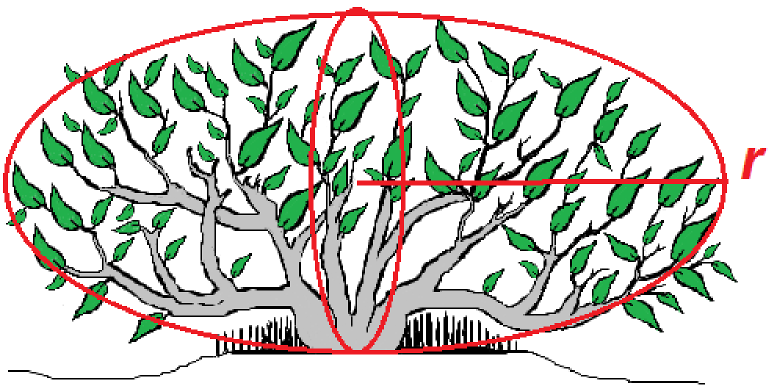

Figure 4.4: The typical shape of the crown of a stunted *A. marina* tree.

Other plant characteristics such as phenology (flowering or propagules) were noted. Mature, fully expanded leaves with similar sizes and thicknesses were collected (sample size n = 50) to analyze the Na^+, K^+, Mg^{2+} and Ca^{2+} cation concentrations in leaves for both the cattle exclusion and control plots. Five leaves from each site were rinsed with distilled water and then oven-dried for 48 hours at 60 °C. After the leaves were completely dry, they were ground to a fine powder and placed in a small air-tight container. 0.25 g of each sample was measured out for analysis using the standard cation extraction method (EPA 3052) and a method adapted from Suarez and Medina (2008). These samples were diluted, using a Compudil 3 Autodilutor, by 100 using 1 % (v/v) sub-boiled HNO_3. Type I ultrapure water was used for all preparations and dilutions. Standard preparations of mixed Na^+, K^+, Mg^{2+} and Ca^{2+} were prepared in 1 % (v/v) sub-boiled HNO_3. The samples were then analysed for cation concentrations using a Perkin-Elmer Sciex 6100 ICP-MS using a plasma power of 1.1 kW and a nebulizer flow rate of 0.93 l min^{-1}. The concentrations were expressed in mg/g dry mass.

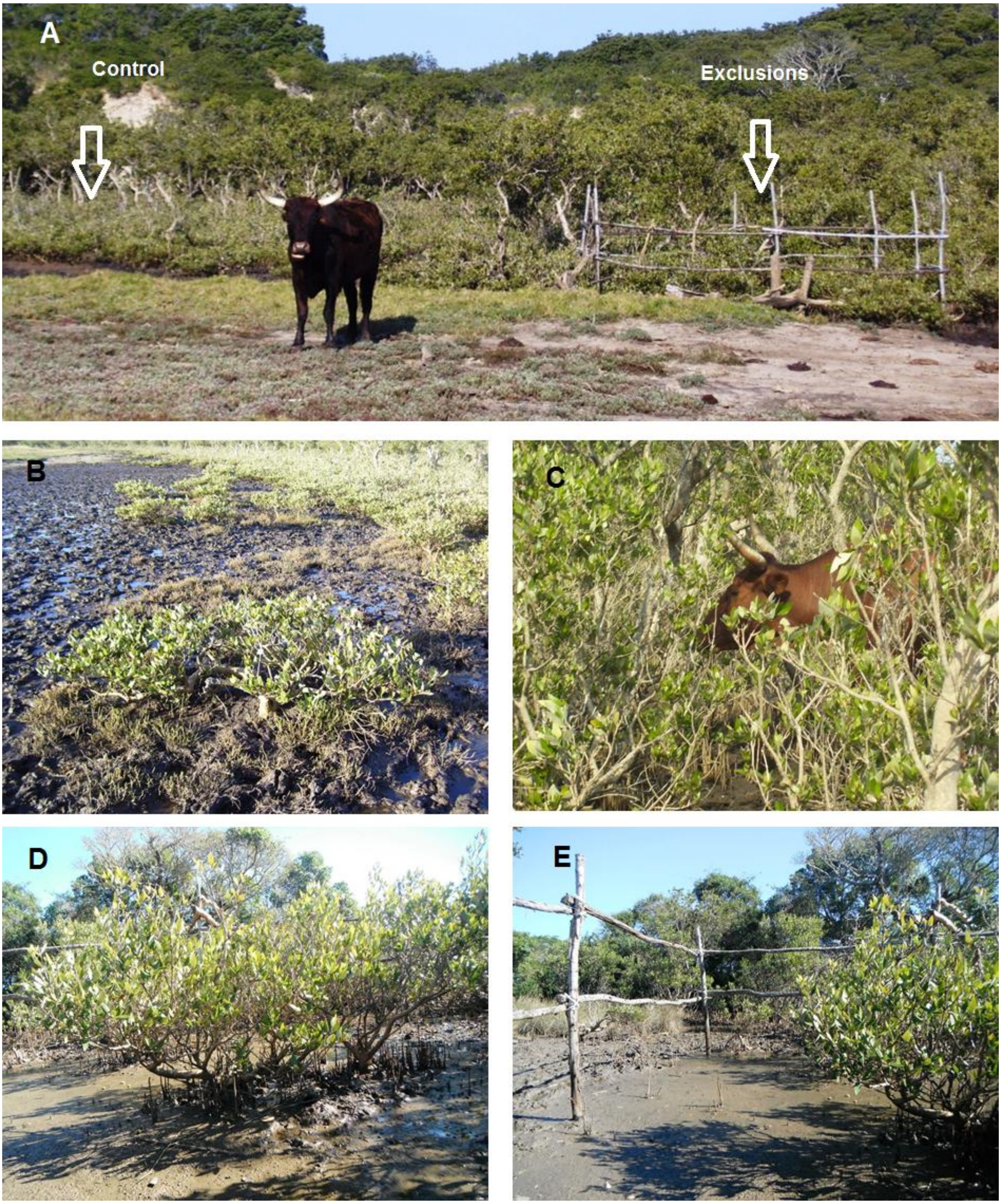

Plate 4.1: A) The plots included 25 m^2 fenced in cattle exclusion plots and 25 m^2 unfenced control plots; B) Cattle movement through mangrove habitat C) The cattle browsing on the *A. marina* D) and E) undisturbed sediment due to fences.

4.2.4 Environmental characteristics

In situ measurements were carried out at each site. Sediment compaction was determined using a DICKEY-John soil compaction meter or penetrometer according to the method described by Randrup and Lichter (2001). Sediment that had a compaction value between 0 and 200 psi was regarded as 'not compacted' and sediment compaction values greater than 200 psi were regarded as 'compacted'. Porewater electrical conductivity, salinity and temperature were measured using a hand held YSI 30M/10 FT conductivity meter, and sediment redox potential and pH were measured using a HANNA redox/pH meter (HANNA Instruments) with a Pt-Au tipped electrode redox probe and a pH probe. These were measured after creating a hole using an Auger and allowing enough water to accumulate to cover the probe. The sediment redox potential was measured immediately after sediment collection.

Sediment samples were collected at each site at both the surface (0 – 5 cm depth) and sub-surface (50 – 55 cm depth) by using an Auger. A total of n = 60 samples were taken and stored in sealed plastic bags, which were then taken to the laboratory for further analysis of sediment salinity, pH, moisture and organic content, and also sediment particle size. Sediment salinity was determined using Barnard's method (Barnard 1990) and a CyberScan, hand-held Conductivity/TDS/ Temperature meter (CON ID/100/200). Sediment pH was measured using a hand-held HANNA redox/pH meter (HANNA Instruments) with a Pt-Au tipped electrode. Sediment moisture content (%) was determined according to the method described by Black (1965). Sediment organic content (%) was determined according to the method described by Briggs (1977), while sediment particle size was determined using the hydrometer method described by Gee and Bauder (1986).

4.2.5 Data analysis

All statistical analysis was carried out using Minitab 15 Statistical Software. (Minitab, Inc. USA). Data were first tested for normality and equal variance. One way ANOVA was done on parametric data, following which a Tukey post hoc Honestly Significant Difference test was used to separate means. The parametric data included: plant growth in height per annum (cm), crown circumference per annum (cm), crown diameter per annum (cm), crown volume per annum (m^3), porewater salinity (PSU) and sediment particle size (%). A Kruskal-Wallis Anova was used to test the differences between environmental conditions which included porewater electrical conductivity (mS), sediment salinity (PSU), sediment pH, sediment redox potential (ORP), sediment moisture (%), sediment organic content (%) and leaf cation concentration (mg/g dry mass). STATISTICA version 10 (StaSoft, Inc.) was used to perform a correlation analysis determining significant correlations between environmental and mangrove characteristics. For all analyses, significance was determined at $\alpha = 0.05$.

4.3 Results

4.3.1 Mangrove characteristics

4.3.1.1 <u>Plant growth</u>

Figure 4.6 compares (A) plant growth height, (B) diameter, (C) circumference and (D) crown volume per annum between the exclusion plots and the controls on the Nxaxo Estuary. The maximum height of *Avicennia marina* was 188 cm in the cattle exclusion plots with an average height of 91 ± 5 cm. For the control plots the average height was 77 ± 10 cm in 2012. It can be seen that there was a significantly greater plant growth in terms of height ($F_{(df = 1, n = 10)} = 40.95$, $p < 0.05$), diameter ($F_{(df = 1, n = 10)} = 40.67$, $p < 0.05$), and crown volume ($F_{(df = 1, n = 10)} = 9.02$ $p < 0.05$) in the cattle exclusion plots compared with the control plots.

Most of the tagged individuals were adult plants and even though they were short they were flowering. It was observed that within the cattle exclusion plots, 67 % of plants were flowering and 39 % were fruiting in 2012. This was significantly higher compared with the control plot plants ($H_{(df = 9, n = 112)} = 125.91$, $p < 0.05$), where only 34 % of the plants were flowering and 5.4 % carried immature propagules. These values were significantly lower at the beginning of the study in 2010, where only 14.3 % of plants showed flowering. This increased to 53 % flowering in 2011 within the cattle exclusion plots, while at the control plots flowering increased from only 7 % in 2010 to 23 % in 2011. No propagules were observed in either the control or cattle exclusion plots in 2010, while in 2011, 2 % of the plants had immature propagules. Germinating seedlings in both cattle exclusion plots and control plots were only recorded in 2012. These mature propagules originated from another area and had been transported by the tides. The cattle exclusion plots had a higher percentage (63 %) of established seedlings, while a lower percentage (37 %) of seedlings was recorded in the control plots. Seedling establishment was variable at the different plots and no significance between cattle exclusion plots and controls was found.

Plate 4.2 reveals the morphological responses to cattle browsing in the control plots. Observations indicated that cattle had trampled a number of seedlings, influencing plant survival. Cattle browsed most of the plants in the control plots to such an extent that the plants had dead branches (Plate 4.2 A), and in response to the browsing, some plants responded with morphological changes such as multiple coppicing (Plate 4.2 B). Most of the plants within the control plots displayed horizontal spreading, while the plants in the cattle exclusion plots displayed a horizontal as well as a vertical expansion.

4.3.1.2 Leaf cation concentration

Figure 4.5 shows the leaf cation concentrations at the start of the experiment (2010) and at the final year of measurement (2012). It can be seen that significantly lower values of leaf cation concentrations for Na^+ ($H_{(df = 3, n = 40)} = 7.44$, $p < 0.05$), were observed in 2012 compared with 2010, and significantly higher concentrations for K^+ ($H_{(df = 3, n = 40)} = 8.73$, $p < 0.05$) and Ca^{2+} ($H_{(df = 3, n = 40)} = 7.76$ $p < 0.05$) in 2012 compared to 2010. Leaf Mg^{2+} concentrations were similar between the years ($H_{(df = 3, n = 40)} = 5.13$, $p > 0.05$).

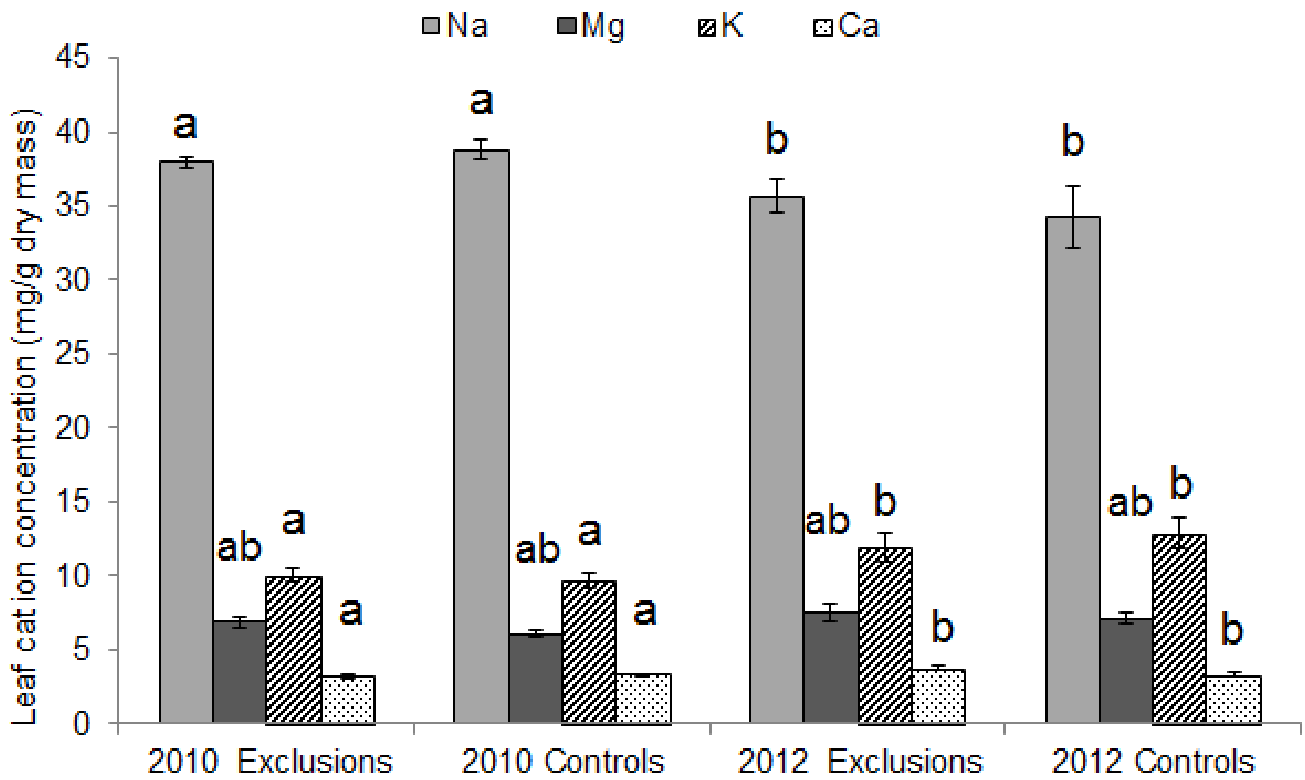

Figure 4.5: Leaf cation concentrations at the start of the experiment (2010) and the final year of measurement (2012). Means with similar superscript letters are not significantly different (Bars = SE).

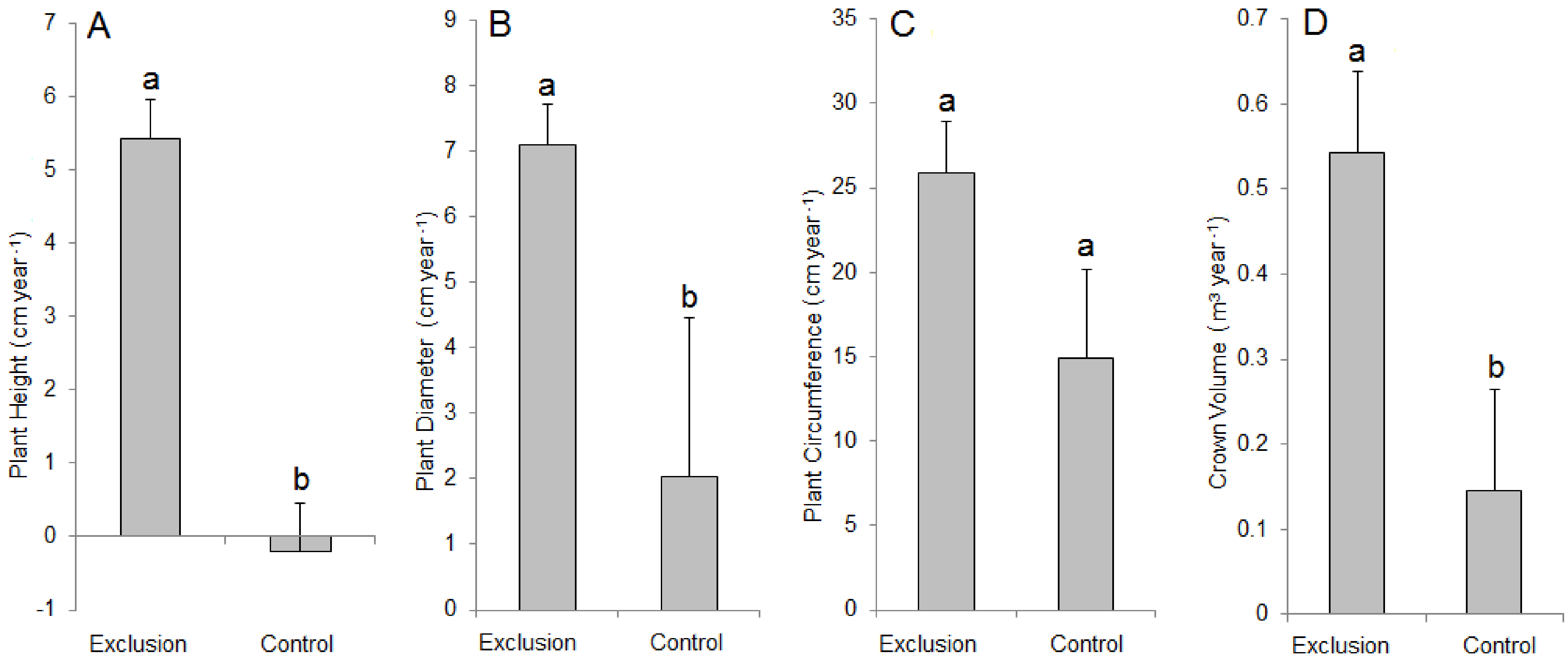

Figure 4.6: A comparison of plant growth in terms of (A) height, (B) diameter, (C) circumference and (D) crown volume per annum between the cattle exclusion plots and the control plots on the Nxaxo Estuary, (Bars = SE). Means with similar superscript letters are not significantly different.

Plate 4.2: The morphological changes as a response to heavy browsing (2012) on *Avicennia marina* were (A) the coppicing of growth buds as the new shoots have been heavily browsed (B) propagules in the exclusion plots, (C) the vertical expansion of the plants in the exclusion plots, (D) the horizontal expansion of the plants in the control plots.

4.3.2 Environmental characteristics

Sediment compaction studies showed that sediment was not compacted during 2010 to 2012, and all values were lower than 200 psi. The sediment was much drier at the start of the experiment since 2010 (< 25 % in sediment moisture) was the end of a drought and in 2011 and 2012 the rainfall was higher (Figure 4.2) therefore this affected groundwater table depth and porewater salinity differently, as shown in Figure 4.7. The groundwater table depth (cm) was significantly lower ($H_{(df = 4, n = 30)} = 1.73$, $p < 0.05$) in 2010 compared with 2011 and 2012 (Figure 4.6 A). However, porewater salinity (PSU) was not significantly different in the cattle exclusion plots and the different years ($F_{(df = 14, n = 30)} = 23.89$, $p < 0.05$) (Figure 4.7 B).

Figure 4.8 A shows that the sediment redox potential of the sub-surface samples was much lower compared with the surface samples, with all sub-surface samples being anoxic (< 200 mV). The redox potentials of the surface sediments in 2010 were significantly lower ($H_{(df = 4, n = 59)} = 28.59$, $p < 0.05$) than in 2011 and 2012. However, sediment pH levels (Figure 4.7 B) were significantly lower in 2011 ($H_{(df = 3, n = 59)} = 29.29$, $p < 0.05$) compared to 2010 and 2012 for both surface and sub-surface samples. In 2010, the annual rainfall was lower than the average annual rainfall and also low compared to 2011 and 2012 (Figure 4.1). The sediment salinity (PSU) (Figure 4.7 C) was thus significantly higher in 2010. The surface sediment salinity was significantly higher in 2010 ($H_{(df = 3, n = 59)} = 82.25$, $p < 0.05$) compared to 2011 ($H_{(df = 3, n = 59)} = 82.25$, $p < 0.05$) and 2012. The combined effects of drought in 2010 and also low rainfall in 2011 resulted in higher evaporation from the sediment, leading to significantly lower sediment moisture content when compared to higher rainfall year 2012.

In 2010 the surface sediment organic content was significantly higher ($H_{(df = 3, n = 59)} = 11.66$, $p < 0.05$) than in 2011 (Figure 4.9 A). It can be seen in Figure 4.9 B that the surface sediment moisture was significantly lower in 2010 and 2011 compared to 2012 ($H_{(df = 3, n = 59)} = 46.97$, $p < 0.05$). Figure 4.10 shows the sediment particle size of the surface and sub-surface samples in the exclusion and control plots from July 2010 to July 2012. For sediment particle size, the sand component (Figure 10A) was significantly higher ($F_{(df = 3, n = 59)} = 6.82$, $p < 0.05$) in 2010 and 2011 in the control plots compared with all the other samples. No significant differences ($F_{(df = 4, n = 59)} = 2.17$, $p > 0.05$) were found in the clay fraction (Figure 4.10 B) between the plots, years and depths. In Figure 4.10 C, the silt fraction in the sub-surface samples was significantly higher ($F_{(df = 3, n = 59)} = 8.92$, $p < 0.05$) compared to that of 2011 and 2012 for both surface and sub-surface samples.

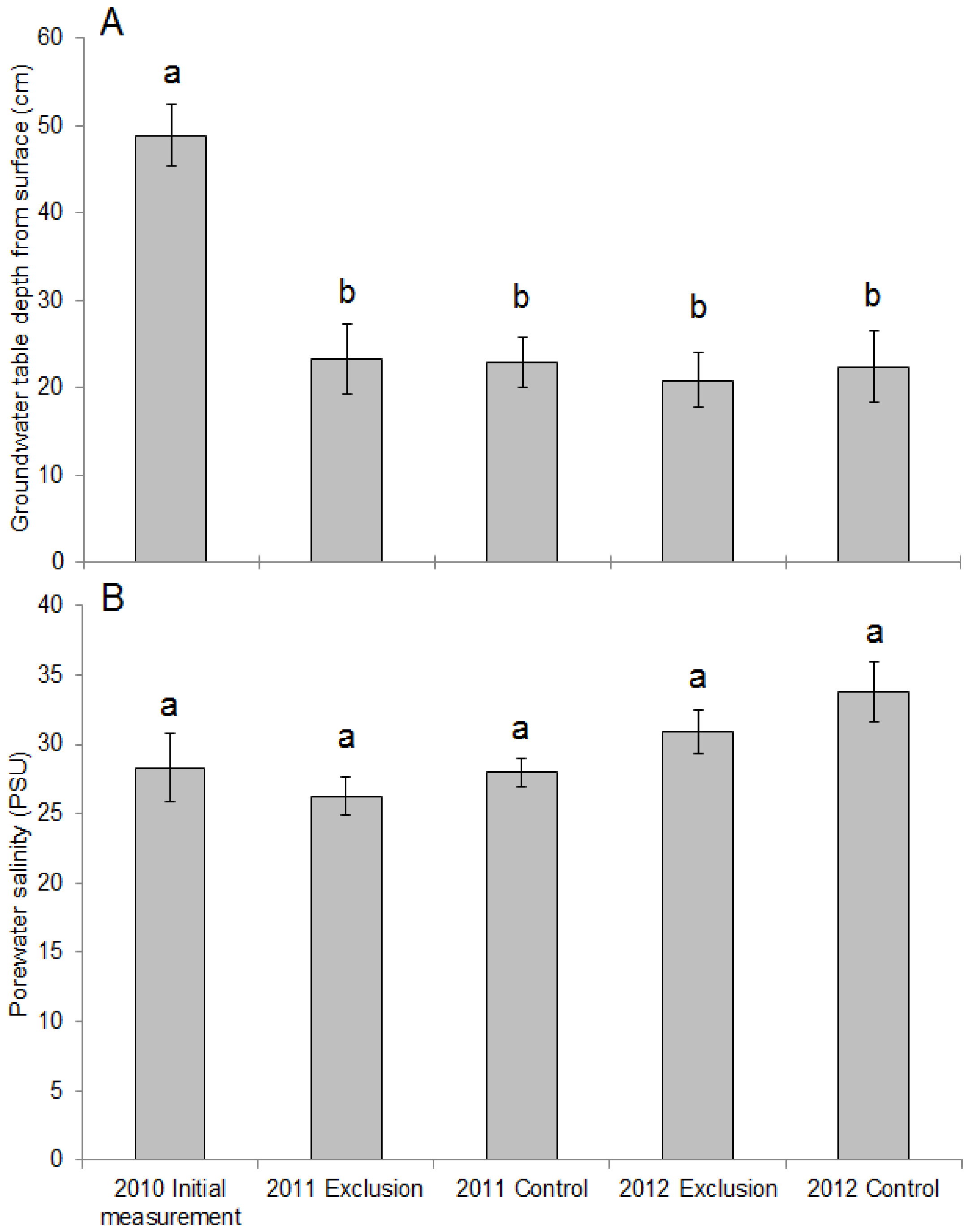

Figure 4.7: (A) Groundwater table depth and (B) Porewater salinity in the cattle exclusion plots and control plots from July 2010 to July 2012. (Bars = SE). Means with similar superscript letters are not significantly different.

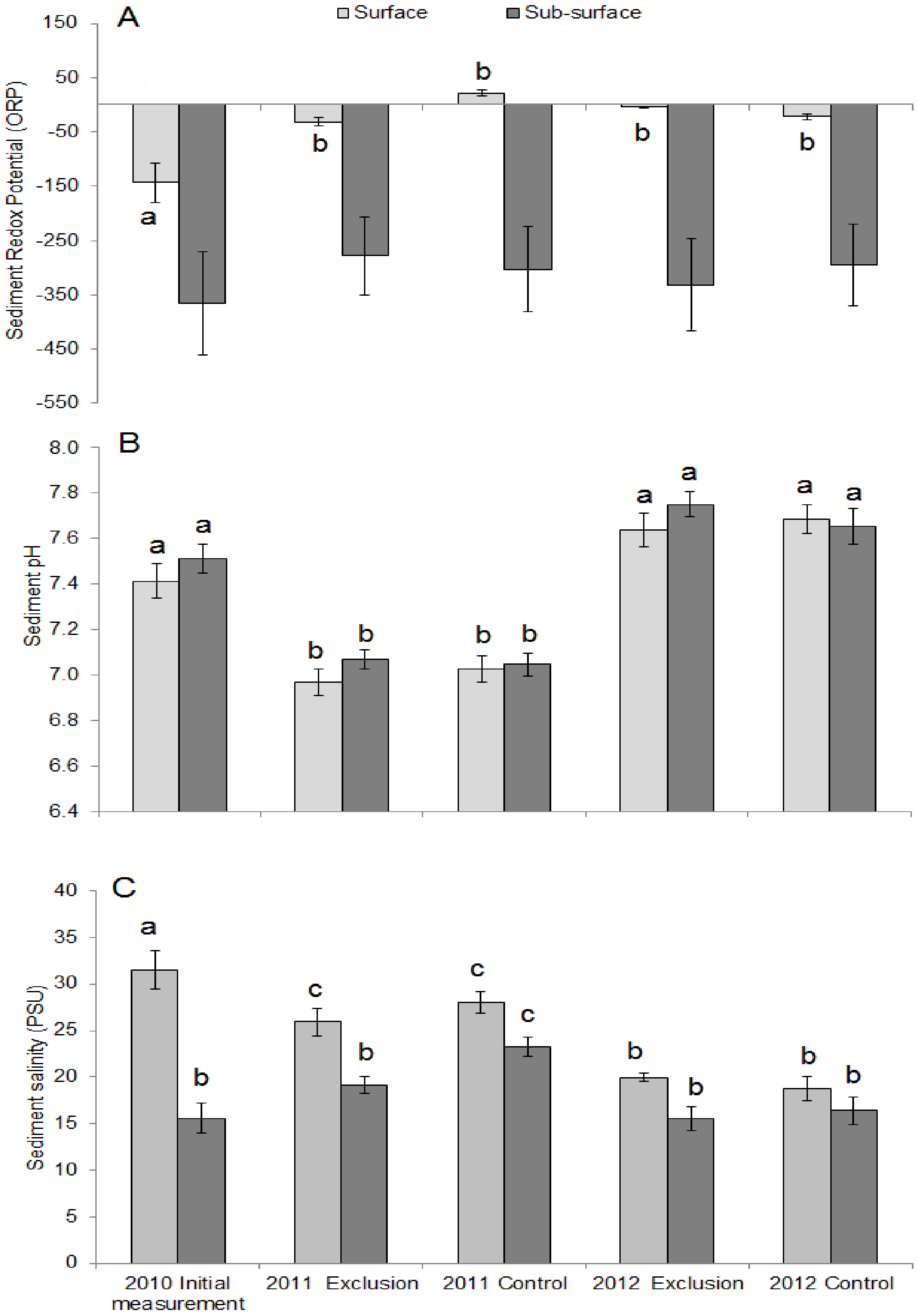

Figure 4.8: Sediment characteristics of the surface and sub-surface samples in the exclusion and control plots from July 2010 to July 2012. (A) Sediment redox potential (ORP), (B) Sediment pH and (C) Sediment salinity (PSU). Means with similar superscript letters are not significantly different. (Bars = SE).

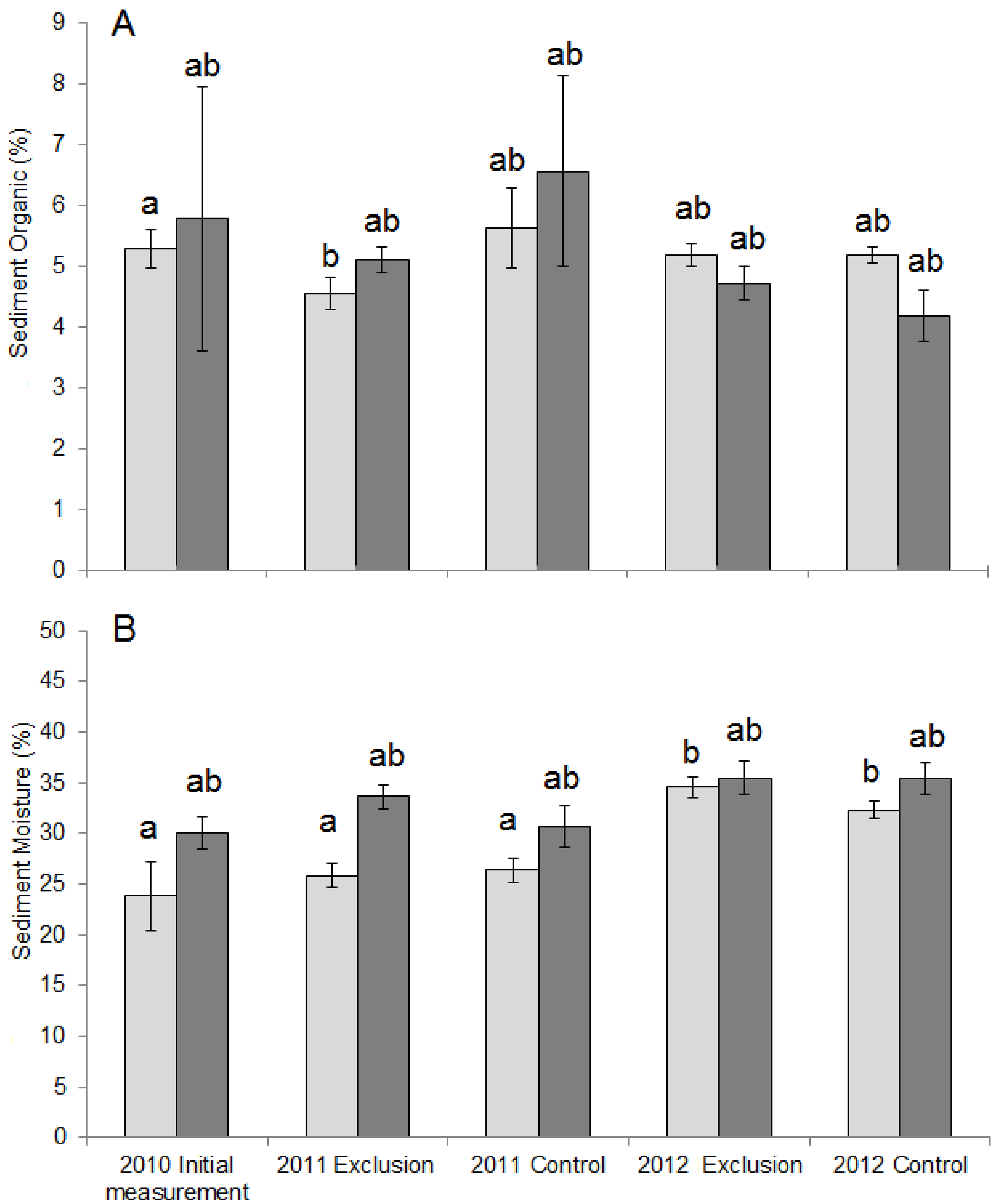

Figure 4.9: Sediment characteristics of the surface and sub-surface samples in the exclusion and control plots from July 2010 to July 2012. (A) Sediment organic content (%) and (B) sediment moisture content (%). Means with similar superscript letters are not significantly different. (Bars = SE).

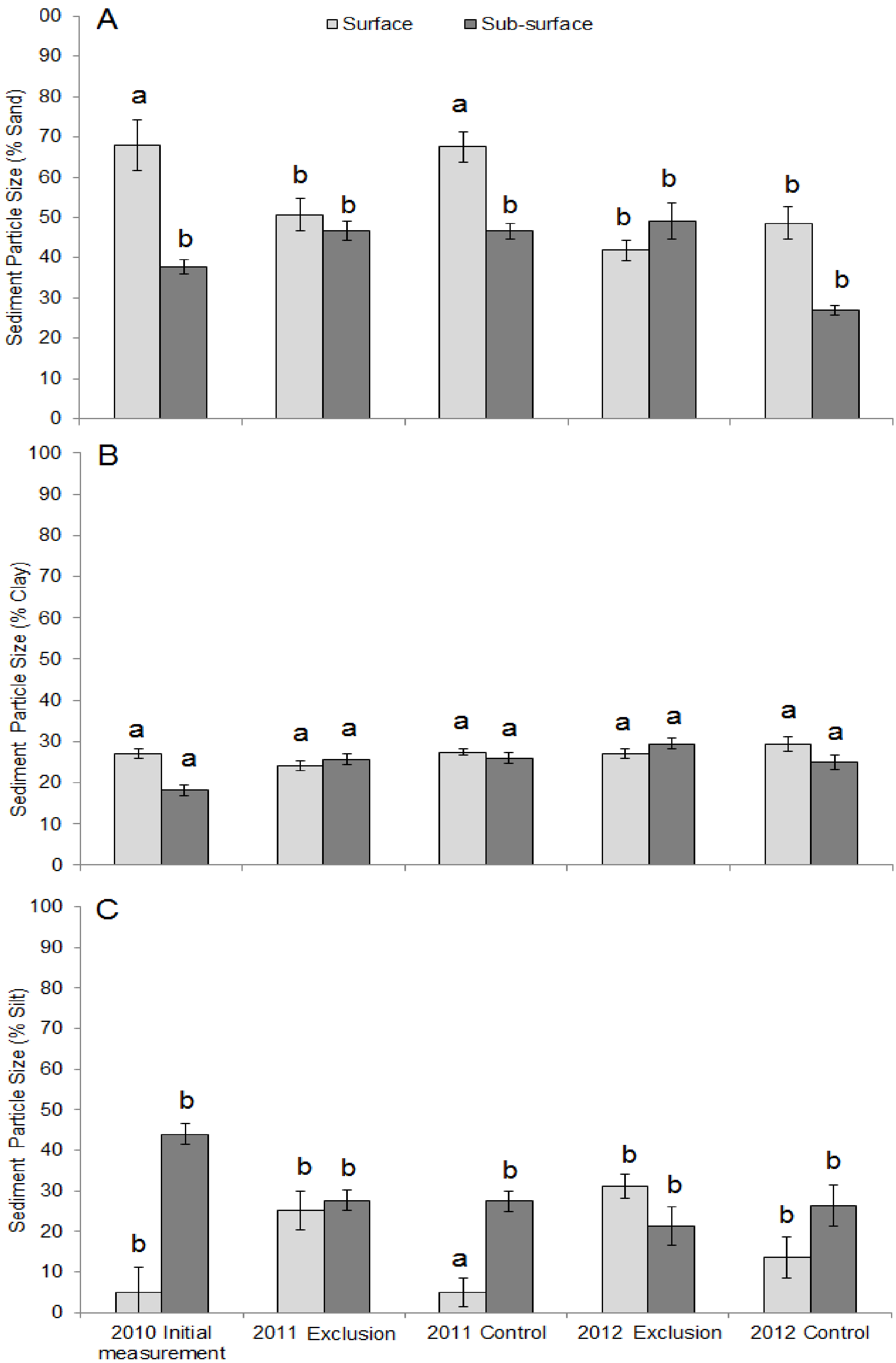

Figure 4.10: Sediment particle size of the surface and sub-surface samples in the exclusion and control plots from July 2010 to July 2012. (A) Sand (%), (B) Clay (%) and (C) Silt (%). Means with similar superscript letters are not significantly different. (Bars = SE).

4.3.3 Correlation analysis

In 2010, the change in crown volume correlated negatively with the sediment moisture content of the sub-surface sediments ($R = -0.55$, $p < 0.05$) as well as with sediment organic content ($R = -0.712$, $p < 0.005$). For the exclusion experiments, there were significant positive correlations between canopy circumference and porewater- and sub-surface sediment salinity during the wet year 2012 ($R = 0.708$ and 0.683 respectively, $p < 0.005$). However there were significant negative correlations between plant growth and the sediment moisture content of the sub-surface sediment in 2012 ($R = -0.545$, $p < 0.005$).

4.4 Discussion

Mangrove classification had been done by Lugo and Snedaker (1974) where mangroves had been classified into six basic classes, which were the (1) riverine mangroves, (2) the fringe mangroves, (3) the over-wash mangrove, (4) the basin mangroves, (5) the hummock mangroves and (6) the dwarf mangroves. These had been classified according to the mangrove responses to different and constant environmental parameters (Lugo and Snedaker 1974).

Much research had been conducted on the morphological and ecophysiological differences between the fringe and dwarf mangrove zones by Dahdouh-Guebas *et al.* (2004b); Coranado-Molin *et al.* (2004); Feller et al. (2004); Naidoo (2006; 2009; 2010); Lovelock *et al.* (2006). For example Dahdouh-Guebas *et al.* (2004b) in Kenya at Gazi Bay, described that the mangrove *A. marina* had been found growing in two distinct zones, these are the "landward zone and the seaward fringing zone". The landward mangroves had reduced plant height (< 8 m) at salinity concentration of greater than 40 PSU, while the seaward mangroves, which had grown in salinity concentration close to seawater (35 PSU) had a tree height greater than 10 m (Dahdouh-Guebas *et al.* 2004b). Dwarfed mangroves have also been observed by Qureshi (2005) in the Indus Delta, Pakistan, where sediment salinity of up to 50 PSU and reduced freshwater has resulted in stunted and sparsely distributed *A. marina* trees. In Florida dwarfed mangroves had a mean height between 0.9 to 1.2 m (Coranado-Molin *et al.* 2004) and Feller *et al.* (2004) described the dwarf zone with a mean height of only 1 ± 0.04 m, while the fringe zone had a mean tree height of 3.8 ± 0.3 m. In South Africa Naidoo (2006; 2010) defined the differences between the fringing trees (*A. marina*) as those with 5 m height, usually between 6 to 10 m, with a healthy appearance and a dense canopy, while the 'dwarf' or 'shrub' mangroves have normally a maximum height not more than 1.5 m, but are usually shorter, with smaller leaf area as well as lower canopy densities and this dwarfing may be attributed to environmental factors such as high salinity (Naidoo 2006; Shah *et al*. 2007). Naidoo (2006) reported that the dwarf zone at Richards Bay had 63 to 70 % higher sediment salinity concentration compared to what was found in the fringing zone, which had been close to seawater. This increase in salinity resulted from reduced tidal influence and reduced sediment moisture due to increased evaporation (Naidoo 2010). Dwarfing could also be a response to excessive inundation and low sediment redox potentials (Naidoo 2010); compacted substratum (Craighead 1971) and nutrient deficient sediments (Lovelock *et al.* 2004; Naidoo 2009). In this study two distinct zones (fringing and dwarf zones) were found at the study site Nxaxo Estuary. The browsing experiment was conducted on dwarf mangroves. The study has shown how browsing can contribute towards maintaining this dwarfed structure. However environmental conditions may have created the distinct zones and this can be deducted from the higher elevation, salinity and lower sediment moisture that was observed in this one compared to the fringing zone in 2010. Additionaly it was observed that a change in environmental conditions such as increased rainfall in 2011/2012 resulted in increased plant growth and phenology in both control and exclusion plots. However the plants in the control plots only had horizontal growth since the cattle browsed these plants from the top down. It was evident that browsing had profound affects on the morphology of the plants as the cattle exclosion plots had horizontal as well as vertical growth.

At Nxaxo Estuary all three species of mangrove were present, namely *Bruguiera gymnorrhiza* and *Rhizophora mucronata*, with *Avicennia marina* being the dominant species. The latter species fringed the estuaries from the mouth to the upper reaches, with exception of individual *B. gymnorrhiza* trees. It was observed that the *A. marina* trees reach up to 3.5 m in height at the water's edge. The stunted *A. marina* plants in the shrub zone behind the fringing trees were found to be less than 1.5 m in plant height. These stunted plants are growing in the upper intertidal region at a higher elevation, bordering salt marsh area and open grasslands. Tidal inundation occurred less at this zone, however the sediment salinity was close to the seawater salinity concentration in 2010 at both the fringing (Rowe 2011) and dwarf zone and decreased with increasing rainfall in 2011 and 2012 to less than 30 PSU. Thus the sediment salinity in the dwarf zone was lower in this study (< 35 PSU) compared to research by Naidoo (2006) and Dahdouh-Guebas *et al.* (2004b), while the sediment salinity (mean 35.5 PSU in 2010) in the fringing mangrove zone in this study was similar to the latter two studies. Thus the *A. marina* at Nxaxo Estuary had been growing well in its optimum salinity range (5 to 35 PSU) (Naidoo 1987).

In this study it was found that the surface sediment was dry in the dwarf zone and this was because 2010 was a drought year in South Africa, thus the groundwater table was significantly deeper (> 0.5 m) and the surface sediment moisture was lower (< 25 %) at the dwarf zone compared to the fringing zone (> 30 %) in 2011 and 2012. After the rain in 2011, there was a significant increase in the groundwater table, which was shallower (< 0.25 cm), the sediment salinity decreased (< 30 PSU), and sediment moisture increased (> 35 %) in the dwarf mangrove zone, while the fringing zone was more constant from 2010 to 2012. The sediment was anoxic in the sub-surface at the dwarf zone (< 200 mV) and recordings had been similar in 2010 but lower in 2011 compared to the fringing zones which had a mean sediment redox potential of -323 mV in 2010 and -107 mS in 2011 (Rowe 2011).

The sediment organic content was low in both the dwarf and fringing zones (< 10 %) and thus this had not affected the sediment pH readings which had also been similar for both the zones where it was neutral to slightly alkaline (7 to 7.8 pH). Low organic content may have resulted in nutrient deficiency in the plants contributing to the dwarfing (Naidoo 2009). Naidoo (2009) reported that stunted *A. marina* may have been nutrient (N and P) limited and that growth is shifted to root rather than to shoot biomass and the plants may thus have a reduced growth and productivity. However it was observed that the droppings of livestock may have introduced nutrients into the site and therefore nutrients were not considered to be a limiting factor. However nutrient input was not covered in this study and this would provide opportunity for future investigations.

Table 4.1: Differences of parameters in the dwarf (this study) and fringing zones (Rowe 2011) at Nxaxo Estuary for 2010

	Dwarf zone	**Fringing zone**
Plant maximum height	1.5 m	3.5 m
Tidal inundation	only spring highs	twice daily
Groundwater depth from surface	± 0.5 m	0 m (intertidal)
Porewater salinity	± 35 PSU	± 35 PSU
Sediment moisture (2010)	< 25 %	> 30 %
Rediment redox potential (2010)	< - 200 mV	- 323 mV
Sediment organic content (2010)	< 10 %	< 10 %
Sediment pH (2010 & 2012)	7 - 7.8	7 - 7.8

At the onset of the experiment all plants in the control and cattle exclusion plots showed signs of stress. These dwarfed plants displayed a creeper-like morphology and stunting (Plate 4.2 D). The higher salinity concentration in the surface sediment (between 30 and 35

PSU at the different plots) in 2010 may have contributed to reduced plant growth (height, crown diameter and crown volume) (Figure 4.4), which would have been also the reason for increased leaf Na^+ concentration in 2010 (Figure 4.5). Inorganic ions or cations are good indicators of mangrove health and physiological adaptation to salinity stress (Pezishki *et al.* 1989; Krauss *et al.* 2008). In hypersaline conditions mangroves tend to increase these cations to counteract water loss and for osmotic regulation within the plant cell (Ball *et al.* 1997; Krauss *et al.* 2008). Naidoo (2010) found that dwarfed *A. marina* that grew in hypersaline sediments had much higher Na^+ : K^+ ratios than those found in the fringe zone. He reported that the mean leaf Na^+ : K^+ ratio for the dwarfed mangrove were 5.6 ± 0.05 : 1, while the fringe mangrove leaf cation ratio was 2.1 ± 0.03. In this study the mean leaf Na^+ : K^+ ratio was similar to that found in the fringe zone by Naidoo (2010), 3.8 ± 0.04 in 2010 and 2.6 ± 0.15 in 2012. These differences could be attributed to the sediment salinity.

In 2010, the change in crown volume correlated negatively with the sediment moisture content of the sub-surface sediments ($R = -0.55$, $p < 0.05$) as well as with sediment organic content ($R = -0.712$, $p < 0.005$), suggesting that the increase in rainfall from 2011 and the increase in sediment moisture increased mangrove growth. However there were significant negative correlations between plant growth and the sediment moisture content of the sub-surface sediment in 2012 ($R = -0.545$, $p < 0.005$), where excessive waterlogging, anoxic condition and increased porewater salinity may have had an effect of plant growth in 2012. Thus the porewater salinity was an important driver of the observed canopy pattern. High salinity conditions are often coupled with dry environments, and in studies conducted by Dodd *et al.* (1999), *A. marina* shrubs grew in arid environments with salinities as high as 70 PSU. *Avicennia marina* can tolerate high salinities, but according to Qureshi (1993), plants are stressed at salinities greater than 35 PSU. They mentioned that high salinity reduces flowering and, ultimately, propagule production. In this study the low flowering and fruiting may be attributed to the low sediment moisture rather that the sediment salinity. However the rainfall in 2011 resulted in rapid growth, increased fruiting and flowering of the mangroves in the cattle exclusion plots. The rainfall decreased salinity in the sediment and increased sediment moisture. However the control plots had similar environmental parameters compared to the cattle exclusion plots but the plants in the control plots had been exposed to intense cattle browsing which had been an additional stressor to these mangroves. These additional stressors arise from herbivores, where stunted growth could be a response to intensive browsing (Qureshi 2005). This last factor was the focus of this study.

Globally, the effects of browsing on mangroves, particularly *Avicennia marina*, have been documented by authors such as Khalil (1999); Sommerlatte and Umar (2000); Qureshi

(2005); Dahdouh-Guebas *et al.* (2006a; 2006b); Shah *et al.* (2007); Shah and Kamaruzaman (2007) and Saifullah *et al.* (2007). However, these studies have mostly focussed on social – environmental conflicts and benefits of natural resources and not much literature is available to underpin the response of mangroves to continuous, long term browsing. There are some examples where the state of the forests have been progressively degraded due to heavy browsing by camels and cattle such as studies presented by Shah *et al.* (2007) who reported that in Pakistan, camels and cattle are responsible for removing about 67 000 tons of foliage per annum from the *A. marina* forests, and suggested that utilization of this natural resource was not sustainable, as consumption by the animals was greater than foliage production. Mangrove forests in Pakistan have a high economic and social importance, as much of the land is arid and the mangrove trees provide much needed fodder in times of drought and during the dry season. These authors also mentioned that overutilization and degradation was coupled with other factors such as wood harvesting, pollution, sea-level rise and hypersalinity (Khalil 1999). These factors contribute greatly to the reduced resilience of the forest for regeneration, and this may lead to the loss of mangrove habitats. The importance of mangroves as fodder for livestock in the dry season was also highlighted by Spugeon (2002) in Egypt, where mangrove browsing by camels and goats becomes increasingly important. However browsing impacts have not yet been well quantified in South Africa and thus this study has demonstrated that browsing pressure does have negative impacts on mangrove morphology, growth and population structure.

It was observed that the cattle of the local farmers from rural communities utilize these forests heavily, especially when the grass is dry and much of the land has been grazed. However, the cattle were observed to move into the mangrove forest continuously, regardless of the season. It was observed that cattle were selective and chose the fresh new shoot instead of the older mature leaves. These older and more mature leaves contain more tannin and ionic compounds than new shoots (Hernes *et al.* 2001), and this may make them less palatable to herbivores, since tannin may result in toxicity of the digestive tract (Neilson *et al.* 1986). Therefore cattle usually browse the short plants from the top down and prefer new shoots that contain less tannin. The lower branches close to the ground were thus left relatively undisturbed, resulting in plants having a horizontal expansion (Plate 4.2 D). Plants in the cattle exclusion plots, however, had horizontal as well as vertical growth increases (Plate 4.2 C). It is thus suggested that the browsed plants respond to ‘top-down’ browsing with a horizontal crown spreading. It was also observed that plants in the control plots responded to browsing by coppicing (Plate 4.2 A). The growth of multiple new shoots (Figure 4.5 A) due to constant browsing is a typical morphological change caused by damage to the growth buds (Siddique and Adrika 2011). Coppicing is known to be a response to stem or branch damage via the production of new shoots at the base of the broken stem or branch, which is the plant's ability to regenerate from such damage (Duke 2001).

In this study it was found that the growth rates with respect to height of the plants within the cattle exclusion plots (5.41 ± 0.53 cm yr^{-1}) were higher compared to the growth rate of the fringing trees in the middle and lower intertidal regions (0.85 ± 0.35 cm yr^{-1}) (Rowe 2011). The growth rate of the control plots however was reduced growth (-0.07 ± 0.67 cm yr^{-1}) compared to both the cattle exclusion plots and the fringing trees. When compared to previous studies at Mngazana Estuary (31°42'S; 29°25'E) by Rajkaran (2011), a growth rate between 3.72 to 14.4 cm yr^{-1} for *A. marina* was recorded, which was similar to the growth rate of the plants in the cattle exclusion plots recorded in this study. This study tested the hypothesis (1) that mangroves responded negatively to cattle browsing, resulting in stunted growth and a shrub-like appearance. It was observed that the plants within the cattle exclusion plots had significant increases in plant growth (plant height and crown volume), while in the control plots the plants had remained short and had negative growth (Figure 4.4 A).

Flowering and fruiting differed between the years, where no fruiting was observed in the study site in 2010 and only a few flowering plants were recorded (14.3 %) for both cattle exclusion plots and control plots. In 2011 these increased to 53 % of flowering plants in the exclusion plots and only 7 % in the control plots. An increase in flowering (67 %) and fruiting (39 %) was observed from 2010 to 2012 in the cattle exclusion plots (Plate 4.2 B) compared with the control plots which had 34 % flowering and only 5.4 % fruiting plants. Thus this supports hypothesis (2) where it is suggested that cattle browsed on new shoots that also contained the flower buds, and thus reduced the numbers of flowers and fruits in the exposed control plots. The cattle exclusion plots were thus mostly undisturbed and had a greater percentage of both flowers and propagules. Additionally an increase in seedling establishment was observed in 2012 at the study site, where spring tides and increased freshwater runoff transported many propagules into the study plots and by July 2012 had established in the plots. It was also observed that the newly established seedlings in the cattle exclusion plots were undisturbed, while cattle had trampled the seedlings deep into the mud in the control plots. Survival of seedlings in the cattle exclusion plots would thus be much higher (63 %) when compared with survival in the control plots (37 %). Important factors that could have influenced recruitment prior to 2012 may have been the recent drought which resulted in dry sediments with low sediment moisture (Hoppe-Speer *et al.* 2012). However, a combination of factors such as sediment erosion (Adame *et al.* 2010b), sediment accretion (Anthony 2004), anoxic sediment (Pezeshki *et al.* 1997), nutrients

(Lovelock *et al.* 2006a), nutrient transportation (Adame and Lovelock 2011), salinity increase and light availability (Lopez-Hoffmann *et al.* 2006) may also have had an effect on seedling establishment. This study showed that cattle browsing had a significant effect on the growth of the dwarfed *A. marina* forest. Trampling by cattle reduced seedling establishment, but results for the study period showed no effect on sediment compaction. In addition to cattle browsing, the already harsh environmental conditions may have contributed to the dwarfing of the mangrove plants. Browsing, however, is an additional stress causing mangrove degradation and loss of habitat.

4.5 Conclusion

The key findings of this study were that mangroves are sensitive to continuous long-term browsing and trampling and that this contributed to mangrove degradation. At Nxaxo Estuary the dwarf mangroves are already under pressure due to the stressful environment of low sediment moisture and high salinity. Thus a long-term monitoring programme of this study would determine if trampling would be a determining factor for seedling establishment and if the mangroves in the cattle exclusion plots will grow to be trees. Most locals practice subsistence farming and depend on the nearby forest resources. For this reason, alternative solutions such as controlled planted woodlots close to the villages for use as firewood and building material would reduce the pressure on the mangrove forest and the adjacent coastal forests. The prevention of cattle movement in sensitive mangrove areas is a possible solution to allow seedlings to establish into new plants. This could be done by actively involving the community and fencing some areas off. Ecotourism has already taken off due to the Wavecrest Lodge and provides some protection to the estuaries, but more sustainable management practices, and community involvement such as mentioned by Shah *et al.* (2007) and Satyanarayana *et al.* (2012) that would involve the local community in protecting and conserving the mangroves, should be investigated. Therefore, additional research investigating socio-economic aspects/impacts would be beneficial in promoting a greater understanding between the stakeholders. This would provide information for the creation of more sustainable management and conservation plans, in order to prevent increased degradation of these mangrove forests.

Chapter 5: Present status of mangroves along the former Transkei Coast, Eastern Cape

5.1 Introduction

Mangrove forests provide many environmental services which have recently received much attention as mangrove ecosystems are threatened and are disappearing fast due to natural changes such as climate change, and anthropogenic impacts (Nicholls *et al.* 2007). Climate change pressures include sea-level rise (SLR) and increased tidal volumes, which cause an increase in salinity within the estuaries. The increased volume of water would push up further into the estuaries as well as into the intertidal areas. Ye *et al.* (2004) predicted a sea level increase of 10 to 20 cm for the last 100 years and a rise of 5 to 11 cm was predicted for the next 100 years (Nicholls *et al.* 2007). The low-lying regions along the coastlines would be most vulnerable to sea-level rise and studies conducted by McFadden *et al.* (2007) suggest that 33 to 44 % of mangrove ecosystems might be lost between 2000 and 2080 due to possible sea level increases as great as 72 cm. However, this is site-specific and does not represent all mangrove areas. For the former Transkei region on the east coast of South Africa, a mean SLR of greater than 2.74 mm yr^{-1} has been reported by Mather *et al.* (2009). This is a more precise estimation compared to the IPCC report (Boko *et al.* 2007), which predicted a global SLR between 0.5 to 2 m increase in 2100 (Van Niekerk and Turpie 2012).

Globally it is also predicted that the impacts of climate change will include increases in the magnitude and frequencies of tropical storms, which includes hurricanes and tsunamis (Boko *et al.* 2007). These have negative social, economic and environmental impacts, particularly for low-lying coastal regions (Nicholls and Lowe 2004). The rapid increase in human population also poses additional threats to the environment, as most of the world's coastal regions are prime residential areas. As much as 23 % of the world's population inhabit the coastlines (Nicholls and Lowe 2004). These populated coastlines are vulnerable to the predicted increases in sea-level, and studies carried out by Kumar, (2006) were undertaken to attempt to find possible solutions, including providing suitable protection for the environment and natural resources, alleviating human poverty, protection of settlements, and protection of the natural vegetation of the coastlines. However, estuaries in particular are changing environments and are constantly being shaped. This makes them vulnerable to climate and sea-level change (Woodroffe 2000). Any vegetation type within these low-lying areas needs be able to cope with longer hydroperiods and increased salinity concentrations due to sea-level rise in order to have a chance of survival in this uncertain future (Woodroffe 2000).

Mangroves can be found throughout tropical, subtropical and warm temperate regions (Steinke 1999). Temperature could be the key factor in the global distribution of different species of mangroves (Duke *et al.* 1998). Mangrove forests are considered to be rare in South Africa and can be found on the east coast of South Africa, within the former Transkei region, which has approximately 270 km of coastline where 17 permanently open estuaries provide habitat for mangrove forests. This part of the Eastern Cape is mostly undeveloped and difficult to access. Only 19 % of the coastline is protected in marine protected areas (MPA's) and nature reserves. The permanently open estuaries that currently fall in protected areas include the Mtamvuna, Mzamba, Mnyameni, Mtentu and Mbashe estuaries. The rest of the Transkei is communal land which is managed by the local people (Colloty *et al.* 2000).

Ward and Steinke, (1982) and Adams *et al.* (2004) conducted detailed studies on the mangroves within the former Transkei region in 1982 and 1999, respectively. Although there are seventy-six estuaries in this area, only seventeen are permanently open and these are thus the estuaries where mangroves are expected. These authors estimated the total mangrove cover in South Africa to be 1 043 ha in 1982 and 1 660.07 ha in 1999, while the total mangrove cover of the Transkei estuaries was estimated at 272.2 ha in 1982 and 270.5 ha in 1999. Ward and Steinke, (1982) observed mangroves within seventeen of the seventy estuaries studied and Adams *et al.* (2004) reported that only fourteen of the seventeen estuaries still had mangroves. Mangrove forests were thus lost in estuaries such as Mnyameni, Mzimvubu and Bulungula due to mouth closures, floods or changes in salinity. The total loss of mangrove area cover for these estuaries in 1999 was 17.6 ha (6.5 %) across the seventeen estuaries, which was a loss of 1.04 ha per annum (Adams *et al.* 2004).

The aim of this study was to investigate the impact of natural changes (drought, floods) and anthropogenic factors (harvesting and browsing) on the population structure and distribution of three different mangrove species *Avicennia marina*, *Bruguiera gymnorrhiza* and *Rhizophora mucronata*. The changes over time were quantified by comparing the findings of this study with previous studies documenting the forests in 1982 (Ward and Steinke 1982) and 1999 (Colloty *et al.* 2000; Adams *et al.* 2004). Further objectives of this study included establishing some future predictions for the changes in the status of mangroves along the former Transkei coast, the influence of different impacts on these changes, and to compare these to the results of studies presented by Adams *et al.* (2004). This would indicate the resilience of the mangroves in the warm temperate region and determine whether mangrove area cover along the east coast showed an increase or decrease.

The following hypotheses were tested (1) Mangrove cover showed an overall decrease since 1999 due to an increase in populated areas. (2) Healthy mangrove forests occur in protected areas, such as the adult to seedling ratio indicates new growth in forests. (3) There has been steady loss in mangrove area cover due to human impacts with harvesting being the dominant factor impacting cover in most estuaries.

This research will provide a better understanding of the present distribution, area cover and structures of the mangrove population, and will also provide greater insight regarding the possible stressors on these forests. The findings will provide necessary information to decision makers who will use this information for implementation of management practices. For these reasons, the research into and management of mangrove forests is of great importance in order to conserve these unique ecosystems.

5.2 Materials and Methods

5.2.1 Site description

Mangroves in South Africa can only be found in permanently open estuaries, which fall into the warm temperate and subtropical regions on the east coast of South Africa. From 1997 to 1999, the average rainfall ranged between 1100 mm y^{-1} in the northern Transkei to 750 mm yr^{-1} in the southern Transkei (Colloty *et al.* 2000). The total annual rainfall between 2000 and 2012 is shown in Figure 5.1 for the Transkei region. It can be calculated from Figure 5.1 A that in the northern Transkei (Port Edward), average rainfall was 1 167 ± 160 mm y^{-1}. Figure 5.1 B shows average rainfall for the southern Transkei (East London) to be 930 ± 303 mm y^{-1} (South African Weather Service 2012). Figure 5.2 summarizes the annual average temperature ranging between 17.4 ± 0.9 °C (Port Edward) in the north and 14.4 °C in the south (East London) in the winter months (June – August) to average annual summer temperatures for the whole coastline of 23 ± 0.5 °C (Figure 5.2). The study area was along the coastline from the Great Kei River (32^{o}41'; S 28^{o}23' E) to the Mtamvuna Estuary (31^{o}04' S 30^{o}11' E) (Figure 6.3), which is the transitional zone between the warm temperate and subtropical biographical region (South African Weather Service 2012).

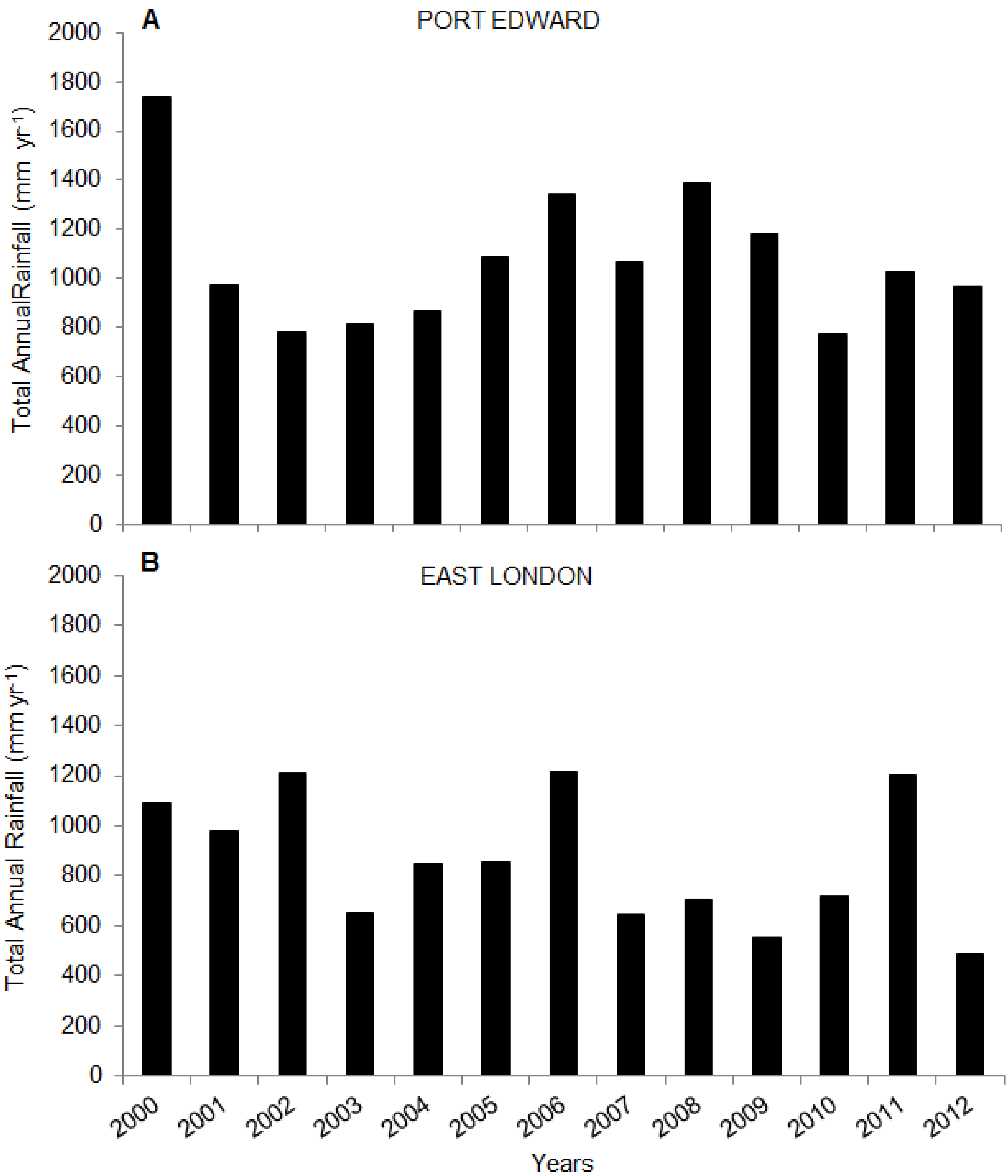

Figure 5.1:The total annual rainfall for the (A) northern (Port Edward) and (B) southern (East London) Transkei (South African Weather Service, 2012).

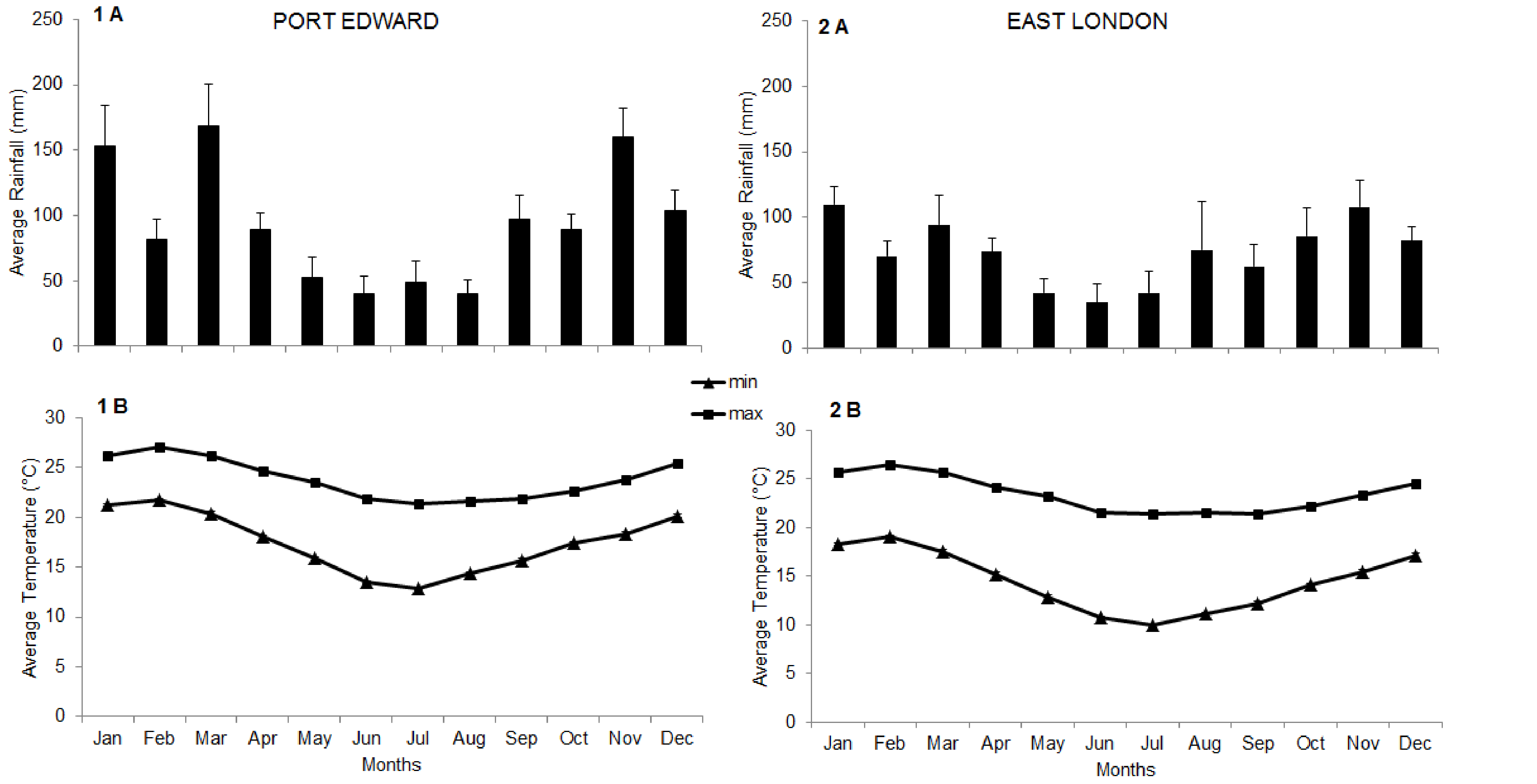

Figure 5.2: The average monthly rainfall and temperature (2000 – 2012) for the (1) northern (Port Edward) and (2) southern (East London) Transkei. (Bars = SE). For temperature, the SE bars are too small to see (South African Weather Service, 2012).

5.2.2 Field surveys

Figure 5.3 depicts the study area, a transitional zone between the warm temperate and subtropical geographical regions, along the coastline from the Great Kei River (32°41' S; 28°23' E) to the Mtamvuna Estuary (31°04' S; 30°11' E). Two field trips were completed in this region, one in November 2011 and one in May 2012, where a total of seventeen estuaries were sampled. The area covered by mangrove and salt marsh habitats was measured by using a hand-held GPS (Garmin). Mangrove species composition, density and population structures were assessed and completed for each estuary. Possible anthropogenic impacts and the general state of the mangroves were also noted in order to identify the various stressors on these habitats and the possible future status of the estuary habitats in response to the current natural impacts. Overall impacts that were noted included changes in freshwater inflow (presence of water pumps and pipelines), pollution, agricultural activities bordering mangrove settings and those on the banks, veld fires that destroyed mangrove stands, mangrove harvesting, disturbance through cattle trampling and browsing, and any other mangrove disturbances that had an impact on the forests (Table 5.3).

5.2.3 Mapping mangrove area cover

Esri ArcGIS Desktop 10 (2010) software, digital satellite images (Spot 5 2010, 2 m spatial resolution), and ground-truthing data from 2011 and 2012 were used to map the extent of the estuary habitats in the different estuaries. This mapping exercise focussed on the distribution of mangroves but also included salt marshes.

5.2.4 Mangrove state and health index

For each estuary, human pressures (harvesting) were recorded to provide an assessment of the state of the mangroves. A heath index (HI) was determined using the pressures summarised in Tables 5.1 and 5.2. Each impact in each estuary and the extent of its influence on the HI was scored separately. This was done by recording the presence / absence of the impacts in all 17 estuaries. Percentage occurrence of impacts in all 17 estuaries was also calculated to identify the dominant impacts. Mangrove harvesting intensity was determined in areas where harvesting was evident, mostly in more accessible areas close to footpaths. This was done by sampling between three and six 25 m^2 quadrats in a stand, depending on the size of the stand. The harvesting intensity was then determined using Table 5.1. A score of 1 to 2 indicates a low harvesting intensity, 3 to 4 a medium intensity and 5 to 6 indicates a high harvesting intensity.

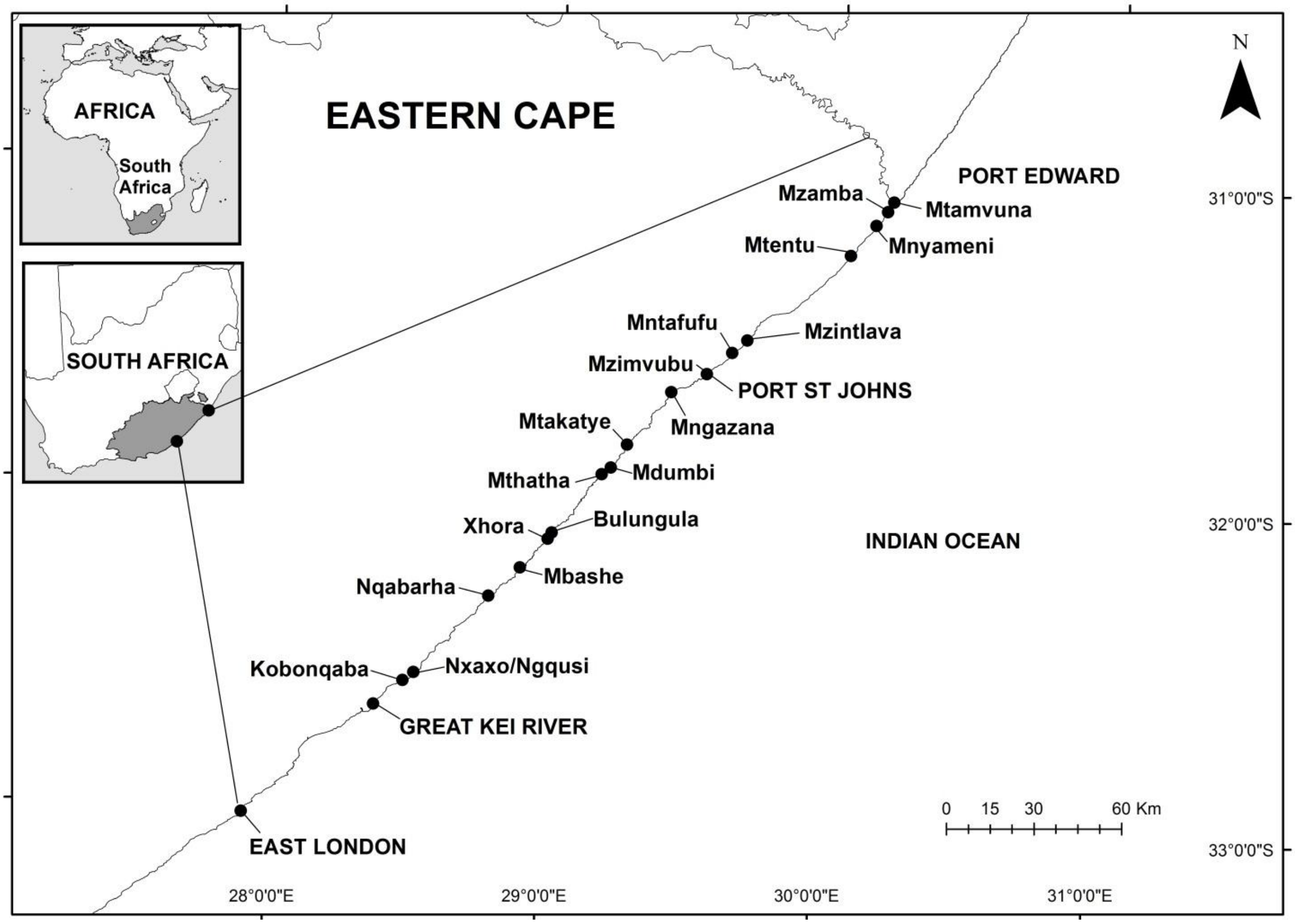

Figure 5.3: The 17 estuaries in the former Transkei within the Eastern Cape Province.

Table 5.1: The Aesthetic Health Index Categories adapted from Turpie, (2002) and the present status categories used in this study for the assessment of the extent of anthropogenic pressures in each estuary.

Aesthetic Health Index Score	**Description**	**Present Status Category**
91 – 100	Unmodified, natural	1
76 – 90	Largely natural with few modifications	2
61 – 75	Moderately modified	3
41 – 60	Largely modified	4
21 – 40	Highly degraded	5
0 – 20	Extremely degraded	6

Table 5.2: The dominant pressures which have been used to determine a Health Index score

Anthropogenic Impacts	**Description**
Footpaths & Trampling	Paths created by harvesters and cattle through mangrove habitat left bare and unvegetated
Livestock browsing	This was recorded when animals were seen in the mangroves or there was evidence of tracks as well as a distinctive browse line; mainly cattle and goats but sheep and pigs were also observed to browse in the mangrove forest
Wood harvesting	Visible stumps were identified and counted in the mangrove forests; for *Rhizophora* and *Bruguiera*, whole trees were felled and for *Avicennia* mostly only branches were taken; this was identified from clear saw cuts
Agriculture	Freshly ploughed fields behind mangrove forests and along banks
Mud disturbance and erosion	Mostly from bait collection and from trampling by cattle as well as clear cut fishing spots in mangroves to gain access to water
Pollution	Litter (plastic pollution) and excessive cow dung in mangrove areas
Alien plant invasion in intertidal	Mostly in the middle and upper reaches where invasive plants were found growing behind or among mangroves
Sand mining and water abstraction	Any signs of these close to mangrove habitat, were sand mining was used for building material and water abstraction mostly for agricultural irrigation
Veld fires	Agricultural fields that were burned adjacent to mangrove forests resulting in fire damage to mangroves

5.2.5 Mangrove characteristics

Mangrove characteristics give insight into species distribution and composition, and the population structure of each species in the estuaries. Within each estuary, transects were placed in each dense mangrove stand, from the water's edge landwards into the upper intertidal region. In estuaries where mangrove area cover was lower than 5 ha, only one transect was placed through the largest stand, and three replicate 25 m^2 quadrats were chosen along a transect (total area 75 m^2). In estuaries with a mangrove cover > 5 ha, three transects with each represented by three 25 m^2 quadrats were chosen along the different transects (total area 225 m^2). Within each quadrat, the mangrove species composition, density and population structure were measured. For population structure, all plant heights were recorded within a quadrat and different height classes were determined. The classes were seedlings (< 50 cm), saplings (50 – 129 cm) and adult trees (> 130 cm). For all adult trees the circumference at breast height of the stem (CBH) was measured at 1.3 m.

CBH was later converted to diameter at breast height (DBH) using the equation:

$$DBH = CBH/\pi$$

The DBH (m) was also used to calculate the total basal area (m^2 ha^{-1}) by using the following equation (Kathiresan and Bingham 2001):

$$\text{Basal area } (m^2) = 0.005 * \text{DBH (m)}$$

$$\text{Total Basal area } (m^2\ ha^{-1}) = (\sum \text{basal area in } m^2/\text{quadrat in } m^2) * 10\,000$$

For estuaries where the mangrove distribution was too sparse to use quadrats and where only a few individuals were present, all plants were measured. These estuaries were the Mntamvuna (data obtained from Rajkaran *et al.* (2009) was used for this purpose), Mzimvubu and Bulungula estuaries.

5.2.6 Data analysis

All statistical analysis was done using Minitab 15 Statistical Software. (Minitab, Inc. USA). All the data was first tested for normality. For all normal data, one way analysis of variance (ANOVA) was used, followed by Tukey post hoc Honestly Significant Difference tests, to separate the means. A Kruskal-Wallis ANOVA was used to test for differences between the density of life forms and across the estuaries. Pearson Correlation analysis was used to determine correlations between estuaries, abiotic characteristics and mangrove population structure. For all analysis, significance was determined at $\alpha = 0.05$.

5.3 Results

5.3.1 Mangrove area cover

Mangroves were recorded in all 17 estuaries. However, Kobonqaba Estuary lost more than 95 % of its mangroves (3.5 ha) due to mouth closure in September 2008 caused by sea storms. The estuary mouth remained closed until the end of July 2011 due to reduced freshwater input caused by a recent drought. Only one adult *Bruguiera gymnorrhiza* plant and some individual *Avicennia marina* trees (< 15 individuals) which carried mature propagules in 2011 were still alive in 2012 (Rajkaran per. comm. 2012). In estuaries such as the Mnyameni, Mzimvubu and Bulungula, where mangroves were completely lost in 1999, new mangroves were recorded in this study in 2011 and 2012.

Table 5.3 summarizes the change in mangrove area cover within the different estuaries in the former Transkei. The dark shading represents an increase and the light shading a decrease in area cover. The increase or decrease in cover was related to past mangrove area cover and also represents the gain and loss of mangrove area cover in each estuary. From the Table 5.3, it can be seen that an increase in mangrove area cover was observed in nine estuaries. Mnyameni Estuary had a 100 % increase from 1999, where mangroves were lost due to floods. Since then, mangroves have re-established into dense, healthy stands. However, compared to 1982, an increase of only 40 % in mangrove cover was observed. Mdumbi Estuary showed an increase of 90 % in mangroves, Mtamvuna Estuary 75 %, Mzamba Estuary 50 %, Xhora Estuary 35 %, Nqabarha/ Nqabarana estuaries 28 % and finally, Mtakatye Estuary showed a 10 % increase in mangrove area cover. Natural re-establishment of mangroves was also found at the Mzimvubu Estuary. However, only a few individuals were recorded in 2011. This was similar for Bulungula Estuary, where only a few individuals were found.

The mangrove area has decreased in seven estuaries since 1999. These were Kobonqaba Estuary (95 %), Mtentu Estuary (70 %), Nxaxo/Ngqusi estuaries (37 %), Mbashe Estuary (34 %), Mthatha Estuary (26 %) and Mngazana Estuary (19 %). Mngazana is an important estuary for mangrove conservation as close to 50 % of the total mangrove cover along the east coast occurs in this system (Table 5.3). Xhora Estuary contained 10.6 % of the mangrove cover and Mthatha Estuary 12.7 %. All other estuaries contributed less than 5 %.

Salt marshes occurred in most estuaries but the area covered was variable. Appendix 17 contains the salt marsh species identified for each estuary. Table 5.4 indicates that in the large estuaries which also had the largest mangrove area cover it was found that these estuaries also contained the largest intertidal salt marsh area except for Mngazana Estuary which only had 1.25 ha. Estuaries with high intertidal salt marsh area included Mthatha Estuary (27.3 ha), Mntafufu Estuary (14 ha) and Mnyameni Estuary (7.4 ha). Smaller mangrove-containing estuaries such as Mbashe and Mdumbi had 4.2 ha and 4 ha of salt marsh cover respectively. All other estuaries had less than 5 ha of salt marsh area.

Table 5.3: The mangrove area cover (ha) within the different estuaries of the former Transkei region.

Light shaded and bold text represents a decrease in area cover, dark shaded and bold text an increase in area cover. Past area cover values were taken from Ward and Steinke (1982) and Adams *et al.* (2004) for 1999.

Estuary	Past cover (1982)(ha)	Percentage (%) of total cover	Past cover (1999) (ha)	Percentage (%) of total cover	Present cover (2012)(ha)	Percentage (%) of total cover
Mtamvuna	1	0.4	**0.25**	0.09	**1**	0.4
Mzamba	1	0.4	**0.15**	0.06	**0.3**	0.1
Mnyameni	3	1.1	**0**	0	**5**	2.1
Mtentu	1	0.4	**2**	0.74	**0.6**	0.2
Mzintlava	1.5	0.5	**1.75**	0.65	1.7	0.7
Mntafufu	10	3.7	**12.4**	4.58	12	5
Mzimvubu	1	0.4	**0**	0	**0.03**	0
Mngazana	150	55	**145**	53.59	**118**	49
Mtakatye	7.7	2.8	**9**	3.33	**10**	4.2
Mdumbi	1	0.4	**0.5**	0.18	**5**	2.1
Mthatha	34	12.5	**42**	15.52	**31**	12.9
Bulungula	3.5	1.3	**0**	0	**0.014**	0
Xhora	16	5.9	**16.5**	6.1	**25.5**	10.6
Mbashe	12.5	4.6	**14**	5.17	**9.2**	3.8
Nqabarana/Nqabarha	9	3.3	**8.5**	3.14	**11.8**	4.9
Nxaxo/Ngqusi	14	5.1	**15**	5.54	**9.5**	3.9
Kobonqaba	6	2.2	**3.5**	1.29	**0**	0
Total	272.2	100	**270.55**	100	**240.64**	100

Table 5.4: The salt marsh area cover (ha) within the different estuaries of the former Transkei region in 2012. Bold and shaded boxes indicate ≥ 4 ha salt marsh area cover.

Estuaries	Salt marsh area cover (ha)
Mzamba	1.7
Mnyameni	**7.4**
Mtentu	< 0.1
Mzintlava	0.12
Mngazana	1.25
Mntafufu	**14**
Mzimbuvu	< 0.1
Mtakatye	2.7
Mdumbi	**4**
Mthatha	**27.3**
Bulungula	< 0.1
Xhora	< 0.1
Mbashe	**4.2**
Nqabarha	< 0.1
Nxaxo/Ngqusi	2.35
Kobonqaba	< 0.1

Table 5.5. shows the mangrove species status in conservation areas. The only estuaries that fall within a nature reserve are Mtamvuna Estuary (KZN Parks on the border of KwaZulu-Natal and the Eastern Cape), Mnyameni and Mtentu estuaries (in a marine protected area, Mkambati Nature Reserve), and Mbashe Estuary (Dwesa/Cebe Nature Reserve). Only 7 % of the mangrove area cover is currently protected. According to Colloty *et al.* (2000), 18 % of the estuaries were protected and at present, additional marine protected areas have increased the percentage of protected estuaries to 29 %. In estuaries where mangroves had been lost previously (1999), species had since then re-established. This had been *A. marina* and *B. gymnorrhiza,* which had recolonized the Mnyameni and Mzimvubu estuaries, while *A. marina* was planted in Bulungula Estuary. *Buguiera gymnorrhiza* increased in Mdumbi Estuary and *Rhizophora mucronata* in Xhora Estuary, while at Mtentu Estuary no *A. marina* plants had been observed in this study.

Table 5.5: Mangrove species present in past literature compared to the present distribution (2012) (shaded and bold are differences in species composition) and their conservation status

Estuary	**Species recorded in Ward and Steinke (1982)**	**Species recorded in Adams *et al.*, (2004)**	**Species present in 2012**	**Conservation status**
Mtamvuna	*B. gymnorrhiza*	*B. gymnorrhiza*	*B. gymnorrhiza*	Protected
Mzamba	*B. gymnorrhiza*	*B. gymnorrhiza*	*B. gymnorrhiza*	Previously unprotected now partly protected by Casino in Port Edwards
Mnyameni	*A. marina* *B. gymnorrhiza*	None	***A. marina*** ***B. gymnorrhiza***	Previously unprotected now protected
Mtentu	*B. gymnorrhiza*	*A. marina* *B. gymnorrhiza*	***B. gymnorrhiza***	Mkambati Nature reserve
Mzintlava	*B. gymnorrhiza*	*B. gymnorrhiza*	*B. gymnorrhiza*	Unprotected
Mntafufu	*A. marina,* *R. mucronata* *B. gymnorrhiza*	*A. marina*, *R. mucronata* *B. gymnorrhiza*	*A. marina*, *R. mucronata* *B. gymnorrhiza*	Unprotected
Mzimvubu	*A. marina* *B. gymnorrhiza*	None	***A. marina,*** ***R. mucronata*** ***B. gymnorrhiza***	Unprotected
Mngazana	*A. marina,* *R. mucronata* *B. gymnorrhiza*	*A. marina,* *R. mucronata* *B. gymnorrhiza*	*A. marina,* *R. mucronata* *B. gymnorrhiza*	Unprotected
Mtakatye	*A. marina,* *R. mucronata* *B. gymnorrhiza*	*A. marina,* *R. mucronata* *B. gymnorrhiza*	*A. marina,* *R. mucronata* *B. gymnorrhiza*	Unprotected
Mdumbi	*A. marina*	*A. marina*	*A. marina* ***B. gymnorrhiza***	Unprotected
Mthahta	*A. marina,* *R. mucronata* *B. gymnorrhiza*	*A. marina,* *R. mucronata* *B. gymnorrhiza*	*A. marina,* *R. mucronata* *B. gymnorrhiza*	Unprotected
Bulungula	*A. marina,* *R. mucronata* *B. gymnorrhiza*	None	***A. marina*** (replanted)	Unprotected
Xora	*A. marina* *B. gymnorrhiza*	*A. marina* *B. gymnorrhiza*	*A. marina,* ***R. mucronata*** *B. gymnorrhiza*	Unprotected
Mbashe	*A. marina* *B. gymnorrhiza*	*A. marina* *B. gymnorrhiza*	*A. marina* *B. gymnorrhiza*	Dwesa and Cebe Nature reserves
Nqabarha	*A. marina*	*A. marina*	*A. marina*	Unprotected
Nxaxo/Ngqusi	*A. marina* *B. gymnorrhiza*	*A. marina,* *B. gymnorrhiza* *R. mucronata*	*A. marina,* *B. gymnorrhiza* *R. mucronata*	Unprotected
Kobonqaba	*A. marina*	*A. marina*	**** A. marina,*** ****B. gymnorrhiza***	Unprotected

* More than 95 % of *A. marina* trees died due to mouth closure and only one small *B. gymnorrhiza* tree was recorded. Less than ten *A. marina* trees were still alive and continuous monitoring of these will determine if the species will persist in Kobonqaba Estuary.

5.3.2 Mangrove state and health index

Figure 5.4 summarises the percentage occurrence of biotic impacts in all 17 estuaries in 2012 and the photographs provide evidence of some of the different impacts. The health index (HI) values are summarised in Table 5.6. Within the estuaries, the different intensities of biotic impacts that affected the mangroves are discussed. The largest impact was the harvesting of mangrove wood (total HI = 51), which was evident in 70.6 % of the estuaries (Plate 5.1 A and B). Livestock browsing, propagule predation and trampling in mangrove stands had a total HI = 44 and were evident in 76.5 % of the estuaries (Plate 5.1 C to F). Foot paths through the habitats, which was coupled often with harvesting, had a total HI = 42 and was evident in 82 % of the estuaries (Plate 5.2 B). The impacts of invasive alien plants such as *Lantana camara* L. and *Opunia ficus-indica* Miller (L.) (prickly pear) in the intertidal habitats, especially in the upper reaches fringing the mangrove areas, was quite low. Alien plants were evident in 47 % of the estuaries and an HI = 27 was determined. Appendix 19 presents a detailed list of invasive alien species observed in the supratidal areas of the estuaries.

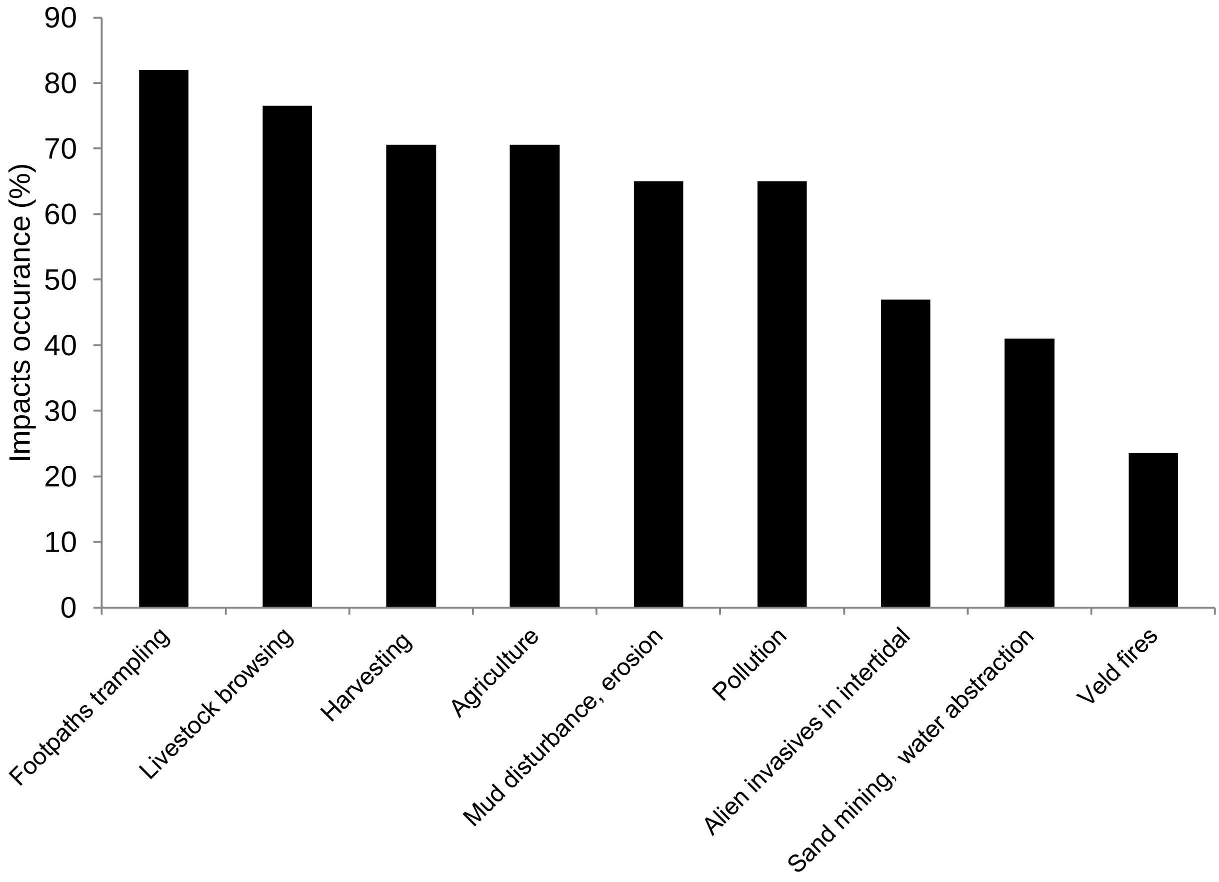

Figure 5.4: Percentage occurrence of impacts in all 17 estuaries in 2012.

Table 5.6: Scores of the biotic pressures in the Transkei estuaries (2012)

Estuaries	Invasive alien plants along estuary banks	Harvesting of mangrove wood	Livestock browsing & trampling in mangrove stands	Footpaths through habitats	Comments
Mtamvuna	-	-	-	-	In a nature reserve
Mzamba	-	4	2	2	-
Mnyameni	-	4	-	2	Healthy stands but over-harvested
Mtentu	-	3	-	2	-
Mzintlava	-	-	2	2	-
Mntafufu	2	4	2	3	Healthy stands but over-harvested
Mzimvubu	5	-	3	3	Alien invasive plants dominate
Mngazana	-	6	2	4	Large forest but stands are degraded
Mtakatye	3	3	3	3	Mostly only old trees
Mdumbi	4	4	5	3	Browsed, propagule predation
Mthatha	4	6	5	3	Larger forest but sparse and degraded
Bulungula	2	5	5	3	Bank erosion
Xora	4	4	4	4	-
Mbashe	-	-	-	-	-
Nqabarha/ Nqabarana	3	4	4	3	-
Nxaxo/Ngqusi	-	4	5	5	Healthy stands but over-utilized
Kobonqaba	-	-	2	-	-
Total scores	**27**	**51**	**44**	**42**	

For the abiotic impacts (Table 5.7), the greatest impacts are activities that cause mud disturbances, erosion and sedimentation resulting in possible smothering of mangrove pneumatophores. These included the bare ground along estuarine banks (HI = 34) evident in 65 % of estuaries, and agriculture behind mangrove stands (HI = 33) in 70.6 % of estuaries. Other impacts included pollution (for example litter dumped or washed down the river, and nutrient input due to excessive cow dung) with a HI score of 25 and evident in 65 % of estuaries, man-made structures, sand-mining and water abstraction that may have had an influence on the estuaries' health was evident in 41 % of the estuaries and had a HI score of 18, and veld fires resulting from burning agricultural fields behind mangrove stands destroyed trees (HI = 10) in 23.5 % of the estuaries.

Table 5.7: Scores of the abiotic pressures in the Transkei estuaries (2012)

Estuaries	Agriculture behind mangroves adding to erosion, sediment deposition	Intertidal bank disturbance (eg from bait collection)	Pollution (eg. Litter & cow dung)	Veld fires from agriculture that destroyed mangrove stands	Other	Comment
Mtamvuna	-	-	-	-	3	Bridge close to mouth
Mzamba	2	-	1	-	-	Sediment accretion in some stands
Mnyameni	3	2	2	-	-	Forest is restricted to high cliffs
Mtentu	2	2	2	-	-	Veld fires destroyed old *Avicennia*
Mzintlava	3	2	2	2	-	Freshwater abstraction
Mntafufu	3	-	1	-	2	Freshwater abstraction
Mzimvubu	3	3	4	-	3	Sand-mining close to mouth
Mngazana	2	2	-	-	-	Retaining walls
Mtakatye	-	3	2	2	1	-
Mdumbi	3	3	3	3	-	Sand mining at launch site
Mthatha	3	3	4	3	2	-
Bulungula	5	6	2	-	-	-
Xora	2	4	2	-	-	Sedimentation at estuary mouth
Mbashe	-	-	-	-	3	Sediment accretion (marine sediment)
Nqabarha/ Nqabarana	2	4	-	-	4	-
Nxaxo/Ngqusi	-	-	-	-	-	Mouth closure resulted in
Kobonqaba	-	-	-	-	6	mass dieback of mangroves
Total scores	**33**	**34**	**25**	**10**	**18**	

Plate 5.1:High intensity harvested sites with many stumps of (A) *Bruguiera gymnorrhiza* at Mthatha Estuary and (B) *Avicennia marina* at Xhora Estuary close to villages. Cattle impacts are shown in (C) cattle browsing at water's edge and (D) Cattle go to great lengths to get to new mangrove growth, and as a result cause trampling in the mangrove forests. Here, the bull was knee-deep in mud. Propagules are affected by: (E) mature propagules are picked up from the 'swash' zone (white arrows) by goats and (F) where most *Avicennia marina* propagules were grazed from the intertidal area (pictures for Xhora Estuary).

Plate 5.2: (A) The absence of seedlings at Mdumbi Estuary with only old large trees. (B) Trampling can result in large bare areas, e.g. at Mtakatye Estuary. (C) The erosion of sediment from previous mangrove habitats (Bulungula Estuary). (D) In Kobonqaba Estuary, mouth closure in 2008 resulted in the mortality of almost all mangroves. The water line can still be seen clearly on the trees (white arrows). (E) One of the few surviving trees in Kobonqaba at higher elevation that carried mature propagules.

Plate 5.3: (A) *Bruguiera gymnorrhiza* (white arrows) growing among *Hibiscus tiliaceus* (Mtamvuna and Mzamba estuaries). (B) Sediment accretion covers roots and leaves adult trees stunted (< 2.5 m), with no seedlings or saplings at these elevated areas (Mzamba Estuary). (C) Tall *B. gymnorrhiza* trees (> 8 m) but only low seedling recruitment at Mtentu Estuary. (D) Healthy *Rhizophora mucronata* stands at Mngazana Estuary.

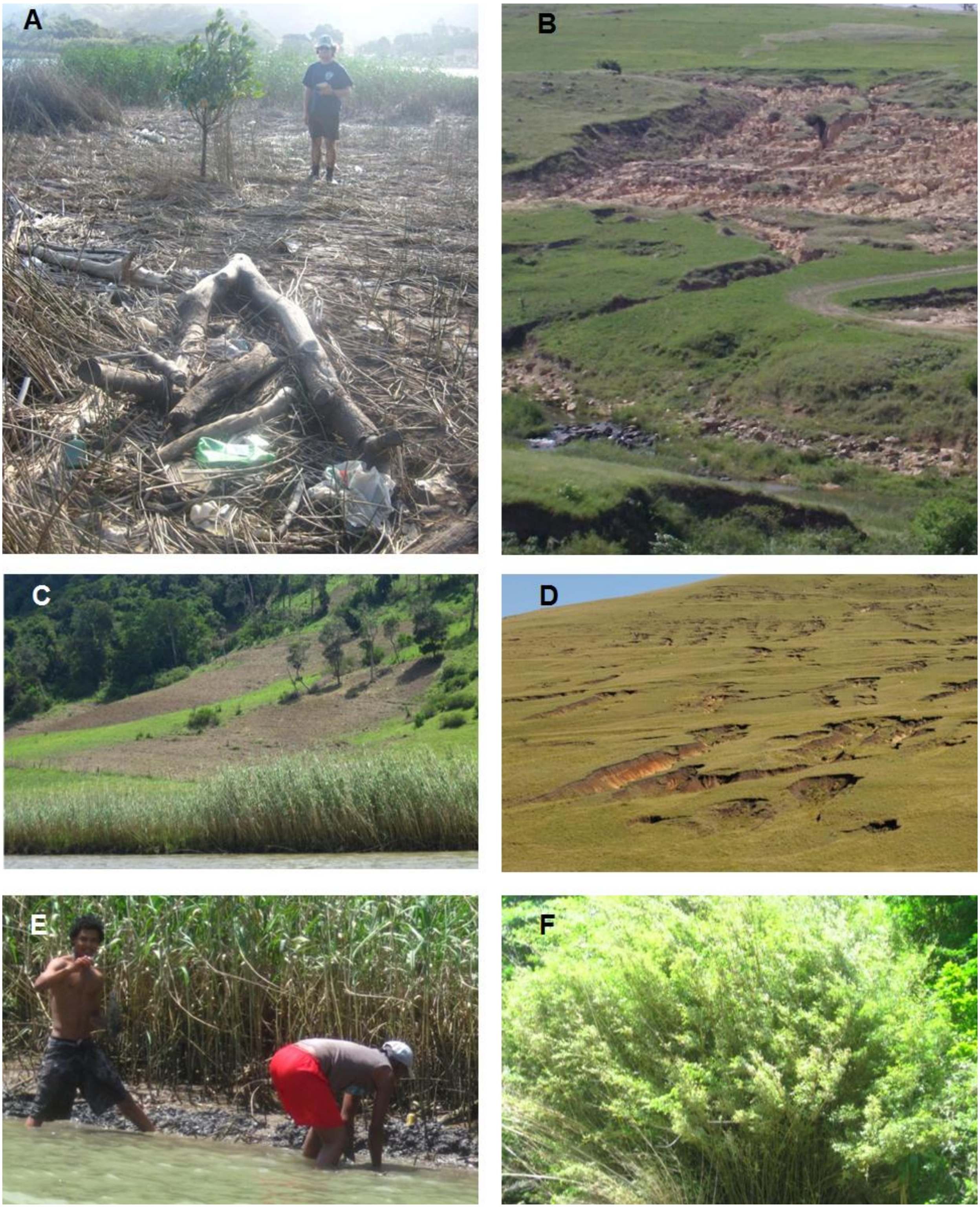

Plate 5.4: (A) Pollution such as litter was evident in most estuaries. (B) Poor agricultural practices results in erosion along rivers banks. (C) Agricultural fields close to the estuary. (D) Extensive erosion in upper catchment areas. (E) Bank disturbance such as bait collection as seen here in Mzimvubu was evident in most estuaries. (F) Alien invasive species such as this weed *Arundo donax* L. (giant reed) have become dominant in the intertidal region from the lower reaches to the upper reaches at Mzimvubu Estuary.

Plate 5.5:(A) The re-established forest at Mnyameni Estuary with tall *A. marina* trees and dense *B. gymnorrhiza* stands. (B) Bulungula Estuary had a few *A. marina* seedlings (arrows indicate mangroves that have survived as short trees). (C) The Nxaxo Estuary with only fringing *A. marina* mangroves in the middle reaches.

In Table 5.8 the estuaries that had gained the most mangrove cover area due to natural regeneration had been for the smaller forests (≤ 5 ha) such as Mdumbi, Mnyameni, Mtamvuna, Mzamba, Mzimvubu, while the larger forests (≥ 10 ha) Mtakatye, Xhora and Nqabarha estuaries had gained only a small percentage of their overall area cover, which was also found to be the case in Colloty *et al.* (2000). From 1999, 24 ha of mangrove area cover were gained, while 53 ha were lost. The mangrove area cover gained was mainly due to mangroves colonizing new habitats such as intertidal islands observed at Mdumbi Estuary (Appendix 10), Xhora Estuary (Appendix 13) and Nqabarana/Nqabarha estuaries (Appendix 15). On these mudbanks domestic livestock were excluded or reduced due to its inacessibility and thus reducing the browsing pressure in those stands. At Mtakatye Estuary mangrove have been expanding in the sheltered creeks and in estuaries where mangroves had been lost in 1999 such as Mzimvubu and Mnyameni estuaries. Mangroves had recolonized those estuaries again, while at Bulungula Estuary, mangroves had been replanted. Highest mangrove area cover loss was due to anthropogenic activities which had been in the larger forest such as observed at Mngazana, Mthata and Nxaxo/Ngqusi estuaries. Mthatha Estuary had the highest HI score (33) while Mngazana Estuary, which had the largest forest, had a HI score of 16 and Nxaxo/Ngqusi estuaries had a HI score of 14. Greatest pressures occurred in the larger estuaries and thus contributed to the total area cover lost and mangrove degradation. In contrast Mdumbi Estuary which has a small forest, had a high HI score (28) due to anthropogenic activities but also had the greatest gain in mangrove area cover since 1999 from 0.5 to 5 ha. However only adult *A. marina* trees had been recorded in this study and a large number of domestic livestock (cattle and goats) were observed browsing within the forest. Bulungula Estuary also had a high HI score (28) because this forest had not yet re-generated since 1999 (See Plate 5.5 B and Appendix 12) and here livestock browsing on the few individuals was evident. The mangrove loss that had been due to natural changes had been observed in Kobonqaba, Mbashe and possibly Mtentu estuaries and these had been in small forests. Bulungula Estuary was the only estuary where mangroves had actively been replanted.

There has been no relationship between HI scores and mangrove cover area loss; however mangrove cover loss had been 2.2 times greater than mangrove cover gain. Thus the combination of natural pressures in the small forests and the anthropogenic pressures in the larger forests contributed to the overall mangrove area cover loss.

Table 5.8: The total mangrove area cover loss or gain and the total pressure scores for each estuary from 1999

Estuary	**Mangrove area cover**		**Total biotic pressures score**	**Total biotic pressures score**	**Total pressures score**	**Pressures A=Anthropogenic N=Natural**
	gain (%)	ha				
Mtamvuna	75	0.75	0	3	3	A & N
Mzamba	50	0.15	8	3	11	A
Mnyameni	100	5	6	7	13	A & N
Mzimvubu	3 (from 1982)	0.03	11	13	24	A & N
Mtakatye	10	1	12	8	20	A
Mdumbi	10	4.5	16	12	28	A
Mzintlava	no change	0	4	9	13	A
Mntafufu	no change	0	11	6	17	A
Bulungula	1	0.014	15	13	28	A & N
Xhora	35	9	16	8	24	A
Nqabarana/ Nqabarha	28	3.3	14	10	24	A & N
		24	**113**	**92**	**205**	
	loss (%)	ha				
Mtentu	70	1.4	5	6	11	N
Mngazana	19	27	12	4	16	A
Mbashe	34	11	0	0	0	N
Mthatha	26	4.8	18	15	33	A
Nxaxo/Ngqusi	37	5.5	14	0	14	A
Kobonqaba	95	3.5	2	6	8	A & N
		53	**51**	**31**	**82**	

*gain from 1982, 1999 had no mangroves area cover in this estuary

Table 5.9 shows the harvesting intensity (ratio of stumps to adult trees per ha) in the former Transkei estuaries (2012). Mangrove harvesting had the highest pressure score, but was the third most frequent pressure and was one of the anthropogenic impacts measured by using the harvested stumps to adult trees ratio per ha of each species to determine the harvesting intensity. Thus a ratio of 1:2 is regarded as low, a 1:1 ratio is regarded as medium and a 2:1 ratio is recorded as a high harvesting intensity (Rajkaran *et al.* 2009). *Avicennia marina* was harvested in 9 of the 17 estuaries (53 %), while *Bruguiera gymnorrhiza* was harvested in 7 of the 17 estuaries (41 %). *Rhizophora mucronata* was harvested in 3 of the 17 estuaries (18 %). Mngazana Estuary had a higher abundance of this species and the harvesting intensity was high in this estuary as seen in a study conducted by Rajkaran *et al.* (2009). However, in all other estuaries, *R. mucronata* had a low density and therefore was not harvested in comparison with the other two species.

Table 5.9: Harvesting intensity (stumps to adult trees per ha) in the former Transkei estuaries (2012). Av = *Avicennia marina*; Bg = *Bruguiera gymnorrhiza*; Rm = *Rhizophora mucronata*. Low harvesting intensities had a HI score of 1 or less, medium score between 2 and 3, and high harvesting intensity had a HI score of 4 to 6

Estuaries	**Bg**	**Av**	**Rm**	**Comment**
Mtamvuna	N/A	-	-	Protected by the reserve
Mzamba	medium	-	-	-
Mnyameni	medium	**high**	-	Harvesting at every site
Mtentu	medium	-	-	Low density of trees
Mbashe	-	-	-	No evidence of harvesting
Mzintlava	low	-	-	-
Mntafufu	**high**	**high**	-	-
Mntafufu	-	-	medium	-
Mzimvubu	N/A	N/A	-	Only a few individuals
Mngazana	low	low	**high**	-
Mtakatye	medium	**high**	-	-
Mdumbi	-	**high**	-	Harvesting at main channel
Mthatha	**high**	**high**	**high**	Harvesting in creeks close to villages
Bulungula	-	N/A	-	Too few individuals
Xhora	-	**high**	-	-
Nqabarana/ Nqabarha	-	medium	-	-
Nxaxo/Ngqusi	low	**high**	N/A	Harvesting at main channel, only a few Rm individuals
Kobonqaba	N/A	**high**	N/A	All but a few are dead due to the closed estuary mouth

5.3.3 Salinity concentrations in estuaries

The salinity concentration (PSU) for the main channel (2000 and 2012) has been summarized in Appendix 19 for the different estuaries. Estuaries with low average water channel salinity were Mbashe Estuary with salinity range from 0.2 PSU in the middle reaches (MR) to 6 PSU at the lower reaches (LR) close to the estuary mouth. Mzimvubu Estuary was also freshwater dominant with low average salinity (8.6 PSU) for most of the estuary and only at the mouth did the salinity increase to 28.8 PSU. Estuaries that had salinity concentrations close to seawater concentrations, were Xhora Estuary (32.1 PSU = MR; 35.2 PSU = LR); Nqabarha Estuary (29.8 PSU = MR; 34.3 PSU = LR) and Mngazana Estuary (27.2 PSU = MR; 36.3 PSU = LR). All other estuaries (Appendix 19) had a gradient of salinity concentrations with brackish waters in the MR (17 to 23 PSU) and higher salinities close to sea water at the mouth (19 to 34.7 PSU).

5.3.4 Mangrove characteristics

The population structure (seedling, sapling and adult density), mean DBH (cm) for the adult trees, total basal area (m^2 ha^{-1}) and the adult to seedling ratio are summarized in Tables 5.10 to 5.13 for each mangrove species in each estuary. *Bruguiera gymnorrhiza* occurred in 13 (76 %) estuaries, *Avicennia marina* was found in 10 (56 %) estuaries and *Rhizophora mucronata* was only represented in 5 (29 %) of the 17 estuaries. The highest adult to seedling ratio was found at Mtakatye Estuary for *B. gymnorrhiza* and at Mnyameni Estuary for *A. marina*. The overall number of seedlings for *R. mucronata* was low as adults exceeded the number of seedlings (Tables 5.10; 5.11; 5.12).

Within the small (< 5 ha) estuaries (Mtentu, Mzamba, Mzintlava and Mzimvubu) (Table 5.10) *B. gymnorrhiza* was significantly higher in number at Mzintlava Estuary for all size classes compared to Mtentu, Mzamba and Mzimvubu ($F_{(df = 11; n = 3)}$ 26.9; $p < 0.05$). In Mzimvubu Estuary, mangrove density was the lowest and was represented only by a few individuals of *B. gymnorrhiza* and *R. mucronata*. Therefore, significantly lower numbers of *B. gymnorrhiza* saplings were recorded in Mzimvubu Estuary ($F_{(df = 11; n = 3)}$ 7.04; $p < 0.05$) and fewer adult trees compared with all the other estuaries (Mtentu, Mzamba and Mzintlava) ($F_{(df = 11; n = 3)}$ 22.49; $p < 0.05$). Only a few (< 5) individual *B. gymnorrhiza* saplings were recorded in Mdumbi Estuary within the creeks, and adult *R. mucronata* trees were found at Mzimvubu Estuary.

When comparing the larger (> 9 ha) estuaries (Mnyameni, Mbashe, Mtakatye, Mntafufu, Mthatha, and Xhora) (Table 5.10) a significantly higher density of *B. gymnorrhiza* seedlings ($H_{(df = 5; n = 9)} = 23.33$; $p < 0.05$) and saplings ($H_{(df = 5; n = 9)} = 33.87$; $p < 0.05$) were recorded in Mnyameni and Mntafufu estuaries, compared to Mthatha and Mbashe estuaries. The density of saplings in Xhora Estuary was significantly lower ($H_{(df = 5; n = 9)} = 33.87$; $p < 0.05$) compared to Mthatha Estuary, while the highest sapling density ($H_{(df = 5; n = 9)} = 33.87$; $p < 0.05$) for *B. gymnorrhiza* was recorded in Mtakatye Estuary, which was also significantly higher compared with Mnyameni Estuary. In these larger mangrove-containing estuaries, the lowest density of *B. gymnorrhiza* in all size classes was recorded in the Nxaxo/Ngqusi estuaries. When comparing the *B. gymnorrhiza* adult tree density between the estuaries, Mnyameni Estuary had the highest density. However, Mntafufu Estuary was the only estuary that had a significantly higher *B. gymnorrhiza* adult tree density ($H_{(df = 5; n = 9)} = 11.44$; $p < 0.05$) than Mthatha and Mbashe estuaries. The highest number of *Avicennia marina* seedlings and the highest adult to seedling ratio were recorded in Mnyameni Estuary. The highest number of saplings were recorded in Mngazana Estuary, but there were none in Mthatha Estuary, which was significant ($H_{(df = 7; n = 9)} = 10.45$; $p < 0.05$) compared to the Nxaxo/Ngqusi estuaries. However, no other significant difference was recorded for *A. marina* seedlings or saplings for the remaining larger mangrove-containing estuaries.

The highest tree density for *A. marina* and *R. mucronata* was recorded in Mngazana Estuary. Nqabarha/Nqabarana Estuary had a significantly higher ($H_{(df = 7; n = 9)} = 21.08$; $p < 0.05$) number of *A. marina* adult trees compared to Mnyameni, Xhora and Mtakatye estuaries, while Mbashe Estuary had a significantly lower ($H_{(df = 7; n = 9)} = 21.08$; $p < 0.05$) adult density compared to the Mtakatye Estuary (Table 5.11). Mbashe Estuary had the highest DBH and basal area ($m^2\ ha^{-1}$) for *B. gymnorrhiza* ($F_{(df = 47; n = 48)} = 4.75$, $p < 0.05$); ($F_{(df = 45; n = 83)} = 6.77$, $p < 0.05$) and *A. marina* trees ($F_{(df = 59; n = 60)} = 6.93$, $p < 0.05$); ($F_{(df = 9; n = 48)} = 4.75$, $p < 0.05$) compared to all other estuaries. However, Mnyameni, Mthatha, and Xhora estuaries had significantly higher basal areas ($m^2\ ha^{-1}$) ($F_{(df = 9; n = 48)} = 4.75$, $p < 0.05$) compared to Mntafufu and Mtakatye estuaries. For *R. mucronata* similar basal area ($m^2\ ha^{-1}$) and DBH were recorded at the Mthatha and Mngazana estuaries ($F_{(df = 9; n = 10)} = 7.11$; $p > 0.05$).

Mnyameni, Nxaxo/Ngqusi, Mdumbi and Nqabarhana/Nqabarha, mangroves increased in area cover due to available mud banks. Appendix 2 shows Mnyameni Estuary and mangroves are expanding with available sedimentation along the channel due to deposition from the creek. In Mdumbi Estuary (Appendix 10) small *A. marina* trees have colonized the mudbank, which will become more stabilized as mangroves expand. These are also out of reach of domesticated livestock. Similarly at Nxaxo/Ngqusi Estuary (Appendix 16) sediment deposits have increased the area that can be colonized by mangroves around the two islands, where no livestock will be able to impact these established stands. At Nqabarhana/Nqabarha (Appendix 15 see enlarged section A) much of the connection with the smaller Nqabarana Estuary has been restricted by sediment and mangroves are expanding on these mudbanks; however excessive sedimentation threatens to close off Nqabarana Estuary from Nqabarha Estuary. Estuaries where mangrove expansion would be limited are Mtentu Estuary (Appendix 3) and Mtamvuna estuaries. In both estuaries mangroves are restricted by high cliffs and only available sedimentation is around the estuary mouth.

5.1 Discussion

5.1.1 Pressures on Transkei mangrove forests

Mangrove ecosystems are protected in South Africa and fall under the protection of the Natural Forest Act, 1998 (Act no. 84 of 1998). Management plans to protect the forests have been put in place by the Department of Agriculture, Forestry and Fisheries (Lewis *et al.* 2002). However, much research as well as sustainable management plans are still needed for more effective conservation of these mangroves. This is not without challenges as many rural people are still directly dependent on estuarine resources, and if these ecosystems change due to natural and/or anthropogenic influences, so does the mangrove species composition, population structure and distribution (Blasco *et al.* 1996). Most of the former Transkei is still undeveloped and difficult to access by dirt road. However, with human population increasing, and where the majority of people are still living in poverty, there is an increasing pressure on natural resources. This increase in demand for firewood, building material, fodder for livestock and subsistence fishing has visible impacts on mangrove habitats. Globally, mangroves are also harvested for charcoal, timber and for their medicinal purposes (Dahdouh-Guebas *et al.* 2006b). In this study, the extent to which human activities contributed to mangrove degradation was investigated. Pressures focussed on in this research included poor agricultural practices which result in increased erosion and sediment loads, harvesting of branches and whole trees, trampling and footpaths in mangrove habitats, livestock browsing on mangrove foliage and propagule predation, alien invasive plants that dominate the estuary banks in some estuaries, and reduced freshwater inflow due to water abstraction.

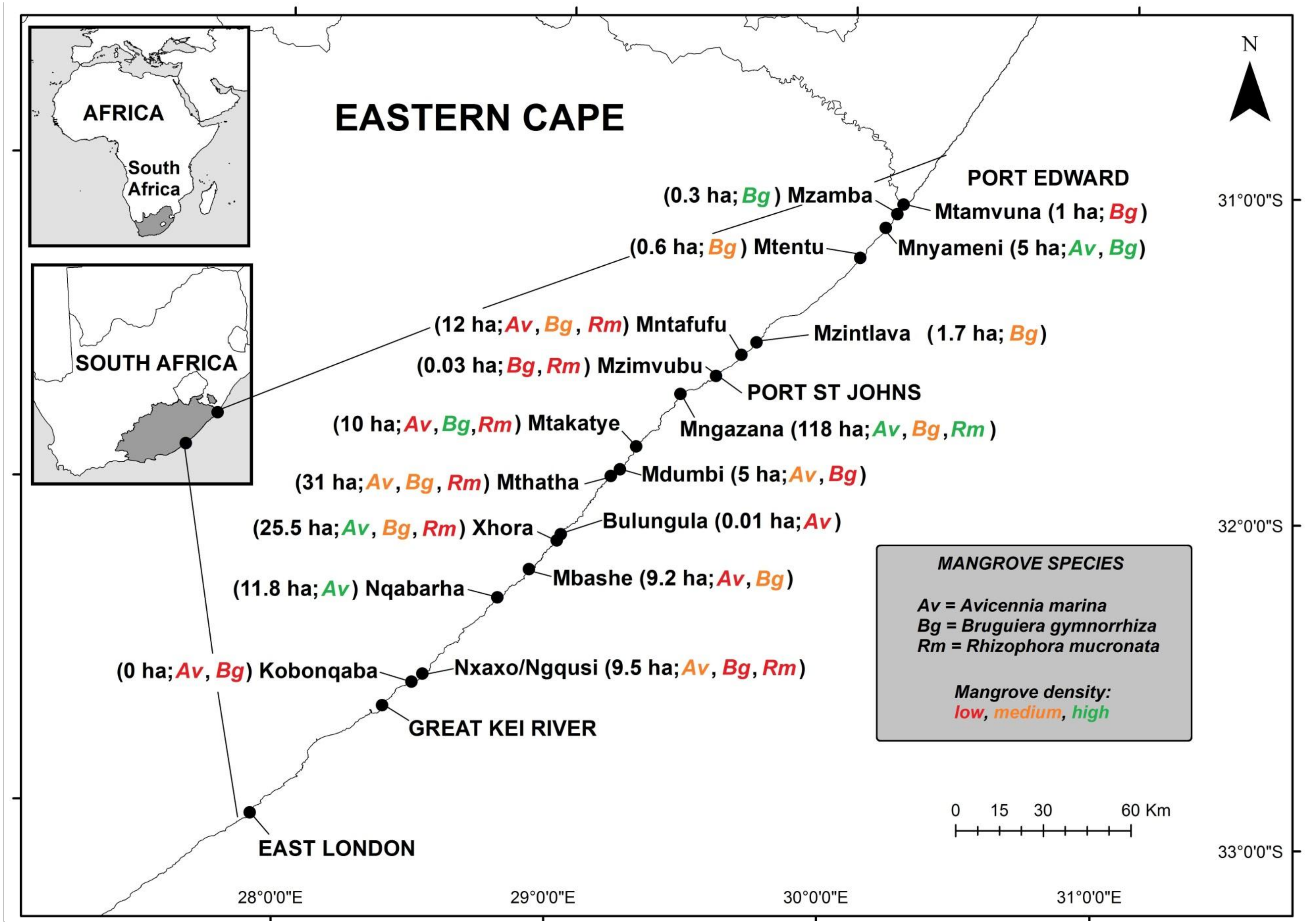

Figure 5.5: The 17 estuaries studied with mangroves indicating species present and the mangrove area present in 2011 / 2012

Table 5.10: The number of seedlings, sapling and adult trees of ***Bruguiera gymnorrhiza*** reported per ha in each estuary (± SE).

Estuaries	Seedlings ha^{-1}	Saplings ha^{-1}	Adults ha^{-1}	Mean DBH ± SE (N)	Basal area (m^2 ha^{-1})	Adult : seedling
LOW DENSITY						
Nxaxo/Ngqusi	0	356 ± 435	689 ± 844	2.52 ± 0.45 (31)	0.32*	2:1 Saplings only
MEDIUM DENSITY						
Mtentu	6 800 ± 2 116	2 533 ± 1733	7 733 ± 1 964	5.40 ± 0.88 (57)	2.00 ± 0.42	1:1
Mbashe	0	0	8 866 ± 8 866	3.42 ± 0.29 (61)	8.88*	no seedlings
Mzintlava	4 933 ± 581	7 200 ± 1 743	14 933 ± 1 922	2.38 ± 0.17 (109)	1.78 ± 0.27	3:1
Mntafufu	6 711 ± 1 050	12 800 ± 6 929	7 377 ± 4 633	2.71 ± 0.34 (267)	1.80 ± 0.45	1:1
Mngazana	1 696 ± 935	6 830 ± 1 442	3 482 ± 1 061	5.9 ± 0.6 (72) [A]	1.99 ± 0.76	2:1
Mthatha	1 155 ± 1 155	0	1 911 ± 1 646	4.62 ± 0.89 (43)	0.59 ± 0.27	2:1
Xhora	8 311 ± 3 330	5 288 ± 1 053	9 244 ± 1 769	3.58 ± 0.49 (147)	1.52 ± 0.42	1:1
HIGH DENSITY						
Mzamba	20 533 ± 2 575	10 400 ± 2 498	12 933 ± 581	2.56 ± 0.05 (131)	0.72 ± 0.15	1:1.5
Mnyameni	28 666 ± 6 944	34 977 ± 3 362	12 400 ± 4 183	2.49 ± 0.34 (264)	2.19 ± 0.84	1:2
Mtakatye	31 022 ± 30 490	5 600 ± 4 948	8 933 ± 4 569	3.14 ± 0.29 (194)	2.02 ± 0.37	1:3

*Individuals only and represents the whole population = total N; [A] = Rajkaran (2011); Low density = < 999; Medium density = 1000-10 000; High density = >10 000 individuals ha^{-1}.

Table 5.11: The number of seedlings, sapling and adult trees of ***Avicennia marina*** reported per ha in each estuary (± = SE).

Estuaries	Seedlings ha^{-1}	Saplings ha^{-1}	Adults ha^{-1}	mean DBH ± SE (N)	Basal area (m^2 ha^{-1})	adult : seedling
LOW DENSITY						
Mbashe	533 ± 352	333 ± 261	9 466 ± 2 764	8.84 ± 1.93 (107)	4.97 ± 1.69	17:1
Mntafufu	711 ± 645	533 ± 407	1 511 ± 1 444	8.10 ± 2.61 (56)	1.19 ± 0.56	2:1
Mtakatye	2 000 ± 1 740	622 ± 438	718 ± 643	5.51 ± 1.27 (130)	1.56 ± 0.44	1:3
MEDIUM DENSITY						
Mthatha	0	0	2 400 ± 961	12.91 ± 1.95 (54)	1.49 ± 0.30	no seedlings
Nxaxo/Ngqusi	2 185 ± 1 093	6 978 ± 2 505	3 311 ± 1 404	1.29 ± 0.15 (561)	0.10 ± 0.02	1:1.5
Mdumbi	0	0	1 644.44 ± 177	10.88 ± 1.4 (37)	0.82 ± 0.09	no seedlings
HIGH DENSITY						
Mnyameni	52 355 ± 29 537	1 733 ± 815	1 066 ± 335	3.11 ± 1.80 (16)	0.30 ± 0.16	1:49
Mngazana	7 062 ± 4 002	16 567 ± 8 750	16 937 ± 8 342	1.9 ± 0.2 (216) [A]	1.71 ± 0.42	2:1
Xhora	10 666 ± 4 222	6 888 ± 2 053	3 377 ± 1 719	2.00 ± 0.45 (100)	17.95 ± 4.52	1:3
Nqabarha/Nqabarana	10 756 ± 5 422	3 244 ± 1 203	5 644 ± 519	1.90 ± 0.31 (131)	0.57 ± 0.17	1:2

*Individuals only and represents the whole population = total N; [A] = Rajkaran (2011); Low density = < 999; Medium density = 1 000-10 000; High density = >10 000 individuals ha^{-1}.

Table 5.12: The number of seedlings, sapling and adult trees of ***Rhizophora mucronata*** reported per ha or as number of individuals in each estuary.

Estuaries	Seedlings ha^{-1}	Saplings ha^{-1}	Adults ha^{-1}	mean DBH ± SE (N)	Basal area (m^2 ha^{-1}) (SE) (N)	adult : seedling
LOW DENSITY						
Mzimvubu	0	0	4	4.03 ± 2.01 (4)	-	no seedlings
Mthatha	44 ± 44	0	622 ± 437	8.47 ± 1.96 (14)	0.44 ± 0.26	14:1
Mtakatye	0	0	5	4.74 ± 1.77 (5)	-	no seedlings
HIGH DENSITY						
Mngazana	812 ± 589	16 812 ± 7653	15 687 ± 9 184	2.4 ± 0.1 (218) [A]	1.62 ± 0.35 (56)	19:1

Table 5.13: **Individuals** of mangrove seedlings, saplings and adults trees. Av = *Avicennia marina*; Bg = *Bruguiera gymnorrhiza* and Rm = *Rhizophora mucronata* (± = SE). Where there were no SE bars the entry / numbers represent the whole population.

Estuaries		Species	Seedlings ha^{-1}	Saplings ha^{-1}	Adults ha^{-1}	mean DBH ± SE (N)	Basal area (m^2 ha^{-1})
Mtamvuna	Bg	21*	55*	24*	2.7 ± 0.5 (24) [A]	x	adult : seedling
Mntafufu	Rm	0	0	3	-	x	1:3
Mzimvubu	Bg	42 ± 5	18 ± 9.7	4 ± 3	3.48 ± 0.90	0.59*	no seedlings
Mdumbi	Bg	0	2	0	-	x	1:10
Bulungula	Av	3.14*	12*	7.14*	1.44 ± 2.01	x	no seedlings

*Individuals only and represents the whole population = total N; [A] = Rajkaran (2011); Low density = < 999; Medium density = 1000 -10 000; High density = >10 000.ha^{-1}. individuals.

These anthropogenic impacts were evident in all estuaries studied, regardless of whether or not they were in protected areas. It was found that the Mzimvubu, Mtakatye, Mntafufu, Mthatha, Mngazana, Mdumbi, Bulungula, and Xhora estuaries had the highest number of pressures with scores greater than 15. The Mthatha and Magazana estuaries had the highest wood harvesting intensities of live trees and branches, while all of the dead trees in Bulungula Estuary had been harvested. Cattle and goat browsing was evident in most estuaries, with Mdumbi and Mthatha estuaries impacted the most (Plate 5.1). Alien invasive plants were transforming the banks of the Mzimvubu Estuary and the upper reaches of Xhora Estuary (Plate 4 F). Traynor and Hill (2008) conducted a study and found that local people at Mngazana Estuary used mainly *R. mucronata and B. gymnorrhiza* for building houses and fences, while *A. marina* was used for firewood.

5.1.2 Mangrove distribution, area cover and population structure

Estuaries with low average water channel salinity included Mbashe Estuary where salinity ranged from 0.2 PSU in the middle reaches (MR) to 6 PSU at the lower reaches (LR) close to the estuary mouth. Mzimvubu Estuary was also freshwater dominant with low average salinity (8.6 PSU) for most of the estuary and only at the mouth the salinity increased to 28.8 PSU. Estuaries that had salinity concentrations close to that of sea water were Xhora Estuary; Nqabarha Estuary and Mngazana Estuary. All other estuaries (Appendix 19) had a gradient of salinity concentrations with brackish waters in the MR (17 to 23 PSU) and higher salinities close to seawater at the mouth (19 to 34.7 PSU). The high salinities explains the wide distribution and dominance of *Avicennia marina* in most of the systems, and in estuaries where there was a low salinity *B. gymnorrhiza* was more dominant, while when there was a salinity gradient from the head of the estuary to the mouth of the estuary, there was a greater richness in mangroves and salt marsh species.

In previous studies, Ward and Steinke (1982) found a total mangrove area of 272.2 ha in these 17 estuaries, which decreased to 270.6 ha in 1999 (Adams *et al.* 2004). Colloty *et al.* (2000) and Adams *et al.* (2004) reported that Mtentu, Mbashe and Mthatha estuaries had the greatest increase in area cover. However, this study (2012) found a loss of mangroves within these three estuaries, with a total area loss of 17 ha between 1982 and 1999 (Table 5.3). A net gain of 1.8 ha yr^{-1} was thus observed, which was an increase compared to the 1982 - 1999 net gain of 0.6 ha yr^{-1}. Over the past 30 years, a total loss of 31.5 ha of mangroves was observed. Since the total mangrove area of all estuaries was 240.6 ha a total of 10.5 ha (11 %) per decade was lost (Table 5.3). In estuaries where mangroves had been lost previously (1999), species had since then re-established. *Avicennia marina* and *B. gymnorrhiza* have recolonized the Mnyameni and Mzimvubu estuaries, while *A. marina* was planted in Bulungula Estuary, *B. gymnorrhiza* in Mdumbi Estuary and *Rhizophora mucronata* in Xhora Estuary, while at Mtentu Estuary no *A. marina* plants were observed.

Key findings in this study were that the greatest losses of mangrove area cover had been due to anthropogenic pressures in the larger forests such as Mngazana and Mthatha estuaries while natural changes had resulted in losses in the smaller mangrove forests such as Kobonqaba and Mbashe estuaries. The natural changes had influenced the mangrove forest structure and distribution. Where sedimentation was evident it promoted mangrove expansion due to new available habitats observed in estuaries such as Mdumbi, Xhora and Nxaxo/Ngqusi estuaries, where intertidal islands had been colonized by mangroves and salt marsh species. Here these stands had high seedling densities since livestock had been excluded from these islands. Stokes *et al.* (2010) mentioned that mangroves expand onto new available habitats and contribute additionally to the sedimentation process by trapping sediment in their roots. However excessive sedimentation in mangrove stands may result in higher elevation and dieback. This is because the sediment accretion causes smothering of the pneumatophores. This excessive sedimentation at the mouth was observed at Mbashe, Mzamba and Nqabarha estuaries. Mangroves are able to adapt to their environment especially species such as *A. marina*, which has a wide tolerance range and can establish fast. It is known to be a pioneer species among the mangrove genera (Steinke 1999). Proisy *et al.* (2009) also mentioned that mangroves such as *Avicennia* species are opportunists and will colonize new available banks and therefore could rapidly increase mangrove area cover. This study showed that *A. marina* opportunistically colonized the Eastern Cape estuaries of South Africa.

Other natural pressures such as droughts and intense sea storms are responsible for restricting water flow at estuarine mouths and for mouth closure. When the estuary mouth is closed off to the sea by a high sand bank or berm, the water level in the estuary will rise. This is known as back-flooding and causes inundation of the mangrove areas in the intertidal area. According to Begg (1978) and Naidoo (1985), mangroves do not tolerate long-term inundation and would die after more than five months of continuous inundation. This occurs due to long term inundation of the pneumatophores, causing the plants to die (Breen and Hill 1969). For example, the Bulungula Estuary mouth closed in the past due to an extended drought prior to 1999. Complete loss of the mangrove forests was reported (Adams *et al.* 2004). Prior to this all three species of mangroves were observed in this estuary in 1982

(Ward and Steinke 1982). Similarly, more than 95 % of mangroves were lost at Kobonqaba Estuary due to estuary mouth closure in 2008. The past mangrove area cover was 6 ha in 1982 (Ward and Steinke 1982) and 3.5 ha in 1999 (Adams *et al.* 2004). Colloty *et al.* (2000) reported that other impacts causing a reduction in mangrove area cover between 1982 and 1999 in this estuary were cattle browsing and wood harvesting.

Adams *et al.* (2004) also reported that Mzimvubu and Mnyameni had lost all mangroves due to massive floods scouring banks and destroying the mangrove habitats. Such flooding events could have been intensified due to poor catchment practices. Since then, mangroves naturally re-colonized these estuaries, and at Mnyameni Estuary, dense and healthy stands were observed. However, at Mzimvubu and Bulungula estuaries, only a few individuals were recorded. At Bulungula Estuary, after 1997, mangroves were planted by the staff of the Bulungula Lodge and the local community members. Around 10 000 *A. marina* seedlings were planted around the dead tree stumps in hopes of regenerating the mangrove forests. However, this rehabilitation effort was mainly unsuccessful, as less than 60 individuals survived (2012) and are under intense pressure from cattle browsing and bank erosion (Plate 5.2 C and 5.4 B). An example of natural regeneration was observed at Mnyameni Estuary, where mangroves had previously been lost. This forest has shown a 100 % recovery over the last 13 years and a 40 % increase in area cover from 1982. The population showed exponential growth for both *A. marina* and *B. gymnorrhiza*, and dense stands of all size classes were recorded in 2012. Harvesting pressures, however, impacted the density of the *A. marina* trees. Maximum tree heights of 5 m (DBH 6.6 ± 1.7) were observed for *A. marina* and 4.5 m (DBH 1.7 ± 0.5) for *B. gymnorrhiza*. *Bruguiera gymnorrhiza* had an adult to seedling ratio of 1:2 and *A. marina* had a ratio of 1:49. A forest with a 1:1 ratio is regarded as stable, while a 2:1 ratio indicates a degrading forest, and therefore seedling numbers can be used to determine the natural regeneration of forests (Ashton and Macintosh 2002; Rajkaran *et al.* 2009).

Colloty *et al.* (2000) found that *A. marina* was the dominant species in 1999; this was similar in this study with *A. marina* being dominant in terms of area cover in the estuaries where this species was found. Increased wave action in the intertidal area may have negatively affected seedling recruitment (Swales *et al.* 2007). *Bruguiera gymnorrhiza* was the most widely distributed species in all estuaries, and the only species in the small estuaries such as the Mtamvuna, Mtentu, Mzamba, and Mzintlava estuaries. Only two estuaries did not have this species, namely Nqabarha/Nqabarana and Bulungula, and only a very few individuals were found at Mdumbi and Kobonquaba estuaries. Plate 5.3 A shows *B. gymnorrhiza* (indicated by white arrows) growing among *Hibiscus tiliaceus*, which eventually out-competes the mangroves for space. Other freshwater species, seen in estuaries such as Mzamba Estuary, as well as invasive alien plants such as those seen at Mtamvuna Estuary, also prevail over *B. gymnorrhiza*. This contributes to the loss of biodiversity and reduces ecosystem functioning in these systems (Braatz *et al.* 2007; Koch *et al.* 2009).

In this study, forests were classified as degrading because of their low adult to seedling ratios, or no seedlings and saplings were observed but only old, large adult *A. marina* trees. However no relationships have been found between seedlings and high harvesting intensity. No seedlings or saplings were observed in 35 % of the studied estuaries, which were Mntafufu, Mtakatye, Mthatha and Mdumbi estuaries. It was suggested that the lack of established seedlings and saplings of mangroves in these estuaries could be due to predation by herbivores on fallen propagules, as most trees had mature propagules, but livestock, particularly goats, have been frequently noted to feed on the propagules in the intertidal zone (Plate 5.1 E). Crabs were also present, but not abundant, in most estuaries and could also have been responsible for the lack of seedlings or saplings. Studies by Chan (1996) and Dahdouh-Guebas *et al.* (1997 and 1998) showed that the failure of seedling establishment was due to many of the propagules and seedling being preyed upon by crabs, insects, livestock and game. A prominent browse line was evident on these old *A. marina* trees in estuaries such as Mtakatye, Mthatha and Mdumbi (Plate 2 A), where all foliage below 2 m has been removed, and the foliage above this height has remained intact. This was also seen in many estuaries in the study carried out by Colloty *et al.* (2000). Most of these *A. marina* trees have had some branch harvesting (Plate 5.2 A) and are thus under intense anthropogenic pressure.

Colloty (2000) mentioned that mangroves at Mbashe, Xhora, Mtakatye, Mthatha, and Nxaxo estuaries were considered to be fringing mangroves that also colonize intertidal islands. In this study, these estuaries are still dominated by fringing mangroves, and at Nxaxo the islands (Appendix 14) and additional created mudflats were colonized by *A. marina*. This was also seen at Mdumbi Estuary, where a few *A. marina* seedlings and saplings established on the island (Appendix 8), out of reach of cattle. In most estuaries including protected estuaries such as Mtentu and Mnyameni, tree harvesting was observed. Colloty *et al.* (2000) found that in estuaries such as Mtentu and Xhora, there was a high abundance of seedlings and no harvesting occurred. However, in this study the opposite was found.

In other estuaries, for example Mzamba Estuary, sediment accretion was possibly responsible for higher elevation and low seedling establishment due to higher elevation, where plants receive less frequent tidal flushing. Aerial roots get covered by sediment which also tends to result in stunting (< 2.5 m) of adult *B. gymnorrhiza* trees, with no seedlings or saplings at these elevated areas. This is also often seen in fringing banks in other estuaries such as Xhora and Mthatha, where only old *A. marina* trees remain. Terrados *et al.* (1997) found that sediment accretion increased seedling mortality and reduced growth in mangrove seedlings.

In estuaries such as in Nxaxo/Ngqusi, Mnyameni and Mntafufu, where high numbers of seedlings and saplings of both *A. marina* and *B. gymnorrhiza* were present, it is suggested that the forests will expand over available mud banks or islands and thus increase in area cover, depending on whether or not conditions prevail. A small population (< 20 individuals) of *R. mucronata* of all size classes had established at Nxaxo Estuary. Colloty *et al.* (2000) reported these to be only small individuals, whereas there are now adult trees of 6 m in height approximately.

In Mtentu and Mtamvuna estuaries where only *B. gymnorrhiza* was observed, the forests are limited by available habitat, since steep cliffs and adjacent sand banks restrict expansion and landwards migration. These areas are thus particularly vulnerable to sea-level rise as mangrove areas become flooded (Gilman *et al.* 2008). At Mtentu Estuary, adult trees greatly exceeded seedling numbers, and very tall trees (> 8 m) have been recorded here, suggesting that it was a degrading forest due to the low adult to seedling ratio (Plate 5.3 C). Therefore, the competition for available space between species such as *Hibiscus tiliaceus* and *Bruguiera gymnorrhiza*, coupled with sea-level rise, would be intense (Abel *et al.* 2011). The highest tree densities (> 10 000 individuals ha^{-1}) for *B. gymnorrhiza* were found in Mzintlava Estuary (14 933 ± 1 922), Mzamba Estuary (12 933 ± 581) and Mnyameni Estuary (12 400 ± 4 183) (Table 5.7). For *A. marina*, the highest tree densities recorded were at Mngazana Estuary (16 937 ± 8 342), Mbashe Estuary (9 466 ± 2 764) and Nqabarha/Nqabarana estuaries (5 644 ± 519) (Table 5.8).

Mngazana Estuary also had the highest *R. mucronata* tree density (15 687 ± 9 184) (Table 5.9 and Plate 5.3 D). This study recorded much higher tree densities compared to research done in 1999 by Colloty *et al.* (2000). The heighest density was found at Mngazana Estuary (2 594 trees ha^{-1}) and Mntafufu Estuary (1402 trees ha^{-1}), while Nqabarha Estuary had low density of 74 trees ha^{-1} and Mtakatye Estuary 162 trees ha^{-1} and all other estuaries had low tree densities between 10 to 296 trees ha^{-1}. One reason for this difference may be because Colloty *et al.* (2000) used the ‘line transect method’ while in this study the ‘plot method’ was used.

In Figure 5.5, the mangrove distribution and the density of each species in each estuary was summarized. Most estuaries had low (< 999 individuals ha^{-1}) to medium (1 000 to 10 000 individuals ha^{-1}) mangrove density. However, Rajkaran *et al.* (2004) found that the harvesting at some sites in Mngazana Estuary resulted in a change in the adult to seedling ratios and height classes, and therefore influenced the population structure in the harvested sites.

For this research it was found that intertidal salt marsh cover was high in estuaries with large mangrove stands. Surprisingly intertidal salt marsh area cover in Mngazana Estuary was low (1.25 ha) even though this estuary had the largest mangrove forest (188 ha) among the former Transkei estuaries. This may have been due to the bare and unvegetated areas in the intertidal areas and Rajkaran (2011) suggested that these are the result of footpaths by people and animals through the habitat. Estuaries that had high mangrove and intertidal salt marsh area cover were Mthatha Estuary (mangroves = 31 ha; salt marsh = 27.3 ha) and Mntafufu Estuary (mangroves = 12 ha; salt marsh = 14 ha), while Mnyameni Estuary has a mangrove area cover of 5 ha and a salt marsh area cover of 7.4 ha. All other estuaries had less than 5 ha of salt marsh area cover. It is expected with sea-level rise that the mangroves could expand into the salt marsh habitats especially at estuaries with large floodplain such as Mthatha, Mntafufu, Mtakatye and Mngazana estuaries.

In this study it was found that the most frequent impacts were trampling (82 %) and livestock browsing (76 %), which may have influenced seedling establishment (Figure 5.6). Browsing of *Avicennia marina*, which seemed to be the preferred species by goats and cattle, corresponded with seedling absence observed in estuaries such as Mdumbi and Mthatha estuaries and the low adult to seedling density in Mntafufu, Mtakatye and Nxaxo/Ngqusi estuaries. Mthatha, Mdumbi and Nxaxo/Ngqusi estuaries all had high HI scores of 5 for browsing and trampling, with Mtakatye and Mtafufu estuaries having a HI score of 3 and 2 respectively.

Wood harvesting was the third most frequent impact occurring in 70% of the estuaries sampled (Table 5.4). However in terms of severity of the pressure harvesting had the highest biotic pressure score (51) for all estuaries sampled. This was followed by browsing and trampling (score of 44) and footpaths (score of 42) (Table 5.6). Table 5.8 summarized the HI scores were Mthatha Estuary (33 = HI); Mdumbi Estuary (28 = HI), Mzimvubu and Xhora (24 = HI) and Mngazana Estuary (16 = HI) had the highest scores attributed to anthropogenic impacts, while Bulungula Estuary (28 = HI) and Kobonqaba Estuary (8 = HI) had lost most of their mangrove area cover due to natural change impacts. Thus it was evident that the smaller forests had been more impacted by infrequent natural changes such as floods and droughts, while the larger forests had been greatest impacted by long-term anthropogenic impacts.

As it was expected in Hypothesis (1) Mangrove cover showed an overall decrease since 1999 due to an increase in populated areas. This hypothesis can be accepted. However, Hypothesis (2), which stated "healthy mangrove forests occur in protected areas, i.e. adult to seedling ratios indicate new growth"; can be rejected because anthropogenic influences were evident in all estuaries, regardless of whether they were close to settlements or if they were in protected areas. Hypothesis (3) stated that the largest mangrove areas experienced the greatest impacts (Table 5.8), with harvesting being the dominant impact in most estuaries. It can be deduced from this study that the estuaries with the highest mangrove area cover were also most affected by human impacts, while smaller estuaries with small forests (< 5 ha) were mostly affected by natural change impacts.

Hypothesis 4 suggested that habitat availability promoted mangrove growth and expansion, and thus the formation of mud banks and intertidal areas allowed mangrove spreading. This statement can be accepted as true, since in estuaries such as Mnyameni, Nxaxo, Mdumbi and Nqabarhana, mangroves increased in area cover due to available mud banks, while in Mtentu and Mtamvuna estuaries mangrove expansion was restricted by high cliffs (Appendix 3).

In South Africa and gobally, anthropogenic impacts on mangrove forests are of great concern, and overutilization and development coupled with natural changes such as sea-level rise (Gilman *et al.* 2008), could result in decreased resilience and adaptations of these forests to these pressures. Forests could thus become lost in more areas in the future (Barbier *et al.* 2011). Mangrove degradation is increasing at an alarming rate due to an increase of human activities (Valiela *et al.* 2001) which threatens many important ecosystem services that are dependent on healthy mangrove forests. Barbier *et al.* (2011) stated that the "widespread and rapid transformation of estuarine and coastal ecosystems" would have an impact on the benefits and values of estuarine services. The dependence on these natural resources is driving subsistence economies in many developing countries where a large portion of the population consists of subsistence farmers (Ewel *et al.* 1998; Kairo *et al.* 2002; Rajkaran *et al.* 2004). In South Africa these resources support the rural coastal communities and thus the estuary health and productivity would be essential for these environmental services that the local people depend on. Estuarine resilience will be greater in healthy estuaries (Van Niekerk and Turpie 2012) and thus stakeholders need to strive towards increasing ecosystem health through better management practices in South Africa.

Natural change had profound impacts on the smaller forests to the extent of removing mangroves from these small estuaries, thus adding to the mangrove area cover loss. However if left undisturbed by human activities and if natural disasters such as floods and mouth closures are infrequent these mangroves will re-generate and thus the gain in mangrove cover. In contrast in the larger forests which are found to be in the larger estuaries, which are also more stable, anthropogenic pressures have had the greatest impacts. The frequency of pressures and their intensity (HI scores) have been the determining factor in shaping these forests. The harvesting, browsing and trampling had determined present population structure and especially seedling establishment resulting in mangrove area cover loss, while environmental conditions such as salinity and temperature determined the distribution of the different mangrove species along the former Transkei coast. Climate change will increase natural disasters along the coasts and thus mangroves will have to be able to adapt to more frequent natural changes. For this reason the anthropogenic activities have to be reduced in order for these forests to be more resilient in case of such natural events.

5.1 Conclusion

It can be concluded that the rainfall had been declining in the past twelve years and this region of southern African region was experiencing a drought cycle which may have influenced mangrove growth and sediment characteristics. However human activities have a profound impact on mangrove forests and that most permanently open estuaries in the former Transkei show mangrove degradation. Movements of people, livestock and harvesting had the highest impacts. However freshwater flow requirements need to be investigated and monitored for each estuary since freshwater flow would determine the mouth dynamics and open mouth conditions are needed for mangrove growth and survival.

Therefore, more sustainable practices and conservation plans are urgently needed to protect South African biodiversity and the ecosystem services that these forests provide. These forests also need good management. Recommendations would be to set up a continuous monitoring and management program for mangrove-containing estuaries, where anthropogenic activities, freshwater requirements and open estuarine mouth conditions are emphasised, in order to reduce the impacts that result in mangrove degradation.

Chapter 6: General discussion, conclusions and management recommendations for mangroves in South Africa

6.1 Synthesis of the research

The aim of this study was to identify possible stressors from natural change (for example floods and droughts) as well as anthropogenic pressures (for example harvesting and livestock browsing) on South African mangroves and investigate the impacts that these pressures had on the degradation of these forests. Thus the general key question was **'Have anthropogenic pressures had more profound impacts on mangrove ecosystems than the pressures caused by natural changes?'**

The study investigated an unusual situation at St. Lucia Estuary in the KwaZulu-Natal Province, where mangroves had survived under estuary mouth closure and low water levels (Chapter 2). The mouth closure was due to prolonged drought and reduced freshwater flow into the estuary. The drought is a natural change but this was coupled with anthropogenic impacts which reduced the water volume considerably due to a diversion of the Umfolozi river water away from St. Lucia Estuary and water abstraction for agricultural fields. This has resulted in the low water levels and closed mouth conditions. The site fringing the main estuary water channel had the highest density of mangrove seedlings and saplings that had established in the standing water (10 600 ± 5093 ha^{-1}) while the dry site had only low density of saplings (4200 ± 2347 ha^{-1}) and no seedlings. The flooded site near the Umfolozi River had significant higher tree density (23 800 ± 2560 ha^{-1}) compared to all the other sites. The study showed that mangrove tree height and density increased significantly with sediment moisture. However the overall adult to seedling ratio was 6:1 and thus the mangroves at St. Lucia Estuary are not regenerating. It can be deduced from this research that sediment characteristics such as low sediment moisture (< 30 %) are unfavourable for mangrove growth at sites now characterized by a lack of tidal flooding and a shift of vegetation types as well as loss of mangroves in some areas may occur if conditions prevail.

This information from Chapter 2 has provided new insights on mangrove survival under non tidal, low water level conditions due to prolonged estuary mouth closure and confirms that the anthropogenic activities in preventing the Umfolozi River from flowing into the St. Lucia Estuary had a negative effect on the mangrove ecosystems, where degradation of these habitats was evident. It is usually expected that during prolonged estuary mouth closure, estuary water level rises and mangrove forest would get flooded and as a result die back of these forests would occur within just a few months due to continued inundation. Examples of such events were presented in studies by Breen and Hill (1969) and Adams *et al.* (2004). Thus St. Lucia Estuary provided a unique opportunity to study these different mangrove habitats and the findings of this chapter provide the initial measurements for a long term monitoring program to assess the influence of mouth closure on recruitment and survival of the mangrove forest at this estuary. If the estuary mouth opens and the Umfolozi River is reconnected to the St. Lucia Estuary then this provides more opportunities to determine the state of the mangroves as they may regenerate in response to the tidal conditions.

At Nahoon Estuary, the planted mangrove forest appears to be thriving (*Avicennia marina*) (Chapter 3) and this study tested the hypothesis that mangroves had expanded and replaced salt marsh habitats over a 33 year period. *Avicennia. marina* was the dominant mangrove in this estuary and had a high seedling density (33 822 ± 16 364 ha^{-1}) which showed a seaward expansion from the creek (Zone 3) which was suspected to be the centre of expansion. Here taller and older trees were recorded and had a mean height of 5.8 ± 0.25 m and a DBH (diameter at breast height) of 10.6 ± 1.4 cm, while at the water's edge (Zone 1) the trees were slightly shorter with a mean height of 4.1 ± 0.02 m and a DBH of 8.3 ± 1.3 cm. The findings for mangrove area cover were that the mangrove forest increased linearly at a rate of 0.06 ha yr^{-1}, while salt marsh had a slightly higher increase of 0.09 ha yr^{-1} but was found to be variable over the 33 years. The results in Chapter 3 are the first studies documented on planted mangrove forests in South Africa, out of its natural distribution and mangroves have been growing exponentially at this site. However in this study, no evidence was found that this mangrove expansion was to the detriment of the salt marsh habitat. Thus the initial hypothesis has to be rejected but at present the mangrove forest is still small (< 2 ha) and continuous monitoring would determine if there is a change in the relationship between mangroves and salt marsh habitats. These mangroves fall in a protected area and thus are useful in determining the possible scenarios of sea-level rise and flood events without the additional anthropogenic impacts such as harvesting or cattle browsing. The only human disturbances that had been observed were bait collection and litter pollution. However these do not seem to have affected mangrove growth. The findings of this study can be used as an example for future predictions of possible shifts in mangrove distribution due to climate change, where mangroves are expected to migrate into higher latitudes out of their present natural distribution (Stokes *et al.* 2010).

At Nxaxo Estuary the response of mangroves (*A. marina*) to long-term cattle browsing and trampling was investigated (Chapter 4) and the hypothesis tested was that (1) 'browsed mangroves will have reduced growth; browsed trees would show horizontal but not vertical growth' and (2) 'increased trampling due to cattle movement in the mangrove forests would result in compact sediments and reduced seedling establishment. This study showed that cattle browsing had a significant effect on the growth of the dwarfed *A. marina* forest, where there was a horizontal and vertical expansion of growth in the mangrove plants in the cattle exclusion plots but only a horizontal expansion in the control browsed sites. The findings showed that plants in the cattle exclusion plots had significantly higher plant height (5.41 ± 0.53 cm), crown volume (0.54 ± 0.01 m^3) and crown diameter (7.09 ± 0.6 cm) compared to the control plots, where plant height decreased significantly (-0.07 ± 0.67 cm) and there was only a small increase in crown volume (0.14 ± 0.1 m^3) and crown diameter (2.03 ± 2.61 cm^{-1}) from 2010 to 2012. The research showed that browsing by cattle on mangroves stunts growth and causes a shrubby appearance as coppicing, while the unbrowsed plants in the cattle exclusion plots had vertical growth of new shoots.

It was also found that the cattle exclusion plots had significant higher percentages of flowering (67 %) and fruiting (39 %), while the control plots had significant lower flowering (34 %) and fruiting plants (5.4 %). The observations in the field also indicated that cattle had trampled a number of seedlings and the findings were that cattle exclusion plots had a higher percentage (63 %) of established seedlings, while a lower percentage (37 %) of seedlings was recorded in the control plots. Thus trampling influenced mangrove seedling establishment but the sediment was not as compacted as expected. Empirical evidence was provided that showed that domesticated livestock browsing and trampling changes the morphological structure of mangroves and reduces growth and seedling survival. However the additional stress from high surface sediment salinity (> 30 PSU) and reduced sediment moisture (< 30 %) in and before 2010 in addition to cattle browsing may have contributed to the dwarfing of the mangroves. Salinity stress and dwarfing has been reported on by Naidoo (2006). The increased rainfall in 2011 and 2012 was beneficial to mangrove growth.

The research presented in Chapter 4 has contributed new knowledge by providing the first documented study on the effects of continuous long-term livestock browsing on mangroves. This is an anthropogenic pressure which had a more profound effect on the mangrove forests than natural changes. The findings showed that mangroves were sensitive to continuous long-term browsing and trampling. This study showed empirical evidence that livestock can significantly impact mangrove ecosystems and contributes to the knowledge on the state, distribution and reproductive capability of these forests in the Eastern Cape Province. Globally, the effects of browsing on mangroves, particularly *Avicennia marina*, have been well documented by authors such as Khalil, (1999); Sommerlatte and Umar, (2000); Qureshi, (2005); Dahdouh-Guebas *et al.* (2006a; 2006b); Shah *et al.* (2007); Shah and Kamaruzaman, (2007) and Saifullah *et al.* (2007). However, these studies have mostly focussed on social – environmental conflicts and benefits of natural resources and not much literature is available to underpin the response of mangroves to continuous, long term browsing. There are some examples where the forests have been progressively degraded due to heavy browsing by camels and cattle such as studies presented by Shah *et al.* (2007) who reported that in Pakistan, camels and cattle are responsible for removing about 67 000 tons of foliage per annum from the *A. marina* forests, and suggested that utilization of this natural resource was not sustainable, as consumption by the animals was greater than foliage production. Mangrove forests in Pakistan have a high economic and social importance, as much of the land is arid and the mangrove trees provide much needed fodder in times of drought and during the dry season. Similarly Dahdouh-Guebas *et al.* (2006) reported that feral water buffalo in the East-Godavari Delta in India were responsible for mangrove degradation and that intensive browsing by water buffalo on mangroves had both environmental and social impacts, as local villagers and conservation officials were in conflict with each other.

Chapter 5 investigated the impacts of natural changes (for example estuary mouth dynamics, storms) and anthropogenic factors (for example wood harvesting and browsing) on the population structure and distribution of three different mangrove species namely *Avicennia marina*, *Bruguiera gymnorrhiza* and *Rhizophora mucronata.* Results from this study have been compared to previous studies conducted by Adams *et al.* (2004). Similar anthropogenic pressures were found to have the greatest impacts on these forests, where wood harvesting, livestock browsing and trampling, as well as foot paths were observed in most estuaries (> 70 %). The findings in this chapter indicated that the overall state of mangroves was deteriorating in most of the 17 estuaries investigated, where 50 % of the estuaries showed a decrease in area cover. The largest mangrove forests such as Mngazana Estuary contributed to 11 % (27 ha) and Mthatha Estuary contributed 4.5 % (11 ha) to the overall losses of mangrove cover. The rate of mangrove area cover loss is concerning as 11 % of mangrove cover was lost for every decade with a total loss of 31.5 ha from 1999 and this was mostly due to anthropogenic activities. Natural change such as mouth closure was observed in Kobonqaba Estuary, which caused a loss of 3.5 ha of mangroves. Additionally the adult to seedling ratio had been low (70 %) for all mangrove species in the Transkei estuaries.

Thus it was found in this study that continuous long-term anthropogenic pressures had profound effects on the mangrove forests in the study area. However the devastating impacts of natural changes such as mouth closures or flood events such as observed at

Kobonqaba Estuary in this study and Myameni as well as Bulungula estuaries in previous studies (Adams *et al.* 2004) showed that sudden catastrophic events can destroy large mangrove areas in a short time. The recovery of mangroves at Mnyameni Estuary provided empirical evidence that even after such devastation these forests are able to regenerate naturally if anthropogenic activities are kept to a minimum. This forest showed 100 % (5 ha) recovery since 1999, when all mangroves had been removed by a large flood. Since then, mangroves have re-established into dense, healthy stands. Such catastrophic environmental events are expected to increase with climate change (Nicholls *et al.* 2007).

In this study it was also found that wood harvesting had contributed most to mangrove degradation, the second largest pressure was livestock browsing and thirdly trampling and /or foot paths which resulted in open forest gaps, and poor agricultural practices, which increased the sediment load and elevation in mangrove forests resulting in forest degradation. Seedling establishment and recruitment are fundamentally important to forest regeneration (Ashton and Macintosh 2002; Rajkaran 2011) and it was evident that predation of propagules and seedlings by goats and cattle have influenced seedling establishment which may be one of the determining factors for mangrove forest degradation in the study area.

The major finding in this chapter was that the largest mangrove forests such as at Mngazana and Mthatha estuaries had been impacted the most by human activities and livestock browsing. Mngazana Estuary has lost 27 ha (19 %) of mangrove area cover, which is significant since Mngazana Estuary has 49 % of the total mangrove area cover in the Eastern Cape Province. Mthatha Estuary has lost 11 ha (26 %) from 1999 to 2012, while smaller forests have been regenerating as observed at Mnyameni Estuary (100 %) and Mdumbi Estuary (90 %) both of which had below 5 ha mangrove area cover. The overall findings highlighted the present pressures, impacts and the state of the mangrove ecosystems in South Africa with emphasis on the Eastern Cape estuaries. The research in Chapters 4 and 5 confirmed that mangrove ecosystems are sensitive to continuous long-term anthropogenic pressures and if coupled with natural change will accelerate mangrove loss in South Africa. However findings in Chapter 3 show that mangroves can thrive if allowed to re-generate as seen in Myameni Estuary where an adult to seedling ratio of 1:2 was found. Where this is no or low anthropogenic disturbances, mangrove forests are more resilient and are able to regenerate. However when anthropogenic pressures are coupled with natural change pressures there is a threshold limit and resilience to these pressures will ultimately determine if mangrove loss will be irreversible (Walker and Salt 2006).

6.2. The DPSIR framework and climate change versus human activities

6.1.1 The drivers, pressures, state of mangrove forest and the social- and environment impacts:

The **DPSIR (Drivers-Pressures-State-Impacts-Responses) framework** has been used widely in ecological studies including some mangrove-related research such as studies presented by Lin *et al.* (2007); Slingengerg *et al.* (2009); Campuzano *et al.* (2011) Rajkaran, (2011) and Wu and Wang, (2011). This conceptual framework has been developed as a tool for describing the links or "relationships" between the environment and the stakeholders (UNEP 2006). It provides an insight into the different links: where the **drivers** are usually the communities or a certain development that put the natural resources under immense **pressure**. These pressures will leave the environment in a certain **state** and result in environmental **impacts**. The impacts then result in a **response** from the society or community, which is often in form of legislation and policies (Maxim *et al.* 2009; Rajkaran 2011).

In this study the major **drivers** include the increased demand for estuary resources by the local communities, catchment activities such as agriculture, developments and climate change. Climate change results in increased sea surface temperatures, sea-level rise and increase intense events such as prolonged droughts, extreme floods and increased sea storms, which influences estuary mouth dynamics (Boesch, 2002; Nicholls *et al.* 2007). The predicted sea-level rise (SLR) by the IPCC, (2007) is between 1.8 to 8 mm yr^{-1} over the next century. With SLR, the incoming tides are expected to push up further into the upper reaches of the estuary and up into the upper intertidal regions (Swales *et al.* 2007). This will cause an increase in salinity and inundation as well as an increase in tidal range (Nicholls *et al.* 2007) and mangroves are expected to have a landward migration into higher elevation and towards the upper reaches (Doyle *et al.* 2003; Swales *et al.* 2007). Migration would only be possible provided there is a gentle slope and available habitat beyond the intertidal region. Therefore in areas which are undeveloped and more or less pristine, mangrove area cover is expected to increase with SLR as trees colonize new habitats that previously hadn't been flooded. However the increase in sea water would have a negative impact on the freshwater marsh area (Doyle *et al.* 2003) and possibly cause a shift in species composition of both salt marsh and mangroves. More salt-tolerant species such as *Avicennia marina* may out-compete the low salinity mangrove species such as *Bruguiera gymnorrhiza*. Thus mangroves are able to adapt to SLR and are able to cope with a steady rate of increase. Salt marsh areas may be replaced by mangroves in future with increased inundation; however this was inconclusive in this study for the Nahoon Estuary study site.

Other drivers are human activities that are responsible for conversion of these habitats, for example, human developments such as infrastructures behind or in present mangrove forest may prevent the landward migration of mangroves into higher elevation areas (Kennedy *et al.* 2002). Thus mangrove forest will be restricted to the lower intertidal areas with higher salinity and with longer inundation or even permanent inundation, which again will result in mangrove ecosystem collapse and ultimately habitat loss (Nicholls and Lowe 2004). Other examples of drivers are the poor agricultural practices in the catchment and on the river banks, which has been seen in many of the Transkei estuaries. These have resulted in an increase of sediment load such as observed in Mzamba Estuary (Plate 5.3. B). This caused sediment accretion in mangrove forests thereby covering the roots and increasing elevation. These areas have become non-tidal except at high spring tides. This has caused a lack of mangrove seedlings and low density stands with only old trees in those areas, similarly as seen in St. Lucia's dry sites. With SLR these higher elevations will become tidal again and may become recolonized by mangroves (Anthony 2004; Stokes *et al.* 2010). Therefore sediment accretion coupled with SLR could be beneficial to the mangroves and could counteract the effects of SLR (Swales *et al.* 2007; McKee *et al.* 2007), since these forests are able to trap a large amount of sediment (Boesch 2002; Stokes *et al.* 2010).

In contrast extreme events such as sea storms and droughts as well as floods are expected to increase (Nicholls and Lowe 2004). These will be to the detriment of these forests, where increased freshwater runoff, tidal prisms and storm surges cause the re-suspension or deposition of large amounts of sediment into and from mangrove forest resulting in excessive sediment accretion or erosion (Marion *et al.* 2009; Adame *et al.* 2010b). The sediment load may also result in mouth closure, which has proven to be detrimental to mangrove forests in some estuaries such as seen at Kobonqaba Estuary, where droughts and reduced freshwater runoff caused mouth closure. This resulted in back-flooding and continuous inundation of the mangrove roots and mass mortality of the trees in this estuary.

Global warming and temperature increase will also result in a steady transgression of mangroves into higher latitudes where they will colonize new habitats. Chapter 2 has shown that mangroves are able to cope well at higher latitudes and had exponential growth outside their natural distribution (Hogarth 2007; López-Hoffman *et al.* 2007; Krauss *et al.* 2008) but often this colonization is only by one species (Tomlinson 1994) such as *A. marina*.

Anthropogenic pressures have been evident in all estuaries and have contributed to mangrove degradation, with a loss in mangrove habitat of 11 % per decade. This is because most of the rural communities that live in close proximity to these forests are directly dependent on the resources and services that these mangroves provide especially in the rural Eastern Cape region. This leads to pressures on these resources, which include fire wood, building material and fodder for livestock as well as fishing, mollusc and bait collection. Thus the major findings in this study were that anthropogenic pressures had more profound effects compared to the natural change pressures. This was similar to studies presented by Clarke and Kerringan (2000) and Duke (2001), because the anthropogenic disturbances had been mostly **selective** (such as harvesting and domestic livestock browsing), more **widespread** (evident in most estuaries) and **continuous** over a long period of time (many decades), while natural change (sudden severe floods, mouth closure, competition with freshwater species) had been observed and reported in only a few estuaries; these however had been **random** and been much **less frequent** over shorter periods of time compared to the anthropogenic disturbances. Natural changes such as mouth closures and floods had been responsible for sudden incidental loss of mangrove areas, even to the extent of causing die-back of all mangroves in an estuary.

Coastal communities utilize mangrove ecosystems for food security (fishing, molluscs, prawn farming, bait collection) and for fuel (firewood and charcoal production) (Mohamed 2008), but also for timber harvesting and building materials, increasing the pressures on these resources. With the increasing human population the natural resources are becoming more important in a more threatening economy where there is a need for cheap domestic fuel such as firewood especially for impoverished communities. South Africa has a human population growth rate of 1.10 % per annum (2011) (DWA 2011). Even though less than half of the South African population (43 %) lives in rural areas (STATSSA 2001), there is much pressure on the forests resources, especially when considering that the total mangrove area cover in South Africa is 1631.7 ha, with the largest area cover in a few estuaries in the KwaZulu Natal Province (1391.1 ha) (Rajkaran 2011), and the rest was recorded in the Eastern Cape Province with 240.6 ha. This is a small area which only represents 0.05 % of Africa's mangrove area cover (FOA 2003). However mangrove ecosystems are protected in South Africa by Department Agriculture, Forestry and Fisheries (DAFF) through the Natural Forest Act, 1998 (Act No. 84 of 1998). Two of the three mangrove species are on the protected tree list; these are *Bruguiera gymnorrhiza* and *Rhizophora mucronata.* Therefore the harvesting of these species is regarded as illegal and a permit is required for the use of these species (Rajkaran 2011). The effectiveness of the legislation in protecting these trees has proven to be unfruitful if not 'policed and enforced'. This would be especially difficult in the impoverished rural communities that make a living on subsistence farming, who have no alternative fuel and because it is a ‘common’ resource it is often undervalued by the local community (Walters *et al.* 2008).

“Tragedy of the commons” applies in the area, where there is a lack of responsibility of individuals in a community, which compromises the resource by taking more than they should (Hardin 1968). This theory didn’t take into account the population increase, poverty pressure and the demand increase on the resources and thus the higher demand will result in the inevitable overutilization. Therefore the tragedy will result in ecosystem alterations and degradation as seen in the mangrove forests. Thus alternative solutions have to be found to alleviate poverty, reduce pressures and needs for mangrove wood, and protect the existing forest by also involving the local people that utilize them in management. The numerous mangrove studies presented in more tropical regions such as Kenya (Bosire *et al.* 2003; Kairo *et al.* 2008; Mohamed 2008), where mangroves growth is much more rapid (2.6 times faster) than in South Africa (Rajkaran *et al.* 2004; Rajkaran 2011) due to more favourable environmental conditions. In addition continuous and persistent anthropogenic disturbances will result in longer recovery time and reduced resilience (Duke 2001; Ellis and Bell 2004; Mohamed 2008). Anthropogenic pressures combined with environmental stresses result in these subtropical and warm temperate forests recovering more slowly than the tropical forests in the equatorial regions. Thus if no intervention is made, mangroves are expected to be lost in more estuaries and in extreme cases may result in local extinctions.

What is needed for forest regeneration? Mangroves are known to naturally recover from disturbance in two important ways (1) by propagules and seedling recruitment and (2) some species such as *A. marina* are able to coppice. The latter was seen in most estuaries where *A. marina* trees send out shoots from harvested branches but when harvesting was too intense, mortality of the overharvested trees was observed. *Rhizophora mucronata* and *B. gymnorrhiza* are species that do not recover from harvesting (except for a few multi stemmed *B. gymnorrhiza*) but usually die, because whole trees are harvested and they do not coppice (Osborne and Berjak 1997). The recruitment strategy of mangroves has the adaptive advantage of producing viviparous propagules. This adaptation allows them to be successful in these harsh habitats (Kathiresan and Bingham 2001). Mature propagules drop into the intertidal area where they may float and be transported for long distances before establishing into seedlings. This enables them to distribute and colonize different available habitats (Kathiresan and Bingham 2001). The forest is said to be regenerating if a typical J-curve exists (Klimas *et al.* 2007; Rajkaran 2011) and the adult to seedling ratio is greater or equal to 1:2. If there is a seedling density above 5 000 ha^{-1} in larger forests then it can be regarded as regenerating (Ashton and Macintosh 2002). Rajkaran *et al.* (2009) suggested that even with a 50 000 ha^{-1} seedling density it is not certain how many of these would survive into the next stage of saplings and then into adult trees. Seedlings will only establish, survive and grow if favourable environmental conditions are present such as temperature, salinity and, inundation. In this study the lack of tidal exchange at St. Lucia and Mzamba estuaries was thought to have reduced seedling recruitment. Different species also have different tolerance ranges to these environmental parameters (Chapter 1) which determine distribution (Naidoo 1985; Holguin *et al.* 2001). Predators such as crabs and insects (Chan 1996; Dahdouh-Guebas *et al.* 1997 and 1998; Steele *et al.* 1999; Lee 2008; Lindquist *et al.* 2009) and domestic livestock in particular goats and to a lesser extent cattle, feed on the propagules and seedlings and have negative impacts on seedling establishment, recruitment, population structure and thus forest regeneration (Lee 2008).

The species composition of a mangrove forest is important to determine its regenerative traits and the needed recovery times for successful regeneration (Mohamed 2008). Vegetative regenerative traits are mostly only through coppicing in mangroves and depending on the severity of the disturbance (Duke 2001) as well as the duration of the disturbances on the plants such as *A. marina* will determine its survival and regenerative growth (Hutchings and Saeger 1987; Snedaker *et al.* 1992; Ellis and Bell 2004). The disturbances, depending on its severity and duration results in the forests becoming more sparse or ‘thinned out’ and this causes the formation of forest gaps in the forest canopy (Duke 2001), where these gaps will result in changes in the sediment characteristic that again may reduce the seedling establishment. This was evident in many estuaries and especially in forests where *A. marina* was dominant. The size of these ‘gaps’ also determines the recovery duration, for example Ellis and Bell, (2004) mentioned that small gaps in the canopy, caused by low intensity harvesting or lightning strikes (Mohamed 2008), may recover in a few months by producing new branches, while large gaps, caused by intense harvesting and browsing, may take up to 10 years to close the canopy again and up to 25 years under constant human disturbance in more tropical regions, which may be longer in the more warm temperate forest such as in the study area. These larger gaps will not just regenerate by vegetative regeneration but require the recruitment strategy to close canopy gaps (Ellis and Bell 2004), which is also depended on the availability of mature propagules and the transportation thereof (Allen and Krauss 2005).

Anthropogenic pressures can be more severe and cause greater degradation of mangrove forests due to the long recovery time required for forest regeneration (Clarke and Kerrigan 2000; Imai *et al.* 2006; López-Hoffeman *et al.* 2007). Forest gaps have been observed in larger mangrove forests where harvesting and browsing was high, such as in Mngazana, Mthatha, Mtakatye and Xhora estuaries. Mngazana Estuary has the third largest mangrove forest in South Africa and the largest forest in the Eastern Cape Province (Rajkaran 2011). Forest gaps in this estuary have been attributed to harvesting and trampling where 27 ha (19%) of mangrove area was lost since 1999 and 32 ha (21 %) since 1982 (Rajkaran and Adams 2009). Rajkaran (2011) had suggested that harvesting in Mngazana Estuary should be reduced to 5 % of trees per ha^{-1} for *R. mucronata* and *Avicennia marina* should not be harvested at more than 10 % of the trees per ha^{-1} year^{-1}, while harvesting of *B. gymnorrhiza* should be prevented and these trees should be completely protected. In this study these forest gaps had also been the result of over utilization of harvesting and browsing seen in most of the former Transkei estuaries. Thus overutilization from harvesting should also be applied to all the other estuaries as *B. gymnorrhiza* are the target species for the harvesters in the estuaries where it is found and domestic livestock should be restricted in mangrove to allow for natural regeneration (Rajkaran and Adams 2012). Rajkaran (2011) suggested that there should be a 'buffer zone' of not less than 10 m for the finging mangroves and 25 m for the creeks.

6.1.2 The responses from stakeholders and management recommendations

The DPSIR framework can be used as a tool in informing managers and decision makers of the social and environmental concerns. To remedy the present state of the forests, interventions by all stakeholders have to be made by developing and implementing policies and legislation, taking all issues of the stakeholders into consideration (Figure 1.2).

Mangroves are protected and legislation has been put in place by the Department of Water Affairs and Forestry (DWAF), the Natural Forest Act, 1998 (Act no. 84 of 1998) (Lewis *et al.* 2002), and all harvesting of mangroves of *B. gymnorrhiza* and *R. mucronata* is illegal. *Avicennia marina,* found to be the dominant species in this study, was also targeted for wood harvesting and is being heavily over-utilized in most of the former Transkei estuaries. However *A. marina* does not receive the same protection as the other two species. In addition even with the protection act in place much of its implementation is not enforced and controlled and thus falls short of its expectations. However a better understanding of the social differences, the diverse interests in the resource use (local communities) as well of its environmental management and conservation (communities' views versus the government official's goals) is needed for more successful management strategies (Leach *et al.* 1999). For this reason specific objectives of all stakeholders need to be addressed. Table 1 would be a basic guide to addressing the important objectives. Macintosh and Ashton, (2003) have provided some important management tools for more sustainable mangrove utilization. They have summarized these steps that need to be taken into account when developing a management plan for mangrove ecosystems. This Code of Conduct was adapted from Macintosh and Ashton, (2003) for this study and the key steps include:

“Mangrove management objectives” (summarized in Table 6.1): These would be all the important objectives that would benefit the sustainable mangrove utilization as well as benefit the protection and biodiversity of these ecosystems. It must be beneficial to all stakeholders involved to avoid conflict of interests.

“Precautionary approach to management”: There has to be relevant and recent knowledge gained from research to avoid possible failures in management strategies, however they urged that this step should not prevent or postpone the implementation process for management plans, but rather to leave room for improvements.

“Legal Framework”: Existing and improved policies should provide the fundamental guidelines for the protection and sustainable utilization of the mangrove ecosystems. This would include the direct (fire wood, building material, fodder for livestock) and indirect uses (fish, molluscs, bait collection) of the mangrove ecosystem resources.

“Implementation”: This would be the enforcement of the legal framework and policing of the utilization of the resources; however in many countries including South Africa this implementation is not effective as illegal activities are still continuing and are overlooked. A general lack of communication between the management officials and the communities results in the increase in mangrove degradation. Thus community involvement and participation is vital in mangrove conservation.

“Mangrove Inventory for Management”: This monitoring would include the socio-economic-environmental aspects and not only the ecological functioning of the mangrove ecosystem, even though they are important; the social research fields should also be included by means of structured interviews with the local communities. Continuous mangrove monitoring is essential in establishing a good baseline data set and will provide an insight on the best management strategies which support policy makers’ decisions.

“Socio-Economic Considerations”: Mangrove ecosystems provide so many benefits for local communities and it would be insensitive to ban the use of the mangrove resources that these subsistence farming communities need to make a living from and are heavily dependent on. Many impoverished families should be provided with alternative resources to alleviate the pressure on the mangrove forests and improve their livelihoods. Therefore the communication and education between the stakeholders is very important.

“Cultural and Community issues”: Social studies and research would identify the communities’ “needs and wants” as well as traditional knowledge passed down through generations and the traditions that are associated with mangroves and mangrove ecosystems. One example would be mangrove medicinal uses (the harvesting of bark of *B. gymnorrhiza*) which are still practiced in South Africa (observed in this study at Mbashe and Mntafufu estuaries) would strengthen the protection of the mangrove ecosystems.

"Capacity Development": This is a fundamental part of the process of sustainable mangrove management, where all stakeholders are involved. This would be the governmental officials, researchers and the local community that is dependent on the mangrove ecosystems. This capacity development is to provide awareness and education on mangroves and their sustainable uses. This will be the tool in linking all parties involved and create better understanding between all.

"Forestry Management and Fisheries": This step would be to manage the resources more sustainably by identifying the environmental services that these forests provide such as sustainable firewood, building material as well as protection against sea storms and severe floods, provision of nursery habitats for biota and, provision of food security from fisheries.

"Agriculture and Mining": Most of the catchment areas and much of the estuary banks in the upper reaches have been converted to agricultural fields, many on steep slopes, where erosion would be inevitable. This erosion is evident and widespread in the Transkei coastal areas due to the hilly nature of the landscape. Many agricultural activities, especially those bordering mangrove settings, resulted in excessive siltation and sediment accretion within the mangrove ecosystems causing mangrove degradation (see Chapter 5). Sand mining close to the estuary mouth was observed in some estuaries (Mngazana, Mnyameni, and Kobonqaba estuaries) and may have had similar impacts compared to the poor agricultural practices. The lack of environmental regulation and enforcement would have to be dealt with as this also is a major threat to mangrove ecosystems.

"Tourism, Recreation and Education": The income generated from tourism and tourist related activities is one of the fastest growing economic assets in the world and mangrove ecosystems provide such a strong opportunity for ecotourism (Satyanarayana *et al.* 2012). This is because these ecosystems are rich in biodiversity and are unique, with a high health index value. This would include activities such as bird watching and boat rides through the mangroves, recreational fishing and discovery of the rich cultural traditions of the local community. The increased awareness among the public will promote greater incentive to protect these ecosystems. The conservation efforts in protecting these unique ecosystems will create a sense of pride and additional income to the guides and local communities. Examples of these can be seen at the Bulungula Estuary and Nxaxo/Nqgusi estuaries, where the Wavecrest Lodge and the Bulungula Lodge have been positively involved in conservation, education and job creation in the community. Education would be a key factor including both school children and adult literacy. This would steer general perception of these habitats as sanctuaries for ecological services they provide and to conserve these for future generations.

"Mangrove Products and Responsible Trade": In South Africa the trade of mangrove wood is relatively uncommon and not comparable to the global timber trade from the tropical regions. The selling of small scale mangrove wood is only between local communities or villages for domestic use. However the mangrove forests in the former Transkei are also small compared to the forest in the subtropical and tropical regions, thus even harvesting of firewood for the domestic household use poses a serious threat to these ecosystems.

"Mangrove Research and Information Exchange": The lack of information and knowledge has resulted in undervaluing of the ecosystem services provided by mangroves as well as the misunderstandings of the struggles and interest of the local communities, by the decision makers. The information on these mangrove ecosystems is essential for more effective management and the sharing of information between researchers, government officials and local communities and would improve the management strategies.

"Integration of Mangrove Management into Coastal Zone and River Basin": This would include the conservation of not only the mangrove forest area but also the whole estuary, the estuary mouth dynamics, up river and the river banks and the catchment area. These would all have to be considered and thus the co-operation of all stakeholders is required for effective coastal management. It is important to increase Marine Protected Areas (MPAs) as well as the incorporate the 5 m topographical contour line, where this boundary allows for estuarine ecological processes to take place, where sea-level rise will be taken into account, and preventing developments or constructions below this 5 m contour. This will protect the estuarine vulnerable biodiversity, regulate coastal developments and conversions, provide buffer zones and connect possible ecological corridors (Van Niekerk and Turpie 2012).

Coastal regions are some of the most populated areas, where climate change such as sea-level rise will have social-economic and environmental consequences (Nicholls and Lowe 2004; Nicholls *et al.* 2007). The sea-level rise will cause a shift in mangrove distribution, species composition and competition as well as recruitment successes or failures (Ye *et al.* 2004; Abel *et al.* 2011). There is an urgent need to protect the mangroves in South Africa in particular in the former Transkei region, even if these comprise only a small percentage of the continent's mangrove area cover. They are highly threatened, unique ecosystems and are one of the most southerly distributed mangrove forests (Macnae 1963; Spalding *et al.* 1997; 2010). A summary of the pressures and the management recommendations are provided in Figure 6.1.

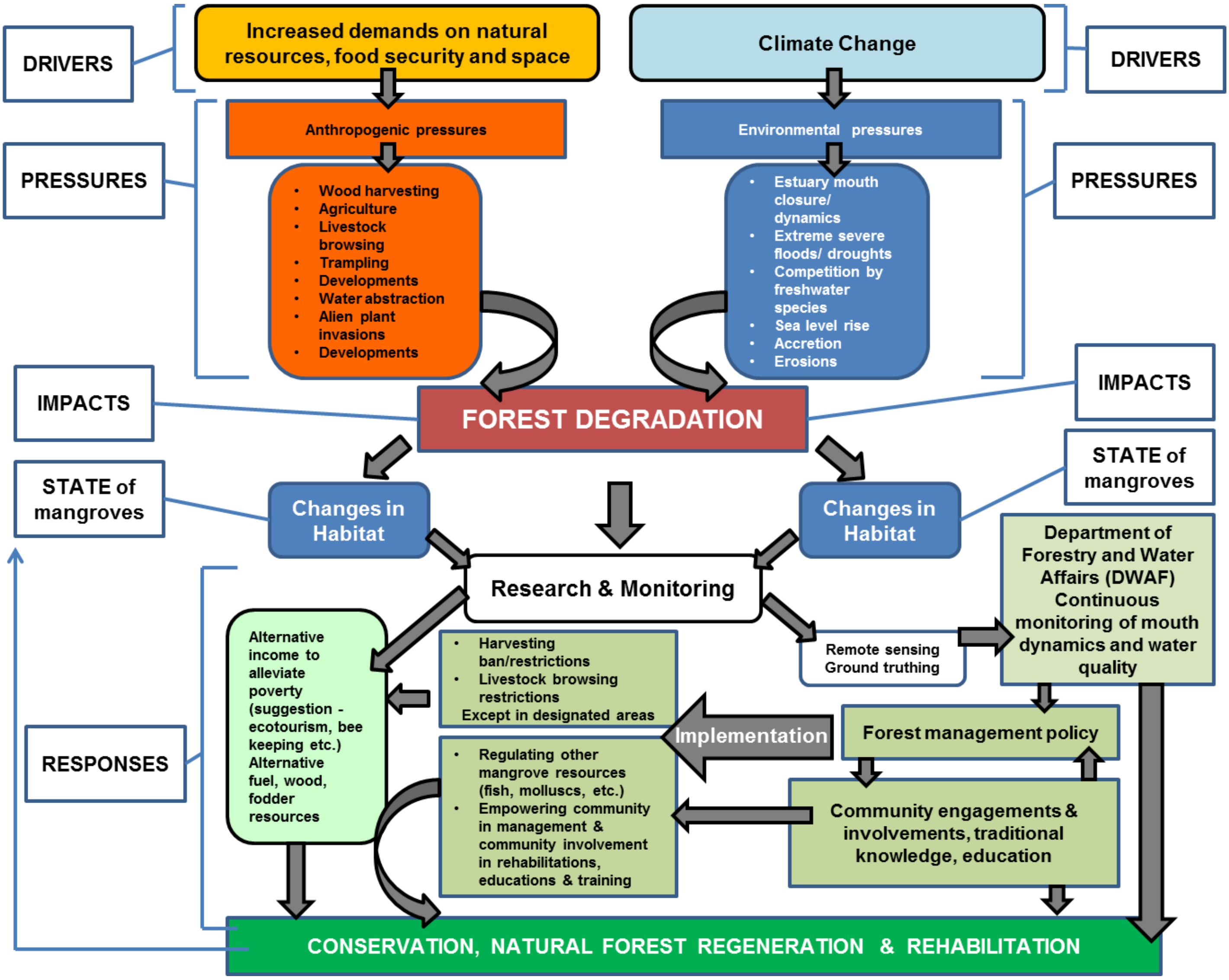

Figure 6.1: A simplified diagrammatic summary of the natural change and anthropogenic pressures with the management recommendations.

Table 6.6.1: Objectives for sustainable management of mangroves (adapted from Macintosh and Ashton 2003)

Objectives (why it needs to be done?)	Interventions (what can be done?)
Development Objective: Promote conservation and biodiversity by preventing further mangrove degradation and to continuously improve management for the protection of these ecosystems.	1. directly protect pristine or near pristine mangrove ecosystems 2. protect mangroves from degradation due to anthropogenic impacts 3. promote natural regeneration where mangroves have the capacity for "self-renewal" 4. rehabilitate degraded mangrove ecosystems where "self-renewal" would be too slow or not possible 5. enforce and protect mangrove/estuarine buffer zones 6. enhance and protect cultural and social values related directly or indirectly to mangroves ecosystems 7. promote and improve sustainable traditional management techniques from communities near forests 8. support co-management with these local communities and government officials
Immediate Objectives: Policies Find more effective policies and management strategies for the sustainable conservation of the mangrove ecosystems and these should be implemented by the people involved; nature conservation officials and local community.	9. continuously improve and adapt management strategies and policies (learned from research and experience) 10. strengthen and harmonise regulations for sustainable mangrove utilization 11. share information for better policy decision (community engagements) 12. share and safeguard traditional knowledge of local communities to strengthen sustainable utilization 13. promote research as new information will improve policies and management strategies
Immediate Objectives: Local communities Alleviate poverty by improving their quality of life of these communities that are directly and heavily depended on mangrove ecosystems.	14. provide alternative wood supply such as planting forests close to villages to promote sustainable utilization 15. provide designated forest areas for browsing domestic livestock and "no browse areas" to allow regeneration 16. identify and resolve ownership debates and involve/ provide communication, education and public awareness 17. be sensitive to equity, gender issues and age old traditions of the communities 18. directly involve communities in mangrove conservation projects; rehabilitation, education etc.
Immediate Objective: Productivity Increased mangrove ecosystem productivity will sustain direct resources such as fire wood and fodder for domestic livestock, while indirect resources includes ecosystem services (fisheries, water quality nursery habitats, biodiversity etc.)	19. identify and improve management strategies through research, education and local community involvements 20. reduce or prevent mangrove over-utilization and protect large parts of these forest to increase productivity 21. identify and promote alternative sustainable uses of resources such as wood from planted forests, alternative methods for cooking (that are inexpensive and maybe subsidised) rather than using wood, alternative income such as ecotourism, production of mangrove related products such as mangrove honey and promoting "green labelling" (eco-friendly) of such products.

6.2. Shortcomings of this study and future research

The study at St. Lucia Estuary only determined the state of the mangroves under closed mouth conditions and it would be recommended to continue this research as part of a long-term monitoring programme. If the estuary mouth opens and the Umfolozi River is reconnected to the St. Lucia Estuary then it will provide more opportunities for research to determine if the mangrove forests will regenerate in response to tidal conditions. In addition the browsing of game such as kudu on these mangroves was evident in this estuary but this was not covered in this study. A detailed study on the browsing and propagule predation from game, crabs as well as insects for South African mangroves would be needed to describe these relationships and compare findings to anthropogenic pressures such as that from domestic livestock.

The limitation of the study at Nahoon Estuary was that this study presented the initial measurement for possible future studies and no sediment accretion rate data has been available for this estuary. Tamin *et al.* (2011) measured the establishment of *Avicennia marina* on accreting coastal regions in Malaysia, where seedling establishment was monitored and regeneration proven to be successful on the “accreted coastline”. A similar study would be suggested for Nahoon Estuary and thus represent a valuable opportunity for future research in fields of climate change through a long term monitoring programme. The higher latitude would provide evidence of mangrove expansion beyond their previous distribution; another opportunity understands the mangrove-salt marsh interaction over the long term research in relation to sedimentation, sea-level rise and erosion from floods. It is recommended to include methodology for measuring sediment elevation and bank erosion in most estuaries as a long-term monitoring programme, where these would provide more insights on the rate of sea-level rise. Other methodology such as data loggers along the east coast that measure ambient and water temperature, pH and CO_2, for continuous monitoring would be useful indicators in determining the rate of climate change.

For both Chapters 4 and 5, this research did not address the social and economic aspects, which would consist of a detailed investigation by means of structured interviews of all people that utilize the mangrove resources directly such as the subsistence farmers or indirectly such as the ecotourism business. Globally much research has been done on mangrove already and much literature is available, with many examples of natural forest regeneration (Mohamed *et al.* 2009), rehabilitation (Tamin *et al.* 2011), community-based natural resource management, community involvement (Shah and Kamaruzaman 2007;

Campuzano *et al.* 2011; Satyanarayana *et al.* 2012) and conservation, successful sustainable mangrove restorations and failures (Primavera and Esteban 2008; Friess *et al.* 2012). These many examples, successes and failures can be considered and adapted in the local conservation plans to strengthen these for a more sustainable mangrove resource. A limitation of this study was that estuaries mentioned in Chapter 5 have been sampled once-off and these should be revisited at different seasons and time spent at each estuary should be increased for a more comprehensive data set of the different taxa, environmental and sediment characteristics.

The preceding chapters have:

(1) determined the present state, distribution, population structures of mangroves and identified causes of mangrove loss or gain.

(2) identified the current threats to mangroves, which are both of anthropogenic and natural change origin. The major findings were that anthropogenic pressures had been more profound and widely distributed than the pressures from natural changes. However if environmental pressures would intensify and coupled with anthropogenic pressures it would accelerate mangrove degradation and loss.

(3) presented management recommendations from the findings of this research that would provide decision-makers with the tools for effective management plans and conservation of mangrove ecosystems.

(4) The findings of this research will provide input to multi-disciplinary forums such SANBI's National Biodiversity Assessment and estuary management plans which are a requirement of the Integrated Coastal Management Act of the Department of Environmental Affairs. An understanding of the responses of plants to changes in environmental conditions also provides important input to the Department of Water Affairs' ecological water requirement studies which are conducted to ensure implementation of the National Water Act (Act 36 of 1998).

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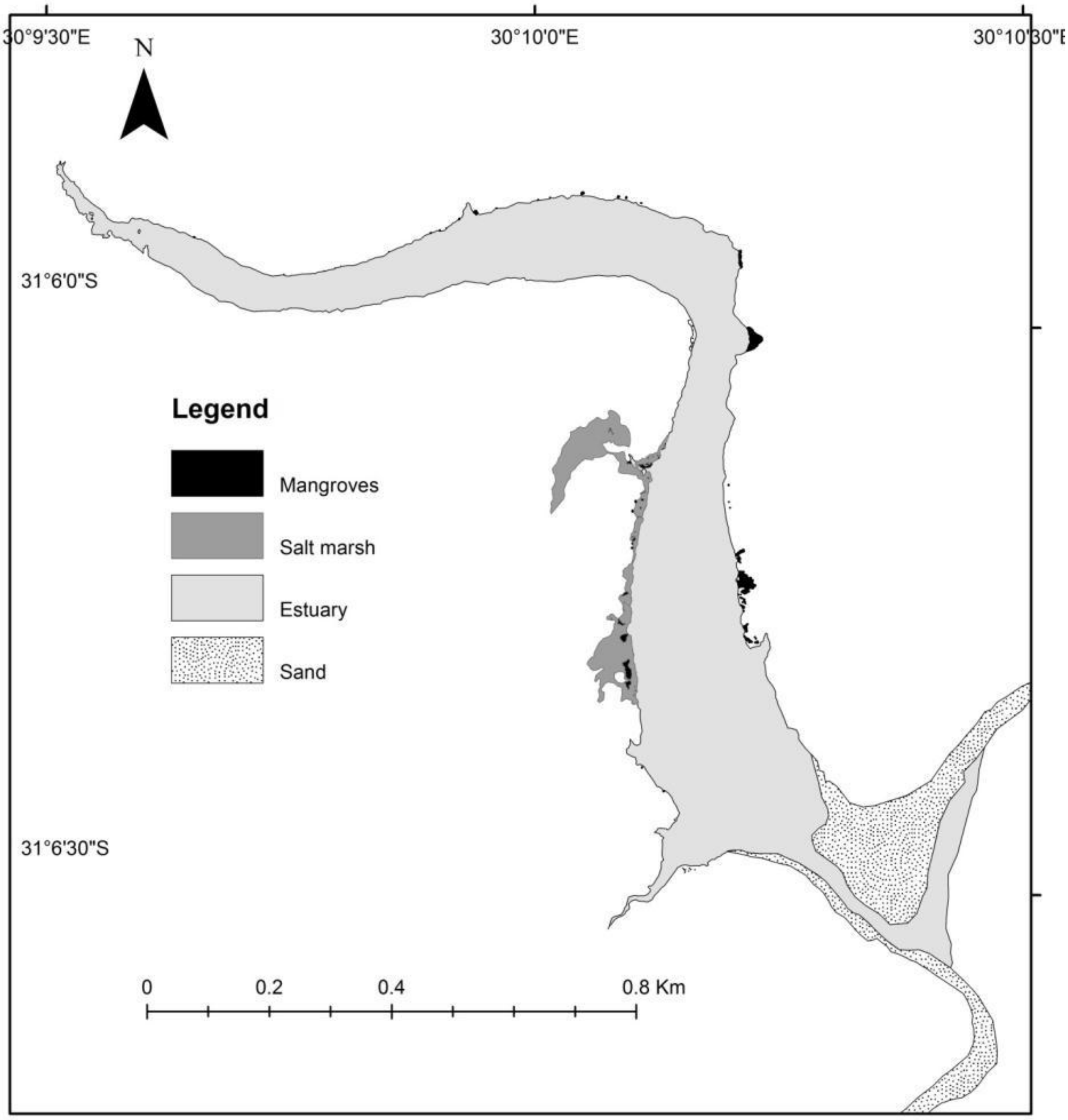
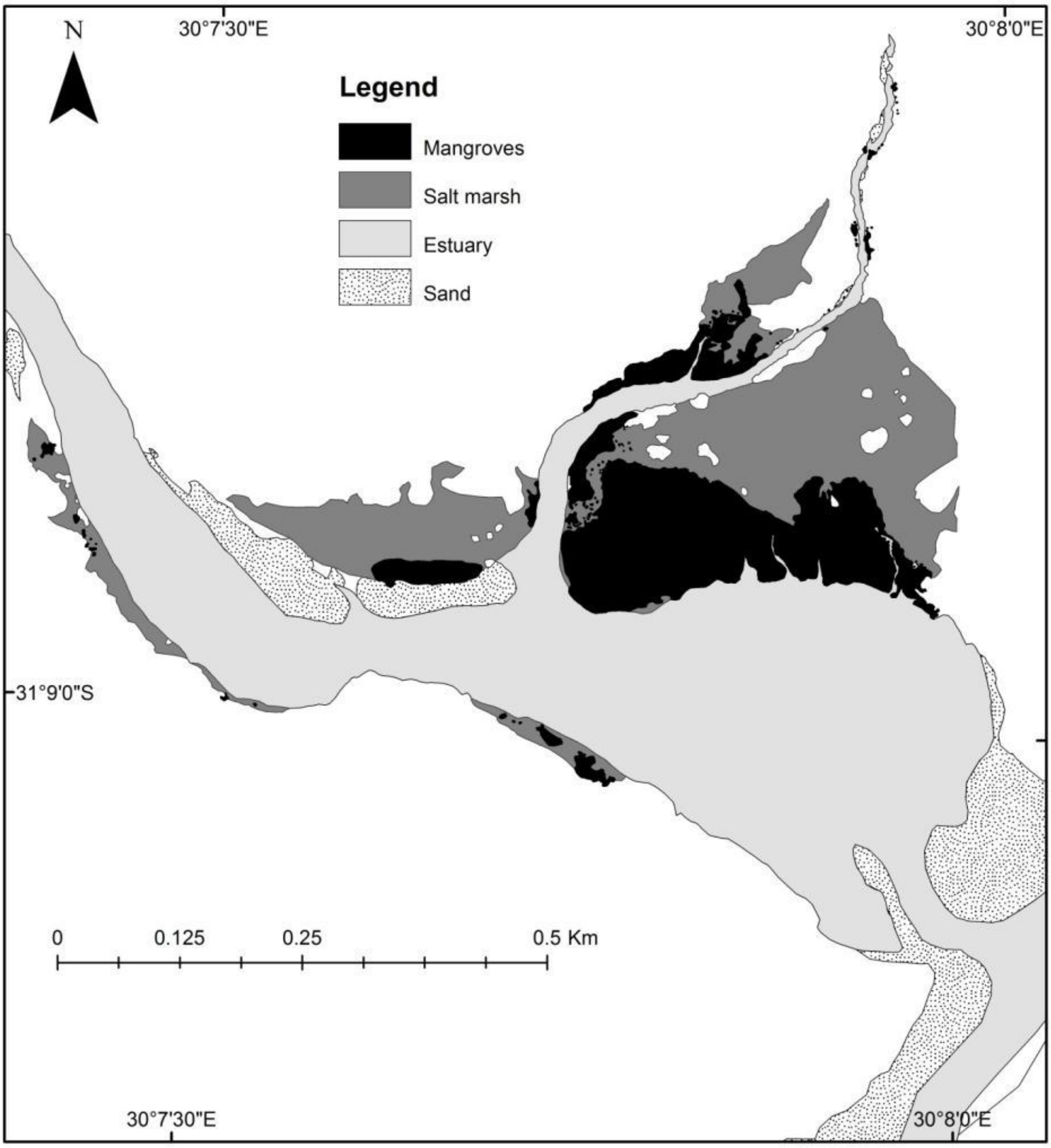
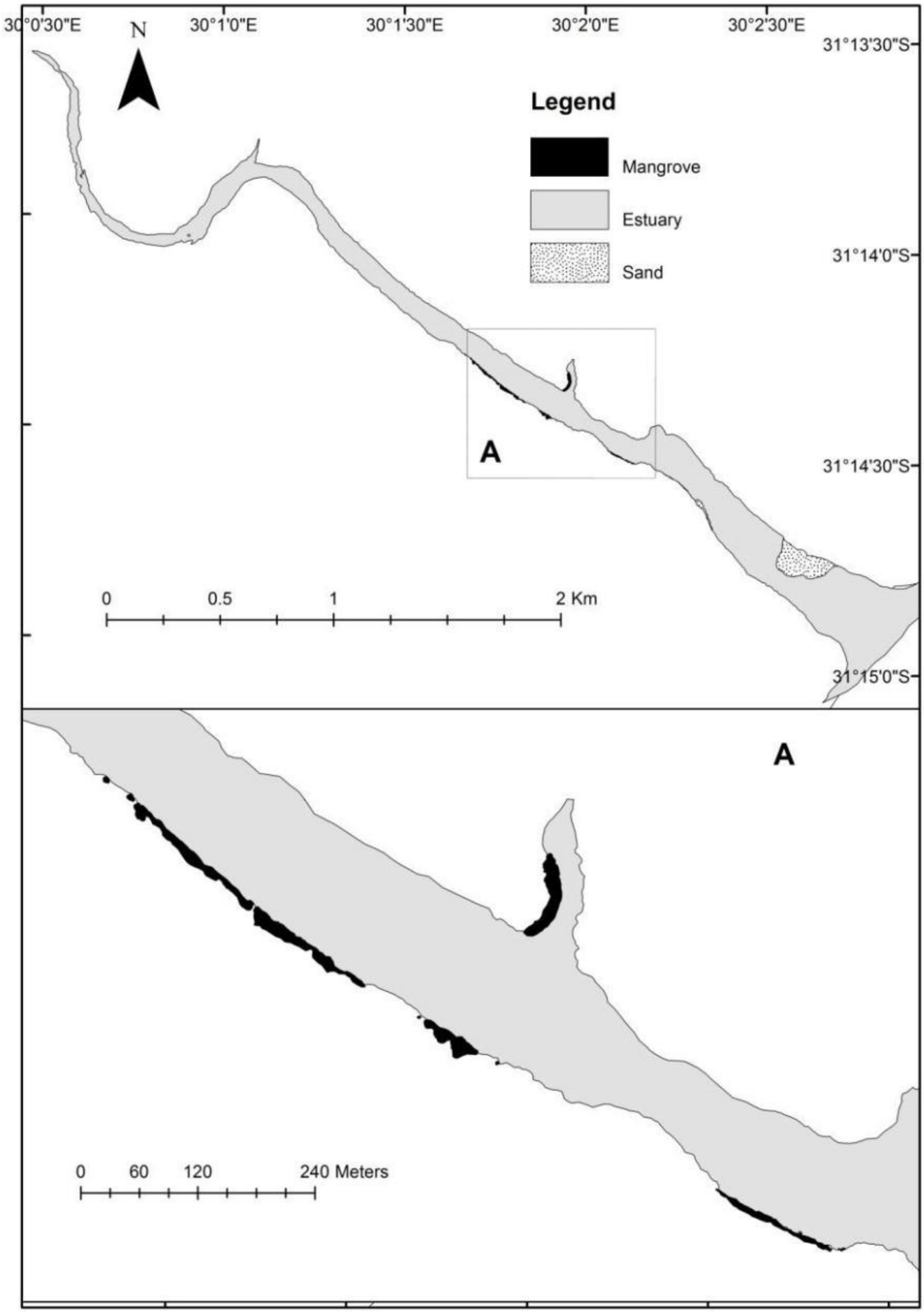
APPENDIX 1: The Mzamba Estuary close to Port Edward



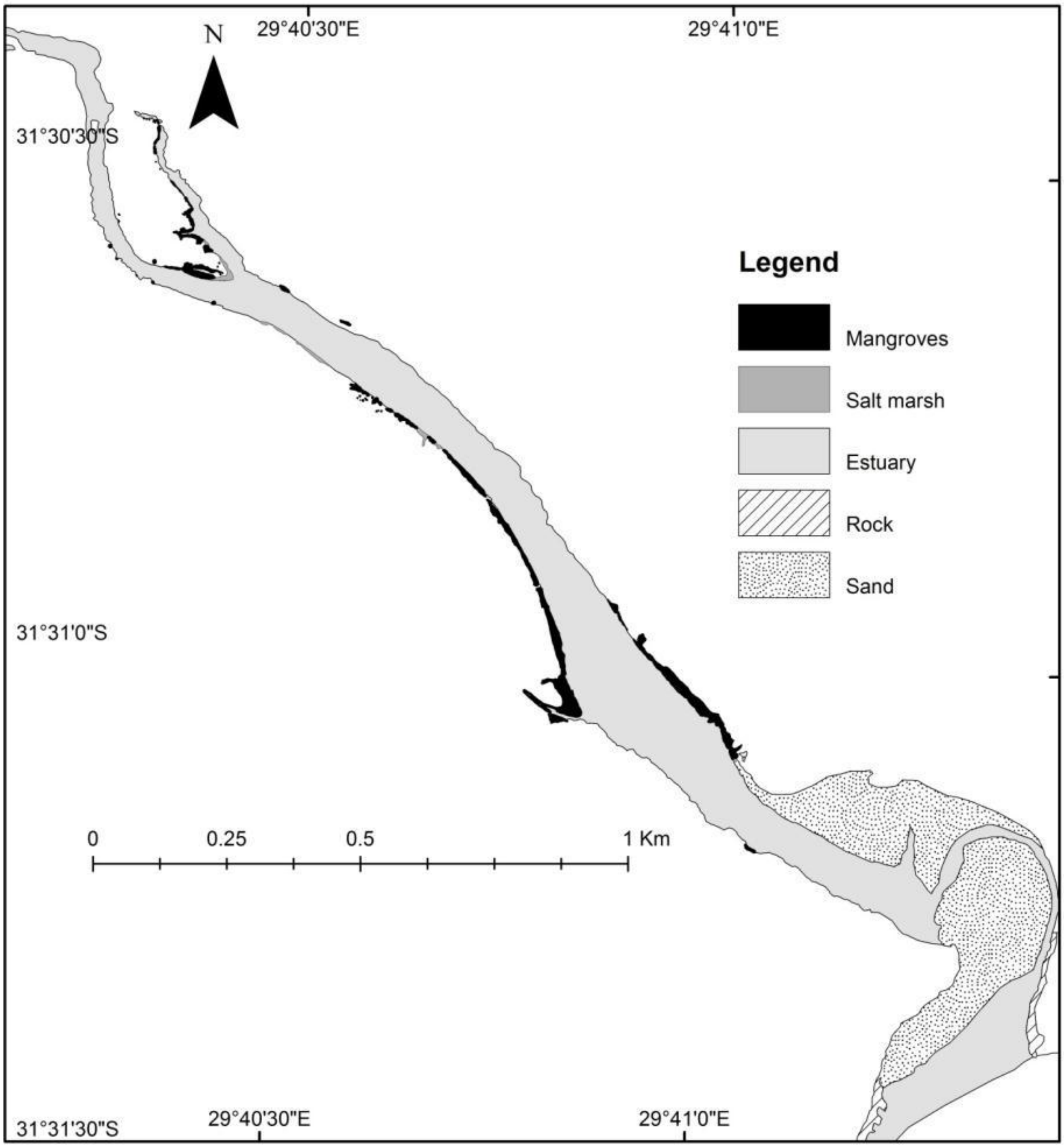
APPENDIX 2: The Mnyameni Estuary which falls within the marine protected area



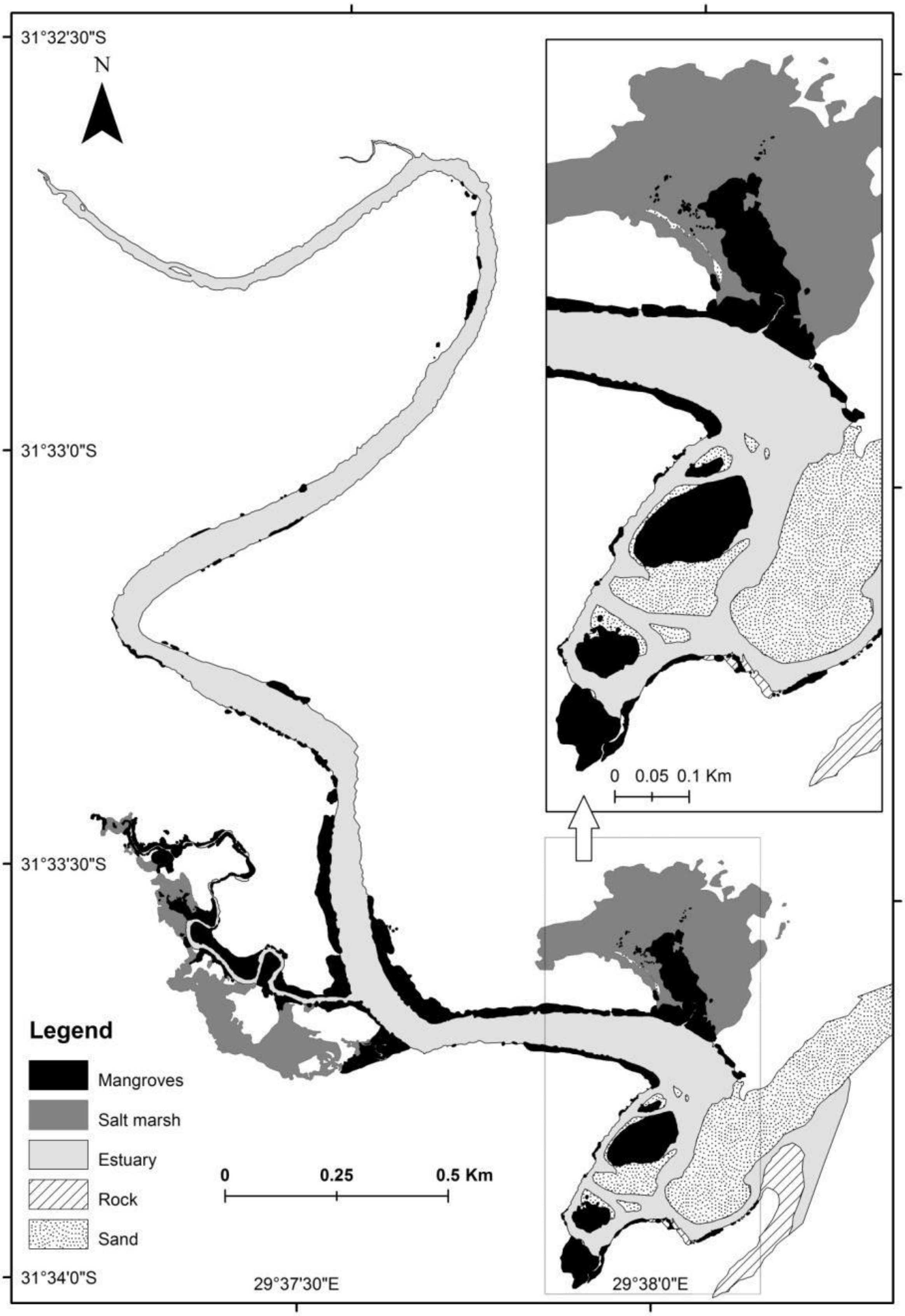
APPENDIX 3: The Mtentu Estuary which falls within a marine protected area



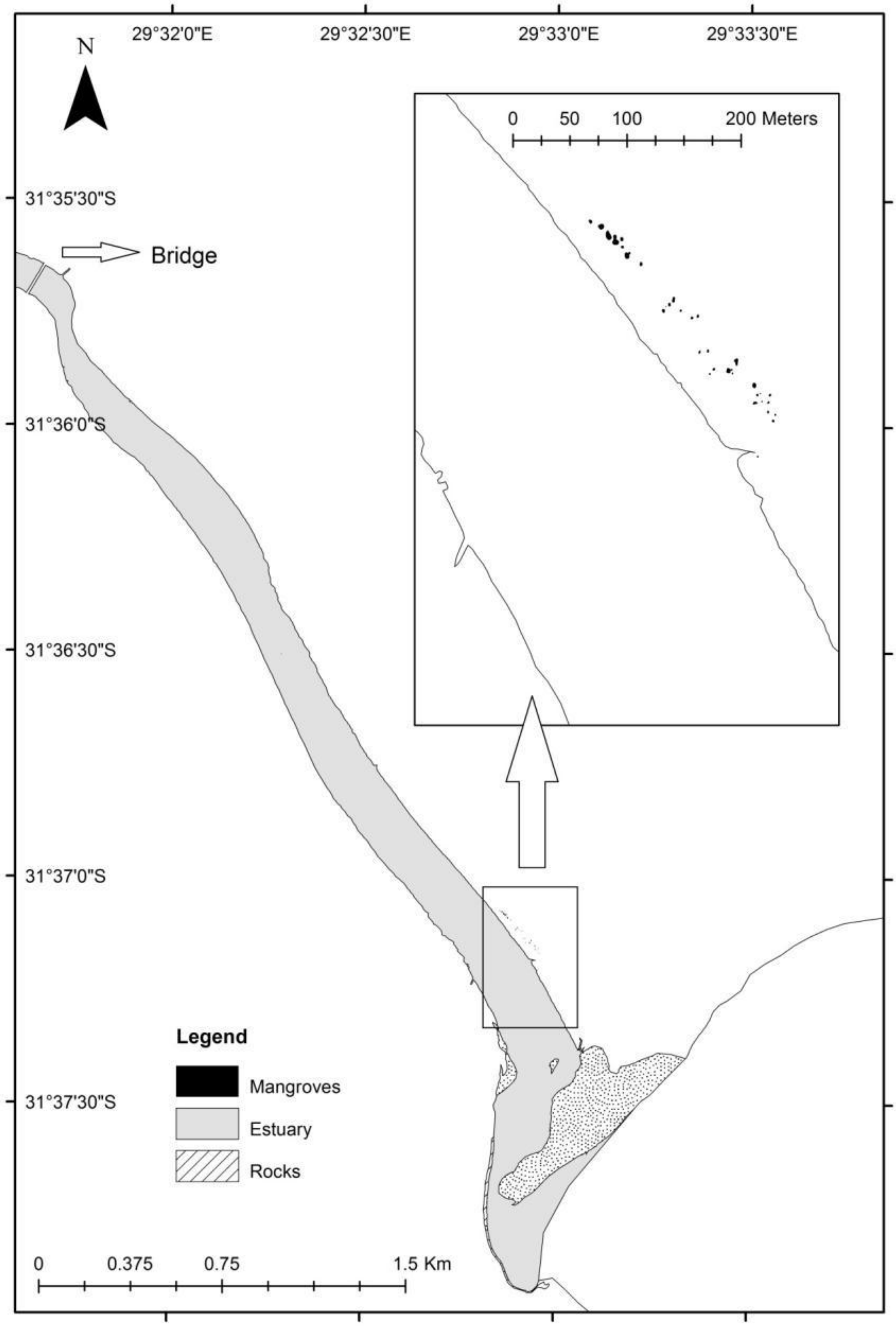
APPENDIX 5: The Mzintlava Estuary



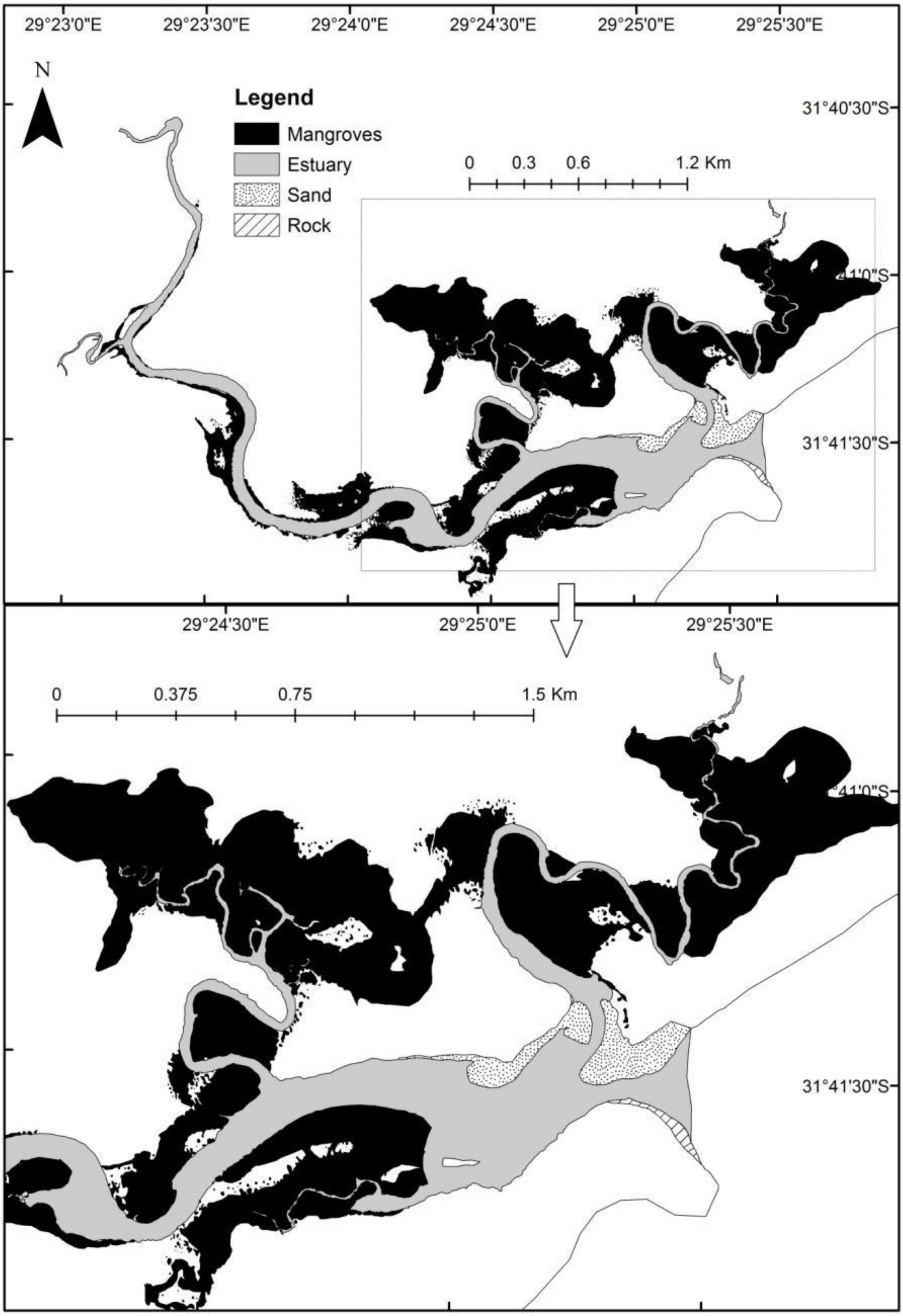
APPENDIX 6: The Mntafufu Estuary



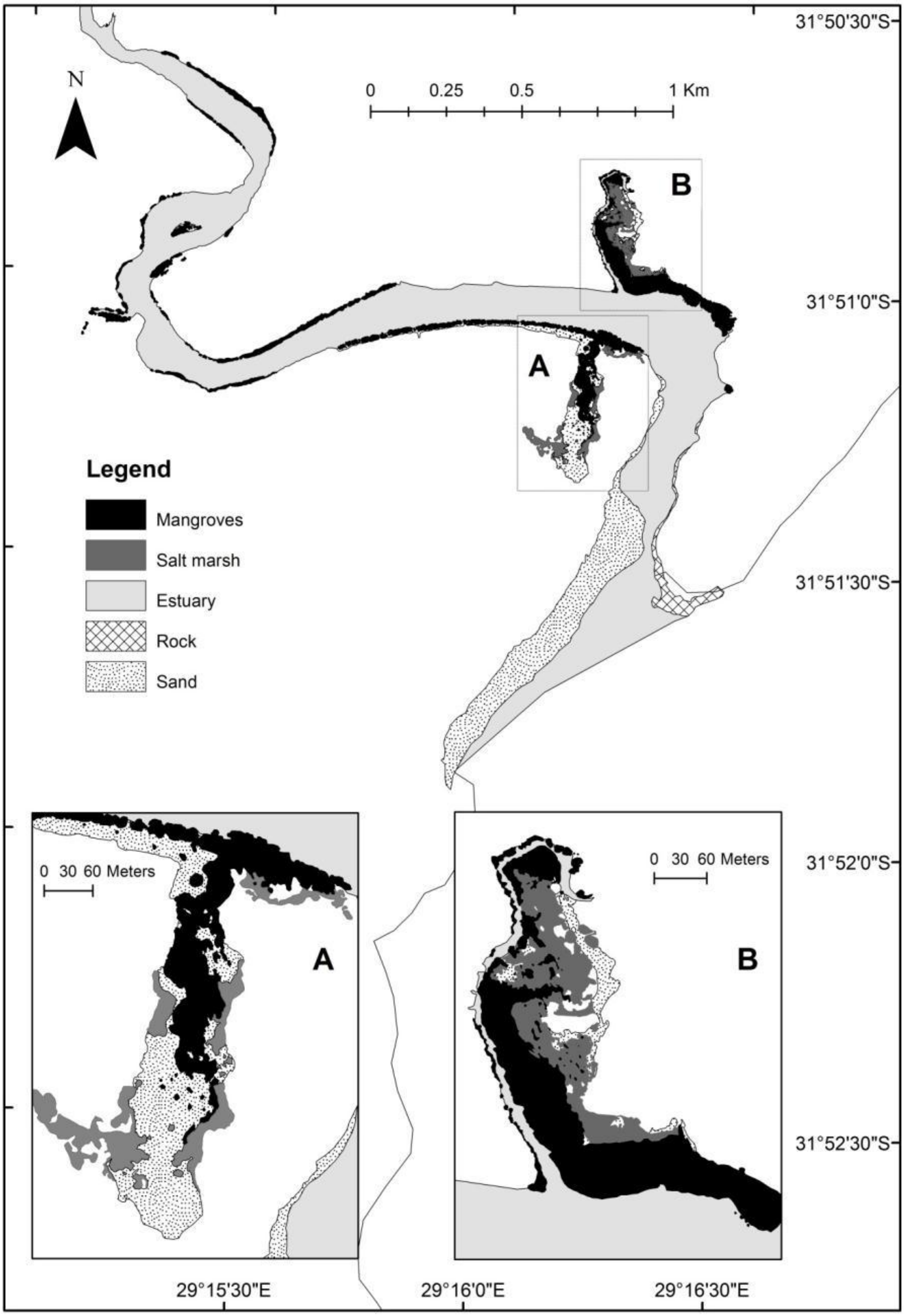
APPENDIX 7: The Mzimvubu Estuary



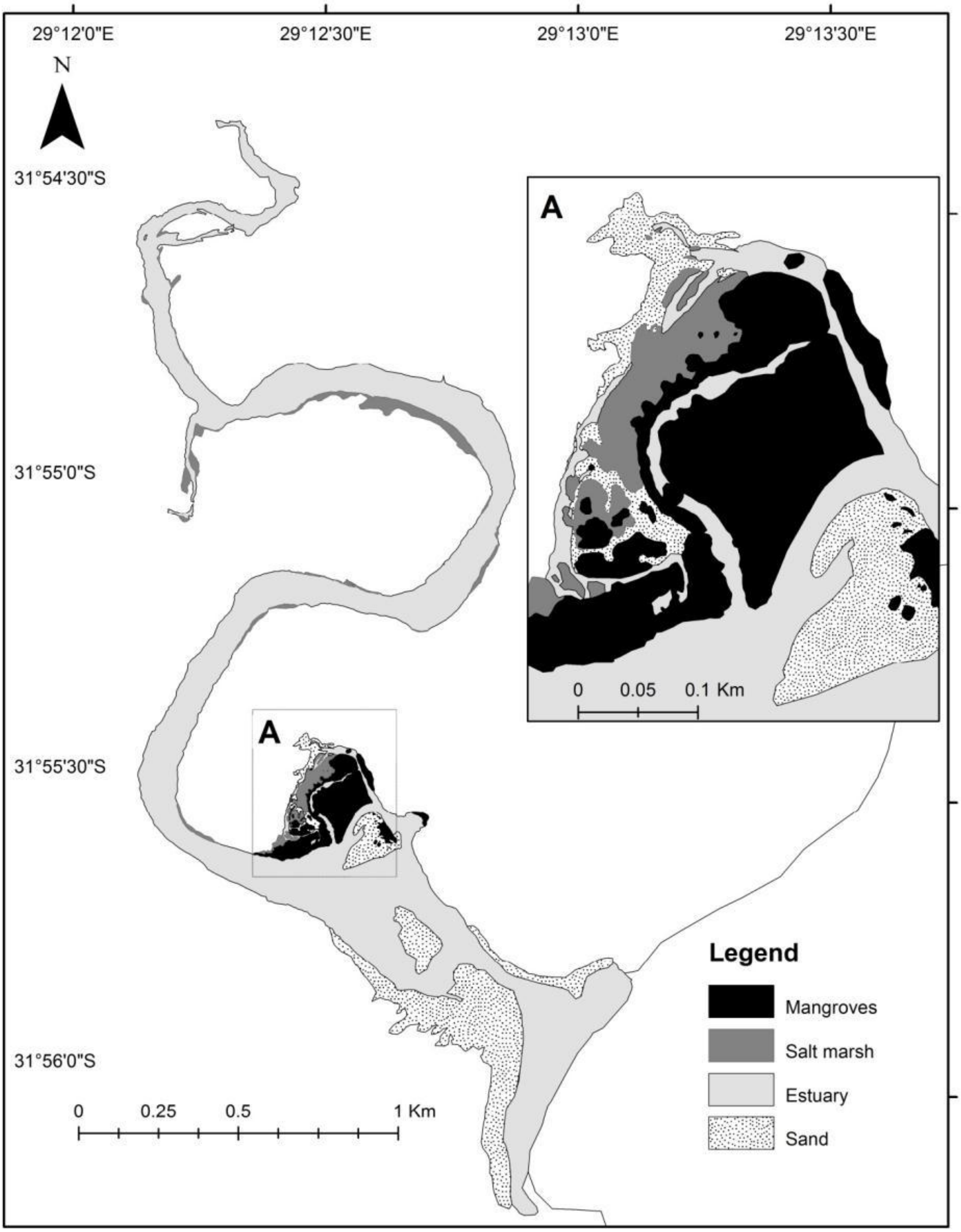
APPENDIX 8: The Mngazana Estuary



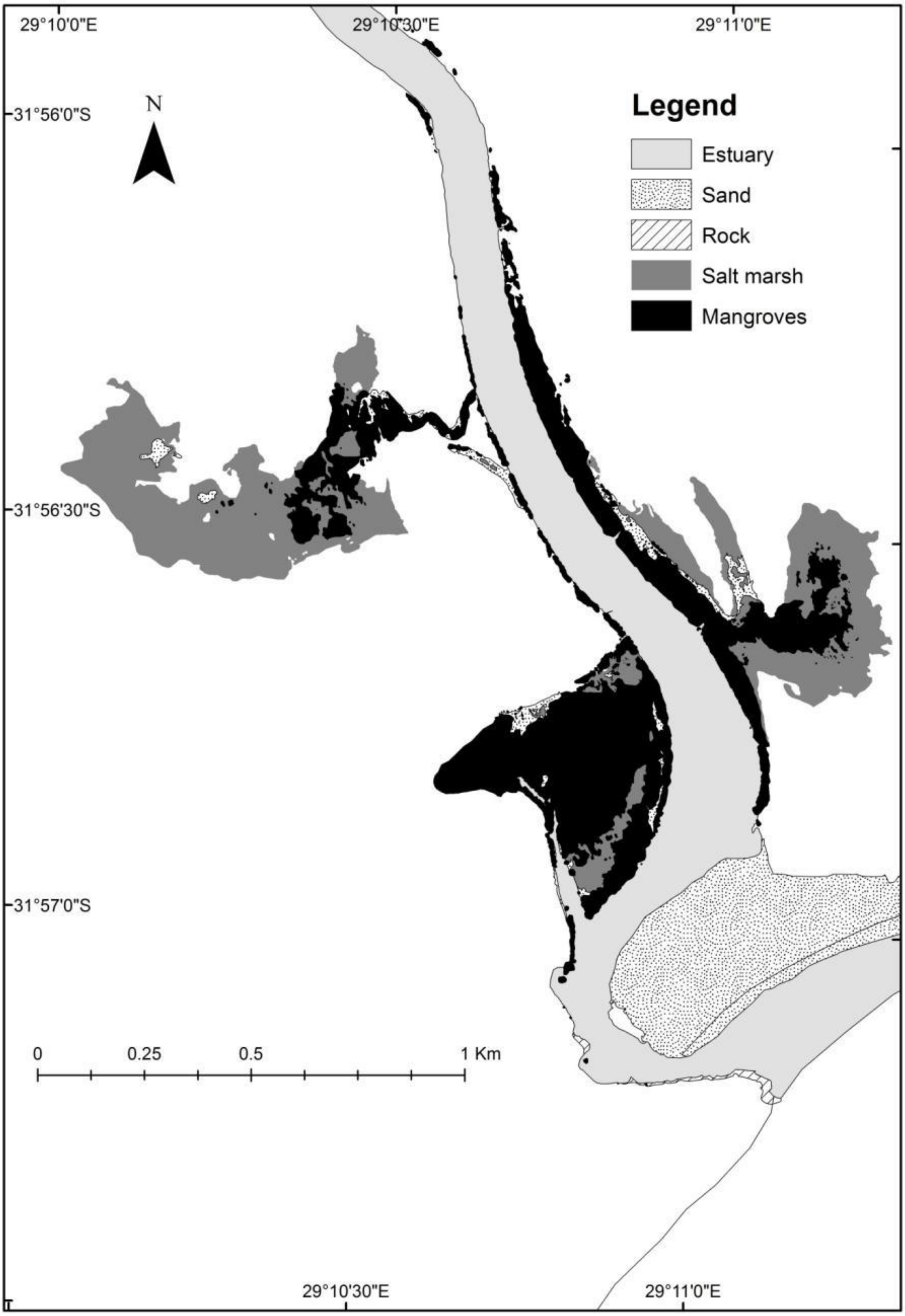
APPENDIX 9: The Mtakatye Estuary



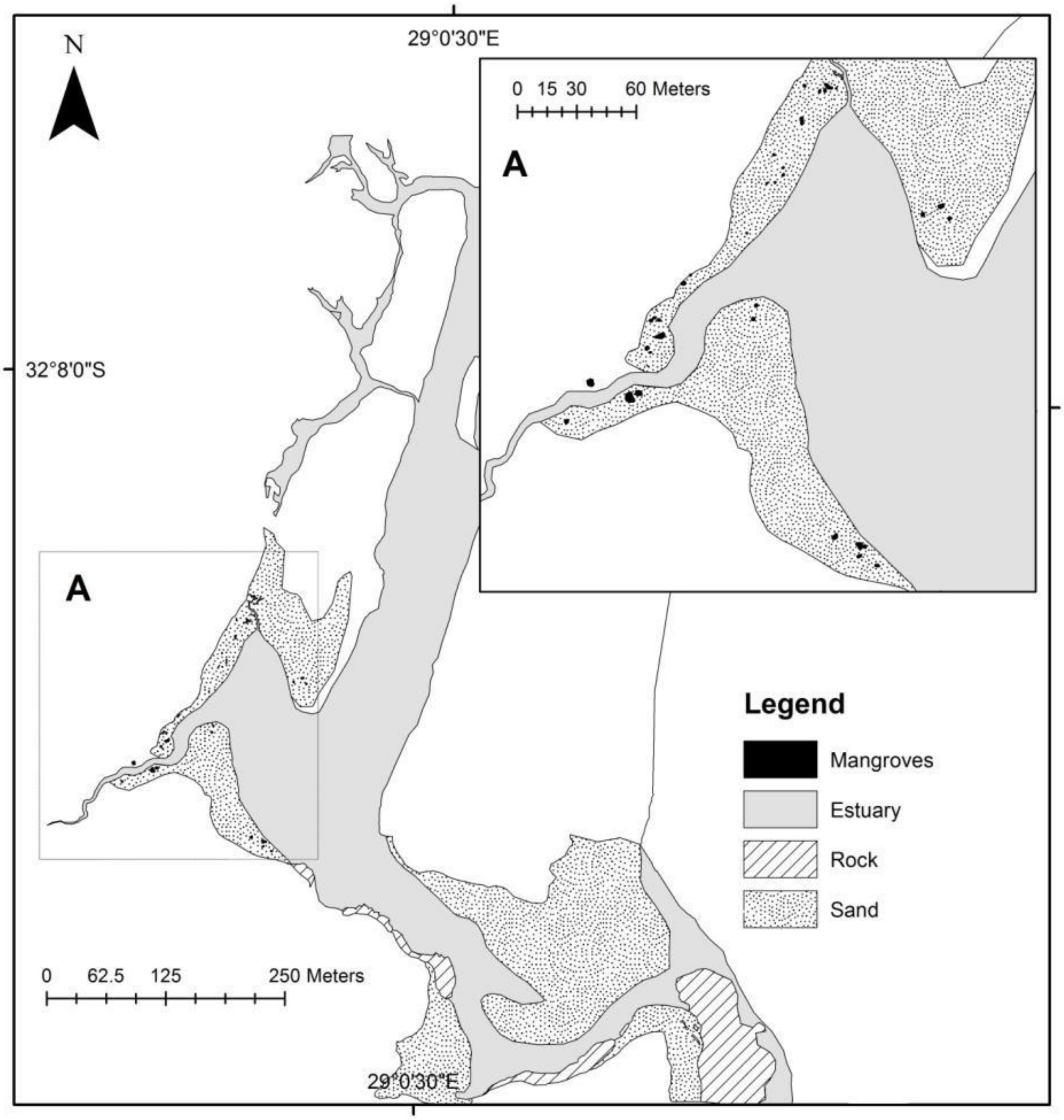
APPENDIX 10: The Mdumbi Estuary



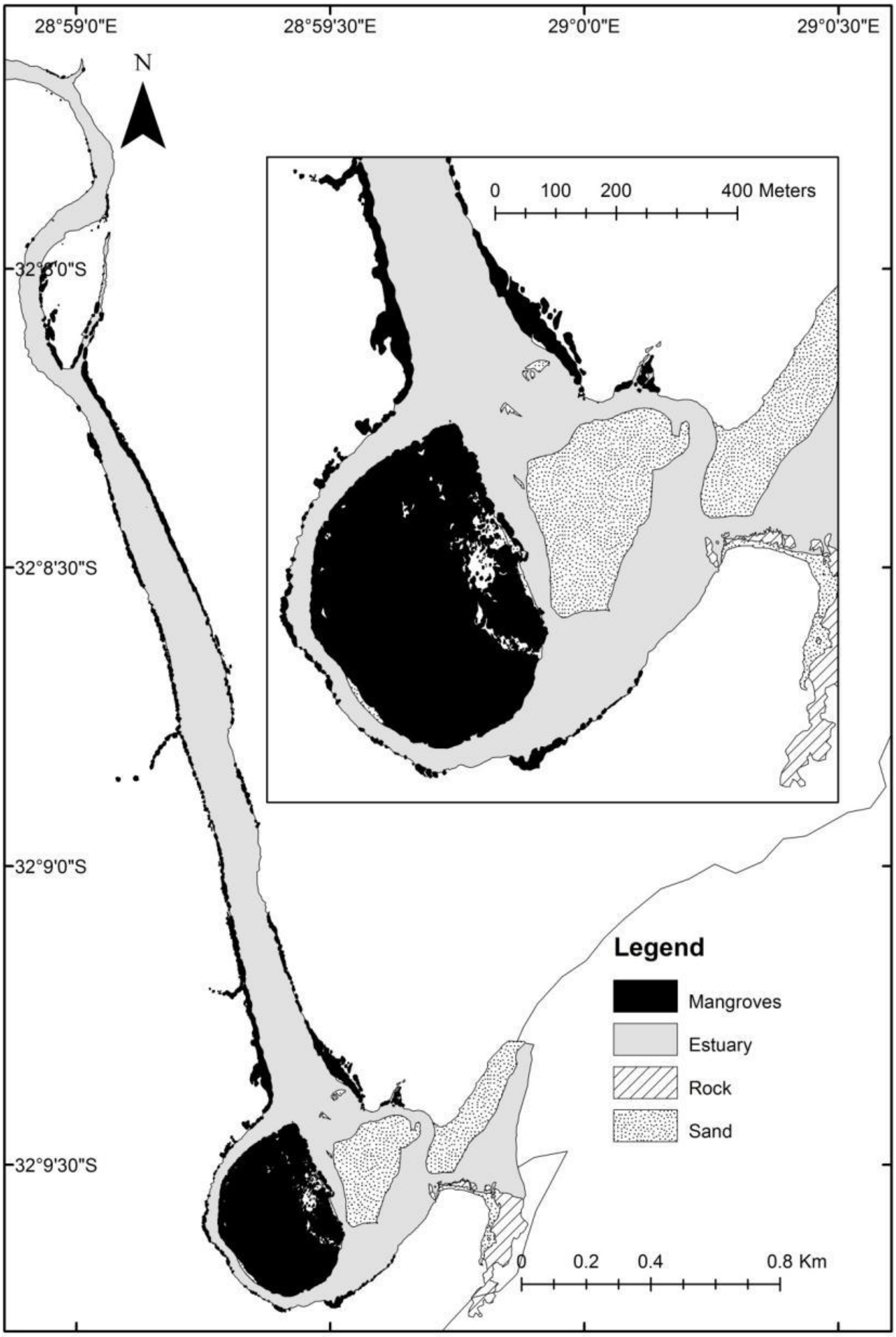
APPENDIX 11: The Mthatha Estuary



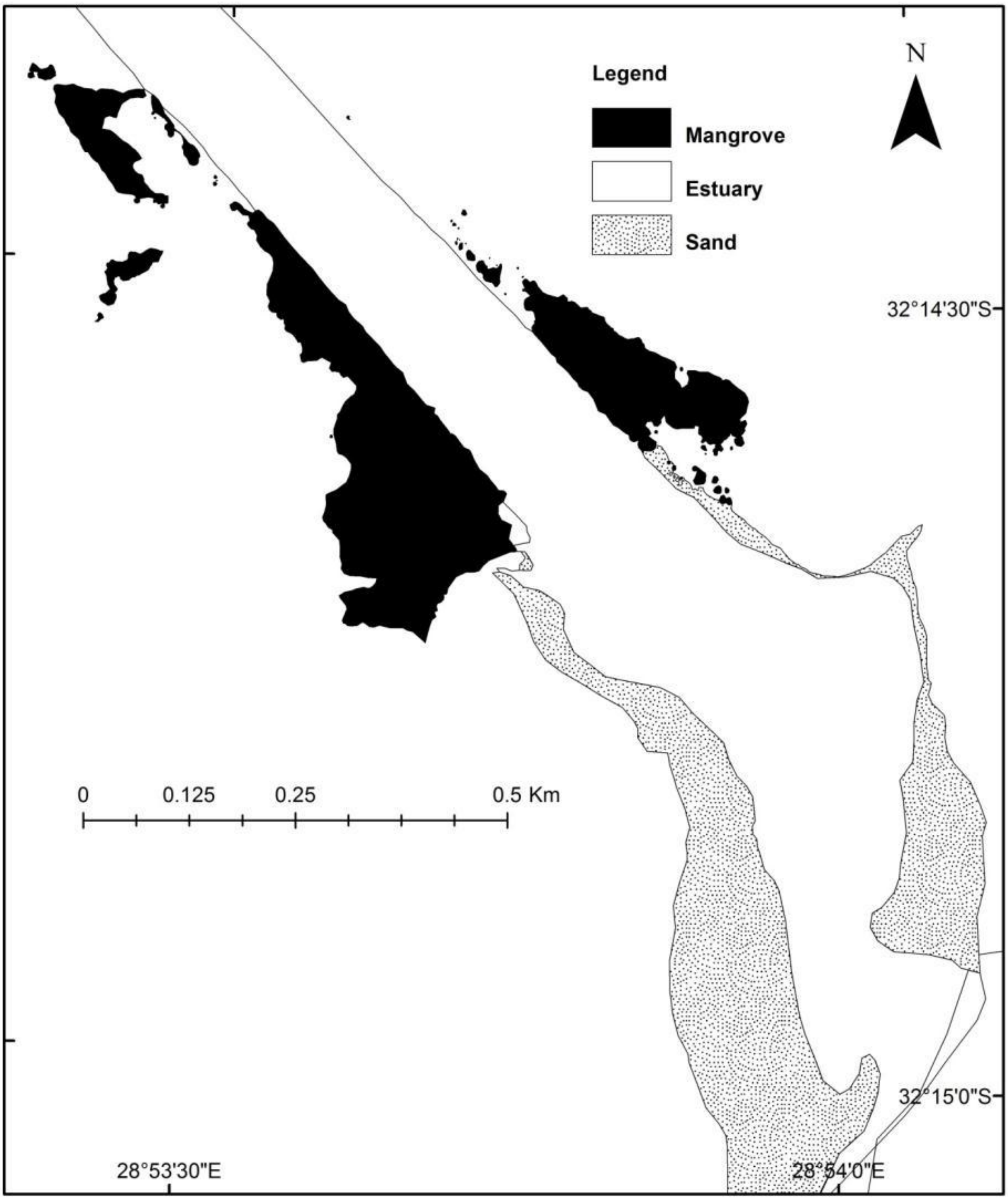
APPENDIX 12: The Bulungula Estuary



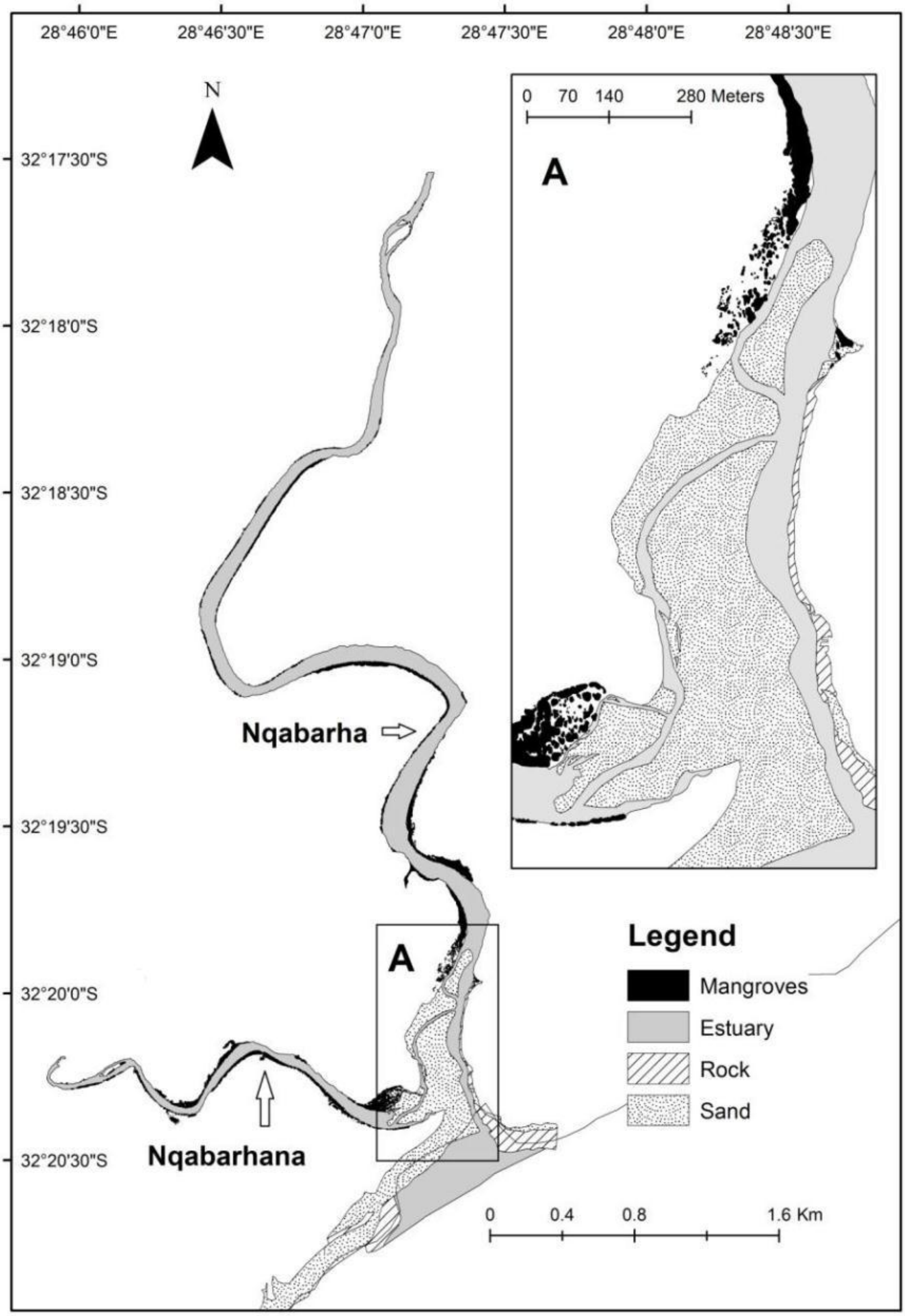
APPENDIX 13: The Xhora Estuary



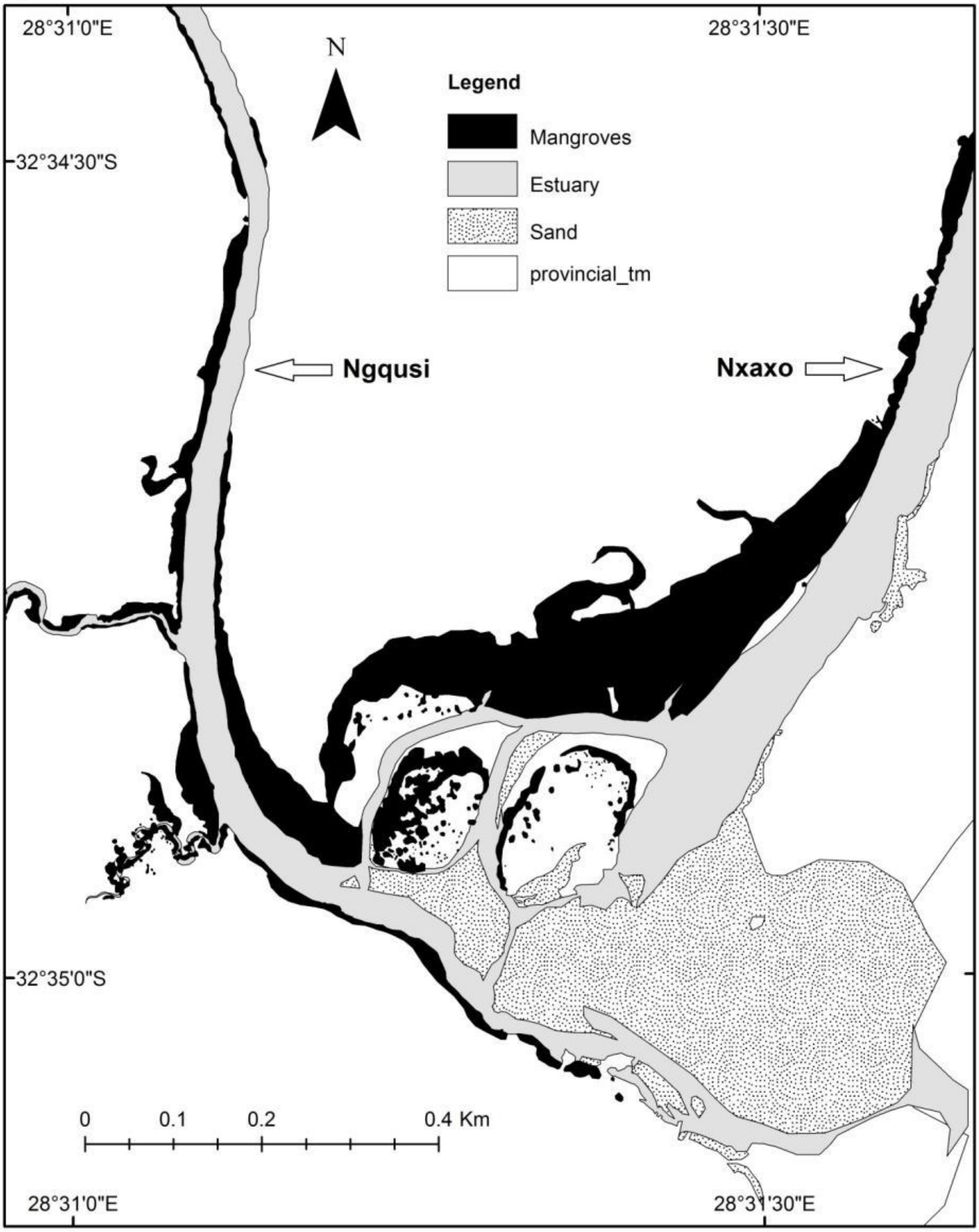
APPENDIX 14: The Mbashe Estuary which falls within the Dwesa and Cebe Nature Reserve



APPENDIX 15: The Nqabarha and Nqabarhana estuaries sharing the same mouth



APPENDIX 16: The Nxaxo and Ngqusi estuaries sharing the same mouth.



APPENDIX 17: The salt marsh and submerged macrophyte species list for the 17 former Transkei estuaries

Salt marsh & submerged species	Mtamvuna	Mzamba	Mnyameni	Mtentu	Mzintlava	Mntafufu	Mzimbuvu	Mnganzana	Mtakatye
Bassia diffusia (Thunb.) Kuntze					√	√		√	√
Sarcoconia natalensis (Bunge ex. Ung-Sternb.) A.J.Scott			√		√	√		√	√
Sarcoconia tegetaria S. Steffen, Mucina & G. Kadereit			√						
Cotula coronopifolia Thunb.				√	√	√			√
Triglochin striata Ruiz & Pav.			√	√	√		√		√
Zostera capensis Setch.		√	√	√	√	√	√	√	√
Phragmites australis (Cav.) Steud	√	√		√	√	√	√		
Stenotaphrum secumdatum (H. Walter) Kuntze	√	√	√	√	√	√	√	√	√
Sporobulus virginicus (L.) Kunth	√	√	√	√	√	√	√	√	√
Cynodon dactylon (L.) Pers.					√		√		√
Juncus effusus L.			√				√		
Juncus kraussii Hochst subsp. *kraussii*	√	√	√		√	√	√	√	√
Juncus rigidus Desf.						√			
Juncus littoralis C.A Mey.		√	√				√		√
*Bolboschoenus maritimus (*L.) Palla			√		√				

*submerged macrophyte

APPENDIX 18: The salt marsh and submerged macrophyte species list for the 17 former Transkei estuaries

Salt marsh & submerged species	Mdumbi	Mthatha	Bulungula	Xhora	Mbashe	Nqabarha	Nxaxo/Ngqusi	Kobonqaba
Bassia diffusia (Thunb.) Kuntze		√		√	√	√	√	
Sarcoconia natalensis (Bunge ex. Ung-Sternb.) A.J.Scott	√	√	√	√	√	√	√	
Sarcoconia tegetaria S. Steffen, Mucina & G. Kadereit	√				√		√	
Cotula coronopifolia Thunb.			√			√		
Triglochin striata Ruiz & Pav.	√	√	√			√		
Zostera capensis Setch.	√		√			√	√	
Phragmites australis (Cav.) Steud	√	√						
Stenotaphrum secumdatum (H. Walter) Kuntze	√	√		√	√	√		√
Sporobulus virginicus (L.) Kunth	√	√	√	√	√	√		√
Cynodon dactylon (L.) Pers.	√							
Juncus effusus L								
Juncus kraussii Hochst subsp. *kraussii*	√	√	√			√		√
Juncus rigidus Desf.								
Juncus littoralis C.A Mey.	√							
*Bolboschoenus maritimus (*L.) Palla						√		√

*submerged macrophyte

APPENDIX 19: Invasive alien plant species list for the 17 former Transkei estuaries

Alien invasive species	Common names	Mzamba	Mnyameni	Mtentu	Mzintlava	Mntafufu	Mzimbuvu	Mtakatye
Lantana camara L.	Lantana						√	√
Sesbania punicea (Cav.) Benth.	red flower "Rattlebox"	√				√	√	
*Senna didymobotrya (*Fresen.) Irwin & Barneby	Peanutbutter cassia							
Psidium guajava L.	Guava					√		
Opuntia ficus-indica (L.) Mill.	Prickley pear							√
Arundo donax L.	Giant reed - declared weed						√	
Chromolaena odorata (L.) R.M. King & H. Rob	Jack-in-the-bush							√

Alien invasive species	Common names	Mdumbi	Mthatha	Bulungula	Xhora	Mbashe	Nqabarana/ Nqabarha	Nxaxo/ Ngqusi
Lantana camara L.	Lantana	√	√	√	√		√	
Sesbania punicea (Cav.) Benth.	red flower "Rattlebox"							
*Senna didymobotrya (*Fresen.) Irwin & Barneby	Peanutbutter cassia		√					
Psidium guajava L.	Guava							
Opuntia ficus-indica (L.) Mill.	Prickley pear		√					√
Arundo donax L.	Giant reed - declared weed							
Chromolaena odorata (L.) R.M. King & H. Rob	Jack-in-the-bush	√		√	√		√	

APPENDIX 20: The past and present salinity ranges (PSU) for the Transkei estuaries.

ESTUARY	Colloty *et al.* (2000)			Present (2012) middle reaches			Present (2012) Lower Reaches at mouth		
	Average PSU	Minimum PSU	Maximum PSU	Average PSU	Minimum PSU	Maximum PSU	Average PSU	Minimum PSU	Maximum PSU
Mnyameni	13	2	21	x	x	x	32	28	35
Mtentu	13	11	16	x	x	x	x	x	x
Mzintlava	x	x	x	19	6	31	19	6	318
Mngazana	x	x	x	27	22	32	36	36	36
Mntafufu	17	15	21	24	10	31	26	26	26
Mzimbuvu	23	12	36	9	5	16	28	28	28
Mtakatye	6	5	10	17	9	25	29	28	30
Mdumbi	5	2	8	20	2	31	30	30	30
Mthatha	14	11	19	21	13	28	31	31	31
Xhora	20	17	23	32	29	34	35	36	35
Mbashe	X	x	X	0	0	0	4	2	6
Nqabarha	X	x	X	30	28	32	33	32	34
Nxaxo/Ngqusi	15	12	18	x	x	x	34	34	34
Kobonqaba	18	15	21	x	x	x	20	7	35

No data were available for Mntamvuna, Mzamba, Mtentu (2012) and Bulungula estuaries.