REVISING THE DISTRIBUTION OF MANGROVE FORESTS IN SOUTH AFRICA AND CHANGES IN GROWTH OF MANGROVE SPECIES ALONG A LATITUDINAL GRADIENT.

A thesis submitted in fulfilment of the requirements for the degree of

MAGISTER SCIENTIAE

at

DEPARTMENT OF BOTANY

RHODES UNIVERSITY

by

UVIWE BOLOSHA

DECEMBER 2016

Abstract

Mangrove forests are a diverse assemblage of trees and shrubs that are adapted to a saline and tidally inundated environment. The global spread of mangrove species is affected by climate, with most confined to areas that are warmer and moist. At a global scale, temperature limits the distribution of mangrove forests but on a regional scale and local scale, rainfall, river flow and tidal exchange have a strong effect on the distribution and biomass of mangrove forests. Other factors that play a role in limiting growth and distribution of mangroves include accessibility of suitable habitats for growth and also conditions that are suitable for propagule dispersal. The objectives of this study were to review the distribution of mangroves in South Africa and determine the number of extreme temperature events that may limit further distribution and secondly to measure growth rates of mangrove species at Mngazana and Nahoon Estuary and the nutrient pools in the sediment of these forests. In 1982, Ward and Steinke published a list of estuaries where mangroves were present. The current study sampled the population structure, microclimate and level of expansion in two estuaries within and outside of the published range. The minimum, maximum temperature and number of extreme temperature events were measured using iButtons. Mangrove expansion will be limited by minimum temperatures (1.1 ° C) and an increase in extreme temperature events (830) (< 5 ° C and 5- 10 ° C) at the latitudinal limits. Expansion of *A. marina* at Kwelera and Tyolomnqa Estuary was evident but sapling survival was low. Mangroves have expanded within and outside the range proposed by Ward and Steinke (1982), but are limited by physical factors, restricted by the presence of saltmarsh and other estuarine macrophytes and natural disturbance regimes.

An increase in population growth was recorded over the years at both Mngazana and Nahoon Estuary. Nutrients, (ammonium, nitrates + nitrites and soluble reactive phosphorus) varied amongst sites and were related to seasonality. Nitrogen in both estuaries was available in the form of ammonium (NH4) and its concentration was generally higher (1.3-76.2 μm) than other forms of nitrogen (0.07-6.3 μm). Soluble reactive phosphorus (SRP) was higher during the wet seasons in both estuaries. An increase in porewater salinity since 2007 (41.3 practical salinity unit (PSU)) was measured at Mngazana and this is a result of freshwater abstraction and low rainfall. *A. marina* saplings and adults grew significantly faster at Nahoon Estuary (the distributional limit) (11.1 ± 1.1 cm year^{-1}) compared to Mngazana Estuary (5.3 ± 1.8 cm year^{-1}). Different mangrove species and forests respond differently to environmental factors and changes in mangrove distribution is expected in South Africa but changes are expected to happen slowly and opportunistically.

Keywords: Mangroves, climate change, expansion, porewater characteristics and growth

Contents

Acknowledgements

In order for this project to be a success, a funding was of importance. I am indebted to the following organisations for their financial support, thank you National Research Foundation (Thuthuka Grant) and Rhodes University for funding this project, much appreciated.

This project would not have been completed without my supervisor Dr A Rajkaran. I would like to thank her for her patience and kindness with me, her input and encouraging me into doing this research. You have made me fall I love with research, you have shown me that in spite of it all I can do it and you have I learnt so much about mangroves, all thanks to you. You were always there for me when I needed your help with anything, you were willing to listen to me and my problems. Allow me to say you are the best supervisor ever.

I would like to acknowledge the Rhodes University, Department of Botany for being generous and allowing me to use their laboratories and equipment. I would also like to thank the following departments for the help with my analyses, Rhodes University Department of Geography for helping me with particle size and allowing me to use their expensive Mastersizer machine. I would also like to thank the Nelson Mandela Metropolitan University Botany Department or their help with nutrient analysis.

I would also like to acknowledge my colleagues and field work sampling buddy, Phumlile Cotiyane… I would not have done it without your help buddy, you were always there for me during the long hot and cold sampling days, helping me with my work, encouraging me and giving me hope. Thanks friend. And to my other friends, thanks guys for not being tired to listen to my complaints, and I also thank you for spoiling me and making me laugh during days of struggle and pain.

To my parents, thank you for the chance and opportunity you gave me to study. You never got tired of my endless studying, you never forced me into working, and you never put pressure on me. You took care of my angel without complaining, you were always there for my throughout my financial shortages. Thanks mom and dad.

To my angel Lilo, you are the one that kept me going. You are the reason I never gave up on studying, you are the reason I always had a smile on my face, whenever I felt too much weight on my shoulders I knew when I see and speak to you I will feel much better.

Saving the best for last……Thank you God Father for everything you have done for me, if it weren't for you all of this would have not been possible.

"I will lift up mine eyes unto the hills, from whence cometh my help.

My help *cometh* from the LORD, which made heaven and earth". (Psalms 121:1-2)

List of Figures

List of Tables

Tables of Plates

Chapter 1: General introduction

The term 'mangrove' describes either an individual or the entire ecosystem (Tomlinson 1986). Mangroves are a group of taxonomically diverse flowering plants that are salt-tolerant and grow predominantly in sub-tropical and tropical regions of the world (Ellison and Stodart 1991). Ellison (2008) states that mangroves have fossil records that date back to the Cretaceous era, that is, at least around 69 million years ago, and their evolution coincides with that of the Angiosperms when the southern super-continent of Gondwana broke apart.

Mangrove forests are a diverse collection of woody plants and trees that have been adapted to a saline and tidally-inundated environment (Wilkie *et al.* 2003). Kristensen *et al.* (2008) states that these mangroves trees are the only halophytes that are found along the tropical and subtropical coasts. Sharma *et al.* (2012) and Saintilan *et al.* (2014) state that mangroves are exclusively tropical, but there are some outliers stretching into temperate zones, particularly in Australia and southern Japan, South Africa and southern Florida. Mangroves generally grow in mud and in environmental settings that are harsh and variable such as high temperature, high sedimentation, high salinity and extreme tides. In order for them to cope in their environment, they evolved a variety of morphological and physiological adaptations (Lugo 1998). Mangroves spread in the intertidal region between the land and sea at the sub-tropical and tropical areas of the world between about 30°N and 30°S (Giri e*t al.* 2011). In the subtropical and tropical regions, they are considered areas that are important biologically, and play a role in providing food and protection for a diversity of organisms (Duke *et al.* 2007).

Duke (1998) stated that mangrove distribution can be based on four geographical scales, namely: their occurrence worldwide, their coastal range within each region, their upstream location within an estuary and their position along the intertidal view. The global distribution of mangroves is affected by climatic factors such as rainfall and temperature (Duke 1992). The spread of mangroves is affected by climate, with most confined to areas that are warmer and moist. Air and water temperature are the main factors that limit the expansion and the growth of mangrove trees, and substantial variation can be found between mangroves species in their vulnerability to changes in temperature (Field 1995, Alongi 2009, Godoy and De Lacerda 2015). However, water temperature in comparison seems to be the most important since some mangrove forests have developed in temperate areas, but only where warm ocean currents are found (Berjak *et al.* 1977).

Generally, temperature limits the distribution of mangroves, however, on a regional and local scale, rainfall, river flow and tidal exchange have a strong influence on the distribution and biomass of mangrove forests (Alongi 2009). Low temperatures limit latitudinal range of different species, while areas that have high rainfall ranges and high runoff from riverine catchments have greater number of species (Duke *et al.* 1998).

The spatial distribution of mangroves is typically influenced by the physiological tolerance of each species to salinity. *Avicennia marina* (Forrsk. Vierh). is usually located at the estuary mouth of most estuaries and it reaches the upstream tidal limits. On a local scale, mangrove trees that are closer seaward become larger and their productivity higher than those in the middle of the forest. According to de Lange and de Lange (1994), accessibility of suitable habitats for growth and conditions that are suitable for propagule dispersal determine the distribution of mangroves. Soares *et al.* (2012) states that another factor limiting mangrove distribution is the cold current that is found in lower latitudes; it limits the dispersal of propagules since propagules cannot endure low temperatures. This is exemplified by the limited expansion of mangroves forests on the west coast of Africa, Oceania and South America.

Ward and Steinke (1982) and Hoope-Speer (2015) presented Nahoon Estuary (33° S) as the southern limit of mangrove forests in South Africa. The three mangrove species *(Avicennia marina, Rhizophora mucronata* Lam and *Bruguiera gymnorrhiza* (L.) Lam.) found in this estuary were transplanted from Durban Bay (29°53′22″ S, 31°00′43″ E) by Professor Steinke in 1969 and various saltmarsh species (*Triglochin* spp., *Sporobolus* spp., *Sarcoconia* spp., and *Juncus* spp.) also occur here naturally. Of the three mangrove species transplanted, Only *A. marina* expanded significantly within the estuary (Saintilan *et al.* 2014). *Rhizophora mucronata* and *Bruguiera gymnorhiza* species also expanded but only a few trees are found in the forest.

Maintaining a long-term monitoring dataset for the growth of mangrove is of importance in mangrove ecology. A long-term dataset of growth rate data assists in predicting changes in growth rates as climate parameters such as minimum and maximum temperature and rainfall begin to change.

We proposed a working hypothesis that expansion is determined by; climate, increases in atmospheric carbon dioxide concentrations, propagule production and suitable habitat while growth is expected to be related to nutrient dynamics (i.e. nutrient availability and concentrations). Therefore this study aims to review mangrove expansion in South African estuaries (Gonubie - Keiskamma Estuary) and describe the importance and occurrence of mangrove expansion as well as investigate the correlation between growth rates and the likely drivers for growth of mangrove trees in the temperate and subtropical regions (Mngazana Estuary and Nahoon Estuary), of South Africa.

The objectives for the review and description of mangrove expansion in South Africa are:

1. To determine changes in the microclimate of two mangrove forests in South Africa and to relate this to biological characteristics.
2. To revise the distribution of mangroves in South Africa.
3. Determine how the spatial and temporal distribution of mangroves has changed overtime
4. To determine and compare propagule weight (maternal investment) at mangrove forests along a latitudinal range.

The objectives for the investigation into the correlation between growth rates and the likely drivers for growth of mangrove trees in South Africa are:

5. To compare the growth rates of mangroves at two forests within the current distribution range and at the edge of the range.
6. To determine the nutrient pool in the pore water of mangroves and to identify the main driver of growth for the mangroves at Mngazana and Nahoon Estuary.

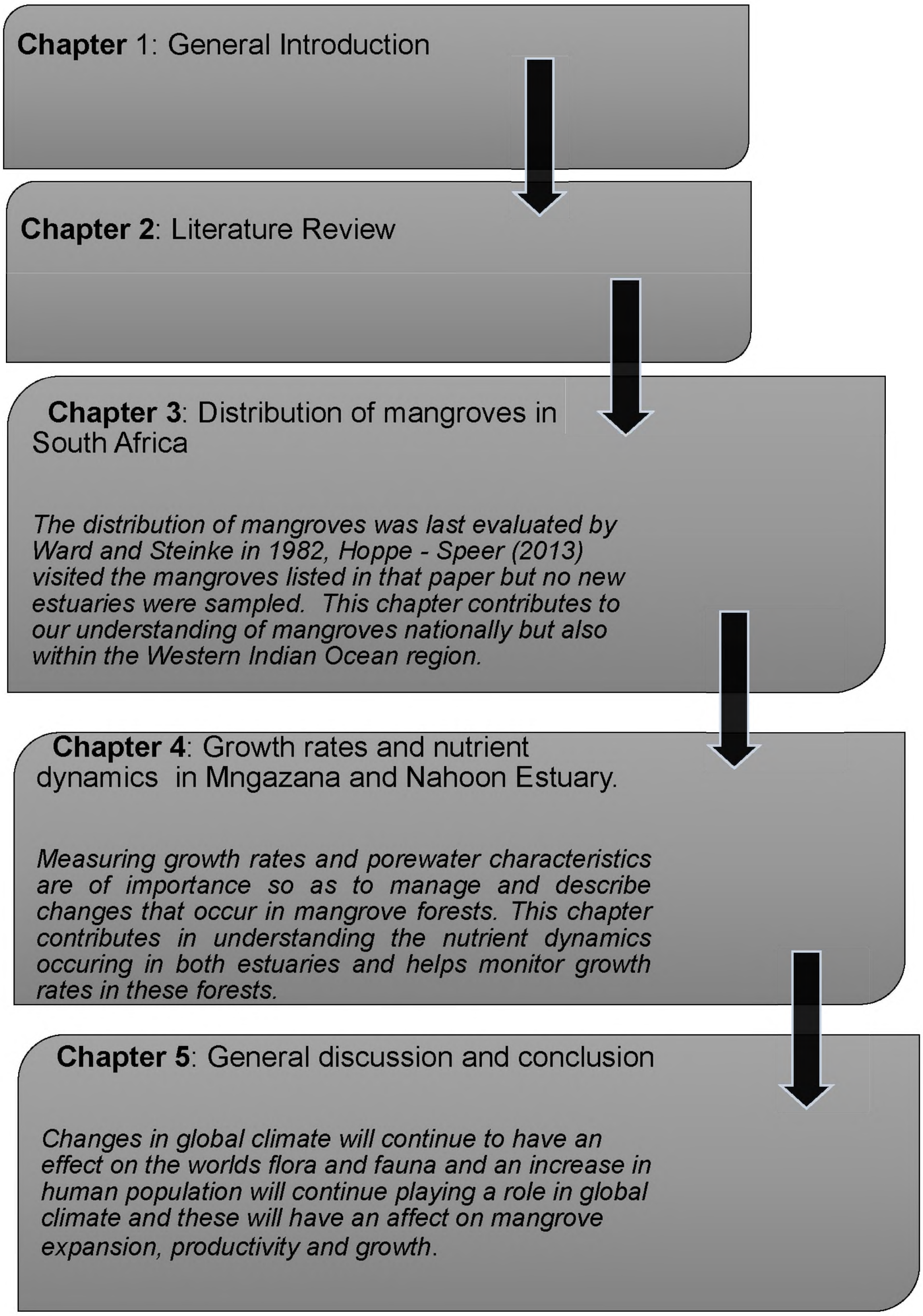
Layout of Thesis

Chapter 2, is the literature review section, a broad description of estuaries is discussed and estuaries in South Africa will be classified according to Whitfield (1992) and the importance of freshwater in estuaries will be described and explained. Hydrological regimes are discussed together with the distribution of mangroves as well as the important drivers for growth in mangroves. Without this understanding, we cannot predict how mangroves will respond to changes.

Chapter 3, re-evaluates the distribution of mangroves in South Africa and explains the causes and limitations of mangrove spread. Four study sites were used in this chapter in conjunction with temperature measurements to describe the influence of this factor on mangrove expansion.

Chapter 4, describes the growth and mortality rates of mangroves at two estuaries in South Africa. The availability of nutrients (ammonium, nitrates + nitrites and soluble reactive phosphorus) and other porewater characteristics such as salinity, pH and temperature are also assessed.

Chapter 5, the final chapter, discusses the final outcomes of the research and provides an explanation of how nutrients, climate and other factors influence the growth, productivity and spread of mangroves in South Africa.



Chapter 1: General Introduction

Chapter 2: Literature Review

Chapter 3: Distribution of mangroves in South Africa

The distribution of mangroves was last evaluated by Ward and Steinke in 1982, Hoppe - Speer (2013) visited the mangroves listed in that paper but no new estuaries were sampled. This chapter contributes to our understanding of mangroves nationally but also within the Western Indian Ocean region.

Chapter 4: Growth rates and nutrient dynamics in Mngazana and Nahoon Estuary.

Measuring growth rates and porewater characteristics are of importance so as to manage and describe changes that occur in mangrove forests. This chapter contributes in understanding the nutrient dynamics occuring in both estuaries and helps monitor growth rates in these forests.

Chapter 5: General discussion and conclusion

Changes in global climate will continue to have an effect on the worlds flora and fauna and an increase in human population will continue playing a role in global climate and these will have an affect on mangrove expansion, productivity and growth.

Chapter 2: Literature review

2.1 Estuaries

According to Miththapala and Sriyanie (2013) estuaries are where rivers and the sea meet and Barrett (1992) states that they are the most biologically productive ecosystem. The word, estuary, is derived from the Latin word *aestuarium* which means 'tidal inlet of the sea'.

Estuaries form a transition zone between river and maritime environments (McLusky and Elliott 2004) and Wolanski *et al.* (2013) states that an estuary in never found in a steady state as the bathymetry of these ecosystems are ever changing. Estuaries are typically classified based on their geomorphology, stratification and water balance (Valle-Levinson 2010 and Wolanski *et al.* 2013).

2.1.1 Geomorphology

Estuaries can be classified based on their geomorphological features (Valle- Levinson 2010):

Coastal plain/Estuarine Lakes

These types of estuaries are formed when sea levels rise and fill an existing river valley. These are common in areas with temperate climates. Examples of the estuaries are Chesapeake Bay, Tampa Bay, Hudson River, Swart Vlei and St Lucia Estuary (Kuneke and Palik 1984).

Bar built estuary/Temporarily open/closed estuaries

These are found in areas where there is a smaller watershed, and where deposition of sediment has occurred. They are shallow and are separated from the sea by sand bars (Heady *et al.* 2015). Bar built estuaries are mostly found along the Atlantic and Gulf coasts of the United States in areas where active coastal deposition occurs and where tidal ranges are less than 4m (NOAA 2010). In South Africa, TOCEs have a low to moderate tidal prism or in some cases no tidal prism; examples of these kinds of estuaries are Kasouga, Old Woman's and Mtati Estuaries (Colloty *et al.* 2002).

Fjord type estuary: These estuaries are those that are created by glaciers. These moving glaciers cut deep valleys and thus form these estuaries. These estuaries are shallow at the mouth where glacier rocks obstruct water flow and they are limited to higher latitudes where glaciers are found. Fjord estuaries can be found along the coasts of Iceland, Alaska, and Eastern Canada amongst others (Perillo 1995).

Tectonically-caused estuary

These are estuaries that are created by tectonic processes which include volcanic eruptions, earthquakes, faulting etc. that result in the shifting and drifting of the earth's crust. Very few of these estuaries are found; an exampled of one can be found in San Francisco Bay. (Kampf 2010).

2.1.2 Stratification

Classification by stratification makes use of the competition between buoyancy, i.e. forcing from river discharge, and mixing from tidal forcing to define estuaries. A salt wedge estuary, for example, is highly stratified and large freshwater inputs are observed. Comparatively, strongly stratified estuaries have a clear stratification of fresh and salt water; the flow of freshwater is stronger than the influx of saltwater. Weakly/vertically-stratified estuaries have strong tides, weak to moderate freshwater discharge and saltwater and freshwater are completely mixed (Valle-Levinson 2010). Salinity structure differs along the estuary and that results in different water columns and stratification. Under some conditions the Orange River (South Africa) has a blocked connection to the sea by sandbars, therefore it is not considered a true estuary but rather a river mouth (SAWCP 2009), therefore the river usually dominates the estuarine physical processes, sea water seldom penetrates upstream areas.

2.1.3 Water balance

Estuary classification is based on the balance between volume inputs (river discharge and precipitation) and river outputs (evaporation and freezing). Water balance is classified into: positive estuaries, negative or inverse estuaries.

Estuaries with positive water balance are those with high river discharge that exceeds evaporation. Those with inverse water balance are found in arid regions; loss of freshwater by evaporation is greater than freshwater addition by precipitation and low or no river discharges occurs in these estuaries (Wolanski 1986; Nunes and Lennon 1986). Estuaries with low inflow, i.e. negative estuaries, are typically found in regions where there is high evaporation and low river discharge; high and dry temperatures may result in high salinity ranges.

2.2. South African estuaries

The coastline of South Africa expands from the mouth of the Orange River (west coast) (29° 35' S, 16 °29' E, to the Kosi Bay (east coast) 26°54′13″ S ,32°52′25″ E (Figure 1b). According to Teske *et al.* (2006), it can be divided into biogeographical provinces that are distinguished by dissimilar hydrological conditions and species assemblages. The three main regions are the temperate cool west coast, the temperate warm south coast and the subtropical east coast. Many estuaries, which are found along the coastline of the temperate region in South Africa, are microtidal and as stated by Allanson (2001) they usually have an entrance channel that is shallow and small and it leads into a large basin. They also have sand bars that usually form at their mouths and this block the exchange of water between the estuary and the ocean.

Estuaries are bodies of waters where freshwater from rivers and streams combine with sea water, their environments are diverse with different types of fauna and flora. Elliot and Whitfield (2011) define an estuary as a body of water that can be connected to the sea either permanently or partially, or it can be disconnected from the sea by a sandbar, thus resulting in lakes or lagoons. Estuaries have different salinity ranges, which depend on freshwater inflow and factors like tidal mixing. During flood events, an estuary becomes a river mouth due to high freshwater flow rate that blocks the flow of sea water. Estuaries are important and essential systems, they provide favourable conditions for recreation, provide habitat and food resources for economically and ecologically important fish (Yoskowitz *et al*. 2010). There are 300 functional estuaries in South Africa and they make up ~70 000 ha of one of South Africa's habitats that are most productive (Turpie *et al*. 2002 and NBA 2011). They have been estimated to be worth R153 000 per hectare (ha) annually, R3 500 per ha in food production, R2 550 per ha in recreation and R141 000 per ha in nutrient cycling (Lamberth *et al*. 2003). At Mngazana, the third largest estuary in South Africa, the lowest minimum value of mangroves was evaluated to be about R1.1 million and the highest minimum value was evaluated to be about R13.6 million (De Wet 2004).

The future health of estuaries in South Africa depends on two principal components: their immediate management, quality and the quantity of freshwater (Turpie *et al*. 2002). The overall estuarine health rating for Mngazana was 7 out of 9 and was described as being in moderately good condition based on the diverse flora and fauna present in the estuary that makes it possible for ecological processes to take place (Harrison *et al*. 1996). Mngazana Estuary might be exposed to environmental distress and other factors if it is not properly managed. These can be as a result of nutrient loading that can occur due to agricultural practises, hydrological modification and changes to land use. The effects of this distress often result in harmful changes in the structure of the ecosystem and this may alter or compromise the goods and services that it provides. The overall health status of Nahoon Estuary was considered to be fair and it had a very good hydrodynamics, but water quality was rated as being moderate to poor (Cotiyane 2016).

The rising impact of human activities in estuarine and marine environments on a global scale, led to the need to assess, manage and monitor estuarine quality and health (Karr 1999) so as to improve and, ultimately, keep it functioning well (Borja *et al*. 2008a). Valiela *et al.* (2001)

mentions how the deterioration of estuarine and coastal ecosystems as a result of human activities is increasing; 29% of sea grasses, 30% of coral reefs, 35% of mangroves and 50% of salt marshes are degraded or lost worldwide. This worldwide decrease in mangroves is known to have effects on important ecosystem services (Worm *et al*. 2006), since they serve as nursery habitats, and areas of filtration and detoxification increasing water quality. In South Africa, human impacts on ecosystems have been documented by the Department of Environmental Affairs where hydrological patterns are altered by human activities and an increase in land use and waste disposal has been recorded.

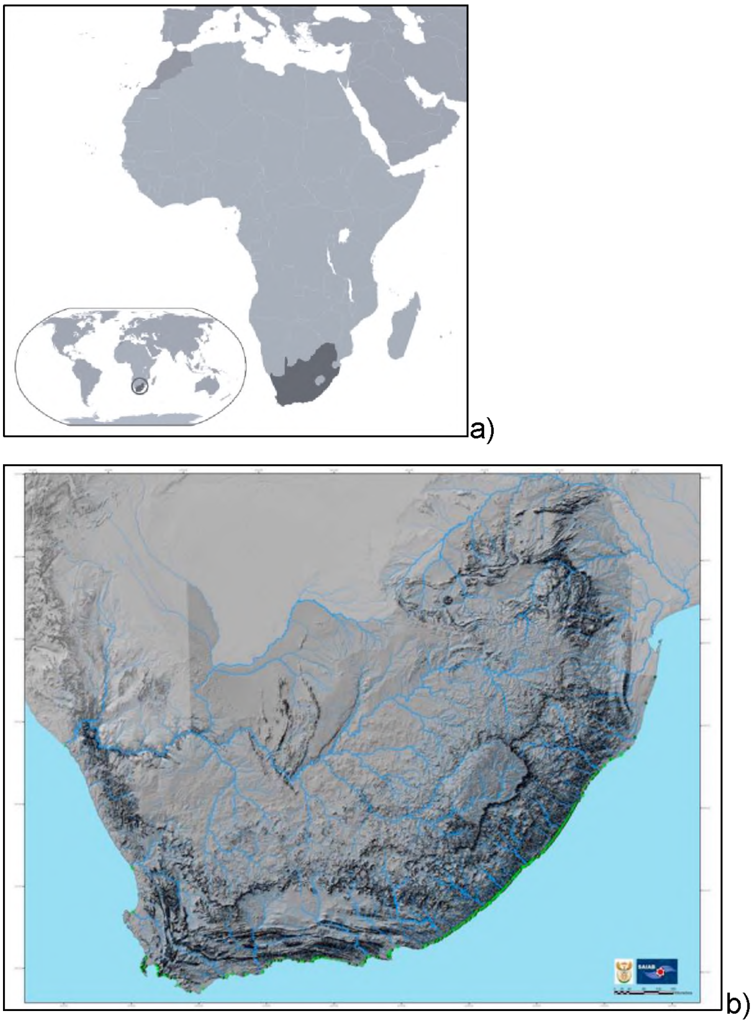


Figure 1 a) Location of South Africa on a world map b) the location and abundance of estuaries found in this country (source: Department of Water Affairs and Forestry).

2.2.1 Classification of South African estuaries

Whitfield (1992) classified estuaries of South Africa into five categories based on the catchment area, tidal prisms, salinity gradient and mixing processes (Table 1). Three hundred estuaries were identified in South Africa by Whitfield (1992), but Turpie et al (2002) identified

255 functional estuaries, therefore system that did not function as stuaries were not considered.

Table 1: Physical parameters used to classify estuaries in South Africa by Whitfield (1992)[1] and Turpie *et al*. (2002)[2].

Type	Number of this type of estuary in South Africa[2]	Mixing process[1]	Average salinity[1] *
Estuarine Bay	4	Tidal	20-35
Permanently open	47	Tidal/riverine	10- >35
River mouth	12	Riverine	<10
Estuarine Lake	8	Wind	1- > 35
Temporarily closed	184	Wind	1->35

* Total amount of dissolved solids in water in parts per thousand by weight (seawater = ~35)

Permanently open estuary

Hyper saline conditions in this type of system occur in the upper reaches when there is drought. Water temperature is managed by the ocean during periods of regular conditions and by river input during flood conditions. Saltmarshes together with other macrophytes are prevalent e.g Mngazana and Nahoon Estuary (EC).

Temporarily open estuary

Within these systems, periodic formation of sandbar (berm) occurs across the mouth, which is strongly linked to rainfall patterns. Hyper saline conditions tend to prevail in these systems during periods where there is drought. The tidal and riverine loads manage water temperature during open mouth conditions, but are self-supporting when the mouth is closed. E.g Mhlanga and Mdloti Estuary (KZN).

Estuarine lake

These systems can either be permanently or temporarily connected to the ocean, their salinity can be extremely variable. They have small tidal prism and the river input has little influence on the water temperature. Organism found depends on the salinity condition of the system e.g. St Lucia (KZN).

Estuarine Bays

These systems are similar to permanently open systems except with a greater marine influence. Hyper saline conditions are not frequent in this system and water temperature is capable of being influenced by the ocean. Organism can occur here and substantial mangrove swamps and wetlands occur e.g. Kosi Bay (KZN).

River mouth

In these systems, the estuarine entrance is normally open and they have a small tidal prism and a strong riverine outflow that blocks marine intrusion. Oligohaline conditions and shallow mouths conditions as a result of heavy silt loads often occur. The water temperature is influenced by river inflow e.g. Orange River Mouth (EC).

2.2.2 Freshwater requirements of estuaries in South Africa

Freshwater in estuaries is an essential component in order for the system to function properly. Jezenski and Roberts (1986) estimated that the total South African freshwater required by estuaries is $2160\times10^6 m^3$ per annum, which represents approximately 4% of mean annual runoff (MAR). Costanza *et al.* (1993) stated that the primary factor that determines the degree to which estuaries are vertically stratified with the lighter freshwater near the surface and the salty denser water near the bottom is the amount and flow rate of freshwater entering the system.

Once there is a decrease in the flow of freshwater into the estuary, extensive changes may occur; at times, they may even be disastrous. The modification of freshwater inflow can result in the change of the hydrodynamics of the system. The reduction in discharge can result in an increased effect of the tide on the kind of circulation, such that a stratified system with well-advanced gravitational distribution can change to a well-mixed system where tidal change increase is of significance (Alber 2002). Freshwater supply abstraction occurs in St Lucia Estuary, there is no tidal exchange and low water level has been recorded. Recently, about 90 % of the water area disappeared in the estuary and that will obviously have an effect on the flora and fauna of that estuary (Whitfield and Taylor 2009). Another consequence of changes in the input of freshwater is that salty water can trespass further upstream and this causes an increase in salinity along the estuarine gradient. In instances where there is increased evaporation taking place and low precipitation, the estuary can become hyper saline as shown by Whitfield and Woolridge (1994) in the Kariega Estuary. There was drought for more than a year and this resulted in no fresh water inputs in the estuary. There was more than 40 PSU salinity recorded in the uppermost range of the estuary. Lengthening of an estuary can also occur where a decrease in outflow leads to the enlargement of the transition zone from zero salinity to full sea water (Alber 2002). A decrease in freshwater flow in the mangrove system may lead to an increase in salinity and this has an effect on the growth and productivity of mangroves. Castaneda–Moya *et al.* (2006) stated that the distribution of mangroves in a forest is regulated by dry saline soils.

2.3 Factors influencing mangrove distribution and growth

Generally, species occupy a distinctive geographical range. Some have geographical ranges that are large and stretch over various continents, while other species exhibit more restricted ranges. Abeli *et al.* (2014) noted the effects of marginality on the performance of plant populations and stated that plants found at the edge of their distribution tend to be smaller as compared to those found in central populations and their success in completing the life cycle may be lower. Plant distribution around the world has been believed to be controlled primarily by climate and soil and as a result changes in climate should guide changes in plant distribution (Richardson and Bond 1989). Pressey *et al.* (2006) speculated that climate change will be one of the vital drivers of biodiversity patterns in the near future. Species seem to be shifting their ranges in both altitude and latitude as a reaction to changes in climate (Parmersain and Yohe 2003, Bellard *et al.* 2012, Beale *et al.* 2013 and Hoppe-Speer *et al.* 2015). An example of this is the northward expansion of pine forests in Europe due to changes in climate (Huntley and Birks 1983). Similarly, Kruger *et al.* (1986) also showed how changes in climate had an effect on limiting the distribution of *Pinus radiatia* (D. Don) over the impact of pathogenic activity in the northern parts of South Africa. Various authors (Hilbish *et al.* 2010, Lenoir *et al.* 2010 and Groom 2012) stated that it is of importance to understand how species ranges will change due to variations in climate.

Mangroves are found throughout the tropical, subtropical and temperate regions of the world, with diversity peaking at about 90 species. The determinants of their distribution are complex and varied and they act over a range of spatial and temporal scales (Manson *et al.* 2003). Duke *et al* (2002) stated that there is absence of mangroves where cold waters are forced towards the equator by western continent current and ocean circulation patterns. Conversely, warm currents on the eastern continental shorelines allow mangrove populations to expand into sub-tropical latitudes. Temperature appears to be what is limiting the overall distribution of mangroves (Saintilan *et al.* 2014). According to Saenger and Moverly (1985); Hutchings and Saenger (1987) nearly all mangrove species, with the exception of *A. marina*, generate maximal shoot density when the mean air temperature increases to 25 °C and they cease to produce leaves when the mean air temperature decreases to 15 °C. Some mangrove species may show a reduction in leaf formation rate. In subtropical areas, especially those that are near the latitudinal limit of mangrove distribution, cold temperatures that are less than 5 °C (freezes/frost) may result in damage or even kill mangroves trees (Liu *et al.* 2014) e.g. Mangroves have been sporadically damaged and killed in Texas, Louisiana and Florida as shown by Ross *et al.* (2006), and Steven *et al.* (2006). According to Stevens *et al.* (2006) *Avicennia germinans* (Author) in North America has a distribution that can be expanded during periods where it is not freezing or when salt marsh vegetation is killed. A study by Cavanaugh *et al.* (2013) at the east coast of Florida from 25 °N to 30 °N showed that mangroves expanded poleward across the east coast of North America. This expansion was related to global warming and it was not due to increases in mean air temperatures, but it was caused by decreasing frequency of individual cold events.

Plant distribution is determined by a variety of complex environmental factors. Many of these factors are known to have an effect on mangrove distributional patterns and growth (Perera *et al.* 2013). Connolly and Lee (2007) and MacIntosh (2002) mentioned the key factors that have an effect on mangrove growth and distribution, and these include temperature, salinity, fresh water supply and soil type. Smith (1992) stated that animals also play a role in influencing mangrove distribution, that is to say, they complicate distribution patterns. A study by McKee *et al.* (2002) together with a subsequent study by Lovelock *et al.* (2004) reported on the effect low nutrient values, which are caused by a decrease or no tidal flushing, have on mangrove growth. Saintilan *et al.* (2014) also mentioned various factors that hinder mangrove distribution and these include mouth closure, hard rock coastlines, unfavourable ocean currents, and the absence of estuaries with depositional environments suitable for growth.

There are two elements in mangroves; mangrove forests and water bodies associated with it (Selvan and Karunagaran 2004). Mangroves have exceptional physiological adaptations that make it possible for them to survive in their environment (Goudkamp and Chin 2006). Mangroves differ in their tolerance depending on the adaptation of that species (Ball 1998).

2.3.1 Temperature

Temperature changes may have an effect on mangrove phenology and productivity especially at subtropical latitude (Dahdouh-Guebas and Satyanarayana 2012). It is unknown whether small changes in temperature or the occurrence of extreme temperature events have a greater effect on mangrove productivity. Mangroves show significant difference in their responsiveness to temperature, but most generate maximum shoot growth when the average air temperature is at 25 ˚C. *Avicennia marina is* the only species that continues to produce leaves even when the availability of air temperature falls lower than 15 ˚C (Field 1995).

Mangrove growth can be inhibited by temperatures of about 5 ˚C (Tomlinson 1986). Ellis *et al.* (2006) stated that early death of leaves of *Laguncularia racemosa* (L.) C.F. Gaertn can occur as a result of frost. Stuart *et al.* (2007) in their study also revealed that frost causes embolism in the secondary xylem, and thus, can cause the death of mangrove trees. Mangroves that are found from Florida to Texas are exposed to recurrent frost. A limit in mangrove establishment, diebacks in mangrove stands and changes in aerial cover between saltmarsh species and mangroves can be caused by frost (Lugo 1996).

According to Deurwaerder (2012) the most favourable temperatures are not only restricted by cold temperatures, but also by high temperatures. Photosynthetic productivity of mangroves can be altered by an increase in temperatures, improving it in colder climates. Higher temperatures can also cause a decrease in net CO_2 assimilation in climates that are warm (Okimoto *et al*. 2007). They also may hinder the tree seedlings and result in seedling mortality. High climates may lead to events which include droughts which are more probable to have considerable effect on mangroves than gradual changes in average conditions.

There are mangrove forests that are found at the edge of their distribution (Figure 2). In Louisiana mangroves merge with temperate salt marsh species. *Avicennia germinans* populations disperse across the deltaic plain (29°N latitude) and they dominate creek banks. Mangrove in this area are sporadically damaged by freezes since mangroves are tropical species and likely to be affected by cold temperatures (Markley *et al*. 1982). There are other factors that may lead to the damage and death of mangroves, e.g. in a study by McKee *et al.* (2004) in the USA, a combination of events which include drought, low river outflow and low sea level led to the widespread dieback of salt marsh *Spartina alterniflora* Loisel.. One mangrove species *A. germinas* and a saltmarsh species, *Juncus roemerianus* Scheele. were not affected during the events as they were drought tolerant and this generated an opportunity for *A. germinans* to expand. In this case the events that took place were favourable for mangrove expansion. This may not be the case in other mangrove species that are sensitive to drought and salinity. Alongi (2008) clearly expressed that the observed data suggested an increase in photosynthetic rates in areas of high rainfall and river inputs such as Southeast Asia and there was a decline in photosynthesis of mangroves in regions that were arid such as parts of northern Australia.

Mangroves in New Zealand are also found at the southern edge of their range, but still their distribution is quickly expanding (Ellis *et al*. 2004). These temperate mangroves are found in estuaries, harbours and creeks north of 38°C. Lovelock *et al.* (2010) stated that mangrove growth occurred in estuaries that previously had coarse sand sediments and had no vegetation, and this was a result of increased sedimentation which is caused by land use change that is related to agricultural development and urbanization. Extensive mangrove forests that had developed at the southern latitude (Firth of Thames) were linked to changes in wind and wave energy (Lovelock *et al.* 2010). According to Gladstone-Gallagher *et al.* (2014) New Zealand has a single species, *A. marina* subspecies *australasica*, and they are stunted in stature. With its capacity to reproduce and grow in various climatic, tidal and edaphic conditions, *A. marina* tends to be found in diverse ranges of littoral habitats and they show considerable fluctuation of growth form (Duke *et al*. 1998). This species can be found in estuaries ranging from Ohiwa Harbour to Northland. Mangrove populations expands near pole-ward limits (Saintilan *et al*. 2014).

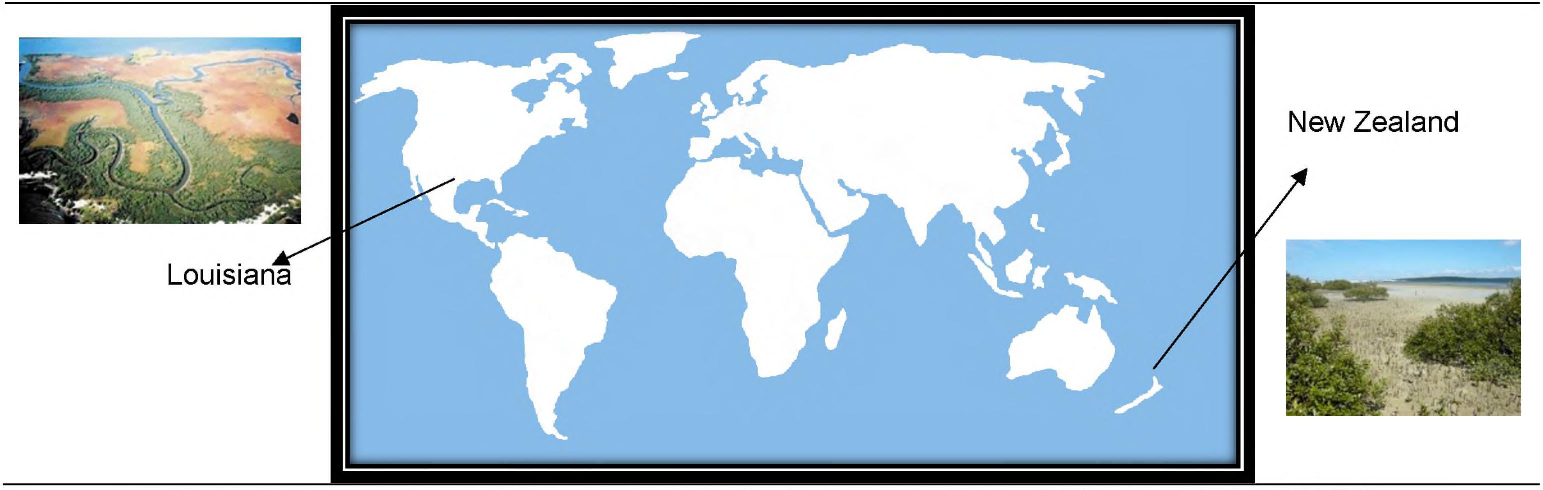


Figure 2: Mangrove populations at the latitudinal extremes of their distribution (USGS, NWRC 2004)

2.3.2 Atmospheric CO_2 Concentrations

According to the IPCC AR4 (2007), atmospheric CO_2 and other greenhouse gases are increasing due to human activities. This is as a result of the nature of CO_2 in the atmosphere. Atmospheric CO_2 has increased from 280 parts per million by volume (ppmv) in 1880 to almost 370 ppmv in 2000. An increase in atmospheric CO_2 by 2 ppmv yr^{-1} was observed by Conway and Tans (2012), and they recorded an annual global mean of 391 ppmv in 2011. Currently, the CDIAC (2016) measured an atmospheric CO_2 of 399.5 ppmv. Hunt *et al.* (1991) stated that plants that are exposed to high atmospheric CO_2 levels tend to be larger than plants that are grown in medium CO_2 concentrations. The consequences of growth stimulation depend on sink strength, phenotypic plasticity, photosynthetic pathways and life history strategies. Elevated CO_2 levels may result in increased productivity of mangroves (Field 1995), alter phenological systems, enlarge mangrove ranges into higher latitudes where lower temperatures would not allow that to occur (Ellison 2000). The metabolic reaction of mangroves to increased atmospheric CO_2 quantity are also likely to increase growth rates. A study by McKee and Rooth (2008) in the northern Gulf of Mexico showed that *Avicennia germinans* increased in growth and biomass production under elevate CO_2 treatments. CO_2

does not influence mangrove growth if salinity is also high, as species need to facilitate the uptake of water to ensure growth. According to Snedaker and Araujo (1998) increases in CO_2 globally may lead to a competitive advantage of mangroves in arid regions. This is caused by their capacity to reduce water use throughout the periods where there is water stress while keeping elevated rates of CO_2 uptake. Gilman *et al.* (2008) stated that the consequences of increased CO_2 on mangroves are poorly understood and there is a need for further research in this area.

2.3.3 Changes in rainfall patterns and salinity

Solomon *et al.* (2007) predicted that rainfall increases in higher latitudes and decreases in areas that are closer to the equator, therefore leading to subtropical regions experiencing low rainfall. Adjustment in rainfall patterns are expected to have an effect on the growth and spatial distribution of mangroves (Ellison 2001). When precipitation decreases it results in low mangrove productivity, mangrove growth and endurance of seedlings and may shift species composition favouring more salt tolerant plants (Ellison 2004). Duke *et al.* (1998) stated that high rainfall shorelines support more diverse and taller mangroves in Australia.

A decrease in rainfall might result in increased salinity which in turn affects the growth of mangrove trees. Sea level rise will also play a role in increasing salinity in mangrove forest. Krauss *et al.* (2008) states that mangroves function better between 5 and 75 PSU. Most mangrove species attain their maximal growth potential under brackish surroundings (modest salinity) between 10 and 20 PSU (Doyle 2003). Salt tolerance appears to vary between species, in a review Clough (1992) showed that *A. marina*, *Bruguiera gymnorrhiza* and *Rhizophora mucronata* are salt tolerant, but the latter is less tolerant compared to the other two mangrove species. *A. marina* has an optimal growth rate that occurs at a salinity of 0-30 PSU and has a maximum porewater salinity of 85 PSU; *R. mucronata* can endure a maximal salinity of 40 PSU but maximum growth occurs between 8-33 PSU. The maximal porewater salinity of *B. gymnorrhiza* is 50 PSU and maximum growth occurs between 8-34 PSU (Robertson and Alongi 1992).

2.3.4 Nutrient availability

Nitrogen and phosphorus are the macronutrients identified as those that may limit primary productivity of mangroves. Most studies have focused on them (Reef *et al.* 2010) (Table 2). Nutrient availability in mangrove forests is administered by a variety of factors (abiotic and biotic) which include soil type, tidal inundation, redox status, elevation in the tidal frame, plant species, litter production and decomposition (Feller *et al.* 2003). Many mangrove soils have exceptionally low nutrient availability. A study by Dangremond and Feller (2014) showed that trees had a mean N concentration of 1.51% and 1.37% in the Caribbean and Pacific respectively, they also had P concentrations of 0.10% and 0.11% in the Caribbean and Pacific. Nutrient availability differs highly between mangroves and in mangrove stands. The deficiency of phosphorus has been set out to be a considerable factor that limits mangrove growth in forests that are dwarfed (Feller 1995). In Australia, mangrove sediments are phosphorus and nitrogen deficient and this has a negative effect on mangroves (Beadle 1953).

The availability of nutrients in mangrove forests is often associated with sedimentation (Alongi 2005). Tilman (1988) states that plant growth is increased by the enrichment of nutrients and this is achieved by lowering the distribution to roots comparable to aboveground tissues. An escalation in nutrient accessibility results in rapid growth rates which are related with an increase in allocation to leaf area respective to roots, also with a collection of changes in physiology that consist of increased hydraulic conductivity and photosynthetic rates, and decreased capability in nutrient resorption and benefit (Krauss 2008). However, autotrophic production can be activated by nutrient enrichment while plant growth can be negatively influenced by supplementing with fine sediments. In a study by Chiu *et al.* (1996) in Taiwan the nitrogen content measured was between 0.82 - 5.98 mg kg-1 soil in Chuwei Tidal compared to Kuanto and Pali (1.68 to 17.10 mg kg^{-1} soil; 1.44 - 8.91 mg kg^{-1} soil), respectively. It was noted to limited growth of *A. marina* in New Zealand and South Africa. In the tropics, phosphorus limited plant growth in high intertidal forest (Lovelock *et al.* 2007, Naidoo 2009).

It is vital to measure the dynamics of the early stages in the life cycles of mangroves to predict the species composition, distribution and structure of mangrove forests (Ha *et al.* 2003). Tomlinson (1994) stated that mangroves spread by sexual reproduction, so the conservation and the recovery of mangrove forests depend on the production of propagules, their dispersal, establishment and the recruitment of seedlings. Population growth of mangrove forest is driven by seedling recruitment and survivorship, and this regulates the quality of the crop and forest stand production (Srivasta and Ball 1984, Burns 1985). If trees producing propagules are continuously being harvested, then that will lead to low propagule outputs, which in turn will lead to low population structure and low seedlings. Various spatial and temporal processes influence mangrove establishment and growth, and these include tidal activity, hydroperiod (flood duration, frequency and depth), sediment characteristics, and stability of landform / geomorphology (Twilley and Day 1999; Krauss *et al.* 2006).

Mangrove forests that have more frequent tides and river inundation tend to be productive as compared to forests that are not (Twilley *et al.* 1997). Study by Milbrant *et al.* (2006) and Harris *et al*. 2008 also showed how tides had an effect on mangrove forests; their results showed that seedling and litter fall were higher in tidally unrestricted areas. Twilley (1986) and Delgado *et al*. (2006) have also documented mangrove growth and productivity due to hydroperiods. Study by Krauss *et al*. (2006) also showed how sediment characteristics (soil fertility) and hyrdoperiods had an effect on mangrove growth.

2.4 Hydrological aspects of mangrove ecosystems

Mangroves are found on the intertidal zone between low and high tide, in a broad spectrum of hydrological settings. Hydrology refers to factors that include tidal current, tidal circulation and water exchange and has an influence on abiotic (salinity, availability of nutrients, soil moisture) and biotic factors (propagule dispersal) (Dunham 2014; Loon *et al*. 2016). It is known to be the key factor in species distribution (Hughes *et al*. 1998). Knowing the hydrological regime of a mangrove ecosystem is critical to understanding its functioning, protection and maintenance (Calderon *et al*. 2014). Saenger (1982) stated that factors like river inflow, frequency and tidal flushing are important factors in the hydrology of mangroves forests. The exchange of water between the mangrove system and the near shore modifies nutrient and salinity concentrations, while the degree of water mixing and exchange is determined by the flow rate of incoming freshwater (Ngesi 2010).

According to Tan Le (2008) flooding frequency is the most important tidal factor that affects mangrove growth, less mixing of water results in high salinity levels in the estuary and thus affects growth and productivity of mangroves, by decreasing water uptake and reducing photosynthesis and transpiration (Parida and Jha 2010). Long periods of inundation of mangroves also lead to anaerobic conditions that influence the root system (Mbense *et al.* 2016).

Table 2: A summary of porewater nutrient concentrations from various estuaries where mangroves are present.

Estuary	**Nitrogen**			**Phosphorus**	**Annotation**	**Reference**
Shark River Estuary, Florida	NO$_2$ & NO$_3$ = < 1µm and < 0.5 U m respectively			PO$_2$ = <2 µm at all the sites	Nitrogen concentrations were lower as compared to other studies	Chen and Twilley (1999)
Gazi Bay, Kenya	NH$_4$ - 3 to 390 µm	NH$_3$ - 1 to 50 µm	NH$_2$ - < 0.1 to 2.3 µm	PO$_4$ - <0.05 to 1 µm	Mangrove sediments were nitrogen rich	Middelburg *et al.* (1996)
Mapopwe Creek, Chwaka Bay	NH$_4$ - 50 & 180 µm	NO$_2$ - 0 & 2.2. µm		PO$_4$ - 40 and 90 µm	Mangrove sediments are the source of ammonium and sink for soluble reactive phosphorus	Mohammed and Johnstone (2002)
Mangrove Forest of Everglades National park, South Florida	NO$_2^-$ & NO$_3^-$ < 0.5 µm across all seasons and years			SRP was higher during the wet season (1.2 ±0.1 µm) compared to the dry season (0.8 ± 0.3 µm).	Different nutrient concentration occurred over the seasons	Castaneda-Moya *et al.* (2013)
Tauranga Harbor,New Zealand (site Te Puna and Waikareao)	NH$_4$ - 2-9 µm	NH$_3$ increased from 3 µm at high tide to 74 µm at low tide in Waikareao		PO$_4$ ranged between 0.3-1.0 µm	Ammonium was the main form of nitrogen at Waikareao and nitrate was the main form of nitrogen at Te Puna	Santos *et al.* 2014
Firth of Thames, New Zealand	High concentration of NH4+	N present as ammonium with low concentrations of nitrate		Low PO$_2$	Low nutrient concentration due to demand by plants and microbial community	Lovelock *et al.* (2010)

Watson (1928) mentions how mangroves are classified hydrologically, where species are classified based on tidal regime, flooding frequency and elevation. Different mangrove species occupy different zones. Watson (1928) and Loon *et al.* (2016) stated that class 1 is too often inundated and that results in in lack of vegetation in that area. In class 2 only pioneer species are able to survive and these include *Avicennia spp.* and *Sonneratia spp.* as they are suitably adapted to hydrological conditions. Class 3 has favourable hydrological conditions for mangrove growth and various species are able to survive there, e.g. *Bruguiera spp.*, *Rhizophora spp.* and Ceriops *spp*. Class 4 is rarely inundated by the tides, therefore species of the genera *Bruguiera* and *Lumnitzera*, which possess features suited for these conditions, are able to survive. Class 5 is never inundated, therefore is species like *Phoenix paludosa* Roxb. are able to thrive there due to their ability to withstand various environmental conditions (frost, moist soils, sunny positions etc). (Table 3). Smith *et al.* (2002) mentions how zonation is allocated to various environmental parameters which include salinity, nutrient content, soil chemistry and type, predation, competition and physiological tolerances.

Table 3: Hydrological classification by Watson (1928).

Class	Flooded by	Height above datum (m)	Flooding frequency (days)
1	All high tide	2.44	56-62
2	Medium high tides	3.35	45-59
3	Normal high tides	3.96	20-45
4	Spring high tides	4.57	2-20
5	Abnormal tides	15	2

Four types of mangrove forests based on their hydrological regimes are classified by Lugo (1989), namely riverine mangrove, basin mangrove, scrub mangrove and fringe mangrove.

2.4.1 Riverine mangrove

These types of mangroves are flooded by river water as well as sea water. Salinity decreases during the summer season. They are influenced by large amount of freshwater entering the forests together with fluvial nutrients. Mangroves may reach a height of 18 to 20 m (Odum *et al*. 1982).

2.4.2 Basin mangrove

These forests occur inland and have a stunted growth. Their location channels the terrestrial runoff to move towards the coast, with a slow velocity water flow. Mangroves of this type may reach a height of 15 m (Odum *et al.* 1982) and species of the genera *Avicennia* and *Rhizophora* dominate here.

2.4.3 Scrub mangrove

These types of mangroves occur in areas where there is low nutrient availability, freshwater and inundation by tides is limited (Gilmore and Snedaker 1993). Individual mangroves rarely exceed 1.5 m in height (Odum *et al.* 1982). Species of the genera *Avicennia, Rhizophora* etc. are found.

2.4.4 Fringe mangrove

These types of mangroves receive a brunt of tides which are often full-strength seawater (Saenger 2013), they grow as a thin fringe along the coast and so are directly exposed to seawater. The maximum recorded height of these mangroves, which are dominated by *Rhizophora* species, is about 10 m (Odum *et al.* 1982).

2.5 Mangrove sediment as sinks of nutrients

Smith *et al.* (2012) describe an estuary as an open, compound system that undergoes various processes such as inputs of inorganic nutrients and organic matter that are deposited from land. The chemical composition, formation and physical properties of mangrove sediments are substantially controlled by long term geochemical, geomorphic and primary production (Chen and Twilley 1999).

Mangrove sediments have been regarded as important sinks for land acquired contaminants (heavy metals), carbon and nutrient sinks. They have been used to distinguish sources of pollution (Tam *et al.* 1995). According to Robertson and Alongi (1992), mangroves have the ability to absorb nutrients directly from the sediment and they accumulate the nutrients as biomass. The absorption level of nutrients in the sediment is substantially determined by microbial activities (Prasad and Ramanathan (2008). Various factors have been identified to play a role in determining the capacity of mangroves as sinks, these factors include rainfall and geomorphology settings. The latter includes physical forces that react on the edge of the water body and they also control processes. Prasad and Ramanathan (2008) further states that in riverine systems there is nutrient inflow and a strong out-welling which results in the system being the source of nutrients, while in tidal systems there is small-scale export thus resulting in the system being a sink of nutrients. Adame and Lovelock (2008) hypothesized that rain will increase the ability of mangrove forests to be sinks of nutrients and their results showed that the ability of mangroves to be sinks was increased during periods of rainfall.

Mangrove sediments have been found to be sinks for ammonium and Si and PO_4^{-3}. The latter was believed to be as a result of utilization by benthic microalgae and mangrove trees (Kaiser *et al*. 2015). Another study in Red River Estuary, Vietnam supported the fact that mangroves were sinks of nutrients. This was as a result of their study showing that mangroves were the net consumer of nitrogen and phosphorus (Wosten *et al*. 2003).

2.6 Keystone species influencing mangrove forests

Keystone species are defined as those species associated with activities taking place in a mangrove forest and they have a major influence on the ecosystem level parameters (Naiman *et al*. 1986). Crabs are known to be the most important organisms that have an impact on a mangrove forest, after microbial processes. The well-being of mangrove ecosystems is stated to be directly dependent on microbial function and dynamics (Bashan *et al*. 2000).

According to Sahoo and Dhal (2009), there are various types of bacteria found in mangrove forests whereby each has its own role in mangrove sediments. These include methanogenic bacteria, phosphate-solubilising bacteria, and enzyme producing bacteria, photosynthetic anoxygenic bacteria and nitrogen fixing bacteria. Nitrogen fixation is defined as the process of changing nitrogen into integrated arrangement, e.g. ammonium and organic nitrogen. Nitrogen fixation in mangrove forests is correlated with decomposing leaves and other litter fall (Sjoling *et al*. 2005). Restrictions by energy sources can have an effect on nitrogen fixation and the main factors that can limit nitrogen fixation are light intensity and water temperature (Sahoo and Dhal 2009).

In mangrove ecosystems, litter from trees (twigs, leaves and propagules) are the main source of nutrients and carbon (Rajkaran and Adams 2007; Reef *et al*. 2010). They become deposited on the forest floor (Alongi 2008) and crabs will utilize some of this litter and the other litter will be stored in crab burrows for further degradation. Moreover, this in turn leads to conserving nutrients within the mangrove forests. Processing of leaves by the crabs increases the rate of nutrient cycling since crabs can break down litter 75 times more than the rate of microbial decay. By that, high rates of productivity will be promoted (Robertson and Daniel 1989a). Other litter is exported from the mangrove ecosystem by tidal currents to other coastal areas. Bioturbation by crabs is also stated to lead to changes in soil chemistry and texture, changes in degree of anoxia and affects surface topography (Alongi 2009).

The absence of crabs in a mangrove forest lead to nutrient deficiency in the forest and thus affecting mangrove productivity and the life supporting capacity of the ecosystem. According to Youssef and Saenger, (1996) the presence of crab burrows in the forest helps in the aeration of the soil and this leads to the establishment of seedling in a less stressful environment than the normal mangrove soil that is anaerobic.

2.7 Factors affecting the population structure of mangroves and production in Mangrove forests

There are various factors that influence the functioning and structure of mangrove forests and they vary in correlation to global, regional and local scales over different time scales (Alongi 2002). Rajkaran (2011) stated that an understanding of the population structure of mangroves is essential and knowing the relationship with the current environmental characteristics is necessary to ensure conservation. Mangrove productivity is higher at latitudes that are low, areas of high temperature and high annual rainfall. As a result, high productivity might result in high qualities of material that is available to be exported (Adame and Lovelock 2011). Reef *et al.* (2010) stated that mangroves are highly productive in the world in terms of gross primary productivity (GPP). They store and fix notable amounts of carbon. Even though mangrove forests are carbon rich, they are also nutrient poor. The main portion of the GPP stored in mangroves is the net primary productivity (NPP) and it constitutes the portion of the GPP that remained after carbon loss by respiration is accumulated and deducted as biomass primarily in wood, leaves (above ground NPP) and roots (below- ground NPP) (Amarasinghe and Balasubramaniam 1992).

About one-third of primary production in mangrove forests is represented mainly by plant litter and this is usually exported to adjacent waters through creeks (Ngesi 2010). The mangrove ecosystem is said to be highly productive and they contribute through their litter fall (Hernandez *et al.* 2011). They have a gross primary production that is estimated at 3-24g C/m^{-2} day $^{-1}$ and net production that is estimated at 1-12gC/m^{-2} day $^{-1}$ (Lugo *et al.* 1976). Mngazana Estuary plays a great part in the production of organic carbon and it is an important source of mangrove litter and also POC for the adjoining marine environment. There is little or no DOC that is exported from it. In their study, leaf litter production was 2.4 ± 0.2 g m $^{-2}$ d $^{-1}$ in summer and it was 0.3 ± 0.1 g m $^{-2}$ d $^{-1}$ (Rajkaran and Adams 2007).

2.8 Importance of mangroves and the influence of mangroves on soil stability

Mangroves usually grow on muddy, waterlogged soil (Northland Regional Council 2005), they play a role in assisting resistance against soil erosion, and this is done in two ways through consolidation and increasing soil rigidity through underground roots. Mangroves are able to colonize recently deposited sediment, by so doing they help consolidate this sediment and this may encourage further sedimentation (Blasco *et al.* 1996). The capacity for further consolidation seems to differ from mangrove to mangrove. Consolidation is better under *Sonneratia* and *Avicennia* species as compared to *Rhizophora* species. This is due to the fact that the roots of *Avicennia* and *Sonneratia* species grow rapidly and thicker. According to

Blasco *et al.* (1996), mangroves do not stop erosion, but their roots are able to slow that process down greatly.

Taylor *et al.* (2003) stated that there is an increase in the attention that mangroves receive from various organizations which include government and nongovernment. This is due to the ecological and environmental services that they provide, and also their ecological importance. Mangrove forests are important ecologically and socioeconomically (Jia *et al*. 2014). Economically, mangroves play a role as the main source of important products of livelihoods in coastal areas. These include wood, dye tannins etc. They also play a role in the stabilization and protection of coastal areas (Lee *et al*. 2014), water quality conservation, storm protection, habitat protection and biodiversity and ground water discharge and restoration (Giri *et al*. 2015). According to Abrantes *et al.* (2015), mangroves are important contributors to estuarine food webs. In systems where extensive mangrove forests are present, mangrove derived carbon can be the main source of nutrients supporting food webs.

2.9 Mangroves and their dynamics

2.9.1 Mangroves distribution in South Africa

Steinke (1995) in his review stated that mangroves in South Africa are only found in estuaries and bays that are usually located between mean sea level and mean high water spring tides since wave energy is regularly high around the coast. Only 0.05% (1631.7 ha) of the African continent's entire mangrove area is found in South Africa, the majority of the continents mangroves are found in Nigeria (Giri *et al*. 2011). Mangroves are only found on the east coast of South Africa and are larger in the north (26ºS), becoming smaller in southern estuaries (33ºS). Mangroves exist along the east coast of South Africa; they are found in 37 estuaries and they cover nearly 1688 ha which is about 0.05% of the African continent's total (Adams *et al*. 2004). They expand from Kosi Bay found in the north to Nahoon Estuary. They established large stands in Kobonqaba, Nqabara, Xora and Bulunga Estuary. The mangroves covered larger areas on the Xhora Estuary (16 ha) than the Mbashe and Nxaxo which covered 12.5 ha and 14 ha respectively (Saintilan *et al.* 2014). Steinke (1986) documented the colonization of *Avicennia marina* - in Kwelera Estuary; it was thought to be as a result of mangrove stands in Kobonqaba Estuary which is found 60 km north of Kwelera. Mangroves also colonised Kei River (17 km away) and Gonubie River (70 km away), their source is believed to be also Kobonqaba Estuary (Steinke 1986). These estuaries have not been sampled since that study. At Nahoon Estuary, Steinke and others planted mangroves (*Avicennia marina, Rhizophora mucronata* and *Bruguiera gymnorrhiza*) at the mouth of the estuary, the forest later spread as a result of natural distribution of propagules, but only *A. marina* managed to spread extensively in the forest (Steinke 1995). Mangroves were also believed to have been planted in Tyolomnqa Estuary.

Mangroves usually show a zonation pattern, for example Watson (1929) showed that in Southeast Asia a *Sonneratia* or *Avicennia* stand is often found on the sea front and *Rhizophora* and *Bruguiera* stands are distributed more inland. In South Africa, since there is low species diversity, zonation patterns are entirely variable. *Avicennia marina* in some of the estuaries is the only mangrove that occurs in this region. It is usually found on the seaward and also on the landward edge of the forests with *B. gymnorrhiza* and *Ceriops tagal* Perr. inland. *Rhizophora mucronata*. in Mngazana and Kosi Bay is found at the edges of channels and *Lumnitzera racemosa* Willd. is mostly common at the landward margin of the forest (FAO 2005).

Von Maltitz *et al.* (2003) states that mangroves are considered to be most threatened and the rarest forest types in South Africa. There are six species of mangrove found in South Africa and two associate species (Steinke 1995). *Avicennia marina* is the widest spread compared to the other six species found in South Africa. It is commonly known as the white mangrove as a result of its greyish-white under surface of the closely packed leaves. This species is considered to be a pioneer species and it can grow in both muddy and sandy soil. It has small buoyant propagules that are dispersed by ocean water. *Avicennia marina* is found in a number of estuaries (24), from Kosi Bay (KwaZulu Natal) to Nahoon Estuary (Eastern Cape) (Rajkaran 2011). *Bruguiera gymnorrhiza* is another common species found in South Africa occurring in 33 estuaries. It is sparse south of Mbashe River and its natural distribution ends at Wavecrest 32° 35' S, 28° 31'E. Trees of this species can reach a height of up to 15 m (Steinke 1995). In southern estuaries, the largest trees do not exceed 5m (Steinke 1995). *Rhizophora mucronata* is native to Africa, and is not as common as *A. marina* and the *B. gymnorrhiza*. It is found in 12 estuaries of KwaZulu Natal and Eastern Cape. *Ceriops tagal is* the least common species in South Africa, trees of this species do not reach a height that is above 2 m to 3 m. *Lumnitzera racemosa* is found in the Kosi system where trees reach heights of 2 m to 3 m. *Xylocarpus granatum* Koen. is also only found in the Kosi system and only one individual is present (Ward and Steinke 1982). Kosi Bay is the only system that has all six mangrove species. Two other species are found in South Africa, associate mangrove *Hibiscus tiliaceus* L. and a mangrove fern *Acrostichum aureum* L. (Rajkaran 2011*).*

The mangrove forest at Mngazana Estuary is the third largest in the country (Rajkaran 2011). According to Rajkaran and Adams (2012) there are three species found at Mngazana*, A. marina*, *R. mucronata* and *B. gymnorrhiza.* *Avicennia marina* is foremost along the main channel and has seedlings that colonised the intertidal areas at the fore of the adult trees.

Rhizophora mucronata is dense along Creek 2 where sediment moisture content and organic content was the highest. *Bruguiera gymnorrhiza* is mostly found in landward edge of Creek 1 and most of its sites are inundated only during spring tide.

All the mangrove species at Mngazana have undergone disturbance by cutting. The three local communities surrounding Mngazana Estuary use the forest for products manufactured out of wood. The main stems of *B. gymnorrhiza* and *R. mucronata* have been cut and collected. From *A. marina* the branches were cut instead of the main stem (Traynor and Hill 2008). *Bruguiera gymnorrhiza* and *R. mucronata* was used by 70% of the homesteads for building and 3% of the homes used them for other construction purposes. Therefore, these two species were the most preferred for building and *A. marina* was used for firewood. Traditional houses, which are structures, huts etc. that are built up from natural material, are the primary kind of housing structures utilized by 94% of the community living at Mngazana (Traynor and Hill 2008)

Rajkaran *et al.* (2004) observed that people who harvest these mangroves have particular points of entry where the removal of the bundle of poles is proper and suitable for them. Harvesting of trees was concentrated in two prime areas in Creek 1 and Creek 2. Both of these Creeks had stump densities of approximately 3240 ha^{-1}. The third area that was harvested was characterised by 2400 stump ha^{-1}. Harvesters selected adult trees that were above 2.5 m in height. Mangrove loss was estimated at one hectare a year (Rajkaran and Adams 2010) and some of these patches are now bare ground indicating that harvesting resulted in changes in environmental conditions. Rajkaran (2011) stated that a decline in mangrove cover in this forest was noticed on the southern bank which is at the lower reaches near the houses and at the northern bank that is beyond Creek 1.

The next chapter will focus on the distribution of mangroves in South Africa, to determine changes in the microclimate within the mangrove forests and to relate these to biological factors. The weight of propagules produced in a forest is considered to be one of the determinants of mangrove distribution, therefore mangrove distribution and expansion is dependent on the ability of mangrove trees to produce propagules, transport of propagules and also on the number of propagules produced.

Chapter 3 –Distribution of mangroves in South Africa

Generally, mangrove species occupy a distinctive geographical range. Some have geographical ranges that are large *(Avicennia marina*) and stretch over various continents while other species exhibit more restricted ranges (*Rhizophora mucronata*). Abeli *et al.* (2014) noted the effects of marginality on the performance of plant populations and stated that plants found at the edge of their distribution tend to be smaller as compared to those found in central populations and their success in completing their life cycle may be lower.

Various factors have been documented to influence mangrove distribution. Changes in rainfall and temperature patterns influence mangrove distribution and growth. Increased sediment, nutrient supplies, reduced salinity (less than 30 PSU) and a decrease in sulphates may be caused by high rainfall. These can lead to an increase in growth rates and productivity of mangroves. Rainfalls that are low increases salinity which in turn decreases productivity, growth, diversity and seedling survival (Eslami-Andargoli *et al*. 2009). In Australia, mangroves grow taller (30 m high), are more productive and more diverse in areas where there is high rainfall. According to Saenger and Moverly (1985); Hutchings and Saenger (1987) nearly all mangrove species with the exception of *A. marina* generate maximal shoots density when daily mean air temperature increases to 25 °C and they cease to produce leaves when the mean air temperature decreases to 15 °C. Some mangrove species may show a reduction in leaf formation rate. In subtropical areas, especially those that are near the latitudinal limit of mangrove distribution, cold temperatures that are less than 5 °C (freezes/frost) may result in damage or even kill mangroves trees (Liu *et al*. 2014). For example, mangroves have been sporadically damaged and killed in Texas, Louisiana and Florida and this is shown by (Ross *et al.* (2006), and Steven *et al.* (2006)). According to Stevens *et al.* (2006) *Avicennia germinans* in North America have a distribution that can be expanded during periods where it is not freezing or when salt marsh vegetation is killed. A study by Cavanaugh *et al*. (2013) at the east coast of Florida from 25 °N to 30 °N showed that mangroves expanded pole ward across the east coast of North America. This expansion was related to global warming and it was not due to increases in mean air temperatures but it was caused by decreasing frequency of individual cold events.

Mcleod and Salm (2006) stated that as a result of climate change, mangroves will be influenced by sea level rise. Mangrove distribution and growth changes when sea level rises; mangroves move landward and those found at or near the sea margin die due to stress caused by increased water levels (Gilman 2008). Increased CO_2 levels linked to climate change may result in an increase in growth and productivity of some mangroves species, and these are most significant when salinity is low (Field 1995; Gilman 2008). Saintilan and Rogers (2015) document the importance of CO_2 enrichment together with other environmental drivers. Increases in atmospheric CO_2 concentration helps improve soil moisture conditions, therefore resulting in seedling survival, maintains water loss by lowering leaf conductance, increase mangrove growth rates and sapling recruitment that is facilitated by floods that occur frequently.

Coastal development has also been recognized as a factor that is of importance to the expansion of mangroves on the east coast of Australia (Saintilan and Wilton 2001). The dredging operations are thought to increase nutrient and sediment loads in estuaries and the succeeding fertilisation of areas with saltmarsh facilitating colonization of mangroves. Some species of mangroves migrate to higher latitudes where range extension is limited by temperature; therefore, cold events are more likely to limit mangrove distribution into higher latitudes (Snedaker 1995).

Seedlings are the key factor for mangrove distribution. Seedling establishment and growth expands the distribution of mangroves (Hastuti *et al*. 2012). The capacity for vegetative propagation in mangroves is diminished and only a few species are capable of this, hence they are dependent on seedlings for forest expansion and regeneration. They reveal two reproductive strategies that are unique: vivipary and hydrochory. Vivipary increases the chances of successful establishment in an environment that is unpredictable (Feller and Sitnik 2002). Saintilan *et al.* (2014) mention that most species of mangroves are hydrochorus and consequently are likely to disperse to new localities by sea and estuarine currents, driven by wind, tides, waves and ocean circulation. Mangrove species are dispersed by propagules that are buoyant, the ability of these propagules to disperse across the water have limits that are determinable and these are species specific (Duke 1995).

3.1.1 Distribution and drivers of mangrove expansion in South Africa

Mangroves in South Africa expand from Kosi Bay in the north in the province of KwaZulu Natal (KZN) to Nahoon Estuary in the Eastern Cape (EC) (Ward and Steinke 1982). The southern limit of mangroves was previously stated to be at Kobonqaba Estuary which is found north-east of Nahoon Estuary (Table 4) (Hoppe-Speer *et al*. 2015). Mangroves have established large stands in Kobonqaba, Nqabara, Xora (16 ha) and Bulunga Estuary and they covered larger areas on the Xhora Estuary than the Mbashe (12.5 ha) and Nxaxo (14 ha) Saintilan *et al.* (2014). At Nahoon Estuary, Steinke and Carl Vernon planted mangroves in 1969 (*Avicennia marina, Bruguiera gymnorrhiza and Rhizophora mucronata*) at the mouth of the estuary, the forest later spread as a result of natural distribution of propagules, but only *A. marina* managed to spread extensively in the forest (Steinke 1972 and Steinke 1995).

Table 4: The current distribution of mangrove species and coverage of mangrove forests in South Africa. Am = *Avicennia marina*, Bg= *Bruguiera gymnorrhiza*, Rm= *Rhizophora mucronata*, Ct= *Ceriops tagal*, Lr= *Lumnitzera racemose*. Estuaries with * indicate loss of mangroves as recorded by Rajkaran *et al*. (2009).

Estuary with mangroves	**Species present**	**Cover (ha) (Ward and Steinke 1982)**	**Cover (ha) Hoppe-Speer *et al*. (2015)**
Kosi Bay	Am,Bg,Rm,Ct,Lr	59.0	ND
Mgobezeleni	Bg	2.5	ND
St Lucia	Am,Bg	160.0	ND
Mfolozi	Am,Bg	26.0	ND
Richards Bay	Am,Bg,Rm	427.5	ND
Mlalazi	Am,Bg,Rm	30	ND
Mhlanga	Bg	0.5	ND
Mgeni	Am,Bg,Rm	44.0	ND
Durban Bay	Am,Bg,Rm	15.0	ND
Sipingo	Am,Bg,Rm	12.5	ND
Little Manzimtoti	Bg	0.5	ND
Lovu	Am,Bg,Rm	2.0	ND
Msimbazi	Am,Bg	0.5	ND
*Mgababa	Bg	0.5	ND
*Ngane	Bg	0.5	ND
Mkomazi	Am,Bg.Rm	2.0	ND
*Mahlongwa	Bg	1.0	ND
*Kongweni	Bg	<0.5	ND
*Bilanhlolo	Bg	0.5	ND
*Mhlangankulu	Bg	<0.5	ND
*Khandandlovu	Bg	<0.5	ND
Mtamvuna	Bg	1.0	1
Mzamba	Bg	1.0	0.3
Mnyameni	Am,Bg	3.0	5
Mtentu	Bg	1.0	0.6
Mzintlava	Bg	1.5	1.7
Mntafufu	Am,Bg,Rm	10.0	12
Mzimvubu	Am,Bg	1.0	0.03
Mngazana	Am,Bg,Rm	150.0	118
Mtakatye	Am,Bg,Rm	7.5	10
Mdumbi	Am	1.0	5
Mtata	Am,Bg,Rm	34.0	31
Bulungula	Am,Bg,Rm	3.5	0.014
Xora	Am,Bg	16.0	25.5
Mbashe	Am,Bg	12.5	9.2
Nqabara	Am	9.0	11.8
Nxanxo/Ngqusi	Am,Bg	14.0	9.5
Kobonqaba	Am	6.0	0
Kwelera	Am	<0.5	ND
Gonubie	Am	<0.5	ND
Nahoon	Am	0.5	1.6

ND= No data

Hoppe-Speer *et al* (2015) stated that mangroves expanded and spread in that area even though they were out of their limit and were at higher latitudes. They speculated that these mangroves expanded as a result of changes in precipitation and temperature and these changes were brought about by the changes in sea level rise and climate. They concluded that as temperature increases so will the south ward expansion of mangroves. Steinke and Ward (2003) used drift cards as indicators of mangrove propagule dispersal and they recovered cards south of East London and implied that these sites were suitable propagule dispersal.

3.1.2 Objectives and aims

Many factors influence mangrove expansion, but for this study we focussed on air temperature. The aim of this study was to revise the distribution limits of mangroves in South Africa, assess temperature limits and spatial changes in mangrove development at Kwelera and Tyolomnqa Estuary and compare propagule weight of mangroves from different locations. The limit of these forests was expected to be at Nahoon Estuary as presented by Ward and Steinke in 1982 and Hoppe-Speer (2015) and that no forests existed further south of this location due to environmental constraints.

3.1.3 Study site description

The length of the Eastern Cape coastline is approximately 815 km (Figure 3); it extends from the mouth of Blaauwkrantz River to the Umtumvuna River (at the border of KZN). It consists of bays, sandy beaches, rocky shores, sharp cliffs and wave cut platforms. From East London lies the wild coast that is found in the former Transkei, is described as undulatory, and very rough (Breetzke *et al*. 2013). Further south of East London lies the Sunshine coast that is less undulating as compared to the wild coast. The coastline in the Eastern Cape is considered to be the least spoilt when compared to other parts of coastlines in the country.

In the Eastern Cape coastline, 139 estuaries are found, 17 of which are permanently open to the sea. The volume of water in permanently open estuaries is large, permitting tidal activity, which maintains open mouth conditions (Whitfield and Bate 2007). These estuaries differ in size, mouth status, salinity, mixing process and ecological importance. According to Grange and Allanson (1995) estuaries differ in salinity ranges, some have less freshwater coming into the estuary which in turn lead to high salinity ranges and others have average to low salinity ranges. This is as a result of semi-arid temperatures and also irregular catchments.

Eastern Cape estuaries are considered to be in good health (29 % excellent, 25 % good, 14 % fair and 6 % poor) (Berliner and Desmet 2007). Wild coast estuaries are stated to be healthier than the others. DEAET (2004a) recorded Mngazana, Mbashe and Keiskamma estuaries as having great botanical importance. Some of these estuaries have mangrove forests that have various importance which include storm wave protection, provide fuel and building materials, maintaining water quality etc. (Adams *et al*. 2004). A study by Adams *et al*. (2004) showed that there are 14 estuaries that have mangrove forests in them. Some forests were lost as a result of various factors which include mouth closure, changes in salinity and also floods (Hoppe-Speer *et al*. 2015). What follows is a description of estuaries north and south of Nahoon Estuary – the current limit of mangroves. These estuaries were selected for sampling.

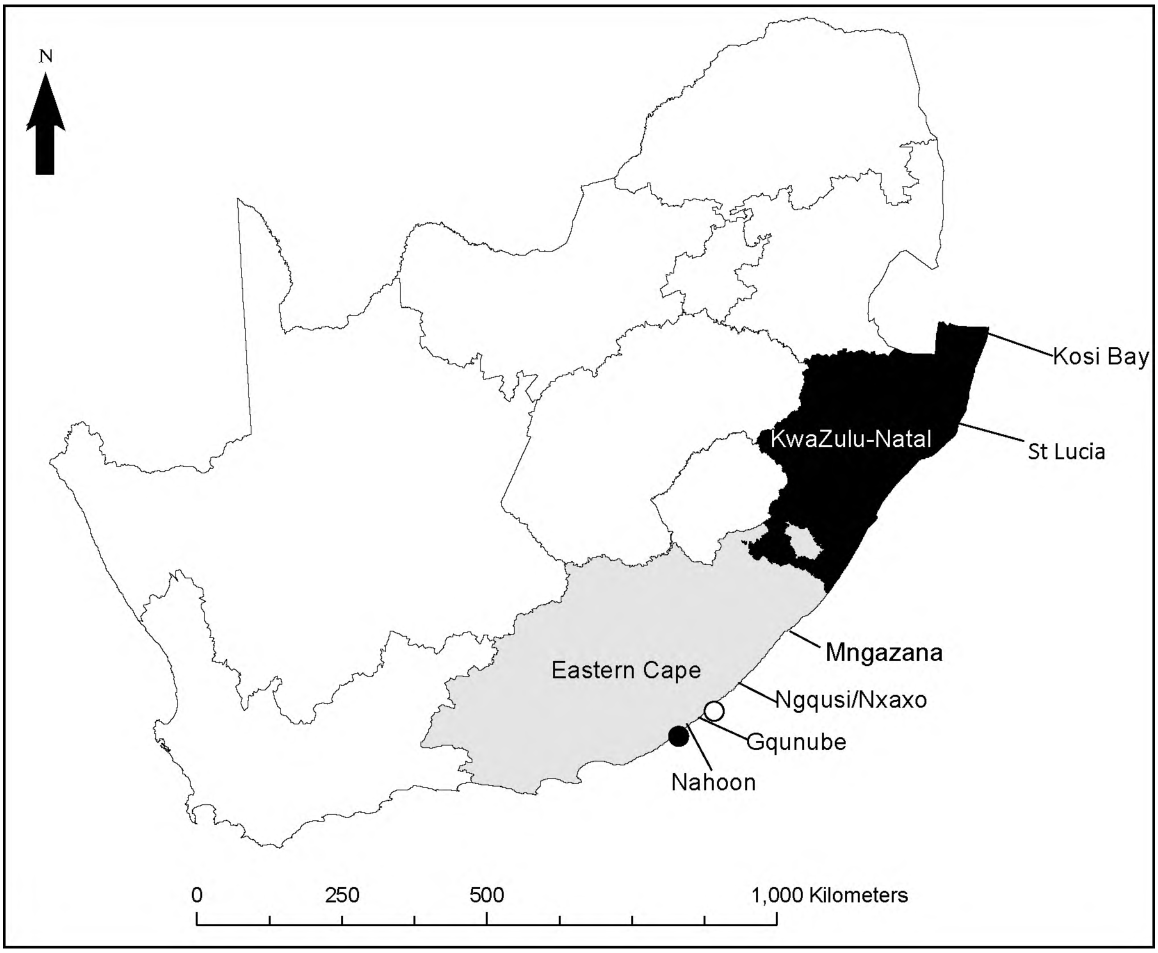


Figure 3:The location of the Eastern Cape province and the estuaries sampled and mentioned in this study (closed circle-Tyolomnqa Estuary, open circle- Kwelera Estuary) (Rajkaran 2011).

Kwelera Estuary

The Kwelera Estuary is 4.9 km long and situated 32º55´26.5˝S 28º04´35.8˝E (Figure 3) (Plate 2 A). It is a permanently open estuary that is close to East London, South Africa (Figure 3, Figure 4). This estuary has a tidal inlet which is 20 m wide and 2 to 3 m deep during high tide conditions (du Preez and Hosking 2010) It has a catchment size of approximately 391 km^2 and a tidal prism of 5.2×10^5 m^3 and a mean annual runoff (MAR) of 42×10^6 m^3 (Talbot *et al.* 1990). A reduction in water inflow was recorded to have increased by 67 % in this estuary (du Preez and Hosking 2010). *Halophila ovalis* (R. Brown) has been found in Kwelera Estuary, this species is stated to be very sensitive to competition with *Zostera capensis* (Setchell) and environmental changes. A study by Talbot *et al.* (1990) showed that floods that occurred frequently for 15-years in this estuary completely removed *Z. capensis* beds.

Figure 4: Map of Kwelera Estuary, from Areena riverside resort to the mouth (Google map).

Gonubie Estuary

The Gonubie Estuary also known as Gqunube Estuary is a permanently open estuary that is found in the warm temperate region. It is situated at 32°55´59˝S; 28°01´59˝E (Figure 3). It has a catchment area of approximately 665 km^2 and the length of the estuary is approximately 109 km. According to Campbell *et al.* (1991), this estuary has a significant freshwater inflow and a large tidal inlet. There are a number of tributaries in its catchment. The largest tributary is the Tanga River which enters the Gqunube from the north. The Ngwevana, Ntena, Qolora, Ncweba and Mtyana tributaries enter the Gqunube River in its lower reaches. The intertidal area of Gqunube Estuary is characterised by various plant communities including perennial herb (*Juncus krausii* Hochst.), submerged macrophytes (*Zostera capensis*) and saltmarsh species (*Sarcoconia* spp., *Triglochlin* spp. etc.) (Coetzee *et al*. 1996).

Tyolomnqa Estuary

The Tyolomnqa Estuary is a large, permanently open estuary that is situated approximately 38 km southwest of East London and it forms the northern border of the Ciskei coast of the Eastern Cape. It is located 33º13'32" S; 27º35'01" (Figure 3, Figure 5). It has a river that is about 78 km long with a catchment area of 441 km^2. James and Harrison (2011) describe the estuary as being narrow (approximately 32 m) and is channel-like in the upper reaches and only widens towards the mouth (200-300 m) (Figure 5). According to Bowd *et al.* (2012) Tyolomnqa Estuary has a large and complex social system and it encompasses important local and provincial governmental bodies, non-governmental agencies and general service users e.g. local and visiting fishermen. Along the estuary and around the catchment, subsistence farming occurs. The estuary is stated to be attractive and has a natural heritage that has been maintained (Maponya and Ngulube 2007). Reeds, sedges and supratidal saltmarsh are found in this estuary and the total current saltmarsh is 19.4 (Table 5) (Mucina *et al*. 2006). The estuary is known for its irregular commercial fishing and poaching by people from other communities. Offshore of this estuary is where the first coelacanth (which was believed to be extinct) was captured and numerous memorial statues was erected by Captain Hendrik Goosen in 1938.

Figure 5: Map of Tyolomnqa Estuary, from furthest mangroves to the mouth (Google map).

Keiskamma Estuary

The Keiskamma Estuary is a large permanently open estuary that is situated at the small town of Hamburg that is approximately 62 km northeast of Port Alfred and 51 km southwest of East London. It is located at 33°16´45˝S; 27°29´50˝E. This system has a large catchment that is 2745 km^2 and a MAR of 142.7 × 10^6. The estuary is narrow in the upper reaches (30 m) and widens towards the mouth (James and Harrison 2011) and it has a length of approximately 12 km and an area of 197 ha. Keiskamma Estuary has the highest density of endemic gobiids and it is a fresh water rich estuary (Strydom and Neira 2006). Saltmarsh species *Sarcoconia tegetaria* S. Steffen, Mucina & G. Kadereit was also recorded by Steffen *et al.* (2009). According to Mhangara and Kakembo (2012), the average annual rainfall and temperature in Keiskamma were 1 900 mm and 11° C for the escarpment zone and 600 mm and 18° for the coastal plateau. Its salinity ranges between 0.00-26.38 PSU, temperature ranged between 15.71-26.36° C and water transparency was indicated by secchi disk depth that ranged between 0.01-0.0 m (Strydom *et al.* 2002).

3.2 Materials and methods

To revise the distribution of mangroves in South Africa sampling took place at Kwelera, Gonubie, Tyolomnqa and Keiskamma. However, data was only collected from Kwelera and Tyolomnqa as only one tree was found at Gonubie and no trees were found at Keiskamma. Sampling occurred in 2014 and 2015, and first sampling took place in May 2014, and the second sampling occurred in April 2015. Mapping was done using Google Earth Pro to determine the temporal and spatial changes of the mangroves at Kwelera and Tyolomnqa Estuary. Aerial photos were examined for Kwelera (2002, 2012 and 2016) and Tyolomnqa (2000, 2011 and 2016) estuaries. Mangrove units were mapped using the polygon feature, where possible and the aerial extent was measured in hectares.

Physical and biological characteristics were measured where mangroves were located. At two estuaries (Kwelera, Tyolomnqa Estuary) temperature was measured hourly using iButtons (Plate 1) (from May 2014 to June 2015). To assess sediment and porewater characteristics, sediment was collected at low tide during May (Kwelera and Tyolomnqa - Site 1 and 2) and September 2014 (Tyolomnqa - Site 3 and 4) using three replicate holes at three zones; it was collected from the upper 30 cm. Porewater salinity, temperature, conductivity and pH was measured from pooled water with a handheld YSI meter.

Plate 1: An iButton recording temperature hourly on an *A. marina* tree at Kwelera Estuary.

Table 5: Site description of the two estuaries sampled.

Name of Estuary	**Site**	**Distance from mouth (km)**	**Distance from channel (m)**	**Tidal regime**	**Associated vegetation**
Kwelera	1	1	<5	Inundated by every tide	*Phragmites australis* (Cav.)Trin.
	2		<5	Inundated by every tide	*Spartina maritima* (Curtis) Fernald
Tyolomnqa	1	5.5	5.6	Inundated by every tide	*Bassia diffusa* (Thunb.) Kuntze , *Sarcocornia spp.*
	2		5.6	Inundated by every tide	
	3	1.5	16.1	Inundated by normal and spring tides	*Spartina maritima* (Curtis) Fernald
	4		47.3	Inundated by every tide	

Sediment Analysis

Redox potential was measured within 12 hrs of sediment collection. Samples were then transported back to the lab where pH, moisture content, organic content and conductivity was measured according to Gee *et al*. (1986).

Redox potential

Redox potential was measured using an oxidation-reduction platinum electrode attached to a pH/ redox meter. Readings were done by placing the probe in the sediment and taking readings every 6 minutes.

Moisture content (Black 1965)

Moisture content (%) of the sediment was measured by weighing 10 g sediment samples in crucibles and drying them to constant weight in an oven at 100° C for 48 hours. The samples were then re-weighed and the difference in weight before and after drying recorded.

$$\frac{(wet\ mass - dry\ mass)}{wet\ mass} \times 100\%$$

Organic content (Briggs 1977; Sanders et al 2012)

To measure the organic matter content (%) of the sediment, sediment that was re-weighed after measuring moisture content was placed in an ashing at 550° C for 5 hours. Readings were then taken before and after ashing, and the organic matter content calculated.

$$\frac{(initial\ dry\ mass - mass\ after\ ashing)}{initial\ dry\ mass} \times 100\%$$

pH

Sediment pH was measured using a multiprobe by weighing 5 g of sediment and 50 ml of distilled water added to this and carefully stirred. The probe was then inserted into the solution and the readings recorded.

Conductivity

To measure conductivity, a 250 g of air-dried sediment samples were placed in beakers. The distilled water was added to the sediment until a paste-like consistency was achieved. The samples were filtered using a Buchman filter and Whatman No. 40 filter paper after being left to stand for an hour. A conductivity meter was used to measure the conductivity of the sediment samples.

Grain Size

Grain size was measured using Mastersizer 2000 which uses a laser diffraction technique. The samples were analysed with a wet analysis using a Hydro S dispersion unit (capacity 100-150 ml). The Mastersizer 2000 had a grain size distribution range of 0.02-2000 µm. When measuring, water was used as a dispersant and each sample was made wet with a small amount of water and it was then added to the Hydro S dispersion unit.

Biological characteristics

Population characteristics were measured to determine if the mangrove populations were regenerating and to attempt to age the trees. To determine the population structure of mangroves, quadrats of 3 m x 3 m were constructed and replicated 5 times in each site (or as many times as the mangrove population allows). The forest at Kwelera is 1 km from the mouth of the estuary and was divided into two sites while four sites (Site 1 and Site 2 – 5.5 km from the mouth, Site 3 and Site 4 – 1.5 km from the mouth) were set out at Tyolomnqa (Table 4). Individual trees were measured (height, circumference at breast height - CBH). CBH was converted to diameter at breast height (DBH). Aspects of phenology were recorded in terms of flowering and propagule production. Propagules were also collected, dried at 70° C for 24 hours and measured to determine if the investment into propagules was lower compared to other mangrove forests. Trees were shaken to retrieve propagules that were about to drop, 10 propagules were collected from 5 different trees at each estuary.

Plate 2: View of Kwelera mangrove forest (A) and the singular *B. gymnorrhiza* sapling found in the estuary (B).

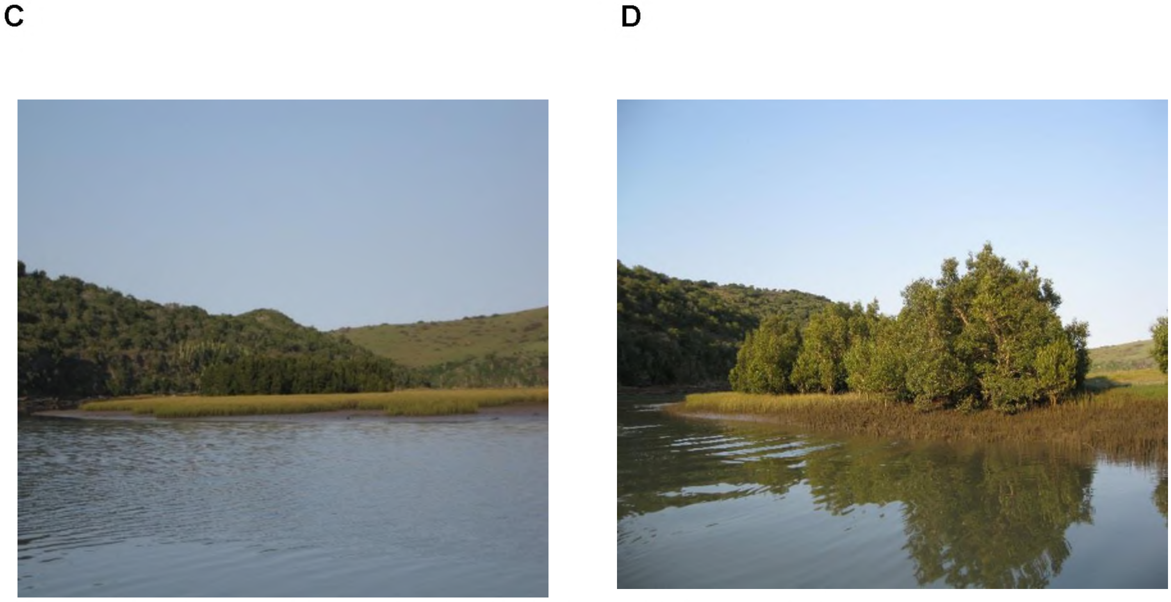


Plate 3: View of Site 3 and 4 at Tyolomnqa mangrove forest (C) and (D).

3.2.1 Data Analysis

Data was analysed using STATISTICA (12 64 -bit). Normality was tested with Shapiro Wilk's test to test if the data were normal at $p>0.05$. If data was found to be non-normal, then non-parametric a Kruskal-Wallis ANOVA was used to compare more than two sets of data for significant differences. If data was found to be normally distributed, then One Way ANOVA tests was used. Correlation matrices were used to determine the relationships between porewater and sediment salinity conductivity and also to determine a relationship between moisture content, organic content and redox potential.

3.3 Results

Spatial and temporal changes have been occurring both at Kwelera (Figure 6) and Tyolomnqa Estuaries (Figure 7 and Figure 8). An increase in mangrove cover was recorded in both estuaries and it occurred continuously. At Kwelera Estuary, mangrove cover increased by 0.3 ha in 2016, while mangrove cover at Tyolomnqa Estuary increased by 0.12 ha from 2012-2016 (Table 6).

In 2002, a few mangroves trees were found growing at Kwelera Estuary, in a subsequent assessment between 2012-2016 the mangrove stand was thicker as compared to the previous years. Google Earth images for the year 2000 at Tyolomnqa Estuary were not clear to record, but mangroves were present in that estuary. The highest increase in mangrove cover was recorded in Site 1 at Tyolomnqa Estuary (0.07 ha) from 2012-2016 (Figure 7). Total mangrove area at Tyolomnqa was higher than that of Kwelera Estuary.

Seaward expansion was recorded at Kwelera Estuary. An increase in mangrove cover at Site 1 and Site 2 at Tyolomnqa Estuary occured along the shore and a landward expansion occurred in Site 3 and Site 4.

Table 6: Changes in mangrove cover (ha) at two estuaries

Year	**2002**	**2012**	**2016**
Kwelera	0.1	0.3	0.4
Year	**2000**	**2011**	**2016**
Tyolomnqa	Not clear to map	0.55	0.67

Air temperature at Kwelera averaged between 14.54 °C to 21.66 °C. Maximum temperature was 30 °C (during summer months) and minimum temperature was 4.5 °C (June 2014 and 2015) (Table 7). At Tyolomnqa, temperature ranged between 13.87 to 22.46 °C. Maximum temperature was 34 °C (summer months) and minimum temperature was 1.1 °C recorded in June and July 2014 (Figure 9). Extreme temperature events were defined when temperatures were < 5 °C (Table 7). Low temperatures mostly occurred at night or during the early mornings. An increase in the number of extreme events was recorded as you move further south. Temperature data from Mngazana and Nahoon are included for comparison.

Figure 6: Development of the mangrove stand (white circle) at Site 1 and 2 at Kwelera. A) 2002, B) 2012 and C) 2016.

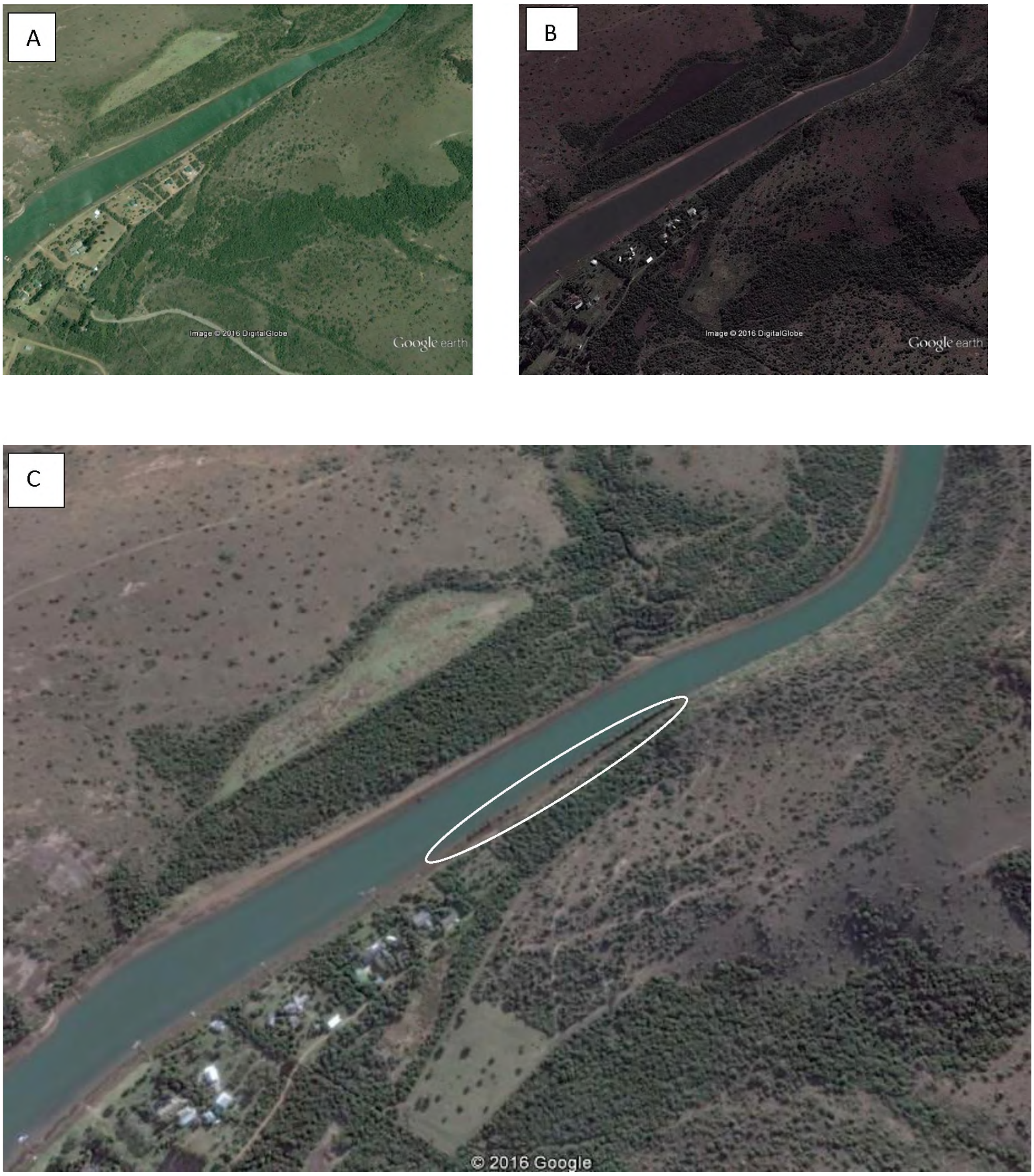


Figure 7: Development of the mangrove stand (white circle) at Site 1 and 2 at Tyolomnqa. A) 2000, B) 2011 and C) 2016.

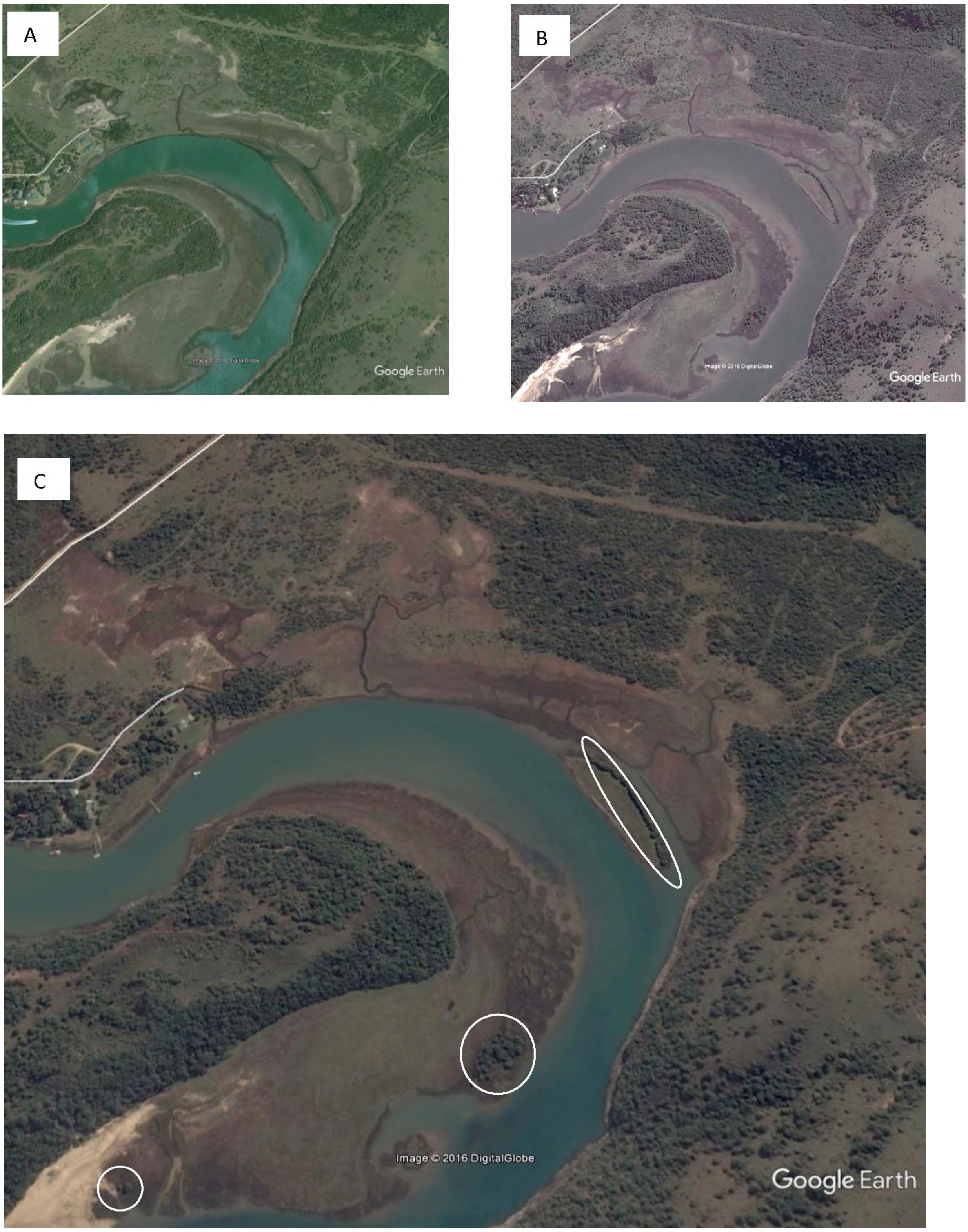


Figure 8: Development of the mangrove stand (white circles) at Site 1 and 2 at Tyolomnqa. A) 2000, B) 2011 and C) 2016.

Table 7: A summary of air temperature and extreme temperature events at four estuaries in South Africa

Estuary	Coordinates	Minimum temperature (°C)	Maximum temperature (°C)	Number of extreme events (5-10 (°C)	Number of extreme events(<5°C)
Mngazana	31 °S	5.9	34.7	338	-
Kwelera	32 ºS	4.5	30	542	3
Nahoon	33 °S	5.3	31.4	130	15
Tyolomnqa	33 ºS	1.1	34	758	72

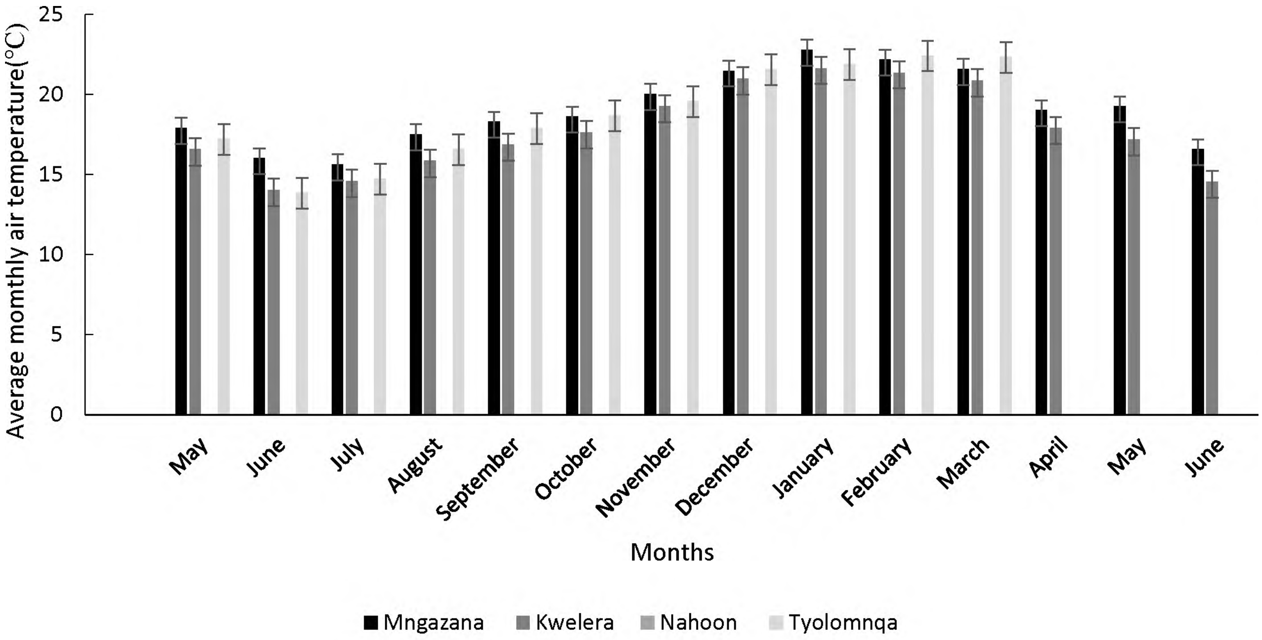


Figure 9: Average monthly air temperature variations at Mngazana, Kwelera, Nahoon and Tyolomnqa.

Sediment characteristics

Moisture content of the sediment was similar between the two estuaries (F $_{(1:38)}$ = 0.0, p>0.05) and ranged from 25.8-47.6 % at Kwelera and 5.6 - 47.02 % at Tyolomnqa. Organic content was significantly lower at Tyolomnqa (1.53 % ± 0.19) (F $_{(1, 38)}$ = 6.4, p<0.05) and ranged from 0.2 – 6.7 %. (Table 8).

Redox potential (mV) of the sediment ranged from 8.8 to 12.4 (mV) at Kwelera and at Tyolomnqa it ranged between -4.2 to 12.5 (mV). Redox at Tyolomnqa was significantly lower than redox at Kwelera (F $_{(1, 18)}$) = 4.28, p< 0.05,). Redox potential was higher for the surface samples at Kwelera compared to Tyolomnqa. Sediment pH at Kwelera and Tyolomnqa was acidic to being neutral. The pH ranged from 2.3-7.17 at Kwelera and, 2.22 -7.09 at Tyolomnqa (Table 8). Mean pH for Kwelera and Tyolomnqa was 4.5 ± 2.17 and 5.78 ± 1.78 respectively. There was no significant difference between the two estuaries ($F_{(1, 38)} = 4.9$), $p > 0.05$).

Average sediment conductivity was similar between the two estuaries (55.3 ± 3.5 mS; 54.1 ± 2.4 mS) and it ranged from 40.7–78.8 mS at Kwelera and 36.4-80.1 mS at Tyolomnqa. Conductivity was not significantly different between sites ($p > 0.05$) and estuaries ($F_{(1, 34)} = 0.30$, $p > 0.05$). Table (8)

There was a positive correlation between organic and moisture content at Kwelera Estuary ($R=0.9$, $p<0.05$). No correlation was found between organic and moisture content at Tyolomnqa Estuary ($R= 0.4$, $p>0.05$) and similarly no correlation was found between moisture content and redox potential at Kwelera and Tyolomnqa ($R=0.6$, $p>0.05$; $R=-0.8$, $p>0.05$ respectively).

Porewater characteristics

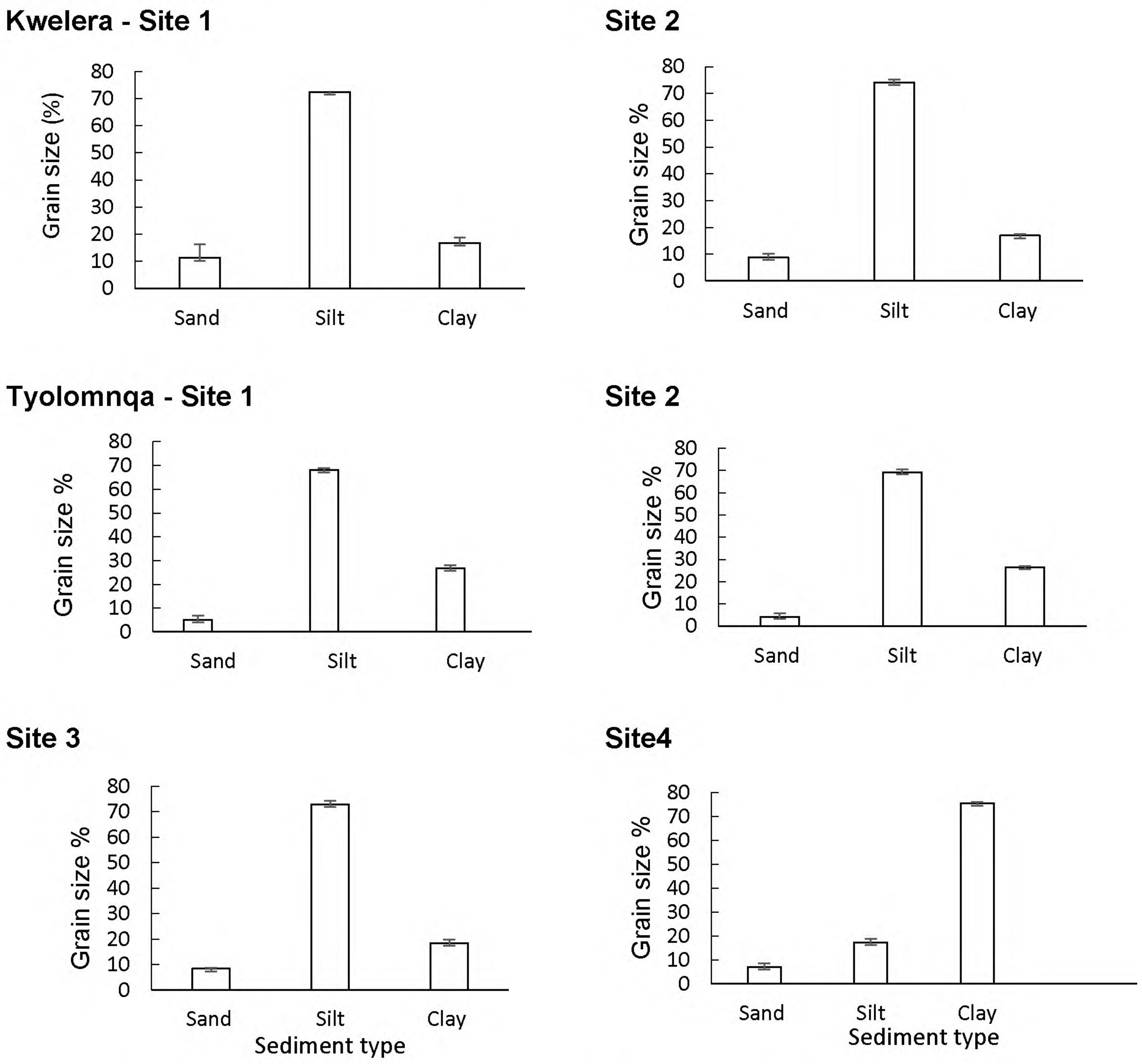
Porewater salinity ranged from 27.4- 31.3 PSU at Kwelera and 21.1-38.8 PSU at Tyolomnqa where it was significantly higher ($F_{(1, 5)} = 5.77$, $p< 0.05$) (Table 8). Porewater salinity was similar between sites at Kwelera Estuary ($F_{(1, 3)} = 1.6$, $p>0$. At Tyolomnqa porewater salinity differed among sites, Site 1 had higher porewater salinity as compared to Site 3 ($F_{(1, 3)} = 244.2$, $p< 0.05$) and Site 2 had higher porewater salinity as compared to Site 3 and 4 ($F_{(1, 4)} = 5.1$, $p < 0.05$). Porewater conductivity was similar in both estuaries. Conductivity ranged from 45.7 ± 0.9 mS at Kwelera and 43.9 ± 2.7 mS at Tyolomnqa. There was no significant difference in conductivity between Kwelera and Tyolomnqa ($F_{(1, 15)} = 3.3$, $p>0.05$). Porewater temperature was similar and ranged between 16.5-23.3ºC for Kwelera and Tyolomnqa ($F_{(1, 15)} = 3.3$, $p> 0.05$). Porewater pH at both Kwelera and Tyolomnqa was less than 7 with an exception of two sites at Tyolomnqa (Site 2 and Site 4). Overall pH was significantly higher at Tyolomnqa ($F_{(1, 15)} = 1.5$, $p < 0.05$). (Table 8).

Table 8: Sediment and porewater characteristics for all sites (average ± SE, N=3).

Estuary	**Site**	**Moisture content (%)**	**Organic content (%)**	**Sediment Redox Potential (mV)**	**Sediment pH**	**Sediment conductivity (mS)**	**Porewater Salinity (PSU)**	**Porewater Conductivity (mS)**	**Porewater pH**	**Porewater Temperature (° C)**
Kwelera	1	33.0 ± 1.9	2.6 ± 0.3	10.8 ±0.5	4.7 ±0.	60.2 ±4.	29.6 ± 1.1	45.7 ± 1.3	6.3 ± 0.2	20.96 ± 0.7
	2	37.0 ± 3.6	3.9 ± 1.1	9.8 ± 0.2	4.5 ±0.9	50.4 ± 4.	28.0 ± 0.6	43.3 ± 0.7	6.4 ± 0.1	20.35 ± 0.4
Tyolomnqa	1	39.0 ± 2.8	0.9 ± 0.2	-3.1 ± 0.4	4.7 ±0.8	47.1 ± 2.5	26.0 ± 4.8	25.2 ± 21.7	6.5 ± 0.2	22.2 ± 1.1
	2	39.6 ±1.8	0.9 ±0.2	2.1 ± 0.2	4.6 ± 1.0	46.39 ± 3.2	23.8 ± 2.4	37.5 ± 3.3	7.1 ± 0.1	21.6 ± 0.4
	3	34.9 ± 4.5	2.2 ± 0.5	3.6 ± 0.5	6.6 ± 0.1	57.04± 2.0	37.9 ± 0.5	56.7 ± 0.5	6.7 ± 0.2	17.7 ± 0.1
	4	38.5 ±2.4	1.7 ±0.3	6.8 ± 1.4	6.7 ± 0.2	65.9± 3.5	34.3 ± 1.5	51.7 ± 2.5	7.4 ± 0.2	18.4 ± 1.3

Sediment grain size

The sediment at Kwelera and Tyolomnqa was mostly composed of silt, while sand and clay was found in small proportions in both estuaries (Figure 10). Grain size ranges for each component of the sediment were designated as follows: sand (2 mm-64 mm), silt (smaller than 0.0625 mm down to 0.004 mm) and clay (< 0.002 mm).


Kwelera - Site 1

Site 2

Tyolomnqa - Site 1

Site 2

Site 3

Site4

Figure 10: Sediment composition of Kwelera and Tyolomnqa Estuary

Population structure of mangroves

One tree was found at Gonubie Estuary, it was 235 cm in height and DBH of 3.3 cm. It had flowers but no propagules were found. Tree height at Kwelera ranged from 12-800 cm. Twelve of the mangrove trees were classed as seedlings (0-50 cm). DBH ranged from 2.4 - 16.49 cm, five trees were flowering but none had propagules. Tree height at Tyolomnqa ranged between 3- 593 cm, 82 trees were seedlings (Table 9). DBH ranged from 0.4 to 10.3 cm, 24 trees were flowering and 8 trees had propagules on them. Total number of trees at Kwelera Estuary and Tyolomnqa were 37 and 164 respectively. The density of individuals was 0.9 trees per m^2 at Kwelera (Plate 2 A) and 1.2 trees per m^2 at Tyolomnqa. The population at both Kwelera and Tyolomnqa Estuary showed a J shape curve and the population found there mostly belonged to the Regeneration size class less than 50 cm (Figure 11). At Kwelera no trees were found in Size class 250-350 cm.

During June 2015, the maximum height at Kwelera was still 800 cm and at Tyolomnqa it was 620 cm. The *Bruguiera gymorrhiza* tree that was recorded in 2012 was still there and it had a height of 88 cm (Table 9). Populations found in both estuaries mostly belonged to the regeneration class size less than 50 cm. When looking at phenology, no trees were flowering at Kwelera, while 23 trees were flowering at Tyolomnqa (Table 10) and no trees had propagules in them at both Kwelera and Tyolomnqa.

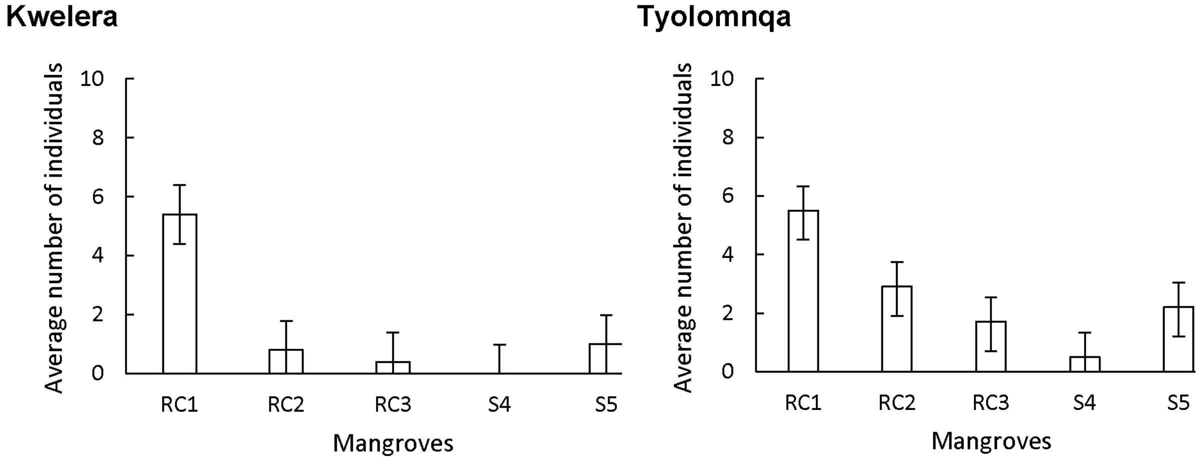


Figure 11: Average number of trees found in each size class. Regeneration size classes (RC1- <50 cm), (RC2 50.5-150 cm), (RC3 151 – 250 cm). Size classes (S4 251-350 cm) and (S5 >350 cm).

Table 9: Population characteristics in 2014 for all the sites (Average ± SE).

Estuary	**Site**	**Density (per m^2)**	**Maximum height (cm)**	**Average Height (cm)**	**Maximum DBH (cm)**	**Average DBH (cm)**	**Adult: seedling ratio**	**Trees with flowering (%)**	**Trees with propagules (%) (Summer 2014 and 2015)**
Kwelera	1	0.9 ± 0.2	800	192 ± 30.8	16.5	2.2 ± 0.9	1:16	0.3 ± 0.3	-
	2	0.8 ± 0.1	240	48.5 ± 16	2.4	0.2 ± 0.2	1:13	-	-
Tyolomnqa	1	1.3 ±0.4	593	202.1 ± 21	8.3	2.9 ± 0.3	1:4	0.3 ± 0.2	-
	2	0.7 ±0.3	457	168.3± 4.4	6.3	2.08 ± 0.6	1:7	1.2 ± 0.1	1.6 ± 0.1
	3	1.3 ± 0.2	521	211.3 ± 32	8.2	2.08± 0.57	1:2	0.2 ± 0.2	-
	4	1.4 ± 0.5	420	71.8 ± 15	10.3	1.07 ± 0.2	1:5	-	-

Table 10: Population characteristics in 2015 for all the sites (Average ± SE).

Estuary	**Site**	**Density (per m^2)**	**Maximum height (cm)**	**Average Height (cm)**	**Maximum DBH (cm)**	**Average DBH (cm)**	**Adult: seedling ratio**	**Trees with flowering (%)**	**Trees with propagules (%) (summer 2014 and 2015)**
Kwelera	1	0.3 ± 0.1	800	643 ± 64. 5	16.7	9.6 ± 1.6	-	-	-
	2	1.4 ± 0.1	88	28.7 ± 2.3	-	-	-	-	-
Tyolomnqa	1	1.3 ± 0.9	580	226.2±24.6	12.7	4.3 ± 0.5	1:3	-	-
	2	0.2 ± 0.1	470	419.3±18.8	12.7	7.3±0.9	1:0	0.9 ± 0	-
	3	1.3 ± 0.6	550	132.9±18.9	9.7	2.8 ± 0.2	1:1	0.5 ± 0.4	-
	4	1.1 ± 0.3	620	185.6±21.3	10.5	1.8 ± 0.3	1:2	0.2 ± 0.2	-

Table 11: Propagule weight of *Avicennia marina* in estuaries of the Eastern Cape (Average ±SE)

Estuary	**Propagule weight before drying (g)**	**Propagule weight after drying (g)**
Mngazana	3.9 ± 0.2	1.4 ± 0.1
Kwelera	5.4 ± 0.2	2.5 ± 0.1
Tyolomnqa	5.1 ± 0.2	3.8 ± 0.1

3.4 Discussion

The objective of this study was to re-evaluate the distribution of mangroves in South Africa. Nahoon Estuary was expected to be the limit of these forests as presented by Ward and Steinke in 1982 and Hoppe-Speer *et al.* (2015) and moreover, that no forests existed further south of this location due to environmental constraints.

Various factors have been documented to have an effect on mangrove expansion, Eslami – Andargoli *et al*. (2009) mentioned that changes in rainfall patterns have played a role in landward mangrove expansion. Gilman *et al*. (2008) also documented how changes in rainfall patterns will have an effect on mangrove distribution and growth. Other factors like sea level rise, extreme high water events, temperature and CO_2 also have an effect on mangrove expansion (Gilman *et al*. 2008) but the last two factors may play a synergist role. Authors, which include Seppa and Birks (2002); Sekhwela and Yates (2007); Saintilan *et al*. (2014) document how temperature has an effect on the expansion of the world's flora. Changes in temperature due to global warming result in these trees growing in areas that are favourable for their growth and productivity. E.g *Rhizophora spp*. now dominates the Richmond River, Australia as compared to the previous decades. Thermal modelling also suggests that expansion of mangroves will still be seen occurring at the Gulf coast with increasing temperatures, subsequently replacing saltmarsh (Osland *et al*. 2013; Saintilan and Rogers 2015).

Changes in atmospheric CO_2 overtime improves soil water connections and facilitates propagule/ sapling recruitment. Increases in atmospheric CO_2 maintains water loss therefore decreasing leaf conductance, which leads to low evapotranspiration. It also improves moisture content, which in turn induces better percolation, therefore allowing seedling survival (Saintilan and Roger 2015).

The study identified four estuaries to determine if changes had occurred in the distribution of mangroves. Kwelera and Tyolomnqa Estuaries were the only two of the four that showed signs of a regenerating forest. Changes in mangrove cover (ha) from 2000-2016 were observed at Kwelera and Tyolomnqa Estuary with both estuaries exhibiting increasing changes in mangrove cover on a spatio-temporal scale. Various mangrove forests have been monitored using spatio-temporal dynamics, an increase in mangrove cover and distribution was recorded by Cuc *et al.* (2008), where mangroves increased from 3 936 ha in 1999 to 5 436 ha in 2006. Eslami-Andargoli *et al.* (2009) also documented an increase in mangove cover and landward expansion of mangroves at Moretons Bay subtropical estuaries, this landward expansion was associated with changes in rainfall patterns. Study by Hoope-Speer (2013) at Nahoon Estuary also documented increase in mangrove cover, where mangrove expansion was occurring toward the where colonisation first occurred on the sides of the wider creek. Hoope- Speer (2013) further studied other estuaries in South Africa where an increase in mangrove cover was occurring, past cover (ha) at Mntafufu in 1982 was 10 ha and in 2012, the cover increased to 12 ha. Mzintlava Estuary increased by 0.2 ha from 1982 to 2012.

Hutchings and Saenger (1987) states that nearly all mangrove species with the exception of *A. marina* generate maximal shoots density when the mean air temperature increases to 25°C and they cease to produce leaves when the mean air temperature decreases to 15° C. In temperate areas, especially those that are near the latitudinal limit of mangrove distribution, cold temperatures that are less than 5 °C (freezes/frost) may result in damage or even kill mangroves trees. Some species are either more or less sensitive to freezing temperatures than the others e.g. *A. germimans* is less sensitive to cold temperatures than *R. mangle* L. (Krauss *et al*. 2008 and Liu *et al*. 2014). *Avicennia marina* is able to tolerate cold temperatures as low as (<5 °C). Quisthoudt *et al.* (2012) states that *A. marina* was able to cope better in colder environments where the mean monthly temperature may drop to 8.1 °C in comparison to the other species assessed. If temperatures drop more than 10 °C within a short period of time or if they are exposed to freezing *A. marina* becomes damaged. Low temperatures lead to a decrease in photosynthetic rates and affects leaf membranes (Kao *et al*. 2006). Once temperatures become cold and freezing, *A. marina* may have dead branches, leaf fall and defoliation of the whole tree. Since *A. marina* species is known to be a pioneer species, once temperatures improve to their normal state these mangrove trees may recover and survive. As one moves further south down the coastline of South Africa the minimum air temperatures decrease and the number of extreme temperature events increase. This is expected to decrease the survival of mangrove species as their distribution is limited by temperature. This study has shown that the two estuaries do not experience the lethal minimum temperature recorded for *A. marina* and therefore the expansion of mangroves is probably further south than Tyolomnqa and is taking place. During this study a surviving individual of *Bruguiera* *gymnorrhiza* was found at Kariega Estuary (33° 41´S; 26 °44´ E) which is ~143 km south of Nahoon.

Two mangrove species were found at Kwelera (*A. marina* and *B. gymnorrhiza* – the latter respresented by one individual, only) (Plate 2 A and B) and only *A. marina* was found at Tyolomnqa. The *Bruguiera gymnorrhiza* tree found at Kwelera was the same one encountered in 2012, that is, no new *B. gymnorrhiza* seedlings were found in the course of this study. Expectations of low genetic diversity as mentioned by De Ryck *et al.* (2016) may occur at these peripheral forests. Adult trees found there were approximately the same heights. Recruitment does occur, but the transition from seedling to sapling is almost zero. Tyolomnqa Estuary had a larger population size and greater distribution along the estuary (Plate 3 C and D) when compared to Kwelera where mangroves are restricted to one area less than 0.5 ha approximately 1 km from the mouth of the estuary. The Northland Regional Council (2015) mention that in some cases mangroves that occupy small estuaries may have their spread being restricted and that results in small scale changes in distribution as mangroves are unable to grow there and having their growth along their landward side being limited by elevation. At Kwelera Estuary, sea grass and saltmarsh species were found on the adjacent bank. Environmental stress affects the balance between species facilitation and competition (Zhang and Shao 2013). A study by Stevens *et al.* (2006) shows how the presence of saltmarsh and other species facilitates mangrove colonization and growth by trapping mangrove propagules. The presence of saltmarsh and *Spartina* at Kwelera and Tyolomnqa may lead to a further colonization of mangroves in those estuaries. Once the mangrove propagules become established and successful they will expand and grow in the area and saltmarsh will be replaced by mangroves as it cannot grow under the shade of a mangrove tree. This colonisation process has been observed at Harbor Island, Texas where A. *germinans* increased in density and subsequently replaced the saltmarsh species present (Montagna *et al*. 2011 and Raabe *et al*. 2012).

The starter population at Tyolomnqa was planted by Duncan Brown sometime in the 1990s, and it has since expanded. Mangrove seedlings are extremely vulnerable and under trying conditions have a high mortality rate. Balke (2013) stated that high tide (currents and waves) can prevent seedlings from becoming established. If establishment does occur they can still be deficient as a result of being buried under the sediment, erosion can also play a role in seedlings not growing and surviving. It is also possible that this is as a result of unfavourable topographical elevations and hydrological patterns that prevent seedlings from establishing there. Another factor that may lead to seedlings being deficient is temperature. Steinke (1975) and Steinke and Charles (1986) stated that low temperatures may limit the early growth of *A. marina* seedlings and temperatures of about 17° C result in slow growth of seedlings. Trees found at both Kwelera and Tyolomnqa had propagules on them during the 2014 and 2015 sampling dates. An increase in average propagule weight was observed as you move further south. Propagules in both estuaries were mostly found in mangrove trees that were exposed to the sun. A study by Steinke (1988) at Mngeni Estuary recorded similar observations namely shoots exposed to the sun had more propagules than those in the shade. A study by Delgado *et al*. (2001) found that average weight before drying for *Laguncularia* was 0.35 g and for *A. germinans* was 1.90 g. Another study by Saifullah *et al*. (2007) at a Pakistan Indus Delta showed that *A. marina* species had a fresh propagule weight of less than 2.0 g, but an increase in propagule weight was observed after September. The average propagule weight of both of the above-mentioned studies were lower as compared to average propagule weight of propagules in our study (Table 11).

A majority of mangroves are hermaphroditic and their germination takes place while the embryo is still attached to the parent tree (Hoff *et al*. 2010). These propagules are dispersal elements of propagules. Once fully developed, these propagules drop from the parent tree and float away to mature somewhere, if conditions are favourable for these propagules they will then establish and grow successfully. Factors such as sediment characteristics, presence of nutrients and porewater characteristics play a role in the establishment and growth of these propagules. According to Steinke and Ward (2003) in South Africa, *A. marina* propagules from many estuaries tend to remain afloat and this suggests that the dispersal range may be large. Studies and distinctive observations by Steinke (1986) showed that *A. marina* propagules tend to travel less than 200 maritime miles before they die. They further state that Clarke (1993) shows how *A. marina* propagules are limited to a range of up to 10 km in Australia; these propagules establish themselves near the parent tree. Steinke (1995) therefore suggested that temperature is not really the most important factor that limits the distribution of mangroves in the southern areas of South Africa. It may be possible for these propagules to be transported to other sites and estuaries, therefore facilitating the expansion and distribution of mangroves. A mangrove propagule can grow close to its parent tree or it may float kilometres away from it. This was proved by mangroves that were able to colonise Kei River (17 km away) and Gonubie River (70 km away), and their source is believed to be Kobonqaba Estuary. No trees were found in Keiskamma, this may be as a result of inability of propagules to establish and grow there in competition with already established vegetation.

Mangrove ecosystems are specialised and as a result any imbalance in the hydrological regimes may lead to notable mortality events. These ecosystems do experience tree mortality that is caused by various human or natural disturbances. All developmental stages (Propagules, seedlings, sapling and trees) are influenced by mortality although mangroves that are in premature stages of development undergo higher mortality rates and mortality is density dependant. The early mortality of seedlings in definite conditions can determine the density of adult plants. Mangrove seedlings that grow from propagules that are small (1 g or less fresh weight) experience greater mortality compared to that that grow from large propagules (more than 10 g fresh weight) (Sousa *et al.* 2003). Our data showed that the average propagule weight at Kwelera was 5.4 g and at Tyolomnqa it was 5.1 g (Table 11), therefore the seedlings have a lower chance of success due to a lack of available resources from the propagule Sousa *et al.* (2003) but interestingly the mass of propagules was greater at the edge of range compared to that at Mngazana.

When comparing between the two estuaries, Kwelera had a higher moisture content and organic content compared to Tyolomnqa. Therefore, it is possible that as you move further south, physical properties of the soil changes. Higher organic content was measured in 2012 as compared to 2014. Higher moisture content may be as a result of tides coming into the mangrove forests from the sea. The amount of organic content found in these estuaries show that these are still young forests. In a study by Rajkaran and Adams (2011), the amount of high organic content that was found at Mhlathuze Estuary was associated with mangrove stands that were older. Mangroves are able to tolerate pH ranging from 6 to 8.5 (Duke 2006). Sediment pH is stated to be a characteristic that plays the principal role in nutrient recycling between the sediment and the water in an estuary (Shaikh and Tiwari 2012). Sediment pH at Kwelera and Tyolomnqa varied with depth, but was within the considered range, therefore sediment pH may not affect establishment of new species. Joshi and Ghose (2003) stated that pH notably influences plant growth and this is as a result of a change in the availability of vital and non-vital element. The growth and physiology of mangroves are affected by salinity. *Avicennia marina* is able to survive and tolerate high levels of salinity as compared to other mangrove species. Since porewater salinity was not greater than 38 PSU, it is not expected to affect the growth and expansion of mangroves. The importance of freshwater flowing into estuaries is highlighted as range. Mangroves require permanently open estuaries to grow successfully. Between Kwelera and Kariega Estuary, there are 6 permanently open estuaries, mangroves occur in 5 of them including Kwelera and Kariega (Table 12). Sediment that builds up in estuaries enabled the spread of mangroves into the intertidal areas. However, the establishment may be restricted by surround vegetation such as intertidal and supratidal salt marsh.

Table 12: Evidence of mangrove expansion and human disturbance in various estuaries in the Eastern Cape within (shaded in grey) and outside of the range of mangroves as per Ward and Steinke (1982).

Name of estuary	**Evidence of mangrove expansion**	**Observations of surrounding vegetation**	**Evidence of human disturbance**
Kwelera	No	*Phragmites australis* around mangroves and salt marsh on adjacent bank. *Spartina maritima* present	Houses with launching facilities near the mangroves
Gonubie	No	Mostly salt marsh	None
Nahoon	Yes	Mangrove and saltmarsh species present	Present, plastic pollution
Tyolomnqa	Yes	*Spartina maritima* present	Present, removal of mangrove seedlings
Keiskamma	No	Saltmarsh present	Present and they lead to increased freshwater abstraction
Great Fish	No		None
Kowie	No	Macrophytes and saltmarsh present	Present in lower reaches
Kariega	Yes (1 individual)	*Spartina maritima* present	Present, urban and agricultural demands leading to reduced freshwater inputs.

3.5 Conclusion

Quisthoudt *et al.* (2013) stated that mangroves in South Africa (namely, *A. marina* & *B. gymnorrhiza*) will colonize sites further than their recorded distribution as a result of global warming associated with climate change. An expansion in the distribution of mangroves forests has been observed in South Africa. Ward and Steinke (1982) noted that the planted mangrove forests at Nahoon was the previous limit of mangroves but this study shows that the successful planting of mangroves at Tyolomnqa and the natural dispersion and establishment of a *Bruguiera* propagule to Kariega Estuary has expanded the range by 141 km and further expansion is possible as the climate continues to change. This chapter agrees with the work done by De Ryck *et al.* (2016) that shows that *A. marina* has the ability to extend beyond their current distributional limit. Propagule supply and ocean currents are the main drivers for propagule dispersal. Changes in ocean currents have been observed and it is stated that they may have an effect on mangrove propagule dispersal and also on mangrove genetic structure (Lovelock and Ellison 2007). Ocean currents have a key role in determining where establishment and successful growth will occur. Two ocean currents bound the coast of Southern Africa, the cold Benguela current is found on the west coast, the warm Agulhas current found on the east coast of South Africa, and it moves south from the tropics. A stronger Agulhas current has been foreseen due to global warming (Lutjeharms and de Ruitjer 1996) and it will have an effect on factors such as rainfall, changes in estuarine functioning and the movement of species from tropical regions to areas that are more temperate.

Chapter 4 - Growth rates and nutrient dynamics of mangroves at Mngazana and Nahoon Estuary

4.1 Introduction

Mangrove forests are found in tropical and subtropical regions of the world, within 30° north and south of the equator (Spalding *et al.* 2010). Like all plants, the growth and productivity of mangroves is restricted by the supply and availability of essential nutrients (Alongi 2011). The availability of nutrients, high primary productivity and shallow waters in mangrove forests make these areas ideal for supporting food webs (Simard *et al.* 2008). Similar to other ecosystems, mangroves are subject to various disruptions that vary in nature (Alongi 2008). Tidal forces, freshwater runoff, rates of sedimentation and changes in sea level tend to influence the growth and survival of mangroves and changes to these factors may result in mangrove tree mortality (Jimenez and Lugo 1985).

Mangroves reproduce by sexual reproduction (Tomlinson 1994). Propagules of mangrove trees float after they have been released and ocean currents allow for their dispersal to various places. In their life cycle, propagule dispersal, establishment and recruitment to the sapling stages are the most critical stages (Padilla *et al.* 2004). This determines their ability to colonize new habitats and ultimately maintains the population. Before establishment takes place, processes such as dispersal and predation affects the patterns of distribution and abundance (McKee 1995). Tomlinson (1986) supported the self-planting theory where propagules fall from the parent plant and establish themselves just under the parent tree. After establishment occurs competition and other related factors are able to limit species to a section of the actual range in an estuary. Harper (1977) states that it is a very small quantity, of dispersed propagules, that survive to establish as seedlings and many of the propagules die soon after establishing. According to Padilla *et al.* (2004) and Alongi (2011), the survival and growth of mangrove seedlings that are recruited to the sapling stage are determined by light and nutrient availability. They also depend on factors like salinity, redox potential, tidal position, sediment disturbance (burial by sedimentation or erosion) and wave exposure. Chen and Ye (2014) also mentioned that the growth and survival of various mangrove species can be influenced by small changes to salinity (e.g. in *Excoecaria agallocha, Acanthus ilicifolius spp.*).

Ukpong (1997) states that the availability of nutrients is one of the three ruling components that determine the performance and productivity of mangroves. As nutrient availability increases in mangrove forests, plant growth also increases as a result of plants investing less in their roots and more on the above ground parts (Lovelock *et al.* 2009). Ukpong (1997) further state that plants exposed to high nutrients may be more sensitive to environmental stressors. In low nutrient environments, some species are more competitive because they have nutrient conservation mechanisms. According to Prasad and Ramanathan (2008), nutrient concentration levels in the sediment of mangrove forests are largely influenced by microbial activities.

An increase in temperature has been predicted with climate change and may lead to increases in growth of mangroves at their limits (southern) through increased photosynthetic rates over extended growing seasons and may also increase productivity. Growth rates of mangroves differ according to locations, species composition and the physical-chemical factors that drive mangrove ecosystems (e.g. salinity, light, nutrient availability etc.). According to Duke *et al.* (1998) mangroves may have stunted growth where salinity is greater than its tolerance range, and while equatorial and wet tropical mangroves are tall and dense this is as a result of high rainfall (1 500 mm), therefore salinity is low to moderate (< sea water).

A primary role of fresh water inflow is the dilution of sea water so as to create brackish conditions (<20 PSU). It also supplies external organic and inorganic materials into the estuaries and thus provides nutrients to the system. This leads to an enrichment and in turn leads to an increase in primary and secondary production. The importance of freshwater inflow is most obvious when there is decreased inflow as this leads to decreased rates of both primary and secondary production. Sediment is also delivered to estuaries from streams and rivers by freshwater inflow and this helps to build and stabilize tidal flats and wetlands (Whitall *et al.* 2007).

Natural tree mortalities have been documented by various authors including Pegg *et al.* (1980); Chapman (1976), Breen and Hill (1969) as a consequence of changes in the natural functioning of mangrove ecosystems. Jimenez and Lugo (1985) stated that temperatures between -3 to -11° C resulted in heavy defoliation and death in many areas of New Zealand, Australia and south eastern United states. In Gambia, mangroves died between January and March as a result of change in hydrological regimes (flooding) that resulted in mangrove hydrological regimes being modified (minor less than 5 cm, 6 weeks rapidly) and that led to mangrove inundation (Blasco 1983). Mangrove mortality (about 117 ha lost) was also recorded in Rufiji Delta (Tanzania), where El Nino floods occurred and inundated mangroves forest in that area (Erftemeauer and Hamerlyck 2005). Mbense *et al.* (2016) documented mangrove mortality at Kobonqaba Estuary (South Africa), whre inundation occurred for long perionds of time and thus lead to mortality. Cardona and Botero (1998) documented that mangrove tree mortality and biomass was associated with soil factors along the Caribbean coast of Colombia. Various soil properties were measured and their results showed that redox potentials were irregular and correlated with the amount of water and organic matter present in the soil; pH was within its normal range for mangrove soils (~6.5). There were sites that had dead and dwarfed mangrove trees and they had an average salinity of 75 PSU (min-52 PSU, max-100 PSU). Naturally, mangroves have a physiological tolerance mechanism when salinity is high. Just like other halophytes, their osmotic and water potentials decrease so as to conserve leaf turgor and excessive salt concentrations (Naidoo 1987). An interaction between nutrients (nitrogen and potassium) was studied by Naidoo (1987) and the results showed that when salinity is high, potassium and nitrogen concentrations in roots and shoots of *A. marina* decreases. When salinity increases, decreased conductance occurs and this indicates that water flux through the plant is minimized and the low osmotic potentials of high water salinity makes it necessary for internal osmotic adaptation (Naidoo 1985; 1986).

Mangroves that have been exposed to high concentrations of nutrients (especially nitrogen) tend to suffer high mortality rates during periods of drought. These concentrations are those that lead to eutrophication and low oxygen in the sediment. This is shown by Lovelock *et al.* (2009) where nutrients prompted mortality in sites that had periods of low humidity, low rainfall and high sediment salinity. Diseases (fungi) can also lead to a dieback or mortality of mangroves and the first reported incident was in Puerto Rico in 1920 on *R. mangle* (Stevens 1920). This is documented by Wier *et al.* (2000) where they did a study at Southwest Puerto Rico. A high frequency of dieback and mortality was found in *R. mangle*, 33% mortality in the field was recorded and 50% mortality in greenhouses was also recorded. *Cytospora rhizophorae* Kohlm. & E. Kohlm was associated with the mortality of mangroves in that area and Gilbert *et al.* (2008) also found butt and heartwood infections on mangrove species of several regions.

According to Mohammed and Johnstone (2002), sediments are known to be key sites for the collection of organic matter, successive remineralisation and also nutrient recycling. They have also been reported by Alongi (1994) to be sinks for nutrients. Not all mangrove ecosystems act the same way, considering nutrients. Mangroves are regarded to be generally oligotrophic ecosystems and so the species that grow there are considered to be adjusted to low nutrient conditions (Lugo 1989), but nutrient availability differs substantially between mangrove stands (Feller *et al.* 2003). Feller *et al.* (1999) state that plant species that have been acclimated to conditions of low nutrient availability in tropical and temperate latitudes, have nutrient conservation that is well advanced which also obtain, use and reuse nutrients productively.

Ilami *et al.* in (2002) at Harney River, Everglade Natural Park showed that porewater nutrients were concentrated at a depth of 30 cm. Phosphorus, nitrates + nitrites had a higher concentration in summer than in winter (0.05 -0.15 mg/L for TOxN and 0.02-0.04 mg/L for P), while the opposite trend was recorded for porewater ammonium. Magni and Montani (2006) in SW Japan showed that ammonium ranged between 75.1 and 908 µm, phosphorus (0.9 and 36.9 µm) and nitrates + nitrites between 0.2 and 21.1 µm. Ammonium had a higher concentration as compared to other nutrients studied and both ammonium and phosphorus were lower in winter in the uppermost layers and their concentration increased in summer.

The study by Penha-Lopes (2009) on East African mangroves showed the effects of domestic sewage on the characteristics of mangroves, high domestic waste leads to high nutrient concentrations and disturbs top sediment. Vaiphasa *et al.* (2007) showed the effect of shrimp derived waste on the growth of mangroves in Thailand. Lower mean monthly growth rates (~ 0.35 cm) were recorded for *A. marina* in the affected areas. Salt pans in Tanzania have been recorded to elevate local soil salinity therefore resulting in hypersaline conditions and this affects growth and productivity of mangroves (Faki 2003). Another study by Hwang and Chen (2001), examined the growth responses of *Kandelia candel* (L.) Druce to variations in nitrogen, phosphorus and salinity showed that both salinity and the nutrients had an effect on the growth of *K. candel* and that salinity > 20 PSU reduced growth. Rajkaran (2010) found that the growth rate of species at Mngazana Estuary, South Africa differed from one species to the other. The growth rate of *A. marina* increased from regeneration size class 1 (RC1) (<50 cm) to size class 5 (S5) (>350 cm) in height, growth rate of *R. mucronata* was the highest in RC3 (151-250 cm) and the growth rate of *B. gymnorrhiza* stopped changing at a height of 151 cm. DBH growth rates ranged between 0.7-2.3 yr^{-1}. No correlation between growth and nutrients was done since nutrient levels in the sediment were not measured. Growth rates of height and DBH were recorded at Nahoon Estuary, South Africa by Geldenhuys (2013) and they were higher than that measured by Rajkaran (2011). Saplings of *Avicennia marina* were growing at a rate of 14.4 cm yr^{-1}. Nutrient deficiency and pore water characteristics (low temperature, high salinity and high pH) were negatively affecting the growth of mangroves in this forest. Apart from Geldenhuys (2013) no other studies done in South Africa have investigated the relationship between nutrients and growth rates.

The aim of this chapter is to determine the growth rates and environmental conditions of mangroves at Mngazana and Nahoon Estuary. By achieving this aim, we will further develop and continue to maintain a long term dataset of growth rates of mangroves in South Africa. A long-term dataset of growth rate data will assist in predicting changes in growth rates as climate parameters such as minimum and maximum temperature and rainfall begin to change.

Objectives of this chapter are:

1. To measure the porewater characteristics (salinity, temperature and electrical conductivity and nutrient pool) in the two mangrove forests.

2. To determine the growth rates of different sizes classes of *Avicennia marina, Bruguiera gymnorrhiza and Rhizophora mucronata* at Mngazana Estuary, the third largest mangrove forest in South Africa and that of *Avicennia marina* at Nahoon Estuary, a planted mangrove forest near the distributional limit of mangroves.
3. To determine if porewater characteristics are the main drivers of growth for the mangroves at Mngazana and Nahoon Estuary.

4.1.2 Study site descriptions

Mngazana Estuary

The Mngazana Estuary is situated at 31°41'29" S; 29°25'24" E, close to the town of Port St Johns. It is found along the Wild coast of the Eastern Cape Province South Africa, close to the biogeographic boundary between the warm temperate and sub-tropical zones (Figure 12). It is a permanently open estuary that is approximately 5.3 km in length that has a river that flows through 275 km^2 of catchment for 150 km, before discharging to the ocean (Rajkaran and Adams 2012). The estuary has large sand banks in the lower reaches and mouth region, as a result of a large tidal prism the mouth is kept permanently open to the sea (Mbande *et al.* 2002). Three species of mangroves *B. gymnorrhiza*, *A. marina* and *R. mucronata* are found in Mngazana Estuary. (Adams *et al.* 2004). It is the third largest area of mangroves in South Africa covering an area of 118 ha (Rajkaran 2011). In the middle and the lower reaches (1 085 m, 300 m from the mouth), two creeks are found (Rajkaran and Adams 2007) and they provide additional sources of fresh water to the estuary and provide additional habitat for flora. It is the largest stand of *Rhizophora mucronata* in the country.

Steinke (1999) stated that it is one of the great southerly mangrove systems in South Africa which bears a diverse marine flora and fauna; it also has a high conservation priority ranking. During winter, water temperatures decrease from the mouth to the head and during summer, water temperatures increase distinctly from the mouth to the head (furthest point of tidal influence). This estuary has a salinity range similar to that of sea water (30-35 % PSU) (de Wet, 2004). According to Sgwabe *et al.* (2004), the estuary is free from pollution, has good water quality and is well oxygenated. In 2005, nine sites where set up in various parts of the estuary to measure the growth rates of the three species (Figure 12).

Weather data (1950 to 2015) was obtained from a weather station near Port St Johns (Figure 13).

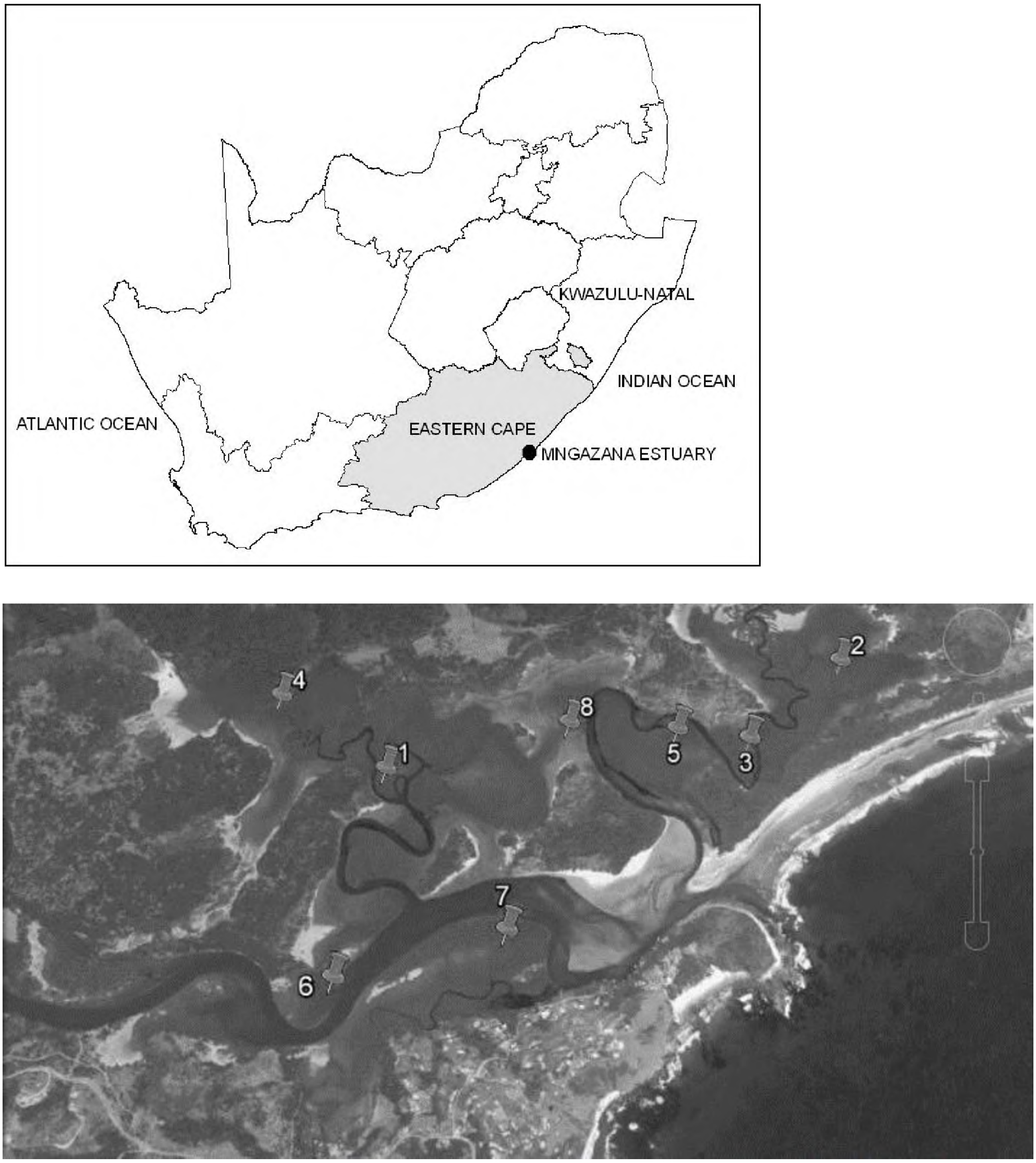


Figure 12: Map of Mngazana Estuary showing sites studied (taken from Google Earth). Sites 2,3,5,8 are situated on Creek 1, Site 1, 2 and 4 on Creek 2 and Sites 6 and 7 are situated on the Main Channel

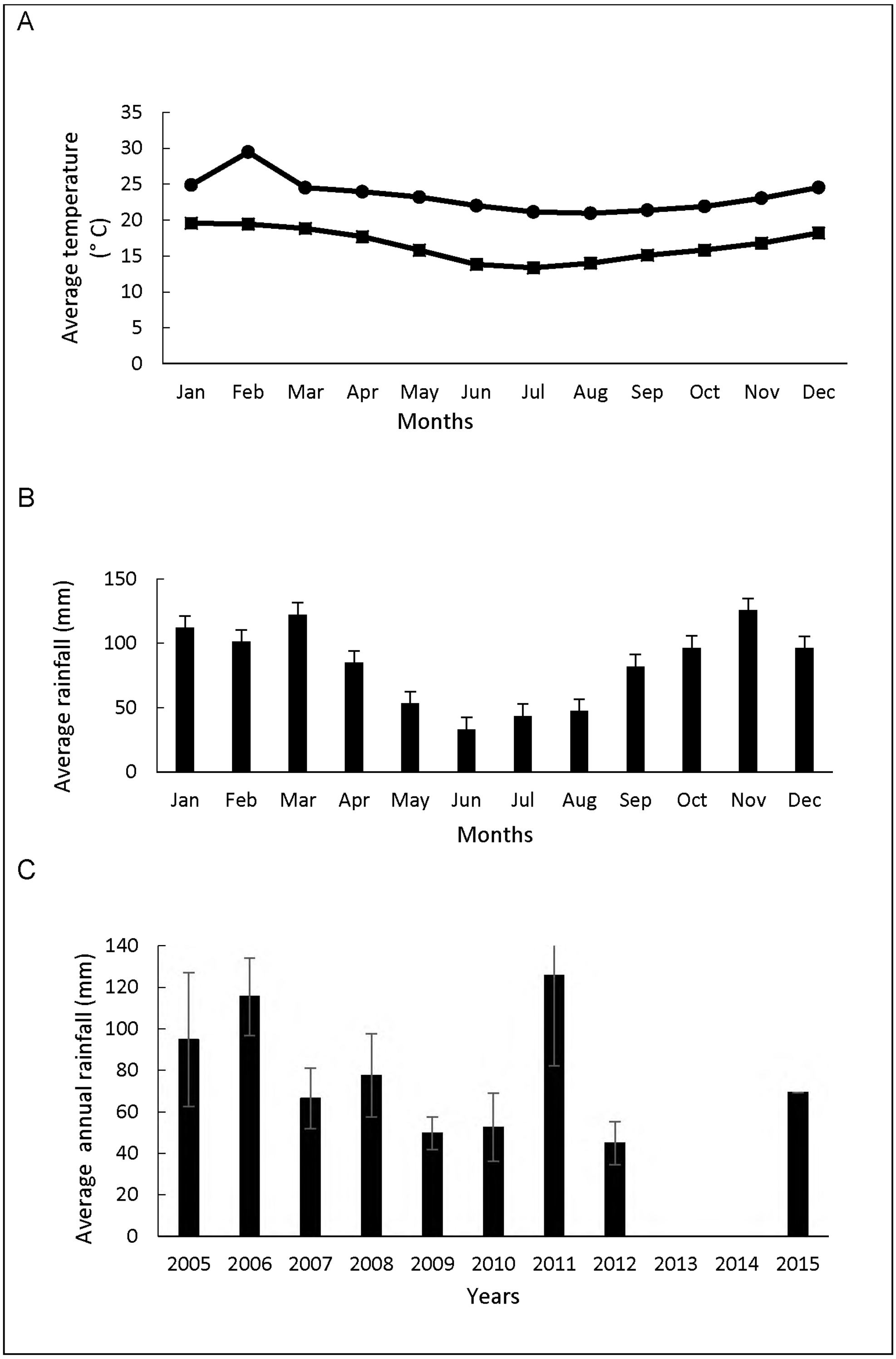


Figure 13: Average rainfall per month over the years and average annual rainfall (± SE) at Mngazana Estuary. No data was found in 2013 and 2014 in figure (9C).

Diagram A: (**Circles= maximum and squares= minimum**)

Nahoon Estuary

Nahoon Estuary is a warm temperate estuary which is permanently open to the sea, situated at 27°57′05″ E, 32°59′05″ S (Figure 14). It is found on the southeast coast of South Africa, which is wave dominated and surrounded by the metropolitan area between East London and Beacon Bay. Waves occur mainly from the southwest and result in long shore sediment movement in a north-easterly direction (Bursey and Wooldridge 2002). It has a catchment area of approximately 547 km^2 and a tidal prism of 6.3 x 10 m^3 (Copper 2002). The estuary measures approximately 4 km in length with an average depth of 4 km. The estuary has a warm temperate climate that has annual temperature ranging from 13 to 25 °C. Maximum temperatures are 31 °C and minimum temperatures in winter are 5.3 °C. The average annual rainfall is 76.10 mm (Figure 15).

Nahoon Estuary receives its freshwater supply from the Nahoon Dam. A decrease in freshwater supply has been recorded in the literatures and it has been attributed to various features e.g. Bridges built, Abbotsford causeway that narrows flow of water downstream (Wiseman *et al.* 1993). Reddering (1988) classified Nahoon as being microtidal (tidal range < 2 m), and has an average tidal range of 0.76 m and a coastal spring tide range of 1.6 m. Decreased freshwater inflow and close residential proximity have an impact on the estuary; and as a result, plant communities cover small areas and are fragmented (Coetzee *et al.* 1996). The estuary is used for recreational activities which include fishing, canoeing etc. (Talbot *et al.* 1985).

Mangrove and saltmarsh species are found at Nahoon Estuary. The mangrove forest was planted by Prof Trevor Steinke in 1969; the reason behind this was to determine if the trees would be able to survive at higher latitudes. Three species from Durban Bay were planted but only two species survived and were able to spread and colonize the area, the third species, *Rhizophora mucronata* is constrained to less than 20 individuals. Later, Prof Steinke hypothesized that the stand further up the estuary may have established due to natural dispersal of propagules and by 1982, an increase in the population size was seen and 25 trees were recorded near the mouth (Morris 1986).

In a study by Hoppe-Speer *et al.* (2015), an interaction between saltmarsh species and mangroves was documented, climate and habitat were the main cause of the interaction. Mangroves were found to be occupying areas with saltmarsh and it was expected that the presence of mangroves in those areas might lead to a decrease in saltmarsh as previous studies have delivered compelling evidence on the inability of saltmarsh to grow under the shade of mangroves. A comparison of climatic measurements is summarised in Table 13, summer rainfalls are higher than winter rainfalls, which is similar to Mngazana Estuary.

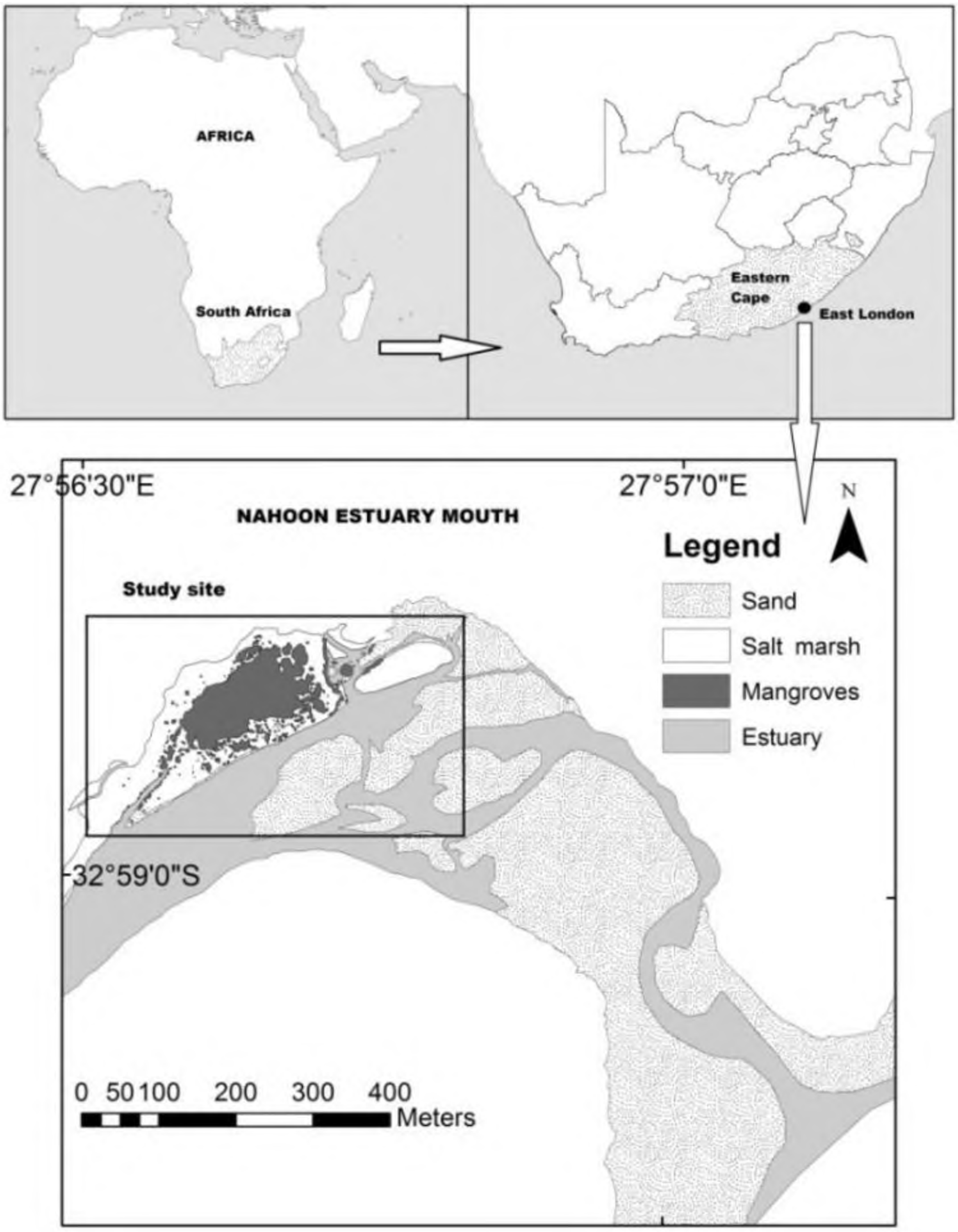


Figure 14: Map of the Nahoon Estuary showing the lower intertidal areas. Three transects were set up in 2011 in the area surrounded by the black box. Growth sites were located along transects. (Taken from: Hoppe-Speer 2015).

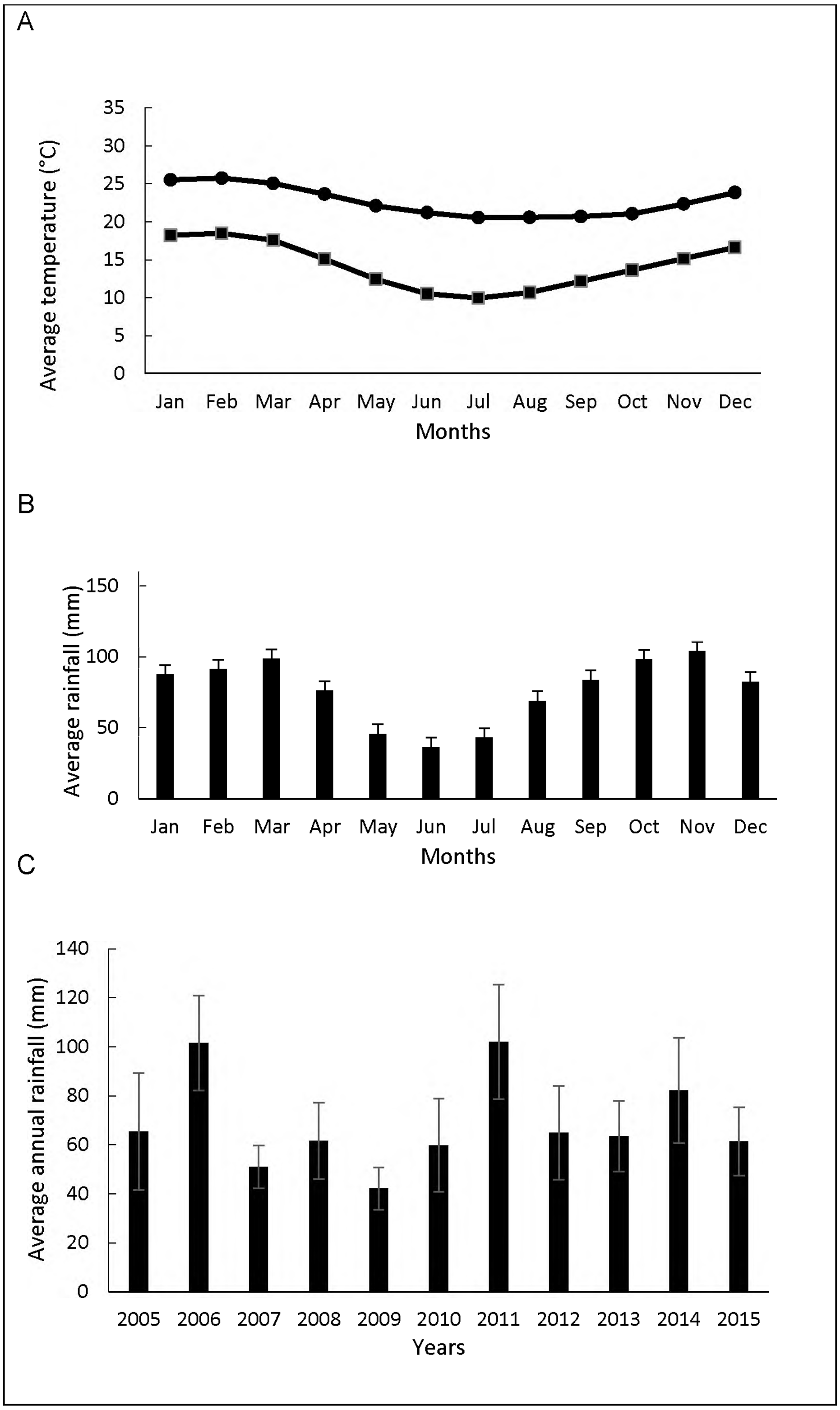


Figure 15: Climate data collected of Nahoon Estuary showing monthly maximum and minimum temperatures (°C), average rainfall (mm) from 1950-2015 and average annual rainfall (Mean ± SE).

Diagram A: **(Circles= maximum and squares= minimum**)

Table 13: Climate data of Mngazana and Nahoon Estuary

Estuary	Minimum yearly temperature (° C)	Maximum yearly temperature (°C)	Average yearly temperature (°C)	Average yearly rainfall (mm)
Mngazana	11.1	28.4	19.8	81.7
Nahoon	9.9	26.2	18.1	76.2

To achieve the objectives of this chapter, eight sites at Mngazana and four sites at Nahoon were used to determine the nutrient availability and growth of the mangroves. These sites were set up in 2005 for Mngazana and 2011 for Nahoon (Plate 4, 5, 6). Table 14: summarises the characteristics of these sites.

Table 14: Summary of mangrove characteristics of Mngazana and Nahoon Estuary

Estuary	Site	Dominant species	Distance to nearest body of water (main channel or creek) (m)	Tidal regime	Location along the creek or estuary
Mngazana Estuary	1	*Rhizophora mucronata*	24	Inundated by every tide	Creek 2 - Upper
	2	*Bruguiera gymnorrhiza*	223	Inundated by normal/spring tide	Creek 1 - Upper
	3	*Bruguiera gymnorrhiza*	122	Inundated by normal/spring tide	Creek 1 - Middle
	4	*Avicennia marina*	125	Inundated by normal/spring tide	Creek 2 - Upper
	5	All species present	71	Inundated by every tide	Creek 1 - Middle
	6	*Avicennia marina*	67	Inundated by every tide	Main channel - Middle
	7	*Avicennia marina*	49	Inundated by every tide	Main channel - Lower
	8	*Bruguiera gymnorrhiza*	86	Inundated by every tide	Creek 1 - Lower
Nahoon Estuary	1	*Avicennia marina*	46	Inundated by spring tide	Main channel - Lower
	2	*Avicennia marina*	105	Inundated by spring tide	Main channel – Upper
	3	*Avicennia marina*	63	Inundated by spring tide	Main channel – Middle
	4	*Avicennia marina*	47	Inundated by spring tide	Main channel – Lower

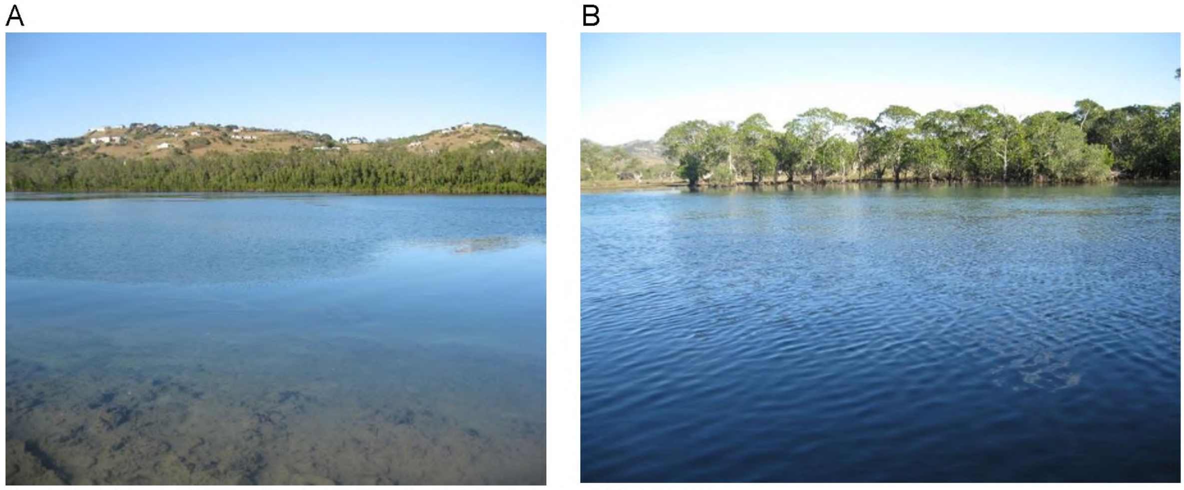


Plate 4: View of Site 7 and Site 8 at Mngazana Estuary

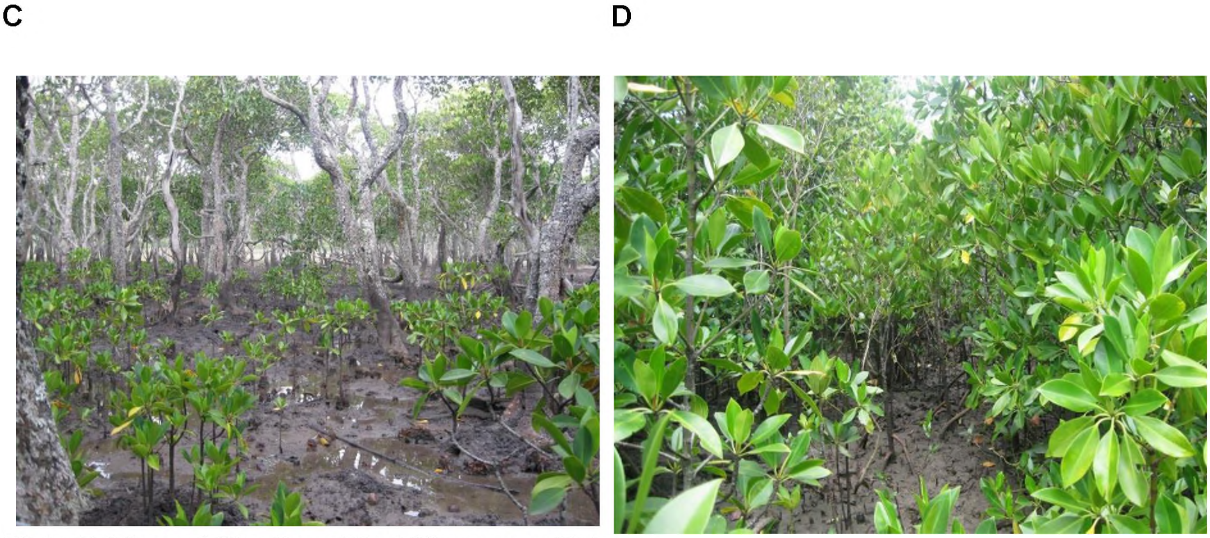


Plate 5: View of Site 3 and 5 at Mngazana Estuary

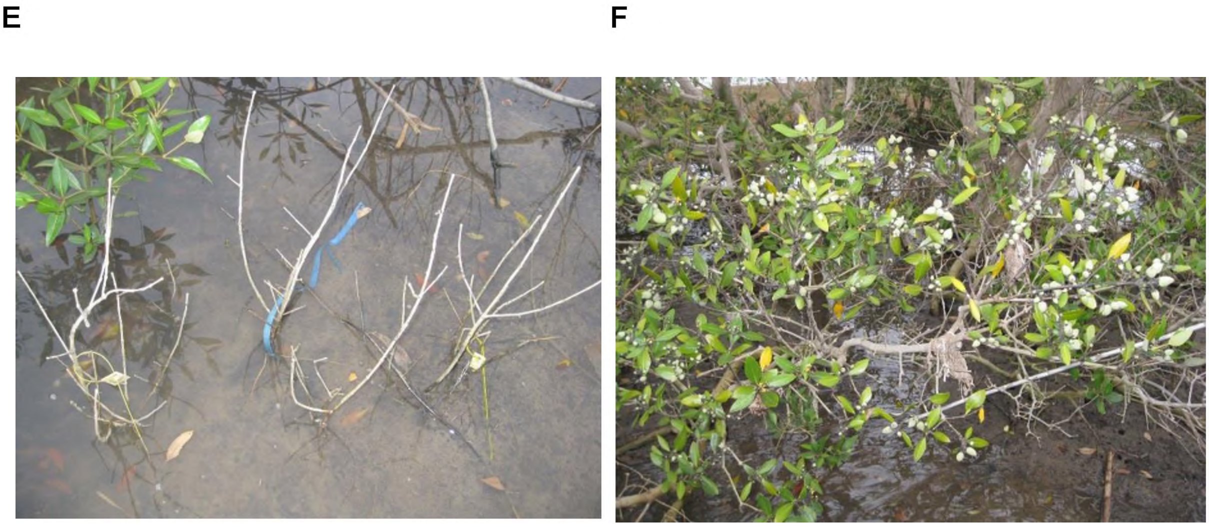


Plate 6: Showing dead *A. marina* trees and propagules at Nahoon Estuary

4.2 Materials and Methods

Sampling was undertaken in summer and winter in 2014 and 2015. Growth was measured once a year while nutrients were collected on all sampling trips.

Porewater and nutrient analysis

Porewater characteristics were measured in triplicate in augured holes at each site. The water was allowed to fill the augured holes so that temperature, salinity and conductivity could be measured using a YSI multimeter or a refractometer and a thermometer. Porewater samples were collected using syringes, filtered with cellulose acetate sterile membrane 0.45 µm pore size syringe filters, they were then frozen and stored <0 °C. They were transported to the lab so as to analyse the concentration of ammonium, nitrates + nitrites and phosphorus nutrients. Ammonium and soluble reactive phosphorus was analysed using standard spectrophotometric methods by Parsons *et al.* (1984) and nitrites and nitrates were analysed using a reduced copper cadmium method by Bates & Heelas (1975). A dataset of sediment characteristics for Mngazana (2010) and Nahoon (2012) already exists and it was not expected to change overtime, these collections were not repeated (Rajkaran, 2011; Hoppe-Speer, 2012).

Growth and mortality rates

To determine the growth and mortality rates of mangroves a number of quadrats (2-3) were set up, trees were tagged with a number plate and cable tie. These were repeatedly measured for vertical growth (height) and DBH for trees > 130 cm. Each site at Nahoon has three quadrats and only *A. marina* is found there. Measurements took place in February 2014 and February 2015. Data from 2011 to 2012 was obtained (Hoppe-Speer, 2012, Geldenhuys, 2013). To determine the mortality in both estuaries, dead tagged trees were counted and recorded (Plate 6E). This did not include lost tags or those that were found lying around or buried in the sediment. Site 1 at Nahoon had a lot of dead trees due to deposition that occurred, new quadrats were created and trees were tagged and measured. The data collected was divided into the following size classes; RC 1 (<50 cm), RC2 (51-150 cm), RC3 (151-250 cm), S4 (251-350 cm) and S5 (>350 cm). Individuals that had a height of > 130 cm were divided into the following size classes to monitor DBH: 0-0.9 cm, 1-1.9 cm, 2-2.9 cm, 3-3.9 cm and >4 cm.

Data Analysis

Data was analysed using STATISTICA (12 64-bit). Normality was tested by looking at the skewness and kurtosis and significance difference was determined at $p < 0.05$. If data was normally distributed One-Way ANOVA was used and if data was not normally distributed, a Kruskal–Wallis ANOVA ($H_{(df,N)}$) and Wilcoxon T test (T-stat=) was used to determine differences while a multiple comparison of mean rank was used to further test between individual means. Correlation analysis was also measured using correlation matrices.

4.3 Results

Porewater salinity varied between seasons (summer and winter) and years. Porewater salinity at Mngazana Estuary ranged from 18.4 to 45 PSU and was significantly higher in 2015 than in 2014 ($F_{(11, 36)}$ =3.9, $p<0.05$). Overall average porewater salinity at Mngazana Estuary was 35.2 ± 0.4 PSU. Variations in salinity within sites across dates were evident and seasonal differences are summarised in Table 15 and Table A1. Salinity at Nahoon Estuary ranged from 16 - 41 PSU, salinity was similar between seasons (summer 33.5 ± 1.3 PSU; winter (31.9 ± 1.4 PSU) ($H_{(10, N=24)}$ =9.5, $p> 0.05$), years ($H_{(10, N=24}$ =11.7, $p>0.05$) and within sites ($p>0.05$) (Table 16). Salinity was similar between the two estuaries in 2014, but was significantly higher at Mngazana in 2015 ($H_{(19, N=48)}$ =25.4, $p<0.05$).

Porewater temperature at Mngazana ranged from 7 – 30 °C. When comparing between seasons, temperature was higher in summer (23.2 ± 0.3 °C) than winter (16.8 ± 0.4 °C) (T=1.5, $p<0.05$) (Table 15). As expected average porewater temperature at Mngazana Estuary was significantly higher than Nahoon Estuary ($H_{(9, N=48)}$ =36, $p<0.05$). Seasonal changes in temperature was also noted and recorded in Table 15 and Table A2. Porewater temperature at Nahoon was similar between the years ($p> 0.05$) and it ranged between 15- 26 °C. Seasonal changes were noted and recorded in Table 16 and Table A4. Temperature was similar in summer 2014 and 2015 in all the sites studied.

Porewater conductivity at Mngazana Estuary ranged between 25.3 - 66.2 mS, conductivity in winter and was significantly higher than summer (T=178.5, $p< 0.05$). Seasonal variations in conductivity was noted, Site 1-8 conductivity was higher in winter 2015 compared to summer 2014. Other seasonal changes and differences are recorded in Table 15 and Table A3. Porewater conductivity at Nahoon Estuary ranged between 24- 61 mS, it was higher in 2014 than in 2015 ($F_{(1, 46)}$ =2.4, $p<0.05$ and was similar between the seasons ($p>0.05$). Seasonal changes in porewater conductivity was mostly recorded in Site 1 and Site 3 (Table A5) and no seasonal variations were recorded in summer 2014 and 2015 in all the sites studied (Table 16 and Table A5).

Table 15: Porewater characteristics measured in 8 sites at Mngazana Estuary over two seasons in 2014 and 2015 (Mean ± SE, N=3).

Site	**Salinity (PSU)**				**Temperature (ºC)**				**Conductivity (mS)**			
	Feb 2014	**Jun 2014**	**Mar 2015**	**Jun 2015**	**Feb 2014**	**Jun 2014**	**Mar 2015**	**Jun 2015**	**Feb 2014**	**Jun 2014**	**Mar 2015**	**Jun2015**
1	29.6±2.73	35.5±0.47	41.3±2.30	39.3±0.7	24±0	16.9±0.86	22.9±0.06	17± 0.6	45.8±3.8	53.4±0.7	56.1±0.0	58.8±0.9
2	28.6±1.86	34.1±0.53	34.6±0.38	39.3±0.7	25.3±0.33	18.7±0.43	22.7±0.38	16±0.2	44.4±2.6	54.6±0.1	53.1±0.2	58.8±0.9
3	33.3±1.45	37.3±0.37	37.4±0.71	39±1	24±0.58	19.1±0.31	22.1±0.3	17.5±0.3	50.8±1.9	55.9±0.5	57.5±0.5	58.4±1.3
4	28.3±1.67	36.3±0.48	39.6±0.42	39.3±1.7	24.5±0.29	16.9±0.55	23.4±0.21	17±0.6	43.9±2.3	54.5±0.8	59±0.5	58.8±2.2
5	32.6±0.33	35.6±0.75	35.6±0.45	37.7±0.3	24.6±0.44	18.3±0.13	18.4±0.23	18.7±0.4	48.9±0.03	54.6±0.4	54.6±0.2	56.6±0.4
6	30.3±0.33	35.9±0.15	37.7±0.35	37.7±1.3	26.6±0.33	18.7±0.58	22.2±0.06	18±0.6	46.7±0.5	47.7±0.6	56.6±0.2	56.6±1.8
7	31±0	35.9±1.23	35.8±0.1	38.3±0.3	26.7±0.3	18.7±0.1	22.1±0.06	18.7±0.7	47.6±0	46.1±9.9	54±0.1	57.5±0.4
8	30.6±0.67	29.4±5.52	33.7±1.00	40.3±0.3	25.3±0.33	11.03±4.1	25.2±2.21	19.1±0.6	47.1±0.9	39.03±6.9	53.5±0.3	60.2±0.4

Table 16: Porewater characteristics measured over two seasons at Nahoon Estuary in 2014 and 2015 (Mean±SE).

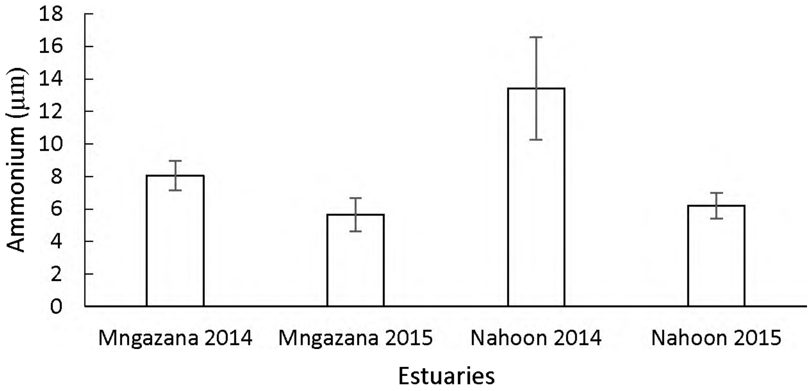
Site	**Salinity (PSU)**				**Temperature (ºC)**				**Conductivity (mS)**			
	Feb 2014	**Jun 2014**	**Mar 2015**	**Jun 2015**	**Feb 2014**	**Jun 2014**	**Mar 2015**	**Jun 2015**	**Feb 2014**	**Jun 2014**	**Mar 2015**	**Jun 2015**
1	36.0±1.3	40.1±0.1	38.4±0.9	27.6±1.7	23.3±0.3	17.6±0.7	22.1±6.4	15.8± 0.2	51.3±1.8	59.8±0.1	54.4±1.7	42.8±2.3
2	29.6±6.9	22.2±0.03	20.3±6.4	27.6±1.3	23.3±0.3	20.4±1	22.3±0.1	15.1± 0.1	45.4±9.7	35.0±0.1	39.5±1.3.	42.7±2.0
3	36±2.1	40.1±0.1	37.8±1.1	27.1±1.5	22±0.6	17.3±0.3	21.8±0.4	15.5± 0.4	54.4±2.8	59.8±0.1	53.6±0	42.2±1.9
4	34±2.5	41.6±1.7	30.7±2.2	31.0 ±0.4	23.6±1.2	17±0.6	23.4±1.2	16.0± 0.1	51.6±3.4	59.7±5.0	44.7±0.1	45.1±1.4

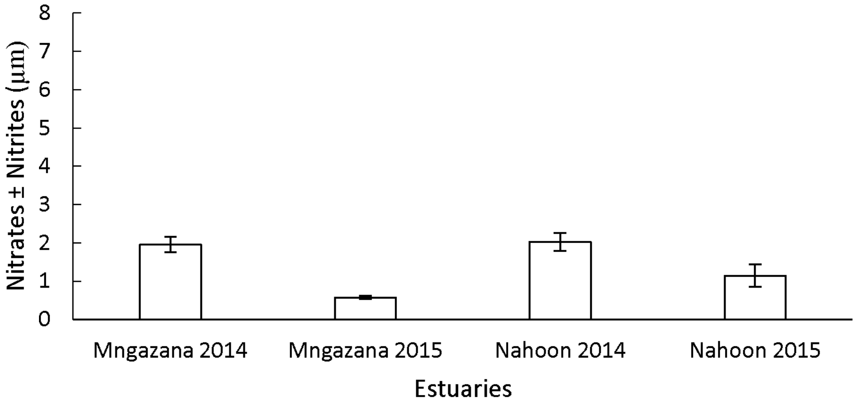
Nutrients

Ammonium concentration was higher at Nahoon Estuary compared to Mngazana Estuary in 2014 (T=23.0, $p < 0.05$), but were similar in 2015 ($p > 0.05$). Overall concentration at Nahoon Estuary was 10.1 ± 1.7 µm compared to Mngazana Estuary (6.9 ± 0.7 µm). Statistical analysis (F-test), between the two estuaries, show that the overall ammonium concentration was similar ($p > 0.05$) (Figure 16).

TOxN concentration at Mngazana and Nahoon Estuary in 2014 ranged between 0.1 - 5.0 µm and 0.2 - 8.5 µm, respectively and concentration levels were similar between the two estuaries in 2014 and 2015 ($p > 0.05$). Overall concentrations averaged 1.3 ± 0.1 µm at Mngazana and 1.6 ± 0.2 µm at Nahoon Estuary.

Phosphorus concentration at Mngazana Estuary ranged between 0 - 9.4 µm and 0.4 - 8.5 µm for Nahoon Estuary. Soluble reactive phosphorus concentration in 2004 and 2015 between the two estuaries was similar ($p > 0.05$). Overall SRP concentrations at Mngazana and Nahoon Estuary averaged 1.3 ± 2.1 and 1.7 ± 2.1 µm respectively. Other seasonal changes in concentrations are recorded in Figure 17(a),(b) and (c) for Mngazana and Figure 18(a),(b) and (c) for Nahoon.





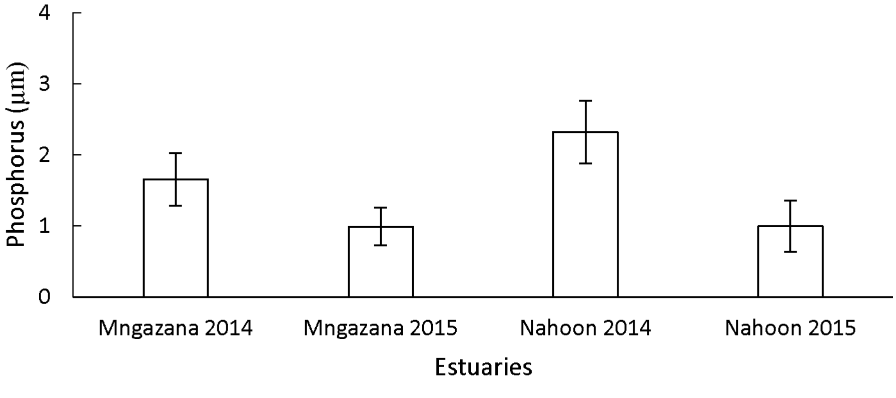


Figure 16: Comparison in NH4, TOxN and SRP concentrations of Mngazana and Nahoon Estuary (Mean ± SE).

Mngazana

Ammonium ranged from 2.4 – 19.6 µm across all sites (Figure 17a). Ammonium concentrations were similar in winter 2014 and 2015 ($p > 0.05$). Seasonal changes in concentration were recorded only in Site 2, Site 5 and Site 6 (Table A6). Overall ammonium concentration was similar in all the sites ($H_{(8,N=12)=}10.2$, $p > 0.05$). When comparing between sites, Site 5 had the highest NH_4 concentration in summer 2014 and 2015 compared to the other sites (Figure 17a).

Porewater TOxN ranged between 0.2 – 4.1 µm in all the sites. Average TOxN in summer 2014 was siginificantly higher (3.3 ± 0.1 µm) than summer 2015 (0.5 µm) (Figure 17b, Table A7). TOxN concentrations were similar in winter 2014/2015 (Table A7). Site 5 had the highest concentration in summer 2014 and 2015 as compared to the other sites (Figure 17b).

Porewater SRP concentration was higher in summer as compared to winter, and concentration were similar between winter sampling times (Figure 17c, Table A8). No seasonal variations were noted in Site 1, Site 3, Site 7 and Site 8 between the sites, and are excluded from Table A8. Site 5 had a higher concentration in winter 2014 and 2015 as compared to the other sites (Table A8).

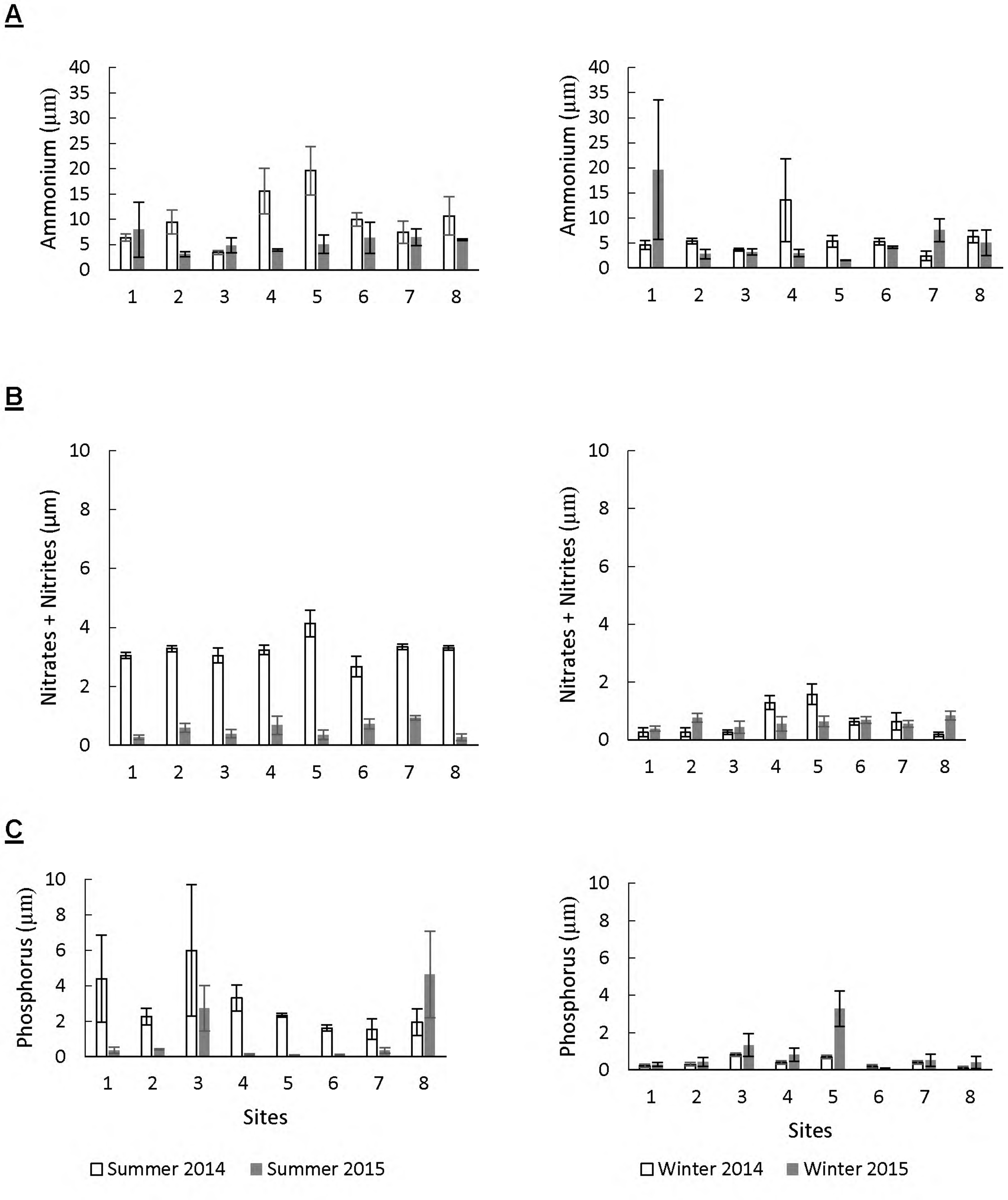


Figure 17: Summer and winter NH4 (A), TOxN (B) and SRP concentrations (C) (Mean ± SE) at Site 1-8, Mngazana Estuary.

Nahoon

Overall ammonium concentrations at Nahoon were higher in 2014 compared to 2015. Highest concentrations between seasons was recorded in winter 2014 (17.39 µm) and generally concentrations were higher in winter (12.1 ± 3.2 µm) as compared to summer (8.1 ± 0.9 µm). A Kruskal-Wallis test showed that NH_4 concentrations were similar between summer 2014 (10.3 ± 1.5 µm) and summer 2015 (5.9 ± 0.9 µm) ($H_{(7,N=22)}$=8.6, p > 0.05) and between winter 2014 and winter 2015 for all sites ($H_{(6,N=12)}$=5.0,p>0.05) (Figure 18). Overall NH_4 concentration in Site 1 was significanly higher than Site 4 (T=9.54,p< 0.05), and concentration between other sites was similar (p>0.05).

Porewater TOxN ranged between 0.23 – 3.25 µm and were higher in 2014 as compared to 2015 at Nahoon (T=58.5,p< 0.05). TOXN concentrations in summer 2014 was higher than summer 2015 (T=0.0, p< 0.05) and was similar in winter 2014 and 2015 ($H_{(4,N=12)}$=9, p> 0.05) (Figure 18). No site differences were recorded (p>0.05).

Porewater phosphorus ranged between 0.2-5.6 µm at Nahoon and the concentration was higher in 2014 (2.3 ± 0.4 µm) compared to 2015 (0.9 ± 0.3 µm). SRP in summer 2014 had a higher concentration than summer 2015 for all sites (T=1.0, p< 0.05) (Figue 18). There was no significant difference of SRP concentration in winter 2014 and winter 2015 (p>0.05) but Site 2 had the highest concentration as compared to other sites.

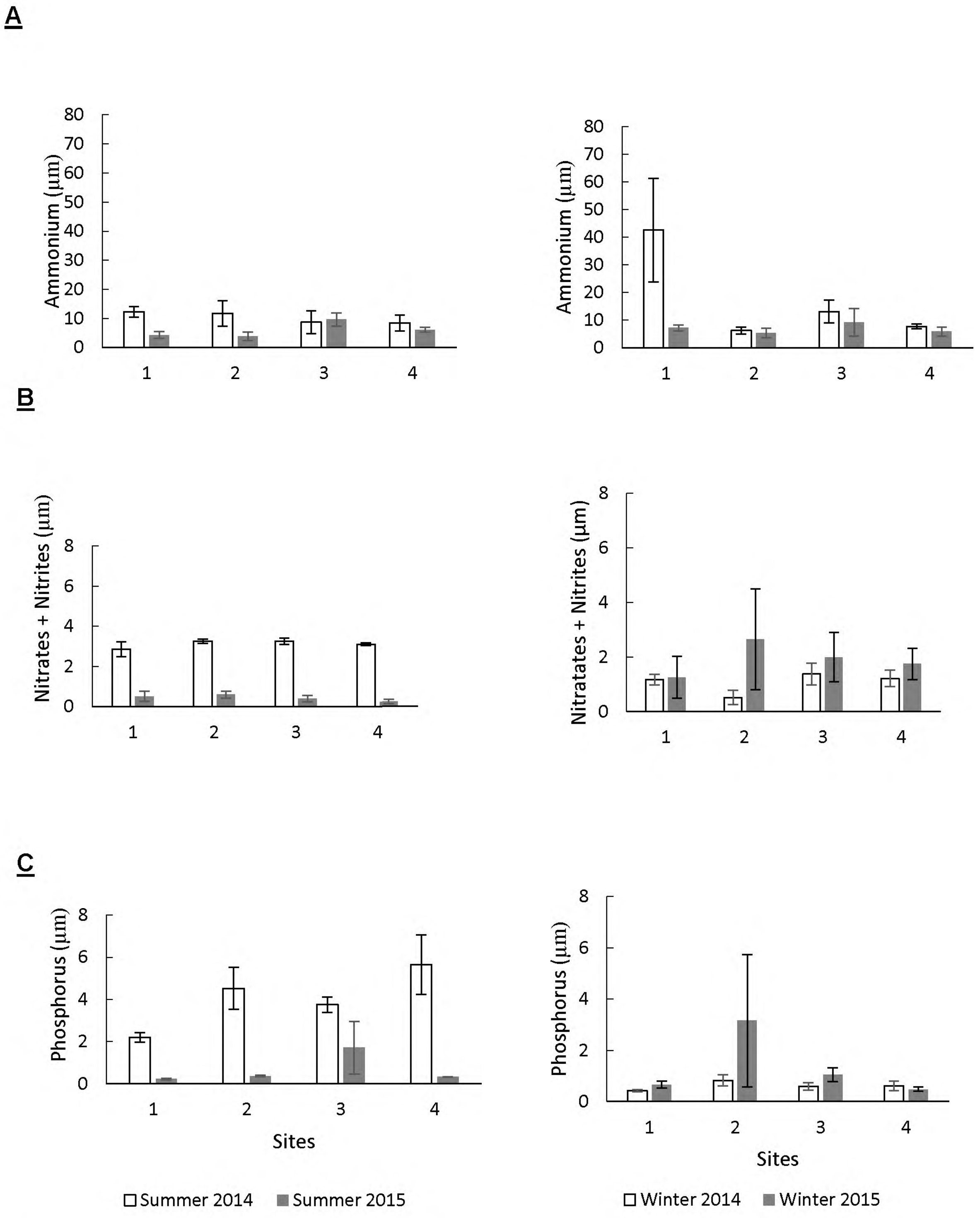


Figure 18: NH4, TOxN and SRP concentrations of Nahoon Estuary. (Mean± SE).

Tree growth (Height and DBH)

An increase in average vertical and horizontal growth rate (cm yr. $^{-1}$) occurred at all sites in the study areas (Table 17). Average vertical growth differed between estuaries and between sites. Overall average vertical growth rate of RC1 *(A. marina)* between Mngazana and Nahoon Estuary was similar ($p > 0.05$), while the overall growth rate of *A. marina* for the other size classes (RC2, RC3, S4 and S5) at Nahoon was higher as compared to Mngazana Estuary ($p < 0.05$) (Table 19). There was a significant difference in overall average vertical growth rate between species at Mngazana. RC3 had a higher growth rate compared to RC1, RC2, S4 and S5 ($H_{(14,\,N=55)} = 12.8$, $p<0.05$) for all the species studied. Average horizontal growth (cm yr^{-1}) was similar between the size classes ($p>0.05$).

Mangrove trees at Nahoon grew faster in Site 1 than in Site 2 ($F_{(25,\,12)} = 4.9$, $p < 0.05$), while growth was similar between Site 3 and Site 4 ($F_{(25,\,34)} = 1.5$, $p>0.05$. Growth rate between RC3 and S5 was similar ($F_{(20,\,13)} = 1.6$, $p>0.05$). S4 had a higher growth rate as compared to the other regeneration size classes, and it had an average growth rate of 23.4 cm yr^{-1}. Average horizontal growth rate was similar between the size classes. Average vertical growth rate was lower in the Regeneration Size Classes (RC1, RC2 and RC3) as compared to the other size Classes (Table 19). For Site 1 and Site 4, S5 had a lower growth rate per year.

Avicennia marina (AM)

A. marina was found in Sites 4, 5, 6 and 7. The average growth rate at Mngazana differed between sites; Site 4 (4.8 cm yr.$^{-1}$) had the highest growth rate as compared to the other sites. In addition, when comparing between the size classes, S5 (Average of *A. marina* sites) (7.2 cm yr.$^{-1}$) had the fastest growth rate as compared to RC1, RC2, RC3 and S4. Overall DHB growth was mostly seen in size class 1-1.9 to >4 cm (Figure 20).

At Nahoon Estuary only *A. marina* was found and S4 had the highest average growth rate (23.4 ± 4.5 cm yr^{-1}) as compared to other size classes (Figure 19, Table 19). When comparing between sites, Site 2 (19.9 cm yr. $^{-1}$) had the highest growth rate as compared to Site 1, Site 3 and Site 4 (Table 18). The horizontal growth rate of trees (DBH) at Nahoon did not follow any uniform rise or fall, and average DBH was different between sites. Site 3 had the highest increase as compared to Site 1, Site 2 and Site 4. Individuals in size class 0-0.9 cm were not found in Site 1, but overall average DBH growth in size class 0-0.9 cm was similar with size class 1-1.9 cm ($F_{(0,82)} = 3.9$, $p>01.05$) (Table 20). Average DBH growth for size class 0-0.9 cm was significantly higher than 2-2.9 and 3-3.9 cm ($F_{(1,\,94)} = 7, 2$, $p<0.05$) and average DBH growth for trees >4 cm was higher than size class 0-0.9 cm ($F_{(1,149)}$, $=17$ $p<0.05$) (Table 20). Overall DBH growth was mostly seen in size class > 4 (0.9 ± 0.1 cm).

Rhizophora mucronata (RM)

R. mucronata was found in Sites 1 and 5. In Site 1, the individuals in S4 had a similar average growth rate as compared to RC1, RC3 and S5 (6.3 ± 9.9 cm yr. $^{-1}$) and S5 had the slowest average growth rate as compared to the other size classes. No significant difference was found between them ($p > 0.05$). Average growth rate was significantly different between the size classes ($p < 0.05$). Average DBH growth in Site 1 and site 5 was similar in all the size classes ($p > 0.05$)

Bruguiera gymnorrhiza (BG)

B. gymnorrhiza was found in Site 2, 3, 5 and 8 (Plate 5). The overall average growth rate of *B. gymnorrhiza* increased from seedlings (2.1 cm yr^{-1}) to adults (6.7 cm yr^{-1}), a decrease in average growth rate was seen in S5 (0.1 cm yr^{-1}). Growth rate was similar in RC1 and RC3 ($F_{(1, 61)}$, =-0.8 $p > 0.05$). Average DBH growth at Mngazana was similar between size class 1-1.9 and 3-3.9 cm ($F_{(1, 69)}$, =0.5 $p > 0.05$) and also in size classes 1-1.9 and >4cm ($F_{(1, 46)}$, $p > 0.05$). *B. gymnorrhiza* trees in Site 3 were lower than 130 cm; therefore, no DBH was found

Table 17: Changes in height and diameter at breast height for each site over time (2012-2015) (Mean ± SE).

Estuary		**Average height (cm)**			**Average DBH (cm)**		
Mngazana	**Site**	**2012**	**2014**	**2015**	**2012**	**2014**	**2015**
	1	151.6±9.5	159.6±9.8	164.7±10.0	9.3±1.4	9.6±1.3	10.1±1.3
	2	187.7±22.1	176.5±19.7	177.8±19.7	20.1±2.5	20.1±2.5	20.6±2.6
	3	57.6±1.4	63.1±1.4	64.8±1.4	-	-	-
	4	174.3±22.2	176.6±19.7	183.5±19.7	12.1±2.1	11.9±1.9	11.1±1.9
	5	136.2±5.4	144.5±5.8	154.8±6.5	5.7±0.5	5.3±0.4	5.8±0.5
	6	256.9±12.3	257.2±12.6	268.6±13.5	4.6±0.3	4.9±0.3	5.6±0.3
	7	171.7±8.5	172.3±8.9	183.2±10.2	4.9±0.6	5.5±0.6	6.1±0.7
	8	249.1±14.8	252.8±14.9	264.3±15.2	14.1±1.7	13.6±1.6	14.1±1.6
Nahoon	1	197.9±13.8	240.6±13.0	259.7±14.0	5.5 ± 0.6	6.4±0.6	7.1±0.6
	2	159.7±11.1	162.7±8.6	183.2±9.7	6.4±0.9	4.7±0.6	5.1±0.5
	3	276.3±21.2	313.5±23.6	323.1±23.5	11.7±1.2	12.7±1.3	13.1±1.3
	4	124.3±7.7	99.8±6.3	112.1±6.9	4.0±0.5	4.4±0.5	4.9±0.6

ND= No Data

Table 18: Average growth in height (cm yr^{-1}) between size classes (RC1-S5) in both estuaries (Mean ± SE (N)).

Estuary	**Site**	**Species**	**RC 1**	**RC 2**	**RC 3**	**S 4**	**S 5**
Mngazana	1	*RM*	5.9±2.9 (3)	3.3±0.3 (85)	5.9±1.1 (19)	6.3±3.8 (7)	3.0±1.1 (11)
	2	*BG*	1.6±0.3 (8)	1.9±0.4 (29)	3.1±1.5 (5)	9.8±6.7 (3)	0.1±1.1 (14)
	3	*BG*	2.2±0.6 (31)	8.7±0.8 (90)	-	-	-
	4	*AM*	7.6±2.4 (9)	4.9±1.1 (40)	4.1±6.2 (4)	9.6±7.9 (3)	-
	5	*BG*	-	4.6±1.2 (9)	11.8±1.3 (12)	-	-
		AM	8.3±5.1 (7)	3.1±0.7 (112)	1.4±4.2 (8)	-	-
		RM	-	5.2±0.9 (68)	7.9±8.3 (31)	6.9±1.7 (6)	6±2.9 (3)
	6	*AM*	-	2.4±1.7 (8)	6.9±3.6 (17)	2.1±1.4 (19)	7.2±2.5 (5)
	7	*AM*	ND	ND	ND	ND	ND
	8	*BG*	-	3.7±1.4 (17)	3.9±2.4 (10)	1.8±1.7 (22)	2.7±1.6 (9)
Nahoon	1	*AM*	-	20.7±2.9 (11)	22.6±2.3 (15)	9.5±2.6 (9)	13.8±0.5 (2)
	2	*AM*	10.9±2.1 (10)	13.5±1.3 (69)	20.8±3.9 (8)	26.3±2.7 (10)	28.3±3.3 (7)
	3	*AM*	11.2±5.4 (4)	10.8±3.2 (18)	11.3±1.7 (9)	25.2±5.8 (8)	19.0±1.4 (23)
	4	*AM*	11.2±1.2 (36)	14.9±1.1 (57)	22.1±2.2 (18)	32.5±3.7 (2)	15.9±4.6 (2)

Table 19: Overall average growth (cm yr^{-1}) (Mean ± SE (N))

Estuary	**Species**	**RC 1**	**RC 2**	**RC 3**	**S 4**	**S 5**
Mngazana	*BG*	2.1±0.5 (38)	6.5±0.6 (145)	7.2±1.3 (27)	2.8±1.7 (25)	1.1±0.9 (23)
	RM	5.9±2.3 (3)	3.8±0.4 (119)	9.1±1.2 (50)	6.6±2.1 (13)	3.7±1.0 (14)
	AM	7.9±2.6 (15)	3.6±0.6 (161)	5.0±2.5 (29)	3.1±1.6 (22)	2.3±1.1(15)
Nahoon	*AM*	11.1±1.1(50)	14.2±0.8(156)	20.1±1.4 (50)	23.4±2.5(29)	19.8±1.5 (34)

Table 20: Overall average growth in diameter at breast height (cm yr-1) (Mean ± SE (N))

Estuary	**Site**	**Species**	**0-0.9**	**1-1.9**	**2-2.9**	**3-3.9**	**>4**
Mngazana	1-8	*BG*	-	-	-	0±0.1 (8)	0.1±0 (77)
		AM	-	0.2±0 (20)	0.1±0 (17)	0.1±0 (22)	0.1±0 (89)
		RM	-	0.2±0.1 (5)	0.1±0 (13)	0.1±0 (17)	0.2±0 (69)
Nahoon	1-4	*AM*	0.8±0.1 (78)	0.6±0.1 (6)	0.5±0.1 (16)	0.5±0.1 (18)	0.9±0.1

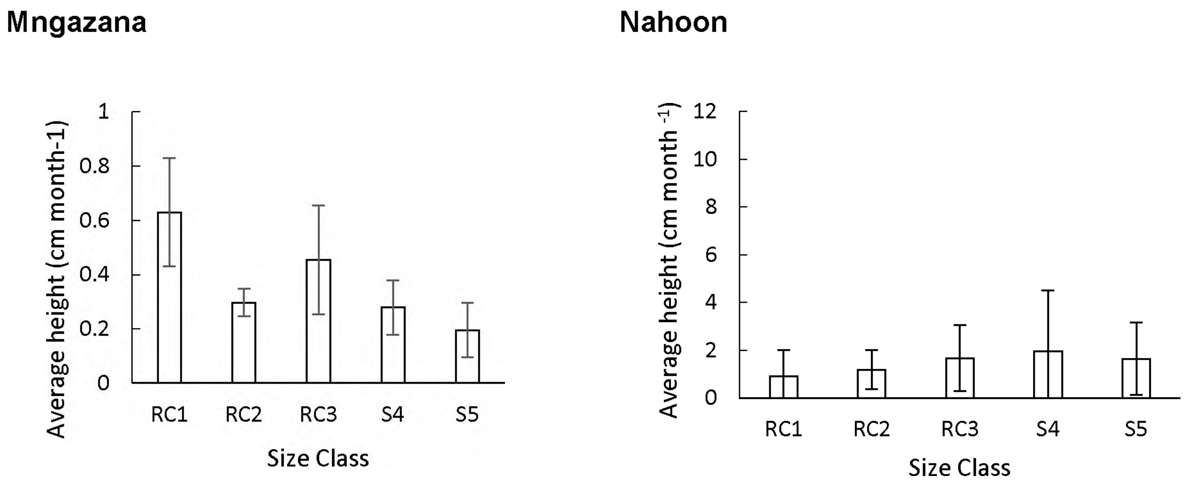


Figure 19: Average vertical growth rate (cm month-1) of A. marina for all the sites at Mngazana and Nahoon Estuary (Mean ± SE).

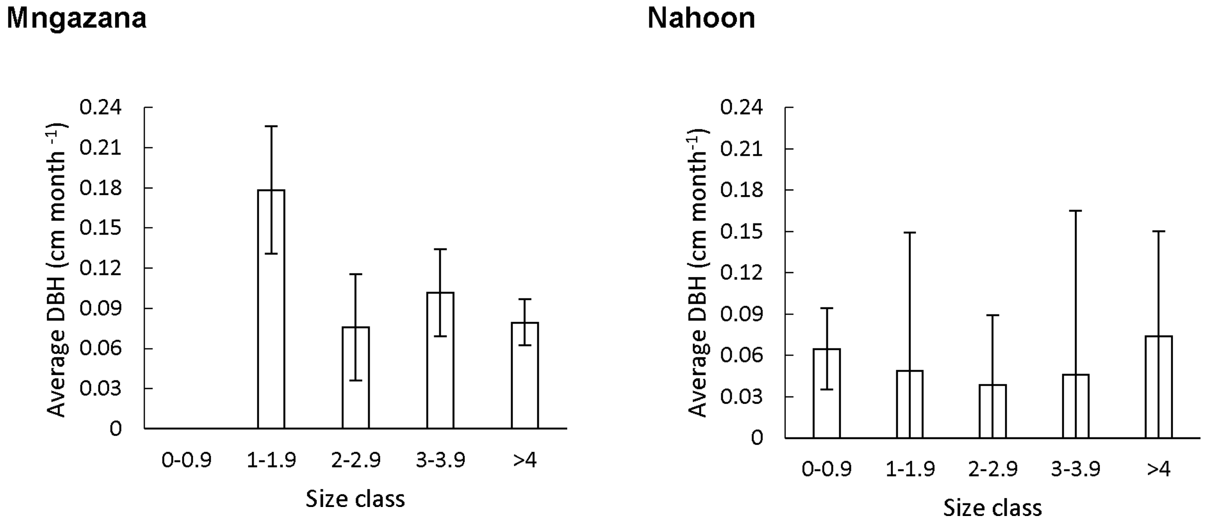


Figure 20: Average horizontal growth rate (cm month-1) for all the sites at Mngazana and Nahoon Estuary (Mean ± SE).

Correlations between growth and porewater characteristics

At Mngazana Estuary, a correlation analysis showed that there were no significant relationships between growth and porewater salinity ($r=0.6$, $p>0.05$), porewater temperature ($r=0.1$, $p> 0.05$) or electrical conductivity ($r=0.1$, $p>0.07$). No correlation was found between porewater characteristics and growth at Nahoon Estuary ($p>0.05$) either.

Correlation between growth and nutrients

Correlation analysis showed that there was no correlation between porewater nutrients and growth at Mngazana Estuary. No correlations were found at Nahoon between S4 and NH_4 ($r=-0.1$, $p>0.05$). A correlation analysis showed that growth of *A. marina* species at Nahoon Estuary was negatively-correlated to the presence of TOXN ($r=-0.4$, $p< 0.05$). No correlation was found between growth and SRP ($r=-0.9$, $p>0.05$) for this species.

Mortality rates

Mangrove death was not uniformly recorded at Mngazana. High mortality was seen in (RC2) as compared to the other size classes ($H_{(13, 38)} = 26.7$, $p<0.05$) and when comparing between sites, mortality was higher in the Mixed (Site 5) and *A. marina* (Site 7) site (Table 21). During the duration of the study, mortality of saplings (RC1) was higher as compared to other regeneration size classes ($F_{(1, 34)} = 17.9$, $p<0.05$) at Nahoon Estuary and more deaths were recorded in 2015 (Table 19). Site 4 had the highest mortality as compared to the other sites.

Table 21: Number of dead mangrove individuals found in different size classes

Estuary	**RC1 (I)**	**RC1 (F)**	**RC2 (I)**	**RC2 (F)**	**RC3 (I)**	**RC3 (F)**	**S4 (I)**	**S4 (F)**	**S5 (I)**	**S5 (F)**
Mngazana	-	3	4	36	-	5	-	4	1	1
Nahoon	-	9	-	2	-	-	-	-	-	-

I=Initial study period, **F**=Final study period

4.4 Discussion

The objectives of this chapter were to measure the porewater characteristics (salinity, temperature, electrical conductivity and nutrient pool) and the growth characteristics of mangrove forests at Mngazana and Nahoon Estuary, to identify the relationships between environmental factors and growth, to determine the growth rates of different sizes classes of *Avicennia marina, Bruguiera gymnorrhiza and Rhizophora mucronata* and lastly, to correlate porewater characteristics to the growth for the mangroves.

High salinity ranges were measured at Mngazana Estuary and this may be due to low or no fresh water coming into the estuary; the highest salinity measured was 41.6 PSU (2015) and it varied between seasons and years. A study done by Rajkaran (2011) recorded salinity values ranging from 28.7 ± 3.3 to 38.2 ± 1.7 PSU for all the sites at Mngazana Estuary, this was lower than what was measured in this study. An increase in salinity was mostly seen in Site 4 where initial (February 2014) salinity was 28.3 PSU and the final (June 2015) salinity was 39.1 PSU. This was a result of decreased freshwater coming into the estuary and lower rainfall (Figure 13c). Salinity greater than 35 PSU may affect growth, production and leaf senescence of mangroves (Naidoo 1990 and Wakushima *et al.*1994a). However, species differ in their salinity preference, with *B. gymnorrhiza* favouring salinity ranges that are reduced (± 10 PSU but less than 30 PSU), while *R. mucronata* prefers salinity ranges of 17.5 PSU. In some cases, however, the latter species was able to survive in high salinities as was seen at Inhaca Island, Mozambique where it survived on salinities of 55 PSU (MacNae 1968). Moore *et al.* (2015) showed survival of mangroves on salinity over 40 PSU at coastal vegetation of United Arab Emirates. *A. marina* is known to be the most tolerant species as compared to other mangroves, it prefers salinity ranges of 5-35 PSU (Naidoo 1987). MacNae (1968) showed how it was able to grow in salinities of over 90 PSU, but this affected the growth and resulted in a stunted morphology.

According to Whitfield (1994), the eastern coast can be divided into subtropical (from the border of Mozambique to Mbashe Estuary) and warm temperate (from south of Mbashe Estuary onwards) regions, and this is controlled by water temperature. Rainfall and temperatures at Mngazana Estuary are higher as compared to Nahoon Estuary. Average rainfall at Mngazana is 81.7 mm and 76.2 mm at Nahoon (Figure 13c and 15c, respectively). Changes in rainfall pattern has been observed especially at Mngazana Estuary, this may have an effect on mangrove growth and productivity. Species that are found in temperate regions are those that have a wide latitudinal distribution (Morrisey *et al.* 2010), Nahoon is no different from other temperate regions hence it has *A. marina* growing there. Low temperatures are known to limit the productivity and distribution of mangroves, but since *A. marina* is the most tolerant species and is able to grow in diverse ranges of habitats, it has the ability to grow in diverse climatic and tidal conditions (Duke *et al.* 1998) as well, and so it is able to survive and grow at Nahoon Estuary and further south as stated in Chapter 3.

Porewater nutrients found in mangrove sediments differ temporally and spatially and this depends on several factors such as redox potential, availability of fresh water, pH and flooding frequency (Bernini *et al.* 2010). Decomposition of mangrove materials is a major source of nutrients in the system. Nitrogen at Mngazana and Nahoon Estuary was available in the form of ammonium; it had a higher concentration as compared to the other nutrients as seen in Figure 17 and 18, respectively Lovelock *et al.* (2010) found a similar trend and recorded higher concentrations at both seaward and landward edges. A study by Emmerson (2005) which focused on the nutrient status of Mngazana also showed that ammonium was found at higher concentrations (153 µg 1^{-1}) compared to the other nutrients. Middelburg *et al.* (1996) recorded ammonium concentrations at Gazi Bay in the range of 3 – 390 µm, and in this study ammonium concentrations were also greater than 3 µm but below 390 µm. Porewater concentration of ammonium, nitrogen and phosphorus in Gazi Bay was lower as compared to those found in temperate regions. Even in this study, ammonium concentration in 2014 was higher at Nahoon Estuary (temperate) as compared to Mngazana Estuary (subtropical). As stated by Emmerson (2005) the high concentration of ammonium at Mngazana may be due to large populations of crabs, which process a substantial quantity of decaying leaf materials. Ammonium concentration at Mngazana and Nahoon Estuary during the wet season ranged between 1.4-27.7 µm (Figure 17) and 1.6-20.4 µm (Figure 18), respectively, and these were lower than concentrations measured by Geldenhuys (2013) (1.5-35.9 µm) in similar areas.

SRP concentrations were higher during the wet seasons than the dry seasons in both estuaries (Figure 17 and 18, respectively). This is in line with work by Castaneda-Moya *et al.* (2013) where SRP concentrations were also higher in the wet seasons (1.2 ± 0.1 µm) than the dry seasons (0.8 ± 0.3 µm). The results opposed what Mohammed and Johnstone (2002) found in Mapopwe Creek, Zanzibar where SRP was higher in the dry seasons than the wet season. SRP was being imported from the ocean by tides, thus leading to seasonal component to nutrient flux (Fong and Zedler 2000) where SRP was released in summer and taken up in winter, therefore resulting in sediment of that estuary was acting as a sink.

TOxN concentration at Mngazana ranged from 0.1-5.0 µm and 0.1- 6.3 µm at Nahoon. It varied between seasons, at Mngazana TOxN was higher in wet seasons than in dry seasons and in contrary TOxN concentrations at Nahoon were higher in dry seasons (Figure 17 and 18, respectively). A study by Geldenhuys (2013) at Nahoon Estuary showed that TOxN concentrations were similar to this study in Transect 1 and 2. Work done by Ilami *et al.* (2002) on the spatial and temporal analysis of soil porewater nutrients verify that most nutrients tend to follow a trend of having higher concentrations in wet seasons as compared to the dry seasons. This may be because of rainfall playing a role by washing away nutrients into the estuary through runoff from the surrounding parts. As stated by (Statham 2012), the supply of nutrients in estuaries is strongly linked to freshwater input which is driven by regional climatic factors.

It is important to measure particle size as this will determine if nutrients are stored in the sediment or easily exported out of the system resulting in a more vulnerable situation for regeneration and growth. Van de Koppel *et al.* (2001) stated that sediments that are mostly dominated by clay or silt are a much better source of nutrients compared to the sediment that is mostly dominated by sand. Study by Rajkaran and Adams (2012) at Mngazana showed that sites found that clay content was significantly higher at Creek 2 while silt was significantly higher in Creek 1. Higher nutrients contents are found mostly in sites where clay and silt are dominant. The high percentage of sand is observed in a study by Bernini *et al*. (2010) and that resulted in a decreased capacity to retain nutrients. At Nahoon Estuary, sand formed the largest component of the sediment in all the sites studied (Geldenhuys 2013). The retention of nutrients in a mangrove forest is dependent on the particle size, the bigger the particle size, the less ability of the sediment to retain nutrients.

According to Thomas (2010) nutrients tend to be available in forests when pH ranges from 6.5 to 7.5 and this range is consistent with plant growth. Most nutrients are not affected by pH but phosphorus is the nutrient that is mostly affected. If pH is above 7.5, they become less available in the sediment. Therefore, the low phosphorus concentration in both estuaries may be due to the range that was measured by Rajkaran (2011) and Geldenhuys (2013) where pH was greater than 6.9.

The organic content found in the mangrove forests affects the presence of nutrients found. High concentrations of organic matter found in the forest as a result of litter fall improve denitrification processes (Mohammed and Johnstone 2002). Therefore, higher organic content will aid the forest in nutrient retention (Geldenhuys 2013) and this is facilitated by moisture content which plays a role in the decomposition of material found in the mangrove forest. Rajkaran (2011) recorded higher moisture content (44.9 ± 1.1 %) in Site 1, as compared to the other sites. Sites where *R. mucronata* (1 & 5) and *A. marina* (4, 5, 6 & 7) was found had significantly higher sediment moisture content and organic matter and our results show that nutrient concentration was higher in these sites. At Nahoon Estuary, moisture content ranged between 26.4 to 39.3 % and organic content was 2.3 to 4.6 % (Hoope- Speer 2015) and these factors also had an influence on nutrient concentrations found there.

The average growth rate of *A. marina* was higher at Nahoon than Mngazana Estuary, it ranged from 9.5 ± 2.6 to 32.5 cm yr^{-1}(Table 18). Growth rate in adult trees was faster than in saplings. An increase in average growth rate was seen over the years and a higher growth rate was recorded from 2012-2014 as deduced from height changes in Table 17. There were trees that experienced a decrease in growth and this was due to breaking of the tree and dead branches, other mangrove trees grew by a cm and this was mostly seen in sites where saplings were dominant. Duarte *et al.* (1998) state that seedlings that grow in sites or areas where there is poor nutrient availability or where coarse sands dominate, lower growth rates are recorded. According to (Kathiresan and Rajendran 2002) seedlings that are found in areas entirely exposed to full sunlight may experience development failure even though other environmental elements may be sufficient. Extreme sunlight may decrease mangrove growth and the photosynthetic rate. There are other factors that play a role in the growth of seedlings and these include nutrient availability, salinity, predation by crabs, tidal position and desiccation, wave exposure etc. (McKee 1995, Duarte *et al.* 1998 and Padilla *et al.* 2004). Beard (2006) state that most stands that have a height of 1.5 m tend to have a vertical growth that is less than 10 cm yr^{-1} (because of the physiological limitations of low temperatures experienced at this latitude). Rajkaran (2011) measured growth rates of *A. marina* to be 6.0 cm yr^{-1}, S4 (251-320 cm) for had the highest growth rates as compared to the other size classes, no increase in height was recorded for trees > 350 cm. In this study, the overall average growth rate of *A. marina* was 5.5 cm yr^{-1} for all the size classes, meaning that a decrease in growth rate was observed in this study (Table 19). *Avicennia marina*'s growth rate was higher in trees > 350 cm which is not different from what Rajkaran (2011) measured. When comparing *A. marina* DBH in a study by Rajkaran (2011) and this study, it was similar (both 0.1 cm yr^{-1}) (Table 20). Overall growth rate at Nahoon Estuary was 17.7 cm yr^{-1} (Table 18) and this was higher than temperate mangroves found in New Zealand where their growth rate was between 3-7.5 cm yr^{-1} and was likely as a result of limitations of low temperatures experienced in those estuaries. When comparing between the size classes, growth was similar in RC1 at Mngazana and Nahoon Estuary. But growth rates in other size classes were higher at Nahoon Estuary as compared to Mngazana Estuary (Table 19). Growth of seedlings rely on various factors which include wave exposure, salinity, light availability and nutrient availability.

Mangrove forests found at higher latitudes are expected to have lower tree growth rates compared to those at lower latitudes (Morrisey *et al.* 2010), but the opposite was found at Nahoon Estuary where mangrove trees grew faster as compared to those found at Mngazana

Estuary. Various studies mentioned how above ground net primary production declines in old mangrove forests after the full canopy leaf areas has been accomplished (Berger *et al*. 2004). Lugo (1997) also mentioned that old mangrove forests tend to have low net production, but nutrient recycling is effective. Poor/ low growth rates may also be as a result of dense tree numbers found in forests, which leads to light completion in the canopy, this has been documented by Okimoto *et al* (2013).

Thinning and resource availability may also play a role in mangrove growth rates. Berger *et al* (2004) did a study on age related decline in forest production and their results showed that growth reduction was as a result of local competition between species and mortality of trees recorded was influenced by self-thinning. Any of these factors, or even a combination of them, could be responsible for the lower growth rates seen at Mngazana Estuary in this study.

Climatic factors made it possible for the growth and productivity of mangroves at Nahoon Estuary; while low temperatures and low rainfall did not have an effect on their growth, temperature extremes might affect it. Plant-to-plant interaction in these mangrove communities might also play a role in mangrove growth, productivity and expansion. At Nahoon Estuary, there is low population density of trees which results in less competition for nutrients and light between species and that in turn leads to high growth rates. The opposite can be seen at Mngazana where the population density is high. The concentration of nutrients found at Nahoon Estuary play a role in the increase in growth rates and also the spread of the forest. This has been seen taking place in New Zealand and documented by Yates *et al*. (2004). Mangroves can also coexist with saltmarsh species and this has been documented by various authors (Stevens 2006 and Zhang *et al*. 2012). The presence of some saltmarsh species can act as a nursery to mangrove seedlings (Lewis 2009), but can also restrict the growth of mangroves (McKee and Rooth 2008). Climate change has increased the competitive advantage of mangroves and as such increased their ability to outgrow saltmarsh species resulting in establishment and growth. According to Soarez *et al*. (2012) if mangrove species are under stress, they normally show low growth rates (horizontally and vertically) and a high degree of branching on trees.

Higher mortality has been recorded in 2015 in both estuaries as compared to the previous years studied. Jimenez and Lugo (1985) in their study mentioned various factors that can lead to high mortality rates and also stated how it can be caused by normal interaction between individuals and their environment and are also density-dependent. Mortality at Mngazana was mostly in RC2 (Table 21), this has been documented by Rajkaran (2011) and it is supported that this mortality is a result of intraspecific competition and self-thinning taking place in this estuary. At Nahoon Estuary, mortality was mostly in RC1 and RC2 no mortality was recorded in the other size classes (Table 21) In both estuaries, seedlings were dying and this might be due to various reasons. Various factors might lead to mangrove mortality, and these include herbivory, competition between species, and the presence of fungi. Fungi, like most biological factors, can likely kill only those hosts weakened through some other cause (Jimenez and Lugo 1985). A study at the Gladstone coast of Queensland, Australia showed that there was an extensive mortality of *A. marina* in that area, which was attributed to a *Phytophthora* species. Only *A. marina* was affected in that area even though there were 12 different types of mangrove species (Osorio *et al*. 2014).

There are other factors that may have played a role in the increased mortality rate observed at Nahoon Estuary. According to (Ellison 1998) some mangrove species are more susceptible compared to others, therefore excessive rates of sedimentation may lead to mortality. High rates of sedimentation obstruct nutrient cycling and gas exchange as a result of rhizophore's lenticels and pneumatophores being covered (Odum and Johannes 1975). And that is the case at Nahoon, as a result of the mangrove location that is close to the mouth of the estuary, Transect 1 suffered a high mortality rate. this was as a result of increased sedimentation in that area which resulted in mangrove pneumatophores being covered which then led to the mortality of mangroves in that area. A study done in New Zealand by Lovelock *et al.* (2007) so as to understand the effect of sedimentation and nutrient availability on growth of mangroves showed the opposite of what occurred at Nahoon, sedimentation enhanced mangrove growth and it also increased microbial mineralization of N which in turn led to nutrient limitation.

4.5 Conclusion

The objective of this chapter was to compare the growth of mangrove species at Mngazana and Nahoon Estuary and to determine the pore water nutrient pool of these estuaries. Climate change will lead to migration of mangroves to higher latitudes, extreme cold events will limit their expansion and growth (Godoy and De Lacerda 2015). Different mangrove species and forests respond differently to various factors. Mangrove growth and productivity is affected by various abiotic and biotic factors. High or low temperatures, increased salinity ranges, availability of nutrients, siltation and sedimentation are among the factors affecting the growth of mangroves (Noor *et al*. 2015).

Both the vertical and the horizontal growth in this study occurred in non-identical rates. Temperate mangroves (Nahoon forest) were growing faster as compared to subtropical mangroves (Mngazana forest). If high salinity ranges at Mngazana Estuary persist, they might have an effect on the growth of mangroves and it will likely be species specific. Sedimentation taking place at Nahoon Estuary is resulting in areas becoming unfavourable for mangrove growth and thus leading to mangrove mortality and inability of mangrove propagules and seedlings to survive. According to Noor *et al*. (2015) high sedimentation tends to have a negative effect on mangrove trees

The presence of nutrients in these forests is important to maintaining growth and productivity of mangroves. Soil type plays a role in the availability of nutrients, therefore it is an important parameter for mangrove growth and production. Hydrological regimes may also have an effect on mangrove growth as inundation and position near to the sea plays a role in growth (sites closer to the channel had higher porewater salinity as compared to the other sites).

Chapter 5 – General discussion and conclusion

5.1 General information

According to Chapman (1977) mangroves are mainly found along tropical and subtropical coastlines. Some species expand and are found in ranges that are cooler (warm temperate). The distribution structure of mangroves is proportional to the availability of warm and cold oceanic patterns (Steinke 1995). Mangrove diversity decreases as you move further south and in some cases only one species is able to establish, grow and become reproductive. *Avicennia marina* is known to be the most common species on the African continent; it can tolerate high salinities, aridity, low water temperatures and frost frequency. It is also able to survive in temperate regions (Morrisey *et al.* 2010). *Avicennia marina* can survive habitats where salinity may be 1.5 times greater than seawater or up to 70 PSU (Martin *et al.* 2010 and Reef *et al.* 2012). Of the six mangrove species found in South Africa, two species are most common, *A. marina* and *Bruguiera gymnorrhiza* (Ward and Steinke 1982).

Global climate change will have a considerable impact on mangrove ecosystems, these effects are most likely to be noticeable in those regions where mangrove ecosystems are close to their limits (Kjefve and Macintosh 1997). There are various elements of climate change that have an effect on the growth and production of plants including mangroves. These include; sea level rise, temperature, precipitation, circulation patterns, storms, high water events and how humans respond to climate change (Gilman *et al.* 2008). Various authors have mentioned how extreme temperatures (cold) may illustrate the current latitudinal limits of mangrove expansion, although these extreme events are certainly not the only factors that limit mangrove expansion to higher latitudes (Snedaker 1995 and Solomon *et al.* 2007). As mangrove expansion occurs, saltmarsh species are expected to be displaced. Several authors report on the expansion of mangroves which occur quickly as result of facilitation by global warming. The change occurring in the mangrove forest might have an effect on these ecosystems since they are ecologically and economically important (Ghose 2013).

5.2 Revising the distribution of mangroves

The limit of the forests was expected to be at Nahoon Estuary as presented by Ward and Steinke in 1982 and Hoppe-Speer (2015) and, moreover, that no forests existed further south of this location due to environmental constraints.

Mangrove response to climate change differs from one forest to the other. In this study, mangroves expanded within and outside the range that was proposed by Ward and Steinke (1982). On the East coast of USA, a poleward limit of mangrove expansion was observed as a result of changes in temperatures and global warming. Mangroves were able to survive in that area as a result of a decrease in extreme cold events (-4 °C) (Godoy and De Lacerda 2015). Expansion of mangroves in New Zealand has been recorded in some locations and this is brought about by high sedimentation rates (Alfaro 2010). High mangrove expansion can occur provided that there is high propagule production rate and low propagule predation. This occurred in estuaries in Hawaii where *R. mangle* was spreading extensively due to high propagule production (Cox and Allen 1999). Notable changes in mangrove cover (ha) have been seen occurring at both Kwelera and Tyolomnqa Estuary over the years. *Avicennia marina* was the most dominant species in these estuaries. Spatial distribution of mangroves is mostly influenced by the physiological tolerance of each species to salinity. As stated by Duke *et al*. (1998), *A. marina* is usually located at the estuary mouth of most estuaries and it reaches the upstream tidal limits; this has been seen occurring at both Kwelera and Tyolomnqa Estuaries.

Literature mentioned the number of extreme events that are more detrimental to mangrove growth. These events were defined at <5 °C. The number of extreme events in this study increased from north to south with a maximum number of events recorded at Tyolomnqa Estuary. The overall density did not change between 2014 and 2015. In Site 2 of Kwelera, the average height decreased in 2015 implying more seedlings had established and some changes in ratio of seedlings to adults were noticed. At Tyolomnqa. Site 1 and Site 2 were dominated by adults and Site 3 and Site 4 had more seedlings present. The presence of propagules found in these estuaries show that mangrove trees there are productive and this in turn may lead to the expansion of the forests. The production of propagules differed in each estuary. If the estuary produces propagules that are more buoyant, either initially or after refloating then that means distributions will be likely to occur (Steinke 1986). Mangroves at Kwelera first established in 1969 from propagules that were believed to have floated into the estuary from further south. No seedlings were establishing even though the trees were producing fruit. From 1982 seedlings were recorded in the estuary, their survival was low and showed signs of dieback or damage. In this study, we found that seedling survival was also low and seedlings were not growing into the next regeneration size classes. Several studies have shown how mangrove seedlings may be outcompeted buy the presence of other species found in the area e.g. Mntamvuna Estuary (Rajkaran *et al*. 2009). At Kwelera, the encroachment of *Phragmites australis* might play a role in the dieback of seedlings in the estuary.

A conversation that we had with the residents of Tyolomnqa Estuary revealed that a removal of mangrove saplings was occurring in the estuary and mangrove expansion was being restricted by the inhabitants. This has been reported in other estuaries, e.g. in New Zealand mangroves are removed legally and illegally. The community/residents believe that by removing them, the sediment that accumulated the banks will be removed and flushed out thus preventing further establishment (Stokes *et al*. 2010). If mangrove removal at Tyolomnqa continues, then the population size at this estuary will be affected.

5.3 Comparison of propagule weight at mangrove forests along a latitudinal range.

Average propagule weight increased as you move further south. Propagule weight after drying at Mngazana Estuary was lower as compared to the other two estuaries studied. Propagules at Kwelera and Tyolomnqa Estuaries were found mostly in mangrove trees that were exposed to the sun.

5.4 Comparing the growth rates of mangroves at two forests within the current distribution range and at the edge of the range.

A decrease in population structure overtime was seen in Site 2 at Mngazana Estuary and Site 4 at Nahoon Estuary. Growth rates also differed from one species to the next. Overall growth rates at Nahoon Estuary were higher as compared to Mngazana Estuary. At Mngazana Estuary RC3 had the highest overall growth rate (7.6 ± 0.9 cm yr^{-1}) as compared to RC1, RC2, S4 and S5 and at Nahoon Estuary S4 had the highest overall growth rate (23.4 ± 2.5 cm yr^{-1}) as compared to RC1, RC2, RC3 and S5. No DBH was found in size class 0-0.9 cm at Mngazana Estuary and at Nahoon Estuary highest DBH growth rate was found in size class >4 (0.9 ± 0.1 cm yr^{-1}).

Mangrove mortality was recorded at both Mngazana and Nahoon Estuary and it was more prevalent in RC1 and RC2. There are various factors that may affect seedling growth. E.g. *A. marina* seedlings can be killed by water temperatures that are between 39 °C and 40 °C; this excludes seedlings and trees that have already been established. Competition between species may also lead to seedling mortality, e.g. competition between *Spartina alterniflora* Loisel. and mangroves was reported by Mckee and Rooth (2008). Physical factors which include tidal actions and waves may wash away seedlings (Walsh 1974, Clarke and Myerscough 1993), the presence of macro algae in a mangrove forests may also reduce seedling establishment, and thus leads to mortality (Clarke and Myerscough 1993). Herbivores such as crabs, snails and insects might also affect seedling survival, and silt being deposited around mangrove seedlings may also affect their establishment. A study by Satumanatpan and Keough (1999) at Rhyll inlet, Phillip Island, Australia showed that the presence of mangrove barnacles did not have much impact on mangrove seedling growth and survival as hypothesized. They discovered that the high mortality rates in that forest might have been as a result of the above-mentioned factors.

An increase in salinity resulted in decreased growth of mangroves at Mngazana Estuary and if these conditions persist it may result in mangrove mortality. Clough (1984) showed how salinity affect mangrove species, specifically *A. marina* and *Rhizophora stylosa* Griff.. Average salinity stimulates the growth of these species, but once an increase in salinity occurs, a decrease in growth will be observed.

5.5 Determining the nutrient pool in the pore water of mangroves and to identify the main driver of growth for the mangroves at Mngazana and Nahoon Estuary.

Gleeson *et al*. (2013) mentions how mangroves remove dissolved nutrients during the inundation process. Mangroves are also able to reserve these nutrients through the process of sediment retention, absorption and carbon sequestration. The ability of the mangroves to remove or retain these nutrients is dependent upon the speed of the current, topographic elevation and how frequent these mangroves become inundated. Mangrove forests are known to be oligotrophic, therefore mangrove species that grow there become adapted to low nutrients availability (Hutchings and Saenger 1987). Nutrients can play a secondary role in mangrove expansion by increasing plant growth (Lovelock *et al*. 2007) and it is available in the system as a result of litter fall in the forest. Various litter studies have been done to show their importance in the system and Lugo and Snedaker (1974) mentioned how its production can have a huge contribution to the nutrient budget of the system.

Nitrogen and phosphorus are known to be the most limiting macro-nutrients in a mangrove forest and this is as a result of geochemical restrictions. Mangrove forests can either be nitrogen limited, phosphorus limited or a forest can be both nitrogen and phosphorus limited. Nitrogen can be limiting due to slow modification from particulate to dissolved form by bacteria and phosphorus may be limiting due to low solubility and sorption onto carbonates (Alongi 2010). Once waterlogging occurs in mangrove sediments, conditions become anaerobic and this prevents nitrification from taking place and thus leads to low nutrient availability (Chen and Ye 2004).

A relationship was found between the presence of TOxN and growth at Nahoon Estuary while no connection was found on the other nutrients studied. Nevertheless, nutrients are found to play a huge role in mangrove production and growth and they have been documented by various authors through nutrient introduction experiments. A study at the Indian River, Florida by Onuf *et al*. (1977) where they compared two mangrove islands. Their results showed that trees had higher growth rates at the high nutrient sites as compared to the low nutrients site. Another study by Duarte *et al*. (1998) on seedlings of *Rhizophora apiculata* Blume showed that seedlings that were growing in coarse and nutrient poor sediments had a very low growth rate as compared to seedlings that were growing in nutrient rich sediment composed of silt, therefore growth was associated with the presence of nitrogen and phosphorus in sediment porewater. Another field observation by Clarke and Allaway (1993) proved the importance of nutrients on growth of mangroves, in their study the survival of *A. marina* seedlings was enhanced when nutrients were added and after a year the nutrient-added seedlings had a higher growth rate as compared to the ones not supplied with nutrients. Other studies focused on dwarf mangroves in Australia and they were associated with low nutrient availability in the forest. A study on nutrient relations in eastern Puerto Rico concluded that the stunted growth of *Rhizophora mangle* was likely caused by a combination of P limitation and seasonal water stress (Medina *et al.* 2010).

The goal of this study was to understand the correlation between growth rates and the drivers for growth of mangrove trees in temperate and subtropical regions (Mngazana and Nahoon) and to see nutrient conditions affect their growth and productivity. It is also to understand the importance of mangrove expansion, factors that lead to their expansion and also what role do various characteristics play in mangrove expansion (Gonubie- Keiskamma).

5.6 The following conclusion can be drawn from this study:

- Mangroves have expanded within and outside the range noted by Ward and Steinke (1982).
- Localised expansion within estuaries is limited by physical factors such as space in the intertidal area.
- Mangrove cover at both Kwelera and Tyolomnqa Estuary increased over the time.
- Mangroves have been seen co-occurring with saltmarsh and *Spartina maritima* at Kwelera Estuary and Tyolomnqa Estuary respectiviely. The presence of these species will facilitate mangrove colonization and establishment by trapping mangrove propagules, but will result in competition thereafter. Therefore, the presence and density of these species and disturbance regimes will determine further expansion in these mangrove forests.
- Nitrogen at Mngazana and Nahoon Estuary was available in the form of NH4 and its concentration was higher than other nutrients. Ammonium concentration at Nahoon Estuary was higher compared to Mngazana Estuary in 2014 sampling dates.
- Mangrove trees in the temperate region grew faster than those found in subtropical regions. RC1 growth rate at Nahoon was similar compared to Mngazana Estuary. Other size classes grew faster at Nahoon Estuary compared to Mngazana Estuary.
- Mangrove mortality at Mngazana and Nahoon Estuary was mostly in RC1 and RC2, no death was recorded in other size classes at Nahoon and No death was recorded in S5 at Mngazana, this might be as a result of competition between species.

5.7 Expectations of mangrove alterations due to the Anthropocene

The increase in global human population plays a role in the loss and fragmentation of habitats around the world. The media and scientists have reported on the losses of these environments as a result of changes in climate and human interventions (Valiela *et al*. 2001) and mangrove ecosystems are also affected by these anthropogenic and natural impacts. Climates have changed extensively in the geological past, and it is unquestionable that they will change again in the future (Bernal 1991).

The expansion of mangrove to areas where they did not previously occur have been documented by various authors (Parmesan and Yohe 2003 and Saintilan *et al*. 2014). Sea level rise as caused by climate change and increase in temperature may be the driving factors for alterations. Sea level rise will affect salinity and inundation periods where mangrove forests will be drowned and this will lead to a landward movement of mangroves (Hoppe-Speer 2013). Natural disasters and storms have been documented in various estuaries around the world, and once the storms become intense and an increase in frequency occurs, damage in mangrove forests will occur which will lead to tree mortality, soil compression, soil erosion, peat collapse etc. (Gilman *et al*. 2008). At Kobonqaba Estuary, South Africa a storm led to mouth closure and that resulted in inundation of mangroves which latter caused a mass mortality of *A. marina*. A hurricane in Honduras was also documented by Cahoon *et al*. (2003) and it led to a collapse of a peat. Urbanization and construction will affect hydrological regimes of the estuary and this in turn will affect mangroves and other coastal systems (Lugo *et al*. 2014). Mangrove harvesting is seen occurring where mangrove trees are being cut for various purposes, this has been document by Rajkaran *et al*. (2004). Cattle grazing in mangrove forests also occurs and this leads to stunted growth and loss of propagules (Hoppe-Speer *et al*. 2013).

Extreme events are predicted to increase over the years (Nicholls and Lowe 2004) and South Africa is expected to have a decrease in rainfall. According to Quisthoudt *et al*. (2013), if there is a decrease in rainfall, an increase in evaporation is observed and this leads to drought that will overall affect mangrove production and growth. The eastern parts are expected to be drier and the mid parts are the only ones that will be wetter (Geldenhuys 2013). Regions with low rainfall are expected to have groundwater inputs that are very low and freshwater surface inputs to mangrove forests will also be lessened and thus increasing salinity in the system (Gilman *et al*. 2008). Some of these events have been documented and have already taken place in estuaries around the world (Table 22)

Table 22: Extreme events and their effect that occurred in various estuaries globally

Country/Estuary	**Year**	**Extreme event**	**Effect**	**References**
Kosi Bay (SA)	1966	Cyclone	Mass mortality of mangroves due to erosion and increased rainfalls	Griffiths 1974 and Rajkaran 2011
Florida	1977	Extreme cold events	Mangrove mortality due to exposure to freezing temperatures.	Lugo and Patterson-Zucca 1977
Philippines	1990	Typhoon	Loss of mangroves due to structural damage	Long *et al*. 2014
Rufiji Delta (Tanzania)	1997/ 1998	EL Nino Floods	Mangrove mortality due to inundation	Erftemeijer and Hamerlynck2005
Honduras	1998	Hurricane Mitch	Mass mangrove mortality that resulted in a peat collapse	Cahoon *et al*. 2003
Mangrove Bight, Guanaja, Central America	1998	Hurricane Mitch	Impact in sediment dynamics which resulted in lack of propagule establishment. No mangrove recovery for nearly 3 years	Asbridge *et al*. 2015
Mississippi River Deltaic	2000	Drought	Extensive dieback of saltmarsh	McKee *et al* 2004
St Lucia (SA)	2002	Drought	non-tidal, dry and hypersaline conditions in the system resulted in decreased productivity of mangroves	Hoppe-Speer *et al.* 2013
Asia	2004	Tsunami	Mangrove loss (ha)	Alongi 2008
Nicobar Islands, (Camorta, Katchal, Nancowry and Trinkat)	2004	Tsunami	Extensive damages to mangrove forests and other aquatic systems	Ramachandran *et al*. 2005
Australia	2011	Flooding	Mangrove mortality and damage	Asbridge *et al*. 2015
Northeast coast of Australia	2011	Cyclone Yasi	Mangrove damage	Ellison 2000, Hopley and Smithers 2010
Kobonqaba (SA)	2012	Flooding that led to inundation of mangroves	Mangrove dieback	Brett 2012; Mbense *et al*. 2016
Mississippi River deltaic plain	2014	Freeze events	*A. germinans* leaf damage and mortality	Osland *et al*. 2015

5.8 Knowledge gained from this study and future research

This research has improved our understanding on mangrove expansion in South Africa. Evidence gathered in this study showed that mangroves expanded within and outside their range. It revealed that *A. marina* has the ability to extend beyond its current limit in South Africa.

The population structure of mangroves at Kwelera and Tyolomnqa measured revealed 1) the low genetic diversity of mangroves in these forests. Only one individual of *B. gymnorrhiza* was found at Kwelera and no new species exists. 2) the effects of mangroves being restricted to one area. This was evident at Kwelera and it results in a small population size. 3) the effects of hydrological regimes leading to the transition of seedling to sapling being almost zero. Sediment and porewater characteristics show that these forests are still young and pH did not affect any new establishments. Different mangrove species respond differently to various factors, high porewater salinity at Mngazana had an effect on growth of mangrove in that forests. Sedimentation at Nahoon Estuary will create unfavourable conditions for mangrove growth and will lead to mortality in the near future as the climate continues to change.

This study has drawn attention to the need for further research in estuaries further south of Nahoon Estuary focusing on their structural changes, shifting environmental conditions and the periodical monitoring of the growth of mangroves, especially seedlings and saplings, since expansion of these forests is dependent on them. A long-term monitoring field based assessment of mangrove distribution is also of importance, and can be another future research point. Planning of educational workshops to teach communities about the importance of mangroves is also important, since they (mangroves) are regarded as being of high conservational value and it is essential for communities around these ecosystems to be informed of the value these ecosystems provide in terms of environmental services.

A study by Hoppe-Speer (2013) investigated the effect of drought on the mangroves of St Lucia Estuary, their results showed that sediment salinity was higher in surface and subsurface samples and in sites where sediment was dry, there was low density of seedlings or no seedlings. High leaf cation was also observed which proves that mangroves at St Lucia are under stress. Mangroves had difficulties surviving these harsh, dry conditions. Yessoufou and Stoffberg (2015) state that the future of mangrove forests is at risk as a result of climate change, however mangrove southward expansion is favoured by climate change. According to Menezes *et al*. (2003), rainfall is regarded as an important factor that allows mangrove growth and production which later results in saltmarsh species being outcompetes and overall leads to mangrove expansion. The impacts of long term drought on South African mangroves needs further investigation particularly how it will affect reproductive outputs and seedling mortality.

5.9 Mangrove ecosystem management

Managing mangrove forests are of importance in the coastal ecosystem. Kairo *et al* (2004) mentions how this management increases fisheries production and biodiversity in the system.

Islam (2008) mentioned how management of mangrove ecosystems can play various roles:

1. Management can minimise negative impacts occurring on mangrove ecosystems.
2. Management can lead to avoiding crisis management (economic, social and ecological) in mangrove ecosystems.
3. Management can lead to sustainable use of natural resources and allow natural resource management.

Mismanaging mangrove systems will not only affect that system, but also surrounding coastal systems. Mangrove forests are economically and ecologically valuable; the low rainfall that is currently received in South Africa will have an effect on the functioning of mangrove forests and other surrounding systems. Therefore, long term monitoring is required to track population and ecosystem level changes in these sensitive ecosystems.

References

Abeli, T., Gentil, R., Mondoni, A., Orsenigo, S., Rossi, G., 2014. Effects of marginality on plant population performance. *Journal of Biogeography* 41: 239-249.

Abrantes, K.G., Johnston, R., Connolly, R.M., Sheaves, M., 2015. Importance of mangrove carbon for aquatic food webs in wet-dry tropical estuaries. *Estuaries and Coasts* 38: 383–399.

Adame, M.F., Lovelock, C.E., 2008. Ecosystem services provided by mangroves: retention of dissolved nutrients. MSc. School of Biological Sciences, University of Queensland.

Adame, M.F., Lovelock, C.E., 2011. Carbon and nutrient exchange of mangrove forests with the coastal ocean. *Hydrobiologia* 663: 23-50.

Adams, J.B., Colloty, B.M., Bate, G.C., 2004. The distribution and state of mangroves along the coast of Transkei, Eastern Cape Province, South Africa. *Wetlands Ecology and Management* 12: 531-541.

Alber, M., 2002. A conceptual model of estuarine inflow management. *Estuaries* 25: 1246-1261.

Alfaro, A.C., 2010. Effects of mangrove removal on benthic communities and sediment characteristics at Mangawhai Harbour, northern New Zealand. – ICES *Journal of Marine Science* 67: 1087–1104.

Allanson, B.R., 2001. Some factors governing the water quality of microtidal estuaries in South Africa. *Water SA* 27(3): 373-386.

Alongi, D.M., 1994. The role of bacteria in nutrient recycling in tropical mangrove and other coastal benthic ecosystems. *Hydrobiologia* 285: 19–32.

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

Alongi, D.M., 2005. Mangrove-microbe soil relations. In: Kristensen E, Haese RR, Kostka JE. (Eds.), Interactions between Macro- and Microorganisms in Marine Sediments. American Geophysical Union, Washington, D.C. 85-103 pp.

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

Alongi, D.M., 2009. Energetics of Mangroves. Springer Science. Dodretch.

Alongi, D.M., 2010. Dissolved iron supply limits early growth of estuarine mangroves. *Ecology* 91(11): 3229-3241.

Alongi, D.M., 2011. Carbon payments for mangrove conservation: ecosystem constraints and uncertainties of sequestration potential. *Environmental Science and Policy* 14: 462–470.

Amarasinghe, M.D., Balasubramanian, S., 1992. Structural properties of two types of mangrove stands on the northwest coast of Sri Lanka. In V. Kaccarin and E Martens 9eds). The ecology of mangrove and related ecosystems. Developments in

Hydrobiology 80. Kluwer Academic Publishers. Dordretch: 17-27. Reprinted from *Hydrobiologia* 247.

Asbridge, E., Lucas, R., Accad, A., Dowling, R., 2015. Mangrove response to Environmental changes predicted under varying climates: case studies from Australia. *Current Forestry Reports* 1(3): 178-194.

Balke, T., Webb, E.L., van den Elzen, E., Galli, D., Herman, P.M.J., Bouma, T.J. 2013., Seedling establishment in a dynamic sedimentary environment: a conceptual framework using mangroves. *Journal of Applied Ecology* 50: 740–747.

Barrett, M.G., 1992. *Coastal zone planning and management*. Thomas Telford services Ltd, Thomas Telford House 1 Heron Quay, London.

Bashan, Y., Morenomm, M., Troyo, E., 2000. Growth promotion of the seawater-irrigated oilseed halophyte *Salicornia bigelovii* inoculated with mangrove rhizosphere bacteria and halotolerant *Azospirillum* spp. *Biology and Fertility of Soils* 32: 265–272.

Bates, G.C., Heelas, B.V.,1975. Studies on the nitrate nutrition of two indigenous Rhodesian grasses. *Journal of Applied Ecology* 12: 941-952.

Beale, C.M., Baker, N.E., Brewer, M.J., Lennon, J.J., 2013. Protected area networks and savannah bird biodiversity in the face of climate change and land degradation. *Ecology Letters* 16: 1061–1068.

Beard, C., 2006. Physiological Constraints on the Latitudinal Distribution of the Mangrove *Avicennia marina* (Forsk.) Vierh. subsp. australasica (Walp.) J. Everett in New Zealand. Hamilton, New Zealand: University of Waikato, Doctoral thesis. 203 pp.

Beadle, N.C.W., 1953. The edaphic factor in palnt ecology with a special note on soil phosphates. *Ecology* 34: 426-428.

Bellard, C., Bertelsmeier, .C, Leadley, P., Thuiler, W., Courchamp, F., 2012. Impacts of climate change on the future of biodiversity. *Ecology Letters* 15: 365-377.

Berger, U.T.A., Hildenbrandt, H., Grimm, V., 2004. Age related decline in forest production: modelling the effects of growth limitation, neighbouring competition and self-thinning. *Journal of ecology* 925: 846-853.

Berjak, P., Campbell, G.K., Huckett, B.I., Pammenter, N.W., 1977. *In the mangroves of Southern Africa,* pp 72*. Wildlife Society*, Durban.

Berliner, D., Desmet, P., 2007. Eastern Cape Biodiversity Conservation Plan: Technical Report. Department of Water Affairs and Forestry Project No 2005- 012, Pretoria. 1 August 2007.

Bernal, P.A., 1991. Consequences of Global Climate Change for Oceans: A Review. *Climate Change* 18: 339-359.

Bernini, E., Silva, M.A.B., Carmo, T.M.S., Cuzzuol, G.R.F., 2010. Spatial and temporal variation of the nutrients in the sediment and leaves of two Brazilian mangrove species and their role in the retention of environmental heavy metals. *Brazilian Journal of Plant Physiology* 22: 177-187.

Black, C.A., 1965. Methods of Soil Analysis: Part I physical and mineralogical properties. American Society of Agronomy, Madison, WI, USA.

Blasco, F., Saenger, P., Janodet, E., 1996. Mangroves as indicators of coastal change. *Catena* 27: 167-178.

Blasco, F., 1983. Climatic factors and the biology of mangroves. In Snedaker SC, Snedaker JG (eds) The mangrove ecosystem research methods. Monographs on oceanographic methodology, vol 8, Unesco, Paris.

Borja, A., Bricker, S.B, Dauer, D.M., Demetriades, N.T., Ferreira, J.G., Forbes, A.T., Hutchings, P., Jia, X., Kenchington, R., Marques, J.C, Zhu, C., 2008a. Overview of integrative tools and methods in assessing ecological integrity in estuarine and coastal systems worldwide. *Marine Pollution Bulletin* 56: 1519–1537.

Boto, K.G, Wellington, J.T., 1984. Soil characteristics and nutrient status in a Northern Australian Mangrove forest. *Estuaries* 7(1): 61-69.

Bowd, R., Breen, C., Hay, D., Kotze, D., Mander, M., 2012. An approach to estuary-based economic empowerment with a particular focus on the Eastern Cape Wild Coast. WRC Report No. 1705/1/11. Water Research Commission, Pretoria, South Africa.

Breen, C.M., Hill, B.J., 1969. A mass mortality of mangroves in Kosi Estuary. *Transactions of the Royal Society of South Africa* 387: 285-303.

Briggs, D., 1977. Sources and Methods in Geography. Butterworths, London.

Breetzke, T., Moore, L., Meyer, C., 2013. Eastern Cape Coastal Management Programme: 2013 Update. Royal HaskoningDHV on behalf of t Department of Economic Development, Environmental Affairs and Tourism. 50pp.

Burns, B.R., Ogden, J., 1985. The demography of the temperate mangrove (*Avicennia marina* forsk Vierh.) at it southern limit in New Zealand. *Australian Journal of Ecology* 10: 125-133.

Bursey, M.L., Wooldridge, T.H., 2002. Diversity of benthic macro fauna of the flood-tidal delta of the Nahoon Estaury and adjacent beach, South Africa. *African Zoology* 37: 231-246. (Invertebrates - Nahoon).

Cahoon, D., Hensel, P., Rybczyk, J., McKee, K., Proffitt, E., Perez, C., 2003. Mass tree mortality leads to mangrove peat collapse at Bay Islands, Honduras after Hurricane Mitch. *Journal of Ecology* 91: 1093–1105.

Calderon, H., Weeda, R., Uhlenbrook, S., 2004. Hydrological and geomorphological controls on the water balance components of a mangrove forest during the dry season in the pacific coast of Nicaragua. *Wetlands* 34: 685.

Carbon Dioxide Information Analysis Center (CDIAC). 2016. Recent greenhouse gas concentrations.

Cardona, P., Botero, L., 1998. Soil characteristics and vegetation structure in a heavily deteriorated mangrove forest in the Caribbean coast of Colombia. *Biotropica* 30: 24-34.

Castañeda-Moya, E., Twilley, R.R., Rivera-Monroy, V.H., 2013. Allocation of biomass and net primary productivity of mangrove forests along environmental gradients in the Florida coastal Everglades, USA. *Forest Ecology and Management* 307: 226–241.

Casteneda-Moya, E., Rivera-Monroy, V.H., Twilley, R.R., 2006. Mangrove zonation in the dry life zone of the Gulf of Fonseca Honduras. *Estuaries and Coasts* 29: 751-764.

Cavanaugh, K.C., Kellner, J.R., Forde, A.J., Gruner, D.S., Parker, J.G., Rodriguez, W., Feller, I.C., 2014. Poleward expansion of mangroves is a threshold response to decreased frequency of extreme cold events. *Proceedings of the National Academy of Science* 111(2): 7223-727.

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

Chapman, V.J. (Ed.)., 1977. Wetland Coastal Ecosystems. Amsterdam: Elsevier.

Chen, R., Twilley, R.R., 1999. A simulation model of organic matter and nutrient accumulation in mangrove wetland soils. *Biogeochemistry* 44: 93-118.

Chen, Y., Ye, Y., 2004. Effects of salinity and nutrient addition on mangrove *Excoecaria agallocha* Plos One 9(4) e93337.

Chiu, C.Y., Lee, S.C., Juang, H.T., Hur, M.T., Hwang, Y.H., 1996. Nitrogen nutritional status and fate of applied N in mangroves soils. *Botanical Bulletin of Academia Sinica* 37: 191-196.

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

Clarke, P.J., Myerscough, P.J., 1993. The intertidal distribution of the grey mangrove *Avicennia marina* in south eastern Australia: the effects of physical conditions, interspecific competition, and predation on propagule establishment and survival*.* *Australian Journal of Ecology* 18: 307-315.

Clough, B.F., 1984. Growth and salt balance of the mangrove *Avicennia marina* (Forsh.) Vierh. And *Rhizophora stylosa* Griff. In relation to salinity. *Australian Journal of Plant Physiology* 11: 419-430.

Coetzee, J.C., Adams, J.B., Bate, G.C., 1996. A botanical importance rating system for estuaries. *Journal of Coastal Conservation* 2: 131-138.

Colloty, B.M., Adams, J.B., Bate, G.C., 1998. The use of a botanical importance rating to assess changes in the flora of the Swartkops Estuary over time. *Water SA* 26: 171-180.

Conway, T., Tans, P., 2012. Globally averaged marine surface annual mean data. U.S. National Oceanic and Atmospheric Administration, Earth System Research Laboratory

Cooper, J.A.G., 2002. The role of extreme floods in estuary-coastal behaviour: contrast between river and tide dominated micro tidal estuaries. *Sedimentary Geology* 150: 123-137.

Costanza, R., Kemp. W.M., Boynton, W.R., 1993. Predictability, scale, and biodiversity in coastal and estuarine ecosystems: implications for management. *Ambio 22*: 88–96.

Cotiyane P. 2016. Rhe response of microalgal biomass and community composition to the chemical and physical dynamics of two Eastern Cape estuaries. Masters Thesis. Rhodes University

Cox, E.F., Allen., 1999. Stand structure and productivity of the introduced *Rhizophora mangle* in Hawaii. *Estuaries* 22: 276–284.

Cuc, N.T.K. Nghia, N., Tam, D.K, Chau, T.M., To, H.H., Cua, N.Q., 2008. Current status of mangrove restoration and management in the southern provinces of Vietnam in PN

Hong. In: Cuc, N.T.K., Hien, V.T. (Eds.), Restoration of Mangroves. Agriculture Publisher House, Ha Noi, Vietnam, pp. 129e138 (In Vietnamese)

Dahdouh-Guebas F, Satyanarayana, B, (Ed.)., 2012. Proceedings of the International Conference 'Meeting on Mangrove ecology, functioning and Management - MMM3', Galle, Sri Lanka, 2-6 July 2012. *VLIZ Special Publication*, 57. Vlaams Instituut voor de Zee (VLIZ): Oostende. ISBN 978-90-817451-6-1. xxxi, 192 p

Dangremond, E.M., Feller, I.C., 2014. Functional traits and nutrient limitation in the rare mangrove *Pelliciera rhizophorae*. *Aquatic Botany* 116: 1-7.

Day, J.W., Hall, C.A.S., Kemp, W.M., Yanez-Arancibia, A., 1989. *Estuarine Ecology*. New York: John Wiley and Sons. 558 pp.

de Lange, W.P., de lange, P.J., 1994. An appraisal of factors controlling latitudinal distribution of mangrove (*Avicennia marina var. resinifera*) in New Zealand. *Journal of Coastal Research* 10: 539-548.

Delgado, P., P. F., Hensel, J. A., Jime´nez and J. W. Day Jr., 2001. The importance of propagule establishment and physical factors in mangrove distributional patterns in a Costa Rican estuary. *Aquatic Botany* 71: 157–178.

De Wet, J.S., 2004. Establishing an economic value for the mangroves of the Mngazana Estuary in the Eastern Cape. MSc Thesis, University of KwaZulu Natal.

DEAT, 2004a.Overview of Integrated Environmental Management, Integrated Environmental Management Information Series 0, Department of Environmental Affairs and Tourism (DEAT), Pretoria.

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

Department of Economic Development, Environmental Affairs and Tourism. 2013. Nahoon Estuarine Management Plan, Report J-616-13.

De Ryck, D.J.R., Koedam, N., Van der Stocken, T., van der Ven, R.S., Adams, J., Triest, L., 2016. Dispersal limitation of the mangrove *Avicennia marina* at its South African range limit in strong contrast to connectivity in its core East African region. *Marine Ecology Progress Series* 545: 124-134.

Deurwarder, H., 2012. How are anatomical and hydraulic features of *Avicennia marina* and *Rhizophora mucronata* trees influenced by siltation? MSc. Universitieit Gent.

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

Dittmar, T., Lara, R.J., 2001. Driving forces behind nutrient and organic matter dynamics in a mangrove tidal creek in north Brazil. *Estuarine Coastal Shelf Science* 52: 249 – 259.

Doyle, T.W., 2003. Predicting Future Mangrove Forest Migration in the Everglades under Rising Sea Level: U.S. Geological Survey Fact Sheet 030-03, 2 p. Available from http://www.nwrc.usgs.gov/factshts/030-03.pdf

Du Preez, M., Hosking, S., 2010. Estimating the recreational value of freshwater inflows into the Klein and Kwelera Estuaries: An application of the zonal travel cost method. *Water SA* 36 (5): 1-9.

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

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

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

Duke, N.C., Yuk Ying Lo, E., Sun, M., 2002. Global distributions and genetic discontinuities of mangroves- emerging patterns in the evolution of *Rhizophora. Trees* 16: 65-79.

Duke, N.C., 1992. Mangrove floristics and biogeography. Pages 63-100 in AI Robertson and DM Alongi, editors. Tropical mangroves ecosystems. American Geophysical Union, Washington, DC.

Duke, N.C., 1995. Genetic diversity, distribution barriers and rafting continents-more thoughts on the evolution of mangroves. *Hydrobiologia* 295: 167-181.

Duke, N.C., 2006. Australia's Mangroves. The Authoritative Guide to Australia's Mangrove Plants. Brisbane: The University of Queensland and Norman C Duke, 200 pp

Dunham, N.R., 2014. Influence of hydrological and environmental conditions on mangrove vegetation at coastal and inland semi-arid areas on the Gascoyne region. Master's Thesis. Edith Cowan University.

Elliot, M., Whitfield, A.K., 2011. Challenging paradigms in estuarine ecology and management. *Estuarine, Coastal and Shelf Science* 94: 306-314.

Ellis, J., Nicholls, P., Craggs, R., Hofstra, D., Hewitt, J., 2004. Effects of terrigenous sedimentation on mangrove physiology and associated macrobenthic communities. *Marine Ecology Progress Series* 207: 71 –82.

Ellis, W.L., Bowles, J.W., Erickson, A.A., Stafford, N., Bell, S.S., Thomas, M., 2006. Alteration of the chemical composition of mangrove (*Laguncularia racemosa*) leaf litter fall by freeze damage. *Estuar. Coast. Shelf Sci.* 68: 363–371.

Ellison J. 2004. Vulnerability of Fiji's Mangroves and Associated Coral Reefs to Climate Change. Review for the World Wildlife Fund. Launceston, Australia: University of Tasmania.

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

Ellison, A.M., 2000. Mangrove restoration: do we know enough? *Restoration Ecology* 8(3): 219–29.

Ellison, J., Stodart, D., 1991. Mangrove ecosystem collapse during predicted sea level rise: Holocene analogues and implications. *Journal of Coastal Research* 7: 151– 165.

Ellison, J., 2000. How South Pacific mangroves may respond to predicted climate change and sea level rise. In: Gillespie, A Burns, W. (Eds.), Climate Change in the South Pacific: Impacts and Responses in Australia, New Zealand, and Small Islands States. Kluwer Academic Publishers, Dordrecht, Netherlands, (Chapter 15), 289–301 pp.

Ellison, J., 2001. Possible impacts of predicted sea level rise on South Pacific mangroves. 289-301pp IN Noye, B., M. Grzechnik (eds.). Sea level Changes and their Effects. Singapore: World Scientific Publishing Company.

Ellison, J., 2004. Vulnerability of Fiji's Mangroves and Associated Coral Reefs to Climate Change. Review for the World Wildlife Fund. Launceston, Australia: University of Tasmania.

Ellison, J.C., 1998. Impacts of sediment burial on mangroves. *Marine Pollution Bulletin* 37: 420–426.

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

Emmerson, W., 2005. The nutrient status of Mngazana, a warm temperate mangrove estuary in the Transkei, Eastern Cape, South Africa. *Wetlands Ecology and Management 13:* 405-418.

Erftemeijier, P.L.A., Hamerlynck, O., 2005. Dieback of the mangrove *Heritiera littoralis* dryland, in the Rufiji Delta (Tanzania) following El Nino Floods. *Journal of Coastal Science* 42: 228-235.

Eslami-Andargoli, L., Dale, P.E.R., Sipe, N., Chaseling, J., 2009. Mangrove expansion and rainfall patterns in Moreton Bay, Southeast Queensland, Australia. *Estuarine, Coastal and Shelf Science* 85: 292-298.

Eslami-Andargoli, L., Dale, P.E.R., Sipe, N., Chaseling, J., 2010. Local and landscape effects on spatial patterns of mangrove forests during wetter and drier periods. Moreton Bay, Southeast Queensland, Australia. *Estuarine, Coastal and Shelf Science* 89: 51-63.

Faki, S.M., 2003. Impacts of Salt Pans to the Mangroves and their Associated Organisms. Dar es Salaam: UDSM University Press.

Food and Agricultural Organization.,2005. Global forest resources assessment 2005. Thematic study on Mangroves-Somalia Country profile. FAO. Rome

Food and Agricultural Organization., 2007. Mangroves of Africa 1980-2005: Country reports. Forest Resources Assessment Programme, Working Paper 135, Rome, Italy.

Feller, I.C., Sitnik, M., (eds) 2002. Mangrove Ecology: A manual for a field course. A field Manual focused on the Biocomplexity of mangrove ecosystems. Smithsonian Institution.

Feller, I.C., McKee, K.L., Whigham, D.F., O'Neill, J.P., 2003. Nitrogen vs. phosphorus limitation across an ecotonal gradient in a mangrove forest. *Biogeochemistry* 62: 145–175.

Feller, I.C., Whigham, D.F., O'Neill, J.P., McKee, K.M., 1999. Effects of Nutrient Enrichment On Within-Stand Nutrient Cycling in Mangrove Ecosystems in Belize. *Ecology* 80: 2193– 2205.

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

Field, C., 1995. Impacts of expected climate change on mangroves. *Hydrobiologia* 295: 75–81.

Fong, P., Zedler, J.B., 2000. Sources, Sinks, and Fluxes of Nutrients (N + P) in a Small Highly-Modified Estuary in Southern California. *Urban Ecosystems* 4: 125-144.

Gee, G.W., Bauder, J.W., 1986. *Particle-size analysis. In:* Klute A. (eds), Methods of soil analysis; Part I- Physical and mineralogical methods. (2nd eds). Agronomy No. 9 Part 1. Madison, Wisconsin, USA. 383-411 pp.

Geldenhuys, C., 2013. Mangroves and saltmarsh dynamics at Nahoon Estuary, Eastern Cape: a planted mangrove forest. MSc Thesis. Botany Department, Rhodes University.

Ghose.,2013. www. Livescience.com

Gilbert, G.S., Gorospe, J., Ryvarden, L., 2008. Host and habitat preferences of polypore fungi in Micronesian tropical flooded forests. *Mycological Research* 112: 674–680.

Gilman, E.L., Ellison, J., Duke, N.C., Field, C., 2008. Threats to mangroves from climate change and adaptation options. *Aquatic Botany* doi: 10.1016/j.aquabot.2007.12.009.

Giri, C., Ochieng, E., Tieszen, L., Zhu, Z., Singh, A., Loveland, T., Masek, J., Duke, N., 2011. Status and distribution of mangrove forests of the world using earth observation satellite data. *Global Ecology and Biogeography* 20(1): 154–159.

Giri, C., Long, J., Abbas, S., Murali, R.M., Qamer, F.M., Pengra, B., Thau, D., 2015. Distribution and dynamics of mangrove forests of South Asia. *Journal of Environmental Management* 148: 101-111.

Gladstone-Gallagher, R.V., Lundquist, C.J., Pilditch, C.A., 2014. Mangrove (*Avicennia marina* subsp. australasica) litter production and decomposition in a temperate estuary. *New Zealand Journal of Marine and Freshwater Research*, *48*(1):24-37. doi:10.1080/00288330.2013.827124

Gleeson, J., Santos, I.R., Maher, D.T., Golsby-Smith, L., 2013. Groundwater-surface water exchange in a mangrove tidal creek: Evidence from natural geochemical tracers and implications for the nutrient budget. *Marine Chemistry* 156: 27-37.

Godoy, M.D.P., De Lacerda, L.D., 2015. Mangroves Responses to climate change: A review of recent findings on mangrove extensions and distribution. *Anias da Academia Brasileira de Ciencias* 87(2): 651-667.

Grange, N., Allanson, B.R., 1995. The influence of freshwater inflow on the nature, amount and distribution of seston in estuaries of the Eastern Cape, South Africa. *Estuarine, Coastal and Shelf Science* 40: 403-420.

Griffiths, C.L., 1974. The amphipoda of Southern Africa. Part 3. CSIR, University of Cape Town.

Groom, Q., 2012. Some poleward movement of British native vascular plants is occurring, but the fingerprint of climate change is not evident. *PeerJ* 1 (e77). doi:10.7717/peerj.77.

Ha, H.T., Duarte, C.M., Tri, N.H., Terrados, J., Borum, J., 2003. Growth and population dynamics during early stages of the mangrove *Kandelia candel* in Halong Bay, North Viet Nam. *Estuarine Coastal and Shelf Science* 58: 435-444.

Harper, J.L., 1977. *Population Biology of Plants*. Academic Press, New York. 600-601pp.

Hariss, R.J, Milbrandt, E.C., Evenhan E.M., and Bovard, B.D., 2008. Tidal influence on mangrove community structure and recovery post hurricane disturbance; Revisiting Hurricane Charlie 2004. Sanibel captiva Conservation Foundation.

Harrison, T.D., Cooper, J.A.G., Ramm, A.E.L., Singh, R.A., 1996. Application of the Estuarine Health Index to South Africa's estuaries, Groot (Wes) - Great Fish. Unpublished Technical Report. CSIR, Durban

Harty, C., 2009. Mangrove planning and management in New Zealand and South East Australia—a reflection on approaches. *Ocean and Coastal Management* 52: 278–286.

Hastuti, E.D., Anggoro, S., Pribadi, R., 2012. The effects of environmental factors on the dynamic growth patterns of mangrove *Avicennia marina. Journal of Coastal Development* 16(1): 57-61.

Heady, W.N., Ross, P.C., O' Connor, K., Clark, C., Endris, C., Ryan, S., Stoner-Duncan, S., 2015. Assessing California's bar built estuaries using the California Rapid Assessment Method. *Ecological Indicators* 58: 300-310.

Herna´ndez-Terrones, L., Rebolledo-Vieyra, M., Merino-Ibarra, M., Soto, M., LeCossec, A., *et al.*, 2011. Groundwater pollution in a karstic region (NE Yucatan): Baseline nutrient content and flux to coastal ecosystems. *Water, Air, and Soil Pollution* 218: 517–528.

Hilbish, T.J., Brannock, P.M., Jones, K.R., Smith, A.B., Bullock, B.N., Wethey, D.S., 2010. Historical changes in the distributions of invasive and endemic marine invertebrates are contrary to global warming predictions: the effects of decadal climate oscillations. *Journal of Biogeography* 37: 423–431.

Hoff, R., Hensel, P., Proffitt, E.C, Delgado, P., 2010. Oil spills in mangroves. US Department of commerce.

Hopley, D., Smithers, S., 2010. Encyclopedia of World's Coastal Landforms. Springer 1260. ISBN 978-1-4020-8638-0. Queensland

Hoppe-Speer, S.C., 2012. Response of mangroves in South Africa to anthropogenic and natural impacts. PhD Thesis. Faculty of Science at Nelson Mandela Metropolitan University.

Hoppe-Speer, S.C.L., Adams, J.B., Rajkaran, A., 2013. Response of mangroves to drought and non-tidal conditions in St Lucia. *African Journal of Aquatic Science* 38(2): 153-162.

Hoppe-Speer, S.C.L., Adams, J.B., Rajkaran, A., 2015. Mangrove expansion and population structure at a planted site, East London, South Africa. *Southern Forests* 1-9.

Hughes, C.E., Binning, P., Willgoose, G.R., 1998. Characterisation of the hydrology of an estuarine wetland. *Journal of Hydrology* 211: 34-39.

Hunt, R., Hand, D.W., Hannah, M.A., Neal, A.M., 1991. Response to CO2 enrichment in 27 herbaceous species. *Functional Ecology* 5: 410-421.

Huntley, B., Birks, H.J.B., 1983. *An Atlas of past and present pollen maps for Europe: 0-13 000 Years Ago*. Cambridge University Press.

Hutchings, P., Saenger, P., 1987. Ecology of mangroves. *University of Queensland Press*, St Lucia, Qld. *Proceeding of Ecological Society of Australia* 13: 257-265.

Hwang, Y., Chen, S., 2001. Effects of ammonium, phosphates and salinity on growth, gas exchange characteristics and ionic contents of seedlings of mangrove *Kandelia candel*(L.). *Botanical Bulletin of Academia Sinica* 42: 131-139.

Ilami, F., Anderson, G.H., Smith iii, T.J.,2002. Spatial and temporal analysis of soil porewater nutrients in an estuarine fringe mangrove forest Harney River Estuary, Everglades National Park.

IPCC Fourth Assessment Report., 2007. Climate change Synthesis Report. Contribution of Working Groups I, II and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, Pachauri, R.K and Reisinger, A. (eds.)]. IPCC, Geneva, Switzerland. 104 pp.

Islam, S.F., 2008. Cultural landscape changing die to anthropogenic influences on surface water and threats to mangrove wetland ecosystems: a case study on the Sundarbans, Bangladesh. PhD Thesis. Brandenburg University of Technology at Cottbus.

Jafta, N., 2010. The botanical importance and health of the Bushmans Estuary, Eastern Cape, South Africa. MSc Thesis. Nelson Mandela Metropolitan University, South Africa.

James, N.C., Harrison, T.D., 2011. A preliminary survey of the estuaries on the southeast coast of South Africa, Old Woman's –Tyolomnqa with particular reference to the fish fauna. *Transactions of the Royal Society of South Africa* 66(2): 59-77.

Jezenski, W.A., Roberts, C.P.R.,1986. Estuarine and Lake Freshwater Requirements. Department of Water Affairs Technical Report No. TR 129. DWAF, South Africa. 36 pp.

Jia, K., Wei, X., Gu, X., Yao, Y., Xie, X., Li, B., 2014. Land cover classification using Landsat 8 operational land imager data in Beijing, China. *Geocarto International* 29: 941–951.

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

Joshi, H., Ghose, M., 2003. Forest structure and species distribution along soil salinity and pH gradient in mangrove swamps of the Sundarbans. *Tropical Ecology* 44 (2): 197-206.

Joye, S.B., Lee, R.Y., Joye, J.L., Feller, I.C., 2005. Sediment water flux of inorganic and organic constituents in mangroves in Belize and Panama. Estuarine. Research. Federation, 18th, Norfolk. Port Republic, MD: ERF

Kairo, J.G., Dahdou-Guebas, F., Bosire, J., Koedam, N., 2004. Restoration and management of mangrove systems- a lesson for and from the East African region. *South African Journal of Botany* 67: 383-389.

Kaiser, D., Kowalski, N., Bottcher, M.E., Yan, B., Unger, D., 2015. Benthic nutrient fluxes from mangrove sediments of an anthropogenically impacted estuary in Southern China. *Journal of Marine Science and Engineering* 3: 466-491.

Kampf, J., 2010. *Advanced ocean modelling: using open- Source Software.* Spronger Heidelberg Dordrecht London, New York.

Kao, W.Y., Tsai, T.T., Tsai, H.C., Shih, C.N., 2006. Response of three Glycine species to salt stress. *Environmental and Experimental Botany* 56: 120–125.

Karr, J.R.,1999. Defining and measuring river health. *Freshwater Biology* 41: 221-234.

Kathiresan, K., Rajendran, N., 2002. Fishery resources and economic gain in three mangrove areas on the south-east coast of India. *Fisheries Management and Ecology* 9: 277-283.

Kjerfve, B., Macintosh, D.J., 1997. Climate change impacts on mangrove ecosystems. In: Kjerfve B, Lacerda LD. and Diop S. (eds), Mangrove Ecosystem Studies in Latin America and Africa. UNESCO, ISME. Paris. 1–7 pp.

Krauss, K.W., Doyle, T.W., Twilley, R.R., Rivera-Monroy, V.H., and Sullivan, J.K., 2006. Evaluating the relative contributions of hydroperiod and soil fertility on growth of south Florida mangroves. Hydrobiologia 569:311–324.

Krauss, K.W., Lovelock, C.E., MCKee, K.L., Lopez-Hoffman, L., Ewe, S.M.L., Sousa, W.P., 2008. Environmental drovers in mangrove establishment and early development: A review. *Aquatic Botany* 89: 105-127.

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

Kruger, F.J., Richardson, D.M., van Wilgen, B.W., 1986. Processes of invasion by alien plants. In: Macdonald IAW, Kruger FJ, Ferrar AA (eds.), the Ecology and Management of Biological Invasions in Southern Africa. Cape Town, South Africa: Oxford University Press, 145-155.

Kunneke, J.T., Palik, T.F., 1984. Tampa Bay Atlas. *U.S Fish Wildl. Serv. Bio. Rep.* 85 (15): 3.

Lamberth, S.L., Turpie, J.K., 2003. The role of estuaries in South African fisheries: economic importance and management implications. *African Journal of Marine Science* 25: 131-157.

Lee, S.Y., Primavera, J.H., Dahdouh-Guebas, F., McKee, K., Bosire, J.O., Cannicci, S., Diele, K., Fromard, F., Koedam, N., Marchand, C., Mendelssohn, I., Mukherjee, N., Record, S., 2014. Ecological role and services of tropical mangrove ecosystems: a reassessment. *Global Ecology and Biogeography* 23: 726-743.

Lenoir, J., Gégout, J.C., Guisan, A., Vittoz, P., Wohlgemuth, T., Zimmermann, N.E., Dullinger, S., Pauli, H,, Willner,W., Svenning, J.C., 2010. Going against the flow: potential mechanisms for unexpected downslope range shifts in a warming climate. *Ecography* 33: 295–303.

Lewis III, RR., 2009. Mangrove Field of Dreams: If we build it will they come? SWS Research Brief. *Wetland Science and Practice* 27 (1): 15-18.

Liu, K., Liu, L., Liu, H., Li, X., Wang, S., 2014. Exploring the effects of biophysical parameters on the spatial pattern of rare cold damage to mangrove forests. *Remote Sensing of Environment* 150: 20-33.

Long, J., Napton, D., Giri, C., Graesser, J., 2014. A mapping and monitoring assessment of the Philippine's mangrove forests from 1990 to 2010. *Journal of Coastal Restoration* 30: 260-271.

Loon, A.F., Brake, T.E., Van Huijgevoort, M.H.J., Dijksma, R., 2016. Hydrological classification, a practical tool for mangrove restoration. Plos One | DOI: 10.1371/journal.pone.0150302

Lovelock, C.E., Ruess, R.W., Feller, I.C., 2006. Fine root respiration in the mangrove *Rhizophora mangle* over variation in forest stature and nutrient availability. *Tree Physiology* 26: 1601-1606.

Lovelock, C.E., Ellison, J.C., 2007. Vulnerability of mangroves and tidal wetlands of the Great Barrier Reef to climate change. In: Johnson, J.E Marshall, P.A. (Eds.), Climate Change and the Great Barrier Reef: A Vulnerability Assessment. Great Barrier Reef Marine Park Authority and Australian Greenhouse Office, Australia. 237–269 pp.

Lovelock, C.E., Feller, I.C., Ball, M.C., Ellis, J., Sorrell, B., 2007. Testing the Growth Rate vs. Geochemical Hypothesis for latitudinal variation in plant nutrients. *Ecology Letters* 10: 1154-1163.

Lovelock, C.E., Feller, I.C., Ellis, J., Schwarz ,A.M., Hancock, N., Nichols, P., Sorell, B., 2007. Mangrove growth in New Zealand estuaries: the role of nutrient enrichment at sites with contrasting rates of sedimentation. *Oecologia* 153: 633-641.

Lovelock, C.E., Ball, M.C., Martin, K.C., Feller, I.C., 2009. Nutrient enrichment increases mortality of mangroves. *Plos One* 4(5): 5600.

Lovelock, C.E., Sorrell, B.K., Hancock, N., Hua, Q., Swales, A., 2010. Mangrove forest and soil development on a rapidly accreting shore in New Zealand. *Ecosystem* 13: 437-451.

Lugo, A.E., Brinson, M.M., Brown, S., 1989. Synthesis and search for paradigms in wetland ecology. Pages 447- 460 in A. E. Lugo, M. Brinson, and S. Brown, editors. Forested wetlands: ecosystems of the world 15. Elsevier, Amsterdam, The Netherlands.

Lugo, A.E., Medina, E., McGinley, K., 2014. Issues and Challenges of Mangrove Conservation in the Anthropocene. *Madera y Bosques* 20(3): 11-38.

Lugo, A.E., Patterson-Zucca, C., 1977. The impact of low temperature stress on mangrove structure and growth. *Journal of Tropical Ecology* 18: 149-161.

Lugo, A.E., Sell, M., Snedaker, S.C., 1976. Mangrove ecosystem analysis: In pattern B.B 9Ed0, Systems Analysis and Simulations in Ecology. Academic Press, New York. 113-145 pp.

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

Lugo, A.E., 1996. Caribbean island landscapes: indicators of the effects of economic growth on the region. *Environment and Development Economics* 1(1): 128-136.

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

Lugo, AE., 1998. Mangrove forests: a tough system to invade but an easy one to rehabilitate. *Marine Pollution Bulletin* 37: 427-430.

Lutjeharms, J.R.E., de Ruitjer, W.P.M., 1996. The influence of the Agulhas Current on the adjacent coastal ocean: Possible impacts of climate change. *Journal of Marine Sys*tems 7: 321-336.

Magni, P., Montani, S., 2006. Seasonal patterns of porewater nutrients, benthic chlorophyll a and sedimentary AVS in a macrobenthos-rich tidal flat. *Hydrobiologia* 571: 297-311.

Manson, F.J., Loneragan, N.R., Phinn, S.R., 2003. Spatial and temporal variation in distribution of mangroves in Moreton Bay, subtropical Australia: a comparison of pattern metrics and change detection analyses based on aerial photographs. *Estuarine, Coastal and Shelf Science* 57: 653-666.

Maponya, P.M., Ngulube, P., 2007. The State of Estuarine Knowledge of the Communities of the Tyolomnqa Estuary in the Eastern Cape, South Africa. *Journal of Library and Information Science* 73(1): 75-83.

Markley, J.L., McMillan, C., Thompson, G.A Jr., 1982. Latitudinal differentiation in response to chilling temperatures among populations of three mangroves, *Avicennia germinans*, *Laguncularia racemosa*, and *Rhizophora mangle*, from the western tropical Atlantic and Pacific Panama. *Canadian Journal of Botany* 60: 2704–2715.

Martin, K.C., Bruhn, D., Lovelock, C.E., Feller, I.C., Evans, J.R., Ball, M.C., 2010. Nitrogen fertilization enhances water-use efficiency in a saline environment. *Plant Cell and Environment* 33: 344–357

Mbande, S., Whitfield, A., Cowley, P., 2002. The ichthyfaunal composition of the Mngazi and Mngazana Estuaries: a comparative study. SAIAB ISSN 1684-4149.

McKee, K.L., Mendelssohn, I.A., Materne, M.D., 2004. Acute salt marsh dieback in the Mississippi River deltaic plain: a drought-induced phenomenon? *Global Ecology and Biogeography* 13: 65-73.

McKee, K.L., Rooth, J.E., 2008. Where temperate meets tropical: multifactorial effects of elevated CO2, nitrogen enrichment, and competition on a mangrove-salt marsh community. *Global Change Biology* 14: 1-14.

McKee, K.L., 1995. Interspecific variation in growth, biomass partitioning, and defensive characteristics of neotropical mangrove seedlings: response to light and nutrient availability. *American Journal of Botany* 82: 299–307.

McLeod, E., Salm, R.V., 2006. Managing mangroves for resilience to climate change (IUCN Resilience Science Group Working Paper Series - No. 2). Gland, The World Conservation Union. Retrieved from CAKE http://www.cakex.org/virtual-library/537

McLusky, D.S., Elliot, M., 2004. The Estuarine Ecosystem: Ecology, Threats and Management. New York: Oxford University Press. ISBN 0-19-852508-7

Mcnae, W.,1968. A general account of the fauna and flora of mangrove swamps and forests in the Indo-West-Pacific region. *Advances in Marine Biology* 673-220.

Medina, E., Cuevas, E., Lugo, A.E., 2010. Nutrient relations of dwarf *Rhizophora mangle L.* mangroves on peat in eastern Puerto Rico. *Plant Ecology* 207: 13-24.

Menezes, M., Berger, U., Worbes, M., 2003. Annual growth rings and long-term patterns of mangrove trees from the Brangança peninsular, North Brazil. *Wetlands Ecology and Management* 11: 233-242.

Mhangara, P., Kakembo, V., 2012. An Object- Based Classification and fragmentation analysis of land use and cover change in the Keiskamma catchment, Eastern Cape, South Africa. *World Applied Sciences Journal* 19 (7): 1018-1029.

Middelburg, J.J., Niewenhuize, J., Slim, F.J., Ohowa, B., 1996. Sediment biogeochemistry in an east African mangrove forest (Gazi Bay, Kenya). *Biogeochemistry* 34: 1323-155.

Milbrandt, E. C., Greenawalt-Boswell, J. M., Sokoloff, P. D. and Bartone, S. A., 2006. Impact and response of Southwest Florida mangroves to the 2004 Hurricane Season. *Estuaries and Coasts*. 29 (6a): 979-984

Miththapala, Sriyanie .2013. *Lagoons and Estuaries.* Coastal Ecosystems Series (Vol 4). vi + 73 pp. IUCN Sri Lanka Country Office, Colombo.

Mohammed, S.M., Johnstone, R.W., 2002. Studies on benthic denitrification in the Chwaka Bay mangrove sediments, Zanzibar. *Tanzania Journal of Science* 28(1).

Montagna, P.A., Brenner, J., Gibeaut, J.C., Morehead, S., 2011. Coastal impacts In. Schmandt J, North GR, Clarkson J, editors. The Impact of Global Warming on Texas. 2nd edition ed. Austin, TX: University of Texas Press.

Moore, G.E., Grizzle, R.E., Ward, K.M., Alshihi, R.M., 2015. Distribution, porewater chemistry and stand characteristics of the mangroves on the United Arab Emirates. *Journal of Coastal Research* 31(4): 957-963.

Morris, R.D., 1986. Recreational capabilities and zoning of the Nahoon Estuary, East London. Honours Thesis. Rhodes University. Grahamstown, South Africa.

Morrisey, D.J., Beard, C., Morrison, M., Craggs, R., Lowe, M., 2007. The New Zealand mangrove: review of the current state of knowledge. Auckland Regional Council Technical Publication, 325

Morrisey, D.J., Swales, A., Dittman, S., Morrison, M.A., Lovelock, C.E., Beard, C.M., 2010. The ecology and management of temperate mangroves. *Oceanography and Marine Biology: An Annual Review* 48: 43-160.

Mucina, L., Jürgens, N., Le Roux, A., Rutherford, M.C., Schmiedel, U., Esler, K.J., Powrie, L.W., Desmet, P.G., Milton, S.J., 2006. Succulent Karoo Biome. In L. Mucina and M. C. Rutherford (editors), The vegetation of South Africa, Lesotho and Swaziland.*Strelitzia* 19: 221–299. South African National Biodiversity Institute, Pretoria.

Naidoo, G., 1985. Effects of waterlogging and salinity on plant water relations and on the accumulation of solutes in three mangrove species. *Aquatic Botany* 22: 133-143.

Naidoo, G., 1986. Responses of the mangrove *Rhizophora mucronata* to high salinities and low osmotic potentials. *South African Journal of Botany* 52: 124-128.

Naidoo, G., 1987. Effects of salinity and nitrogen on growth and plant water relations in the mangrove *Avicennia marina* (Forsk.) Vierh. *New Phytologist* 107: 317–326.

Naidoo, G., 1990. Effects of nitrate, ammonium and salinity on growth of the mangrove *Bruguiera gymnorrhiza* (L) Lam. *Aquatic Botany* 38: 209-219.

Naidoo, G., 2009. Differential effects of nitrogen and phosphorus enrichment on growth of dwarf *Avicennia marina* mangroves. *Aquatic Botany* 90: 184–190.

Naiman, R.J., Mellilo, J.M., Hobbie, J.E., 1986. Ecosystem alteration of boreal forest streams by beaver (Castor canadensis). *Ecology* 67: 1254-1269.

National Ocean and Atmospheric Administration., 2010. Classifying estuaries. http://oceanservice.noaa.gov/education/tutorial_estuaries/media/supp_est04_barbuilt.html

Noor, T., Batool, N., Mazhar, R., Ilyas, N., 2015. Effects of siltation, temperature and salinity on mangrove plants. *European Academic Research* Vol II (1) ISSN 2286-4822

Van Niekerk, L., Turpie, J.K., 2012. South African National Biodeoversity Assesment 2011: Technical Report. Volume 3: Estuary Component. CSIR Report Number. CSIR/NRF/ECOS/ER/2011/0045/B. Council for Scientific and Industrial Research, Stellenbosch.

Ngesi, H.N., 2010. Microalgal biomass and distribution in the Mngazi and Mngazana Estuaries. MSc Thesis. NMMU.

Nicholls, R.J., Lowe, J.A., 2004. Benefits of mitigation of climate change for coastal areas. *Global Environmental Change* 14: 229-244.

Northland Regional Council., 2005. http://www.nrc.govt.nz/for-schools/school-information-packs/mangroves/

Nunes, R.A., Lennon, G.W., 1986. Physical property distributions and seasonal trends in Spencer Gulf, South Australia: an inverse estuary. *Australian Journal of Marine and freshwater Research* 37: 39-53.

Odum, W.E., Johannes, R.E., 1975. The response of mangroves to man-induced stress. In: Odum WE, Johannes RE (eds) Tropical marine pollution. Oceanography Series. No. 12. Elsevier, Amsterdam. 52–62 pp.

Okimoto, Y., Nose, A., Katsuta, Y., Tateda, Y., Agarie, S., Ikeda, K., 2007. Gas exchange analysis for estimating net CO2 fixation capacity of mangrove (*Rhizophora stylosa*) forest in the mouth of River Fukido, Ishigaki Island, Japan. *Plant Production Science* 10: 303-313.

Okimoto, Y., Nose, A., Murdiyarso, D., Purbopuspito, J., Sasmito, S.D., 2013. Thinning practises in rehabilitated mangroves: Opportunity to synergize climate change mitigation and adaptation. Proceedings of the 7th International Conference on Asian and Pacific Coasts Bali, Indonesia, September 24-26.

Onuf, C.P., Teal, J.M., Valiela, I., 1977. Interactions of nutrients, plant growth and herbivory in a mangrove ecosystem. *Ecology* 58: 5 14-526.

Osland, M.J., Enwright, N., Day, R.H., Doyle, T.W., 2013. Winter climate change and coastal wetland foundation species: salt marshes vs. mangrove forests in the southeastern United States. *Global Change Biology* 19: 1482–1494.

Osland, M.J., Day, R.H., From AS, McCoy, M.I., MCLeod, J.L., Kelleway, J.J., 2015. Life stage influences the resistance and resilence of black mangrove forests to winter climate extremes. *Ecosphere* 6(9): 160.

Osorio, J.A., Wingfield, M.J., Roux, J., 2014. A review of factors associated with decline and death of mangroves, with particular reference to fungal pathogens. *South African Journal of Botany* 103: 295-301.

Padilla, C., Fortes, M.D., Duarte, C.M., Terrados, J., Kamp-Nielsen, L., 2004. Recruitment, mortality and growth of mangroves (*Rhizophora* sp.) seedlings in Ulugan Bay, Palawan, Philippines. *Trees* 18: 589-595.

Parida, A.K., Jha, B., 2010. Salt tolerance mechanisms in mangroves: a review. *Trees* 24: 199-217.

Parmesan, C., Yohe, G., 2003. A globally coherent fingerprint of climate change impacts across natural systems. *Nature* 421 (6918): 37–42.

Parsons, T.R., Maita, Y., Lalli, C.M., 1984. A Manual of Chemical and Biological Methods for Seawater Analysis. Pergamon Press, New York. 173 pp.

Pegg, K.G., Gillespie, N.C., Forsberg, N.C., 1980. Phytophthora sp. associated with mangrove death in central coastal Queensland. *Australasian Plant Pathology* 9(3): 6-7.

Penha-Lopes, G.P., 2009. Effect of domestic sewage on characteristics of mangrove communities and their functioning in East Africa. PhD Thesis, University of Lisbon, Portugal.

Perillo, G.M.E., 1995. Geomorphology and sedimentology of estuaries. Elsevier. Amsterdam. Lausanne, New York.

Prasad, M.B., Ramanathan, A.L., 2008. Sedimentary nutrient dynamics in a tropical estuarine mangrove ecosystem. *Estuarine, Coastal and Shelf Science* 80: 60-66.

Pressey RL, Cabeza M, Watts ME, Cowling RM, Wilson KA. 2007. "Conservation planning in a changing world". *Trends in Ecology and Evolution* 22 (11): 583–92. doi:10.1016/j.tree.2007.10.001. PMID 17981360.

Quisthoudt, K., Schmitz, N., Randin, C.F., Dahdou-Guebas, F., Robert, E.M., Koedam, N., 2012. Temperate variation among mangrove latitudinal range limits worldwide. *Trees* 26: 1918-1931.

Quisthoudt, K., Adams, J., Rajkaran, A., Dahdouh-Guebas, F., Koedam, N., Randin, C.F., 2013. Disentangling the effects of global climate and regional land use change on the current and future distribution of mangroves in South Africa. *Biodiversity Conservation* 22: 1369-1390.

Raabe, E.A., Roy, L.C., McIvor, C.C., 2012. Tampa Bay coastal wetlands. Nineteenth to twentieth century tidal marsh-to-mangrove conversion. *Estuaries and Coasts* 35(5): 1145-1162.

Rajkaran, A., Adams, J.B., du Preez, D.R., 2004. A method for monitoring harvesting at the Mngazana Estuary, South Africa. African *Journal of Aquatic Science* 29(1): 57-68.

Rajkaran, A., Adams, J.B.,2007. Mangrove litter production and organic carbon pools in the Mngazana Estuary, *South Africa. African Journal of Aquatic Sciences* 32 (1): 17–25.

Rajkaran, A., Adams, J., Taylor, R., 2009. Historic and recent (2006) state of mangroves in small estuaries from Mlalazi to Mtamvuna in KwaZulu-Natal, South Africa. *Southern Forests* 71: 287–296.

Rajkaran, A., Adams, J.B., 2010. The implications of harvesting on the population structure and sediment characteristics of the mangroves at Mngazana Estuary, Eastern Cape, South Africa. *Wetlands Management* 18: 79-89.

Rajkaran, A., 2011. A status assessment of mangrove forests in South Africa and the utilization of mangroves at Mngazana Estuary. PhD Thesis. Faculty of Science at Nelson Mandela Metropolitan University

Rajkaran, A., Adams, J.B., 2011. Mangrove forests of Northern Kwa-Zulu Natal: sediment conditions and population structure of the largest mangrove forests in South Africa. *Western Indian Ocean Journal of Marine Science* 10(1): 25– 38

Rajkaran, A., Adams, J.B., 2012. The effects of environmental variables on mortality and growth of mangroves at Mngazana Estuary, Eastern cape, South Africa. *Wetlands Ecology and Management* 20: 297-312.

Ramachandran, S., Anitha, S., Balamurugan, V., Dharanirajan, K., Vendhan, K.E., Divien, K.I.P., Vel, A.S., Hussain, I.S., Udayaraj, A., 2005. Ecological impact of tsunami on Nicobar Islands (Camorta, Katchal, Nancowry and Trinkat). *Current Science* 89(1).

Reddering, J.S.V., 1988. Coastal and catchment basin controls on estuary morphology of the south-eastern cape coast. *South African Journal of Science* 84: 154-157.

Reef, R., Ball, M.C., Lovelock, C.E., 2012. The impact of a locust plague on mangroves of the arid Western Australia coast. *Journal of Tropical Ecology* 28: 307–311.

Reef, R., Feller, I.C., Lovelock, C.E., 2010. Nutrition of mangroves. *Tree Physiology* 30:1148–1160.

Richardson, D.M., Bond, W.J., 1989. Determinants of plant distribution: Evidence from pine invasions. *The American Naturalist* 137 (5): 639-668.

Robert, E.M.R., Oste, J., Van der Stocken, T., de Ryck, D.J.R., Quisthoudt, K., Kairo, J.G., Dahdouh-Guebas, F., Koedam, N., Schmitz, N., 2015. Viviparous mangrove propagules of *Ceriops tagal* and *Rhizophora mucronata*, where both Rhizophoraceae show different dispersal and establishment strategies. *Journal of Experimental Marine Biology and Ecology* 468: 45-524.

Robertson, A.I., Alongi, D.M. (eds.)., 1992.Tropical Mangrove Ecosystems. *Coastal and Estuarine Studies* 41. American Geophysical Union, Washington, DC.329 pp.

Robertson, A.I, Daniel, P.A., 1989a. The influence of crabs on litter processing in high intertidal mangrove forests in tropical Australia. *Oecologia* 78: 191-198.

Rodriguez, J., Vos, F., Below, R., Guha-Sapir, D., 2009. Annual disaster statistical review 2008: the numbers and trends. Centre for research on the epidemiology of disaster, Jacoffset Printers, Melin, Belgium.

Ross, P.M., 2006. Macrofaunal loss and microhabitat destruction: the impact of trampling in a temperate mangrove forest, NSW, Australia. *Wetland Ecology and Managemen*t 14: 167-184.

Saenger, P., Moverly, J., 1985. Vegetative phenology of mangroves along the Queensland coastline. *Proceeding Ecological Society of Australia* 13: 257-265.

Sahoo, K., Dhal, N.K., 2009. Potential microbial diversity in mangrove ecosystems: A review. *Indian Journal of Marine Sciences* 38(2): 249-256.

Saifullah, S.M., Chaghtai, F., Akhtar, S., 2007. Dispersal and establishment of mangrove propagules in an exposed coastal habitat of Indus Delta. *Pakistan Journal of Botany* 39(2): 577-582.

Saintilan, N., Wilton, K., 2001. Changes in the distribution of mangroves and saltmarshes in Jervis Bay, Australia. *Wetlands Ecology and Management* 9: 409-420.

Saintilan, N., Rogers, K., 2015. Woody plant encroachment of grasslands: a comparison of terrestrial and wetland settings. New Phytologist 205: 1062–1070.

Saintilan, N., Wilson, N., Rogers, K., Rajkaran, A., Krauss, K.W., 2014. Mangrove expansion and saltmarsh decline at mangrove poleward limits. *Global Change Biology* 20 (1): 147-157.

Sanders, C.J., Smoak, J.M., Waters, M.N., Sanders, L.M., Brandini, N., Patchineelam, R., 2012. Organic matter content and particle size modifications in mangrove sediments as responses to sea level rise. *Marine Environmental Research* 77: 150-155.

Santos, L.C.M., Matos, H.R., Schaeffer-Novelli, Y., Cunha-Lignon, M., Bitencourt, M.D., Koedam, N., Dahdouh-Guebas, F., 2014. Assessment of anthropogenic pressures in the São Francisco River Estuary, (Northeast Brazil): a GIS-based analysis of CBERS and SPOT satellite images to aid local mangrove forest management. *Ocean and Coastal Management* 89: 39-50.

Satumanatpan, S., Keough, M.J., 1999. Effect of barnacles on the survival and growth of temperate mangrove seedlings. *Marine Ecology Progress Series* 181: 189-199.

Sekhwela, M.B.M., Yates, D.J., 2007. A phenological study of dominant acacia tree species in areas with different rainfall regimes in the Kalahari of Botswana.*Journal of Arid Environments* 70: 1–17.

Sepp€a, H., Birks, H.J.B., 2002. Holocene climate reconstructions from the Fennoscandian tree-line area based on pollen fata from Toskaljavri. *Quaternary Research* 57: 191–199.

Sgwabe, G., Vermeulen, W., van der Merwe, I., 2004. Report on the Mngazana mangrove forest area: a case study. Unpublished first draft: Department of Water Affairs and Forestry.

Shaick, M.Z., Tiwari, L.R., 2012. Sediment quality of Sewri mudflats, Mumbai, West Coast of India. International Journal of Scientific and Research Publications 2(10): 2250-3153.

Sharma, S., Hoque, A.T.M.R., Analuddin, K., Hagihara, A., 2012. Litter fall dynamics in an overcrowded mangrove *Kandelia obovate* (S.L) Yong stand over five years. *Estuarine Coastal and Shelf Science* 98: 31-41.

Simard, M., Rivera-Monroy, V.H., Mancera-Pineda, J.E., Castaneda Moya, E., Twilley, R.R., 2008. A systematic method for 3d mapping of mangrove forests based on shuttle radar topography mission elevation data, ICESat/GLAS waveforms and field data: Application to Cienaga Grande De Santa Marta, Colombia. *Remote Sensing of Environment* 112 (2008): 2131–2144

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

Smith, J., Michele, A., Burford, M.A., Revill, A.T., Haese, R.R., Fortune, J., 2012. Effect of nutrient loading on biogeochemical processes in tropical tidal creeks. *Biogeochemistry* 108: 359-380.

Snedaker, S., 1995. Mangroves and climate change in the Florida and Caribbean region: scenarios and hypotheses. *Hydrobiologia* 295: 43-49.

Snedaker, S.C., Araujo, R.J., 1998. Stomatal conductance and gas exchange in four species of Caribbean mangroves exposed to ambient and increased CO2. *Marine and Freshwater Research Journal* 49: 325-327.

Soares, M.L.G., Estrada, G.C.D., Fernandez, V., Tognella, M.M.P., 2012. Southern limit of the Western South Atlantic mangroves: Assessment of the potential effects of global warming from a biogeographical perspective. *Estuarine, Coastal and Shelf Science* 101: 44-53.

Solomon, S., Qin, D., Manning, M., Alley, R.B., Berntsen, T., Bindoff, N.L., Chen, Z., Chidthaisong, A., Gregory, J.M., Hegerl, G.C., Heimann, M., Hewitson, B., Hoskins, B.J., Joos, F., Jouzel, J., Kattsov, V., Lohmann, U., Matsuno, T., Molina, M, Nicholls, N., Overpeck, J., Raga, G., Ramaswamy, V., Ren, J., Rusticucci, M., Somerville, R., Stocker, T.F., Whetton, P., Wood, R.A., Wratt, D., 2007. Technical summary. In: Solomon S, Qin D, Manning M, Chen Z, Marquis M, Averyt KB, Tignor M, Miller HL. (Eds.), Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Sousa, W.P., Kennedy, P.G., Mitchell, B.J., 2003. Propagule size and predispersal damage by insects affect establishment and early growth of mangrove seedlings. *Oecologia* 135 (4): 564–575.

Spalding, M., Kainuma, M., Collins, L., 2010. *World atlas of mangroves*. Earth scan, London. Sullivan C. 2005. The importance of mangroves. <http://ufdc.ufl.edu/UF00093446/00028/1x> (Accessed 13.08.13).

Spencer, T., Moller, I., 2013. *Mangrove Systems*. In: Treatise on Geomorphology (ed. J.F. Shroder). San Diego, Academic Press. Volume 10. 360-391 pp.

Srivastava, P.B.L., Bal, H.S., 1984. Composition and distribution pattern of natural regeneration after second thinning in Matang mangrove reserve, Perak, Malaysia. In: Soepadimo E, Rao AN, Macintosh DJ (eds) Proc Asian Symp Mangrove Environ – Res Manage, Kuala Lumpur. 761–784 pp.

Statham, P.J., 2012. Nutrient in estuaries- An overview and the potential impacts of climate change. *Science of the Total Environment* 434: 213-227.

Steffen, S., Mucina, L., Kaereit, G., 2009. Three new species of *Sarcocornia* (Chenopodiaceae) from South Africa. *KEW Bulletin* 64: 447-459.

Steinke, T.D., 1972. Further observations on the distribution of mangroves in the Eastern Cape Province. *Journal of South African Botany* 38: 165-178.

Steinke, T.D., 1975. — some factors affecting dispersal and establishment of propagules of *Avicennia marina* (Forsk.) Vierh. In Proceedings of the International Symposium on the Biology and Management of Mangroves, Hawaii, October 1974. Walsh, G. E., Snedaker, S. C. and H. J. Teas (Eds). Gainesville; University of Florida. 846 pp.

Steinke, T.D., 1986. Mangroves of the East London area. *The Naturalist* 30(2): 50-53.

Steinke, T.D., Charles, L.M., 1986. Litter production by mangroves. I: Mngeni Estuary. *South African Journal of Botany* 52: 552-558.

Steinke, T.D., Ward, C.J., 1987. Degradation of mangrove leaf litter in the St Lucia Estuary as influenced by season and exposure. *South African Journal of Botany 53(5): 323-328.*

Steinke, T.D., 1988. Vegetative and floral phenology of three mangroves in Mngeni Estuary. *South African Journal of Botany* 54(2): 97-102.

Steinke, T.D., 1995. A general review of the mangroves of South Africa. In: Cowan GI. 1995. *Wetlands of South Africa*. Department of Environmental Affairs and Tourism, Pretoria. 53-73 pp.

Steinke, T.D., 1999. Mangroves in South African Estuaries. In. Allanson BR, Baird D. (eds), Estuaries of South Africa. Cambridge University Press. 119-140 pp.

Steinke, T.D., Ward, C.J., 2003. Use of plastic drift cards as indicators of possible dispersal of propagules of the mangrove *Avicennia marina* by ocean currents*. African Journal of Marine Science* 25: 169-176.

Stevens, F.L., 1920. New or noteworthy Porto Rican fungi. *Botanical Gazette* 70: 399-402.

Stevens, P.W., Fox, S, L., Montague, C.L., 2006. The interplay between mangroves and saltmarshes at the transition between temperate and subtropical climate in Florida. *Wetlands Ecology and Management* 14: 435-444.

Stokes, D.B., Healy, T.R., Cooke, P.J., 2010. Expansion dynamics of monospecific, temperate mangroves and sedimentation in two embayment's of a barrier-enclosed lagoon, Tauranga Harbour, New Zealand. *Journal of Coastal Research* 26(1): 113-122.

Strydom, N.A., Neira, F.J., 2006. Description and ecology of larvae of *Glossogobius callidus* and *Redigobius dewaali* (Gobiidae) from temperate South African estuaries. *African Zoology* 41(2): 240-251.

Strydom, N.A., Whitfield, A.K., Paterson, A.W., 2002. Influence of altered freshwater flow regimes on abundance of larval and juvenile *Gilchristella aestuaria* (Pisces: Clupeidae) in the upper reaches of two South African estuaries. *Marine and Freshwater Research* 53: 431-438.

Stuart, S.A., Choat, B., Martin, K.C., Holbrook, N.M., Ball, M.C., 2007. The role of freezing in setting the latitudinal limits of mangrove forests. *New Phytol*ogist 173: 576–583.

Talbot, M.B., Bate, G.C, Campbell, E.E., 1990. A review of the ecology of surf-zone diatoms, with special reference to *Anaulus australis*. Oceanography and Marine Biology: An Annual Review 28:155-185.

Talbot, S.S., Shasby, M.B., Bailey, T.N., 1985. Landsat-facilitated vegetation classification of the Kenai National Wildlife and adjacent areas, Alaska. Pages 333-345 in Pecora 10, remote sensing in forest and range resource management: proceedings, August 20-22, 1985, Colorado State University, Fort Collins, Colorado.

Tam, N.F.Y., Wong, Y.S., 1995. Spatial and temporal variations of heavy metal contamination in sediments of a mangrove swamp in Hong Kong. *Marine Pollution Bulletin* 31: 254-261.

Tan Le, L., 2008. Louisiana State University and Agricultural and Mechanical College, Doctor of Philosophy

Taylor, M., Ravilious, C., Green, E.P., 2003. Mangroves of East Africa. UNEP-WCMC Biodiversity Series, 13. UNEP/WCMC: Cambridge. 24 pp.

Teske, P.R., McQuad, C.D., Froneman, P.W., Barker, N.P., 2006. Impacts of marine biogeographic boundaries on phylographic patterns of three South African estuarine crustaceans. *Marine Ecology Progress Series* 314: 283-293.

Thomas, L., 2010. Soil pH and the availability of plant nutrients. IPNI plant nutrition today. No 2. www.ipni.net/pnt.

Tilman, D., 1988. Plant strategies and the dynamics of plant communities. Princeton University Press, Princeton

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

Tomlinson, P.B., 1994. The botany of mangroves, New York: Cambridge University Press. 436 pp.

Traynor, C.H., Hill, T., 2008. Mangrove utilisation and implications for participatory forest management, South African. *Conservation and Society* 6(2): 109-116.

Turpie, J.K., Adams, J.B., Joubert, A., Harrison, T.D., Colloty, B.M., Maree, R.C., Whitfield, A.K., Wooldridge, T.H., Lamberth, S.J., Taljaard, S., Van Niekerk, L., 2002. Assessment of the conservation priority status of South African estuaries for use in management and water allocation. *Water SA* 28(2). ISSN 0378-4738.

Twilley, R.R., 1997. Mangrove wetlands. In: Messina M and Connor W (eds.) Southern Forested Wetlands: *Ecology and Management* 445–473. Boca Raton, FL: CRC Press.

Twilley, R. R. and J. W. Day, Jr., 1999. The productivity and nutrient cycling of mangrove ecosystems, p. 127-152. In: A. Yáñez-Arancibia y A. L. LaraDomínguez (eds.). Ecosistemas de Manglar en América Tropical. Instituto de Ecología A.C. México, UICN/ORMA, Costa Rica, NOAA/NMFS Silver Spring MD USA. 380 p

Twilley, R.R., Lugo, A.E., Patterson-Zucca, C., 1986. Litter production and turnover in basin mangrove forests in southwest Florida. *Ecology* 67: 670 – 683.

Ukpong, I.E., 1997. Vegetation and its relation to soil nutrient and salinity in the Calabar mangrove swamp, Nigeria*. Mangroves Salt Marshes* 1: 211–218.

USGS, National Wetlands Research Centre. 2004. Global change impacts of mangrove ecosystems.

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

Valle-Levinson, A., 2010. Definition and classification of estuaries, in: Contemporary Issues in Estuarine Physics, A. Valle-Levinson (eds.). Cambridge, UK: Cambridge University Press.1-11 pp.

Valiela, I., Bowen, J., York, J., 2001. Mangrove forests: one of the world's threatened major tropical environments. *Bioscience* 51: 807–815.

Van de Koppel, J., Herman, P.M.J., Thoolen, P., Heip, C.H.R., 2001. Do alternate stable states occur in natural ecosystems? Evidence from a tidal flat. *Ecology* 82: 3449–3461.

Van Steenis, C.G.G.J., 1962. The distribution of mangrove plant genera and its significance for paleogeography. *Proceedings of the Section of Sciences, Koninklijke Akademie van Wetenschappen te Amsterdam 65: 164-169.*

von Maltitz, G., Mucina, L., Geldenhuys, C. *et al.*, 2003. Classification system for South African indigenous forests. Pretoria: Department of Water Affairs and Forestry.

Wakushima, S., Kuraishi, S., Sakurai, N., 1994a. Soil salinity and pH in Japanese mangrove forests and Joshi and Ghose 207 growth of cultivated mangrove plants in different soil conditions. *Journal of Plant Research* 107: 39- 46.

Walsh, G.E., 1974. Mangroves: a review. In: Reimold RJ, Queen WH (eds) Ecology of halophytes. Academic Press. New York. 51-174 pp.

Ward, C.J., Steinke, T.D., 1982. A note on the distribution and approximate arears of mangroves in South Africa. *South African Journal of Botany* 3: 51-53.

Watson, J.G., 1929. Mangrove forests of the Malay Peninsula. *Malayan Forest Records* no 6.

Wentworth, C.K., 1922. A scale of grade and class terms for clastic sediments. *The Journal of Geology* 30(5): 377–92.

Whitall, D., Bricker, S., Ferreira, J., Nobre, A.M., Simas, T., Silva, M., 2007. Assessment of eutrophication in estuaries: pressure-state-response and nitrogen source apportionment. *Environmental Management* 40(4): 678–690.

Whitfield, A.K., 1992. A characterisation of Southern African estuarine systems. *Southern African Journal of Aquatic Sciences* 12: 89-103.

Whitfield, A.K., Wooldridge, T.H., 1994. Changes to Freshwater Supplies to Southern African Estuaries: Some Theoretical and Practical Considerations. Olsen and Olsen, Fredensborg, Sweden.

Whitfield, A., Bate, G.C., 2007. A review of information on temporarily open/closed estuaries in the warm and cool temperate biogeographic regions of South Africa with particular emphasis on the influence of river flow on these systems. WRC Report 1581/1/07. Water Research Commission, Pretoria, South Africa.

Whitfield, A.K., Taylor, R.H., 2009. A review of the importance of freshwater inflow to the future conservation of Lake St Lucia. *Aquatic Conservation and Freshwater Ecosystems* 19: 838-848.

Wier, A.M., Tattar, T.A., Kleklowiski Jr, E.J., 2000. Disease of red mangrove (*Rhizophora mangle*) in *Rhizophorae*. *Biotropica* 32(2): 299-306

Wilkie, M.L., Fortuna, S., Souksavat, O., 2003. FAO's database on mangrove area estimates. Forest Resources Assessment Working Paper no. 62. Food and Agriculture Organization of the United Nations (FAO), Rome.

Wilkie, M.L., Fortuna, S., 2003. Status and trends of mangrove area worldwide. Forest Resources Assessment Working Paper no. 63. Food and Agriculture Organization of the United Nations (FAO), Rome.

Wiseman, K., Burns, M., Vernon, C., 1993. Nahoon (CSE 44), Qinira (CSE 45) and Gqunube (CSE 46). Stellenbosch: Estuarine and coastal research unit, division of Earth, Marine and Atmospheric Science and Technology, CSIR.

Worm, B., Barbier, E.B., Beaumont, N., Duffy, J.E., Folke, C., Halpern, B.S., Jackson, J.B.C., Lotze, HK., Micheli, F., Palumbi, S.R., Sala, E., Selkoe, K.A., Stachowicz, J.J., Watson, R., 2006. Impacts of biodiversity loss on ocean ecosystem services. *Science* 314: 787–790.

Wolanski, E., 1986. An evaporation driven salinity maximum zine in Australian tropical estuaries. Estuarine, Coastal and Shelf Science 22: 415-424.

Wolanski, E., Andutta, F.P., Delhez, E., 2012. Estuarine Hydrology. DOI: 10.1007/978-1-4020-4410-6_77

Wosten, J.H.M., de Willigen, P., Tri, N.H., Lien, T.V., Smith, S.V., 2003. Nutrient dynamics in mangrove areas of the Red River Estuary in Vietnam. *Estuarine, Coastal and Shelf Science* 57: 65-72.

Yang, S., Rachel, L.F.L., Sheue, C., Yong, J.W.H., 2013. The current status of mangrove forests in Singapore. Proceedings of NSS Conference: Nature Conservation for a Sustainable Singapore.

Yates, E.J., Ashwath, N., Midmore, DJ., 2004. Responses to nitrogen, phosphorus, potassium and sodium chloride by three mangrove species in pot culture. *Tree Structure and Function* 16: 120-125.

Yessoufou, K., Stoffeberg, G.H., 2015. Biogeography, threats and phylogenetic structure of mangrove forest globally and in South Africa: A review. *South African Journal of Botany*. http://dx.doi.org/10.1016/j.sajb.2015.11.002.

Yoskowitz, D.W., Santos, C., Allee, R.J., Carollo, C., Henderson, J., Jordan, S., Ritchie, J., 2010. Proceedings of the Gulf of Mexico ecosystem services workshop, Bay St. Louis, Mississippi; 2010 June 16–18. October. Texas A&M University-Corpus Christi: Harte Research Institute for Gulf of Mexico Studies.

Youssef, T., Saenger, P., 1996. Anatomical adaptive strategies to flooding and rhizosphere oxidation in mangrove seedlings. *Australian Journal of Botany* 44(3): 297-313.

Zhang, Y., Huang, G., Wang, W., Chen, L., Lin, G., 2012. Interactions between mangroves and exotic *Spartina* in an anthropogenically disturbed estuary in southern China. *Ecology* 93(3): 588-597.

Zhang, L., Shao, H., 2013. Direct plant-plant facilitation in coastal wetlands: A review. Estuarine, *Coastal and Shelf Science* 119: 1-6.

Appendices

Table A1: Summary of statistical analyses for seasonal changes in porewater salinity between sites at Mngazana Estuary.

	Site 1	**Site 2**	**Site 3**	**Site 4**	**Site 5**	**Site 6**	**Site 7**	**Site 8**
Summer 2014 vs Summer 2015	F=3.2, p<0.05, df=4	F=2.9, p<0.05, df= 4		F= 6.7, p<0.05, df=4		F=15.5, p<0.05, df=4	NS	NS
Winter 2014 vs Winter 2015	F=5.4,p<0.05, df=4	NS	NS	NS	NS	NS	NS	NS
Summer 2014 vs Winter 2014	NS	F=4.7 ,p<0.05, df=4	F=2.9, p< 0.05, df=4	F=4.7, p<0.05, df=4	F= 3.2, p< 0.05, df=4	F=17.0,p<0.05, df=4	F= 5.0 , p< 0.05, df=4	NS
Summer 2014 vs Winter 2015	F= 3.4, p< 0.05, df=4	F= 4.7, p<0.05, df= 4	F=3.2, p< 0.05, df=4	F= 4.7, p<0.05, df=4	F= 10.6, p< 0.05, df=4	F= 5.3, p<0.05, df=4	F= 22.0 , p< 0.05, df=4	F= 12.9, p< 0.05, df =4
Summer 2015 vs Winter 2014	NS	NS	NS	F=7.1, p<0.05, df=4	NS	F=5.0, p<0.05, df=4	NS	NS
Summer 2015 vs Winter 2015	NS	F=4.22, p<0.05, df=4	NS	NS	F= 6.7, p< 0.05, df= 4	NS	F= 7.0, p< 0.05, df=14	F=6.0, p< 0.05, df=4

NS: No significant difference was found

Table A2: Summary of statistical analyses for seasonal changes in porewater temperature between sites at Mngazana Estuary.

	Site 1	**Site 2**	**Site 3**	**Site 4**	**Site 5**	**Site 6**	**Site 7**	**Site 8**
Summer 2014 vs Summer 2015	NS	F=5.7, p<0.05, df=4	NS	NS	NS	F=14.0,p<0.05, df=4	F=10.0, p<0.05, df=4	NS
Winter 2014 vs Winter 2015	NS	F=8.0, p<0.05, df=4	F= 4.0, p<0.05, df=4	NS	NS	NS	NS	NS
Summer 2014 vs Winter 2014	F=8.0, p<0.05, df=4	F=14.1,p<0.05, df=4	F=8.7, p<0.05, df=4	F=11.5,p<0.05, df=4	F= 10.0, p<0.05, df=4	F=23, p<0.05, df=4	F=3.6, p<0.05, df=4	F=14.8,p<0.05, df=4
Summer 2014 vs Winter 2015	F=12.1, p<0.05, df=4	F=28.0, p<0.05, df=4	F=9.5, p< 0.05, df=4	F=11.5,p<0.05, df=4	F=7.3, p<0.05, df=4	F=13.0,p<0.05, df=4	F=8.9, p<0.05, df=4	F=9.5,p<0.05, df=4
Summer 2015 vs Winter 2014	F=7.0, p<0.05, df=4	F=8.5, p<0.05, df=4	F=10.0, p<0.05, df=4	F=10.0,p<0.05, df=4	F=10.6, p<0.05, df=4	NS	NS	F=2.9, p<0.05, df=4
Summer 2015 vs Winter 2015	F=10.4, p<0.05, df=4	F=20.0, p<0.05, df=4	F=9.9, p<0.05,df=4	F=10.0,p<0.05, df=4	F=6.5,p<0.05, df=4	F=6.9, p<0.05, df=4	F=5.0,p<0.05, df=4	NS

NS: No significant difference was found

Table A3: Summary of statistical analyses for seasonal changes in electrical conductivity between sites at Mngazana Estuary.

	Site 1	**Site 2**	**Site 3**	**Site 4**	**Site 5**	**Site 6**	**Site 7**	**Site 8**
Summer 2014 vs Summer 2015	F=3.3, p<0.05, df=4	F=2.9, p<0.05, df=4	F=3.7, p< 0.05, df=4	F=6.6, p<0.05, df=4	NS	F=11.7, p<0.05,df=4	NS	F=6.3, p<0.05, df=4
Winter 2014 vs Winter 2015	NS	F=4.1, p<0.05, df=4	NS	NS	F=4.2, p<0.05, df=4	F=5.0, p<0.05, df=4	NS	F=3.0, p<0.05, df=4
Summer 2014 vs Winter 2014	NS	F=3.8, p<0.05, df=4	F=3.0, p<0.05, df=4	F=4.3, p<0.05, df=4	F=17.0, p<0.05, df=4	NS	NS	NS

Summer 2014 vs Winter 2015	F=2.9, p<0.05, df=4	F=5.0, p<0.05, df=4	F=3.6, p<0.05, df=4	F=4.5, p<0.05, df=4	F=23.0, p<0.05, df=4	F=5.4, p<0.05, df=4	F=28.0, p<0.05, df=4	F=12.6, p<0.05, df=4
Summer 2015 vs Winter 2014	NS	NS	NS	F=5.3, p<0.05, df=4	F=5.6, p<0.05, df=4	F=15.6, p<0.05, df=4	NS	NS
Summer 2015 vs Winter 2015	NS	F=4.2, p<0.05, df=4	NS	NS	F=7.2,p<0.05, df=4	NS	F=10.0, p<0.05,df=4	F=14.1, p<0.05, df=4

NS: No significant difference was found

Table A4: Summary of statistical analyses for seasonal changes in porewater temperature between sites at Nahoon Estuary.

	Site 1	**Site 2**	**Site 3**	**Site 4**
Winter 2014 vs Winter 2015	NS	F=4.0, p<0.05, df=4	F=4.2, p<0.05,df=4	NS
Summer 2014 vs Winter 2014	F= 7.6, p< 0.05, df=4	F=4.1, p<0.05, df=4	F=7.0, p<0.05, df=4	F=5.0, p<0.05,df=4
Summer 2014 vs Winter 2015	F=16.3, p< 0.05, df=4	F=25.0, p<0.05,df=4	F=10.0, p<0.05,df=4	F=6.4, p<0.05,df=4
Summer 2015 vs Winter 2014	F=6.1, p<0.05, df=4	F=3.2, p<0.05,df=4	F=7.0, p<0.05,df=4	F=5.0, p<0.05,df=4
Summer 2015 vs Winter 2015	F=8.9, p< 0.05, df=4	F=22.0, p<0.05,df=4	F=10.0, p<0.05,df=4	F=6.4, p<0.05,df=4

NS: No significant difference was found

Table A5: Summary of statistical analyses for seasonal changes in porewater conductivity between sites at Nahoon Estuary.

	Site 1	**Site 2**	**Site 3**	**Site 4**
Winter 2014 vs Winter 2015	F=7.4, p<0.05,df=4	F=3.8, p<0.05,df=4	F=9.0, p<0.05,df=4	F=8.8, p<0.05,df=4

Summer 2014 vs Winter 2014	F=5.2, p<0.05,df=4	NS	NS	NS
Summer 2014 vs Winter 2015	F=3.0, p<0.05,df=4	NS	F=3.5, p<0.05,df=4	NS
Summer 2015 vs Winter 2014	F=3.2, p<0.05,df=4	NS	NS	NS
Summer 2015 vs Winter 2015	F=4.2, p<0.05,df=4	NS	F=5.5, p<0.05,df=4	NS

NS: No significant difference was found

Table A6: Summary of statistical analyses for seasonal changes of ammonium between sites at Mngazana Estuary.

	Site 2	**Site 5**	**Site 6**
Summer 2014 vs Summer 2015	NS	F=2.8, p<0.05, df=4	NS
Winter 2014 vs Winter 2015	NS	F=3.7, p<0.05, df=4	NS
Summer 2014 vs Winter 2014	NS	F=2.9, p<0.05, df=4	F=3.2, p<0.05, df=4
Summer 2014 vs Winter 2015	NS	F=3.8,p<0.05,df=4	F=4.4, p<0.05, df=4
Summer 2015 vs Winter 2014	F= 3.2, p<0.05, df=4	NS	NS

NS: No significant difference was found

Table A7: Summary of statistical analyses for seasonal changes in nitrates and nitrites between sites at Mngazana Estuary.

	Site 1	**Site 2**	**Site 3**	**Site 4**	**Site 5**	**Site 6**	**Site 7**	**Site 8**

Summer 2014 vs Summer 2015	F=20.9, p<0.05, df=4	F=14.9,p< 0.05, df=4	F=9.2,p< 0.05, df=4	F=7.3, p<0.05, df=4	F=7.9, p<0.05, df=4	F=5.1, p<0.05, df=4	F=19.6, p<0.05, df=4	F=22.9, p<0.05, df=4
Summer 2014 vs Winter 2014	F=15.3, p<0.05, df=4	F=16.6,p<0.05, df=4	F=10.2, p<0.05, df=4	F=6.6, p<0.05, df=4	F=4.5,p<0.05, df-4	F=5.6, p<0.05, df=4	F=8.7, p<0.05, df=4	F=29.8, p<0.05, df=4
Summer 2014 vs Winter 2015	F=19.9, p<0.05, df=4	F=13.9,p<0.05, df=4	F=7.8, p<0.05, df=4	F=8.8, p<0.05, df=4	F=7.2,p<0.05, df=4	F=5.4, p<0.05, df=4	F=17.8, p<0.05, df=4	F=14.9, p<0.05, df=4

NS: No significant difference was found

Table A8: Summary of statistical analyses for seasonal changes in phosphorus between sites at Mngazana Estuary.

	Site 2	**Site 4**	**Site 5**	**Site 6**
Summer 2014 vs Summer 2015	F=3.9, p<0.05, df=4	F=4.4, p<0.05, df=4	F=20.5, p<0.05, df=4	F= 8.1, p<0.05, df=4
Summer 2014 vs Winter 2014	F=4.2, p<0.05, df=4	F=3.9, p<0.05, df=4	F=5.7, p<0.05, df=4	F=7.4, p<0.05, df=4
Summer 2014 vs Winter 2015	F=3.5, p<0.05, df=4	F=3.1, p<0.05, df=4	NS	F=8.4, p<0.05, df=4
Summer 2015 vs Winter 2015	NS	NS	F=3.4, p<0.05, df=4	NS

NS: No significant difference was found