CLIMATE CHANGE AND SOUTH AFRICA'S BLUE CARBON ECOSYSTEMS

Report to the

WATER RESEARCH COMMISSION

by

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EXECUTIVE SUMMARY

BACKGROUND AND RATIONALE

Blue carbon habitats include living and non-living biomass of mangroves, salt marshes and seagrasses (Howard et al., 2014). These habitats form the interface between land and sea, and provide numerous ecosystem services such as coastal protection, fish nursery habitats, nutrient filters and carbon storage. The term ‘blue carbon’ refers to the carbon sequestered and stored by these systems. In South Africa, blue carbon habitats occur in sheltered estuarine environments where their integrity and biodiversity are threatened by increasing freshwater abstraction, development pressures, poor water quality, climate change and sea level rise. The loss of estuarine ecosystems reduces their capacity to act as carbon sinks and has the potential to release large quantities of carbon into the atmosphere as CO_2. and

Carbon emissions are recognized by the Inter-Governmental Panel on Climate Change (IPCC) and the UN Framework Convention on Climate Change (UNFCCC) as significant sources of greenhouse gases (GHG). Global CO_2 emissions from the degradation and destruction of blue carbon ecosystems are estimated at 45 billion metric tons annually, with an associated economic cost approaching $20 billion each year (Pendleton et al., 2012). Consequently, protecting these ecosystems has become a commonly accepted way of paying for various ecosystem services. The Paris Agreement within the UNFCCC allows for the mitigation and financing of GHG emissions and has included an offset mechanism that secures their sequestration through protecting blue carbon ecosystems and paying for ecosystem services. Coastal wetland conservation needs to be linked to carbon markets and payments that benefit local communities, such as in Kenya, Madagascar and Indonesia (Wylie et al., 2016). Our research will provide input to South Africa’s climate policies, incentivising the need to protect and restore coastal wetlands and estuaries.

Effective management and conservation of coastal wetlands is a critical priority, especially in areas where people’s livelihoods depend on these ecosystems. Apart from the immediate effects of anthropogenic activities, coastal wetlands also face threats related to global climate change. The rate of sea-level rise will determine the persistence of these ecosystems. Past research efforts have focused on understanding the response of estuaries to changes in freshwater inflow. Currently, there is an urgent need to determine their response to development and rising sea-levels to prevent habitat loss through “coastal squeeze”. An assessment of the vulnerability of blue carbon ecosystems to sea-level rise needs to be conducted and if necessary, mitigation plans developed.

The southern range limit for east African mangroves on the South African coastline provides a unique opportunity to study their response to climate change. Globally, mangroves are expanding into warm-temperate salt marshes in response to rising temperatures. This can significantly impact carbon storage and other ecosystem services. Recent research has identified distinct natural and anthropogenic factors that influence the resilience and stability of mangrove ecosystems. It is therefore necessary to investigate whether mangroves can potentially expand their habitat range and how these communities and salt marshes respond to climate change.

This study generates knowledge on blue carbon stored by South Africa's estuarine habitats. It quantifies the extent and loss of these habitats and their ecosystem services and describes responses to climate change. It presents the trends of surface elevation in two salt marsh and two mangrove systems and discusses the long-term survival and expansion of mangrove forests and surrounding salt marsh areas.

The research contributed to the National Biodiversity Assessment of 2018 and included information on the distribution and health of estuarine habitats. It also informed the compilation of estuary management plans required by the Integrated Coastal Management Act. Habitat loss was related to the corresponding loss of ecosystem services such as carbon storage, coastal protection, nutrient filtration and fish nursery function. Since healthy estuaries ensure ecosystem resilience and enable adaptation to global change, those with intact, undisturbed habitat need to be identified and prioritized for biodiversity conservation.

To support adaptation to a changing climate, sites need to be identified for future coastal wetland expansion. Superimposed on rising sea levels, estuaries experience significant development pressure and loss of estuarine habitat due to "coastal squeeze". Mangrove species are particularly vulnerable to local extinction and it is therefore essential to monitor and report on these ecosystems.

The study contributes to a global understanding of mangrove, salt marsh and seagrass carbon storage and sequestration. Our mangrove ecosystems occur at a southerly limit and the rate and volume of carbon sequestration may be reduced compared to more tropical systems. The work is a first step to leveraging South Africa's estuaries as part of a potential carbon credit trading scheme. Coastal wetland protection and carbon sequestration can be included in payment for ecosystem services.

OBJECTIVES AND AIMS

The aims of this project were described in the original proposal document as follows:

a) To determine the extent of blue carbon ecosystems in South Africa and estimate blue carbon storage using the IPCC assessment methods.
b) To quantify the loss of blue carbon habitats and associated ecosystem services.
c) To predict the responses of blue carbon ecosystems to climate change in the form of sea-level rise and increased global temperatures.

The following tasks were developed to achieve the project aims:

- Assess the extent of blue carbon habitats and their associated ecosystem services in South Africa. Quantify changes in blue carbon habitats and ecosystem services over time.
- Directly quantify blue carbon storage in mangrove, salt marsh, and seagrass habitats at a representative study site. Compare carbon storage between mangrove and salt marsh habitats at the mangrove distributional range limit.
- Measure surface elevation change in mangrove and salt marsh habitats and relate this response to sea-level rise threats.
- Predict changes in mangrove distribution along the South African coastline in response to rising temperatures and changes in precipitation regimes associated with climate change. Review the implications of climate change on salt marsh along the South African coastline.
- Determine the viability of a carbon offset mechanism for South Africa’s blue carbon ecosystems.

METHODOLOGY

TASK 1 ASSESS THE EXTENT OF BLUE CARBON ECOSYSTEMS IN SOUTH AFRICA

The estuary botanical database was updated to provide data on the area covered in each estuary of South Africa for seagrasses, salt marsh and mangroves. Ecosystem services were linked to the available habitats. Visible changes in estuary habitats were identified from aerial photographs and from Google Earth and mapped. The changes were associated with development, roads, housing, grassed areas, grazing and agriculture. The type of habitat lost (e.g. salt marsh, mangroves, and reeds/sedges) was determined by comparing historical aerial photographs and literature as well as physical features such as surrounding habitat, elevation, biogeographic zone and estuary type.

Blue carbon habitats were digitized for 115 estuaries using ESRI™ ArcMap 10.1 (2012) from orthorectified aerial photographs obtained from the Chief Directorate: National Geo-spatial Information (CD:NGI). These images have a 50 cm spatial resolution. The earliest images dated back to 1934/1937. Past macrophyte cover (salt marsh, mangrove, seagrass, reeds & sedges) thus represents the situation in the 1930s and in 2018. Macrophyte cover in each image was mapped and the difference in area between images taken as habitat loss or gain. Recent Google Earth images and field work were used to update macrophyte cover following a similar approach to that of Fernandes and Adams (2016). Arcpad 10.1 loaded on Trimble Juno GPS was used to map the distribution of macrophytes in the field. GIS vegetation maps are now available for approximately 40% of the estuaries in the country.

TASK 2 ESTIMATE BLUE CARBON STORAGE USING THE IPCC ASSESSMENT METHODS

Tier 1 Assessment

Available area cover data were used to make a Tier 1 assessment. Globally averaged estimates were used to quantify carbon stocks in a given area. The global averages for carbon stocks (to 1 m depth) were 386 Mg/ha for mangroves, 255 Mg/ha for salt marshes and 108 Mg/ha for seagrasses. The carbon stock was determined by multiplying the area of the ecosystem by the mean estimated carbon stock for that ecosystem type (IPCC, 2013).

Tier 2 Assessment

In situ measurements were made at Nxaxo Estuary to quantify carbon sequestration for a Tier 2 assessment. Nxaxo is a permanently open estuary located at Wavecrest near Butterworth, Eastern Cape (32°35'S; 28° 31' E). This site was selected as the estuary has an equal area of mangrove and salt marsh as well as patches of seagrass (*Zostera capensis*). The focus species for this study were the white mangrove *Avicennia marina,* salt marsh succulent *Sarcocornia tegetaria* and seagrass *Zostera capensis*.

The carbon pools assessed included living above-ground biomass (woody plant mass of mangroves, herbaceous plant mass, grasses for seagrasses and salt marshes), below-ground carbon (soil organic matter), and dead above-ground biomass (litter or wood). Sediment characteristics, such as particle size and moisture content, were also measured.

<u>TASK 3 QUANTIFY THE LOSS OF BLUE CARBON HABITATS AND ASSOCIATED ECOSYSTEM SERVICES</u>

Available data on changes over time in areas covered by seagrasses, salt marshes and mangroves were collated. Detailed mapping was completed to assess changes in blue carbon habitats. Ecosystem services associated with these habitats were identified and loss of services quantified for all estuaries along the coastline.

Habitats were mapped in GIS and changes over time assessed using imagery provided by Google Earth, Department of Water Affairs (2006) aerial survey, Department of Environmental Affairs (2008) coastal survey (5 m x 5 m resolution), and historical coastal photographs archived in the SAEON Elwandle Estuarine Database. GIS maps were collated in a single database now available on the SANBI website.

<u>TASK 4 PREDICT THE RESPONSE OF BLUE CARBON HABITATS TO CLIMATE CHANGE</u>

A qualitative assessment was made of salt marsh response to climate change, and detailed modelling studies were completed for mangrove habitats. Structural equation modelling was used to determine the importance of environmental factors influencing mangrove distribution patterns. This graph-theoretic approach was used to understand and delineate causative relationships between multiple variables. The high energy, wave-dominated coast restricts mangroves to sheltered estuarine areas creating a discontinuous distribution. This study provides the first quantitative assessment of factors influencing current mangrove distribution in this region.

The potential for mangrove expansion in response to global climate change was investigated using a species distribution modelling approach in the open-source software MaxEnt. This approach was based on ecological niche models that allow occurrence locations to be used alongside environmental variables to identify suitable habitats. Distributions can be determined for new locations or under new environmental conditions. This study identified potentially suitable estuaries for mangroves beyond the current coastal distribution. It investigated whether estuaries that currently support mangroves could become unsuitable under predicted climate change scenarios and whether estuaries beyond the current distribution of mangroves could become suitable under predicted climate change scenarios.

The study reviewed and synthesized existing scientific information to provide an up-to-date understanding of the patterns and processes influencing salt marshes to enable an accurate prediction of their responses to climate change. Firstly, an assessment was made of available information on distributional patterns and importance in terms of ecosystem services. Secondly, a present state assessment of salt marsh was conducted and changes in area cover described in relation to threats and pressures. Expected responses to climate change and implications for ecosystem services were summarised, and a framework was provided for understanding climate change responses in open and closed estuaries.

TASK 5 RESPONSE OF BLUE CARBON ECOSYSTEMS TO SEA LEVEL RISE

Seasonal changes in surface elevation were investigated for salt marsh and mangrove habitats at the Knysna, Nahoon and Nxaxo estuaries. Sites were selected from long-term monitoring data on salt marshes and mangroves. They were also chosen based on ease of access to facilitate a long-term monitoring programme aimed at identifying responses to sea level rise. Annual measurements are needed to understand the dynamic changes in erosion and accretion.

Surface Elevation Table (RSET) devices were set up (Cahoon et al., 2002; Lynch et al., 2015) following the globally standardized method for measuring surface elevation trends relative to sea-level rise; this work was completed in collaboration with the South African Environmental Observation Network (SAEON). A total of 18 new RSET benchmarks were deployed in the Nahoon and Nxaxo estuaries (9 benchmarks per estuary) as trends are known to be spatially variable. At the Knysna Estuary, seven benchmarks originally installed by SAEON in 2009 (Schmidt, 2013) were re-visited to assess surface elevation change over the past decade. These measurements could be related to sea-level rise measured at the nearest tide gauge. The surface elevation dynamics of the salt marshes in the Knysna Estuary were also used to inform a predictive model (Sea-Level Affecting Marshes Model; Clough et al., 2016). This assessed the threat of ‘coastal squeeze’ to salt marshes at Thesen’s Island. These RSET stations will be maintained in future by SAEON in collaboration with Nelson Mandela University.

TASK 6 DETERMINE THE VIABILITY OF A CARBON OFFSET MECHANISM FOR SOUTH AFRICA'S BLUE CARBON ECOSYSTEMS

To determine the viability of blue carbon or the economic benefit of intact mangroves, salt marsh and seagrass habitats, we used a Net Present Value (NPV) analysis following the methods of Pendleton et al. (2014), Thompson et al. (2014), Chang et al. (2015) and UNEP (2016). An NPV determines the revenue (BC financial value) generated by blue carbon schemes over a certain period. This should be greater than the cost of establishing and managing the project by either protection or restoration. The NPV analysis factored in the price of carbon (at different market prices), carbon sequestration rates, avoided CO_2 emissions and the establishment, opportunity and management costs at two different discount rates. Carbon sequestration rates were calculated from carbon density and surface elevation measurements at Nxaxo, Nahoon, Swartkops and Knysna estuaries while emission reductions were determined from quantified carbon stocks. A blue carbon offset project is considered viable if the net benefit of conservation (expressed as the blue financial value) is larger than the sum of management/protection and opportunity cost in alternative use over the next 20-year horizon (UNEP, 2016).

RESULTS AND DISCUSSION

Details are provided for each of the tasks covering the three different research aims;

1. To determine the extent of blue carbon ecosystems in South Africa and estimate blue carbon storage using the IPCC assessment methods.
2. To quantify the loss of blue carbon habitats and associated ecosystem services.
3. To predict the responses of blue carbon ecosystems to climate change in the form of sea-level rise and increased global temperatures.

Aim 1: To determine the extent of blue carbon ecosystems in South Africa and estimate blue carbon storage

TASK 1 ASSESS THE EXTENT OF BLUE CARBON ECOSYSTEMS IN SOUTH AFRICA

The estuary botanical database was updated to provide data on the area covered in each estuary of South Africa for seagrasses, salt marsh and mangroves. Ecosystem services were linked to the available habitats. To date 122 estuaries have GIS vegetation maps available. The geodatabase containing these shapefiles are available from the SANBI website or on request from the Botany Department, Nelson Mandela University. The geodatabase contains the distribution and area covered by the different macrophyte habitats in South African estuaries. The associated metadata provides necessary details pertaining to the method followed. Estuary habitat covers a total area of 103 500 ha. Reeds and sedges (17 530 ha) were the dominant habitat type overall. Supratidal salt marsh was dominant in the cool temperate region (6 300 ha) and warm temperate region (2 400 ha) and reeds and sedges in the subtropical region (10 800 ha). Mangroves and swamp forest are restricted to the subtropical region of South Africa, with a few estuaries in the warm temperate supporting habitat. Present mangrove extent is 1673 ha recorded for 31 estuaries in the country. uMhlathuze Estuary supports the largest mangrove habitat (793 ha), with the second highest area occurring at St Lucia Estuary (288 ha). Only 20% (70 estuaries) of estuaries supported submerged aquatic vegetation as these species are sensitive to changes to water level, turbidity, nutrients and salinity. Estuarine lakes provide the most suitable conditions for establishment and the largest areas are found in this scarce estuary type. These results are reported on in the National Biodiversity Assessment.

TASK 2 ESTIMATE BLUE CARBON STORAGE USING THE IPCC ASSESSMENT METHODS

Tier 1 Assessment

Available area cover data derived from the updated estuarine botanical database and the global average estimates for carbon reported in Howard et al. (2014) were used in this assessment. The area cover for mangroves, salt marshes and seagrasses per estuary (ha) was multiplied by the global average estimates for carbon stocks (to 1 m depth), which were 386 MgC ha^{-1} for mangroves, 255 MgC ha^{-1} for salt marshes and 108 MgC ha^{-1} for seagrasses. This estimated the carbon stock (MgC) for those estuaries supporting the above-mentioned habitats.

Subtropical estuaries with the largest mangrove areas, such as uMhlathuze and St Lucia, had the largest sediment carbon stores at 251710.6 MgC and 80867 MgC respectively. In contrast, estuaries such as Bulungula and Mzimvubu stored less carbon (from 5.404 MgC to 115.8 MgC). For salt marsh ecosystems, the Langebaan and Groot Berg estuaries had the highest carbon storage (335427 MgC and 1074060 MgC respectively). Seagrass carbon storage was highest at St Lucia (46602 MgC) and Kosi (70416 MgC) estuaries and lowest at uMlalazi (0.108 MgC) and Klein Palmiet (2.16 MgC) estuaries.

Tier 2 Assessment

Results for the Nxaxo Estuary showed that sediment organic carbon was significantly higher in mangroves (228.08 ± 27.99 MgC ha^{-1}) compared to salt marshes (2.61 ± 0.19 MgC ha^{-1}) and seagrasses (1.67 ± 0.81 MgC ha^{-1}). However, these values were less than the reported global means for blue carbon because of slower growth and productivity at latitudinal limits for mangroves, a drier climate and coarse-grained sediment. Even though carbon storage was limited, this study provided baseline data for South Africa's estuaries to partake in potential carbon credit trading schemes which will aid with the management and protection of blue carbon habitats in future.

There is significant spatial variability in carbon storage as shown by the measurements for five *Avicennia marina* sites at Nxaxo Estuary. The sediment carbon pool made the largest contribution to total carbon storage at each site and ranged from 176.91 ± 4.5 MgC ha^{-1} to 262.53 ± 18.8 MgC ha^{-1}. Across all sites, average carbon storage for all pools (above-ground biomass and sediment) was 234.9 ± 39.16 MgC ha^{-1}, which falls within the range reported for mangroves at other southern hemisphere range limits.

Results for the Nxaxo Estuary were reported in the chapter (Blue carbon storage comparing mangroves with salt marsh and seagrass habitats at a warm temperate continental limit) and article in the journal Estuarine, Coastal and Shelf Science (Quantification of blue carbon storage in a warm temperate estuary). In addition, carbon storage at the Nahoon Estuary was described in an article published in South African Journal of Botany "A comparison of soil carbon pools across a mangrove-salt marsh ecotone at the southern African warm-temperate range limit. An increase in sediment carbon has been found at some range limits where mangroves are expanding along ecotones into salt marsh; mangroves store more carbon per unit area. However, this study did not support this finding as the mangrove habitat was limited in extent and was encroaching slowly. Twenty-seven sediment cores (0.5 m in depth) were taken along the mangrove-salt marsh ecotone to compare sediment characteristics. Sediment carbon density was highest at 30-50 cm below the surface. The average sediment carbon to 0.5 m depth was similar in mangrove (110.14 MgC ha^{-1}), ecotone (114.50 MgC ha^{-1}), and salt marsh habitats (109.62 MgC ha^{-1}). Similarities in sediment carbon content indicate that longer time scales are required for mangrove carbon to accumulate in this region.

Although blue carbon studies have become increasingly topical, few studies have comprehensively assessed carbon storage in all three habitats. The findings of our study contribute towards the global understanding of blue carbon storage in warm temperate habitats and mangroves at their distributional limits. Future studies should expand to other estuaries in South Africa for a comparative analysis and to create a more robust blue carbon inventory.

Aim 2: To quantify the loss of blue carbon habitats and associated ecosystem services

TASK 3 QUANTIFY THE LOSS OF BLUE CARBON HABITATS AND ASSOCIATED ECOSYSTEM SERVICES

Available data were collated on changes over time in the area covered by seagrasses, salt marshes and mangroves. Detailed mapping was completed to assess changes in blue carbon habitats over time. Ecosystem services associated with these habitats were identified and loss of ecosystem services quantified for all estuaries.

Habitats were mapped in GIS and changes over time assessed using imagery such as Google Earth, DWA 2006 aerial survey, DEA 2008 coastal survey (5 m x 5 m resolution), and the historical coastal photographs archived in the SAEON Elwandle Estuarine Database. GIS maps were collated in a single database available on the SANBI website.

Although ecosystem properties are a single aspect of the ecosystem services cascade, they are generally the most commonly used since there is a lack of data on direct measurements of ecosystem processes and functions. This study found that blue carbon habitats in South African estuaries have experienced cumulative loss in the potential to provide services and there is a need for restoration to regain the potential to provide them in future. Mangroves in the subtropical region have had the greatest losses in potential to supply ES attributed to habitat loss, artificial mouth breaching and impacts from agricultural practices.

Aim 3: To predict the responses of blue carbon ecosystems to climate change in the form of sea-level rise and increased global temperatures

TASK 4 PREDICT THE RESPONSE OF BLUE CARBON HABITATS TO CLIMATE CHANGE

A qualitative assessment of salt marsh responses to climate change was made and detailed modelling studies completed for mangrove habitats. This is reported in three sub-chapters/scientific articles: 1) Drivers of mangrove distribution at a high-energy, wave-dominated, southern distribution limit, 2) Potential for mangrove range expansion in response to global climate change, and 3) Salt marsh at the tip of Africa: patterns, processes and changes in ecosystem services in response to climate change.

Structural Equation Modelling showed that floodplain area, mean annual runoff, daily flushing rate, and a permanently open estuary mouth were significant predictors of mangrove area. Floodplain area was the strongest predictor, and this was evident as the largest mangrove forests in South Africa occur in the few coastal plain estuaries and embayments along the KwaZulu-Natal coast. Both geomorphology and physical characteristics of estuaries influence mangrove occurrence. Limited accommodation space in conjunction with estuary mouth stability and connection to the marine environment determine the estuaries that can support persistent mangrove habitats.

Mangrove expansion along the east coast in response to global climate change was investigated using species distribution models and MaxEnt software. Distributions can be determined for new locations or under new environmental conditions. This study identified three potentially suitable estuaries (Gqunube, Qora, uMzimkhulu) for mangroves beyond the current distribution. Only one *A. marina* individual has been recorded in the Gqunube Estuary, and mouth restriction at the uMzimkhulu Estuary limits suitable saline intertidal habitats. The research identified a new mangrove location for the Qora Estuary. Google Earth satellite imagery showed the presence of mangroves not previously recorded. An additional model predicted high habitat suitability for mangroves at the Keiskamma Estuary located ~10 km south of the Tyolomnqa Estuary. This estuary has large salt marsh areas where mangroves could establish. Models 3 and 4 showed the potential habitat suitability for mangroves in 2050 under the two different IPCC scenarios for the Bushmans, Kowie, Great Fish and Keiskamma estuaries located further south than the current distribution limit. These predictions need to be verified with mangrove propagule dispersion and recruitment models.

A conceptual understanding of climate change responses of salt marsh in open and closed estuaries was developed and changes in terms of implications for ecosystem services described. Salt marshes occur in sheltered estuaries distributed along the ~3,000 km South African coastline. Supratidal salt marsh occurs at elevations greater than 1.5 m amsl (above mean sea-level) and are dominant in the cool temperate (5328 ha) and intertidal salt marsh in the warm temperate region (2093 ha). Although relatively small in total extent (14 955 ha), salt marshes play a central role in biodiversity conservation because they provide critical habitat for migratory fish and birds. Approximately 43% of salt marsh habitat has been lost due to encroaching development and agriculture. In addition, salinization and desiccation resulting from upstream freshwater abstraction reduces freshwater inflow, which extends periods of mouth closure in temporarily closed estuaries causing salt marsh inundation and flooding. Plant ecophysiological studies have informed our future predictions and shown that lower intertidal salt marsh species can survive conditions typical of upper intertidal ranges. However, the reverse is not true of upper intertidal species sensitive to waterlogging that will occur in response to an increase in sea level. Predicting the combined effects of multiple stressors (increased storm surges, floods, droughts and reduced river flow) is critical to conserve these important habitats. Research and monitoring to understand salt marsh responses is ongoing because the interface between the subtropical and warm temperate coastal regions is expected to be significantly affected by expected future climate change. This study is globally relevant as little is known about southern hemisphere salt marshes in Africa and data are needed for comparative purposes.

TASK 5 RESPONSE OF BLUE CARBON ECOSYSTEMS TO SEA LEVEL RISE

This task was expanded to include salt marsh responses to sea level rise. Seasonal changes in surface elevation were investigated for salt marsh and mangrove habitats at the Knysna, Nahoon and Nxaxo estuaries. Surface Elevation Table (RSET) devices were set up at the Nahoon and Nxaxo estuaries. This research provides a first report on surface elevation change in these estuaries. Salt marsh surface elevation changes and sea level rise were investigated at the Knysna Estuary. Here RSET sites were established in the lower intertidal *Spartina maritima* zone. Results indicated the influence of migratory estuary channels, and sedimentation patterns were related to surface elevation change. Site-specific data are needed for interpretation of changes. For example, Site NH9 at Nahoon Estuary is adjacent to a walkway and human disturbance.

At both the Nahoon and Nxaxo estuaries, the largest increase in surface elevation height over time was measured at the mangrove RSETs. This could be related to local hydrodynamics and the deposition of inorganic and organic material. Mangrove roots and salt marsh plants trap material which increases surface elevation. Surface elevation loss was recorded in the salt marsh sites at the Nxaxo Estuary. This was related to cattle disturbance and trampling that compacts the sediment.

The Knysna study is important as it is the first report in South Africa integrating salt marsh surface elevation dynamics, predicted responses to sea-level rise, and threats from coastal development. This estuary is ranked the highest nationally in terms of estuarine biodiversity supported by salt marsh habitats. The study showed that 60.4 ha of salt marsh would be lost due to sea level rise and coastal development. Management interventions are needed to prevent further development and make suitable areas available for landward migration of salt marsh.

TASK 6 DETERMINE THE VIABILITY OF A CARBON OFFSET MECHANISM FOR SOUTH AFRICA'S BLUE CARBON ECOSYSTEMS

Carbon markets are founded on the concept of producing carbon credits through ensuring avoidance of greenhouse gas (GHG) emissions or atmospheric removal of these gases through the action of a project. Carbon credits can be resold or used to offset carbon dioxide (CO_2) emissions (Wylie et al., 2016). Acceptable offsets must show 'additionality' demonstrating that reduced carbon emissions (either through ecosystem protection or restoration) would not be possible without the funds generated by selling carbon credits. Our study estimated that blue carbon habitats in South Africa can potentially remove 10.3 million $tCO_2eq\ yr^{-1}$, an amount higher than the projected sequestration potential from terrestrial habitats (8 million $tCO_2eq\ yr^{-1}$ reported in the National Terrestrial Carbon Sink Assessment (DEA, 2014). Salt marsh habitats made the highest contribution to this value. Our value was comparable to the emissions reduction potential of blue carbon habitats in southeast Australia, reported to be 9.63 million $tCO_2eq\ yr^{-1}$ (Lewis et al., 2018). The value of blue carbon habitats was estimated as R1.2-R10.6 billion per year when carbon is traded at a high price and ~R120-R150 million per year when carbon is traded at lower carbon prices.

Based on our results, we suggest that blue carbon Payment for Ecosystem Services (PES) or offset projects will only be economically viable in South Africa, if the credits are traded at a higher price in the voluntary markets which at present are the more attainable options for blue carbon pilot projects (Ullman et al., 2013). Additional ecosystem service values should be added in this analysis in future, such as the value of mangrove coastal storm protection and other regulatory services which are still understudied in South Africa (Turpie et al., 2017).

CONCLUSIONS AND RESEARCH IMPACT

Our research is the first in the country to quantify carbon capture and storage by blue carbon habitats. These cover < 2% of oceans worldwide but capture up to 70% of carbon (~770 Gt). Our study provides baseline data for potential carbon credit trading schemes as a strategy to reduce greenhouse gas emissions globally. It demonstrates South Africa's commitment to the Paris Agreement signed in April 2016 to limit the increase in global average temperature this century to less than 2°C. Our preliminary data were included in a recent *Nature* publication (Rogers et al., 2019) highlighting their relevance and international significance.

At a national level, our research outcomes have been included in the country's National Biodiversity Assessment that set essential baselines against which future environmental change can be measured. The outcomes also show the country's commitment to the 2015 UN Sustainable Development Goals related specifically to Climate Action, Life below Water, and Quality Education, goals that aim to end poverty, protect the planet and work towards universal peace and prosperity. The pursuit of active partnerships between academics, government and civil society, a key element in our research, builds capacity by sharing relevant, useful and reliable knowledge in support of prudent policy formulation.

This research project has endeavoured to achieve innovation through the application of existing techniques and protocols to address new research questions and generate knowledge in the following manner:

- First application of the Blue Carbon Initiative protocol for quantifying carbon storage in mangroves, salt marshes and seagrasses in South Africa.
- Development of an Ecosystem Services Index for South African estuaries.
- Establishment of the first mangrove RSET network to measure surface elevation change and responses to sea-level rise on the African continent.
- First quantitative assessment of mangrove distribution to incorporate geomorphological features and using the structural equation modelling approach.
- Development of a predictive model for mangrove expansion under climate change in South Africa that is not only dependent on climatic variables.

There has been important human capital development in the water and science sectors as the primary researchers were Ms Sinegugu Mbense (PhD candidate) and Dr Jacqueline Raw (post-doctoral fellow). They benefitted from the multi-disciplinary input that a WRC project can provide as well as gained valuable skills in the research management process. In addition, an MSc student, Ms Jamie Johnson, completed her study on this project. Three Honours students were also involved in the research programme as well as interns from SAEON and Nelson Mandela University. Joint field trips for mangrove and salt marsh elevation measurements developed capacity and knowledge sharing among researchers from these institutes.

The project has resulted in the publication of eight scientific articles (three published, four accepted for publication, and one under review) and two book chapters. Dissemination of knowledge also took place through ~10 conference presentations over the three-year period.

The ecological importance of mangroves, salt marshes and seagrasses in South Africa has been acknowledged since the earliest estuarine surveys along the coastline. Primarily, these vegetation types have been recognized for providing structural habitat to characteristic estuarine species of birds, fish, and invertebrates. Their importance to people has also been acknowledged, and resource harvesting remains an important part of subsistence community livelihoods. Their ‘unseen’ ecological services in terms of carbon storage are globally recognized as one of their most important ecosystem services.

Our research project has generated the first comprehensive assessment of blue carbon habitats in South Africa. It has used some existing data, but a large amount of new knowledge has been generated by collecting data using innovative approaches. Overall, the outcomes fill a significant knowledge gap in southern Africa and are of national interest with many components forming part of the 2018 National Biodiversity Assessment.

Results achieved are however not without limitations. Recommendations to address gaps are provided throughout the report. The project has provided new baseline data for blue carbon ecosystems that need to be incorporated into long-term monitoring and research programs. This is essential because data series of at least 5 years duration are required if threats from sea-level rise are to be assessed. Variability in sediment carbon across the mangrove-salt marsh ecotone also needs to be re-visited as the mangrove habitat continues to encroach into the salt marsh at Nahoon Estuary. Ecosystem services assessment provides a tool to compare temporal change for specific estuaries and could be valuable for future restoration research. The predictive models for mangrove expansion do not currently incorporate anthropogenic stressors that could limit or facilitate the future persistence of these ecosystems. This requires further investigation so that long-term restoration sites can be identified.

The study provided a preliminary assessment of the viability of a carbon offset mechanism for South Africa using new data collected for this purpose. This is an emerging field globally and its methodology is yet to be standardized. Additional studies to quantify carbon storage in other South African estuaries (particularly large mangrove ecosystems) will provide a more comprehensive base for the carbon inventory that underpins the offset mechanism.

RECOMMENDATIONS FOR FUTURE RESEARCH

This project has identified new areas of research and monitoring needed to understand future climate change responses. Monitoring is necessary as responses to climate change are long-term and outside the duration of a three-year research programme. Management actions have been identified to protect blue carbon habitats and their associated ecosystem services.

Ecosystem services

Research on ecosystem services promotes inter- and transdisciplinary approaches and assists in understanding and conserving complex interconnected ecological systems. Quantification of these services should remain a research priority. Coastal protection and the role of blue carbon habitats in attenuating floods and storms have not been investigated in South Africa. There are ongoing studies on fish nursery habitats, but little research has addressed the role of estuaries as pollutant filters. Ecosystem services are dynamic varying in space and time and require a deeper understanding and quantification.

Carbon storage and sequestration

This study assessed blue carbon storage for one estuary. These measurements need to be expanded to additional study sites as little data are available for southern African systems, and representation in global data sets is poor. Future research that quantifies carbon storage in South African blue carbon ecosystems should focus on the larger mangrove areas in estuaries of KwaZulu-Natal. As these occur at subtropical latitudes and receive large sediment inputs (from erodible catchments), carbon storage is likely to be much greater than that measured at the Nxaxo and Nahoon estuaries.

The influence of environmental variables on carbon storage at different spatial scales needs further investigation. Quantification of vertical accretion rates is needed to estimate carbon sequestration potential. RSETs are now in place for two salt marsh and two mangrove estuaries and need to be expanded nationally. Historical accretion should be measured using Pb^{210} isotope dating of the soil.

Habitat extent and trajectory of change

Regular measurements of the extent of blue carbon habitats are needed to identify the trajectory of change. These data have been used in the National Biodiversity Assessment (NBA) and guide future blue carbon habitat conservation and restoration initiatives. The NBA 2018 (Van Niekerk et al., 2019) assessed the pressures, states, trends, responses and benefits of estuaries, directs the future management of pressures, and highlights estuaries in need of restoration. Data on the area coverage and species composition of macrophyte habitats are used to determine the Estuarine Health Index for South Africa estuaries. It is also used to inform management (Estuary Management Plans) and conservation priorities and needs regular updating.

Buffer areas for landward expansion

Buffer areas need to be identified to allow for landward expansion of salt marsh and mangrove habitats in response to sea level rise. To conserve blue carbon habitats, estuaries where these habitats can expand inland need to be identified for climate change adaptation and protection. Coastal squeeze due to surrounding development will cause a loss of salt marsh and mangroves. The predicted habitat changes in response to sea level rise need to be modelled for each estuary in the country using the approach developed in this research (Raw et al., in press). Adjacent properties for landward migration need to be identified and protected and, in some cases, purchased from landowners. Policies and planning mechanisms to set aside buffers for landward migration need to be developed.

Sea level rise

Field studies are needed to monitor changes in habitats in both expansion and elevation of different zones particularly at ecotones. It is critically important to know whether these habitats are eroding or accreting because for their survival they must accrete at a rate that allows them to keep abreast of sea level rise. RSET measurements need to be scaled up to all estuaries with large salt marsh and mangrove areas (e.g. Olifants, Groot Berg, Keiskamma, Mngazana).

Site-specific data on sediment budgets are required to interpret RSET results such as the source of sediment and the influence of water level fluctuations on sediment transport. The rate of sedimentation determines the capacity of mangroves and salt marshes to resist rising sea-levels through surface elevation gain.

Additional monitoring data for RSET sites

Additional environmental data are needed for the RSET sites to interpret their erosion/accretion status. These include an understanding of the local topography, i.e. elevation gradient from the shore/water column to the terrestrial interface. Vegetation cover of dominant plants, water level readings, sediment particle size, and organic content needs to be measured annually. It is also necessary to use a high-resolution digital elevation model, preferably collected by a light-detection and ranging (LiDAR) remote sensing survey. LiDAR surveys have been carried out over the Nahoon Estuary as it is within the Buffalo City Municipality, although the most recently available data are from 2013. Obtaining a LiDAR survey for the Nxaxo Estuary would be a priority for future research on sea-level rise at this site. The RSET positions in relation to mean sea level need to be finalised for the Nahoon and Nxaxo estuaries.

Multiple stressors

Predicting the combined effects of multiple stressors associated with environmental instability (increase in storm surges, floods, droughts and reduced river flow) is critical to conserve blue carbon habitats. A study is underway (WRC research project K5/2931: Development of climate change mitigation and adaptation strategies for South Africa's estuarine lakes) to understand climate change responses of estuarine lakes as these systems are in a poor state. Pressures such as infrastructure development, flow reduction, artificial breaching, mouth manipulation and overfishing have increased their vulnerability to climate change. A strategic programme is needed to restore health so that these blue carbon habitats can continue to provide the ecosystem services of flood regulation, nutrient cycling, nursery habitat, and recreational and tourism opportunities.

Salinisation

Climate change is an additional stress to that of human pressures and subtle changes are occurring such as desiccation of salt marshes and salinization. Long-term monitoring of permanent plots and transects are needed to identify these changes. Remote sensing needs to be used to measure the integrity of mangrove and salt marsh habitats.

Range expansions

South Africa's geographic zones provide an outdoor laboratory to investigate range expansions between, for example, the subtropical and warm temperate regions. Long-term monitoring can measure the expansion of mangroves into salt marsh areas and the resultant changes in ecosystem services. Mangroves are expanding globally towards higher latitudes in response to increasing temperature (tropicalization). Research can provide an understanding of factors governing the dynamics between salt marsh and mangrove habitats in transitional zones. Research outcomes indicate that some estuaries will provide suitable habitat for mangroves under future climate change. This is possible because increased rainfall will promote higher mean annual runoff, and this will allow some temporarily closed systems to develop more permanent connections to the sea. Increased intensity of rainfall events is however expected to exacerbate flooding, and this could negatively impact mangroves; this aspect needs more research. Local coastal hydrology and geomorphology influence mangrove distribution patterns by restricting propagule dispersal and limiting suitable areas for establishment. Propagule dispersal and recruitment models are needed to predict movement patterns.

Greater protection of blue carbon habitats

Salt marsh, seagrass and mangrove habitats require protection as they are threatened by increasing human pressures and climate change. Priority estuaries are the Groot Berg, Knysna, Mngazana, uMlalazi, St Lucia and Kosi estuaries. The Groot Berg Estuary, with its expansive floodplain marshes, is especially unique and must be prioritised for rehabilitation and protection status. The Knysna Estuary has important large intertidal salt marshes and the most extensive population of the endangered seagrass *Zostera capensis.* The Mngazana Estuary supports the largest mangrove area in the temperate/subtropical transition zone and the largest red mangrove (*Rhizophora mucronata*) stand in the country. This estuary needs to be conserved and protected as a matter of priority. Estuarine lakes support the largest areas of submerged macrophytes and require special protection as they have weak resetting mechanisms and represent nutrient sinks.

Estuary restoration

South Africa has a long estuary research history that can inform future restoration plans. A comprehensive programme is needed to restore estuary health and ecosystem services. A National Estuary Restoration and Research Programme (NERPP) must be implemented. This can include action research and a learning-by-doing approach in a strategic adaptive management cycle. This project has identified mangrove and salt marsh sites that require such interventions, and these will address the objectives of the UN Decade of Ecosystem

Restoration (2021-2030). Applied solutions in the form of innovative methods for water quality improvement need to be developed so that estuary health can be restored and resilience to future climate change impacts maintained. The National Biodiversity Assessment of 2018 (Van Niekerk et al., 2019) indicated our estuaries to be under severe pollution pressure and that improved water quality as a key intervention would lead to a significant improvement in estuary health and associated services. The WRC Project K5/2736 (The Blue Economy from an ecosystem perspective with a specific focus on coastal resources and communities) identified estuary restoration as an important Blue Economy activity. A socio-ecological systems approach to restoration will be key in addressing alignment between legislation, governance, implementation and social commitment.

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DECLARATION OF PUBLICATIONS

The research conducted for this WRC project has in some cases been published or at the time of writing this report was under review for publication in peer-reviewed scientific journals. Where applicable, published research is acknowledged with the appropriate citation below the heading of the component.

The following publications form research components that are presented in this report:

<u>Published/In Press journal articles:</u>

Raw JL, Godbold JA, Van Niekerk L, Adams JB. 2019. Drivers of mangrove distribution at a high-energy, wave-dominated, southern distribution limit. *Estuarine Coastal and Shelf Science* 226, 106296.

Raw JL, Julie CL, Adams JB. 2019. A comparison of soil carbon pools across a mangrove-salt marsh ecotone at the southern African warm-temperate range limit. *South African Journal of Botany* 107, 301-307.

Raw JL, Riddin T, Wasserman J, Lehman TWK, Bornman TG, Adams JB. 2020. Salt marsh surface elevation and sea-level rise at the Knysna Estuary. *African Journal of Aquatic Science* 45, doi: 10.2989/16085914.2019.1662763

<u>Accepted for publication:</u>

Adams, JB. Salt marsh at the tip of Africa: patterns, processes and changes in ecosystem services in response to climate change. *Estuarine, Coastal & Shelf Science*

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LIST OF ABBREVIATIONS

AFOLU	Agriculture, Forestry and Other Land Uses
AGB	Aboveground Biomass
AGC	Aboveground Carbon
AIC	Akaike Information Criterion
AMSL	Above Mean Sea-Level
ARIMA	Auto-Regressive Integrated Moving Average
BGB	Belowground Biomass
BGC	Belowground Carbon
CCBS	Community and Biodiversity Standard
CD	Carbon Density
CDM	Clean Development Mechanism
CERs	Certified Emissions Reductions
CHN	Carbon, Hydrogen, Nitrogen
CICES	Common International Classification of Ecosystem Services
CIW	Conservation of Intact Wetlands
CO_2	Carbon dioxide
C_{org}	Organic Carbon
CSR	Corporate Social Responsibility
DBD	Dry Bulk Density
DBH	Diameter at Breast Height
DEM	Digital Elevation Model
DFR	Daily Flushing Rate
EF	Ecosystem Function
EFZ	Estuarine Functional Zone
ENM	Ecological Niche Model
ES	Ecosystem Services
GHG	Greenhouse Gases
GLS	Generalized Least Squares
GS	Gold Standard
ICER	Long-term Certified Emission Reduction

In_{org}	Inorganic carbon
IPCC	Intergovernmental Panel on Climate Change
IRMS	Isotope Ratio Mass Spectrometry
ITMOs	Internationally traded mitigation obligations
LiDAR	Light Detection and Ranging
LOI	Loss on Ignition
MAR	Mean Annual Runoff
MEA	Millennium Ecosystem Assessment
ML	Maximum Likelihood
MSL	Mean Sea-Level
NBA	National Biodiversity Assessment
NDC	Nationally Determined Contributions
NEMPAA	National Environmental Management: Protected Areas Act
NPV	Net Present Value
OM	Organic Matter
PERs/AvCE	Potential Emissions Reductions/Avoided carbon emissions
PES	Payments for Ecosystem Services
PriceCt	Price of Carbon
PSMSL	Permanent Service for Mean Sea Level
PVC	Polyvinyl Chloride
REDD+	Reducing Emissions from Deforestation and Forest Degradation
REI	River Estuarine Interface
REML	Restricted Maximum-Likelihood
RCP	Representative Concentration Pathway
RSET	Rod Surface Elevation Table
RSLR	Relative Sea-Level Rise
RWE	Restoring Wetlands Ecosystems
SANHO	South African Navy Hydrographic Office
SANParks	South African National Parks
SDM	Species Distribution Model
SEEA	System of Environmental Economic Accounting
SEM	Structural Equation Model

SLAMM	Sea-Level Affecting Marshes Model
SLR	Sea-Level Rise
tCERs	Temporary Certified Emissions Reductions
tCO_2eq	One metric tonne of carbon dioxide
TEEB	The Economics of Ecosystems and Biodiversity
TIDE	Tidal River Development Project
TN	Total Nitrogen
UN	United Nations
UNFCCC	United Nations Framework Convention on Climate Change
VCMs	Voluntary carbon markets
VCS	Verified Carbon Standard
VLM_w	Vertical Land Movement at the Wetland
WIO	West Indian Ocean
WRC	Wetlands Restoration and Conservation

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1 INTRODUCTION AND OBJECTIVES

Vegetated coastal ecosystems, including mangroves, salt marshes and seagrasses, are ecologically valuable as they exist between terrestrial, estuarine and near-shore marine environments (Barbier et al., 2011). These ecosystems provide natural protection against coastal erosion and storm surges (Arkema et al., 2013; Barbier, 2015; Perkins et al., 2015), as well as essential services such as flood attenuation, nutrient processing and sediment retention (Alongi, 2002; Zedler and Kercher, 2005; Bos et al., 2007). The plants that colonize these coastal interfaces also provide essential structural habitat to a various other species and thus form the basis of unique ecological communities (Bell et al., 1978; Whitcraft and Levin, 2007; Nagelkerken et al., 2008; Graham et al., 2016). These coastal ecosystems are however among the most threatened natural systems globally and estimates show that 50% of salt marshes, 35% of mangroves, and 29% of seagrasses have been lost or degraded by human activities (Van Katwijk et al., 2016; Feller et al., 2017; Li et al., 2018).

1.1 Defining Blue Carbon and the Importance of Blue Carbon Ecosystems

One of the most valued ecological roles of vegetated coastal habitats is carbon sequestration and storage. All plants accumulate organic carbon through the process of capturing carbon dioxide from the atmosphere and using it for growth; however, carbon is also released during the process of respiration. The carbon budget of vegetated habitats therefore relates to the build-up and discharge of carbon within the system and provides an indication of whether particular environments are carbon “sources” or “sinks” (Sitch et al., 2015). Although mangroves, salt marshes and seagrasses cover less than 2% of the area of the global ocean, these habitats are critical carbon sinks (Mcleod et al., 2011; Duarte, 2017). Carbon sequestered and stored by these ecosystems is known as “blue carbon” (Nellemann et al., 2009) and includes carbon stored in the soil, in the living biomass both aboveground (leaves, stems, branches) and belowground (roots), as well as in non-living biomass (leaf litter and dead wood) (Mcleod et al., 2011). Carbon stored in this manner is an important ecosystem service because it is a key component of the global carbon cycle (Bouillon et al., 2008; Ray et al., 2011; Keller et al., 2018). Recent research indicates that blue carbon habitats have the potential to act both as net carbon sinks when conserved and net carbon sources when degraded (Lovelock et al., 2017; Hamilton and Friess, 2018; Spivak et al., 2019).

The global loss of blue carbon ecosystems not only reduces their capacity to act as natural carbon sinks, but degradation and disturbance of these habitats also directly releases large amounts of carbon back into the atmosphere in the form of CO_2 emissions (Pendleton et al., 2012; Siikamäki et al., 2012). These emissions have been estimated at 45 billion metric tonnes annually, with an associated economic cost approaching $20 billion (International Blue Carbon Scientific Working Group, 2015). There has been a global effort to quantify intact blue carbon stocks, prevent further degradation and promote their restoration and rehabilitation (Sutton-Grier and Moore, 2016; Wylie et al., 2016; Lovelock and Duarte, 2019; Macreadie et al., 2019).

Preventing the degradation of coastal ecosystems to minimize CO_2 emissions has been advocated as a reason to conserve these habitats even though the ecological services supplied by mangroves, salt marshes and seagrasses have been widely acknowledged for decades (Lugo and Snedaker, 1974; Frey and Basan, 1978; Hatcher et al., 1989). These ecosystems support food production in the form of fisheries (Barbier et al., 2008; Liquete et al., 2013) by providing important nurseries and resting areas for commercially exploited marine species (Lamberth and Turpie, 2003; Nagelkerken et al., 2015; Muller and Strydom, 2017; Whitfield, 2017; James et al., 2019). As coastal vegetated habitats occur at the interface between marine and terrestrial environments, they provide natural coastal protection as their vegetation structures efficiently attenuate wave energy (Barbier, 2016; Montgomery et al., 2018). Mangroves limit the impact of storms by reducing the energy of wind-generated surface waves by 20% per 100 m (Das and Vincent, 2009). Despite scientific recognition of their ecological services, economic valuations for blue carbon ecosystems remain lacking for many regions. Quantification of these services can also serve as an essential motivation for protecting blue carbon habitats.

1.2 Blue Carbon Ecosystems and Climate Change

Globally, coastal wetlands cover extensive areas with estimates of up to 150 000 km^2 for mangroves and 45 000 km^2 for salt marsh in tropical and temperate regions respectively (Greenberg et al., 2006; Spalding et al., 2010). These ecosystems are threatened by climate change and are particularly vulnerable to sea-level rise, increasing temperatures, and changes in precipitation regimes (Webb et al., 2013; Ward et al., 2016; Arias-Ortiz et al., 2018).

Sea-level rise is the most significant threat to mangrove and salt marsh ecosystems around the worldwide and their vulnerability to rising sea-levels has been extensively reviewed (Webb et al., 2013; Alongi, 2015; Sasmito et al., 2016; Borchert et al., 2018). Factors affecting their susceptibility include geographic locality and elements such as coastal geomorphology, regional oceanographic properties, and sedimentation patterns (Soares, 2009; Mcleod et al., 2010). The intrinsic resilience of mangroves and salt marshes to sea-level rise is determined by positive surface elevation change and unrestricted landward migration (Di Nitto et al., 2014;

Woodroffe et al., 2016; Schuerch et al., 2018). Local topography and the extent of coastal development regulate the availability of areas for landward migration, but the rate of sedimentation determines the capacity of mangroves and salt marshes to resist rising sea-levels through surface elevation gain. Drainage basin geology and local coastal dynamics determine whether sediment is retained within the intertidal region. Local geomorphology contributes significantly towards resilience as the structure of a wetland ecosystem influences resistance to, or recovery from, a disturbance (Phillips, 2017a). The responses of mangrove and salt marsh ecosystems to sea-level rise are therefore not uniform between different regions, and variability exists between sites within the same mangrove forest or salt marsh platform (Lovelock et al., 2011; Rogers et al., 2013; Passeri et al., 2015).

The response of mangrove and salt marsh to rising temperatures and changes in precipitation regimes as a result of climate change are regionally variable. Mangroves are particularly sensitive to these changes as the global distribution range of this ecotype is linked to sea-surface temperature, with occurrence limited to tropical and subtropical regions by the winter 20°C isotherm (Tomlinson, 1999). Mangrove species are generally distributed between 30°N and 30°S (Giri et al., 2011). Rising temperatures are associated with the expansion of mangroves towards higher latitudes (polewards), described as ‘tropicalization’, and several studies have reviewed these range shifts in different regions around the world (Saintilan et al., 2014; Osland et al., 2017a; Cavanaugh et al., 2018). Expansion has often been accompanied by loss of salt marsh habitats and this can lead to large ecological shifts and changes in ecosystem service provisioning (Doughty et al., 2016; Kelleway et al., 2017a). The factors that govern the dynamics between salt marsh and mangrove transitional zones need to be better understood.

Assessing the responses of blue carbon ecosystems to climate change has been described as a priority for civil society, particularly in relation to sea-level rise in areas where the livelihoods of people are threatened (FitzGerald et al., 2008; Hall, 2011; Webb et al., 2013). Blue carbon ecosystems are resilient and ecologically stable as these ecosystems have persisted through extreme environmental variability during prehistoric periods (Timpane-Padgham et al., 2017) and, in certain regions for millennia through changes in global sea level (Morris et al., 2002; Alongi, 2015). Recent assessments have however highlighted that the ability of mangroves and salt marshes to respond to sea-level rise is regionally variable and influenced by several factors (Webb et al., 2013; Ward et al., 2016). Examining this variability is therefore necessary to underpin effective management and conservation plans for blue carbon habitats.

1.3 Research Aims and Objectives for South African Blue Carbon Ecosystems

Along the high-energy South African coastline, blue carbon ecosystems are restricted to sheltered estuarine areas. As a result, they face anthropogenic pressures such as freshwater abstraction and poor water quality associated with agricultural and urban developments throughout the catchments (Turpie et al., 2002; Van Niekerk et al., 2013). The conservation of coastal wetlands is prioritized through the National Biodiversity Act (2004) (Act No. 10 of 2004) which is part of the National Environmental Management Act. As global climate change accelerates, the potential impact of threats on blue carbon habitats in must be assessed.

Recent studies have highlighted the global variability of blue carbon ecosystems in terms of vulnerability and responses to climate change. The biogeographical patterns observed along the South African coastline present an opportunity for climate change research as the interface between subtropical and warm temperate regions is expected to be influenced significantly (Whitfield et al., 2016). This research project has therefore been developed with the following aims:

1) To determine the extent of blue carbon ecosystems in South Africa and estimate blue carbon storage using the IPCC assessment methods.
2) To quantify the loss of blue carbon habitats and associated ecosystem services.
3) To predict the responses of blue carbon ecosystems to climate change in the form of sea-level rise and increased global temperatures.

To achieve these aims, the following objectives were developed and carried out as different components of this research project. These components are presented in detail as separate chapters of this report:

1) Assess the extent of blue carbon habitats and their associated ecosystem services in South Africa. Quantify changes in blue carbon habitats and ecosystem services over time.
2) Directly quantify blue carbon storage in mangrove, salt marsh, and seagrass habitats at a representative study site. Compare carbon storage between mangrove and salt marsh habitats at the mangrove distributional range limit.
3) Measure surface elevation change in mangrove and salt marsh habitats and relate this response to sea-level rise threats.
4) Predict changes in mangrove distribution along the South African coastline in response to rising temperatures and changes in precipitation regimes associated with climate

change. Review the implications of climate change on salt marsh along the South African coastline.

5) Determine the viability of a carbon offset mechanism for South Africa's blue carbon ecosystems.

Tidal creek at Nahoon Estuary, East London. Photo: J Raw

2 BLUE CARBON HABITATS AND ECOSYSTEM SERVICES

2.1 Introduction

Blue carbon habitats (salt marshes, seagrasses and mangroves) provide many ecosystem services (ES), broadly defined as benefits people derive from the ecosystem's ecological structure and function that sustains human life (Daly, 1997; Costanza et al., 1997; MEA, 2005; Barbier et al., 2011). The quantification of these benefits is referred to as the valuation of ES, which can be done in two ways (Barbier, 2017). Firstly, valuation is monetary, and ES is expressed in monetary units derived from cost-benefit analyses and macroeconomic indicators. Secondly, it is assessed using biophysical quantification based on empirical data (i.e. ecosystems accounting) (Mancini et al., 2018). Of the two, monetary valuation is often the most common and preferred method (Costanza et al., 1997; 2014b; Mancini et al., 2018) because assigning a monetary value to an ES has a higher impact, especially when arguing for the conservation of natural systems to policymakers, the business sector and stakeholders (TEEB, 2010; Hanson et al., 2012; Mancini et al., 2018).

Several ES classifications have emerged including the Millennium Ecosystems Assessment (MEA) (MEA, 2005), Economics of Ecosystems and Biodiversity (TEEB, 2010), Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (Díaz et al., 2015), System of Environmental-Economic Accounting (SEEA; UN et al., 2014) and the Common International Classification of Ecosystem Services (CICES)(Haines-Young and Potschin, 2013). The MEA was the first to group ecosystem services into four categories: provisioning, regulating, supporting and cultural services and this classification scheme led to the exponential increase in ES research and its applications. Most recent literature has since shifted towards CICES and much recent research is based on the 2010 ES cascade framework (Potschin and Haines-Young, 2016). The CICES approach reverted to three major sections: provisioning, regulating/maintenance and cultural services, where the 'supporting' service is excluded to avoid double counting. The cascade framework also details the pathway of ES from ecological structures to processes to human well-being and separates these entities into measurable units. For example, ES can be divided into ecosystem properties (biophysical structures or stocks) that produce ecosystem functions (flows) that provide ES of benefit to humans and can be valued (Potschin and Haines-Young, 2011; Boerema et al., 2017).

The ecosystem services of coastal habitats have been reviewed extensively (Lau, 2013; Liquete et al., 2013; Cullen-Unsworth et al., 2014; Vegh et al., 2014; Campagne et al., 2015; Barbier, 2016; Dewsbury et al., 2016; Torres and Hanley, 2016; Nordlund et al., 2016, 2017) with a new systematic review by Himes-Cornell et al. (2018) focusing on the valuation of blue forests. This review emphasized the lack of ES studies in blue forests and argued that current studies are often inaccurate (based on outdated data older than 10 years) or difficult to find as data are not always presented in the context of ES and often focused on something else. Moreover, major gaps still exist in the methods used, past and present habitat area cover data, and estimated economic values. The authors also reported that of the blue carbon habitats, the ES of salt marsh are the most understudied, despite having a greater cover than mangrove and seagrass habitats. Although there are more studies for mangroves and seagrasses, these are still generalized for all species, genera and geographical location. For example, for seagrasses, the general consensus is that leaf area of shoots is positively correlated to ES provision, but most data are generated from larger species (i.e. *Posidonia*) and not smaller ones (i.e. *Halophilia* and *Zostera)* further highlighting the gaps in ES research mentioned previously (Nordlund et al., 2016).

Blue carbon habitats deliver numerous ES including the provision of raw materials, nursery habitats for juvenile fish, coastal protection, nutrient retention and water quality enhancement (Elliot and Whitfield, 2011). Globally, the most studied and cited of these services is nursery function (Lefcheck et al., 2019). This is also true for estuaries in South Africa with the nursery value/function for juvenile fish most researched (i.e. Strydom, 2015; Whitfield and Pattrick, 2015; Edworthy and Strydom, 2016; Leslie et al., 2017; Whitfield, 2017; Bornman et al., 2018; McGregor and Strydom, 2018; Nel et al., 2018). Turpie et al. (2017) estimated the nursery value of estuaries at close to R803 million per year, followed closely by the provision of habitats for fish (Potter et al., 2015; Grant et al., 2017; Muller and Strydom, 2017; Pollard et al., 2017) and invertebrates such as crabs and snails, which play an important role in ecosystem engineering and influence benthic primary productivity (Cannicci et al., 2008; Raw et al., 2017; Peer et al., 2018). Natural resource provision and subsistence harvesting of coastal habitats is also an important ecosystem service, valued at approximately R35.7 million per year (Turpie et al., 2017). Mangroves in the Eastern Cape have been harvested by rural coastal communities for firewood, house building materials and fences, and through animal browsing (Rajkaran and Adams, 2010; Hoppe-Speer et al., 2015a); mangrove fauna are also collected for food or used as bait for fishing (Rajkaran and Adams, 2010). Nutrient cycling is another important ecosystem service, provided specifically by *Zostera capensis* Setchell seagrass beds which store, release and cycle large amounts of nitrogen and phosphorus (Tibbles et al., 1994; Lemley et al., 2014; Human et al., 2015; Adams, 2016).

Natural habitats and their associated ecological processes are being altered and/or destroyed worldwide mainly as a result of human population growth and land development (MEA, 2005; Sala et al., 2000; Tardieu et al., 2015). In accordance with these global trends, coastal habitats in South Africa are under threat due to various anthropogenic pressures including freshwater abstraction, agriculture, pollution from surrounding industries, resource over-utilization and coastal squeeze (Van Niekerk et al., 2013). With an increase in coastal development, there is an increase in land-cover conversion that leads to the loss of key habitats (Schwarz et al., 2017). Land conversion directly impacts an ecosystem because the loss of habitat results in a loss of ecosystem function and thus services (Kreuter et al., 2001; Fahrig, 2002; Gascoigne et al., 2011; Broekx et al., 2013; Geneletti et al., 2013; Kumar et al., 2013; Tardieu et al., 2015). The loss of ecosystem services is largely dependent on habitat type and spatial extent. Even though ES quantification and valuation studies have increased rapidly, those that assess the direct loss of ES resulting from habitat change remain insufficient (Tardieu et al., 2015).

In South Africa especially, ecosystem services in estuaries play an important role in addressing the country's prevalent poverty issues (Turpie et al., 2017). Quantification of these services is important for informing policy and highlighting the role of natural capital as an incentive for coastal conservation. The main aims of the research project presented in this section were to:

1) Quantify the potential supply of ecosystem services by blue carbon habitats in South African estuaries; and

2) Evaluate how the potential supply of ES by blue carbon habitats has changed over time due to land transformation.

Recent studies show that approximately 50% of blue carbon habitats have been lost in South Africa, and within the Eastern Cape 1.04 ha of mangrove forest is lost every year (Adams et al., 2004; Rajkaran and Adams, 2012; Hoppe-Speer et al., 2015a). We hypothesize that in most estuaries there has been a reduction in the potential to contribute to ES due to habitat transformation.

2.2 Materials and Methods

2.2.1 Study Areas

South Africa has 290 estuaries unequally distributed along its ~ 3000 km coastline. These have been newly classified into nine types, namely estuarine lakes, estuarine bays, estuarine lagoons, predominantly open estuaries, large and small temporarily closed estuaries, large and small fluvially dominated estuaries, and arid predominantly closed estuaries (Van Niekerk et al., in press). Estuarine habitats cover a total area of 95 657 ha and include open water area/channel, intertidal and supratidal salt marsh, submerged macrophytes, reeds and sedges, mangroves, sand/mud banks, rocks and swamp forest (Coetzee et al., 1997; Colloty et al., 1998; Adams et al., 2016).

Recently Adams et al. (2018) reviewed the current state of estuarine habitats in South Africa in terms of extent/habitat area cover and species richness in the recent National Biodiversity Assessment (2018). The study found that the total area cover for intertidal and supratidal salt marsh was 6 189.78 ha and 2 564.78 ha respectively, while submerged macrophytes had an area cover of 2 564.78 ha, and mangroves an area cover of 1 631.03 ha. Reeds and sedges had the highest cover (14 732.6 ha). The definition of blue carbon habitats includes only salt marsh, seagrasses and mangroves (Howard et al., 2014). Therefore, we decided to exclude reeds and sedges even though they are the dominant habitat type.

2.2.2 Assessing the Areal Extent of Blue Carbon Habitats

Status of current blue carbon habitats and reeds were identified using the Estuary Botanical database, an Excel workbook with three worksheets. The first worksheet contains habitat cover data for each estuary including location, habitat area and anthropogenic pressures (i.e. development pressure). The other two worksheets comprise a species list, and habitat richness (number of habitats in each estuary) (Turpie et al., 2002; Turpie and Clark 2007; NBA, 2011; Adams et al., 2016). The area cover data in the database has been sourced from vegetation maps of estuaries used as part of environmental flow requirement studies and estuary management plans (available as GIS shapefiles in the estuarine database). It is constantly updated using available information from site visits and research projects.

2.2.3 Identification of Potential ES and Their Classification

The study identified five potential ecosystem services that can be provided by blue carbon habitats. They were classified based on the revised version of the Common International Classification of Ecosystem Services (CICES Version 5.1, Haines-Young & Potschin, 2018). CICES is hierarchical with the highest level being “Sections” which is divided into three known MEA (2005) categories, namely provisioning, regulating and maintenance, and cultural. This approach includes habitat services under regulating and maintenance and excludes biodiversity to avoid double counting of benefits (Haines-Young and Potschin, 2013). Aspects of biodiversity are important for supporting ecosystem function and service delivery, such as habitat structure and area in addition to abundance, biomass and density, all direct indicators or metrics of biodiversity (Schwarz et al., 2017). Some metrics of biodiversity will be included in this study even though biodiversity itself as an ES will not be measured. Also, cultural services (i.e. recreation and tourism) have been excluded as these have been assessed extensively in blue carbon habitats (Himes-Cornell et al., 2018). The major “Sections” are further divided into “Divisions”, “Groups”, “Class” and “Class type”. For this study, the classification system was slightly modified, ES was classified down to “Class” and then a short description of the service was given instead of “Class Type” (Postchin and Haines-Young, 2016) (Table 2.1).

Table 2.1. Classification of potential ecosystem services of blue carbon habitats based on the CICES classification scheme.

<table>
<tr><td>Section</td><td colspan="6">Regulating and maintenance</td><td>Provisioning</td></tr>
<tr><td>Division</td><td colspan="2">Regulation of physical, chemical, biological conditions</td><td colspan="4">Transformation of biochemical or physical inputs to ecosystems</td><td>Biomass</td></tr>
<tr><td>Group</td><td>Life cycle maintenance, habitat and gene pool protection</td><td colspan="3">Mediation of wastes or toxic substances of anthropogenic origin by living processes</td><td colspan="3">Wild plants (terrestrial and aquatic) for nutrition, materials or energy</td></tr>
<tr><td>Class</td><td>Maintaining nursery populations and habitats (including gene pool protection</td><td colspan="3">Filtration/sequestration/storage/accumulation by micro-organisms, algae, plants, and animals</td><td>Wild plants (terrestrial and aquatic, including fungi, algae) used as a source of energy</td><td colspan="2">Wild plants (terrestrial and aquatic, including fungi, algae) used for nutrition</td></tr>
<tr><td>Description</td><td>Nursery function
Habitat for invertebrates</td><td colspan="2">Carbon storage</td><td>Nutrient filtration</td><td colspan="3">Mangrove harvesting (for fuelwood)</td></tr>
</table>

2.2.4 Blue Carbon Habitats and Their Potential to Supply ES

The Ecosystem Services (ES) concept provides a useful method to determine the potential natural capital provided by ecosystem properties. The ecosystem service cascade comprises four parts (properties, functions, benefits and values) where ecosystem properties (EP) are the biophysical structures of an ecosystem (ecosystem stocks) while ecosystem function (EF) or processes are ecological interactions between ecosystem components over time that generate flows of services. Ecosystem services are determined by functions, which are a collection of structures and processes within an ecosystem (Boerema et al., 2017). However, few studies quantify the full ES cascade and its required aspects (Van Oudenhoven et al., 2012; Boerema et al., 2017) and numerous studies are based only on part of the ES cascade (properties). The most common measure/unit for ecosystem properties is land use/land cover or habitat area (ha), which is used as a proxy for the potential supply of ES. Another measure is the stock of various ecosystem properties (e.g. vegetation biomass, soil type and volume) or ecosystem functions. Ecosystem properties/structures/stocks are used as proxies for ES as they allow service to be supplied (La Notte et al., 2017). In our study, we assigned scores to represent the overall potential to supply ES based on habitat area (Table 2.2). Each habitat type required its own ranking as there was large variability in area cover (e.g. salt marsh cover of >1000 ha versus mangrove cover of <1 ha).

Table 2.2. Overall potential to supply ES ranks based on habitat area (ha).

Ranking	Mangroves	Salt marsh	Seagrass
1	< 50	<500	<300
2	50-100	500-2000	300-400
3	100-500	2000-5000	400-500
4	500-1000	5000-10 000	500-600
5	>1000	> 10 000	>600

Blue carbon habitats were aggregated into four biogeographic regions and assigned a past and present score. In total there were 33, 105, 59, and 2 estuaries that were included from the cool temperate, warm temperate, subtropical and tropical regions respectively. The scores represented an approximate quantification of the potential supply for each ecosystem service (Goldenberg, 2017). Scores were used in an ecosystem service matrix, a useful tool that allows large amounts of information to be presented in a compressed way that is simple and easily understood. Typically, this matrix has two-dimensions with column and rows that form grids where indicators scores are entered (Burkhard et al., 2009; 2012; Mangi, 2016). For the assessment matrix, we used values from 1 to 5; 0 indicated there was no habitat available to supply the ES, as suggested by Burkhard et al. (2012).The transformation in area of each habitat type (either gain or loss) was translated into the loss or gain in the potential for a particular habitat to supply the ES using available data.

2.2.5 Ecosystem Services in Detail and Derivation from Literature

Mangrove Harvesting for Fuel/Energy

Due to their proximity to rural communities, mangroves are a valuable natural resource in estuaries and there is a large demand for firewood and building material. Harvesting was about 75% in a total of 17 estuaries in the Eastern Cape (Hoppe-Speer et al., 2015a). Rajkaran and Adams (2010) quantified the rate of mangrove harvesting at Mngazana Estuary in the Eastern Cape to be approximately 1 ha^{-1} kg^{-1} yr^{-1}. We therefore scored the potential for mangroves to supply this service based on mangrove habitat cover data and the recorded mangrove harvesting rate.

Nutrient Filtration

The filtration function of estuaries is most important as it serves as an additional wastewater treatment service (Costanza et al., 1997; Dähnke et al., 2008; TIDE, 2013). In estuaries, submerged macrophytes are important for nutrient cycling by acting as both sources and sinks for nutrients (Lemley et al., 2014). The removal of nitrogen and phosphorus from the water reduces eutrophication and improves water quality. Inorganic nutrients in the porewater of sediments are absorbed through roots and rhizomes, and surface water is taken in through the blades of seagrasses (Van der Heide et al., 2008; Bulmer et al., 2018). Human et al. (2015) found that after a year of mouth closure at the Groot Brak Estuary, *Zostera capensis* filtered and released more total nitrogen (TN) (6 180 kg and 5 352 kg) than other submerged macrophytes and macroalgae (3 863 kg) and *R. cirrhosa* (2 828 kg). Nutrient filtration will therefore be quantified based on seagrass area cover according to information from Human et al. (2015).

Nitrogen and Phosphorus Uptake

Scores were calculated based on data from Human et al. (2015) from the Groot Brak Estuary. We calculated the rate of nitrogen and phosphorus uptake by *Z. capensis* using reported uptake values and seagrass area cover. The rate of nitrogen uptake was 2 122 kg N ha^{-1} yr^{-1} and phosphorus was 604.66 kg P ha^{-1} yr^{-1} during the closed estuary mouth conditions and 873.42 kg N ha^{-1} yr^{-1} and 432.02 kg P ha^{-1} yr^{-1} when the mouth was open. Area cover of seagrasses per estuary was multiplied by the rate of nitrogen and phosphorus uptake to determine the nutrient filtration. The mouth state of the estuary was also considered.

Macrophyte Habitat Contribution to the Nursery Function

Many studies have highlighted estuarine macrophytes as important habitats for juvenile fish (Whitfield et al., 2016) and have shown that more complex habitats provide a greater nursery value (Hovel and Lipcius, 2001; Jackson et al., 2006; Whitfield, 2017; Leslie et al., 2017) Recently, Potter et al. (2015) reported mangrove forests to be the most important nursery habitat followed by seagrasses and salt marshes, while Leslie et al. (2017) at Bushmans Estuary found a higher abundance of juvenile *Rhabosargus holubi* (Steindachner 1881) fish in the seagrass beds of *Z. capensis* compared to the salt marsh *Spartina maritima* (Curtis) Fernald.

To quantify macrophyte contribution to the nursery role, we used the desktop provisional eco-classification of temperate estuaries in South Africa to identify estuaries that are important nursery areas (Van Niekerk et al., 2015). The selected estuaries were grouped into the different estuary types; most were either predominately open or temporarily closed. Scores for blue carbon habitat contribution to the 'nursery function' ES were generated based on salt marsh and seagrass area in the cool and warm temperate region. The quantitative estimate for the abundance of juvenile fish in the seagrass beds of southern Australia was used which was 9 260 individuals ha^{-1} yr^{-1} (Blandon et al., 2014). We applied this rate of juvenile recruitment to our habitats since juvenile abundance data in our blue habitats is scarce.

Carbon Storage

Estuaries are biologically important systems because of their high productivity and their importance as carbon sinks (Chmura et al., 2003). We used the IPCC tiers of assessment methods to determine the carbon stored in the mangrove, salt marsh and seagrass habitats. The IPCC has three tiers of assessment and Tier 1 was used for our purpose (Howard et al., 2014). Available habitat area data from the botanical database was multiplied by the carbon estimate for each habitat type (mangroves, salt marsh and seagrasses). Carbon stored was expressed in Megagrams Carbon (Mg C).

Habitat for Invertebrates

Mangroves have been identified as important habitats for faunal species such as fiddler crabs and gastropods. A comprehensive assessment of the current state of mangrove-associated fauna was recently done by Peer et al. (2018) and was comparable to the first survey published by Macnae (1963). This study used the abundance of brachyuran crabs and gastropods (no.

individuals m^{-2}) counted at mangrove estuaries and this data was combined with present and past mangrove area cover data to quantify this service.

2.3 Results

Based on habitat area, salt marshes and seagrasses in the cool temperate region scored 4 in their potential to supply ES; salt marshes however decreased from a ranking of 4 over time. In the warm temperate region, mangroves ranked the lowest in terms of present and past potential ES supply. Salt marshes scored 3 for their present potential to supply ES; this decreased from a past ranking of 4. Seagrasses in the warm temperate region were identified as having the highest potential to supply ES but this has decreased over time. In the subtropical and tropical regions, mangroves in the past had the highest potential to supply ES but this has decreased while seagrass habitats have increased their potential. Salt marshes in the tropical region have increased their potential to supply ES (Table 2.3).

Table 2.2. Overall present and past potential to supply ES based on area cover by mangroves, salt marshes and seagrasses in different biogeographic regions.

	Cool temperate		**Warm temperate**		**Subtropical**		**Tropical**	
	Present	Past	Present	Past	Present	Past	Present	Past
Mangrove	-	-	1	1	2	5	1	2
Salt marsh	4	5	3	4	3	2	5	1
Seagrass	4	4	5	4	5	1	2	0

2.3.1 Changes in Ecosystem Services over Time

Estuaries in the tropical and subtropical region increased their potential to supply mangroves for harvesting by ~2-100 kg. yr^{-1} while mangroves in the warm temperate region decreased their potential (Figure 2.1). The potential to supply habitats for mangrove fauna has largely increased in estuaries in the subtropical region and increased slightly in the tropical region. This is indicated by a potential increase in abundance of crabs and snails in this region. The warm temperate region has declined in its potential to supply habitats for mangrove fauna (Figure 2.2).

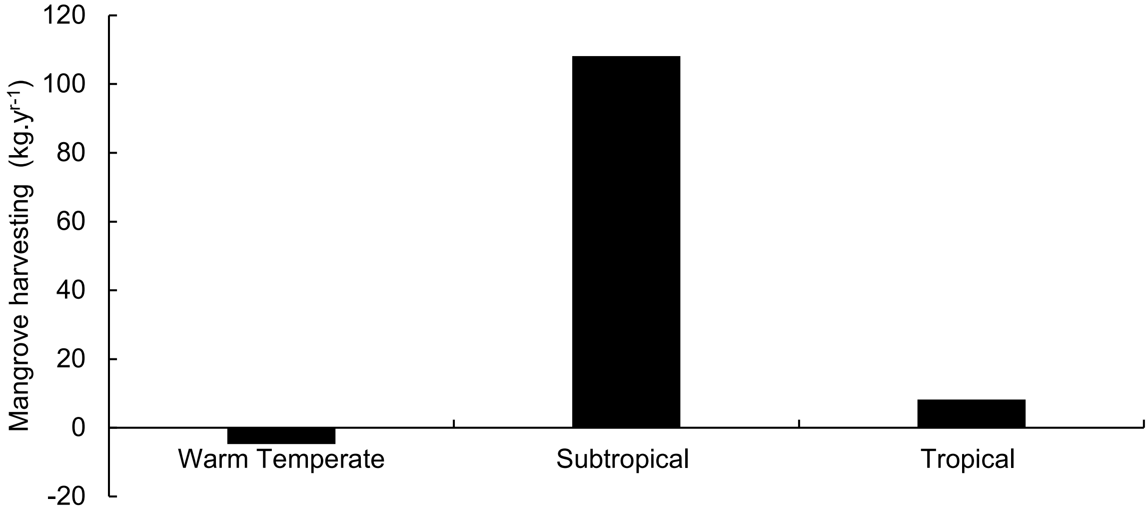


Figure 2.1. Changes in potential for mangrove harvesting (kg. yr^{-1}) over time in the different biogeographic regions.

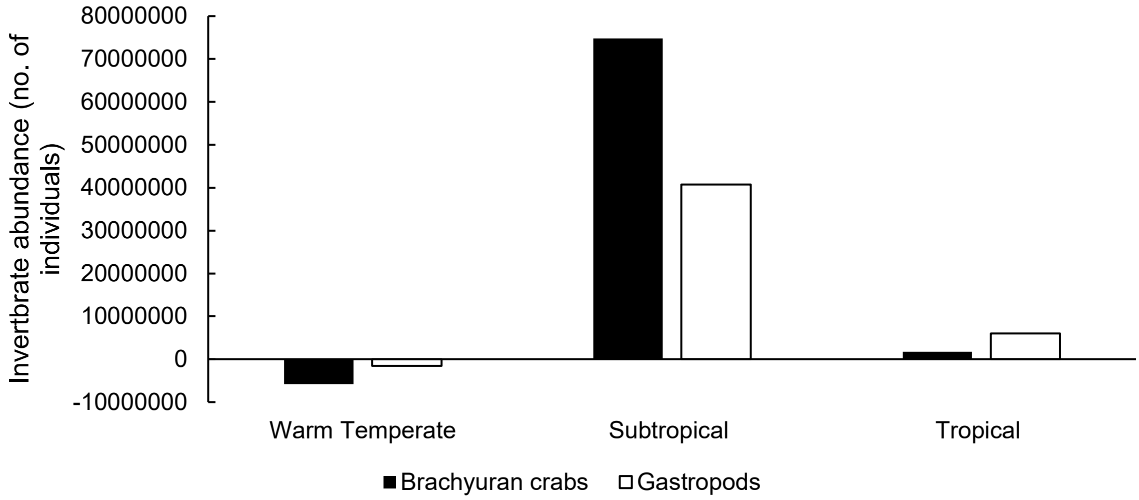


Figure 2.2. Changes in potential for invertebrate habitat supply (measured as abundance) in the different biogeographic regions.

Overall the potential for nutrient filtration by blue carbon habitats has increased over time, especially in the tropical region. Seagrasses in the tropical region have the potential to filter ~1600 tons of N and P per year while seagrasses in the cool temperate region have gained the potential to filter only 200 tons of N and P per year (Figure 2.3). The contribution of salt marshes in the nursery function has decreased especially in predominantly open estuaries in both cool and warm temperate regions. There has been an increase in this service in seagrasses in the warm temperate region (Figure 2.4).

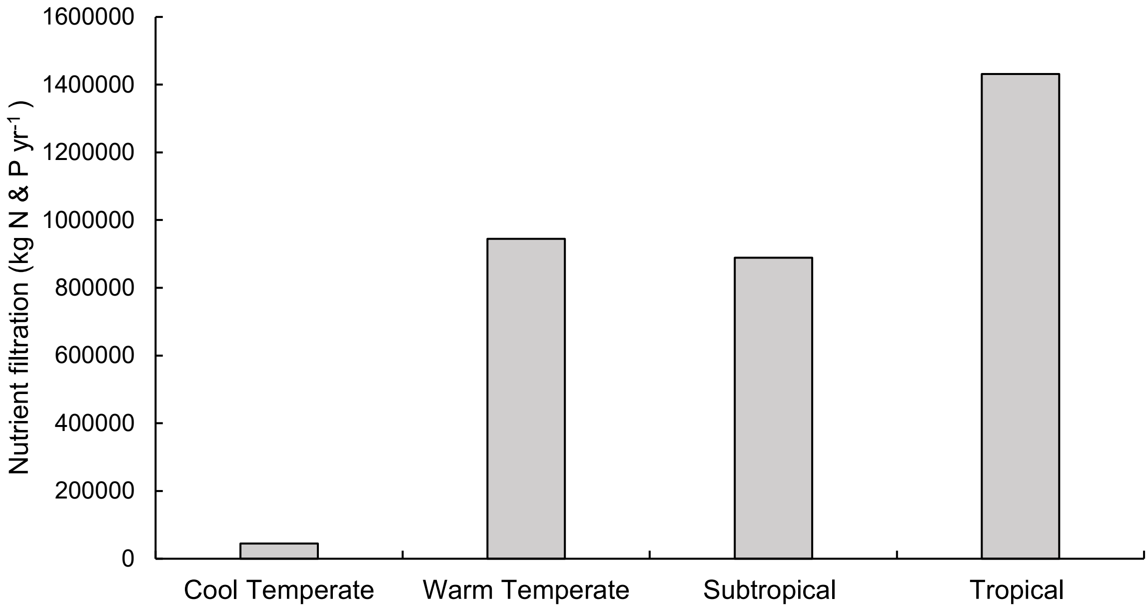


Figure 2.3. Changes in potential to supply nutrient filtration (kg N & P yr$^{-1)}$ over time at the different biogeographical regions.

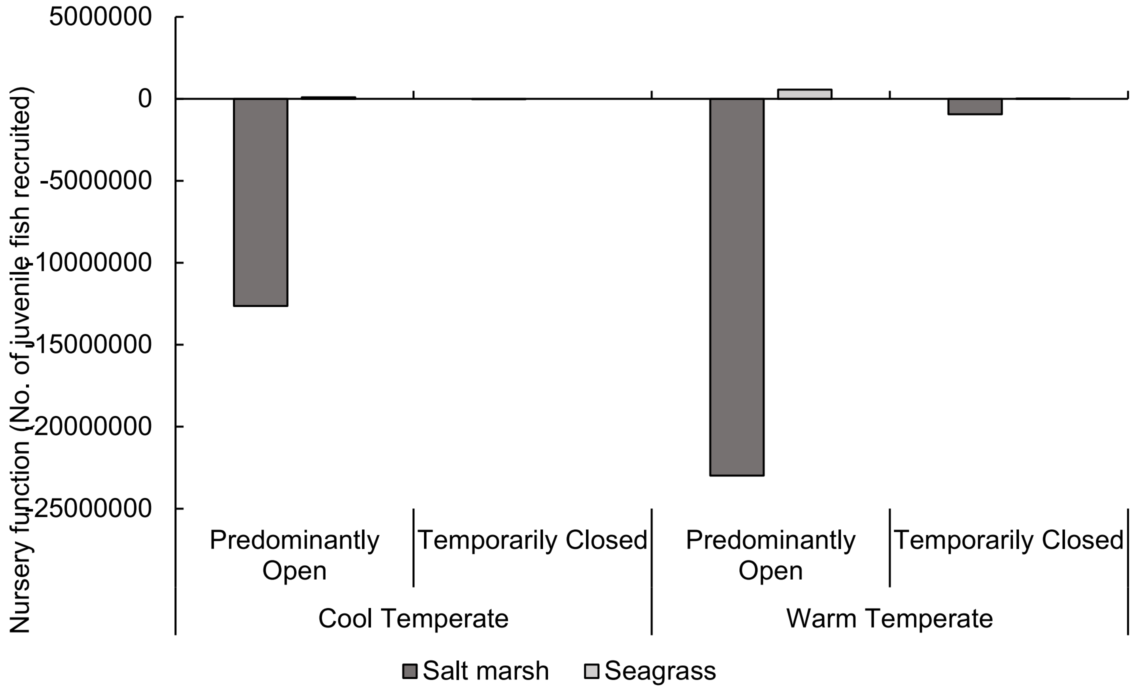


Figure 2.4. Changes in the blue carbon habitat potential contribution to nursery function in the cool and warm temperate regions for predominantly open and closed estuaries.

There have mainly been losses in potential supply carbon storage, especially in the warm temperate salt marsh habitats followed by salt marshes in the cool temperate region. These salt marshes have potentially lost about 400 000-800 000 Mg C. There have been slight gains in carbon storage potential in the subtropical and tropical regions of less than 200 000 Mg C (Figure 2.5).

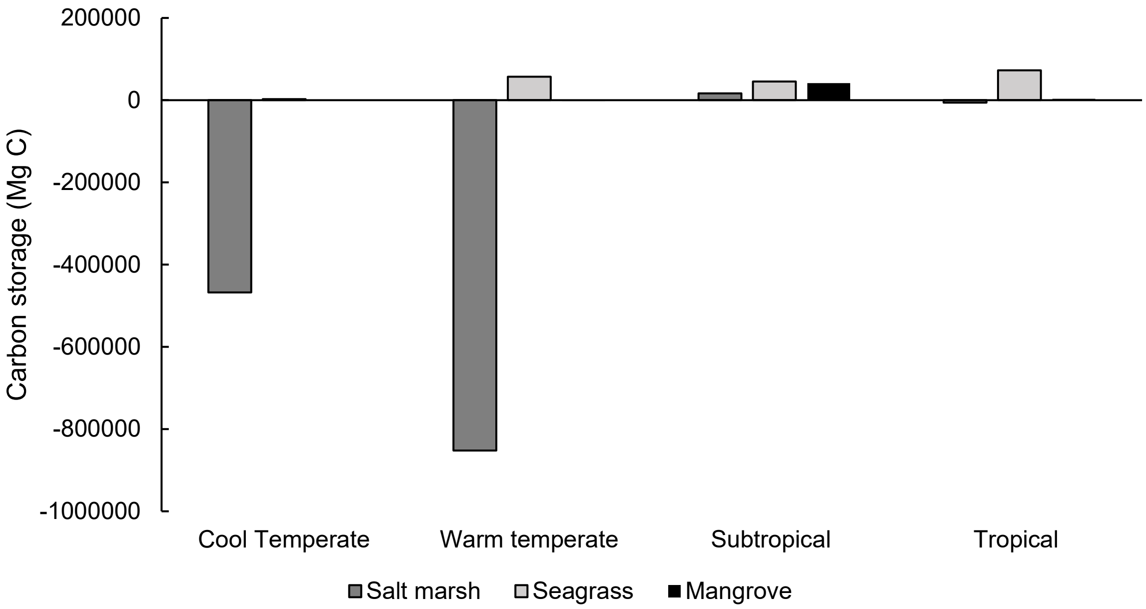


Figure 2.5. Changes in potential for carbon storage at different biogeographical regions.

2.4 Discussion

The management and conservation of ecosystem services has become paramount because these services are under threat (Sutherland et al., 2018). Ecosystem services are affected by a multitude of natural and human induced disturbances, the most common of which is land use change, recently termed an "ecosystem disservice" (Leh et al., 2013). Biophysical structures and processes of an ecosystem can be changed by large perturbations caused by the removal of vegetation. These changes can modify ecosystem functioning and ultimately the potential to supply ES (Haines-Young and Postchin, 2010; Davidson et al., 2017). Our study was the first to quantify ecosystem services (provided by mangrove, salt marsh and seagrass habitats) using area cover as the main ecosystem property serving as an indicator for the potential to provide ES since there is little data for direct measurement of ecosystem processes, functions and services (Van Oudenhoven et al., 2012; Davidson et al., 2017). We also assessed how the potential to provide ES changed over time as a result of habitat transformation from either land use changes, natural perturbations or other factors.

We found our estuarine blue carbon habitats to have experienced a cumulative loss in potential to provide ES over time. In the most recent National Biodiversity Assessment, Adams et al. (2018) reported 10 000 ha loss of blue carbon habitats in estuaries specifically salt marshes (4312 ha) and mangroves (250 ha). Estuaries that displayed cumulative losses in ES were those from warm temperate and subtropical regions. Typically, warm temperate estuaries are in good to excellent condition because they occur in mainly rural and underdeveloped areas (Van Niekerk et al., 2013); habitat loss is a result of natural disturbances (i.e. floods, mouth closures and prolonged inundation of mangrove pneumatophores). These natural pressures have caused large-scale destruction of blue carbon habitat, predominantly mangroves, as reported for Bulungula, Kobonqaba and Mbashe estuaries (Hoppe-Speer et al., 2015a). These disturbances will occur more often with global warming, resulting in further habitat losses (Hoppe-Speer et al., 2015a) and ultimately losses in ES. Subtropical mangroves have experienced the greatest losses in potential to supply ES. This result was probable as many subtropical estuaries are in poor condition, attributable to habitat loss, artificial mouth breaching and agricultural practices (Van Niekerk et al., 2013).

Few estuaries have the habitat requirements for a good nursery area. Submerged macrophytes are important to provide this service and their extent changes in response to turbidity and sediment type, were not included in our study. Estuary type has been linked to the potential to provide nursery services; larger, permanently open estuaries are better nursery areas (Lamberth and Turpie, 2003; Van Niekerk et al., 2017). Contrastingly, Strydom (2015) reported similar larval fish density between estuarine lakes, permanently open and temporarily open-closed estuaries in both the cool temperate and warm temperate regions. We found that blue carbon habitats in the cool and warm temperate regions have experienced the largest losses to potential nursery function, which agrees with Turpie et al. (2017) who reported that almost half of our estuaries have lost their nursery function. Larvae and juvenile fishes at Kowie Estuary have also declined in response to reduced salt marsh habitats when the estuary was transformed into a marina (Kruger and Strydom, 2010). The loss in habitat area (especially seagrasses) reduced food and protection from predation in these nursery areas (James et al., 2018).

In the Eastern Cape, Kobonqaba, and Bulungula estuaries have lost their potential to supply mangroves for harvesting. At Kobonqaba Estuary, 92% of mangroves were lost owing to long-term mouth closure, which increased water levels and inundated pneumatophores (Mbense et al., 2016). Likewise, at Bulungula Estuary, drought caused the mouth to close, flooding pneumatophores (Adams et al., 2004; Hoppe-Speer et al., 2015a).

Coastal blue carbon habitats are declining at a rate ranging 0.5-3% per year. Habitat destruction and the disturbance of sediment causes the remineralization of carbon dioxide that has been stored over long periods (Mcleod et al., 2011; Pendleton et al., 2012; Lovelock et al., 2017). We found the potential for carbon storage has mostly declined in salt marshes in the Western Cape region and this could be attributed to habitat losses from land transformation for development and agriculture (Adams et al., 2019). Although development is less likely to alter carbon storage per unit area, it significantly decreases carbon stocks in estuarine habitats (Conrad et al., 2019). Changes in blue carbon per unit area however depends more on local conditions such as tidal inundation and nutrient inputs (Mcleod et al., 2011; Sanders et al., 2014; Conrad et al., 2019). Seagrass habitats at Knysna Estuary have increased despite pollution in the Ashmead Channel (Adams, 2016) and the proliferation of opportunistic macroalgae due to nutrient inputs from a wastewater treatment works close to the estuary (Allanson et al., 2016; Human et al., 2016a). Industrial development near estuaries, increases suspended sediment loads and nutrient discharge, which encourages algal growth often at the expense of seagrasses (Orth et al., 2006; Burkholder et al., 2007; Conrad et al., 2019). At Swartkops Estuary, *Z. capensis* has expanded by ~20 ha despite increased development in the surrounding area and the complete removal of seagrass after flooding in 1984 (Bornman et al., 2016).

Brachyuran crabs are the most abundant macrofauna in mangrove systems. Their bioturbation influences abiotic and biotic sediment properties, nutrient cycling and productivity (Amaral et al., 2009). Anthropogenic disturbances such as wastewater discharge have been shown to influence crab abundance (Wear and Tanner, 2007), bioturbation activities (Bartolini et al., 2011) and crab community structure (Capdeville et al., 2018). In South Africa crab and gastropod abundance decreased in the warm temperate mangroves and increased in the mangroves of the subtropical and tropical regions. Although subtropical estuaries are associated with poor water quality, chemical pollution and exposure to sewage (which results in higher nutrient levels) due to urbanization (Naidoo, 2016; Peer et al., 2018), Peer et al. (2018) found that crabs were still abundant in these mangrove forests. These include mangrove forests in the harbours of Richards Bay and Durban. While exposure to chemicals reduces the survival and health of mangroves (Naidoo et al., 2010; Lewis et al., 2013; Naidoo, 2016), increased nutrient levels can have a positive effect on crab diversity and abundance. This has been shown in earlier studies by Cannicci et al. (2009) and Bartolini et al. (2011) which suggested that wastewater input increased mangrove crab abundance in Kenya and Mozambique. Contrastingly, a recent study by Theuerkauff et al. (2020) report that pollution from urban runoff significantly decreased mangrove crab abundance as exposure to wastewater results in rapid oxygen consumption in addition to osmotic and redox imbalances (Theuerkauff et al., 2018). However, the effects of wastewater pollution on mangroves forest still requires further investigation in South Africa (Naidoo, 2016) and what effects this pollution has on the associated mangrove fauna.

2.5 Conclusions and Study Limitations

In this study we used ecosystem properties (habitat area) to quantify the provision of ecosystem services by blue carbon habitats and assess how ES provision has changed over time. In some instances, provided data were available, ecosystem properties were linked to processes and functions. However, all links of the ecosystem service cascade were not quantified and remain not fully understood. Generally, ecosystem services and benefits have been based on qualitative data that has been widely used in current conservation efforts and policies because quantitative data is scarce (Koch et al., 2009). Therefore, we proposed a modified method to quantify ecosystem services and their temporal changes, which needs to be expanded. Future research should include quantitative data when they become available and qualitative future predictions of the impacts of natural pressures, development and land use changes on ES provision in these habitats.

Some ES provision may have been slightly over- or underestimated. For example, the rates used for nutrient filtration from the Groot Brak Estuary (Human et al., 2015) were notably higher than values reported in literature, considering that these were the nutrient filtration rates for seagrass alone (specifically *Z. capensis.*). Turpie et al. (2017) suggested that wetlands can filter between 10 kg N ha^{-1} yr^{-1} and 25 kg N ha^{-1} yr^{-1} while Human et al. (2015) reported a range of 432-2122 kg N ha^{-1} yr^{-1} at the Groot Brak Estuary. Nutrient filtration could therefore be overestimated. In a study on the quantification of nutrient filtration in the Scheldt Estuary (Belgium), the rate of nitrogen removal was also low (between 11 and 347 kg N ha^{-1} yr^{-1}) (Van Damme et al., 2010; TIDE 2013).

The potential for carbon storage may also have been an overestimation. For this study, we conducted a Tier 1 blue carbon assessment using the IPCC default factors for mangroves, salt marshes and seagrasses outlined in Howard et al. (2014). The Tier 1 assessment is the least accurate and is based on simplified mean and range values of organic carbon to 1 m depth. The Tier 1 assessment is compared to site-specific data from a Tier 2 assessment for the blue carbon habitats at Nxaxo Estuary in the Eastern Cape. The preliminary results show Tier 1 assessments to be overestimates for the amount of carbon stored in habitats especially mangroves and salt marshes as the values were significantly higher than those obtained for Tier 2. The potential for carbon storage can therefore only be quantified accurately when there are better blue carbon storage data available (using the Tier 2 assessment) for more estuaries.

Mangroves and salt marshes contribute to coastal protection through the reduction of wave energy and erosion (Spalding et al., 2014). Wave attenuation is a function of plant cover and other material obstructing the water column in addition to the bathymetry of the area (Koch et al., 2009). Mangroves can reduce wave heights by 13% to 66% and salt marshes by 61-82% (Möller et al., 1999; McIvor et al., 2012; Spalding et al., 2014). In this study, coastal protection/wave attenuation was not included as one of the quantified services. Although in theory, blue carbon habitats provide this service, it is insufficient to quantify this service based on habitat area alone (Barbier et al., 2008; Feagin, 2008; Koch et al., 2009). To properly quantify coastal protection, the area of the mangrove belt or the coastline is required as well as other parameters such as wave depth (Shuto, 1987; Koch et al., 2009).

Generally measuring ecosystem services (specifically the above-mentioned regulating services) is difficult because these services do not remain constant in space and time and are non-linear (Farnsworth, 1998; Koch et al., 2009). One of the major assumptions in ecosystem service quantification and valuation studies is that ecosystem structure/functions/processes are linear with independent characteristics and variables (i.e. size of the ecosystem, seasons, pressures and interactions between species) (Barbier et al., 2008; Koch et al., 2009). There is often a disjunct between the interactions of specific aspects and processes that allow for regulation (Sutherland et al., 2018). For example, carbon storage is driven by the carbon cycle and is influenced by remineralisation that is dependent on temperature, rainfall and a whole suite of other factors. This makes it hard to quantify service provision using a simple score. In addition, ecosystem functions reach a threshold or change drastically, and we still need to improve on incorporating non-linearity in ecosystem services which will give more realistic values that can be used in ecosystem based management practices and decision making (Barbier et al., 2008; Koch et al., 2009). Our study approach can however be used in future efforts to identify, quantify, value and manage ES in South African blue carbon habitats and provide a direction for policy and decision making to help preserve services that are pivotal in promoting and supporting human wellbeing.

3 QUANTIFICATION OF BLUE CARBON IN SOUTH AFRICA

3.1 Quantification of Blue Carbon Storage in a Warm-Temperate Estuary

Mbense SP, Adams JB, Rajkaran A, Johnson JL, Raw JL (accepted) Blue carbon storage comparing mangroves with salt marsh and seagrass habitats at a warm-temperate continental limit. In: Sidik F, Friess D, *Dynamic Sedimentary Environments of Mangrove Coasts*. Elsevier

3.1.1 Introduction

Coastal habitats such as mangroves, salt marshes and seagrasses have been recognized for their high productivity and biodiversity, however interest has recently been shifted towards the services these habitats provide and their responses to changing climate and sea level rise (Duarte et al., 2013b; Rogers et al., 2014a; Kelleway et al., 2017b). An important service includes blue carbon storage, and these habitats have been termed ‘blue carbon ecosystems’ or ‘blue forests’ (Mcleod et al., 2011). Blue carbon is the carbon stored in the sediment, living and non-living above and below ground biomass of mangrove, salt marsh and seagrass habitats (Nellemann et al., 2009). Although these habitats cover less than 2% of the area of the oceans worldwide, they capture up to 70% of carbon (~770 Gt), making blue forests more intensive carbon stores than other habitats (Nellemann et al., 2009; Macreadie et al., 2014). In these blue carbon habitats, carbon capture and storage depend on three main factors: high productivity and carbon dioxide conversion into plant biomass (Alongi, 2002); the ability to retain particulate organic carbon that is either autochthonous or allochthonous; and sedimentary conditions related to biogeochemistry which result in the slowed decay of organic material (Kelleway et al., 2017c).

Carbon storage in salt marsh and mangroves has not been clearly linked to a specific environmental or biological factor (Alongi, 2018). However, environmental factors such as variations in nutrients, salinity, moisture, acidity and sediment supply are important contributors to plant primary production and decomposition (Lovelock et al., 2007; Kelleway et al., 2016b). Carbon storage in salt marshes has also been linked to factors related to their hydro-geomorphic settings (Donato et al., 2011; Breithaupt et al., 2012; Saintilan et al., 2013; Kelleway et al., 2016b; Samper-Villarreal et al., 2016) including tidal elevation and frequency of inundation (Alongi, 2018), sediment depth, deposition and more importantly sediment type (Kelleway et al., 2016b). A large body of literature agrees that carbon storage is greater in fine-grained sediment because porosity and oxygen exchange is generally low resulting in lower sediment redox potential and rates of remineralization (Hedges and Keil, 1995; Kelleway et al., 2016b). Contrastingly, salt marsh with sandy sediments drain water faster, have a higher oxygen exchange capacity and rapid sediment carbon remineralisation (Crooks et al., 2002; Kelleway et al., 2016b). Long term carbon storage, however, depends on the input of organic matter and how quickly it decays. In addition, global carbon stock data has shown that carbon storage is higher in mature salt marshes as opposed to relatively new or restored salt marsh (Alongi, 2018). Higher carbon storage is evident in salt marshes where erosion is limited and there is little mangrove encroachment (Kelleway et al., 2016b; Wang et al., 2016; Alongi, 2018). In mangrove forests, carbon stocks have been linked to latitude, vegetation type, productivity rates and forest age (Radabaugh et al., 2018) along with tidal amplitude and forest elevation.

Despite their importance as carbon stores, large areas of these blue forests continue to be degraded. This is due to human induced pressures that have reduced their ability to sequester atmospheric carbon dioxide for climate change mitigation (Donato et al., 2011; Pidgeon et al., 2011; Fourqurean et al., 2012). Carbon offset mechanisms were introduced in 1997 to decrease global carbon dioxide emissions through the conservation of coastal vegetated habitats and this increased the assessment of carbon storage in these systems. As a result, substantial research effort has been put into blue carbon quantification with attempts to improve carbon sequestration and storage estimates over local, regional and global scales. While most of these studies have largely focused around North America, Europe and Australia (Duarte, 2017), Africa has also made some significant strides. Carbon assessments and trading projects have become increasingly popular in the Western Indian Ocean (WIO) Region (Bosire et al., 2016). The Mikoko Pamoja project in Gazi Bay, Kenya and The Blue Forests Initiative in Madagascar are two eminent projects. Their aim was to enhance mangrove ecosystem services (mainly carbon sequestration) through working with local communities and to implement carbon financing schemes while conserving mangroves (Wylie et al., 2016).

The region also boasts sound published scientific research on carbon storage potential in mangroves habitats. Steinke et al. (1995), Kirui et al. (2006), Kairo et al. (2008, 2009) and Bosire et al. (2012) focused mainly on biomass and productivity. Studies on mangrove carbon stock assessments have emerged for countries such as Madagascar (Jones et al., 2014; Benson et al., 2017), Mozambique (Sitoe et al., 2014; Shapiro et al., 2015; Stringer et al., 2015), Kenya (Gress et al., 2018) and Tanzania (Njana et al., 2015). However, blue carbon research for seagrass habitats is still lacking despite the region containing vast meadows that may contribute greatly to global carbon storage (Duarte et al., 2011). To date, only one quantitative assessment of sedimentary organic and inorganic carbon has been done for Zanzibar, mainland Tanzania and Mozambique (Gullström et al., 2017) in addition to a study by Dahl et al. (2016) on the effects of shading and grazing on carbon sequestration in seagrass meadows in Zanzibar. For salt marsh habitats, the situation is direr, with no record of published data on carbon stocks and accumulation rates in Africa, India and South America (Chastain et al., 2018). Most salt marsh studies are concentrated in Eastern and North-eastern United

States (Drake et al., 2015; Tripathee and Schäfer, 2015), Europe (Beaumont et al., 2014) and Australia (Saintilan et al., 2013; Saintilan et al., 2013; Rogers et al., 2014b, Macreadie et al., 2014, 2017b; Kelleway et al., 2016a, 2017).

In South Africa, blue carbon habitats occur in sheltered estuarine environments where little is currently known about carbon storage. Research on carbon only exists for terrestrial systems estimated as part of the South African National Carbon Sink Assessment (DEA, 2015; Turpie et al., 2017). Consequently, this study aimed to assess blue carbon storage at a warm temperate estuary in the Eastern Cape Province of South Africa and to compare these findings with global datasets. Its main objectives were to 1) quantify sediment carbon stocks following the blue carbon quantification methods of Howard et al. (2014); 2) determine differences in carbon storage between habitat types within the estuary (salt marsh, seagrass and mangroves); and 3) quantify carbon stocks of above and belowground biomass and establish allometric equations for salt marsh aboveground biomass.

Although much carbon in blue carbon ecosystems is stored in the sediment and vegetation carbon is considered a small component, it remains a significant carbon pool that requires accurate quantification. Aboveground biomass is usually measured using metrics that can be easily determined, related to total plant biomass (i.e. canopy volume and stem height) and used to form species-specific allometric equations (Kauffman and Donato, 2012; Howard et al., 2014; Radabaugh et al., 2017). The equations required for biomass carbon estimates are generally limited in the literature except for well-studied species (e.g. *Spartina alterniflora*) because creating these equations is laborious and destructive. Thus, when an allometric equation has been determined, it can be used in other studies to rapidly quantify biomass with less destruction (Radabaugh et al., 2017). This study will provide species-specific carbon data and allometric equations and will be the first to provide an equation for the salt marsh species *Salicornia tegetaria* (S. Steffen, Mucina & G. Kadereit) Piirainen & G. Kadereit in South Africa. The paucity of blue carbon prompted the present study, which is a component of a larger study on climate change and blue carbon habitats.

3.1.2 Materials and Methods

3.1.2.1 Study Site

Nxaxo Estuary is a permanently open estuary located at Wavecrest near Butterworth in the Eastern Cape (32°35'S; 28°31'E) within the warm temperate bioclimatic region (Harrison, 2004). This area also marks the southern distributional limit for mangroves along the east coast (Steinke et al., 1995; Adams et al., 2004). Nxaxo Estuary was selected for study as it has the largest area of salt marsh (10.9 ha) co-occurring with mangroves of a similar area (9.5 ha) (salt marsh: mangrove ratio < 1) (Adams et al., 2016) with patches of seagrass.

This estuary has four salt marsh species including *Salicornia tegetaria* and *S. natalensis* (Bunge ex. Ung-Sternb.) in the lower intertidal area and *Bassia diffusa* (Thunb.) Kuntze and *Sporobolus virginicus* (L.) Kunth in the supratidal area (Hoppe-Speer, 2013). There are three mangroves species, namely *Bruguiera gymnorrhiza* (L.) Lamk., *Rhizophora mucronata* (L.) and *Avicennia marina* (Forssk.), the latter of which is dominant and found fringing the estuary water channel from the mouth to the upper reaches (Hoppe-Speer and Adams, 2015). There are also two species of seagrass, *Halophila ovalis* (R. Brown) J. D Hooker and the endangered lower intertidal seagrass species *Zostera capensis* Setchell (Cape dwarf-eelgrass). *Z. capensis* co-occurs with *A. marina* in most estuaries along the east coast owing to their preference for saline habitats. At Nxaxo Estuary, *Z. capensis* may be involved in the facilitation of mangrove recruitment as it sometimes found growing with germinating propagules (Adams, 2016). *Avicennia marina* trees at the estuary are stunted due to previous harvesting, heavy browsing and trampling by cattle during drought periods. Cattle freely enter the area from neighbouring rural communities (Hoppe-Speer and Adams, 2015). Browsing pressure was removed for a short period when maize fields were planted adjacent to the mangrove area which prevented cattle from entering. Within this time, the trees showed some signs of recovery by increasing their growth rate and seedling establishment (Mbense, 2017). The mangrove population at this estuary consists of seedlings (< 50 cm), juveniles (101-150 cm) and adults (> 151 cm). Some adult trees along the main channel are taller (3-4 m in height) and the stunted adults occur more landwards (Hoppe-Speer and Adams, 2015).

This study focused on the quantification of a salt marsh (*S. tegeteria)* (the most abundant shrub) and a seagrass *Z. capensis* species. Two plots were strategically selected within the *Salicornia* dominated salt marsh and *Z. capensis* patches. The salt marsh plots were located behind the mangrove area while the seagrass plots were in front of the mangroves on an island close to the main estuary channel. For mangroves, five already existing *A. marina* sites along the main channel were used for this study. Site 1 (a combination of three sites in previous studies) occurred in the upper reaches of the estuary. One part of the site comprised mainly adult trees with few seedlings and juveniles. Another site was located close to a smaller channel and contained mostly saplings and adult trees. Site 2, situated more landward, was waterlogged and comprised juveniles with some adults and a stand of *B. gymnorrhiza* trees. Site 3 was located on the island with mostly adult trees and Site 4 was close to the mouth of the main channel with a population made up of both adult and juvenile trees.

3.1.2.2 Sediment Collection

Two salt marsh and seagrass plots were randomly chosen within the estuary; two 25 m^2 quadrats were laid in each plot. Within each quadrat, six sediment cores of approximately 1 m depth were extracted using PVC tube (11 cm diameter) at different areas within the quadrat. The core was divided into 8 depth intervals (0-5 cm, 5-10 cm, 10-20 cm, 20-30 cm, 30-40 cm, 40-50 cm, 50-70 cm, and 70-100 cm) (Plate 3.1.1A). A subsample was taken in the middle of each depth interval by inserting a syringe into pre-drilled holes in the PVC tube (Plate 3.1.1B). Individual soil increments were packed into small, pre-labelled plastic bags and transferred into a cooler box for storage until further analysis in the laboratory.

Each mangrove site was divided into six (5 m^2) plots placed in different zones, the lower being along the bank of the estuary and the upper in the landward edge of the mangrove forest. One sediment core was extracted per plot in the middle of each site using a Russian Peat Corer (Plate 3.1.1C). Additionally, three cores were extracted in the area between the lower and upper zone. Each core was divided into four sections at depths intervals of 0-15 cm, 15-30 cm, 30-50 cm and 50-100 cm. Sediment core samples were taken in the middle of each site at 25 cm, 50 cm, 75 cm and 1 m depth for redox measurements and sediment analyses. The samples were also stored in plastic bags in cooler boxes and processed in the laboratory.

Plate 3.1.1. A) An example of an extracted sediment core using a PVC pipe with pre-drilled holes. **B**) Extracting a subsample from the core using a syringe. **C**) A sediment core extracted from a mangrove plot using a Russian Peat Corer. Photos: S. Mbense and J. Raw, 2017.

3.1.2.3 Sediment Analysis

Dry Bulk Density and Organic Matter

Each sediment subsample for each depth increment was dispensed into a glass petri-dish and oven dried at 60°C until a constant weight was reached. Dry bulk density (DBD) was then determined by dividing the mass of the dried soil (g) by the original volume sampled (cm^3) because the sample was taken with a syringe and volume was measured directly (where 1 cc = 1 cm^3) (Howard et al., 2014). The dried sediment was further divided into two subsamples, one was used to determine organic matter using the percent mass loss on ignition (LOI), which quantifies percentage organic matter (OM%). The loss on ignition procedure is a cheap and effective predicator of sediment organic carbon (Craft et al., 1991, Macreadie et al., 2013). The ashing method of Briggs (1977) was used to determine OM%. This involved placing the already dried sediment subsamples in a muffle furnace or an ashing oven for 8 hours at 550°C. Percentage organic matter was calculated using dry mass (*Md)* and mass after ashing (*Ma)* using the following equation:

$$\left(\frac{Md - Ma}{Md}\right) \times 100$$

where *Md* is dry mass and *Ma* is mass after ashing.

Organic and Inorganic Carbon Analysis

The second subsample was homogenized into a fine powder using a mortar and pestle and used to determine percentage organic carbon (C_{org}%) through elemental analysis. This procedure measures total carbon including organic and inorganic carbon; samples therefore required correction for inorganic carbon. Inorganic carbon as carbonates or calcium carbonates ($CaCO_3$) are found in shells often abundant in seagrasses. Before sample correction for organic carbon, the Champagne test (Jaschinski et al., 2008) was conducted to determine if samples contained carbonates. This involved testing some sediment subsamples by treating them with dilute 1 M hydrochloric acid (HCL). Visual observations showed that the seagrass sediment samples contained specks of calcium carbonate shells and effervesced immediately upon addition of acid; correcting for carbonates was therefore deemed necessary for all seagrass sediment samples. There were no visible shells in the salt marsh sediment and no effervescence occurred when some samples were treated; all salt marsh sediment samples were nevertheless acidified for consistency. The acidification process included adding dilute acid to a known weight of homogenized subsample until the acid covered the sample. The mixture was shaken manually for approximately 10-15 minutes to break any large clumps so the carbonates were properly removed. When the effervescing ended, the samples were left overnight (18 hours) and the same process was repeated until no further effervescence occurred. The remaining acid was removed using a plastic pipette and the samples were rinsed three times with distilled water and left to dry overnight at 60°C. The samples were weighed again and the amount of calcium carbonate (In_{org}%) in each sample calculated using the methods in Howard et al. (2014). Inorganic carbon in salt marsh and mangroves was undetectable (mass remained the same after acidification) and thus not reported.

After acidification and re-drying, samples were prepared for elemental analysis following standard protocol. For samples with organic matter between 0.5% and 1%, an amount of 25-30 mg was weighed, and for sediment with organic matter between 2% and 11%, an amount of 9-10 mg was weighed using a fine scale (UniBloc, Shimadzu, AUWW220D). Samples were then packed into pressed tin capsules (8 x 5 mm) (Elemental Microanalysis, UK). Three replicates were analysed for total carbon using a CHN analyser at the Wetland Biochemistry laboratory at the Louisiana State University in the United States.

Carbon Density and Total Carbon

Carbon density (CD) was calculated using organic carbon (C_{org}%) and dry bulk density (DBD) for each depth interval from the surface to 1 m depth. The carbon density of each interval of each core was calculated as follows:

$$\text{Soil carbon density (g cm}^{-3}\text{)} = \text{dry bulk density (g cm}^{-3}\text{)} * (C_{org}/100)$$

Total Carbon Stocks for Salt Marsh, Seagrass and Mangrove Soil

The amount of carbon per core section was determined by multiplying each soil carbon density value by the thickness of the sample interval (cm) where:

$$\text{Amount of carbon in core section (g cm}^{-2}\text{)} = \text{Soil carbon density (g cm}^{-3}\text{)} * \text{thickness interval (cm).}$$

The amount of carbon per core was determined as the sum of carbon per section over the total sampling depth (~1 m). The total core carbon was converted into carbon stock assessment units where:

$$\text{Total core carbon (MgC ha}^{-1}\text{)} = \text{Total carbon for all cores (g cm}^{-3}\text{)} * 1\ \text{Mg}/1000000\ \text{g}) *(100000000\ \text{cm}^2/1\ \text{ha}).$$

This was repeated for all cores in the two quadrats for both the salt marsh and seagrass cores and the average amount of carbon per plot was determined by summing all cores and dividing by the total number of cores per quadrat. The total amount of carbon per ecosystem was determined by multiplying the average carbon value (MgC ha^{-1}) for all cores by the area of the ecosystem in hectares to determine total carbon for the sediment pool for each habitat (MgC) (Howard et al., 2014).

3.1.2.4 Above and Belowground Biomass Collection

For the vegetative carbon component, the same 25 m^2 quadrats as mentioned above were divided into smaller quadrats (1 m^2) because Howard et al. (2014) suggested that for certain vegetation types (e.g. salt marsh shrubs and seagrasses), a small area is required to capture more variation and accuracy in quantification. They also recommended that a similar method be used to determine biomass carbon for salt marsh shrubs. However, in the field, we found that the selected salt marsh species (*S. tegeteria*) was growing flat and not in shrub form (Plate 3.1.2B) and we were unable to measure crown diameter, width, volume and area.

Within the smaller quadrats, three replicates of living aboveground and belowground material were collected by inserting the same PVC tube (11 cm diameter) into the sediment surface (Plate 3.1.2 A, B). Care was taken to ensure the core was inserted deep enough to harvest and capture both the aboveground and below ground components. For salt marsh vegetation, any remaining sediment was removed, and plant material was put in labelled plastic bags. For the seagrass, plant material in the core was transferred into a mesh sieve, washed until free of sediment and packed into pre-labelled plastic bags and stored in cooler boxes. Seagrass biomass should be separated immediately after collection into above- and belowground components but this was not possible due to limited time in the field. The processing of biomass samples began within 2 days of collection as seagrasses have a high decay rate.

For the mangroves, aboveground biomass included living trees, leaf litter and pneumatophores. Eighteen living trees were chosen at each site (n = 3) in each plot based on the average tree height within the plot. For each tree, height and diameter at breast height (DBH) were measured and wood samples were collected from the nearest branch to 1.3 m using a small axe (Plate 3.1.2C). A subsample (from the branch) that weighed ~ 26.7 g was taken for laboratory analysis. Three smaller plots (1 m^2) were measured within the larger 5 m^2 area for the collection of leaf litter. Three subplots (50 cm^2) were demarcated inside the 1 m^2 plot and 50 individual pneumatophores were collected from each subplot through cutting at the base.

3.1.2.5 Above and Belowground Plant Material Analysis

In the laboratory, salt marsh plants were separated into aboveground material and roots. The height of each aboveground *S. tegetaria* branch was measured to the nearest 0.1 cm. The method of plant measurement usually depends on the growth pattern of the species. For succulent shrubs with a diverse growth pattern, measurements of individual stems within a shrub is required to properly determine the stem height and biomass relationship (Radabaugh et al., 2017). Stems slightly dead or senescing were also included in measurements. Following height measurements, the plant material was oven-dried to a constant weight for approximately 72 hours at 60°C to determine biomass (as dry weight).

For seagrasses, the plant material was checked carefully for epiphytes and as none were present, epiphyte removal and measurements were not required. The seagrass was separated into shoots and rhizomes, the shoots were measured for height, and both components were dried using the same protocol as for the salt marsh to determine biomass as dry weight. The above- and belowground carbon of salt marsh and seagrass were estimated by multiplying the biomass per area (g cm^{-2}) by the organic carbon determined from CHN analysis.

Plate 3.1.2. Above and belowground biomass collection within the smaller quadrats for **A**) seagrass *Zostera capensis*; **B**) salt marsh *Salicornia tegetaria*; **C**) and the mangrove *Avicennia marina* at the Nxaxo Estuary. Photos: S. Mbense and J. Johnson, 2017.

The formula from Chave et al. (2014) was used to calculate mangrove aboveground biomass:

$$ABG = 0.0559 * (\rho D^2 H)$$

where AGB is aboveground biomass (kg), ρ is wood density (g cm^{-3}), D is DBH (cm) and H is height (m).

Wood density was calculated from dry weight and volume of fresh wood.

$$\text{Wood density (g cm}^{-3}\text{)} = \text{Dry weight (g)/volume of fresh wood (cm}^{3}\text{)}$$

where the volume of fresh wood was measured by immersing the fresh sample (after weighing) into a beaker with 1 L of water and the difference in mass was measured as:

$$\Delta \text{ Mass (g)/water density (g cm}^{-3}\text{)}$$

Each sample wood density was determined, and above ground biomass was calculated using the equation above. Each value (representing each tree) was added to determine the total biomass for all five sites. Total biomass was then multiplied by the organic carbon value obtained from CHN analysis.

Organic and Inorganic Carbon

A subsample of dried above- and belowground vegetation was ground to a fine powder using a mortar and pestle. The champagne test was performed on a few seagrass and salt marsh samples and there was no effervescence rendering sample acidification unnecessary. Each subsample of plant material was weighed to be between 4 mg and 4.5 mg and then packaged into small tin capsules (8 x 5 mm) (Elemental Microanalysis, UK). Like the sediment, three replicates samples of the subsample were analysed for total carbon using a CHN analyser at the Wetland Biochemistry laboratory at the Louisiana State University in the United States.

3.1.2.6 Data Analysis

Analysis of variance (ANOVA) was used to test for significant differences in sediment parameters (organic matter, bulk density, organic and inorganic carbon and carbon density) with changes in depth (depth section as a fixed factor) using the MASS package in RStudio, R version 3.5.1 (Copyright 2018, The R Foundation for Statistical Computing).

Equations were created to determine the relationship between sediment organic matter (LOI%) and organic carbon concentration (C_{org}%) by plotting the values against each other. Natural logarithmic transformations were performed for *Z. capensis* sediment values to improve the fit of the linear relationship between organic matter and organic carbon. The values were further transformed from negative to positive using the modulus. A similar linear equation was conducted for salt marsh to determine the relationship between stem height and stem biomass (as dry weight).

3.1.3 Results

Sediment cores for the salt marsh species *S. tegetaria,* seagrass *Z. capensis* and mangrove *A. marina* penetrated to a maximum of 100 cm. Salt marsh sediment dry bulk density (DBD) ranged 0.43-0.76 g cm^{-3} with a mean of 0.62 ± 0.13 g cm^{-3}; DBD was correlated and increased significantly with depth ($P < 0.05$, r = 0.63) (Figure 3.1.1 A, Table 3.1.2). For sediment underlying the *Z. capensis* seagrass beds, the bulk density ranged 0.68-0.98 g cm^{-3} with a mean of 0.89 ± 0.1 g cm^{-3}. In *A. marina* sediment, dry bulk density ranged 0.79-0.89 g cm^{-3} with a mean of 0.81 ± 0.04 g cm^{-3} (Figure 3.1.1A). In both *Z. capensis* and *A. marina* sediment, DBD showed no significant trend or correlation with depth (Table 3.1.2). Sediment organic matter for salt marsh ranged from 4% at the surface and decreased to 0.8% at the bottom with a mean of 1.7 ± 0.34% (Figure 3.1.1B). Organic matter in salt marsh also decreased significantly with depth ($P < .05$) (Figure 3.1.1B, Table 3.1.2). *Z. capensis* sediment OM% ranged 1.58-3.06% (Figure 3.1.1B) and varied with depth from the surface to the bottom (Table 3.1.2). Mangrove sediment organic content was lower than both salt marsh and seagrass with a mean of 0.63 ± 0.15% and showed no significant change with depth (Figure 3.1.1B, Table 3.1.2).

Sediment organic carbon showed no significant change with depth in all three habitats (Figure 3.1.1C, Table 3.1.2). Organic carbon ranged 0.85-1.96% with means of 0.86 ± 0.07%, 1.4 ± 0.29% and 1.96 ± 0.31% for seagrass, salt marsh and mangroves respectively. In the seagrass sediment, inorganic carbon ranged 0.08-0. 2% and showed a slight decrease with depth but this was not significant (Table 3.1.1, 3.1.2). Sediment nitrogen for all three habitats ranged 0.06 to 0.31% and showed no significant change with depth. The ratio between carbon and nitrogen ranged 6.91-16.5 and there was no consistent change in C: N with depth in all three habitats. Sediment carbon density was higher in mangroves (1.55 ± 0.09 $g.cm^{-3}$) and much lower in both salt marsh and seagrass (~0.01 $g.cm^{-3}$). In all three habitats, sediment carbon density also showed no significant trend with depth.

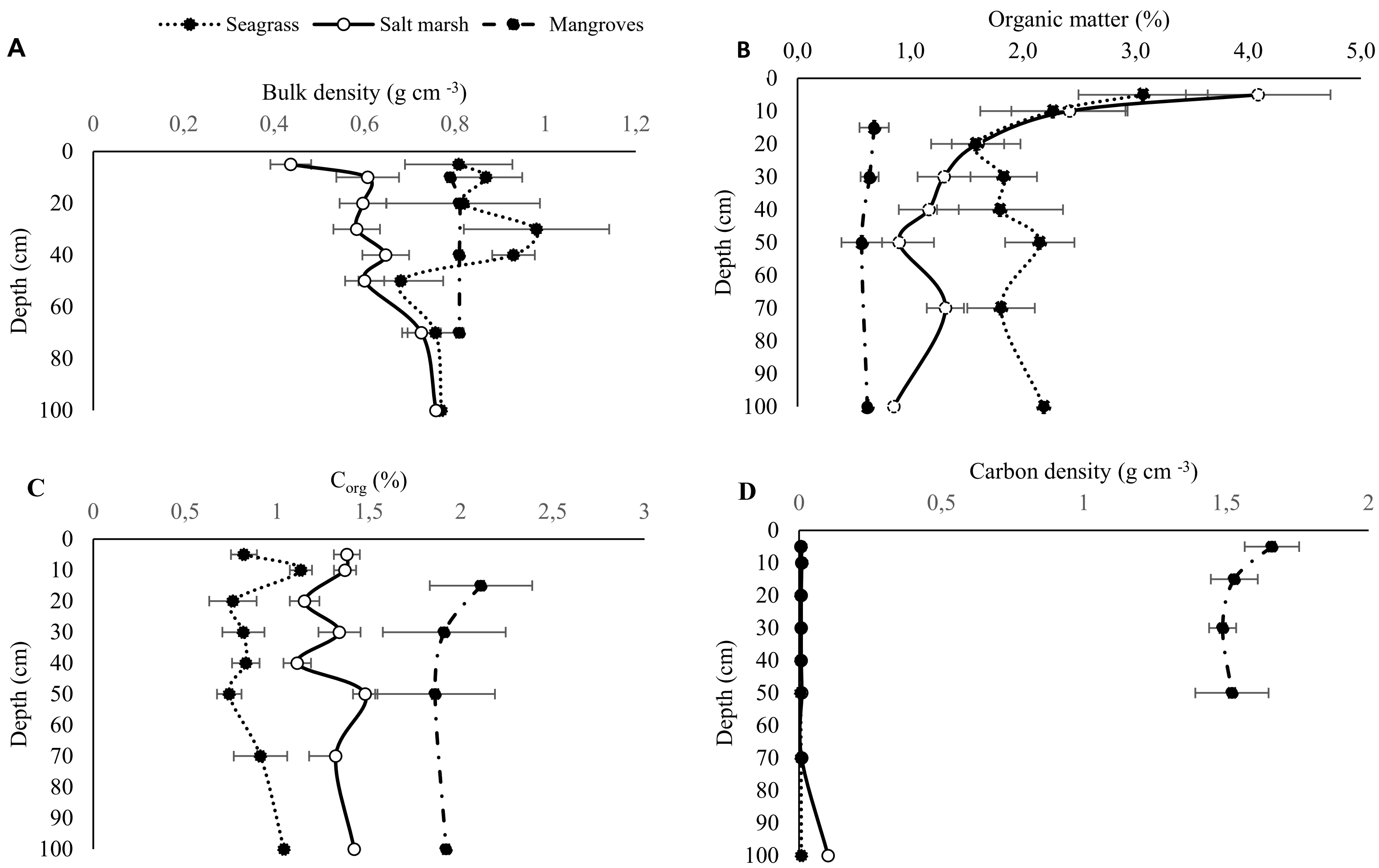


Figure 3.1.1. A) Dry bulk density (g cm^{-3}), **B**) organic matter (OM%), **C**) organic carbon (C_{org} %) and **D**) carbon density (g cm^{-3}) depth profiles for sediment cores extracted in salt marsh, seagrasses and mangroves at Nxaxo Estuary (Mean ± SE, n = 9 mangroves, n = 12 salt marsh and seagrass).

Table 3.1.1. Inorganic carbon (%), nitrogen (%) and the ratio between organic carbon and nitrogen of salt marsh, seagrass and mangrove sediment cores at different depths at Nxaxo Estuary (Mean ± SE).

Depth	Inorganic carbon (%)			Nitrogen (%)			Organic carbon: Nitrogen ratio		
	Salt marsh	Seagrass	Mangroves	Salt marsh	Seagrass	Mangroves	Salt marsh	Seagrass	Mangroves
0-5	Undetectable	0.22 ± 0.06		0.08 ± 0.04	0.07 ± 0.01		16.54	11.7	
5-10	Undetectable	0.28 ± 0.1		0.09 ± 0.05	0.09 ± 0.01		15.08	12.02	
10-20	Undetectable	0.16 ± 0.05	Undetectable	0.1 ± 0.06	0.06 ± 0.01	0.17 ± 0.02	11.24	11.9	12.4
20-30	Undetectable	0.17 ± 0.06		0.09 ± 0.05	0.07 ± 0.01		14.61	12.2	
30-40	Undetectable	0.11 ± 0.02	Undetectable	0.07 ± 0.04	0.07 ± 0.01	0.21 ± 0.01	14.97	12.6	9.1
40-50	Undetectable	0.17 ± 0.08		0.09 ± 0.06	0.08 ± 0.01		16.34	11.6	
50-70	Undetectable	0.08 ± 0.05	Undetectable	0.08 ± 0.05	0.06 ± 0.01	0.16 ± 0.07	15.37	15.3	11.6
70-100	Undetectable	0.13 ± 0.06	Undetectable	0.15 ± 0.09	0.08 ± 0.02	0.31 ± 0.02	9.29	12.9	6.19

Table 3.1.2. Results of the Analysis of Variance (ANOVA) for salt marsh, seagrass and mangrove sediment core variables (dry bulk density, organic content, organic and inorganic carbon content, nitrogen and carbon density), with depth as a fixed factor.

		df	SS	F-ratio	P	r
Sediment core variables						
Dry bulk density ($g\ cm^{-3}$)	Salt marsh	7	52.95	23.83	**<0.05**	0.63
	Seagrass	7	40.7	15.93	0.13	0.46
	Mangrove	3	15.32	55.41	0.1	0.36
Organic matter (OM%)	Salt marsh	7	4.62	16.6	**<0.05**	0.62
	Seagrass	7	4.99	16.2	**<0.05**	0.14
	Mangrove	3	8.61	1.47	0.22	0.23
Carbon content (C_{org}%)	Salt marsh	7	12.6	1	0.49	0.02
	Seagrass	7	31.19	0.99	0.51	0.27
	Mangrove	3	56.23	8.35	0.87	0.36
Inorganic carbon content (In_{org}%)	Salt marsh	7	N. D	N. D	N. D	N. D
	Seagrass	7	32.29	18.47	0.07	0.61
	Mangrove	3	N. D	N. D	N. D	N. D
Nitrogen (%)	Salt marsh	7	33.41	21.16	0.62	0.25
	Seagrass	7	19.2	5.98	0.08	0.64
	Mangrove	3	54.89	20.67	0.82	0.34
Carbon density ($g\ cm^{-3}$)	Salt marsh	7	20.9	17.59	0.49	0.44
	Seagrass	7	55.2	9.91	0.51	0.48
	Mangrove	3	45.98	11.27	0.28	0.38

Sediment organic carbon values were plotted against organic matter. A linear regression analysis determined the relationship in salt marsh (1), seagrass (2) and mangroves (3) as:

1) C_{org} (%) = 0.7898(LOI%) + 0.0438

2) $\ln C_{org}$ (%) = 0.9799 ln (LOI%) + 0.0438

3) C_{org} (%) = 0.3358(LOI%) + 0.1884

There was a strong relationship between OM and C_{org} in salt marsh sediment (R^2 = 0.88) and approximately 78% of the organic matter was organic carbon (Figure 3.1.2A). In both mangrove and seagrass, the relationship between C_{org} and OM was weak (Figure 3.1.2B,C).

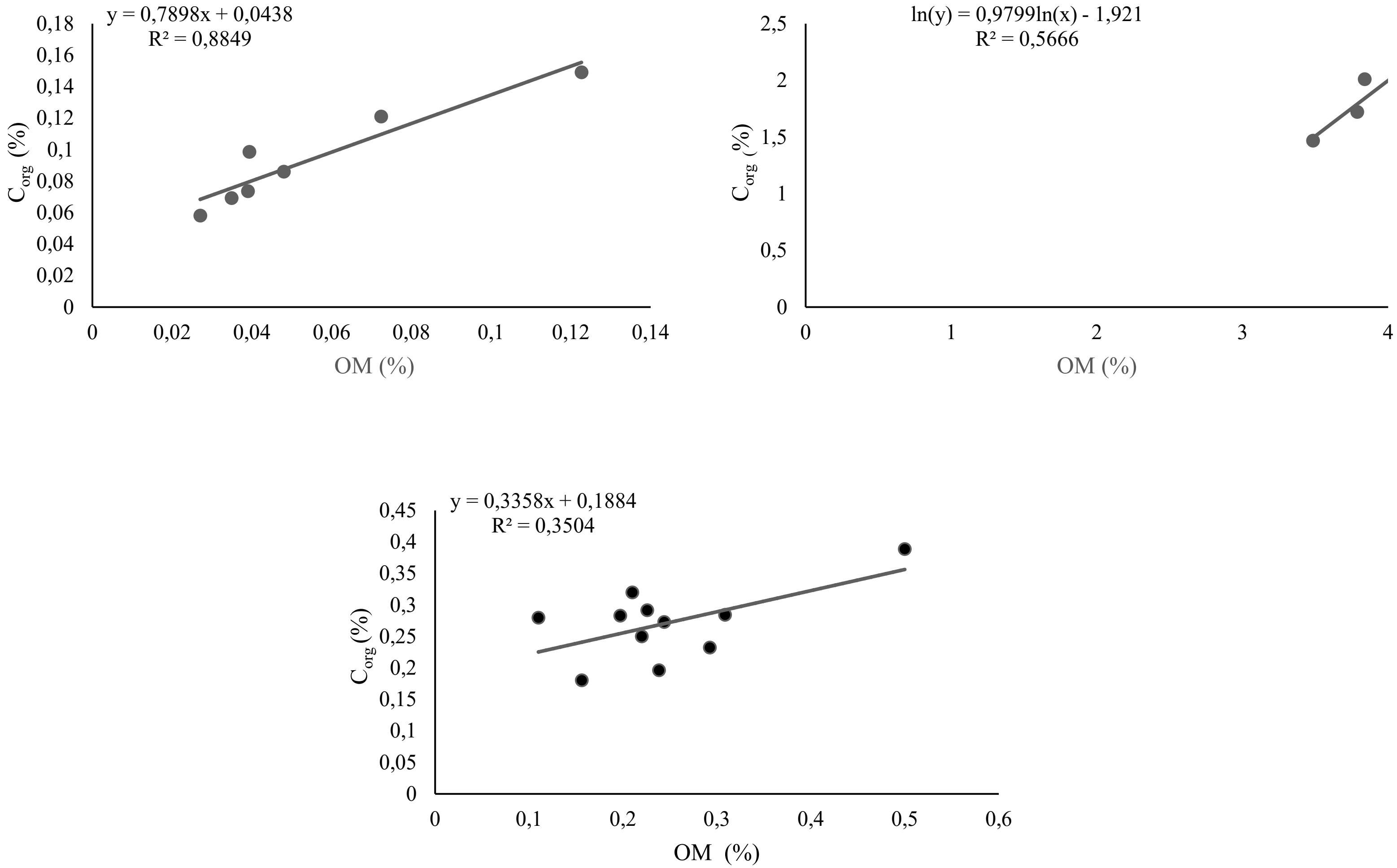


Figure 3.1.2. The relationship between sediment carbon concentration (C_{org}%) and organic content (OM%) in **A**) salt marsh **B**) seagrass and **C**) mangroves at the Nxaxo Estuary.

Total sediment organic carbon (to 1 m depth) was significantly higher in mangroves (228.05 ± 27.99 Mg C ha^{-1}) compared to salt marsh (2.61 ± 0.19 Mg C ha^{-1}) and seagrass (1.67 ± 0.81 Mg C ha^{-1}) (Table 3) (F = 180.92, df = 2, $P < 0.0001$). The total carbon for the entire sediment carbon pool was also significantly greater in mangroves (2166.48 ± 265.91 Mg C) compared to salt marsh (28.5 ± 2.11 Mg C) and seagrass (0.06 ± 0.01 Mg C) (F = 78.12, df = 2, $P < 0.0001$).

Table 3.1.3. Total organic carbon sediment stocks for salt marsh and seagrasses for 1 m core depth at Nxaxo Estuary (Mean ± SE).

	Total carbon (Mg C ha^{-1})	**Area (ha)**	**Total carbon for soil carbon pool (Mg C)**
Salt marsh	2.61 ± 0.19	10.9	28.55 ± 2.1
Seagrass	1.67 ± 0.01	0.04	0.06 ± 0.01
Mangroves	228.05 ± 27.99	9.5	2166.48 ± 265.91

Total aboveground biomass carbon was significantly higher in mangroves (40.1 ± 0.81%) compared to salt marsh (25.3 ± 0.62%) and seagrass (22.49 ± 0.75%) (Figure 3.1.3A) (F = 21, $P < 0.001$). Nitrogen composition was similar in all three habitat types (~ 1%) and there were no noticeable differences in nitrogen composition between above- and belowground biomass.

Salicornia tegetaria had a good degree of fit ($R^2 = 0.707$) between stem height and biomass even though it is a herbaceous shrub that generally has a high degree of branching. Stem height and biomass ranged 11-60 cm and 1-11 g respectively (Figure 3.1.4).

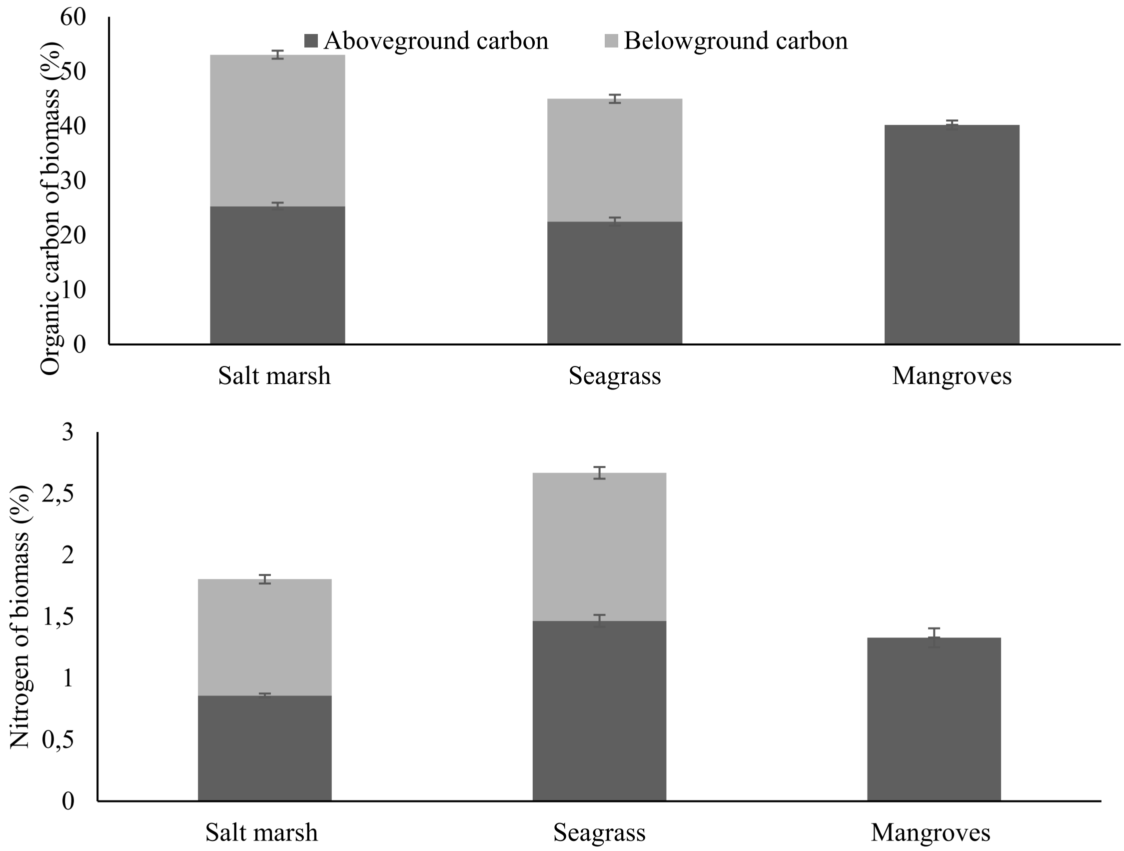


Figure 3.1.3. Percentage organic carbon and nitrogen for salt marsh, seagrass and mangrove plant biomass at Nxaxo Estuary (Mean ± SE).

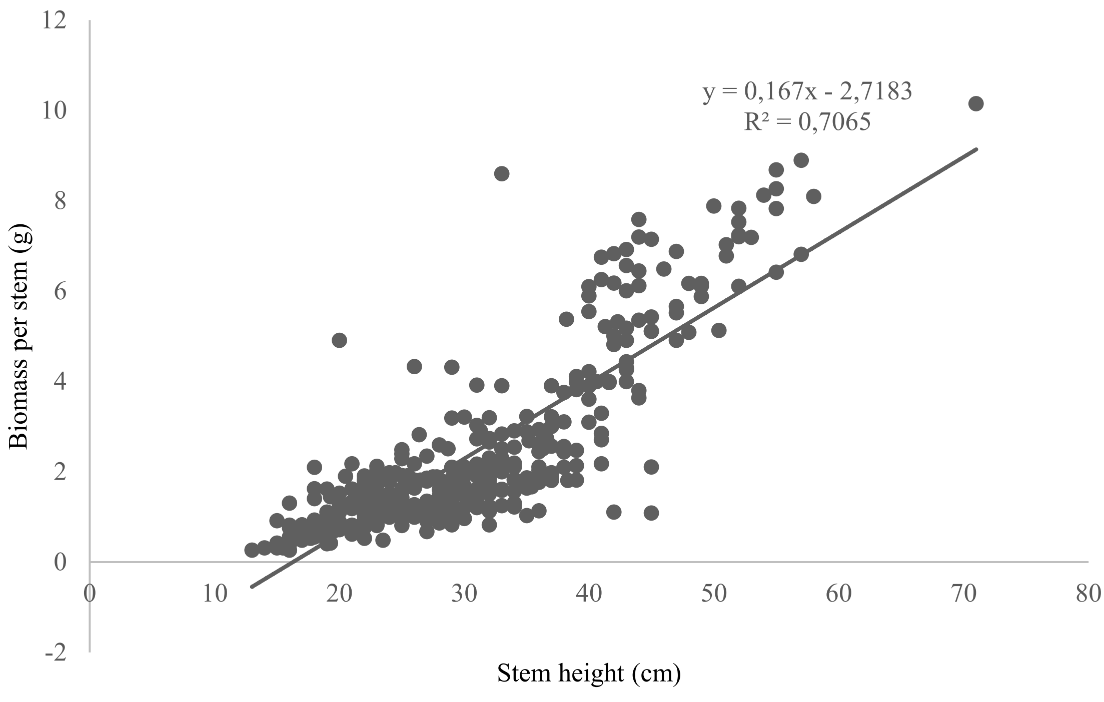


Figure 3.1.4. The relationship between biomass (g) (as dry weight) versus height (cm) for salt marsh *Salicornia tegetaria* at Nxaxo Estuary.

For salt marsh, above- and belowground biomass ranged 715-1157.62 g m^{-2} and 604-832 g m^{-2} respectively. Belowground biomass (714 ± 16.7 g m^{-2}) was significantly higher than seagrass (314.6 ± 76.65 g m^{-2}) ($F = 143.51$, df = 1, $P < 0.0001$). For both salt marsh and seagrasses, there were no significant differences between the quadrats. Mangroves had a higher aboveground biomass (6526.3 ± 117 g m^{-2}) compared to salt marsh (872.4 ± 39.81 g m^{-2}) and seagrasses (354.5 ± 86.47 g m^{-2}) ($F = 4.41$, df = 2, $P < 0.0001$).

Total biomass carbon storage estimates were influenced by biomass estimates and carbon content. Mangrove biomass carbon (71.45 ± 2.47 Mg ha^{-1}) was higher than salt marsh (2.5 ± 0.05 Mg ha^{-1}) and seagrass (1.26 ± 8.1 x10^{-5} Mg ha^{-1}) ($F = 59.99$, df = 2, $P < 0.0001$). As a result, overall carbon storage was significantly higher in mangroves (678.75 ± 17.37 Mg C) compared to salt marsh (27.63 ± 0.62 Mg C) and seagrass (0.1 ± 3.2 x 10^{-6}) ($F = 38.2$, df = 2, $P < 0.0001$) (Table 3.1.4). Although the areal extent of mangroves and salt marshes was similar at Nxaxo Estuary, mangroves stored the most carbon and seagrasses the least (expected due to small area).

Table 3.1.4. Above and below ground biomass (g m^{-2}), total vegetative carbon (Mg C ha^{-1}) and total carbon for the carbon pool for seagrass, salt marsh and mangroves at Nxaxo Estuary (Mean ± SE). *Mangrove above ground biomass included living trees, leaf litter and pneumatophores.

	Biomass (g m^{-2})		**Total biomass carbon (Mg C ha^{-1})**	**Area (ha)**	**Total carbon for vegetative carbon pool (Mg C)**
	AGB	*BGB*			
Salt marsh	872.4 ± 39.81	714 ± 16.7	3.96 ± 0.78	10.9	39.66 ± 1.05
Seagrass	354.5 ± 86.47	314.6 ± 76.65	1.26 ± 8.1x10^{-5}	0.04	0.1 ± 3.2x10^{-6}
Mangroves	*6526.3 ± 117*	ND	71.45 ± 2.47	9.5	678.45 ± 24.15

3.1.4 Discussion

This study is the first quantitative report on carbon stored in salt marsh, seagrasses and mangroves habitats in a South African estuary. Numerous international studies have reported that carbon storage and associated sediment properties are remarkably different between blue carbon habitats (Radabaugh et al., 2017; Schile et al., 2017; Cusack et al., 2018; Lewis et al., 2018). This study found that mangroves had the highest sediment organic carbon stocks, followed by salt marshes and seagrasses, an outcome in agreement with comparative studies showing mangroves to have the largest carbon stocks (Chmura et al., 2003; Pendleton et al., 2012; Siikamäki et al., 2012; Radabaugh et al., 2017; Schile et al., 2017). The large difference in carbon stock between the mangrove and the salt marsh habitats was however not expected since Kelleway et al. (2016a), Lewis et al. (2018) and Hayes et al. (2017) recently showed no notable differences between mangrove and salt marsh carbon storage in Australia. This was attributed to mangroves at their limits having slower growth (with similar ecological functions to young forests) and therefore accumulating carbon stocks equivalent to salt marshes (Kelleway et al., 2017c).

3.1.4.1 Sediment Carbon Storage Capacity of Blue Carbon Habitats at Nxaxo Estuary and Comparisons with Other Studies

Sediment organic matter was low compared to other studies for salt marshes (Macreadie et al., 2013; Radabaugh et al., 2017), seagrasses (Lavery et al., 2013; Rozaimi et al., 2016; Potouroglou, 2017; Green et al., 2018) and mangroves (Radabaugh et al., 2017), and decreased significantly with depth in seagrasses and salt marshes, which was evident in other studies (Serrano et al., 2012; Potouroglou, 2017; Hayes et al., 2017). Organic content generally decreases with depth because of the diagenetic loss of carbon due to sediment remineralisation by bacteria and decomposition in the soil profile with time (Saintilan et al., 2013). Dry bulk density was lower for seagrasses compared to the global mean (Fourqurean et al., 2012) and other studies (Röhr et al., 2016; Potouroglou, 2017; Green et al., 2018), but similar values were reported for salt marsh (Beaumont et al., 2014; Radabaugh et al., 2017; Lewis et al., 2018; Hayes et al., 2017) and mangroves (Radabaugh et al., 2017; Hayes et al., 2017; Lewis et al., 2018). According to Stringer et al. (2015), mangrove sediment carbon density (CD) is usually not discussed in carbon stock studies but Chmura et al. (2003) found it to range from 0.023 to 0.114 g cm^{-3}. Our values for CD were higher than this range and other studies (Jones et al., 2014; Benson et al., 2018; Stringer et al., 2015). This may indicate that our mangrove sediment is mineral rich as this is associated with low organic carbon content and high bulk density while peat or organic rich sediment is associated with low bulk density and high organic carbon (Stringer et al., 2015). Sediment organic carbon corresponded with other values reported for seagrasses (Lavery et al., 2013; Potouroglou, 2017; Howard et al., 2017a; Gullström et al., 2017; Schile et al., 2017; Green et al., 2018) and mangroves (Jones et al., 2014; Stringer et al., 2015; Radabaugh et al., 2017; Hayes et al., 2017; Lewis et al., 2018). In contrast, organic carbon was much lower in the salt marsh sediment compared to values reported by Drake et al. (2015) and Lewis et al. (2018) but was comparable to Radabaugh et al. (2017) and Hayes et al. (2017).

As expected, total sediment carbon stocks in our study were much lower than the global means for salt marshes (Duarte et al., 2013b; Alongi, 2018) along with other studies (Macreadie et al., 2013; Drake et al., 2015; Kelleway et al., 2016; Radabaugh et al., 2017; Hayes et al., 2017; Lewis et al., 2018;). Total sediment stocks were also lower for seagrasses compared to other studies (e.g. Fourqurean et al., 2012; Lavery et al., 2013; Rozaimi et al., 2016; Gullström et al., 2017; Potouroglou, 2017; Radabaugh et al., 2017; Schile et al., 2017; Alongi, 2018; Green et al., 2018). Total carbon stocks in mangroves were similar to values reported by Radabaugh et al. (2017); Schile et al. (2017) and Hayes et al. (2017), higher than Lewis et al. (2018) and Cusack et al. (2018) but still lower than the global mean (Alongi et al., 2018) and other studies (Jones et al., 2014; Stringer et al., 2015; Gress et al., 2016; Benson et al., 2018).

The relationship between C_{org}% and OM% was strong for salt marsh sediment and like the relationship ($R^2 = 0.99$) reported by Craft et al. (1991) and Howard et al. (2014), we also found that 78% of the organic content was carbon, which was more than double the 46% reported by Howard et al. (2014). This indicated the importance of region-specific data because using the conversion factor of 0.46 would have further reduced the carbon stock value. For seagrass, this relationship was weak and required natural logarithmic transformation to increase its strength, but it was still less than the global value ($R^2 = 0.56$ vs $R^2 = 0.96$) (Fourqurean et al., 2012; Howard et al., 2014) but like that of Green et al. (2018). In mangroves, the relationship was also weak compared to the global equation ($R^2 = 0.59$ vs $R^2 = 0.35$) (Kauffman et al., 2011; Howard et al., 2014).

Table 3.1.5. Summary data from literature for organic matter (OM%), organic carbon (C_{org}%) and total carbon (Mg ha^{-1}) for salt marshes. All studies determined OM% using LOI (Loss on ignition) and C_{Org} (%) using various methods.

Region/location	Species	Latitude	Longitude	Core Depth (cm)	OM (%)	Corg (%)	Total carbon stock (Mg ha^{-1})	Reference
United States	Multiple species	82°W	27°N	50	9.4	4.7 [a b]	66	Radabaugh et al. (2017)
United States	Multiple species	85°W	29°N	15	5.1	Data not available	29.8	Macreadie et al. (2013)
United States	Multiple species	70°W 70°W	43°N 42°N	60	Data not available	16.3 [a]	Data not available	Drake et al. (2015)
South Africa	*Salicornia tegeteria*	28°E	32°S	100	1.7	1.3	2.61	This study
Australia	*Sarcocornia-Sporobolus*	145° W	31°S	100	Data not available	Data not available	164	Kelleway et al. (2016a)
Australia	*Sarcocornia quinqueflora*	144° E	36° S	30	Data not available	53[c]	87.1	Lewis et al. (2017)
Australia	Multiple species	153°E	27°S	150	Data not available	2.3	239	Hayes et al. (2018)
Global	Multiple species			100	Data not available	Data not available	162	Duarte et al. (2013b)
Global	Multiple species			100	Data not available	Data not available	593	Alongi (2018)

[a] Elemental analyser
[b] Isotopic Ratio Mass Spectrometry
[c] Diffuse reflectant mid-infrared spectra (MIR-PLSR)

Table 3.1.6. Summary data from literature for organic matter (OM%), organic carbon (Corg%) and total carbon (Mg ha^{-1}) in seagrass sediment. All studies determined %OM using LOI (Loss on ignition) and Corg (%) using an elemental analyser [a], Isotopic Ratio Mass Spectrometry [b] or using Diffuse reflectant mid-infrared spectra (MIR-PLSR) [c].

Region/location	Species	Longitude (degrees)	Latitude (degrees)	Depth (cm)	Organic matter (%)	Organic carbon (%)	Total carbon stock (Mg ha^{-1})	Reference
Global mean		-	-	100	-	2.2	-	Kristensen et al. (2008)
Global mean	Multiple	-	-	-	-	-	761.4	Alongi (2018)
United Arab Emirates	*A. marina*	23°N	53°E	300	Data not available	2.2	156.3	Schile (2017)
Arabian gulf	*A. marina* *R. mucronata*	25-27°N	48-50°E	100	Data not available	0.36-0.87 [a]	76	Cusack et al. (2017)
United States	Multiple	27°N	82°W	50	12.02	5.09 [ab]	133.6	Radabaugh et al. (2017)
Madagascar	Multiple			100		3.4[e]	457	Jones et al. (2014)
Madagascar	Multiple	18°S	46°E	200	Data not available	3.5[f]	113-797	Benson et al. (2017)
Mozambique	Multiple	11°S	24°E	200	Data not available	1.8[a]	484	Stringer et al. (2015)
Australia	*A. marina*	36°S	144°E	30	Data not available	Data not available	65.6	Lewis et al. (2018)
Australia	*A. marina*	27°S	153°E	150	Data not available	3.9[c]	242	Hayes et al. (2018)
South Africa	*A. marina*	28°E	32°S	100	0.63	1.96[a]	151.67	This study

[a] Elemental analyser
[b] Isotopic Ratio Mass Spectrometry
[c] Diffuse reflectant mid-infrared spectra (MIR-PLSR)
[d] Stable Isotope Analyser
[e] Walkley Black Method (modified dichromate titration)
[f] converted values from OM% to Corg (%) using the conversion factor : 2.06 (Kauffman and Donato, 2012)

Table 3.1.7. Summary data from literature for organic matter (OM%), organic carbon (C_{org}%) and total carbon (Mg ha^{-1}) in mangrove sediment. All studies determined %OM using LOI (Loss on ignition) and C_{org} (%) using various methods.

Region/location	Species	Longitude (degrees)	Latitude (degrees)	Depth (cm)	Organic content (%)	Organic carbon (%)	Total carbon stock (Mg ha^{-1})	Reference
United Kingdom	*Zostera marina*	50°N	1-4°W	100	3.61	1.7 [a]	140	Green et al. (2018)
Scotland	*Zostera* sp.	56°N	3°W	30-50	2.58	0.98 [a]	57	Potouroglou (2017)
United Arab Emirates	Multiple species	23°N	53°E	300	Data not available	0.6	49.1	Schile et al. (2017)
Tanzania, Mozambique	Multiple species	6°S 25-26°S	38-39°E 32°E	50-68	Data not available	0.20 to 1.44[a]	85.6	Gullström et al. (2017)
South Africa	*Zostera capensis*	32°S	28° E	100	1.7	0.86	5.1	This study
Australia	*Posidonia australis*	34°S	117°E	150	9.07	2.24 [a]	108	Rozaimi et al. (2016)
Australia	*Zostera muelleri*	26°S	121°E	25	4.48	1.33	Data not available	Lavery et al. (2013)
Australia	Multiple species	34°S	137°E	30	Data not available	Data not available	24.3	Lewis et al. (2018)
Global	Multiple species			< 100	Data not available	2.5 [d]	< 100	Fourqurean et al. (2012a,b)

[a] Elemental analyser
[b] Isotopic Ratio Mass Spectrometry
[c] Mid-Infrared Spectroscopy
[d] Stable Isotope Analyser

3.1.4.2 Potential Factors Limiting Carbon Storage

There are numerous factors that may be responsible for the small amount of carbon stored in the blue carbon habitats at Nxaxo Estuary. Salt marsh sediment carbon storage is influenced by plant community composition (different species have variances in leaf and root morphology), tidal inundation, elevation and sediment grain size (Macreadie et al., 2017a). Generally, sediment carbon stocks are low for shrubby salt marsh vegetation and this was supported by Saintilan et al. (2013) who found low carbon stocks in *Sarcocornia-Sporobolus* sites versus other salt marshes along with Lovelock et al. (2014) who found that sediment carbon storage was lowest for *Sarcocornia quiqueflora.* This is due to *Sarcocornia* occupying the upper intertidal zone, where inundation occurs for shorter periods and less frequently than in the lower intertidal (where mangroves are situated). As a result, vertical accretion of sediment is limited. Salt marshes are also less productive in comparison to mangroves (Saintilan et al., 2013) and this was supported by Schile et al. (2017) who found that salt marsh had significantly lower carbon compared to mangroves. They attributed this to higher tidal elevations, lower redox potential and less anaerobic conditions which are not conducive to carbon storage. Seagrass carbon storage was also low, and this was expected as *Z. capensis* is a smaller, short lived species which is highly unlikely to accumulate large carbon stores (Fourqurean et al., 2012). Our findings were like Schile et al. (2017) who concluded that low carbon storage was because of smaller sized seagrasses. Studies have also shown a clear difference in sediment carbon storage between meadows with larger seagrass species (e.g. *Posidonia* spp. and *Amphibolis* spp.) and those with smaller ones (e.g. *Zostera* spp.) (Dahl et al., 2016). *Posidonia oceanica* meadows for example, have been reported in several studies for having the largest carbon sequestration and storage capacity (Duarte et al., 2005; Lavery et al., 2013; Röhr et al., 2016). This is because larger seagrass species develop higher belowground biomass compared to smaller sized seagrasses and therefore contribute to higher sedimentary organic carbon (Duarte and Chiscano, 1999; Güllström et al., 2017)

We found that the *Zostera capensis* beds at Nxaxo Estuary occurred in large patches. According to Ricart et al. (2015, 2017), patchy seagrass meadows tend to store less carbon per unit area compared to continuous meadows as a result of less retention of autochthonous material (i.e. leaf detritus) Additionally, in patchy meadows, the age and development of the patch potentially influences carbon storage. Older and more stable patches store larger amounts of carbon than younger and more dynamic patches (Duarte et al., 2013c; Greiner et al., 2013; Marbà et al., 2015; Ricart et al., 2015). For this study, there is uncertainty regarding the age and stability of the seagrass sampled due to limited availability of past areal data. In South African estuaries, *Z. capensis* beds are highly dynamic and it is often difficult to map their habitat changes over time as area cover alternates in response to freshwater inflow and marine sediment input (Adams, 2016). Ricart et al. (2015) suggested that newly established seagrass patches store carbon mostly in aboveground biomass and as the patches aged, carbon storage in the sediment increases. Our findings showed that organic carbon was higher in aboveground biomass (~20%) compared to seagrass sediment (< 1%), which may indicate that seagrass patches at Nxaxo Estuary are relatively young.

In seagrasses, carbon storage has been related to structural complexity. Seagrass meadows with low structurally complexity (such as those with a pioneer species that is often small and occurs in patches) are prone to increased erosion and subsequent loss of particulate organic carbon. In our study, seagrass meadow was of low structural complexity as it had only one species (*Z. capensis*). This species has short and narrow leaves at shallow, exposed sites as a morphological adaptation to drying out from exposure, and long and broad leaves in sites that are calm with deeper depths (Talbot and Bate, 1987; Adams, 2016). Meadows with low structural complexity such as at Nxaxo Estuary are associated with low carbon storage. This is aligned with findings by both Rozaimi et al. (2013) and Samper-Villarreal et al. (2016) who reported low carbon storage in meadows with a single small pioneer species. These researchers linked low structural complexity to increased flow rates, less deposition and increased carbon loss from erosion (Fonseca and Fisher, 1986; Bos et al., 2007; Samper-Villarreal et al., 2016). Despite the growing body of literature that quantifies carbon storage in seagrasses, many studies could not adequately explain factors that influence carbon accumulation and storage. A recent study on blue carbon storage in *Z. marina* meadows in the UK showed that even differences in landscapes and habitat conditions may not necessarily influence carbon storage and concluded that determinative factors remain poorly understood (Green et al., 2018). When global seagrass carbon is estimated, to improve accuracy, there should be a clear distinction between patchy versus continuous meadows, and smaller versus larger species.

There have been few studies on carbon storage in arid and semi-arid regions. In these areas, low rainfall and evapotranspiration potentially reduce carbon storage in blue carbon habitats (Adame et al., 2013; Ezcurra et al., 2016; Schile et al., 2017) and is due to differences in decomposition rates between dry and waterlogged sediment (Sanders et al., 2016; Radabaugh et al., 2017). Low carbon stocks in our study could be linked to low rainfall as Nxaxo Estuary is in the warm temperate region and close to the southern latitudinal limit for mangroves along the east coast of Africa (Steinke et al., 1995; Adams et al., 2004). This region receives an annual average rainfall of ~900 mm $year^{-1}$, which is drier than subtropical and tropical sites. Our findings were similar to Radabaugh et al. (2017) who found low carbon stocks for mangroves in Tampa Bay and attributed this to dry climate. Likewise, Schile et al. (2017) reported that carbon stocks in arid regions (rainfall ~100 mm) had significantly lower mangrove sediment carbon stocks due to low rainfall coupled with well drained sandy soils. Similarly, Benson et al. (2017) found low mangrove sediment carbon stocks in southwest Madagascar and concluded that this was due to aridity and low rainfall in the region.

Sediment grain size has been highlighted as a strong predictor of carbon storage in coastal habitats as it influences accumulation of organic particles. Numerous studies have directly correlated sediment grain size and carbon storage in seagrasses, salt marshes and mangroves (i.e. Saintilan, 1997; Saintilan et al., 2013; Dahl et al., 2016; Kelleway et al., 2016a; Röhr et al., 2016; Serrano et al., 2016; Van Ardenne et al., 2018). Soil carbon concentration has been connected to sand and silt content along with soil bulk density. Sandy or coarse-grained sediments are highly permeable with more aeration, which increases remineralization of carbon due to the environment becoming more reduced (Howarth and Hobbie, 1982; Van Ardenne et al., 2018). Fine grained sediment accumulates and preserves more organic matter due to their high surface area. Therefore, sediment with fine particles and high organic matter reduces oxygen (which is consumed by detritovores) in the sediment, decreasing the permeability and decomposition of organic matter (Dahl et al., 2016 and references therein). While our study not specifically examine grain size and other environmental factors, Hoppe-Speer and Adams (2015) found that mangrove sediment at Nxaxo Estuary consisted mainly of sand (~ 55%). Likewise, Julie (2018) found that Nxaxo Estuary salt marsh sediment was sandy (42%). We speculate that sandy soils combined with low rainfall in both mangroves and salt marshes could be partly responsible for the low carbon stocks at this estuary.

Carbon storage in salt marsh and mangroves can also be related to the age of the habitat. For example, in stable and mature salt marshes, carbon storage is typically high while newly established or restored salt marshes have less carbon (Artigas et al., 2015; Alongi, 2018). In mangroves studies, carbon concentration has been positively related to stand age (Lunstrum and Chen, 2014; Xiong et al., 2018). Younger, smaller stands and scrub mangroves store less carbon while older, mature stands with taller trees contain larger carbon stocks (Alongi, 2018). Mangrove and salt marsh sediment carbon stocks also differ with latitude and latitudinal gradients (Twilley et al., 1992; Hayes et al., 2017). Subtropical and tropical mangroves tend to have higher productivity and carbon storage compared to warm temperate mangroves and salt marshes. Mangroves at their southern distributional limit are generally slow growing and their ecological functioning is similar to that of young forests (Alongi, 2009; Alongi et al., 2016; Hayes et al., 2017). However, Alongi (2018) argued that age and size of mangrove forests can be extremely variable and there is some degree of uncertainty with relating carbon storage to latitude especially because young stands occur close to the equator and older, mature forests occur at higher latitudes.

3.1.4.3 Aboveground and Belowground Biomass Carbon Storage of Habitats at Nxaxo Estuary

Although much carbon in blue forests is stored in the sediment, the vegetative component is small but significant and requires accurate quantification (Radabaugh et al., 2017). This study was the first attempt to develop an allometric equation for salt marsh carbon estimates in South Africa’s blue carbon habitats. This contributed to the global availability of allometric equations and carbon data for salt marsh to aid with quicker and less destructive quantification of above-ground vegetative stocks in future. The method used to establish the allometric equation for salt marsh *S. tegeteria* aboveground biomass in this study was consistent with other studies (Radabaugh et al., 2017; Owers et al., 2018). We found a good relationship between stem height and biomass (R^2 = 0.706) which aligned with findings by Radabaugh et al. (2017) for other salt marsh species with high degrees of fit including *Salicornia virginica* (R^2 = 0.935), *Sporobolus virginicus* (R^2 = 0.553) and *Spartina alterniflora* (R^2 = 0.767) in Tampa Bay, Florida. The high R^2 value of the equation suggests that the biomass of *S. tegeteria* can be accurately estimated at other estuaries where this species is present. However, plant height and biomass may vary seasonally because of changes in growth rates, flowering and senescence (Reidenbaugh, 1983; Gonzalez Trilla et al., 2013; Radabaugh et al., 2017) which may influence such allometric equations. Although the sampling for this study was done in winter, the allometric relationship was strong and was in contrast to studies that reported a peak in biomass and stem height in mid-summer compared to winter (Radabaugh et al., 2017). The presence (or absence) of an inflorescence at the time of collection has been shown to influence plant height and biomass, decreasing the predictive ability of the allometric equation (Gonzalez Trilla et al., 2013; Radabaugh et al., 2017). In this study, there is no deviation from the expected biomass value as a result of flowering since *S. tegeteria* has no large flowers but rather terminal inflorescences that are generally small (8 mm in length and 2 mm in diameter). Other factors which contribute to the strength of the allometric equation include plant elevation and other abiotic factors (Morris and Haskin, 1990; Radabaugh et al., 2017). For example, *S. alterniflora* is taller in tidal creeks and shorter in more landward areas due to differences in soil parameters such as nutrient availability, salinity and moisture content (Reidenbaugh, 1983; Clewell, 1997; Wieski and Pennings, 2014). Since *S. tegeteria* usually occurs in the low to mid intertidal zone in most South African estuaries (Steffen et al., 2009), it is generally the same size and the allometric equation we developed is accurate. Although all factors are important, the variations are usually minor considerations, making the relationship between plant height and biomass robust (Gonzalez Trilla et al., 2013; Radabaugh et al., 2017).

According to Radabaugh et al. (2017), few published datasets exist for carbon content for salt marsh. As a result, the most common value for comparison is 45% from Howard et al. (2014). However, this value is not the most accurate as it is based on a study done in China for terrestrial systems (Fang et al., 1996). We found that salt marsh organic carbon content was much lower (25.34%) than the value from Howard et al. (2014). Elemental composition of *S. tegeteria* was also less than salt marsh species in Tampa Bay, where 18 salt marsh species were found to comprise 41.1 ± 5.5% carbon. Additionally, Owers et al. (2018) reported an average carbon content of 38.3 ± 2.5% for all salt marsh species and 34.4 ± 0.8% for *S. quinqueflora* in south east Australia, which is a higher intertidal species and shrubbier than the succulent low growing *S. tegetaria.* Salt marsh above ground biomass estimates in this study were like studies done in Australia since both are classified within the temperate salt marsh group (Adam, 1990; Kelleway et al., 2016b). Our results showed aboveground biomass carbon to be 26.1± 2.47 Mg C ha^{-1} which was within the range (22-88 Mg C ha^{-1}) reported by Owers et al. (2018).

For seagrasses, the average global standing stock has been estimated at 460 g DW m^{-2} and this value was derived from a few studies from Africa (Duarte and Chiscano, 1999; Githaiga et al., 2016) that emphasized the need for more comprehensive assessments of seagrass biomass and blue carbon storage on this continent. We found that *Z. capensis* aboveground biomass averaged 354.5 ± 86.47 g m^{-2} which was lower than the global estimate. The biomass in our study was however larger than what has been reported for *Z. capensis* in other estuaries in South Africa. Grindley et al. (1976) reported aboveground biomass to be 206 g m^{-2} at Knysna Estuary, 217 g m^{-2} at Langebaan lagoon (Christie, 1989) and 55-105 g m^{-2} at Kromme Estuary (Hanekom and Baird, 1988). According to Githaiga et al. (2016) seagrass aboveground biomass for South Africa averaged at 413.3 g m^{-2} which was larger than the biomass at Nxaxo Estuary. Githaiga et al. (2016) also reported that belowground biomass was generally higher (474.6 g m^{-2}) than aboveground biomass (174.4 g m^{-2}); in this study however seagrass above- and belowground biomass was similar. Biomass was generally less in smaller species such as *Z. capensis* compared to larger species due to a slow turnover of their belowground material (Duarte and Chiscano, 1999; Githaiga et al., 2016). Seagrass biomass carbon in our study was like the global average (1.76 Mg C ha^{-1}) reported by Fourqurean et al. (2012) but the aboveground biomass carbon in mangroves was smaller (6.08 ± 1.83 Mg C ha^{-1}) compared to other regions.

Mangroves aboveground biomass can vary but the general trend is that larger mangrove forests in the tropical regions have an aboveground biomass of ~500 Mg C ha^{-1} , while smaller dwarfed mangroves have a biomass of ~8 Mg C ha^{-1} (Kauffman and Cole, 2010; Kauffman et al., 2011; Howard et al., 2014; Alongi et al., 2016). Data for mangrove aboveground biomass was scarce compared to the other two habitats and this was mainly because the aboveground biomass was difficult to quantify and required extensive resources (Owers et al., 2018). The available estimates can vary and largely depend on mangrove forest structure, species composition and sampling design (Stringer et al., 2015). For example, in Gazi Bay in Kenya (a well-studied mangrove area), earlier estimates reported 125 Mg C ha^{-1} and 226 Mg C ha^{-1} (Slim et al., 1996; Kirui et al., 2006) but later, Cohen et al. (2013) reported only 67 Mg C ha^{-1} (Stringer et al., 2015), showing that these estimates can be variable.

The aboveground biomass carbon stock in our study was slightly less than *A. marina* in the United Arab Emirates (77.6 Mg C ha^{-1}) (Schile et al., 2017) and we suspected that this was due to the stunted nature of the trees at Nxaxo Estuary. Stunting has been linked to the occurrence of trees at higher elevations (that lack tidal influence), areas that are limited by nutrients (such as nitrogen and phosphorus) (Feller et al., 2003; Naidoo, 2016), hyper salinity, soil redox potential and waterlogging (Naidoo, 2016). The stunting at Nxaxo Estuary was further exacerbated by cattle browsing, which limited growth and caused changes in mangrove morphology because of top-bottom browsing (Hoppe-Speer and Adams, 2015). Although cattle have been excluded from the mangrove area and the forest has shown signs of recovery (Mbense, 2017), we speculated that cattle browsing in the past may have contributed to such low biomass carbon stocks although this has not been explicitly tested. In agreement with our findings, Benson et al. (2018) reported that harvesting pressure by local communities in Madagascar led to major alterations in mangrove forest structure and a significant reduction in aboveground biomass carbon storage. Harvesting pressure changed dense, closed canopies with tall trees to sparse, short trees with open canopies. The former had significantly more above ground carbon (127.95 Mg C ha^{-1}) and the latter had much less carbon (4.87 Mg C ha^{-1}). This showed that degradation can indeed alter mangrove forest structure and ultimately carbon storage.

The overall biomass carbon stock in our study was within range of that reported for *A. marina* in Australia (495-1047 Mg C) by Owers et al. (2018). We expected similar values to Australia, since mangrove productivity (based on litter fall) is less at higher latitudes with low temperatures (< 30°C) (Bouillon et al., 2008; Naidoo, 2016) and resource allocation towards leaf production is limited by the short growth season (Morrisey et al., 2010; Naidoo, 2016). However, since our mangroves are stunted and occur at their distributional limit, resources could be shifted towards belowground components which potentially have a greater proportion of carbon belowground (Naidoo, 2009, 2016). However, belowground biomass (i.e. roots) was not measured because it is a difficult and destructive to sample.

3.1.4.4 Limitations of the Study

Our study suggested that there are numerous complicated, local and regional processes underlying carbon accumulation and standing stocks in blue habitats. One limitations of the study was that core compaction was not corrected for in our sediment samples and there might be a small margin of error. Also, these carbon stock assessments may have their limitations especially in terms of methodologies used to scale up carbon data (i.e. taking sediment samples from a small area using a corer and then scaling up to landscape level) which may lead to over- or underestimation of carbon stocks (Schile et al., 2017). Since this is a baseline study, there may also have been some errors during field sampling, processing and in calculating carbon. For example, separation of above- and belowground biomass of seagrasses is sometimes ambiguous (Howard et al., 2014) as it is based on how well one can distinguish the "green" stem parts from "brown" root structures, which is sometimes difficult in small species such as *Z. capensis.* We also acknowledge that the organic carbon stored in the sediment of blue habitats is not a true reflection of present vegetation but is a result of habitat changes over time which will be influenced by sea level rise and sedimentation rates in the future (Mudd et al., 2009).

3.1.5 Conclusion

Although blue carbon studies have become increasingly topical, few studies have done a comprehensive assessment of carbon storage in all three habitats. We found that values of carbon storage for salt marshes, seagrasses and mangroves were significantly lower than global means from other studies. However, these findings will contribute towards global understanding of blue carbon storage in warm temperate habitats and mangroves at their distributional limits. While current South African policies have not directly addressed the value of carbon storage in coastal wetlands, there are future opportunities for the protection of these habitats through participation in carbon trading, offset schemes and payments for ecosystems services. As outlined in our findings, carbon storage at higher latitudes was low because of less productivity, thus making these habitats more vulnerable to threats such as global warming and sea-level rise. According to Saintilan et al. (2014) sea-level rise and climate change might result in habitat switching and encroachment by mangroves on salt marsh habitats. Future studies should expand to other estuaries in South Africa for comparative analyses and to create a robust blue carbon inventory.

3.2 Application of the Blue Carbon Protocol to Assess Variability across the Mangrove-Salt Marsh Ecotone

Raw JL, Julie CL, Adams JB (2019) A comparison of soil carbon pools across a mangrove-salt marsh ecotone at the southern African warm-temperate range limit. *South African Journal of Botany* 127, 301-307.

3.2.1 Introduction

Vegetated coastal ecosystems, including mangroves, salt marshes and seagrasses, are globally recognized for their carbon storage capacity, and are therefore collectively referred to as 'blue carbon' ecosystems (Nellemann et al., 2009; Mcleod et al., 2011). In comparison to terrestrial ecosystems, these blue carbon ecosystems are known to exhibit higher productivity and greater efficiency in trapping carbon-associated sediment from outside their ecosystem boundaries, making them very efficient carbon sinks (Alongi, 2015; Kelleway et al., 2017a). Despite covering a restricted global area, mangroves are better carbon sequesters than terrestrial ecosystems as they accumulate and store carbon over a longer period of time (Lovelock and Duarte, 2019). Mangroves in particular exhibit a higher carbon storage capacity due to the high above- and belowground biomass of the trees which store biological carbon (Donato et al., 2011). Furthermore, the aerial roots decrease water velocity and increase the deposition of carbon-rich sediment (autochthonous and allochthonous), and plant matter (leaf litter, branches, roots, and woody debris) and promote the formation of carbon-rich soils (Saintilan et al., 2013; Horstman et al., 2015).

Generally, research has focussed on the quantification of carbon storage between different habitats (Brown et al., 2016; Radabaugh et al., 2017; Simpson et al., 2017), and factors driving soil, vegetation, and above- and belowground carbon dynamics (Perry and Mendelssohn, 2009; Comeaux et al., 2012). More recently, the dynamics of carbon storage has been assessed at different scales along the ecotones of blue carbon habitats (Huxham et al., 2018). Ecotones are defined as the transitional areas between two adjacent ecosystems and they are characterized by the interactions between these ecosystems over different spatial and temporal scales (Holland, 1988; Gosz, 1993). Ecotones occur naturally either as steep gradients in environmental variables that influence species' distributions, or in gradual gradients resulting from nonlinear responses to the physical environment (Risser, 1995). Abiotic drivers that define ecotones include climate, fire regime or edaphic conditions, while biotic drivers include competition and herbivory (Yando et al., 2018). Gradual gradients produce ecotones that are characterized as mosaics of the adjacent ecosystems (Gosz, 1993; Risser, 1995). This type of ecotone is observed between mangrove and salt marsh habitats (Raabe et al., 2012; Rodriguez et al., 2016; Yando et al., 2018).

Mangrove-salt marsh ecotones are of interest when they occur at the range limits for mangrove tree species (Rogers and Krauss, 2018). Range limits are related to the ecological niche and they represent the spatial edge of a species' distributional range (Sexton et al., 2009; Louthan et al., 2015). For mangroves, range limits are globally delineated by the 20°C winter isotherm for sea-surface temperature (Duke et al., 1998; Tomlinson, 1999), and therefore occur at subtropical/warm-temperate biogeographic boundaries. At regional scales, mangrove distribution patterns can be determined by freeze events, precipitation, and ocean currents (Stuart et al., 2007; Soares et al., 2012; Cavanaugh et al., 2014; Osland et al., 2017a). Local coastal hydrology and geomorphology can also influence mangrove distribution patterns by restricting propagule dispersal and limiting suitable areas for establishment (Semeniuk, 1983; Schaeffer-Novelli et al., 1990; Stevens et al., 2006; Leong et al., 2018; Raw et al., 2019a). As ecotones and range limits are influenced by their sensitivity to change, this may be used to understand and predict future shifts in response to climate change (Yando et al., 2018).

At local scales, plant-soil interactions at mangrove range limits are not clearly related to soil carbon storage as an increase in aboveground biomass does not directly result in increased belowground carbon (Perry and Mendelssohn, 2009; Yando et al., 2018). This could be because soil carbon can be influenced by other soil characteristics such as moisture content and organic matter content. Over time there can be an increase in soil carbon storage in areas to which mangroves have expanded at the expense of salt marsh (Kelleway et al., 2016b; Yando et al., 2016, Doughty et al., 2016; Simpson et al., 2017). Besides increased carbon storage, mangrove expansion is also associated with shifts in ecosystem service provision (Kelleway et al., 2017a). Mangrove expansions have generally been associated with rising temperatures as these species are not frost tolerant (Cavanaugh et al., 2014; Saintilan et al., 2014; Alongi, 2015). However, as mangrove distribution limits are determined by different environmental drivers around the world (Osland et al., 2017b), it is important to investigate the effects of mangrove expansion into salt marsh in different regions and at different spatial scales. Like many other southern hemisphere range limits for mangroves, the distribution limit along the South African coastline is controlled by coastal geomorphology, rather than winter temperatures (Osland et al., 2017b; Raw et al., 2019a). This study provides the first assessment of carbon storage along a salt marsh-mangrove ecotone occurring at the southern range limit for mangroves on the east coast of Africa.

The Nahoon Estuary, located along the warm-temperate coast of South Africa, provides a unique opportunity to assess salt marsh-mangrove dynamics as *Avicennia marina* (Forssk.) Vierh. mangrove trees were planted here in 1969 (Hoppe-Speer et al., 2015b). Since then, the forest has naturally expanded through seedling regeneration over time, encroaching into the salt marsh (predominantly *Triglochin striata* Ruiz López & Pavón; *Bassia diffusa* (Thunb.)

Kuntze; and *Salicornia tegetaria* S.Steffen, Mucina & G.Kadereit). This mangrove forest is now the southernmost occurrence for this ecotype in Africa, although this range limit was artificially established. The aims of this study were 1) to compare soil carbon storage across the salt marsh-mangrove ecotone at the Nahoon Estuary; 2) to examine how soil carbon changes with depth and determine if soil carbon variability is related to soil moisture content. It was hypothesized that 1) soil carbon storage would be greatest in the mangrove habitat; 2) soil carbon would increase with sampling depth; 3) soil carbon would be linearly and directly related to soil moisture content. The results of this study can provide some insight to the short-term effects of mangrove expansion into salt marsh habitat for this region.

3.2.2 Methods

3.2.2.1 Study Area

The Nahoon Estuary (32°59′09″ S, 27°57′03″ E) (Figure 3.2.1), is a permanently open estuary that experiences a warm-temperate climate, with annual temperatures ranging from 13-25°C (winter minimum of 5.3°C and summer maximum of 31.4°C). The annual rainfall ranges from 200-600 mm, with most of the rainfall occurring in austral spring and summer (Hoppe-Speer et al., 2015b; Geldenhuys et al., 2016). The Nahoon River has a total catchment area of ~ 564 km^2, including the Nahoon Dam which occurs 27 km upstream of the estuary. The estuary is microtidal with an average tidal range of 0.76 m and a spring tide range of 1.6 m respectively (Wiseman et al., 1993). A tidal creek facilitates connectivity to the main estuary channel (Geldenhuys et al., 2016; Cotiyane et al., 2017).

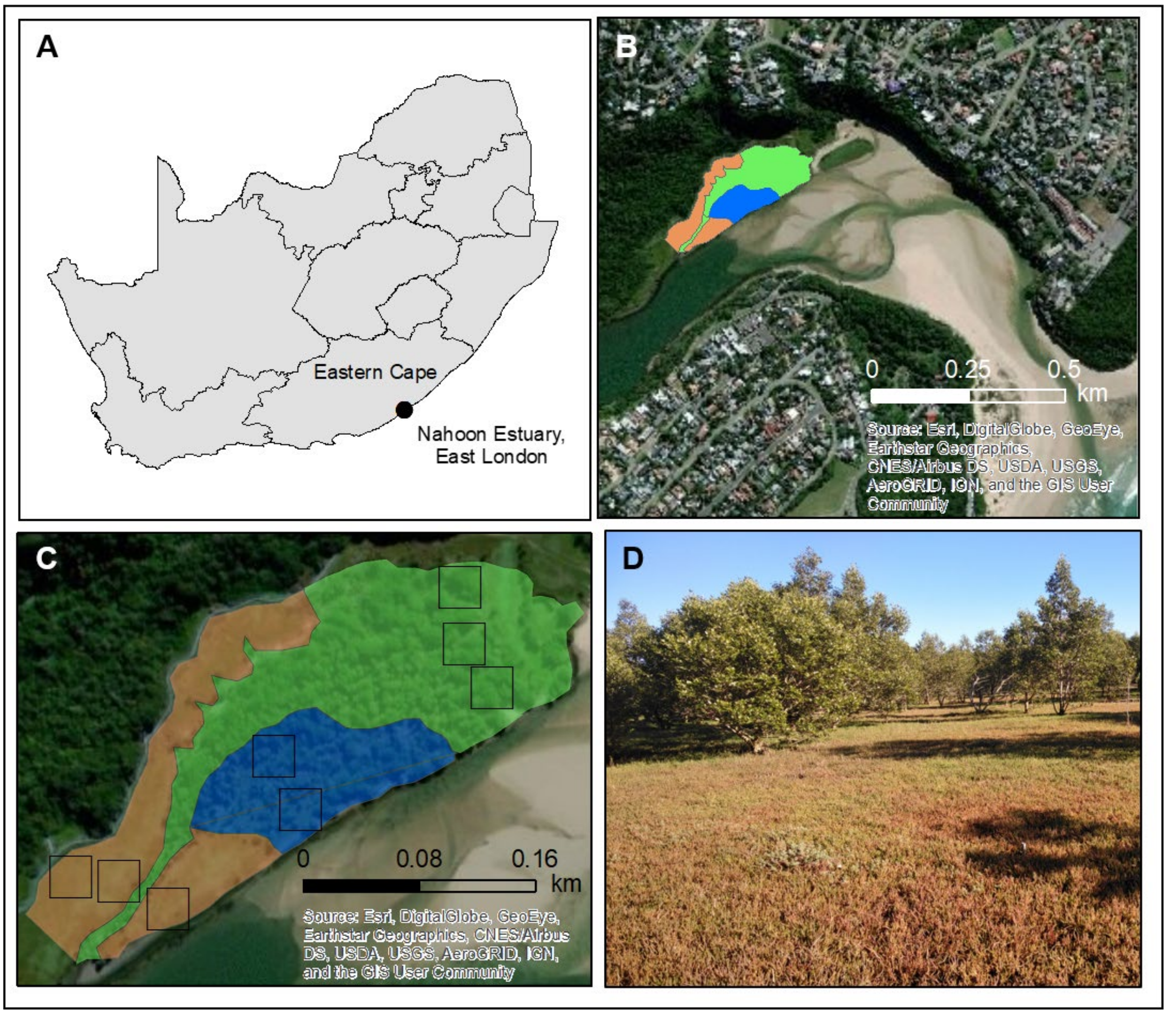


Figure 3.2.1. Location of the Nahoon Estuary (**A**); Mangrove (green), Ecotone (blue), Salt Marsh (orange) habitats sampled in this study (**B**); location and position of the sampling sites in each habitat (**C**); photograph of the ecotone habitat in the Nahoon Estuary (**D**).

The mangrove forest at the Nahoon Estuary was artificially established in 1969 when *A. marina* mangrove trees were transplanted here from Durban Bay (Ward and Steinke, 1982). Over time the mangrove area has expanded into adjacent tidal flats and salt marsh areas (Hoppe-Speer et al., 2015b). The salt marsh area into which mangroves are slowly encroaching is termed the 'ecotone' and it occurs between the two established habitats. In the ecotone zone it is typical to find singular mangrove trees with a salt marsh understorey (Figure 3.2.1). Sampling was carried out in established mangrove, salt marsh and ecotone habitats.

3.2.2.2 Sample Collection

All samples were collected over a two-day period during the low tide of a neap tide in June 2018. To account for within-habitat spatial variability, sampling sites were established within each of the habitat types. Three sampling sites were established in the mangrove and salt marsh habitats, and two sampling sites were established in the ecotone habitat (*n* = 8) (Figure 3.2.1). The sampling sites were located across the shore profile relative to the estuary channel.

To characterize the vegetation for each habitat, 2 sampling quadrats were measured out at each sampling site. In the mangrove habitat, heights of all trees located within 25 m^{-2} quadrats were measured (*n* = 6). In the salt marsh habitat, the percentage cover of different species was recorded in 1 m^{-2} quadrats (*n* = 6). In the ecotone habitat, mangrove and salt marsh quadrats were both used (*n* = 4 mangrove quadrats, *n* = 4 salt marsh quadrats), with the salt marsh quadrat placed consistently within the larger mangrove one.

Soil cores were collected in triplicate at each site (*n* = 24) to 0.5 m depth using a Russian Peat Corer (internal diameter = 5.4 cm). The cores were sectioned in the field at intervals of 0-15 cm; 15-30 cm; and 30-50 cm (Howard et al., 2014). The sections were kept cool in zip-sealed plastic bags until laboratory processing.

3.2.2.3 Laboratory Processing

Soil moisture and organic content were determined in the laboratory. Samples were first pre-weighed and then oven-dried at 100°C for 72 hours to determine the dry weight and to calculate moisture content as the percentage mass lost from the original wet weight (Black, 1965). To determine organic content, the oven-dried samples were pre-weighed and then combusted in a muffle furnace at 550°C for 6 hours. The percentage mass lost from the original dry weight could then be calculated as the organic content lost due to combustion (Briggs, 1977).

3.2.2.4 Carbon Calculations

Soil carbon density for each core section was calculated following the equation provided by Howard et al. (2014) as follows:

$$Soil\ carbon\ density\ (g.cm^{-3}) = Dry\ bulk\ density\ (g.cm^{-3}) \times (\frac{\%\ C_{org}}{100})$$

Dry bulk density is calculated by dividing the dry mass of the soil sample (g) by the volume of the soil sample (cm^3). The % C_{org} is the percentage of organic carbon, which was assumed to be equal to the organic content determined by the loss-on-ignition (LOI) method described above (Howard et al., 2014).

The carbon content ($MgC.ha^{-1}$) of the soil in each of the habitats was then calculated. The carbon content of each core section was first calculated as the product of the carbon density and the thickness interval – which is the length (cm) of the respective core section. Next, the carbon content of each core was calculated by summing the respective sections. The total core carbon is reported as $MgC.ha^{-1}$ for the top 0.5 m soil. The carbon content at this depth was then calculated for each habitat area.

The area of the habitats was determined from Google Earth satellite imagery from the closest date to the sampling occasion (16 July 2018). The ecotone was defined as the area where salt marsh and mangrove vegetation occur together. GPS points at the ecotone "boundaries" were recorded in the field and were considered when the aerial extent was mapped. The ecotone boundary with the salt marsh habitat was defined by the occurrence of mangrove trees.

3.2.2.5 Statistical Analyses

All statistical analyses were carried out in R version 3.6.1 for Windows (R Core Team, 2019). To compare soil carbon storage between habitats, the total carbon per 0.5 m core ($MgC.ha^{-1}$) was compared between the mangrove (n = 9), salt marsh (n = 9), and ecotone (n = 6) habitats using a one-way analysis of variance (ANOVA) test. After checking the normality and equal variance assumptions (using Shapiro Wilk and Levene's tests respectively), a Type III sum of squares ANOVA was run to account for the unequal sample size using the 'car' package (Fox and Weisberg, 2019). To examine the relationship between soil carbon and soil moisture content, a linear regression was used. The data for the carbon per core section ($g.cm^{-2}$) and the moisture content (%), measured from different depth intervals (0-15 cm, 15-30 cm, 30-50 cm), were analysed for all cores sampled (n = 72). The model was built to describe soil carbon as the dependent variable with soil moisture content (continuous predictor) and depth (categorical predictor) as independent variables. The assumptions for linear regression were verified graphically following Zuur et al. (2009). The graphical output of the linear model was generated using the 'ggplot2' package (Wickham, 2016).

3.2.3 Results

3.2.3.1 Vegetation Characteristics

The habitat areas used for comparison in this study are defined by the characteristics of the vegetation. The ecotone habitat resembles a combination of salt marsh and mangrove habitats and so the data are presented so that comparisons can be made between the transitional ecotone habitat and the two established habitats respectively. Salt marsh species composition was different between the salt marsh and ecotone habitats, as *Triglochin striata* did not occur in the ecotone (Figure 3.2.2). *Bassia diffusa* was more prevalent in the salt marsh habitat, but *Salicornia tegetaria* occurred most frequently in both habitats. *Avicennia marina* was the only mangrove species present at the sampling locations in both the mangrove and ecotone habitats. (Figure 3.2.2).

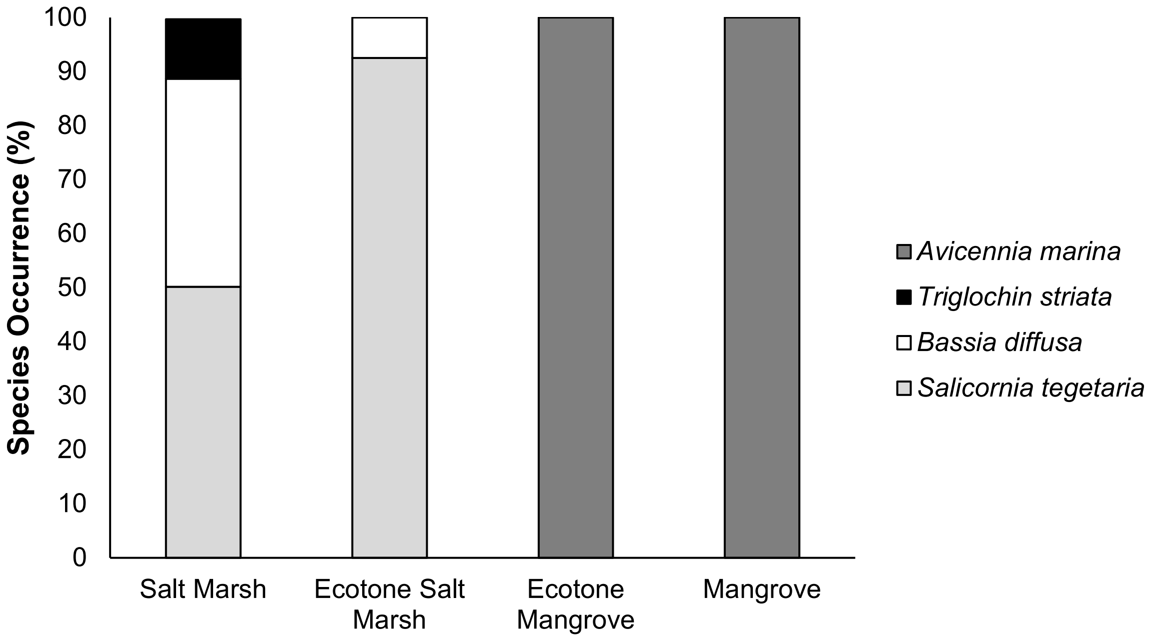


Figure 3.2.2. Vegetation species composition assessed from quadrats placed in salt marsh, ecotone, and mangrove habitats of the Nahoon Estuary. Salt marsh occurred as an understorey to the *A. marina* mangrove trees in the ecotone habitat.

Comparing the mangrove tree density and population structure showed some differences between the ecotone and mangrove habitats (Figure 3.2.3). Seedlings (< 50 cm) were common in both habitats and occurred at the highest density. Large trees (> 200 cm) occurred in both habitats but were more common in the mangrove habitat. Trees in the intermediate size categories (50-200 cm) were absent from the ecotone habitat.

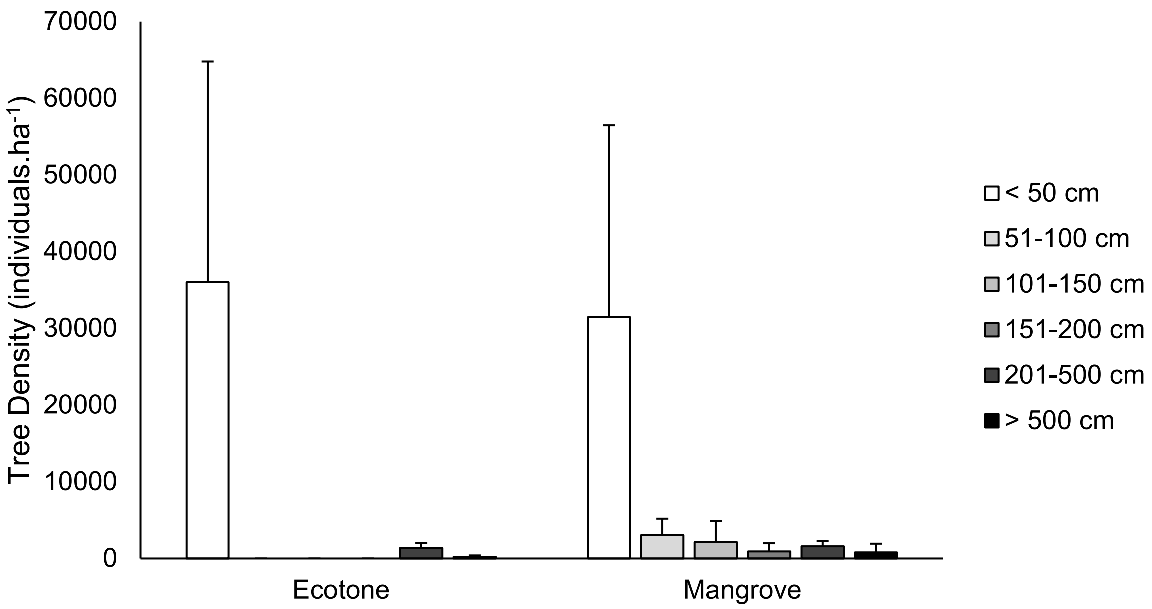


Figure 3.2.3. Average (± SD) tree density of *Avicennia marina* per size category estimated from 25 m^2 quadrats set up in the ecotone and mangrove habitats of the Nahoon Estuary.

3.2.3.2 Soil Carbon Variability

Soil carbon storage, calculated as the total carbon per 0.5 m core, was not significantly different between the salt marsh, ecotone and mangrove habitats ($F_{(21, 2)} = 0.159$, $p = 0.854$) (Table 3.2.1). As the mangrove habitat covered the largest area, the total carbon storage in the sediment was higher (194.94 ± 19.5 MgC) in comparison to the salt marsh (164.42 ± 33.0 MgC) and ecotone (54.96 ± 6.2 MgC) habitats.

Table 3.2.1. Comparison of total carbon per core to 0.5 m depth ($MgC.ha^{-1}$) collected in salt marsh, ecotone, and mangrove habitats at the Nahoon Estuary. The total carbon per habitat area is calculated as the sum of the cores collected in each habitat divided by the habitat area (ha).

	Salt Marsh	Ecotone	Mangrove
Average (±SD) Carbon per 0.5 m core ($MgC.ha^{-1}$)	109.62 ± 22.0	114.50 ± 12.8	110.14 ± 11.0
Habitat area (ha)	1.45	0.71	2.55
Total Carbon (MgC)	158.94 ± 31.9	81.29 ± 9.1	280.86 ± 28.1

As the total carbon was not significantly different between habitats, the data were pooled to examine trends in soil carbon variability in relation to depth and moisture content. The relationship between soil carbon and soil moisture content was different between the depth intervals that were sampled ($F_{(68, 4)}$ = 645.4, p < 0.0001) (Figure 3.2.4, Table 3.2.2). Soil carbon and soil moisture content were directly related across all depth intervals that were sampled. Soil carbon was highest in the 30-50 cm depth interval (Figure 3.2.4).

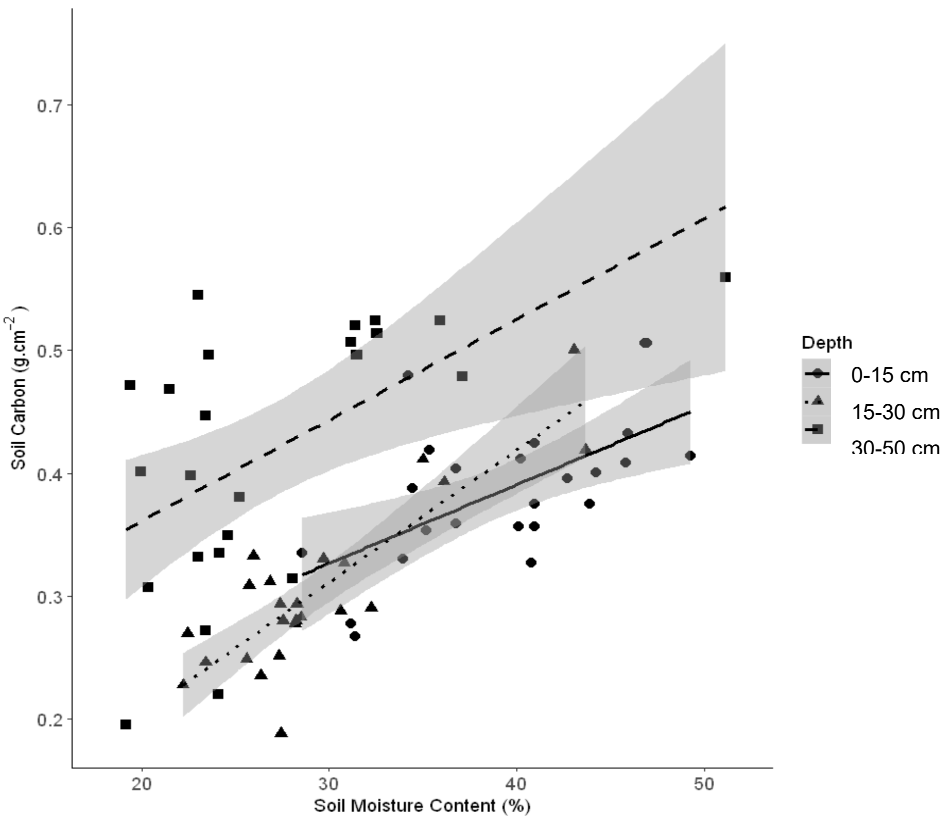


Figure 3.2.4. Soil carbon in relation to soil moisture content at specific depth intervals sampled from soil cores taken at the Nahoon Estuary. The linear relationships for each depth interval are expressed by the black lines and the grey shaded areas denote the 95% confidence intervals for each linear relationship.

Table 3.2.2. Summary of the linear regression model for soil carbon, soil moisture content, and depth.

Coefficient	Value	SE	df	*t*	*p*-value
Depth 0-15 cm	0.057	0.049	-	1.158	0.251
Depth 15-30 cm	0.059	0.038	-	1.560	0.123
Depth 30-50 cm	0.193	0.035	-	5.464	< 0.001
Soil Moisture Content	0.0008	0.001	-	6.858	< 0.001
Residual	-	0.062	4	-	-
Total	-		68	-	-

3.2.4 Discussion

The shift of a wetland from salt marsh to mangrove vegetation is expected to increase the aboveground carbon storage capacity of the habitat, as mangrove trees account for a greater biomass than salt marsh plant species (Kelleway et al., 2016b; Yando et al., 2018; Simpson et al., 2019). However, changes in belowground carbon storage following mangrove encroachment into salt marsh habitats are more complex. Variability in soil carbon has been related to vegetation structure (Owers et al., 2018), sources of allocthonous organic matter (Bouillon et al., 2003) as well as elevation gradients and tidal regimes (Hayes et al., 2017). These factors operate on different spatial and temporal scales.

Vegetation species composition could also be a factor that influences soil carbon shifts following mangrove encroachment. *Avicennia* is considered a pioneer mangrove genus, and globally it is these species that have been most observed expanding into salt marshes (Saintilan et al., 2014; Osland et al., 2017a; Rogers and Krauss, 2018). However, the salt marsh species composition is quite different between range expansion sites around the world. A 70-year encroachment of *A. marina* into salt marshes of *Sarcocornia quinqueflora* (Bunge ex Ung.-Sternb.) A. J. Scott and *Sporobolus virginicus* (L.) Kunth in NSW, Australia led to the conversion of 30% of the salt marsh into mangroves with belowground carbon stocks increasing by 230 ± 62 Mg C km^{-2} yr^{-1} (Kelleway et al., 2016b). A more rapid response has been measured along the Atlantic coast of Florida, USA where a 3-year expansion of *Avicennia germinans* (L.) L. and *Laguncularia racemosa* (L.) C. F. Gaertn. mangroves into salt marsh habitats of *Salicornia* (L.) spp., *Batis maritima* (L.), *Spartina alterniflora* Loisel and *Distichlis spicata* (L.) resulted in a 59% increase in carbon stocks from 31.2 Mg C ha^{-1} to 49.5 Mg C ha^{-1} (Simpson et al., 2019). In contrast, two separate studies done at the mangrove range limit in Louisiana, USA found the expansion/encroachment of *A. germinans* into *S. alterniflora* salt marsh habitat did not result in significant changes in soil carbon content

(Perry and Mendelssohn 2009; Yando et al., 2018). Similar trends were found at the Nahoon Estuary, the southerly distribution limit for mangroves on the east coast of Africa, where the overall average soil carbon was similar between the natural *S. tegetaria*, *B. diffusa* and *T. striata* salt marsh habitat (109.62 ± 22.0 Mg C ha^{-1}) and the slowly expanding *A. marina* mangrove habitat (110.14 ± 11.0 Mg C ha^{-1}).

In mangrove habitats carbon storage is related to the age of the forest as older, well-established forests store more carbon which has accumulated in biomass and soils over a longer time period (Kathiresan et al., 2013; Kelleway et al., 2016b; Chen et al., 2018). The comparatively short history of the mangrove forest at the Nahoon Estuary could therefore explain the similarities in average soil carbon across the salt marsh-mangrove ecotone. Here the mangrove trees have expanded mostly onto adjacent tidal flats, which would have had negligible carbon storage capacity prior to mangrove establishment. In contrast, encroachment into the salt marsh habitat has been slower (Hoppe-Speer et al., 2015b). It is therefore likely that the soil carbon content measured in the ecotone zone is representative of historic salt marsh deposits. Ongoing research at this estuary that includes analysing Pb^{210} isotopes of the soil along the depth profile to determine accretion rates will provide more conclusive evidence of whether these layers were present prior to the establishment of the mangroves in 1969.

Tree size and density are also related to carbon storage in mangrove forests (Kathiresan et al., 2013; Njana et al., 2018). At the Nahoon Estuary, the density of large trees was higher in the mangrove habitat in comparison to the adjacent ecotone habitat, but overall these values are comparable to other mangrove estuaries along the warm-temperate to subtropical South African coastline. For example, the density of *A. marina* mangroves has been reported to range from 62 trees.ha^{-1} at the Nqabara Estuary to 952 trees.ha^{-1} at the Mngazana Estuary (Adams et al., 2004). Higher tree density promotes soil carbon storage if leaf litter production is higher and a greater number of pneumatophores assist with trapping and deposition of allochthonous and autochthonous material. The height of the *A. marina* mangrove trees at the Nahoon Estuary are comparable to those in other South African estuaries. However, overall *A. marina* reaches much larger sizes in warmer tropical regions of the species' range across the Indo-Pacific region (O'Grady et al., 2006; Quisthoudt et al., 2012; Mbense, 2017). There is a clear latitudinal effect on the growth and productivity of mangrove forests which has been linked to their carbon storage capacity (Bouillon et al., 2008; Morrisey et al., 2010; Radabaugh et al., 2017; Simpson et al., 2017; Twilley et al., 2017). The low soil carbon content measured at the Nahoon Estuary is therefore not unexpected for a continental range limit.

Similarities in soil carbon between the mangrove, ecotone, and salt marsh habitats can be explained by local hydrodynamics. Tidal exchange has a large influence on carbon accumulation in blue carbon habitats. Allochthonous organic material from marine or riverine sources can be deposited on surface sediments during tidal exchange (Bouillon et al., 2003; Saintilan et al., 2013). Mangrove rooting structures and salt marsh plants then serve to trap this material and accrete organic sediment, with minimal losses to the system (Krauss et al., 2014; Lovelock et al., 2015a). All areas of the salt marsh, ecotone, and mangrove habitats sampled in this study are tidally influenced, however, as the intertidal zone at the Nahoon Estuary is very small (< 100 m), the potential for spatial heterogeneity is limited. Similarities in soil moisture content could also indicate that all three habitats experience the same inundation regime as soil moisture content was mostly highest in samples collected above 15 cm depth. However, more detailed studies on tidal height variability and accretion rates across these habitats are needed. This is the focus of ongoing research at this estuary.

Soil moisture content can be an important factor influencing soil carbon because it is directly related to soil organic matter. High moisture content promotes organic matter retention in the soil as anoxic conditions form under a negative oxidation-reduction potential, thus reducing soil respiration, organic matter decomposition, and carbon mineralisation (Alongi et al., 2001; Sasaki et al., 2009; Lewis et al., 2014). Drier conditions, in contrast, induce aerobic conditions that facilitate carbon mineralisation by microbial communities. Organic matter decreases with depth as labile carbon is broken down relatively quickly (O’Rourke et al., 2015). In this study, the soil organic carbon (C_{org} %) was estimated from organic matter content measured using the LOI approach. This method is considered to be appropriate, as most studies that have compared organic matter from LOI to C_{org} % measured using elemental carbon analysis report a positive correlative relationship (R^2 = 0.59) (Kauffman et al., 2011; Howard et al., 2014). Estimating soil carbon content also accounts for bulk density, which is an indication of compaction (Keller and Håkansson, 2010; Holmquist et al., 2018). Therefore, soil carbon, decreases with depth as bulk density acts an indicator of pore space, and thus, soil water-holding capacity in the soil profile (Wang et al., 2016). An increase in compacted soil with depth, reduces pore spaces, and thus moisture, which consequently lowers overall soil organic carbon. Differences in bulk density along the depth profile could therefore account for variability in soil carbon. Additionally, in tidal habitats, surface soils are influenced by a number of processes that influence moisture and organic content at short temporal scales, while soil carbon represents longer term carbon accumulation.

The results of this study have important implications for the potential use of mangrove planting strategies as a carbon offset mechanism. Several investors, multinationals, and governments invest in mangrove expansion or conservation projects to offset their local CO_2 emissions through the use of carbon credits (Taillardat et al., 2018). This has been implemented in many countries, such as India through the Sundarbans mangrove restoration and reforestation project, where 6000 ha of mangroves were planted. This initiative was projected to store over 700,000 t of carbon over 20 years in both vegetation biomass and soil carbon pools (Wylie et al., 2016). Similarly, the Mikoko Pamoja project in Gazi Bay, Kenya is aimed at mangrove restoration, enhancement of ecosystem services, and provision of mangrove-related income to locals (Lovell, 2010). Through this project, a total of 2215 carbon credits (equivalent to one metric ton of carbon) is sold annually by the Edinburgh organisation, Plan Vivo (Lau, 2013; Wylie et al., 2016). However, the sale of carbon credits for the Mikoko Pamoja project only begun 13 years after the approval of the project as this was deemed a feasible timescale for generating the carbon credits through mangrove restoration (Huxham, 2013). The results from this study suggest that a much longer period is required for carbon accumulation by warm-temperate mangroves along the South African coastline. The feasibility of a carbon credit system based on mangrove restoration for South Africa is therefore uncertain. More baseline studies on carbon storage capacity of established South African mangroves are needed, and these should be focused on those systems along the subtropical east coast of KwaZulu-Natal.

The changes in soil carbon due to the expansion of mangroves into salt marsh habitats is variable. At the Nahoon Estuary, we found no significant differences between the soil carbon storage at the mangrove, ecotone, and salt marsh habitats. This implies that a longer time frame is required to allow for substantial organic matter accumulation. Other factors that influence soil carbon need to be further investigated such as local hydrologic and tidal patterns, surface elevation change and vertical accretion as well as microbial activity in the soil. It is therefore important to conduct further studies to observe any possible changes in soil carbon content along the salt marsh-mangrove ecotone, as well as to measure changes in soil carbon over time following mangrove expansion at different geographic locations. Carbon isotope analyses may also be considered to identify organic carbon sources to the system.

4 RESPONSES OF BLUE CARBON HABITATS TO SEA-LEVEL RISE

4.1 First Report on Surface Elevation Change in Mangrove Estuaries of South Africa

4.1.1 Introduction

Mangroves are ecologically resilient and stable as they have persisted through extreme environmental variability since prehistoric times (Alongi, 2008; Osland et al., 2016). The occurrence of these habitats within intertidal zones makes them vulnerable to sea-level rise (Mcleod and Salm, 2006; Webb et al., 2013). Rapid sea-level rise (SLR) has been correlated with large-scale losses of prehistoric mangroves (Ellison, 2008) and is one of the major threats to these ecosystems under contemporary global climate change scenarios (Webb et al., 2013; Alongi, 2015; Sasmito et al., 2016; Ward et al., 2016). Assessing the responses of mangroves to sea-level rise has been described as a priority for civil society, particularly in areas where the livelihoods of many people are threatened (FitzGerald et al., 2008; Hall, 2011; Webb et al., 2013). Quantifying the responses of coastal wetlands will assist in identifying sites under threat and this information can be used to inform conservation, mitigation and adaptation strategies.

During periods of sea-level rise, mangroves at the lowest intertidal zones are lost first through drowning as a result of increased length and frequency of inundation as rising sea-level shifts the position of the tidal frame relative to land (He et al., 2007; Grenfell et al., 2016). Change in salinity also causes losses as some species are particularly sensitive to these shifts (Lovelock et al., 2016). This leads to changes in species composition and loss of productivity and ecosystem services (Castañeda-Moya et al., 2013). The vulnerability of mangroves to sea-level rise is determined by several factors related to coastal geomorphology, regional oceanographic properties, local tidal range and sedimentation (Soares, 2009; Mcleod et al., 2010; Lovelock et al., 2015b). Under certain conditions, these habitats can also be resilient to changes associated with sea-level rise (French, 2006; Gedan et al., 2011). Where long-term monitoring has provided sufficient data series for analyses, the response of mangroves to sea-level rise has been attributed to two processes: positive surface elevation change (Soares, 2009; Woodroffe et al., 2016; Cahoon et al., 2019), and unrestricted landward expansion (Wasson et al., 2013; Di Nitto et al., 2014; Krauss et al., 2014).

Surface elevation change in coastal wetlands occurs through the processes of deposition or erosion of inorganic sediment and sedimentary organic matter (Krauss et al., 2014). Surface elevation change can be positive or negative, and coastal habitats under these different conditions are respectively described as undergoing accretion or subsidence (Cahoon et al., 1999, 2006). If the rate of positive surface elevation change is equal to or greater than the rate of sea-level rise, mangrove area will persist (Cahoon et al., 2006; Shepard et al., 2011). A positive surface elevation change rate lower than the rate of sea-level rise will result in drowning (Morris et al., 2002), and if there is a zero or negative surface elevation change, drowning can be expected at an even faster rate (Day et al., 2011; Voss et al., 2013). As surface elevation change is a process, several factors will determine whether mangrove ecosystems are accreting or subsiding. Feedbacks between physical and biological processes enhance the complexity of mangrove responses to sea-level rise and therefore prevent simple extrapolation of trends beyond the systems in which they have been measured (Cahoon et al., 2003; McKee et al., 2007; Krauss et al., 2014). These trends are also spatially variable, even within a single estuarine system as the positive surface elevation rate that must be achieved to avoid drowning is influenced by both sediment and tidal hydrodynamics.

Given the variability associated with mangrove responses to sea-level rise, it is important that baseline assessments of potential resilience are carried out in specific priority areas. South African mangroves are subtropical and occur at one of the southernmost limits of the global distribution range for this ecotype (Traynor and Hill, 2008). This region therefore represents an important ecotone, where mangrove habitats transition into salt marsh habitats at the climatic warm-temperate boundary. Surface elevation change has been measured in some salt marsh estuaries (Schmidt, 2013; Bornman et al., 2016; Raw et al., 2020) but, to date, there have been no studies reporting on these trends for mangroves. South Africa prioritizes the conservation of coastal wetlands through the National Biodiversity Act (Act No. 10 of 2004). Besides their biodiversity value, mangroves are also recognized for their economic value as they provide many ecosystem goods and services (Barbier et al., 2011). Mangrove habitats sustain rural livelihoods as they provide food resources as well as materials for building (Turpie et al., 2002; Adams et al., 2004). As global climate change continues to accelerate, the potential impact of the associated threats at a regional scale must be assessed. Climate change is one of many pressures acting on estuaries and should be viewed as an additional form of anthropogenic alteration. However, it is not always logistically feasible to carry out detailed regional-scale assessments on these threats. Selecting key representative areas or systems that can provide general information that can be extrapolated is therefore an important strategy.

The aim of this research was to set up a monitoring program for estimating long-term surface elevation trends in mangrove habitats along the South African coastline. As responses to sea-level rise are measured over the medium to long term (> 5 years), this research reports only on preliminary seasonal variability measured over two years of the WRC project period. The research has been set up in collaboration with the South African Environmental Observation

Network (SAEON), with the agreement that monitoring will continue beyond the timeframe of this specific project.

4.1.2 Methods

4.1.2.1 Selection Process for Estuaries to be Included in the Study

A few primary factors were considered in the selection process for the estuaries to be included in this study (Table 4.1.1). Other factors considered but not explicitly used to determine estuary selection included the following: proximity to a tide gauge; whether the mangroves are naturally occurring or planted; estuary geomorphology; size of mangrove area; availability of other data for the estuary (mangrove forest structure, sediment characteristics, LiDAR).

Table 4.1.1. Characteristics used to select estuaries suitable for inclusion in the surface elevation change study.

Factor	Suitable characteristics
Vegetation type	Mangrove trees are the dominant vegetation type in the estuary.
Accessibility	The estuary is relatively easy to access – it is not remote and the mangrove areas into which equipment must be carried are accessible.
Variability	The estuary is not extremely variable and there are no large-scale development or structural management plans in place for the foreseen future.
Disturbance	The estuary is not experiencing extreme events that will create large disturbances or destabilization of the sediment (extensive cattle browsing, footpaths, bait collection, etc.)
Security	The estuary is relatively safe for researchers to conduct the work and to ensure that benchmarks are not removed or vandalized by the public.

4.1.2.2 Selection Process for Location of Monitoring Stations at Each Estuary

Surface elevation change was measured at a scale of millimetres using a high-precision, portable mechanical levelling device. This device has been named the rod surface elevation table (RSET). It was designed for measuring elevation change occurring in different regions of the sediment profile as it can be attached to either shallow (< 1 m) or deep (driven to refusal) benchmarks.

The locations of the RSET benchmarks within each estuary were determined using hypothesis-based approaches while keeping certain limiting criteria in mind (Table 4.1.2). Each estuary was considered independently, and specific aims were used to develop the study design. The number of benchmarks placed within each estuary was determined by the study design to ensure replication while collecting data that are spatially representative for the mangrove area.

Available spatial data on the extent and distribution of the vegetation for each estuary and relevant information from recent publications were used to determine the location of the stations. In certain cases, the locations of sampling plots for other long-term studies were also considered.

Table 4.1.2. Criteria that influenced the placement of RSET benchmarks within each estuary.

Criterion	Description
Habitat type	Dense stands of mangrove trees are not suitable for placement of RSET benchmarks. Transition zones between mangroves and salt marsh were preferably selected.
Tidal frame position	Tidal position of the benchmarks was comparable so that this can be used as a factor to compare surface elevation change measurements.
Variability & Replication	Benchmarks were placed in triplicate to allow for replication within certain areas of interest. The number of replicated sites depends on the specific aim for each estuary.
Disturbance	Eroding banks or areas that had evidence of frequent access by people were avoided when placing the benchmarks. Areas with extreme bioturbation were also avoided.
Related studies	Areas where long-term monitoring for mangrove tree population structure data has been collected. The benchmarks were also placed to coincide with sampling locations for carbon storage.

4.1.2.3 Placement of the Monitoring Stations and Measuring Surface Elevation Change

The RSET benchmarks consisted of stainless-steel survey rods which were driven into the sediment until refusal (Cahoon et al., 1995, 1999, 2006). The surface elevation measurements were taken using a high-precision, portable mechanical levelling device that consisted of a horizontal arm with nine measuring pins which were lowered onto the surface of the wetland substrate (Figure 4.1.1). The distance from the top of each measuring pin to the arm was measured. Recordings were taken at eight fixed positions around each benchmark to provide a reference for the elevation change, and at each RSET benchmark every six months to capture short-term seasonal variability.

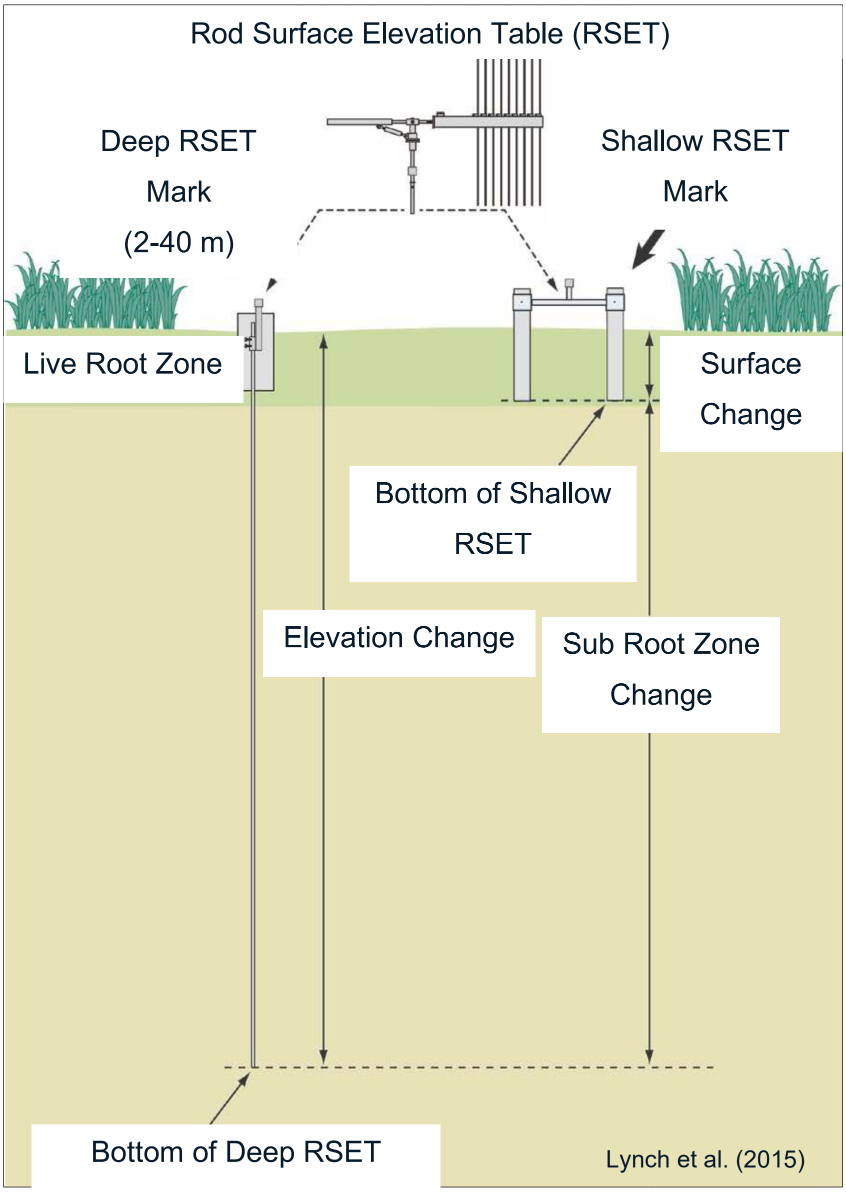


Figure 4.1.1. Schematic diagram of RSET benchmark components adapted from Lynch et al. (2015). For this study, only the deep RSET benchmarks were used.

4.1.3 Results and Discussion

4.1.3.1 Selected Estuaries and Locations of the RSET Benchmarks

Two mangrove estuaries in the Eastern Cape (Nahoon and Nxaxo) were selected for the placement of RSET monitoring stations. A field trip to groundtruth each estuary and identify the potential locations for individual stations was carried out in September 2017. A total of nine RSET benchmarks were placed in each estuary in January 2018 (Figure 4.1.2, Table 4.1.3).

Table 4.1.3. Location of RSET benchmarks in the Nahoon and Nxaxo estuaries.

Nahoon Estuary			
RSET	**GPS**		**Habitat Type**
1	32° 58′ 56.71″ S	27° 56′ 31.63″ E	Salt Marsh
2	32° 58′ 56.8″ S	27° 56′ 32.57″ E	Salt Marsh
3	32° 58′ 57.38″ S	27° 56′ 33″ E	Salt Marsh
4	32° 58′ 53.16″ S	27° 56′ 36.01″ E	Mixed/Ecotone
5	32° 58′ 53.76″ S	27° 56′ 36.57″ E	Mixed/Ecotone
6	32° 58′ 54.75″ S	27° 56′ 37.17″ E	Mixed/Ecotone
7	32° 58′ 50.58″ S	27° 56′ 40.73″ E	Mangrove
8	32° 58′ 51.65 S	27° 56′ 40.82″ E	Mangrove
9	32° 58′ 52.48″ S	27° 56′ 41.43″ E	Mangrove
Nxaxo Estuary			
RSET	**GPS**		**Habitat Type**
1	32° 34′ 46.95″ S	28° 31′ 19.38″ E	Salt Marsh (Landward Edge)
2	32° 34′ 45.69″ S	28° 31′ 21.56″ E	Salt Marsh (Landward Edge)
3	32° 34′ 45.47″ S	28° 31′ 23.73″ E	Salt Marsh (Landward Edge)
4	32° 34′ 43.92″ S	28° 31′ 32.96″ E	Mangrove (Main Channel)
5	32° 34′ 45.11″ S	28° 31′ 32.11″ E	Mangrove (Main Channel)
6	32° 34′ 45.94″ S	28° 31′ 30.86″ E	Mangrove (Main Channel)
7	32° 34′ 54.34″ S	28° 31′ 18.88″ E	Mangrove (Island)
8	32° 34′ 52.63″ S	28° 31′ 20.84″ E	Salt Marsh (Island)
9	32° 34′ 53.49″ S	28° 31′ 24.21″ E	Mangrove (Island)

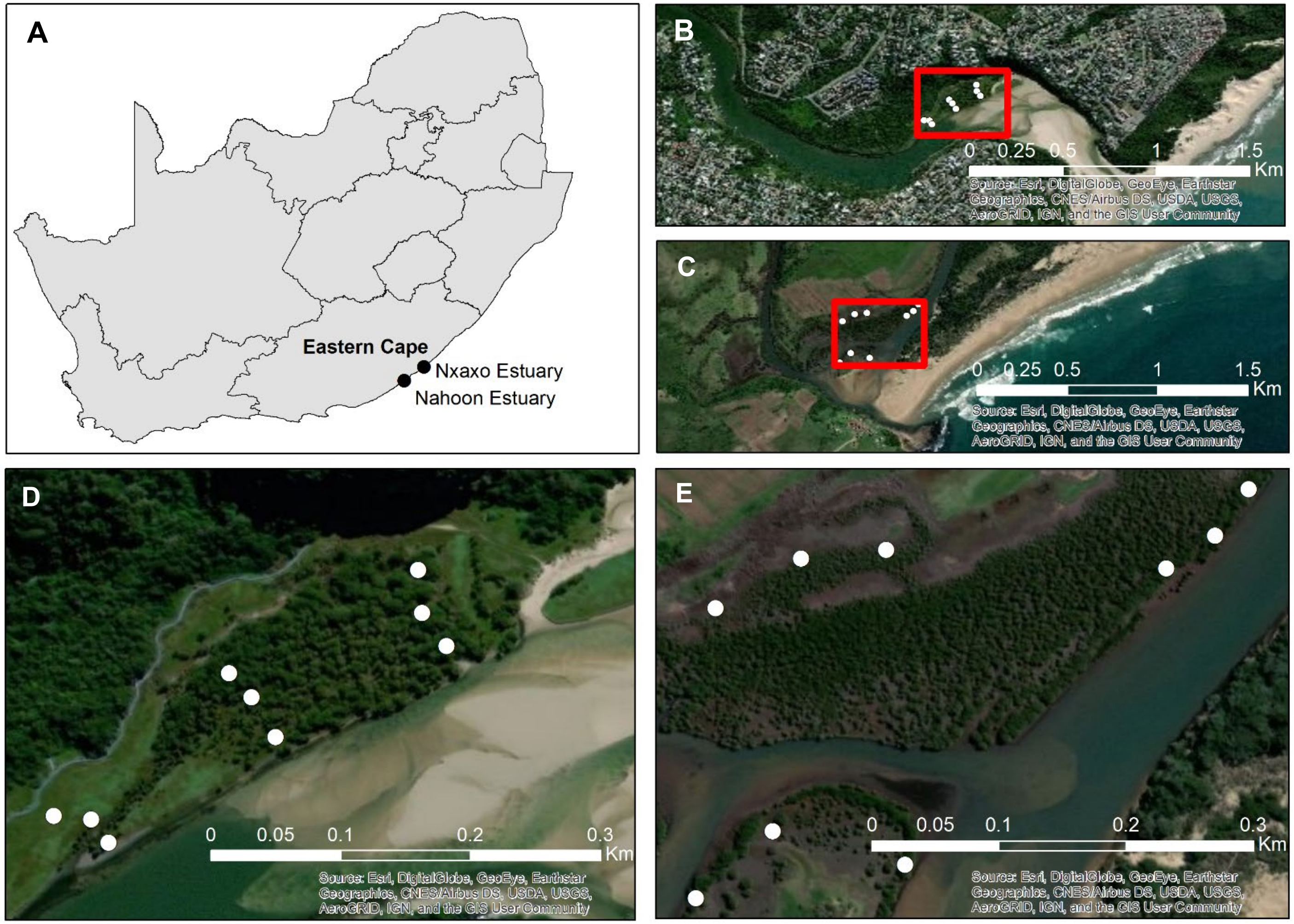


Figure 4.1.2. Mangrove estuaries selected for monitoring surface elevation change in South Africa (**A**); Location of RSET benchmarks placed in the Nahoon (**B**, **D**) and Nxaxo (**C**, **E**) estuaries.

The Nahoon Estuary is a permanently open, urban estuary within the Buffalo City Municipality of East London (Figure 4.1.2). The estuary falls within the warm temperate biogeographic region and is classified as microtidal, with an average tidal range of 0.76 m and a spring tide range of 1.6 m (Reddering, 1988). The estuary is recognized as the artificial southern distribution limit for mangroves in South Africa (Ward and Steinke, 1982). Annual precipitation varies between 200 mm and 600 mm, with most rainfall occurring during spring and summer. Geldenhuys et al. (2016) reported annual temperatures ranging from a minimum of 4.6°C to a maximum of 31.1°C recorded by SA Weather Services. The geomorphology of the estuary is described as a 'drowned river valley' and steep cliffs limit the floodplain areas.

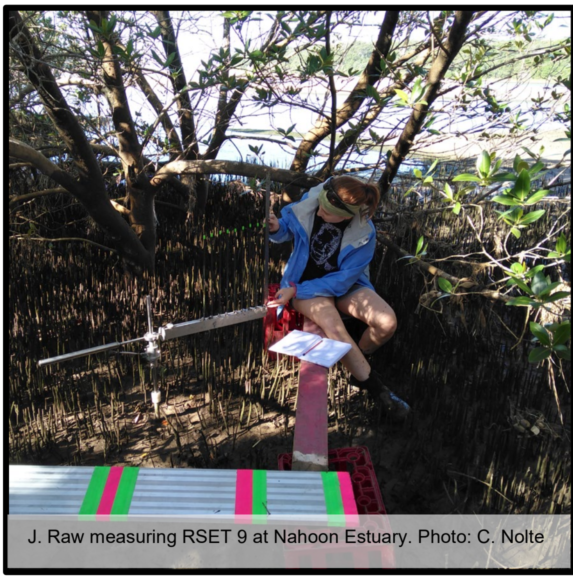
At the Nahoon Estuary, RSET monitoring stations were placed in adjacent mangrove and salt marsh habitats (Figure 4.1.2 D). The mangrove forest at this estuary was artificially established by transplanting trees from Durban Bay in 1969 (Hoppe-Speer et al., 2015b). Since then, the mangrove habitat has expanded by natural regeneration onto the adjacent tidal flat and salt marsh areas. At other study sites, mangrove encroachment into salt marsh has been associated with significant changes in surface elevation. The placement of the RSETs in this configuration at the Nahoon Estuary will allow for monitoring of changes in surface elevation over time if there is further mangrove expansion in the long term.

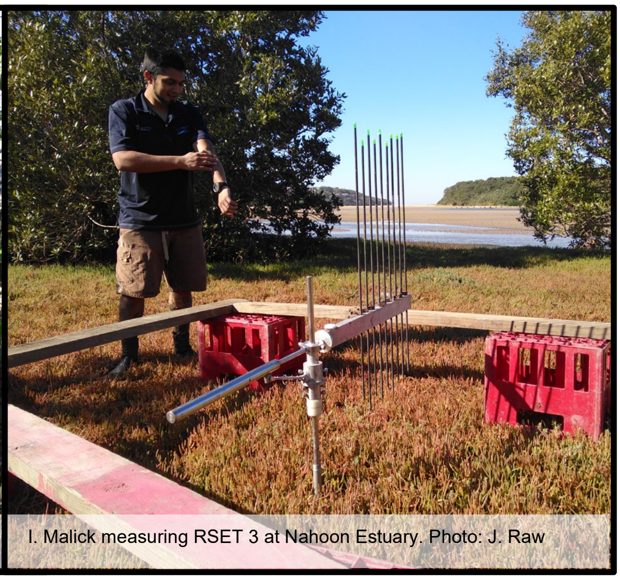
The Ngqusi/Nxaxo Estuary is a permanently open, rural estuary in KwaGcaleka, a former Transkei region of the Eastern Cape. The estuary falls within the warm temperate biogeographic region and has a catchment of ~ 134 km^2 (Harrison et al., 2001). The two arms of the estuary share a common mouth that is stabilized by a rocky promontory. The Nxaxo arm to the north is approximately 2 km in length, with an average depth of 1.4 m (Wasserman et al., 2010). The east bank of the Nxaxo arm supports a fringing mangrove forest adjacent to open grasslands that are farmed by the local community, while the west bank is covered by coastal dune forest.

At the Nxaxo Estuary, RSET monitoring stations were placed relative to long-term mangrove monitoring plots (Hoppe-Speer and Adams, 2015; Mbense, 2017) and the blue carbon sampling locations (Johnson et al., 2020; Section 3.1 of this report). Surface elevation dynamics at the Nxaxo Estuary could therefore be related to carbon storage and accommodation space, defined as "the space available for potential sediment accumulation" (Jervey, 1988). Within a mangrove environment, this space is greatest where mangroves occur closest to MSL, which is on the seaward edge (Woodroffe et al., 2016). The space is depicted in a vertical dimension and depends on the location of the sediment surface and the upper limiting tidal level (Woodroffe et al., 2016). The establishment of mangroves influences accommodation space as they promote sediment and organic matter accumulation.

Mangroves are restricted to a certain space within the tidal frame. The lower limit is defined by the lowest elevation at which seedlings can establish and persist to become trees (Balke et al., 2013). The upper limit is defined by the highest level of storm tides (Woodroffe et al., 2016). Mangroves have persisted in relation to relative sea-level rise as a result of continuous increases in accommodation space, which occurs as the ocean height increases and the land subsides as this enables vertical accretion (Woodroffe et al., 2016).

RSET benchmarks were placed so that the measured surface elevation trends were representative of the mangrove habitat at the Nxaxo Estuary (Figure 4.1.2 E, Table 4.1.3). Surface elevation change measured within the fringing mangrove area alongside the main channel provided an indication of accretion or subsidence, and information on available accommodation space at the lower limit of the mangroves' occurrence in the tidal frame. Surface elevation change measured at the mangrove-salt marsh interface provided information on the available accommodation space at the upper limit for mangroves. Finally, surface elevation change measured across the island, where there is a mosaic of mangrove and salt marsh habitats, provided comparisons between these habitats under the same inundation regime.


J. Raw measuring RSET 9 at Nahoon Estuary. Photo: C. Nolte


I. Malick measuring RSET 3 at Nahoon Estuary. Photo: J. Raw

4.1.3.2 Seasonal Surface Elevation Measurements

At the Nahoon Estuary, surface elevation change measured from 2018 to 2019 was similar between the mangrove, ecotone, and salt marsh habitats (Table 4.1.4). Surface elevation was relatively stable during the sampling period. Overall, at the mangrove RSETs, an average height increase of 0.464 cm was recorded, in comparison to average decreases of 0.1 cm and 0.06 cm at the ecotone and salt marsh RSETs respectively. Data could not be collected at RSET Station 4 in June 2018 as the benchmark unfortunately could not be located during the field sampling period. This benchmark was relocated and measured in subsequent sampling occasions.

At the Nxaxo Estuary, surface elevation change was more variable between mangrove and salt marsh habitats (Table 4.1.5). Surface elevation change was greater compared to the Nahoon Estuary over the sampling period. Overall, at the fringing mangrove RSETs, an average height increase of 1.35 cm was recorded, compared to an average height increase of 0.99 cm for RSETs on the island, and an average decrease of 0.137 cm for RSETs in the salt marsh habitat.

Evidence of sediment deposition at mangrove RSET benchmark (left) in comparison to salt marsh RSET benchmark(right) at the Nxaxo Estuary in November 2018. Photos: J. Raw

At both the Nahoon and Nxaxo estuaries, the largest increase in surface elevation height was measured at the mangrove RSETs. This could be related to local hydrodynamics. Tidal exchange influences surface elevation directly by deposition or erosion of organic and inorganic material (Cahoon et al., 2019). Mangrove rooting structures and salt marsh plants trap this material and accrete organic sediment which increases surface elevation (Krauss et al., 2014; C. Lovelock et al., 2015a). Vertical accretion in coastal wetlands is generally proportional to inundation volume when the source material is provided by tidal exchange (in comparison to land-derived material) (Rogers et al., 2006; Kirwan and Guntenspergen, 2010). Mangroves located in a lower portion of the tidal frame than salt marshes exhibit greater vertical accretion due to more frequent and greater depth of inundation (Rogers et al., 2006; Saintilan et al., 2009; Rogers et al., 2013). Differences between surface elevation heights measured at mangrove and salt marsh RSETs were less pronounced at the Nahoon Estuary. Here the intertidal area was small (< 100 m wide) and the potential for spatial heterogeneity was limited. In contrast, at the Nxaxo Estuary, the salt marsh habitat where the RSETs were located occurs behind the mangrove forest in an area that is only inundated by spring high tides. Ongoing work to measure the spatial variability of tidal inundation at the Nxaxo Estuary will provide suitable data to relate to surface elevation trends measured at the RSETs.

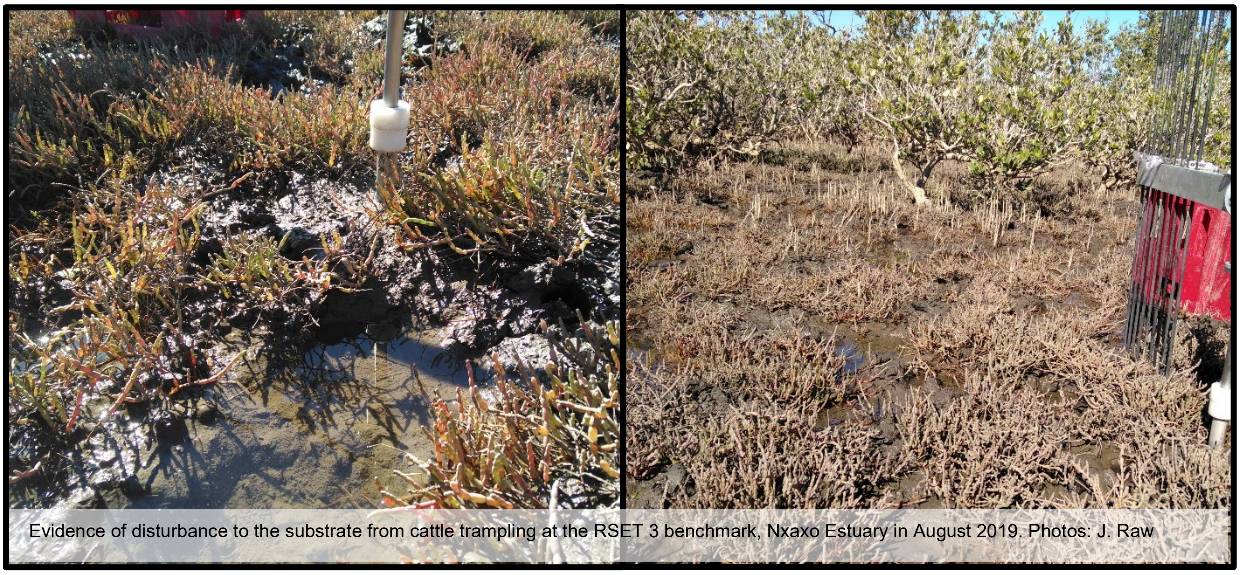
Table 4.1.4. Average (± SD) height measured at RSET benchmarks at the Nahoon Estuary.

RSET	**Habitat**	**Average (± SD) height (cm)**			
		June 2018	November 2018	June 2019	Difference/yr
1	Salt Marsh	36.7 ± 0.5	36.7 ± 0.5	36.3 ± 0.5	- 0.413
2	Salt Marsh	38.7 ± 0.9	39.0 ± 0.9	38.7 ± 0.9	+ 0.028
3	Salt Marsh	36.5 ± 1.2	36.7 ± 1.3	36.7 ± 1.3	+ 0.198
4	Ecotone	(no data)	39.4 ± 0.7	39.3 ± 0.7	- 0.131
5	Ecotone	39.1 ± 0.9	39.2 ± 0.7	39.0 ± 0.8	- 0.148
6	Ecotone	36.6 ± 1.1	36.7 ± 1.2	36.5 ± 1.1	- 0.022
7	Mangrove	36.9 ± 0.4	37.5 ± 0.4	37.4 ± 0.4	+ 0.524
8	Mangrove	34.4 ± 0.8	39.4 ± 0.8	39.6 ± 0.6	+ 0.360
9	Mangrove	37.8 ± 1.5	37.6 ± 2.1	38.3 ± 1.9	+ 0.507

Table 4.1.5. Average (± SD) height measured at RSET benchmarks at the Nxaxo Estuary.

RSET	Habitat	Average (± SD) height (cm)			
		June 2018	November 2018	August 2019	Difference/yr
1	Salt Marsh	39.2 ± 0.5	39.2 ± 1.2	39.2 ± 0.7	- 0.019
2	Salt Marsh	39.0 ± 1.1	38.6 ± 0.9	39.2 ± 1.1	+ 0.194
3	Salt Marsh	39.4 ± 1.5	38.8 ± 2.4	38.8 ± 1.9	- 0.587
4	Mangrove	38.4 ± 0.7	38.5 ± 0.8	40.2 ± 0.6	+ 1.75
5	Mangrove	37.8 ± 1.0	38.1 ± 0.9	39.1 ± 1.0	+ 1.86
6	Mangrove	36.1 ± 1.2	36.5 ± 1.3	37.2 ± 1.1	+ 1.72
7	Island (Mangrove)	38.1 ± 1.4	38.3 ± 1.2	39.2 ± 1.5	+ 1.07
8	Island (Salt Marsh)	35.9 ± 0.7	36.1 ± 0.7	36.6 ± 0.7	+ 0.653
9	Island (Mangrove)	40.2 ± 2.2	40.4 ± 2.1	41.5 ± 1.8	+ 1.27

Surface elevation loss recorded in the salt marsh habitat at the Nxaxo Estuary (Table 4.1.5) is expected to be the result of disturbance to the sediment surface. On each sampling occasion, disturbance to the surface sediment in the form of trampling by cattle has been recorded at these sites, particularly at RSET 3. Trampling by large grazers, such as cattle, compacts the soil and can have a direct negative effect on salt marsh accretion (Elschot et al., 2013; Nolte et al., 2015). Salt marsh areas trampled by cattle can have significantly higher dry bulk density as increased pressure on the soil surface reduces soil air spaces (Nolte et al., 2013; Rodríguez-Medina and Moreno-Casasola, 2013). Higher bulk density in salt marsh soils exposed to cattle has been related to reduced soil organic matter (Di Bella et al., 2015), and this could further be related to reduced vertical accretion (Cahoon et al., 2011). Ongoing monitoring of disturbance by cattle at these sites is essential to interpret the surface elevation data recorded over the long term (> 5 years). Salt marshes that receive enough sediment deposition through tidal inundation can exhibit positive surface elevation change, even if there are trampling effects by cattle, depending on the stocking density (Nolte et al., 2013). However, as the salt marsh area at the Nxaxo Estuary is positioned relatively high within the tidal frame, it is unlikely this will be the case here. Trampling by cattle could therefore significantly reduce the potential for these areas to respond to sea-level rise in the long term.



Evidence of disturbance to the substrate from cattle trampling at the RSET 3 benchmark, Nxaxo Estuary in August 2019. Photos: J. Raw

4.1.4 Conclusion

The surface elevation data collected has the potential to be used for ongoing and future projects at both the Nahoon and Nxaxo estuaries. This short-term dataset can be used to develop new research questions aimed at investigating local drivers of vertical accretion and surface elevation change. One of the most critical gaps that needs to be filled so that this data can be directly interpreted and related to sea-level rise trends is in the form of digital elevation models (DEMs). For the Nahoon Estuary, a DEM can be constructed from LiDAR data (surveyed in 2013) made available by the Buffalo City Municipality. The relative height of each RSET benchmark to mean sea-level has also been measured by surveyors of DME Geomatics. The next phase for research at this estuary would be to model changes in elevation over time using a predictive inundation model (such as the Sea-Level Affecting Marshes Model, SLAMM).

For the Nxaxo Estuary, future research can take a different approach, as LiDAR data are not currently available for this remote system. As this estuary has been selected as the case study site for measuring blue carbon (Section 3.1 of this report), the surface elevation data can be interpreted alongside these estimates. Vertical accretion rates can also be used to estimate carbon sequestration potential (Section 6 of this report). Longer-term accretion can also be determined through Pb^{210} isotope dating of the soil and this can be compared to current rates of change. Finally, this data can be interpreted alongside ongoing studies that are focused on the roles of inundation and sedimentation on mangrove growth and dynamics.

4.2 Salt Marsh Surface Elevation and Sea-Level Rise at the Knysna Estuary

Raw JL, Riddin T, Wasserman J, Lehman TWK, Bornman TG, Adams JB (2020) Salt marsh elevation and responses to future sea-level rise in the Knysna Estuary. *African Journal of Aquatic Sciences* 45(1), doi: 10.2989/16085914.2019.1662763

4.2.1 Introduction

Salt marshes are highly productive coastal wetlands that form along low-energy continental shorelines or as back-barrier environments in sheltered estuaries. Globally, these habitats are valued for providing ecological services that include raw materials and food, coastal protection in the form of wave attenuation and erosion control, breeding grounds and nursery habitats for threatened and commercially important species, water purification, and carbon sequestration (Barbier et al., 2011; Shepard et al., 2011; Costanza et al., 2014a; Barbier, 2015). Along with other types of coastal wetlands, salt marshes face pressures from global climate change and local anthropogenic impacts. Specifically, accelerated sea-level rise in combination with human impacts on sediment supply and hydrology have been identified as the largest threats (Cahoon et al., 2019). Despite this, recent studies have shown that at regional and local scales, salt marshes can respond to sea-level rise and under certain environmental conditions can exhibit resilience to this threat (Kirwan and Megonigal, 2013; Baustian and Mendelssohn, 2018; Schuerch et al., 2018).

The long-term persistence of salt marsh habitats is determined by interactions between sea level, surface elevation, primary production and sediment accretion (Morris et al., 2002). Biogeomorphic controls that maintain a surface elevation equilibrium relative to sea-level therefore determine the responses of salt marshes to sea-level rise (Cahoon et al., 2006; Cahoon, 2015). Significant loss of surface elevation can lead to shifts in salinity and inundation regimes that regulate salt marsh zonation, growth and productivity (Mendelssohn and Morris, 2000; Contreras-Cruzado et al., 2017). Vegetation die-back following abiotic shifts reduces soil volume and organic matter oxidation which directly facilitates elevation loss (Day et al., 2011; Temmerman et al., 2012). These effects can lead to complete conversion to mudflat and open water areas over time. Maintaining surface elevation is therefore critical for salt marsh survival.

Besides surface elevation responses, salt marshes are also expected to migrate landwards in response to sea-level rise (Enwright et al., 2016; Kirwan et al., 2016b). If sea-level rise rates surpass salt marsh surface elevation gains, a shift in the tidal frame will drive habitat conversions, with lower intertidal areas becoming subtidal, and upper intertidal species encroaching the terrestrial boundary (Wasson et al., 2013; Cahoon, 2015; Fagherazzi et al., 2019). When sea-level rise accelerates and surface elevation gain is restricted, landward migration becomes the only adaptation option available for salt marshes (Borchert et al., 2018). Salt marsh landward migration is limited by anthropogenic developments at the terrestrial boundary. These barriers introduce an additional pressure to salt marsh habitats by compounding the effects of sea-level rise through ‘coastal squeeze’ (Pontee, 2013; Borchert et al., 2018). Accelerated sea-level rise has been predicted to drive significant global losses of coastal wetlands, including salt marshes (Crosby et al., 2016; Spencer et al., 2016; Raposa et al., 2016). As physiographic settings and anthropogenic developments are extremely variable between salt marsh habitats globally, it is important that baseline assessments of salt marsh resilience to sea-level rise are carried out in specific conservation priority areas.

The Knysna Estuary, situated on the southern Cape coastline, has long been recognized as one of the most ecologically important estuaries in South Africa and is therefore prioritized for conservation (Turpie et al., 2002). The salt marsh habitats of this estuary comprise 546 ha of intertidal salt marsh and 120.7 ha of supratidal salt marsh (Adams et al., 2019), which together constitute the third largest salt marsh area in South Africa. The Knysna Estuary is protected within the Garden Route National Park which is managed by South African National Parks (SANParks) under the National Environmental Management: Protected Areas Act (NEMPAA) 57 of 2003. Despite the ecological importance of the salt marsh habitats, the Knysna Estuary is subject to various pressures, including the construction and development of marinas, roads and bridges. This leads to degradation of the estuarine functional zones by direct habitat loss that in turn threatens to the health of the estuary. The recent National Biodiversity Assessment has highlighted the need for greater protection of sensitive salt marsh habitats at this estuary (Adams et al., 2019).

Sea-level rise is recognized as a significant threat to the South African coastline (Theron and Rossouw, 2008) with the eustatic sea-level trend for the southern Cape region estimated as increasing by 1.57 $mm.yr^{-1}$ (Mather et al., 2009). A comprehensive assessment of salt marsh responses to sea-level rise has previously been carried out for the Swartkops Estuary in the Eastern Cape (Bornman et al., 2016). The adaptive capacity of salt marsh to sea-level rise for the broader southern Cape region is however largely unknown. Given the ecological and economic importance of salt marsh habitats at the Knysna Estuary, the aims of this study were 1) to provide an updated assessment of salt marsh area cover and distribution; 2) to assess surface elevation change, sediment characteristics, and relative sea-level rise experienced at salt marsh habitats; 3) to determine the potential loss of salt marsh to new developments and coastal squeeze. The results of this study can be used to inform predictions of salt marsh responses to sea-level rise that will contribute towards resilience of this habitat in the future.

4.2.2 Materials and Methods

4.2.2.1 Study Site Description

The Knysna Estuary in the Western Cape Province occurs within the warm-temperate biogeographic region of the South African coastline (Figure 4.2.1). It is classified as an estuarine bay and is tidally dominated. Salt marsh occurs along an elevation gradient ranging from 0.4 to 3.1 m MSL (above mean sea-level) with distinct subtidal, intertidal and supratidal zones (Schmidt, 2013).

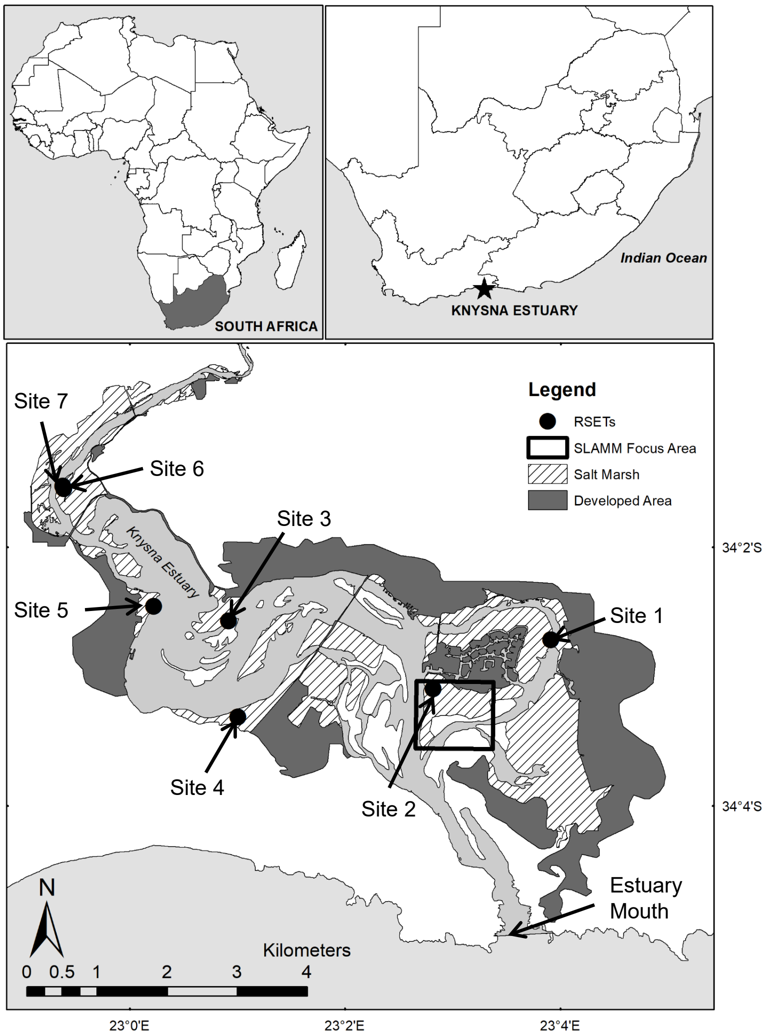


Figure 4.2.1. Location of the Knysna Estuary, South Africa. Black circles indicate Rod Surface Elevation Table (RSET) monitoring sites. Hatched areas indicate salt marsh and grey areas indicate development. The black square delineates the area in which the Sea-Level Affecting Marshes Model (SLAMM) was applied.

4.2.2.2 Salt Marsh Habitats

A desktop assessment in 2018 consisting of past studies (Maree, 2000; Schmidt, 2013) and extensive ground truthing were used to map the current extent of salt marsh habitats as well as the adjacent intertidal areas of the Knysna Estuary. The desktop assessment used satellite imagery from Google Earth (Google Inc., 2018) to identify intertidal habitats. Following this, Avenza Maps version 3.2.3 (Avenza Systems Inc., 2017) was used on a digital tablet to record geotagged notes and ground control points in the field. The study area was manually digitized in ArcMap version 10.6 (ESRI, 2018). Error was assessed using the ground control points to verify habitat boundaries in the field. Vegetated habitats were classified based on the dominant plant species, while unvegetated habitats were classified based on substrate type. The study area extended up to the N2 bridge.

The extent of the developed perimeter was determined by identifying the presence of artificial structures in the study area using Google Earth satellite imagery. These included bridges and causeways, harbours and marinas, jetties, riprap and slipways. Only structures that extended into the subtidal zone were included as part of this assessment. The locations of these structures were also confirmed with ground truthing using Avenza Maps and digitized in ArcMap as described above. The total area of these structures, and their coverage along the perimeter of the estuary were then calculated to assess restrictions to salt marsh landward migration under sea-level rise.

4.2.2.3 Relative Sea-Level Rise

The relative sea-level rise (RSLR) for the Knysna Estuary was calculated from tidal height data measured at a tide gauge located at Thesen's Island from 1960 to 2017. Vertical crustal movements, barometric pressure, mesoscale oceanographic events and the metonic cycle were not considered as we did not aim to investigate factors driving sea-level rise. Instead, we related the measured tidal height data to salt marsh elevations. These data were collected by the South African Navy Hydrographic Office (SANHO) and accessed through the Permanent Service for Mean Sea Level database (Holgate et al., 2012; PSMSL, 2018). The tidal height data retrieved from this database have been corrected to a revised local reference so that all records for the tide gauge are relative to the same chart datum. This allows modelling of sea-level relative to land. The RSLR per annum ($mm.yr^{-1}$) was estimated using the time series of monthly relative sea-level heights (mm above the revised local reference). Monthly averages smooth out short-term tidal harmonics but retain inter-annual variability. Missing values in the time series were imputed using a simple linear interpolation (Moritz and Bartz-Beielstein, 2017).

After checking the assumption of stationarity, the data were fit with an Auto-Regressive Integrated Moving Average (ARIMA) model. This approach allows the analysis of a trend in data that have a moving average and is a more robust alternative to fitting a linear regression. This method has previously been used for tide gauge data that is related to surface elevation trends in wetlands (Rogers et al., 2012; Bornman et al., 2016). The order of the autoregressive component was estimated by maximum likelihood using stepwise selection (Hyndman and Khandakar, 2008; Hyndman et al., 2018). The Akaike Information Criterion (AIC) was used to determine the best fit model. The relative sea-level data were then fit with a generalized least squares (GLS) model (Pinheiro et al., 2018) that included a year/month predictor variable and the autoregressive component identified in the ARIMA model as a correlation structure. The slope of the model provided the value for the RSLR trend. The RSLR for the Knysna Estuary was calculated separately for the whole time series (1960-2017) and for the time period that surface elevation change has been measured (2009-2017). Although the RSET period does not cover a full metonic cycle, this time frame was used to relate recent trends to the longer tide gauge series.

4.2.2.4 Surface Elevation Change and Sediment Characteristics

Surface elevation change was measured using the Rod Surface Elevation Table (RSET) method (Cahoon et al., 2002). The elevation of each RSET was calculated by a land surveyor based on a reference point in the town of Knysna, i.e. a National Geo-Spatial Information (NGI) survey mark (brass stud in metal casing) at 1.715 MSL. Seven permanent RSET stations were measured annually from 2009-2011 (Schmidt, 2013). These sampling stations were re-visited in 2018 to assess long-term surface elevation change. This method is generally supplemented by establishing a marker horizon to estimate vertical accretion (Cahoon et al., 2006; Lynch et al., 2015). Multiple attempts to create an artificial soil layer with Feldspar, calcium carbonate shell fragments, quartz granules and kaolin clay were all unsuccessful. We suspect these materials did not persist as a result of thorough tidal flushing experienced in the lower intertidal zone.

The average rate of surface elevation change per site was calculated by subtracting the average surface elevation for the first measurements (2009) from the average surface elevation for the most recent measurements (2018) and dividing this by the elapsed time in years.

The relative sea-level rise experienced at each of the RSET stations was calculated using the method of Cahoon (2015) as follows:

$$RSLR_{wet} = RSLR - VLM_w$$

where $RSLR_{wet}$ is the relative sea-level rise experienced at the wetland, RSLR is the relative sea-level rise measured by the tide gauge, and VLM_w is the surface elevation change which has been measured at each RSET station.

Sediment samples were collected in April 2018 at the RSET sampling stations in the morning of the spring low tide. Samples were collected as a component of the long-term monitoring of salt marsh responses to sea-level rise at the Knysna Estuary which began in 2009 (Schmidt, 2013). In the original study, sediment cores were sampled along elevation transects. As the current study was focused on lower intertidal salt marsh, the location of the 2018 sediment cores was restricted to the *Spartina maritima* zone and in proximity to the RSET benchmark. The 2009 core of closest proximity was used for comparison at the respective sites.

Sediment cores were collected in triplicate to 50 cm depth at each sampling location using a Russian peat corer (internal radius = 27 mm). Cores were sectioned in the field at 0-15 cm, 15-30 cm, and 30-50 cm intervals, placed into sealed plastic bags, and kept cool at 4°C until laboratory processing. Sediment moisture content was measured by oven-drying samples at 100°C for 48 hours and calculating the percentage mass lost from the original wet weight (Gardner, 1965). Sediment organic content was measured by subsequently combusting the oven-dried samples in a muffle furnace at 550°C for 8 hours and calculating the percentage mass lost from the original dry weight (Briggs, 1977).

4.2.2.5 Spatial Assessment of Coastal Squeeze

The Sea-Level Affecting Marshes Model (SLAMM) version 6.7 was used to assess the potential for coastal squeeze at the Knysna Estuary (Clough et al., 2016). This model has been extensively applied to provide spatially explicit predictions that indicate the responses of coastal habitats to the IPCC sea-level rise scenarios. It allows for the incorporation of additional site-specific data, described below. A specific focus area of the estuary was selected for the SLAMM approach as modelling the entire estuary was not feasible due to the resolution of the available data. The focus area was located in front of Thesen’s Island, where residential properties occur at the salt marsh-terrestrial boundary (Figure 4.2.1).

The SLAMM approach simulates the primary processes that affect the survival of salt marsh habitats under threat from sea-level rise, namely inundation, erosion, overwash, saturation, salinity and accretion. To achieve this, SLAMM requires information on the elevation, slope, vegetation communities, accretion rates and historic sea-level rise. For the Knysna Estuary, elevation (in meters) was derived from a digital elevation model (DEM) obtained from 2013 LiDAR surveys at 5 m spatial resolution with a 0.2 m vertical accuracy. The slope surface (in degrees) could then be obtained from this DEM using the Spatial Analysis Toolbox in ArcMap. The digitized salt marsh distribution data described above was used to describe the vegetation community for SLAMM. As SLAMM is based on vegetation classes identified by the US National Wetlands Inventory, analogous vegetation classes at Knysna were identified based on similarities in their distribution along the tidal frame as identified from the vegetation map and the DEM (Table 4.2.1). To incorporate developed areas into the spatial model, the South African cadastral layer obtained from the Chief Surveyor-General (http://csg.dla.gov.za/) for the focus area was joined to the vegetation layer. The surface elevation measured at the RSET station within the focus area for SLAMM was used to inform the accretion feedback in the model. The RSLR calculated from the tide gauge was used as the historic sea-level trend for the site.

The SLAMM software allows the selection of SLR scenarios from the IPCC 2001 Special Report on Emissions Scenarios (SRES) (Clough et al., 2016). In this study, we selected the maximum SLR predicted under the IPCC A1B scenario. This translates to a 0.694 m rise in sea-level by 2100 in the SLAMM modelling framework. This scenario was selected because it approximates a SLR estimate between 0.4 m and 1.6 m, which is the range that has been used previously to assess sea-level vulnerability in coastal towns along the southern coast of South Africa. The 2014 IPCC Assessment Report provides sea-level rise predictions related to scenarios based on Representative Concentration Pathways (RCPs) (Church et al., 2013). A comparison between predictions under the SRES and RCP frameworks is provided in the IPCC 2013 Annex II report. However, comparing salt marsh responses under different SLR scenarios was not within the scope of this study. The simulations were run at a 25-year time step using alternatives to either protect, or not protect, the developed areas from sea-level rise. Details and justifications for inputs specified for the SLAMM are provided in the Supplementary Material (Appendix I, Section A1.2.2).

Table 4.2.1. SLAMM categories from Clough et al. (2016) assigned to analogous salt marsh vegetation recorded and mapped in the Knysna Estuary in 2018.

SLAMM category	SLAMM description	Analogous habitat
Developed dry land	Can be defended against sea-level rise	Area covered by South African cadastre
Transitional Marsh	Estuarine intertidal shrub-scrub	Upper intertidal salt marsh (*Bassia diffusa*)
Regularly Flooded Marsh	Receives regular tidal flooding	Lower intertidal salt marsh (*Salicornia* spp., *Spartina maritima*, *Triglochin* spp.)
Tidal Flat	Estuarine intertidal mud or organic aquatic bed	Exposed intertidal seagrass (*Zostera capensis*)
Estuarine Open Water	Estuarine subtidal areas	Estuarine open water
Tidal Creek	Estuarine intertidal streambed	Tidal creek
Irregularly Flooded Marsh	Estuarine intertidal salt marsh	Upper intertidal salt marsh (*Plantago crassifolia*)

Besides coastal squeeze, salt marsh habitats in the Knysna Estuary also face threats from future developments within the tidal frame. To assess these threats over the extent of the entire estuary, the current South African cadastral layer was overlaid onto the vegetation distribution layer in ArcMap. Areas currently zoned within the cadastre but undeveloped were identified. The total area of salt marsh habitat types that occurs in these areas was calculated as a percentage that could be lost if these areas were developed.

4.2.2.6 Statistical Analyses

Data for sediment moisture and organic content did not meet the normality assumption for parametric statistical tests. Instead, Kruskal Wallis tests were used to compare main effects and interactions between "Site" and "Depth" for each of these dependent variables. If significant differences were found between groups, the Nemenyi post-hoc test with Tukey distribution was used for pairwise comparisons (Pohlert, 2014). The Chi-square distribution was used if there were ties in the pairwise groupings. Sediment moisture and organic content were also compared with those from samples taken in 2009 (Schmidt, 2013).

The surface elevation data also did not meet the normality assumption for parametric statistical tests. The surface elevation change ($mm.yr^{-1}$) was compared over time at each RSET station using a Friedman Rank Sum test. All analyses, including the time-series analysis for tide gauge data, were performed in R version 3.5.2 for Windows (R Core Team, 2018).

4.2.3 Results

4.2.3.1 Intertidal Habitats and Artificial Structures

An area just over 888 ha was mapped for the Knysna Estuary below the N2 bridge. In this area, 11 distinct habitats were identified and classified (Figure 4.2.2, Table 4.2.2). Eight of these were dominated by typical South African salt marsh species, such as *Bassia diffusa*, *Plantago crassifolia* and *Salicornia* spp. The salt marsh habitats covered an area of 778.6 ha (Table 4.2.2). Submerged macrophyte areas dominated by *Zostera capensis* covered an extensive area (447.3 ha) followed by the salt marsh species *Triglochin* spp. (155.7 ha) and *Spartina maritima* (107.4 ha) respectively. Unvegetated areas (sediments and natural rocky areas) covered an area of 94.4 ha. Reeds and sedges (*Juncus kraussii* and *Phragmites australis*) accounted for the remaining habitat.

Table 4.2.2. Area cover by intertidal habitats within the Knysna Estuary.

	Habitat/species	**Area (ha)**	**Total Area (ha)**
Salt marsh	*Bassia diffusa*	30.9	778.6
	Plantago crassifolia	30.5	
	Salicornia spp.	1.5	
	Spartina maritima	107.4	
	Sporobolus virginicus	5.3	
	Triglochin spp.	155.7	
Submerged macrophytes	*Zostera capensis*	447.3	
Reeds and Sedges	*Juncus kraussii*	14.1	14.9
	Phragmites australis	0.8	
Unvegetated areas	Natural rocky area	1.4	94.4
	Unvegetated sediment	93.0	
			887.9

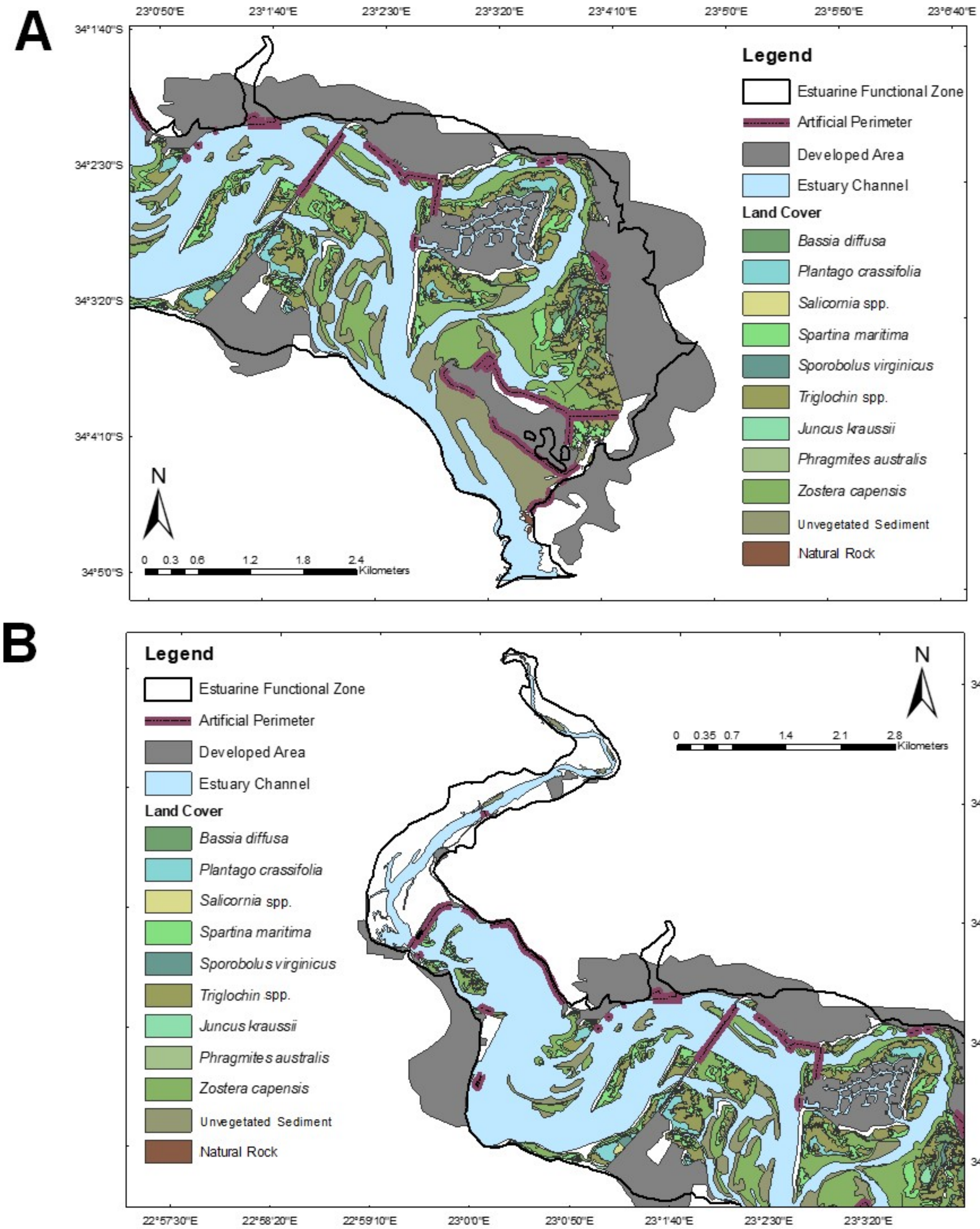


Figure 4.2.2. Distribution of the intertidal habitats characterized by the dominant salt marsh species in the lower (**A**), middle and upper (**B**) reaches of the Knysna Estuary. Artificial structures along the perimeter are also demarcated. Only areas within the Estuarine Functional Zone (delineated by the 5 m contour) are mapped.

Artificial structures accounted for 20.73 km of the perimeter along the Knysna Estuary and made up a total area of 106.58 ha (Table 4.2.3, Figure 4.2.2). There are two major marinas in the estuary (Thesen's Island and Waterfront marinas) and two small harbours (Leisure Island and Ashmead Resort). The marinas and harbours comprised most (103.41 ha) of the total artificial structure area in the estuary. Besides these areas, 27 wooden jetties were also identified, and these had a combined perimeter of 4.02 km. Finally, the seven slipways that were identified contributed the least towards artificial structures with a combined perimeter of 423 m.

Table 4.2.3. Area and perimeter of artificial structures in the Knysna Estuary.

	Number of structures	**Total Perimeter (km)**	**Total Area (ha)**
Bridges and causeways	6	2.69	1.09
Harbours and marinas	4	7.54	103.41
Jetties	27	4.02	0.66
Riprap	7	6.06	1.31
Slipways	7	0.42	0.11
		20.73	**106.58**

4.2.3.2 Relative Sea-Level Rise

The gaps in the tide gauge data series (Figure 4.2.3) represent errors in the dataset that could not be corrected and are common across multiple tide gauge datasets along the South African coastline (Mather et al., 2009). The peak values recorded between 1998 and 2002 have also previously been noted (Mather et al., 2009). These values were retained in the dataset as they did not have a large effect on the overall trend and removing them reduced the data coverage to below 60% of the time period.

The autoregressive components estimated in the ARIMA models were different when analysing the tide gauge time series data over the different periods. For the tide gauge record period (1960-2017), the model was defined as a differenced first-order autoregressive model as the series was not stationary (the mean was not constant over time). For the study period (2009-2017), the model was defined only with a first order autoregressive component as the series was stationary.

The RSLR trend for the tide gauge record period (1960-2017) was calculated as 2.19 ± 1.3 $mm.yr^{-1}$ while the RSLR for the study period was calculated as 0.25 ± 0.8 $mm.yr^{-1}$ (Figure 4.2.3). Detailed results for the ARIMA models are provided in the Supplementary Material (Appendix I, Section A1.2.2).

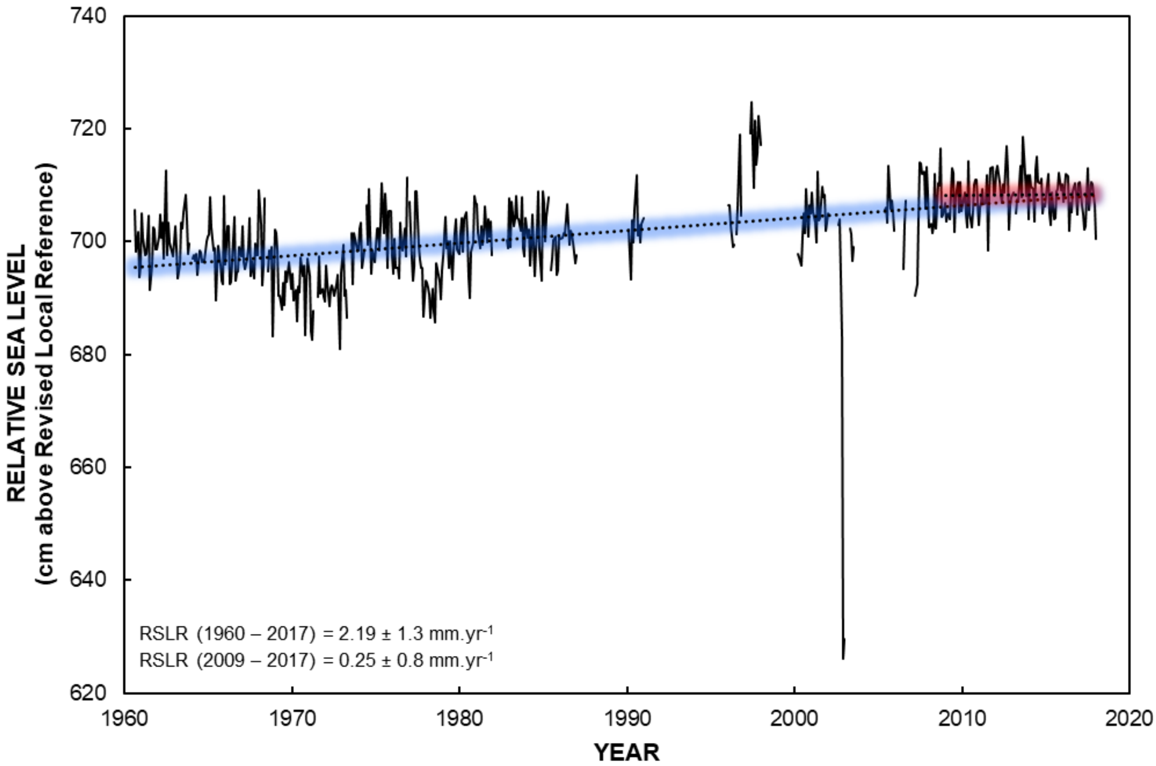


Figure 4.2.3. Monthly mean relative sea-level for the Knysna Estuary from 1960 to 2017 provided for the tide gauge at Thesen's Island from the Permanent Service for Mean Sea Level database (PSMSL 2018). The blue line indicates the trend for the entire tide gauge record period, while the red line indicates the trend for the period over which salt marsh surface elevation has been measured.

4.2.3.3 Surface Elevation Change and Sediment Characteristics

At the Knysna Estuary, all RSET stations were located within the lower intertidal salt marsh (*Spartina maritima* zone). Therefore, only two stations (those in the upper reaches of the estuary) were above the surveyed mean sea-level (Figure 4.2.4A). Surface elevation was significantly different over time between the RSET stations (Friedman χ^2 = 23.673, *df* = 6, $p < 0.05$), this is further indicated by the rates of change at each RSET (Figure 4.2.4B).

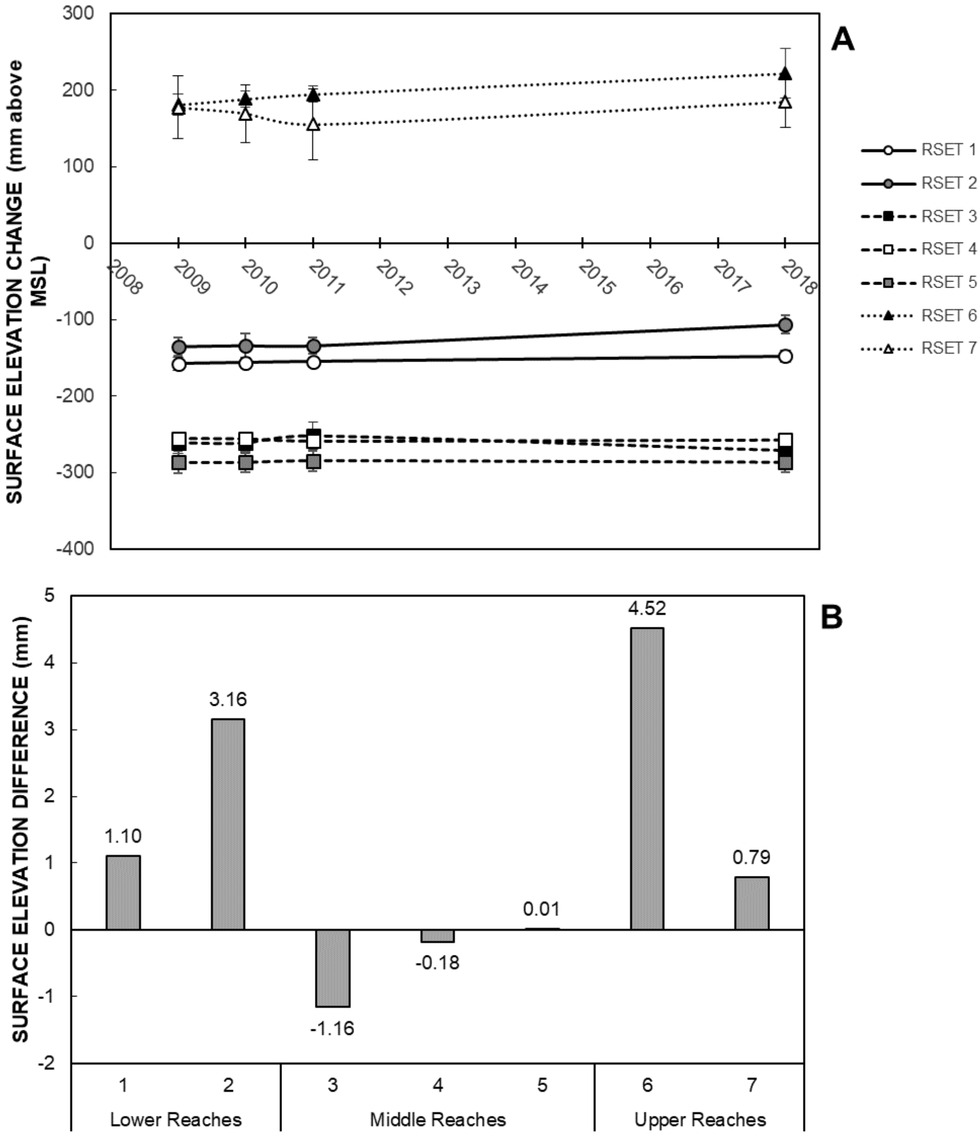


Figure 4.2.4. Surface elevation change (mm) measured at Rod Surface Elevation Table (RSET) stations in the Knysna Estuary presented as the average height above mean sea-level (MSL) at different sampling occasions (**A**), and the average change per station over the 9-year period (**B**).

The largest increase in surface elevation (VLM_w) was measured at Site 6 in the upper reaches, while the largest deficit was measured at Site 3 in the middle reaches (Figure 4.2.4, Table 4.24). The $RSLR_{wet}$ indicates the net change in elevation at a specific site given the surface elevation at that site and the relative rate of sea-level rise (RSLR) over different time periods. Overall, $RSLR_{wet}$ was lower when calculated using the RSLR estimate for the study period (2009-2017). Negative values for $RSLR_{wet}$ indicate surface elevation change at a rate above that of the RSLR. These sites are therefore gaining surface elevation (by accretion) at a rate faster than the RSLR. Sites 3 and 4 experienced $RSLR_{wet}$ rates higher than the RSLR due to loss in surface elevation. These sites are losing surface elevation (by erosion or subsidence) at a rate faster than RSLR. At Sites 1 and 7, the rate of surface elevation change reduced $RSLR_{wet}$ over the study period, but to a lesser effect when considering the long-term trend in RSLR (1960-2017). These sites are gaining surface elevation, but at a rate not fast enough to keep pace with RSLR.

Table 4.2.4 Wetland relative sea-level rise ($RSLR_{wet}$) for the seven Rod Surface Elevation Table (RSET) stations located in the Knysna Estuary. $RSLR_{wet}$ is calculated using surface elevation change (VLM_w) and the relative sea-level rise (RSLR) measured at the tide gauge over different time periods.

			1960-2017 (RSLR = 2.19 mm.yr^{-1})	**2009-2017 (RSLR = 0.25 mm.yr^{-1})**
	RSET	**VLM_w (mm.yr^{-1})**	**$RSLR_{wet}$ (mm.yr^{-1})**	**$RSLR_{wet}$ (mm.yr^{-1})**
Lower Reaches	1	1.10	1.09	-0.85
	2	3.16	-0.97	-2.91
Middle Reaches	3	-1.16	3.35	1.41
	4	-0.18	2.37	0.43
	5	0.01	2.18	0.24
Upper Reaches	6	4.52	-2.33	-4.27
	7	0.78	1.40	-0.53

Sediment organic content was variable between the lower, middle, and upper reaches at the Knysna Estuary (Figure 4.2.5A). There was an overall increase in the average (± SD) sediment organic matter in the surface layer (0-15 cm) at all sites from 2009 to 2018 (Kruskal Wallis $\chi^2 = 28.11$, $df = 13$, $p = 0.008$). As only surface sediment was collected in 2009, depth intervals could not be compared over time. In 2009, the highest sediment organic content was 2.5 ± 0.6%, measured at Site 5 in the middle reaches. Sediment organic content was similar between sites in 2009 (Kruskal Wallis $\chi^2 = 10.839$, $df = 6$, $p = 0.09$). In contrast, in 2018 the surface sediment organic content was highest at Site 6 in the upper reaches (12.7 ± 2.4%) (Kruskal Wallis $\chi^2 = 11.983$, $df = 6$, $p = 0.06$). This site also had the highest sediment organic content across all depth intervals and sites (17.2 ± 7.5% at 15-30 cm) (Kruskal Wallis $\chi^2 = 42.974$, $df = 20$, $p = 0.002$).

Sediment moisture content was less variable than organic content between the lower, middle, and upper reaches of the Knysna Estuary (Figure 4.2.5B). The average (± SD) moisture content in the surface layer was similar between sites in 2009 (Kruskal Wallis χ^2 = 8.809, *df* = 6, *p* = 0.18). As in sediment organic content, moisture content was highest at Site 5 in 2009 (2.50 ± 0.6%) and at Site 6 in 2018 (24.4 ± 3.0%) (Kruskal Wallis χ^2 = 17.835, *df* = 6, *p* = 0.006). Site 6 also had the highest sediment moisture content across all depth intervals and sites (27.7 ± 3.8% at 15-30 cm) (Kruskal Wallis χ^2 = 51.802, *df* = 20, *p* = 0.0001).

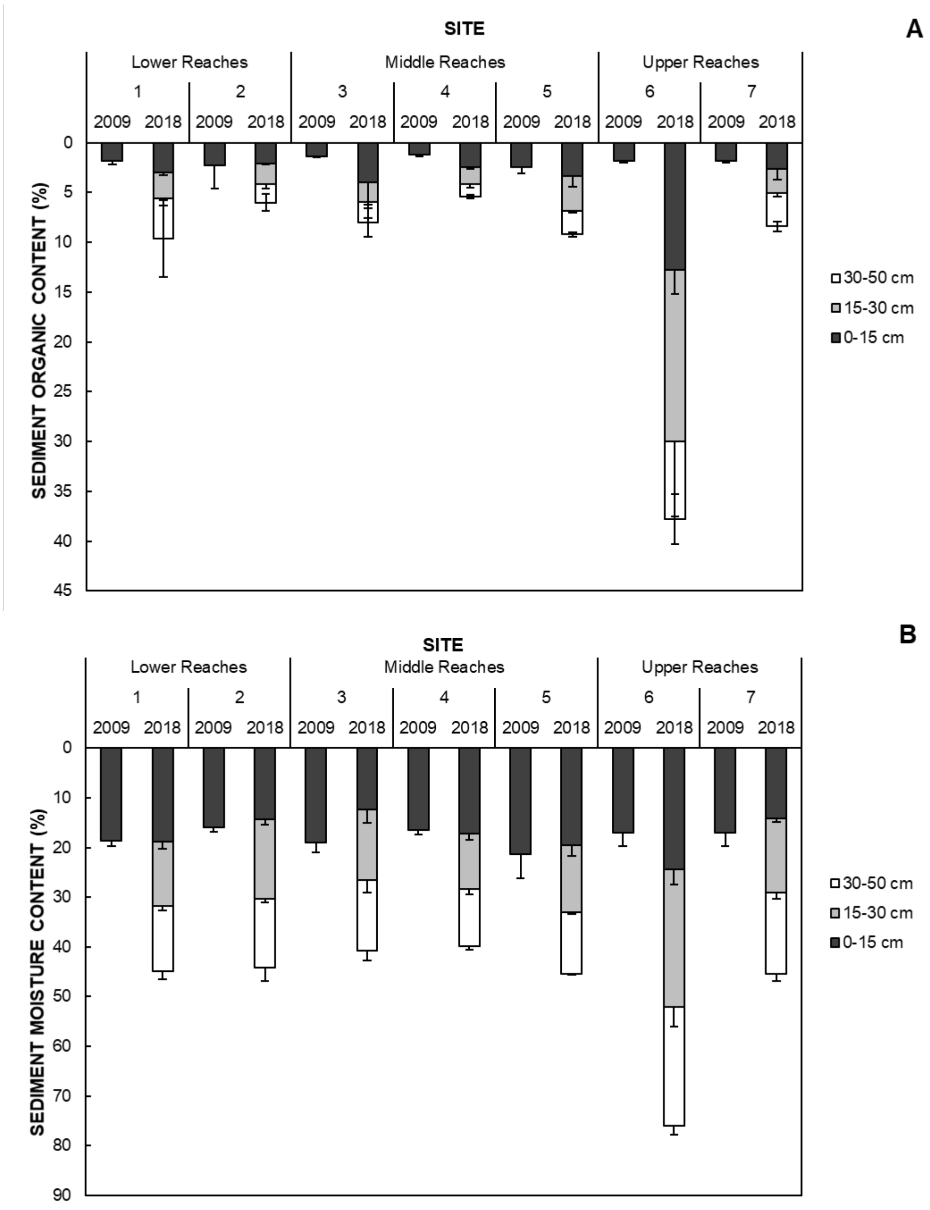


Figure 4.2.5. Cumulative average (± SD) sediment organic content (**A**) and sediment moisture content (**B**) measured in 2009 by Schmidt (2013) and in 2018 by this study. Samples were collected at the location of the Rod Surface Elevation Table (RSET) stations in the Knysna Estuary.

4.2.3.4 Sea-Level Affecting Marshes Model and Development Threats to Salt Marshes

The SLAMM for the focus area at Thesen's Island showed different outcomes depending on the selected SLR scenario and whether developed areas would be protected or not from SLR in the future (Figure 4.2.6, Figure 4.2.7). Protecting developed land against SLR in the model results in a rapid decline in transitional salt marsh (upper intertidal salt marsh species such as *Bassia diffusa*) as landward migration is completely restricted. By 2100, it is predicted that transitional salt marsh will cover 0.465 ha in this focus area, a 40.2% loss from the current (2018) cover of 0.775 ha (Figure 4.2.7). However, if developed areas can convert to salt marsh in the future, SLAMM predicts that transitional salt marsh will increase in area until 2050 to cover 1.225 ha, which is a 57.5% increase of the current area. After this period a subsequent decline in transitional salt marsh is predicted so that by 2100 transitional salt marsh covers 0.915 ha which translates to a 17.7% increase of the current transitional salt marsh area. Irregularly flooded marsh (upper intertidal salt marsh species such as *Plantago* sp.) and regularly flooded marsh (lower intertidal salt marsh species such as *Spartina maritima*, *Triglochin* spp. and *Salicornia* spp.) show similar trends whether developed areas are protected or not against SLR. Irregularly flooded marsh and regularly flooded marsh are predicted to decrease in area over time to 2100 by 24.5% and 19.9% respectively if developed areas are protected from SLR, and 27.5% and 15.9% respectively if developed areas are not protected (Figure 4.2.6, Figure 4.2.7). *Zostera capensis*, an endangered seagrass, occurs at an intertidal elevation range below the salt marsh grass *Spartina maritima*. It was therefore possible to model changes in response to an increase in SLR as it occupies the tidal flat area. This area is predicted to increase in area at almost the same rate regardless of whether developed areas are protected from SLR or not (Figure 4.2.7). The tidal flat area is predicted to increase from 11.35 ha in 2018 to 17.41 ha in 2100 if developed areas are protected or to 17.67 ha in 2100 if developed areas are not protected. This translates respectively to an increase of 53.4% or 55.7% of the current *Zostera capensis* cover for the focus area.

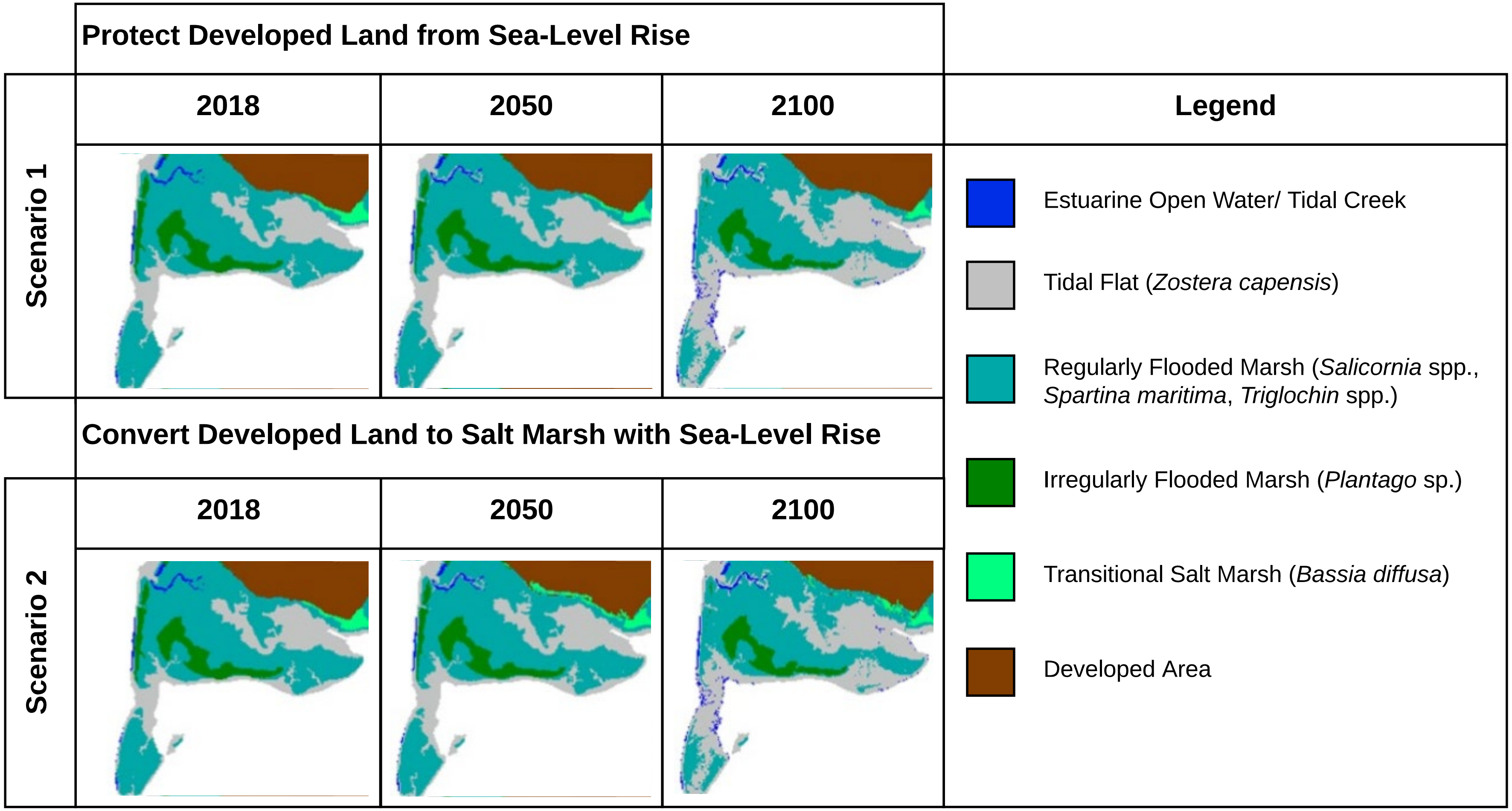


Figure 4.2.6. Future salt marsh habitat distribution predicted by SLAMM for Thesen's Island focus area in the Knysna Estuary under 0.7 m sea-level rise by 2100. The effect of protecting (Scenario 1) or not protecting (Scenario 2) developed areas is shown.

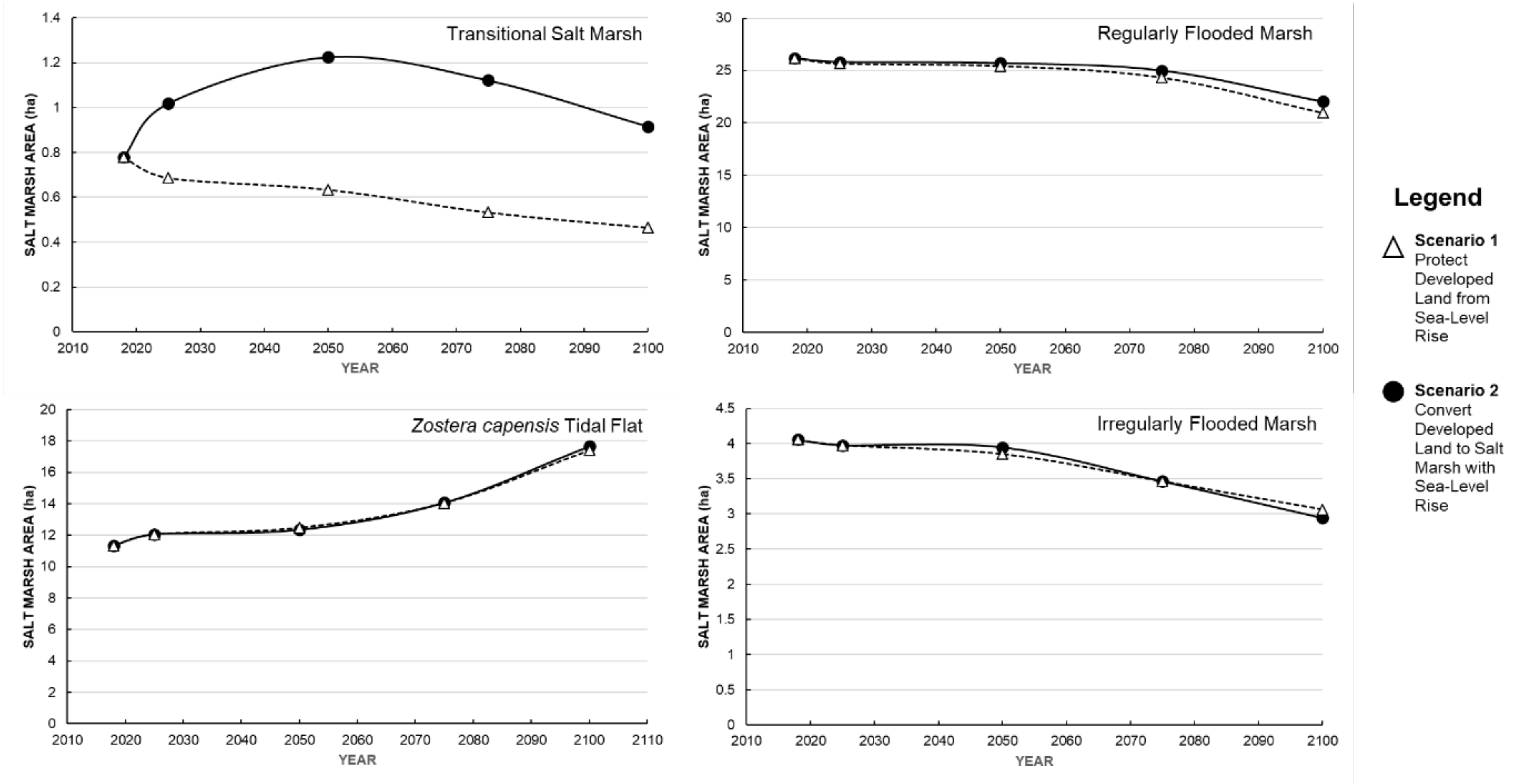


Figure 4.2.7. Future changes in salt marsh habitat areas over time predicted by SLAMM for sea-level rise by Thesen's Island focus area in the Knysna Estuary. The simulation was run under a 0.7 m sea-level rise scenario by 2100 with a 25-year time step.

Salt marsh areas below the N2 bridge that are currently zoned within the South African cadastral layer, but have not been developed, cover 60.4 ha (Figure 4.2.8). These areas consist of erven and farms and beyond the N2 bridge they account for an additional 81 ha. Within potential development zones, *Plantago crassifolia* (13.35 ha), *Zostera capensis* (11.99 ha) and *Triglochin* spp. (11.47 ha) currently cover the largest areas (Table 4.2.5). The largest potential loss will be *Salicornia* spp. (81%) as it currently covers a small area across the entire estuary (Table 4.2.5). Salt marsh species that typically occur towards the lower intertidal zone, such as *Spartina maritima* and *Triglochin* spp., will be less impacted by future developments (Table 4.2.5).

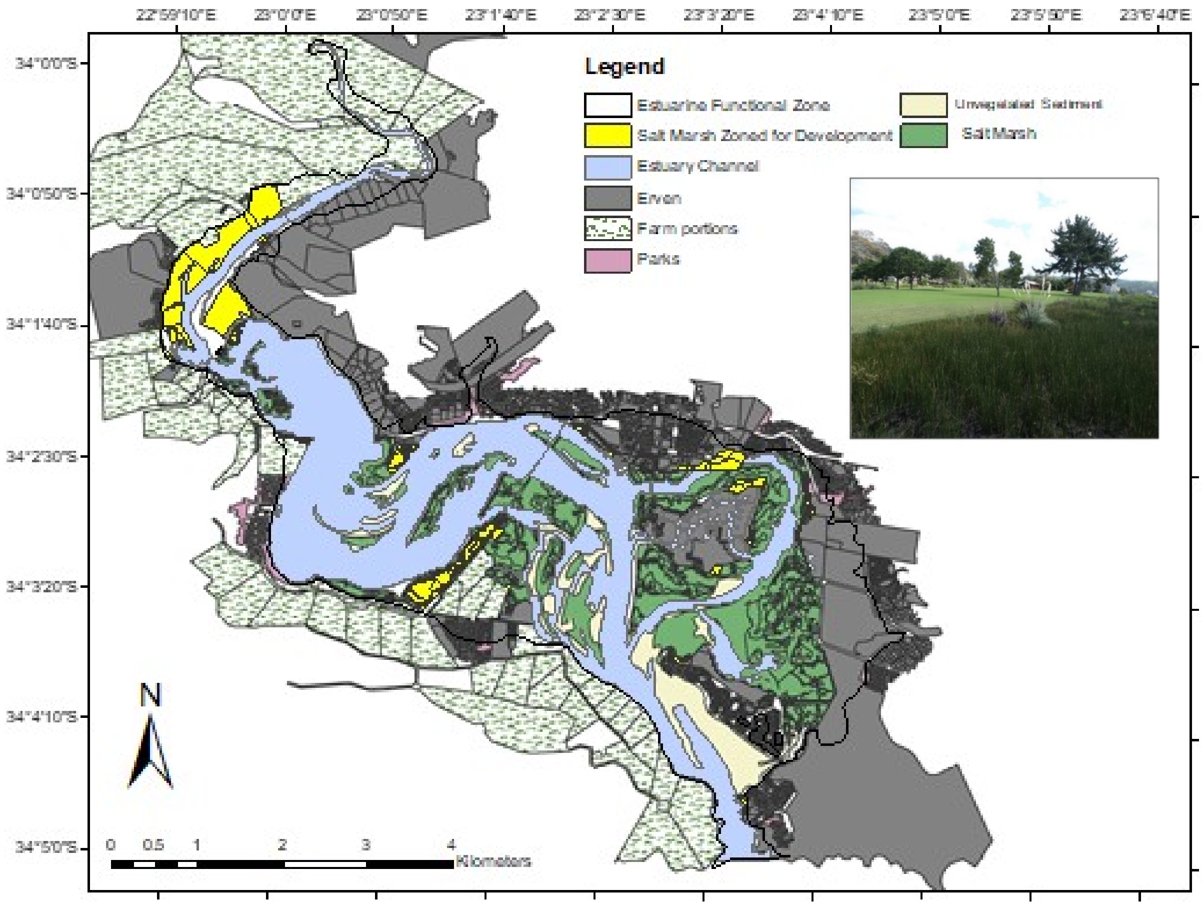


Figure 4.2.8. Development zones and salt marsh distribution in the Knysna Estuary. Yellow areas indicate existing salt marsh habitats that occur within zones allocated for future development within the South African cadastre. An example of lawn encroachment into salt marsh (*Juncus kraussii*) at Belvidere in 2018 is shown on the right.

Table 4.2.5 Estuarine vegetation below the N2 bridge under threat from development in the Knysna Estuary.

Habitat/species	Current area (ha)	Area zoned for development (ha)	% possible loss
Bassia diffusa	30.9	8.8	28.5
Juncus kraussii	14	3.2	22.8
Phragmites australis	0.8	0.53	66.25
Plantago crassifolia	30.5	13.35	37.2
Salicornia spp.	1.5	1.22	81.3
Spartina maritima	107.2	6.25	5.8
Sporobolus virginicus	5.3	2.59	48.8
Triglochin spp.	155.5	11.47	7.4
Zostera capensis	446.8	11.99	2.7
Natural rocky area	3.7	1.01	27.3
TOTAL	**796.2**	**60.41**	**7.6**

4.2.4 Discussion

The high-energy and swell-dominated South African coastline restricts salt marshes to sheltered estuaries as back-barrier environments (Cooper, 2001; Ramsay and Cooper, 2002). Estuarine salt marshes are vulnerable to threats such as sea-level rise and anthropogenic catchment pressures such as freshwater abstraction, urban and agricultural development, nutrient inputs and disruption of sedimentation and hydrologic regimes (Nicholls et al., 2008; Kirwan and Megonigal, 2013; Van Niekerk et al., 2013). As salt marshes are of critical importance to maintaining biodiversity in estuarine systems (Adams et al., 2016), quantifying the threats and responses of these habitats to global environment change is essential to inform management practices. We provide the first report on salt marsh surface elevation dynamics, predicted responses to sea-level rise, and threats from coastal development in the Knysna Estuary. This estuary is nationally ranked the highest in terms of estuarine biodiversity supported by salt marsh habitats. The results of this study indicate significant loss of salt marsh habitat (60.4 ha) without management interventions to prevent further developments and make suitable areas available for salt marsh landward migration.

4.2.4.1 Contemporary Salt Marsh Dynamics and Sediment Characteristics

The relationship between tidal flux, vertical accretion and salt marsh surface elevation is correlated to historic sea-level rise for many locations around the world (Valiela et al., 2018; Cahoon et al., 2019; FitzGerald and Hughes, 2019). The rate at which salt marsh habitats accumulate inorganic sediment and organic material is dependent on intrinsic and extrinsic factors that can be variable at different spatial scales ranging from within the marsh platform to regional site locations (Kirwan et al., 2010; D'Alpaos and Marani, 2016). These factors include sediment supply, vegetation structure and growth as well as relative sea-level rise (Cahoon et al., 2019). At the Knysna Estuary, surface elevation change in lower intertidal salt marsh was spatially variable along the length of the estuary channel. Only two of the sites exhibited positive surface elevation change (accretion) at a rate high enough to keep pace with the calculated RSLR.

Vegetation structure and density have a strong influence on hydrodynamic flows and sedimentation patterns in tidal marshes (Temmerman et al., 2005). Vegetation can facilitate sedimentation by enhancing direct particle capture, reducing flows to promote settling of suspended sediment and contributing directly to organic sediment deposition (Mudd et al., 2010). However, at the Knysna Estuary, all RSET sites were in lower intertidal *Spartina maritima* salt marsh. Vegetation density was also similar between sites, with none situated in open mud flats and generally little bare ground surrounding the RSET benchmarks. Variability in vegetation structure and density is unlikely to drive spatial differences in sedimentation due to enhanced particle capture or flow reduction. Spatial variability in RSET surface elevation at the Knysna Estuary can be related to salt marsh accretion of organic material. Sediment organic content has increased since 2009 and the highest sediment organic content in 2018 was measured at the RSET location with the highest increase in surface elevation (Site 6 in the upper reaches). This could be related to primary productivity as sediment accretion in the congeneric *S. alterniflora* increases with enhanced primary production (Morris et al., 2002). Under natural settings, primary productivity in salt marsh plants is influenced by many factors, but the relative height to mean sea-level is one of the most important (Mendelssohn and Morris, 2000; Morris et al., 2013). As the upper reaches of the estuary are located above mean sea-level, this could indicate a positive feedback between primary productivity, organic sediment deposition and vertical accretion for these salt marsh sites. Surface elevation gains are therefore enhanced despite similarities in vegetation structure and density to other sites in the lower and middle reaches of the estuary.

Biomorphodynamic models of salt marshes have been used to show that in response to sea-level rise, marshes that occur along the edges of channels are limited by inorganic sediment supply, while those that occur further away towards the middle of the platform depend on organic sediment accretion (Belliard et al., 2016; D'Alpaos and Marani, 2016). Inorganic sediment supply could therefore account for differences in surface elevation between *S. maritima* sites in this study, which all occurred adjacent to the estuary channel. Sedimentation patterns in the Knysna Estuary are determined by the distinct "S" shape meander and the respective fluvial and marine sediment inputs (Reddering, 1994; Marker, 2000). Sedimentation is however limited as there is a low influx of inorganic material from marine and fluvial sources while deposition and erosion in the middle reaches are strongly influenced by channel meandering (Reddering and Esterhuysen, 1987). The upper reaches are dominated by fluvial sediment as the N2 bridge restricts tidal influence in these areas (Marker, 2000). The accumulation of predominantly silty sediment at this location could be related to surface elevation gains measured at these salt marsh sites (Schmidt, 2013). In contrast, increased surface elevation in the lower and middle reaches is likely to be the result of sediment deposition at the formation of a point bar (Poff et al., 1997). Salt marshes on Thesen's Island could also receive sediments remobilized from sandbanks as the lower reaches of the estuary are significantly influenced by tidal and wave energy from the marine environment (Reddering, 1994). Sediment erosion on meanders likely explains the slow rates of surface elevation change and high subsidence rates occurring at salt marshes in the mid reaches. The highest subsidence rate was measured at the outside extent of a convex bank on The Point, where flow and subsequent erosion is highest (Poff et al., 1997). This area is prone to erosion and scouring by the incoming tide (Marker, 2000) and there was clear evidence of erosion around the RSET benchmark. The narrow width of the *Zostera capensis* tidal flat at this site further suggests that the area is characterized by higher flows and lower sediment deposition rates.

The variability in surface elevation dynamics between salt marsh sites at the Knysna Estuary influences their susceptibility to sea-level rise as indicated by the relative sea-level rise experienced at each wetland ($RSLR_{wet}$). The historic sea-level rise trend as estimated from the tide gauge on Thesen's Island is higher than the eustatic sea-level rise for the southern Cape region (Mather et al., 2009). This makes estuarine ecosystems particularly vulnerable to sea-level rise as they are experiencing an accelerated rate of sea-level rise (Rilo et al., 2013; Passeri et al., 2015). If sedimentation and accretion regimes at the Knysna Estuary remain the same, areas within the lower and middle reaches of the estuary could be lost to flooding.

4.2.4.2 Future Responses of Salt Marsh to Sea-Level Rise and Threats from Development

Understanding sedimentation, erosion and vegetation dynamics in salt marshes as responses to predicted sea-level rise often requires detailed modelling approaches that are location specific (Rybczyk and Cahoon, 2002; D'Alpaos et al., 2007). These biophysical feedbacks complicate larger scale predictions on the effect of sea-level rise on coastal wetlands, including salt marshes (Belliard et al., 2016; D'Alpaos and Marani, 2016; Raposa et al., 2016). Reliable predictions of salt marsh responses to sea-level rise can be carried out using models at smaller spatial scales. The Sea-Level Affecting Marshes Model (SLAMM) is a readily available model that can be applied to project the spatial distribution of salt marsh under different sea-level rise scenarios (Clough et al., 2016). SLAMM has received numerous criticisms and the model has been revised over recent years to improve assessments of coastal wetland responses (Mcleod et al., 2010; Wu et al., 2015; Kirwan et al., 2016a). SLAMM models salt marsh distribution as a function of elevation change, but the predictions should be validated for reliability if the models are applied to salt marshes that have elevation gradients over small areas, such as in estuaries on wave-dominated coasts (Mogensen and Rogers, 2018).

The application of SLAMM to the focus area at Thesen's Island in the Knysna Estuary showed there would be significant losses of transitional salt marsh under future sea-level rise scenarios. If developed areas convert to salt marsh over time, there would be an initial gain in area for this habitat type, declining towards 2100. SLAMM most likely predicts this outcome because the elevation of the developed area (as determined by the LiDAR digital elevation model) exceeds the maximum elevation or slope at which transitional salt marsh can occur. Although some lower portions of the developed areas may convert to salt marsh in future, raised elevations of properties will prevent a further upslope migration. The height of a wetland within the tidal frame is determined by elevation capital, which is formed through the accumulation of inorganic sediment and organic matter by the process of sediment accretion (Cahoon et al., 2019). SLAMM relies on user inputs to determine the height of salt marsh habitats as well as the accretion and erosion rates. Validation of the SLAMM for Knysna would therefore involve determining the accuracy with which the model can predict current (2018) salt marsh distribution when provided with historic data. Although beyond the scope of this study, it is a necessary step if this approach is to be used to predict salt marsh responses to sea-level rise across the estuary.

The salt marshes in the Knysna Estuary are threatened by sea level rise as well as future development. Since 1942, large areas of salt marsh have been removed by development. Claassens et al. (2020) report losses of 32.5% and 68.5% of intertidal and supratidal salt marsh respectively. Future developments adjacent to the Knysna Estuary will result in further direct losses of salt marsh habitats, and particularly of salt marsh species that occur towards the terrestrial boundary. These areas promote biodiversity by providing nesting and brooding grounds for bird species, including the African Black Oystercatcher, *Haematopus moquini* (Martin et al., 2000). Future developments will also restrict the potential for salt marsh landward migration. Tabot and Adams (2013) showed that in permanently open South African estuaries, a landward migration of salt marsh would be possible under predicted conditions, if coastal squeeze was limited and the rate of landward recruitment sufficient relative to sea level rise. The 60.4 ha of habitat situated on private erven and farms needs to be protected to allow for landward migration in response to sea level rise. These are the only areas available for future salt marsh migration in relation to sea-level rise. Salt marsh plants will spread vegetatively or through seed production into these new habitats. For example, the succulent intertidal salt marsh plant *Salicornia tegetaria* (previously *Sarcornia*, Piirainen et al., 2017) produces thousands of seeds per m^2 and is remarkably adaptive, growing over a wide range of physico-chemical conditions that will allow the species to persist in response to sea-level rise (Riddin and Adams, 2019). If the areas zoned in the cadastral layer are developed, this will represent a significant coastal squeeze and will threaten the survival of the salt marsh.

Salt marshes in the Knysna Estuary face both anthropogenic and climate threats that can exert combined pressures on these habitats. This estuary is of national conservation importance due to its rich biodiversity as 42% of estuarine plant and animal species in South Africa occur there (Allanson, 2000; Turpie et al., 2002; Vromans et al., 2010). Additionally, the economic valuation of coastal wetlands as natural flood control within the estuary has been estimated at R2.8 to R3.4 million per annum (Vromans et al., 2010). The importance of sustaining freshwater inflow into this estuary to maintain biodiversity and habitat structure is well known (Allanson, 2000); an Ecological Water Requirement under the National Water Act of 1998 was determined for the system in 2005 (DWA, 2009). Despite this, river inflow has been decreasing over time (Claassens et al., 2020). Lower flows are expected to influence salt marsh resilience to sea-level rise as transportation of fluvial sediments that contribute significantly towards accretion processes could be severely reduced. Although SANParks has part-jurisdiction to a 50 m buffer zone adjacent to the estuary (Section 41 of National Environmental Management Act: Protected Areas Act), salt marshes outside of this zone are not included in the formally protected area. The surrounding areas up to the mean spring high water mark are under the jurisdiction of the Knysna municipality (Claassens et al. 2020). Delineating boundaries of important habitats adjacent to estuarine areas has been a critical task in South Africa, where urban development has occurred even within the estuarine functional zone as is evident at Knysna (Veldkornet et al., 2015b). Conservation of salt marsh areas presents a challenge for co-operative management, as resilience to sea-level rise strongly depends on the availability of adjacent areas for landward migration. We recommend that future developments in areas adjacent to the estuary not be permitted and that these areas should be conserved as essential estuarine habitat.

5 IMPACT OF CLIMATE CHANGE ON BLUE CARBON HABITATS

5.1 Drivers of Mangrove Distribution at the High-Energy, Wave-Dominated, Southern African Range Limit

Raw JL, Godbold, JA, Van Niekerk L, Adams JB (2019) Drivers of mangrove distribution at the high-energy, wave-dominated southern African range limit. *Estuarine, Coastal and Shelf Science* 226: 106296.

5.1.1 Introduction

Mangrove forests are iconic features of tropical coasts. The range limits of these ecosystems occur at subtropical/warm-temperate boundaries which are characterized by the 20°C winter isotherm for sea-surface temperature (Duke et al., 1998; Tomlinson, 1999). Temperature and precipitation regimes are therefore considered as the most important factors driving contemporary global distribution patterns for mangrove forests (Giri et al., 2011; Osland et al., 2017b). At regional scales, mangrove distribution patterns have been related to particular aspects of climatic regimes (Quisthoudt et al., 2012; Ward et al., 2016; Cavanaugh et al., 2018). For example; freeze events, precipitation, and ocean currents have been identified as drivers of mangrove distribution patterns respectively in the eastern United States, Australia, and Brazil (Saintilan and Williams, 1999; Stuart et al., 2007; Soares et al., 2012; Cavanaugh et al., 2014; Osland et al., 2017b). However, local physical factors such as coastal hydrology and geomorphology can also influence mangrove distribution patterns (Semeniuk, 1983; Schaeffer-Novelli et al., 1990; Stevens et al., 2006; Leong et al., 2018). This mismatch between regional and global drivers could explain why studies carried out at these different scales disagree on whether or not mangroves are already expanding polewards in response to global climate change (Comeaux et al., 2012; Boon, 2017; Hickey et al., 2017).

At local scales, factors that drive mangrove establishment and distribution are well-documented and can be categorized in relation to geomorphology, abiotic controls, physical barriers, and biotic interactions (Krauss et al., 2008; Arrivabene et al., 2014). The hydro-geomorphological settings in which mangroves occur are defined in relation to landform and sedimentation patterns (Woodroffe, 1992). On wave-dominated coasts mangroves are restricted to occurring in sheltered estuaries or lagoons (Woodroffe, 1992; Cooper, 2001). The degree to which these coastal embayments have been infilled by fluvial sediments determines the size of the floodplain available for mangrove establishment (Roy et al., 2001). The hydro-geomorphological settings of a mangrove forest also determine local scale variability in physical and abiotic factors, which in turn control vegetation distribution patterns (Berger et al., 2006). Physical barriers to mangrove distribution on wave-dominated coasts are related to the dynamics and stability of the inlet connection to the ocean. Higher nearshore energy promotes marine sediment deposition, the formation of a barrier, and closure of the inlet (Dalrymple et al., 1992; Heap et al., 2004). Prolonged closures cause inundation stress and dieback of estuarine mangroves (Mbense et al., 2016), but this can be prevented by sufficient fluvial inputs to re-instate the inlet connection (Van Niekerk et al., 2013). Additionally, fluvial inputs regulate soil salinity and pH and, also transport allochthonous organic matter into mangrove habitats (Urrego et al., 2009; Rajkaran and Adams, 2012; Suárez et al., 2015). These processes help maintain suitable ecophysiological conditions for mangroves by preventing salinity stress and promoting nutrient availability through soil microbial activity (Rajkaran et al., 2009; Naidoo, 2016). Relating all these factors to larger scale biogeographic patterns for mangroves can however be challenging, as there can be local variability as well as interactions between different factors.

Understanding the regional drivers of mangrove distribution is important for developing management strategies as the potential for range expansion is associated with ecological shifts at the expense of warm-temperate salt marsh habitats (Kelleway et al., 2017a). These shifts can result in changes to faunal community composition, sea-level rise vulnerability, and carbon storage capacity (Kelleway et al., 2016b; Guo et al., 2017; Hayes et al., 2017; McKee and Vervaeke, 2017; Smee et al., 2017). A holistic approach, such as that offered by structural equation modelling, is therefore required. This technique allows for the investigation of complex networks of relationships between variables and the representation of theoretical concepts (Grace et al., 2010). This approach has become increasingly popular in ecological research to incorporate direct and indirect effects between variables, to assess the strength of positive feedbacks, and to investigate changes across different scales (Van der Heide et al., 2011; Austin et al., 2017, Ouyang et al., 2017).

This study aimed to apply the structural equation modelling approach to determine the importance of different local drivers of mangrove area patterns along the South African coastline. Here mangroves reach a continental range limit, which is one of the southernmost locations in the global distribution for this ecotype. The high energy, wave-dominated coast restricts the occurrence of mangroves to sheltered estuarine areas, resulting in a discontinuous distribution along the coast. Mangrove expansion has been documented at this range limit and generalizations on the driving factors have been made based on decades of observations (Whitfield et al., 2016; Peer et al., 2018). This study provides the first quantitative assessment of factors influencing current mangrove distribution in this region.

5.1.2 Methods

5.1.2.1 Conceptual Framework

A Structural Equation Model (SEM) was implemented following the guidelines of Grace et al. (2012). This is a graph-theoretic approach used to understand and delineate causative relationships between multiple variables. Climate and geomorphology are the two most important drivers of mangrove distribution (Woodroffe and Grindrod, 1991; Duke et al., 1998). At the most general conceptual level, climate and estuarine geomorphology determine abiotic factors and physical barriers that directly influence mangrove biogeographic patterns (Figure 5.1.1). These are the underlying theoretical constructs on which the structural model was built.

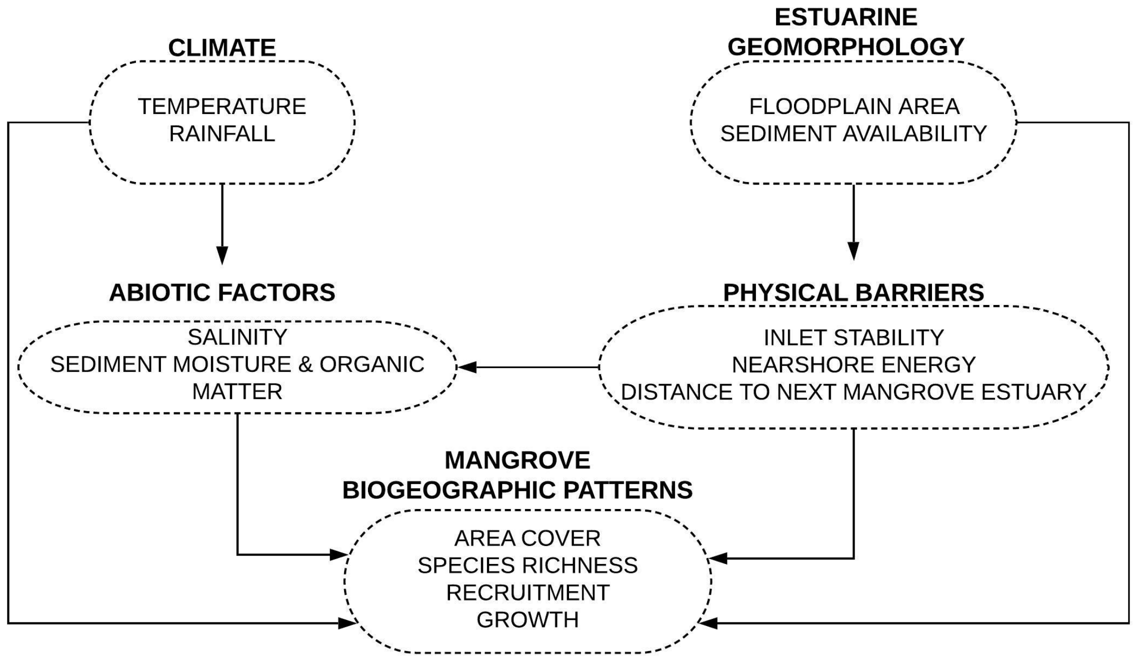


Figure 5.1.1. Conceptual diagram of hypothesized links between the theoretical constructs investigated in this study. Each theoretical construct represents a group of observable variables while the links guide the construction of the structural equations.

Following this, a causal diagram summarizing the hypothesized connections among all potential observable variables was generated (Figure A1.3.1). Each link in the causal diagram was evaluated as an assumption or hypothesis using *a priori* ecological knowledge as well as evidence from published scientific research (Table A1.3.1). This process allows for consideration of which variables are essential and which variables could be omitted to limit model complexity in the SEM (Grace et al., 2012). A full description of the process to develop the links for the SEM is provided in the Supplementary Material (Appendix I, Section A1.3).

5.1.2.2 Data Collation

Regional data for mangrove area cover (ha) in South African estuaries were collated from the most recent (2018) version of the National Estuary Botanical Database maintained by the Nelson Mandela University (Adams et al., 2016). Of the ~ 300 estuaries along the South African coastline, mangroves are restricted to occurring along the east coast in 31 systems that fall within the warm-temperate and subtropical to tropical biogeographic regions (Figure 5.1.2). The two dominant species are *Avicennia marina* and *Bruguiera gymnorhiza* that occur in 27 estuaries, although *A. marina* covers the largest area (Adams et al., 2016). *Rhizophora mucronata* occurs in 14 estuaries and is associated with tidal creeks and channels. An additional three species (*Ceriops tagal*, *Lumnitzera racemosa*, and *Xylocarpus granatum*) only occur in the tropical biogeographic zone at the Kosi Estuary.

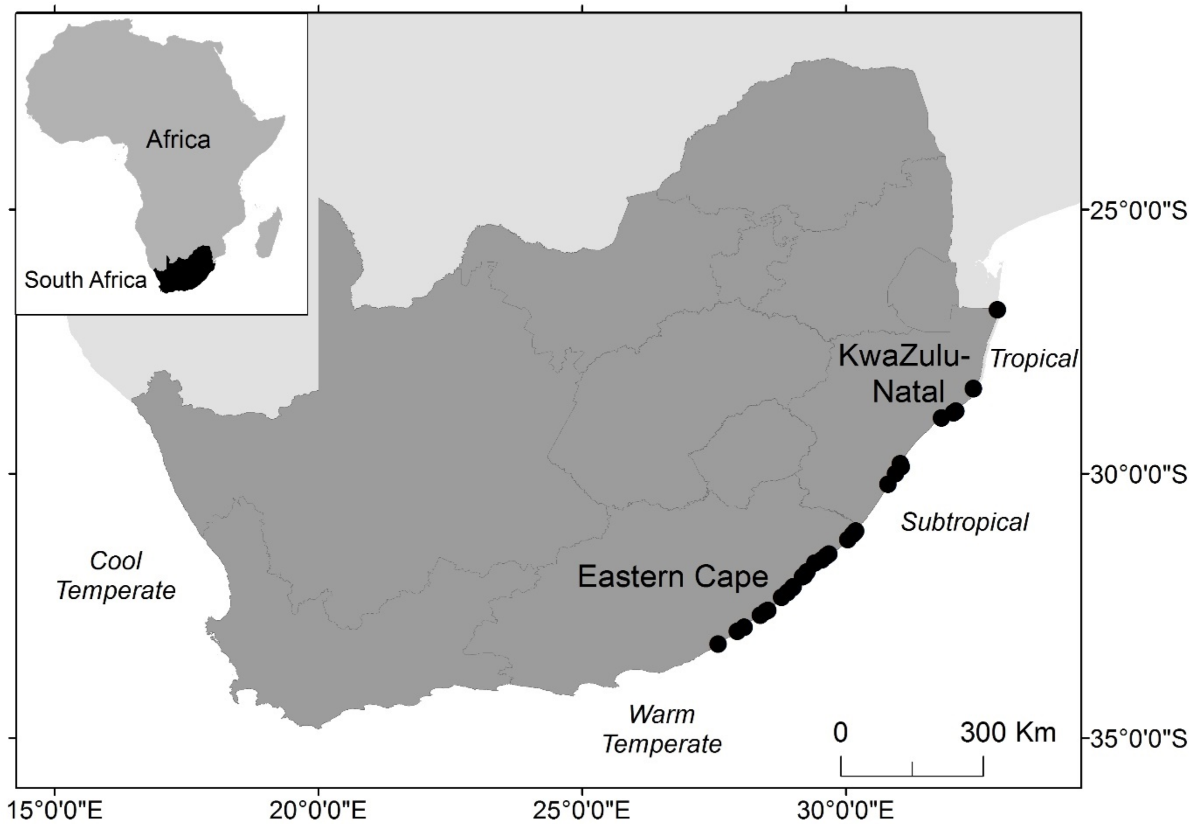


Figure 5.1.2. Locations of estuaries that support mangrove forests along the South African coastline.

Estuaries that support mangrove areas greater than 0.5 ha (*n* = 29) were identified from the botanical database. Data for suitable variables that represent climate and geomorphological drivers were then extracted for these estuaries from the most recent version of the Council of Scientific and Industrial Research (CSIR) National database for the physical characteristics of South Africa's estuaries (Van Niekerk et al., 2017; 2018) (Table 5.1.1). The extracted variables were: Average Annual Land Temperature, Catchment Erodibility, Daily Flushing Rate, Distance to Next Mangrove Estuary, Floodplain Area, Inciseness, Inlet Stability, Mean Annual Runoff, Seasonal Difference in Land Temperature, and Surfzone Width. Rainfall data were extracted from the WR2012 hydrology project.

Preliminary data exploration identified two estuaries as significant outliers with Cook's distance > 1 (Cook, 1979). The outliers were based on the data points for mangrove area cover and floodplain size. These two estuaries (uMhlathuze and St Lucia) were excluded from the final analysis. The large mangrove areas supported by these estuaries are not truly representative of the conditions along the South African coastline and, furthermore, both of these estuaries have been transformed from their baseline ecological states. uMhlathuze has been modified to develop an industrial port while the St Lucia estuary was artificially opened for more than 50 years (Whitfield and Taylor, 2009; Elliott et al., 2016).

Table 5.1.1. Observed variables that were considered for modelling South African mangrove distribution based on the SEM conceptual model.

Theoretical construct	Observed variables	Properties of data
Climatic Variables	Temperature (°C)	
	Average Annual Land Temperature	Continuous
	Seasonal Difference in Land Temperature	Continuous
	Rainfall	0 to +∞; Continuous
	Mean Annual Precipitation (mm)	
Geomorphological Variables	Floodplain Area (ha)	0 to +∞; Continuous
	the area between the 0 and 5 m contour, minus all open water area, it reflects the estuarine functional zone	
	Inciseness	0-1 ratio data, Continuous
	ratio of open water area to floodplain area	
	Catchment Erodibility Index	Categorical (1 (low) to 5 (high))
Abiotic Variables	Salinity Category	Categorical (fresh, mixed, marine)
	the dominant salinity regime of the estuary which reflects the degree of fluvial input as ranging from freshwater-dominated to marine	
	Mean Annual Runoff ($m^3 \times 10^6$)	0 to +∞; Continuous
Physical Variables	Inlet Stability	
	condition of the estuary inlet that reflects periods of altered or low marine connectivity	Binary (1 = open 100% of the time; 0 = experiences closure up to 25% of the time)
	Daily Flushing Rate ($m^3 \times 10^6.d$)	0 to +∞; Continuous
	Distance along the coast to next mangrove estuary (km)	0 to +∞; Continuous
	Nearshore Energy	0 to +∞; Continuous
	Surfzone Width (m)	
Response Variables	Mangrove Area (ha)	0 to +∞; Continuous

5.1.2.3 Data Analysis

To specify the initial SEM, a series of equations describing the relationships between predictor and response variables were defined. These relationships were determined using correlations between the variables and the hypothesized links in the causal diagram. The properties of the available data, such as the type and distribution (Table 5.1.1), were considered to mathematically specify each relationship as a statistical model. Five models were initially specified for the response variables: Inciseness (topography slope), Mean Annual Runoff, Daily Flushing Rate, Inlet Stability, Mangrove Species Richness, and Mangrove Area (Table 5.1.2).

Table 5.1.2. Initial models that were specified based on the *a priori* hypothesized links describing the causative relationships between observed variables.

Model	Response variable	Predictor variables	Statistical model form
(1)	Inciseness	Floodplain Area	Linear regression
(2)	Mean Annual Runoff	Mean Annual Precipitation	Generalized least squares
(3)	Daily Flushing Rate	Floodplain Area, Mean Annual Runoff	Generalized least squares
(4)	Inlet Stability	Daily Flushing Rate, Incisiveness, Surfzone Width	Logistic regression (binomial)
(5)	Mangrove Area	Daily Flushing Rate, Floodplain Area, Inlet Stability, Temperature,	Generalized least squares

Models with continuous response variables (Inciseness, Mean Annual Runoff, Daily Flushing Rate, Mangrove Area) were first specified as linear regression models and model assumptions were verified graphically following Zuur et al. (2009) by assessing (1) the spread in a plot of the standardized residuals and the fitted values (homogeneity of variance); (2) the distribution of the standardized residuals in a frequency histogram (normality); (3) the relationship between the standardized residuals and each of the explanatory variables in the model (independence). When there was evidence of a violation of homogeneity of variances, the data were analysed by incorporating a variance covariate structure and using a generalised least-squares (GLS) estimation procedure (following Zuur et al., 2009; Godbold and Solan, 2013) to allow the residual spread to vary within individual explanatory variables. The optimal variance covariate structure was determined using restricted maximum-likelihood (REML) estimation; the initial linear regression model without a variance structure was compared with the equivalent GLS model incorporating specific variance structures using AIC (Aikaike Information criterion) and visualisation of model residuals. The significance of the fixed-effects structure in the GLS models was determined using maximum likelihood (ML) estimation and was on based log likelihood ratios. Non-significant terms were only retained in the model if they improved the model validation graphs (Zuur et al., 2009).

For the model defining Inlet Stability, logistic regression with a logit link was used as this response variable had a binary distribution. The Nagelkerke (1991) method was used to calculate R^2 for the generalized linear model. Assumptions were validated graphically using the standardized and deviance residuals (Zuur et al., 2009).

The significant models specified for Daily Flushing Rate, Inlet Stability, and Mangrove Area were fit to the SEM using the piecewise estimation method as it allows for the implementation of non-normal distributions, random effects and different correlation structures using local estimation (Lefcheck, 2016; Andriuzzi et al., 2018; Campanati et al., 2018). The fit of the equations within the SEM was tested using Fishers goodness of fit and D-separation (Shipley, 2013). This allowed for missing links in the specified models to be identified and added to the models accordingly. Variables were only added as predictors if the relationship with the response variable was ecologically sound. The statistical assumptions and fit based on AIC were re-assessed for any models to which predictor variables were added. The refined models were then returned to specify the SEM and the process was repeated until the Fisher's goodness of fit indicated $p > 0.05$. A summary of all the models making up the final SEM, the R^2 values, and the coefficients was then generated and presented in a graphical form.

The beta coefficients allow for an interpretation of the relationships between variables in terms of standard deviation units (Grace and Bollen, 2005). In the case of multiple predictors, the standardized coefficients reflect the effect of the variable as well as the variances and covariances of the other variables in the model (Pedhazur, 1997). However, standardizing the coefficients permits direct comparisons between the different models, regardless of the units in which the response variables are measured (Grace and Bollen, 2005). The estimated coefficients are therefore displayed as path coefficients, and these indicate the difference in the predicted value of the response variable for each unit difference in the predictor variable when all other predictor variables remain constant (Grace and Bollen, 2005).

All statistical analyses were performed using R version 3.5.1 (R Core Team, 2018). The SEM was performed using the package "piecewiseSEM" (Lefcheck, 2016). The "faraway" package (Faraway, 2016) was used to test for dispersion in the logistic regression model. The "nlme" package (Pinheiro et al., 2018) was used to specify generalized least squares models.

5.1.3 Results

The models defining Inciseness and Mean Annual Runoff were insignificant and, therefore, were not included in the SEM analysis. For the remaining three models, the coefficients indicate the significance of the individual structural paths in the SEM (Table 5.1.3).

Significant coefficients ($p < 0.05$) were found for the models defining Daily Flushing Rate, and Mangrove Area (Table 5.1.3, Figure 5.1.3). The results show that Floodplain Area and Mangrove Area are closely related. If there is a 1-unit standard deviation change in Floodplain Area, it is predicted that there will be a change in 0.890 standard deviation units for Mangrove Area. A similar relationship occurs for Mean Annual Runoff as a predictor of Daily Flushing Rate. The magnitude of the beta coefficients was used to scale the links between predictor and response variables in the final SEM (Figure 5.1.3).

The graphical representation of the final SEM (Figure 5.1.3) shows the relationships between all variables defined in the models. For the model defining Mangrove Area ($R^2 = 0.57$), Floodplain Area, Inlet Stability, Daily Flushing Rate and Mean Annual Runoff were all found to be significant predictors ($p < 0.05$) (Table 5.1.3, Figure 5.1.3, Table A1.3.4). The model predicted that larger Floodplain Area and higher Mean Annual Runoff would increase Mangrove Area, while an increase in Daily Flushing Rate is associated with a decrease in Mangrove Area. A 1 ha increase in Floodplain Area is predicted to increase Mangrove Area by 0.0154 ha, while an increase in Mean Annual Runoff of 1 $m^3 \times 10^6$ is predicted to only increase Mangrove Area by 0.00091 ha. In contrast, an increase of Daily Flushing Rate of 1 $m^3 \times 10^6$/d is predicted to decrease Mangrove Area by 6.407 ha. Additionally, if Inlet Stability changes so that there is no longer a permanent connection to the ocean, Mangrove Area is predicted to decrease by 5.350 ha.

Table 5.1.3. Summary of the coefficients derived from statistical models defined for climatic and geomorphological drivers of mangrove biogeographic patterns along the South African coastline. Significant (p < 0.05) coefficients are indicated by asterisks. Daggers indicate range standardization of the beta coefficients for categorical predictors. Only models with suitable fit were used in the final Structural Equation Model.

Model 1: Inciseness (R^2 < 0.01)							
Predictor variable	**Variance structure**	**Coefficient**	**Std Error**	**df**	***t***	***p***	**Beta coefficient**
Floodplain Area	N/A	-2.28×10^{-6}	1.58×10^{-5}	25	-0.145	0.886	-0.0289
Model 2: Mean Annual Runoff (R^2 = 0.21)							
Predictor variable	**Variance structure**	**Coefficient**	**Std Error**	**df**	***t***	***p***	**Beta coefficient**
Rainfall	Power = 2.371	0.3199	0.2055	27	1.557	0.133	0.1923
Floodplain Area	N/A	0.0538	0.0219	27	2.450	0.022	0.3910*
Model 3: Daily Flushing Rate (R^2 = 0.62)							
Predictor variable	**Variance structure**	**Coefficient**	**Std Error**	**df**	***t***	***p***	**Beta coefficient**
Floodplain Area	N/A	-1.18×10^{-4}	3.97×10^{-5}	27	-2.972	0.007	-0.3914*
Mean Annual Runoff	Constant = 1.35×10^{-5} Power = -9.645	2.08×10^{-3}	2.79×10^{-4}	27	7.446	0.0001	0.9474*
Surfzone Width	N/A	-3.14×10^{-3}	9.45×10^{-4}	27	-3.324	0.003	-0.3992*
Model 4: Inlet Stability (R^2 = 0.50)							
Predictor variable	**Variance structure**	**Coefficient**	**Std Error**	**df**	***t***	***p***	**Beta coefficient**
Daily Flushing Rate	N/A	-1.667	1.067	23	-1.562	0.118	-0.388
Surfzone Width	N/A	0.013	0.009	23	1.445	0.148	0.370
Model 5: Mangrove Area (R^2 = 0.57)							
Predictor variable	**Variance structure**	**Coefficient**	**Std Error**	**df**	***t***	***p***	**Beta coefficient**
Mean Annual Runoff	N/A	9.1×10^{-3}	3.08×10^{-3}	27	2.984	0.007	0.0736*
Temperature	Power = 5.854	0.7303	1.842	27	0.396	0.696	0.0187
Floodplain Area	Power = 0.570	0.0154	3.79×10^{-3}	27	4.066	0.0006	0.8970*
Marine Connectivity	N/A	-5.350	1.5851	27	-3.375	0.0029	-0.0412*†

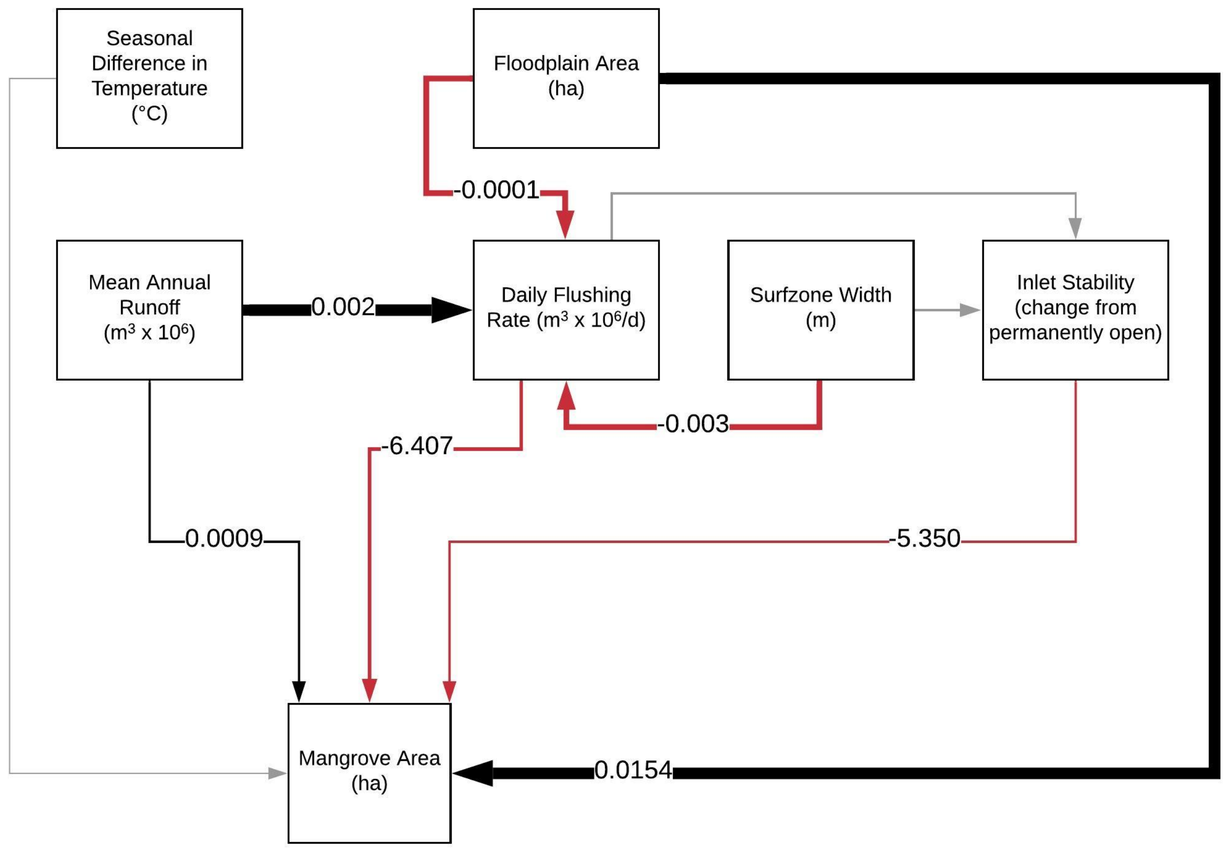


Figure 5.1.3. Final Structural Equation Model for variables representing climatic and geomorphological drivers of mangrove distribution patterns along the South African coastline. Path coefficients are estimated values in the units of the respective response variables. Black lines indicate positive significant effects, red lines indicate negative significant effects, and grey lines indicate non-significant effects ($p > 0.05$). Arrows are scaled by the magnitude of the beta coefficients.

5.1.4 Discussion

Global datasets of temperature and rainfall gradients have been used to examine regional biogeographic patterns in mangrove distribution and species richness (Hickey et al., 2017; Osland et al., 2017b). However, regional scale studies have identified that local conditions can have strong control over mangrove biogeographic patterns (Schaeffer-Novelli et al., 1990; Duke et al., 1998, Stevens et al., 2006). Here we provide the first report on the importance of regional and local factors as drivers of mangrove distribution along the coastline of South Africa.

5.1.4.1 Importance of Geomorphology as a Control on Mangrove Distribution

In this study, Floodplain Area was a significant predictor of Mangrove Area, and these two variables were closely correlated. The gradient of the shoreline significantly influences the development of mangrove forests as different geologic settings and hydrological fluxes drive establishment patterns in relation to species tolerance ranges (Semeniuk, 1985; Woodroffe, 1992; Balke and Friess, 2016). Areas within estuaries that are most suitable for mangrove habitat have low gradients and low energy, and these conditions promote sediment deposition (Boyd et al., 1992; Dalrymple et al., 1992) and facilitate mangrove establishment (Woodroffe, 1992; Roy et al., 2001; Krauss et al., 2008). The suitability of these areas to mangroves is determined by the relationship between the surface sediment height and the upper tidal level limit (Woodroffe et al., 2016). This zone is geologically defined in terms of 'accommodation space' which is "the space available for potential sediment accumulation" (Jervey, 1988).

Tidal and fluvial hydrodynamics affect accommodation space by driving erosion or deposition of sediment, which determines accretion and subsidence dynamics (FitzGerald, 1996; Adame et al., 2010). Net sediment accumulation is expected in microtidal estuaries, such as those along the South African coastline, as inputs received from marine and fluvial sources are only periodically scoured by river floods (Cooper, 2001). Sediment accumulation can restrict the occurrence of coastal wetlands such as mangroves when high accretion rates build elevations above the suitable intertidal zone (Saintilan et al., 2016). In this study, we attempted to include an index of Catchment Erodibility as a proxy for sediment inputs, but it was not a significant predictor in the mangrove area model. However, the restricted occurrence of mangroves in this region has been attributed to the dominance of coarse-grained terrigenous sediment and limited accommodation space in relation to the stage of geomorphological development of these estuaries (Cooper et al., 2018).

Along the South African coastline, almost all estuaries occur in incised bedrock valleys of drowned rivers following post-glacial sea-level change in the Mid-Holocene 6000-7300 years ago (Whitfield, 1992; Cooper et al., 2018). The evolution of modern estuaries from drowned river valleys to mature systems infilled with sediment characterizes the development of the floodplain (Skilbeck et al., 2017). The young geological age of South African estuaries is therefore reflected in the importance of floodplain development as a predictor of mangrove area as the largest forests occur within the few coastal plain estuaries and natural embayments along the KwaZulu-Natal coast (Rajkaran and Adams, 2011; Adams et al., 2016).

The transition of an estuary from one geomorphic state to another is determined by sea level, waves, tides, and river processes (Boyd et al., 1992; Dalrymple et al., 1992). In this study, Inlet Stability and Daily Flushing Rate were significant predictors of Mangrove Area. The interplay between nearshore energy as well as tidal and fluvial mechanisms determines whether an estuary is wave-dominated or tide-dominated (Daidu et al., 2013; Skilbeck et al., 2017). Most estuaries along the eastern South African coastline are wave-dominated, as high nearshore energy promotes deposition of sediment as a barrier at the opening, and this reduces the stability of the inlet, particularly if fluvial flows are relatively weak (Whitfield, 1992; Cooper, 2001; McSweeney et al., 2017). This can lead to prolonged closure, which can significantly impact mangroves as they are sensitive to changes in the inundation regime (Breen and Hill, 1969; Adams and Human, 2016; Mbense et al., 2016).

Although increased Daily Flushing Rate was hypothesized to maintain Inlet Stability, and therefore indirectly support mangrove habitat persistence, a significant negative direct effect on Mangrove Area was predicted by the model. In this study, Daily Flushing Rate was defined by Mean Annual Runoff, Floodplain Area, and Surfzone Width, and this provides a proxy of the flow regime for an estuary as influenced by both fluvial and marine inputs. The importance of hydrology in maintaining mangrove forests has been highlighted by research on effective mangrove restoration (Lewis, 2005; Ferreira et al., 2015; Howard et al., 2017b). In confined estuarine settings, the combined effects of fluvial and tidal discharges on sedimentation patterns determine mangrove establishment and persistence (Wolanski, 1992). This results in different responses to extreme events in river- and tide-dominated estuaries on high energy coastlines (Cooper, 2002; Rogers and Woodroffe, 2016). Mangroves that occur in river-dominated estuaries are expected to be more prone to destabilization by flood events as high energy flows erode the length of channels, while in tide-dominated estuaries the flood energy results in erosion of the barrier and tidal delta (Cooper, 2002).

In combination, the geomorphological and physical characteristics of estuaries along the South African coastline have a strong influence on mangrove occurrence. Limited accommodation space in conjunction with the stability of the estuary inlet and connection to the marine environment determine which estuaries can support persistent mangrove habitats.

5.1.4.2 Climatic Controls on Mangrove Biogeographic Patterns

Mangroves occur at a southern continental limit and extend into a warm temperate biogeographic region along the South African coastline. In this study, temperature was not a significant predictor in the model describing Mangrove Area. The use of average minimum temperatures as proxies for climatic thresholds at range limits has been criticized as they are not informative when there is a considerable range in extreme values (Osland et al., 2017b). For mangrove species, freeze events have been documented to control current distribution patterns as these species are not frost-tolerant (Stevens et al., 2006; Stuart et al., 2007; Cavanaugh et al., 2014; Schaeffer-Novelli et al., 2016).

The absolute coldest air temperature that has occurred is recommended for models that predict range expansions for mangroves (Cavanaugh et al., 2015). However, many southern hemisphere range limits have small differences between the lowest absolute minimum temperature and the daily winter minima, and the absolute minima for these range limits are relatively warmer in comparison to those in the northern hemisphere (Osland et al., 2017b; Rogers and Krauss, 2018). Mangrove trees at southern hemisphere range limits are therefore not likely to be occurring at their physiological limits. For the southeast African range limit, the lowest absolute minimum temperature over the past four decades has been estimated at -0.5°C using gridded daily minimum air temperature data (Osland et al., 2017b), and this could explain the weak causal link between Temperature and Mangrove Area in our model. The reduced effect of temperature as a control on contemporary mangrove distribution in South Africa is also evident from the success of planted mangroves beyond their natural latitudinal limit (Hoppe-Speer et al., 2015b).

Although mangrove distribution is not currently limited by temperature, increases associated with global climate change could still influence mangrove expansion at the southern African distribution limit. Increasing temperature be influencing mangrove expansion at spatial scales within individual estuaries and as the SEM did not consider mangrove areas < 0.5 ha, estuaries where pioneer trees have been reported were not included. Predictive modelling of mangrove species distribution has shown that increased temperature and rainfall will promote mangrove expansion in this region (Quisthoudt et al., 2013). Although individual pioneers have been recently recorded beyond the natural distribution (Whitfield et al., 2016), expansion into salt marsh areas has not occurred at the scales reported on at other range limits in Australia and the eastern United States (Kelleway et al., 2017a; Smee et al., 2017). Latitudinal gradients are weakly associated with species diversity of mangrove fauna and flora for this region (Peer et al., 2018) which further indicates that expansion is likely to be limited by dispersal (Whitfield et al., 2016). Investigating the potential for expansion should account for this and also include an assessment of the suitability of estuaries beyond the current distribution in terms of geomorphology and inlet stability.

5.1.4.3 Structural equation Modelling and Future Applications

This is the first study to use a quantitative modelling approach to describe mangrove distribution patterns for this region. The SEM approach is used to develop and evaluate models so that underlying causal processes can be represented as a network of relationships among the variables of interest (Grace et al., 2012). This is achieved by using *a priori* ecological knowledge to provide context to the causal relationships (Grace, 2008). One of the strengths of the SEM approach is that additional paths can be added and the effect on the system can be evaluated if more data becomes available (Grace et al., 2012). For this study, the addition of data on sediment characteristics (particle size, moisture content, redox potential), high resolution topography data (<10 cm accuracy), and biological interactions (interspecific competition, predation by crabs), could improve the strength of the models as these factors are known to influence mangrove distribution patterns (Dahdouh-Guebas et al., 1998; Berger et al., 2006; Rajkaran and Adams, 2012). The SEM can also be validated by using the models to predict mangrove area for estuaries that have lost mangrove habitat due to anthropogenic disturbances, and the predictions can be compared to historical data.

The conclusive findings on the importance of geomorphology for controlling mangrove distribution in this region can be used as an important baseline for improving climate change predictive models. The results from this study indicate that including floodplain area as part of a geomorphological index for predicting mangrove expansion is essential. The topography of the coast has long been noted as a significant control on mangrove distribution under wave-dominated, high-energy conditions at southern hemisphere range limits. The interplay between geomorphology and hydrodynamic characteristics should not be overlooked when considering potentially suitable habitats for mangroves beyond their current distribution along the South African coastline.

5.2 Potential for Mangrove Range Expansion in Response to Global Climate Change along the South African Coastline

5.2.1 Introduction

Mangroves are typically tropical ecosystems and the global distribution of these forests is delineated by the 20°C winter isotherm for sea-surface temperature (Duke et al., 1998; Tomlinson, 1999). The range limits for mangroves occur at the boundaries between subtropical and warm-temperate biogeographic regions (Spalding et al., 2010; Giri et al., 2011). Rising global temperatures over recent decades have driven what has been termed "tropicalization", as species with tropical affinities have gradually expanded their distribution ranges to historically warm-temperate regions (Wernberg et al., 2013; Hyndes et al., 2016; Vergés et al., 2014). Mangroves are no exception, and expansions of mangroves into salt marsh habitats have been recorded at five continental range limits (Saintilan et al., 2014). These shifts from salt marsh to mangrove vegetation have been associated with changes to the delivery of ecosystem services (Kelleway et al., 2017a). Research that can predict mangrove range expansions could therefore be informative for mitigation and conservation strategies.

Before predictions of range expansions, it is first necessary to determine the drivers of current distribution patterns. This is particularly important because factors that limit mangroves are variable between the different range limits around the world (Quisthoudt et al., 2012; Hickey et al., 2017; Osland et al., 2017b). At northern hemisphere range limits, climate variability associated with freeze events and absolute winter temperature minima have been identified as the most significant control on mangrove distribution (Stuart et al., 2007; Cavanaugh et al., 2014). However, at southern hemisphere range limits the role of climate is less pronounced, and the physical and geomorphological features of these coastlines limit the distribution of mangroves (Semeniuk, 1983; Osland et al., 2017b; Ximenes et al., 2018). Recent work on the South African distribution has shown that physical features and environmental conditions of estuaries are significant predictors of mangrove area (Raw et al., 2019a) (Section 5.1 of this report). The potential for mangrove range expansion along the South African coastline has previously been discussed, and individual trees have been recorded as pioneers at new locations (Whitfield et al., 2016; Peer et al., 2018). However, a quantitative approach is needed if the potential for mangrove expansion under climate change is to be considered.

Species distribution models (SDMs) and ecological niche model (ENMs) have become common in research related to applied ecology and biogeography (Sillero, 2011; Peterson et al., 2015). These models are broadly grouped as correlative models, mechanistic models, and process-oriented models. Correlative models are the most common and are developed by relating known locations of a species to the environmental conditions at those locations so that ecological requirements can be estimated (Araújo and Guisan, 2006; Mesgaran et al., 2014; Melo-Merino et al., 2020). One of the most widespread tools used for SDMs is the MaxEnt software package (Phillips et al., 2006; Merow et al., 2013). Besides providing a straightforward interface for use, MaxEnt has also been found to have a greater predictive accuracy than other methods (Merow et al., 2013). However, this approach is not without its' critics, particularly because results can be generated using various default settings and without careful consideration of the model inputs for specific ecological research questions (Warren and Seifert, 2011; Fourcade et al., 2014; Radosavljevic and Anderson, 2014; Morales et al., 2017). Developing better models with MaxEnt therefore requires transparency in selecting specific settings and validating the results when possible.

As the drivers of mangrove distribution along the South African coast have been quantitatively identified, this study aimed to apply an SDM to predict mangrove distribution for this region under future climate change. The aims of this study were to 1) identify potentially suitable estuaries beyond the current distribution of mangroves along the South African coastline, 2) determine whether estuaries that currently support mangroves could become unsuitable predicted climate change scenarios; and 3) determine whether estuaries beyond the current distribution of mangroves could become suitable under predicted climate change scenarios.

5.2.2 Methods

MaxEnt is an open-source software that uses the principles of maximum entropy to develop spatial species distribution models (SDMs) (Phillips et al., 2004, 2006; Phillips and Dudík, 2008; Phillips et al., 2017). This is achieved by using known occurrence locations and the environmental variables at these locations as “background data” to identify suitable habitat areas based on environmental variables within a broader “landscape”. The distribution model can then be projected to new locations or under new environmental conditions.

Mangroves are established (area > 0.5 ha) in 29 estuaries along the east coast of South Africa (Adams et al., 2016), and the environmental drivers of this distribution pattern have been quantitatively identified (Raw et al., 2019a; Section 5.1 of this report). For the development of a mangrove SDM in MaxEnt, the modelling “landscape”, or background data, consisted of points representing individual estuaries as potential habitats for mangroves as they do not occur on the open coast. The environmental data (particularly physical characteristics) associated with each estuary were extracted from the most recent version of the CSIR national database. The extracted variables included average annual temperature (°C), floodplain area (ha), mean annual runoff (m^3 x 10^6), daily flushing rate (m^3 x 10^6/d), and mouth opening frequency (% time the mouth is open) (Van Niekerk et al., 2017; Raw et al., 2019a; Van Niekerk et al., 2019) (Figure 5.2.1).

Four mangrove distribution models were developed with MaxEnt (Figure 5.2.2). In each model, the 29 estuaries where mangroves are established were used as the presence locations. Separate models were not run for the different mangrove species as diversity was low in this region. There were two dominant species, *Avicennia marina* and *Bruguiera gymnorhiza* that occur in 27 estuaries; *A. marina* covers the largest area. *Rhizophora mucronata* occurs in 14 estuaries and is associated with tidal creeks and channels. An additional three species (*Ceriops tagal, Lumnitzera racemosa*, and *Xylocarpus granatum*) occur only at the Kosi Estuary (Adams et al., 2016).

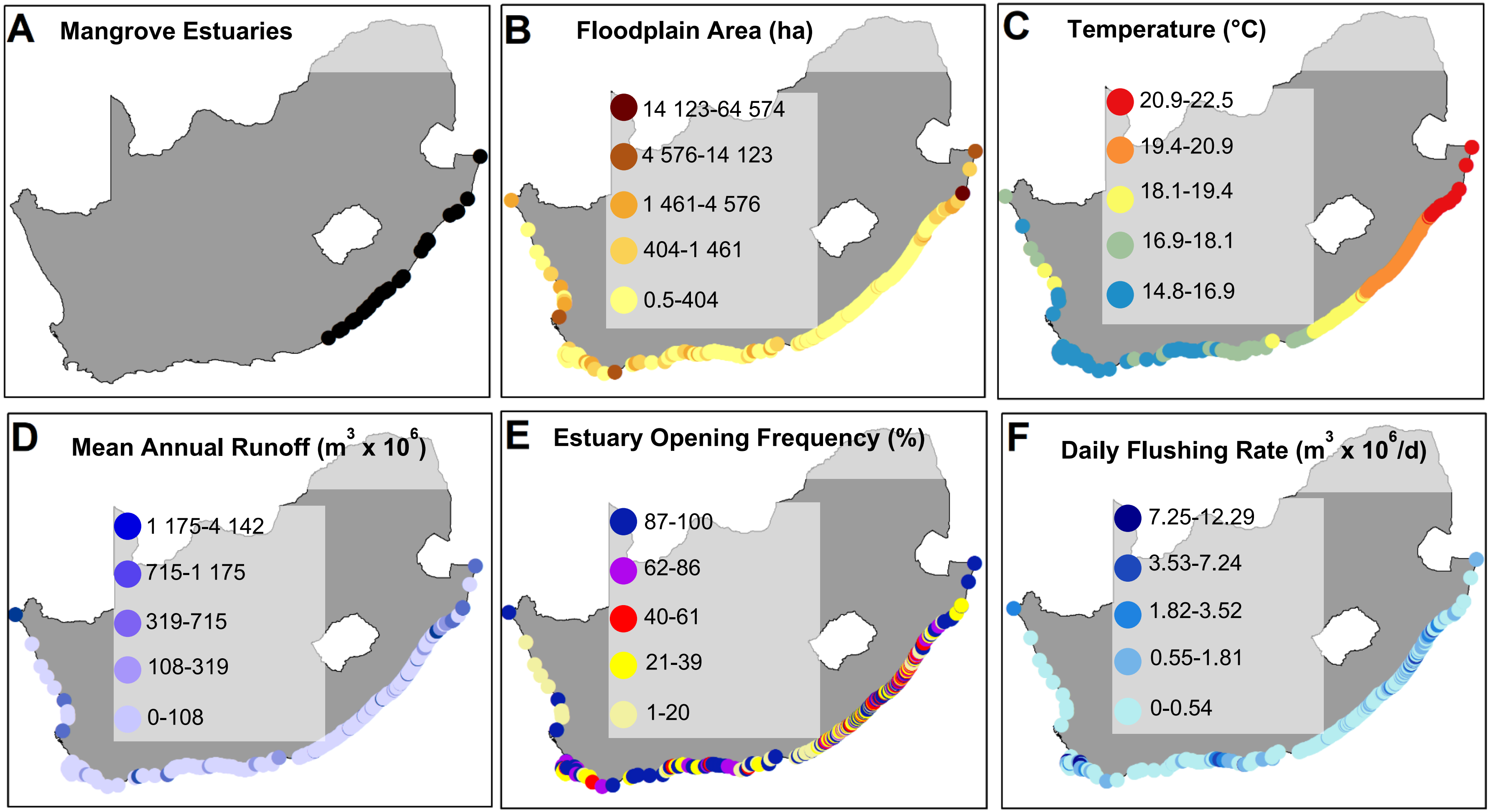


Figure 5.2.1. Estuaries along the South African coastline that support mangroves (**A**), Ranges of environmental variables: Floodplain Area (ha) (**B**), Average Annual Land Temperature (°C) (**C**), Mean Annual Runoff ($m^3 \times 10^6$) (**D**), Inlet Stability/Frequency that the Estuary Mouth is Open (%) (**E**), Daily Flushing Rate ($m^3 \times 10^6$/d) (**F**) extracted for individual estuaries from the CSIR database (Van Niekerk et al., 2017, 2019)

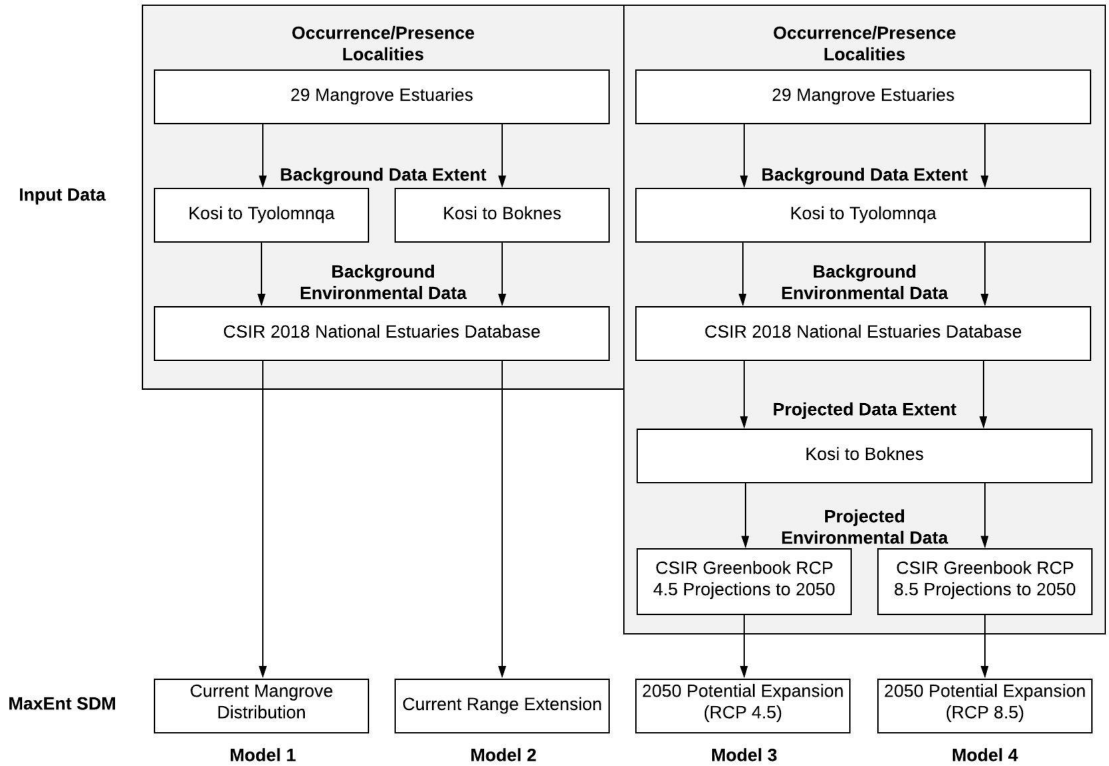


Figure 5.2.2. Schematic of the inputs to species distribution models developed in MaxEnt to predict habitat suitability of estuaries for mangroves along the east coast of South Africa.

For the initial model (Model 1) to compare the MaxEnt predicted distribution to the known occurrence localities of mangroves, the modelling landscape consisted of all estuaries within the current distribution (from Kosi to Tylomnqa). To determine the suitability of estuaries currently beyond mangrove occurrence (Model 2), the landscape was extended along the warm-temperate coastline. This provided an indication of whether there are already suitable estuaries beyond the current distribution into which mangroves have not been able to establish themselves. Algoa Bay forms a significant geomorphological and oceanographic feature that is be expected to influence dispersal of mangrove propagules. The modelling landscape was therefore not extended beyond the Boknes Estuary.

To project the suitability of estuaries for mangroves under climate change, the extended east coast range was used as the projected landscape (Kosi to Boknes), and projected environmental variables were developed by extracting data from the CSIR Greenbook (Engelbrecht et al., 2019). Temperature and rainfall predictions for 2050 under the RCP 4.5 and RCP 8.5 IPCC scenarios were extracted for the geographical location of each estuary. The average temperature increase was calculated for each scenario based on the 10th and 90th percentile predictions. These values were added to the current average annual land temperature at each estuary to obtain a predicted temperature by 2050. The average predicted change in rainfall (in mm) by 2050 was calculated in the same way from the 10^{th} and 90^{th} percentiles under each IPCC scenario. The percentage difference in this predicted value from the current rainfall was then calculated. This percentage difference value was used to scale the mean annual runoff, daily flushing rate, and mouth opening frequency variables accordingly to provide estimates under each IPCC scenario for these variables.

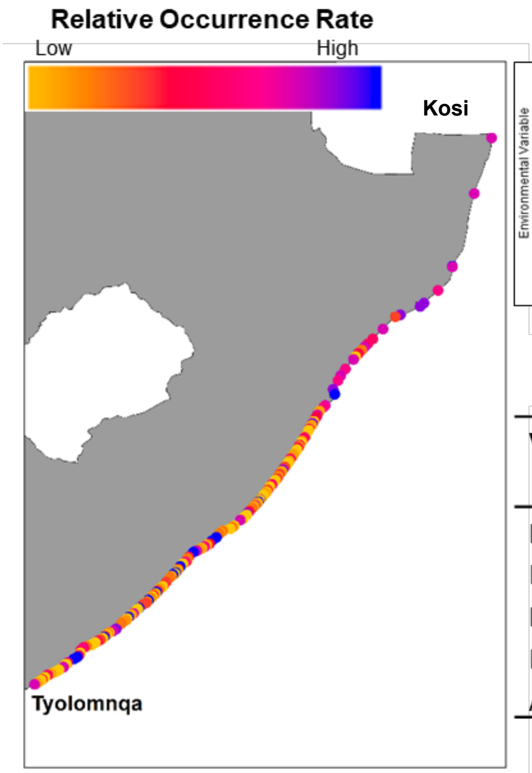
For this study, the species distribution models were initially run with the data randomly partitioned into a 75% training set, and a 25% testing set. After confirming the fit of the models, they were re-run using the whole dataset as the testing set. The species-with-data (SWD) format (Phillips et al., 2006; Elith et al., 2011) was used as both the occurrence localities and the background environmental data were point attributes. All models were run in MaxEnt version 3.4.1 (https://biodiversityinformatics.amnh.org/open_source/maxent/) with linear, quadratic, and hinge features. These features were selected based on the known relationships between mangrove area and the environmental variables used in the models (Raw et al., 2019a). Jack-knife tests of variable importance were used in each model and the AUC (area under curve) statistic was used as a measure of model performance (Phillips, 2017b) For each model, the raw model output (relative occurrence rate) was extracted as this provides an indication of habitat suitability (Phillips, 2017b). The relative occurrence rates calculated for each estuary under each model were spatially represented using ArcMap version 10.6 (ESRI, 2018). For each model, the relative occurrence rate values were imported as attributes to a point shape file of the estuary locations along the coastline. These values were displayed using “natural jenks” to categorize the estuaries in terms of habitat suitability. This allowed for comparisons between estuaries within the same modelling scenario.

5.2.3 Results and Discussion

5.2.3.1 Predicting Current Mangrove Distribution (Model 1)

A MaxEnt model to predict the current mangrove distribution (Figure 5.2.3) was run for comparison with the known occurrence localities in order to validate the method. The model performance, as indicated by AUC = 0.909, had a strong measure of separability and can be used to distinguish presence locations with high certainty (Fielding and Bell, 1997). The jack-knife analysis of variable importance shows that the environmental variable with the highest gain was Floodplain Area (Figure 5.2.3). The gain of the model is related to deviance, which is a measure of goodness of fit. The variable with the highest gain therefore makes the highest contribution towards improving the fit and explaining the variability in the model (Phillips, 2017b).

Floodplain Area was also identified as the most significant predictor of mangrove area along the South African coastline (Raw et al., 2019a; Section 5.1 of this report). The importance of this variable in the MaxEnt model therefore provides validation for this approach, as it agrees with previous quantitative methods to define mangrove distribution patterns along the South African coastline.



Analysis of Variable Contributions

Variable	Percent Contribution	Permutation Importance
MAR	40.9	9.6
FP_AREA	38.4	67
MTH_OP	18	13.6
DFR	2.6	9.3
AALT	0	0.4

Figure 5.2.3. MaxEnt results for Model 1, indicating the habitat suitability for mangroves in estuaries within the current distribution range and the importance of the environmental variables in the model (AALT = Average Annual Land Temperature, DFR = Daily Flushing Rate, FP_AREA = Floodplain Area, MAR = Mean Annual Runoff, MTH_OP = Estuary Mouth Opening Frequency).

Further validation of this approach is shown in Table 5.2.1. Model 1 predicted a medium to high relative occurrence rate for all estuaries that currently support mangroves. Estuary grouping based on relative occurrence rate showed those that currently support mangroves were most frequently predicted by the model to have suitable habitat (Groups 1 and 2 in Table 5.2.1).

Table 5.2.1. Estuaries predicted to have high (Group 1) to medium (Group 5) habitat suitability for mangroves by Model 1 as indicated by the first five (out of ten) groups formed by natural jenks of the relative occurrence rates. Estuaries that currently support mangrove habitats are shaded in grey.

Group 1	Group 2	Group 3	Group 4	Group 5
Durban Bay	uMhlathuze	Buffalo	Cefane	Bulura
Gqunube	uMkhomazi	Kobonqaba	Kwenxura	Cintsa
Great Kei	uMlalazi	Kosi	iZimbokodo	iFafa
Kwelera	uMhtavuna	iLovu	uMhlanga	Gxulu
Mbashe	Mtentu	aMatigulu/iNyoni	Mnenu	Lwandile
Mdumbi	uMzimkhulu	uMdloti	Mzimvubu	aMahlongwa
uMngeni	Nahoon	iMfolozi/uMsunduze	iNhlabane	iMbizana
Mngazana	Nxaxo/Ngqusi	uMgobezeleni		Mncwasa
Mntafufu	Richards Bay	uMhlali		Mpako
Mtakatye	Tyolomnqa	Mngazi		uMsimbazi
Mtata		Mnyameni		uMthwalume
Mzamba		Msikaba		uMvoti
Mzintlava		iNonoti		uMuziwezinto
Nqabara/Nqabarana		Shixini		uMzumbe
Qora		iSiphingo		Ncera
St Lucia		uThongathi		Nkodusweni
Xora				Ntlonyane
				Qinira
				Quko
				Sikombe
				Sinangwana
				uThukela
				uMgababa
				iZinkwasi

Model 1 predicted three estuaries (Gqunube, Qora, uMzimkhulu) to have suitable available habitat although mangroves have not been recorded in these systems (Table 5.2.1). The Gqunube Estuary is the furthest south, situated beyond the natural range limit, which occurs at the Nxaxo/Ngqusi Estuary. Some mangroves further south of the Gqunube Estuary (at Nahoon and Tyolomnqa estuaries) have been artificially established by planting. A latitudinal gradient in winter temperature minima does not control mangrove distribution patterns along the South African coastline (Quisthoudt et al., 2012; Osland et al., 2017b; Raw et al., 2019a). The low importance of the temperature variable in Model 1 confirms this (Figure 5.2.3). If physical conditions of an estuary are suitable for mangroves, and there are no climatic controls on distribution, it is most likely that establishment has not occurred further south as a result of limitations on propagule dispersal. The Kwelera Estuary ~ 5 km north of Gqunube Estuary (by straight line distance along the coast), supports a stand of *A. marina* close to the estuary mouth (Bolosha, 2017). It is unclear whether these trees have established naturally or have been planted, but seedlings have recruited and begun to establish upstream (Bolosha, 2017). The closest confirmed naturally established mangroves occur at Great Kei Estuary, ~ 40 km north of Gqunube Estuary. These were recently recorded as a range expansion along the coast (Saintilan et al., 2014; Whitfield et al., 2016).

The Qora Estuary is located within the current distribution range for mangroves. For this study, only estuaries with mangrove areas > 0.5 ha were used as mangrove occurrence records. Upon inspection of the Qora Estuary using Google Earth satellite imagery (Figure 5.2.4), a small mangrove area is evident in this system. The prediction of high habitat suitability for mangroves at Qora Estuary by Model 1 is therefore confirmed. This is a new record of mangrove occurrence at this estuary.

The uMzimkhulu Estuary, located in KwaZulu-Natal, was also predicted to have suitable mangrove habitat in Model 1. There are no previous records of mangroves occurring in the uMzimkhulu Estuary from early surveys (Macnae, 1963) or recent studies (Adams et al., 2016). Macnae (1963) attributes the absence of mangroves to limited floodplain area, but an area of ~ 251 ha has been recorded. As Floodplain Area is a significant predictor of mangrove occurrence, this is the likely reason Model 1 predicted relatively high habitat suitability for this estuary. Google Earth satellite imagery (Figure 5.2.5) shows the floodplain is dominated by reeds. This is typical of estuaries with high freshwater inflow.

Figure 5.2.4. Google Earth satellite imagery (2019) confirms the occurrence of a small mangrove area in the Qora Estuary, Eastern Cape.

Figure 5.2.5. Google Earth satellite imagery (2019) of the uMzimkhulu Estuary in Port Shepstone, KwaZulu-Natal.

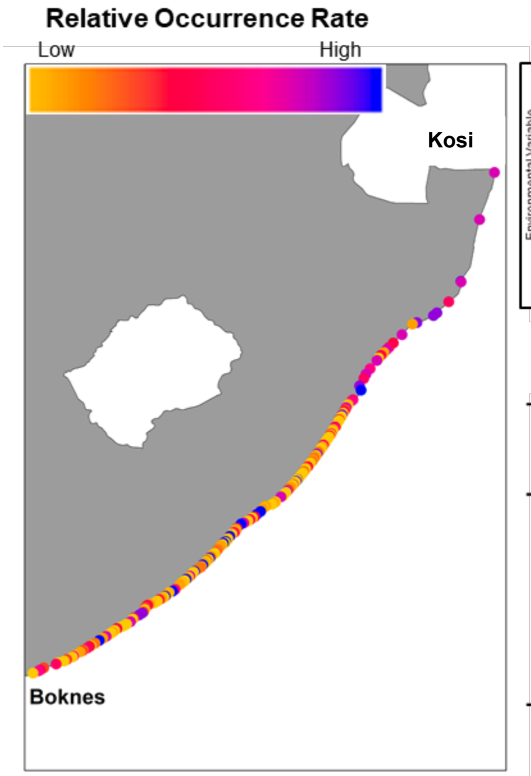
Two estuaries with large mangrove areas (Kosi and iMfolozi/uMsunduze) were not predicted to have the highest habitat suitability. This can be explained by the importance of other variables, besides Floodplain Area, in the model. Mean Annual Runoff (MAR) and Daily Flushing Rate (DFR) are also significant predictors of mangrove area (Raw et al., 2019a; Section 5.1 of this report).

Kosi is an estuarine lake system and has a large floodplain area (~ 7000 ha). For most estuaries, the relationship between MAR and Floodplain Area is linear, and this defines the DFR (higher MAR in a smaller estuary leads to a greater DFR). At the Kosi Estuary, the MAR is naturally very low but as the system receives groundwater inflow and there is a large tidal exchange volume, the calculated DFR is higher than expected. As MAR was the only proxy for freshwater input used in these modelling scenarios, it could explain why Model 1 predicted a lower habitat suitability for Kosi Estuary. Alternatively, the prediction could be related to the unique distribution of mangrove area at the estuary. Its lower reaches support a mangrove stand dominated by *A. marina*, but mangroves also occur further upstream along the lake system where the tidal influence is significantly reduced (DWS, 2016). Most notably, large stands of *L. racemosa* are established along the Mtando Channel that connects Lake Mpungwini to Lake Makhawulane. Areas of *R. mucronata* and *B. gymnorhiza* also occur along the lake system. This unique geomorphology and distribution of mangroves is unlikely to be accurately captured by the model, which is trained to focus on similarities between systems to predict mangrove occurrence.

The iMfolozi/uMsunduze Estuary also has a large floodplain area (~ 8000 ha), but in contrast to Kosi Estuary, it is characterized by a high MAR (~ 800 $m^3 \times 10^6$). DFR is a significant predictor of mangrove area as higher flushing rates translate to water movement that keeps the estuary mouth open to maintain tidal fluctuations. A low DFR can result in mouth closure, especially common along the high energy coastline of KwaZulu-Natal. Prolonged mouth closure can cause mangrove dieback due to inundation stress (Breen and Hill, 1969; Mbense et al., 2016). However, a very high DFR can lead to scouring and a reduction of available habitat for mangroves. This threshold at which DFR is predicted by the model to have a negative impact on mangroves should be identified. It is likely that Model 1 predicted a lower habitat suitability at the iMfolozi/uMsunduze Estuary because of a high DFR.

5.2.3.2 Predicting Current Mangrove Distribution over Extended Range (Model 2)

As indicated in Model 1, some estuaries that occur beyond the natural distribution limit for mangroves provide suitable habitat, and in some cases, mangroves have been successfully planted in these estuaries. A MaxEnt model was run to predict the suitability of estuaries beyond the current mangrove distribution (Figure 5.2.6). Model 2 had a strong measure of separability (AUC = 0.914) and the jack-knife analysis of variable importance shows Mean Annual Runoff to be the environmental variable with the highest gain (Figure 5.2.3). This suggests it to have the most useful information for the model if it is used in isolation. However, Floodplain Area had the highest permutation importance indicating that the model gain decreases the most if this variable is removed.



Analysis of Variable Contributions

Variable	Percent Contribution	Permutation Importance
MAR	52.6	11.1
FP_AREA	26.8	70.0
MTH_OP	15.5	13.5
AALT	2.7	2.6
DFR	2.4	2.8

Figure 5.2.6. MaxEnt results for Model 2, indicating the habitat suitability for mangroves in estuaries within the extended distribution range and the importance of the environmental variables in the model (AALT = Average Annual Land Temperature, DFR = Daily Flushing Rate, FP_AREA = Floodplain Area, MAR = Mean Annual Runoff, MTH_OP = Estuary Mouth Opening Frequency).

As in Model 1, Model 2 predicted a medium to high relative occurrence rate for all estuaries that support mangroves (Table 5.2.2). By extending the modelling landscape beyond the current distribution limit (at Tylomnqa Estuary) to Boknes Estuary, Model 2 predicted high habitat suitability for mangroves at the Keiskamma Estuary. This estuary has a large intertidal area supporting salt marsh habitat that would also be suitable for mangroves (Figure 5.2.7). The Keiskamma Estuary is ~ 10 km south of the Tyolomnqa Estuary and ~ 100 km south of the Great Kei Estuary.

Model 2 predicted a relatively lower habitat suitability for more mangrove estuaries (in Groups 4 and 5) compared to Model 1. This could be an artefact of extending the modelling landscape beyond the occurrence localities.

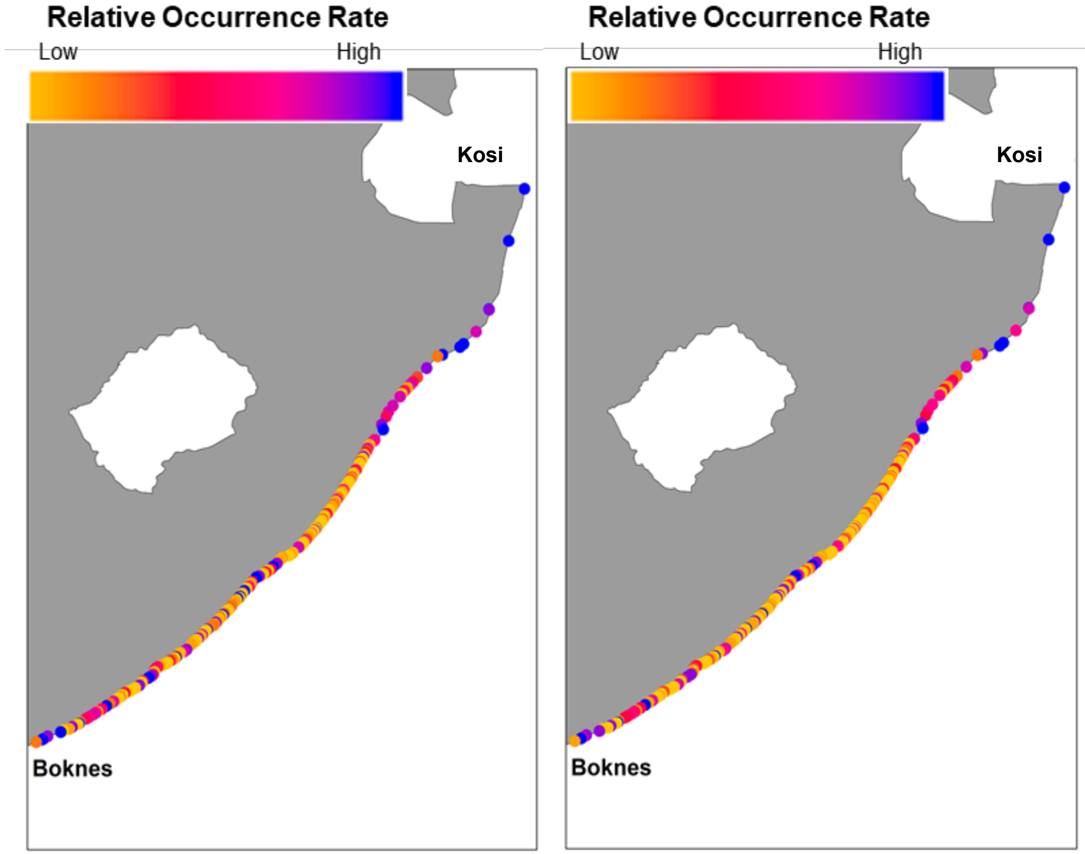
Table 5.2.2. Estuaries predicted to have high (Group 1) to medium (Group 5) habitat suitability for mangroves by Model 2 as indicated by the first five (out of ten) groups formed by natural jenks of the relative occurrence rates. Estuaries that currently support mangrove habitats are shaded in grey.

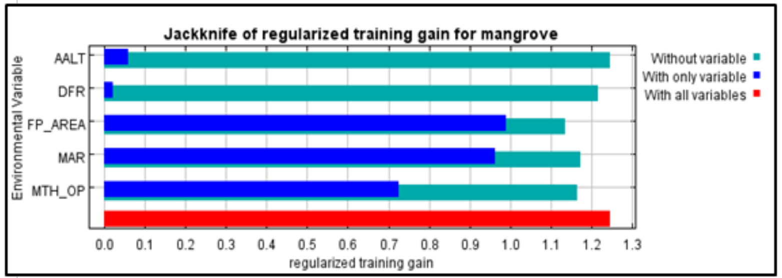
Group 1	Group 2	Group 3	Group 4	Group 5
Durban Bay	Kwelera	Buffalo	Bushmans	Bira
Gqunube	iMfolozi/uMsunduze	Kosi	Great Fish	Kowie
Great Kei	uMngeni	iLovu	iZimbokodo	Kwenxura
Keiskamma	uMhlathuze	aMatigulu/iNyoni	uMhlanga	Mncwasa
Kobonqaba	uMkhomazi	uMdloti	Nxaxo/Ngqusi	Mnenu
Mbashe	uMlalazi	uMgobezeleni		Mzimvubu
Mdumbi	uMhtavuna	uMhlali		uMuziwezinto
Mngazana	Mtentu	Mngazi		uMzumbe
Mntafufu	uMzimkhulu	Mnyameni		iNhlabane
Mtakatye	Nahoon	Msikaba		Sinangwana
Mtata	Richards Bay	iNonoti		iSiphingo
Mzamba		Shixini		iZinkwasi
Mzintlava		uThongathi		
Nqabara/Nqabarana		Tyolomnqa		
Qora				
St Lucia				
Xora				

Figure 5.2.7. Google Earth satellite imagery (2019) of the Keiskamma Estuary.

5.2.3.3 Predicting 2050 Potential Mangrove Distribution: RCP 4.5 and 8.5 (Models 3 and 4)

Models 3 and 4 show the potential habitat suitability for mangroves in 2050 under the two different IPCC scenarios. Both these models use the same inputs for Model 1 as a base, and therefore have the same outcome in terms of AUC, variable importance, and variable contribution to gain (Figure 5.2.8).





Analysis of Variable Contributions

Variable	Percent Contribution	Permutation Importance
MAR	40.9	9.6
FP_AREA	38.4	67
MTH_OP	18	13.6
DFR	2.6	9.3
AALT	0	0.4

Figure 5.2.8. MaxEnt results for Model 3 (left) and Model 4 (right), indicating the habitat suitability for mangroves in estuaries within the extended distribution range by 2050 under the RCP 4.5 and RCP 8.5 IPCC scenarios respectively. The importance of the environmental variables was the same as Model 1, which was used as a base for the respective projections (AALT = Average Annual Land Temperature, DFR = Daily Flushing Rate, FP_AREA = Floodplain Area, MAR = Mean Annual Runoff, MTH_OP = Estuary Mouth Opening Frequency).

Table 5.2.3. Estuaries predicted to have high (Group 1) to medium (Group 5) habitat suitability for mangroves by Model 3 as indicated by the first five (out of ten) groups formed by natural jenks of the relative occurrence rates. Estuaries that currently support mangrove habitats are shaded in grey.

Group 1	**Group 2**	**Group 3**	**Group 4**	*Group 5*
Bushmans	Buffalo	Bira	Cefane	Bulura
Durban Bay	Kariega	Gqutywa	East Kleinemonde	Cintsa
Gqunube	Kasouga	Kobonqaba	iZimbokodo	iFafa
Great Fish	West Kleinemonde	iLovu	uMhlanga	Gxulu
Great Kei	Kwelera	uMdloti	Mnenu	Kwenxura
Keiskamma	aMatigulu/iNyoni	uMhlali	Mtati	aMahlongwa
Kosi	Mbashe	Mnyameni	uMzumbe	iMbizana
Kowie	iMfolozi/uMsunduze	Msikaba		Mgwalana
Mdumbi	uMngeni	Mtentu		Mncwasa
uMgobezeleni	uMkhomazi	iNhlabane		Mpekweni
uMhlathuze	Mngazi	iNonoti		uMsimbazi
uMlalazi	uMhtavuna	Shixini		Mtana
Mngazana	Mzamba	iSiphingo		uMvoti
Mntafufu	uMzimkhulu	uThongathi		Mzimvubu
Mtakatye	Mzintlava	Tyolomnqa		uMuziwezinto
Mtata	Nxaxo/Ngqusi			Nkodusweni
Nahoon	Qora			Qinira
Nqabara/Nqabarana				Quko
Richards Bay				Sinangwana
St Lucia				uMgababa
Xora				iZinkwasi

Both Models 3 and 4 predicted a high habitat suitability for mangroves at the Bushmans, Great Fish, and Keiskamma estuaries, which are all further south than the current distribution limit (Tables 5.2.3 and 5.2.4). Model 3 also predicted a high habitat suitability for the Kowie Estuary. Although this estuary has been developed into a marina for Port Alfred, there is still salt marsh area that could be potentially suitable for mangroves (Figure 5.2.9). Interestingly, the Kariega

Estuary immediately adjacent and similar to the Bushmans Estuary, was not placed in the group of highest habitat suitability. Upon inspection of input data, the floodplain area for the Bushmans Estuary is recorded as greater than the Kariega Estuary (~ 850 ha and ~ 500 ha respectively), although the MAR (~ 15.5 $m^3 \times 10^6$) is lower than the Bushmans Estuary (~ 36.8 $m^3 \times 10^6$). This could account for differences in predicted habitat suitability for mangroves under the different climate change scenarios as the projected environmental variables are directly related to the current records.

Table 5.2.4. Estuaries predicted to have high (Group 1) to medium (Group 5) habitat suitability for mangroves by Model 4 as indicated by the first five (out of ten) groups formed by natural jenks of the relative occurrence rates. Estuaries that currently support mangrove habitats are shaded in grey.

Group 1	Group 2	Group 3	Group 4	Group 5
Bushmans	Mtakatye	Mzamba	Bira	Cefane
Durban Bay	Gqunube	Buffalo	Gqutywa	East Kleinemonde
Great Fish	Kasouga	Kariega	Kobonqaba	iZimbokodo
Great Kei	West Kleinemonde	aMatigulu/iNyoni	iLovu	uMhlanga
Keiskamma	Kowie	uMkhomazi	uMdloti	Mnenu
Kosi	Kwelera	Mngazi	uMhlali	Mtati
uMgobezeleni	Mbashe	uMhtavuna	Mnyameni	uMzumbe
uMhlathuze	Mdumbi	Mtentu	Msikaba	
Mngazana	iMfolozi/uMsunduze	uMzimkhulu	iNhlabane	
Mntafufu	uMngeni	Nxaxo/Ngqusi	iNonoti	
Mtata	uMlalazi	iSiphingo	Shixini	
Nqabara/Nqabarana	Mzintlava	Tyolomnqa	uThongathi	
Richards Bay	Nahoon			
St Lucia	Qora			
Xora				

* Mzimvubu Estuary, which currently supports mangroves, was placed in Group 7 by Model 4.

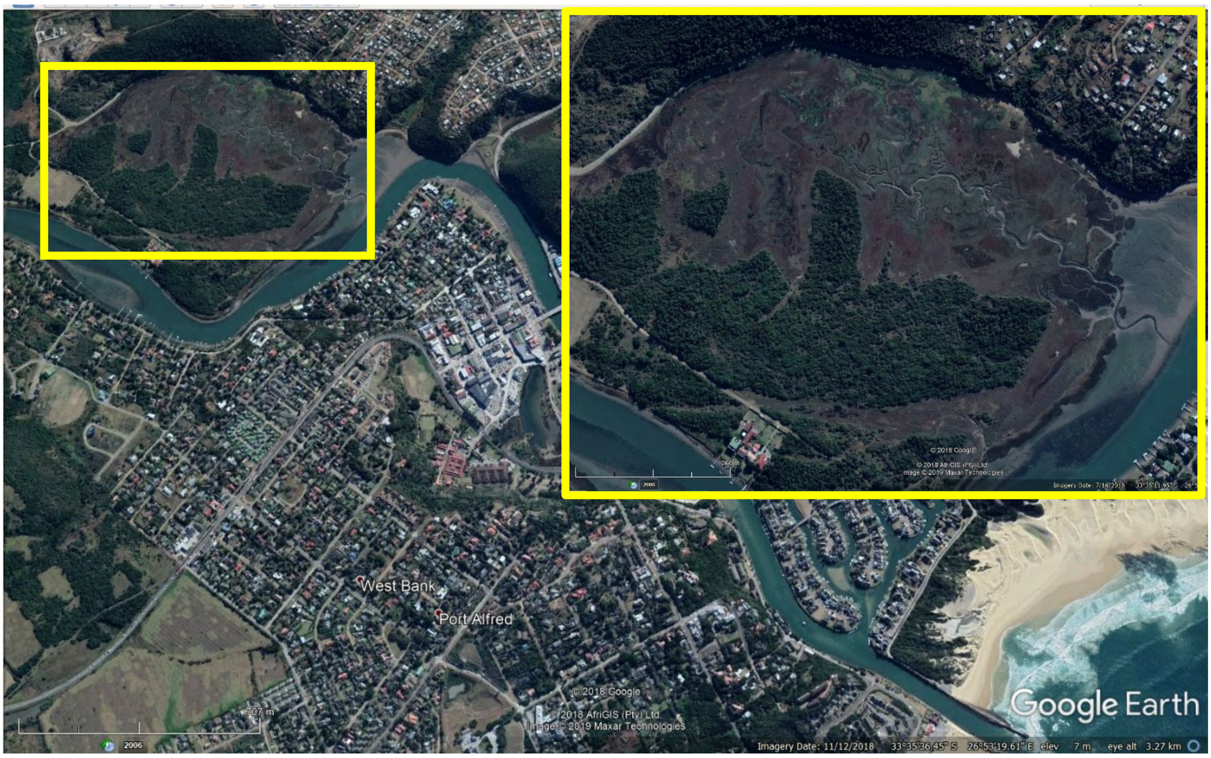


Figure 5.2.9. Google Earth satellite imagery (2019) for the Kowie Estuary in Port Alfred, Eastern Cape.

The uMgobezeleni Estuary was also predicted to have high habitat suitability under both models. This estuary, situated in KwaZulu-Natal, previously supported mangrove habitat (Macnae, 1963). However, a road bridge built across the estuary in the 1970s raised the water level, and the prolonged inundation caused the death of the mangrove trees (Bruton and Appleton, 1975; Taylor, 2016). Currently, a few tall (> 18 m) *B. gymnorhiza* trees and some seedlings have been reported in the system (Peer et al., 2018).

Under both the RCP 4.5 and RCP 8.5 IPCC scenarios, the east coast of South Africa is predicted to experience an increase in temperature as well as an increase in the volume and frequency of rainfall (Engelbrecht et al., 2019; Figure 5.2.10). An increasing temperature is not expected to have a negative effect on South African mangroves, as these species have distribution ranges that extend to warmer tropical latitudes (Duke et al., 1998; Giri et al., 2011). In combination, an increase in temperature and rainfall could potentially result in species range expansions. Currently, two of the five mangrove species (*Ceriops tagal* and *Lumnitzera racemosa*) in South Africa are restricted to the northernmost Kosi Estuary. These species occur more frequently along the east African coastline (Bosire et al., 2016) and could therefore be expected to expand southwards if climatic conditions become more suitable. Modelling the potential for range expansion of these species would require extending the modelling landscape along the coastline of Mozambique, and perhaps even further north, to represent the current distribution range. However, this presents an additional challenge as it is unlikely that the environmental variables used in this study influence mangrove distribution to the same effect as along the high energy South African coastline. For example, differences in coastal geomorphology, including the occurrence of larger deltas and bays, allow for the development of much larger mangrove forests. It would therefore be necessary to first investigate drivers of distribution patterns along this extended range.

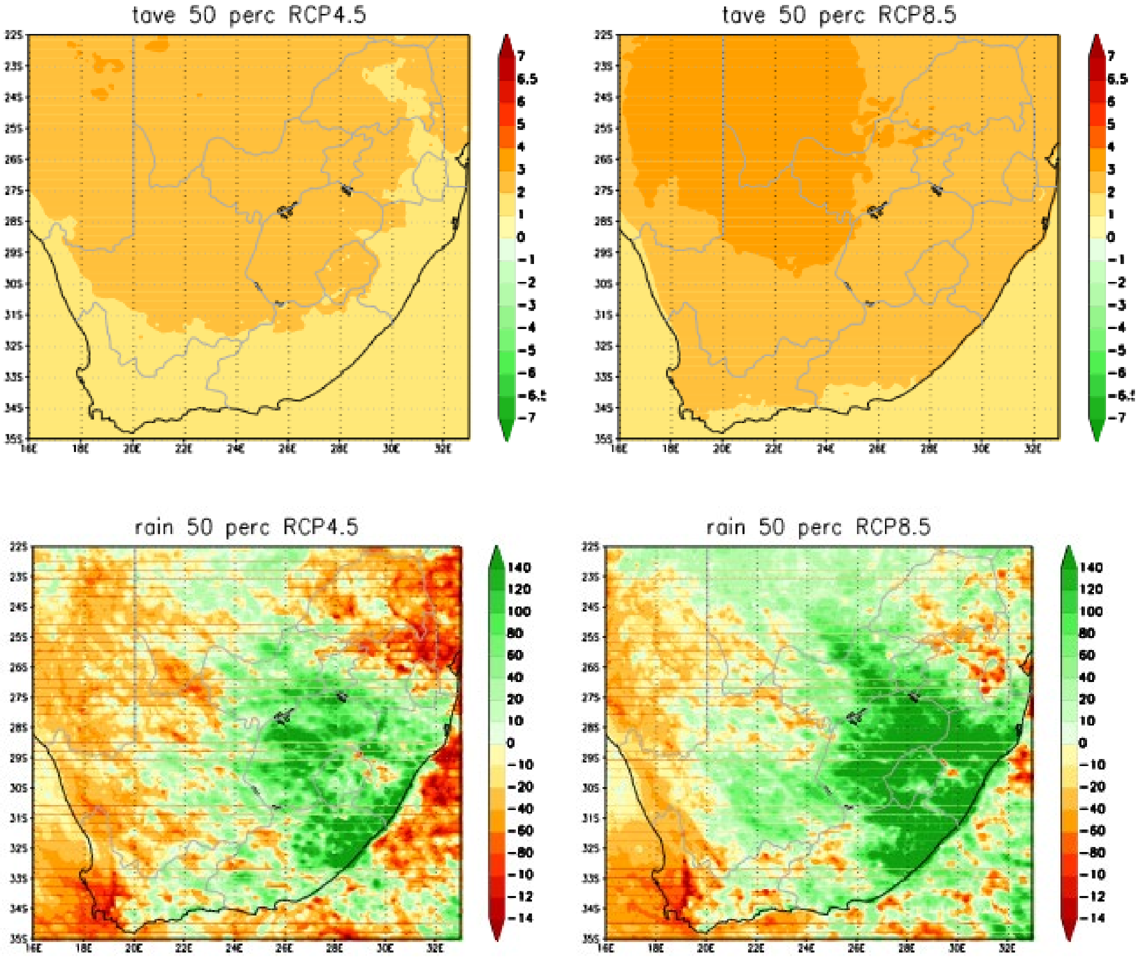


Figure 5.2.10. Projected change for 2021-2050 in temperature (top panels) and rainfall (bottom panels) relative to the period from 1961-1991 under RCP 4.5 (left panels) and RCP 8.5 (right panels) IPCC scenarios. Adapted from Engelbrecht et al. (2019).

The SDMs developed in this study indicate that there are estuaries beyond the current mangrove distribution that have suitable habitat, and that more of these estuaries will become suitable under both climate change scenarios investigated. However, the natural dispersal of mangrove propagules is a significantly limiting factor to extending the distribution of this habitat along the east coast of South Africa. Previous work on the dispersal of mangrove propagules in this region has focused on *A. marina*, as it occurs in the highest number of estuaries and is a pioneer species. One factor to limit dispersal of *A. marina* propagules in this region is the tendency of propagules to sink instead of float. Early work by Steinke (1986) showed a high percentage of propagules collected from mangrove forests at southern estuaries, such as Mngazana (83.3%), Mbashe (98.2%), Nxaxo/Ngqusi (94.6%), and Nahoon (100%), would sink. In contrast, a high percentage from northern estuaries, such as St Lucia (66.7%), Richards Bay (78%), Durban Bay (67.7%), would float. Mangrove forests in southern estuaries are thus more likely to be self-seeding, and these forests could have been established by a founding event of "sinker" trees, which further limits dispersal. A recent genetic study for *A. marina* along the east African range supports this idea, as trees at southern locations showed reduced mixing with northern populations (De Ryck et al., 2016). In addition, the local coastal geomorphology and oceanographic conditions are also expected to limit the potential of mangrove propagule dispersal. Steinke and Ward (2003) used plastic drift cards to estimate potential dispersal and transport rates for *A. marina* propagules from different estuaries. The highest transport rates were recorded for cards released from Richards Bay and Durban Bay. Cards released from Richards Bay were retrieved as far south as Kwelera Estuary 22 days after being released. This dispersal distance has been recently reported for plastic nurdles spilled from a shipping container during rough storm conditions in Durban on 17 October 2017 (Schumann et al., 2019). Nurdles were collected in East London on 30 October 2017, 13 days after the spill. These nurdles are considerably smaller than mangrove propagules (~ 100 nm-5 mm in diameter) and would account for the faster dispersal time. In combination, these studies indicate that mangrove propagules have the potential to move south from northern estuaries. Future work should focus on identifying whether local-scale physical features (related to flow and mouth morphology) are preventing recruitment into suitable estuaries, or whether there are physiological limitations on establishment.

5.3 Salt Marsh at the Tip of Africa: A Review of Patterns, Processes and Changes in Response to Climate Change

Adams JB (accepted) Salt marsh at the tip of Africa: A review of patterns, processes and changes in response to climate change. *Estuarine, Coastal and Shelf Science*

5.3.1 Introduction

Salt marshes in South Africa occur in the sheltered estuaries distributed along the ~3000 km coastline. They lie alongside saline water bodies and support vegetation communities of herbs, grasses or low shrubs. Although mostly exposed to the air, the plants experience periodic flooding from tidal or non-tidal variations in water level of the adjacent water body (Adams et al., 2016). Some define salt marshes as areas subject to tidal influences (e.g. Weis and Butler, 2009) but in South Africa, they are also taken to include seldom flooded supratidal habitat that supports halophytic macrophyte communities. Supratidal salt marsh occurs at > 1.5 m amsl and leads into an ecotone area > 2.5 m amsl that is inhabited by terrestrial plant species (Veldkornet et al., 2015a). The supratidal salt marsh may be flooded as little as twice a year during exceptional spring tide events (Adams et al., 1999).

This study contributes to an understanding of the future of salt marshes and human benefits under climate change. Expected climate change conditions having a particular influence on salt marshes are sea level rise, increase in sea storms and wave height, changes in river discharge (droughts/floods), increased CO_2 levels and higher temperatures. Research has shown how this alters the key abiotic stressors – changing inundation patterns, salinity gradients and sediment biogeochemistry (e.g. organic matter supply). In this study, the relationship between these stressors, ecological processes and ecosystem attributes is described. The biogeographical patterns observed along the South African coastline present an important opportunity for climate change research as the transition between subtropical and warm temperate regions and between cool temperate and warm temperate regions are expected to be significantly influenced. Range expansions are already occurring due to warming as described in Whitfield et al. (2016). Van Niekerk (2018) has completed a recent assessment of climate change effects on South African estuaries providing an opportunity to evaluate salt marsh responses. Changes in ocean circulation processes are driving shifts in the coastal temperature regimes of the transitional zones, with related biological responses such as range extensions and contractions. The largest changes are expected along the cool temperate west coast and subtropical east coast with changes in the frequency and duration of mouth closure and salinity regimes, which in turn will affect critical ecosystem services such as nursery function for fish.

This study is globally relevant as little is known about southern hemisphere salt marshes in Africa, and data are needed for comparative purposes. Southern hemisphere estuaries differ from northern hemisphere systems in that they are predominantly microtidal (tidal range < 2 m) and small. At the southern tip of Africa 70% of estuaries are less than 50 ha and are influenced by high wave energy from the wave-dominated coast. Owing to strong wave action and high sediment availability, more than 90% of the estuaries have restricted inlets, with more than 75% closing for varying periods of time when a sandbar forms across the mouth (Whitfield, 1992; Cooper, 2001; Van Niekerk, 2018). The mean annual run-off of most South African rivers is variable, fluctuating between floods and extremely low to zero flow; during low flow the mouth of the estuary remains closed to the sea. These factors and the dearth of coastal plain estuaries limit the establishment and development of salt marsh. Most of the estuaries are laterally confined as they are located in bedrock valleys (Cooper, 2001).

Identification of ecosystem services (ES) is an important step towards the conservation of salt marsh habitats. Ecosystem goods and services represent the flow of materials, energy and information from both natural and human-modified ecosystems to society (Costanza et al., 1997). A brief history of ecosystem services and natural capital can be found in Costanza et al. (2017). This study identifies the ecosystem services associated with salt marshes and an important next step will be to place a monetary value on these services. Although this approach is often questioned, government departments and other decision makers in South Africa are calling for this information. The country is economically divided, and the vulnerable majority of the population has a high reliance on ecosystem services (Van Niekerk, 2018). Thus, the value of ecosystems to society should be communicated using monetary and non-monetary means for improved decision making.

Barbier et al. (2011) outlined the most important ecosystem services provided by salt marsh habitats that include coastal protection, erosion regulation, water purification, maintenance of fisheries and carbon sequestration. Salt marsh provides coastal protection by stabilizing the sediment, increasing the height of the intertidal zone and reducing the duration of storm surges. Vegetation slows down terrestrial runoff and suspended sediments are deposited allowing for nutrient uptake. Salt marshes are important for the production of fisheries species including shrimp, oysters, clams and other fishes. Due to their complex and rigid plant structure, salt marshes make it difficult for large fish to enter and thus provide protection and shelter for juvenile fish, shrimp and shellfish (Barbier et al., 2011). A global model for wetlands ecosystem services was developed (Janse et al., 2018) but this excluded coastal wetlands. Himes-Cornell et al. (2018) completed a systematic review focussing on the valuation of blue forests and concluded that the ecosystem services of salt marsh are the most understudied, despite them having a greater global coverage than mangrove and seagrass habitats.

Recently, Davidson et al. (2019) assessed annual changes in the monetary value of coastal ecosystems based on their annual rates of change. The largest monetary values in temperate regions came from salt marshes, seagrasses and tidal flats. In tropical regions, coral reefs and mangroves were important coastal habitats.

This review identifies past patterns of change so these can be used to predict potential future change, and in so doing guide salt marsh conservation and restoration initiatives. It has the added value of generating data on southern hemisphere salt marshes that will allow the inclusion in global comparisons of ecosystems about which relatively little is known. According to Davidson et al. (2018) little reliable wetland data exists for Africa and the Neotropics. The responses of salt marsh to climate change in open and closed estuaries is outlined. No single study has provided these relationships and therefore these results will be useful in a global context.

5.3.2 Salt marsh in South Africa

Although salt marshes in South Africa have larger spatial extents in estuaries with regular tidal exchange and gentle topographic gradients, they also occur in temporarily closed estuaries (Adams et al., 2016). Here the frequency and duration of open mouth conditions determines the extent of the salt marsh and the species found typically form a sub-set of the full suite of species occurring in permanently open estuaries. Similarly, in Australia fringing salt marsh persists during the closed phase of intermittently open coastal lagoons (Ross and Adam, 2013). High salinity conditions as a result of barrier overwash can result in the growth of large salt marsh areas in the temporarily open/closed estuaries along the south east coast of South Africa (Colloty et al., 2002). In comparison, high rainfall (> 800 mm/annum) and leaching of salts favour the development of brackish reeds, sedges and grasses in the temporarily closed estuaries in the subtropical province of KwaZulu-Natal situated further north along the east coast of South Africa. An electrical conductivity value in the sediment of 0.780 mS/cm (80 mM NaCl) has been suggested as the threshold between halophytes and salt-sensitive plants (glycophytes) (Flowers and Colmer, 2015).

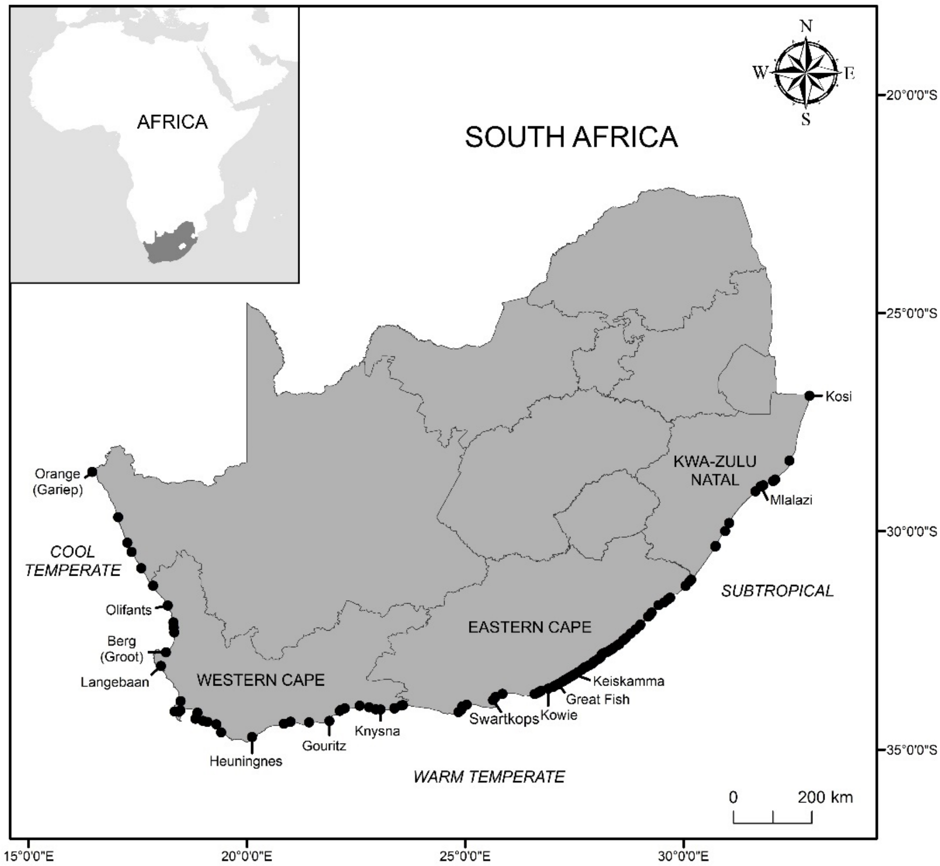


Figure 5.3.1. Distribution of salt marsh along the coast of South Africa.

Distribution of salt marsh along the South African coastline follows biogeographic zoning (Figure 5.3.1); cool temperate on the west coast, the warm temperate zone extends approximately from Cape Point to the Mbashe River in the Eastern Cape. The subtropical zone is on the east coast (Harrison, 2004). Mangroves replace salt marsh in the intertidal zone of open estuaries in the subtropical zone (Adams et al., 2016). Salt marsh species that have the widest distribution include *Bassia diffusa* (Thunb.) Kuntze, *Cotula coronopifolia* L., *Limonium linifolium* (L.f.) Kuntze, *Juncus kraussii* Hochst., *Phragmites australis* (Cav.) Steud and *Triglochin striata* Ruiz & Pav. Phylogeographic breaks in *Juncus kraussii* and *Triglochin striata* corresponded with the temperate and subtropical boundary, whereas the phylogeographic patterns identified in *Sarcocornia pillansii* (Moss) A.J.Scott and *S. tegetaria* (Steffen, Mucina & Kadereit) reflect the boundary between cool and warm temperate provinces (Veldkornet, 2016, Veldkornet et al., 2019). Species occur in specific zones where the tidal elevation gradient is distinct; otherwise they form mosaics (Adams et al., 2016; Figure 5.3.2).

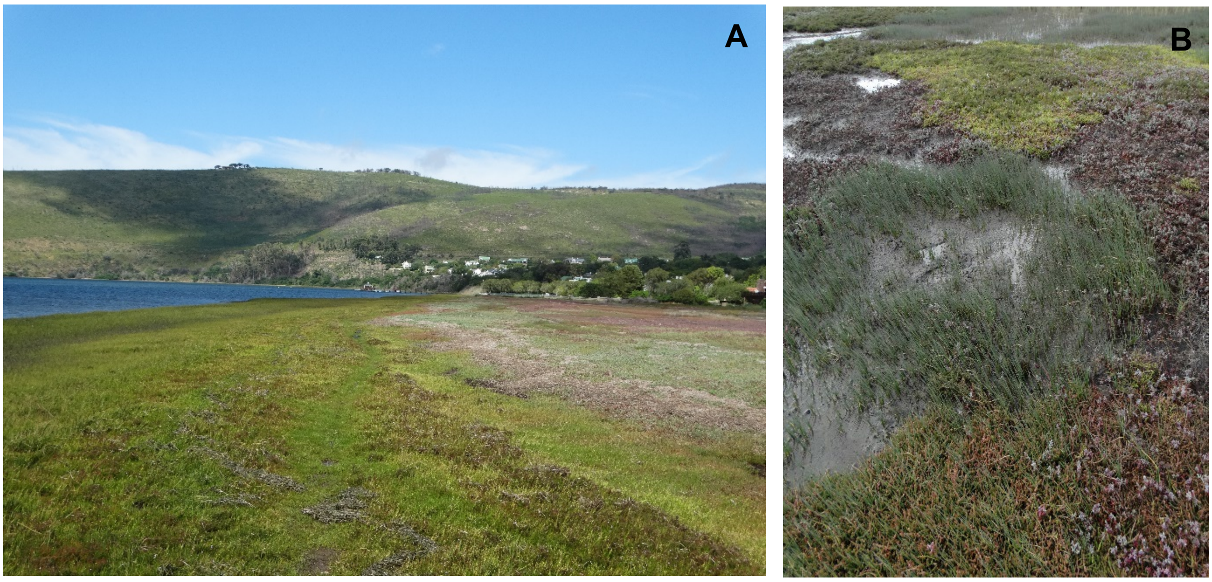


Figure 5.3.2. Salt marsh plants occur in distinct zones along a tidal inundation gradient (Knysna Estuary) (**A**) or as mosaics (Swartkops Estuary) (**B**).

South Africa is a semi-arid country and the National Water Act 1998 has motivated research on the ecological water requirements of estuaries (Adams, 2013). This required an understanding of the ecophysiological tolerances of salt marsh plants in order to predict future responses but is not the focus of this review (see Tabot and Adams, 2013). Field studies have complemented laboratory research to provide a comprehensive understanding of responses of estuarine macrophytes to stress (Table 5.3.1). Detailed studies in the Olifants (Bornman et al., 2008) and Orange estuaries (Shaw et al., 2008, Bornman and Adams, 2010) provided an understanding of the responses of salt marsh to changes in freshwater inflow and extreme saline conditions. Overabstraction of freshwater has resulted in an increase in the duration and frequency of mouth closure in temporarily closed estuaries. The dynamics of macrophytes including salt marsh was investigated in the East Kleinemonde Estuary and showed that water level rather than salinity was most important in influencing macrophyte responses (Riddin and Adams, 2008). The majority of South Africa's estuaries are closed to the sea and research has contributed to an understanding of the structure and function of these systems globally (Whitfield et al., 2008; Whitfield et al., 2012).

Table 5.3.1. Published studies on responses of salt marshes to pressures in South African estuaries.

Pressure	Abiotic Change	Biotic Response	Reference
Restriction in tidal exchange	Freshening	Competition and loss of halophytes/salt marsh	O'Callaghan (1990)
Freshwater abstraction Mining and wind-blown dust	Salinisation, loss of flooding, drop in water table depth	Influence on dominant supratidal salt marsh species *Sarcocornia pillansii*	Bornman et al. (2002) Bornman et al. (2004)
Storm surge, Mouth closure and increase in water level	Increase in salinity & inundation	Increase in water level and loss of supratidal salt marsh	Riddin and Adams (2010, 2012)
Eutrophication	Increase in nutrients	Growth of macroalgae, shading and dieback of salt marsh	Nunes and Adams (2014) Human et al. (2016a)
Spread of Invasive alien plants	Increase in marsh elevation	Spread of *Spartina alterniflora* and loss of indigenous salt marsh, ~ 5 different species	Adams et al. (2012) Adams et al. (2016) Riddin et al. (2016)
Livestock browsing and trampling	Sediment compaction	Breaking of plants, patchy bare areas form	Hoppe-Speer et al. (2013)

Salt marshes are ephemeral landforms constantly being recycled, destroyed and reformed by coastal processes. Their geological life span is very short and estimated at only a few thousand years (Van de Koppel et al., 2005; Fagherazzi, 2013). Sediment core analysis using foraminifera and facies distribution in the Langebaan and Kariega estuaries provided information on sea level changes and marsh development during the Holocene period (Compton, 2001; Strachan et al., 2014). After 300 cal years BP relative sea level rise has remained stable and therefore salt marsh expansion has remained stable (Strachan et al.,

2014) which is similar to other southern hemisphere systems (Rogers et al., 2019). This review focusses on changes in salt marsh extent over the last ~90 years.

5.3.3 Importance and ecosystem services

This section describes Ecosystem Services (ES) and available information for South Africa whereas the next section relates salt marsh structure and function to the delivery of ES. In order to deliver ecosystem services, the salt marsh ecosystem requires functional links with the surrounding terrestrial, freshwater and marine ecosystems. For example, riparian vegetation buffers sediment input and seagrass habitats attenuate flows and prevent salt marsh erosion. Few South African studies have quantified or described salt marsh services; these are usually described for the entire estuary rather than just the salt marsh. For example, Turpie et al. (2017) reported that the value of subsistence harvesting of estuarine and coastal habitats was approximately R35.7 million per year (*circa* USD 2.4 million) and the nursery value of estuaries was estimated at close to R803 million per year (*circa* USD 54 million).

Salt marshes have some provisioning (provision of food and feed) and cultural services that include recreation, spiritual and historical experiences (Figure 5.3.3). Other services are bio-filtration, coastal protection, carbon storage and habitat functions which includes nursery and refuge functions. Although small in total extent South African salt marshes are important for biodiversity conservation as they serve as critical habitat for migratory fish and birds. Estuaries contain much of the only sheltered habitat along the highly exposed linear coastline (Beckley, 1984). Global comparisons also indicate that South African salt marshes are species rich as there are sharp transitions from fresh to saline and from lowland to upland areas or aquatic to terrestrial. At the land-estuary interface Veldkornet et al. (2015b) recorded over 95 different plant species many of which are halophytic in the supratidal salt marsh-terrestrial ecotone. Adam (1990) found only 45 species in saline areas in Britain whereas in the Georgia salt marshes (USA), Kunza and Pennings (2008) found 43 salt marsh species.

Provisioning	Supporting	Cultural	Regulating
Invertebrates – bait	Nutrient cycling	Aesthetic qualities	Flood control
Fishing – commercial & recreational	Nursery habitat	Spiritual value	Erosion control
Edible plants	Biodiversity maintenance	Recreation	Coastal protection
Grazing	Important Bird Areas	Tourism	Carbon storage

Figure 5.3.3. Ecosystem services associated with salt marshes in South Africa

In South African estuaries, there is direct extraction of mangroves for wood and reeds and sedges as a material resource. Although there is no direct use of salt marsh, there are certainly opportunities to use some of the succulent species as a food source (Mitra, 2018). Grazing takes place in the rural areas where livestock roam free. Waste remediation and nutrient cycling are the focus of a new study that will also investigate carbon storage and nutrient uptake as salt marsh ecosystem services. Where salt marsh has been removed the services of erosion control, coastal protection and flood attenuation are visible.

Bait organisms occurring in the intertidal and subtidal habitats adjacent to salt marsh are harvested in most estuaries. These include *Upogebia africana* (mud prawn), *Callichirus kraussi* (common sand prawn or pink prawn) and *Solen capensis* (pencil bait). Estuaries have been identified as important nursery habitats for juvenile marine fish species (Wallace et al., 1984; Beck et al., 2001) many of which are caught in coastal commercial and recreational fisheries (Lamberth et al., 2009; Whitfield and Pattrick, 2015; Edworthy and Strydom, 2016). Over 61 species of marine fishes use estuaries as nursery and/or foraging areas and of those 16 species are entirely dependent on estuaries as nursery areas (Wallace et al., 1984, Whitfield, 1994). Recently, Whitfield (2017) reviewed the role of estuarine macrophytes as nursery areas for fishes and showed that although juveniles make use of the salt marsh habitat during inundation there is a lower diversity of fish here compared to seagrass habitats.

Salt marshes serve as habitats for birds – breeding, roosting and feeding areas. Birds prey on prawns, marsh crabs, pencil bait & fish and include herons, gulls, waders, terns and cormorants. Salt marsh provide important high tide and night time roosting areas and secondary feeding habitat (Saintilan et al., 2018). In the Berg Estuary more than 8000 migrant waders such as Plovers and Curlews have been reported during summer. In the *Sarcocornia* intertidal salt marsh of the Langebaan Lagoon, up to 5 056 birds/km waders have been observed using the habitat as a food source at high tide (Summers and Kalejta-Summers, 1996). Little is known about the importance of the supratidal/high marsh but it is a home to a diversity of spiders, rodents and reptiles. Peringuey's Leaf-toed Gecko (*Cryptactities peringueyi*) is the only gecko in the world that lives in salt marshes and is known only from the Kromme Estuary and a few sites near Port Elizabeth, South Africa. As salt marshes disappear under bulldozers, so will this unique gecko (Van Niekerk and Turpie, 2012).

Carbon sequestered by salt marshes ecosystems has been termed "blue carbon" (Nellemann et al., 2009) and includes carbon stored in the sediments, in the living biomass both aboveground (leaves, stems, branches) and belowground (roots), as well as carbon stored in non-living biomass (leaf litter and dead wood) (Mcleod et al., 2011). The global loss of these coastal ecosystems not only reduces the capacity of natural carbon sinks, but degradation and disturbance of these habitats also directly releases large amounts of carbon back into the atmosphere in the form of CO_2 emissions (Pendleton et al., 2012; Siikamäki et al., 2012). Blue carbon quantification studies are new in South Africa with one detailed in situ study completed at Nxaxo Estuary (Johnson et al., 2020).

In terms of recreation and tourism, scenic views and vistas are created by salt marsh and this is desirable for coastal properties. For example, the Thesen's Island properties (Turpie et al., 2017) at Knysna Estuary are marketed on their salt marsh views (http://www.thesenislandsliving.co.za/*).*

5.3.4 Study Approach

This study reviewed and synthesized the existing scientific information to provide an up-to-date understanding of the patterns and processes influencing salt marshes so that an accurate prediction can be made of their response to climate change. Firstly, available information on distributional patterns and importance in terms of ecosystem services is addressed. A present status assessment of salt marsh is then made and changes in area cover described in relation to threats and pressures. Thereafter, expected responses to climate change and implications for ecosystem services are summarised using a comparative approach with information from other better studied areas, as there is limited information on local responses. Management responses in terms of existing protected areas are assessed and knowledge gaps and future research highlighted. A framework for understanding climate change responses in open and closed estuaries is provided.

The Millennium Ecosystem Assessment (2005) classified ecosystem services into four broad categories: supporting, provisioning, regulating and cultural services that are used in this study. It is recognised that this provides inadequate differentiation between actual ecosystem services, ecosystem functions and ecosystem benefits (Wallace, 2007). However, this classification was used in this study as it is globally applied in wetland research and therefore recognizable to all.

A present-state assessment of salt marsh covering the South African coastline was conducted by measuring change in area cover in relation to prevailing and future threats and pressures. The study area represents the estuaries from west (Orange River mouth) to east (Kosi Bay) (Figure 5.3.1). Adams et al. (2016) provides detail on the study area. Visible changes in salt marsh area cover were identified from aerial photographs and from Google Earth and mapped. The changes were associated with development, roads, housing, grassed areas, grazing and agriculture. This represented direct and indirect anthropogenic changes. The type of habitat lost (e.g. salt marsh, mangroves, and reeds/sedges) was determined by comparing historical aerial photographs and literature as well as physical features such as surrounding habitat, elevation, biogeographic zone and estuary type. Only salt marsh habitat destroyed to make way for development or related human activities was included. Habitats indirectly impacted through, for example, changes in freshwater inflow or water quality were excluded.

Salt marsh habitat was digitized for 115 estuaries using ESRI™ ArcMap 10.1 (2012) from orthorectified aerial photographs obtained from the Chief Directorate: National Geo-spatial Information (CD:NGI). These images have a 50 cm spatial resolution. The earliest images dated back to 1934/1937. Past salt marsh area cover thus represents the situation in the 1930s and present in 2018 (Tables 5.3.2 and 5.3.3). Salt marsh cover in each image was mapped and the difference in area between images taken as habitat loss or gain. Recent satellite imagery (Google Earth) and field work were used to update present salt marsh cover following a similar approach to that of Fernandes and Adams (2016). Arcpad 10.1 loaded on Trimble Juno GPS was used to map the distribution of salt marsh in the field. GIS vegetation maps are now available for approximately 40% of the estuaries in the country.

Table 5.3.2. Intertidal salt marsh area (ha), habitat trend (D: decreasing, I: increasing, S: stable), pressures and protection status. Shaded rows indicate those estuaries where the greatest loss has occurred.

Estuary	Present area 2018	Past area (% lost) 1930s	Habitat trend	Pressures	Protection status
Orange	144	154 (7%)	D	Salinisation	Ramsar site
Olifants	97	97	S	Salinisation	None
Berg	1182	1573 (23%)	D	Agriculture	Partial, CapeNature
Langebaan	792	792	S	Grazing pressure removed	South African National Parks (SANParks)
Knysna	552	794 (30.5%)	D	Development	Partial SANParks
Swartkops	209	215 (3%)	D	Development and industry	None
Kowie	35	83 (58%)	D	Development	None
Great Fish	133	144 (8%)	D	Disturbance	None
Kosi	58	58	S	Grazing, trampling, fires	iSimangaliso Wetland Park, World Heritage site

Trends in salt marsh habitat were classified as stable (S), decreasing (D) or increasing (I). Different estuaries are presented in Tables 5.3.2 and 5.3.3 to indicate those systems with the largest salt marsh or where the greatest loss in area has occurred. To determine the proportion of salt marsh under legal protection, relevant data were extracted from the estuary component of South Africa's 2012 National Biodiversity Report (Van Niekerk and Turpie, 2012). This was however an over-estimation as large areas occur in provincial reserves where there is little active protection.

Table 5.3.3. Supratidal salt marsh area (ha), habitat trend (D: decreasing, I: increasing, S: stable), pressures and protection status. Shaded rows indicate those estuaries where the greatest loss has occurred.

Estuary	Present area 2018	Past area (% lost) 1930s	Habitat trend	Pressures	Protection status
Orange	627	1319 (52%)	D	Salinisation	Ramsar site
Olifants	910	1529 (40.5%)	D	Salinisation	None
Berg	3226	4589 (30%)	D	Agriculture	Partial – CapeNature
Langebaan	557	524 (33% increase)	I	Grazing pressure removed	South African National Parks
Gouritz	123	662 (81.4%)	D	Agriculture	None
Knysna	133	375 (64.5%)	S	Development	Partial SANParks
Gamtoos	81	711 (89%)	D	Agriculture	None
Swartkops	338	1013 (67%)	D	Development and industry	None
Kosi	229	229	S	Grazing, trampling, fires	iSimangaliso Wetland Park, World Heritage site

5.3.5 Present Status: Area Cover and Loss of Salt Marsh Habitat

Total salt marsh area is 14955 ha of which 10169 is supratidal salt marsh and 4786 is intertidal salt marsh. Nine estuaries in the country support greater than 100 ha of intertidal salt marsh and 17 estuaries over 100 ha of supratidal salt marsh. The largest intertidal salt marsh area occurs in the Groot Berg Estuary followed by Langebaan and Knysna estuaries (Table 5.3.2). The Groot Berg Estuary also has the largest supratidal salt marsh area followed by the Olifants, Heuningnes and Orange estuaries (Table 5.3.3). Approximately 27% of salt marsh habitat has been lost due to encroaching development and agriculture. The greatest loss of macrophyte habitats in the country has been of supratidal salt marsh (4881 ha, Table 5.3.3) as it forms the ecotone between salt marsh and terrestrial vegetation and therefore is the most likely habitat to be developed.

Intertidal salt marsh mostly occurs in permanently open estuaries and only 13% (37 of 289 estuaries between the Orange River and Kosi Bay) of estuaries are permanently open. Loss of intertidal salt marsh habitat is usually due to development such as causeways, bridges or encroaching housing and business developments. Intertidal salt marsh has been lost from Knysna (242 ha), Kowie (48 ha) and Great Fish estuaries (11 ha). Development such as the town of Knysna, Thesen's Island, Leisure Isle, the N2 road bridge and embankments has removed large areas of salt marsh. The Port Alfred Marina, the town of Port Alfred and houses along the banks with jetties has removed intertidal salt marsh habitat from the Kowie Estuary. Footpaths and the caravan park in the lower reaches has disturbed habitat in the Great Fish Estuary. These estuaries can also be identified as those vulnerable to coastal squeeze; due to surrounding development there will be little chance of inland migration of salt marsh.

Agricultural impacts are largely responsible for the loss of supratidal salt marsh. In the floodplains of the Gouritz and Gamtoos estuaries, between 80% and 90% of habitat has been lost to vegetable cultivation and cattle grazing. Salinization has also caused large habitat losses in the Orange River and Olifants estuaries (Table 5.3.4). The Orange River Estuary, lying at the boundary between South African and Namibia, is a Ramsar wetland of international importance that was placed on the Montreux Record in 1995 because 300 ha of salt marsh had become desertified (Shaw et al., 2008). This loss was attributed to factors such as leakage of diamond mine water, the impact of windblown dried slimes (waste) dam sediment on marsh vegetation, construction of flood protection works and the elimination of tidal exchange into the wetland by a causeway constructed at the river mouth (Shaw et al., 2008). Due to low rainfall on the west coast and the highly salinized nature of the desertified marsh area, there has been little change in the salt marsh status of this estuary over the past 10 years (Bornman and Adams, 2010).

Table 5.3.4. Area cover of estuary habitats in South Africa, changes in area and percentage area occurring in protected areas.

Habitat	Past (ha)	Present (ha)	Area (ha) lost & % change	Protected area (ha) & % protected
Intertidal salt marsh	5 354	4 786	568 (10.6% loss)	1 221 (25.5%)
Supratidal salt marsh	15 050	10 169	4 881 (32.4% loss)	1 744 (17.1%)
Submerged macrophytes	2 515	2 695	180 (7.2% gain)	1 416 (52.5%)
Mangroves	1576	1 664	88 (5.6% gain)	317 (19%)

Most of this coastal development took place between the years 1950s to 1990s. Since then stricter implementation of legislation has limited removal of salt marsh. Significant loss of salt marsh in Europe and North America took place between the 1930s and the 1970s; thereafter international conventions such as Ramsar and vigorous protection laws provided some protection (Adam, 1990; Gedan et al., 2009). There is a clear evolution from a development narrative of exploitation and transformation in the industrial period (c. 1800-1980) to a crisis narrative of protection and risk in the post-industrial period (c. 1980-present) (Hatvany, 2008). This is true for South Africa too but with a delayed response because of our developing status. Further analysis is needed but preliminary data indicate that the loss of salt marsh has declined as the initial removal was related to development of coastal towns, encroachment by development and agricultural expansion from the 1950s to the 1970s. In some areas such as Langebaan Lagoon there has been an increase in salt marsh due to removal of farm animals and grazing; and expansion of the West Coast National Park (Table 5.3.3).

Other stressors that have been described (Table 5.3.1) but not quantified in terms of loss of salt marsh area include salinization and desiccation due to upstream freshwater abstraction (Bornman et al., 2002, 2004). Reduced freshwater inflow causes extended mouth closure of temporarily open/closed estuaries, inundation and flooding of salt marsh (Riddin and Adams, 2010, 2012). In urbanized estuaries, salt marsh loss is related to a restriction of tidal exchange, freshening and invasion by alien invasive plants (O'Callaghan, 1990). Eutrophication, macroalgal blooms and smothering of salt marsh is a growing concern in South African estuaries (Nunes and Adams, 2014; Human et al., 2016b). In the rural areas, livestock browsing and trampling of the salt marsh is extensive but largely unquantified. There are some success stories; for example, early detection of the invasive grass *Spartina alterniflora* Loisel in the Groot Brak Estuary resulted in successful eradication (Table 5.3.1).

Table 5.3.2-4 indicate that there is some protection for estuaries with large salt marsh areas in South Africa. Approximately 25.5% of the total intertidal salt marsh area occurs in protected areas and 17.1% of the supratidal salt marsh (Table 5.3.3). In addition, estuary management plans are a requirement of the National Environmental Management: Integrated Coastal Management Act (Act 24 of 2008). These plans can be effective in protecting sensitive habitats, e.g. salt marsh through zonation of destructive activities such as boating that leads to erosion. There is a need for formal protection status for the Groot Berg Estuary. The estuary is currently designated as an IBA (Important Bird Area) where the water and intertidal habitat is managed by CapeNature and the local municipality. Restoration of the salt marsh at Orange River mouth is also needed as well as greater protection for the large intertidal salt marshes of Knysna Estuary. In the South African National Estuary Biodiversity Plan (Turpie et al., 2012) habitat targets were set as 20% of the total area of each estuarine habitat type but this has not been implemented or addressed in any way.

5.3.6 Future climate change responses and implications for ecosystem services

Climate change conditions having a particular influence on salt marshes are sea level rise, increase in sea storms and wave height, changes in river discharge (droughts/floods), increased CO_2 levels and higher temperatures. Research has shown how this alters the key abiotic stressors in salt marshes – changing inundation patterns, salinity gradients and sediment biogeochemistry (e.g. organic matter supply). There is some research to indicate how salt marshes may respond to climate change. In South African estuaries responses in estuaries permanently open to the sea will be different compared to temporarily closed estuaries (Tables 5.3.5 and 5.3.6).

Table 5.3.5. Climate change responses of salt marsh and implications for ecosystem services in permanently open estuaries.

Abiotic Change	Ecological Processes	Ecosystem Services
Sea level rise +1.5-2.7 $mm.yr^{-1}$	Salt marsh subsidence	Change in biodiversity provision.
Inundation & waterlogging	Dieback and salt marsh loss	Nutrient cycling affected.
Change in sediment biogeochemistry	Changes in species composition.	
↑Sea storms & wave height	Loss of salt marsh	Loss of bank stabilization, possible flooding of surrounding properties and loss of economic value.
Erosion		
↑Floods		
↑ Nutrient inputs & eutrophication	Macroalgal growth, smothering of salt marsh	Loss of waste assimilative capacity. Negative effect on human health and wellbeing.
↑Sediment input	Salt marsh accretion	Reduced recreation and tourism value. Decreased value of surrounding real estate.
↑Droughts		
↑Salinity	Change in species and community composition.	Aesthetic qualities reduced.
	Decrease in productivity. Loss of salt marsh cover	Loss of nursery habitat for fish.
↑CO_2		
Higher C availability	Increase in plant growth & productivity	Change in biodiversity provision.
	Shift from C3 to C4 plants	Increase in weedy, invasive species.
↑Temperature		
Warming Higher aridity	Increase in plant growth & productivity.	Change in carbon storage and biodiversity provision. Loss of habitat for threatened species.
	Mangroves replace salt marsh.	Mangrove expansion and change in habitat for other biota.
	Distributional range shifts and change in habitat diversity.	
	Increase in invasive species.	
	Change in salt marsh phenology. Extinctions.	

Table 5.3.6. Climate change responses of salt marsh and implications for ecosystem services in predominantly closed estuaries.

Abiotic Change	Ecological Processes	Ecosystem Services
↑**Sea level rise** ↑Open mouth condition	Expansion of salt marsh in intertidal.	Change in biodiversity provision. Potential increase in carbon storage.
↑**Sea storms & wave height** ↑Sediment deposition, constricted mouth	Increase in water level, flooding and dieback of salt marsh.	Possible erosion and loss of bank stabilization, possible flooding of surrounding properties and loss of economic value.
↑**Floods** Scouring, decrease in salinity	Change in species composition. Loss of salt marsh cover.	Change in biodiversity and habitat provision. Loss of marsh nutrient processing.
↑**Droughts** ↑Closed mouth condition	Increase in water level, flooding and dieback of salt marsh. Loss of intertidal habitat. Loss of marine connectivity, fish and invertebrate recruitment.	Reduced habitat for wading birds impacting bird and wildlife viewing. Loss of tourist appeal. Reduced fisheries due to loss of nursery function. Bank destabilization and erosion
↑**CO_2** ↑**Temperature** Warming Higher aridity	Increase in plant growth & productivity	Possible salt marsh expansion and loss of other habitats such as open water surface area. Reduced ecotourism.

5.3.6.1 Sea-Level Rise

Present South African sea level rise rates fall within the range of global trends and are approximately: west coast +1.9 $mm.yr^{-1}$, south coast +1.5 $mm.yr^{-1}$, and east coast +2.7 $mm.yr^{-1}$ (Mather et al., 2009; Mather and Stretch, 2012). Sea-level rise will increase inundation and waterlogging altering sediment biogeochemistry, moisture and salinity (Table 5.3.5). This is the predicted scenario, however, if the salt marshes build elevation at a sufficient rate then inundation and waterlogging may not increase (Rogers et al., 2019). Plant ecophysiology studies have informed our future predictions and shown that lower intertidal salt marsh species will be able to survive conditions typical of upper intertidal ranges, however the reverse is not true of upper intertidal species that are sensitive to waterlogging (Tabot and Adams, 2013). Depending on the plant's tolerance to flooding dieback will result; for example, lower intertidal *Sarcocornia* spp. can die after 3 months of complete submergence (Adams and Bate, 1994). Studies completed on other local species can also inform these predictions (e.g. Naidoo and Mundree, 1993; Naidoo and Kift, 2003). If there is available land, and the elevation is suitable, then salt marsh will migrate inland (Tabot and Adams, 2013; Veldkornet et al., 2015a). In some cases, hard structures may need to be removed to allow upland salt marsh migration. The majority of South African estuaries occupy drowned river valleys that offer limited upland area into which to migrate and those estuaries that do have low-lying adjacent habitat have mostly been developed, creating a physical barrier to potential migration.

In temporarily closed estuaries, the abiotic conditions are controlled by the condition of the mouth; whether it is open or closed to the sea (Table 5.3.6). Sea level rise will result in more open conditions through an increase in the tidal prism particularly if the mouth of the estuary is sheltered from wave action and little sediment is available (Van Niekerk, 2018). However, drought and a reduction in freshwater inflow will result in mouth closure, flooding and die-back of salt marsh plants (Table 5.3.6). There will be temporal and spatial variability associated with these processes.

5.3.6.2 Sea Storms

The Fourth Assessment Report of the IPCC predicts an increase in the frequency and intensity of coastal storms and high-water events in the 21st Century (IPCC, 2007). South Africa is a wave-dominated coast sensitive to increased sea storminess that can result in erosion or sediment deposition and accretion (Mather and Stretch, 2012). Sea storms and high waves can deposit sediment and close an estuary mouth; but storms can also increase the tidal amplitude eroding the mouth area and increase the duration of open mouth conditions (Van Niekerk, 2018). However, an increase in storminess will mostly lead to steeper beach slopes, more constricted mouths, a smaller tidal amplitude and thus less intertidal area for salt marsh growth.

Increased storminess could increase erosion of salt marshes although there is little evidence of this yet in South Africa. Marshes can keep pace with sea level rise if there is available sediment and land for expansion inland. Bornman et al. (2016) showed that in the Swartkops Estuary salt marsh surface elevation was keeping pace with historic sea level rise in the estuary. Subsidence/accretion of the marsh was measured using the Rod Surface Elevation

Table (RSET) method. RSET changes are also being monitored in the Knysna and Kromme estuaries as well as in the salt marsh/mangrove transition zones of the Nahoon and Nxaxo estuaries. In Australia, RSET studies have shown that there is a lower rate of vertical elevation gain in salt marsh compared to mangrove relative to sea level rise (Rogers et al., 2005, 2013, 2014). This accretion difference results in the consistent trend of mangrove encroachment and replacement of salt marsh in the south of the country (Saintilan et al., 2018).

5.3.6.3 Floods

To understand the response of salt marsh to climate change it is important to consider extreme events (Morzaria-Luna et al., 2014). According to Zedler (2009) coastal wetlands could suffer catastrophic effects from sequential extreme events (e.g. floods) when the systems do not have time to recover before the next extreme event. Climate extremes are likely to be through sedimentary processes (Boorman, 2003). In the nearby Limpopo River Estuary, Mozambique, extensive river flooding in 2000 halved the original mangrove area enabling salt marsh colonization in the bare areas. This habitat now consists of extensive grassy *Sporobolus virginicus* salt marshes (Bandeira and Balidy, 2016).

An increase in extreme rainfall events is projected to occur along the southern and eastern coasts of South Africa during spring and summer (Engelbrecht et al., 2013). An increase in runoff also affects the nutrient load entering estuaries, with inflow being an important source of dissolved and particulate nutrients. As such, increased run-off (and nutrient input) from disturbed catchments may result in eutrophication (James et al., 2013). Estuarine ecosystems are increasingly no longer able to assimilate nutrient loads resulting in eutrophication (Lemley et al., 2015, 2017). In temporarily closed estuaries (Table 5.3.6) flooding will open the estuary mouth and scour sediments deposited during periods of low flow. This will influence salinity, sediment supply and a number of other abiotic conditions. Salt marsh is likely to be reduced as the systems become fresher.

In South Africa episodic flood events were studied at the Orange and East Kleinemonde estuaries. The Orange River Estuary, an important Ramsar site, has a large desertified salt marsh area due to restriction of tidal and flood waters by a causeway. The sediment and groundwater are hypersaline and despite predictions that a flood would dilute salts and promote salt marsh germination this did not happen as the causeway prevented the floods from reaching the degraded salt marsh area to reduce salinity (Bornman and Adams, 2010). In the East Kleinemonde Estuary, a sea storm surge increased salinity and inundation resulting in die-back of reeds, sedges and supratidal salt marsh (Riddin and Adams, 2010) providing insight on potential future responses to climate change. Salt marshes have shown to be resilient; massive germination from a large seedbank occurred at the East Kleinemonde Estuary when conditions were favourable (Riddin and Adams, 2009). At the Groot Brak Estuary the indigenous salt marsh returned following removal of the invasive grass *Spartina alterniflora* (Adams et al., 2016). *In situ* long-term monitoring is needed to understand these dynamic processes.

5.3.6.4 Drought

Downscaled regional climate models project slightly drier conditions for the winter rainfall region of South Africa with an increase in inter-annual variability (Hewitson and Crane, 2006; Engelbrecht et al., 2009, 2013). This may result in a decrease in flows and in increase in flow variability (droughts) in estuaries along the west coast (James et al., 2013), with the west coast a 'hotspot' of hydrological change (Schulze et al., 2005). A decrease in freshwater inflow will increase salinity and decrease plant productivity in open estuaries (Table 5.3.5). In addition, an increase in the period between rainfall events, particularly along the west coast, could lead to reduced sediment moisture and higher salinity. Desertification has occurred at the Orange River mouth due to freshwater inflow reduction as well as the Berg Estuary. South Africa is a semi-arid country and salt accumulation is common in dry areas.

Where there is lower freshwater inflow salinity will move further upstream in open estuaries, resulting in a reduction in the extent of the river-estuary interface (REI) zone as well changing the location of the REI and moving the zone of reeds and sedges further upstream (Adams and Bate, 1999). Major reductions in river flow can result in the complete elimination of this zone (James et al., 2013). There will be an initial increase in detritus associate with this loss, and then in the long term a reduction as the reed/sedge habitats have high biomass and productivity. Increases in groundwater and sediment salinity can lead to extirpation of species because migration into less saline lower tidal zones is not possible in the event of a drought (Semeniuk, 2013; Wasson et al., 2013).

Reductions in the amount of freshwater entering temporarily closed estuaries will lead to an increase in the frequency and duration of closed mouth conditions (Table 5.3.6). Depending on the rainfall there is either a decrease or increase in water level in the estuary that will influence the salt marsh (Riddin and Adams, 2008, 2012). Loss of this vegetation in response to flooding can result in bank destabilisation and erosion.

5.3.6.5 Temperature

Estuaries will be affected by changes in both surface air and ocean temperatures. Global surface air temperatures have increased by about 0.8°C over the last century, in response to the enhanced greenhouse effect. However, recent climate trend analyses indicate that South Africa has been warming more than twice the global rate of temperature increase over the past five decades (Engelbrecht et al., 2015; Kruger and Nxumalo, 2016). An increase in temperature will increase plant growth and productivity. Mangrove expansion into salt marsh is occurring on multiple continents (Osland et al., 2014; Saintilan et al., 2018). This is known as tropicalization whereas desertification occurs where there is an expansion of hypersaline coastal wetland ecosystems common to arid and semi-arid climates (Osland et al., 2014, 2017).

A recent species distribution modelling study (Quisthoudt et al., 2013) showed that climate change would create climatically suitable sites for the expansion of mangroves, particularly *Avicennia marina* (Forsk.) Vierh and *Bruguiera gymnorrhiza* (L.) Lam, south of their current limits in South Africa. Expansion could be into salt marsh habitats, as has been occurring elsewhere (Saintilan and Williams, 2000; Stevens et al., 2006; Rogers et al., 2014a; Saintilan et al., 2014). However, at the Nahoon Estuary Hoppe-Speer et al. (2013) showed that expansion at planted mangroves was into bare sandflat areas rather than salt marsh habitats. These data plus that for all estuaries in South Africa are collated in a botanical database (Adams et al., 2016) to provide a baseline for future monitoring and research on these dynamic processes.

An increase in temperature may also increase the number of invasive plant species as well as insect abundance and feeding thus impacting salt marsh growth and reproduction. The influence of environmental cues on life cycle strategies and plant phenology is not well known. As temperature changes, the geographical distribution of species, depending on their tolerances or preferences, may contract or expand, leading to new and unpredictable species interactions (Murawski, 1993; Perry et al., 2005; Harley et al., 2006; USEPA, 2009).

5.3.6.6 CO_2 and pH

Concentrations of CO_2 in the atmosphere have increased exponentially (~ 40%) since the industrial revolution from 280 to 387 ppm, with 50% of this increase having occurred in the last 30 years (Feely et al., 2009). Increases in atmospheric CO_2 levels will stimulate above and below ground salt marsh productivity (Anderson et al., 2010; Morzaria-Luna et al., 2014). Higher rates of organic matter production may lead to sediment accretion and changes in elevation. These feedback loops need to be understood in order to predict the response of salt marsh to climate change.

The pH of surface open ocean waters may decrease by 0.3-0.4 units by 2100 under the influence of rising atmospheric CO_2 levels (Caldeira and Wickett, 2003). Changes in pH in coastal ecosystems may be caused by ocean acidification as well as a multitude of other (natural or anthropogenic) factors such as eutrophication, upwelling and freshwater inflow (Duarte et al., 2013a), which cause greater pH variability than in the open ocean (Strong et al., 2014; Cai et al., 2017). Upwelling can create hotspots of coastal pH change. This is because of naturally high levels of CO_2 combined with increased anthropogenic CO_2, as well as an increase in the intensity of upwelling in some regions (Strong et al., 2014). Thus, by the end of this century, acidification may become a dominant process in permanently open estuaries, especially on the west coast of South Africa where there is regular upwelling (Van Niekerk, 2018). Lower pH will affect all calcifying organisms as structures made of calcium carbonate dissolve requiring more metabolic energy for an organism to maintain the integrity of its exoskeleton (Azevedo et al., 2015). This may influence biotic controls and species interactions in salt marshes. Tidally inundated salt marshes play an important role in oxygenating the water column and buffering acidification by withdrawing excess CO_2 (Duarte et al., 2013c).

5.3.7 Changes in Ecosystem Services

To support sustainable ecosystem service provision, we need to know which species and communities drive ecosystem processes that underlie ecosystem services under particular abiotic conditions (Helfer and Zimmer, 2018). Functional relationships between ecosystem services and marsh characteristics have not yet been developed (Skov pers comm. SaltmarshNET 2018). Table 5.3.7 indicates those studies that have described or quantified changes in salt marsh ecosystem services in response to climate change. This was used to infer changes in the ecosystem services of South African salt marshes (Tables 5.3.5 and 5.3.6).

A decrease in the spatial extent of salt marshes will result in a change in ecosystem service provision. For example, the loss of salt marsh extent not only reduces the capacity to act as a natural carbon sinks, but degradation and disturbance of these habitats also directly releases large amounts of carbon back into the atmosphere in the form of CO_2 emissions (Pendleton et al., 2012; Siikamäki et al., 2012). Changes in species and community composition of the salt marsh will influence many different ecosystem services such as grazing and fishing. For example, the grass *Spartina* versus the succulent *Sarcocornia* species offer different food sources and structural protection for fish (Whitfield, 2017). Understanding the traits of different species rather than the species themselves will foster our understanding of the link between biodiversity and ecosystem processes and services (Helfer and Zimmer, 2018).

Table 5.3.7. Studies that have described or quantified changes in salt marsh ecosystem services in response to climate change. (Supporting = blue, Regulating = yellow, Provisioning = green, Cultural = pink).

Climate change parameter	Ecosystem Service (supporting)	References
Sea level rise, loss of intertidal habitat	Reduced habitat for wading birds impacting bird and wildlife viewing. Loss of habitat threatened bird species. Loss of nursery function.	Clausen and Clausen (2014); Guo et al. (2017); Rosencrantz et al. (2018); Boesch and Turner (1984)
Sea level rise, drowning of salt marsh	Reduced N uptake, buffering from eutrophication and coastal filtering function.	Nelson and Zavaleta (2012); Wasson et al. (2017).
Sea level rise	Change in carbon sequestration. Landscape SLAMM model used to show effect of sea level rise on carbon sequestration and denitrification. Restoration of tidal exchange and increase in carbon storage.	Craft et al. (2009); Theuerkauf et al. (2015) Kirwan and Mudd (2012) Macreadie et al. (2017b)
Change in sediment supply	Needed to maintain salt marsh habitat elevation.	Thorne et al. (2014); Carrasco (2019)
Increase in floods, nutrient input and eutrophication	Loss of marsh nutrient processing.	Deegan et al. (2012); Caçador et al. (2016), Wasson et al. (2017)
Increase in droughts	Loss of foundation species and biodiversity. Loss of species due to range distribution change	Angelini and Silliman (2012); McFarlin et al. (2015) Prahalad and Kirkpatrick (2019)
Increase in CO_2	Marsh growth, marsh accretion, reduction in erosion and improvement in coastal protection.	Langley et al. (2009) Ratliff et al. (2015)
Increase in storms and erosion	Loss of protective barrier function	Shepard et al. (2011); Siikamäki et al. (2012), Spencer et al. (2016); Leonardi et al. (2018).

Climate change parameter	Ecosystem Service (regulating)	References
Increase in temperature Reduction in freezing events	Mangrove expansion and change in ecosystem services such as carbon storage, habitat for waterbirds and threatened species, cultural services and values. Commercially important species influenced.	Santilan and Williams (1999); Morzaria-Luna et al. (2014); Kelleway et al. (2016b); Kelleway et al. (2017a); Smee et al. (2017).
Sea level rise	Loss of grazing, crops, saline agriculture.	Bless et al. (2018).
Reduced rainfall and freshwater input	Salinization and increase in unvegetated salt flats, loss of ecosystem services.	Osland et al. (2014), Osland et al. (2016).

Loss of salt marsh habitat and habitat diversity in response to sea level rise reduces the available area for wading birds reducing activities such as bird and wildlife viewing (Guo et al., 2017). This also influences regulating services such as the filtering function through reduced nitrogen uptake (Nelson and Zavaleta, 2012). Salinisation and an increase in unvegetated salt marsh in response to reduced rainfall and freshwater input results in a loss of ecosystem services (Osland et al., 2014; Osland et al., 2016). The dieback of foundation salt marsh species and a loss of biodiversity has been reported in response to drought (Angelini and Silliman, 2012; McFarlin et al., 2015). Warming will increase tidal wetland productivity and decomposition resulting in enhanced carbon storage and vertical accretion. However, the net outcome depends on the change in balance between productivity and decomposition. For example, if warming enhances decomposition more than it does to productivity, then we can expect a new loss in organic substrate, decline in carbon storage and surface elevation. However long-term temperature responses will be more complex due to species replacements and interactions with rates of sea-level rise (Kirwan and Megonigal, 2013). According to Langley et al. (2009) an increase in CO_2 will increase marsh growth, reducing erosion and improving coastal protection.

The provision of ecosystem services will change in response to distributional range shifts. As temperature changes, the geographical distribution of species, depending on their tolerances or preferences, may contract or expand, leading to new and unpredictable species interactions (Harley et al., 2006). Range shifts can bring about changes in ecosystem services; for example, in response to increasing temperature and sea level rise the replacement of salt marsh by mangroves leads to an increase in carbon storage but a loss of biodiversity (Kelleway et al., 2016b; Saintilan et al., 2018). The habitat available for waterbirds and threatened species changes as well as cultural services and values (Kelleway et al., 2017a). In South Africa there are a number of mangrove-associated invertebrates that have already shifted further than mangroves and colonised “surrogate” salt marsh and sedge habitat to the south. This includes the tropical fiddler crab *Austruca annulipes* and mangrove snail *Cerithidea decollata* in the Knysna Estuary, a new southernmost limit for both genera (Hodgson and Dickens, 2012; Peer et al., 2015; Whitfield et al., 2016).

In order to protect salt marsh and their services we need to understand the connectivity and exchanges with adjacent ecosystems as salt marshes are at the interface of marine, freshwater and terrestrial influences. For example, sea level rise and an increase in water level could distribute contaminated salt marsh material to other habitats (Duarte et al., 2014). South Africa has world-class environmental legislation; so, it is not this that fails to protect estuaries and their salt marshes but rather the implementation thereof. Legislation and policy protecting South African salt marshes and ecosystem services includes the National Water Act, Integrated Coastal Management Act and National Biodiversity Act. However, as highlighted for Australia (Rogers, 2016) there are limited policies and planning mechanisms to set aside buffers for landward migration under sea level rise. For migration of intertidal salt marsh, the important estuaries would be those where there is currently intertidal and supratidal marsh as intertidal salt marsh would migrate into the supratidal zone. This is likely to occur at the Groot Berg, Olifants and Langebaan estuaries on the west coast where there is well-developed salt marsh zonation along an elevation gradient. Along the south and east coasts, important systems would be Gouritz, Keurbooms, Swartkops, Bushmans and Kariega estuaries. The supratidal areas of Heuningnes, Great Fish and Gamtoos are disturbed by cattle, people and agriculture and the compacted sediment may initially inhibit intertidal salt marsh expansion. The expansion of supratidal areas landward will need to be monitored as this will be into mostly transformed areas.

5.3.8 Conclusion

The loss of salt marsh habitat has been quantified and estuaries with intact undisturbed habitat identified. This is to allow for landward migration in response to sea level rise. We have been able to map our salt marshes and provide a baseline for future monitoring and assessment of changes over time. This also provides a baseline for future research that can quantify changes in ecosystem services over longer time scales. Protecting salt marsh ecosystems requires recognizing the patterns, processes and expected responses to disturbance events.

Predicting the combined effects of multiple stressors associated with environmental instability (increase in storm surges, floods, droughts and reduced river flow) is critical to conserve salt marsh habitats. Research and monitoring to understand salt marsh responses is ongoing because the interface between the subtropical and warm temperate coastal regions of South Africa is expected to be significantly affected by changes predicted for the future (Quisthoudt et al., 2013; Whitfield et al., 2016). Climate change is an additional stress to that of human pressures and in this study, the latter was quantified as the loss of salt marsh habitat due to development or agriculture. More subtle changes are occurring in response to climate change such as drying out of salt marshes and salinisation. Long-term monitoring of permanent plots and transects are needed to identify these changes. The traditionally strong focus on salt marsh adaptability in the vertical dimension should be complemented by a better understanding of the processes that control the lateral position of marsh boundaries (Kirwan et al., 2016b). In South Africa, field studies need to monitor changes in both expansion and elevation of different zones, and changes in ecotone habitats. It is critically important to know whether these habitats are eroding or accreting because for their survival salt marshes need to accrete at a rate that allows them to keep abreast of sea level rise. RSET measurements have started in some South African estuaries (Bornman et al., 2016).

6 THE POTENTIAL FOR A CARBON OFFSET MECHANISM FOR SOUTH AFRICA

6.1 Introduction to Carbon Finance and Trading

Carbon markets are founded on the concept of producing carbon credits through ensuring avoidance of greenhouse gas (GHG) emissions or atmospheric removal of these gases through the action of a project. These carbon credits can be resold or used to offset carbon dioxide (CO_2) emissions (Wylie et al., 2016). Acceptable offsets must show "additionality'' which is demonstrating that reduced carbon emissions (that can either be through protection or restoration of ecosystems) would not be possible without the funding made through the selling of carbon credits. Additionality can be shown through national accounting, where a baseline of carbon emissions is established for a country (based on historical trends) and reductions higher than this baseline qualify as offsets, which can be then be sold as carbon credits (Ullman et al., 2013). A carbon credit is the equivalent of one metric tonne of carbon dioxide (tCO_2eq) that is sequestered (or not emitted), which can be traded (bought and sold at a market-determined price) through numerous carbon finance markets (Thompson et al., 2014). Carbon markets can generally be divided into two broad categories, compliance/regulatory and voluntary markets which will be discussed in the section below.

6.1.1 Compliance or Regulatory Markets

Compliance markets are regulated cap-and-trade schemes which were established by the Kyoto Protocol and the European Union Emissions Trading Scheme with the aim of controlling GHG emissions through providing monetary incentives for emissions reductions. These usually involve an authority who sets a limit (cap) on the amount of gases that can be released. The cap is allocated or sold to specific entities as carbon credits which represent the right to emit a specific volume of gas. The entities that are emitting the gas are required to have credits that are equal to their emissions and the total amount of credits held cannot exceed the cap that is set by the authority. Therefore, entities that buy credits are essentially paying to emit GHG, while those that sell credits are being rewarded for their emissions reductions (Ullman et al., 2013). The South African carbon tax, which has been in effect from June 2019 is another regulatory mechanism that allows entities to offset a specified amount of their carbon tax liability and this will be discussed in Section 6.1.3.

Clean Development Mechanism and REDD+ Scheme

The Clean Development Mechanism (CDM) and the Reducing Emissions from Deforestation and Forest Degradation (REDD+) schemes are the most common voluntary market mechanisms. The CDM was established in 2012 through the Kyoto Protocol and is a global treaty that requires countries to reduce GHG emissions through trading (Wylie et al., 2016). The CDM was developed to facilitate compliance with emissions reduction objectives, firstly, through decreasing mitigation cost and secondly, through promoting development and sustainability in emerging countries (Corbera et al., 2009). The carbon credits that are traded in the CDM are referred to as Certified Emissions Reductions (CERs) which are used by developing countries to either meet their own Kyoto targets or for trading in carbon markets (Parnphumeesup and Kerr, 2015). Examples of projects under the CDM include the Cameroon CDM Mangrove Project which was established to contribute towards the sustainable use of mangroves for the preparation of fish which is commonly known as fish smoking (UNFCCC, 2010a; Herr et al., 2017). Typically, a traditional fish smoking oven consumes 1205 kgs of red mangrove wood as it takes 53 hours to prepare the fish, while a modern cinderblock oven consumes 122 kgs of mangrove wood and only takes 5 hours (Dongmo Keumo Jiazet, 2019). The CDM project aimed to promote the use of cinderblock ovens in 350 traditional smoke houses across nine villages which will significantly improve mangrove conservation efforts in this region (UNFCCC, 2010; Herr et al., 2017).Though, Vanderklift et al. (2019) reported that only the mangrove afforestation project in Indonesia has obtained CDM certification thus information on blue carbon projects under the CDM remains uncertain.

Under the CDM, sequestration projects falling under Agriculture, Forestry and Other Land Uses (AFOLU) and mangrove restoration (the only extant blue carbon CDM methodology) provide temporary CERS (tCERs, or lCERs for longer term projects) which are only valid for a specified duration. Once the period of validity has expired, these credits must be replaced by the liable entity by purchasing additional permanent CERs, making such projects less attractive on the CDM market (UNEP and CIFOR, 2014) The CDM as a mechanism formally expires in 2020 with the inception of the Paris Agreement, but currently countries are under negotiation to establish how the CDM will align with Article 6 of the Paris Agreement, which allows for countries to exchange “internationally traded mitigation obligations” (ITMOs) through a non-market based approach that includes both public and private entities (Herr et al., 2017; Vanderklift et al., 2019). These ITMOs have been referred to as the new carbon commodity (Herr et al., 2017; Sharma, 2016). This new mechanism is required to go beyond offsetting and rather demonstrate positive net mitigation that is non-reliant on the transfer and trade of units (Herr et al., 2017).

The Reducing Emissions from Deforestation and Forest Degradation scheme is a policy that was established to reduce forest destruction and degradation while decreasing associated carbon emissions in developing countries (Ullman et al., 2013). REDD+ works similarly to CDM but concentrates more on projects related to reducing emissions from land transformation (Wylie et al., 2016).

Compliance markets have been criticized extensively over the years, especially the CDM. Firstly, for its very high administration costs, secondly, for its stringent requirements and lastly, for the issuing of temporary carbon credits that are sometimes difficult to trade (Joosten et al., 2016). Over time the CDM has become more suitable for lengthy large-scale projects which focus on improving energy efficiency and minimising waste management rather than afforestation and reforestation (Vanderklift et al., 2019). Also, for a project to obtain CDM approval, it costs an estimated US$50 000 to US$250 000. Due to this high administrative cost, only one blue carbon project on mangrove afforestation in Indonesia has received certification from the CDM (Vanderklift et al., 2018). Likewise, transactions via the CDM are very expensive and validation is obtained after long waiting periods, which often creates a barrier for the success of small-scale projects. Although there have been attempts to simplify the procedures and reduce the project validation times, the costs for smaller projects are still on the rise because investors view smaller projects as highly risky with major uncertainty (Boyd et al., 2007; Benessaiah, 2012).

The REDD+ scheme is also quite a slow process and has been majorly critiqued in the blue carbon literature. Firstly, for only including mangroves (specifically in countries where mangroves are defined as forests) (Vanderkflit et al., 2018) and secondly, for excluding other habitats (e.g. salt marshes and seagrasses). The scheme also disregards belowground carbon (from the roots and soil) which has been recognized as the dominant carbon pool compared to aboveground biomass (Herr et al., 2017). As evidence of the difficulty of REDD+, there are few examples of mangrove projects under this framework as not many countries have included mangroves in their baseline inventories and verification systems (Stringer et al., 2014). So far, the Blue Forests initiative (under Blue Ventures) in Madagascar has made significant strides towards integrating the protection and restoration of mangroves in the country's REDD+ strategies (Wylie et al., 2016). In Mozambique, the Zambezi River Delta mangrove carbon project is a pilot baseline assessment for the implementation of mangroves in the Mozambique REDD+ National Program and there is no record of its implementation yet (Stringer et al., 2014). Therefore, Wylie et al. (2016) suggested that alternative standards to CDM and REDD+ should be used until the process becomes more streamlined especially in its inclusion of smaller community projects which are more common in Africa.

South Africa (like many other countries in Southern Africa) has been excluded from the REDD+ scheme, mainly because South Africa has a small forest area, low national deforestation rates and complicated land and environmental issues, which collectively hinder the implementation of REDD+. Nevertheless, studies have suggested that REDD+ should still be part of the national climate change conversations in future (Rahlao et al., 2012). Especially with the application of the new REDD+ extension framework that will include salt marsh and seagrass habitats which are more extensive than mangroves in South Africa.

6.1.2 Voluntary Markets

Voluntary carbon markets (VCMs) were created for individuals, companies or governments who want to voluntarily offset their own GHG emissions (Ullman et al., 2013). Buyers who purchase offsets through voluntary markets generally do so to contribute towards corporate social responsibility (CSR). These markets allow entities who are not regulated by the Kyoto Protocol to reduce their carbon emissions to still offset their emission using various certified methods (Vanderklift et al., 2019). Voluntary carbon markets are significantly smaller than regulatory schemes as they make up only 0.1% of the global carbon markets, but their contribution to the overall carbon markets is not negligible (Peters-Stanley et al., 2011; Benessaiah, 2012). Voluntary markets are good for financing pilot case studies, small scale and community-based projects (Wylie et al., 2016; Vanderklift et al., 2019). Methodologies or verification/certification standards that are used in voluntary markets include the Verified Carbon Standard (VCS), the Gold Standard (GS), Plan Vivo, Climate, Community and Biodiversity Standard (CCBS) and Social Carbon.

Plan Vivo

Plan Vivo is a voluntary registry specifically for land use projects that work closely with rural communities who rely on natural resources (,). The main aim of the Plan Vivo foundation is to relieve poverty in developing countries by working closely with these rural communities in sustainable land use projects (Herr et al., 2016; UNEP, 2016). The activities that are allowed under Plan Vivo include afforestation, forest conservation, restoration and avoided deforestation (He, 2016) The Mikoko Pamoja mangrove restoration and reforestation project in Kenya is an example of a community-based scheme that is run by the Gazi Bay locals. The project is financed by voluntary carbon credits that are managed by Plan Vivo under Payments for Ecosystem Services (PES) (Wylie et al., 2016).

Verified Carbon Standard

The Verified Carbon Standard (VCS) is a programme of the Verra standard which issues verified carbon units to projects that focuses primarily on GHG reductions, without requiring additional environmental or social benefits other than key social safeguards. Often projects that are under VCS add other certifications such as the CCBS (also a Verra programme) and social carbon to obtain good prices in the market. The VCS has implemented more than 10 methods for wetland ecosystems. Two of these are relevant to blue carbon projects, since they include mainly mangroves, salt marshes and seagrasses under their methods (He, 2016). The India Sundarbans Mangrove Restoration project is part of VCS under the AFOLU scope for Afforestation, Reforestation and Restoration and soil carbon is included under the Wetlands Restoration and Conservation (WRC) (Wylie et al., 2016). Globally, VCS accounts for the bulk of voluntary carbon market transactions, with 58% of all certification in 2016 (Hamrick and Gallant, 2017).

Gold Standard

The Gold Standard (GS) issues Voluntary Emission Reductions and Emission Reductions Units and to obtain certification by GS, the project must have other environmental, social and economic benefits and are required to be sustainable. The GS does however provide sustainability guidelines to assist project developers to uphold their sustainability requirements. The GS also has guidelines on mangrove afforestation and reforestation but to date no mangrove blue carbon project has obtained carbon credits from the GS since the method started in 2013 (He, 2016).

Many studies have assessed the role of PES on poor communities and the relationship between carbon sequestration and reducing poverty. The expectation is that carbon is sequestered, the blue carbon habitat is conserved, poverty is reduced, and everybody benefits (Muradian et al., 2010; Benessaiah, 2012). PES, in theory are expected to grow human and financial capital, include community training, increase revenue by creating employment opportunities in addition to providing other socio-ecological benefits and services (i.e. ecotourism or coastal protection) (Pagiola et al., 2005, Wunder, 2008; Benessaiah, 2012). According to Hamrick and Gallant (2017) Africa has one of the lowest Gross Domestic Products (GDPs), fast growing population rates but limited resources and infrastructure to develop climate change mitigation strategies. In Africa thus far for terrestrial systems, the GS, VCS and CCB Standards accounted for 96% of the total African offset transactions. Out of the three, GS was the most used in the African carbon markets (~98%) outweighing the VCS and the CCBS. Presently, there are no blue carbon projects under GS (as mentioned above) along with CCBS in Africa.

6.1.3 Towards Blue Carbon Trading in South African Estuaries

The South African Context

South Africa is one of the top 20 most carbon intensive countries in the world (currently ranked number 13) (UNFCCC, 2011; Klausbruckner et al., 2016) because of a high dependence on industrial activities that rely on the burning of coal, crude oil and natural gas (Arndt et al., 2013). South Africa is also the largest CO_2 emitter in Africa, (UNFCCC, 2011; Klausbruckner et al., 2016) and is rated number 27 in the Global Climate Risk Index (Kreft et al., 2014; Klausbruckner et al., 2016).

In 2016 South Africa signed the UNFCCC Paris Agreement which is a voluntary agreement aimed at limiting GHG emissions which will begin in 2020 (Amusan and Olutola, 2016). South Africa was also the first country in Africa to pass a carbon tax. The Carbon Tax Bill was passed in February 2019 and came into effect on the 1st of June. Carbon taxes were implemented as part of South Africa's Paris Agreement pledge or Nationally Determined Contributions (NDC) which where to obtain GHG emission reductions of 34% below a business-as-usual trajectory by 2020, and 42% by 2025. Also as part of the objectives of the NDC, the country has committed to peak GHG emissions between 2020 and 2025 (increasing from approximately 583 Mt CO_2 eq. to 614 Mt CO_2 eq) (DEA, 2011, 2014; Klausbruckner et al., 2016), plateau emissions between 2026 and 2035 and decline these from 2036 going forward. This is referred to as the peak, plateau and decline emissions trajectory range of the NDC mitigation component. Next year parties can submit new NDCs to the UNFCCC (Parr et al., 2018).

Carbon tax is defined as putting a price on carbon emissions for companies that emit GHG. The more a company emits, the higher the payable tax and this is useful because it shifts the otherwise-unpriced emissions costs from society at large to those industries responsible for the emissions. According to the carbon tax bill, companies are required to pay a tax rate of R120 per tonne of carbon emissions released and from June 2019 until 2022 the tax rate will increase by 2% per year. From 2023 the rate will increase according to inflation (WWF, 2018). However, the first phase until 2022 has extensive allowances for specific emitter circumstances, lowering the effective rate to between R6 and R48 per tonne. Although South Africa has implemented various policies and legislation to deal with climate change issues, the country has a history of little to no policy implementation. Sometimes the lack of capacity within government institutions results in non-compliance to climate change laws (Craigie et al., 2009; Leiman, 2014, Klausbruckner et al., 2016) and the fate of the money that will be generated from the carbon tax remains unclear.

The debate as to whether carbon tax or carbon trade is the best alternative for putting a price on carbon emissions has been ongoing for a while now (Carl and Fedor, 2016). The first emissions trading systems were developed in the 1980s but the world's first and biggest emissions trading system to focus on CO_2 that included many countries was the European Union Emissions Trading System (EU-ETS) which had a pilot phase 1 that was implemented in 2005. In the second phase in 2008, other GHGs were included (Schmalensee and Stavin, 2009). Carbon tax on the other hand was first introduced in the northern European countries, firstly in Finland in January 1990 followed by Netherland, Norway, Sweden and Denmark (Sumner et al., 2009). The main differences between the two is that with carbon tax, the price is fixed and certain but the quantity changes. In carbon cap and trade the quantity is fixed and certain, but the price is variable (Pizer, 2002; Carl and Fedor, 2015). Some have argued for and against carbon tax (Weisbach and Metcalf, 2009; Nordhaus, 2007; Keohane, 2009; Stavins, 2007) whereas others have suggested that the two can be merged as they function in similar ways (Aldy et al., 2010). For example, in the United Kingdom, carbon tax was introduced to support cap and trade as the cap and trade permits are often unstable and unpredictable (Carl and Fedor, 2016). South Africa is looking to something similar, with the draft Climate Change Act proposing the establishment of cap-and-fine mechanisms for sectors and companies (gazette available of www.gpwonline.co.za).

Carbon tax has generated interest again as it is perceived as simpler to implement and more stable than cap and trade. Although carbon markets are slightly riskier, they are still the preferred method by many countries compared to carbon tax (Pollitt, 2015). In a computable general equilibrium (CGE) model, Alton et al. (2014) looked at the potential socioeconomic impacts of introducing carbon tax in South Africa. The results indicated that introducing carbon tax will negatively affect the economy through decreases in exports which will ultimately result in job losses and increased energy prices, it may also result in slower employment growth for less educated workers and slower wage growth for the more educated. The model also showed that employment losses will also increase with an increase in carbon tax by the year 2025. Despite this, the South African government stated that carbon taxes are preferred over cap and trade (National Treasury, 2013; 2014). However, there has been some carbon trading in South Africa, and this will be discussed in the section below.

Carbon Trading – What Has Been Done?

The South African government released a draft document which reported that companies can buy offsets verified under the CDM, VCS, GS and CCBS to meet a 5-10% emissions reduction (National Treasury, 2014; Hamrick and Gallant, 2017). According to the carbon offset paper by the government in 2014, there are 111 registered carbon offset projects in South Africa. As of February 2013, there were 80 under CDM (with 12 issued with CERs) with 58 are still undergoing various stages of the project cycle, six projects under VCS, 22 projects under GS and only 3 under CCBS. It has been predicted that the potential overall national demand for offset could be up to 30 million $tCO_2eq.yr^{-1}$ (30 million carbon credits) (Camco Clean Energy, 2012). According to another study, the level of greenhouse gas emission could be up to 64 million tCO_2 eq. yr^{-1} and only half of these emissions can be covered by the carbon tax (Promethium Carbon, 2012, 2014). CDM projects are expected to reduce 17.2 million tCO_2 eq yr^{-1}. According to the South African National Carbon Sink Assessment (DEA, 2015), terrestrial carbon projects could only account for 8 million tCO_2 eq yr^{-1}. But what about blue carbon in coastal and marine habitats?

Policy Considerations and Possible Blue Carbon Project Methodologies

There are general considerations that are required for a successful carbon project such as developing the project boundary and baseline GHG accounting that accounts for leakage, permanence and the origin and fate of carbon (Emmer et al., 2015).

Before the start of a project, the project boundaries need to be established spatially and temporally. Also deciding which carbon pools and GHG will be measured forms part of establishing the project boundaries. When deciding on boundaries of wetlands, it is important to factor in the sea-level rise as the original boundaries may shift landwards due to wetland migration, inundation and erosion. Temporal boundaries are used to set the crediting period which is correlated to permanence (UNEP and CIFOR, 2014). Permanence refers to how long the carbon pool will be present within the project area (UNEP and CIFOR, 2014). This is important to minimize the risk of CO_2 emissions occurring after the carbon offset has been sold (Ullman et al., 2013). Permanence is especially important when emission reductions are used as offsets because if the current carbon stock disappears then the offset will be affected. The two major threats to permanence are anthropogenic activities and sea-level rise; it is important to select sites where there is no projected development and sites that will be resilient to sea-level rise such as those with a gradual slope for wetland migration.

Another precondition for acceptable offsets is to minimize the risk of leakage, which is when the activities of the project result in the increase of carbon dioxide emissions somewhere else (often outside of the project boundary). If an emissions reduction project results in emissions in another area, the project is rendered ineligible for offsets (Ullman et al., 2013). One of the common types of leakage is ecological leakage. For projects that are aimed at conservation, the project area remains intact and ecological leakage is less likely to occur (UNEP and CIFOR, 2014). The last important consideration for a blue carbon project is distinguishing between autochthonous and allochthonous carbon, specifically the carbon that is generated within (through photosynthesis) and outside the project area (UNEP and CIFOR, 2014).

Autochthonous carbon consists of the carbon in the organic matter from the dead vegetation in and around the wetland, while allochthonous carbon is mainly that which is dissolved and suspended in the runoff (Villa and Bernal, 2018). In carbon projects, only autochthonous carbon is considered for carbon credits and it is generally assumed that if the soil has very low organic matter then the carbon is generated outside the system due to a large deposition of mineral rich carbon. Therefore, projects with highly mineral soils are obligated to account for allochthonous carbon (UNEP and CIFOR, 2014). Many of the abovementioned aspects are based on a REDD+ type of framework which still needs to be properly established and implemented successfully in blue carbon projects (Wylie et al., 2016; Villa and Bernal, 2018). Factoring in all these considerations into the methodology can be very technical and expensive, notwithstanding the many legal components that are also required in preparation (Emmer et al., 2015).

Blue carbon activities include conservation/protection (avoiding the release of GHGs into the atmosphere) or restoration and it is important to determine the appropriate standard and methodology based on the activity. Generally, conservation of intact blue carbon habitats is simpler and cheaper than restoration (as habitat destruction often alters environmental conditions). Projects that are geared towards avoided carbon losses are the projects that most commonly generate blue carbon credits. For blue carbon activities the most suitable carbon market would be the VCS as it is the leading carbon market globally and covers a variety of blue carbon activities. The VCS includes both restoration and conservation activities under its AFOLU requirements and has recently introduced an additional category which specifically targets wetlands. This is known as the Wetlands Restoration and Conservation (WRC) which has two categories; Restoring Wetlands Ecosystems (RWE) and Conservation of Intact Wetlands (CIW). A variety of carbon accounting methodologies are available for AFOLU projects. These include VCS Methodology for Tidal Wetland and Seagrass Restoration (VM0033 V1.0) (Emmer et al., 2015) and the extension of REDD+ modular methodology

(VM0007) that will include both restoration and conservation of wetlands under WRC soon (UNEP and CIFOR, 2014). To date, methodologies that are specifically for wetland (mangrove, salt marsh and mangrove) conservation are yet to be implemented and only Macreadie et al. (2017a) has proposed a framework that includes best-management practises for the protection of existing wetlands for carbon offsets in carbon markets (Villa and Bernal, 2018). At present, there is no existing information on the value of blue carbon or the potential for carbon trading in South African estuaries and this study will be the first to leverage the country towards that goal.

Therefore, the first objective of this study was to determine the blue carbon sequestration and potential emissions reductions (PERs) of mangroves, salt marshes and seagrasses using data from the existing carbon inventory at four South African estuaries. The carbon sequestration and PERs were compared to terrestrial habitats and studies done internationally. The second objective was to determine the value of blue carbon and assess the viability of a carbon trading using a Net Present Value Analysis with the aim of conserving intact blue carbon habitats in South African estuaries over a 20-year period.

6.2 Materials and Methods

6.2.1 Study Site Descriptions

For this study, carbon sediment and biomass data were available for four sites, Nxaxo, Nahoon, Swartkops and Knysna estuaries. All the estuaries fall within the warm temperate bioclimate region in the Eastern and Western Cape of South Africa. Nxaxo, Nahoon and Swartkops estuaries are predominately open while Knysna is a tidally dominant estuarine bay. The area covered by mangrove, salt marsh and seagrasses at each estuary is described in Table 6.1 below. The Nxaxo Estuary (32°35’ S; 28°31’ E) is located at Wavecrest close to Butterworth in the Eastern Cape of South Africa. The estuary has mangroves and salt marshes co-occurring with patches of seagrass. Blue carbon (biomass and soil) was quantified using the methods outlined in Howard et al. (2014) and detailed sampling protocol was outlined in Johnson et al. (2020) and Section 3.1 of this report. The Nahoon Estuary (32°59′ S, 27°56′ E) is a predominately open estuary that is found at East London within the East London Coastal Nature Reserve also in the Eastern Cape. It is the southern distributional limit for mangrove distribution in South Africa (Hoppe-Speer et al., 2015b; Geldenhuys et al., 2016). The mangroves at Nahoon Estuary were planted in 1969 and the forest has since expanded into the tidal flats and salt marsh areas (Ward and Steinke, 1982; Hoppe-Speer et al., 2015b). Only the soil carbon for salt marshes and mangroves was available for this estuary and was quantified by Raw et al. (2019b). The Swartkops Estuary (33°51'54"S; 25°38'00"E) is located near Port Elizabeth in Algoa Bay (Bornman et al., 2016). The Knysna Estuary (34°43′S; 23°04′E) was the only Western Cape Province estuary that was included in our assessment. Both Swartkops and Knysna estuaries contain the largest seagrass areas and Swartkops the third largest salt marsh area (Adams et al., 2016; Adams and Veldkornet, 2016). The sampling protocol for soil and biomass carbon for Swartkops Estuary was outlined in Els (2019) and for Knysna Estuary in Raw et al. (2020) and Section 4.2 of this report.

6.2.2 Carbon Stocks and Potential Emissions Reductions

Soil cores were collected at a depth of 1 m at the Nxaxo Estuary and 0.5 m at Nahoon, Swartkops and Knysna estuaries. At Nxaxo and Swartkops estuaries soil organic carbon (C_{org}%) was determined from CHN analysis while at Nahoon and Knysna estuaries organic matter (OM%) was measured from the loss on ignition (LOI) method and was used as a proxy for C_{org}%. For mangroves, C_{org}% was 2% and 5% at Nxaxo and Nahoon estuaries respectively, while salt marshes ranged from 1-6% and seagrasses 0.5-3% (Table 6.2). Generally at all the estuaries C_{org}% was quite low and this indicated the dominance of mineral rich soil because according to the IPCC (2006) guidelines, mineral rich soils are those that are either a) coarse grained with more than 12% C_{org} or ~20% OM or b) fine grained (>60% clay) with 18% C_{org} or 30% OM. Total soil carbon stocks for mangroves at Nahoon and Nxaxo estuaries were 110 Mg C ha^{-1} and 230 Mg C ha^{-1}, 3-5 Mg C ha^{-1} for salt marshes at Nxaxo, Nahoon and Swartkops estuaries and ranged from 2-220 Mg C ha^{-1} for seagrasses at Nxaxo, Swartkops and Knysna estuaries (Table 6.3). In all the studies, biomass carbon was much higher than the carbon stock in the soil. Biomass carbon was 40% for mangroves at Nxaxo Estuary and ranged from 29-32% for salt marshes while seagrasses ranged from 22-29% at Nxaxo and Swartkops estuaries (Table 6.2). Total biomass carbon stocks were ~71 Mg C ha^{-1} for mangroves, ~4 Mg C ha^{-1} for salt marshes and between 1-2 Mg C ha^{-1} for seagrasses (Table 6.2).

Direct measurements of CO_2 emissions can be quite variable and expensive, so alternative methods are used such as the conversion of total carbon stocks using a conversion factor. The carbon stored in ecosystems is measured in units of megagram of carbon per hectare (Mg C ha^{-1}) where 1 Megagram (Mg) is equal to 1 metric tonne. Carbon emissions are measured in units of megagrams of carbon dioxide per hectare (Mg CO_2 ha^{-1}) often referred to as Mg CO_2 equivalent (UNFCCC, 2011). One carbon credit represents one tonne of carbon dioxide equivalent (expressed as tCO_2eq). The carbon stock (Mg C ha^{-1}) for the top 1 m of the sediment core was converted into carbon dioxide emissions using the conversion factor of 3.67 (the carbon to CO_2 ratio is 44:12) (Murray et al., 2011, Howard et al., 2014; Thompson et al., 2014, Adame et al., 2018) and because 1 metric tonne of carbon emitted is equal to 3.67 (UFCCC, 2011).

Table 6.1. The areal extent (ha) of blue carbon habitats at each of the estuaries in this study.

Estuary	Mangroves	Salt marsh	Seagrass
Nxaxo	9.5	10.91	0.34
Nahoon	1.62	2.99	2.3
Swartkops	-	547.35	62
Knysna	-	684.93	447.3

Table 6.2. Soil and biomass organic carbon (%Corg) and total carbon stocks (Mg C ha^{-1})) for mangroves, salt marshes and seagrass at four warm temperate South African estuaries (Mean ± SE). %Corg was determined using organic matter (OM%) from the loss on ignition method (LOI).

Site	Habitat type	Depth (cm)	$\%C_{org}$		Total carbon stock (Mg C ha^{-1})		Reference
			Biomass	Soil	Biomass	Soil	
Nxaxo Estuary	Mangrove	100	40.1 ± 0.81	1.96 ± 0.31	71.45 ± 2.47	228.05 ± 27.99	Johnson et al. (2020)
	Salt marsh	100	26.53 ± 0.68	1.4 ± 0.29	3.96 ± 0.78	2.61± 0.19	Section 3.1 of this report
	Seagrass	100	22.49 ± 0.75	0.86 ± 0.07	1.26 ± 8.1 x 10^{-5}	1.67 ± 0.01	Section 3.1 of this report
Nahoon Estuary	Mangrove	50	-	5.66 ± 0.56	-	110.14 ± 11.02	Raw et al. (2019b)
	Saltmarsh	50	-	6.18 ± 0.46	-	109.62 ± 22.03	Raw et al. (2019b)
Swartkops Estuary	Saltmarsh	50	31.9 ± 1.64	3.51 ± 0.34	4.28 ± 0.72	212.26 ± 43.99	Els (2019)
		50	29.1 ± 0.66	4.15 ± 0.41	16.27 ± 2.86	247.13 ± 47.71	Els (2019)
	Seagrass	50	27.2 ± 2.29	2.85 ± 0.24	2.08 ± 0.49	224.14 ± 37.93	Els (2019)
Knysna Estuary	Salt marsh	50	-	4.02 ± 1.47	-	-	Raw et al. (2020)
	Seagrass	50	-	0.46 ± 0.09	-	24.96 ± 6.43	Els (2017)

6.2.3 Carbon Sequestration Rates

Carbon sequestration (g C m^{-2} yr^{-1}) was calculated using soil carbon density (g cm^{-3}) multiplied by surface elevation change (cm yr^{-1}) which is obtained from the Rod Elevation Surface Tables (RSETs) (Lovelock et al., 2014; Rogers et al., 2014b). Three RSET stations were set up in the mangrove and salt marsh area at the Nxaxo and Nahoon estuaries in June 2018 (Section 4.1 of this report).and the most recent measurements were taken in June and August 2019. At Swartkops Estuary, eight permanent RSET stations were set up in the lower intertidal *Spartina maritima* zone in July 2009 and sampled seasonally until 2015 (Bornman et al., 2016). Likewise, at Knysna Estuary, seven permanent RSET stations were established in 2009 (also in the *S. maritima* zone) and surface elevation change was measured every year by Schmidt (2013). These RSET stations were resampled by Raw et al. (2020) to determine the surface elevation change from 2009 to 2017. At present, no RSET stations have been established for seagrass habitats in South Africa and are commonly very limited in seagrass habitats (Alongi, 2018). So, for this study we used the available global mean of 2 mm yr^{-1} that has been reported in literature (Duarte et al., 2013b; Röhr et al., 2016; Alongi, 2018). Carbon density ranged from 0.02-0.05 g cm^{-3} for mangroves and salt marshes and 0.007-0.05 g cm^{-3} for seagrasses. Surface elevation change ranged from 0.5-1 cm yr^{-1} for mangroves and -0.06-0.3 cm yr^{-1} for salt marshes (Table 6.3).

Table 6.3. Carbon density (g cm^{3}) and surface elevation change for mangroves, salt marsh and seagrass habitats at four South African warm temperate estuaries (Mean ± SE).

Site	Habitat type	Species	Carbon density (g cm^{3})	Surface elevation change (cm yr^{-1})	Reference
Nxaxo Estuary	Mangrove	*Avicennia marina*	0.01610 ± 0.0005	1.35 ± 0.181	Section 4.1 of this report; Johnson et al. (2020)
	Salt marsh	*Salicornia tegetaria*	0.0199 ± 0.00114	- 0.137 ± 0.233	Sections 4.1 and 3.1 of this report
	Seagrass	*Zostera capensis*	0.00714 ± 0.001403	0.22 ± 0.04	Alongi (2018); Section 3.1 of this report
Nahoon Estuary	Mangrove	*Avicennia marina*	0.02203 ± 0.001	0.464 ± 0.052	Raw et al. (2019b)
	Salt marsh	*Salicornia tegetaria, Bassia diffusa, Triglochin striata*	0.0222 ± 0.0017	-0.062 ± 0.018	Section 4.1 of this report; Raw et al. (2019b)
Swartkops Estuary	Salt marsh	*Salicornia tegetaria*	0.0461 ± 0.0043	0.298 ± 0.234	Bornman et al. (2016); Els (2019)
		Spartina maritima	0.0472 ± 0.0048	0.298 ± 0.234	Els (2019); Bornman et al. (2016)
	Seagrass	*Zostera capensis*	0.047 ± 0.00523	0.22 ± 0.04	Alongi (2018); Els (2019)
Knysna Estuary	Salt marsh	*Spartina maritima*	0.0207 ± 0.0011	0.124 ± 0.0754	Raw et al. (2020)
	Seagrass	*Zostera capensis*	0.0062 ± 0.00107	0.22 ± 0.04	Els (2017)

6.2.4 Blue Carbon Financial Value and the Viability of Carbon Trading

The financial value of blue carbon (revenue) was calculated from the cost of each carbon offset (representing a tonne of decreased emissions) multiplied by the volume of offsets. The cost of carbon is determined by carbon markets. The volume of the offset sold was derived from the carbon sequestration and the emissions (Ullman et al., 2013). Voluntary carbon credits for coastal wetlands range from US$5 to US$90 (R72-R1 291)/tCO_2eq (Emmer et al., 2015). Two reasonable carbon offset prices of US$3 and US$5/tCO_2eq were suggested by Goldstein and Gonzalez (2014) or US$6 to upwards of US$125/tCO_2 by Carr et al. (2018). To calculate the present financial value of blue carbon in this study, we used the minimum, maximum and average cost of carbon in the voluntary market according to the World Bank (2019) which were US$0.1/$tCO_2eq, US70/tCO_2eq and US$1/$tCO_2$eq. The selected prices were then converted to Rands at the conversion of US$ 1 = R14 which resulted in the prices being R1.4/tCO_2eq, R980/tCO_2eq, and R14/tCO_2eq. We also calculated the blue carbon value according to the income from carbon offsets or carbon tax in South Africa which was R120/tCO_2eq.The price of carbon (PriceCt) under these different rates was then multiplied by the carbon sequestration (CS) rate (Mg C ha^{-1} yr^{-1}) and the avoided carbon emissions (AvCE)/PER(tCO_2eq) calculated from the total carbon stock of each blue carbon habitat as shown in equation (1) below adapted from Beaumont et al. (2014) and UNEP (2016):

$$\mathbf{1)}\ \boldsymbol{BC\ Financial\ Value} = (\boldsymbol{CS} + \boldsymbol{AvCE}) \times \boldsymbol{PriceC}$$

To determine the viability of blue carbon or the economic benefit of intact mangroves, salt marsh and seagrass habitats, we used a Net Present Value (NPV) analysis following the methods of Pendleton et al. (2014), Thompson et al. (2014), Chang et al. (2015) and UNEP (2016). An NPV determines the revenue (BC financial value) generated by blue carbon schemes over a certain period. This must be greater than the cost of establishing and managing the project by either protection or restoration. For the protection of existing blue carbon habitats, the volume of the offset is a sum of avoided emissions as a result of habitat destruction and carbon sequestration (Ullman et al., 2013). The NPV analysis calculated the future values and annual costs and benefits in present value using 5% and 8% discount rates over a 20-year horizon period which is the period suggested by UNEP and CIFOR (2014). The process of discounting in economics is simply the conversion of future prices into the present value (Thompson et al., 2014). Although choosing a “real” discount value is not straight forward, it has been recommended that for near future projections (timeframes of 6-25 years), discount rates of between 3% and 10% should be used (Murray et al., 2011; Barbier, 2012; Thompson et al., 2014). The NPV analysis included blue carbon payments, establishment cost, opportunity cost and management cost as shown in equation (2) from UNEP (2016):

$$2)\ \boldsymbol{BC\ Financial\ Value_{it}} = \sum_{t=0}^{20} \frac{(\boldsymbol{CSit} + \boldsymbol{AvCEit})\ \boldsymbol{x\ PriceCt}}{(\boldsymbol{1}+\boldsymbol{d})^{t}} + \boldsymbol{PAE_{stab+}}\ \boldsymbol{PAM_{gmt+}}\ \boldsymbol{OppoCost_i}$$

Carbon sequestration (CS) was calculated from carbon density and surface elevation and AvCE was the avoided cost of emissions/PER ($MgCO_2eq$) calculated from the total carbon stock. PriceCt is the carbon market price and we used the abovementioned prices of carbon which were R1.40/tCO_2eq (min), R980/tCO_2eq (max) R14/tCO_2eq (avg), and R120/tCO_2eq (carbon tax). PAE_{stab} refers to the one-time cost of establishing protected areas (where the blue carbon ecosystems will be conserved for avoided emissions) and PAM_{gmt} is the annual cost of managing these protected areas. The opportunity cost of conservation (OppoCost) is the value per hectare of alternative use (UNEP, 2016). In this case, alternative use was assumed to be agriculture and agricultural returns per hectare were estimated to be R89 900 (DAFF, 2014). According to the 2018 National Biodiversity Assessment, one of the key pressures on estuaries in South Africa is agriculture (clearing for crops and grazing). This is in accordance with international findings which agree that the main threat to wetlands is conversion to agriculture and pastures for livestock (Villa and Bernal, 2018). The offsetting opportunity cost is a requirement to determine whether the blue carbon value (creditable emissions reductions) could be economically viable (Siikamäki et al., 2012; Thompson et al., 2014). It is recommended that two sets of establishment, maintenance and management costs be used based on global data since there is usually no region or country specific data. It is generally assumed that the cost of establishing a blue carbon project is between US$25/ha and US$232/ha (low and high estimates) (McCrea-Strub et al., 2011; Pendleton et al., 2014; Vasconcelos et al., 2014; Chang et al., 2015). Ongoing project management/maintenance costs have been estimated at between US$1/ha (~R14/ha) and US$ 7/ha (R98/ha) (Balmford et al., 2003; Vasconcelos et al., 2014). Blue carbon is considered viable if the net benefit of conservation (expressed as the BC financial value) is larger than the sum of management/protection and opportunity cost in alternative use (agriculture) over the 20-year horizon period (UNEP, 2016).

6.3 Results

6.3.1 Carbon sequestration rates and potential emissions reductions by South Africa's blue carbon habitats

The potential carbon sequestration rate for mangroves at Nxaxo Estuary was higher (217 ± 0.9 g C m^{-2} yr^{-1}) than at Nahoon Estuary (102.26 ± 0.5 g C m^{-2} yr^{-1}). At both Nxaxo and Nahoon estuaries the potential carbon sequestration for salt marsh habitats was 26.44 ± 2.66 and 13.76 ± 3.09 g C m^{-2} yr^{-1} and this was lower than Swartkops (101.9 ± 10.63 g C m^{-2} yr^{-1}) and Knysna (25.01 ± 8.29 g C m^{-2} yr^{-1}) estuaries. Seagrass carbon sequestration was highest at Swartkops Estuary (94 ± 2.09 g C m^{-2} yr^{-1}) compared to Nxaxo (14 ± 0.8 g C m^{-2} yr^{-1}) and Knysna (12 ± 0.42 g C m^{-2} yr^{-1}) (Figure 6.1).

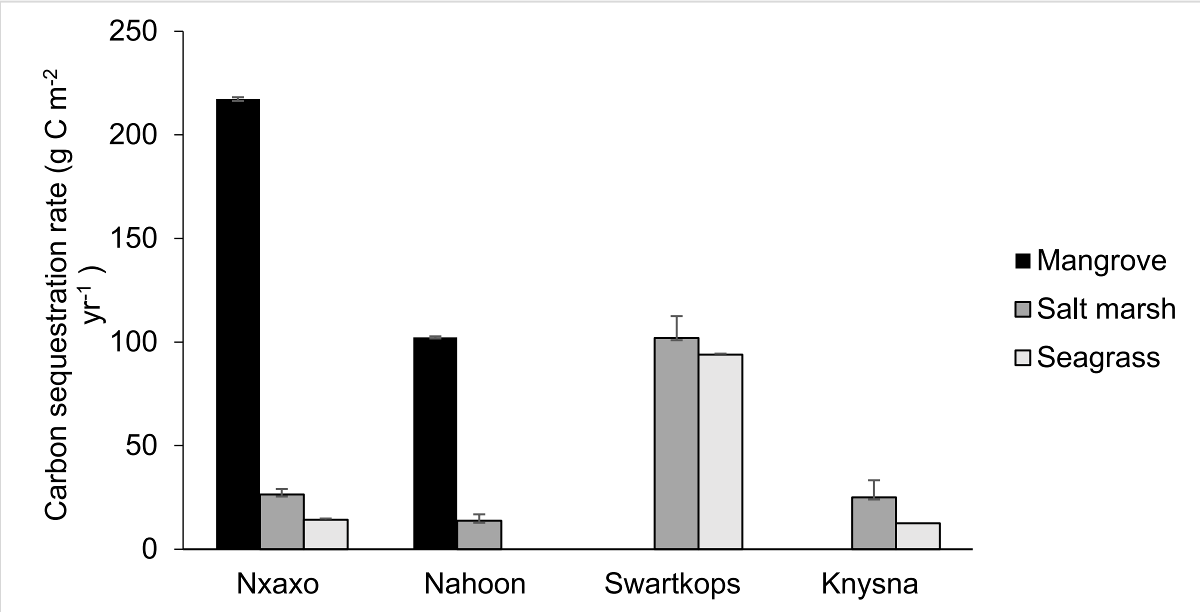


Figure 6.6 Carbon sequestration rates of blue carbon habitats in some of the South African estuaries.

Carbon dioxide potential emissions reductions were highest for the mangroves (1099.17 ± 111.79 tCO_2eq) at Nxaxo Estuary and lower for mangroves at Nahoon (404.21 ± 40.44 tCO_2eq). Swartkops Estuary had the highest carbon dioxide potential emissions reduction for both the salt marsh (880.69 ± 178 tCO_2eq) and seagrass (830.23 ± 141 tCO_2eq) habitats compared to Nxaxo and Nahoon salt marsh (24.11 ± 3.56 tCO_2eq and 402.21 ± 40.44 tCO_2eq) and Nxaxo and Knysna seagrass (10.75 ± 0.04 tCO_2eq and 91.62 ± 23.6 tCO_2eq) (Figure 6.2).

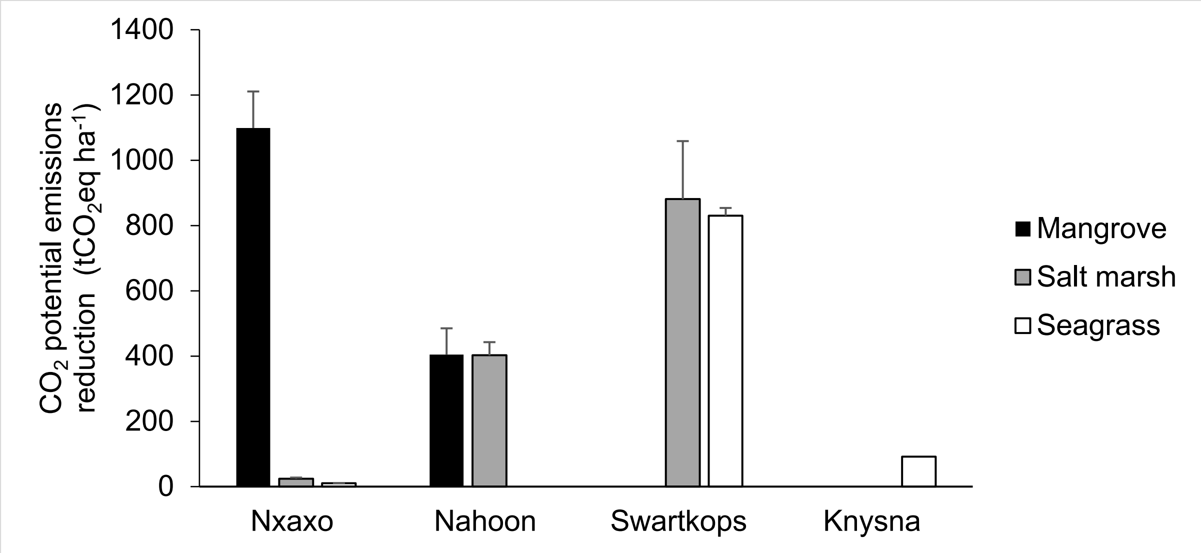


Figure 6.7. Carbon dioxide reduction emissions potential of blue carbon habitats in some of the South African estuaries.

In terms of country wide carbon dioxide reduction emissions potential, salt marshes had the highest value (8 million $tCO_2eq\ yr^{-1}$), and this was followed by mangroves (1.2 million $tCO_2eq\ yr^{-1}$) and then seagrasses (Table 6.4). The total carbon dioxide reduction emissions potential for all the blue carbon habitats in South Africa was 10.4 million $tCO_2eq\ yr^{-1}$.

Table 6.4. Annual carbon dioxide emission potential reduction for different habitat types (mangroves, salt marshes and seagrasses) in South African estuaries.

Habitat	**Area (ha)**	**Potential emissions reductions ($tCO_2eq.\ yr^{-1}$)**
Mangrove	2806.75	1259865.20
Salt marsh	15050.87	8237025.66
Seagrass	1672.49	885684.17

6.3.2 The Value of Blue Carbon in South African Estuaries

The value of blue carbon was highest in the mangrove habitats per hectare compared to salt marshes and seagrasses (Table 6.5). The blue carbon value of mangroves ranged from R1000-R10 000 ha^{-1} yr^{-1} , salt marshes between (R700-R7000 ha^{-1} yr^{-1}) and seagrass carbon (R400-R4000 ha^{-1} yr^{-1}) if carbon is traded at a low price (R1-R14/tCO_2eq). If carbon is traded at the same rate as the carbon tax and higher (R120-R980/tCO_2eq) then the total South African mangrove blue carbon value was between R90 000-R740 000.ha^{-1} yr^{-1}. Additionally, the salt marsh carbon value ranged from R66 000-R536 000 ha^{-1} yr^{-1} and seagrasses were valued at R32 000-R304 000 ha^{-1} yr^{-1}. Overall the total estimated carbon value of blue carbon habitats was ~R1.2-10.6 billion yr^{-1} when carbon is traded at a high price and ~R120-R150 million yr^{-1} at low carbon prices.

Table 6.5. The gross value of blue carbon in South African estuaries (Rands. ha^{-1} yr^{-1}) under various prices of carbon (PriceCt) per ton of carbon dioxide equivalent including the carbon tax (R120), the lowest (R1.4), highest (R980) and average (R14) prices of carbon.

	Carbon tax	**Low PriceCt**	**High PriceCt**	**Average PriceCt**
Mangroves	90 394.46	1054.60	738 221.39	10 546.02
Salt marsh	65 673.48	766.19	536 333.46	7661.91
Seagrass	37 327.39	435.49	304 840.36	4354.86

6.3.3 The Viability of Blue Carbon in South African Estuaries

A preliminary economic analysis of the NPV of carbon from mangrove, salt marsh and seagrass conservation was done in South Africa which factored in the cost of blue carbon, low and high establishment and maintenance/conservation and opportunity cost at 5% and 8% discount rates over 20 years (Table 6.6). However, other economic benefits were not included. Results from the analysis showed that blue carbon was only viable (expressed as a positive value for the net benefit of conservation in Table 6.7) under two scenarios. Firstly, when carbon is traded at the carbon tax PriceCt (8% discount rate) for both high and low establishment and maintenance costs which generated ~R200 000 per hectare. Secondly, at the maximum PriceCt (R980/tCO_2eq) (at 5 and 8% discount rates) for both high and low carbon project establishment and maintenance costs which generated ~ R1.5 million per hectare over the 20-year period. Under low carbon trading prices (minimum and average), a carbon offset project would not be viable even under low establishment and maintenance costs. The negative values indicated a loss in revenue instead of a gain. The costs outweighed the profits with the possibility that funds generated from a blue carbon project would be insufficient to support mangrove, salt marsh and seagrass conservation. Under low PriceCt (R1.4-R14/tCO_2eq) scenarios it was shown that the loss could be ~R1.2 million.

Table 6.6. The Net Present Value (NPV) analysis for blue carbon habitats in South Africa under various scenarios over a 20-year horizon period.

	Years				
	Present value	**1**	**2**	**3**	**20**
C value at R120/tCO_2e, 5%	R61901.03	R58953.36	R56146.06	R53472.44	R23329.85
C value at R120/tCO_2e, 8%	R61901.03	R57315.77	R53070.15	R49139.03	R13280.75
C value at R1.40/tCO_2e, 5%	R722.18	R687.79	R655.04	R623.85	R272.18
C value at R1.40/tCO_2e, 8%	R722.18	R668.68	R619.15	R573.29	R154.94
C value at R980/tCO_2e, 5%	R505525.06	R481452.44	R458526.13	R436691.56	R190527.08
C value at R980/tCO_2e, 8%	R505525.06	R468078.76	R433406.26	R401302.09	R108459.50
C value at R14/tCO_2e, 5%	R7221.79	R6877.89	R6550.37	R6238.45	R2721.82
C value at R14/tCO_2e, 8%	R7221.79	R6686.84	R6191.52	R5732.89	R1549.42
Low cost to manage, 5%	R14.98	R14.27	R13.59	R12.94	R5.65
High cost to manage, 5%	R104	R99.06	R94.34	R89.84	R39.20
Low cost to manage, 8%	R14.98	R13.87	R12.84	R11.89	R3.21
High cost to manage, 8%	R104	R99.05	R89.16	R82.56	R22.31
Opportunity cost, 5%	R89900	R85619.05	R81541.95	R77659	R33882.36
Opportunity cost, 8%	R89900	R83240.74	R77074.76	R71365.52	R19287.88

Table 6.7. The viability (net benefit of conservation) of blue carbon using an NVP analysis in South African estuaries under various scenarios (different PriceCt per hectare, carbon, establishment and conservation/management costs). A positive value indicated that the net benefit of conservation was larger than the sum of the costs associated with a blue carbon project over the 20-year period.

Blue Carbon Viability (Rands)	**R120**		**R1.40**		**R980**		**R14**	
	5	8	5	8	5	8	5	8
Low management + establishment cost	-R377504	R212391	-R1201106	-R1202976	R1483355	R2066677	-R1113607	-R379136
High management + establishment cost	-R381699	R208324	-R1205302	-R453515	R1479159	R2062610	-R1117802	-R383203

6.4 Discussion

6.4.1 Carbon Sequestration in South African Blue Carbon Habitats and Global Comparisons

Coastal vegetated habitats are declining at an increasing rate due to anthropogenic and climate change related pressures. Especially in South Africa where already 44.5% of our salt marsh habitat has been lost due to development (Adams et al., 2019). In this study we quantified the carbon sequestration value of blue carbon habitats in South Africa. We introduced a discussion on how these blue carbon habitats can be valued and the viability of a blue carbon offset project in the country to value the long-term benefits of the ecosystem service of carbon sequestration. Our analysis is a further step towards the participation of South Africa's blue carbon habitats in voluntary carbon markets.

Carbon sequestration rates for salt marshes in this study were very low (18.57 ± 6.16 g C m^{-2} yr^{-1}) compared to the global mean (Ouyang and Lee, 2014; Alongi, 2018) and the mean for southern Africa (Ouyang and Lee, 2014) (Table 6.8). These global estimates, however, are limited by the paucity of carbon sequestration data especially from Africa. Carbon sequestration rates were much higher for *Spartina maritima* (especially at Swartkops Estuary) compared to *Salicornia tegetaria* and the other salt marsh species at Nxaxo and Nahoon estuaries. *Spartina* marshes generally have a significantly higher carbon sequestration rate compared to other species (Ouyang and Lee, 2014) which was sometimes as high as 750 g C m^{-2} yr^{-1} (Sousa et al., 2017). Lovelock et al. (2014) also reported low carbon sequestration rates for *Sarcocornia* marshes in Australia but Saintilan et al. (2013) and Howe et al. (2009) suggested rates that were slightly higher. Apart from species composition, other important factors that influence the carbon sequestration rates of salt marshes include tidal range and elevation (Alongi, 2018). Ouyang and Lee (2014) reported that carbon sequestration decreases with increased elevation. Salt marshes at higher elevations in the high intertidal area have less sediment accretion and lower carbon sequestration because of inundation for shorter periods and limited sediment input (Chmura and Hung, 2004; Ouyang and Lee, 2014; Alongi, 2018).

The average surface elevation for salt marshes in our study was negative (-1.37 ± 0.23 mm. yr^{-1}) due to negative surface elevation measurements at Nxaxo and Nahoon estuaries. Typically, negative elevation change indicated a loss of elevation over time (Lovelock et al., 2014) however, our measurements were short term and it could not be inferred whether this was due to surface erosion. Nevertheless, short term changes in marsh elevation have been linked to various factors related to hydrological processes such as inundation and tidal regime shifts and biotic interactions (Cahoon et al., 2011). Firstly, OM accumulation and the deposition of inorganic soil lead to gains or removal of the substrate or scouring which may seem to be surface elevation loss (McKee et al., 2007; Cahoon et al., 2011; McKee, 2011). Secondly, the shrinking and swelling of the soil due to fluctuations in the water table can also lead to positive and negative surface elevation change (Cahoon et al., 1999; Webb et al., 2013). Lastly, biotic contributions such as root production, bioturbation and soil compaction are also important factors (Webb et al., 2013). We suspect that at Nxaxo Estuary the negative surface elevation maybe due to surface disturbance by cattle which browse in the adjacent mangrove area (Hoppe-Speer and Adams, 2015; Mbense, 2017) in addition to bioturbation by crabs which form burrows and mounds. The above factors still need to be investigated and related to surface elevation change in future research.

Mangrove carbon sequestration was less than the global mean (Alongi, 2018) and the value reported by Saintilan et al. (2013) in Australia. However, our value was similar to that of Howe et al. (2009) and higher than Lovelock et al. (2014) also in Australia (Table 6.8). Alongi (2018) suggested that mangrove carbon sequestration rates can be highly variable as they are largely influenced by numerous factors. These include the age of the forest, the frequency of tidal inundation and elevation, species composition, sediment particle size and anthropogenic pressure. In terms of forest age, older forests have higher carbon sequestration rates than younger ones. This was in accordance with our findings where mangroves at Nxaxo Estuary, which is an older, naturally established forest which had a higher carbon sequestration rate than Nahoon Estuary which has a younger mangrove forest (Hoppe-Speer et al., 2015a).

The seagrass carbon sequestration rate was much less than the global mean (Alongi, 2018). Though, it was reported that the global value was based on metabolic measurements (i.e. respiration rates) and not actual measurements of carbon storage and accumulation, due to a lack of long-term monitoring datasets worldwide (Duarte et al., 2013c; Alongi, 2018). The limited dataset has also shown that species variation and seasonality has not been accounted for in global estimates (Lavery et al., 2013; Rozaimi et al., 2016). In terms of seasons, in summer, sequestration rates rise with increased plant density while in winter sediment resuspension supersedes sediment accumulation (Alongi, 2018). Unlike the global mean value, data from empirical studies described considerably lower carbon sequestration rates (Table 6.8). For example, Serrano et al. (2014) and Röhr et al. (2016) found that seagrass carbon sequestration rates were highly variable while Rozaimi et al. (2016) and Cusack et al. (2018) reported lower rates than our finding.

Table 6.8. A comparison of carbon sequestration rates of blue carbon habitats from different countries regions obtained in literature.

Habitat	*Country/Region*	*Mean Temperature (°C)*	*Mean Rainfall (mm. yr^{-1})*	*Species*	*Carbon sequestration rate (g C m^{-2} yr^{-1})*	*Reference*
Salt marsh	South Africa	18	464	Multiple	18.57	This study
	Global			Multiple	212	Alongi (2018)
	Southern Africa	-	-	Multiple	200.9	Ouyang and Lee (2014)
	Portugal	19.1	1327	*Spartina maritima*	750	Sousa et al. (2017)
	Australia/Moreton Bay	17	150-200	*Sarcocornia quinqueflora*	8.6	Lovelock et al. (2014)
	South east Australia	22	248.9	*Sarcocornia*	46	Saintilan et al. (2013)
	South east Australia	22	248.9	*Sarcocornia*	137	Howe et al. (2009)
Seagrass	South Africa	18	464	*Zostera capensis*	30	This study
	Global			Multiple	220.7	Alongi (2018)
	Denmark & Finland	15.2 &7.5	552 & 700	*Zostera marina*	5.2-49	Röhr et al. (2016)
	Mediterranean	22	350-900	*Posidonia oceanica*	6-175	Serrano et al. (2014)
	Arabian Gulf	27	101.3	Multiple	9	Cusack et al. (2018)
	Western Australia	19.2	168	*Posidonia australis*	3.45	Rozaimi et al. (2016)
Mangroves	South Africa	18	464	*Avicennia marina*	106.75	This study
	Global			Multiple	220.7	Alongi (2018)
	Southern Australia	22	248.9	*Avicennia marina*	256	Saintilan et al. (2013)
	Southern Australia	22	248.9	*Avicennia marina*	105	Howe et al. (2009)
	Australia/Moreton Bay	17	150-200	*Avicennia marina*	76.16	Lovelock et al. (2014)

6.4.2 Potential Emissions Reductions and the Value and Viability of Blue Carbon in South Africa

Blue carbon habitats in South Africa can potentially remove 10.3 million $tCO_2eq\ yr^{-1}$ and this is higher than the projected sequestration potential from terrestrial habitats (8 million $tCO_2eq\ yr^{-1}$ reported in the National Terrestrial Carbon Sink Assessment (DEA, 2014). Salt marshes were the habitats with the highest contribution to this value as they have the largest habitat area. Our value was comparable to PERs of blue carbon habitats in southeast Australia which was reported to be 9.63 million $tCO_2eq\ yr^{-1}$ (Lewis et al., 2018). Thompson et al. (2014) reported a value of 61 $tCO_2eq\ ha^{-1}\ yr^{-1}$ for mangroves in the Panay Islands, Philippines which was much lower than our findings. A study done at Kosi Bay in KwaZulu-Natal showed that mangroves in this system had a PER of 40.13 million $tCO_2eq\ yr^{-1}$ which was significantly more than findings from our study (Hicks, 2019).

The value of blue carbon habitats was R1.2-R10.6 billion per year when carbon is traded at a high price and ~R120-R150 million per year when carbon is traded at lower prices. Our findings were less than the carbon value of R40.7 billion reported in Turpie et al. (2017) which was calculated using the 0.35% of the social cost of carbon (US$31.25/$tCO_2eq$). Although this estimate only used a fraction of the social cost of carbon, it was still higher than the value for blue carbon habitats at a high cost of carbon. In a recent study done on the value of carbon for mangroves at Kosi Bay in KwaZulu-Natal, South Africa, it was found that blue carbon was valued at R22 495 per year and this was calculated using the social cost of carbon of $40/$MtCO_2eq$ (Hicks, 2019) which was less than the values reported in this study. In terms of the annual carbon value per hectare under the various PriceCt scenarios, the value of blue carbon in our study ranged from R750-R500 000. At the low PriceCt this falls within the range estimated for coastal and terrestrial habitats in Turpie et al. (2017) (R500-R1000+). Lewis et al. (2018) found that blue carbon habitats in Australia were valued at $AUD 130 million (~R13 million) calculated using $AUD10-13/$tCO_2eq$ from the Australian Reductions Emissions Fund guidelines.

Other preliminary findings using NPVs from Murray et al. (2011); Siikamäki et al. (2012) and UNEP (2016) showed that carbon offset projects can be viable even if the PriceCt is below US$10/$tCO_2eq$ (~R150) in compliance markets. However, in voluntary markets the PriceCt can be much less than this, making other land use options such as agriculture, development and even aquaculture more profitable for some countries. For example, in Bangladesh, shrimp aquaculture was the more profitable option over a blue carbon offset project as it generated ~1000 US$ $ha^{-1}\ yr^{-1}$ because shrimps were sold at higher prices than carbon in international markets (Ahmed, 2008; Thompson et al., 2014).

Based on our results, we suggest that blue offset projects will only be economically viable in South Africa if the PriceCt is high in the voluntary markets, which at present are the more attainable options for pilot projects (Ullman et al., 2013). When the net benefit of conservation of blue carbon has a low return on investment, this simply means that the project would require other streams of revenue associated with the project such as PES for other services that these habitats provide which could potentially alleviate the higher establishment, management and opportunity cost (UNEP, 2016). For instance, Turpie and Clarke (2007) estimated that the recreational and nursery values of Knysna Estuary were >R1 billion and >R100 million per year while Swartkops was valued at R20-R100 million per year. The estimated values for Nxaxo and Nahoon estuaries were lower (only R1-R20 million per year) but could still be significant in their contribution to the conservation of blue carbon habitats. With these additional ecosystem service values, the net benefit of the conservation of these systems would be higher and trading in the voluntary markets would be a viable option.

It is also important to note that in South Africa, our blue carbon habitat area is quite small compared to the rest of the globe and major habitat losses (6.9% and 44.5% of mangroves and salt marshes) have occurred due to development and agriculture (Adams et al., 2019). While this was not explicitly included, loss of habitat could have influenced the PERs, the overall carbon value and ultimately the viability of carbon trading in our study.

6.5 Limitations to This Study

The results presented here, were from measurements that where done on a regional scale and were applied at a national scale to contribute towards local and national policy planning. The warm temperate region in isolation is not a true representation of the country's blue carbon storage and additional field measurements in different climatic regions can vastly improve these estimates of carbon sequestration rates, potential emissions and the value of blue carbon. The length of the surface elevation measurements at Nxaxo and Nahoon estuaries was also a limiting factor. Short term measurements are less reliable to assess trends (Rybczyk and Cahoon, 2002; Cahoon et al., 2002) but more helpful to determine seasonal patterns and document extreme events (Whelan et al., 2005, 2009; Osland et al., 2017c). Ideally, datasets should be between 5 and 10 years to be considered long term and more 3 years to be short term (Webb et al., 2013).

The NPV analysis assumed no net loss of carbon and deforestation and degradation were not included. Mangrove loss was last reported to be 0.38% per year (6.4% in 17 years) accounting for 1.04 ha per year (Adams et al., 2004). For the scenario where the South African carbon tax was used, we did not account for the tax rate increase up to 2022 (which is inflation + 2%) and the increase from 2023 which will increase only by inflation every year (WWF, 2018).

Other GHGs such as methane and nitrous oxide where excluded from this study and their inclusion was negligible as they are generally low (<1%) in mangrove systems (Adame et al., 2018). We also compared the potential emissions reductions of blue carbon habitats to terrestrial systems, this not entirely correct. There is a higher degree of uncertainty in the carbon stock assessments of mangroves forest because of dynamics that are still misunderstood. For instance, natural variability is large even at relatively localised scales and there are still large inconsistences in the carbon accounting approaches and methods therefore making blue carbon credits a higher risk than their terrestrial counterparts (Thompson et al., 2014). We also just discussed financial viability only and excluded social and legal viability that is associated with carbon projects including public law, issues pertaining to land and carbon rights as outline in the UNEP (2016). We also did not account for the overlap between marine and terrestrial (ecotones) even though carbon measurements for the ecotone area were available at Nxaxo and Nahoon estuaries (Raw et al., 2019b).

In conclusion this study has shown that there is considerable uncertainty in the estimation of the value of carbon as suggested by Beaumont et al. (2014). This is due to the fluctuation of the price of carbon and the utilisation of different discount rates and the method used. However, this study has moved South African estuaries a step further towards carbon trading which will be aid with the protection and conservation of our blue carbon habitats.

7 CONCLUSIONS AND RECOMMENDATIONS

The ecological importance of mangroves, salt marshes, and seagrasses, in South Africa has been acknowledged since early estuarine surveys that were carried out along this coastline (Day, 1951; Macnae, 1963; Begg, 1978). Primarily, these vegetation types were recognized for providing structural habitat to characteristic estuarine species of birds, fish, and invertebrates (Macnae, 1957; Branch and Grindley, 1979; Whitfield, 1983). The importance of these areas to people was also acknowledged (Berjak et al., 1977), and resource harvesting still forms an important part of the livelihoods of subsistence communities today (Rajkaran et al., 2004; Traynor and Hill, 2008). Over time, the importance of these areas to provide 'unseen' ecological services has been increasingly recognized (Turpie et al., 2017). In the past decade, global research efforts have focussed on assessing carbon storage and sequestration as one of the most important ecological services provided by mangroves, salt marshes, and seagrasses, following the 2009 UNEP report that termed these as "blue carbon" ecosystems (Nellemann et al., 2009; Mcleod et al., 2011; Lovelock and Duarte, 2019). This research has been advocated to be important in relation to contemporary climate change, as blue carbon ecosystems have the potential to contribute natural infrastructure for mitigation (Sidik et al., 2018; Taillardat et al., 2018; Serrano et al., 2019). This research project has therefore been carried out to provide the first comprehensive assessment of blue carbon habitats in South Africa. This research has used some existing data, but a large amount of new knowledge has been generated by collecting data using approaches that have not previously been applied in South Africa. The aims and objectives of this research were related to assessing changes in ecosystem services provided by blue carbon ecosystems; quantifying carbon storage in blue carbon ecosystems; and predicting the responses of blue carbon ecosystems to climate change.

Overall, the results of this research project fill an important global knowledge gap on blue carbon research in southern Africa. However, these results are also of national interest and many components of this research have been incorporated into the 2018 National Biodiversity Assessment (Van Niekerk et al., 2019). This research is a first step to leveraging South Africa's estuaries as part of a potential carbon credit trading scheme. Coastal wetland protection and carbon sequestration can be included in payment for ecosystem services. Predicting the combined effects of multiple stressors associated with environmental instability is critical to conserve blue carbon habitats. Pressures such as infrastructure development, flow reduction, artificial breaching, mouth manipulation and overfishing have made estuarine ecosystems that support blue carbon habitats more vulnerable to climate change. A strategic program is thus needed to restore health so that blue carbon habitats continue to provide ecosystem services of flood regulation, nutrient cycling, nursery habitat and recreational and tourism opportunities.

7.1 Ecosystem Services

The current extent of blue carbon ecosystems and their associated ecosystem services in South Africa was quantified as part of this research. Assessments for individual estuaries indicated that there have been significant changes to habitat areas over time. Some of the largest losses of mangroves in South Africa occurred following development of the industrial port in Durban (Moll et al., 1971; Rajkaran et al., 2009). Smaller mangrove forests in the Eastern Cape have been degraded by harvesting and cattle browsing practices (Rajkaran and Adams, 2010; Hoppe-Speer and Adams, 2015; Mbense, 2017). In contrast, salt marsh areas have been reduced by urban development at supratidal-terrestrial boundaries (Adams et al., 2016; Bornman et al., 2016). Losses of seagrass areas has been associated with changes in hydrology that have caused sedimentation or resulted in limited tidal exchange (Adams, 2016). As part of this research, changes in habitat from one vegetation type to another were also recorded. Overall, these changes were related to shifts in ecosystem services which were quantified using standardized approaches that have been applied in other regions (Haines-Young and Potschin, 2018).

Zostera capensis in the intertidal zone of the Nahoon Estuary. Photo: J Raw

Ecosystem services have been previously assessed for South African estuarine ecosystems (Turpie et al., 2017). However, this research project has provided a revised assessment, with a focus on blue carbon ecosystems. This work has also created an ecosystem service scoring system that has been successfully applied to assess shifts in provisioning following changes to vegetation cover over time. This information can provide input to existing policy frameworks, in particular, the estuary management plan mandate of DEFF (Department of Environment, Forestry, and Fisheries) under the National Environmental Management Act. A tangible ecosystem service score is also an attribute that can be considered as part of the Estuarine Health Index (Van Niekerk et al., 2013). This work also provides a baseline for any future research on ecosystem services for South African estuaries. Research on ecosystem services promotes inter- and transdisciplinary approaches and assists in the understanding and conservation of our complex interconnected systems in the Anthropocene. Quantification of ecosystem services should thus remain a research priority. Ecosystem services are dynamic varying in space and time; this requires a deeper understanding and quantification.

7.2 Quantification of Carbon Storage

This research project has provided the first comprehensive quantification of carbon storage in blue carbon ecosystems along the South African coastline using the globally recommended protocol developed by the Blue Carbon Initiative (Howard et al., 2014). For this research, carbon content was directly quantified from vegetation and soil carbon pools in mangrove, salt marsh, and seagrass habitats of the Nxaxo Estuary, Eastern Cape. This was the first study to use elemental carbon analysis to measure the percentage of organic carbon in these pools and to relate this to ecosystem-level carbon storage values for an estuary in South Africa. The results of this research were aligned with global trends, with mangroves having significantly higher carbon storage than salt marsh and seagrass areas. Overall, carbon storage in mangroves at the Nxaxo Estuary was relatively low but it was still similar to reports from warm-temperate range limits in Australia (Johnson et al., 2020). These results are globally relevant as they contribute new data towards variability in carbon storage between different regions.

The quantification of carbon storage was also important so that this new information could be used in other components of this research project. Direct quantification allows for a more accurate estimation of carbon storage as an ecosystem service in other estuaries by extrapolation instead of using global values that relate carbon storage to habitat area. In this project, the carbon storage values measured for seagrasses were particularly much lower than the global average. This was expected to be a result of the variability in vegetation structure and size, as the South African seagrass, *Zostera capensis*, has small leaf blades and generally does not cover extensive areas. The extent of these seagrass habitats is variable and fluctuates in response to floods and droughts (Adams, 2016). Carbon storage data were used to estimate carbon sequestration rates and assess the viability of a carbon offset mechanism for South Africa. Future research should investigate the influence of environmental variables on carbon storage at different spatial scales. Quantification of vertical accretion rates is also needed to provide better estimates of carbon sequestration potential. In the long term this can be achieved through the data collected from the RSET benchmarks, but historical accretion rates can also be measured using Pb^{210} isotope dating of the sediment.

The occurrence of the east African continental warm-temperate range limit for mangroves on the South African coastline provides an important opportunity to conduct research relating to range expansions in response to global climate change. Globally, mangroves have been expanding towards higher latitudes with increased warming, and this can often be at the expense of salt marsh habitats (Saintilan et al., 2014; Osland et al., 2017a). Conversion of salt marsh to mangrove has in some cases been associated with a significant increase in soil carbon storage (Kelleway et al., 2016b; Simpson et al., 2019). However, examining this trend at the South African range limit at Nahoon Estuary found that soil carbon storage was similar across the mangrove-salt marsh ecotone (Raw et al., 2019b).

Future research that quantifies carbon storage in South African blue carbon ecosystems should focus on the larger mangrove areas in estuaries of KwaZulu-Natal. As these mangroves occur at subtropical latitudes and these systems receive larger sediment inputs (from more erodible catchments), it is likely that carbon storage will be much greater than what was measured at the Nxaxo and Nahoon estuaries. This research has also highlighted the importance of using elemental analysis to quantify organic carbon, as the alternative loss-on-ignition approach can provide an overestimate. Future research should therefore also follow the recommended protocol so that results will be directly comparable between different estuarine systems.

7.3 Responses to Sea-Level Rise

Mangroves and salt marshes are vulnerable to threats from sea-level rise as they are restricted to occurring within certain extents of the tidal frame. This research project has established the first monitoring stations to measure the responses of mangroves to sea-level rise in South Africa using the globally standardized Rod Surface Elevation Table (RSET) method. This work, carried out in collaboration with the South African Environmental Observation Network (SEAON) Elwandle Node, provides a platform for continued data collection so that these trends can be assessed in the long term, beyond the timeframe of this specific project. A total of 18 RSET benchmarks were set up in the Nahoon and Nxaxo estuaries (9 per estuary). The placement of the benchmarks allows for spatial representation of the mangrove and salt marsh areas, and also considers the need for replication. The preliminary results presented as part of this research project indicate that over short, seasonal timescales, there has been a larger increase in surface elevation at mangrove sites, in comparison to salt marsh sites, at both estuaries.

J. Raw and T. Lehman measuring surface elevation at an RSET benchmark in the Knysna Estuary. Photo: C. Nolte

Besides the establishment of the mangrove RSET network, this research project also re-visited RSET benchmarks that were placed in the salt marsh of the Knysna Estuary by SAEON in 2009 (Schmidt, 2013). Changes in surface elevation over the past decade from when the original measurements were taken could then be related to sea-level rise trends measured at the nearest tide gauge (Raw et al., 2020). These results indicated that elevation gains and losses in *Spartina maritima* salt marsh habitats were variable along the length of the estuary, and most likely depended on local sedimentation and hydrodynamic patterns. The responses of salt marsh habitats in the Knysna Estuary could also be related to projected sea-level rise using the Sea-Level Affecting Marshes Model (SLAMM) approach (Clough et al., 2016). Threats to salt marshes from encroaching development at the salt marsh-terrestrial boundary were also assessed. The results from this research strongly suggest that no further development should be permitted along the estuary perimeter at Knysna as this could reduce the impacts of sea-level rise by allowing the salt marsh to migrate landwards, thereby avoiding coastal squeeze.

The collaboration with SAEON on this research component has been essential and it has ensured that these data will continue to be collected through the Shallow Marine and Coastal Research Infrastructure (SMCRI) that is supported through SAEON and the National Research Foundation (NRF). Future work to assess the responses of blue carbon habitats to sea-level rise could apply a similar approach to what was carried out at Knysna. The original study that placed the benchmarks at the Knysna Estuary, also placed benchmarks in *S. maritima* salt marsh areas of both the Kromme and Swartkops estuaries. Follow-up studies at these locations will allow for a comparison of responses over the last decade between these three estuaries that have different ecological conditions and anthropogenic impacts.

Future research in the mangrove estuaries can also take this approach, although it is recommended that data on surface elevation change are collected for at least 5 years before these trends can be related to sea-level rise (Cahoon et al., 2000; Lynch et al., 2015). Additionally, the mangrove RSET benchmarks need to be surveyed relative to mean sea-level (MSL) so that this data can be interpreted in this way. It is also necessary to have a high-resolution digital elevation model (DEM), preferably collected by a light-detection and ranging (LiDAR) remote sensing survey. LiDAR surveys have been carried out over the Nahoon Estuary as it is within the Buffalo City Municipality, although the most recently available data are from 2013. Obtaining a LiDAR survey for the Nxaxo Estuary would be a priority for future research on sea-level rise at this site. These data could also be applied to different research questions related to spatial variability in tidal inundation patterns or vegetation zonation patterns.

It is recommended that sea-level rise monitoring using the RSET approach should be scaled up to all estuaries with large salt marsh and mangrove areas in the country (e.g. Olifants, Groot Berg, Keiskamma, Mngazana). Site-specific data on sediment budgets, such as the source of sediment and the influence of water level fluctuations on sediment transport, are also required to interpret RSET results as the rate of sedimentation determines the capacity of mangroves and salt marshes to resist rising sea-levels through surface elevation gain.

7.4 Impacts of Climate Change

Mangrove range expansion in response to contemporary and predicted climate change has become an important area of research as these habitats are often characterized by unique species assemblages that replace warm temperate salt marsh habitats. This can lead to significant shifts in ecosystem services, as these vegetation types are structurally distinct (Kelleway et al., 2017a). However, mangrove range expansion is not occurring at the same rate at different range limits around the world (Hickey et al., 2017). This has been related to the importance of different environmental and physical drivers that control distribution at range limits, with northern hemisphere range limits for mangroves being controlled by freeze events (Cavanaugh et al., 2014; Osland et al., 2017b; Cavanaugh et al., 2018), while southern hemisphere limits are controlled by coastal geomorphology (Adams et al., 2004; Osland et al., 2017b; Raw et al., 2019a). This research project provided the first quantitative assessment of drivers that control the distribution of mangroves along the South African coastline (Raw et al., 2019a). It was found that the floodplain area, mean annual runoff, daily flushing rate, and the percentage of the time that the estuary remains open to the sea, are all significant predictors of mangrove area in this region. Temperature was not a significant predictor, but this was not unexpected, as mangroves that were planted beyond the natural southern distribution at the Nahoon Estuary have successfully persisted for the past 50 years (Ward and Steinke, 1982; Hoppe-Speer et al., 2015b).

Following this, the potential for mangrove range expansion along the South African coastline was predicted using a species distribution modelling approach. Latitudinal trends in mangrove establishment have been previously recorded and reviewed (Whitfield et al., 2016; Peer et al., 2018), and some predictions for future range expansion have also been made (Quisthoudt et al., 2013). However, this research project has focused on using the environmental variables that were quantitatively identified as significant drivers of mangrove distribution patterns to build predictive models. This is the first application of these statistical techniques to explain the potential for mangrove range expansion in this region. A collaboration with the Council for Scientific and Industrial Research (CSIR) has made this research possible as the environmental data for the future scenarios were obtained from the climate modelling projections provided in the CSIR Greenbook (Engelbrecht et al., 2019). Overall, the results from this research indicate that some estuaries will provide suitable habitat for mangroves under future climate change. This is possibly because increased rainfall will promote higher mean annual runoff, and this will allow some systems that are temporarily closed to have a more permanent connection to the marine environment. However, an increased intensity of rainfall events is expected to lead to more intense flooding, and this could negatively impact mangroves. This is an area that needs more research. As local coastal hydrology and geomorphology influence mangrove distribution patterns, future research should also incorporate propagule dispersal and recruitment models into these predictions.

Mangroves along the Nxaxo Estuary, Eastern Cape. Photo: J Raw

This research project has also reviewed the state of salt marsh habitats in South Africa and provided an up-to-date understanding of the patterns and processes influencing salt marshes so that an accurate prediction can be made of their responses to climate change. This status assessment was based on distributional patterns and importance of certain salt marsh habitats in relation to ecosystem services. The available information has been formed into a framework for understanding the implications of climate change on salt marsh ecosystem services. It is critical that future research predicts the combined effects of multiple stressors (storm surges, floods, droughts, and reduced river flow) in order to conserve these important habitats. Research and monitoring to understand salt marsh responses is therefore ongoing. This research has made an important global contribution as relatively little is known about salt marshes in southern Africa, as most research has focussed on mangroves.

7.5 Carbon Offset Mechanism

The final component of this research project was to use the new information generated as part of this work to provide a preliminary assessment of the viability for a carbon offset mechanism using blue carbon ecosystems in South Africa. The methodology for this approach is relatively new and some recent studies have investigated and discussed the need for a standardized framework (Steven et al., 2019; Sapkota and White, 2020). Based on the approaches recommended in the available literature at the time of this study, preliminary calculations show that blue carbon habitats in South Africa can potentially remove 10.3 million $tCO_2eq\ yr^{-1}$ and this is higher than the projected sequestration potential from terrestrial habitats (8 million $tCO_2eq\ yr^{-1}$ reported in the National Terrestrial Carbon Sink Assessment (DEA, 2014). These results are comparable to emissions reductions potential of blue carbon habitats in southeast Australia, which were reported to be 9.63 million $tCO_2eq\ yr^{-1}$ (Lewis et al., 2018). The value of blue carbon habitats was ~ R 1.2-10.6 billion per year if carbon is traded at relatively higher prices and ~ R 120-150 million per year if carbon is traded at lower carbon prices.

Based on our results, we suggest that blue carbon Payment for Ecosystem Services (PES) or offset projects will only be economically viable in South Africa, if the credits are traded at a higher price in the voluntary markets which at present are the more attainable options for blue carbon pilot projects (Ullman et al., 2013). Additional ecosystem service values should be added in this analysis in future, such as the value of mangrove coastal storm protection and other regulatory services which are still understudied in South Africa (Turpie et al., 2017). Verification of the results of this study could also provide support for PES once a globally standardized methodology for this approach is adopted. In South Africa, the development of a full carbon inventory, that includes new data collected on carbon storage in the larger subtropical mangrove systems of KwaZulu-Natal, will provide a better assessment of stocks. More robust data on sequestration rates for this region will also add a higher level of certainty to the estimates.

7.6 Final Remarks

Salt marsh, seagrass and mangrove habitats require protection as they are threatened by increasing human pressures and climate change. Priority estuaries are the Groot Berg, Knysna, Mngazana, uMlalazi, St Lucia and Kosi estuaries. The Groot Berg Estuary, with its expansive floodplain marshes, is especially unique and must be prioritised for rehabilitation and protection status. The Knysna Estuary has important large intertidal salt marsh areas and the most extensive population of the endangered seagrass *Zostera capensis*. The Mngazana Estuary supports the largest mangrove area in the temperate-subtropical transition zone and the largest red mangrove (*Rhizophora mucronata*) stand in the country. This estuary needs to be conserved and protected as a matter of priority. Estuarine lakes support the largest areas of submerged macrophytes and require special protection as they have weak resetting mechanisms and represent nutrient sinks.

We have a long history of research on estuaries in South Africa and a comprehensive program is now needed to restore the health and ecosystem services of South Africa's estuaries. A National Estuary Restoration and Research Programme (NERPP) must be implemented. This can include action research and a learning-by-doing approach in a strategic adaptive management cycle. This project has identified those mangrove and salt marsh sites for such interventions. This will address the objectives of the UN Decade of Ecosystem Restoration (2021-2030). In particular, applied solutions in the form of innovative methods for water quality improvement need to be developed so that estuary health can be restored and resilience to future climate change impacts maintained. The National Biodiversity Assessment 2018 (Van Niekerk et al., 2019) indicated that our estuaries are under severe pollution pressure and that improvement of water quality as a key intervention would lead to significant improvement in estuary health and associated benefits that society derive from them. WRC Project K5/2736 (The Blue Economy from an ecosystem perspective with a particular focus on coastal resources and communities) identified estuary restoration in South Africa as an important Blue Economy activity. A socio-ecological systems approach to restoration will be key in addressing alignment between legislation, governance, implementation and social commitment.

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APPENDIX I

A1.1 Supplementary Material for Section 3.2: Application of the Blue Carbon Protocol to Assess Variability Across the Mangrove-Salt Marsh Ecotone

Table A1.1.1 Average (± SD) soil moisture content (%) at Nahoon Estuary

	0-15 cm	**15-30 cm**	**30-50 cm**
Ecotone	43.95 ± 3.0	28.52 ± 1.87	26.1 ± 5.58
Mangrove	39.79 ± 6.1	28.47 ± 3.4	24.69 ± 3.95
Salt Marsh	36.22 ± 4.7	30.62 ± 8.32	30.02 ± 10.15

Table A1.1.2 Average (± SD) soil organic content (%) at Nahoon Estuary

	0-15 cm	**15-30 cm**	**30-50 cm**
Ecotone	10.34 ± 1.94	4.1 ± 0.48	4.84 ± 1.45
Mangrove	7.61 ± 1.95	4.8 ± 1.0	4.57 ± 1.13
Salt Marsh	7.72 ± 1.79	5.79 ± 2.34	5.02 ± 3.32

Table A1.1.3 Average (± SD) soil bulk density ($g.cm^{-3}$) at Nahoon Estuary

	0-15 cm	**15-30 cm**	**30-50 cm**
Ecotone	0.292 ± 0.032	0.428 ± 0.035	0.464 ± 0.061
Mangrove	0.323 ± 0.053	0.430 ± 0.044	0.488 ± 0.072
Salt Marsh	0.342 ± 0.077	0.393 ± 0.052	0.444 ± 0.112

Table A1.1.4 Average (± SD) soil carbon per core section ($g.cm^{-2}$) at Nahoon Estuary

	0-15 cm	**15-30 cm**	**30-50 cm**
Ecotone	0.446 ± 0.048	0.264 ± 0.040	0.435 ± 0.084
Mangrove	0.357 ± 0.056	0.306 ± 0.048	0.439 ± 0.087
Salt Marsh	0.379 ± 0.047	0.329 ± 0.093	0.389 ± 0.138

Table A1.1.5 Species occurrence (%) in vegetation quadrats at Nahoon Estuary

	Salicornia tegetaria	***Bassia diffusa***	***Triglochin striata***	***Avicennia marina***
Salt Marsh	50.16	38.5	11	0
Ecotone (Salt Marsh)	92.5	7.5	0	0
Ecotone (Mangrove)	0	0	0	100
Mangrove	0	0	0	100

Table A1.1.6 Average (± SD) mangrove density (individuals.ha^{-1}) per size category at Nahoon Estuary

	Ecotone	**Mangrove**
< 50 cm	36000 ± 2880	31467 ± 2501
51-100 cm	0	3067 ± 212
101-150 cm	0	2133 ± 273
151-200 cm	0	933 ± 105
201-500 cm	1400 ± 600	1600 ± 654
> 500 cm	200 ± 100	800 ± 113

A1.2 Supplementary Material for Section 4: Responses of Blue Carbon Habitats to Sea-Level Rise

A1.2.1 Supplementary Material for Section 4.1: First Report on Surface Elevation Change in Mangrove Estuaries of South Africa

Table A1.2.1.1 Total height (mm) and height of receiver groove (mm) from cement base on RSETs at Nahoon and Nxaxo estuaries.

	Nahoon Estuary		Nxaxo Estuary	
RSET number	**Total Height**	**Groove Height**	**Total Height**	**Groove Height**
1	121	84	109	70
2	120	86	100	60
3	133	94	91	51
4	128	85	112	74
5	122	84	130	92
6	135	97	130	90
7	131	93	132	92
8	-	-	144	105
9	140	99	106	66

* No measurements taken at Nahoon RSET 8 as the benchmark could not be located at the time of sampling

Table A1.2.1.2 Levelling results for RSETs at the Nahoon Estuary as reported by DME Geomatics.

RSET number	Height above MSL	Description
1	1.599	Stainless steel pipe in concrete
2	1.447	Stainless steel pipe in concrete
3	1.348	Stainless steel pipe in concrete
4	1.341	Stainless steel pipe in concrete
5	1.375	Stainless steel pipe in concrete
6	1.415	Stainless steel pipe in concrete
7	1.342	Stainless steel pipe in concrete
8	1.286	Stainless steel pipe in concrete
9	1.166	Stainless steel pipe in concrete
REFERENCE POINT		
BM1	1.870	12 mm Round steel peg in concrete

* A misclosure of 0.0017 m was observed and the differences between the measurement and check measurements were typically 0.0008 m.

A1.2.2 Supplementary Material for Section 4.2: Salt Marsh Surface Elevation and Sea-Level Rise at the Knysna Estuary

The Supplementary Material that appears in this section is obtained from the material provided with the paper: Raw JL, Riddin T, Wasserman J, Lehman TWK, Bornman TG, Adams JB (2020) Salt marsh elevation and responses to future sea-level rise in the Knysna Estuary. *African Journal of Aquatic Sciences* 45(1) doi: 10.2989/16085914.2019.1662763

Autoregressive Moving Average (ARIMA) Results

Table A1.2.2.1 Estimated autoregressive (AR) components for ARIMA models fit to tide gauge time series data for the Knysna Estuary over different periods. AR components were included as correlation structures in Generalized Least Squares (GLS) models to estimate the RSLR trend (slope) for each time period.

Tide Gauge Record Period 1960-2017			
ARIMA (1, 1, 0) model coefficients			
AR1	-0.211 ± 0.04		
Drift	-0.041 ± 1.98		
GLS coefficients		t	p
Intercept	2644.1 ± 359.1	7.362	< 0.05
Time	2.19 ± 0.18	12.167	<0.05
Study Period 2009-2017			
ARIMA (1, 0, 0) model coefficients			
AR1	0.133 ± 0.09		
GLS coefficients		t	p
Intercept	6579.7 ± 2802.5	2.347	< 0.05
Time	0.249 ± 1.39	0.179	> 0.05

Specification of the Sea-Level Affecting Marshes Model for Thesen's Island

1) Background to SLAMM

The Sea-Level Affecting Marshes Model (SLAMM) allows for the simulation of the dominant processes that determine whether or not coastal wetlands will persist under long-term sea-level rise. This is achieved by using a complex decision tree that incorporates spatial grids and the pre-defined qualitative relationships of different coastal habitat classes to inundation, erosion, overwash, saturation, salinity, and accretion (Clough et al., 2016). The software provides map distributions of coastal habitats as predicted under the conditions for sea-level rise specified in the model. SLAMM and all supporting materials are freely available at https://github.com/WarrenPinnacle/SLAMM6.7.

SLAMM considers each grid cell within the study area individually by calculating the relative sea-level change at each time step. This relative sea-level change is determined by the sum of the historic eustatic sea-level trend, the site-specific rate of change of elevation and the accelerated sea-level rise rate specified in the model scenario. As sea-level rise is offset by sedimentation and accretion, these parameters can also be specified. At each time step there is a fractional conversion from one habitat class to another which is determined by the relative elevation change as sea-level increases. The specific mathematical models that define these changes are detailed in the SLAMM technical documentation and are not provided here (Clough et al., 2016).

2) SLAMM inputs for Thesen's Island focus area

In this study, a focus area at Thesen's Island was selected for the SLAMM approach (Figure A1.2.2.1). This focus area was selected based on the suitability of the available data which included an RSET benchmark in the lower intertidal salt marsh as well as a tide gauge located on the edge of the developed area. This focus area was also of interest as it is representative of other areas within the Knysna Estuary where residential properties are directly adjacent to the salt marsh-terrestrial boundary.

Figure A1.2.2.1 Focus area for the SLAMM carried out at Thesen's Island indicated by the white square on Google Earth imagery. Photograph taken in 2018 at the location of the Site 2 RSET facing north.

All areas expected to be influenced by sea-level rise in this focus area were included in the model. These areas were identified based on the extent of the estuarine functional zone (EFZ) as well as any adjacent low-lying land that may not have been classified as estuarine habitat. The DEM was derived from 2013 LiDAR data which was collected at 5 m spatial resolution and 0.2 m vertical accuracy for the Environmental Affairs and Development Planning departments of the Western Cape Government. However, the SLAMM is limited by the DEM generated from LiDAR data. As a result, most of the open water areas are not included in the SLAMM, as they are returned as “no data” values in LiDAR surveys due to reflectance off the surface of the water. The classification of habitats for SLAMM is described in the Methods section.

As the SLAMM is site-specific, inputs used for this study were derived from a number of different sources (Table A1.2.2.2). The historic sea-level trend was calculated from 1960-2017 in this study from the Knysna tide gauge data as 2.19 ± 1.3 $mm.yr^{-1}$ using the autoregressive integrated moving average (ARIMA) approach for time series analysis (Hyndman and Khandakar, 2008; Hyndman et al., 2018). This approach was previously used to estimate sea-level trends for Port Elizabeth (Bornman et al., 2016) and is more robust than a simple linear regression for analysis of time series data. However, the trend value obtained from the ARIMA approach is higher than what was previously reported by Mather et al. (2009) where the observed annual sea-level trend using monthly data for Knysna from 1960-2007 was calculated as 1.27 ± 0.50 $mm.yr^{-1}$ from the slope of a linear regression. In their paper, Mather et al. (2009) provide a barometric correction of annual sea-level trends which then results in a 1.60 ± 0.76 $mm.yr^{-1}$ sea-level trend for the Knysna tide gauge. The annual eustatic sea-level trend for Knysna using monthly tide gauge data which accounts for vertical crustal movement and barometric changes is then reported as 2.14 $mm.yr^{-1}$ (Mather et al., 2009). This value is within the error associated with our value calculated using the ARIMA approach. These values were used accordingly in the SLAMM analysis.

The generalized accretion feedback model was used in this study. This model applies a simple constant accretion response. The accretion rate at the RSET was used as the input for the “Regularly Flooded Marsh” and an accretion rate of zero was used for the upland developed areas. Accretion rates for the intermediate salt marsh elevations were calculated by extrapolation from their elevations on the DEM.

The digital elevation model (DEM) was used to derive other site-specific parameters required for SLAMM in the Thesen’s Island focus area. The salt elevation, as defined by SLAMM, is a measure of the salt boundary which is the height beyond which there is no longer a saline influence. For this study site, the salt elevation was determined by identifying the boundary of the salt marsh vegetation on the vegetation map and then extracting the elevation height in meters from the DEM at the same point. All other inputs to the site parameters were derived from existing literature.

Table A1.2.2.2 Input parameters for SLAMM at Thesen's Island in the Knysna Estuary

Parameter	Value	Information source
Historic sea-level trend ($mm.yr^{-1}$)	1.6	(Mather et al., 2009)
Historic eustatic sea-level trend ($mm.yr^{-1}$)	2.14	(Mather et al., 2009)
Mean Tide Level (m)	0.2	Derived from DEM in this study
Diurnal tide range (m)	1.8	(Maree, 2000)
Salt elevation (m above MTL)	1.36	Derived from DEM in this study
Regularly flooded marsh accretion ($mm.yr^{-1}$)	3.16	Calculated from RSET data 2009-2018 (this study)
Irregularly flooded marsh accretion ($mm.yr^{-1}$)	1.46	Estimated from linear extrapolation between regularly flooded marsh accretion and terrestrial boundary (this study)
30-day inundation height (m above MTL)	1	(Maree, 2000)
60-day inundation height (m above MTL)	1.3	(Maree, 2000)
90-day inundation height (m above MTL)	1.3	(Maree, 2000)
10-year storm height (m above MTL)	2	(Fraser, 2018)
100-year storm height (m above MTL)	2.2	(Fraser, 2018)

3) Sea-level rise scenarios

SLAMM allows the specification of sea-level rise scenarios based on those provided by the 2001 Intergovernmental Panel on Climate Change (IPCC) Special Report on Emissions Scenarios. However, following criticism of the SRES system for not incorporating controls on emissions, the IPCC developed a new set of metrics – the Representative Concentration Pathways (RCPs) in the 2014 Assessment Report.

A comparison of SLR scenarios predicted under the SRES and RCP frameworks is provided in the IPCC 2013 Annex II. With respect to 1986-2005, the median [and range] values for global mean SLR (m) are given as 0.6 [0.42-0.8] under the A1B scenario, 0.44 [0.28-0.61] under RCP2.6, 0.53 [0.36-0.71] under RCP4.5, 0.55 [0.38-0.73] under RCP6.0, and 0.74 [0.53-0.98] under RCP8.5.

The most recent version of SLAMM also allows the specification of any sea-level rise rate (in meters) to 2100. The aim of this study was to use SLAMM to assess coastal squeeze of salt marsh habitats at the Thesen's Island focus area, and not to compare habitat responses under different sea-level rise scenarios. Previously, sea-level rise projections of 0.4 m by 2050 and 1.6 m by 2100 were used to assess sea-level vulnerability in coastal towns along the South African coastline using simple inundation models (Mcleod et al., 2010; Fitchett et al., 2016). These models assumed an average rate of change of 0.3 $mm.yr^{-1}$ along the south coast of South Africa (Fitchett et al., 2016). To specify a set sea-level rise by 2100, SLAMM scales the A1B scenario to estimate sea-level rise over time that will result in the specified height by the end of the simulation. The relative rate of sea-level rise is therefore the same between the A1B scenario and the 1, 1.5 and 2 m scenarios but the extent of sea-level rise by 2100 varies (Clough et al., 2016). For this study the A1B maximum scenario was applied in SLAMM for the Thesen's Island focus area so that the maximum eustatic sea-level rise by 2100 is 0.7 m (IPCC, 2001). Realistically, this is an intermediate scenario as follow up studies suggest the IPCC rates are underestimated and that sea-level is actually projected to range from 0.9 to 1.3 m by 2100 under the A1B scenario (Grinsted et al., 2010). The 0.7 m increase by 2100 is however suitable as an intermediate scenario for the focus area at Thesen's Island as surveyor reports have indicated that the best estimate of projected sea-level rise for the Knysna Estuary does not exceed 1 m by 2100 (Barwell, 2016; Fraser, 2018).

4) Elevation data inputs

The high-resolution digital elevation models derived from LiDAR are an essential component to SLAMM. This information is used to demarcate saltwater intrusion and inundation frequencies for coastal habitats can be determined when combined with tidal data. The elevation data are used to determine the elevation ranges within which different habitats occur so that the model can predict shifts in relation to inundation (Clough et al., 2016).

SLAMM includes default elevation ranges at which specific habitat types are expected to occur within the tidal frame. However, these can be individually edited to suit the specific site that is being modelled. For this study the elevation ranges of the different habitats were derived directly from the DEM by extracting the elevation at points that coincided with a change in salt marsh species dominance on the vegetation map. Table A1.2.2.3 shows these extracted values that were used in the SLAMM for the Thesen's Island focus area.

To visualize the elevation distribution of the different habitats, SLAMM provides a histogram output that can be used to verify the conceptual model of habitat distribution along the elevation gradients provided. The elevation histogram for this study is shown in Figure A1.2.2.2.

Table A1.2.2.3. Minimum and maximum elevations for habitat types used in the SLAMM for Thesen's Island focus area in the Knysna Estuary.

SLAMM Category	**Minimum elevation (m)**	**Maximum elevation (m)**
Developed Dry Land	0.6	6.2
Irregularly Flooded Marsh	0.3	1.6
Transitional Salt Marsh	0.4	1.3
Regularly Flooded Marsh	0	1
Tidal Flat	-1	0

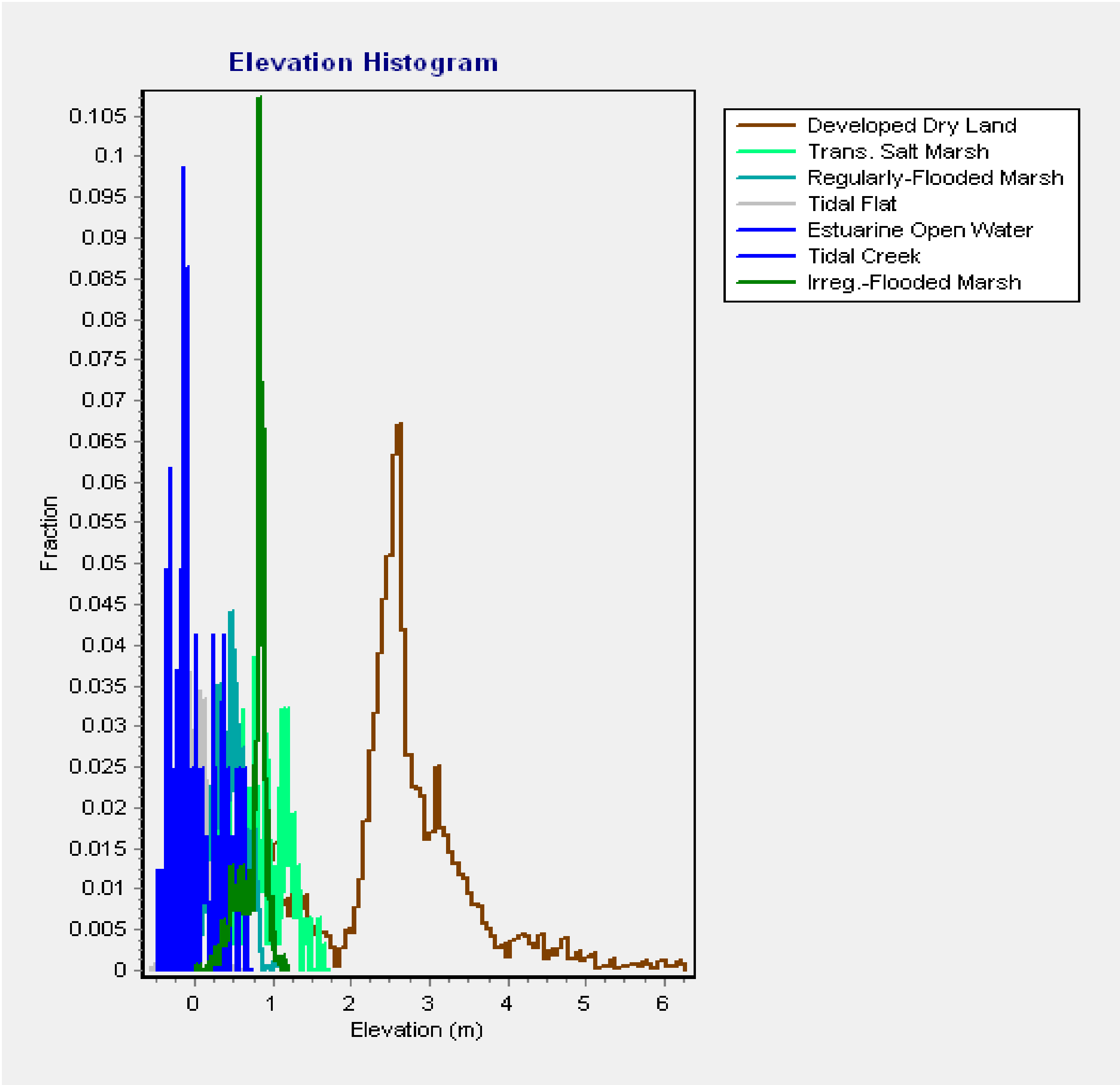

Figure A1.2.2.2. Elevation histogram for habitat types present within the focus area for the SLAMM at Thesen's Island. Produced by SLAMM v 6.7

Finally, SLAMM also allows the simulation to be run under scenarios with optional protection for developed dry land. Areas that are protected in the simulation are not converted into other habitat types over time even if the elevation and inundation are suitable (Clough et al., 2016). This provides a simulation of coastal squeeze as salt marsh is unable to migrate landward due to these protected barriers. In this study, both the dry land protection scenarios were run simultaneously to assess coastal squeeze and determine whether allowing dry land to convert to salt marsh over time does facilitate landward expansion.

A1.3 Supplementary Material for Section 5.1: Drivers of Mangrove Distribution at the High Energy, Wave-Dominated, Southern African Range Limit

The Supplementary Material that appears in this section is obtained from the material provided with the paper: Raw JL, Godbold, JA, Van Niekerk L, Adams JB (2019) Drivers of mangrove distribution at the high-energy, wave-dominated southern African range limit. *Estuarine, Coastal and Shelf Science* 226: 106296.

Detailed description of the SEM development process.

A Structural Equation Model (SEM) is used to develop and evaluate models to represent underlying causal processes. Grace et al. (2012) lists three important components of a network model such as SEM: 1) a definition of graphical relations, including the nodes and links in the graph; 2) a vector of observed variables used to specify the graphical model; 3) the statistical/mathematical functions to convey the information flow in the network. The application of these components to the study on mangrove biogeographic patterns along the South African coastline is provided below.

The conceptual model (Section 5.1 – Methods of this report) represents theoretical expectations and it provides a summarization of theoretical concepts underlying the model development. It is noncommittal, as any potential theoretical relationships and potential observable variables can be included, regardless of whether there are available data. The potential observed variables included in the conceptual model were defined based on *a priori* knowledge of ecological theory.

A causal diagram is developed following the metamodel. The causal diagram summarizes the connections among variables that could be included in the SEM. The links in a causal diagram represent hypothesized causal connections among the variables (Grace et al., 2012). Figure A1.3.1 shows the causal diagram generated for this study which includes nodes that were not used to develop the final SEM. These nodes represent areas where additional data could be collected and added to improve and refine the model in the future.

As the links in the causal diagram represent hypothesized causal relationships, these should be verified and based on sound theoretical principles as well as existing empirical evidence. For every link in the causal diagram it is necessary to define a causal assumption or hypothesis. This is presented in Table A1.3.1.

The causal diagram can then be used as the basis to specify the SEM by developing the models that will be used in the analysis. The focus of the SEM for this study was to investigate the causal network of connections between climatic and geomorphological characteristics of estuaries to explain the observed biogeographical patterns of mangroves along the South Africa coastline. Understanding linkages between climatic and geomorphological characteristics was also important. Mangrove area was considered as the most important observed biotic response.

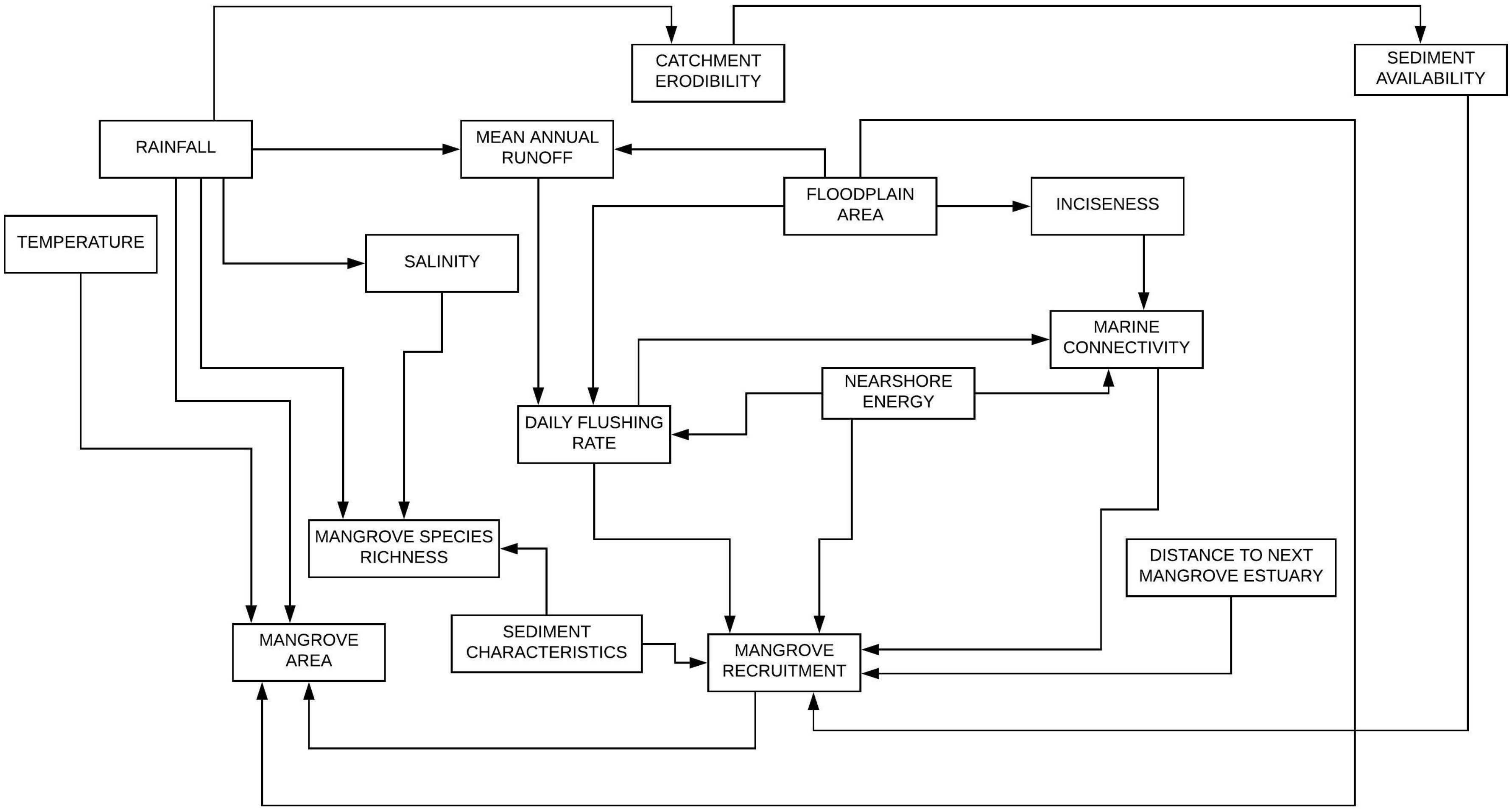

Figure A1.3.1. Initial causal diagram for hypothesized causal relationships between variables that influence mangrove biogeographic patterns along the South African coastline.

Table A1.3.1. Causal links and hypotheses represented in the causal diagram of observable variables that potentially influence mangrove distribution patterns along the South African coastline.

Conceptual Grouping	Causal Link Relationship	Hypothesis	References
Climate	*Temperature → Mangrove Area*	Higher temperature will be correlated with a greater mangrove area.	(Clough, 1992; Gilman et al., 2008; Soares et al., 2012; Quisthoudt et al., 2013; Hoppe-Speer et al., 2015a)
	Rainfall →Mangrove Area	Higher rainfall will be correlated with greater mangrove area.	(Clough, 1992; Rajkaran and Adams, 2011; Quisthoudt et al., 2012; Soares et al., 2012)
	Rainfall → Mangrove Species Richness	Higher rainfall will be correlated with greater mangrove species richness.	(Smith and Duke, 1987; Ball, 1988; Rajkaran and Adams, 2011)
	Rainfall → Catchment Erodibility	Higher rainfall will be correlated with higher catchment erodibility.	(Langbein and Schumm, 1958; Wilson, 1973; López-Tarazón et al., 2009; Buendia et al., 2016)
Geomorphology	*Floodplain Area → Mangrove Area*	Greater floodplain area will be correlated with greater mangrove area.	(Woodroffe, 1992, 1995; Adams et al., 2004; Adame et al., 2010; Rajkaran and Adams, 2011; Yang et al., 2014; Peer et al., 2018)
	Floodplain Area → Daily Flushing Rate	Floodplain size will be negatively correlated with daily flushing rate.	(Heap et al., 2004; Opperman et al., 2010; Whitfield et al., 2012; Rolls et al., 2012)
	Floodplain Area → Inciseness	Inciseness is a ratio of floodplain area to open-water area.	(Nichol, 1991; Dalrymple et al., 1992; Colloty et al., 2002; Skilbeck et al., 2017)
	Catchment Erodibility → Sediment Availability	Higher erodibility will be correlated with higher sediment availability.	(Woodroffe, 1992; Kingsford, 2001; Thrush et al., 2004; le Roux et al., 2008; Weston, 2014)
	Sediment Availability → Mangrove Recruitment	Higher sediment availability will be correlated with higher mangrove recruitment.	(McKee, 1995; Lovelock et al., 2007; Rajkaran et al., 2009; Mudd et al., 2010; Macamo et al., 2016)

Abiotic Features	*Salinity → Mangrove Species Richness*	Lower salinity will be correlated with higher mangrove species richness.	(Smith and Duke,1987; Ball, 1998; Taylor et al., 2006; Rajkaran et al., 2009; Naidoo, 2016)
	Salinity → Mangrove Area	Hypersaline conditions will be negatively correlated with mangrove area.	(Taylor et al., 2006; Rajkaran et al., 2009; Hoppe-Speer et al., 2011; Naidoo, 2016; Peer et al., 2018)
	Mean Annual Runoff → Daily Flushing Rate	Higher mean annual runoff will be correlated with higher daily flushing rate.	(Hagy et al., 2000; Snow et al., 2000; Scharler and Baird, 2005; Malhadas et al., 2010; Van Niekerk et al., 2013)
	Sediment Characteristics → Mangrove Recruitment	Sediment characteristics (moisture content, particle size, pH, redox potential) will influence mangrove recruitment.	(Clarke and Allaway, 1993; Krauss et al., 2008; Rajkaran and Adams, 2011, 2012; Hoppe-Speer et al., 2013; Yang et al., 2014; Macamo et al., 2016; Geldenhuys et al., 2016)
	Sediment Characteristics → Mangrove Species Richness	Sediment characteristics (moisture content, particle size, pH, redox potential) will influence mangrove species richness.	(Ball, 1998; Rajkaran and Adams, 2011, 2012; Urrego et al., 2014; Atwell et al., 2016)
Physical Features	*Distance to next mangrove estuary → Mangrove Recruitment*	Shorter distance to the next northern estuary with mangroves is correlated with greater mangrove recruitment.	(Clarke, 1993; Steinke and Ward, 2003; Nettel and Dodd, 2007; De Ryck et al., 2016; Lo et al., 2014)
	Nearshore energy (surfzone width) → Marine Connectivity	Higher nearshore energy will be correlated with lower mouth opening frequency.	(Dalrymple et al., 1992; Cooper, 2001; Heap et al., 2004; Van Niekerk et al., 2013)
	Nearshore energy → Daily Flushing Rate	Higher nearshore energy will be negatively correlated with daily flushing rate.	(Dalrymple et al., 1992; Kench, 1999; Heap et al., 2004; Malhadas et al., 2010)
	Nearshore energy → Mangrove Recruitment	Higher nearshore energy will be negatively correlated with mangrove recruitment.	(Riley and Kent, 1999; Steinke and Ward, 2003; Kamali and Hashim, 2011; De Ryck et al., 2016; Flores-de-Santiago et al., 2017)
	Daily Flushing Rate → Marine Connectivity	Daily flushing rate will be correlated with marine connectivity.	(Kench, 1999; Monsen et al., 2002; Van Niekerk et al., 2013)
	Daily Flushing Rate → Mangrove Recruitment	Daily flushing rate will be negatively correlated with mangrove recruitment.	(Clarke and Myerscough, 1991; Lewis, 2005; Van der Stocken et al., 2015;

		Hoppe-Speer et al., 2015; Balke et al., 2015; Asbridge et al., 2016)
Marine Connectivity → Mangrove Recruitment	Permanent marine connectivity will be correlated with higher mangrove propagule recruitment and successful establishment	(McKee, 1995; Colloty et al., 2002; Steinke and Ward, 2003; Krauss et al., 2008; Hoppe-Speer et al., 2013; De Ryck et al., 2016; Macamo et al., 2016)
Marine Connectivity → Mangrove Area	Restricted marine connectivity will be negatively correlated with mangrove area due to inundation stress.	(Taylor et al., 2006; Hoppe-Speer et al., 2011, 2013; Rajkaran and Adams, 2011; Yang et al., 2014; Adams and Human, 2016; Mbense et al., 2016; Peer et al., 2018)

The availability of data that can be used in SEM analyses is often a limiting factor and this was also the case for this study. Although mangrove recruitment was identified as a response variable in the causal diagram, there were no quantitative data for this variable for every estuary in this region. Instead, mangrove area was used as the most important response variable as it is expected to be correlated with recruitment success over time. Data on the characteristics of the sediment, such as moisture content, pH, redox potential and particle size, were also not available for each mangrove estuary. Sediment characteristics were therefore not included in the model. As “Nearshore Energy” is difficult to quantify and not currently available at the national scale, Surfzone Width (m) was used as a proxy. Once the availability of the data for the different variables had been determined, the units of measurement, type, and distribution were recorded (Section 5.1 – Methods). An initial SEM that shows the linkages between these variables (Figure A1.3.2).

The linkages represented in the initial SEM were then defined as described in the Methods of Section 5.1 of this report and are presented in Table A1.3.2. Statistical models were not specified for response variables in categorical form (Salinity Category, Erodibility Index) as the piecewise estimation method (Lefcheck, 2016) does not support multinomial regression models. In cases where the predictor variables were correlated or representative of the same net effect (e.g. Rainfall and Mean Annual Runoff are both indicators of freshwater input), the model defining the response variable was run with each predictor separately and the Akaike Information Criterion (AIC) was used to select the optimal model. To reduce complexity and prevent over-fitting, not all of the hypothesized links from the causal diagram were included to define the response of ‘Mangrove Area’. The optimal model was selected by dropping individual predictors and comparing the fit using AIC while also assessing whether the coefficients of the predictors reflected the hypothesized links in a way that was scientifically sound. Variables dropped included Erodibility, Distance to next mangrove estuary, and Inciseness.

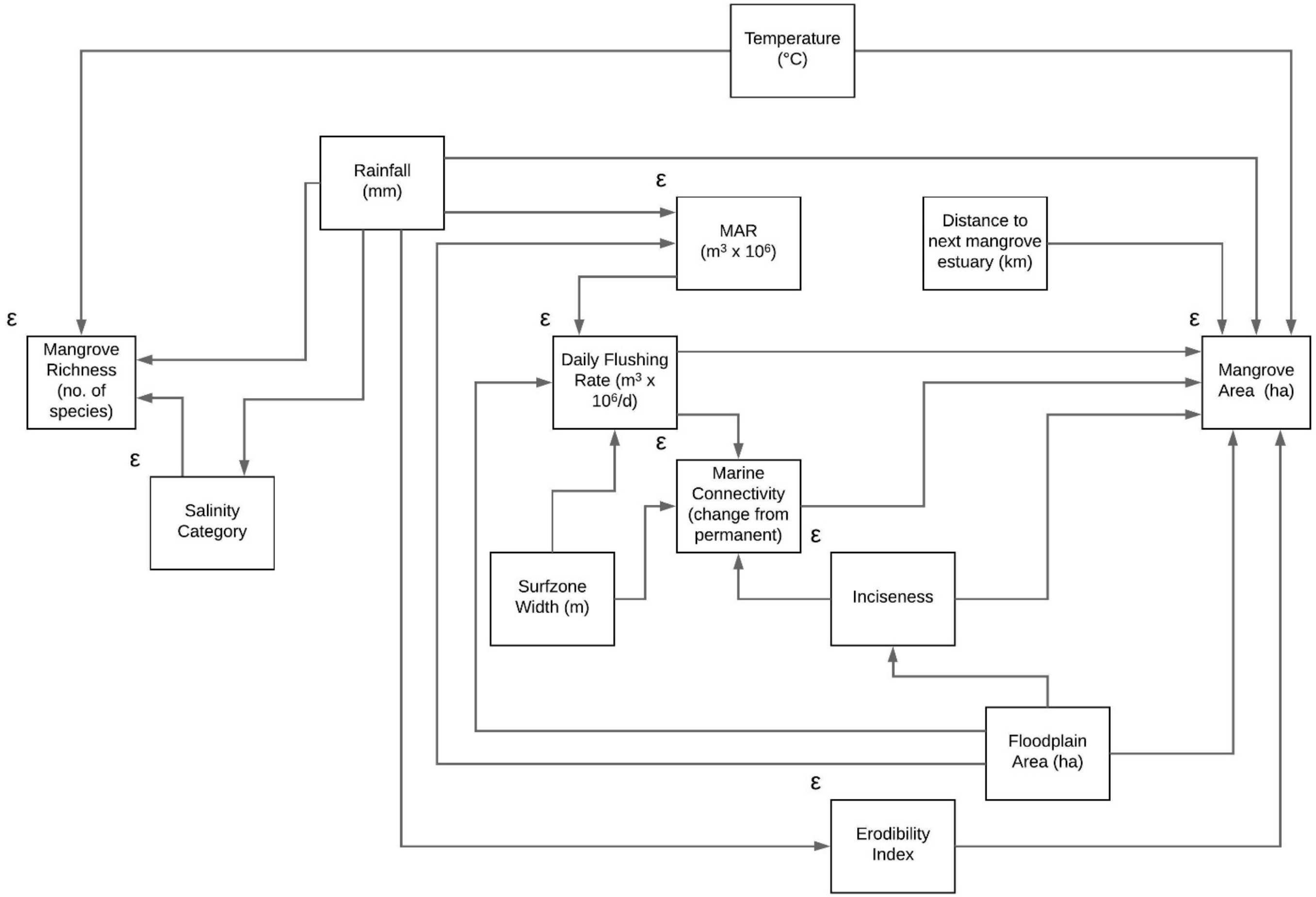

Figure A1.3.2. Initial Structural Equation Model (SEM) showing specific variables with sufficient data and their hypothesized linkages to be defined using statistical models. Epsilons represent error terms which indicate the influence of factors that are uncorrelated with the predictors of a variable.

Table A1.3.2. Initial models specified based on the *a priori* hypothesized links describing the causative relationships between observed variables.

Model number	Response variable	Predictor variables	Statistical model form	Variance structure
(1)	Inciseness	Floodplain Area	Linear regression	N/A
(2)	Mean Annual Runoff	Rainfall	Generalized least squares	Power of covariate Power (Rainfall) = 2.430
(3)	Daily Flushing Rate	Floodplain Area, Mean Annual Runoff	Generalized least squares	Constant + power of covariate Constant = 9.65×10^{-6} Power (MAR) = -9.86×10^{-6}
(4)	Inlet Stability	Daily Flushing Rate, Inciseness, Surfzone Width	Logistic regression (binomial)	N/A
(5)	Mangrove Area	Temperature, Daily Flushing Rate, Floodplain Area, Inlet Stability	Generalized least squares	Combined power of covariates (Temperature, Floodplain Area, DFR) Power (Temperature) = 5.841 Power (Floodplain Area) = 5.854 Power (DFR) = -0.303

The piecewise estimation method (Lefcheck, 2016) was used in this study. This allows for models with different functional forms to be incorporated into the SEM. Although the piecewise estimation approach can be used with small sample sizes (Shipley, 2000), it is still recommended that the ratio of the total number of samples to the number of variables should not be below 5 (Grace et al., 2015). The models defining Inciseness and Mean Annual Runoff were insignificant and therefore not included in the final SEM. Table A1.3.3 shows the Fishers goodness of fit for the initial SEM and the significant independent claims identified by D-separation after specification. This is an indication of which variables can be added to the specified models in order to improve their fit.

Table A1.3.3. Goodness of fit of the initial SEM and significant independent claims for the specified models.

Goodness of Fit					
Fisher C	**df**	**p-value**			
22.71	12	0.03			
D-separation					
Independent Claim	**Estimate**	**Std Error**	**df**	**Critical value**	**p-value**
DFR ~ ADLT	0.089	0.098	27	0.914	0.370
DFR ~ S_WIDTH	-3.14×10^{-3}	9.54×10^{-4}	27	-3.325	0.003
INLET ~ ADLT	0.415	0.580	23	0.717	0.473
INLET ~ FP_AREA	8.05×10^{-5}	2.19×10^{-4}	23	0.366	0.714
INLET ~ MAR	1.56×10^{-3}	2.62×10^{-3}	23	0.594	0.553
MN_AREA ~ S_WIDTH	0.031	0.015	27	2.016	0.057

Following the assessment of the independent claims, the models were re-specified to include additional significant variables only if the relationship was logically sound. For example, Temperature (ADLT) was not included as a predictor of Daily Flushing Rate (DFR) as these variables are most likely spatially correlated. Estuaries occurring in warmer subtropical areas also receive more rainfall and runoff and therefore have larger flushing rates.

The effect of the re-specified model for Daily Flushing Rate on the fit of the SEM was then checked. Adding the re-specified model for Daily Flushing Rate improved the fit of the SEM (Fisher C = 13.63, df = 10, p = 0.191).

Table A1.3.4 provides a summary of the three models fitted to the final SEM. The significance of each predictor variable in the models was determined through appropriate comparisons between the full and nested models.

Table A1.3.4. Significance of individual terms (indicated by asterisks) for statistical models fitted into the final Structural Equation Model (SEM). Degrees of freedom indicate difference between full and nested models for each predictor variable.

Model 1: Daily Flushing Rate				
Model form	**Predictor variable**	**Likelihood ratio**	**df**	**p-value**
Generalized	Floodplain Area	8.7883	1	0.003*
Least Squares	Mean Annual Runoff	33.7180	1	0.0001*
	Surfzone Width	10.5748	1	0.0011
Model 2: Inlet Stability				
Model form	**Predictor variable**	**Deviance**	**df**	**p-value (χ^2)**
Logistic	Daily Flushing Rate	-3.727	-1	0.0535
Regression	Surfzone Width	-1.4832	-1	0.2233
Model 3: Mangrove Area				
Model form	**Predictor variable**	**Likelihood ratio**	**df**	**p-value**
Generalized	Mean Annual Runoff	7.7950	1	0.0052*
Least Squares	Temperature	0.0055	1	0.9407
	Floodplain Area	17.6137	1	0.0001*
	Inlet Stability	12.9127	1	< 0.0001*
	Daily Flushing Rate	11.7112	1	< 0.0001*

APPENDIX II

The following postgraduate student research projects were carried out within this WRC Project (K5/2769). Components of these research projects have been incorporated into this report.

	Name and Surname	Student Number	Degree	Academic Supervisors	Project title	Degree Year/Status
2017	Sinenjongo Gcina	212250515	BSc Hons	JB Adams SP Mbense	Growth and survival of the white mangrove, *Avicennia marina*.	Completed
	Jaime Johnson	212284975	MSc	JB Adams, JL Raw	First report of carbon storage in a warm-temperate mangrove forest in South Africa.	Year 1
	Sinegugu Mbense	210235438	PhD	JB Adams, A Rajkaran	Blue carbon habitats in South African estuaries, ecosystem services and responses to global change.	Year 1
	Jacqueline Raw	213476967	Post-doc	JB Adams	Responses of South African mangroves and salt marshes to contemporary global change.	N/A
2018	Corianna Julie	215120566	BSc Hons	JB Adams, JL Raw	Carbon storage across a salt marsh-mangrove ecotone (A comparative study at the Nxaxo and Nahoon estuaries).	Completed
	Tevan Lehman	214266001	BSc Hons	JB Adams, JL Raw	Evaluating changes in surface elevation, sediment characteristics and species composition for lower intertidal salt marsh in the Knysna Estuary, South Africa.	Completed
	Jaime Johnson	212284975	MSc	JB Adams, JL Raw	First report of carbon storage in a warm-temperate mangrove forest in South Africa.	Completed
	Sinegugu Mbense	210235438	PhD	JB Adams, A Rajkaran	Blue carbon habitats in South African estuaries, ecosystem services and responses to global change.	Year 2
	Jacqueline Raw	213476967	Post-doc	JB Adams	Responses of South African mangroves and salt marshes to contemporary global change.	N/A

	Name and Surname	Student Number	Degree	Academic Supervisors	Project title	Degree Year/Status
2019	Corianna Julie	215120566	MSc	JB Adams, JL Raw	Mangrove responses to sedimentation and water level fluctuations: A comparison between a tidal and a non-tidal estuary	Year 1
	Sinegugu Mbense	210235438	PhD	JB Adams, A Rajkaran	Blue carbon habitats in South African estuaries, ecosystem services and responses to global change.	Year 3
	Jacqueline Raw	213476967	Post-doc	JB Adams	Responses of South African mangroves and salt marshes to contemporary global change.	N/A

APPENDIX III

The following research outputs were produced as part of this WRC Project (K5/2769).

Published journal articles:

Johnson JL, Raw JL, Adams JB. 2020. First report on carbon storage in warm-temperate mangroves of South Africa. *Estuarine Coastal and Shelf Science* 235: 106566.

Riddin, T and JB Adams. 2019. Water level fluctuations and phenological responses in a salt marsh succulent. *Aquatic Botany* 153:58-66.

Raw JL, Godbold JA, Van Niekerk L, Adams JB. 2019 Drivers of mangrove distribution at a high-energy, wave-dominated, southern distribution limit. *Estuarine Coastal and Shelf Science* 226: 106296.

Raw JL, Julie CL, Adams JB. 2019. A comparison of soil carbon pools across a mangrove-salt marsh ecotone at the southern African warm-temperate range limit. *South African Journal of Botany* 107, 301-307.

Raw JL, Riddin T, Wasserman J, Lehman TWK, Bornman TG, Adams JB. 2020. Salt marsh surface elevation and sea-level rise at the Knysna Estuary. *African Journal of Aquatic Science* 45(1) doi: 10.2989/16085914.2019.1662763

Rogers K, Kelleway JJ, Saintilan N, Megonigal P, Adams JB, Holmquist JR, Lu M, Schile-Beers L, Zawadzki A, Mazumder D, Woodroffe CD. 2019 Wetland carbon storage controlled by millennial-scale variation in relative sea-level rise. *Nature* 567: 91-95.

Articles accepted for publication:

Adams, JB. Salt marsh at the tip of Africa: patterns, processes and changes in ecosystem services in response to climate change. *Estuarine, Coastal & Shelf Science*

Wasserman J, Claassen L, Adams JB. Mapping subtidal estuarine habitats using a remotely operated underwater vehicle (ROV). *African Journal of Marine Science*

Book chapters:

Mbense SP, Johnson JL, Raw JL, Rajkaran A, Adams JB. Blue carbon storage comparing mangroves with salt marsh and seagrass habitats at a warm temperate continental limit. In “Dynamic Sedimentary Environment of Mangrove Coasts”.

Tabot, T and J.B. Adams. 2019. South African salt marshes: Ecophysiology and ecology in the context of climate change (Chapter 5). In: Halophytes and Climate Change. Adaptive Mechanisms and Potential Uses. Editors: H Hassanuzzaman, S Shabala & M Fujita. CABI. 416 pg https://www.cabi.org/bookshop/book/9781786394330

Popular articles:

Human L, Raw JL, Els J, Mbense S, Adams JB, Bornman TG "Blue Carbon Ecosystems, the SAEON and Nelson Mandela University initiative" – South African Environmental Observation Network (SAEON) Newsletter, April 2018.

Raw JL "Nahoon mangroves in national sea-level rise program" – Tiptol Newsletter of Birdlife Border, Affiliate of Birdlife South Africa, Issue 124, October 2017.

Raw JL "Mangroves monitored for sea-level rise" – Nahooner News, Newsletter of the Friends of the Nahoon Estuary Nature Reserve, Issue 31, February 2018.

Raw JL "Update on Nahoon Mangrove Research" – Nahooner News, Newsletter of the Friends of the Nahoon Estuary Nature Reserve, Issue 32, February 2019.

Conference presentations:

2017

Mbense S, Adams JB, Rajkaran A "Ecosystem services and blue carbon habitats in South African estuaries" – 2017 South African Marine Science Symposium (SAMSS) in Port Elizabeth, South Africa (4-7 July 2017).

Mbense S, Adams JB, Rajkaran A "Ecosystem services and blue carbon habitats in South African estuaries" – 10th Western Indian Ocean Marine Science Association (WIOMSA) Scientific Symposium in Dar es Salaam, Tanzania (30 October-4 November 2017).

Raw JL, Adams JB, Bornman TG "Measuring the responses of South African mangroves to sea-level rise: Better late than never" – 2017 South African Marine Science Symposium (SAMSS) in Port Elizabeth, South Africa (4-7 July 2017).

2018

Adams JB “Salt marsh in South Africa: patterns, processes and predicted responses to climate change.” – 57th Estuarine and Coastal Science Association (ECSA) Conference in Perth, Australia (3-6 September 2018), oral presentation.

Adams JB “Status, threats and management situation of mangrove ecosystems in South Africa.” – Keynote presentation invite for ZMT Leibniz Centre for Tropical Marine Research workshop on mangroves (Durban, April 2018).

Adams JB “The response of South Africa’s salt marshes to global change: implications for ecosystem service delivery.” – 2018 South African Society of Aquatic Sciences (SASAqS) Congress in Cape St Francis, South Africa (24-28 June 2018).

Adams JB, Van Niekerk L, Taljaard S “Bridging the science-policy-practice divide for the management of estuaries”. – 57th Estuarine and Coastal Science Association (ECSA) Conference in Perth, Australia (3-6 September 2018), poster presentation.

Johnson JL, Adams JB, Raw JL "First report on carbon storage in warm-temperate mangroves of South Africa" – 57th Estuarine and Coastal Science Association (ECSA) Conference in Perth, Australia (3-6 September 2018).

Raw JL, Adams JB, Van Niekerk L, Godbold JA "Drivers of mangrove distribution along the high-energy coastline of South Africa" – 57th Estuarine and Coastal Science Association (ECSA) Conference in Perth, Australia (3-6 September 2018).

2019

Adams JB “An assessment of restoration efforts in South African estuaries” Society for Ecological Restoration 8th World Conference on Ecological Restoration in Cape Town, South Africa (24-28 September 2019).

Adams JB “Key principles for determining environmental flow requirements in closed estuaries” Coastal and Estuarine Research Federation 25th biennial conference in Mobile, Alabama, USA (3-7 November 2019).

Adams JB “Environmental flow requirements for a diversity of estuary types” Plenary address at the Australian Freshwater Sciences Association in Geelong, Victoria, Australia (1-4 December 2019).

Adams JB, Rajkaran A “Status, threats and management of mangroves at a southern distributional limit” MMM5 (5th Mangrove, Macrobenthos & Management Meeting) in Singapore (1-5 July 2019).

Mbense SP, Adams JB, Rajkaran A. “The quantification of blue carbon in a South African Warm Temperate Estuary” MMM5 (5th Mangrove, Macrobenthos & Management Meeting) in Singapore (1-5 July 2019).

Raw JL, Mbense SP, Van Niekerk L, Adams JB “Potential for mangrove range expansion in South Africa and links to ecosystem service delivery” MMM5 (5th Mangrove, Macrobenthos & Management Meeting) in Singapore (1-5 July 2019).

Inputs made to reports using project data:

Van Niekerk, L., Adams, J.B., Lamberth, S.J., MacKay, C.F., Taljaard, S., Turpie, J.K., Weerts S.P. & Raimondo, D.C., 2019 (eds). South African National Biodiversity Assessment 2018: Technical Report. Volume 3: Estuarine Realm. CSIR report number CSIR/SPLA/EM/EXP/2019/0062/A. South African National Biodiversity Institute,

Pretoria. Report Number: SANBI/NAT/NBA2018/2019/Vol3/A. http://hdl.handle.net/20.500.12143/6373 (Unproofed version).

Workshops and symposiums:

Snow, GC, JB Adams and B Snow. 2019. Swartkops Estuary Research Symposium – Improving estuary health for the delivery of multiple ecosystem services Issue 224. pp: 29-32. April 2019. ISSN 03700-9026

Estuaries: science, management & implications for the future. Institute for Coastal and Marine Research. Discovering Diversity Symposium.

APPENDIX IV

All data that were generated as part of this research project are available upon request from the project contributors.

The following datasets are available

- ASCII files for DEM, vegetation, and slope used as inputs to the SLAMM, ASCII file outputs of the SLAMM-predicted salt marsh distributions under the modelling scenarios.
- Calculated environmental data for South African estuaries used for projected models.
- Calculated soil carbon per 0.5 m core for each core samples
- Data used to calculate blue carbon offset viability
- Environmental data for South African estuaries (CSIR)
- Mangrove occurrence locations (NMU Botanical Database)
- Measured surface elevation heights at RSET stations (72 data points per RSET on each sampling occasion) (SAEON)
- Raw blue carbon data
- Raw values for predicted habitat area under sea-level rise (outputs from the SLAMM).
- Relative occurrence rates (raw MaxEnt output) for each estuary under each modelling scenario.
- Sediment parameters (moisture content, organic content, soil bulk density, soil carbon density, soil carbon per core section).
- Temperature and rainfall projections to 2050 under RCP 4.5 and RCP 8.5 (CSIR Greenbook)
- Vegetation map of the Knysna Estuary (NMU Botanical Database)