South African Journal of Botany 107 (2016) 137–147

Contents lists available at ScienceDirect

South African Journal of Botany

journal homepage: www.elsevier.com/locate/sajb

Understanding the creek dynamics and environmental characteristics that determine the distribution of mangrove and salt marsh communities at Nahoon Estuary

Chanel Geldenhuys [a], Phumlile Cotiyane [a], Anusha Rajkaran [b,*]

[a] Department of Botany, Rhodes University,P.O. Box 94, , Grahamstown 6140, South Africa
[b] Department of Biodiversity and Conservation Biology, Faculty of Natural Science, University of the Western Cape Private Bag X17, Bellville 7535, South Africa

ARTICLE INFO

Article history:
Received 20 April 2015
Received in revised form 28 February 2016
Accepted 29 April 2016
Available online 27 May 2016

Edited by JB Adams

Keywords:
Avicennia marina
Elevation
Freshwater requirements
Environmental constraints

ABSTRACT

The southern distributional limit for mangroves on the east coast of Africa is thought to be at the planted mangrove forest at Nahoon Estuary (33° S) in the Eastern Cape, South Africa. This study investigated the influence of a tidal creek on the intertidal zone and the physical and biological differences between the salt marsh and mangrove forest communities at Nahoon Estuary. Three transects were established across the tidal creek and one transect in each of the following habitats mangrove, mangrove–salt marsh, and the salt marsh area. The tidal creek introduced oxygenated (~6 $mg.l^{-1}$) and saline water with high levels of total suspended solids (120–424 $g.l^{-1}$) into the intertidal zone. In areas where tidal water was retained, algal mats formed over pneumatophores during summer. The vegetation distribution in the mangrove–salt marsh community was significantly affected by elevation, ammonium concentration, and porewater temperature while the salt marsh vegetation distribution was influenced by porewater salinity, sediment, pH and the percentage of sand content. Porewater nitrogen was mostly present as ammonium, and phosphate concentrations were moderate ranging from 1.3 μM in the salt marsh to 3.7 μM in the mangrove community. Mangrove and salt marsh communities are clearly constrained by the physical characteristics of the intertidal area (elevation) and this will ensure that both communities will be maintained at Nahoon Estuary. However with climate change and sea level rise, this may change in the long term with mangroves expanding into elevated areas.

1. Introduction

Mangroves are defined as trees and shrubs that grow in saline coastal habitats (Giri et al., 2011) while salt marsh habitats are defined as stands of salt-tolerant plants such as herbs, grasses, and shrubs that occur in the upper intertidal zone (Adam, 1990). Mangroves characteristically dominate lower elevation zones, where they are frequently inundated by tides, while salt marsh communities occupy the higher elevation zones, which are less commonly flooded (Chapman, 1974, Adam, 2002). On a global scale, mangroves are generally limited to the warmer coastal tropical regions of the world and extend into subtropical and occasionally even temperate regions >33° S (Clarke and Hannon, 1967; Adam, 2002; Stevens et al., 2006; Adame et al., 2010; Morrisey et al., 2010; Giri et al., 2011), while salt marsh are found at most latitudes but are largely replaced by mangrove forests in the tropical latitudes (Chapman, 1960, Chapman, 1975). At the transition zone between temperate and subtropical climate regions salt marsh and mangrove communities can co-exist (Clarke and Hannon, 1967; Steinke, 1995; Adam, 2002; Stevens et al., 2006; Adame et al., 2010; Morrisey et al., 2010). Where these communities co-exist more local conditions such as elevation, sediment characteristics, freshwater, and nutrients influence their distribution and ability to compete. In South Africa, mangroves occur in estuaries from Kosi Bay (KwaZulu–Natal) to Nahoon Estuary (Eastern Cape) where they were planted in 1969 (Steinke, 1972); salt marsh communities are dominant in estuaries further south from Nahoon to the Orange River (Western Cape).

Local parameters such as sediment particle size has a major influence on sediment biogeochemical characteristics such as redox potential, pH and organic and moisture content (Clarke and Kerrigan, 2000) and these in turn will affect the plant community that grows on it. Fine sediments are less permeable than course sediments which have higher infiltration rate of water. Clarke and Kerrigan (2000) found that conductivity, pH, nitrogen, and phosphorus were associated with sediments of different particle size. Mangrove sediments are characterized as being fine grained, poorly drained, saline, anoxic, and rich in organic matter (Lear and Turner, 1977). Fine particles are introduced into mangroves from riverine sources while course marine sediments are washed in through the mouth from the marine environment (Lovelock et al., 2007a). In general, mangroves are low-lying systems with a flat topography. They are thus regularly inundated by tides and

---

* Corresponding author.
E-mail address: arajkaran@uwc.ac.za (A. Rajkaran).

http://dx.doi.org/10.1016/j.sajb.2016.04.013
0254-6299/

remain saturated with water even during low tide due to poor drainage (Clarke and Hannon, 1967). As a result of being permanently waterlogged mangrove sediments are anoxic; with oxygen only being present in the surface layers around roots. The physiochemical characteristics of such waterlogged, anoxic soils are different to those of aerated soils. Salt marshes in contrast generally occur at higher elevations than mangroves. Although salt marsh soils can be highly saline, they are generally better drained and are more oxic. The sediment characteristics and nutrient content influence species distribution, as well as long-term growth and survival of plants. How the sediment changes when salt marsh areas are invaded by mangroves will affect a wide range of process as the sediment becomes more anoxic and may affect nutrient availability; this is the main focus of this study.

Soil nutrient availability is variable within and among estuarine ecosystems, including mangrove forests and salt marsh, and these dynamics are mostly unstudied in South Africa. Nutrient concentrations and species can vary spatially along a tidal gradient as well as temporally with seasons depending on the source and rate of cycling in each system (Lovelock et al., 2007a). Micro and macronutrients such as nitrogen, phosphorus, and potassium are essential to a variety of biological and chemical processes, both at the organism level (e.g. somatic growth, reproduction) and on the scale of ecosystems (Nirmal Kumar et al., 2011). Nutrients enter estuarine systems through numerous pathways: upland runoff, precipitation, and tidal input (Nirmal Kumar et al., 2011). Nutrient inputs may increase noticeably following rainfall events, as nutrients are washed from catchments and adjacent areas into the coastal zone. The availability of nutrients within the estuary is further influenced by various biotic factors including microbial activities in the soil, litter production, and rates of decomposition (Prasad and Ramanathan, 2008, Reef et al., 2010, Nirmal Kumar et al., 2011) as well as anthropogenic factors such as sewage runoff.

Nitrogen and phosphorus are two of the major plant nutrients determining plant growth. In mangrove sediments, nitrogen becomes available through microbial fixation of atmospheric nitrogen and through the biological decomposition of organic matter in the soil. In anaerobic soils most nitrogen is available in the form of ammonium ions (Armstrong, 1982). With nutrients being trapped in sediments and often little surface drainage entering from the surrounding environment, mangrove forests depend largely on nutrients from the sediment (released from decomposed organisms) and trapped in sediment porewater (Nirmal Kumar et al., 2011). The movement of nutrients and terrestrial sediment to the landward edges of intertidal range, both from freshwater sources and from the intertidal area, is often facilitated by tidal creeks (Green and Hancock, 2012). Tidal creeks transport oxygenated seawater, unicellular organisms, suspended solids (TSS), dissolved substances, and nutrients into sediments and facilitates the movement of degraded products and organisms from sediments (Santos et al., 2012). Porewater circulation through permeable sediments thus has a major influence on the biogeochemistry of sediments as it influences the porewater composition and the time it resides in sediments. Total suspended solids (TSS) is the concentration ($\mathrm{mg \cdot l^{-1}}$) of organic and inorganic matter which is held in the water column by turbulence and alters the water column both physically and chemically.

Very little is known about the nutrient dynamics of Southern African mangrove and salt marsh systems (Emmerson, 2005). A few studies have looked at channel water nutrients in salt-marsh-specific estuaries (Emmerson and Erasmus, 1987; Emmerson, 1989), but have not considered the importance of nutrients in the porewater which is directly available to the plants. This paper specifically aims to determine the role of a tidal creek in the intertidal zone by measuring the adjacent plant communities and the physical parameters of the water entering the intertidal area. Secondly, we aim to determine differences between the physico-chemical conditions in the porewater and sediment of the mangrove and the salt marsh communities. It is important to understand the physiochemical conditions in these sediments as this has a major effect on the vegetative growth response and survival of these vegetation types.

2. Study site

The Nahoon Estuary (32°59′09″ S, 27°57′03″ E) is a permanently open estuary situated in East London and falls within the East London Coastal Nature Reserve in the Eastern Cape Province of South Africa (Fig 1). Nahoon falls into the warm temperate biogeographic region and is 5 km long and the main tributary, the Nahoon River is approximately 70 km in length, with a catchment area between 547 and 625 $\mathrm{km^2}$ (CSIR 2000; Harrison et al., 2001). The Nahoon Estuary is microtidal with an average tidal range of 0.76 m and a coastal spring tide range of 1.6 m (Reddering, 1988), and is historically prone to periodic droughts and floods. Based on the last review, Nahoon was in fair condition and was prioritized at number 70 in the importance rating of estuaries in South Africa (Turpie et al., 2002). This estuary is recognized as the southern limit of the distribution of mangroves in South Africa (Ward and Steinke, 1982). The annual precipitation varies between 200 and 600 mm and most rainfall occurs during the spring and summer months. During this study, approximately 249 mm was received in summer while only 122 mm fell during the winter season. The annual temperatures ranged from a minimum of 4.6 °C to a maximum of 31.1 °C during the study (South African Weather Services 2012). The Nahoon River is a ‘drowned river valley’ since it is surrounded by steep cliffs or slopes which occur along certain lengths of the river. The steep cliffs which reach up to 105 m high in places, limit the access to the estuary and to some floodplain areas (Mega, 2013). The tidal creek at Nahoon Estuary is small and narrow in comparison to other creeks in South Africa.

3. Materials and methods

To determine the influence of the creek on the biological characteristics of the intertidal zone at Nahoon Estuary, water samples were taken at the mouth of the creek over a half tidal cycle and compared to that in the main channel in September 2013. Total suspended solids expressed as ($\mathrm{mg\ l^{-1}}$) was measured by filtering 250 ml of water samples through pre-dried GFC filters (at 103 – 105 °C). The residue retained on the filter were dried in an oven at 103–105 °C until the weight of the filter no longer changed. The increase in weight of the filter represented the total suspended solids (Trott and Alongi, 2000). Dissolved oxygen ($\mathrm{mg/L^{-1}}$), pH, temperature and redox was directly measured in the water column using a YSI ProPlus Multimeter. Three transects were then setup along the creek from the lower to the upper parts of the creek. These are labeled A, B, and C. Along each transect (0–5 m, 5–10 m, and >15 m), sediment was collected (3 replicates per depth, $n = 18$ per transect) at the surface and extracted from 20 cm depth to determine the sediment organic and moisture content, redox potential, particle size. The sediment redox potential was measured within 24 hours of collecting the sediment, and pH, moisture content and organic matter were measured within 48 hours. The hydrometer method was used to determine sediment particle size (Gee and Bauder, 1986). The proportions of sand, silt, and clay were then calculated. Redox potential was measured *in situ* with a multiprobe (a HANNA redox/pH meter (HANNA Instruments) and a platinum–gold tipped electrode).The pH of the sediment was measured using a multiprobe (a HANNA redox/pH meter (HANNA Instruments) and a platinum-gold tipped electrode). Moisture content (Black, 1965), organic matter content (Briggs, 1977) and electrical conductivity (The Non-Affiliated Soil Analyses Working Committee, 1990) of the soil was measured in a laboratory according to methods cited.

To determine differences between the physico-chemical conditions in the porewater and sediment; transects were setup perpendicular to the main channel in the mangrove, mangrove–salt marsh and the salt marsh communities. The mangrove community (Fig. 1—Transect

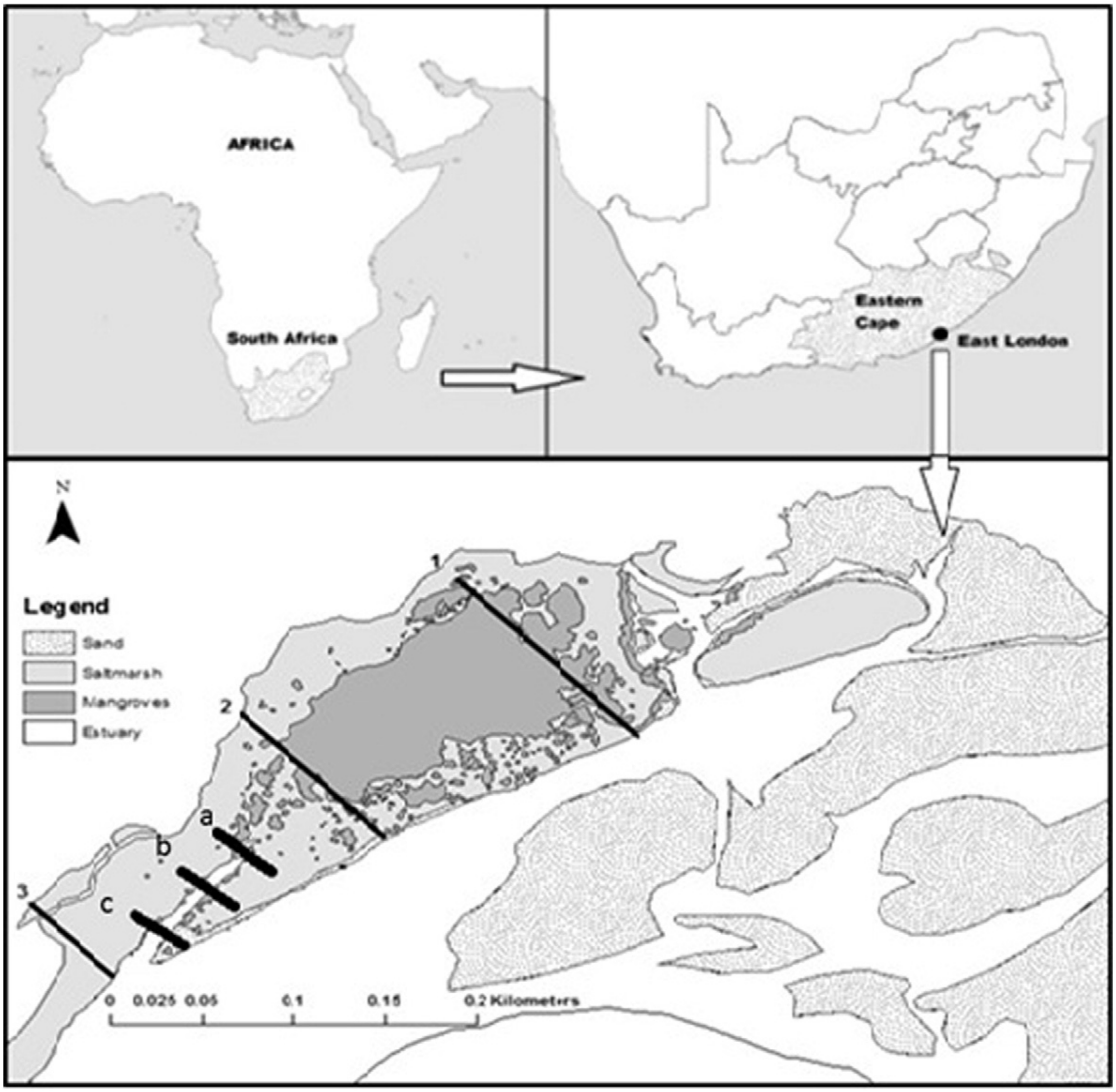

Fig. 1. Location of the Nahoon Estuary in East London, South Africa. Transects a, b, c are associated with the tidal creek while Transect 1, 2, 3 are associated with each habitat type.

1) supports three mangrove species although *Avicennia marina* (*Forrak.*) *Vierh* is the clear dominant. The mangrove–salt marsh (Transect 2) is an intermediate community where mangrove and salt marsh species co-occur and compete directly; in this area a large, dense algal mat was also present. The salt marsh community (Transect 3) consists of 20 salt marsh species including reeds. *Sarcocornia tegetaria* S. Steffen, Mucina & G. Kadereit, *Sporobolus virginicus* (L.) Kunth., *Bassia diffusa* (Thumb.) Kuntze and the freshwater species *Nasturtium officinale* W.T. Aiton are the dominant species in this community. Sediment and nutrient samples were collected during a spring and neap tide in each of the summer (November–March) and winter (June–August) seasons in 2012.

Three replicate holes were augured in each zone (lower, middle and upper intertidal) along each transect. Sediment was collected from the surface and at a depth of 50 cm, holes were then left to fill with water. If no porewater was visible holes were augered further generally to a depth of 1 m. All sediment and porewater samples were collected at low tide so as not to measure marine water entering the habitat at high tide. The porewater was collected using a 50 ml syringe and filtered using GF/C (1.2–2.7 μM) filters in the field. Porewater samples were kept in 50 ml dark, sterilized plastic bottles and frozen in order to preserve the ammonium (NH_4). Porewater samples were sent to the Postgraduate Research Laboratory of the Department of Botany at the Nelson Mandela Metropolitan University and analyzed to determine the concentrations of orthophosphate (PO_4^{3-}), ammonium (NH_4), and total oxidized nitrogen (NO_3–N). Filtered water samples were analyzed for total oxidized nitrogen (TOxN) (nitrate $\pm$ nitrite) using the reduced copper cadmium method as described by Bate and Heelas (1975). Ammonium and soluble reactive phosphorus (SRP) were analyzed using standard spectrophotometric methods (Parsons et al. 1984). Fresh water samples were collected upstream of the Abbotsford causeway (32°57′53″ S 27°54′55″ E) as this is a physical barrier limiting the tidal flow upstream (MEGA, 2013). Marine water samples were collected outside the mouth of the Nahoon River. In certain mid and upper salt marsh sites, no samples could be collected as the depth of porewater was not accessible. Where no samples could be collected this was denoted as 'no data' in the results. In the field, porewater salinity and temperature were measured in each augured hole using a refractometer and thermometer respectively.

The skewness and kurtosis of the data were tested to determine the normality of the data. If the data were not normally distributed, non-parametric tests were used to determine differences between tidal zones (lower/mid/upper), seasons (summer/winter), tides (spring/neap), plant community (mangrove/mangrove–salt marsh/salt marsh) and in the case of sediment-depth (surface/bottom). A Kruskal–Wallis ANOVA ($H_{(df,N)}$) was used to determine differences while a multiple comparison of mean ranks was used to test for mean separation. If data were normally distributed, One-Way ANOVA tests along with Tukey HSD post hoc tests were used. All statistical analyses were run using Statistica (Version 11, 2011) and significance was determined at $p < 0.05$. A Spearman rank correlation was used to determine relationships between characteristics.

A canonical correspondence analysis (CCA) was used to obtain an ordination showing how vegetation data were constrained by environmental variables. Data collected from Transects 1, 2, and 3 were used for this test. Methods used to collect vegetation data are outlined in Geldenhuys (2014). Monte Carlo permutation tests (999 permutations) were used to assess the significance of the canonical axis showing the relationship between species and environmental variables. Only the vegetation cover at 35, 75, and 115 m along T2 (mangrove–salt marsh transect) and at 15, 30, 45-m along T3 (salt marsh transect) were used in the CCA to match and determine the influence of environmental factors on the vegetation distribution. The environmental variables were plotted as arrows originating from the centre of the CCA ordination. The direction of each environmental arrow represents an increase in the value of that particular characteristic. The length of the environmental arrow expresses the importance of the characteristic. Statistical results are indicated in a table below each CCA ordination diagram (ter Braak and Šmilauer, 2002).

4. Results

The tidal creek introduced oxygenated (6.0 ± 0.5 mg/l) water with salinity (33.7 ± 1.3) similar to seawater. The pH of water was 8.0 ± 0.1 and the redox potential was 103.9 ± 15.6 mV. Total suspended solids (TSS) peaked at 424 g/l during the tidal cycle. The concentration of dissolved oxygen was similar over the 6 hours ranging from 3.78 to 7.79 mg/l and was similar to channel water ($H_{(5, 6)} = 5$, $p > 0.05$). There was no significant difference in porewater characteristics in the creek as shown in Table 1 (temperature, salinity, and pH) along all transects across the tidal creek at Nahoon Estuary. Transects A, B, and C showed that the sediment characteristics such as electrical conductivity were similar along the creek, while moisture and organic matter changed from the lower to the upper parts but was generally lower in Transect C (Table 1). This was associated with changes in biological components such as algal biomass, crab burrows, and population structure of mangroves.

Sediment composition was not significantly different between the three different community types (Transect 1–3), the different sampling sessions nor did it change with depth ($p > 0.05$). Sand (51.6–56.3%) formed the largest component of the sediment across all three transects with clay (33–35%) and silt (8.8–12%) making up the smallest percentage.

Overall, there was no significant difference in porewater temperature between the different vegetation communities ($p > 0.05$). Porewater temperature across all three transects was significantly lower during winter ($F = 21.03$, $p < 0.001$, $df = 17$) (Tables 2, 3, 4) and was significantly correlated to the maximum daily air temperatures ($r = 0.8$, $p < 0.05$). Porewater temperature was also significantly lower during the neap tide than during the spring tide ($F = 29.88$, $p < 0.05$, $df = 1$) (Tables 2, 3, 4) for each season. The salinity measurements are a broad representation of the amount of freshwater received by the estuary. Mean porewater salinity ranged from 26 ± 0 PSU in the upper salt marsh to a maximum of 45.8 ± 1.8 PSU in the mid salt marsh (Tables 2, 3, 4). Porewater salinity along the salt marsh transect was significantly higher than in the mangrove and mangrove–salt marsh ($H_{(2, 202)} = 7.87$, $p < 0.05$). Porewater salinity was similar between spring and neap tides ($H_{(1, 102)} = 0.36$, $p > 0.05$), but changed seasonal ($H_{(1, 102)} = 29.65$, $p < 0.05$) illustrating the effect of seasonal rainfall. The porewater in the mangrove–salt marsh community and the mangrove community was more saline in winter (mean 42 ± 0.8 PSU) than those in summer (mean 33 ± 1.1 PSU). Overall the upper tidal zone had a significantly lower salinity than the mid and lower tidal zones ($H_{(2, 102)} = 31.875$, $p < 0.05$).

Table 1
Sediment characteristics along transects associated with the tidal creek at Nahoon Estuary.

	Transect A	Transect B	Transect C
Porewater characteristics (N = 9)			
Porewater salinity (‰)	29.0 ± 1.4	31.5 ± 2.2	29.4 ± 1.7
Porewater temperature (°C)	19.3 ± 0.3	20.2 ± 0.9	20.0 ± 0.3
pH	6.1 ± 0.1	5.6 ± 0.2	6.4 ± 0.1
Sediment characteristics (N = 18)			
Sediment redox potential (mV)	−140.8 ± 13.1	−85.4 ± 24.0	−106.5 ± 14.3
Sediment electrical conductivity (mS)	38.3 ± 1.7	45.2 ± 4.0	41.4 ± 0.9
Moisture content (%)	34.1 ± 2.5	40.5 ± 3.2	36.0 ± 2.7
Organic matter (%)	7.7 ± 1.0	10.2 ± 2.2	3.6 ± 0.8

Overall, the electrical conductivity (EC) in the salt marsh was significantly lower than that in the mangrove and mangrove–salt marsh ($F = 26.06$, $p < 0.05$, $df = 2$). The increased rainfall during the summer season significantly reduced the electrical conductivity of the soil as EC was significantly higher in the winter season than during the summer season ($F = 19.95$, $p < 0.05$, $df = 1$), particularly in the mangrove sites. EC was significantly lower during the neap tide than during the spring tide ($F = 23.67$, $p < 0.05$, $df = 1$), particularly in the mangrove where the EC during the neap tide was significantly lower than in the other transects ($F = 9.38$, $p < 0.05$, $df = 17$) (Tables 2, 3, 4).

The pH in salt marsh were significantly lower than the mangrove ($H_{(2, 216)} = 15.88$, $p < 0.05$) with the salt marsh average (±SE) pH being 7.6 ± 0.1 and the mangrove average pH being 8 ± 0.1 (Tables 2, 3, 4). In general, sediment pH was significantly influenced by tides and seasons. A Spearman rank correlation showed correlations between maximum air temperature and sediment pH ($r = -0.59$, $p < 0.05$) as well as between porewater temperature ($r = -0.51$, $p < 0.05$) and rainfall and sediment pH ($r = -0.64$, $p < 0.05$).

In general moisture content and organic matter was significantly higher in the mangrove and mangrove–salt marsh habitats ($F = 6.19$, $p < 0.05$, $df = 2$ and $F = 5.27$, $p < 0.05$, $df = 2$, respectively) than in the salt marsh habitat (Tables 2, 3, 4). The surface sediments were significantly more moist in the mangrove and the mangrove–salt marsh ($F = 13.79$, $p < 0.05$, $df = 5$) and in general had a higher percentage of organic matter ($H_{(1, 216)} = 34.84$, $p < 0.05$) than the bottom sediments. A Spearman rank correlation showed that there was a positive correlation between moisture content and organic matter ($R = 0.90$, $p < 0.05$). Overall the sediment in the salt marsh was least reduced (i.e. Redox potential was highest) and significantly higher to that of the mangrove and mangrove–salt marsh ($F = 78.36$, $p<0.05$, $df = 2$).

Porewater ammonium concentrations were significantly higher in the salt marsh than in the mangrove or mangrove–salt marsh ($H_{(2, 92)} = 18.79$, $p < 0.05$) (Fig. 2). Within each transect, there was seasonal or tidal difference in ammonium (Table 5) ($p > 0.05$). Porewater ammonium concentration at the mouth of the estuary was lower than in the transects and was measured in the range of 0.9–3.1 μM (NH_4). Ammonium concentration in the river water ranged from 1.2 to 28.2 μM (NH_4). Porewater TOxN was not significantly different between the transects or between the tidal zones ($p > 0.05$) (Fig. 20). Total oxidized nitrogen in the porewater under the salt marsh was significantly higher during the winter season than during summer ($H_{(5, 96)} = 23.32$, $p < 0.05$) (Table 5) and generally higher during the spring tide (mean: 1.57 ± 0.7) than during the neap tide (mean: 1.35 ± 0.1, $p < 0.05$). Total oxidized nitrogen was highest in the river where readings varied from 9 to 33.9 μM ($NO_{3-} + NO_{2-}$). Porewater orthophosphate was significantly different between the mid and the upper tidal zones ($H_{(2, 92)} = 5.89$, $p < 0.05$) but was similar between transects (Fig. 2). The highest phosphate concentrations were measured in the porewater of the upper tidal zones with the highest reading occurring in the mangrove upper during the summer season (10.6 ± 6.7 μM) and the lowest

Table 2

Sediment characteristics of Transect 1: mangrove habitat in the lower, mid, and upper tidal zones measured over four sampling seasons in 2012 (±SE, N = 3).

	Summer Spring (Feb 2012)			Winter Spring (June 2012)			Winter Neap (July 2012)			Summer Neap (Nov 2012)		
	Lower	Mid	Upper	Lower	Mid	Upper	Lower	Mid	Upper	Lower	Mid	Upper
Porewater salinity (PSU)	37 ± 0.6	36 ± 0.3	26 ± 5.0	43 ± 1.2	44 ± 0.6	38 ± 3.1	43 ± 0.9	45 ± 0.3	32 ± 4.4	41 ± 2.4	38 ± 1.8	28 ± 4.8
Porewater temperature (°C)	23.8 ± 0.3	24.5 ± 0.3	26.2 ± 0.2	16.8 ± 0.2	16.5 ± 0.3	14.5 ± 0.3	8.7 ± 0.3	7.8 ± 0.6	8.7 ± 0.3	19 ± 1.2	22.3 ± 1.2	21.7 ± 0.3
pH	6.9 ± 0.1	6.3 ± 0.1	7.2 ± 0.2	8.8 ± 0.1	8.6 ± 0.0	8.6 ± 0.0	8.5 ± 0.0	8.5 ± 0.0	8.5 ± 0.1	8.2 ± 0.0	7.8 ± 0.1	7.8 ± 0.1
Sediment redox potential (mV)	−188.5 ± 17.8	−192.3 ± 18.7	−181.3 ± 24.04	−120.2 ± 64.02	−225.8 ± 19.3	−153.2 ± 31.01	−67.5 ± 60.8	−56.8 ± 27.1	−209.4 ± 80.3	−124.5 ± 8.4	−183.7 ± 29.9	−259.4 ± 31.5
Sediment electrical conductivity (mS)	17 ± 5.7	15.8 ± 7.5	23 ± 10.7	64.1 ± 4.7	50.9 ± 2.8	49.4 ± 5.8	36.9 ± 1.2	36.3 ± 6.1	33.8 ± 3.5	22.9 ± 1.1	33.1 ± 3.7	42.7 ± 5.7
Moisture content (%)	24.8 ± 0.8	26.4 ± 1.3	26.4 ± 2.7	26.7 ± 3.5	31.5 ± 3.03	28.6 ± 2.8	32.3 ± 5.5	22.4 ± 3.7	29.5 ± 3.0	26.7 ± 2.3	29.3 ± 4.3	33 ± 5.5
Organic matter (%)	11.7 ± 0.2	12.8 ± 0.7	12.9 ± 1.4	14.1 ± 1.5	16.9 ± 1.9	15.6 ± 1.2	18.2 ± 3.7	12.4 ± 2.01	15.5 ± 1.6	13.2 ± 1.2	18.2 ± 2.3	22.8 ± 4.1

Table 3

Sediment characteristics of along Transect 2—the mangrove–salt marsh habitat in the lower, mid and upper tidal zones measured in the four sampling seasons in 2012 (±SE, N = 3).

	Summer Spring (Feb 2012)			Winter Spring (June 2012)			Winter Neap (July 2012)			Summer Neap (Nov 2012)		
	Lower	Mid	Upper	Lower	Mid	Upper	Lower	Mid	Upper	Lower	Mid	Upper
Porewater salinity (PSU)	36 ± 1.2	34 ± 0.9	14 ± 6.9	42 ± 1.2	46 ± 0.6	36 ± 3.3	38 ± 0	40 ± 0.3	42 ± 1.7	35 ± 2.4	40 ± 1	30 ± 2.6
Porewater temperature (°C)	24.2 ± 0.2	24.5 ± 0.3	26 ± 0.5	16.5 ± 0.3	15.8 ± 0.3	16.8 ± 0.6	7.8 ± 0.2	7.7 ± 0.3	8 ± 0	24 ± 0.6	23.7 ± 1.2	25 ± 1
pH	7.5 ± 0.07	7.1 ± 0.1	7.3 ± 0.2	8.6 ± 0.1	8.4 ± 0.1	8.4 ± 0.11	7.27 ± 0.1	7.8 ± 0.1	8.6 ± 0.1	7.5 ± 0.1	7.2 ± 0.1	8 ± 0.1
Sediment redox potential (mV)	−160.5 ± 22.7	−235.3 ± 22.9	−205 ± 22.0	19.5 ± 65.4	−82.8 ± 57.2	−104.8 ± 51.0	−198.0 ± 78.3	−329.1 ± 23.4	−241.6 ± 23.0	−125.5 ± 44.9	−180 ± 38.2	−195.9 ± 77.6
Sediment electrical conductivity (mS)	44 ± 3.6	47.1 ± 2.1	32.1 ± 5.7	49.1 ± 3.7	52.5 ± 2.4	55 ± 4.5	35.5 ± 3.1	36.1 ± 2.1	38 ± 2.8	28.5 ± 3.4	21 ± 1.9	20.2 ± 2.1
Moisture content (%)	26.8 ± 1.9	31.6 ± 3.5	26.9 ± 2.6	26.6 ± 1.8	40.9 ± 5.6	41.2 ± 6.6	38.2 ±4.2	30.8 ± 4.4	26.2 ± 2.6	34.7 ± 2.3	47.5 ± 1.4	46.6 ± 2.7
Organic matter (%)	15.7 ± 1.3	19.5 ± 2.3	15.2 ± 1.7	16.9 ± 1.6	31.2 ± 4.5	28 ± 5.1	26.3 ± 3.5	19.9 ± 3.1	16 ± 1.7	23.2 ± 1.6	32.7 ± 1.1	25.1 ± 1.7

Table 4
Sediment characteristics along Transect 3: salt marsh habitat in the lower, mid, and upper tidal zones measured in the four sampling seasons in 2012 (±SE, N = 3). No data = ND.

	Summer Spring (Feb 2012)			Winter Spring (June 2012)			Winter Neap (July 2012)			Summer Neap (Nov 2012)		
	Lower	Mid	Upper	Lower	Mid	Upper	Lower	Mid	Upper	Lower	Mid	Upper
Porewater salinity (PSU)	38 ± 0.9	41 ± 1.5	26 ± 0	42 ± 2.5	55 ± 0	ND	43 ± 1.5	46 ± 0	ND	35 ± 0.3	ND	ND
Porewater temperature (°C)	24.2 ± 0.4	23.7 ± 0.2	24 ± 0	16 ± 0.8	14.5 ± 0	ND	8.9 ± 0.5	8 ± 0	ND	23 ± 1.7	ND	ND
pH	7.4 ± 0.1	6.8 ± 0.1	6.5 ± 0.1	7.7 ± 0.2	8.2 ± 0.2	7.9 ± 0.1	7.5 ± 0.1	8.3 ± 0.2	8.3 ± 0.1	7.5 ± 0.1	7.6 ± 0.1	7.4 ± 0.1
Sediment redox potential (mV)	−133.3 ± 39.0	66.7 ± 37.3	167.2 ± 25.3	−12.8 ± 68.2	198 ± 39.7	233.2 ± 12.3	−187.85 ± 61.8	174.9 ± 50.9	302.5 ± 8.8	− 47 ± 81.7	182.6 ± 19.2	232.3 ± 17.7
Sediment electrical conductivity (mS)	39.7 ± 5.5	36.3 ± 3.6	9.2 ± 4.7	39.3 ± 3.5	24.8 ± 6.4	16.2 ± 2.9	27.8 ± 2.8	6.9 ± 2.6	5.2 ± 1.5	18 ± 2.3	16.5 ± 4.4	5.5 ± 1.0
Moisture content (%)	35.6 ± 3.0	32.1 ± 3.6	28.1 ± 3.9	30 ± 4.8	21.8 ± 3.0	31.2 ± 2.7	44.3 ± 6.1	34.3 ± 3.7	27.3 ± 2.3	25.9 ± 2.8	23.1 ± 4.3	16.7 ± 1.1
Organic matter (%)	27.3 ± 5.2	16.1 ± 2.2	14.7 ± 2.5	15.7 ± 2.6	11.8 ± 1.5	19.3 ± 2.1	25 ± 4.4	17.7 ± 2.4	15.2 ± 2	14 ± 1.6	13.3 ± 2.0	10.5 ± 0.8

recordings occurring in the salt marsh (0.1–3.2 μM) (PO_4^{3-}) (Table 5). Overall orthophosphate was significantly higher during summer than during winter ($H_{(5,\ 96)} = 41.88$, $p < 0.05$). An overall total inorganic nitrogen (total ammonium ± total oxidized nitrogen) to total phosphorus ratio (N:P) were calculated for the three plant communities in the Nahoon Estuary. The N:P ratio for the mangrove was 1:0.7, the mangrove–salt marsh 1:0.2 and the salt marsh 1:0.04. These ratios showed that all three plant communities are phosphate limited.

The numerical results of the canonical correspondence analysis (CCA) are shown in Table 6 and Figs. 3 and 4 for Transect 2 and Transect 3. A Monte Carlo permutation test of the trace (sum of eigenvalues of all canonical axis; 999 permutations) showed the vegetation distribution in the Nahoon Estuary. In the mangrove–salt marsh community (T2) (Fig. 3) the second canonical axis described 100% of the variation of the species—environment relation (Table 6). Along this axis parameters were negatively correlated with soil moisture content (−0.41), soil organic matter (−0.54), sediment electrical conductivity (-0.62) and the sand percentage (−0.64) and positively correlated with sediment pH (0.10), porewater temperature (0.19), porewater ammonium (0.94) and elevation (0.34). Fig. 3 indicates that in the mangrove–salt marsh community the more terrestrial salt marsh species including *Juncus kraussii*, *Limonium scabrum*, *Stenotaphrum secundatum*, and *Bassia diffusa* were associated with orthophosphates and the percentage of clay. The algae, mangroves, and *Sarcocornia* species—the dominant low tidal salt marsh species were found at lower elevations, and associated with high porewater salinity.

In the salt marsh (T3) habitat at Nahoon Estuary (Fig. 4), the second canonical axis described 100% of the variation of the species—environment relation (Table 6). Along this axis parameters were negatively correlated with porewater temperature (−0.73) and porewater ammonium (−0.48) and positively correlated with porewater salinity (0.83), sediment pH (0.87), sand percentage (0.43) and sediment electrical conductivity (0.26). Fig. 4 indicates that in the salt marsh community the species commonly occurring in the upper tidal zone including *Disphyma crassifolium*, *Cyperus laevigatus* and *Phragmites australis* were strongly associated with oxidized sediments, while the lower intertidal *Sarcocornia* species were influenced by sediment electrical conductivity. Fig. 4 shows that algal mat and *Triglochin* was associated with high porewater nitrate and ammonium

5. Discussion

Globally, mangrove distribution has been shown to be largely limited by temperature at a 15–20 °C isotherm, with mangrove species preferring the tropical and sub-tropical climates. However, on a more local scale species distribution and species diversity have proven to be strongly influenced by salinity, pH, and sediment redox potential (Hogarth, 1999; Alongi, 2002). More importantly the persistence of mangrove forests at this scale depends on sediment stability and the availability of water and nutrients (Krauss et al., 2008; Alongi, 2009). Salt marshes alternatively prefer more temperate climates where they inhabit estuaries with stable sediments where they are less inundated in comparison to mangroves (Mitchell and Adam, 1989; Adame et al.; 2010, Barbier et al., 2011). Sediment characteristics strongly influence the survival of salt marshes. The most important characteristics being salinity, this determines the vertical and horizontal zonation of salt marsh species. Sediment texture and compaction influences the level of water drainage and salt retention (Adam, 2002). Nahoon Estuary is permanently open with the width of the mouth varying between m and 40 m. Sandstone outcrops on the eastern bank of the river mouth result in tidal flow causing scouring and maintaining an open mouth. The sediment near the estuary mouth is composed primarily of sand. During periods of drought or low rainfall marine sediments are deposited in the lower estuary up to 1.2 km upstream from the mouth. Further upstream the sediment becomes more cohesive as a result of clay sediment being washed from the eastern bank (MEGA, 2013). The intertidal area under investigation in this study was equally affected by river and the ocean.

Sediment particle size plays a significant role in determining the moisture, nutrients, and organic matter content which in turn influence the redox potential and salinity of the sediment (Hegazy, 1998). Bornman et al. (2002) found that sediments with a high clay percentage had little infiltration of water resulting in high surface moisture contents. Prasad and Ramanathan (2008) found that in the Pichavaram mangroves, fine-grained sediments retain higher levels of organic carbon due to higher surface area. The sediment at Nahoon Estuary constituted primarily sand particles but there was no significant difference in particle size between the three vegetation types with sand (50 μM) making up approximately 50% of the sediment, clay (2 μM) 35% and silt 15%. Some sediment samples in the mangrove and mangrove–salt marsh habitats contained oxidized dark grey-brown mud. In 2011, a flood deposited sediment and large amounts of debris in the estuary, particularly in the area of the channel in the mangrove–salt marsh area. Incoming tide deposited marine sediment across the estuary (Vernon, 2013), between June 2011 and January 2013 resulting in more sandy sediments in the estuary. Although the sediment particle size was not significantly different between the three plant communities, sediment moisture content and organic matter were significantly lower in the salt marsh (27% moisture content, 16% organic matter) compared to in the mangrove (33% moisture content, 19 % organic matter) and mangrove–salt marsh communities. This could also be related to elevation. Mangrove sediments are generally high in organic litter and organic carbon as a result of high levels of litter fall, mostly in the form of senescent leaves. In a study in New Zealand, Alfaro (2010) found a generally higher organic matter in mangrove (±5–22 %) and pneumatophore (±5–25 %) sites than in marsh grass (±3–7 %) and sandflat (± 1 – 8 %) sites. Bornman and Adams (2008) found organic matter to vary between 0.35 and 7.09% and moisture content to vary between 0.54 and 27.43% in a salt marsh habitat in the Orange River Estuary in South Africa. Hoppe-Speer (2015) measured the organic matter (1–6 % and 2.3–4 %) and moisture (25–55 % and 26.4–39.9 %) at the mangrove system in the St Lucia Estuary and Nahoon Estuary respectively. Organic matter was higher in the current study compared to Hoppe-Speer et al. (2015). It was found here that organic matter is important as it influences the capacity of the sediment to hold water and nutrients. Higher organic matter has a greater potential to hold more water and retain nutrients.

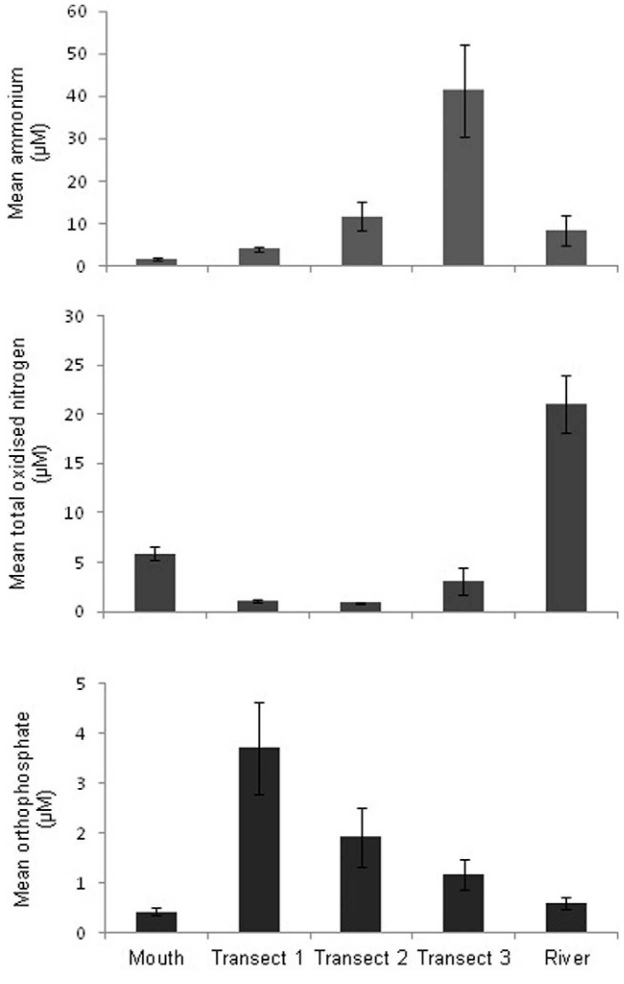

Fig. 2. Mean ammonium (NH_4), TOxN ($NO_{3-} \underline{+} NO_{2-}$) and orthophosphate ($PO_4^{3-}$) concentrations in the three plant communities in the Nahoon Estuary and in the Nahoon river and estuary mouth in 2012.

Sediment redox potential was highest in the mid and upper salt marsh and decreased in the mangrove communities. This shows that the mangrove and mangrove–salt marsh areas which are inundated by tidal water on a daily basis experience higher levels of anoxia. Hoppe-Speer (2015) found redox in the mangroves at St Lucia to range from approximately −190 to +250 mV and Rajkaran and Adams (2012) found the redox potential to vary between approximately +39 to +193 mV at Mngazana Estuary in South Africa. Lovelock et al. (2007a) found similar redox potentials in two mangrove sites in New Zealand–Whangapoua± +100 to +128 mV and Waikopua± –34 to +76 mV. These results were comparatively high compared to those found at Nahoon Estuary where the redox potential varied from −259 to −120 mV in the mangrove to −329 to +19.5 mV in the mangrove–salt marsh. Tidal flow into and out of the estuary strongly affects temperature and salinity characteristics along with total suspended solids transported via the tidal creek.

The Nahoon Estuary is considered to be a microtidal and flood-dominant estuary with a spring range of 1.6 m and an average tidal range of 0.76 m. The mixing of marine and estuarine water occurs until 1.4 km from the mouth where the Abbotsford causeway forms a physical barrier preventing mixing further upstream. The freshwater flow into the estuary has been reduced by the development of the Nahoon Dam. The reduced freshwater inflow has had significant influences on salinity, oxygen, algal biomass, bacteria, and biodiversity of zooplankton (MEGA, 2013). Algal mats influence sediment chemistry (Raffaelli et al., 1999), and the biomass is governed and affected by fluctuations in light attenuation, temperature, micronutrients (Gubelit, 2009), and water retention. Algal species identified during the study were *Ulva intestinalis* Linnaeus and. *Cladophora glomerata* (L.) Kutz. Its presence at Nahoon and may indicate periods of nutrient enrichment. The area where it occurs links the tidal creek to the main mangrove area and water retention in this area is higher to due to a lower elevation compared to the surrounding area. This implies that the tidal creek is supplying the mangrove area with nutrients but competition between mangroves and algae may be high and establishment of seedlings may be reduced.

Porewater salinity in the mangrove sites ranged from a low 2 PSU to a high of 47 PSU but averaged at 37 PSU. This is similar to that reported by the Nahoon Estuary Management Plan (MEGA, 2013) which has reported the average salinity in the Nahoon Estuary to vary between 34 and 37 PSU. In the Eastern Cape, the mangrove-dominant Mgazana Estuary had an average salinity below 26 PSU (Rajkaran and Adams, 2012) and two estuaries in New Zealand North Island had a salinity ranging from 19.6 to 29.8 PSU (Lovelock et al., 2007a). Robertson and Alongi (1992) found that *A. marina* has a tolerance for a maximum porewater salinity of 85 PSU, with optimal growth occurring between a salinity of 0–30 PSU. Qureshi (1993) similarly found that most mangrove species surviving in salinity concentrations greater than 35 PSU show signs of stress, such as reduced propagule production. Salinity in the mangrove communities at Nahoon Estuary is on average 38 ± 0.8 PSU, higher than the optimal salinity range for *A. marina*, which may inhibit growth but this depends on how long salinity remains at this level.

Table 5
Concentrations of ammonium, total oxidized nitrogen, and orthophosphate in the different tidal zones in the mangrove, mangrove–salt marsh, and salt marsh communities during four sampling seasons in 2012 (±SE, N = 3). No data = ND.

Transect		Ammonium (NH_4)				Total oxidized nitrogen ($NO_{3-} + NO_{2-}$)				Orthophosphate (PO_4^{3-})			
		Feb	June	July	Nov	Feb	June	July	Nov	Feb	June	July	Nov
1 (Mangrove)	Lower	9.1 ± 2.3	3.4 ± 0.0	1.6 ± 0.0	1.5 ± 0.8	0	1.3 ± 0.3	1.2 ± 0.5	2.1 ± 0.1	9.9 ± 0.8	4.8 ± 1.8	0.2 ± 0.1	1.1 ± 0.7
	Mid	5.0 ± 1.2	2.4 ± 0.5	1.8 ± 0.2	2.6 ± 0.8	0	1.5 ± 0.3	0.8 ± 0.1	1.8 ± 0.1	4.9 ± 0.2	0.04 ± 0.0	0.04 ±0.0	0.7 ± 0.3
	Upper	8.6 ± 0.7	2.9 ± 0.9	2.5 ± 1.3	7.0 ± 2.9	0	2.3 ±0.9	0.4 ± 0.2	1.3 ± 0.2	7.2 ± 4.8	0.6 ± 0.5	4.6 ± 4.4	10.6± 6.7
2 (Mangrove–salt marsh)	Lower	2.9 ± 0.1	1.6 ± 0.2	48.4 ±6.0	35.9 ± 18.2	0	0.9 ±0.1	0.5 ± 0.1	2.3 ± 0.3	3.2 ± 0.8	0.04 ± 0.0	0.1 ± 0	1.3 ±1.2
	Mid	2.8 ± 0.6	2.5 ± 1.0	11.1 ± 6.1	1.3 ± 0.4	0	1.4 ± 0.4	0.8 ± 0.4	1.6 ± 0.1	3.3 ± 1.0	0.01 ± 0.01	0.02± 0.0	0.9 ± 0.8
	Upper	5.3 ± 0.3	1.8 ± 0.4	3.6 ± 0.2	24.1 ± 20.4	0	0.7 ± 0.1	0.4 ± 0.1	1.8 ± 0.2	7.6 ± 4.4	0.2 ± 0.1	2.1 ± 2.0	4.4 ± 3.5
3 (Salt marsh)	Lower	146.6 ±57.6	31.5 ± 14.2	31.1 ± 22.8	30.8 ± 3.8	0	12.9 ± 10.9	1.6 ± 0.7	2.4 ± 0.2	3.2 ± 0.8	0.3 ± 0.2	0.2 ± 0.1	0.1 ± 0.0
	Mid	5.4 ± 1.1	35.9 ± 3.7	39.0 ± 0	ND	0.2 ± 0.2	4.0 ± 0.1	0.6 ± 0	ND	3.3 ± 1.0	0.02 ± 0.0	0.1 ± 0	ND
	Upper	5.8 ± 0	ND	ND	ND	0	ND	ND	ND	3.8 ± 0	ND	ND	ND
Mouth		0.9 ± 0.1	0.7 ± 0.0	3.1 ± 0.3	1.9 ± 1.1	6.8 ± 0.3	2.2 ± 0.1	5.6 ± 0.2	8.9 ± 0.1	0.6 ±0.05	0.1 ± 0.0	0.3 ± 0.0	0.6 ± 0.0
River		2.9 ± 0.2	1.2 ± 0.14	28.2 ± 1.9	1.2 ± 0.4	17.5 ± 0.4	23.8 ± 1.7	33.9 ± 4.5	9.0 ± 1.6	0.8 ± 0.04	0.03 ± 0.0	0.9 ± 0.1	0.7 ± 0.1

The increase in freshwater through rainfall in the summer months resulted in an overall decrease in the sediment EC. Freshwater runoff at the base of the cliff on the landward side of the Nahoon Estuary played an important role in the salinity dynamics of the study site. The upper landward zones had a significantly lower salinity than the mid and lower tidal zones. The high sand content of the sediment and the high percentage of plant cover allow good infiltration of water though the sediment resulting in a uniform EC from the surface to deeper sediments. The construction of the Nahoon Dam in 1966 and the causeway limit freshwater reaching the estuary and could be important factors that result in an increase in salinity at Nahoon Estuary. The decrease in EC and salinity in the summer months with the relief of increased rainfall suggests the importance of freshwater inputs into the estuary. This further emphasizes the management of freshwater abstraction and the role of the different structures along the estuary.

At Nahoon Estuary, porewater nitrogen was mostly present as ammonium. Levels of ammonium were higher than the total oxidized nitrogen in both summer and winter. This is due to rapid decomposition of leaf matter, probably by macro-fauna. Although no quantitative studies have been done on the crab populations at Nahoon Estuary, studies such as Smith et al. (1991) have shown direct correlations between the size of crab populations and ammonium concentrations. The mean ammonium levels increased from the mangrove (4 ± 0.5 μM) to the salt marsh (44.4 ± 0.4 μM). This was similar to the findings of Clarke (1985) who found the mean ammonium concentrations in mangroves to be 18 μM and increased in the salt marsh to 20 μM. In a South African study, Emmerson (2005) found the mean ammonium in the mangrove-dominant Mngazana Estuary to be 10.9 μM, while Emmerson (1989) found the mean ammonium concentration in the salt-marsh-dominant Sundays and Swartkops to be 2.9 and 14.6 μM, respectively. Morris (1980) states that in general salt marsh sediments contain high concentrations of dissolved inorganic nitrogen, mostly as ammonium. Anaerobic soils are more often nitrogen limited and higher in ammonium. Bava and Seralathan (1999) and Lovelock et al. (2010) similarly found that porewater N was mostly present as ammonium in the mangrove with low concentrations of nitrate. Total dissolved N, however, increased toward the landward edge. Total oxidized nitrogen (TOxN) includes both nitrates and nitrites. TOxN at Nahoon Estuary was low in all three community types (range 0.2–12.9 μM) and was lowest in the mangrove–salt marsh. TOxN was in a similar range at the Mngazana Estuary (nitrite 0.9 μM + nitrate 7.0 μM) (Emmerson, 2005) and the Kromme (nitrite 0.4 μM + nitrate 4.6 μM) but lower than at the Sundays Estuary (nitrite 12.8 μM + nitrate 44.9 μM). Total oxidized nitrogen concentrations were significantly higher during the spring than the neap tide. This is similar to studies by Bava and Seralathan (1999) who suggest that at low tides, water seeps out of the sediment into the adjoining creek and estuary. This 'tidal pumping' exports dissolved nutrients from mangrove sediments to overlying water. Phosphate concentrations were moderate (1.3 and 1.9 μM) in the salt marsh and mangrove–salt marsh respectively and increased in the mangrove (3.7 μM). This was lower than at the Kromme (mean 3.9 μM), Sundays (mean 7.5 μM) (Emmerson, 1989), and Mngazana estuaries (mean 5.07 μM) (Emmerson, 2005). The rate of phosphate adsorption from the sediment and the form of dissolved inorganic nitrogen are dependent on the oxic state of the sediment (Lillebo et al., 2006). Phosphate was found to increase in the summer season and to be highest in the upper tidal zone. Phosphate originates from the breakdown of rocks in catchments and so increases with river flow (Grobler and Silberbauer, 1985). Runoff from the cliff on the landward side of the estuary would also result in increased phosphate concentrations in the upper tidal zone. This is supported by Grobler and Silberbauer (1985), who have shown a positive correlation between phosphate export and run off in several South African catchments. Emmerson (1989) reported N:P ratios for salt marsh dominated estuaries of 0.8:1 at the Kromme and 1.12:1 at the Swartkops while Emmerson (2005) reported an N:P ratio of 2.7:1 at Mngazana—a mangrove

Table 6
Summary of the CCA results for Transect 2 and 3 for the Nahoon Estuary macrophyte species and environmental data from February to November 2011 (P = 1.000).

		Axis1	Axis2	Total inertia
Transect 2				
Eigenvalues		0.369	0.052	0.420
Species-environment correlation		1.000	1.000	
Cumulative percentage variance	Of species data	87.800	100.000	
	Of species–environment relation	87.800	100.000	
Sum of all eigenvalues				0.420
Transect 3				
Eigenvalues		0.376	0.103	0.478
Species–environment correlation		1.000	1.000	
Cumulative percentage variance	Of species data	78.500	100.000	
	Of species–environment relation	78.500	100.000	
Sum of all eigenvalues				0.478

dominated estuary. This suggests that mangrove communities are generally more phosphate limited than salt marsh communities. At the Nahoon Estuary, however, the N:P ratios of the porewater suggested that both the mangrove and salt marsh communities were phosphate limited, particularly the salt marsh community which had the highest ammonium concentration.

A number of studies including Emmerson (1989), Davis et al. (2001), Scharler and Baird (2003), and Emmerson (2005) found positive correlations between river flow and nutrient concentrations. In the Nahoon River, ammonium concentration ranged from 1.2 to 28.2 μM and TOxN ranged from 0.8 to 33.9 μM having on average higher readings than the sampled sites in the estuary. High rainfall in the summer months increases river flow and terrestrial runoff into the estuary. We noticed increased levels of orthophosphate and ammonium in the porewater during the summer sampling seasons. Total oxidized nitrogen, however, appeared to decrease in the summer seasons, particularly in the mangroves. This is proposed to be as a result of the occurrence of dense algal mats. Nahoon Estuary supports a high biomass of macroalgae which form dense algal mats on the sediment surface in the mangrove and mangrove–salt marsh areas, particularly in the mid and upper tidal zones. Algal densities appeared to increase in the warmer summer months and decrease in the winter months. High sediment pH values across all three communities could result in a further loss of nitrogen. Reef et al. (2010) found that sediment pH greater than 7 could lead to a loss of nitrogen due to ammonia volatilization.

The results of the CCA plots showed that the vegetation distribution at the Nahoon Estuary might be strongly influenced by the physico-chemical factors. The distribution of *A. marina* is largely influenced by lower elevation conditions, further from the channel (Fig. 3), while the salt marsh species are generally more strongly influenced by higher elevation, porewater salinity, pH, and ammonium concentrations. Both mangroves and salt marsh are known to be influenced by elevation with most salt marsh species preferring higher elevations (Adam, 1990; Adam, 2002; Adams and Ngesi, 2002) and mangroves occupying lower in more regularly tidally inundated sediments (Steinke, 1995; Adame et al., 2010; Barbier et al., 2011). Mid–low tidal salt marsh species such as *Sarcocornia tegetaria* were found in areas where the soil moisture was high, closer to the channel, and at lower elevation above MSL. Similar trends were found by Bornman et al. (2008) in the Olifants Estuary and by Bezuidenhout (2011). Bornman et al. (2008) and Bezuidenhout (2011) similarly found that the upper tidal, terrestrial salt marsh communities were associated with more elevated soils with low moisture content and high salinity.

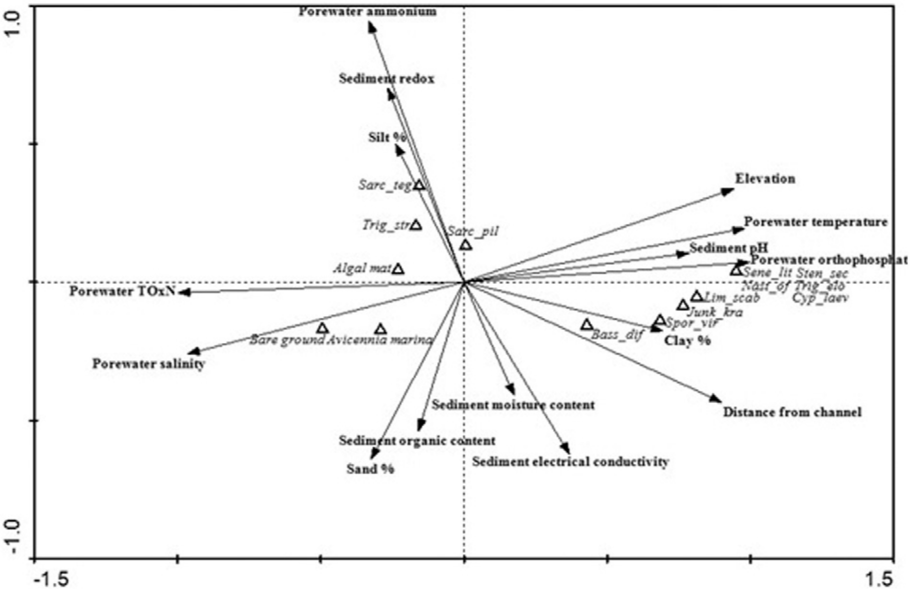

Fig. 3. CCA ordination plot of macrophyte species cover and environmental data of the mangrove–salt marsh habitat at Nahoon Estuary from February to November 2011. The arrows represent each environmental variable pointing in the direction of its maximum change. Plant names are abbreviated as follows: Sarc_teg = *Sarcocornia tegetaria*; Trig_Str = *Triglochin striata*; Sarc_pil = *Sarcocornia pillansii*; Sene_lit = *Senecio litorosus*; Nast_of = *Nasturtium officinale*; Lim_scab = *Limonium scabrum*; Junk_kra = *Juncus kraussii*; Spor_vir = *Sporobolus virginicus*; Bass_dif = *Bassia diffusa*; Sten_sec = *Stenotaphrum secundatum*; Trig_elo = *Triglochin elongate*; Cyp_laev = *Cyperus laevigatus*; Algal mat = Unidentified microalgae species).

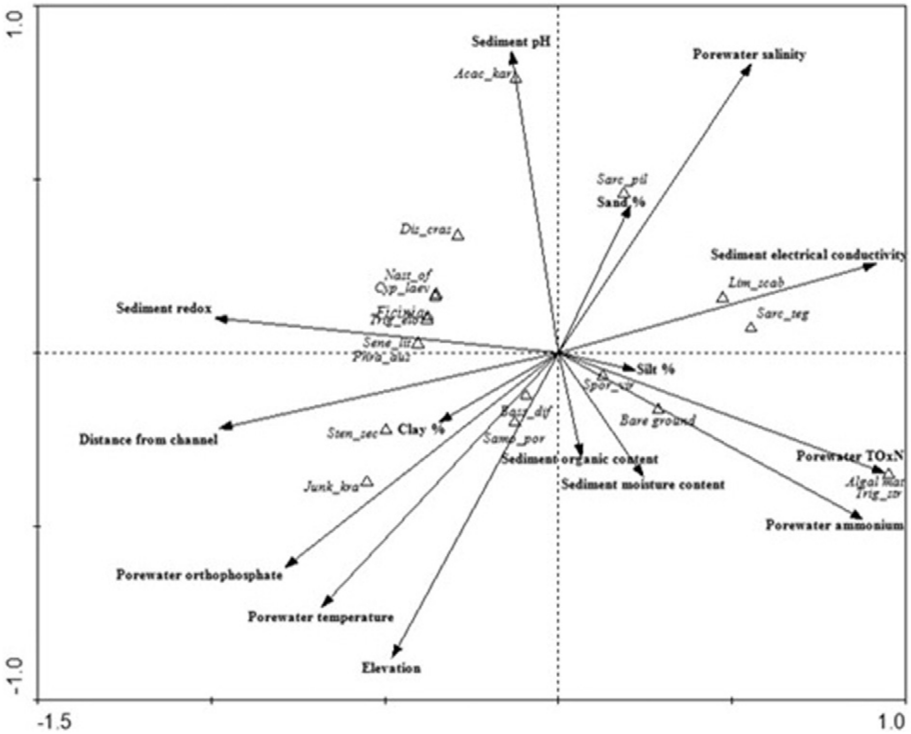

Fig. 4. CCA ordination plot of macrophyte species cover and environmental data of the salt marsh habitat at Nahoon Estuary from February to November 2011. The arrows represent each environmental variable pointing in the direction of its maximum change. Plant names are abbreviated as follows: Dis_cras = *Disphyma crassifolium*; Nast_of = *Nasturtium officinale*; Cyp_laev = *Cyperus laevigatus*; Ficinia = *Ficinia* spp.; Trig_elo = *Triglochin elongate*; Sene_lit = *Senecio litorosus*; Phrag_aus = *Phragmites australis*; Sarc_pil = *Sarcocornia pillansii*; Lim_scab = *Limonium scabrum*; Sarc_teg = *Sarcocornia tegetaria*; Spor_vir = *Sporobolus virginicus*; Trig_Str = *Triglochin striata*; Bass_dif = *Bassia diffusa*; Samo_por = *Samolus porous*; Sten_sec = *Stenotaphrum secundatum*; Junk_kra = *Juncus kraussii*; Algal mat = Unidentified microalgae species).

6. Conclusion

The objective of this study was to determine whether nutrient and sediment physio-chemical characteristics positively influence mangrove migration into the salt marsh and exclude salt marsh. This study showed that in the Nahoon Estuary, the vegetation distribution within the mangrove–salt marsh was primarily determined by the ammonium concentrations and the elevation above sea level while in the salt marsh porewater salinity and sediment pH were the most important factors determining species distribution. It was further found that salinity and temperature may be two major factors influencing growth and expansion of the mangroves at Nahoon Estuary. Further assessment of sedimentation rates combined with continuous assessment of nutrient fluctuations could reveal further insights into the potential for mangrove to expand into the salt marsh.

Acknowledgments

Rhodes University (JRC Funding) and the National Research Foundation (Thuthuka Grant–TTK20110819000025185) are thanked for the funding of this research. The Nahoon Estuary Nature Reserve is thanked for allowing us to work in the reserve. The Department of Botany at Rhodes University and the Nelson Mandela Metropolitan University is thanked for the use of equipment and facilities. The South African Weather Bureau provided the historical rainfall and temperature data. We also thank Dr Gavin Snow and Dimitri Veldkornet at the Nelson Mandela Metropolitan University. The authors thank all the field assistants who assisted in sampling. Finally, we thank the two anonymous reviewers for helping us improve the quality of the paper.

References

Adam, P., 1990. Saltmarsh Ecology. Cambridge University Press, Australia.

Adam, P., 2002. Saltmarshes in a time of change. Environmental Conservation 29, 39–61.

Adame, M.F., Neil, D., Wright, S.F., Lovelock, C.E., 2010. Sedimentation within and among mangrove forests along a gradient of geomorphological settings. *Estuarine, Coastal and Shelf Science* 86, 21–30.

Adams, J.B., Ngesi, N., 2002. An Assessment of the Salt Marsh of the Great Brak Estuary CSIR.

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

Alongi, D.M., 2002. Present state and future of the world's mangrove forests. Environmental Conservation 12, 331–349.

Alongi, D.M., 2009. The Energetics of Mangrove Forests. NY, USA, New York.

Armstrong, W., 1982. Waterlogged soils. In: Etherington, J.R. (Ed.), Environment and Plant Ecology. John Wiley & Sons, Chichester, pp. 290–330.

Barbier, E.B., Hacker, S.D., Kennedy, C., Koch, E.W., Stier, A.C., Silliman, B.R., 2011. The value of estuarine and coastal ecosystem services. Ecological Society of America 81, 169–193.

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

Bava, K.A., Seralathan, P., 1999. Interstitial water and hydrochemistry of a mangrove forest and adjoining water system, south west coast of India. Environmental Geology 38, 47–52.

Bezuidenhout, C., 2011. Macrophytes as Indicators of Physic–Chemical Factors in South African Estuaries. PhD. Nelson Mandela Metropolitan University, South Africa.

Black, C.A., 1965. Methods of Soil Analysis: Part I Physical and Mineralogical Properties. American Society of Agronomy, Madison, WI, USA.

Bornman, T.G., Adams, J.B., 2008. Response of a hypersaline salt marsh to a large flood and rainfall event along the west coast of southern Africa. Estuarine, Coastal and Shelf Science 87, 378–386.

Bornman, T.G., Adams, J.B., Bate, G.C., 2002. Freshwater requirements of a semi-arid supratidal and floodplain salt marsh. Estuaries 25, 1394–1405.

Bornman, T.G., Adams, J.B., Bate, G.C., 2008. Environmental factors controlling the vegetation zonation patterns and distribution of vegetation types in the Olifants Estuary, South Africa. South African Journal of Botany 74, 685–695.

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

Chapman, V.J., 1960. Salt Marshes and Salt Deserts of the World. Interscience Publishers, New York, p. 928.

Chapman, V.J., 1974. Salt Marshes and Salt Deserts of the World. second ed. Leonard Hill, London, UK.

Chapman, V.J., 1975. Mangrove Biogeography. In: International Symposium on Biology and Management of Mangroves. University of Florida Press, Miami, pp. 179–212.

Clarke, P.J., 1985. Nitrogen pools and soil characteristics of a temperate estuarine wetland in Eastern Australia. Aquatic Botany 23, 275–290.

Clarke, L.D., Hannon, N.J., 1967. The mangrove swamp and salt marsh communities of the Sydney district. I. Vegetation, soils and climate. Journal of Ecology 55, 753–771.

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

CSIR Report ENV-S, Wooldridge, T., Taljaard, S., Huizinga, P., van Niekerk, L., Reddering, K., Baird, D., Adams, J., Bate, G., Walker, D., Whitfield, A., J., T., 2000. Determination of Intermediate Resource Directed Measures: Nahoon Estuary. pp. 1–81.

Davis, S.E., Childers, D.L., Day, J.W., Rudnick, D.T., Sklar, F.H., 2001. Nutrient dynamics in vegetated and unvegetated areas of a Southern Everglades mangrove creek. *Estuarine, Coastal and Shelf Science* 52, 753–768.

Emmerson, W.D., 1989. The nutrient status of the Sundays river estuary South Africa. Water Resources 23, 1059–1067.

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

Emmerson, W.D., Erasmus, T., 1987. The nutrient status of the Kromme river estuary, South Africa. Hydrobiologia 148, 87–96.

Gee, G.W., Bauder, J.W., 1986. Particle-size analysis. In: Klute, A. (Ed.), Methods of Soil Analysis; Part I—Physical and Mineralogical Methods. (2nd eds). Agronomy Nr9 Part 1. Madison, Wisconsin, USA, pp. 383–411.

Geldenhuys, C., 2014. Mangrove and salt marsh dynamics at Nahoon Estuary, Eastern Cape: a planted mangrove forest. MSc Thesis. Botany Department, Rhodes University.

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

Green, M.O., Hancock, N.J., 2012. Sediment transport through a tidal creek. *Estuarine, Coastal and Shelf Science* 109, 116–132.

Grobler, D.C., Silberbauer, M.J., 1985. The combined effect of geology, phosphate sources and runoff on phosphate export from drainage basins. Water Research 19, 975–981.

Gubelit, Y.I., 2009. Biomass and primary production of *Cladophora glomerata* (L.) Kutz. In the Neva Estuary. Inland Water Biology 4, 300–304.

Harrison, T.D., Hohls, D.R., Meara, T.P., Webster, M.S., 2001. South African estuaries: catchment land-cover. National Summary Report. Department of Environmental Affairs and Tourism. Pretoria 1–30.

Hegazy, A.K., 1998. Perspectives on survival, phenology, litter fall and decomposition, and caloric content of *Avicennia marina* in the Arabian Gulf region. Journal of Arid Environments 40, 417–429.

Hogarth, P.J., 1999. The biology of mangroves. Oxford University Press, Oxford.

Hoppe-Speer, S.C.L., 2015. Response of Mangroves in South Africa to Anthropogenic and Natural Impacts. PhD. Nelson Mandela Metropolitan University, Port Elizabeth, South Africa.

Hoppe-Speer, S.C.L., Adams, J.B., Rajkaran, A., 2015. Mangrove expansion and population sturcutre at a planted site, East London. South Africa. Southern Forests: a Journal of Forest Science. http://dx.doi.org/10.2989/20702620.2014.1001622.

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

Lear, R.J.C., Turner, T., 1977. Mangroves of Australia. University of Queensland Press, St. Lucia Australia.

Lillebo, A.I., Flindt, M.R., Pardal, M.A., Marques, J.C., 2006. The effect of *Zostera noltii*, *Spartina maritima* and *Scirpus maritimus* on sediment pore-water profiles in a temperate intertidal estuary. Hydrobiologia 555, 175–183.

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

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

MEGA, 2013. Estuary management plan for the Nahoon Estuary. Parts I and II. Economic Development Environmental Affairs and Tourism, Eastern Cape. Draft Report: J-616-13.

Mitchell, M.L., Adam, P., 1989. The relationship between mangrove and saltmarsh communities in the Sydney region. *Wetlands* (*Australia*) 8, 37–46.

Morris, J.T., 1980. The nitrogen uptake kinetics of *Spartina altemiflora* in culture. Ecology 61, 1114–1121.

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

Nirmal Kumar, I.J., Sajish, P.R., Nirmal Kumar, R., Basil, G., Shailendra, V., 2011. Nutrient dynamics in an *Avicennia marina* (Forsk.) Vierh., mangrove forest in Vamleshwar, Gujarat, India. Notulae Scientia Biologicae 3, 51–56.

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

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

Qureshi, M.T., 1993. Rehabilitation and management of mangrove forests of Pakistan. In: Lieth, H., Al Masoom, A. (Eds.), Towards the Rational Use of High Salinity Tolerant Plants 1. Kluwer Acadamic Publishers, The Netherlands, pp. 89–95.

Raffaelli, D., Raven, J.A., Poole, L.I., 1999. Ecological impact of green macroalgal blooms. Oceanogr Mar Biol Ann Rev 36, 97–125.

Rajkaran, A., Adams, J., 2012. The effects of environmental variables on mortality and growth of mangroves at Mngazana Estuary, EasternCape, South Africa. Wetlands Ecology Management 20, 297–312.

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

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

Robertson, A.I., Alongi, D.M., 1992. Tropical Mangrove Ecosystems. American Geographical Union, Washington DC.

Santos, I.R., Eyre, B.D., Huettel, M., 2012. The driving forces of porewater and groundwater flow in permeable coastal sediments: a review. *Estuarine, Coastal and Shelf Science* 98, 1–15.

Scharler, U.M., Baird, D., 2003. The influence of catchment management on salinity, nutrient stochiometry and phytoplankton biomass of Eastern Cape estuaries, South Africa. *Estuarine, Coastal and Shelf Science* 56, 735–748.

Smith III, T.J., Boto, K.G., Fruscher, S.D., Giddins, R.L., 1991. Keystone species and mangrove forest dynamics: the influence of burrowing by crabs on soil nutrient status and forest productivity. *Estuarine, Coastal and Shelf Science* 33, 419–432.

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

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

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

ter Braak, C.J.F., Šmilauer, P., 2002. CANOCO reference manual CanoDraw for Windows user's guide: software for canonical community ordination (version 4.5). Microcomputer Power, Ithaca, NY, US.

The Non-Affiliated Soil Analyses Working Committee, 1990. Handbook of Standard Soil Testing Methods for Advisory Purposes. Soil Science Society of South Africa, Pretoria.

Trott, L.A., Alongi, D.M., 2000. The impact of shrimp pond effluent on water quality and phytoplankton biomass in a tropical mangrove estuary. Marine Pollution Bulletin 40, 947–951.

Turpie, J.K., Adams, J.B., Joubert, A., Harrison, T.D., Colloty, B.M., Maree, R.C., Whitfield, A.K., Wooldridge, T.H., Lamberth, S.J., Taljaard, S., Van Niekerk, L., 2002. Assessment of the conservation priority status of South African estuaries for use in management and water allocation. Water SA 28, 191–206.

Vernon, C.J., 2013. Drowning in mud: impact by the 2011 flood on the Avicennia at Nahoon Estuary (Unpublished).

Ward, C.J., Steinke, T.D., 1982. A note on the distribution and approximate areas of mangroves in South Africa. South African Journal of Botany 1, 51–53.