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Blue Carbon at the southern tip of Africa: current knowledge and future perspectives for dynamic estuarine environments

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A R T I C L E I N F O

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Tidal ecosystems
Salt marsh
Seagrass

A B S T R A C T

Blue Carbon Ecosystems (BCEs), specifically salt marsh, seagrass, mangroves, occur in South Africa's relatively small, sheltered estuaries that are often disconnected from the ocean. These are dynamic environments where shifts between BCEs and other habitats along ecotones occur in response to mouth changes, floods and droughts, as well as anthropogenic pressures. Although Blue Carbon is becoming well established in South Africa, critical knowledge gaps remain; these are summarised under seven themes and future research and management actions identified. A holistic approach is recommended for Blue Carbon studies in estuaries to measure across elevation gradients (rather than focusing on individual vegetation types) and to include reeds, sedges and forested wetlands. Additionally, quantifying data deficient carbon stocks and processes, modelling future climate change impacts, instilling a sustainable long-term monitoring program, incorporating relevant emerging blue carbon stocks, realizing nationally inclusive restoration and protection co-management plans, and aligning local approaches with global frameworks of reporting are advocated as future recommendations with respect to South African BCEs. South Africa has high biodiversity and unique pressures influencing BCEs and is well positioned to inform the global research agenda. While the limited spatial extent of BCEs restricts the feasibility of carbon credit opportunities, high biodiversity values of these ecosystems hold potential under emerging 'nature credit' frameworks.

1. Introduction

Blue Carbon' Ecosystems (BCEs) are differentiated from the 'green carbon' of terrestrial ecosystems due to their marine association and include habitats such as mangroves, salt marshes and seagrasses (Nellemann et al., 2009). One of the most valued ecosystem services by BCEs is their carbon storage and sequestration (Macreadie et al., 2021). The vegetation within these ecosystems takes up atmospheric carbon dioxide by means of photosynthesis and then stores this organic carbon within the sediments, living biomass and non-living biomass with the

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sediments storing the highest proportion of carbon (McLeod et al., 2011; Howard et al., 2014; Duarte, 2017). Trapping of suspended particles during tidal inundation allows BCEs to store more carbon than that of the above-ground standing biomass, with vertical accumulation of sediment preventing the saturation of soils with carbon (McKee et al., 2007; McLeod et al., 2011).

Restoration actions in BCEs have direct biodiversity and socio-ecological benefits (Macreadie et al., 2021, 2022; Pétillon et al., 2023). These benefits are lost when the vegetation and associated sediment of BCEs are degraded, completely removed, or converted, at which point these ecosystems become sources of Greenhouse Gas (GHG) emissions (CO_2, CH_4 and N_2O). There is the potential for large releases of GHGs from the organic matter stored over centuries, thus making BCE preservation and restoration crucial (Pendleton et al., 2012; Duarte et al., 2013; Rosentreter et al., 2023). A global effort to quantify carbon stocks and sequestration rates has therefore become established, with the aim to understand the role that BCEs can play simultaneously in climate mitigation and conservation initiatives (i.e. https://www.thebluecarboninitiative.org/). Additionally, there is increased scientific, political and economic interest to investigate the potential of blue carbon for emissions offsetting, either through the voluntary carbon market, or as part of Nationally Determined Contributions (NDCs) (Macreadie et al., 2019; Friess et al., 2022). To be truly effective however, this requires clear and unambiguous blue carbon definitions, with data-driven science and actionable human management as a cornerstone of blue carbon sink protection and restoration (Howard et al., 2023; Sheehy et al., 2024).

Carbon stock estimates in Africa are limited, with very few studies represented in global datasets, despite BCEs being present along much of the continent's coastline (Friess et al., 2019; Adams, 2020; Mwikamba et al., 2024). South Africa has ~3000 km of high-energy coastline with BCEs, including salt marshes, seagrasses and mangroves, occurring in sheltered estuaries. Blue carbon research has been a focus in South Africa since 2017 (Adams et al., 2019), with regional studies based on the international standard methodology outlined in the Blue Carbon Manual developed by Conservation International and the International Union for Conservation of Nature (Howard et al., 2014). South African research has quantified blue carbon stocks at specific locations, such as the Knysna and Swartkops estuaries (Wasserman et al., 2023), and for BCE types across sites in the cool temperate, warm temperate and subtropical biogeographical regions of South Africa (Johnson et al., 2020; Ndhlovu et al., 2024). Habitat-specific impacts of climate change have been investigated for both salt marshes (Adams, 2020; Raw et al., 2020, 2021) and mangroves (Raw et al. 2019a, 2023b).

A national blue carbon sinks assessment adhering to Greenhouse Gas Inventories (GHGI) reporting was completed in South Africa in 2021 (DFFE, 2021) along with a policy document to integrate the findings for the conservation and restoration of BCEs. This first mapped the existing distribution of BCEs and quantified their carbon storage, sequestration and sink potential, created a GHG-CO_2 emissions and removal baseline for BCEs and identified opportunities to reduce GHG emissions from sources while enhancing carbon sinks. Once the potential for restoration of BCEs was established, the research focus shifted to restoration and Target 2 and 3 of the 2030 Kunming-Montreal Global Biodiversity Framework to expand conservation and restoration to 30 % by 2030 (CBD, 2022; Adams et al., 2023). Research relevant to BCEs has been published on the need for restoration (Raw et al., 2023b; Mokumo et al.), and the development of restoration plans (Adams et al., 2023). Despite this early research in South Africa, better estimates of carbon stocks, sequestration rates and GHG fluxes are needed to facilitate comparisons and inclusion into global datasets as the assessments described were based on a limited field data set of *in situ* carbon measurements. Expanding BCE research in South Africa will need to include representation of all bioregions and diverse estuarine types, and coverage on larger and regular temporal scales for national monitoring and reporting. Globally, Blue Carbon research is unequally distributed as research from the northern hemisphere dominates outputs (Hatje et al., 2023; Holmquist et al., 2024), while Australia is the hub of activity in the southern hemisphere, with comparatively little thus far from Africa (Quevedo et al., 2023). Considering these challenges locally and globally, this study contributes by investigating the current knowledge of Blue Carbon research and management in South Africa, and identifies knowledge gaps for future research, policy and management action. This has relevance for similar developing nations of the global south with emerging BCE research agendas. Improved monitoring, assessment and reporting is needed, to support reporting to various global indicators within the next decade, including the GHGIs, Global Biodiversity Framework and Sustainable Development Goals.

2. Study approach

A workshop of specialists was convened from universities and institutes conducting BCE research in South Africa to provide input on ongoing Blue Carbon studies and identify research gaps. All participants at the workshop are authors of this manuscript. The specialists included have a track record as a lead author on Blue Carbon topics in South Africa or are busy with research on the topic (i.e. postgraduate students). Prior to the workshop, participants submitted research questions to start the discussion and inform research themes (Supplementary Table 1). From the research questions submitted and discussions at the workshop, seven broad Blue Carbon themes and recommendations for future research and management action were identified (Table 1). Additional recommendations were provided to improve monitoring and reporting of BCE extent and stocks and facilitate reporting to national and global indicators. Measured carbon stocks of each BCE from published studies were collated and presented in Table 2 to identify ecosystem type and regional data gaps. Aerial images and field photographs were used to illustrate the distribution and dynamics of BCEs along narrow intertidal

Table 1
Recommendations for future Blue Carbon Ecosystem research and management actions.

Keywords	Details
Holistic estuary assessments	• Adopt holistic approach for Blue Carbon studies measuring across different estuary habitat types as seagrass, salt marsh and mangroves overlap with reeds, sedges and forested wetlands.
Carbon stocks, sequestration & GHG fluxes	• Quantify BCE carbon stocks, sequestration and GHG fluxes in ecosystems and regions that are data deficient using standardized approaches. Data deficient regions are cool temperate, subtropical and tropical bioregions.
Drivers & responses	• Drivers and responses of BCE stocks, sequestration and GHG fluxes to be considered across different spatio-temporal scales and modelled for future climate change scenarios.
Climate change	• Implement long-term monitoring and research to understand how BCEs respond to climate change stressors like rising temperatures and sea level, storm surges, floods, droughts and reduced freshwater inflow. Range expansions and shifts between BCE types and other habitats along ecotones are expected.
Emerging BCEs	• Investigate whether emerging BCEs (seaweeds/kelps, biofilms/mudflats) have long term organic carbon storage in their soils and low greenhouse gas emissions due to high primary productivity and low decomposition rates in waterlogged soils.
Protection & restoration	• Design and implement a national protection and restoration plan for BCEs focusing on co-management. Formal protection of BCEs must be increased to meet the GBF targets.
Monitoring & reporting systems	• Develop a national monitoring and reporting system to understand current and future threats to BCEs and track changes to facilitate reporting to national and global indicators.

Table 2
Measured carbon stocks of each Blue Carbon Ecosystem from different studies conducted in South Africa.

Estuaries (listed from west to east)	Habitat extent (ha)	Measured Sediment C (Mg C ha $^{-1}$)	Biomass C (Mg C ha $^{-1}$)		Total C pool (Mg C)	Reference
			Above ground	Below ground		
SEAGRASS - *Zostera capensis*						
Olifants	47.74	10.15 ± 3.3	–	–	484.47 ± 157.3	Ndhlovu et al. (2024)
Groot Berg	206	18.4 ± 13.8	–	–	3789.78 ± 2842.8	Ndhlovu et al. (2024)
Breede	2.5	19.71 ± 2.93	–	–	49.28 ± 7.33	Ndhlovu et al. (2024)
Knysna	353	30.54 ± 11.37	–	–	10780.97 ± 4012.2	Ndhlovu et al. (2024)
Knysna	353	56.18 ± 31.16	1.02 ± 0.04	1.56 ± 0.7	19832.2 ± 10 999.5	Wasserman et al. (2023)
Swartkops	44.7	280.69 ± 84.56	1.47 ± 0.85	0.6 ± 0.36	12546.8 ± 3779.9	Wasserman et al. (2023)
Swartkops	62.3	181.37 ± 45.41	–	–	11299.09 ± 2828.88	Human et al. (2022)
Swartkops	62.3	224.14 ± 37.93			129.58 ± 30.53	Els (2019)
Swartkops	44.7	104.78 ± 102.37	–	–	4683.8 ± 4575.94	Ndhlovu et al. (2024)
Mngazana	2	46.18 ± 25.3	–	–	92.37 ± 50.6	Ndhlovu et al. (2024)
Nxaxo	0.04	115.45 ± 33.9	1.66 ± 0.66	1.34 ± 0.57	4.26 ± 1.36	Wasserman et al. (2023)
Nxaxo	0.04	1.67 ± 0.01	–	–	–	Banda et al. (2021)
SALT MARSH – dominant species as footnote						
Swartkops[2]	96	247.13 ± 47.71	10.63 ± 0.66	6.58 ± 0.56	1561.92 ± 274.56	Els (2019)
Swartkops[3]	27.32	212.26 ± 43.99	–	–	116.93 ± 19.67	Els (2019)
Swartkops[2]	96	225.45 ± 101.12	–	–	21643.07 ± 9706.24	Human et al. (2022)
Swartkops[3]	27.32	210.67 ± 72.19	–	–	5755.62 ± 1972.12	Human et al. (2022)
Nahoon[5]	1.45	109.62 ± 22.0	–	–	158.94 ± 31.9	Raw et al. (2019b)
Nxaxo[6]	10.9	2.61 ± 0.19	–	–	–	Banda et al. (2021)
MANGROVES – *Avicennia marina*						
Nahoon	2.55	110.14 ± 11.0	–	–	280.86 ± 28.1	Raw et al. (2019b)
Nxaxo	9.5	228.05 ± 27.99	–	–	–	Banda et al. (2021)
Nxaxo	9.2	186.79 ± 8.12 to 270.71 ± 18.62	–	176.91 ± 4.48 to 262.53 ± 18.87	–	Johnson et al. (2020)

Table footnote …

	Salt marsh species
Swartkops[1]	*Salicornia tegetaria*
Swartkops[2]	*Spartina maritima*
Swartkops[3]	*Salicornia tegetaria*
Swartkops[4]	*Triglochin striata, Bassia diffusa* and *Salicornia tegetaria*
Nahoon[5]	
Nxaxo[6]	

gradients and ecotonal habitats in South African estuaries.

Interacting factors that influence carbon stocks and sequestration at different scales to be considered in future investigations were identified. The responses of BCEs to climate change stressors (CO_2, temperature, floods, droughts, freshwater inflow, sea-level rise, sea storms and wave height) were synthesized from available knowledge and used to propose future research to measure and model climate change impacts. In addition, a flow chart was constructed to show the steps to consider when planning, sampling, measuring and reporting Blue Carbon studies and how to compile metadata. This study is the first step towards implementation of a standardized approach for Blue Carbon assessments in South Africa based on global best practice. It maps out a pathway to action for management that would increase protection of BCEs if existing policy and sustainable land management practices were effectively implemented.

3. Distribution and features of Blue Carbon Ecosystems in South Africa

The southern tip of Africa has high biodiversity influenced by two contrasting ocean currents: the cooler Benguela Current along the western and the warm Agulhas Current on the eastern coastline (Griffiths et al., 2010; Harris et al., 2025). The South African coastline comprises four biogeographic regions (Fig. 1) and the distribution of BCEs is influenced by temperature; mangroves only occur along the tropical and subtropical bioregions on the east coast, with salt marshes occurring more extensively in estuaries along the southern and western coastline. Seagrasses are restricted to 37 estuaries spanning the biogeographic provinces (Fig. 1). All BCEs provide important ecosystem services as indicated in Fig. 1. Carbon storage per unit area (biomass and sediment) has been reported as mangroves with the highest values (253–534 Mg C ha^{-1}), followed by salt marshes (100–199 Mg C ha^{-1}) and seagrasses (45–144 Mg C ha^{-1}) (Raw et al., 2023a).

Southern hemisphere estuaries differ from northern hemisphere systems in that they are predominantly microtidal (tidal range < 2 m) and small in extent. In South Africa ~70 % of estuaries are less than 50 ha with more than 75 % closing to the sea for varying periods of time when a sandbar forms across the mouth and prevents marine connectivity (Van Niekerk et al., 2019). These are wave dominated systems with limited floodplain and intertidal area as they are incised, drowned river valleys that are immature in terms of geological development (Cooper, 2001). The mean annual run-off of most rivers is variable, fluctuating between floods and extremely low to zero flow; during low flow many smaller estuaries remain closed to the sea (Van Niekerk et al., 2019). These factors limit the establishment of BCEs in estuaries, but these dynamic ecosystems are characterized by high biodiversity even along the narrow intertidal zone (Fig. 2a–f, Supplementary Fig. 1).

In South Africa, estuarine habitat is considered to include all areas in the estuarine functional zone (EFZ) i.e. areas below the 5 m above mean sea level (MSL) contour line mapped at a scale of 1:10 000 orthophoto

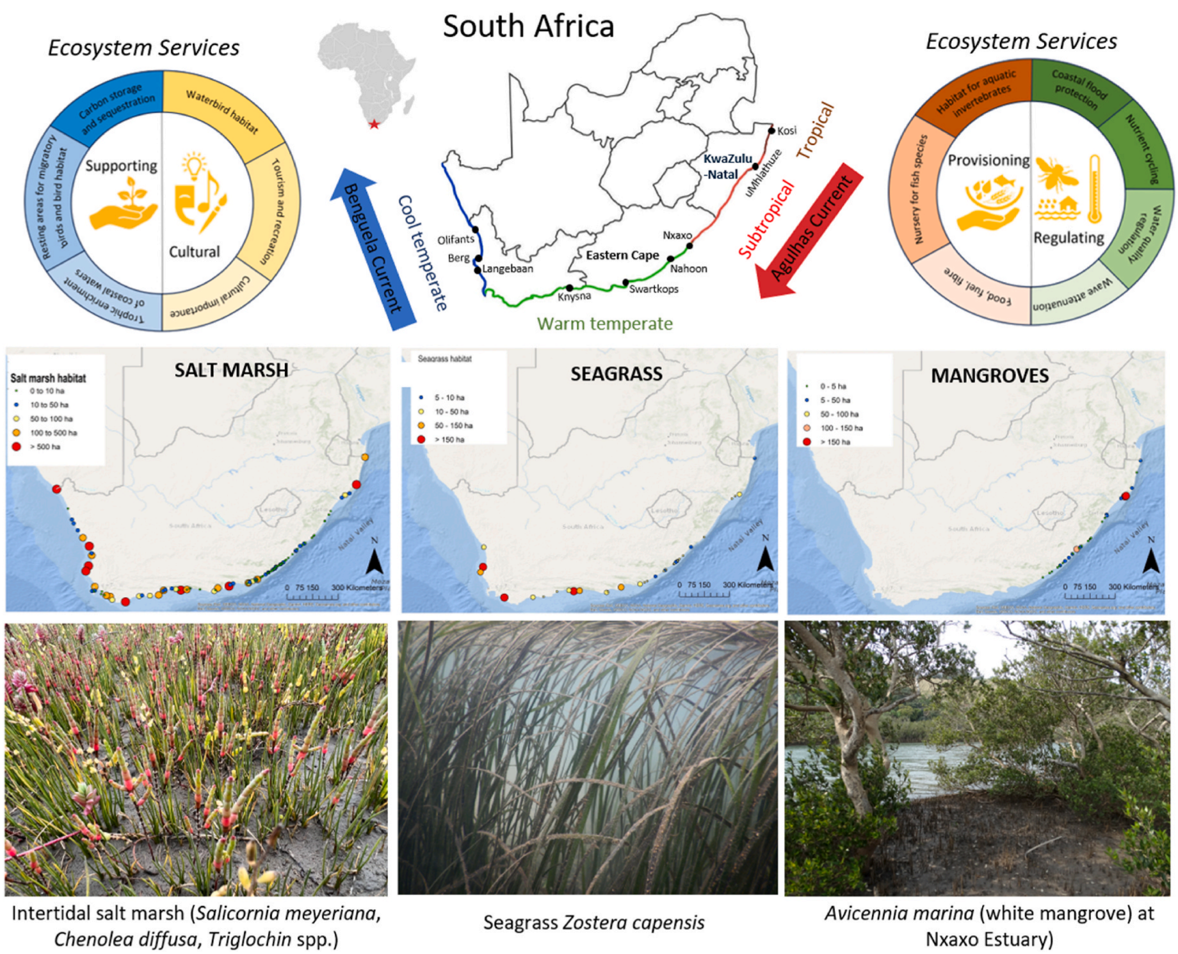

Fig. 1. Blue Carbon Ecosystems in South African estuaries across the four biogeographic zones and key services and benefits of these ecosystems. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

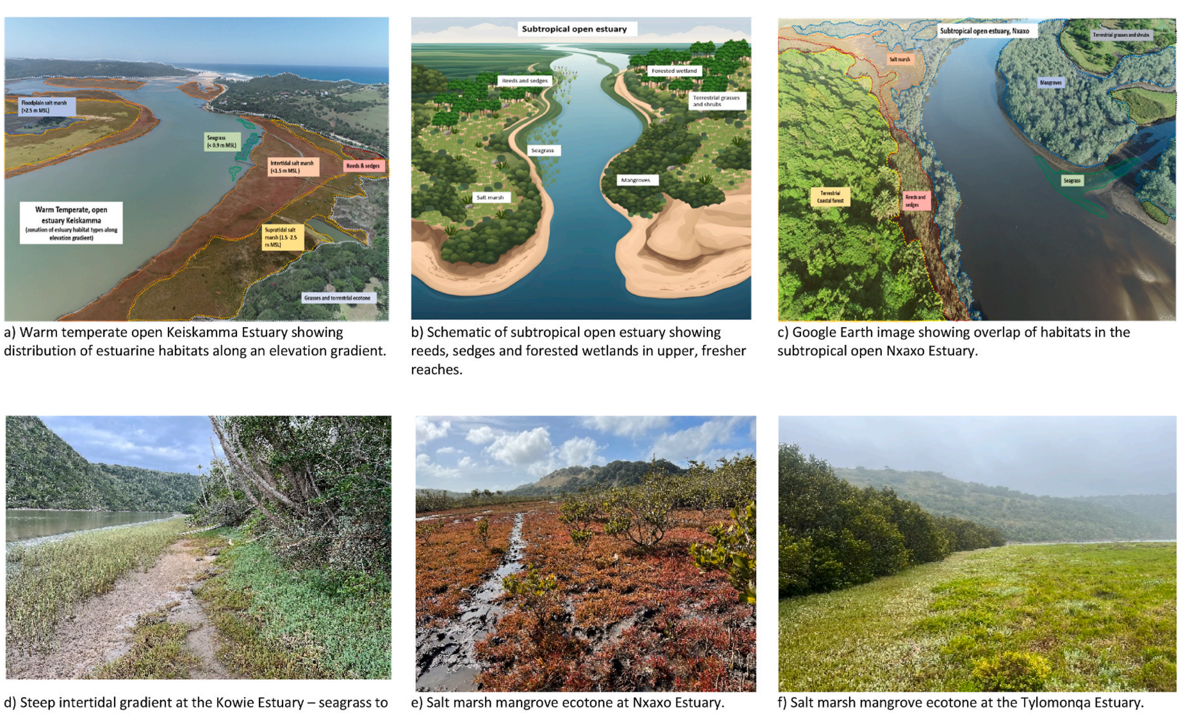

Fig. 2. Distribution of estuarine and ecotonal habitats in temperate and subtropical South African estuaries.

maps (Van Niekerk et al., 2019). The EFZ is naturally dynamic, and therefore the BCEs described below do naturally shift across alternative stable states, for example transitioning between seagrass and mudflats. The consequences of this in terms of carbon sequestration and storage is an emerging research topic. In some steeply incised estuaries along the subtropical region the estuarine functional zone extends to the 10 m contour above MSL to accommodate large floods and dynamic erosion/depositional cycles. Reeds, sedges, and forested wetlands (swamp forest) are a common feature and are therefore included in the summary below.

3.1. Salt marsh

Salt marsh ecosystems occur across all biogeographic regions of the South African coastline and cover an aggregate area of ~15 000 ha with the largest cover in the Groot Berg, Langebaan, Olifants, Knysna and St Lucia estuaries (Fig. 1). Salt marshes include herbs, grasses, or low shrubs (Supplementary Fig. 1) that occur from the intertidal zone to the terrestrial ecotone (Adams, 2020) and are therefore exposed to flooding from tidal or non-tidal variation in water level (Yando et al., 2023). Intertidal salt marsh is dominant in the warm temperate region, while supratidal salt marsh is more abundant in the cool temperate (Fig. 1). Although supratidal and floodplain marshes are perched and seldom flooded, these areas support halophytic macrophyte communities and are therefore included as 'salt marsh' ecosystems (Veldkornet et al. 2015a, 2015b). Floodplain salt marsh relies on groundwater during dry months of the year to survive, but water table levels are linked to tidal rise and fall as well as the water level in the estuary. Short seasonal rainfall is important to recharge groundwater, reduce salinity and thereby allow the plants to grow and reproduce (Bornman et al., 2002).

Anthropogenic pressures in South African estuaries that impact salt marsh ecosystems include: restricted tidal exchange, freshwater abstraction (Bornman et al., 2002), mining and windblown dust (Bornman et al., 2004), estuary mouth closure and associated increase in water level (Riddin and Adams, 2019), eutrophication (Nunes and Adams, 2014), spread of invasive plants (Riddin et al., 2016), and livestock browsing and trampling (Adams, 2020) (Supplementary Fig. 1). These pressures are often overlapping and cumulative. Approximately 43 % of salt marsh area in South Africa has been lost mostly due to encroaching development and agriculture since the 1930s (Adams, 2020). Furthermore, invasive species are common in disturbed riparian areas and in catchments where they can reduce freshwater inflow to the downstream estuary. However, halophytic invasive plants are not common in South Africa salt marshes, except for the marsh grass *Spartina alterniflora* (Supplementary Fig. 1) which was successfully removed from the Groot Brak Estuary and has not been noted in South Africa since 2016 (Adams et al., 2016a; Riddin et al., 2016).

3.2. Seagrass

Seagrass in South Africa is dominated by *Zostera capensis* Setchell (*Nanozostera capensis;* Cape Dwarf-eelgrass) that covers ~1500 ha. Of the 37 estuaries it occurs in, it is most extensive in the Olifants, Langebaan, Groot Berg, Knysna, Keurbooms and Swartkops estuaries (Adams, 2016). It extends from the cool-temperate west coast to the subtropical and tropical east coast of South Africa and has also been recorded from the coasts of Mozambique and Kenya (Phair et al., 2019). *Zostera capensis* is abundant in the intertidal zone of estuaries predominantly open to the sea but can also be found in temporarily closed estuaries when the conditions remain saline (>15; Adams, 2016). In temporarily closed estuaries *Z. capensis* occurs with other submerged macrophyte species (e.g. *Ruppia cirrhosa*, *Stuckenia pectinata*) and algae, which creates uncertainty and challenges around the mapping of the extent of a specific seagrass species (Supplementary Fig. 2). *Halophila ovalis* (Supplementary Fig. 2) occurs below the *Z. capensis* zone and its distribution is undescribed. South African seagrass habitats are highly dynamic and vary in extent due to floods, droughts and human pressures (Talbot et al., 1990; Adams, 2016). In the KwaZulu-Natal Province along the east coast, *Z. capensis* is mostly absent (Fig. 1) due to the dominance of closed estuaries which are fresh, characterized by high turbidity and frequent freshwater flooding, preventing its establishment and persistence.

Recent research has shown that *Z. capensis* populations are severely fragmented because of the patchiness of suitable habitat (Smit et al., 2023; Combrink et al., 2024; von der Heyden et al.). *Zostera capensis* is slow to colonise and is influenced by increasing anthropogenic threats which lead to listing as Endangered on the South African Red List of Ecosystems in 2018 (Adams and Van der Colff, 2018) and globally, according to IUCN species criteria, given its very small area of occupancy mapped to ~11–13 km^2 (Watson et al., 2024). Seagrass meadows continue to experience loss and degradation from extended estuary closures, dredging, bait digging, trampling, competition from invasive aquatic species, recreational disturbances (Supplementary Fig. 2), and pollution through persistent organic pollutants and microplastics (Olisah et al., 2021; Van Wyk et al., 2022; Boshoff et al., 2025). Blooms of filamentous green algae (e.g. *Cladophora* and *Ulva* spp.) in response to eutrophication have caused declines in *Z. capensis* area in the Groot Brak and Knysna estuaries (Nunes and Adams, 2014; Human et al., 2016), with population losses in Langebaan driven by changes in hydrodynamic patterns and more direct pressures such as bait digging (Pillay et al., 2010). In South Africa, Estuary Management Plans are used to zone estuaries to reduce impacts on BCEs; while 46 % of the estuaries where *Z. capensis* occurs have these plans, there is minimal enforcement of activities impacting seagrass (Adams, 2016) resulting in *Z. capensis* having little direct protection.

3.3. Mangroves

Mangroves in South Africa occur along the eastern coastline in the warm temperate, subtropical and tropical bioregions from the Tyolomnqa to Kosi estuaries, with a total cover of ~2400 ha (Fig. 1, Supplementary Fig. 4). Mangroves here are at one of the southernmost locations in the world and occur in different estuarine types ranging from predominantly open estuaries, large temporarily closed estuaries, estuarine lakes, bays, and large fluvially dominated estuaries (Van Niekerk et al., 2019; Adams and Rajkaran, 2021). The dominant species are *Avicennia marina* (White mangrove), *Bruguiera gymnorrhiza* (Black mangrove), and *Rhizophora mucronata* (Red mangrove). Three additional species, *Ceriops tagal* (Indian mangrove), *Lumnitzera racemosa* (White-flowered black mangrove), and *Xylocarpus granatum* (Mangrove cannonball tree) only occur in the tropical Kosi Estuary (Fig. 1). *Bruguiera gymnorrhiza* has the widest distribution of the five mangrove species in South Africa and is found in all estuaries that support mangroves in the Eastern Cape Province and some estuaries of the KwaZulu-Natal Province (Fig. 1). This species can grow in areas with low salinity and in drier sediments (Hoppe-Speer et al., 2015a).

Mangroves have been recorded in 31 estuaries along the South African coastline, with the largest coverage, at just over 60 % of the country's mangrove expanse, in the uMhlathuze Estuary (Adams and Rajkaran, 2021; Riddin et al., 2024), with other large mangrove areas found in Richards Bay harbour and the Mngazana Estuary. Pressures faced by mangrove ecosystems in South Africa include: coastal development (Rajkaran and Adams, 2011), harvesting for building material and firewood (Adams et al., 2004; Rajkaran and Adams, 2010), livestock browsing and trampling (Hoppe-Speer and Adams, 2015), restricted tidal exchange (Hoppe-Speer et al., 2013), freshwater abstraction (Mbense et al., 2016), heavy metal pollution (Naidoo et al., 2014), oil pollution (Naidoo et al., 2010), micro- and macroplastic pollution (Govender et al., 2020; Johnson et al., 2023), and eutrophication (Geldenhuys et al., 2016). Stochastic events such as estuary mouth closure and back flooding can cause the collapse of the entire mangrove ecosystem as has been observed at 11 small estuaries in KwaZulu-Natal.

The mangroves are sensitive to inundation caused by the high-water levels as this causes anoxia interrupting gas exchange (Rajkaran et al., 2009; Adams and Rajkaran, 2021). While natural regeneration is taking place in some estuaries, mangrove losses are expected to increase under climate change as extreme events such as storm surges and reduced freshwater inflow are predicted to increase in frequency and intensity (Raw et al., 2023a).

3.4. Reeds and sedges

Estuarine habitats (those occurring below the 5 m above MSL contour line) that should be measured routinely for carbon storage but have to date not been included in whole estuary assessments are reeds, sedges and forested wetlands (specifically swamp forests). Reeds and sedges cover an area of ~18 000 ha and are the most extensive habitat type in the brackish and freshwater reaches of estuaries (Supplementary Fig. 3), providing shelter and food for many bird, fish and invertebrate species. When these plants die-back and decompose there is a peak in organic load and a release of particulate matter into the water (Hemminga et al., 1993; Kirwan et al., 2012). Dominant species are common reed (*Phragmites australis*) (Bulrush) and the sedges *Schoenoplectus scirpoides* (Club-rush) and *Bolboschoenus maritimus* (Alkali bullrush) while *Cyperus papyrus* (Papyrus sedge) occurs predominantly in subtropical to tropical systems. Pressures on these vegetation types include removal and habitat loss from encroaching coastal and agricultural development, uncontrolled fires, invasive species and general disturbance of riparian zones. These plants can indicate sites of freshwater seepage in estuaries (Adams et al., 2016b, Fig. 2b and c and Supplementary Fig. 3) which may hold substantial carbon stocks.

3.5. Forested wetlands (swamp forest)

The botanical classification of estuaries has always included forested wetlands particularly swamp forest as estuarine habitats (Adams et al., 2016b). Forested wetlands are recognised as an IUCN ecosystem functional group (Keith et al., 2022). They can occur in the fresh upper reaches of an estuary, but more typically are found in temporarily closed estuaries that are brackish (Supplementary Table 3). Forested wetlands thrive under these conditions and 13 associated tree species have been identified within these ecosystems in South Africa (Van Deventer et al., 2021). They cover an area of ~5400 ha in South African estuaries. The dominant species in the brackish environments of the estuarine functional zone are *Hibiscus tiliaceaus* (Lagoon hibiscus) and *Barringtonia racemosa* (Powder puff tree/Freshwater swamp tree). Forested wetland ecosystems are identified as Critically Endangered in South Africa with increasing development, subsistence agriculture and groundwater pressures resulting in high rates of transformation, despite 67 % of their extent occurring in protected areas (Van Deventer et al., 2021). Forested wetlands have the potential for high storage of carbon (Adame et al., 2024), but this has not yet been measured in South Africa.

4. Blue carbon research and recommendations for future research and management actions

4.1. Measuring carbon storage and fluxes (theme 1)

Quantification of Blue Carbon stocks in South Africa (Table 2) has focussed on mangroves, salt marshes and seagrasses based on protocols provided by the Blue Carbon manual (Howard et al., 2014). Available data have been integrated in global databases and analyses (e.g. Holmquist et al., 2024; Maxwell et al., 2024). Although there has been some progression in South Africa since 2017 on carbon-related research in BCEs there remains a large gap in data on carbon stocks and fluxes for different estuary types and biogeographic regions for national coverage.

4.1.1. Sampling estuarine and ecotonal habitats

There is a case for broadening the Blue Carbon outlook for South African estuaries to include all areas and vegetation types in the estuarine functional zones (Van Niekerk et al., 2019) as these are all tidally influenced ecosystems (Fig. 2a–f). This is in line with Adame et al.'s (2024) definition that all wetlands below the highest astronomical tide directly or indirectly influenced by tides should be considered BCEs. In South Africa, this would include estuary ecotonal habitats consisting of grass, shrubs, reeds, sedges, and forested wetlands. However, it remains to be tested whether these ecosystems have (1) long-term organic carbon storage in their soils, and (2) low GHG emissions because of high primary productivity and low decomposition rates in waterlogged soils (Adame et al., 2024). The latter is mainly caused by sulphates in marine water, which outcompete carbon as an electron acceptor inhibiting methanogenesis. Adame et al. (2024) exclude macroalgal beds within their BCE definition as they do not store carbon in their sediments. Similarly, ancient peat formations are not considered BCE as these ecosystems are not currently fixing and accumulating carbon.

South African estuaries have high spatiotemporal variability in physical and biological features that causes a rapid turnover between habitats, with documented die-back of mangroves and replacement by salt marsh (Mbense et al., 2016), dieback of mangroves and replacement by reeds and sedges (Adams and Human, 2016), and encroachment of mangroves into salt marsh and mudflats (Hoppe-Speer et al., 2015c; Katharoyan et al., 2024) (Supplementary Fig. 5). In a closed estuary, mouth opening due to a sea storm caused an increase in salinity and a change from submerged macrophytes to macroalgae (Riddin and Adams, 2010). In addition, BCEs change in response to anthropogenic impacts; macro- and microalgae replace submerged macrophytes (*Ruppia cirrhosa*) and seagrass (*Zostera capensis*) in response to nutrient enrichment and reeds replace salt marsh when there is a restriction in tidal connectivity and lower salinity (Nunes and Adams, 2014; Adams, 2020). For these reasons it is recommended that future estuarine carbon studies in South Africa have a holistic approach extending across different estuary habitat types. Carbon measurements must be taken across transects and elevation gradients to include all estuary habitat types (Fig. 2a–f) and meet the regional goal of producing an Estuary Carbon Atlas. This is in line with Carpenter et al. (2023) who call for a more context-specific whole-system approach to carbon stock assessment that guides ecosystem management. Recent studies now adopt a seascape approach acknowledging connectivity of habitats and lateral carbon fluxes (Kirwan et al., 2023; Engelbrecht et al., 2024; Queirós et al., 2024). Osland et al. (2016) stated that although mangroves, salt marshes, and salt flats are often treated as entirely different ecosystems, these need to be considered via a holistic lens particularly in relation to climate change.

Climate change is affecting the distribution of BCEs in South Africa; mangroves have the potential to expand their distribution range down the coast in a southerly direction; this a process known as tropicalization (Osland et al., 2016). Shifts in the vegetation can occur along the estuarine - terrestrial ecotone (Fig. 2a–d) (Veldkornet et al. 2015a, 2015b; Whitfield et al., 2016). Reeds and sedges can expand into salt marsh areas in response to a decrease in salinity (Fig. 2a) and mangroves can expand into terrestrial grasses and shrubs with an increase in inundation and salinity (Fig. 2b). Ecotones may be sites of high carbon sequestration and storage as was found for the mangrove-salt marsh ecotone at Nahoon Estuary in South Africa (Raw et al., 2019b) (Supplementary Fig. 1) and as such carbon measurements should also include ecotonal transitions as these may shift in response to variable estuarine physico-chemical conditions. Given that South Africa has a diversity of estuary types across different biogeographic regions this provides exciting opportunities for tracking global change.

4.1.2. Estuary sampling sites

The quantified carbon stocks are mostly concentrated within the warm temperate bioregion (Table 2) while published data from other bioregions are rare (Fig. 1). Furthermore, the choice of quantified BCE is also driven by the research question, for example, Banda et al. (2021) sampled all three BCEs at Nxaxo Estuary while Raw et al. (2019a) focused on the mangrove-salt marsh ecotone at Nahoon Estuary. At the Swartkops Estuary, because the estuary falls outside the mangrove range limit, Human et al. (2022) focused on a salt marsh (*Spartina maritima* and *Salicornia tegetaria*) and seagrass beds (*Zostera capensis)* comparison and Engelbrecht et al. (2024) included *Z. capensis* and salt marsh species for an estuary on the west coast. Other studies have focused on just one BCE across multiple estuaries, for example, the multi-estuary assessments of *Z. capensis* stocks of Wasserman et al. (2023) and Ndhlovu et al. (2024).

Finer scale studies have been conducted at individual estuaries across the South African coastline including Olifants, Groot Berg, Breede, Knysna, Swartkops, Nahoon, Nxaxo and Mngazana estuaries (Table 2), where sediment carbon was shown to differ within and between study sites. Carbon within the above-ground plant biomass was measured at Knysna, Swartkops and Nxaxo estuaries and this remains a knowledge gap beyond these three estuaries. As total Blue Carbon stocks (above, below-ground and sedimentary) have only been measured in a few estuaries, there is a need for a more comprehensive dataset to better understand carbon stocks across the South African coastline. Data deficient regions are particularly the cool temperate, subtropical and tropical bioregions.

4.1.3. Planning, sampling, measuring and reporting blue carbon

To guide future sampling and measurements, steps to consider when planning, sampling, measuring and reporting Blue Carbon studies were identified (Supplementary Fig. 6), with standardized metadata for reporting on collection and analysis of study samples (Supplementary Table 2). To capture intra-estuary variability of carbon stocks, sampling sites are identified based on a systematic or random sampling approach, with South African studies reporting between one and six sampling sites. For example, Human et al. (2022) collected sediment and plant biomass for salt marsh and seagrass at six sites from the middle reaches of the Swartkops Estuary to the mouth to cover the full range of their distributional limits. The spatial extent of BCEs in estuaries must be taken into consideration when selecting sites as there is significant spatio-temporal variability in BCEs. For example, it has been found that *Z. capensis* sediment carbon stocks are highly variable ranging from 1.67 to 180 Mg C ha^{-1} for the top 50 cm (Table 2). Interestingly, variability appears low at small (1–5 m) to medium (50–300 m) spatial scales, with significant differences only appearing between meadows separated by greater than 1 km, attributed to habitat characteristics and estuary hydrodynamics (Engelbrecht et al., 2024; Bossert et al., 2025).

The main exercise of a Blue Carbon assessment is the collection of sediment cores using a coring device. In South Africa comparable sample collection protocols have been used for collecting and processing salt marsh and mangrove samples according to Howard et al. (2014), while there have been variations for seagrass sediment samples due to substrate-related considerations. Russian peat corers were used for the collection of sediment cores in mangroves (Johnson et al., 2020), salt marsh and seagrass (Human et al., 2022; Ndhlovu et al., 2024); however, a PolyVinyl Chloride (PVC) corer (11 cm diameter and 1 m depth) with predrilled holes was used to sample seagrass sediments (Banda et al., 2021). The number of cores and depth section varies between studies, although for most studies this is down to 50 cm, subsequently sectioned into depth intervals. Due to the high variability associated with BCE measurements (Ricart et al., 2020; Williamson and Gattuso, 2022), in future cores should be collected at three to six replicates across several sites within an estuary and at least to 1 m depth where possible.

The methods for determining carbon content in a Blue Carbon assessment have ranged from determination of organic carbon using the loss on ignition (%LOI) method (Raw et al., 2019b), validated by elemental analysis (Johnson et al., 2020; Human et al., 2022; Wasserman et al., 2023) and elemental analysis (Banda et al., 2021; Ndhlovu et al., 2024). An important step in preparing these samples is pretreatment with hydrogen chloride (HCl) to remove inorganic carbonates, although this may underestimate carbon content (Serrano et al., 2023). As concerns about the reliability of carbon removals by BCEs are raised (Williamson and Gattuso, 2022; Johannessen and Christian, 2023; Kristensen et al., 2025), sources of methodological errors need to be identified, and protocols standardized across studies. The final step is to standardize reporting of results. Metadata provides important background information that includes the site details, methods used to collect the data and repositories, such as a national dataset, which is not available in South Africa, however the format of the Smithsonian Coastal Carbon Network and Atlas is recommended for future studies (Supplementary Table 2). The Smithsonian Institute assisted in the compilation of a Blue Carbon sink database for South Africa, as a component of the Coastal Carbon Network and Atlas (Holmquist et al., 2024).

There are no data on the turnover rate and permanence of carbon within different BCEs in South Africa. Carbon accumulation in sediment represents the active carbon sequestration of BCEs and is thus important to quantify (Jennerjahn, 2021). Several radionuclides can be used to assess sediment and carbon accumulation rates for the last decades-century, such as ^{210}Pb and ^{137}Cs (Arias-Ortiz et al., 2018) and ^{14}C for longer timescales. However, the low fluxes of artificial radionuclides due to global fallout in these latitudes make ^{137}Cs less useful. Application of dating techniques based on ^{210}Pb and C^{14} should be explored within a South African context together with the ongoing RSET (rod surface elevation table) monitoring that measures long-term surface elevation changes (Bornman et al., 2016; Raw et al., 2020).

Studies identifying the origin of carbon sources are needed to quantify the allochthonous versus autochthonous contributions to carbon stocks (including the influence of wastewater and eutrophication). Autochthonous organic carbon is produced within an ecosystem, whereas allochthonous carbon is transported from another environment, such as adjacent sinks to the depositional sources (Saintilan et al., 2013). The quality and quantity of the source of organic matter accumulated with the sediment of a BCE can be determined with the use of both elemental ratios (total organic carbon and total nitrogen), with natural stable isotopes ($\delta^{13}C$, $\delta^{15}N$, and $\delta^{2}H$), eDNA metabarcoding (Ortega et al., 2020) and with lipid biomarkers (Dahl et al., 2025). Identifying the sources of carbon is crucial to understanding carbon cycling and the influence by external factors such as wastewater and eutrophication, as shown for data from the eutrophic Swartkops Estuary (Human et al., 2022; Ndhlovu et al., 2024). A preliminary study in the Breede and Groot Berg estuaries (South Africa) employing isotope analysis suggested that autochthonous carbon accounts between 35 and 42 % of total seagrass meadow carbon (Bossert et al., 2024), but further studies are required. Building an estuarine isotope database to elucidate carbon sources and sinks in South Africa's BCEs should form part of the national research roadmap for these ecosystems.

4.1.4. Estimating GHG emissions and fluxes in South African BCEs

The contribution of BCEs to climate change mitigation requires information on the extent of the ecosystems, the carbon stocks present, and the rate at which carbon is either emitted or sequestered — their fluxes. Direct estimates of CO_2 emissions are complicated and require specialized equipment, therefore alternative methods based on the conversion of total carbon stocks have been developed (Emmer et al., 2015). Understanding the management of BCEs to help maximise carbon storage while simultaneously reducing GHG emissions is one of the major challenges to BCEs restoration and conservation globally. Uncertainties and biases surround the use of generalised emission factors and not critically accounting for anthropogenic and natural sources or sinks (Bastviken et al., 2022). Recent infrastructure and capacity building in South Africa will allow for the future determination of GHG (CH_4, CO_2 and N_2O) emissions from BCEs based on international best practice. Lateral carbon exports, such as the movement of dissolved or particulate organic carbon from BCEs to adjacent marine systems, represent a potentially significant component of carbon cycling, but they pose a major challenge for verifying long-term carbon sequestration potential and ensuring carbon permanence. Globally, the magnitude of lateral carbon (DOC, DIC, POC, PIC, and Alkalinity) exchange (2.35 ± 0.7 PgC yr^{-1}) is equivalent to the net carbon sink (2.3 ± 1.5 PgC yr^{-1}) of terrestrial ecosystems (Regnier et al., 2022). Thus, elevating the need for holistic budgeting in net ecosystem carbon inventories integrating lateral carbon exchange and recognizing its importance to global climate feedbacks. Another aspect to investigate is estuary mouth closure, salinity declines and whether BCEs become net emitters of GHGs.

4.1.5. *The role of microbiomes in preserving blue carbon*

Carbon in BCEs is preserved by the anoxic and aquatic conditions that slow down and prevent microbial decomposition of organic matter (Trevathan-Tackett et al., 2017), but microbial communities also contribute directly to the organic matter in the soil by producing a wide range of stable organic products (Kallenbach et al., 2016). Processes such as disturbances to sediments can drive the decomposition of organic carbon by microbial communities (Wainwright et al., 2023). Our understanding of the role of the microbial communities in biogeochemical cycling in BCEs remains limited and in South Africa, only Searle (2023) used a metabarcoding approach to characterize the sediment microbiome in the three BCEs. This work showed that microbial community abundance and diversity differ significantly between salt marsh, seagrass and mangrove ecosystems within an estuary, with core taxa (including Armatimonadota, Euryarchaeota, Halanaerobiaeota and Halobacterota) found in all three BCEs (Searle, 2023). Studies are required to understand the association of core microbial taxa with sediment characteristics and environmental conditions, particularly as relevant to BCE carbon sequestration and fluxes. Understanding microbial mineralisation of plastic carbon has also been identified as a globally important research topic (Noman et al., 2024).

4.1.6. *Remote sensing and mapping of blue carbon ecosystem extent*

Recent improvements to satellite images have strengthened global assessments of BCEs relying on freely available Landsat and Sentinel-1 and –2 sensors, in mapping the extent of mangroves (Bunting et al., 2022) and salt marshes (Campbell et al., 2022; Van Deventer et al., 2025). However, there are challenges in providing accurate extent values for South African BCEs. For example, salt marshes are overestimated in the subtropical regions because of spectral confusion with sugarcane (Bessinger et al., 2022). Van Deventer et al. (2025) showed an overestimation of salt marsh and seagrass using 30- and 10-m spatial resolutions of Landsat and Sentinel-1 and –2 sensors, respectively. Fine-scale mapping of small spatial BCE extents by proprietary space-borne sensors with spatial resolutions ranging from $\geq$1 m to $\leq$5 m, including the *Satellite pour l'Observation de la Terre* (SPOT), RapidEye and WorldView, suggests that some BCEs are separable, but inconsistencies in classification of vegetation classes remain (Lück-Vogel et al., 2016; Van Deventer et al., 2019). The traditional global datasets are too coarse in spatial resolution for the small and narrow BCEs of South Africa (e.g. above-ground biomass for Africa is at a 3 km spatial resolution; Spawn et al., 2020), however, the new PlanetScope satellite imagery, with spatial resolutions between 3 and 5 m, remains to be assessed for improved mapping (Campbell et al. in review). The use of drones could potentially address the minimum spatial resolution required to ensure the accurate representativity of the extent of ecosystem types and their above-ground carbon stock but requires further evaluation. Consistent airborne and space-borne sensors contribute to continuous monitoring of BCEs as highly dynamic environments, with the aim of quantifying the geographic extent and changes over time.

4.2. *Drivers and responses of Blue Carbon Ecosystems (theme 2)*

Important global, regional, ecosystem and site-specific factors that influence carbon stocks and sequestration and need to be considered in future BCE studies in South Africa were identified (Fig. 3). The future of coastal wetlands depends on understanding their resilience and on implementing actions to enhance their extent and quality across local, regional and global scales (He et al.). Stocks of BCEs are influenced by characteristics of the vegetation, and the availability of allochthonous organic material and sedimentary processes at the ecosystem scale (Fig. 3). For example, in the Swartkops Estuary high *Z. capensis* carbon stocks were attributed to inputs from municipal wastewater treatment works and associated eutrophic conditions (Human et al., 2022). At the same estuary depositional sites in estuary creeks had higher sediment carbon compared to the main estuary channel with stronger tidal flows (Human et al., 2022; Ndhlovu et al., 2024). While previous studies have established the role of seagrass shoot density and leaf length in assisting deposition rates and carbon stocks, studies on *Z. capensis* have not yielded significant results (Bossert et al., 2024a; Engelbrecht et al., 2024; Ndhlovu et al., 2024). It is likely that a complex interplay of site-specific factors interacts with large-scale environmental factors in driving the variability of carbon stocks in *Z. capensis* meadows.

Age of the BCE (Fig. 3) is also an important consideration; for example, older mangrove forests have been recorded to have larger carbon stocks on account of their greater above-ground biomass and the longer time periods over which carbon stocks accumulate (Kathiresan et al., 2013). When comparing two South African mangrove forests, the Nxaxo Estuary which has an older, naturally established forest, had higher sediment carbon storage compared to Nahoon Estuary with a younger planted forest (Hoppe-Speer et al., 2015a). In salt marshes, those consisting of taller plants (such as rushes) at higher density have elevated carbon stocks compared with more herbaceous species (Owers et al., 2018). In South Africa, the intertidal grass *Spartina maritima* had higher sediment carbon stocks compared to the low growing succulent *Salicornia tegetaria* (Adams et al., 2023).

The accumulation of sediment in all BCEs is critical for maintaining and building soil carbon stocks (Lovelock et al., 2014), with gradual accretion of sediments forming the response of BCEs to sea-level rise (Lovelock et al., 2015; Cahoon et al., 2019). Evidence for sediment accumulation is sparse, but in the Swartkops Estuary, salt marsh surface elevation is keeping pace with historic sea level rise at certain sites (Bornman et al., 2016; Raw et al., 2021). Estuaries are threatened by excessive sediment accumulation due to erosion in the catchment. In more arid regions along the South African west coast, loss of vegetation, combined with an expected rise in high-intensity rain events, will likely increase sediment deposition in estuaries (Skowno et al., 2021). Research is needed to understand sediment availability (catchment to coast) to inform coastal carbon sequestration and evaluate within-estuary variability and sedimentation rates.

Bioturbation - the reworking of sediment through animal activities (Kristensen et al., 2012) - is also an important consideration as this can accelerate nutrient cycling, transport sediment and modify sediment texture (Wang et al., 2010; Xie et al., 2020), all of which influence carbon stocks and sequestration processes. Crabs are of interest, given their ability to actively or passively trap organic matter in their burrows in tidal environments (Martinetto et al., 2023). For example, this can add modern carbon to old stocks, up to depths of 115 cm because of crab activity (Andreetta et al., 2014). However, because bioturbation and burrow networks expose more sediment surface area, anoxic processes that sequester carbon can be reversed. The role of bioturbation in carbon sequestration is therefore complex as a facilitative or suppressive process and research is underway to study this in South Africa.

4.3. *Climate change prediction and impacts on BCEs (theme 3)*

Key processes that impact BCEs include atmospheric, hydrological

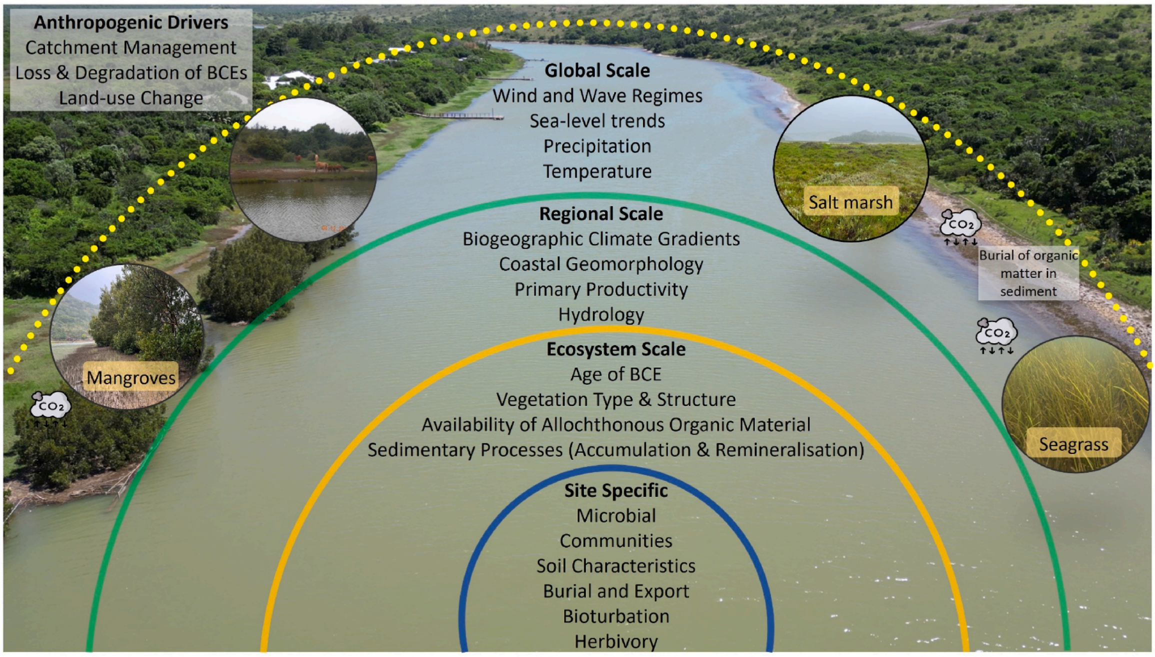

Fig. 3. Interacting factors that influence carbon stocks and sequestration in Blue Carbon Ecosystems at different scales. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

and oceanographic factors that vary across biogeographical regions (Fig. 3). Atmospheric climate stressors include increased CO_2 and temperature, while hydrological stressors refer to increased occurrence of droughts and floods and change in streamflow (Rogers et al. 2023a, 2023b). The abiotic changes associated with these climate change stressors and the ecological responses of BCEs to these changes in South Africa was described and future research was proposed to measure and model climate change (Table 3). Globally and for South Africa long-term field observations and *in situ* experiments are needed to predict and model changes in carbon storage and sequestration (Gu et al., 2023), with a stronger understanding of antagonistic (e.g. greater carbon loss to decomposition) and synergistic responses (e.g. sea level rise and greater soil carbon burial). Henry et al. (2024) identified organic carbon burial, a fundamental process in carbon exchange, as a key parameter neglected from global empirical Earth System models used to understand carbon fluxes across the land-ocean aquatic continuum.

An increase in storms along with changes in wind and wave regimes is expected along the South African coast that could release sediment carbon stocks (Rogers and Woodroffe, 2016). Droughts can have a negative influence on mangroves and are associated with hypersaline porewater and reduced sediment inputs. As diebacks of mangroves have been associated with an El Niño-Southern Oscillation (ENSO) event in Australia (Duke et al., 2017; Lovelock et al., 2017; Asbridge et al., 2019) the influence of these events and their associated low rainfall and increased temperatures on Southern African mangroves must be determined. The encroachment of mangroves into salt marsh habitat is also expected to increase carbon stocks (Kelleway et al., 2016).

Sea-level rise varies regionally and controls the accommodation space of BCEs that require sediment deposition and landward movement for the intertidal zone to persist (Spencer et al., 2016; Schuerch et al., 2018; Cahoon et al., 2019; Rogers et al., 2019; Lovelock and Reef, 2020).

The ability of BCEs to accumulate sediment as well as available land for landward migration is required for their long-term persistence (Saintilan et al., 2022). Sediment accumulation and elevation changes are tracked on the South African coastline using RSETs at only four estuaries (Knysna, Swartkops, Nahoon and Nxaxo), with the RSET network clearly requiring expansion. Sea-level rise could allow for expansion and enhancing of carbon sinks when supratidal marsh habitat is inundated (Adams, 2020). This will likely naturally transform into estuarine habitat due to waterlogging and not be suitable for other purposes e.g. agriculture. The implementation of buffer areas for the expansion of salt marsh and mangroves in response to climate change and sea level rise is essential. Long-term monitoring and research are needed to understand how BCEs respond to climate change stressors as these expansions and shifts between BCEs and other habitats along ecotones are expected. For this reason, Blue Carbon measurements should be made across elevation gradients such as seagrass, salt marsh and mangrove habitats overlap with reeds, sedges and forested wetlands (Fig. 2a–f). In addition, the response of South African BCEs to climate change impacts is expected to be variable as these responses are intricately linked with anthropogenic pressures as most estuaries are small with limited potential for resilience or adaptation capacity.

4.4. Emerging ecosystems to consider for carbon storage and sequestration (theme 4)

In South Africa, the Blue Carbon focus has been almost exclusively on mangroves, salt marsh and seagrass, with little attention given to other potential BCEs, such as seaweeds, kelps, coastal sediments and biofilms (Supplementary Table 3) that co-occur alongside traditional BCEs in estuaries (except for kelps that are restricted to shallow, rocky coasts). Filamentous green algae proliferate in South African estuaries particularly in response to nutrient input (Human et al., 2016), but it is not known how this translates to carbon storage in the sediment (Krause-Jensen et al., 2022). Similar knowledge gaps exist in South Africa although the macroalgae carbon stock is potentially significant in the coastal environment given the large spatial extent of the dominant kelp species, *Ecklonia maxima* and *Laminaria pallida*. However, their carbon sequestration has not been measured (Prew et al., 2024). To be considered a BCE; carbon must be sequestered and buried over timescales of decades (James et al., 2024).

Within estuaries there has been comparatively little attention given to mudflats which can host extensive microphytobenthic biofilms that potentially facilitate carbon capture and storage (Dalu et al., 2018; Nunes et al., 2024). Given the sediment mobility in South African estuaries, shifts between mudflat, intertidal marsh or seagrass habitat

Table 3
Overview of abiotic changes associated with climate change stressors and the ecological responses of Blue Carbon Ecosystems to these changes (Adapted from Adams et al., 2022). Future research needs related to predicting responses to stressors are also summarised. (DEM = Digital Elevation Model).

Abiotic Change	Ecological responses of Blue Carbon Ecosystems			Proposed research to measure and model Climate Change impacts
	Seagrass	Salt Marsh	Mangroves	
Increased CO_2 Higher C availability	Increase in plant growth and productivity	Increase in plant growth and productivity	Increase in plant growth and productivity	Mesocosm experiments to assess short-term responses to elevated CO_2
Increased Temperature Warming Higher aridity Increasing marine heatwaves	Competition with macroalgae Decreased growth due to epiphyte cover Impact on seagrass photosynthetic responses and genomic resilience	Increase in plant growth and productivity Reduced distribution range if replaced by mangroves Increased invasive species Change in phenology and potential for extinctions	Increase in plant growth and productivity Distributional range shifts and changes in habitat diversity	Species distribution models that include projections under different climate change scenarios to predict changes in distribution ranges for each BCE
Increased Floods Increased nutrient inputs and eutrophication Scouring of estuary and decrease in salinity	Seagrass loss/reduced extent due to scouring, sediment deposition and smothering	Increased macroalgal growth in response to increased nutrients leading to smothering. Lower intertidal salt marsh also at risk of scouring and sediment deposition	Mangrove loss due to scouring, sediment deposition and smothering	Effect of floods as extreme events on extent of different BCE has not been investigated – there are a few case studies but no quantitative assessments. Hydrodynamic and sediment modelling approaches would be needed
Increased droughts Increased salinity and aridity Closed estuary mouth conditions Increased water level and inundation, loss of intertidal habitat Increased concentration of pollutants	Seagrass extent and distribution within estuary increases towards the upper reaches in response to higher salinity Pollutants (including heavy metals, herbicides) impact seagrass fitness and survival	Changes in species and community composition Decreased productivity and extent Closed mouth conditions can lead to higher water level which can inundate salt marsh and cause die back Mouth closure also restricts marine connectivity	Decrease in productivity and cover Closed mouth conditions can lead to higher water level which can inundate mangroves and cause die back	Physiological limits related to inundation tolerance are broadly known, these could be used in species distribution/habitat suitability models, particularly to identify estuaries at risk for losing BCEs under this climate stressor
Change in stream flow/freshwater inflow Change in mouth condition (open or closed frequency) Change in salinity penetration upstream	Shift in water level will cause changes in seagrass extent Higher salinity will allow for seagrass expansion while lower salinity will lead to replacement by other submerged macrophyte species	Shifts in water level or flooding will cause an increase or decrease in intertidal area available for salt marsh. Salt marsh may be replaced by mangroves Changes in species composition depending on inundation and salinity tolerance	Mangroves may encroach into salt marsh if water level regime is significantly altered Decreased productivity and extent associated with increased salinity or prolonged inundation	Predicting catchment level changes in streamflow in combination with present level of water resource development and dam infrastructure Predicting regional scale responses to flow modification with a focus on salinity regimes and estuary mouth states
Sea-level rise Inundation and water logging, coastal squeeze Increased open mouth conditions Increased saline conditions	Increase in intertidal area for seagrass colonisation Seagrasses can become established further upstream with increased saline conditions	Salt marsh can experience subsidence if sediment accretion rate is below the rate of relative sea-level rise. Salt marsh may be subjected to coastal squeeze in systems that are naturally incised, or those that have hard infrastructure. Salt marshes may expand onto exposed mudflats and sand flats with increased open mouth conditions.	Mangroves can experience subsidence if sediment accretion rate is below the rate of relative sea-level rise. Mangroves may be subjected to coastal squeeze in systems that are naturally incised, or those that have hard infrastructure limiting the potential for landward migration	Site-specific sea level rise models have been developed for a few estuaries; these have used detailed data inputs and *in-situ* measurements of surface elevation change with RSETs. Alternative methods include Pb210 for sediment accumulation rates. High resolution datasets are needed for modelling new systems
Increased sea storms and wave height Erosion Increased sediment deposition leading to constricted mouth condition	Scouring of seagrasses leading to reduce extent	Smothering and loss of salt marsh Increases in water level, flooding and dieback of salt marsh	Smothering of pneumatophores by marine sediment Loss of mangroves	Increasing capabilities to model and predict changes in storms and wave height including downscaling to site-specific changes in sedimentation rates and possible impacts on estuary mouth state
Ocean acidification	Possible increase in seagrass production and carbon storage	N.A.	N.A.	Lab and *in situ* studies to understand local species sensitivity to changing pH conditions Develop models to predict vulnerability to ocean acidification

would be expected to impact carbon storage and sequestration. Mud flats as well as fjords, coralline algal (rhodolith) beds, and some coral reef systems have been identified as other coastal ecosystems that can act as Blue Carbon sinks in certain situations (James et al., 2024). For example, Carpenter et al. (2023) showed that mudflats store almost as much carbon in the soil as vegetated habitats and microbial mats in the arid coastal ecosystems of the United Arab Emirates.

Emerging Blue Carbon stocks in South Africa also include inconspicuous tidal habitats such as the supratidal microbialite pools fed by groundwater springs that form on rocky coasts and mirror estuarine-like conditions (Rishworth et al., 2020). These unique habitats are constructed by microalgal biofilms that deposit calcium carbonate. Although the biogenic calcification process releases carbon dioxide (Dupraz et al., 2009), high levels of primary production in these habitats could make them a net carbon sink, although this is unknown at present. Finally, novel anthropogenic ecosystems, such as artificial wetlands to improve water quality, support macrophytes specialized for nutrient uptake (Lemley et al., 2022), but how these function with regard to ecosystem services such as Blue Carbon storage is unclear.

4.5. Management, protection and restoration (theme 5)

South Africa has recognised estuary co-management policies and practices, but efforts are needed to mainstream BCE values in these. Therefore, it is important to link protection and restoration of BCEs to other estuary management processes such as setting environmental flow requirements, implementation of Estuary Management Plans and Critical Biodiversity Areas. Although there are existing mechanisms, such as the Integrated Coastal Management Act (2008) and National Estuarine Management Protocol (NEMP), these need to be implemented and streamlined from the national to local level to ensure the future productivity and persistence of BCEs (Van Niekerk et al., 2021). For example, the Integrated Coastal Management Act aids in cooperative governance with supporting norms, standards and policies for the management and conservation of the coastal zone, whilst ensuring ecologically sustainable development. This is then supported by the NEMP that ensures the ongoing provision of resources, both human and funding, to sustain this effort. Estuary Management Plans, a requirement of the NEMP, converge all existing management actions in estuaries to achieve integrated environmental management. The NEMP also calls for the establishment of Estuary Advisory Forums which form a vital communication platform between coastal communities, non-governmental organisations and various government departments that play a role in estuarine management.

4.5.1. Land ownership

Much of the BCE extent (~50 %) is privately owned or under the jurisdiction of tribal authorities, with the most vulnerable being salt marsh, as ~58 % of the extent is under private ownership, followed by mangroves at 43 % (Adams et al., 2023). Also concerning is that 28 % of seagrass areas are delineated as under private ownership in old title deeds, even though these habitat types are associated with open water areas (deemed part of South Africa's coastal waters in more recent legislation) (Adams et al., 2023). Given the ongoing trend in BCE degradation, particularly in salt marshes, the high percentage of private land ownership is concerning and requires a strong focus on building awareness through multi-stakeholder engagement (Fig. 4). South Africa could also develop a land-exchange program to reclaim land that is currently privately owned and regularly inundated by flooding. This would gain state owned 'accommodation space' and ensure BCE persistence under rising sea level conditions. Policies that support upslope landward migration to avoid 'coastal squeeze' (Raw et al., 2020) as well as development of protocols for the setting of conservative estuary flood lines and buffer zones are needed for inclusion in municipal Integrated Development Plans.

4.5.2. Protection

Fig. 4 was compiled as an output of this study to highlight how implementation of policy and sustainable land management through engagement of multiple stakeholders can increase protection of BCEs. Overall, 45 % of BCE extent is under formal protection, unsurprisingly salt marsh, which is the most degraded BCE type, is also the least protected with only 39 % of its extent under formal protection in proclaimed Marine Protected Areas or Protected Areas (Adams et al., 2023). While these represent high levels of protection, degradation still takes place in the protected areas and the establishment of more formally protected areas will be a critical response to protecting BCEs from human activities. Stewardship or contract parks have shown to be effective and allow for rapid increase in protection levels (Skowno et al., 2019). More recently, Other Effective Area-based Conservation Measures (OECMs) also offer new opportunities for a broader range of co-management measures to contribute to biodiversity conservation beyond declared protected areas. For example, in South Africa there is an urgent need to develop a 'bottom-up' community conservation model to protect BCEs with a focus on mangroves, which facilitates the inclusion of Indigenous and Local Knowledge Systems (ILKS), integration of local communities and traditions in management, monitoring and restoration (IUCN-WCPA Task Force on OECMs, 2019; Paterson, 2023; Adams et al., 2023).

The National Estuary Biodiversity Plan, as well as future refinements of it, prioritizes which estuaries should be assigned protection status and provides the 'lens' through which all present and future resource

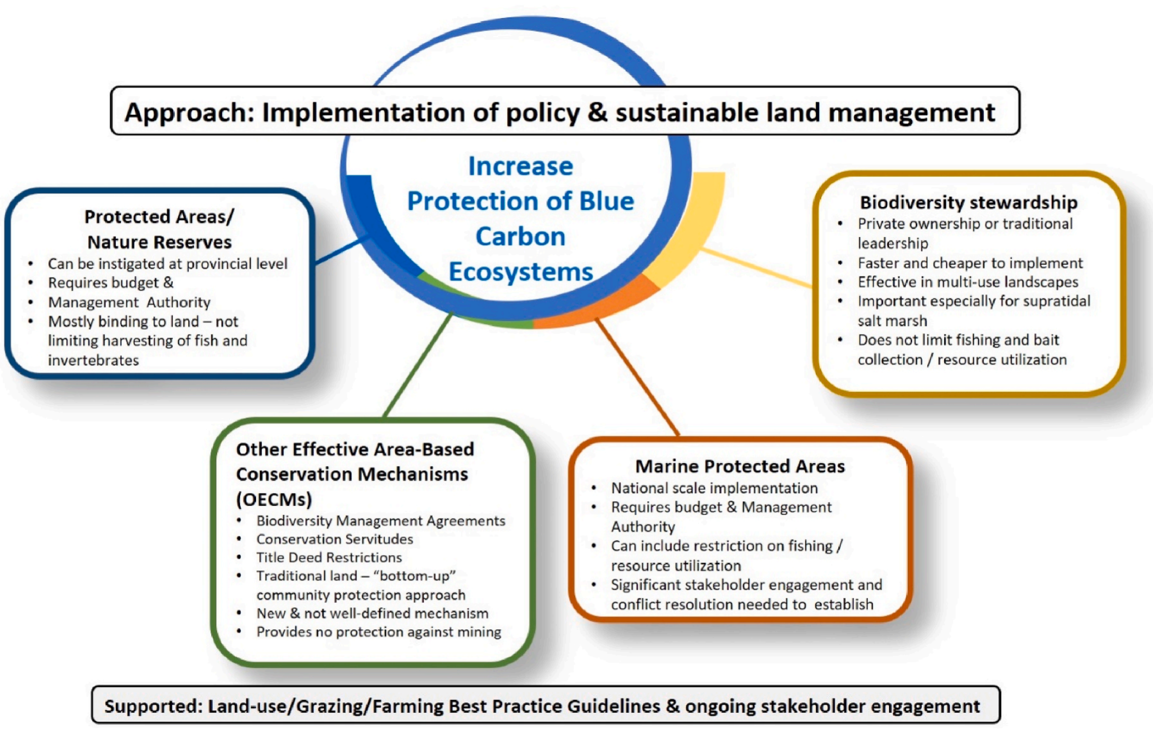

Fig. 4. Implementation of policy and sustainable land management through engagement of multiple stakeholders to increase protection of Blue Carbon Ecosystems. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

allocations should be evaluated to ensure that national and international biodiversity targets are achieved. Future plans must increase area targets for protection to 100 % of mangroves and seagrass, and at least 30 % of salt marsh to ensure adequate protection and representation (Adams et al., 2023). All BCEs should be demarcated as Critical Biodiversity Areas to increase protection and mainstreaming into planning processes. These are legally declared areas critical for conservation and maintenance of ecosystem services with accompanying land-use planning and decision-making guidelines. Formal protection of BCEs must be increased to meet the GBF targets. Furthermore, in order to ensure persistence of the sequestration functionality of BCEs there is also a need to develop policies that support the reinstating of a minimum amount of freshwater flows required by estuaries and BCEs, prevent over-abstraction and lowering of the groundwater table, and development of regional scale sediment management plans (Van Niekerk et al., 2021). The socio-ecological and economic implications of these policies will need investigation as part of water resource allocation processes. BCEs can be explicitly incorporated in the determination of environmental flows gazetted by the South African Department of Water and Sanitation. Also necessary is a reduction in nutrient pollution pressures and impacts of eutrophication on BCEs such as seagrass (Wasserman et al., 2023) by reducing the volume of effluent from wastewater treatment works into estuaries.

4.5.3. Restoration of Blue Carbon Ecosystems

Ecological restoration has become increasingly relevant across the world's ecosystems, with this decade (2021–2030) being declared the 'UN Decade on Ecological Restoration' by the United Nations General Assembly (March 2019) (Waltham et al., 2020). Restoration is the process of assisting the recovery of damaged, degraded or destroyed systems and is increasingly important in mitigating the loss of biodiversity to improve ecosystem services and societal benefits (Moritsch et al., 2021). As member to the Convention on Biological Diversity, South Africa should aim to have 30 % of the extent of degraded BCEs under restoration by 2030, for reporting to Target 2 of the Global Biodiversity Framework (CBD, 2022; de Paula Costa et al., 2023). However, restoration of coastal and marine ecosystems, including those important for carbon sequestration, is neglected when compared to restoration of inland ecosystems (Benayas et al., 2009). South African BCEs are relatively small, in addition to future pressures such as Sea Level Rise, there may not always be sufficient space for the restoration of BCEs. As such, pressures and threats on BCEs must be minimized to prevent their loss, as restoration of habitat will not be a suitable stand-alone option. In South Africa, there have been two documented attempts at restoring seagrass meadows (Mokumo et al.; Watson et al., 2023), limited activities for salt marsh (although through the identification of sites and frameworks there are considerations for salt marsh restoration; Adams et al., 2021) or mangroves, leaving significant gaps in regional capacity and methodology for BCE restoration. Connectivity to tides and freshwater inflow is critical to restore these habitats, but ongoing and future impacts on these systems, as well as their management, must be considered so that restored sites can persist. An important research focus is hydrological connectivity, fragmentation and restoration of carbon stocks in salt marsh and estuary degraded areas. Once a restoration intervention has taken place, its outcome must be determined through monitoring of key ecological indicators, e.g. hydrological conditions, growth and expansion of restored and other vegetation or functional indicators such as carbon sinks and estimations of plant biomass (e.g. Poppe and Rybczyk, 2021). One such approach is underway globally and provides a basis for tracking restoration e.g. the Mangrove Restoration Tracking Tool (Gatt et al., 2024) that could be adapted for South Africa.

Restoration efforts will need to be site-specific and account for seasonal dynamics. For example, although Watson et al. (2023) and Mokumo et al. followed similar approaches in restoration of *Z. capensis* in Langebaan, Knysna and Klein Brak estuaries, restoration outcomes were vastly different. Langebaan restored sites showed positive increases in area cover (up to >400 % area cover) and macroinvertebrate diversity, while none of the transplants in Knysna or the Klein Brak survived. Adams et al. (2023) investigated priority restoration sites for BCEs with nine estuaries prioritized for salt marsh (from west to east: Orange, Olifants, Groot Berg, Gouritz, Klein Brak, Knysna, Keurbooms, Swartkops and Gamtoos), five estuaries for seagrass (Olifants, Groot Berg, Knynsa, Keurbooms, Swartkops) and six estuaries for mangroves (Nxaxo, Mbashe, Mtata, Mngazana, Mntafufu, uMlalazi). These estuaries had the largest existing habitats as well as the largest degraded areas with potential for restoration and as such a greater opportunity to gain benefits and support multiple ecosystem services.

Overall, an 'Integrated Estuarine Restoration Strategy' is needed to coordinate and direct restoration of BCEs at national, provincial or even municipal levels, with links to community support, political buy-in in the form of estuarine management plans, and available technical expertise to maximise success of restoration activities and ensure co-benefit for nature and people.

4.6. Monitoring and reporting (theme 6)

More frequent quantification of changes in the spatial extent of BCEs and the impact on carbon storage over time is necessary to monitor seasonal dynamics and the success of restoration activities as well as identify areas where degradation or range expansion of BCEs is occurring. Global datasets are not adequately representative of extent and rate of loss, compared to the finer-scale maps carried out through field validation (Van Deventer et al., 2025). Future monitoring programmes to determine changes in BCEs require stronger in field validation of both extent and the integrity of habitats. Results then need to be integrated in a coordinated National Wetland Monitoring Programme that allows for efficient and accurate change detection and quantification. Table 4 indicates some key recommendations for sustainable and improved monitoring and reporting that includes the use of citizen science. A national monitoring programme is needed to track changes in BCEs in response to climate change and human pressures to facilitate early response and intervention.

Future monitoring programmes to determine changes in BCE extent require stronger combination of *in situ* and *ex situ* monitoring tools. In field validation of both extent and the integrity of habitats and their components, and the synergistic alignment of these with ex situ remote sensing mapping and monitoring are crucial. The approaches are both important to integrate in a coordinated National Wetland Monitoring Programme that allows for efficient, accurate and temporally consistent monitoring that would facilitate change detection and quantification.

Changes in BCE extent and ecological condition are evaluated and reported at 5 to 7-year intervals as part of the South African National Biodiversity Assessment: Estuary Realm (Van Niekerk et al., 2019). As part of this country-level assessment, key pressures such as flow modification, pollution, land-use change, biological invasions are evaluated per estuary and conditions are assessed for estuary components, including that of BCEs as macrophyte habitats (Van Niekerk et al., 2013, 2019). This forms the basis for South Africa's reporting to national and global commitments, e.g. Ramsar and the Convention on Biological Diversity and the 2030 Kunming-Montreal Global Biodiversity Framework targets. Therefore, extent and condition monitoring of BCEs is ongoing, but repetitive carbon measurements are not done which is important for carbon accounting and reporting on targets.

4.7. Carbon finance and policy (theme 7)

The effective integration of BCEs into climate mitigation strategies requires increased protection, sustainable management, and restoration of degraded ecosystems (Howard et al., 2023), creating mitigation options to reduce GHG emissions, or enhance CO_2 sinks. BCEs should be included into Nationally Determined Contributions (NDCs), which are countries' commitments to the Paris Agreement. However, the inclusion

Table 4
Key recommendations for improved monitoring and reporting of Blue Carbon Ecosystem (BCE) extents and stocks in South Africa that can facilitate reporting to national and global indicators.

Recommendation	Details
Embed the monitoring of BCEs in a national monitoring programme	Infield monitoring sites must be selected appropriately, with consideration of suitable sensors, methods and temporal assessments of carbon stocks per estuarine type across bioregions. This will require a consortium of partners, including national departments (e.g., DFFE, DWS), state-funded organisations (e.g., SANSA) and research institutes (e.g., SANBI, SAEON). Budget allocation to these activities should be formalised by the South African government.
Create a centralised data repository of infield measurements and extent mapping	This can be facilitated through governmental department collaboration (DWS, DFFE, SANBI and SAEON), as well as complimentary options such as the Smithsonian data repository. As a minimum, a standardized metadata reporting template and accuracy reporting of equipment used should be documented in a guideline (see Supplementary Table 3).
Improve representation of true extent and trends	Monitoring should ideally represent ≥70 % of the extent of natural BCEs. Error reporting should be formalised for earth observation and other applications. Assessment of suitable subcategories related to the ecosystem functional groups (Keith et al., 2022) of the IUCN should be considered. The use of horizontal earth observation devices should also be assessed for monitoring estuary mouth and tidal changes influencing Blue Carbon Ecosystems (e.g. InletTracker.
Inclusion of citizen science	Citizen science monitoring can be beneficial in reporting of unique and extreme events, rare species or distribution. Applications such as CoastSnap can be used to monitor changes in shorelines and estuary mouth condition (Harvey & Kinsela 2022).

DFFE = Department of Forestry, Fisheries & Environment, DWS = Department of Water & Sanitation, SANSA = South African National Space Agency, SANBI = South African National Biodiversity Institute, SAEON = South African Observation network, IUCN = International Union for Conservation of Nature.

of BCEs into GHG accounting for NDCs requires appropriate national datasets as well as supporting policies for implementing the proposed protection or restoration actions (Vanderklift et al., 2022). Initial efforts towards Blue Carbon accounts for South Africa (Taljaard et al., 2023) have shown there are still several challenges preventing national implementation. A collaborative effort between the South African Department of Fisheries, Forestry and Environment, Statistics South Africa and the South African National Biodiversity Institute will be needed to develop an approach that will follow the required international guidelines and standards.

BCEs can also be leveraged for climate mitigation by inclusion into carbon credit projects developed under the voluntary carbon market (VCM). These site-specific projects also reduce the need for national scale datasets. Carbon credit projects have the potential to be part of the solution to access climate finance and drive implementation of actions that enhance carbon storage and sequestration potential in BCEs while NDCs are still under development (Friess et al., 2022). Carbon credit projects must follow methodologies that are defined by the specific standard which will verify and issue the credits to the VCM. There are methodologies that are specific to BCEs (such as Verra's VM0033), but there have been relatively few projects that have been successfully registered to issue credits. A recent review by Jones et al. (2024) has highlighted the uncertainty on whether such projects will meet requirements for additionality (need for management action), feasibility, and permanence. This is also directly linked to the technical complexity of methodologies to accommodate natural processes in BCEs as transitional ecosystems between land and sea. For example, sea-level rise may shift the spatial extent of BCEs – thus influencing the project boundary or permanence of the carbon sequestered (Moritsch et al., 2021; Williamson and Gattuso, 2022); BCEs accumulate allochthonous carbon and export carbon to marine ecosystems – thus requiring separate carbon stock accounting (Krause et al., 2022); BCEs may also be significant sources of methane, thus reducing their net GHG sink potential (Rosentreter et al., 2018). Thus, requiring a systems approach to account for climate mitigation benefits across multiple ecosystem types. There is also the opportunity to combine different methodologies for carbon credit projects across landscape scales (Glass et al., 2024).

The restricted coastal geomorphology and limited extent of BCEs in South Africa reduces their feasibility for BCE-specific carbon credit project development and creates challenges for monitoring and management if these ecosystems were to be incorporated into the national GHG inventory as part of the mitigation component of the NDC. However, the critical importance of BCEs beyond carbon sequestration potential is recognised by national policy, which is an important step towards channelling finance for successful restoration and conservation (Merk et al., 2022). As Blue Carbon science in South Africa continues to develop, new opportunities for accessing climate finance are emerging. For example, the development of a 'nature credit' under Verra's Nature Framework (VerraThe Biodiversity Consultancy, 2024) is intended to finance conservation and restoration activities by recognizing the ecosystem and climate benefits of protecting and restoring biodiversity. The development of a South African local carbon standard (Inclusive Carbon Standard) may significantly reduce costs for carbon credit project development in the country. Blue Carbon science in South Africa is already used to inform on restoration priorities (Adams et al., 2021) and coupled with existing information on known threats and pressures to specific estuaries (Van Niekerk et al. 2022a, 2022b), provide a strong foundational framework which can be used to guide the development of effective restoration projects. In this context, integrating biodiversity and carbon finance streams, through mechanisms that reward co-benefits, may unlock greater investment in South African BCEs. Systems with low carbon potential that have high biodiversity value could then be considered when carbon returns alone would otherwise be insufficient (Webb et al., 2025).

By aligning emerging Blue Carbon opportunities with South Africa's unique ecological and socio-economic conditions, researchers, policymakers, and project developers can unlock the full potential of BCEs to contribute meaningfully to climate mitigation, biodiversity conservation, and sustainable development goals.

5. Future perspectives

Management actions, recommendations and research priorities identified in this study provide a roadmap for future activities that link policy and management (Table 1). There are ongoing challenges for Blue Carbon as Nature-based Solutions, particularly as South African BCEs are declining in extent and ecological condition due to increasing anthropogenic pressures and impacts from climate change. Carbon markets are prepared to incentivise restoration of BCEs as they adapt to climate change. This potentially places South African BCEs in a unique position given their high biodiversity and socio-ecological importance, considering that carbon markets are likely to soon be dovetailed by other incentivised schemes such as nature credit frameworks. However, knowledge gaps remain, particularly regarding the dynamics of BCEs in the global south which can only be informed by dedicated studies to collate the relevant datasets on carbon stocks, sequestration and GHG fluxes. Standardized approaches are needed; this article presents the steps to consider when planning, sampling, measuring and reporting Blue Carbon studies. We have also highlighted the need for a broader Blue Carbon outlook considering estuarine and ecotonal habitats that are currently not represented.

South Africa has high biodiversity and unique pressures influencing BCEs and is well positioned to help inform the global research agenda. Understanding the persistence of BCEs under future climate regimes and sea level rise will be essential to ensure long-term conservation. Effective co-management of BCEs in South Africa provides an opportunity for adopting communities and Indigenous and Local Knowledge Systems into a framework of co-benefit for nature and people. The priority research areas outlined in this article align with international Blue Carbon research agendas providing novel opportunities for collaboration. This study also has relevance for similar global south developing nations with emerging BCE research agendas.

CRediT authorship contribution statement

Janine B. Adams: Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Daniel Buttner:** Writing – review & editing, Writing – original draft, Resources, Investigation. **Sarah Hawkes:** Writing – review & editing, Writing – original draft, Resources, Investigation. **Lucienne R.D. Human:** Writing – review & editing, Writing – original draft, Investigation, Conceptualization. **Anesu Machite:** Writing – review & editing, Writing – original draft, Resources, Investigation. **Athi N. Mfikili:** Writing – review & editing, Writing – original draft, Resources, Investigation. **Andrew Ndhlovu:** Writing – review & editing, Writing – original draft, Resources, Investigation. **Leigh-Ann Smit:** Writing – review & editing, Writing – original draft, Resources, Investigation. **Anusha Rajkaran:** Writing – review & editing, Writing – original draft, Investigation, Conceptualization. **Taryn Riddin:** Writing – review & editing, Writing – original draft, Resources, Investigation. **Gavin M. Rishworth:** Writing – review & editing, Writing – original draft, Visualization, Investigation. **Heidi van Deventer:** Writing – review & editing, Writing – original draft, Methodology, Investigation. **Lara Van Niekerk:** Writing – review & editing, Writing – original draft, Visualization, Methodology, Investigation, Conceptualization. **Sophie von der Heyden:** Writing – review & editing, Writing – original draft, Visualization, Investigation, Conceptualization. **Emily C. Whitfield:** Writing – review & editing, Writing – original draft, Investigation, Data curation. **Jacqueline L. Raw:** Writing – review & editing, Writing – original draft, Visualization, Investigation, Conceptualization.

Data availability

Data are avaialble from the authors upon request.

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Janine Adams reports financial support was provided by National Research Foundation. If there are other authors, they declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecss.2025.109360.

Data availability

Data will be made available on request.

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