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Vulnerability to sea-level rise and the potential for restoration to enhance blue carbon storage in salt marshes of an urban estuary

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ABSTRACT

Coastal wetlands are vulnerable to sea-level rise (SLR) but are also valued for their potential to provide effective nature-based solutions to climate change mitigation and adaptation. Ecological benefits from these ecosystems can be constrained under urban settings by anthropogenic disturbances and pressures, so restoration activities are promoted as a management approach. Here we report on the potential for restoration of disused commercial salt extraction pans to enhance carbon (C) sequestration in the urban Swartkops Estuary, South Africa. We also considered the impact of SLR to 2100 on the distribution of estuarine habitats, the vulnerability of built infrastructure to tidal flooding, and how C sequestration is projected to change over time using the Sea-Level Affecting Marshes Model (SLAMM). Potential restoration of all salt pans (320 ha) to estuarine habitat was estimated to result in a gain of 67 850 Mg C. Establishing tidal connectivity was investigated as a potential restoration action, but most of the salt pan area was above the elevation of the current tidal range and would require excavation. Although conversion of the salt pans to estuarine habitat was predicted to occur without intervention under SLR, 44% of the original area would remain unchanged. Restoring hydrological connectivity to the estuary for these salt pans would significantly increase the extent of transitional/floodplain marsh, even under SLR to 2100. C sequestration was predicted to be 15% higher (54 614.8 Mg C) by 2100 if the salt pans could be restored, compared to if no action is taken. Overall, restoration of the salt pans has the potential to enhance C sequestration, but SLR will still cause large losses of supratidal marsh due to 'coastal squeeze' and extensive tidal flooding of developed areas by 2100 in the lower reaches of the estuary. A full-scale restoration approach for the Swartkops Estuary could use C sequestration potential to fund the project through carbon offsetting if the revenue exceeds the cost of the restoration activities, but additional social and ecological goals also need to be incorporated if the outcome is to be holistic and beneficial.

1. Introduction

Sea-level rise (SLR) is a significant threat to coasts around the world. Even if a low carbon emissions trajectory is followed global mean sea-level is projected to rise between 0.29 and 0.59 m by 2100 relative to 1986–2005 (Oppenheimer et al., 2019). Mean Higher High Water (MHHW) levels by 2100 are predicted to reach areas that are currently occupied by 150–250 million people around the world (Kulp and Strauss, 2019). Many people in coastal areas will be threatened with displacement and will incur financial costs associated with repairing, replacing or relocating built infrastructure (McMichael et al., 2020; Scata, 2020). Coastal wetlands can ameliorate the impacts of SLR on built infrastructure by providing protection from both flooding and erosion (Möller et al., 2014; Hijuelos et al., 2019).

Alongside socio-economic impacts, SLR also threatens coastal ecosystems and the services that they provide (Crosby et al., 2016; Raposa et al., 2016). Coastal wetlands, including mangroves and tidal marshes, are vulnerable to SLR because the survival of the plant species that form the foundation of these habitats are linked to their elevation relative to the tidal frame (Best et al., 2018; Valiela et al., 2018). However, the severity of SLR impacts on coastal wetlands depends on interactions between sea-level, surface elevation, primary productivity, and sediment accretion (Kirwan et al., 2010; Cahoon et al., 2019; FitzGerald and Hughes, 2019). Global estimates of coastal wetland vulnerability to SLR

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are variable, with estimates ranging from large losses or gains in total area by 2100 (Kirwan et al., 2016; Spencer et al., 2016; Schuerch et al., 2018). Downscaled assessments of vulnerability to SLR are needed to provide detailed information for adaptation and risk-avoidance strategies (Mcleod et al., 2010; Davis et al., 2019).

Restoration of degraded coastal wetlands is a nature-based solution for achieving coastal protection from SLR and associated extreme events such as flooding from storm surges (Sutton-Grier et al., 2015; Narayan et al., 2016). Holistic restoration of coastal wetlands is also associated with co-benefits. One of the most highly-valued ecosystem services provided by coastal wetlands is carbon (C) storage as mangroves, tidal marshes and seagrasses (so-called "blue carbon ecosystems") can sequester and store more C per unit area than terrestrial forests (Nellemann et al., 2009; McLeod et al., 2011). Hypersaline tidal flats have also been identified as important coastal carbon storage systems (Brown et al., 2021). Enhancing C storage and sequestration contributes to climate change mitigation, making this a primary goal of many coastal wetland restoration projects. The valuation of blue carbon has the potential to create opportunities to fund the restoration, conservation, and protection of coastal wetlands (Ullman et al., 2013; Sutton-Grier and Moore, 2016). This is achieved by generating carbon credits through restoration activities that follow an official methodology approved by carbon offset mechanisms, such as the Verified Carbon Standard (https://verra.org/) (Emmer et al., 2015; Needelman et al., 2018). Feasibility assessments that estimate the potential effectiveness of proposed restoration activities (for example, the amount of C that could be sequestered) can help maximize benefits.

Restoration activities seek to return an ecosystem to a previous state or trajectory, or to return the processes that existed before by 1) improving the condition of existing habitats (if they are degraded), 2) creating new habitats (without replacing other natural habitats), and 3) returning impacted areas to a natural state following cessation of use for economic activities. Improving the condition of existing coastal wetlands to enhance C storage and sequestration involves reducing activities that contribute towards degradation, such as overharvesting of plant species (Rajkaran et al., 2004); trampling (Mabula et al., 2017); grazing by livestock (Nolte et al., 2013); sediment destabilization caused by bait digging in intertidal seagrass beds (Adams, 2016); and pollution (Häder et al., 2020). Management objectives focus on reducing these activities, but it can be challenging to monitor whether there has been effective restoration for C storage and sequestration without a comprehensive baseline for comparison. Constructed wetlands can enhance C sequestration from a zero baseline (Were et al., 2019). However, the design and function of constructed wetlands should allow for a C sequestration rate that is equal to or greater than that of natural habitats (Madrid et al., 2012; Yang and Yuan, 2019). Restoration of coastal wetlands that have been lost or modified for economic activities (such as shrimp ponds and salt extraction pans) can enhance C storage and sequestration if the areas can be returned to a natural state (Keller et al., 2012; Dittmann et al., 2019; Noll et al., 2019).

The Swartkops Estuary (South Africa) presents an opportunity to assess the potential for the restoration of estuarine habitats (salt marsh) to enhance C sequestration in an urban estuary. This study uses a desktop modelling approach to provide a preliminary ecological assessment of the potential C sequestration that could be gained through the restoration of disused commercial salt extraction pans in the estuary. The objectives were 1) to quantify the potential C storage gains following restoration of the disused commercial salt extraction pans to estuarine habitat; 2) to determine the effect of establishing tidal connectivity on C sequestration in one of the salt extraction pans; 3) to compare the effect of SLR on the distribution of estuarine habitats and C sequestration for the Swartkops Estuarine Functional Zone (EFZ) to the year 2100 either with or without restoration of the salt pans; and 4) to identify the extent of built infrastructure that is vulnerable to SLR within the EFZ.

2. Methods

2.1. Site description

The Swartkops Estuary (33°51′58.48″S, 25°37′58.96″E) is an urban estuary in Nelson Mandela Bay, South Africa. This estuary is nationally ranked as important for biodiversity and as a nursery for commercially and non-commercially harvested fish species (Strydom, 2015; Van Niekerk et al., 2019). However, this estuary is also subject to cumulative anthropogenic pressures that include flow modification, pollution, habitat loss, and high fishing effort (Lemley et al., 2017; Adams et al., 2019b; Van Niekerk et al., 2019; Olisah et al., 2020). The human settlements within the Swartkops River catchment are densely populated and are occupied mostly by people who either receive very low income or are unemployed (NMBM, 2017; Zuze, 2018). The location of these communities also makes them prone to urban flooding as well as impacts from future SLR (NMBM, 2015; Siyongwana et al., 2015). A full-scale restoration project must incorporate the socio-economic needs of these communities by including them as participants and beneficiaries of the process.

The Swartkops Estuarine Functional Zone (EFZ), which is determined by the 5 m contour (Van Niekerk and Turpie, 2012), contains 209.2 ha of intertidal marsh and 338.2 ha of supratidal marsh (Adams, 2020). The estuary is permanently open to the Indian Ocean with marine intrusion occurring 13.6 km upstream. There is a salinity gradient from marine conditions at the mouth (35.5) to slightly brackish conditions at the tidal limit (1.2) (Adams et al., 2019b).

The present extent of the supratidal marsh represents 33% of its original area (1013.15 ha), the rest of which has been lost to industrial and residential development (Bornman et al., 2016; Adams, 2020). Within the EFZ, 316.9 ha consists of estuarine habitat that was modified for use as part of a commercial salt extraction operation in the early 1960s (Martin and Randall, 1987; Adams et al., 2016, 2019a). These areas have become completely desiccated since pumping operations by the salt works ceased in 2019 (Fig. 1).

The developed residential areas (Amsterdamhoek, Swartkops Village, and Redhouse) situated within the EFZ (Fig. 1), are already susceptible to 1 in 50 year and 1 in 100 year flood events and are therefore also expected to be vulnerable to SLR (S.R.K. Consulting, 2010; C.A.P.E. Estuary Management Plan, 2011).

2.2. Estuarine habitats for restoration

The most recent (2018) spatial extent of estuarine habitats as well as the developed areas within the Swartkops EFZ were obtained from the National Estuary Botanical Database (Adams et al., 2016, 2019a,b). Areas within the EFZ classified as "salt pans" in the database were used to identify potential restoration sites. To quantify the potential C storage (Mg C) that could be gained by restoring all disused salt pans, it was first necessary to identify the estuarine habitat types that could occur in these areas, as C storage is variable between habitats and across the tidal elevation gradient. Data generated from a 2017 LiDAR survey were provided by the Nelson Mandela Bay Municipality to build a digital elevation model (DEM). The LiDAR was validated with test points obtained by field surveys and compared to elevations from measured data. The test points were distributed across the mapping area and located on open ground patches. Classification of points was checked and validated against orthorectified imagery. The elevation of all the salt pan areas was extracted from the DEM at 1 m spatial resolution using the Spatial Analyst Toolbox in ArcMap 10.6. The total salt pan area was then classified into estuarine habitat categories based on known elevation ranges. Potential carbon storage (Mg C) was calculated using the mean values for soil and biomass C (Mg C ha^{-1}) for each habitat category. For intertidal and supratidal marsh, soil and biomass C stock was calculated using values measured by Els (2019) in this estuary for *Spartina maritima* and *Salicornia* spp. respectively. For floodplain marsh, soil and biomass

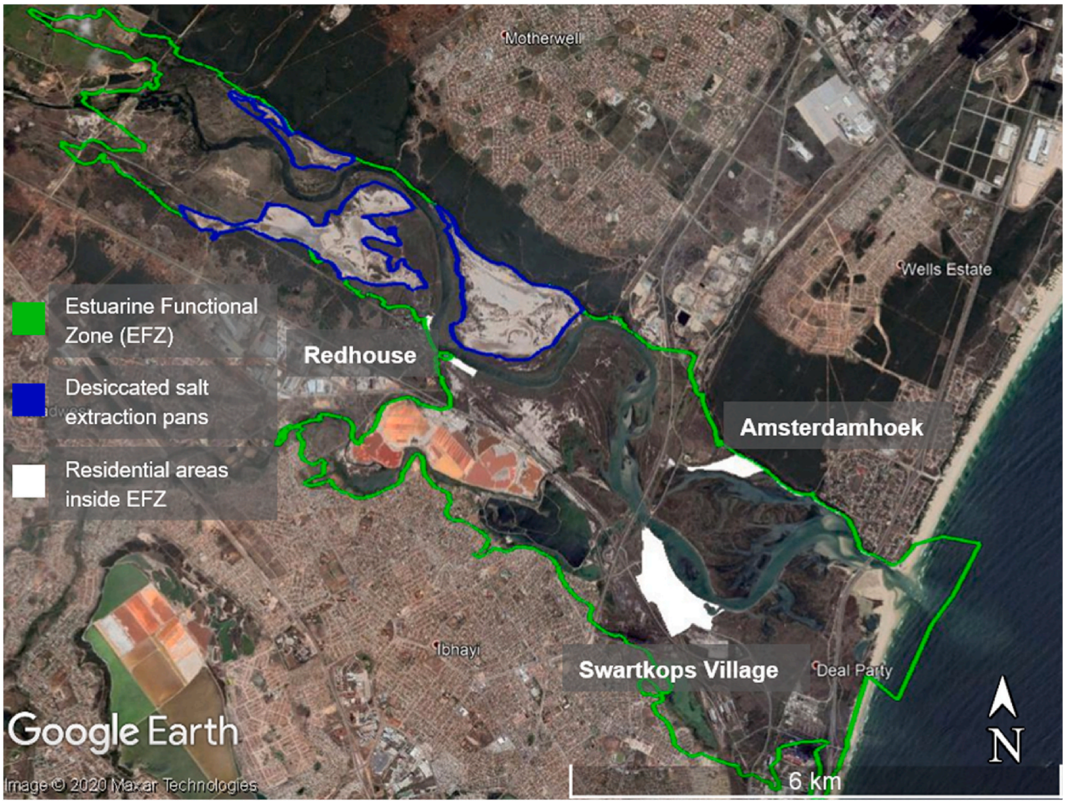

Fig. 1. Google Earth satellite imagery for the Swartkops Estuary (29-01-2020). The green line delineates the Estuarine Functional Zone (EFZ), which is determined by the 5 m contour (Van Niekerk and Turpie, 2012). Residential areas within the EFZ are shaded white and disused salt pan areas are outlined in blue. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

C storage was calculated using values measured by Wasserman (2021) from *Sarcocornia pillansii* at this estuary. Biomass C was 50% of the value measured for supratidal marsh as the area cover of the vegetation is patchy, while for soil C the values were the same as for supratidal marsh. Soil C stocks were estimated to 0.5 m depth.

2.3. Modelling the establishment of tidal connectivity

Establishing tidal connectivity was investigated as a primary restoration action for the disused salt pan areas. A spatial model to assess this was developed for one of the salt pans as a pilot study. The large salt pan opposite Redhouse (see Fig. 1) (referred to from here as "Redhouse Salt Pan") in the middle reaches of the estuary on the northern bank was selected as it covers 145.25 ha of the total disused salt pan area. This was previously a natural depression where a temporary wetland would form. To develop the salt extraction operation, the area was excavated, and the spoil was used to form several islands within the pan and to create the retaining walls. Estuarine water was actively pumped into the pan as the first step in the salt extraction process (Martin and Randall, 1987).

Some small patches of supratidal marsh that formed on the dredge spoil islands are still present. The salt pan area is currently within the elevation range at which supratidal marsh would naturally occur in South African estuaries (Veldkornet et al., 2015, 2016).

The Sea-Level Affecting Marshes Model (SLAMM) Version 6.7 (Clough et al., 2016) was used to model the effect of establishing a tidal connection to the Redhouse Salt Pan site. This provides a detailed approach to assessing habitat distribution. The SLAMM framework provides spatially explicit predictions by simulating the primary processes that control estuarine habitat distribution: inundation, erosion, overwash, saturation, salinity, and accretion. SLAMM requires information on the elevation, slope, vegetation distribution, accretion rates and historic SLR for the focus area. For the Redhouse Salt Pan, elevation (in meters) was derived from the available DEM for the salt pan and surrounding estuarine habitats. The slope (in degrees) was obtained from the DEM (Spatial Analyst, Slope tool). The vegetation distribution for the area was digitally mapped using ESRI World Imagery and Google Earth satellite imagery from November 2019. The 2018 habitat map was used for verification as it incorporates ground control points at the habitat boundaries. The habitat types were cross-walked into the land cover categories required by SLAMM (Table 1). To guide our habitat crosswalk, we consulted previous studies that have applied SLAMM to coastal wetlands on the east coast of Australia (Runting et al., 2017; Mogensen and Rogers, 2018), as the estuarine settings and species composition are comparable to those in South Africa. The SLAMM input parameters are described in full in the Supplementary Material.

SLAMM was used to predict whether estuarine habitat distribution would change following the establishment of a tidal connection between the salt pan and the estuary main channel. To simulate this, a new DEM was created in which the elevation was lowered to create two channels (combined area of 10.6 ha) within the salt pan. The channels were created over areas with the lowest elevations within the salt pan as this would minimize the amount of material to be removed if the activity was carried out. The total height and gradient of the slope between the salt pan and the estuary channel were reduced so that the new area did not exceed 0.5 m above the Mean Tide Level (MTL). This elevation was applied as the tidal range at this site has been reported as ~1 m (Huizinga, 1984; Schumann, 2013). Raster modification was carried out with the Raster Calculator function (Spatial Analyst). To determine the effects of the DEM modification only (i.e.: simulation of channels and not SLR), SLAMM was run to predict habitat distribution with 0 m SLR to the year 2020, i.e. assuming instantaneous habitat conversion based on elevation relative to the MTL.

Table 1
Crosswalk from SLAMM categories to analogous habitats at the Redhouse Salt Pans site. Categories were assigned to habitats based on elevation range and the "conversion under inundation" category which are both built into SLAMM.

SLAMM category	Analogous habitat	Under inundation, converts to	Min Elevation	Max Elevation
Developed dry land	Developed area, including gravel roads, jetties and marinas	Nearest transitional salt marsh, ocean beach or estuarine beach	Determined from DEM	Determined from DEM
Undeveloped dry land	Dry land, includes natural and disturbed terrestrial vegetation	Nearest transitional salt marsh, ocean beach or estuarine beach	Determined from DEM	Determined from DEM
Transitional marsh	Includes floodplain and supratidal marsh on the saltpan islands	Regularly flooded marsh	Mean Higher High Water (MHHW)	Salt boundary (95th percentile of Transitional Marsh elevation on DEM)
Regularly flooded marsh	Intertidal marsh	Tidal flat	Mean Tide Level (MTL)	Mean of MHW and salt boundary
Tidal flat	Estuarine intertidal shore	Estuarine open water	Mean Lower Low Water (MLLW)	MTL
Estuarine open water	Estuary channel/estuarine open water	Ocean	–	–
Irregularly flooded marsh	Desiccated salt pans	Regularly flooded marsh	Mean of MHW and salt boundary	Salt boundary

2.4. Restoration for carbon sequestration

The potential for restoration activities to contribute towards increased C sequestration at the Redhouse Salt Pan was modelled as a function of land cover change as predicted by SLAMM following establishment of tidal connectivity (Clough et al., 2016). C emissions through the loss of methane were not included as the sediment in the salt pan is saline due to residual salts. Tidal marshes with salinity >18 are considered to have negligible methane emissions (Poffenbarger et al., 2011; Needelman et al., 2018), making this is a conservative estimate. The following equations are provided by Clough et al. (2016) and directly represent the C sequestration model applied in SLAMM v6.7.

At each time step (t) the rate of CO_2 sequestration $\left(\frac{dM_{CO_2}}{dt}\right)$ was calculated for each land cover category as:

$$\frac{dM_{CO_2}}{dt} = \frac{44}{12} \times \left(\frac{dM_C^{ag}}{dt} + \frac{dM_C^S}{dt}\right) \quad \text{Equation 1}$$

where $M^{ag}{}_C(t)$ is the aboveground C mass sequestered at time t (Mg C).

$M^S{}_C(t)$ is the soil C mass (including belowground biomass) sequestered at time t (Mg C),
44/12 is the ratio of molecular weight of CO_2 to C

The aboveground C mass sequestered at time t ($M_C^{ag}(t)$)was calculated as:

$$M_C^{ag}(t) = F_C \times m_{ab} \times A(t) \quad \text{Equation 2}$$

where F_c is the C fraction of dry biomass (assumed to be 0.47 in SLAMM following IPCC (2006).

m_{ab} is the aboveground biomass of the vegetation (Mg ha^{-1}),
$A(t)$ is the area (ha) of the land cover category at time t.

The rate of soil C mass sequestered at time t $\left(\frac{dM_C^S}{dt}\right)$ was calculated as:

$$\frac{dM_C^S}{dt} = R_C^S \times A(t) \quad \text{(Equation 3)}$$

where $R^S{}_C$ is the soil carbon mass storage rate per unit area of the land cover category (Mg C ha^{-1} yr^{-1}).

$A(t)$ is the area (ha) of the land cover category at time t.
The carbon mass storage rate is estimated by multiplying the carbon density ($g.cm^{-3}$) by the surface elevation ($cm.yr^{-1}$) and scaling by the habitat area. Details are provided in Table S6.

Substituting equations (2) and (3) into equation (1) allows the CO_2 sequestration rate to be expressed as a function of area for each respective land cover category:

$$\frac{dM_{CO_2}}{dt} = \left(K_1 \times \frac{dA(t)}{dt}\right) + (K_2 \times A(t)) \quad \text{(Equation 4)}$$

where:

$$K_1 = \frac{44}{12} \times 0.47 \times m_{ab}$$

$$K_2 = \frac{44}{12} \times R_c^s$$

Finally, by using the first order derivative approximation:

$$\dot{x} = \frac{x(t+\Delta t) - x(t)}{\Delta t}$$

an approximated solution of the equation above can be obtained as:

$$M_{CO_2}(t_0) = 0 \quad \text{(Equation 5)}$$

$$M_{CO_2}(t_n) = M_{CO_2}(t_{n-1}) + K_1 * A(t_n) + [K_2 * \Delta t - K_1] * A(t_{n-1}) \quad \text{(Equation 6)}$$

Site-specific values for aboveground biomass and soil C storage were used for the Transitional Marsh, Regularly Flooded Marsh, and Irregularly Flooded Marsh land-cover categories (Els, 2019; Wasserman, 2021). When site-specific data were not available, the SLAMM default values were applied, but this was only for land-cover categories that had relatively small area coverage (Undeveloped Dry Land (5% of modelled area) and Inland Fresh Marsh (1% of modelled area) (see details in Supplementary Material).

2.5. Sea-level rise vulnerability of developed residential areas

Vulnerability to SLR was assessed for the developed residential areas adjacent to the Swartkops Estuary using an approach adapted from Lovelock et al. (2015). The height of the MHWS tide was predicted every decade from 2020 to 2100 for Amsterdamhoek, Swartkops Village, and Redhouse. The model was built with the SLR rate increasing every decade by the current rate of 1.82 $mm.yr^{-1}$ (Bornman et al., 2016). This rate projects an increase in sea-level of 0.65 m by 2100. To model vulnerability to SLR under a scenario of 1 m increase by 2100, the SLR rate was set to begin at 3 $mm.yr^{-1}$, which is the global average reported by the IPCC AR5 (Church et al., 2013). The mean sea-level was calculated for each decade with accelerating SLR. The elevation of the MHWS at the end of each decade was calculated as half the tidal range (in meters) above the mean sea level.

A spatial representation of the vulnerability assessment was carried out in ArcMap 10.6. First, separate rasters for each of the residential areas were extracted from the DEM. The rasters were corrected relative to the mean elevation measured for the open ocean on the original DEM, so that mean sea-level was set to 1.04 m, which is the mean tide level above Chart Datum for the Port Elizabeth tide gauge (Rautenbach et al., 2019). The rasters were then reclassified (Spatial Analyst, Reclassify tool) based on the height of the predicted MHWS at the end of each decade. The area predicted to be inundated by the MHWS at the end of each decade was then calculated (Spatial Analyst, Zonal Geometry tool). The current elevation of the residential areas as well as the tidal range were assumed to remain constant over time.

2.6. Estuarine habitats and carbon sequestration under sea-level rise

The SLAMM was applied to the entire Swartkops EFZ to predict estuarine habitat distribution under SLR and to assess the potential C sequestration if all the salt pan areas were restored to estuarine habitat. The inputs for elevation, slope, and the vegetation distribution were derived in the same way as described above for the Redhouse Salt Pan. Areas classified only as "Disturbed" in the 2018 habitat map represent estuarine habitat that is degraded, but not modified – we therefore used the same classification based on elevation to allocate these areas into the respective habitats. The metadata on the state of the habitat from the National Estuarine Botanical Database (http://bgis.sanbi.org/SpatialDataset/Detail/2687) were used to corroborate these categories. All habitats were then cross-walked to the SLAMM land cover categories. As the model now covered the EFZ, additional land cover categories were incorporated to represent ocean- and freshwater-related habitats respectively (see Supplementary Material).

Estuarine habitat distribution and area cover as well as the associated carbon sequestration were modelled under an SLR projection of 1 m by 2100 with a 10-year time step. The change in area cover over time was visualized with "ggplot2" (Wickham, 2016) using R version 3.6.3 (R Core Team, 2020). The vulnerability of built infrastructure was assessed by highlighting flooded developed dry land.

3. Results

3.1. Restoration of salt pans for carbon storage

Most of the salt pans are within the expected elevation ranges for supratidal marsh (183.29 ha), and floodplain marsh (131.20 ha) (Table 2). The total (biomass and soil) potential C storage if the salt pans were to be restored to estuarine habitats was calculated as 67 850.66 Mg C (Table 2).

An additional 2.37 ha of the salt pans, with elevation exceeding that of the floodplain marsh, was classified as potential terrestrial vegetation, but this area was not included in the C storage estimates as there are no site-specific data for this habitat type.

3.2. Tidal connectivity and carbon sequestration at Redhouse Salt Pan

Establishing tidal connectivity to the Redhouse Salt Pan by creating two channels to the estuary was simulated as the primary restoration activity. Reducing the elevation of the 10.6 ha area to 0.5 m above the MTL was estimated to require the removal of 32 402 t of material (bulk density = 1.4 g cm^{-3}).

SLAMM predicted a change in habitat distribution following the channel simulation (Fig. 2). This is an instantaneous change, as SLAMM predicts the initial (no SLR) habitat distribution based solely on elevation. The change in habitat distribution at the Redhouse Salt Pan was largely represented by a conversion of the salt pan area (represented by the Irregularly Flooded Marsh category) into Regularly Flooded Marsh. This category represents the intertidal marsh zone that is dominated by *Spartina maritima* in the Swartkops Estuary with an elevation range between MTL and MHHW, therefore this area is predicted to become tidally inundated. Areas of the salt pan that are at an elevation above the MHHW level were not predicted to be affected by the activity.

Carbon stock over one year (assuming an immediate conversion of habitats) was predicted by SLAMM to be greater (276.37 Mg C) if tidal connectivity was established at the Redhouse Salt Pan, compared to the present state (232.65 Mg C). Total carbon stock in marsh habitats (Regularly Flooded Marsh, and Transitional Marsh land cover categories) was greater if tidal connectivity was established (Table 3), and this was due to the increase in area of Regularly Flooded Marsh. However, because most areas of the salt pan remained above the elevation of the MHHW, the change in carbon stock was limited to the 10.6 ha area of the proposed excavation channels.

3.3. Sea-level rise vulnerability of residential areas

Vulnerability to inundation by the MHWS tide under SLR was different between the three residential areas (Fig. 3). All three residential areas were predicted to have increased area below the elevation of MHWS by 2100 under the 3 mm.yr^{-1} SLR projection, in comparison to the 1.82 mm.yr^{-1} projection. Overall, vulnerability to inundation was predicted to be much lower at Redhouse, with a maximum of only 0.3 ha below the elevation of MHWS. This area is in the middle reaches of the estuary and experiences a smaller tidal range (1 m) compared to that of Amsterdamhoek and Swartkops Village (both 1.65 m) in the lower reaches.

Swartkops Village and Amsterdamhoek were predicted to follow similar trends under both SLR projections, but there was a much faster increase in vulnerable areas at Swartkops Village after 2090 under the 3 mm.yr^{-1} projection. With a SLR rate of 1.82 mm.yr^{-1}, the total developed land cover of the three residential areas within the EFZ predicted to be below the elevation of MHWS by 2100 was 2.82 ha, but at a rate of 3 mm.yr^{-1}, this was predicted to increase almost tenfold to 20.3 ha.

Table 2
Classification of disused salt pan area into potential estuarine habitat types based on elevation ranges for the Swartkops Estuary from the National Estuary Botanical Database. The dominant salt marsh vegetation species and corresponding SLAMM habitat category is provided for comparison. Average ± SD carbon for biomass and soil C pools derived from Els (2019).

				Average Carbon ± SD		Total Carbon ± SD	
Botanical Database Habitat Category	SLAMM Habitat Category	Elevation Range (m)	Estimated Area (ha)	Biomass C (Mg C ha^{-1})	Soil C (Mg C ha^{-1})	Total Biomass C (Mg C)	Total Soil C (Mg C)
Intertidal marsh (*Spartina maritima*)	Regularly Flooded Marsh	0.9–1.2	0.024	16.27 ± 2.86	247.13 ± 47.71	0.39 ± 0.07	5.88 ± 1.15
Supratidal marsh (*Salicornia* spp., *Sporobolus virginicus*)	Irregularly Flooded Marsh	1.2–2.5	183.29	4.28 ± 0.72	212.26 ± 43.99	784.48 ± 131.97	38 904.46 ± 8062.93
Floodplain marsh (*Sarcocornia pillansii*)	Transitional Marsh	2.5–4.9	131.20	2.14 ± 0.36	212.26 ± 43.99	280.77 ± 47.23	27 874.68 ± 5771.49
Total						**67 850.66 Mg C**	

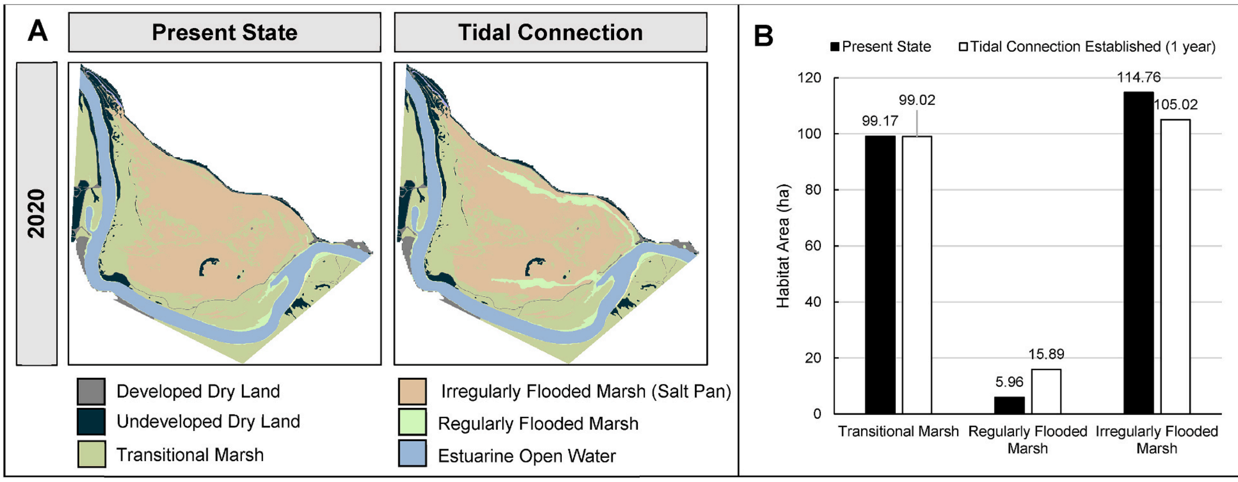

Fig. 2. Comparison of estuarine habitat distribution predicted by SLAMM for the Redhouse Salt Pan. The original vegetation map is predicted to change following the modification of the DEM to simulate the establishment of tidal connectivity. The desiccated area of the salt pan (represented by "Irregularly Flooded Marsh") is predicted to be replaced by "Regularly Flooded Marsh", which is flooded at Mean High Water.

Table 3
Comparison of carbon storage for marsh habitats at the Redhouse Salt Pan site (2019–2020). Average carbon values for biomass and soil C pools derived from Els (2019) were multiplied by the area for each habitat predicted by SLAMM.

		Regularly Flooded Marsh			Transitional Marsh			Total C storage (Mg C)	Total C stock (Mg C ha^{-1})
		Area (ha)	Biomass (Mg C)	Soil C (Mg C)	Area (ha)	Biomass C (Mg C)	Soil C (Mg C)		
2019	Present State	5.50	89.40	1358.05	99.67	426.58	21 155.42	23 029.46	218.99
	Tidal Connection	15.36	249.80	3795.03	99.57	426.18	21 135.73	25 606.78	222.80
2020	Present State	5.96	97.03	1473.88	99.17	424.46	21 050.44	23 045.82	219.20
	Tidal Connection	15.89	258.51	3926.65	99.02	423.79	21 017.01	25 625.96	223.02

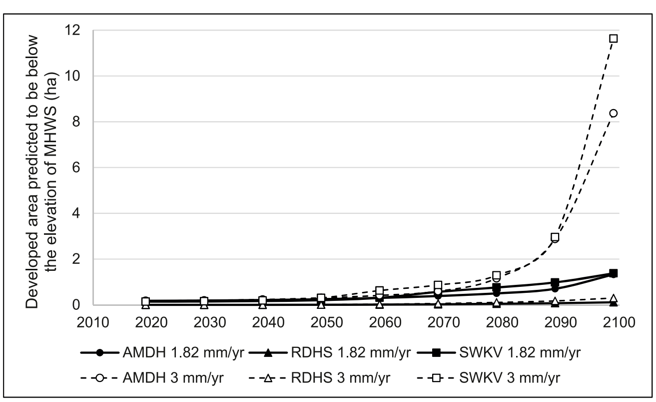

Fig. 3. Comparison of vulnerability to inundation under sea-level rise projections for residential areas adjacent to the Swartkops Estuary (Amsterdamhoek (AMDH), Redhouse (RDHS), and Swatkops Village (SWKV)).

3.4. Predicted sea-level rise at the Swartkops Estuary

Estuarine habitats were predicted to have different responses to the 1 m SLR projection modelled in SLAMM (Fig. 4). The potential restoration of all the disused salt extraction pans had the largest effect on the trajectories of area cover change over time for the Transitional Marsh and Irregularly Flooded Marsh land cover categories, as these were the habitats that were assigned to the salt pans based on the present elevation of these areas. Although there is an initial gain in Irregularly Flooded Marsh if the salt pans are restored, by 2100 the total area in the EFZ for this habitat is predicted to be the same as the scenario in which no action is taken. In contrast, Transitional Marsh is gained by the restoration activity and the larger area is mostly maintained by 2100. If no action is taken, the salt pans are predicted to decrease in area over time with SLR as there is conversion to estuarine habitat when the relative elevation becomes suitable for the respective vegetation. However, by 2100, 142.2 ha of the salt pans (44% of original area), is predicted to still be unchanged, and above the MHW.

Under 1 m SLR, Tidal Flat and Regularly Flooded Marsh were predicted to have the largest percentage increase in area coverage (up to 124.6% and 121.9% respectively) (Fig. 5, Fig. 6). Estuarine Open Water was also predicted to increase in extent by up to 73.4% by 2100. In contrast, Irregularly Flooded Marsh was predicted to decrease in area by up to 68.5% of the current extent (Fig. 5), which was largely due to replacement by Regularly Flooded Marsh and Tidal Flat habitats in the lower and middle reaches of the estuary (Fig. 6). The decrease in Irregularly Flooded Marsh was predicted to be relatively higher if the salt pans are restored because in this scenario there is a greater area of this habitat at the start of the simulation (2020). Up to 24% of Developed Dry Land was predicted to be flooded by 2100 (Figs. 5 and 6).

The three residential areas within the Swartkops EFZ cover a total of 78.68 ha of Developed Dry Land. SLAMM predicted that up to ~70% of this area will be flooded under a projection of 1 m SLR by 2100. For Swartkops Village, 12.05 ha was predicted to be flooded by 2100, in comparison to 4.58 ha for Amsterdamhoek, and 0.97 ha for Redhouse.

Cumulative C storage by 2100 for the Swartkops Estuary was predicted to be greater if all the disused salt extraction pans are restored to estuarine habitat, compared to if no action is taken (Fig. 7). However,

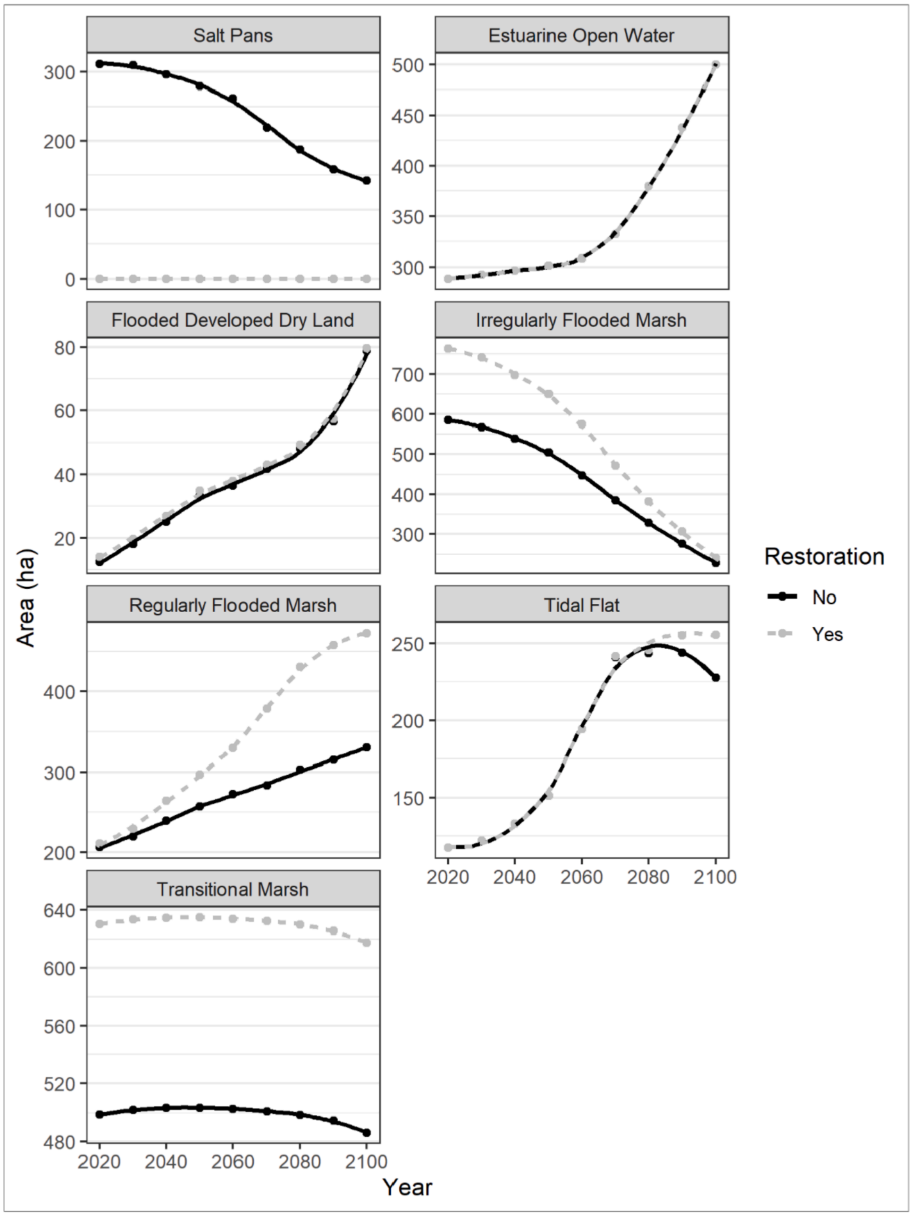

Fig. 4. Change in area over time for land cover categories in the Swartkops Estuary predicted by the Sea-Level Affecting Marshes Model under a sea-level rise projection of 1 m by 2100. The potential effect of restoring all salt pans (Restoration = Yes) to estuarine habitat is compared to taking no action (Restoration = No), for each land cover category.

these gains are likely to be lower, as SLAMM assumes instantaneous conversion between habitat types, while in reality this transition could take time.

If the salt pans are restored, the total C storage in 2020 is predicted to increase by 767.9 Mg C, but by 2100 the difference between restoring these areas and taking no action is much higher, i.e. 54 614.8 Mg C. Under the restoration scenario, the salt pans would already be converted to vegetated habitats and therefore experience a longer accumulation period as opposed to conversion only occurring with SLR.

4. Discussion

Urban coastal wetlands experience many anthropogenic impacts that can directly, or indirectly lead to degradation, reduced ecological benefits, or complete loss of these ecosystems (Lee et al., 2006). Restoration measures can significantly improve ecological functioning of degraded coastal wetlands (Abbott et al., 2020; Orth et al., 2020). Assessing the potential of a proposed restoration activity can provide practical information on the expected success, and this can be leveraged towards achieving the restoration goal. Here we provide an ecological assessment on the potential for restoration of commercial salt extraction pans to estuarine habitat in the urban Swartkops Estuary, South Africa.

4.1. Potential for blue carbon restoration

The restoration of coastal wetlands to support climate change mitigation through C sequestration is an integral component of policy approaches towards achieving the Sustainable Development Goals (Steven et al., 2019). Blue carbon can be used in market-based mechanisms to create economic value. There is also an opportunity to link with other types of funding instruments such as green bonds and insurances (Vanderklift et al., 2019; Stewart-Sinclair et al., 2020). The realisation of private capital is constrained by multiple requirements, including robust quantification of the project's costs and deliverables, and assessments of

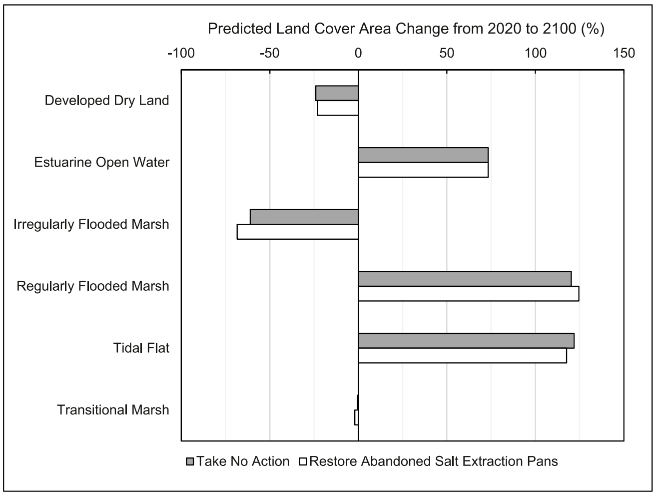

Fig. 5. Percentage difference in area cover for land cover categories in the Swartkops Estuary between 2020 and 2100 predicted by the Sea-Level Affecting Marshes Model under a sea-level rise projection of 1 m by 2100. The potential effect of restoring all salt pans to estuarine habitat is compared to taking no action, for each land cover category.

risk management (Blignaut and van der Elst, 2014; Waltham et al., 2020). Before carrying out a comprehensive cost-benefit analysis, there is a need to assess the ecological feasibility of proposed restoration actions.

For the Swartkops Estuary, there is potential to gain up to 67 850 Mg C if the desiccated salt pans could be instantaneously restored to estuarine habitat. The SLAMM carbon sequestration model predicts a cumulative linear increase in carbon storage over time based on accretion in response to sea-level rise. However, other detailed studies have shown that carbon storage in restored tidal wetlands can be dynamic and depend on site-specific factors including bathymetry, soil nutrient levels, and seasonal environmental changes (Artigas et al., 2015; Valach et al., 2021). Additionally, carbon credit projects need to provide estimates on how long it will take for this C stock to be realized in the form of a C stock gain per year. For example, the Tahiry Honko project in Madagascar aims to generate credits for >1000 t CO_2 equivalent per year (Blue Ventures, 2020), while the Mikoko Pamoja project in Kenya

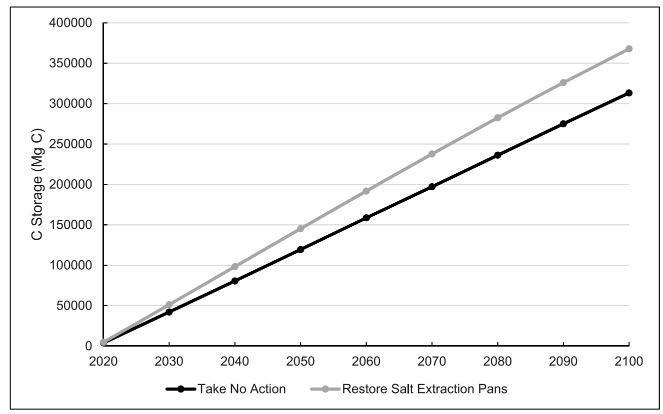

Fig. 7. The effect of restoring all disused salt extraction pans on predicted cumulative carbon stock over time for estuarine habitats of the Swartkops Estuary. Carbon storage was predicted by the Sea-Level Affecting Marshes model under a sea-level rise projection of 1 m by 2100.

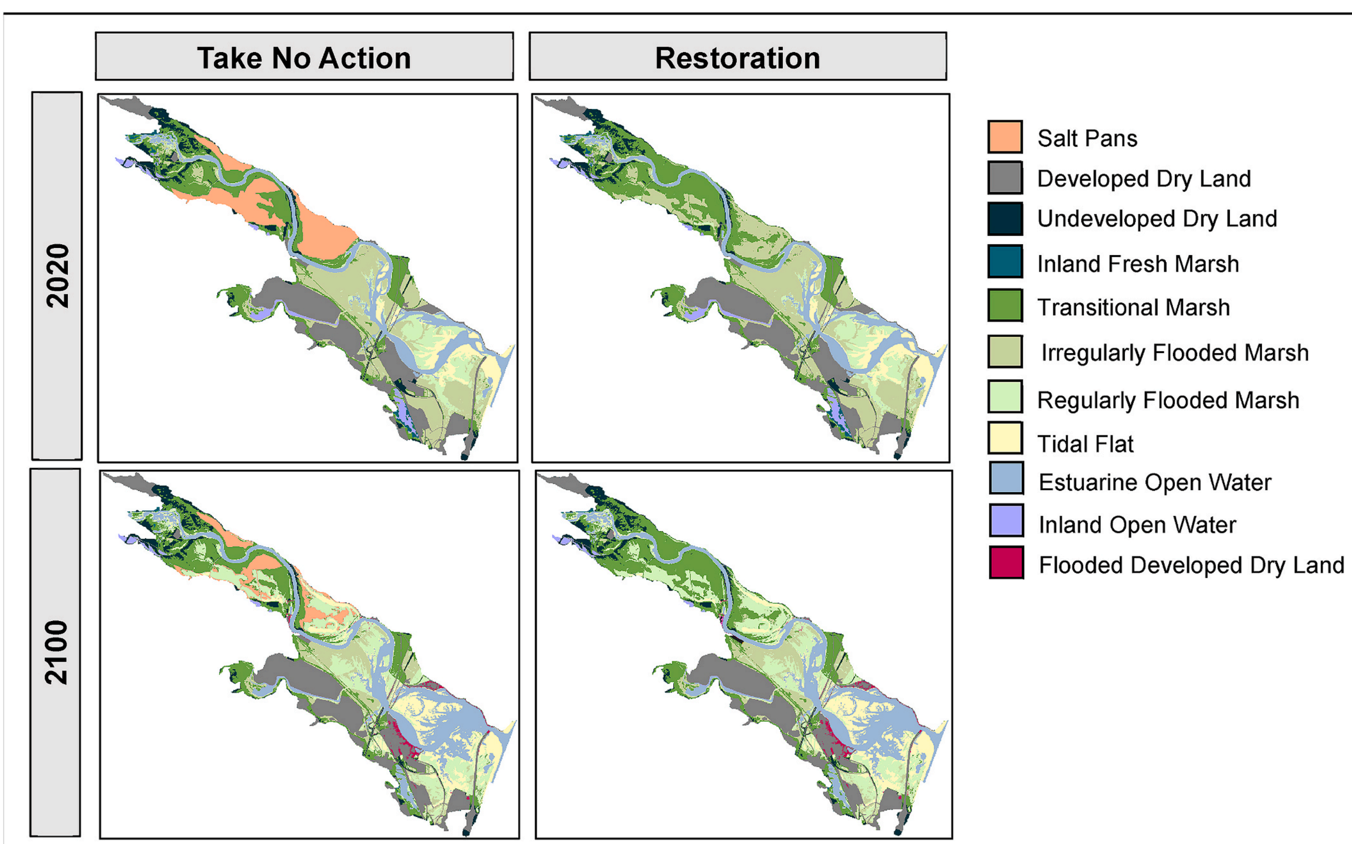

Fig. 6. Distribution of estuarine habitats for the Swartkops Estuary as predicted by the Sea-Level Affecting Marshes model in 2020 and by 2100 under 1 m sea-level rise by 2100. The effect of restoring all salt pans to estuarine habitat is compared as the restoration scenario.

has generated ~ 10 000 t CO_2 equivalent in five years (Wanjiru et al., 2019). In comparison, for the Swartkops Estuary we estimated with SLAMM that restoring the salt pans would increase C stocks by ~54 000 Mg C over 80 years (2020–2100) which is ~2500 t CO_2 equivalent per year.

Whether this amount is considered feasible for offsetting will depend on how much the project will cost as an initial injection of funds from government or philanthropic sources will be needed. Private sector investment in ecological restoration is limited in South Africa (Blignaut and van der Elst, 2014) and an economic assessment would need to consider whether the capital investment can be returned or made profitable. The carbon price at the time at which the restoration project is carried out will diminish the financial viability of the project if there are no other sources of funds. The price of carbon on the international voluntary market has been stable and relatively low (US $4–5 per tonne), but there is potential to catalyse local investments through the implementation of South Africa's Carbon Tax Act (Alton et al., 2014). The value of the Swartkops Estuary for recreational use and as a nursery for fish species has been estimated at ZAR 20–100 million yr^{-1} (~US $ 1.4–73 million yr^{-1}) (Turpie and Clark 2007), indicating the potential to include co-benefits and other forms of valuation for this project.

4.2. Available restoration actions

Key drivers of degradation in coastal ecosystems include agriculture, coastal commercial developments, and increased urbanization (Freeman et al., 2019). Disruption of hydrological flows (including changes to tidal influences) directly leads to the loss of estuarine habitat and associated ecosystem services (Gilby et al., 2020), thus restoration actions for fragmented areas are focused on re-instating hydrology.

The topography of the salt pans has been relatively stable as observed from the available historical aerial imagery. Restoration of tidal flow is expected to result in sediment deposition and accretion in the salt marshes and erosion in the immediate vicinity of the tidal channel. As the proposed tidal channel will be blind (it will not extend through to the other side of the salt pan), this should result in more deposition than erosion. It is anticipated that surface elevation will increase with SLR as long as there is sufficient sediment supply. Recent research has shown that the minerals in the sediment would be dissolved, the higher salinity may initially influence plant growth but with tidal flushing this would be diluted. In mesocosm experiments, submerged macrophytes (*Ruppia cirrhosa*) germinated from salt pan sediments that had been inundated with estuarine water (Wasserman, 2021). Construction of a culvert to re-establish tidal connectivity can be achieved with moderate costs (Sheaves et al., 2014), but in this case excavation of the site to lower the elevation would incur a much higher cost and could even create further damage. Alternative restoration actions need to be considered, or these desiccated salt pans will remain ecologically non-functional for decades.

While the commercial salt extraction process was still operational, estuarine water was actively pumped into the salt pans and this created large areas of suitable water bird habitat (Martin and Baird, 1987; Birdlife International, 2012). Diverting urban stormwater runoff into the Redhouse Salt Pan has recently been proposed as a rehabilitation option, as filling the desiccated pans will promote bird nesting on the emergent salt marsh islands. The long-term effects of introducing stormwater runoff need to be carefully considered, especially as there could be potential for nutrient and heavy metal accumulation over time without regular flushing (Yang and Lusk, 2018). If the area can be sustained as a freshwater pan (as an alternative restoration action to establishing tidal connectivity), there is potential for growth of associated vegetation (such as reeds, sedges, and rushes), that could contribute towards C sequestration (Owers et al., 2020). Preliminary research indicates that the dissolution of precipitated salts from the sediment would not result in persistent hypersaline conditions. The proposed management plan should include the monitoring of water levels and salinity to determine when more water should be pumped into the salt pan area (Wasserman, 2021). If this restoration action is carried out, a re-assessment of C sequestration potential to 2100 would need to be determined, as methane emissions from freshwater marshes are not negligible (Poffenbarger et al., 2011) and this would need to be accounted for if a carbon-crediting scheme is intended to be used to fund the restoration project (Needelman et al., 2018).

In this study, we have provided the first evaluation of the opportunity for restoration of the disused salt pan areas to estuarine habitat that will result in significant C stock gains over 80 years. For 'blue carbon' restoration projects, it is important to consider the time frame over which there will be significant gains in C stock. Our restoration scenario represented a complete and immediate change to estuarine habitat, but C sequestration would be limited by vegetative growth of the plants into these areas and the processes that control sediment accumulation over time (Dittmann et al., 2019). However, C sequestration is only one potential objective for a restoration plan at the Swartkops Estuary, and a comprehensive restoration proposal would include the concerns and priorities of all affected stakeholders. Existing blue carbon offset projects are almost exclusively community-based meaning they are implemented by locals, but are financed from elsewhere (Herr et al., 2017), and a similar approach could be advocated for this project. Besides local communities, the commercial operator that utilised the salt pan areas should be included as a stakeholder in the restoration project. In South Africa, any holder of mining rights or permits must develop plans for rehabilitating decommissioned sites and provide financial support for the rehabilitation. The restoration of the salt pan to natural habitat aligns with Section 56e (principles for mine closure) in the Mineral and Petroleum Resources Development Act (Act No 28 of 2002). This act states that "the land is rehabilitated, as far as is practicable, to its natural state, or to a predetermined and agreed standard of land use which conforms with the concept of sustainable development". This rehabilitation would be governed by the Regulations for Financial Provision for Prospecting, Exploration, Mining and Productions Operations in South Africa (GNR 1147) and should be implemented under the National Environmental Management Act (Act No 107 of 1998).

4.3. Incorporating sea-level rise into coastal wetland restoration projects

Sea-level, in combination with local geomorphology, is a principal driver of coastal wetland distribution and resilience (Rogers et al., 2014; Cahoon et al., 2019). Coastal wetland restoration projects must include the effects of SLR, as resilience can be site-specific (Raposa et al., 2016) and this will have a significant effect on the feasibility of the project (Emmer et al., 2015). At the Swartkops Estuary, SLR was predicted to influence the distribution of estuarine habitats, particularly in the lower and middle reaches of the estuary. By 2100, an expansion of Tidal Flat and Estuarine Open Water was predicted in areas currently occupied by Regularly Flooded Marsh (representing intertidal marsh, *Spartina maritima*). *Spartina maritima* habitats in the lower reaches of the estuary are already tending towards subsidence, rather than surface elevation gain (Bornman et al., 2016), suggesting that they will not keep pace with SLR. Overall, it was predicted that the area of Regularly Flooded Marsh would increase by 2100, but this was largely due to replacement of Irregularly Flooded Marsh (representing supratidal marsh, *Salicornia* spp. *Sporobolus virginicus*) habitats. Restoration of the salt pan areas was predicted to increase the areal extent of Regularly Flooded Marsh by 2100. This would be in support of a restoration goal to enhance C stock, as the *S. maritima* habitats have been reported to store and sequester more C than *Salicornia* spp. habitats (Els, 2019). Expansion of the Tidal Flat and Estuarine Open Water habitats under SLR could reduce C sequestration following marsh die-back and erosion of marsh platforms, which could release stored C back into the environment (Sapkota and White, 2021). However, there is also potential to increase C sequestration, as intertidal and subtidal beds of the seagrass *Zostera capensis* are relatively expansive in the lower reaches of the Swartkops Estuary, with average biomass C to be 2.08 ± 0.49 Mg C.ha^{-1}, and the average soil C to be 224.14 ± 37.93 Mg C. ha^{-1} (Adams, 2016; Els, 2019). We did not account for *Z. capensis* habitat change or C storage in this study, as this seagrass is dynamic and easily influenced by changes in hydrodynamic flows and suspended sediment which are not modelled in SLAMM (Adams, 2016; Clough et al., 2016).

Restoration of the salt pan areas in the middle to upper reaches of the estuary had no effect on the area of Irregularly Flooded Marsh predicted to still be intact by 2100, representing a ~60% reduction of the current extent. This decline predicted for Irregularly Flooded Marsh with SLR is the result of two factors: limited potential for surface elevation gain, and the unavailability of adjacent upland areas for potential landward migration. Sediment deposition can be significantly lower in supratidal marshes in comparison to those occurring lower in the tidal frame (Butzeck et al., 2015). The supply of mineral sediment is a key contributor to surface elevation (Cahoon et al., 2019), and is a significant predictor of resilience against SLR (Best et al., 2018; Mariotti, 2020). Transitions from supratidal marsh to intertidal marsh occur when sediment supply and vertical accretion are low (FitzGerald and Hughes, 2019). When SLR rates exceed surface elevation gain, landward migration becomes the primary mechanism by which coastal wetlands can respond and adapt to SLR (Borchert et al., 2018). However, this option is significantly restricted in urban areas, as hard infrastructure often occurs at the marsh boundary where the current elevation is above tidal influence (Valiela et al., 2018; Rogers et al., 2019; Raw et al., 2020). This ultimately leads to loss of supratidal marshes through "coastal squeeze" (Pontee, 2013). Although a C sequestration goal for restoration can still be fulfilled if supratidal marshes are lost, this would not be a suitable outcome for a biodiversity restoration goal, as supratidal marshes are formed by specific plant species (Veldkornet et al., 2016; Adams, 2020) and they provide critical habitat to fauna above the intertidal zone (Martin and Baird, 1987; Van Niekerk and Turpie, 2012). To avoid this biodiversity loss as a result of SLR, a full scale restoration proposal for the Swartkops Estuary will need to recommend for adjacent upland areas to be made available for landward migration of supratidal marsh.

The vulnerability of residential areas to tidal flooding with SLR was not affected by restoring the salt pans to estuarine habitat as this action did not influence habitat distribution in the lower reaches of the estuary where built infrastructure is most vulnerable. Besides restoring existing habitats, an alternative approach to provide coastal protection from SLR, flooding, and storm surges is to employ a nature-based solution using hybrid designs of natural habitat and built infrastructure (Sutton-Grier et al., 2015; Möller, 2019). These options include "living shorelines" which can protect waterfront infrastructure from erosion, and biomimicry approaches to designing culverts that reduce flood damage to roads (Davis et al., 2015; Sutton-Grier et al., 2015). These approaches can be considered as adaptation options for the residential areas in the lower reaches of the Swartkops Estuary. However, it is likely that a managed retreat will be necessary, as our vulnerability model is only a conservative representation of the actual flooding risk that would be associated with SLR. Tidal flooding of the roads and property edges indicates daily inundation of these areas, so any additional higher tide or storm event will lead to significant flooding and damage to existing infrastructure. These events will be more severe and will happen sooner in areas where hard infrastructure is immediately adjacent to the estuary channel, such as at Swartkops Village.

5. Conclusion

Urban coastal wetlands require careful management approaches that consider the numerous, and often interacting, anthropogenic pressures on these ecosystems. It is essential that restoration projects for these ecosystems are holistic, and that they include both social and ecological restoration goals. Restoration to enhance C stock in the Swartkops Estuary can be carried out by rehabilitating the desiccated salt pans to natural estuarine vegetation. This has the potential to add 67 850 Mg C to the total carbon stock for the estuary. However, because of SLR, this action will not contribute towards biodiversity conservation of supratidal (irregularly flooded) marshes which are predicted to decline by up to 60% in area by 2100. Similarly, restoration of the salt pans did not reduce the extent of tidal flooding in adjacent residential areas and developed dry land by 2100. Up to 70% of these areas were predicted to be flooded under a projection of 1 m SLR by 2100. The SLAMM framework has allowed us to predict the distribution of estuarine habitats under SLR and this information is useful to identify areas that are most vulnerable to change.

CRediT authorship contribution statement

J.L. Raw: Conceptualization, Methodology, Formal analysis, Investigation, Writing – original draft, Writing – review & editing, Visualization. **J.B. Adams:** Conceptualization, Resources, Writing – review & editing, Funding acquisition. **T.G. Bornman:** Resources, Writing – review & editing, Funding acquisition. **T. Riddin:** Methodology, Writing – review & editing. **M.A. Vanderklift:** Conceptualization, Resources, Writing – review & editing, Funding acquisition.

Declaration of competing interest

The authors 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.2021.107495.

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