RECREATING A WETLAND AT AN ABANDONED SALTWORKS: TOWARDS A REHABILITATION PLAN

By

Johan Wasserman

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Supervisor: Prof. Janine B. Adams

Declaration

NELSON MANDELA
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DECLARATION BY CANDIDATE

NAME: Johan Wasserman

STUDENT NUMBER: 214250687

QUALIFICATION: MSc (Botany)

TITLE OF PROJECT: Recreating a wetland at an abandoned saltworks: Towards a rehabilitation plan

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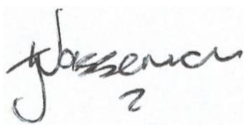
SIGNATURE: 

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Summary

A saltworks at Swartkops Estuary was abandoned in 2018. While operational, the saltworks hosted some of the largest breeding colonies of several shorebird species in southern Africa and hosted thousands of Palearctic migrant waterbirds annually. The abandonment of the saltworks has resulted in the loss of the artificially managed hydrological regime and therefore the wetland function and habitat value of the site, and the rich and diverse avifauna that once occurred at the site have not returned. The rehabilitation of the saltworks as a wetland that functions as a waterbird sanctuary is currently being organised, and this research aimed to create a plan for implementing and monitoring the rehabilitation. In order to do so, the baseline environmental condition of the abandoned saltworks was established, the possible rehabilitation interventions necessary for rehabilitating the site were assessed, and the potential ecological implications of any interventions were investigated.

The assessment of the saltworks' baseline condition revealed that the site is now characterised by vast expanses of dry hypersaline sediment with sparse patches of monospecific vegetation and depauperate avifauna. The once rich and diverse waterbird communities have all but disappeared since the site was abandoned and are unlikely to return unless a managed hydrological regime is reinstated. Furthermore, it is improbable that salt marsh vegetation will cover the abandoned saltworks primarily due to the high sediment salinity that will persist unless the saltpans are flooded. The area will likely remain barren with little ecological value if no rehabilitation action is taken.

Two potential rehabilitation options for reinstating a hydrological regime at the saltworks were identified: (1) pumping estuary water into all of the saltpans; or (2) pumping estuary water into some of the saltpans, while allowing the largest one to be filled with stormwater. Both options were deemed to be feasible; however, the second option will likely have lower running costs. The use of stormwater to fill the one saltpan is expected to result in brackish conditions initially, while the saltpans filled with estuary water would have salinity levels ranging from euhaline to slightly hypersaline. Both the stormwater and estuary water are rich in inorganic nutrients – the estuary water is rich in both dissolved inorganic nitrogen (DIN) and dissolved inorganic phosphorus (DIP), while the stormwater has an exceptionally high DIN content. This raised concerns of creating eutrophic wetlands with detrimental conditions such as algal blooms and a hypoxic water column.

A microcosm experiment investigating the response of the dry hypersaline sediment to inundation with either stormwater or estuary water revealed that initially eutrophic conditions with phytoplankton blooms (accounted for by the proliferation of diatoms) and a turbid water

column can be expected if the saltpans are inundated with either water source. The bloom was most notable in a stormwater treatment with the highest total oxidised nitrogen (NO_x) concentration. However, in both the stormwater and estuary treatments, these blooms collapsed within a week of being recorded after depleting the nutrients from the water column. Within two more weeks, the water column became clear and primary production was mostly accounted for by microphytobenthos (MPB) and submerged macrophytes (*Ruppia cirrhosa*). This oligotrophic, benthically-driven regime with a clear water column persisted until the end of the experiment. However, one of the replicate tanks with estuary water persisted in a turbid state due to a *Tetraselmis* sp. bloom throughout the study, suggesting that phytoplankton blooms are more likely to occur if estuary water is used to fill the saltpans. However, as it will be required to repeatedly fill the saltpans with nutrient-rich water to maintain water levels, recurrent phytoplankton blooms may occur. It is therefore expected that the rehabilitated saltpans might cycle between a regime characterised by a turbid water column with phytoplankton as the dominant primary producers while the saltpans are being filled with water, and a clear water-benthically driven regime once the blooms collapse. Lastly, there was no evidence suggesting that potentially harmful algal bloom (HAB) forming species would proliferate at the saltpans using either water source.

As the use of stormwater to fill the Redhouse saltpan would have lower maintenance costs, and are not expected to cause any ecological damage, the second rehabilitation option was recommended. These research findings were used to develop a plan for rehabilitating the saltworks. The plan established a vision and relevant ecological targets, as well as recommendations for implementing the rehabilitation and managing the site. Finally, a monitoring plan was developed, and the Society for Ecological Restoration's Five-Star system was adapted for evaluating the success of this project over time.

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List of Abbreviations and Acronyms

CCA	Canonical correspondence analysis
CWAC	Coordinated Waterbird Counts
DIN	Dissolved inorganic nitrogen
DIP	Dissolved inorganic phosphorus
DO	Dissolved oxygen
EFZ	Estuary functional zone
EHI	Estuarine health index
HAB	Harmful algal blooms
IBA	Important Bird and Biodiversity Area
LMM	Linear mixed-effect model
MAP	Mean annual precipitation
MAR	Mean annual rainfall
MPB	Microphytobenthos
MWAW	Motherwell artificial wetland
MWC	Motherwell Canal
NBA	National Biodiversity Assessment
NGO	Non-governmental organisation
NMBM	Nelson Mandela Bay Municipality
NOAA	National Oceanic and Atmospheric Administration
PES	Present ecological state
REC	Recommended ecological category
SER	Society for Ecological Restoration
WWTW	Wastewater treatment works

1. General introduction

A saltworks at Swartkops Estuary, Port Elizabeth was abandoned by Cerebos Ltd in 2018 after over 50 years of salt production. While operational, the saltworks hosted thousands of waterbirds annually and was an important component of the Swartkops Estuary-Redhouse and Chatty Salt Pans Important Bird and Biodiversity Area (IBA) complex. However, the abandonment has resulted in the loss of the artificially managed hydrological regime of the saltworks, and the site has been left to dry. The wetland function once provided by the site has been lost and the waterbirds have not returned.

Swartkops Estuary is nationally important and boasts a high habitat diversity and biodiversity (van Niekerk et al., 2015). The estuary has a recommended ecological category (REC) of ‘C’, but a present ecological state (PES) of ‘D’ as it has been largely modified by urbanisation. Although the saltworks was artificially created, it contributed toward the biodiversity (particularly in regard to waterbirds) of the estuary and the abandonment thereof thus contributes toward the loss of habitat and a decline in biodiversity from the estuary. Rehabilitating the saltworks will therefore assist in improving estuary’s PES to a ‘C’. Furthermore, the loss of waterbird habitat from the estuary was identified prior to the abandonment of the saltworks (van Niekerk et al., 2015), and long-term declines in waterbird populations are an issue nationally and globally (Wetlands International, 2012; Raimondo et al., 2019). Rehabilitating the saltworks will thus be beneficial locally by improving the ecological health of Swartkops Estuary and will be relevant globally in the face of declining waterbird populations.

The rehabilitation of the saltworks is currently being planned by the Zwartkops Conservancy (a local NGO) and the previous operating company Cerebos Ltd. It is envisioned to recreate a wetland at the saltworks that functions as a sanctuary for waterbirds. In order to achieve this, a hydrological regime needs to be reinstated at the site. It is not possible to create a tidal connection with Swartkops Estuary due to the elevation at which the saltworks lies, so the hydrology needs to be artificially managed. However, the pumphouses once used to manage the hydrology of the saltworks were subject to frequent vandalism and theft of equipment, and it is highly likely that this would be a recurrent problem if the same approach is taken in this rehabilitation project. Therefore, a new innovative approach is necessary to ensure that the saltworks can be rehabilitated successfully.

Two options have been identified for reinstating the hydrology of the saltworks. The first option is to fill the saltworks with estuary water, as was previously done, but using a pump mounted on a pontoon that would float in the estuary as opposed to using the onshore pumphouses.

This would allow the pump to be stored in a safe area while not in use. The second option is to fill the saltworks with urban stormwater from a nearby stormwater canal. Stormwater discharge from this canal has been identified as a significant contribution to the continual degradation of water quality at Swartkops Estuary (Adams et al., 2019b). Diverting the stormwater into the saltworks thus presents an opportunity to recreate the important waterbird habitat while simultaneously alleviating the estuary from a major source of pollution.

Although the saltworks was filled with estuary water while the site was being used for salt production, no information on the ecology of the system was available. Furthermore, the use of stormwater to fill the saltworks would be a novel approach and the possible ecological impacts thereof are uncertain. In order to ensure that the rehabilitation project will be successful, thorough planning is necessary. Such planning should include an assessment of the site in its degraded state, an investigation of possible rehabilitation interventions, and an understanding of the possible ecological trajectories that may result from those interventions (Gann et al., 2019). The Zwartkops Conservancy approached Nelson Mandela University to develop such a rehabilitation plan for the abandoned saltworks. This research thus aimed to:

1. Establish the baseline environmental conditions of the saltworks in its abandoned state;
2. Assess the feasibility of the proposed rehabilitation options (the use of either estuary water or stormwater to fill the saltworks) and any potential undesirable impacts that may arise; and
3. Create a plan for implementing, managing and monitoring the rehabilitation of the saltworks.

To achieve these aims:

i. Field sampling was carried out at the saltworks;
ii. The feasibility of the two rehabilitation options were assessed through site visits and desktop assessments;
iii. The latest available water quality data for the estuary and the stormwater were collated to compare the two rehabilitation options; and
iv. A nanocosm experiment was conducted to investigate the ecological response of inundating the dry hypersaline sediment of the saltworks with estuary water and stormwater.

2. Literature review

2.1. Planning estuary restoration: A synopsis of the process

Coastal wetland ecosystems like estuaries and associated habitats such as salt marshes, seagrasses and mangroves have long been recognised for their high productivity and biodiversity (Day et al., 1989; Costanza et al., 1997; Barbier et al., 2011). Despite covering less than 5% of Earth's land surface area, coastal wetlands provide a disproportionately high amount of ecosystem services including the provision of food and raw materials, coastal protection against storms, biogeochemical cycling and the provision of culturally important sites (Costanza et al., 1997; MEA, 2005; Barbier et al., 2011; Thrush et al., 2013). However, coastal areas have been subjected to long-term degradation from numerous anthropogenic pressures, and coastal ecosystems are now among the most degraded ecosystems worldwide (Lotze et al., 2006; Halpern et al., 2008; 2015). The degradation of these ecosystems can be expected to become increasingly more severe as human populations in coastal areas continue to grow coupled with the effects of climate change and sea-level rise (Morris et al., 2002; Neumann et al., 2015). Current management approaches for maintaining the health of coastal ecosystems have not been sufficient overall (Borja, 2014). Ecological restoration will be necessary for the effective conservation of these important ecosystems and has become increasingly relevant, with this decade (2021 – 2030) being declared the "UN Decade on Ecological Restoration" by the United Nations General Assembly.

Ecological restoration is defined as "the process of assisting the recovery of an ecosystem that has been damaged, degraded, or destroyed" (Gann et al., 2019). Although ecological restoration has commonly referred to recovering an ecosystem to some pristine pre-disturbance state, it is seldom possible to accomplish this (Davis, 2000b). Instead, restoration projects should aim to recover a degraded ecosystem to a self-sustaining, functioning state that is resistant and resilient to disturbance (Ruiz-Jaen and Aide, 2005; Harris and van Diggelen, 2006; Choi et al., 2008). There are various other restorative activities that may not result in the full recovery of an ecosystem including reduced societal impacts, remediation (management activities that remove causes of degradation), and rehabilitation (the reinstatement of some ecosystem functioning in degraded areas) (Gann et al., 2019).

The ecological restoration of coastal ecosystems is a particularly challenging undertaking. These ecosystems represent the interface between terrestrial and marine realms and are characterised by distinct gradients in physical and ecological processes and patterns that are both spatially and temporally dynamic (Cloern and Jassby 2008; Jennerjahn and Mitchell, 2013). Furthermore, coastal areas are subjected to a multitude of anthropogenic pressures depending on land use in the area (Wolanski et al., 2009; Mitchell et al., 2015). As such, coastal restoration projects are often highly idiosyncratic and require site-specific planning to ensure their success. Various guidelines and frameworks exist for planning and implementing ecological restoration projects. The most widely recognised is likely the International Principles and Standards for the Practice of Ecological Restoration (Gann et al., 2019) from the Society of Ecological Restoration (SER), an authoritative organisation in the field of ecological restoration. However, numerous other guides have been developed (e.g. Hobbs and Norton, 1996; Galatowitsch, 2012; Howell et al., 2012; Rieger et al., 2014). Such guides and frameworks generally have the following core components in common: a preliminary assessment of the degraded site, identification of the pressures causing ecosystem degradation, a vision and relevant restoration targets, a plan for implementing and managing restoration interventions, and a monitoring plan. These components are briefly discussed below.

Preliminary assessments of sites to be restored are valuable in that they provide a baseline of the site, which is fundamental to informing the interventions necessary to restore an ecosystem and allows for monitoring the success of any interventions applied (Gann et al., 2019). Such baseline assessments should consider the ecological and socioeconomic characteristics that may affect the outcomes of restoration and should identify all relevant opportunities and constraints for restoration (Howell et al., 2012). During preliminary assessments, it is essential to identify and understand the pressures responsible for degradation not only locally, but at a landscape-scale (Simenstad et al., 2006; Shackelford et al., 2013). This is particularly important for coastal ecosystems as they are highly connected to their surrounding landscapes and sensitive to activities that occur in catchments and along coastlines (Wolanski et al., 2009). In some cases, the full recovery of coastal ecosystems may not be achievable due to prolonged degradation from multiple pressures (Duarte et al., 2009; 2015). In addition to anthropogenic pressures, natural disturbances (such as droughts, floods and storms) that may threaten the success of restoration should also be identified (van Diggelen et al., 1995; Simenstad et al., 2006).

Once a baseline for the degraded site has been established, and the pressures acting upon it understood, a realistic vision for ecological restoration can be established. The vision broadly describes the desired endpoint that a restoration project aims to achieve. To achieve the vision of a restoration project, specific targets and objectives need to be established. Many restoration projects have commonly set restoration targets based on historical baseline datasets or pristine analogous reference sites nearby the site to be restored (Harris and van Diggelen, 2006; Howell et al., 2012; Gann et al., 2019). However, setting static targets based on some historical condition or a present-day reference site is troublesome in that temporal variation in environmental conditions (or states) is not accounted for (Pickett and Parker, 1994). Furthermore, baseline datasets of pre-disturbance conditions of degraded ecosystems are often lacking and appropriate contemporary reference sites may not exist for certain restoration projects (Clewell and Rieger, 1997; Jones and Schmitz, 2009). Therefore, dynamic yet achievable restoration targets should be established (Hobbs and Norton, 1996; Choi et al., 2008). Such targets can be based ecological processes and functions and should anticipate the possibility of multiple ecological trajectories that may arise from restoration interventions as well as the socio-economic implications thereof (Choi, 2004; 2007; Hobbs, 2007). However, the most appropriate approach for setting restoration targets currently remains unresolved (Jones et al., 2018). Nonetheless, targets are a necessity in ecological restoration and should be tailored for individual projects among all involved stakeholders.

Restoration targets are often set from an ecological perspective. Ecological targets that are commonly used include community patterns (e.g. community structure, species composition and heterogeneity) and ecosystem function (e.g. physical processes and energy transfers) (Hobbs and Norton, 1996; Ehrenfeld and Toth, 1997). However, ecological restoration often has social and economic implications, and socioeconomic targets can be greatly beneficial in a restoration project (Cairns, 2000; Choi et al., 2008). Ideally, a project can incorporate ecological and socioeconomic targets. For example, the regionwide Chesapeake Bay Program on the east coast of the USA has ecological (e.g. improving water quality through various means, increasing habitat area), social (e.g. promoting opportunities for minority stakeholders to be involved with the restoration) and regulatory goals (e.g. increasing the area of land that is legally protected), among others.

After a vision and the associated restoration targets have been established, the necessary restoration interventions can be identified. Restoration interventions can be active or passive (Simenstad et al., 2006; Suding, 2011). Passive interventions entail reducing or removing the degrading pressure and allowing the ecosystem to recover on a ‘natural’ trajectory without further active management. In the restoration of aquatic environments, passive restoration is often realised as the recovery of ‘natural hydrological cycles’, which reinstate the ecological processes and eventually the ecosystem structure (Simenstad and Warren, 2002; Simenstad et al., 2006). Examples include the removal of artificial structures (e.g. dikes, weirs) to restore hydrological connectivity, improving infrastructure that discharges treated effluent into estuaries to improve water quality, and restrictions on land-use or pollution. Active restoration interventions consist of ongoing anthropogenic actions and management that aim to re-establish the ecological structure and function of an ecosystem. Examples include artificial management of an estuary mouth, drawdown programmes and active revegetation.

Restoration interventions should be sustainable and care must be taken that any interventions do not result in undesirable consequences (Gann et al., 2019).

Prior to any restoration interventions being implemented, the potential responses that may be prompted should be investigated (Choi, 2004). This is challenging but provides valuable insight into the possible ecological trajectory (i.e. ecosystem development) that may be prompted by restoration interventions. Restoration attempts may shift ecosystems to alternative states or regimes (sensu Scheffer and Carpenter, 2003) and in some cases this can be undesirable. Furthermore, there are often time lags in the recovery of coastal and marine ecosystems often as various feedback loops can buffer responses to restoration interventions (Duarte et al., 2015). These time lags are variable depending on the nature and extent of pressures and the recovery of coastal ecosystems can range from months to decades (Jones and Schmitz, 2009; Borja et al., 2010; Verdonschot et al., 2013). Furthermore, the dynamic nature of these ecosystems, especially in light of global change, compound the uncertainty of ecosystem recovery in response to restoration attempts (Simenstad et al., 2006). Appropriate restoration interventions are necessary if an ecosystem is to recover (Zedler and Callaway, 1999), thus careful consideration and planning of restoration interventions are necessary.

A crucial, albeit often overlooked, component of any restoration project is monitoring after restoration has been implemented. Monitoring allows for evaluating the success of achieving the restoration targets and to adaptively manage restoration projects (Hobbs and Norton, 1996; Zedler and Callaway, 1999). Prior to restoration interventions taking place, a monitoring plan should be developed that addresses the restoration targets laid out at the onset of the restoration project. Successful monitoring requires the selection of appropriate indicators over appropriate spatial and temporal scales (Prach et al., 2019). Monitoring is not only helpful for specific projects, but also allows for the sharing of knowledge that can assist with other restoration projects that may take place in the future. Monitoring plans should be developed specifically to meet the requirements of individual projects, although general guidelines to assist with doing so are available (e.g. Galatowitsch, 2012; Howell et al., 2012; Gann et al., 2019). The Society for Ecological Restoration have provided useful tools for monitoring the progress of restoration projects: the Five-Star System and the Ecological Recovery Wheel (Gann et al., 2019). These tools allow for effectively evaluating and communicating the recovery of ecosystems and are valuable additions to monitoring plans for ecological restoration projects.

In summary, coastal ecosystems are under high pressure globally and ecological restoration is the new frontier in their conservation. Effective restoration requires comprehensive planning that includes an understanding of ecosystems in their degraded state and the pressures responsible for degradation. Realistic visions must be set as ecosystems cannot always fully recover due to prolonged or ongoing degradation, often from multiple pressures. Restoration targets and the necessary interventions to achieve the vision must be thoroughly deliberated so as to maximise the benefits of restoration and to avoid any potentially detrimental impacts. Lastly, monitoring the progress and success of restoration projects is essential for adaptive management and generating knowledge that can inform other ecological restoration attempts.

2.2. Estuarine management and restoration in South Africa

Along South Africa's 3000 km stretch of coastline lie 290 estuaries across four biogeographic regions. South African estuaries are diverse in their physical and biological characteristics, with nine distinct ecosystem types being recognised (van Niekerk et al., 2020). These ecosystems are highly productive and support remarkable biodiversity despite covering just 2% of the country's territory — they are thus considered 'super ecosystems' (van Niekerk et al., 2019b). However, South Africa's estuaries are under high pressure (van Niekerk et al., 2019a). The ecology of South Africa's estuaries has been thoroughly studied, with studies dating back to as early as the first half of the 20th century (Allanson and Baird, 1999; Whitfield and Buliwe, 2013). The latest trends in estuarine research and management, as well as recommendations for the future, are synthesised in the regularly updated Estuarine Component of the National Biodiversity Assessment (NBA) (van Niekerk et al., 2019b).

This long history of research and management of estuaries in South Africa has led to the development of comprehensive legislature and management tools relevant to the national protection of these ecosystems. Environmental management in South Africa is guided by a well-established suite of legislation that covers the sustainable use of natural resources and land use planning and development. Most notably, the natural environment is protected under Section 24 of the Constitution. However, until recently South African estuaries have received little legislative protection from development. The country's estuaries have been highly impacted by development, with over 90% of estuaries being affected by urban encroachment (Veldkornet et al., 2015). Stricter regulations regarding development in coastal areas were put in place in 1998 with the proclamation of the National Water Act (Act No. 36 of 1998) and the National Environmental Management Act (Act No. 107 of 1998), and specific environmental acts that followed such as the National Environmental Management: Integrated Coastal Management Act (Act No. 24 of 2008). A National Estuarine Management Protocol has also been developed and is supplemented by the 'Guideline for the development and implementation of Estuarine Management Plans'. Additionally, there are numerous policies, provincial legislations and international conventions applicable to estuarine management in

South Africa. Comprehensive summaries of all relevant legislation are provided in van Niekerk and Taljaard (2003) and Taljaard et al. (2019).

Various tools have been developed to aid estuarine management in South Africa. The country's estuaries have been ranked in terms of importance using the estuary importance score index (van Niekerk et al., 2015). This index accounts for the estuary size, rarity within its biogeographical zone, habitat diversity, biodiversity and functional importance and allows for the allocation of management priorities among the nation's estuaries. The ecological condition of individual estuaries can be determined using the estuarine health index (EHI). The EHI scores changes in the abiotic (hydrology, hydrodynamics and mouth condition, water quality and physical habitat alteration) and biotic (microalgae, macrophytes, invertebrates, fish and birds) characteristics of estuaries from a reference natural condition before anthropogenic impacts (approximately 100 years ago) to the present day. These scores are aggregated to determine the health of the estuary (referred to as the present ecological state, or PES) as an overall score on a scale from 'A' (natural) to an 'F' (critically modified). Estuaries are also assigned a recommended ecological category (REC), which represents the condition towards which the system should be managed and determines the level of protection that should be allocated to the estuary. The minimum REC of any estuary is set as that estuary's PES meaning that an estuary should not be further degraded. These tools are fundamental in estuarine management and provide a standardised approach for determining the current status of estuaries and management targets and allow for the identification of management objectives and priorities.

Despite the existing legislature and management tools, the conservation of estuaries in South Africa is still a major challenge. Nearly 65% of the country's estuarine area has been highly altered by anthropogenic activities, placing pressure on ecological processes and reducing the provision of ecosystem services by estuaries (van Niekerk et al., 2019e). Major pressures on South African estuaries include the modification of flow (particularly decreases in freshwater inflow due to water abstraction and damming), pollution (especially wastewater discharge, stormwater inputs and litter), overexploitation of living resources (such as bait collection and fishing) and physical habitat alteration (due to development and land use) (van Niekerk et al., 2013). Furthermore, the country's estuaries are poorly protected. Currently, less than 1% of the country's estuarine area (representative of less than 10% of estuarine ecosystem types) is well protected (van Niekerk et al., 2019c). Even more concerning is the fact that estuaries that are highly protected (e.g. Knysna, St Lucia) continue to be subjected to degradation due to anthropogenic impacts (Claassens et al., 2020). Estuaries are now considered the most threatened ecological realm in South Africa (van Niekerk et al., 2019c).

Ecological restoration has been identified as a critical knowledge gap and management strategy that needs to be undertaken in South Africa to address the deterioration of the country's estuaries (van Niekerk et al., 2019d). The most common problems that need to be addressed nationally are alterations in water quantity and quality, followed by the restoration of riparian areas and wetlands, and the removal of alien vegetation. Interventions that are necessary at fewer estuaries include reducing the exploitation of living resources (bait collection, fishing), restoring connectivity in estuarine systems, regulating recreational activities and addressing mining impacts. Estuary restoration is a relatively young and uncharted discipline in South Africa. While several projects have been undertaken over the last few decades, these have consisted of ad hoc and site-specific interventions and have achieved various levels of success. Furthermore, few primary publications on estuary restoration projects are available and most relevant information is limited to grey literature. A summary of estuary restoration attempts in the country is provided in Table 2.1.

Table 2.1 Summary of attempted estuary restoration projects in South Africa and their level of success (modified from Claassens et al., under review).

Anthropogenic driver	Anthropogenic pressure	Resultant state	Ecological impact	Restoration activity	Degree of success
St Lucia					
Agriculture	Reduced freshwater inflow related to river flow diversions and restrictions	Abnormally long periods of mouth closure, lake desiccation and hypersalinity	Biodiversity loss, compromised nursery function, fish kills, eutrophication, algal blooms	Reconnection of the Mfolozi River to the St Lucia Estuary	Medium success. Normal water levels restored but system has shifted into an alternate oligohaline, silt laden state.
Zandvlei					
Urbanisation	Water level regulation and nutrient loading	Mouth closure and poor water quality	Eutrophication, spread of pondweed, algal blooms, reduced connectivity, anoxia, fish kills	Actively manage the mouth area by artificial breaching when required	Medium success. More saline conditions and flushing of estuary, reduction in algal blooms.
Zeekoevlei					
Urbanisation	Water level regulation and nutrient loading	Poor water quality	Eutrophication and persistent microalgal blooms	Flushing nutrient-rich water through an annual drawdown programme	Low success. Water quality and estuary health remain poor.
Sipingo					
Urbanisation	Decreased freshwater input	Mouth behaviour altered	Eutrophication and anoxia	Install pipes at the mouth to facilitate tidal exchange	Medium success. Pipes do not allow sufficient flushing of the estuary and connectivity but mangroves thriving.
Sezela					
Pollution	Waste material entering the closed estuary	Anoxic conditions prevail throughout the estuary	Loss of resident and migrant aquatic fauna	Improved processing of sugar mill effluent to reduce biological oxygen demand	High success. Fish and aquatic invertebrates have somewhat recolonised the estuary.

Anthropogenic driver	Anthropogenic pressure	Resultant state	Ecological impact	Restoration activity	Degree of success
Siyaya					
Agriculture	Destruction of riparian vegetation	Erosion of streambanks and deposition of sediments and pollutants in estuary	Filling of estuary with sediments, nutrient loading, reed encroachment and anoxia	Restore riparian vegetation, control reed expansion and update land use plans	Low success. The project failed due to poor coordination and implementation of rehabilitation plan.
Mgobozeleni					
Engineering	Building of bridge above estuary	Tidal and riverine exchange between estuary and inflowing stream constrained	Mass mortality of mangroves upstream of the bridge, reduction in longitudinal fish and invertebrate movements	Bridge demolished and replaced with a redesigned and more appropriate structure	Medium success. Connectivity restored but full tidal range and upstream penetration still not achieved.
Orange					
Urbanisation and Mining	Reduced river flow and loss of connectivity with floodplain salt marshes	System could not be properly flushed, resulting in excessively saline water and soil on floodplain	Approximately 300 ha of estuarine salt marsh lost, impacting negatively on fish, invertebrate and bird populations	Breaching of artificial barriers to facilitate tidal and river inundation of the floodplain	Low success. Connectivity has not been restored and the salt marsh has therefore not recovered.
Great Brak					
Urbanisation and major dam on river	Reduced river inputs to estuary, eutrophication and estuary disturbance	Increased mouth closure, invasion of intertidal habitats by *Spartina alterniflora*	Reduced estuarine-marine connectivity, loss of natural intertidal habitats to an invasive salt marsh species	Scientifically designed dam release policy to promote mouth breaching, implementation of invasive plant control programme	High success. Mouth breaching policy implemented and successful eradication of the alien invasive *Spartina alterniflora*.
Knysna					
Urbanisation and eutrophication	Urban impacts on littoral zone, eutrophication arising from nutrient inputs	Loss of natural supratidal habitat, invasion of eelgrass beds by macroalgae	Reduction in supratidal estuarine habitat, loss of certain eelgrass *Zostera capensis* habitat	Improvement in nutrient discharges from the town's sewage works	Medium success. Protection of remaining supratidal habitats and upgrading of sewage works in progress.
Keurbooms					
Urbanisation	Urban impacts on littoral zone	Loss of natural habitats where bank erosion occurs	Reduction in overall eelgrass *Zostera capensis* habitat for fish and invertebrates	Use of Reno mattresses to reduce flood induced bank erosion	Medium success. Reno mattresses appear to be a suitable habitat for fish and invertebrates.
Nhlabane					
Dune mining, damming and freshwater abstraction.	Reduced freshwater supply to estuary and weir barrier to fish access to estuarine lake	Prolonged estuarine mouth closure and greatly reduced estuarine habitat	Lack of connectivity for fish and invertebrates between the sea and estuary and lack of biotic access to estuarine lake	Artificial breaching of the estuary mouth and construction of a fishway to facilitate access to and from the lake	Low success. Insufficient water to breach mouth at appropriate times and fishway has not restored connectivity between the estuary and lake.

Most estuary restoration measures implemented in South Africa have been active interventions such as artificial management of the estuary mouth (Zandvlei, Orange and Nhlabane) and the construction of artificial structures to restore tidal action (Sipingo) and reduce bank erosion (Keurbooms). Passive restoration approaches that have been implemented include restoring hydrological processes (St Lucia) and improving water quality through addressing point sources of polluted inflowing waters (Sezela, Knysna). Little physical habitat restoration has been attempted, despite large areas of estuarine habitats being degraded or lost nationally (Adams et al., 2019a).

Very few restoration interventions have been highly successful – this level of success was only apparent at the Sezela and Great Brak estuaries. In these cases, it was only necessary to consider a single pressure; however, they highlight the importance of addressing pressures appropriately and promptly, and coordination among the involved parties. The success of several restoration attempts was limited in cases where interventions addressed just one of several pressures, or only the symptoms of degradation and not the cause. Some of these were somewhat successful (Sipingo, Mgobozeleni), while others were not (Nhlabane, Orange, Zeekoevlei). The potential for ecoengineering approaches has been highlighted in the Keurbooms and Knysna estuaries, where artificial structures like Reno mattresses prevent bank erosion while providing a habitat for estuarine fauna. Furthermore, it is possible to lessen the deterioration of urban estuaries through active management, as seen at Zandvlei. Failures can be attributed to the implementation of interventions that do not address the degrading pressures appropriately (Nhlabane, Orange, Zeekoevlei) and a lack of coordination and cooperation among the involved parties (Siyaya).

A major shortcoming in estuary restoration in South Africa is the lack of implementation of projects in a national framework. However, standardised procedures and frameworks have been proposed. Brownlie (1988) proposed a practical approach based on four underlying principles: (1) it is necessary to identify the symptoms of degradation and (2) the causes of these symptoms; (3) a vision for a restored state of the estuary must be determined; and (4) various strategies for attaining the vision must be identified and reviewed. These principles are also stated in more recent international restoration frameworks (e.g. Hobbs and Norton, 1996; Choi et al., 2008; Gann et al., 2019). More recently, a framework for estuary restoration was proposed by Marneweck et al. (2004). This framework covers the context and relevant principles for restoring estuaries and presents a protocol for planning restoration projects. This protocol has been modified from a framework developed for rehabilitating streams in Australia (Rutherford et al., 2000) and is presented in Figure 2.1. However, this framework has not been applied to any estuary restoration projects in South Africa thus far. Recently Adams et al. (2020a) outlined a socio-ecological systems approach for the restoration of South Africa's estuaries, which emphasises the use of an adaptive management cycle (Figure 2.2).

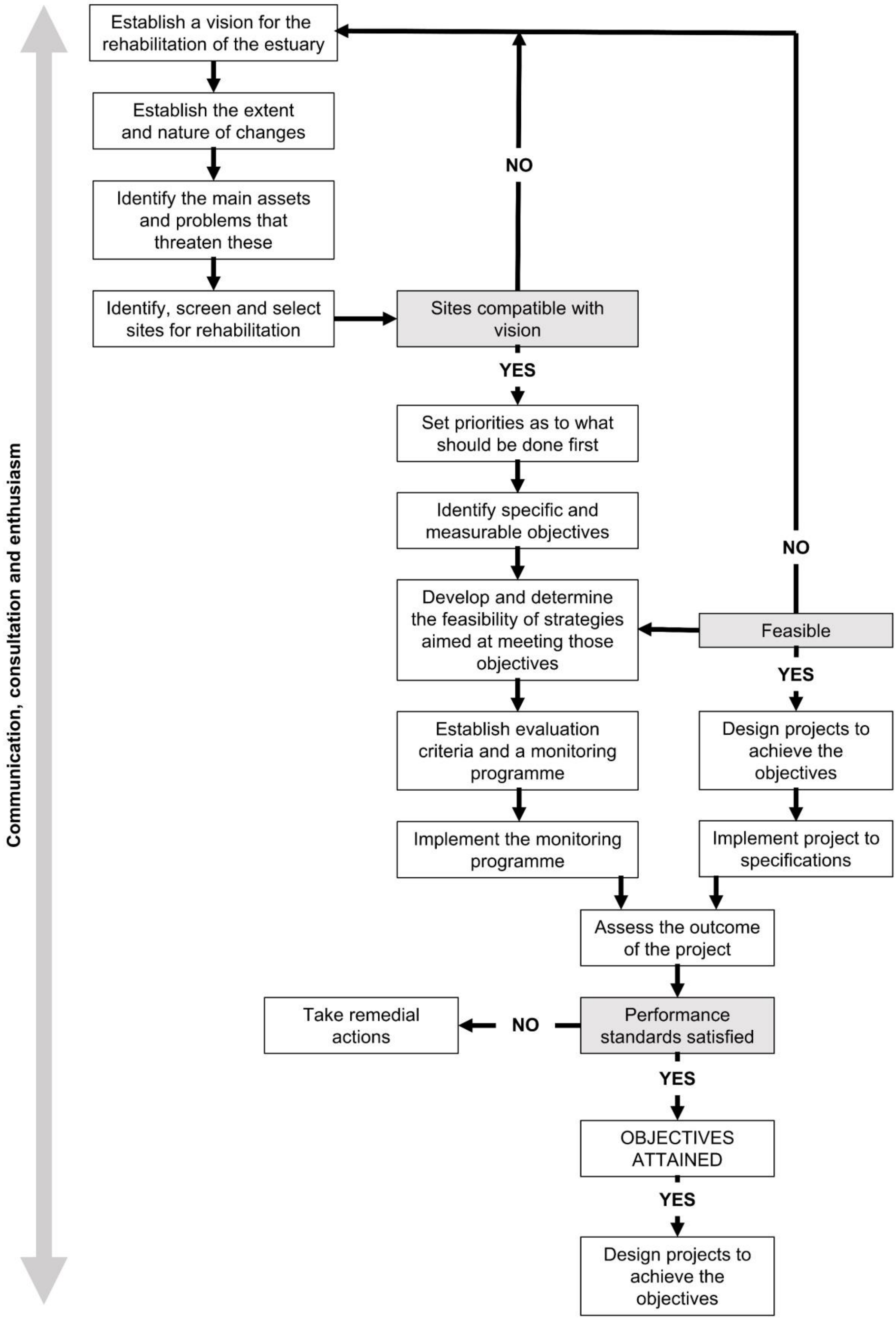

Figure 2.1 Protocol for developing an estuary restoration plan (Marneweck et al., 2004).

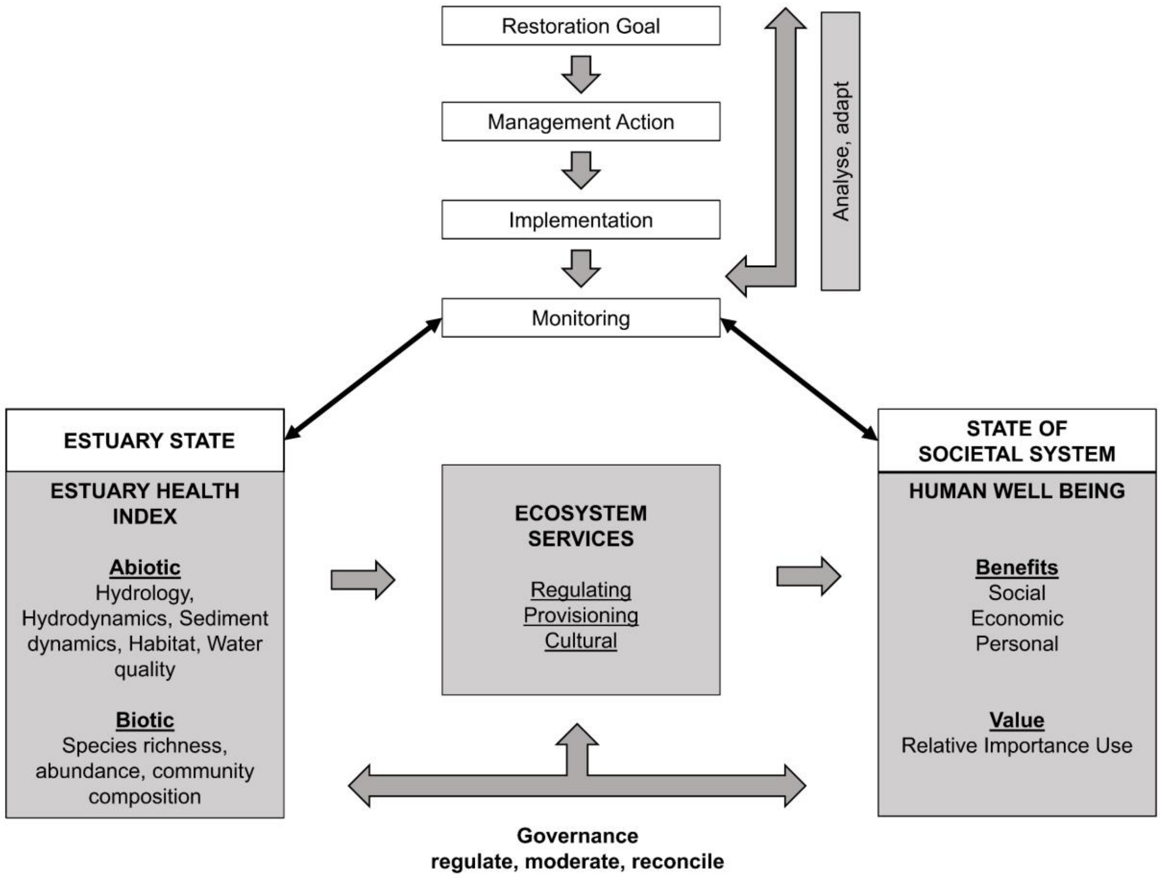

Figure 2.2 A socio-ecological systems framework for estuarine research, management and restoration (Adams et al., 2020).

Various opportunities and constraints exist for restoring South Africa's estuaries. Some of these were identified by Wiseman and Sowman (1992). Some legal mechanisms were identified as opportunities that could enable ecological restoration; however, most of the policies and legislation referred to have since changed since the study had been published. One major policy remains – the Receiving Water Quality Objectives, which stipulates the water quality standards of effluents discharged into estuaries, thus enabling an improvement in estuarine water quality. More recent legal mechanisms that may enable estuary restoration include the requirement for estuaries to be maintained at a PES of 'D' or better in accordance with the National Water Act (Act No. 36 of 1998). Furthermore, opportunities with socioeconomic benefits could arise through Blue Economy initiatives that support circular economies and Ecosystem-based Adaptation initiatives. One such major initiative currently being carried out in South Africa is Operation Phakisa, although there are currently no ecological restoration components to this operation. There is scope for government-driven initiatives similar to the Working for Water and Working for Wetlands programmes that have exhibited ecological and social success.

Constraints to restoration identified by Wiseman and Sowman (1992) included the high cost of restoration interventions, the extent to which estuaries had already been developed and a division of administrative control over estuaries. Although national environmental legislation and estuarine management and administration have undergone many changes in the decades since this study, these opportunities and constraints are still relevant today. The cooperative governance of estuaries remains a challenge. The low level of protection and continual pressure placed on estuaries increase the difficulty of restoring and maintaining these ecosystems. Furthermore, ecological restoration projects are generally costly and it may be difficult to justify allocating resources to such projects in a developing nation fraught with economic disparity and social issues. However, it is often these nations that have the greatest need for ecological restoration initiatives (Laurance, 2001). Prioritising ecological restoration may provide important opportunities for poverty alleviation and the improvement of livelihoods through accelerating socio-economic development (Le et al., 2012; Rohr et al., 2018).

Estuary restoration in South Africa has a long way to go. Strong foundations exist in that the present health of active pressures have been identified, as well as priority actions that need to be taken in the NBA (van Niekerk et al., 2019b). Protocols for planning estuary restoration exist (Marneweck et al., 2004; Adams et al., 2020), but are yet to be implemented. Restoration projects should fall under a national framework, instead of an ad hoc, site specific approach as has been done to date. While such smaller restoration projects may be somewhat successful, it is more beneficial to integrate them into larger overarching projects (Hobbs and Norton, 1996). The protection of estuarine habitats that have not yet been degraded also deserves far more attention as restoring highly degraded estuaries is often unfeasible. Alternatively, the use of artificial habitats should be considered in highly developed estuaries. National environmental legislation should be expanded to include acts that call for ecological restoration of estuaries such as the Clean Water Act and the Estuary Restoration Acts in the USA and the Water Framework Directive and the Marine Strategy Framework Directive in the European Union. The socioeconomic value of the country’s estuaries should be realised and the impact of their degradation can be used as an incentive to protect and restore the ecosystem services they provide.

2.3. Saltworks as ecosystems

Amid the diversity of coastal ecosystems that exist worldwide, saltworks are among the most inconspicuous. Saltworks (also known as salinas, salterns and saltfields) are man-made systems created for extracting salts from saline water through evaporation, but also serve an ecological function as semi-natural wetlands. They are often created in coastal areas, where they are ecologically unique and valuable ecosystems despite their artificial origin. Saltworks have been constructed around the world in various climates. In areas with warm, dry climates (e.g. South Africa, Western Australia, Mexico) salt can be produced year-round; while in areas with a higher annual rainfall (e.g. California, the Atlantic coast of Europe, South China Sea), salt is produced only during summer months (Davis, 2000a). These systems are economically important as they account for a third of global salt production and ecologically important as they host rare and diverse biota (Davis, 1999). Saltworks thus present a rare example of harmony between nature and industry.

Saltworks are comprised of a series of several connected shallow ponds (referred to as saltpans in this thesis). Seawater is pumped into the initial pond and then through a series of concentrating ponds where salts concentrate as the water evaporates. This concentrated saline water, or brine, finally flows into crystalliser ponds from which salt is harvested. During this process, a series of precipitation reactions take place. Firstly, calcium carbonate ($CaCO_3$) precipitates (at salinity levels of ~70 - 105), then calcium sulfate (as gypsum: $CaSO_4.2H_2O$) at a salinity of ~150 and lastly sodium chloride (NaCl) generally at salinity levels higher than 250 (Williams, 1998; Oren, 2002a; Segal et al., 2006). Once most of the NaCl has precipitated on the floors of the crystalliser ponds, concentrated brines known as bitterns remain, which contain mostly Mg^{2+}, K^+, Cl^- and SO_4^{2-} (Oren, 2002a). These are occasionally processed to obtain potash (KCl) but are generally returned to the adjacent coastal waterbody.

The high salinity levels at saltworks place numerous stresses on the organisms present — the most obvious being osmotic stress. Additional stress is caused by changes in the ionic composition of brine as different salts precipitate with increasing salinity through the series of ponds (Copeland and Nixon, 1974; Oren, 2002b; 2013). Due to the high salt stress, biota are limited to halotolerant and halophilic groups (primarily microorganisms), but diversity and productivity are surprisingly high (Pedrós-Alió, 2004). The strong salinity gradient of saltworks creates a series of microenvironments, each characterised by distinct physical features and biotic assemblages (Joint et al., 2002). Several general trends are apparent along the salinity gradient, including a shift from halotolerant species in the concentrating pans to exclusively halophilic species in the extraction pans, a decrease in diversity with a concurrent increase in population densities, and alternations between benthically- and planktonically-driven primary production (Davis, 1990; Des Marais, 1995; Davis and Giordano, 1996; Oren, 2002a; Pedrós-Alió, 2004; Khemakhem et al., 2010).

The primary concentration ponds of saltworks, which receive water directly from an adjacent saline waterbody, have the lowest salinity levels which generally range from 35 to 80. The biota in these ponds resemble those found in natural nearshore communities (Oren, 2002b;

Evagelopoulos et al., 2009). These ponds host the most diverse assemblages of primary producers, with submerged macrophytes, macroalgae, phytoplankton and microphytobenthos (MPB) often present (Davis, 2000a). Phytoplankton and MPB community compositions are generally similar and comprised mostly of various diatoms, dinoflagellates and cyanobacteria (Clavero et al., 2000; Oren, 2002b). These ponds often have a brown or green water column due to the abundance of planktonic diatoms and dinoflagellates (Davis, 1990; Ayadi et al., 2004). Macrophytes can spread across the lower salinity ponds, creating habitats utilised by fauna occurring here (Britton and Johnson, 1987; Davis, 2000a; Sánchez et al., 2006). Diverse invertebrate assemblages are often found in these ponds and are an important food source to higher trophic levels such as fishes and waterbirds (Evagelopoulos et al., 2009; de Melo Soares et al., 2018).

In the ponds of intermediate salinity (80 – 150), submerged macrophytes and macroalgae disappear and MPB become the dominant primary producers, forming benthic mats on the pond floors (Javor, 1983; Davis, 1990; Pedrós-Alió et al., 2000). Benthic mats can outcompete phytoplankton and greatly limit their growth (Fong et al., 1993). However, phytoplankton may still be abundant, with biomass and production rates often peaking at salinities around 100 (Pedrós-Alió, 2004). This is generally due to the proliferation of a single species – the halophilic chlorophyte *Dunaliella salina*. This species grows optimally in this salinity range and often colours water in these ponds an orange-red hue due to its high β-Carotene content (Oren, 2014). Diatoms and dinoflagellates are scarce in these ponds, but cyanobacteria such as species of *Aphanothece*, *Spirulina* and *Phormidium* are often present (Khemakhem et al., 2010). Zooplankton are often abundant in this salinity range, especially brine shrimp (*Artemia salina*) and brine fly larvae, but copepods, rotifers and ciliates are also common (Davis, 2000a; Toumi et al., 2005). These zooplankton communities are important in controlling phytoplankton abundance through grazing and are an important food source to higher trophic levels like waterbirds (Davis, 1978; 2000a).

In the ponds with salinity > 150, gypsum crusts form on the pond floors and resultantly planktonic organisms become the dominant primary producers as opposed to the benthically-driven primary production in intermediate salinity ponds (Oren, 2009). Layered MPB communities form within the gypsum crusts, but the most abundant organisms in these ponds are planktonic and include *D. salina* and a few species of halophilic archaea and bacteria (Davis, 2000a; Litchfield et al., 2000; Oren, 2009). These carotenoid-rich plankton colour the water a pink-red hue (Davis, 2000a). Zooplankton communities in these ponds are far less diverse and generally consist of just the brine shrimp *A. salina* and the ciliate *Fabrea salina* (Toumi et al., 2005). In the crystalliser ponds with salinities > 250 (where NaCl begins to precipitate), heterotrophic halophilic archaea and bacteria become prevalent, but *D. salina* may still be abundant (Davis, 1990; Des Marais, 1995; Litchfield et al., 2000). The halobacteria and *D. salina* colour the brine a deep red hue (Oren and Dubinsky, 1994). After NaCl precipitation, no biota can survive in the remaining bitterns due to the intolerably high Mg^{2+} concentrations (Javor, 1983).

Although salinity is the primary driver of community structure in saltworks, other physico-chemical characteristics like nutrients, temperature, pH and dissolved oxygen also play a role, particularly for microbial communities (Ayadi et al., 2004; Khemakhem et al., 2010; Zhang et al., 2016). In turn, these variables are influenced by the biota that characterise the different ponds in a saltworks (Chien, 1992; Davis, 2000a). Resultantly, these characteristics vary spatially and temporally. In fact, temporal variability can be far higher in saltworks than in other coastal and marine ecosystems, often with large diurnal, daily and seasonal fluctuations (Britton and Johnson, 1987). Saltworks thus represent highly complex and heterogenous aquatic ecosystems.

Nutrient availability is particularly important as it drives productivity. Most nutrients in a saltworks originate from the water initially pumped into the system. However, nutrient availability in saltworks is affected by many variables, including salinity. Salinity influences nutrient availability through causing changes in the types and activity of primary producers and other organisms present, the relative abundance of different ions in the water column and nutrient fluxes between the sediment and the water column (Clavero et al., 1993; Joint et al., 2002; Cotner et al., 2004). Nutrient availability also varies among ponds as water is pumped from one to the next, with organisms often dying and decomposing as they reach ponds that exceed their salinity tolerance limit, thereby adding organic matter and nutrients to subsequent ponds (Davis, 1993). Nutrient concentration can also be influenced by runoff, dilution by rainfall and evapoconcentration. Due to all of these factors, nutrient limitation in saltworks is complex and idiosyncratic. In some systems, primary production is limited by N, while P is limiting in others, but temporal or spatial shifts in nutrient limitation may occur (Javor, 1989; Joint et al., 2002; Dolapsakis et al., 2005; Oren, 2009).

Water temperature can also be highly variable in saltworks, which may have various impacts on organisms. For example, water temperature influences the metabolic activity of organisms and the solubility of compounds in water. Excessively high temperatures are especially unfavourable as this places heat stress on organisms, leads to decreased availability of dissolved oxygen in the water column and may lead to algal blooms if sufficient nutrients are available, which could further deplete oxygen from the water (Copeland, 1967; Evagelopoulos et al., 2009). Water temperature is largely influenced by depth in that shallower waterbodies can heat up more rapidly and to higher temperatures, and higher vapour pressures are maintained for longer periods of time (Garret, 1966). Salinity also influences temperature as a higher salt content decreases the specific heat capacity of the water, thus the more saline water is, the quicker it can reach high temperatures (Copeland, 1967). Furthermore, various planktonic species found in saltworks can colour the brine in various hues, increasing light absorption and thus brine temperature (Sammy, 1983).

Saltworks also display spatial and temporal variability in pH. Water in saltworks is generally alkaline, resembling that of the influent seawater (~7.5 – 8.4) in the initial ponds and gradually decreases through the series of ponds of increasing salinity, usually reaching a pH between 7 and 8 in the crystalliser ponds (Du Toit, 2001; Dolapsakis et al., 2005; Costa et al., 2015). Primary production causes diurnal changes in pH and these changes are more pronounced in the highly productive low salinity ponds (Rahaman et al., 1993; Du Toit, 2001; Ayadi et al., 2004). Slightly alkaline water is beneficial for algal growth, but excessive alkalinity present considerable bioenergetic challenges to organisms, and may threaten them with ammonia toxicity (Oren, 1999; 2013; Smyth and Elliot, 2016).

Overall, saltworks are unique ecosystems with many intricacies and notable heterogeneity. This heterogeneity creates numerous niches ranging from environments similar to natural coastal waterbodies to harsh environments hospitable only to extremophiles. Resultantly, saltworks support extremely diverse communities in a small area and these communities can be unexpectedly productive. Additionally, these unique habitats provide numerous ecosystem services including the provisioning of raw materials, supporting biodiversity, regulating water quality, and have a high cultural value (Crisman et al., 2009; de Melo Soares et al., 2018). Consequently, saltworks are valuable ecosystems that deserve proper management and conservation despite being artificially created.

2.4. The case for rehabilitating the abandoned saltworks

Although saltworks are of economic and ecological importance, they have become more frequently abandoned in many parts of the world (Bouzillé et al., 2001; Crisman *et al*., 2009; Dittmann et al., 2019; Dos Reis-Neto et al., 2019; Tran et al., 2019). This is due to various changes in the salt industry since the mid-1900s, such as increased competition and advances in refrigeration lessening the need for salt to preserve food products, resulting in the decreased profitability of saltworks. The abandonment of saltworks results in a loss of rare ecosystems and the various services they provide and often leaves behind barren landscapes with little ecological value (Crisman *et al*., 2009).

Various opportunities exist for the ecological restoration of abandoned saltworks and several such projects have been or are currently being carried out around the world. Some of these projects focus on the environmental benefits of restoring saltworks, such as the notable South Bay Salt Pond Restoration Project in San Francisco Bay, which aims to restore over 15000 acres of saltpans to natural tidal wetlands and associated habitats (South Bay Salt Pond Restoration Project, no date). In Southern Europe and North Africa, BirdLife Partners are carrying out the Saltpan Recovery Project that aims to restore and maintain saltpans to provide stopover sites for waterbirds migrating along the East Atlantic Flyway (BirdLife International, no date b). Dittmann et al. (2019) have shown the potential of restoring tidal connectivity to an abandoned saltworks in South Australia as a blue carbon project. Defunct saltworks in Europe have been restored for cultural and economic reasons, such as the salinas of Guérande, France, which have been restored for artisanal salt production and as tourist attractions (Neves, 2002), and the Lion Salt Works in Marston, England, which has been restored as a museum (Hewitson, 2017). Other economic opportunities for abandoned saltworks include the creation of rice fields and aquaculture prospects such as fish and oyster farms as has been done in western France (Crisman *et al*., 2009). Saltworks can also be restored for recreational purposes, such as a defunct saltworks in Italy that has been transformed into a popular urban park (Lai, 2013).

Little information is available on the salt industry in South Africa. Most salt production occurs inland where salt is harvested from underground rock salt deposits. Coastal saltworks are very rare, with just two of these systems at the Berg Estuary and one at the Sout Estuary on the west coast, as well as three at Swartkops Estuary and one at Coega Estuary on the south-eastern coast (Du Toit, 2001; DMR, 2011). Local salt production is in decline, with production companies having decreased from 86 to 17 within the last decade due to strong competition with salt producers in neighbouring countries (Who Owns Whom, 2019). However, just one reported case of an abandoned coastal saltworks has been found – it is currently being planned to demolish an unused saltworks at Coega Estuary to expand the adjacent port (SAHRIS, no date). If local salt production continues to decline, it can be expected that more saltworks will be abandoned, leading to the loss of these rare ecosystems and leaving behind large barren areas that provide little ecosystem services. Rehabilitating abandoned saltworks could thus become increasingly necessary. No recorded cases of the ecological restoration of any saltworks in South Africa have been found.

This thesis focuses on a saltworks at Swartkops Estuary on the south-eastern coast of South Africa that has been abandoned after over 50 years of salt production. Excluding the economic value of the saltworks while salt was being produced there, the site had a high ecological value, particularly for waterbirds. The saltworks hosted thousands of birds annually and was an important component of the Swartkops Estuary – Redhouse and Chatty Salt Pans IBA complex (BirdLife International, no date a). The saltworks hosted thousands of waterbirds annually and triggered various criteria to be classified as a Global IBA: A1 (globally threatened species); A4i (congregations of waterbird species); and A4iii (wetland of international importance). The site also accommodated some of the largest breeding colonies of several waterbird species in southern Africa (Martin and Randall, 1987). However, the latest assessment of the IBA by BirdLife International (in 2012) found the site to be in a very poor condition and under high threat with low management response. The condition has worsened since the saltworks has been abandoned as thousands of waterbirds have since been lost from the IBA (P Martin, unpubl. data, 2020).

The rehabilitation of the saltworks as a waterbird sanctuary will be important in safeguarding the IBA status of the Swartkops area. The loss of waterbird habitat from the estuary has already been identified as an issue prior to the abandonment of the saltworks (van Niekerk et al., 2015). The rehabilitation will assist with the Zwartkops Conservancy's current plans to have Swartkops Estuary designated as the first urban Ramsar Site in South Africa. Furthermore, rehabilitating the saltworks can also assist in improving the PES of Swartkops Estuary from a 'D' to the REC of 'C'. The national importance of the estuary is largely to the diversity of habitats and biodiversity it supports (van Niekerk et al., 2015). The abandonment of the saltworks has resulted in the loss of a rare (albeit artificial) habitat type and a notable decrease in the abundance and diversity of waterbirds. Therefore, there are multiple local benefits to rehabilitating the saltworks. However, the rehabilitation would also be relevant nationally and internationally, as there have been long-term declines in waterbird populations in South Africa as well as in Palearctic-African migrant bird populations (Sanderson et al., 2006; Raimondo et al., 2019). Recreating a wetland at the saltworks would provide a regionally important breeding site for resident waterbird species and stopover sites for that are vital to migratory species (Martin and Randall, 1987; Kirby et al., 2008).

The rehabilitation of the saltworks as a waterbird habitat aligns with Section 56e (principles for mine closure) in the Mineral and Petroleum Resources Development Act (Act No 28 of 2002), which states that "the land is rehabilitated, as far as is practicable, to its natural state, or to a predetermined and agreed standard or land use which conforms with the concept of sustainable development". The rehabilitation of mines in South Africa is now governed by the Regulations for Financial Provision for Prospecting, Exploration, Mining and Production Operations issued in Government Notice R1147 (GNR 1147) and promulgated under the National Environmental Management Act (Act No 107 of 1998). The regulations also stipulate that any holder of mining rights or permits must develop plans for rehabilitating any decommissioned mining sites and financial provision for said rehabilitation.

The saltworks lies in a remote area away from most urban and residential areas of Port Elizabeth and has little apparent social or cultural value. It is unlikely that restoring the saltworks as a cultural or historical attraction will be successful, especially since there is little infrastructure remaining and the saltworks are now completely defunct. However, the site does have some recreational value. The dirt roads between the saltpans are used by cyclists, hikers sometimes visit the area, and car and motorcycle tracks are apparent in the now-dry saltpans. The primary motivation for rehabilitating the site would thus be ecological, although various opportunities may arise from this project. One opportunity may be the alleviation of stormwater pollution entering the nationally important Swartkops Estuary. A possible intervention to recreate wetlands at the saltworks is the use of stormwater from the nearby Motherwell stormwater canal to fill the saltpans. This urban stormwater discharge has been recognised as an issue at the estuary that needs to be addressed (Adams et al., 2019b). Some economic opportunities may arise, such as using the saltpans for aquaculture or ecotourism, particularly birding (Crisman et al., 2009). The saltworks may also be used as sites for research and education. The rehabilitation of the saltworks will require active management, which may have high running costs, and the potential for using the site to generate income to cover these costs should be explored.

In conclusion, saltworks are unique ecosystems of high ecological importance. Coastal saltworks are particularly rare in South Africa, and national salt production is in decline. The abandonment of the saltworks at Swartkops Estuary may be foreshadowing the fate of the country's few saltworks associated with estuaries. The rehabilitation of this saltworks as a waterbird habitat is of local and national importance and is globally relevant. This would be the first recorded case of the rehabilitation of a saltworks in the country and would serve as a case study of habitat restoration of South African estuaries, of which there have been few attempts. In order to ensure that this rehabilitation project is a success, thorough planning is necessary. A vision needs to be established and must be accompanied with realistic targets and appropriate rehabilitation actions. Lastly, monitoring will be crucial to facilitate adaptive management and to generate knowledge on the rehabilitation of these rare ecosystems.

3. Study site

3.1. Swartkops Estuary

Swartkops Estuary lies on the coast of Algoa Bay in the Eastern Cape of South Africa. The estuary is centrally located in the metropolitan Nelson Mandela Bay Municipality (NMBM), which includes the city of Port Elizabeth, the towns of Uitenhage and Despatch and various other residential, industrial and rural areas. The estuary is fed by the Swartkops River and its main tributary, the Elands River, both originating some 150 km inland in the Great Winterhoek mountains near Uitenhage (Reddering and Esterhuysen, 1981). The mouth of the Swartkops Estuary is permanently open and meets the Indian Ocean just south of Bluewater Beach. The lower reaches of the estuary contain expansive intertidal salt marshes, mudflats, sandbanks and islands. Further upstream, the estuary is narrower with large meanders and steeper banks that are flanked by extensive floodplains.

Swartkops Estuary falls in the warm temperate biogeographic region of southern Africa. Mean temperatures range from 15°C to 32°C in the middle of summer and from 5°C to 18°C in the middle of winter (Haigh, 2002). Large daily variations in temperature are not uncommon. Strong winds are a frequent occurrence along the coastline here, including at Swartkops Estuary, and often reach gale force. A south-easterly wind direction is prevalent in summer, while strong south-westerly winds are typical in winter. Swartkops Estuary lies in South Africa's year-round rainfall zone, with the highest rainfall typically occurring in June and October while summer months are the driest (Klages et al., 2011). Mean annual precipitation (MAP) is 636 mm, ranging from 500 – 1000 mm (Baird, 2001). As with the rest of southern Africa, droughts are common in the region. At the time that this thesis was being written (2019/2020), a prolonged drought has persisted since 2015. The Eastern Cape has been especially affected, and this drought has been regarded as the most severe occurring here in the last millennium (Ellis, 2019).

Swartkops Estuary receives little freshwater input from the catchment. Median flow is low (0.3 $m\ s^{-1}$) and there is effectively no baseflow (van Niekerk et al., 2015). Mean annual runoff (MAR) in the catchment is $84.2 \times 10^6\ m^3$ (Reddering and Esterhuysen, 1981). Roughly 16% of the MAR in the catchment is retained by the Groendal Dam on the Swartkops River and minor impoundments on the tributaries of the Elands River; however, these obstructions reduce freshwater inflow by just 5% (Baird et al., 1986). Freshwater flow is further impeded by several obstructions including causeways on the Swartkops River (Enviro-Fish Africa, 2009). Occasionally there are small floods in the system, usually 40 to $80 \times 10^6\ m^3$ in volume (DWAF, 1999), but floods of up to $160 \times 10^6\ m^3$ have been recorded (HKS, 1974). Saltpans in the middle reaches of the estuary generally hold back floodwaters from the lower reaches. Several bridges span the upper, middle and lower reaches of the estuary, but do not significantly disrupt flow. Despite the low level of natural freshwater input, the estuary receives considerable amounts of treated effluent from three wastewater treatment works (WWTWs) that discharge into the Swartkops River and together contribute to roughly half of the estuary's inflow (DWAF, 1999).

As the estuary mouth is permanently open to the sea, the system is exposed to semidiurnal tidal cycles. The tidal range varies from 0.5 m during neap tides up to 2 m during spring tides and the tidal prism is approximately 2.9 x 10^6 m^3 (Reddering and Esterhuysen, 1981). The tide intrudes roughly 16.4 km into the estuary where it reaches its head at Perseverance (Baird et al., 1986). The lower reaches of the estuary are marine dominated and are regularly flushed and well-mixed by the tides. The upper reaches experience much poorer water circulation as sandbanks that formed behind a bridge constrict flow, resulting in long water residence times (10 – 14 days) (Goschen and MacKay, 1992; MacKay, 1993). A distinct longitudinal salinity gradient is commonly present in the estuary, with salinity ranging from typical marine levels at the mouth (~35) to fresh at the tidal head (> 1.5) (Adams et al., 2019b). Throughout most of the estuary, vertical stratification of the water column is a common occurrence during neap tides and under high freshwater inflow; while spring tides promote vertical mixing of the water column (Pretorius, 2015).

Swartkops Estuary is one of the largest estuaries in southern Africa's warm temperate region. It has a total habitat area of 908 ha comprised of five distinct estuarine habitats: Intertidal and supratidal salt marsh (209.2 and 338.15 ha, respectively), submerged macrophytes (44.7 ha), reeds and sedges (4.5 ha) and open water (135 ha) (Adams et al., 2019a). The estuary's salt marshes are among the largest and most diverse in the country (Baird et al., 1986; Adams et al., 2016; 2019a). Faunal diversity at the estuary is also high, including communities of invertebrates, ichthyofauna and waterbirds (Baird et al., 1986; Martin and Baird, 1987; Hanekom et al., 1988; Scharler et al., 1997).

The large size, habitat diversity and biodiversity of Swartkops Estuary give it a high ecological importance, and the system has been ranked as the 11th most important estuary in South Africa in terms of conservation value (Turpie et al., 2002; van Niekerk et al., 2015). The ecological significance of the estuary has been recognised through other designations as well. It has been designated as a Critical Biodiversity Area (category CB1) and formal protection of the system has been recommended (Enviro-Fish Africa, 2011). Additionally, the Swartkops Estuary and several adjacent saltpans has been designated by BirdLife International as a globally Important Bird and Biodiversity Area (IBA) complex.

As Swartkops Estuary is centrally located within the metropolis Nelson Mandela Bay it is the centre of many anthropogenic activities, giving the estuary a high socio-economic value. The estuary generates nearly R85 million annually in the local economy primarily through tourism and its function as a nursery area, but it is also important for subsistence fishing and bait collection (Turpie and Clark, 2007). Additionally, the total value of properties around the estuary is appraised at R155 million. Swartkops Estuary is also valuable as a popular site for recreation (e.g. fishing, boating and birdwatching) and for cultural and spiritual ceremonies.

Despite the conservation importance of Swartkops Estuary, it has been subjected to long-term anthropogenic pressures. Various industrial activities such as sand mining, saltworks, brickworks, the automotive industry, wool washeries and tanneries have impacted the estuary for decades (Baird et al., 1986). Several residential areas, ranging from small villages to densely populated informal settlements, surrounding the estuary also place pressure on the system. Due to the prolonged degradation of the estuary, it has a present ecological state (PES) of 'D'[1], while the recommended ecological category (REC) has been set as a 'C'[2] (van Niekerk et al., 2015).

A key pressure that has led to the degradation of the estuary is the deterioration of water quality due to inflowing polluted effluent from numerous sources (Adams et al., 2019b). Industrial wastewater and stormwater runoff often contaminated with untreated residential sewage enter the system through the Chatty River and the Markman and Motherwell stormwater canals in the middle and lower reaches of the estuary. Three wastewater treatment works (WWTWs) discharge into the Swartkops River and account for roughly half of the estuary's inflow (DWAF, 1999). Poor water quality has been a persistent issue in Swartkops Estuary that has had negative ecological impacts (e.g. eutrophication and algal blooms) and social consequences (e.g. the cancellation of a popular swimming event) (Adams et al., 2019b). Physical habitat loss due to development is another key issue at Swartkops Estuary. Development in the estuarine functional zone (EFZ) include numerous bridges in the middle and lower reaches of the estuary, and commercial developments such as solar saltworks and various residential areas. These developments have primarily impacted the estuary's important salt marshes, especially the supratidal salt marsh which has been reduced from 1013.15 ha to just 338.15 ha (Adams, 2020).

[1] Largely modified estuary: A large loss of natural habitat, biota and basic ecosystem functions has occurred.

[2] Moderately modified estuary: A loss and change of natural habitat and biota have occurred, but the basic ecosystem functions are still predominantly unchanged.

The administrative authority responsible for the management of Swartkops Estuary is the NMBM. Other stakeholders are involved in the conservation of the estuary, such as the Zwartkops Conservancy, a local non-governmental organisation. Two protected areas, the Swartkops River Valley and Aloes Nature reserves, lie just north of the estuary, but the estuary itself is not formally protected under the National Environmental Management: Protected Areas Act (Act No. 57 of 2003). As required by the Integrated Coastal Management Act (Act No. 24 of 2008), a Swartkops River Valley Management Forum had been established in the past but has since disbanded. The Forum, together with the C.A.P.E. Estuaries Programme developed an integrated environmental management plan for the Swartkops River valley including the estuary and adjacent nature reserves (Enviro-Fish Africa, 2011). However, this management plan has not been gazetted and as such has it has not been implemented.

3.2. Redhouse and Bar None saltpans

The abandoned solar saltworks lies on the floodplain of Swartkops Estuary approximately 5 km upstream of the mouth and consists of two main areas: the Redhouse saltpan (33°50'10"S 25°35'E) and the Bar None saltpans (33°49'16"S 25°33'42"E) (Figure 3.1). The Redhouse saltpan (Pan 1) lies on the northern bank of the estuary and is approximately 98.6 ha in size. The Bar None saltpans consist of three saltpans (Pans 2 – 4) on the southern bank (14.7 ha, 21.9 ha 33.2 ha in size, respectively). Another series of smaller saltpans lie on the northern side of the estuary opposite the Bar None saltpans, but they are not included in this rehabilitation project.

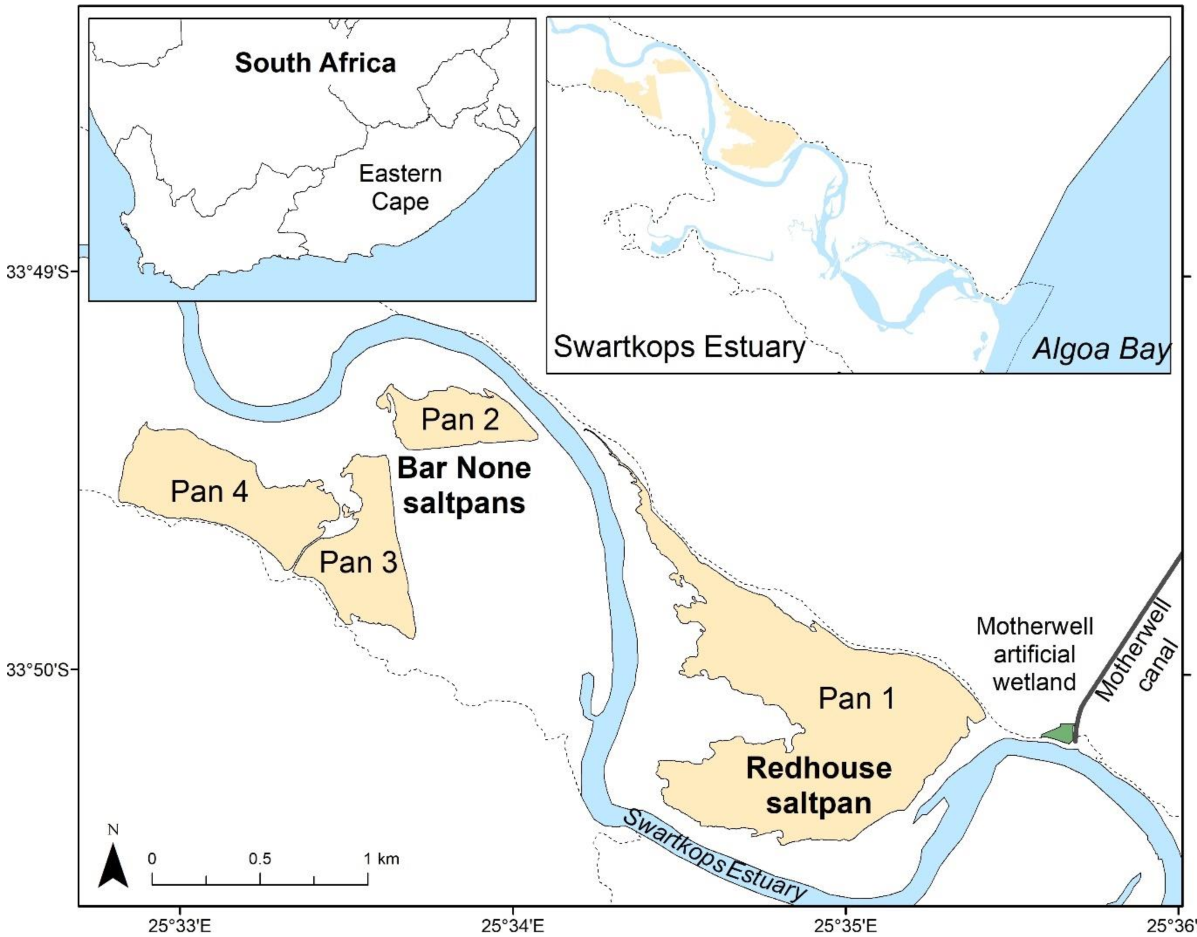

Figure 3.1 The abandoned solar saltworks at Swartkops Estuary. Refer to Appendix 1 for a detailed habitat map.

Little information is available on the history and functioning of the saltworks. Available information is limited to a brief description of the Redhouse saltpan by Martin and Randall (1987). Included here are also personal communications from P. Martin who has carried out bird counts at the site since the late 1980s. Originally a small depression where seasonal vleis occasionally formed, Pan 1 was excavated in 1961 – 1962. The excavated spoil was used for building retaining walls around the saltpan and creating several islands within it. Pan 1 is the primary concentration pan into which estuary water is pumped via a pumphouse on the northern-eastern corner. The saltpan was generally kept full, with salinity ranging from 30 to 50, but periodic floods would result in salinity decreasing to below 10. The retaining walls were sometimes breached during floods to release the excess water, and the saltpan would be left with low water levels for months. The saltpan was refilled when the salinity of the estuary water in the area increased to near that of seawater again.

The same pumphouse used to fill Pan 1 was used to pump water from Pan 1 underneath the estuary into Pan 2. Water was then gravity fed into Pans 3 and 4 consecutively, with salinity increasing in each successive pan. To the south of Pan 2 and to the east of Pan 3 lies a depression into which brine occasionally leached, resulting in areas of bare salinised sediment. Another pumphouse is located in the corner of Pan 4, which was used to pump the brine under the estuary once again to the smaller saltpans on the northern side of the estuary. The brine was pumped through this last series of evaporation ponds and to the top of the escarpment to the north into an extraction pan from which the salt was harvested.

Although manmade, the saltworks provided an otherwise rare wetland habitat type that was of high ecological value, especially to waterbirds (Figure 3.2). The site was an essential component of the Swartkops Estuary – Redhouse and Chatty Salt Pans IBA complex and hosted thousands of waterbirds annually (Marnewick et al., 2015). When water levels were high (Figure 3.3A), the islands within this saltpan provided safe breeding areas for waterbirds isolated from egg poachers and small mammalian predators. Resultantly, this saltpan was considered the most important mainland breeding ground in the Eastern Cape for various resident waterbird species including the Caspian tern, white-breasted cormorant, kelp gull, grey-headed gull and sacred ibis (Martin and Randall 1987). Similar islands were present in some of the Bar None saltpans. Summer months saw a large influx of Palearctic migrant waterbirds, especially waders, which would feed at the saltpans as water levels would decrease to expose wet mudflats providing feeding areas (Figure 3.3B). The saltworks were abandoned in 2018, partly due to continual theft and vandalism of the infrastructure and have since been left to dry.

Vegetation cover within the saltpans is sparse and patchy and mostly limited to hummocks. Most areas of lower elevation lie bare and a salt layer is visible on the surface of the sediment in some areas. The vegetation composition within the saltpans resembles the surrounding supratidal and floodplain salt marsh. Communities are highly monospecific with *Salicornia pillansii* being the dominant species. Other species include *Disphyma crassifolium*, *Suaeda fruticosa*, *Atriplex lindleyi* subsp. *inflata*, *Bassia diffusa*, *Psilocaulon dinteri*, *Spergularia media* and *Sporobolus virginicus*. Vegetation cover is mostly limited to the hummocks in the saltpans. Prior to abandonment, submerged macrophytes (*Ruppia cirrhosa* and *Enteromorpha* spp.) had been recorded in the Redhouse saltpan (Martin and Randall, 1987). Another striking feature at the saltworks is the widespread presence of remnant reef clusters formed by the invasive Australian tubeworm *Ficopomatus enigmaticus* (Figure 3.4).

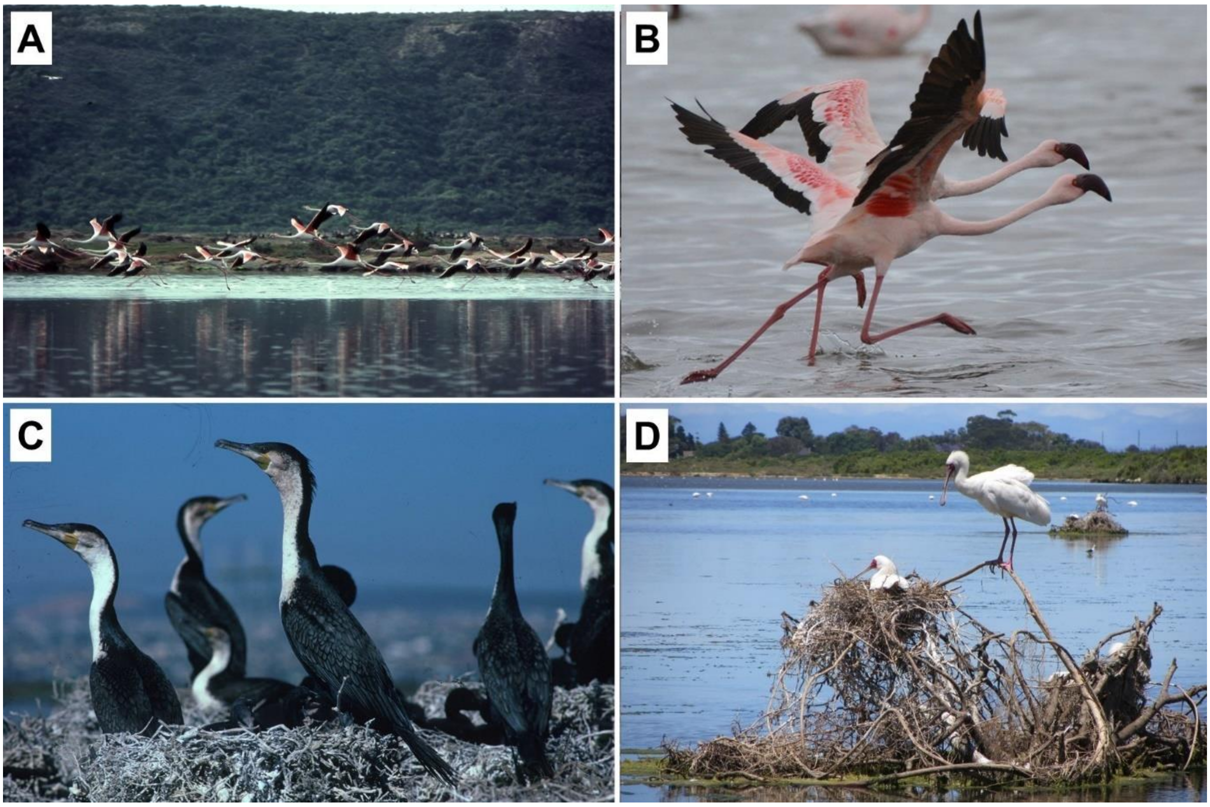

Figure 3.2 The saltworks hosted various waterbird species such as greater and lesser flamingos (A and B, respectively), white-breasted cormorants (C) and African spoonbills (D). Photos courtesy of Paul Martin.

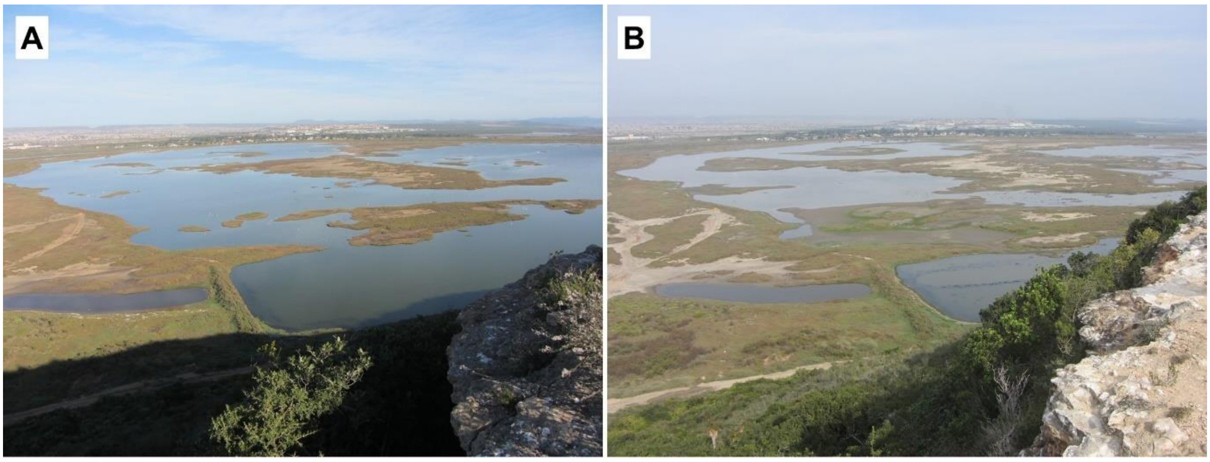

Figure 3.3 The Redhouse saltpan filled to maximum capacity (A) and with water at the lowest allowable level (B). Photos courtesy of Paul Martin and captured in 2013 and 2014, respectively.

Figure 3.4 The salt marsh vegetation at the abandoned saltworks.

Figure 3.5 Remnant reef structures formed by the Australian tubeworm *Ficopomatus enigmaticus* lie scattered across the abandoned saltworks.

3.3. Motherwell artificial wetland

The Motherwell Canal Artificial Wetland System (33°50'10"S 25°35'39"E), hereafter referred to as the Motherwell artificial wetland (or MWAW), lies on the northern bank of the middle reaches of Swartkops Estuary, roughly 400 m east of the Redhouse saltpan (Figure 3.6A). The MWAW was commissioned by NMBM and designed by SRK Consulting as a pilot project to evaluate the use of artificial wetlands to treat polluted water entering rivers and estuaries in South Africa.

The MWAW is a hybrid system consisting of a primary subsurface flow cell (a pebble bed) and two secondary free water surface cells (comprised of stands of bulrush *Typha capensis*). It was constructed adjacent to the outflow of the Motherwell Canal (Figure 3.6B), a 4.2 km long canal that drains stormwater from the densely populated Motherwell residential area that lies on top of an escarpment on the northern side of the estuary. Urban runoff persistently flows through the Motherwell Canal, even during dry weather, with consistently high concentrations of contaminants including nutrients, bacteriological pollutants and heavy metals (Stroebel, 2019; Lakane, 2020). Additionally, raw sewage is regularly discharged into the canal and debris is illegally dumped into it (Adams et al., 2019b).

The MWAW became operational in early 2010, prior to which the polluted Motherwell stormwater canal discharged straight into the estuary. The stormwater greatly exceeded the general guidelines for recreational use as stipulated by the Department of Water Affairs, and the canal had long been recognised as a major point source of water pollution to the estuary (Emmerson, 1985; Scharler, et al., 1997). In order to effectively treat the stormwater, SRK Consulting (Pty) Ltd (2010) determined that the wetland would need to be approximately 4.5 ha in size in order to meet the hydraulic requirements of the Motherwell Canal. However, the wetland constructed was just 0.8 ha in size due to a lack of space, and just 20% of the runoff from the canal was diverted into it. Initial monitoring revealed that the artificial wetland was successfully decreasing concentrations of bacteriological pollutants (total coliforms, faecal coliforms and *Escherichia coli*) (SRK Consulting (Pty) Ltd, 2010). However, nearly all of the runoff from the Motherwell Canal has since been diverted into the artificial wetland and the efficiency of the wetland has been reduced. The MWAW is currently still effective at removing bacteriological pollutants, but nutrient concentrations are not being decreased before the stormwater is discharged into the estuary (Stroebel, 2019; Lakane, 2020).

Figure 3.6 The Motherwell artificial wetland, with the Redhouse saltpan in the distance (A) and the adjacent Motherwell Canal (B).

4. Baseline conditions at the abandoned saltworks

4.1. Introduction

Saltworks are man-made systems created to harvest salts from brine. These systems are comprised of a series of connected saltpans through which the brine flows. Brine becomes more saline in each successive saltpan due to evapoconcentration, resulting in a distinct salinity gradient (Korovessis and Lekkas, 2009). Due to the high salt stress these hypersaline environments impose on organisms as well as the pronounced salinity gradient, saltworks host unique and surprisingly diverse biotic communities (Oren, 2009). Along with supporting biodiversity, saltworks provide a variety of other ecosystem services such as the provisioning of natural resources, improving water quality, and cultural value (Crisman et al., 2009; de Melo Soares et al., 2018). Despite their man-made origin, saltworks are unique and relatively rare ecosystems with high conservation value.

Despite their importance, saltworks have been abandoned in various regions around the world (Crisman et al., 2009; Dos Reis-Neto et al., 2013; Dittmann et al., 2019). The abandonment of these systems leaves behind large areas of disturbed and highly salinised land and leads to the loss of most, if not all, of the ecosystem services they provide (Crisman et al., 2009). In some cases, a tidal connection at abandoned saltworks can allow natural coastal ecosystems to recover (Dos Reis-Neto et al., 2019). However, saltworks that have lost hydrological functioning maintain high salinity levels in the sediment, resulting in highly stressful conditions for vegetation to spread and potentially providing opportunities for non-native plants to thrive (Dethier and Hacker, 2005; Chefaoui and Chozas, 2019).

A solar saltworks at Swartkops Estuary was abandoned in 2018. The hydrological function of this saltworks was actively managed by pumping water into and through the saltpans but since the abandonment, the saltpans have been left to dry. The saltworks used to host thousands of waterbirds annually and was an important component of the Swartkops Estuary-Redhouse and Chatty Saltpans IBA complex (Marnewick et al., 2015). However, without any water, the saltpans have little habitat value for waterbirds. The rehabilitation of the site as a waterbird sanctuary is currently being planned by reinstating a managed hydrological regime.

Before any restoration interventions commence, it is important to develop a comprehensive plan. A fundamental component of any restoration plan is a baseline inventory which is a comprehensive assessment of the site to be restored (Gann et al., 2019). A baseline inventory should include biotic and abiotic characteristics of the site, any threats to the ecosystem and aim to provide understanding of the nature of degradation (i.e. causes and effects) at the site (Hobbs and Norton, 1996; Howell et al., 2012; Gann et al., 2019). Taking inventory of the site allows for the setting of realistic restoration goals, provides insight into the required restoration actions, and allows for monitoring the results of any actions taken.

Little information is available on the ecology of the abandoned saltworks at Swartkops Estuary, particularly in its current desiccated state (see Chapter 3.2 for a collation of available information). While the cause of the degradation is clear (the loss of hydrologic functioning), the effects have not yet been studied. This study aimed to establish the baseline environmental conditions of the abandoned solar saltworks by collecting and collating data on the environmental (sediment and groundwater) and biotic characteristics (vegetation and waterbirds) at the site in its current degraded state. This information was collected to establish a basis for a rehabilitation plan and to provide insight into the expected ecological trajectory of the ecosystem if no rehabilitation action is taken.

4.2. Materials and methods

4.2.1. Sampling design

Environmental conditions (groundwater and sediment characteristics) and vegetation distribution were investigated across nine transects at the abandoned saltworks (Figure 4.1). Four transects were laid out in the large Redhouse saltpan and five at the smaller Bar None saltpans. All data were captured during a single sampling occasion on 1 September 2019. Sampling points were subjectively chosen to represent vegetated and unvegetated areas and areas of dead vegetation. The coordinates of the sampling points are presented in Appendix 2. At each sampling point, groundwater characteristics were captured, sediment samples collected, and vegetation composition and cover recorded. The elevation at each sampling point was derived from a digital terrain model (Appendix 3), which was obtained from a LiDAR survey (at 1 m spatial resolution) carried out in 2019 by Nelson Mandela Bay Municipality (NMBM).

Figure 4.1 Layout of sampling points along the nine transects at the abandoned saltworks. The boundaries of the saltpans were delineated based on the water level when they were filled to maximum capacity.

4.2.2. Groundwater characteristics

At each sampling point, a hole was manually augered to the groundwater table. The auger hole was left for approximately 30 minutes allowing the water level to stabilise. A metre stick was used to measure the depth to the top of the groundwater table. The salinity of the groundwater was recorded using a Hanna HI98194 multiparameter meter.

4.2.3. Sediment characteristics

Sediments were analysed for determination of moisture content, organic content, salinity, particle size and redox potential. These characteristics were determined for sediment at the surface of the saltworks (hereafter referred to as 'surface sediment') and at the top of the groundwater table (hereafter referred to as 'bottom sediment') at each of the sampling points. Sediment redox potential was recorded in situ using a Hanna HI918121 meter, while sediment samples were taken back to the laboratory for determination of the other characteristics. Sediment moisture content was determined using the method presented in Gardner (1965) (Appendix 4A), organic content was determined using the method presented in Briggs (1977)

(Appendix 4B), salinity was determined using the ‘saturated paste’ method (Barnard, 1990: Appendix 4C) and particle size was determined using the hydrometer method (Gee and Bauder, 1986: Appendix 4D).

4.2.4. Vegetation distribution

Vegetation composition and percentage cover were recorded at each sampling point, as well as at 20 m intervals along each transect using a 1 m^2 quadrat on the left and right side of the transect. Areas with vegetation cover < 10% were classified as ‘bare ground’; areas where vegetation cover was > 10% were classified as ‘vegetated’; and areas where the cover of dead plant material exceeded that of living plants were classified as “dead vegetation”. Additionally, the annual change in vegetation cover since the abandonment of the saltworks was analysed through a desktop assessment using satellite imagery from Google Earth (Google, 2020). These images were used to digitise vegetation cover as polygons in ArcMap (ESRI, 2018) and were dated 19 May 2018, 14 June 2019 and 8 June 2020, respectively.

4.2.5. Waterbird counts

Waterbird abundance and species diversity data were obtained from the Coordinated Waterbird Counts (CWAC) programme. Counts took place biannually – once in summer (January/February) and once in winter (June/July). The Redhouse salt pan has been registered to the programme since 1994 (CWAC, no date b) and the Bar None pans since 1995 (CWAC, no date a).

4.2.6. Data analysis

All statistical analyses were performed in R (R Core Team, 2020). Data were assessed for normality graphically as well as with the Shapiro-Wilks test, and for homogeneity of variance using the Brown-Forsythe test. As the assumptions of normality and homogeneity of variance were not met, the recorded variables for the groundwater and sediment characteristics were tested for significant differences with Kruskal-Wallis tests. For pairwise comparisons where significant differences were found, Nemenyi post hoc tests (“PMCMR” package: Pohlert, 2014) with Chi-square distribution to account for ties in pairwise groupings were used. A canonical correspondence analysis (CCA) was carried out on the recorded vegetation cover data and all measured environmental data (sediment and groundwater characteristics and elevation) using the “vegan” package (Oksanen et al., 2020). Additionally, Spearman’s rank correlation tests were carried out to test the correlation between all environmental factors and the cover of bare ground, dead vegetation and living vegetation, respectively.

4.3. Results

4.3.1. Groundwater characteristics

Mean depth to groundwater and groundwater salinity were similar among the saltpans but showed notable variation within each saltpan (Figure 4.2). Groundwater depths ranged from 0.18 to 1.48 m at the Redhouse saltpan and 0.28 to 2.2 m at the Bar None saltpans. The groundwater table was shallower at areas of lower elevation where water was observed to accumulate after rainfall. The groundwater at each saltpan was mostly hypersaline and ranged from 15 to 107 at the Redhouse saltpan and from 31 to 106 at the Bar None saltpans. The groundwater was generally more saline at sampling points where the water table was shallower, although no significant correlation between groundwater depth and salinity was found.

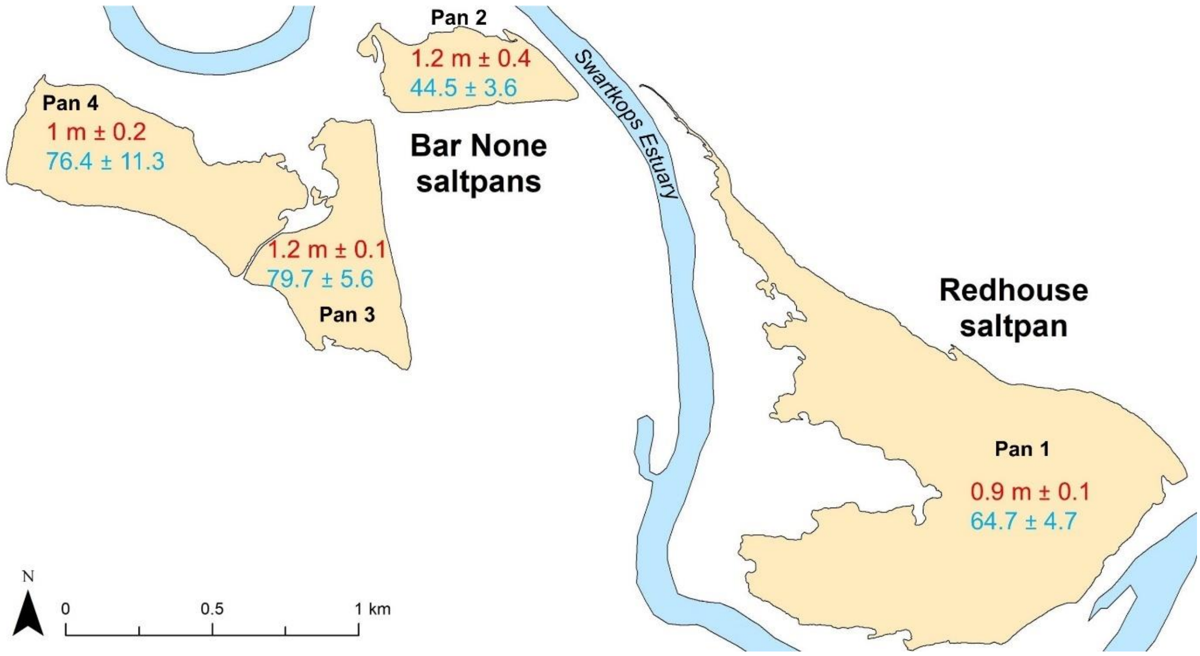

Figure 4.2 Mean depth to groundwater (red text) and groundwater salinity (blue text) (± SE) of the abandoned saltpans (n = 20 for Pan 1, n = 6 for Pan 2, n = 5 for Pan 3 and n = 6 for Pan 4).

4.3.2. Sediment characteristics

Sediment characteristics generally varied among the saltpans and between the surface and bottom sediments. Moisture content was somewhat similar between the four saltpans and was slightly higher at the top of the groundwater table than at the surface as could be expected (Figure 4.3A), while organic content (Figure 4.3B) was significantly higher in surface sediments than bottom sediments (Kruskal–Wallis $\chi^2 = 27.47$, df = 1, $p < 0.001$). The sediment was hypersaline in each saltpan and salinity was generally higher at the surface than at the top of the groundwater table (Figure 4.3C). Sediment salinity differed significantly among the saltpans (Kruskal–Wallis $\chi^2 = 11.42$, df = 3, $p = 0.01$), particularly between Pans 1 and 2 ($p = 0.01$), but not between surface and bottom sediments. The surface sediments were well oxygenated and had a significantly higher redox potential than the anoxic bottom sediments (Kruskal–Wallis $\chi^2 = 44.72$, df = 1, $p < 0.001$) (Figure 4.3D). Sediment at each of the saltpans was comprised mainly of clay and silt (Figure 4.3D). Surface sediments generally consisted of more clay (relative to silt) than bottom sediments. Bottom sediments also had a slightly higher sand content than surface sediments.

4.3.3. Vegetation distribution

Vegetation at the defunct saltworks was sparse and patchy (Figure 4.4, see also Figure 10.3). In the first year of abandonment, 30.8 ha (11.8% of the total area) of the saltpan area was vegetated. This decreased slightly to 29.75 ha by 2019 due to a loss of vegetation in Pan 1; however, vegetation cover visibly increased in all other saltpans. By 2020, vegetation cover had increased to a total area of 52.1 ha. This increase was most noticeable in Pans 1 to 3 while there was relatively little change in Pan 4.

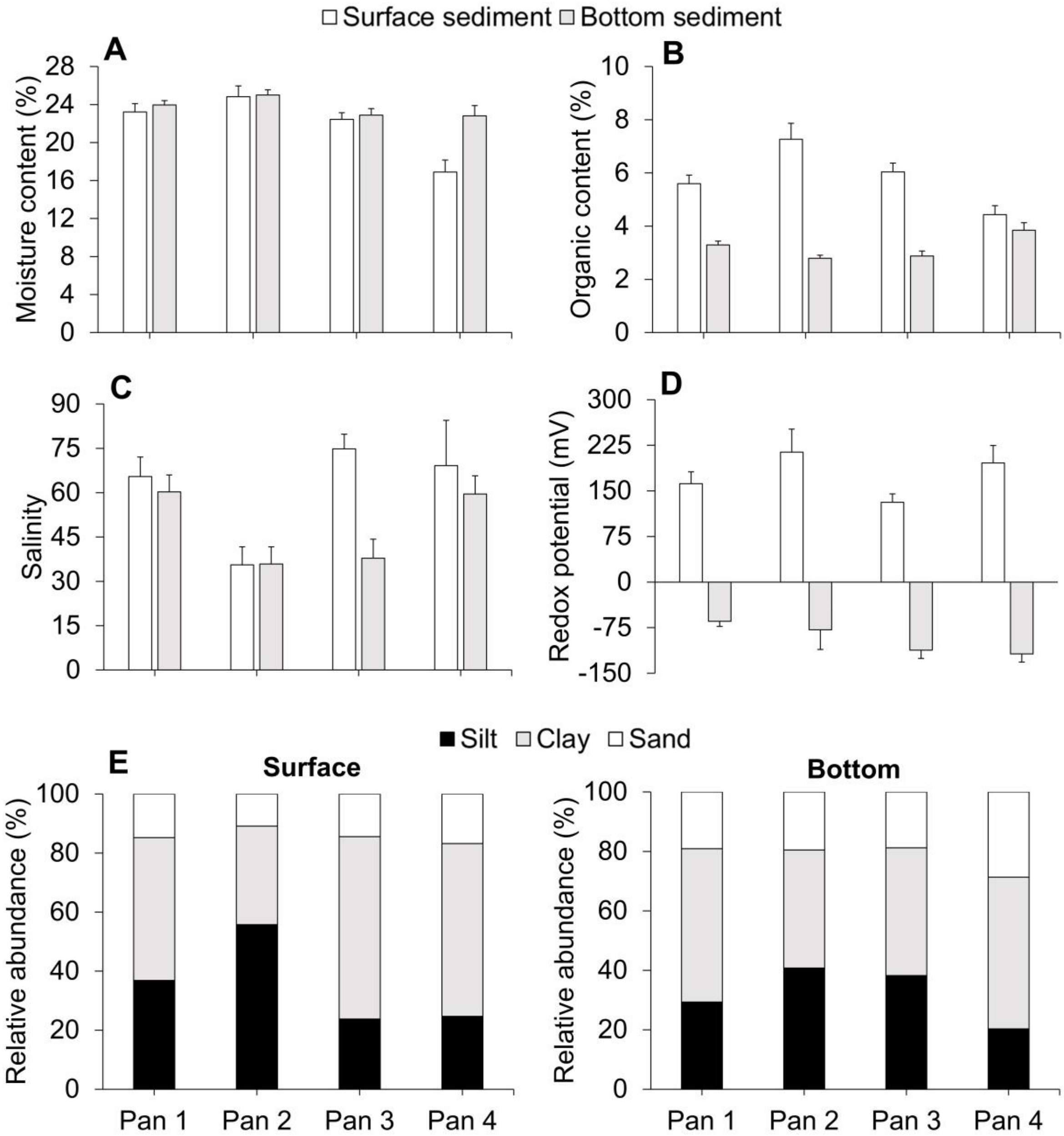

Figure 4.3 Sediment moisture content (A), organic content (B), salinity (C), redox potential (D) (mean ± SE) and particle size (E) at the abandoned saltworks (n = 60 for Pan 1, n = 18 for Pan 2, n = 15 for Pan 3 and n = 18 for Pan 4).

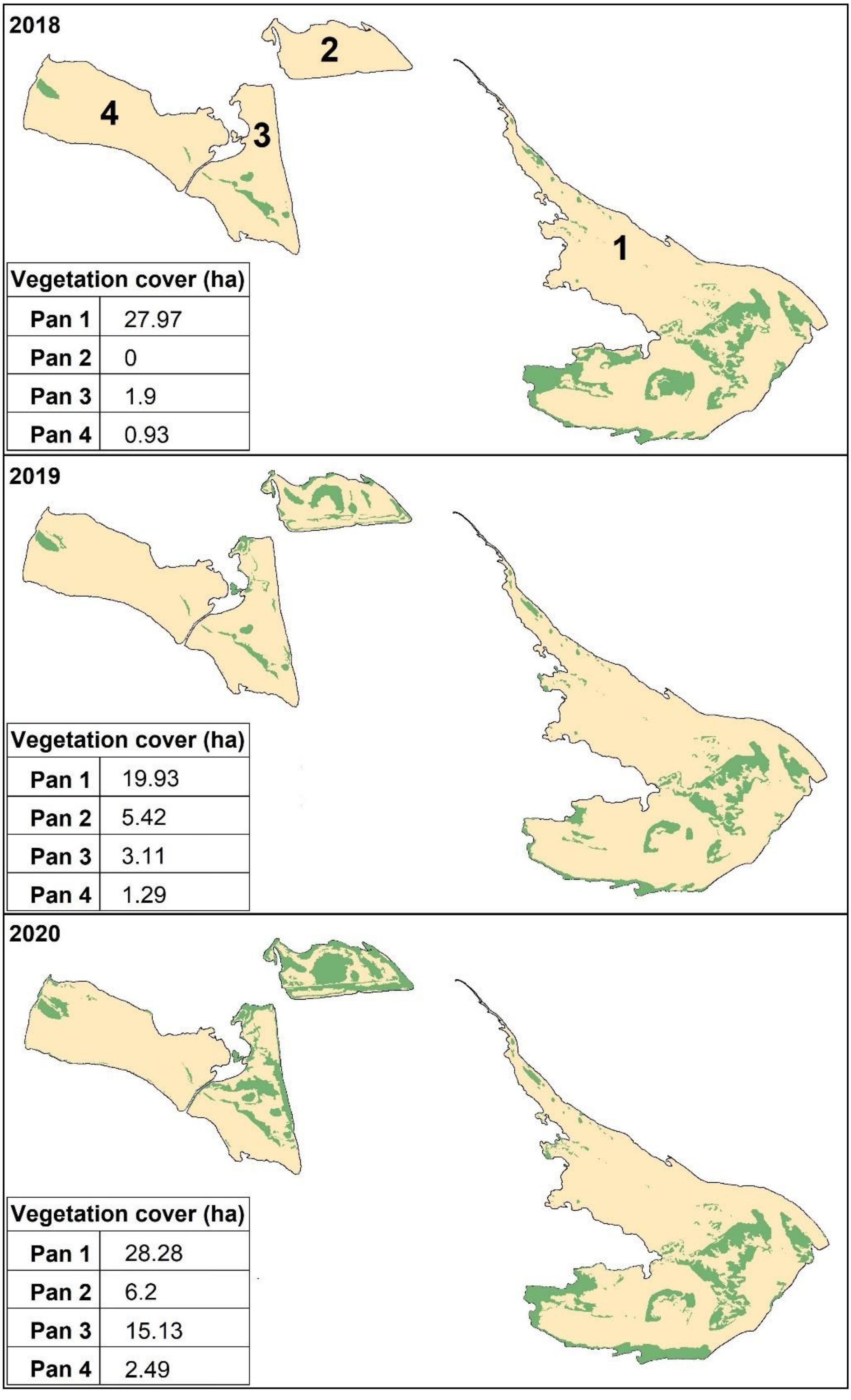

Figure 4.4 Annual change in vegetation cover at the defunct saltworks since abandonment.

Canonical correspondence analysis (CCA) captured just 29% of the total variability in vegetation distribution at the abandoned saltworks (Figure 4.5). The presence of most plant species (excluding *Suaeda fruticosa* and *Senecio* sp.) found at the site were associated with an increase in elevation, depth to groundwater and silt content in the sediment. Unvegetated areas were associated with an increase in the salinity of groundwater and sediment as well as an increase in sand and clay content in the sediment. Areas that contained dead vegetation were associated with an increase in sediment redox potential and a decrease in groundwater salinity, sediment salinity and sediment moisture content, and organic content. However, the only significant environmental factors were elevation ($p < 0.01$), groundwater salinity ($p < 0.01$), sediment salinity ($p = 0.04$) and sediment organic content ($p = 0.03$).

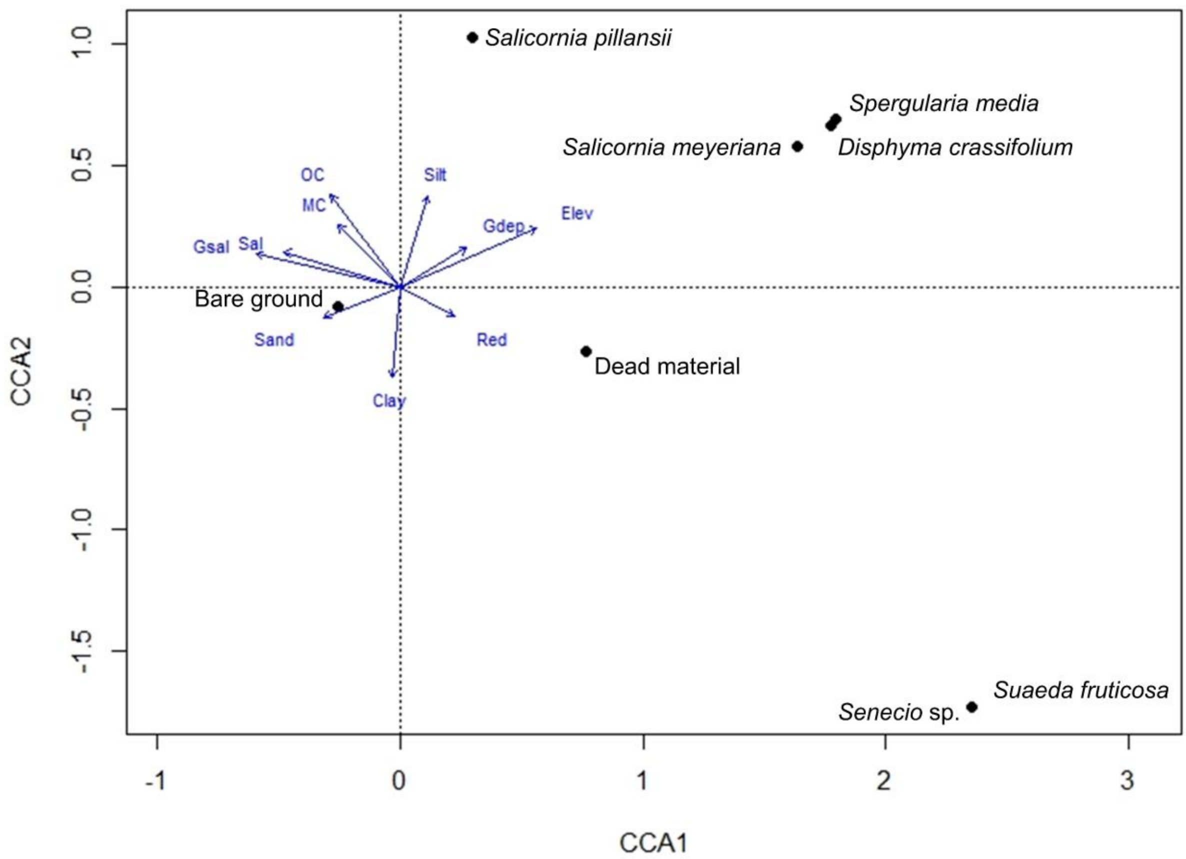

Figure 4.5 Canonical correspondence analysis (CCA) plot of plant species at the abandoned saltworks based on abundance. Blue arrows indicate the constraining environmental variables. Total inertia = 2.14. Eigenvalues for constrained axes: CA1 = 0.27, CA2 = 0.16, CA3 = 0.07, CA4 = 0.06, CA5 = 0.04, CA6 = 0.01.

The distribution of living vegetation was most strongly correlated with elevation, but also with the salinity of the sediment and groundwater (Table 4.1). However, elevation was correlated with both sediment salinity (r = -0.36, p = 0.02) and groundwater salinity (r = -0.4, p = 0.01). Large areas of the saltpans, excluding the elevated hummocks, were almost permanently inundated with brine. Consequently, salts accumulated in the sediment of lower-lying areas within the saltpans. These lower areas also had a significantly shallower groundwater table (r = 0.43, p < 0.01) and thus the salts from the sediment leached more easily into the groundwater. It is most probable that the primary factor limiting the distribution of living vegetation was sediment salinity, and not elevation per se.

Table 4.1 Spearman's rank correlations between environmental factors and the cover of bare ground, dead material, and living vegetation at the abandoned saltworks. Bold font indicates significant correlations.

		Bare ground		**Dead material**		**Living vegetation**	
		r_s	**p**	r_s	**p**	r_s	**p**
Surface sediment	**Moisture content (%)**	0.18	0.28	-0.14	0.39	-0.11	0.51
	Organic content (%)	0.07	0.68	-0.17	0.31	0.09	0.60
	Salinity	0.42	**< 0.001**	-0.32	**0.04**	-0.41	**0.01**
	Redox potential	-0.14	0.41	0.01	0.95	0.19	0.25
	% silt	-0.17	0.31	-0.02	0.90	0.30	0.07
	% clay	0.13	0.44	0.04	0.82	-0.22	0.18
	% sand	0.18	0.28	0.00	0.98	-0.36	**0.03**
Groundwater	**Depth to groundwater**	-0.24	0.14	0.21	0.20	0.24	0.14
	Salinity	0.46	**< 0.01**	-0.40	**0.01**	-0.46	**< 0.01**
Elevation		-0.59	**< 0.001**	0.41	**0.01**	0.64	**< 0.001**

4.3.4. Waterbird counts

Prior to 2018, while the saltworks was operational, the site regularly hosted thousands of waterbirds each year (Figure 4.6). Annual waterbird abundance was generally higher at the Redhouse saltpan (which had a maximum count of 7030 birds in the summer of 2005) than the Bar None saltpans (which had a maximum count of 2783 birds in the summer of 2007). A total of 60 species have been recorded at each site. Furthermore, waterbird abundance and species diversity were generally higher in summer than winter.

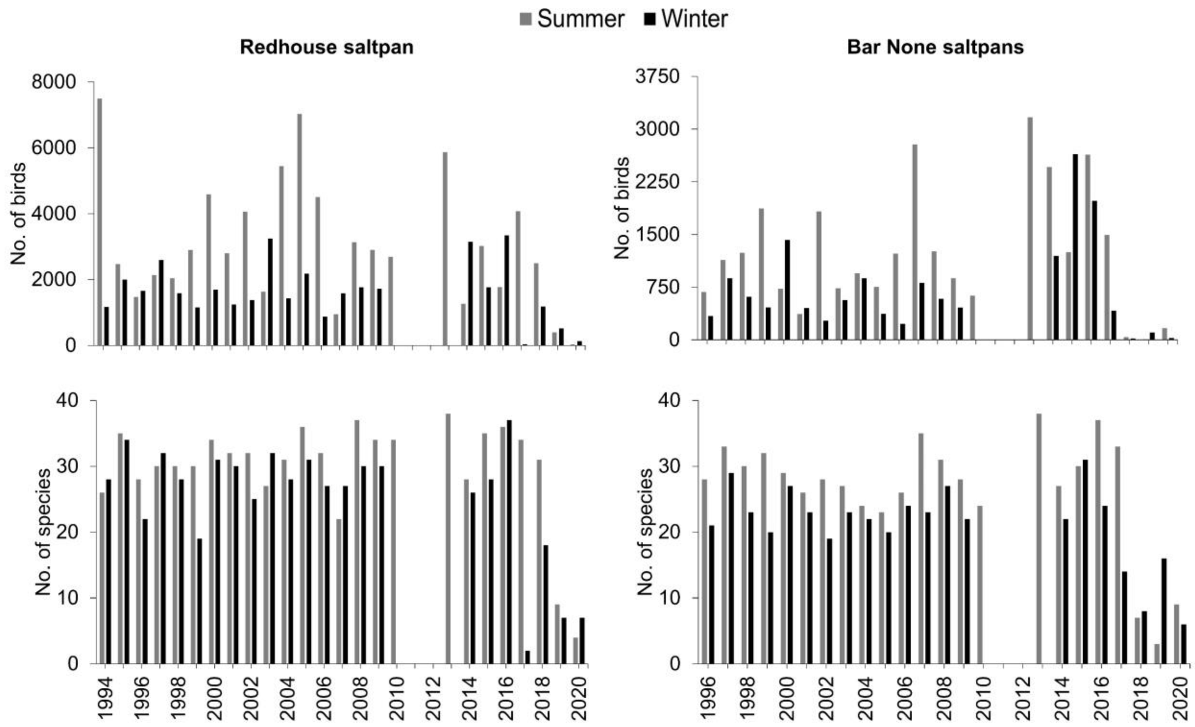

Figure 4.6 Biannual counts of waterbird abundance and species richness at the abandoned saltworks. Data for 2010 to 2013 were not available.

4.4. Discussion

The abandonment of the saltworks has left behind large expanses of desiccated hypersaline sediment (Figure 4.7A). The distribution of vegetation at the saltpans was sparse and patchy. The dry and hypersaline conditions at the Redhouse and Bar None saltpans have resulted in highly monospecific vegetation with *Salicornia pillansii* as the dominant species (Figure 4.7B). The composition of vegetation at abandoned saltworks is primarily driven by sediment salinity and the hydrologic regime, which are a result of the previous management programme (i.e. the salt production process) of the site (Bouzillé et al., 2001; González-Alcaraz et al., 2014; Chefaoui and Chozas, 2019). *Salicornia pillansii* is typically the dominant species in many supratidal salt marshes along the South African coastline and is well adapted to the challenging environment presented by the abandoned saltworks as it can extend its roots to the water table and is able to tolerate hypersaline groundwater (Bornman et al., 2004; Steffen et al., 2010; Veldkornet et al., 2016). Various other halophytic salt marsh species such as *Suaeda fruticosa*, *Bassia diffusa* and *Disphyma crassifolium* also occurred at the abandoned saltworks but in far lower abundance (see Appendix 5 for a comprehensive plant species list and species abundance along the transects). It appeared that these other species had a higher relative abundance on the periphery of the saltpans and on the most elevated areas within the saltpans where the sediment salinity was lower. Overall, species composition was similar to that of the supratidal and floodplain salt marsh and the salt marsh – terrestrial ecotone habitats of the surrounding Swartkops Estuary and along the south-eastern coast of South Africa (Schmidt, 2013; Veldkornet, 2016).

Overall, vegetation cover has increased since the saltworks has been abandoned. Vegetation cover was greatest on the elevated hummocks (Figure 4.7C), where plants have persisted since the saltworks was operational. In the lower lying areas, patches of new growth were observed, but upon later site visits, many such patches appeared to be dying (Figure 4.7D). Dieback was likely a consequence of salts concentrating in the surface sediments due the capillary rise of the hypersaline groundwater. Periodic flooding is necessary in supratidal salt marshes to flush excess salts from the sediment (Jolly et al., 1993; Neill, 1993). Occasional floods have been reported to occur at the Redhouse saltpan (Baird et al., 1986) but the Bar None saltpans lie at a markedly higher elevation and will likely not be flushed by flood events. Another factor limiting the distribution of vegetation is the accumulation of rainwater within depressions in the saltpans (Figure 4.7E). This results in sediment in some areas of the saltpans being waterlogged for prolonged periods, preventing the establishment of supratidal salt marsh in these areas (González-Alcaraz et al., 2014).

An increase in supratidal salt marsh at the abandoned saltworks would be a beneficial offset to the loss of this vegetation from Swartkops Estuary. Approximately 67% has already been lost from the estuary due to development (Adams, 2020). If vegetation could cover the entire area of the saltworks, the estuary could gain back approximately 160 ha of supratidal salt marsh (nearly 50% of the current extent). If the hydrology of the saltworks is not reinstated, it is unlikely that vegetation will cover the entire expanse of the saltpans due to the high sediment salinity, which will persist unless flushing events take place. However, if the saltpans are flooded once again, supratidal salt marsh will also not spread as supratidal species cannot tolerate prolonged inundation (González-Alcaraz et al., 2014). Nevertheless, supratidal salt marsh can be expected to persist on the hummocks in the saltpans with *S. pillansii* as the dominant species.

One concern that was observed in the plant communities was the presence of the invasive sponge-fruit saltbush (*Atriplex lindleyi* subsp. *inflata*) (Figure 4.7F). The stressful conditions at disturbed environments like abandoned saltworks often facilitate invasion by exotic plants (Dethier and Hacker, 2005; Almeida et al., 2014; Chefaoui and Chozas, 2019). *Atriplex lindleyi* subsp. *inflata* is well adapted to persist in such environments through fruit dimorphism – it produces both dormant and non-dormant fruits ensuring swift germination and a lasting seedbank (Atia et al., 2011). Furthermore, *Atriplex* species are highly tolerant to salinity and drought (more so than most other species found at the saltworks) (Glenn et al., 2012; Veldkornet, 2016) and will likely spread throughout the abandoned saltpans. This specific species has been declared an invader category 3 in the Conservation of Agriculture Resources Act (Act 43 of 1983, as amended in 2001) and thus does not have to be legally removed but reasonable measures should be taken to prevent it from spreading further.

The most concerning change at the saltworks since abandonment is the decline of the avifaunal communities. The once abundant birdlife has all but disappeared since the saltpans have dried up. This includes the loss of several species that are listed in the South African Red Data Book and those that trigger the criteria for an IBA and/or a Ramsar designation (refer to Appendix 6 for a comprehensive species list). Most waterbirds that were recorded at the saltworks used the site for feeding, with the majority of species (including greater flamingos and various waders) feeding on aquatic invertebrates but piscovorous species (such as cormorants, egrets and herons) as well as herbivorous species (such as ducks and geese) also fed at the saltpans (Martin and Randall, 1987). The saltpans were very important feeding habitats over summer when large influxes of Palearctic migrant species would occur such as curlew sandpipers and little stints. Decreasing water levels in summer exposed wet mudflats that were utilised by these migrant wader species (P. Martin, pers. comm., 2020). Waterbirds respond strongly to changes in water levels in saltpans. Decreasing water levels (especially in systems containing an abundance of benthic macrofauna) increases the foraging value of saltpans, resulting in higher densities of waterbirds feeding there (Velasquez, 1992). Wetland habitats with high foraging value are especially important to migratory waterbirds due to their high energetic needs. However, the saltworks no longer provides the feeding opportunities it once did since it has become desiccated.

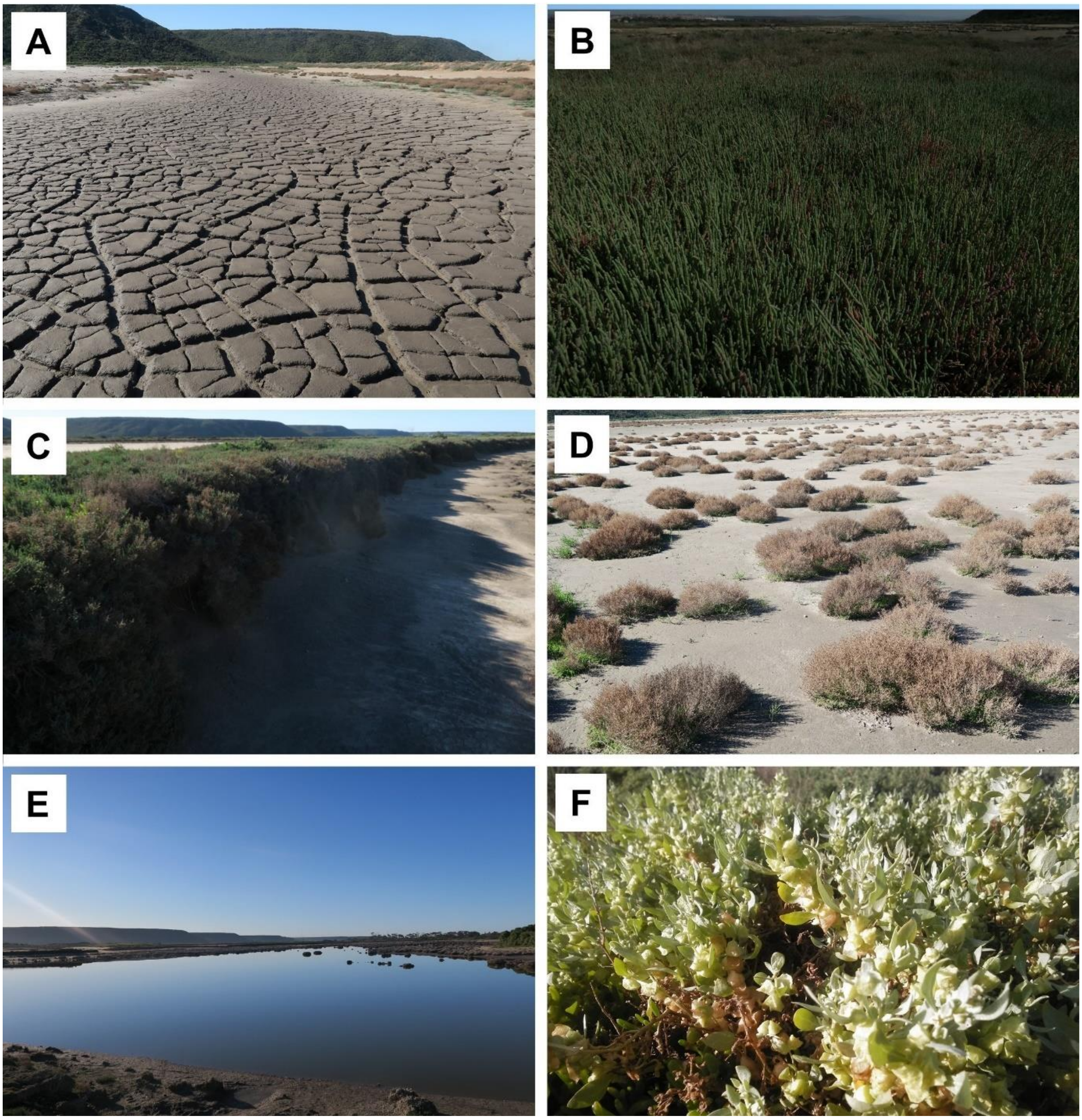

Figure 4.7 The abandonment of the saltworks has left behind vast expanses of dry hypersaline sediment (A). The salt marsh vegetation that is present is highly monospecific with *Salicornia pillansii* being the dominant species (B). Vegetation within the saltpans is mostly limited to the elevated hummocks (C). Vegetation has established in the lower lying areas, but dieback has been observed (D). Rainfall accumulates in depressions within the saltpans (E). The invasive sponge-fruit saltbush *Atriplex lindleyi* subsp. *inflata* is present at the saltworks (F).

The high water levels at the saltworks were also important during winter when various resident waterbird species would breed on islands within the saltpans. The islands provided areas protected from human egg poachers and small mammalian predators (P. Martin, pers. comm., 2020). The Redhouse saltpan in particular was considered one of the most important mainland breeding sites in the Eastern Cape Province for shorebirds and hosted some of the largest breeding colonies of Caspian terns, whitebreasted cormorants and kelp gulls in southern Africa (Martin and Randall, 1987). Additionally, five species have been recorded as regularly breeding at the Bar None saltpans (Crawford et al., 2009). Various other species also made use of these islands and the periphery walls of the saltpans for roosting during high tide in the adjacent Swartkops Estuary. Just one colony of sacred ibis has been recorded roosting at the Redhouse saltpan since the site's abandonment, otherwise no colonies have returned for roosting or breeding.

The saltwork's habitat value to waterbirds has been lost due its abandonment. Without any hydrological function, waterbirds will not return for feeding or breeding. Furthermore, without any hydrological functioning, sediment at the saltworks will remain hypersaline, likely limiting the spread of the native salt marsh vegetation and potentially facilitating the proliferation of the invasive *A. lindleyi* subsp. *inflata*. Recreating wetlands at the saltworks will greatly increase the ecological value of the site relative to the current depauperate state. The reinstatement of a managed hydrological regime is therefore strongly recommended.

5. Characterisation and comparison of the potential water sources for restoring the abandoned saltworks

5.1. Introduction

In order to rehabilitate the abandoned saltworks at Swartkops Estuary as a waterbird habitat, it is necessary to restore the hydrology of the system. While the saltworks was operational, water was pumped in from the estuary via a pumphouse. However, it is not feasible to use the pumphouse once again to fill the saltpans, as it has been continuously targeted by vandalism and theft. An alternative option has been proposed to mount a pump (the same pump that was originally used at the saltworks) on a pontoon which would float in the estuary (F. Collier, pers. comm., 2020). This allows for the pump to be stored safely while not in operation. However, this option would require high running costs for fuel and maintaining the pontoon and pump. Another option for restoring the hydrology for the saltworks is diverting stormwater into the saltpans. The Motherwell stormwater canal (MWC) lies near the abandoned saltworks and diverting the canal's outflow may save on costs of this rehabilitation project in the long-term.

Swartkops Estuary is heavily urbanised and receives anthropogenic inputs from various sources. The estuary and the lower catchment of the Swartkops River receive high loads of point source pollution from three wastewater treatment works (WWTWs), two stormwater canals and the Chatty River, which flows through an informal residential area. Furthermore, diffuse pollution enters the estuary from the densely populated residential areas and a variety of industrial activities that surround it. These various sources of pollution (but particularly the point sources) have resulted in long-term anthropogenic nutrient loading of the estuary. Swartkops Estuary has been classified as eutrophic in terms of dissolved inorganic phosphorus (DIP, > 0.1 mg P l^{-1}), while dissolved inorganic nitrogen (DIN) levels have been categorised as mesotrophic (≥ 0.1 mg N l^{-1}) to eutrophic (> 1 mg N l^{-1}) (Lemley et al., 2015; Adams et al., 2019b). The prolonged degradation of the water quality in the estuary has had negative ecological (such as phytoplankton blooms) and socio-economic consequences such as the loss of a popular swimming event, the Redhouse River Mile, from the estuary (Adams et al., 2019b).

The MWC, which discharges urban stormwater directly into the middle reaches of Swartkops Estuary, has been identified as a major source of pollution (particularly DIN) to the estuary (Adams et al., 2019b). The MWC receives stormwater from the large Motherwell residential area and is often subjected to raw sewage spills and illegal dumping. In an attempt to alleviate the high nutrient inputs from the MWC, the Motherwell Canal Artificial Wetland System (referred to as the MWAW in this thesis) was constructed in 2010 (SRK Consulting (Pty) Ltd, 2010). The MWAW was a pilot project designed to filter just 20% of the stormwater runoff from MWC. However, all of the stormwater is currently being diverted through the wetland, which is not large enough to effectively remove nutrients from the stormwater prior to discharge into Swartkops Estuary (Stroebel, 2019; Lakane, 2020). Diverting the stormwater from the MWAW to the abandoned saltworks provides the opportunity to further treat the stormwater, relieving the estuary of a major source of nutrient inputs, while also rehabilitating the saltpans as a waterbird habitat.

Planning is essential to the success of any ecological restoration projects and is crucial to avoid any unintended consequences that may result from restoration interventions (Gann et al., 2019). Investigating the water quality of both potential sources is fundamental to developing a rehabilitation plan for the saltworks. This study aimed to characterise and compare the water quality of the two potential water sources (Swartkops Estuary and the MWAW). The most recent available water quality data for Swartkops Estuary and the MWAW were collated and analysed. Furthermore, the practicality of the two rehabilitation options (i.e. the different water sources) was assessed in terms of the quantity of water necessary to restore the hydrology of the saltworks and the how to execute the actions needed to do so.

5.2. Materials and methods

5.2.1. Hydrological considerations

To quantify the amount of water necessary to fill the saltpans, the volume of each saltpan was calculated using the Surface Volume tool (in the 3D Analyst toolbox) in ArcMap (ESRI, 2018). The reference plane to calculate the volume (i.e. the height of the water level) was visually determined through assessing a contour map superimposed over Google Earth satellite imagery (Google, 2020) of the saltpans when they were completely filled with water (28 March 2016). The predicted water levels were estimated by calculating the minimum and mean elevation (derived from a digital terrain model supplied by NMBM in 2019: Appendix 3) within each saltpan and subtracting these values from the height of the contour line that historically represented the water level at each saltpan. All analyses were carried out under the Africa Albers Equal Area conic projection.

The quantity of water discharged by the MWAW was determined by measuring the head of water above the V notch weir at the inlet of the wetland, as there is no such weir at the outlet. It was assumed that the discharge at the outlet would be equivalent to the discharge from the MWC into the MWAW. The measurements of the head of water at the weir were applied to the Kindsvater-Shen equation (USBR, 2001: Appendix 4E) to calculate the discharge. Rainfall data were obtained for the Port Elizabeth International Airport weather station from the National Oceanic and Atmospheric Administration (NOAA) Global Surface Summary of the Day dataset.

5.2.2. Water quality

All available water quality data were collated from several sampling trips for six sites throughout Swartkops Estuary and two sites at the MWAW. The six sites at Swartkops Estuary covered the estuary from the mouth to the head in order to inform the selection of a site to pump estuary water into the saltpans. The sites were named as follows (in order from the mouth to the head of the estuary): Settlers Bridge: SB; Swartkops Village: SV; Brickfields: BF; Redhouse: RH; Bar None: BN; and Perseverance: PS. All the procedures described below were carried out at the surface, middle and bottom of the water column of the estuary, except for PS due to the site's shallowness. The two sites sampled at the MWAW were the inlet, where stormwater enters the wetland from the MWC, and the outlet of the wetland from which the stormwater discharges into the estuary. These two sites were sampled to determine the efficiency at which the MWAW is currently removing nutrients from the stormwater. The collated water quality data included physico-chemical characteristics and inorganic nutrient concentrations for both Swartkops Estuary and the MWAW; while phytoplankton biomass, cell abundance and community composition were only recorded for the estuary. The sites sampled at Swartkops Estuary and the MWAW are shown in Figure 5.1 and details on the sampling occasions are presented in Appendix 7.

Physico-chemical characteristics (salinity, dissolved oxygen, pH, and temperature) were recorded in situ using a YSI ProDSS Multiparameter Water Quality Meter. Inorganic nutrient concentrations were determined by collecting water samples from Swartkops Estuary and the MWAW for analysis in the laboratory. For the estuary samples, dissolved inorganic phosphorus (DIP = PO_4^{3-}-P), ammonium (NH_4^+-N) and total oxidised nitrogen (NO_x-N = NO_2^- + NO_3^-) concentrations were determined using a SEAL AutoAnalyzer 3 HR (SEAL Analytical, Inc.). For the MWAW samples, DIP and NH_4^+ concentrations were determined using standard spectrophotometric methods (oxidation method: Parsons et al., 1984) and NO_x concentrations were determined using the reduced copper cadmium method (Bate and Heelas, 1975: Appendix 4F). Inorganic N:P molar ratios were calculated using the concomitant DIN and DIP concentrations. The estuary samples were analysed for phytoplankton biomass (determined as chlorophyll-*a* concentration) using spectrophotometric methods as per Nusch (1980) (Appendix 4G) and phytoplankton communities were enumerated and identified to the class level (Appendix 4H).

The salinity levels that can be expected at the saltpans once refilled with either stormwater or estuary water were estimated using a method adapted from Bate and Taylor (2008). Dry surface sediment (top 20 cm) was collected from the saltpans (see Appendix 7 for the location of sediment samples) using an auger with a diameter of 5.5 cm, which therefore covers a surface area of 23 cm^2. Each sediment sample thus had a volume of 0.4752 l. Sediment salinity was determined using the 'saturated paste' method (Barnard, 1990: Appendix 4C). These salinity values (in psu; equivalent to ppt) were averaged for each saltpan and converted to kg l^{-1}. The mean salinity values (kg l^{-1}) were multiplied by the volume of the auger (0.4752 l) to determine the mean amount of salt (in kg) per sample. Recalling that the samples each represented an area of 23 cm^2, the mean amount of salt per sample (kg) for each saltpan was multiplied to 1 m^2 to determine the amount of salt in kg m^{-2}. The amount of salt (kg) per m^2 was subsequently multiplied by the area of the respective saltpan (in m^2) to determine the total sediment salt load (in kg) in the top 20 cm of each saltpan. The total sediment salt load (kg) for each saltpan was then divided by the volume of water each saltpan is to be filled with (in l – these volumes are provided in Table 5.1 in m^3), providing a salinity value in kg l^{-1}, which was converted to psu. This, however, does not account for the salinity of the water prior to the water entering the saltpans. As such, the salinity of the water to be used to fill the saltpans (refer to Tables 5.3 and 5.4) was added to the last calculated values (in psu) to provide an estimate for the salinity that can be expected in the rehabilitated saltpans.

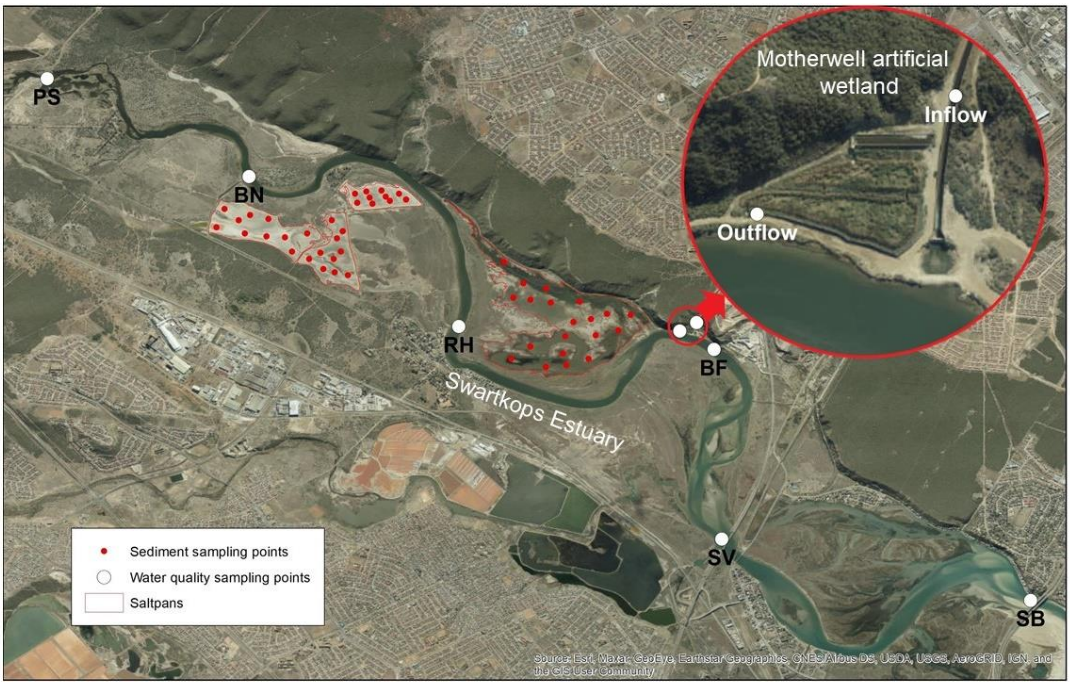

Figure 5.1 Location of sampling points for water quality parameters at the Motherwell artificial wetland (MWAW) and Swartkops Estuary and sediment samples from the saltpans for Chapter 5.

5.2.3. Data analysis

All statistical analyses were performed in R (R Core Team, 2020). The collated data were graphically tested for normality as well as with the Shapiro-Wilks test, and assessed for homogeneity of variance using the Brown-Forsythe test. For the MWAW water quality data, Wilcoxon signed-rank tests were carried out to determine whether there were significant differences in water quality characteristics before and after the stormwater had passed through the artificial wetland (i.e. at the inlet and the outlet). Additionally, one-way ANOVAs were performed to test if there were significant differences between sampling occasions. One-way ANOVAs were also carried out for the Swartkops Estuary data to detect significant differences between sampling occasions, sites and depths, respectively. If significant differences were found, post hoc Tukey's range tests were also carried out to determine where means differed significantly. Linear regressions were carried out for salinity and DIN and DIP concentrations with distance from mouth as the predictor variable. The assumptions of residual normality and homogeneity ANOVAs and linear regressions were assessed graphically, and all assumptions were met.

5.3. Results

5.3.1. Hydrological considerations

As mentioned previously, the pumphouses used to fill the saltpans with estuary water are no longer in operation and it would not be feasible to use them in the rehabilitation project due to the threat of vandalism and theft. However, it will still be possible to fill the saltpans with estuary water using a pump mounted on a pontoon that floats in the estuary. One consequence of the pumphouses not being functional, however, is that water can no longer be pumped from the Redhouse saltpan into the Bar None saltpans as was under the previous management regime. These two sections of the saltworks will essentially be disconnected and will have to be managed separately.

The use of a pontoon-mounted pump to fill the saltpans with estuary water allows for water to be pumped from various locations within the estuary. Three sites have been identified from which water can be pumped into the saltpans (Sites 1, 2 and 3 in Figure 5.2). Site 1 lies 11.6 km from the mouth of the estuary and is the site from which estuary water was previously pumped into the saltworks. Site 1 is the only site from which estuary water can be pumped into the Bar None saltpans. Water can be pumped from this site into Pan 2 (which has a mean elevation of 3 MSL) and allowed to flow through existing culverts into the lower lying Pans 3 and 4 which have mean elevations of 2.35 MSL and 2.22 MSL, respectively. This is the same approach taken to fill the Bar None saltpans under the previous management regime. The Redhouse saltpan, however, can be filled from any three of the sites. While the saltworks was operational, water was pumped from Site 1 into a channel that fed water into the saltpan. Sites 2 and 3 lie 9.7 km and 7.6 km from the estuary mouth, respectively, and were selected due to the close proximity between the estuary channel and the Redhouse saltpan at these points.

While estuary water can be used to fill all of the saltpans, stormwater could only feasibly be used to fill the Redhouse saltpan. The Redhouse saltpan lies just 400 m from the outlet of the MWAW from which the stormwater would be diverted, while the Bar None saltpans lie nearly 3 km away on the opposite side of the estuary and at a higher elevation. As no land-based pump will be used in this rehabilitation project, it will not be practical to divert stormwater to the Bar None saltpans. For the Redhouse saltpan, however, a pipe could be built to divert the stormwater into the north-eastern corner of the saltpan. This point lies approximately 1 m lower than the outlet of the MWAW, allowing the stormwater to be gravity-fed into the saltpan.

Figure 5.2 The abandoned saltworks with the proposed water sources for filling the saltpans indicated. The Motherwell artificial wetland (MWAW) is represented by a green polygon, while the proposed sites to pump estuary water are indicated with the red numbers. The yellow arrow indicates the orientation of the pipe necessary to fill the Redhouse saltpan with stormwater and the red arrows indicate how estuary water can be pumped into, and flow through, the saltpans.

To fill all of the saltpans to maximum capacity, a total water volume of 1016491.17 m^3 is necessary (Table 5.1). The Redhouse saltpan is notably larger than all of the Bar None saltpans and has a total storage capacity of approximately 758100 m^3, nearly three times that of all of the Bar None saltpans together. Furthermore, the water column at the Redhouse saltpan has a greater mean depth when filled to maximum capacity than each of the Bar None saltpans (see also Figure 7.1).

Table 5.1 Estimated water surface area, volume, and depth when the saltpans are filled to maximum capacity.

Saltpan	Water surface area (ha)	Water volume (m^3)	Water depth (m)	
			Mean	Max
1	92.56	758100.13	0.9	1.3
2	14.10	52248.41	0.37	0.79
3	18.99	79683.41	0.35	0.85
4	29.65	126459.23	0.38	0.7

The rate of stormwater discharge from the artificial wetland was temporally stochastic (Figure 5.3). Discharge was positively correlated to rainfall, although the relationship was neither strong nor significant ($r = 0.57$, $p = 0.08$). The mean discharge of stormwater exiting the MWAW was 0.073 $m^3\ s^{-1}$ which is equivalent to approximately 264.3 $m^3\ hr^{-1}$or 6442.2 $m^3\ day^{-1}$. From the mean rate of discharge, it can be estimated that it would take approximately 120 days to fill the Redhouse saltpan with stormwater. However, discharge is not constant and filling the saltpan can be expected to take between 93 to 162 days if the maximum and minimum discharge values recorded in this study are considered, respectively. Comparatively, it would take one week of continuous pumping (i.e. 168 hrs) to fill the Redhouse saltpan with the pump that was previously used at the saltworks – it is being planned to use this same pump for the rehabilitation project (F. Collier, pers. comm., 2020). At this pumping rate, it would take approximately 67.4 hrs (nearly three days of continuous pumping) to fill all three of the Bar None saltpans. It should be noted that these estimates do not account for loss through evaporation or gain through precipitation.

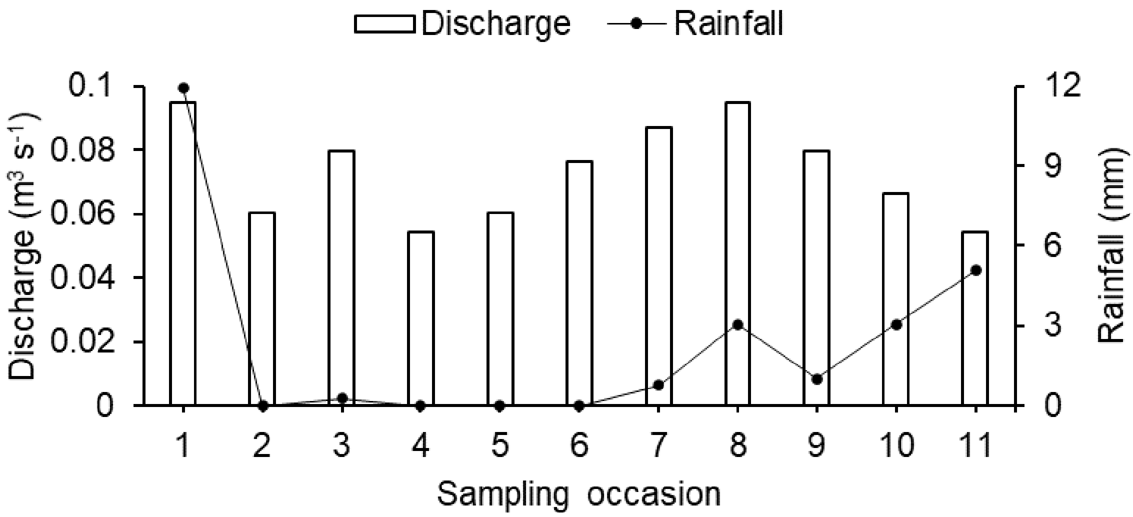

Figure 5.3 Discharge of stormwater into the Motherwell artificial wetland (MWAW). Rainfall on the day of sampling is indicated on the secondary axis. Details of sampling occasions are provided in Appendix 7.

5.3.2. Water quality

5.3.2.1. Water quality at Swartkops Estuary

The physico-chemical characteristics of the water column at Swartkops Estuary varied among the sites sampled (Table 5.2). Dissolved oxygen (DO) varied significantly between sites ($F = 4.37$, $p = 0.001$) and sampling trips ($F = 4.36$, $p = 0.001$), and vertical differences in the water column were recorded at each site on each sampling occasion ($F = 4.17$, $p = 0.02$). These vertical differences were generally more pronounced at sites further upstream. Hypoxic conditions (DO < 3 mg l^{-1}) were occasionally recorded at the upstream sites (SV, BF, RH and BN), particularly in the bottom waters. pH did not display any spatial variability, but did vary temporally ($F = 33.15$, $p = < 0.001$) and was significantly positively correlated to chlorophyll-*a* concentrations in the water column ($r = 0.56$, $p < 0.001$), suggesting that phytoplankton productivity influenced pH along the estuary. pH was also notably higher at the site furthest upstream (PS) than throughout the rest of the estuary. The high pH and DO values and standard errors at site PS are indicative of eutrophic conditions and phytoplankton blooms (refer to Figures 5.5 and 5.6). Water temperatures increased upstream from the mouth of the estuary. The lowest mean temperatures were recorded in July 2019 (14.2°C), while the highest were recorded in February 2020 (27°C). Vertical differences in temperature were more apparent at sites further upstream, particularly during summer months.

Table 5.2 Mean physico-chemical parameters (n = 18) measured at the various sites sampled at Swartkops Estuary.

Site	DO (mg l^{-1})		pH		Temperature (°C)	
	Mean	**± SE**	**Mean**	**± SE**	**Mean**	**± SE**
SB	7.19	0.18	8.1	0.05	18.27	0.65
SV	5.77	0.35	8	0.05	18.34	0.97
BF	5.73	0.56	7.96	0.07	18.79	1.20
RH	5.41	0.57	7.96	0.08	19.06	1.22
BN	6.28	1.25	8.05	0.12	19.74	1.2
PS	12.25	3.22	8.47	0.27	23.03	3.01

A distinct longitudinal salinity gradient persisted in Swartkops Estuary ($F = 82.31$, $p = < 0.001$), ranging from salinity typical of marine environments at the mouth (SB) to near fresh at the head of the estuary (PS) (Figure 5.4). This is typical of permanently open estuaries that receive persistent riverine input. The highest mean salinity levels were recorded in summer months (November 2019 and February 2020), while the lowest were recorded in winter months (July 2019 and June 2020). A vertical salinity gradient was also recorded ($F = 5.32$, $p < 0.01$), particularly between surface and bottom waters ($p < 0.01$) but also between surface and middle waters ($p = 0.04$). The vertical salinity gradient was more pronounced in winter months, especially at sites further upstream (RH and BN).

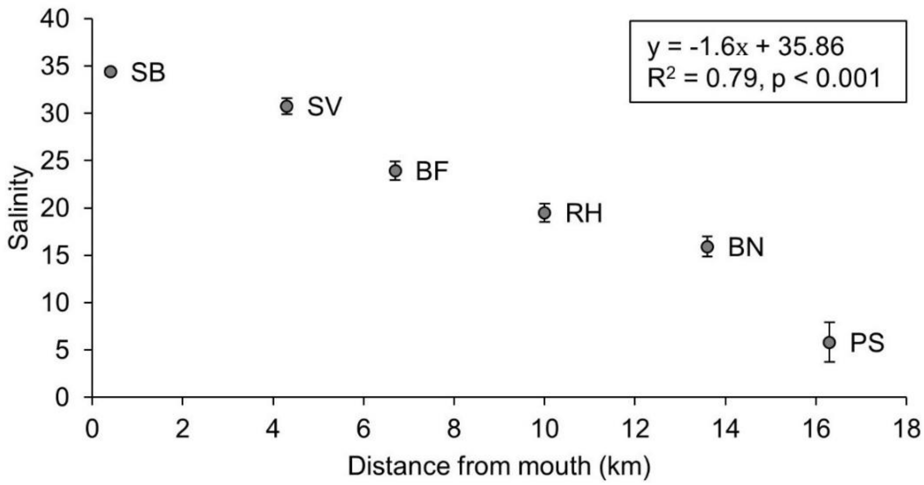

Figure 5.4 Salinity at the six sites sampled in Swartkops Estuary (mean ± SE, n = 18). The relationship of salinity with distance from the estuary mouth is shown in the textbox.

Both DIN and DIP increased significantly from the mouth to the head of the estuary ($F = 10.93$, $p = < 0.001$ and 66.91, $p = < 0.001$, respectively) (Figure 5.5). Furthermore, there were significant vertical differences in the water column for both DIN ($F = 3.7$, $p = 0.03$) and DIP ($F = 6.64$, $p = 0.03$) concentrations. In particular, nutrient concentrations were significantly higher on the surface than at the bottom of the water column ($p = 0.05$ for DIN and $p = 0.03$ for DIP). There was also significant temporal variation in DIN concentrations ($F = 12.3$, $p = < 0.001$), which were higher in winter months (July 2019 and June 2020) than summer months. However, DIP concentrations displayed no clear temporal trends. Over the entire sampling period, DIN mostly comprised of NO_3^- (51%), followed by NH_4^+ (38%) and NO_2^- (11%). Inorganic N:P molar ratios ranged from as low as 0.04 to 9.3 with an overall mean of 2.1 across all sampling occasions.

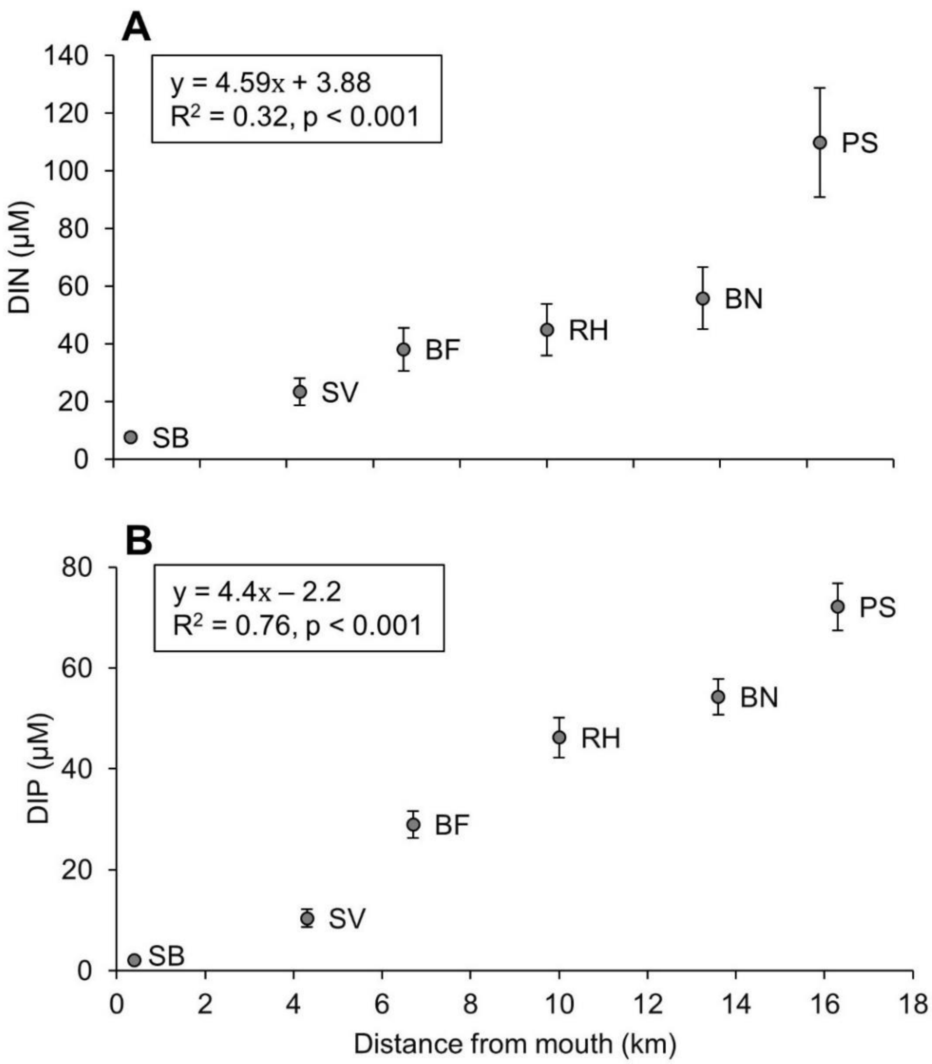

Figure 5.5 Dissolved inorganic nitrogen (DIN) (A) and dissolved inorganic phosphorus (DIP) (B) concentrations (mean ± SE, n = 18) recorded at Swartkops Estuary. The relationship of DIN and DIP with distance from the estuary mouth are shown in the respective textboxes.

Slight differences in water quality (salinity and DIN and DIP concentrations) can be anticipated at the three sites from which estuary water can be pumped into the saltpans (Table 5.3). From Site 1 toward the other sites downstream, salinity increases while nutrient concentrations decrease. It should be noted that the distance of the sites from the estuary mouth alone accounts for just 32% of the variation in DIN concentrations. As such, there are other variables that will influence the DIN levels of estuary water being pumped into the saltpans. Nonetheless, the relationship is still significant ($p < 0.001$).

Table 5.3 Interpolated values for salinity, dissolved inorganic nitrogen (DIN) and dissolved inorganic phosphorus (DIP) for the three sites from which estuary water can be pumped into the saltpans. Values were interpolated from separate linear regressions of the respective variables as a function of distance from the mouth of the estuary. The location of the sites can be seen in Figure 5.2 and the outputs from the linear regressions can be seen in Figures 5.4 and 5.5.

	Distance from mouth (km)	**Salinity**	**DIN (μM)**	**DIP (μM)**
Site 1	11.6	17.3	57.12	48.84
Site 2	9.7	20.3	48.4	40.84
Site 3	7.6	23.7	38.76	31.24

Significant spatial variability was recorded in both phytoplankton biomass ($F = 6.08$, $p < 0.01$) and cell abundance ($F = 2.39$, $p = 0.04$), with both increasing upstream from the mouth of the estuary (Figure 5.6A and B). Phytoplankton blooms (> 20 μg Chl-*a* l^{-1}) were recorded on four of the six sampling dates. On two of these occasions the blooms were limited to the site furthest upstream (PS) and on the other two occasions the blooms were recorded in the middle reaches of the estuary, with one bloom in February 2020 occurring as far downstream as the site BF. The occurrence of phytoplankton blooms in the sites near the head of the estuary are reflected in the large standard error bars in Figure 5.6A and B. Furthermore, significant temporal variability was also evident for both phytoplankton biomass ($F = 1.23$, $p < 0.001$) and cell abundance ($F = 10.09$, $p < 0.001$). The lowest values were recorded during winter months (July 2019 and June 2020) and the highest in summer (particularly February 2020). Biomass had a significant positive correlation with temperature ($r = 0.43$, $p < 0.001$) and DIP concentrations ($r = 0.37$, $p < 0.001$). Similarly, cell abundance was positively correlated to temperature ($r = 0.54$, $p < 0.001$) and less so to DIP levels ($p = 0.28$, $p < 0.01$).

While phytoplankton biomass and abundance increased with distance from the mouth, functional group diversity tended to decrease (Figure 5.6C). Diatoms (Bacillariophyceae) were generally the dominant group throughout the estuary over the study period and were responsible for the two recorded blooms that reached the middle reaches of the estuary. These blooms were comprised mostly of the centric diatom *Cyclotella atomus*.

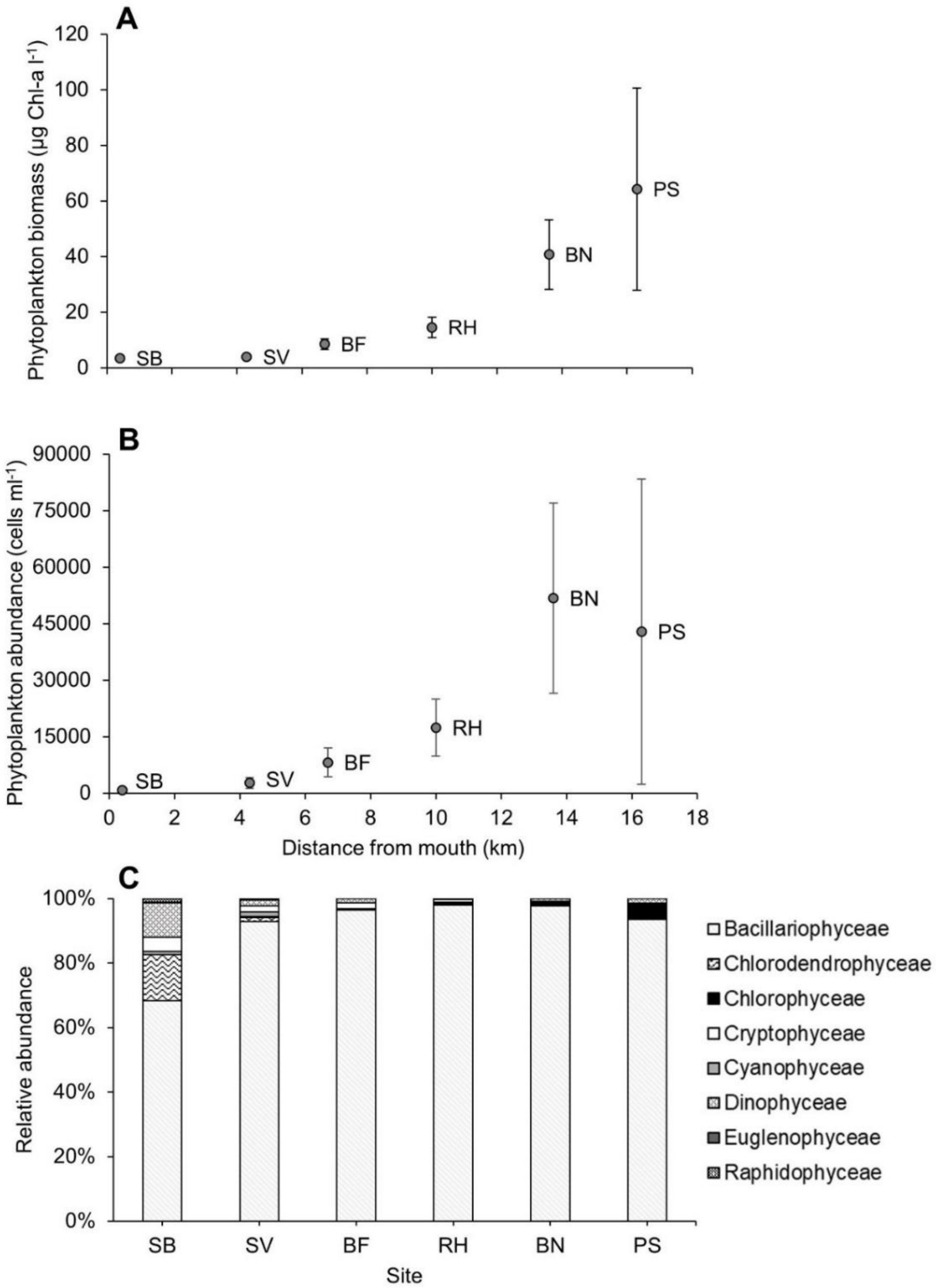

Figure 5.6 Phytoplankton biomass (A) and cell abundance (B) (mean ± SE, n = 18), and community composition (C) recorded at Swartkops Estuary.

5.3.2.2. Water quality at the MWAW

The physico-chemical parameters of the stormwater, excluding dissolved oxygen (DO), remained relatively consistent as the water passed through the artificial wetland (Table 5.4). Salt content remained consistently low in the stormwater, with salinity ranging from 0.5 to 2.4 between sampling dates. Water temperature generally decreased slightly by the time the stormwater had reached the outlet and ranged from 15 to 22°C across sampling trips. The pH decreased as stormwater passed through the MWAW on every occasion but remained basic (pH ≥ 7.9 on every occasion) in the outflowing water. The stormwater was well oxygenated as it entered the wetland but DO was significantly lower at the outlet ($W = 90$, $p = 0.002$).

Table 5.4 Mean physico-chemical parameters ($n = 10$) measured at the inlet and outlet of the Motherwell artificial wetland (MWAW).

	Salinity		Temperature (°C)		DO (mg l^{-1})		pH	
	Mean	**± SE**	**Mean**	**± SE**	**Mean**	**± SE**	**Mean**	**± SE**
Inlet	1.7	0.14	18.93	0.66	10.29	0.91	8.67	0.11
Outlet	1.67	0.11	17.95	0.73	4.47	0.72	8.23	0.09

As stormwater passed through the MWAW, mean DIN concentrations decreased by less than 5% (Figure 5.7A) and mean DIP concentrations increased by nearly 73% (Figure 5.7B). Both DIN and DIP concentrations had significant temporal variability ($F = 5.638$, $p < 0.01$ for DIN and $F = 4.58$, $p = 0.01$ for DIP), but no clear temporal trend was discerned and neither correlated with the flow rate of the stormwater. At the outlet of the MWAW, DIN ranged from 158 µM to 469 µM and DIP ranged from 1.3 µM to 15.5 µM over the study period. Total oxidised nitrogen (NO_x) constituted most (93%) of the DIN over the study period, with NH_4^+ accounting for just 7%, and the MWAW decreased the concentration of these forms of N equivalently.

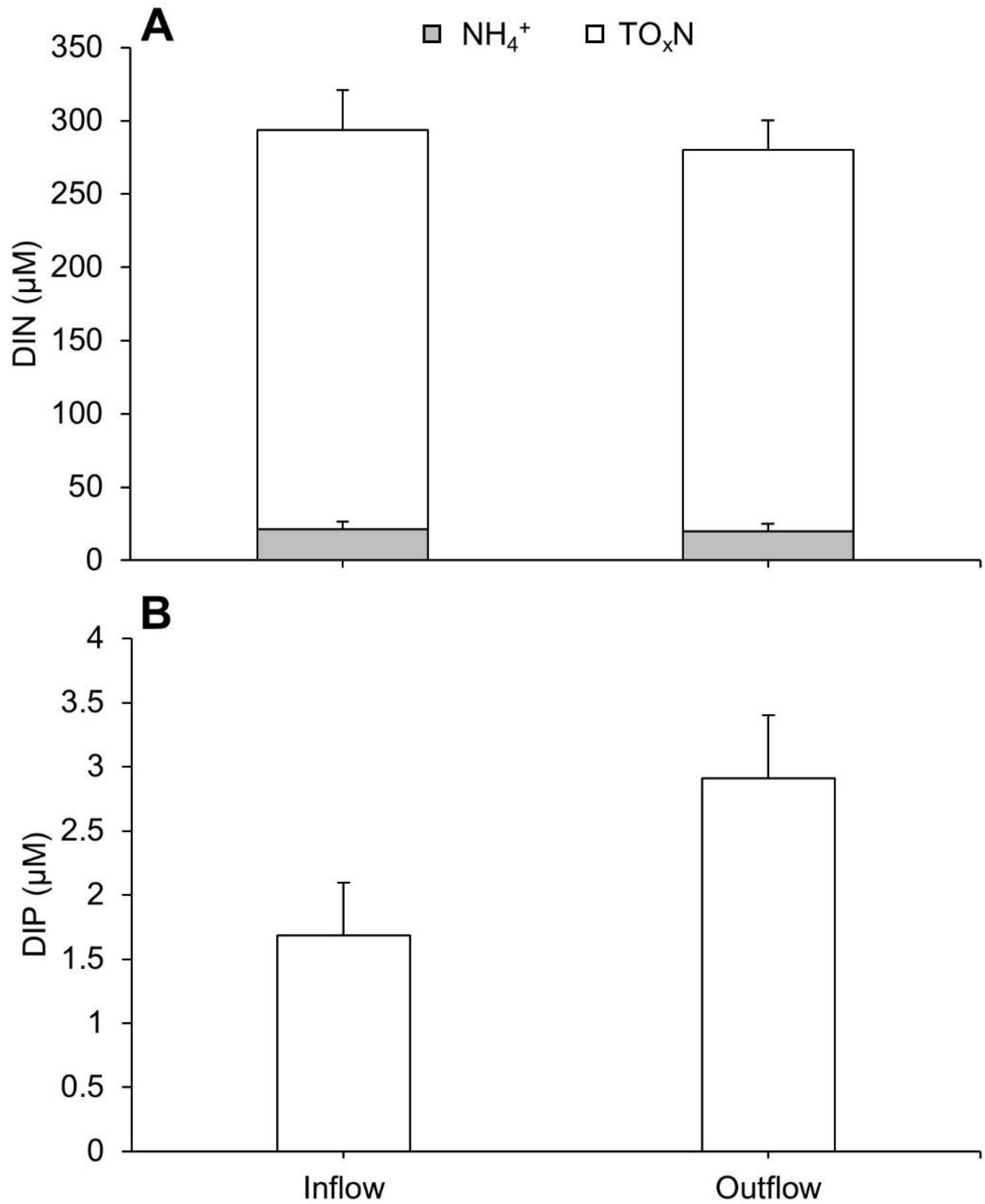

Figure 5.7 Dissolved inorganic nitrogen (DIN) (A) and dissolved inorganic phosphorus (DIP) (B) concentrations (mean ± SE, n = 28) at the inflow and outflow of the Motherwell artificial wetland (MWAW).

5.3.2.3. Expected salinity at the rehabilitated saltpans

The salinity of water expected in the saltpans is dependent on the sediment salt load in each saltpan, the volume of water each saltpan is filled with, and the salinity of that water. As these variables differ for each of the saltpans, it is anticipated that each one will have a different salinity once filled with water (Table 5.5). Unexpectedly, a reverse salinity gradient may form at the Bar None saltpans with the salinity decreasing form Pan 2 to Pan 4. However, this reverse gradient would likely change so that salinity increases for Pan 2 to Pan 4 as the hypersaline water from Pan 2 would flow through to Pans 3 and 4 (see Chapter 5.3.1).

It should be noted that these estimates have several assumptions and limitations. These values would only represent salinity levels within the first few weeks or months of the saltpans being filled with water, as various factors can influence salinity that cannot be accounted for. Firstly, the method used assumes that each saltpan is filled to maximum capacity (see Table 5.1) and that the maximum depth of sediment from which salt can enter the water column (through diffusion and wind/wave action) is 20 cm. These estimates also do not account for changes in salinity due to evapoconcentration or dilution by rainfall, nor for the exchange of water between saltpans as would occur at the Bar None saltpans. Lastly, these estimates can be affected by the accumulation of salt (through capillary rise from the hypersaline groundwater: Figure 4.2), or the flushing by rainfall from the surface sediment at the saltpans before any rehabilitation measures are taken. These limitations must be considered if these estimates are being used to inform any decision making in this rehabilitation project.

Table 5.5 Estimated initial salinity of water in the saltpans based on the source of water used to fill them. The 'estuary water' sites are shown in Figure 5.2.

	Stormwater	Estuary water		
		Site 1	Site 2	Site 3
Pan 1	21	36	39	43
Pan 2	-	56	59	63
Pan 3	-	50	53	57
Pan 4	-	48	51	54

5.4. Discussion

This study has shown that there are two feasible options for recreating a wetland habitat at the Redhouse saltpan, but just one option for doing so at the Bar None saltpans. The Redhouse saltpan can be filled with either estuary water using a pontoon-mounted pump afloat in the estuary or be fed stormwater via a pipe from the outlet of the MWAW, while the Bar None saltpans could only be filled by actively pumping estuary water into them. Unlike the management of the saltworks when it was functional, The Redhouse and Bar None saltpans would now have to be managed as separate entities. As such, the Bar None saltpans will receive water straight from the estuary. Only one site was identified from which water could be pumped into the Bar None saltpans. Estuary water can also be pumped into the Redhouse saltpan from this site, but it will also be possible to make use of two other sites further downstream.

One major difference between the two rehabilitation options that can be expected would be the hydrological regime of the saltworks. The use of a pontoon-mounted pump would allow for quickly filling the saltpans and provides the opportunity to adaptively manage the hydrological regime to maintain water levels. Contrastingly, the volume of stormwater discharged by the MWAW is temporally variable and unpredictable. The Redhouse saltpan would take far longer to fill with this rehabilitation option and the management of water levels will likely be challenging. It should also be noted that stormwater flows continuously in the MWC and is occasionally subject to sewage spills. If it is decided to fill the Redhouse saltpan with stormwater, the pipe constructed to do so should have a mechanism that stops flow into the saltpan and instead diverts stormwater into the estuary once again. This will prevent the saltpan from overflowing and contamination from raw sewage. Further monitoring of the rate and volume of stormwater discharged by the MWAW is recommended to discern any potential temporal patterns that may inform this rehabilitation option.

The use of estuary water to fill the saltpans is expected to result in euhaline to slightly hypersaline conditions at the Redhouse saltpan (salinity 36 - 43) and more hypersaline conditions at the Bar None saltpans. The salinity within the Bar None saltpans is expected to range from 50 to 60 initially but is likely to reach higher levels as water evaporates and passes through the series of saltpans (as occurs in operational saltworks). These expected salinity ranges are typical of the initial ponds of solar saltworks (Davis, 2000a; 2009). If stormwater is used to fill the Redhouse saltpan, brackish conditions (salinity ~ 20) can be expected. Such brackish conditions resemble salinity levels typical of natural lagoons and estuaries.

The brackish to slightly hypersaline conditions expected at the saltworks would facilitate the vision of restoring the site as a waterbird sanctuary. Lower salinity (< 80) saltpans support diverse primary producer communities including submerged macrophytes, macroalgae and benthic and planktonic microalgae (Britton and Johnson, 1987; Davis, 2000a). Submerged macrophytes are particularly important for most waterbird species that would occur at the rehabilitated saltworks (invertebrate feeders) as they support abundant and diverse invertebrate fauna (Martin and Randall, 1987; Kingsford and Porter, 1994; Anderson and Smith, 1999; Wolfram et al., 1999; Sánchez et al., 2006). Furthermore, saltpans of differing salinities provide greater habitat diversity for different feeding guilds of waterbirds and secures food availability over time by accounting for temporal differences in the densities of invertebrate populations (Britton and Johnson, 1987; Sánchez et al., 2006). Therefore, it would be favourable to achieve a salinity gradient along the series of saltpans and this can be accomplished using either rehabilitation option.

Whichever rehabilitation option is taken, it will be necessary to maintain the salinity of each saltpan within relatively narrow ranges. Large fluctuations in salinity can disturb biota that waterbirds will be dependent on, particularly macrophytes and invertebrates (Rubega and Robinson, 1997; Sánchez et al., 2006). The maintenance of salinity in functional saltworks is controlled primarily through the management of the hydrological regime. The rates at which water is pumped into saltworks are regulated to match the rate of evaporation (Davis, 2000a). Unfortunately, no information was available regarding how salinity was managed at this saltworks while salt was being produced but it is recommended that a pumping regime is determined before the rehabilitation project is initiated. If estuary water is used, the pumping regime should consider seasonality and the vertical position in the estuary water column from which water is pumped into the saltpans. Although there were no significant temporal differences in salinity in the estuary, a vertical salinity gradient was present in the water column. This gradient was relatively faint in summer months and generally quite distinct in winter months, especially in the area from which water would be pumped into the saltpans. It is therefore recommended that estuary water is not pumped from the top of the water column, especially after high rainfall, as this may cause sudden decreases in salinity in the saltpans which may disturb biota. While the saltworks was functional, pumping was reportedly ceased after floods until water in the estuary would return to brackish levels (Martin and Randall, 1987). If the stormwater is used to fill the Redhouse saltpan, salinity may be managed through controlling flow from the MWAW using a mechanism that can regulate flow as recommended earlier.

One cause of concern with either rehabilitation option is the high nutrient concentrations of both potential water sources. The estuary water was particularly high in DIP, but also had high DIN concentrations. Applying the estuarine eutrophic condition index developed by Lemley et al. (2015) to the results of this study, Swartkops Estuary can be classified as a eutrophic system, with DIN levels categorised as mesotrophic (≥ 0.1 but < 1 mg N l^{-1}, equivalent to ≥ 7.1 but < 71.4 µM) to eutrophic (> 1 mg N l^{-1}, or > 71.4 µM) and DIP levels as eutrophic category (> 0.1 mg P l^{-1}, or > 3.2 µM). Furthermore, phytoplankton biomass was mostly classified as mesotrophic (> 5 but ≤ 20 µg Chl-*a* l^{-1}), although eutrophic (> 20 but ≤ 60 µg Chl-*a* l^{-1}) to hypertrophic (> 60 µg Chl-*a* l^{-1}) levels were recorded on several occasions. The eutrophic condition of Swartkops Estuary described in this study reflect the findings of several past studies at the estuary (Emmerson, 1985; Scharler et al., 1997; Binning, 1999; Adams et al., 2019b). Swartkops Estuary receives high anthropogenic nutrient inputs from the lower catchment, as evident by the longitudinal gradient in DIN and DIP. These inputs are primarily from three WWTWs upstream (Adams et al., 2019b; Lemley et al., 2019). The significantly higher nutrient concentrations are likely a result of the fresh (i.e. less dense) wastewater flowing into the brackish estuary. Pumping estuary water from the top of the water column into the saltpans is thus also recommended against, as not only would it alter salinity in the saltpans (as mentioned earlier), but also result in a higher nutrient input.

The upper reaches of the Swartkops Estuary, where the proposed sites for pumping water into the saltpans lie, are not sufficiently flushed by tides or freshwater inflow and there is vertical stratification of the water column, which reduces mixing. The combination of these factors with the high nutrient inputs create conditions conducive to phytoplankton blooms (Ferreira et al., 2011; Lemley et al., 2015). During this study, blooms of the diatom *C. atomus* were recorded close to the proposed pumping sites on two occasions. More concerning is the recent occurrence of blooms of the raphidophyte *Heterosigma akashiwo*, which has the potential to proliferate into HABs. The estuary experienced the first recorded bloom of this species during the study period (May/June 2020). The bloom spread nearly throughout the entire estuary and coloured the water black (Rogers, 2020). Although the peak of the bloom was not captured during this study, sampling took place two days after the bloom reportedly decayed and hypoxic conditions (DO > 3 mg l^{-1}) were recorded throughout most of the estuary (from sites SV to BN). Blooms of this species are often ichthyotoxic and have been responsible for fish kills around the world (Khan et al., 1997; Cochlan et al., 2013). Another bloom of this species was recorded in November 2020 (D. Lemley, pers. comm., 2020). Blooms of the mixotrophic ciliate *Mesodinium rubrum* were also recorded at the site BN and occasionally at site PS during this study. Although *M. rubrum* blooms are generally not harmful, the collapse of such blooms may disturb benthic algal communities (Yih et al., 2013). A bloom of a species of the dinoflagellate *Peridinium quadridentatum* was found in the upper reaches (site PS) on one sampling date of the study. Another dinoflagellate species, *Gymnodinium* sp. has been reported to bloom in the estuary and the presence of a potentially toxic bloom-forming cyanophyte (belonging to the genus *Anabaena*) has been noted by Lemley et al. (2017).

In the stormwater, inorganic nutrient concentrations were highly stochastic. Relative to the estuary water, DIN concentrations were an order of magnitude higher and DIP concentrations were an order of magnitude lower. These highly variable, but overall high, levels of N are a common challenge in the management of stormwater (Taylor et al., 2005). The MWAW was found to be ineffective at removing nutrients from the stormwater – in fact, it acted as a source of DIP to Swartkops Estuary. The increase in DIP within the MWAW was likely due to the die-back of *Typha capensis*, which was abundant and not regularly harvested, resulting in the release of P into the water column (Vyzamal, 2007; Stroebel, 2019). Diverting the stormwater to the Redhouse saltpan would essentially transform the saltpan into a retention pond. Stormwater retention ponds (also known as wet detention ponds) are artificial ponds used to retain and remove contaminants from stormwater. Filling the Redhouse saltpan with stormwater may be beneficial in alleviating the persistent deterioration of water quality in the nationally important Swartkops Estuary by essentially acting as a retention pond. Although these ponds are considered a best management practise for stormwater management around the world, the efficiency at which they remove nutrients from stormwater is highly variable (Koch et al., 2014). Various factors influence nutrient removal processes in retention ponds (Collins et al., 2010; Gold et al., 2019) and the distinct characteristics of biogeochemical cycling in hypersaline environments make it difficult to speculate the efficiency at which the Redhouse saltpan could remove nutrients from the stormwater.

As both the estuary water and stormwater are high in inorganic nutrients, eutrophic conditions and phytoplankton blooms may be expected to occur regardless of which rehabilitation option is taken. Saltpans are susceptible to eutrophication due to the high water residence times in combination with the stratification and poor flushing of the water column (Du Toit and Campbell, 2002). The continuous pumping of nutrient-rich water saltpans often results in phytoplankton or macroalgal blooms (Abid et al., 2008; Difford, 2008; Elloumi et al., 2009; Evagelopoulos et al., 2009). As such, this rehabilitation project raises concerns such as potentially toxic cyanobacterial blooms or the proliferation of nuisance macroalgal blooms. There is also the possibility of creating wetlands that incubate HABs, which has been recorded at stormwater ponds elsewhere (Lewitus et al., 2008). This may potentially occur if either stormwater or estuary water is used and requires further research prior to the commencement of any rehabilitation measures.

In summary, there are two viable options for reinstating the hydrology of the saltworks: (1) the Redhouse and Bar None saltpans can both be filled with estuary water, or (2) the Redhouse saltpan can be filled with stormwater while the Bar None saltpans are filled with estuary water. Both rehabilitation options are expected to result in a salinity gradient, which would be favourable to rehabilitating the site as a waterbird habitat. At the Redhouse saltpan, brackish conditions can be expected if the stormwater is used and salinity resembling marine waters if estuary water is used. At the Bar None saltpans, hypersaline conditions can be expected. Both water sources have high inorganic nutrient concentrations and may result in eutrophication and algal blooms at the rehabilitated saltworks, although this necessitates further research.

6. Response of hypersaline sediment to stormwater input: A nanocosm experiment to determine primary producer dynamics.

6.1. Introduction

Two potential water sources for restoring the abandoned saltworks' hydrology have been identified: estuary water and stormwater. These two water sources differ notably in salinity and inorganic nutrient content (refer to Chapter 5). Salinity is the primary driver of community structure in waterbodies including saltworks (Segal et al., 2006; Smyth and Elliot, 2016), while nutrient availability (particularly N and P) regulates primary production (Fong et al., 1993; Anderson et al., 2002). As such, the two water sources may elicit distinct ecological trajectories that should be investigated prior to rehabilitation actions being taken (Choi, 2004). Although the saltworks was previously filled with estuary water, no information on the ecosystem structure or function is available. Furthermore, filling a hypersaline environment like the saltpans appears to be a novel approach and no studies detailing the ecological implications thereof are available.

Numerous studies have investigated the primary producer communities of saltworks and their response to changes in salinity (Segal et al., 2006; Oren, 2009; Khemakhem et al., 2010) and nutrient availability (Davis, 1978; Dolapsakis et al., 2005; Segal et al., 2009). The spatial patterns of salinity in saltworks and the associated changes in community structure in saltworks have been thoroughly documented and are fairly ubiquitous (Pedrós-Alió, 2004; Segal et al., 2006; Oren, 2009; Khemakhem et al., 2010); however, nutrient dynamics in these systems are highly idiosyncratic. Primary production in some saltworks is N-limited while others are P-limited and moreover, spatial or temporal shifts in nutrient limitation may occur within these systems (Javor, 1989; Joint et al., 2002; Dolapsakis et al., 2005; Segal et al., 2009). An understanding of primary production at the rehabilitated saltworks in response to the differing salinity and nutrient concentrations of stormwater and estuary is important as it will influence the value of the site to higher trophic levels, particularly waterbirds. The use of either rehabilitation option may result in different ecological regimes at the saltworks (Scheffer and Carpenter, 2003). Models for such regimes have been developed for aquatic environments experiencing salinisation (e.g. Davis et al., 2003; Strehlow et al., 2005) and eutrophication (e.g. Scheffer, 1989; Moss et al., 1996). Models for both processes typically include a state characterised by a clear water column and primary production being driven by benthic communities and a turbid state where phytoplankton are the dominant primary producers. In order to maximise the value of the saltworks to waterbirds, a clear water state that encourages the growth of submerged macrophytes and the establishment of diverse invertebrate communities would be favourable (Britton and Johnson, 1987; Strehlow et al., 2005; Sánchez et al., 2006).

In addition to forecasting the potential ecological trajectories that may result from ecological restoration activities, any unintended consequences should be identified before these activities take place (Howell et al., 2012). Saltpans are confined waterbodies that generally have a shallow water column and long water residence times, thus the use of nutrient-rich water to restore the hydrology of the saltworks raises concerns of eutrophication and algal blooms (Du Toit and Campbell, 2002; Ferreira et al., 2011). Undesirable impacts from eutrophication that may affect the success of this particular rehabilitation project include the depletion of oxygen from the water column, decreases in the abundance and diversity of food sources of waterbirds (macrophytes and invertebrates) and the proliferation of HABs that may be toxic to the waterbirds or their prey (Cloern, 2001; Shumway et al., 2003; Salgado et al., 2003). Disturbances (such as eutrophication) in this case are unfavourable in any ecological restoration project and should be avoided; instead, such projects should aim to recover a degraded ecosystem to a self-sustaining and resilient state (Ruiz-Jaen and Aide, 2005; Choi et al., 2008). It is uncertain if eutrophication and algal blooms occurred at the saltworks while it was functional, or such disturbances may become problematic in the rehabilitation of the site.

To inform decision making surrounding the potential rehabilitation measures that can be taken in this project, this study aimed to investigate the response of primary producers and nutrient dynamics resulting from the inundation of the dry hypersaline sediment of the saltworks with estuary water and stormwater. To achieve this, a nanocosm experiment (sensu Solomon and Hanson, 2014) using the different water sources as treatments was carried out. Similar experiments have been used in previous studies to elucidate relationships between primary producers and nutrients in aquatic environments (e.g. Baldwin et al., 2006; Pannard et al., 2007; Lemley et al., 2018). These types of experiments provide “model ecosystems” that provide a practical approach to understanding ecological responses that would be otherwise difficult or unmanageable to investigate (Benton et al., 2007). If well-executed, such “model ecosystem” studies are able to provide an accurate representation of ecological complexity (such as biomass and taxa diversity) in aquatic environments despite the small-scale of these experiments relative to natural systems (Belanger, 1997).

6.2. Materials and Methods

6.2.1. Sampling and experimental design

The nanocosm experiment consisted of 54 L plastic tanks, each containing dry surface (top 10 cm) sediment (roughly 13 kg) collected from the Redhouse saltpan. The sediment was kept intact to replicate in situ conditions as closely as possible (Figure 6.1A). Prior to the experiment, the plastic tanks were degreased with a 1 M NaOH solution, soaked in a 1% HCl solution for one day, and subsequently soaked in distilled water for an additional day. The tanks were kept in a glasshouse to expose them to natural light and temperature variability (Figure 6.1B). The sediment in each tank was inundated with 40 L of water (with an approximate depth of 16.5 cm from the top of the sediment) from the different sources (i.e. different treatments), with three replicates per treatment. The treatments included two stormwater treatments: “Stormwater 1” and “Stormwater 2”, which were comprised of semi-treated stormwater collected from the outlet of an artificial wetland nearby the saltpan (the MWAW: Figures 3.1 and 3.6). The “Stormwater 1” treatment consisted of unaltered stormwater as collected on the day of sampling, while the “Stormwater 2” treatment received a one-off addition of 200 µM NO_3^- (as KNO_3) to mimic the high nitrate loads often found in the stormwater (e.g. sewage spills). Additionally, an “Estuary” treatment was included which consisted of water collected from the Swartkops Estuary adjacent to the saltpan. This treatment mimicked the previous management regime of the saltworks (which was filled with estuary water while functional) for which no ecological information was available. River water collected in the upper catchment of the Swartkops River (near Groendal) was used as the control.

Once filled with water, the tanks were left for 24 h to allow disturbed sediments to settle. After the 24 h period, data collection commenced and continued on a weekly basis for three weeks (i.e. 21 days). A 28 day period with weekly sampling has commonly been used in similar experiments (e.g. Seitzinger et al., 2001; Porter et al., 2018; Lemley et al., 2018), but the fourth week of sampling in this study could not be carried out due to the onset of the nationwide COVID-19 lockdown. Additional sampling was carried out after 14 weeks to determine a ‘stable state’ of biotic communities in each treatment. Throughout the study period, each tank was supplied with filtered air to create some turbulence, allow CO_2 saturation and to avert diffusion-limitation of the phytoplankton. The tanks were kept covered with lids throughout the study to limit evaporation and prevent contamination.

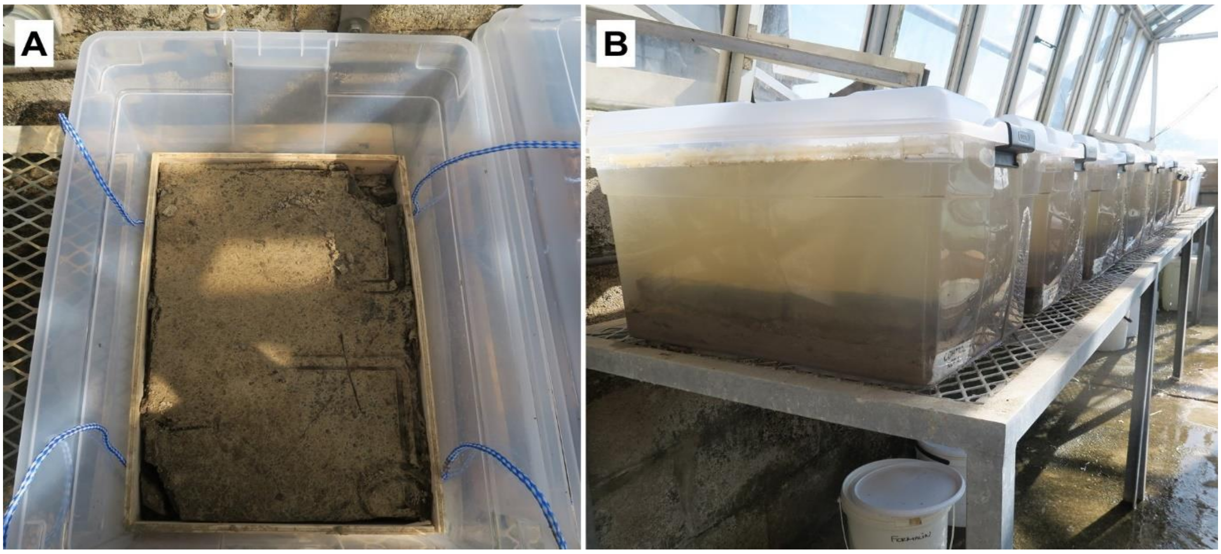

Figure 6.1 The dry intact sediment from the saltpan collected for the nanocosm experiment (A) and the nanocosm experiment setup in the glasshouse (B).

6.2.2. Data collection

6.2.2.1. Physico-chemical characteristics and inorganic nutrient concentrations

On each sampling occasion, physico-chemical parameters (salinity, temperature, pH and dissolved oxygen) of the water column were recorded in each tank using a YSI ProDSS Multiparameter Water Quality Meter. These characteristics were measured at the mid-point of the water column. Inorganic nutrient concentrations in the water column were also determined. Dissolved inorganic phosphorus (DIP = PO_4^{3-}-P) and ammonium (NH_4^+-N) concentrations were determined using standard spectrophotometric methods (oxidation method as per Parsons et al., 1984: Appendix 4F). Total oxidised nitrogen (NO_x-N = NO_2^- + NO_3^-) concentrations were determined using the reduced copper cadmium method (Bate and Heelas, 1975: Appendix 4F).

6.2.2.2. Primary producer communities

Replicate water samples (100 ml) were collected from each tank for the determination of phytoplankton biomass (determined as µg chlorophyll-*a* l^{-1}) using spectrophotometric methods as per Nusch (1980) (Appendix 4G) and the identification and enumeration of plankton cells (Appendix 4H). Sediment samples were collected from the top cm of the sediment of each tank using a 20 mm internal diameter corer and microphytobenthos (MPB) biomass was determined (as mg chlorophyll-*a* m^{-2}) using the procedure described in Appendix 4G. The relative abundance of MPB functional groups (green microalgae, cyanobacteria and diatoms) was recorded using a Moldaenke bbe BenthoTorch. Lastly, the percentage cover of macrophytes and floating macroalgae (which was apparent on the last sampling occasion, but not during the initial 3-week period) was recorded at the end of the study period.

6.2.3. Data analysis

All statistical analyses were carried out in R (R Core Team, 2020). The data were tested for normality graphically and using the Shapiro-Wilks test, and tested for homogeneity of variance using the Brown-Forsythe test. Friedman tests were carried out to detect significant differences among variables in treatments across the study period. Additionally, linear mixed-effects models (LMMs) were applied separately to phytoplankton and MPB biomass using the "glmmTMB" package (Brooks et al., 2017) to account for zero-inflated response variables. A gamma distribution was used for the models to account for the right skewed distribution of the response variables. Models using all possible predictor variables (temperature, salinity, NH_4^+, NO_x, DIP and treatment regime) and all possible combinations of predictor variables were considered. A first-order autoregressive (AR1) variance structure was included as a random effect in the models to account for repeated measures of treatments (Zuur et al., 2009). The models were assessed for violations of any assumptions following the protocol described by Zuur et al. (2010) to select the most appropriate combination of predictor variables.

6.3. Results

6.3.1. Physico-chemical characteristics and inorganic nutrient concentrations

The water sourced for the different treatments varied in both physico-chemistry and inorganic nutrient concentrations (Table 6.1). Expectedly, the Control and Stormwater were far less saline than the Estuary water. Unlike the Control, the Stormwater and Estuary water was alkaline. Additionally, DIP concentration was high in the Estuary water and NO_x was high in the Stormwater.

Table 6.1 Physico-chemical variables, inorganic nutrient concentrations, and phytoplankton biomass and abundance (mean ± SE, n = 3) of the water sampled from different sources (i.e. different treatments) prior to the study.

	Physicochemical parameters							
Treatment	**Temperature (°C)**	**± SE**	**Salinity**	**± SE**	**DO (%)**	**± SE**	**pH**	**± SE**
Control	22.9	0	0.12	0	75.43	0.54	6.6	0.09
Stormwater 1 and 2	26.43	0.09	1.75	0	36.33	3.11	8.5	0.03
Estuary	25.67	0.03	21.99	0.38	166.12	0.14	8.24	0.02
Nutrient concentrations (μM)								
	DIP	**± SE**	**NH_4^+**	**± SE**	**NO_x**	**± SE**		
Control	0.42	0.08	0	0	3.21	0.06		
Stormwater 1 and 2	1.07	0.21	4.56	0.76	115.38*	7.38		
Estuary	25.9	0.37	1.03	0.09	2.53	1.22		
*Prior to the 200 μM NO_3^- addition in the Stormwater 2 treatment								

The weekly variation in physico-chemical parameters was similar across the treatments over the study period (Figure 6.2A). Salinity continually increased in each treatment throughout the study period, with the greatest increase (> 10 in each treatment) in the first week. This was followed by a more gradual weekly increase that was similar across treatments for the remainder of the study. As expected, salinity in the Estuary treatment was markedly higher than the other treatments on each sampling occasion and reached approximately twice the salinity of all other treatments by the end of the study.

Dissolved oxygen (DO) concentrations were markedly higher in the Stormwater 1 and 2 treatments than the Estuary treatment in the beginning of the experiment. However, DO levels, and temporal trends thereof, became more similar among treatments by Week 2. The Estuary treatment had the lowest DO concentrations throughout the study period. All treatments showed a similar temporal trend in pH. There was a slight decrease in Week 2, but pH was similar across all treatments by the end of the study (8.7 – 9), with the most notable overall increase recorded in the Estuary treatment. Water temperatures remained similar among treatments each week and gradually decreased over the study period from ca. 34 to 23°C, as would be expected with the subjection to natural variability as well as the change in season (summer to autumn) over the period of the study.

Inorganic nutrient concentrations also showed generally similar temporal trends across all treatments over the study period (Figure 6.2B). As expected, the concentration of NO_x was initially highest in the Stormwater 1 and 2 treatments (98.6 μM and 128.8 μM, respectively), while the concentrations of DIP and NH_4^+ were initially highest in the Estuary treatment (3 μM and 0.1 μM, respectively). Both inorganic forms of N (NO_x and NH_4^+) decreased rapidly by Week 1 across all treatments. Total oxidised nitrogen (NO_x) remained at negligible concentrations for the remainder of the study in all treatments. Ammonium (NH_4^+) was below detectable limits for all treatments in Weeks 2 and 3; however, there was a sudden increase in the Stormwater 2 treatment in Week 3. Here, NH_4^+ reached over five times the concentration recorded at the onset of the study and the concentration appeared to increase slightly by Week 14. Dissolved inorganic phosphorus (DIP) generally decreased across all treatments over the study period to near negligible concentrations. Overall, DIP was significantly higher in the Estuary treatment than both Stormwater treatments ($p = 0.02$ in both cases). There was a notable increase in the Estuary treatment from Weeks 2 to 3 (97%), but DIP had decreased by Week 14 again, reaching a concentration similar to that recorded in the Stormwater 1 and 2 treatments.

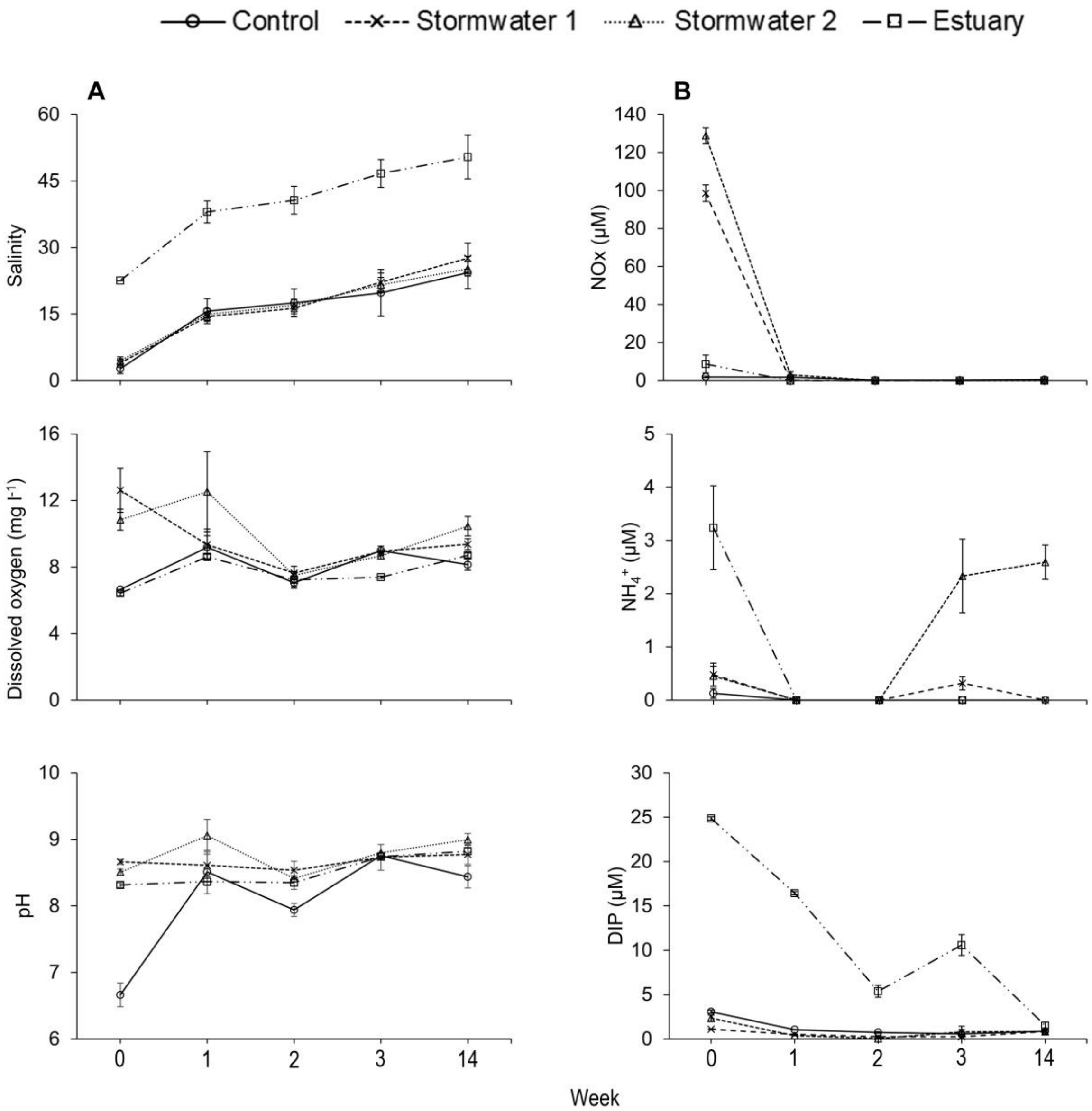

Figure 6.2 Physico-chemical variables (salinity, dissolved oxygen and pH) (A) and dissolved inorganic nutrient (total oxidised nitrogen: NO_x, ammonium: NH_4^+, and dissolved inorganic phosphorus: DIP) concentrations (B) (mean ± SE, n = 15 for physico-chemical parameters and n = 30 for inorganic nutrients) recorded in each treatment over the study period.

6.3.2. Primary producer communities

Phytoplankton biomass was initially low in the Stormwater treatments (< 15 µg Chl-*a* l^{-1}) and below detectable limits in the Estuary treatment (even though the highest Chl-*a* concentration was recorded in the estuary water prior to the start of the experiment: Table 6.2). Within one week of the sediment being inundated with water, phytoplankton biomass sharply increased to a maximum in all treatments (Figure 6.3A). Across all treatments, this increase in biomass was indicative of a phytoplankton bloom (> 20 µg Chl-*a* l^{-1}) and was concurrent with the rapid decrease of inorganic N (Figure 6.2B). The bloom was far more conspicuous in Stormwater 2 (which received a one-off NO_3^- addition) than all other treatments but, surprisingly, the peak in biomass was far higher in the Control than in the Stormwater 1 and Estuary treatments. The blooms collapsed in all treatments by Week 2. Chlorophyll-*a* remained low (≤ 8 µg Chl-*a* l^{-1}) in the Stormwater treatments for the remainder of the study period (lower than the Control, unexpectedly), and was even below detectable limits in some weeks. Biomass remained highest in the Estuary treatment from Week 2 onwards, and a bloom was recorded in the last week of the study (33 µg Chl-*a* l^{-1} – comparable to the bloom in this treatment in Week 1). Unexpectedly, there were no significant differences in phytoplankton biomass between treatments over the course of the study (Friedman $\chi^2 = 3$, $p = 0.39$).

Phytoplankton cell abundance increased greatly in all treatments when the sediment was first inundated (Table 6.2 and Figure 6.3B). In the first week, cell abundance continued to increase sharply in the Stormwater 2 and Estuary treatments, but nearly halved in the Stormwater 1 treatment. In Week 2, the highest abundance for the Stormwater 1 treatment was recorded, but declines occurred in the Stormwater 2 and Estuary treatments. Cell abundance declined from Week 3 onward in both Stormwater treatments, but increased in the Estuary treatment once again, where a high cell abundance (> 40 000 cells ml^{-1}) concurred with the bloom in Week 14.

The phytoplankton communities in the water sources for all treatments predominantly consisted of diatoms (Figure 6.3C). Diatoms remained the dominant phytoplankton group in all treatments until at least Week 3. The blooms in Week 1 were accounted for by diatoms (particularly the species *Cyclotella atomus* and *C. menenghiana*), although dinoflagellates and cryptophytes also contributed to the bloom in the Estuary treatment. However, by Week 14, cyanophytes had replaced diatoms as the dominant group. This shift from diatom- to cyanophyte-dominance by the end of the study period was also evident in the Estuary treatment, although it is not as visible in Figure 6.3C. This is a result of a major bloom of the chlorodendrophyte *Tetraselmis* sp. in one of the replicate tanks that began proliferating in Week 3. The bloom continued to proliferate until the end of the study period, where abundance reached > 80000 cells ml^{-1}. Interestingly, this species (although present in the other Estuary treatment replicates) bloomed only in one tank, and the relative abundance of cyanophytes was also notably higher in this replicate.

Table 6.2 Mean phytoplankton biomass (mean ± SE, n = 3) and cell abundance of the water sampled from different sources (i.e. different treatments) prior to the study.

	Phytoplankton		
	Biomass (µg Chl-*a* l^{-1})	**± SE**	**Cells ml^{-1}**
Control	0	-	108
Stormwater 1 and 2	29.6	2.96	299
Estuary	183.52	2.96	226

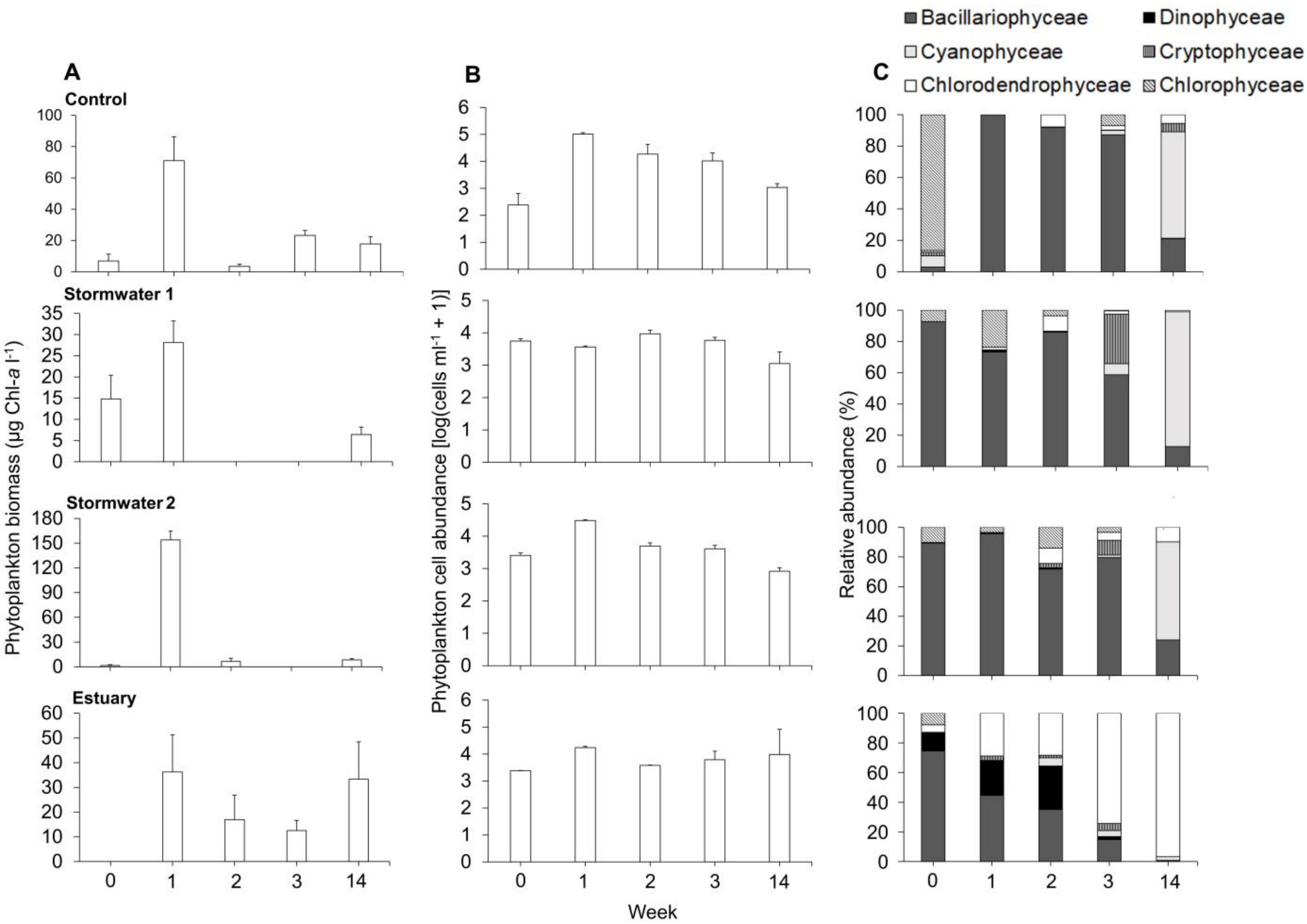

Figure 6.3 Phytoplankton biomass (A) and cell abundance (B) (mean ± SE, n = 30), and community composition (C) observed in the various treatments throughout the study period.

Microphytobenthos (MPB) biomass displayed slightly different temporal responses among treatments over the study period, but similar changes in community composition (Figure 6.4). Initially, MPB biomass was highest in the Stormwater 2 and Estuary treatments. In these two treatments, biomass decreased steadily through to Week 2, but increased again in Week 3 reaching a similar value to that of the initial week in each respective treatment. In the Stormwater 1 treatment, biomass increased gradually until Week 3 and remained consistent until Week 14. Surprisingly, MPB biomass was highest in the Control by the end of the study. Similar to the phytoplankton communities, MPB communities in all treatments shifted from diatom- to cyanophyte-dominance. This shift was least pronounced in the Stormwater 1 treatment, where diatoms and cyanobacteria were equally abundant at the end of the study. Furthermore, green microalgae were more abundant in the Stormwater treatments than in the Estuary treatment at the end of the study.

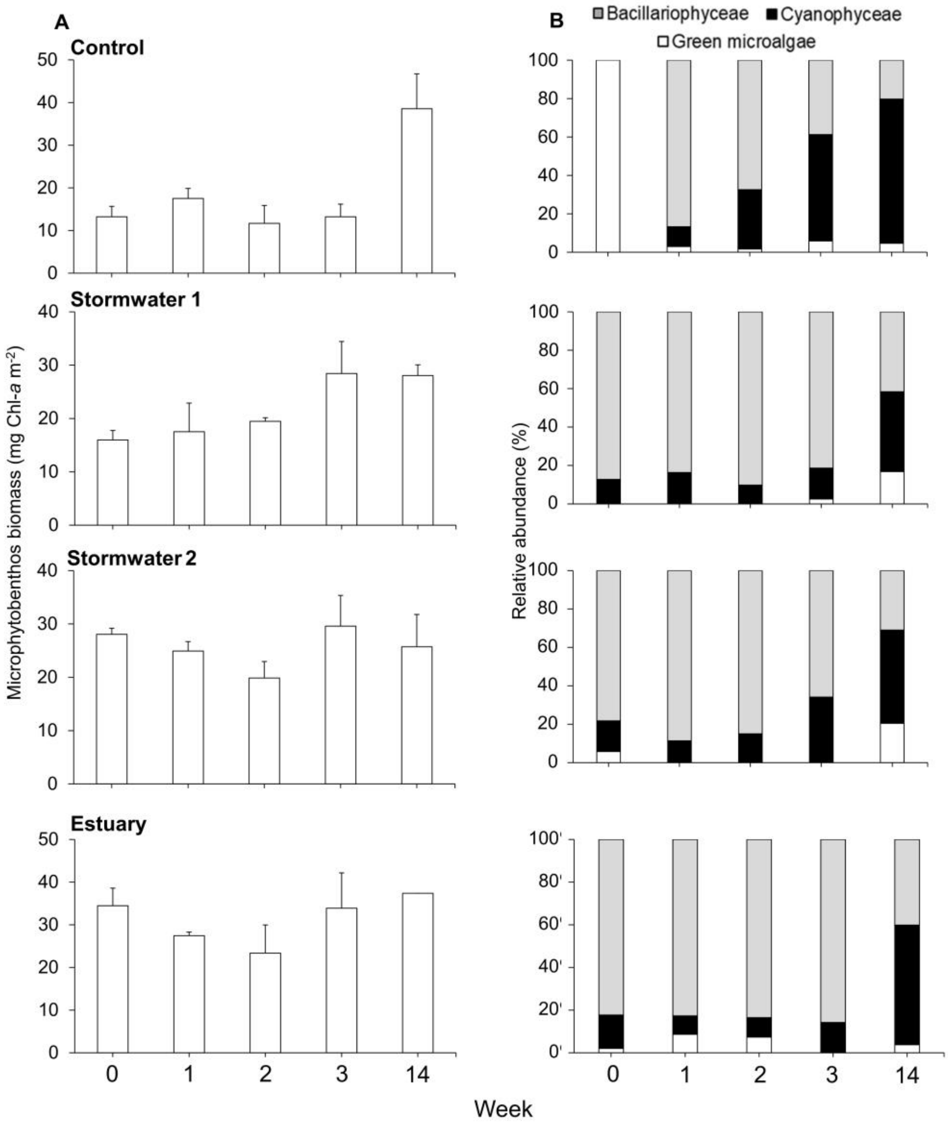

Figure 6.4 Microphytobenthos (MPB) biomass (A) (mean ± SE, n = 30) and community composition (B) over the study period.

Inter-treatment model-based analyses (Table 6.3) showed that the increasing salinity in all treatments over the study period was associated with a decrease in phytoplankton biomass and a significant increase in MPB biomass ($p = 0.02$). Furthermore, an increase in SRP was associated with an increase in both phytoplankton and MPB biomass, while a decrease in NO_x was associated with an increase in phytoplankton biomass and a decrease in MPB biomass. The Stormwater 1 and 2 treatments (particularly the Stormwater 2 treatment) were associated with lower phytoplankton biomass and higher MPB biomass than the Control, although the only significant response was an increase in MPB biomass in Stormwater 2 treatment ($p = 0.03$). The Estuary treatment was associated with an increase in both phytoplankton and MPB biomass.

It should be noted that the LMM for phytoplankton biomass violated the assumption homogeneity of variance (i.e. there was heteroscedasticity of the residuals). Consequently, the standard errors for this model are biased and the coefficient estimates may be less precise. Heteroscedasticity may also produce erroneously low p-values, but nevertheless, none of the p-values are below the 0.05 significance threshold. Heteroscedasticity was an issue in all of the models tested, as well as when transformations were applied to phytoplankton biomass. It may have been a result of the great difference in phytoplankton biomass in Week 1 (during the bloom) among treatments and relative to the other weeks, or it may suggest that phytoplankton biomass was influenced by a variable not considered in this study. No assumptions were violated for the MPB biomass model.

Table 6.3 Results of linear mixed-effects models (LMM) of phytoplankton and MPB biomass as a function of fixed-effect predictor variables (salinity, DIP, NO_x and treatment regime). A first-order autoregressive (AR1) variance structure was included as a random effect in the models to account for repeated measures of treatments. Significant p-values are indicated in bold font.

	Phytoplankton biomass			**MPB biomass**		
	C	**± SE**	**p**	**C**	**± SE**	**p**
Salinity	-0.07	0.48	0.89	0.32	0.14	**0.02**
DIP	0.14	1.14	0.90	0.35	0.35	0.32
NO_x	-0.05	0.08	0.52	0.03	0.04	0.49
Treatment						
Stormwater 1	-0.12	6.48	0.99	3.26	3.10	0.29
Stormwater 2	-9.81	7.28	0.18	6.94	3.26	**0.03**
Estuary	8.39	19.25	0.66	1.85	6.92	0.79

By the end of the study, the submerged macrophyte *Ruppia cirrhosa* and floating macroalgae (primarily *Enteromorpha* sp.) had appeared in the nanocosm tanks. The Stormwater treatments had the highest cover of *R. cirrhosa* and the lowest cover of macroalgae (Table 6.4). Interestingly, the cover of macroalgae was greatest in the replicate tank of the Estuary treatment in which the prolonged *Tetraselmis* sp. bloom was recorded, while the macroalgal cover in the other replicates of this treatment was similar to that recorded in the Stormwater treatments.

Table 6.4 Mean percent cover (± SE, n = 3) of submerged macrophytes and algae in each treatment at the end of the study period.

	Macrophytes		**Floating algae**	
	% cover	**± SE**	**% cover**	**± SE**
Control	30	5.8	12	5.7
Stormwater 1	43.3	3.3	2.3	1.3
Stormwater 2	26.7	13.3	8.7	5.8
Estuary	15	5	25.5	24.5

6.4. Discussion

The ecological response prompted by inundating dry hypersaline sediment from the saltworks was similar among the Stormwater and Estuary treatments. The most notable difference was salinity, which increased across all treatments as expected but the Stormwater treatments reached a brackish state (salinity 20 - 30) while the Estuary treatment became truly hypersaline (salinity 45 - 55). This range of salinity levels are typical of the initial ponds of saltworks, as were the primary producers that established in the treatments (Britton and Johnson, 1987; Davis, 2000a). Furthermore, temporal trends in physico-chemical characteristics and inorganic nutrient concentrations were similar among treatments. Phytoplankton blooms occurred within the first week in all treatments, making the water column turbid (Figure 6.5A), but collapsed within one week after depleting the water column of nutrients and shifting the treatments from a eutrophic to an oligotrophic state. Subsequently, phytoplankton biomass remained lower in the Stormwater treatments than in the Estuary treatment. Across all treatments, the water column reached a clear state with MPB growth becoming more apparent by Week 3 (Figure 6.5B). The submerged macrophyte *R. cirrhosa* began germinating and periphytic growth started becoming apparent by this time. *Ruppia cirrhosa* proliferated throughout the rest of the study and by Week 14 the Stormwater treatments reached a regime characterised by clear water and benthically-driven primary production (Figure 6.5C). The same regime was reached using estuary water, although one replicate tank in this treatment persisted in a turbid regime with high concentrations of phytoplankton.

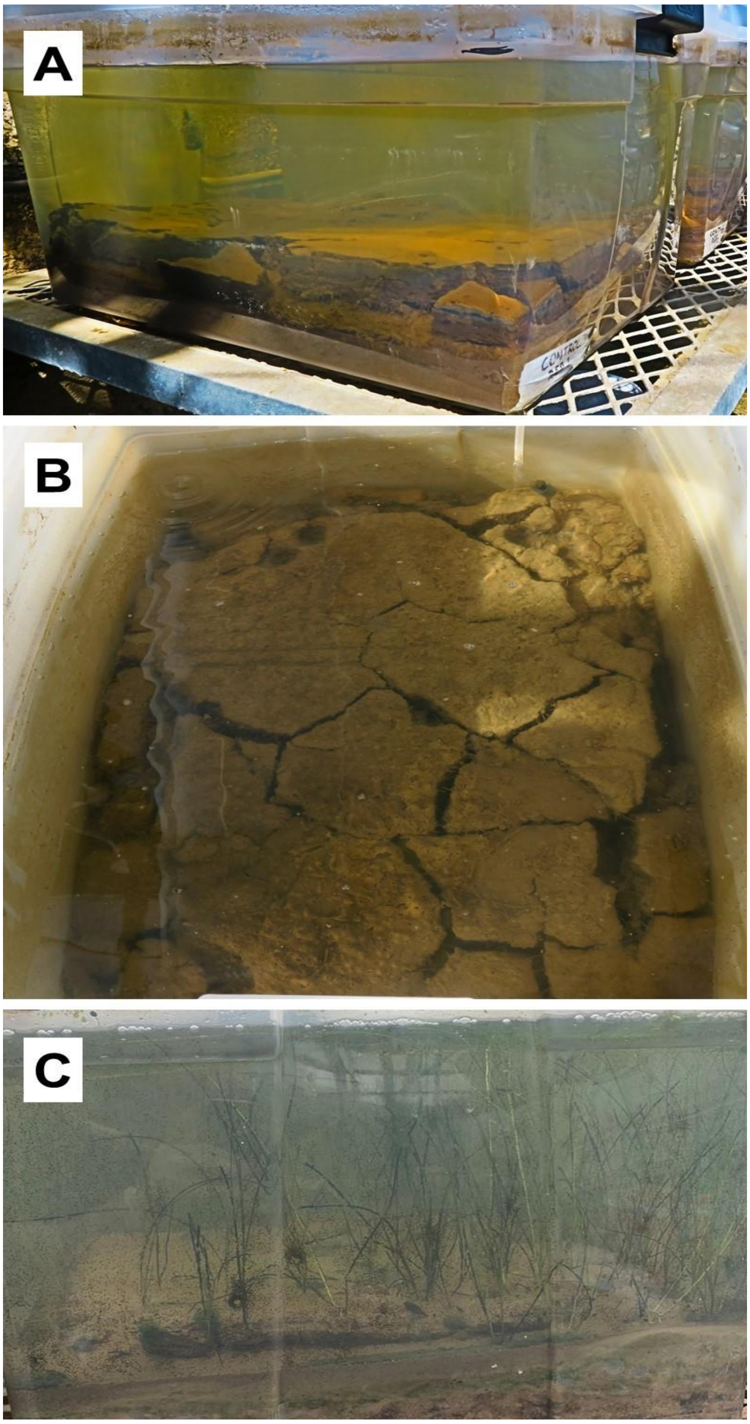

Figure 6.5 Across all treatments, phytoplankton blooms discoloured the water column in the first week (A) but following their collapse, the water column reached a clear state by the third week (B). At the end of the study, the water column remained clear and MPB and the submerged macrophyte *Ruppia cirrhosa* were the dominant primary producers (C).

It is important to note that there were two differences between this nanocosm experiment and the conditions that can be expected at the rehabilitated saltworks. Firstly, the inorganic nutrient concentrations in the collected water used for the stormwater and estuary treatments (Table 6.1) were markedly lower than the concentrations recorded in Chapter 5. For the stormwater, the experiment's initial DIP concentrations were similar to those recorded in the previous chapter (Figure 5.7B), but NH_4^+ levels were two orders of magnitude lower and NO_x concentrations were less than half those reported in Figure 5.7A, even in the Stormwater 2 treatment which received a one-off NO_3^- addition. In the estuary water, initial DIP concentrations in the experiment were somewhat lower and DIN levels were an order of magnitude lower than the expected values (Table 5.3).

Secondly, the nanocosm experiment did not mimic changes in water levels that will occur at the rehabilitated saltworks. In reality, the saltpans will experience decreases in water levels due to evaporation. To counter this, the saltpans will have to be periodically filled with nutrient-rich estuary water or stormwater. Such alterations in water levels, accompanied by the periodic nutrient loading of the saltpans, may result in intermittent phytoplankton blooms. During periods of drying and filling, wetlands undergo a turbid phytoplankton-dominated regime in contrast to a regime characterised by a clear water column and benthically-driven primary production by either submerged macrophytes or MPB when water levels are stable (Strehlow et al., 2005). However, such regime shifts are unlikely to occur within the short time period (weeks) that nutrients would be depleted within the saltpans as indicated by the experiment.

At the onset of this study, diatom blooms occurred across all treatments. These blooms quickly assimilated the excess inorganic nutrients in the first week and collapsed by the end of the following week. Throughout the rest of the study, diatom abundance decreased while cyanobacteria became the dominant phytoplankton group. Such a shift from diatom- to cyanobacterial-dominance is common as salinity increases in saltworks, but generally only occurs at salinity levels far higher than those recorded in this study (salinity ≥ 100) (Oren, 2000; 2015). Instead, this shift in community structure could be attributed to the decreasing DIN concentrations over the study period limiting the growth of most phytoplankton functional groups present. The cyanobacterial species recorded (*Anabaena* sp., *Aphanothece* sp. and *Chroococcus* sp.), however, are capable of N-fixation and were thus less affected by the depletion of DIN. At the rehabilitated saltworks, similar shifts in phytoplankton dynamics may occur in a cyclic manner. Shortly after nutrient-rich water is pumped into the saltpans, diatom blooms form and quickly collapse as the nutrients are depleted from the water column followed by an increase in the abundance of N-fixing cyanobacteria. Once pumping recommences, this cycle may be repeated. This is similar to the model of the seasonal succession of phytoplankton postulated by Margalef (1968), where the growth of large nonmotile taxa like diatoms is favoured under turbid high-nutrient conditions (i.e. after water is pumped into the saltpans), while motile taxa (cyanobacteria in this case) can become dominant in nonturbulent, nutrient-depleted conditions (i.e. after pumping has ceased).

Unexpectedly, the Estuary treatment maintained higher phytoplankton biomass throughout the study and was the only treatment to experience a bloom after the first week. This may be due to the higher availability of DIP in the estuary water. Furthermore, a phytoplankton bloom was recorded in the Control, which was notably larger (in terms of Chl-*a* l^{-1}) than the bloom in the Stormwater 1 treatment. This may also have been due to a higher availability of DIP, which was slightly greater in the Control than in the Stormwater treatments. These results suggest that phytoplankton will be limited by the availability of both DIN and DIP if the stormwater is used to fill the Redhouse saltpan, but just by the availability of DIN if estuary water is used. The inter-treatment analysis (Table 6.3) suggested that an increased availability of DIP was associated with an increase in phytoplankton biomass. However, a second phytoplankton bloom was also recorded in the Week 3 in the Control, in which inorganic nutrient concentrations and physico-chemical variables were similar to that of the Stormwater treatments (in which no further blooms occurred). This, along with the heteroscedasticity in the phytoplankton biomass LMM, suggests that phytoplankton biomass may have responded to a variable not recorded during the experiment.

Nonetheless, this experiment suggested that a turbid, phytoplankton-dominated regime is more likely to exist if estuary water is used than stormwater. In one of the replicate tanks of the Estuary treatment, a *Tetraselmis* sp. bloom was first recorded in Week 3 and continued to proliferate until the end of the study, by which time the species comprised 98% of the phytoplankton community. This bloom coloured the water a deep green as opposed to the clear water that was characteristic of the other treatments. It is uncertain why this bloom occurred in just one tank as the species was present across all replicates for the Estuary treatment since the onset of the study, and physico-chemical parameters and nutrient concentrations were similar across all weeks for the duration of the study. The bloom may have been a result of the position of the replicate tank in the glasshouse — unlike the other tanks it was situated directly next to a window (Figure 6.1B), allowing a larger surface area to be irradiated and thereby stimulating *Tetraselmis* growth (Kristoffersen et al., 2016). In this replicate tank, the cover of *Enteromorpha* sp. was also notably higher than the other replicates and notably higher than the Stormwater treatments. The higher salinity and availability of DIP, in combination with high irradiance, likely created ideal conditions for the germination and growth of *Enteromorpha* sp. (Sousa et al., 2007). This suggests that macroalgal blooms may be more likely to occur if estuary water is used at the Redhouse saltpan as opposed to stormwater. It is also interesting to note that the HAB-forming raphidophyte *Heterosigma* *akashiwo* (which has recently formed blooms in Swartkops Estuary, see Chapter 5.4) was abundant in the estuary water used for this experiment, but was not recorded once the sediment was inundated with the estuary water.

Once the saltpans have been filled with water and macrophytes have established, phytoplankton blooms may be supressed despite the continuous introduction of nutrient-rich water. It can be expected that macrophytes, particularly *R. cirrhosa* and *Enteromorpha* sp. will be able to establish and persist at the rehabilitated saltworks, as occurred in the experimental tanks. Martin and Randall (1987) also recorded these two macrophyte species at the Redhouse saltpan while it was operational. The presence of macrophytes, especially if they densely cover large areas of the saltworks, will be favourable in maintaining the saltpans in a clear water regime. Macrophytes can supress phytoplankton blooms (and the associated turbidity) through quickly depleting the water column of nutrients, resource competition, allelopathy and shading (Ozimek et al., 1990; Fong et al., 1993; Mulderij et al., 2007). To ensure that macrophytes persist at the site, management of the water and salinity levels within the saltpans will be necessary. The submerged macrophyte *R. cirrhosa* is sensitive to desiccation and will die-back within one week of continued exposure (Adams and Bate, 1994b), therefore the saltpans should be kept inundated permanently. The maintenance of steady water levels will also decrease the likelihood of an ecological regime shift, thereby maintaining the saltpans' habitat value to higher trophic levels. Additionally, *R. cirrhosa* germinates at salinity levels ≤ 35, but once established, the species can grow up to a salinity of 75 (Adams and Bate, 1994a). Macroalgae, however, have a remarkedly wide salinity tolerance range (Kirst, 1990). To ensure the establishment and persistence of macrophytes, salinity levels should be kept low initially (which would likely not need management intervention) and be kept below 75 once the saltpans have been filled with water.

Another factor that could maintain the desired clear water regime at the saltworks but was not accounted for in the nanocosm experiment (and should not be overlooked) is the potential presence of the Australian tubeworm *Ficopomatus enigmaticus* at the saltworks. These polychaetes aggregate in colonies and can form calcareous reef-like structures. Remnants of such structures cover extensive areas of all of the saltpans (Figure 3.5). The tubeworms are efficient filter feeders and can filter large volumes of water quickly and improve water quality by decreasing turbidity and chlorophyll-*a* concentrations very effectively (Davies et al., 1989; Bruschetti et al., 2009). At high densities, they have the potential to inhibit eutrophication and phytoplankton blooms (Bruschetti et al., 2008; 2018). Such top-down control has been shown to be able to hinder the symptoms of eutrophication in a shallow coastal lagoon for over two decades (Pérez-Ruzafa et al., 2019).

Furthermore, these tubeworm reefs create habitats that can support high abundances of organisms, particularly invertebrates and they have been shown to provide highly valuable feeding sites for waterbirds (Heiman et al., 2008; Luppi and Bas, 2009; McQuaid and Griffiths, 2014). Waterbirds also use these reefs as resting areas (Bruschetti et al., 2009). Despite being invasive, this species may be desirable at the rehabilitated saltworks due to its potential to limit phytoplankton blooms as well as the habitat value it presents to waterbirds. However, the tubeworms did not survive the desiccation of the saltworks, but they will likely be reintroduced to the system as estuary water is pumped into the saltpans or through floods. The species can tolerate hypersaline environments (Dittmann et al., 2009) and can be expected to persist in the saltworks if reintroduced.

This study has shown that the use of either stormwater or estuary water to reinstate the hydrology of the Redhouse saltpan can result in a regime characterised by a clear water column and benthically-driven primary production. The use of estuary water appeared to pose a higher risk of micro- and macroalgal blooms resulting in a turbid regime than the use of stormwater. However, this study did not mimic the dynamic conditions that may be expected at the saltworks that result from periods of drying or filling with nutrient-rich water. During these times, a turbid regime with phytoplankton blooms may arise. The nanocosm experiment, however, indicated that the primary producers that can be expected to arise at the saltpans can quickly assimilate the excess nutrients in both potential water sources. Furthermore, there was no evidence suggesting that potentially HAB-forming species would proliferate at the saltworks. In line with these findings, either water source can be used in the rehabilitation of the saltworks. Whichever option is taken, the rehabilitation should aim to maintain clear water, benthically-driven regime in the saltpans. To accomplish this, it is recommended that water levels are managed so as to not rapidly fluctuate, and that salinity within the saltpans is kept low enough to encourage the growth of macrophytes.

7. Towards the rehabilitation of the abandoned saltworks

7.1. Recommendations for planning and implementing the rehabilitation of the saltworks

In this thesis, any information relevant to the saltworks and its surroundings has been collated (Chapter 3) and the current condition of the site has been established (Chapter 4). This provided a basis to establish a vision for the proposed rehabilitation project and ecological targets to achieve this vision. The feasibility of the possible rehabilitation measures was investigated in Chapter 5, and the potential ecological consequences of these measures was investigated in Chapter 6. These data informed a plan for implementing and monitoring the rehabilitation of the saltworks. The rehabilitation plan is summarised in Box 7.1.

Box 7.1 The scope and current condition of the abandoned saltworks, the vision for the rehabilitated site, and the implementation and monitoring measures necessary to achieve the vision.

Study site

An abandoned saltworks at Swartkops Estuary (Eastern Cape, South Africa) comprised of four saltpans covering a total area of 163 ha.

Current condition

Since the pumping of estuary water into the saltworks ceased in 2018, the area has been left dry. The site is now characterised by vast expanses of hypersaline sediment with sparse patches of halophytic vegetation and hypersaline pools that occasionally form after rainfall. The once abundant and diverse birdlife of the site has all but disappeared.

Vision

The creation of four wetlands at the Redhouse and Bar None saltpans with a salinity gradient ranging from brackish to marine conditions. The wetlands will provide a regionally important mainland breeding ground for various shorebird species throughout most of the year and provide a foraging habitat for Palearctic migrant waterbirds over summer. Additionally, the Redhouse saltpan will be transformed into an extension of the Motherwell artificial wetland in order to effectively treat urban stormwater that is currently impacting water quality in the nationally important Swartkops Estuary.

Ecological targets

- Saltpans to host breeding waterbird colonies and Palearctic migrant species increasing over summer
- ≥ 80% of the saltpans' area inundated from February to October, decreasing to ≥ 60% from November to January to maximise habitat value for waterbirds
- Maintain a salinity gradient throughout the saltpans
- Presence of various primary producer functional groups (phytoplankton, microphytobenthos, submerged macrophytes and floating macroalgae)
- Absence of harmful algal blooms

Implementation of rehabilitation measures

Option 1: Fill the Redhouse and Bar None saltpans with estuary water; or
Option 2: Redhouse saltpan filled with stormwater from the outlet of the Motherwell artificial wetland and the Bar None saltpans filled with estuary water.

Monitoring

Monitoring is to be carried out over three temporal scales (see Table 7.3) and to cover the physical conditions and biotic characteristics (Table 7.4) of the saltworks.

7.1.1. Waterbird targets

The recommended targets for waterbirds at the rehabilitated saltworks are 80% of the annual mean number of individuals and species counted at the site (Table 7.4). Waterbird abundance and species richness varied seasonally and annually at the saltwork (Table 7.1, Figure 4.6) and the temporal variability should be considered when these targets are assessed – some years the target may not be met, but other years it may be exceeded. In line with Zwartkops Conservancy's plans to have Swartkops Estuary designated as a Ramsar Site and to safeguard the estuary's IBA status, management should prioritise the return of bird species that trigger the criteria for these designations (Table 7.2). A comprehensive list of waterbird species recorded at the saltworks is provided in Appendix 6.

Table 7.1 Waterbird numbers previously recorded at the saltworks. Counts have been conducted at Redhouse saltpan since 1994 and at the Bar None saltpans since 1995.

		Winter			**Summer**		
		Min	**Max**	**Mean**	**Min**	**Max**	**Mean**
Redhouse (Pan 1)	**No. of birds**	871	3339	1835	948	7494	3375
	No. of species	18	38	28	22	38	32
Bar None (Pans 2 – 4)	**No. of birds**	228	2645	800	368	3169	1404
	No. of species	14	31	23	23	38	29

Table 7.2 Waterbird species found at the saltworks prior to abandonment that triggered the criteria for Ramsar Sites and Important Bird and Biodiversity Areas (IBAs). Information collated from Martin and Randall (1987), Crawford et al. (2009) and P. Martin (unpubl. Data, 2020).

Species	Criteria met	Comments
Caspian tern	Ramsar Site (> 8 individuals)	Second largest colony in South Africa breeds at Redhouse saltpan in February to August and in October. Red List species (near-threatened).
Whitebreasted cormorant	Ramsar Site (> 130 individuals)	Breed at Redhouse and Bar None saltpans in February to September. Redhouse saltpan hosts second largest breeding colony in southern Africa.
Greater flamingo	Ramsar Site (> 760 individuals) and sub-Regional IBA (> 625 individuals)	May occasionally exceed criteria for a Globally IBA (1250 individuals). Red List species (near-threatened).
South African shelduck	Globally IBA (> 420 individuals)	Frequent the saltpans, most abundant in summer.
Pied avocet	Globally IBA (250 individuals)	Non-breeding, present year-round.
Kelp gull	Globally IBA (> 700 individuals)	Redhouse saltpan hosts one of the largest breeding colonies in southern Africa. Breed from September to January.
Cape shoveler	Sub-Regional IBA (> 175 individuals)	Breeds in October.
Curlew sandpiper	Sub-Regional IBA (> 3750 individuals)	Highly abundant in summer, found foraging on mudflats for invertebrates.
African spoonbill	Sub-Regional IBA (> 75 individuals)	Regularly breeds at Bar None saltpans.

7.1.2. Rehabilitation options

There are two options (i.e. rehabilitation measures) for inundating the saltpans to achieve the desired water levels: (1) filling the Redhouse and Bar None saltpans with estuary water; or (2) filling the Redhouse saltpan with stormwater from the outlet of the Motherwell artificial wetland (MWAW) and filling the Bar None saltpans with estuary water. Under the previous management regime of the saltworks, estuary water was pumped into the Redhouse saltpan, and from there through the series of Bar None saltpans. The old pumphouses are no longer operational, so estuary water will be pumped into the saltpans using a diesel pump mounted on a pontoon that floats in the estuary. Furthermore, it will no longer be practical to pump water from the Redhouse saltpan to the Bar None saltpans, instead these two areas will have to be managed separately.

The first option is similar to the previous management regime of the saltworks. However, since salt production has ceased, no income is being generated from continuing such a regime, which is costly due to the operation and maintenance of the pumps. The use and maintenance of the pontoon on which the pump is mounted will add additional costs to this rehabilitation project. As such, it is recommended that the second option is taken, as it will reduce these costs. This option requires the construction of a pipe from the outflow of the MWAW to the Redhouse saltpan. This may be costly initially, but the costs of fuel and pump maintenance will be greatly reduced as active pumping into the Redhouse saltpan (which is larger than all of the Bar None saltpans combined) will not be necessary. This will also ensure that the Redhouse saltpan will provide a wetland habitat that can host nationally important breeding waterbird colonies regardless of any funding challenges that may arise in the future.

In both rehabilitation options, the saltpans will be repeatedly filled with nutrient-rich water (the estuary water is rich in both N and P while the stormwater is exceptionally N-rich: Figures 5.4 and 5.6), which may result in recurrent phytoplankton blooms. In a nanocosm experiment (Chapter 6), phytoplankton blooms occurred regardless of whether stormwater or estuary water was used (a bloom was also recorded in the oligotrophic control) within a week of the dry saltpan sediment being inundated with water. However, these blooms took up the excess nutrients and collapsed within one week of being recorded (Figures 6.2B and 6.3A). Within two more weeks, the water column reached a clear state and primary production was driven by benthic communities (microphytobenthos and submerged macrophytes) and floating macroalgae across all treatments (Table 6.4 and Figure 6.4). Interestingly, it appeared that phytoplankton blooms may be more likely to reoccur if estuary water is used, potentially due to the higher P availability, as indicated by the higher phytoplankton biomass across the study period and a persistent *Tetraselmis* sp. bloom in one of the replicate experimental tanks filled with estuary water. Furthermore, there was no evidence indicating that HAB-forming species will proliferate at the saltworks with either estuary water or stormwater.

As this study has not demonstrated any potentially negative ecological impacts of using the stormwater to fill the Redhouse saltpan and the lower running costs associated with this approach, the second rehabilitation measure is recommended for the saltworks. This option also has the benefit of removing a major nutrient source contributing to the deterioration of water quality at the nationally important Swartkops Estuary (Adams et al., 2019b). If this option would ever result in undesirable consequences, or if it is decided against, the option of filling the Redhouse saltpan with estuary water can be adopted.

An alternative option that was not explored in this study is the use of estuary water in combination with stormwater to fill the Redhouse saltpan. However, this would likely result in high concentrations of both DIN and DIP from both water sources and could result in unmanageable nuisance macroalgal growth as recorded at the nearby Swartkops Solar Saltworks (Difford, 2008). As the combination of water sources was not included as a treatment in the nanocosm experiment and the ecological response thereof is unknown, mixing the two water sources is not currently recommended, but could be explored in future research.

7.1.3. Implementation of the rehabilitation activities

Three potential sites have been identified for filling the Redhouse saltpan with water and one for the Bar None saltpans (Figure 5.2). The three sites differ slightly in salinity and inorganic nutrient concentrations, with salinity increasing and nutrient concentrations decreasing from the site furthest upstream. Site 1 (the furthest site upstream, also the same site from which water was pumped into the saltworks previously) is recommended due to the accessibility of the saltpan from the estuary – water can be pumped directly into a channel near the estuary that feeds into the saltpan. This is also the only site into which water can be pumped into the Bar None saltpans. From here, water would be pumped into Pan 2 and allowed to flow into the lower lying Pans 3 and 4. For the second rehabilitation option, filling the Redhouse saltpan with stormwater, a pipe would need to be constructed from the outlet of the MWAW to the saltpan. The most practical approach would be to build a pipe from the MWAW to the north-eastern corner of the saltpan which lies roughly 300 m away. Outflows at the saltpans are also recommended for the management of water levels and salinity (see Chapter 7.1.4). Before any rehabilitation interventions commence, it is recommended that a hydrological model is developed to determine the timing and frequency at which pumping and outflows should occur to maintain a favourable environment for waterbirds.

7.1.4. Managing the rehabilitated saltworks

To meet the targets set out for waterbirds, it will be necessary to manage the hydrological regime of the saltworks at the saltpans to maximise the site's habitat value to the target species. Firstly, the saltpans need to be inundated but subsequent management of water levels will be necessary. The management of physico-chemical characteristics of the rehabilitated saltpans will largely be governed by the management of the water levels.

7.1.4.1. Water levels

Water levels were generally kept high throughout most of the year while the saltworks was operational, but the site should now be managed to maintain water levels suitable to the needs of waterbirds. Water depth is an important factor influencing the abundance and distribution of waterbirds at saltworks (Velasquez, 1992; López et al., 2010). Most importantly, high water levels result in the formation of islands within the saltpans utilised by various breeding colonies. However, spatial and temporal heterogeneity in water depth will be favourable as it will allow for a variety of foraging guilds to utilise the site for feeding (Takekawa et al., 2006; Ma et al., 2010). Spatial heterogeneity in depth can be expected due to the topography of the saltpans (Figure 7.1), but a managed hydrological regime will be required for temporal heterogeneity. The management of water levels should be adaptable with priority given to the breeding activity of waterbirds, particularly rare species such as Caspian terns and greyheaded gulls (Martin and Randall, 1987).

The hydrological regime should be managed to keep water levels high (≥ 80% of the saltpans' area inundated) from February to the end of October to keep islands within the saltpans protected from the mainland. Predation by small mammals and egg poaching were the primary causes of breeding failure at the saltworks in the past (Martin and Randall, 1987), so these islands should be surrounded by water to facilitate the breeding activity of most of the target waterbird species (Table 7.2). Islands 1, 2 and 3 in the Redhouse saltpan (Figure 7.1) were particularly important as they host some of the largest breeding colonies of certain shorebird species (Martin and Randall, 1987). To keep the islands protected from predators and poachers, channels around these islands should be kept at least 1 m deep as has been recommended by Martin and Randall (1987). In particular, the narrow channel between Island

1 and the mainland should be expanded. Such channels should be excavated prior to the saltpans being inundated. Additionally, more islands could be created within the saltpans to provide more breeding sites for waterbirds.

Water levels can be allowed to decrease from November to January. Generally during these months most waterbirds are not breeding at the site; instead, there is an influx of Palearctic migrant species (Table 7.1; Figure 4.6). Many of these species are invertebrate-feeding waders that would benefit from a lower water level as this increases the size of shallow areas and exposes mudflats, providing ample foraging habitat for these species (Martin and Randall, 1987; Sánchez et al., 2006). A minimum water level of 60% of each saltpan's area is recommended as some islands will still remain protected and the saltpans could still be filled relatively quickly if necessary.

Although water levels should be kept high throughout most of the year, water levels should not be allowed to get too high. This will decrease the area of the islands used by waterbirds for breeding and possibly lead to the abandonment of breeding attempts (Ma et al., 2010). If estuary water is used to fill the saltpans, the pumping of water can be managed adaptively, so pumping can cease once the desired water levels have been achieved. If stormwater is used to fill the Redhouse saltpan via a pipe, a mechanism is necessary that can control the flow and divert it into the estuary as necessary because the MWAW has a permanent outflow. Furthermore, it is recommended that the saltpans have an outflow (e.g. outlet weirs) to regulate water levels and to prevent the concentration of salt within the saltpans (see the section below), especially if brackish estuary water is repeatedly pumped into the saltpans. While the saltworks was operational, water would be pumped out of the Bar None saltpans toward an extraction pan as part of the salt production process, thus limiting the accumulation of salt. For the Bar None saltpans, an outflow can be created on the western side of Pan 4, where a channel has already been dredged and has been used in the previous management regime to release water from the saltpans if necessary. The Redhouse saltpan should also have an outflow point, particularly if it is filled with estuary water. However, the discharge of hypersaline water into the estuary would have impacts on the ecology that would need to be assessed and may require a Coastal Waters Discharge Permit in line with the National Environmental Management: Integrated Coastal Management Act (Act No. 24 of 2008). It is also recommended that floodgates are built between the Bar None saltpans that allow for more efficient regulation of water levels within each individual saltpan.

Unfortunately, no detailed information regarding the previous management of the hydrological regime was acquired. Figure 7.1 depicts the saltpans filled to maximum capacity and at the lowest allowable water levels observed at the saltworks via a desktop assessment of satellite images spanning from 2014 to 2017. Although this figure does not precisely represent the 60% and 80% water surface area targets, it can be used as a spatial reference to assess water levels at the site.

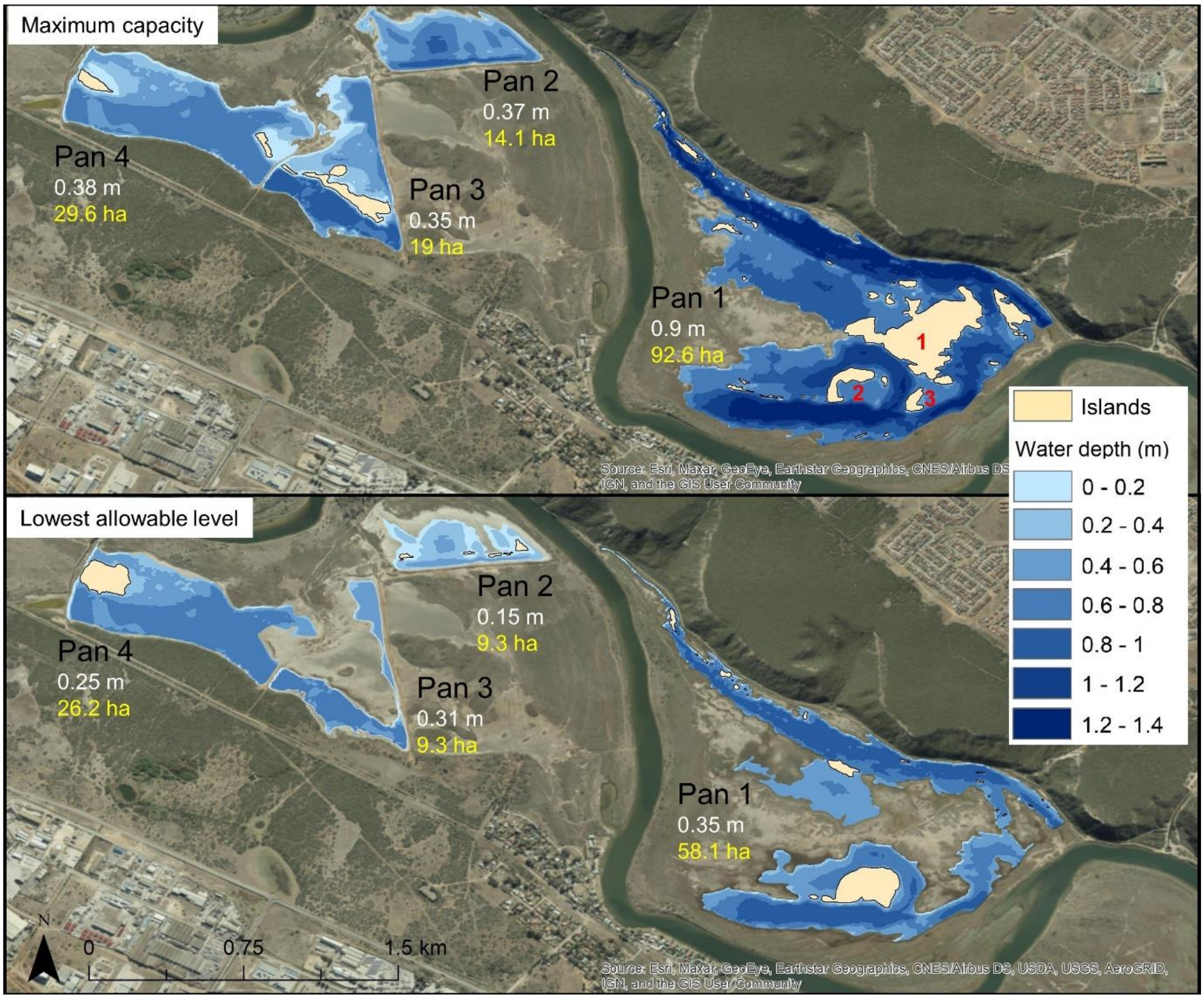

Figure 7.1 The saltpans filled to maximum capacity and at their lowest allowable level while the saltworks was functional (excluding times when the pumps were not operational). Red numbers indicate islands that were particularly important breeding sites for water birds; white text indicates mean water depth and yellow text indicates water surface area.

7.1.4.2. Salinity management

In operational saltworks, saltpans are managed within narrow salinity ranges (Davis, 2009). This creates a salinity gradient ranging from salinity levels similar to nearshore water up to NaCl saturation (Davis, 2000a). Ideally, salinity levels at the rehabilitated saltworks should be similar to the those while salt production was occurring. The Redhouse saltpan was previously maintained at a salinity between 30 and 50 (Martin and Randall, 1987), but the Bar None saltpans' previous salinity range is not known. This makes it challenging to set salinity targets for the Bar None saltpans. Although the initial (i.e. first few weeks or months) salinity levels of the saltworks have been estimated, salinity will be variable in response to several factors once the saltpans have been filled with water (Chapter 5.3.2.3). The minimum targets for salinity have been set as the initial expected salinity at the saltworks (Table 5.5) and it is recommended that the maximum salinity in any of the saltpans does not exceed 80 to encourage the growth of diverse primary producer communities, particularly macrophytes to maintain abundant and diverse prey populations for waterbirds (Anderson and Smith, 1999; Wolfram et al., 1999; Sánchez et al., 2006). However, due to the uncertainty surrounding salinity levels that will occur at the saltworks, it is recommended that appropriate targets are set after one year of monitoring. It would also be favourable to reinstate a salinity gradient to provide foraging opportunities for different guilds of waterbirds over different spatial and temporal scales (Britton and Johnson, 1987; Sánchez et al., 2006). A salinity gradient can be expected to form at the saltworks (Table 5.5) but may require active management to be maintained. Lastly, it is also recommended that large salinity fluctuations within saltpans is avoided so that primary producer and invertebrate communities are not disturbed (Sánchez et al., 2006).

Managing the salinity will also be accomplished through a managed hydrological regime. This will be more pertinent for the Bar None saltpans, where estuary water flow through a series of saltpans, as opposed to the Redhouse saltpan that will be managed as a standalone waterbody. To achieve the desired salinity levels, the rate of pumping of estuary water would have to be regulated according to the rate of evaporation (Davis, 2000a). An undesirable consequence that may occur, particularly if the saltpans are filled with estuary water, is the excess concentration of salts within the saltpans if the water is continually left to evaporate. A continual increase in salinity will lead to changes (and potentially declines) in prey availability and composition which may be detrimental to waterbirds (Britton and Johnson, 1987; Velasquez, 1992; Warnock et al., 2002). As previously mentioned, outflows will be necessary at the saltworks, particularly at the Bar None saltpans. This will allow for the release of water it becomes excessively hypersaline and the saltpans can be subsequently refilled with less saline estuary water or stormwater. Floodgates between the saltpans will also allow more precise control of salinity levels, as water can be allowed to pass to the subsequent saltpan once at the desired salinity. Lastly, to avoid large fluctuations in salinity when estuary water is being pumped into the saltpans, water should be pumped from the bottom of the water column as occasional vertical stratification occurs (especially in winter or after rainfall events) where the surface waters are very fresh (Figure 5.4).

7.1.4.3. Nutrient management

To maintain stable water levels, the saltpans will have to be periodically filled with nutrient-rich water. Although the nanocosm experiment indicated that nutrients could quickly be assimilated at the saltpans, care must be taken to avoid eutrophic conditions. If nutrient concentrations exceed the target values (see Table 7.4), pumping can be stopped to cease excessive nutrient loading. Once nutrient concentrations have decreased to below the target values, pumping can recommence. However, if waterbirds are breeding, the saltpans should first be filled to a level that ensures that the islands utilised by breeding colonies are protected from the mainland.

As recommended to avoid fluctuations in salinity if estuary water is pumped into the saltpans, water should be pumped from the bottom of the water column to limit nutrient loading. The vertical stratification of the water column results in significantly higher inorganic nutrient concentrations in the surface waters of Swartkops Estuary (Figure 5.5). Another measure that could be taken to limit nutrient loading is the construction of artificial wetlands at the inflow of the saltpans. Artificial wetlands are globally considered a best management practise for alleviating nutrient enrichment of natural waterbodies but need to be carefully designed and constructed as they can occasionally contribute to nutrient loading of receiving waters (Fisher and Acreman, 2004). An appropriate design was presented by Du Toit and Campbell (2002), which made use of halotolerant plant to efficiently decrease nutrient loading at a saltworks at Swartkops Estuary.

While the nanocosm experiment indicated that primary producers in the saltpans could quickly assimilate nutrients, the use of an artificial wetland or other means to decrease nutrient loading is recommended. If nutrient-rich water is consistently pumped into the saltpans, nutrients would likely accumulate in the sediment and continually be recycled within the system, thereby driving eutrophication and algal blooms. This has occurred in several South African estuaries such as Knysna and Great Brak where nutrient enrichment in shallow, poorly flushed areas has resulted in regular nuisance blooms of filamentous macroalgae (Human et al., 2015; 2016). Such blooms can displace beds of submerged macrophytes, which may decrease the foraging value of the saltpans to waterbirds and can also be aesthetically unappealing and create foul odours when they decompose. If such a situation arises, then regular harvesting of macroalgae or dredging of the sediment may become necessary.

7.2. Monitoring plan for the rehabilitated saltworks

Monitoring is an essential component of any ecological restoration project that not only allows for evaluating the success of restoration measures, but also allows for adaptive management actions to be taken. A monitoring plan has been developed which adopts a three-tiered approach (Table 7.3) as incorporated in the National Estuarine Monitoring Programme (Cilliers and Adams, 2016), but adapted for the purpose of this project. Tiers 1 and 2 together comprise the fundamental environmental data to develop an understanding of the ecology of the restored saltworks in order to inform any necessary management decisions. Tier 1 incorporates short-term environmental responses while Tier 2 is comprised of responses that are expected to take longer. Tier 3 sampling is limited to the Redhouse saltpan after heavy rainfall events or reported sewage spills when the characteristics (e.g. volume, nutrient loads) of the stormwater runoff are more extreme, and rapid and distinct ecological responses such as algal blooms are expected as a result.

Table 7.3 Sampling frequency for the different tiers of the monitoring plan.

Tier	Sampling frequency
Tier 1	Monthly for the first year. Frequency in following years contingent on the findings of the first year, but minimum biannually.
Tier 2	Biannually – once in summer and once in winter.
Tier 3	Following any major storm events or reported sewage spills at the MWC for Redhouse saltpan only.

The parameters to be monitored and their specific targets are stipulated in Table 7.4. As this rehabilitation project will result in the creation of a novel semi-natural ecosystem, a baseline to inform targets has not yet been established. A baseline will be established after one year of Tier 1 and Tier 2 sampling has been completed, after which targets can be adjusted if necessary. The targets for salinity were set to resemble the lower salinity ponds of saltworks and the salinity expected from the findings of this thesis (Table 5.5 and Figure 6.1A), with the upper limit set at 80 to encourage macrophyte growth throughout the saltpans. The target for pH was also based on the findings of this thesis (Figure 6.1A) as well as the typical conditions reported at other saltworks. The target for turbidity was set at > 0.5 m Secchi depth to encourage benthic growth throughout most of the saltpans, although turbidity may be high during periods of filling of the saltpans. The targets for temperature were set on anticipated seasonal variation. The targets for DIN, DIP, DO and Chl-*a* have been set based on the criteria for a “fair” condition according to the estuary eutrophic condition index (Lemley et al., 2015).

However, the physical and biological processes that affect these variables cannot be expected to be equivalent between saltworks and estuaries. Furthermore, in accordance with the South African Water Quality Guidelines for aquatic ecosystems (DWAF, 1996), targets for DIN, DIP and DO should be set only after case-specific studies. Additionally, management responses to any of the major targets or concerning issues that may occur are outlined in Figure 7.2.

Table 7.4 Parameters to be monitored and the associated targets to be achieved at the rehabilitated saltworks.

Parameter	Targets	Tier
Abiotic		
Water surface area and depth	From February to October, water should cover > 80% of each saltpan. From November to January, water should cover ≥ 60% of each saltpan. See Figure 7.1 for reference.	1
Salinity	80^{th} percentiles: Redhouse saltpan: 15 – 30 if stormwater is used; 30 – 50 if estuary water is used. Bar None saltpans: 50 - 80	1 & 3
DIN	80^{th} percentile < 1 mg l^{-1}	1 & 3
DIP	80^{th} percentile < 0.1 mg l^{-1}	1 & 3
DO	10^{th} percentile > 5 mg l^{-1}	1 & 3
pH	80^{th} percentile ranging between 8 – 9. pH monitoring should also account for temperature to inform the risk of ammonia toxicity.	1 & 3
Turbidity	80^{th} percentile > 50 cm Secchi disk depth	1 & 3
Temperature	5^{th} and 95^{th} percentile ranging from 10 to 30°C	1 & 3
Stormwater discharge at MWAW	If sewage spills at the MWC are reported, then flow to Pan 1 is stopped.	3
Biotic		
Submerged macrophytes and macroalgal cover	Pan 1: 50 to 75% cover Pans 2 – 4: 25 – 75% cover	2
Phytoplankton biomass (measured as Chl-*a*)	90^{th} percentile: < 20 µg l^{-1}	1 & 3
Phytoplankton identification	Assessed for any potentially HAB-forming species	1
Microphytobenthos biomass	90^{th} percentile: Chl-*a* < 50 mg m^{-2}	2
Waterbird counts	Annual abundance and species richness ≥ 80% of historical levels (see Figure 4.6 and Table 7.1): Redhouse saltpan: ≥ 1500 birds of 23 species in winter; ≥ 2700 birds of 26 species in summer. Bar None saltpans: ≥ 640 birds of 18 species in winter; 1120 birds of 23 species in summer.	2

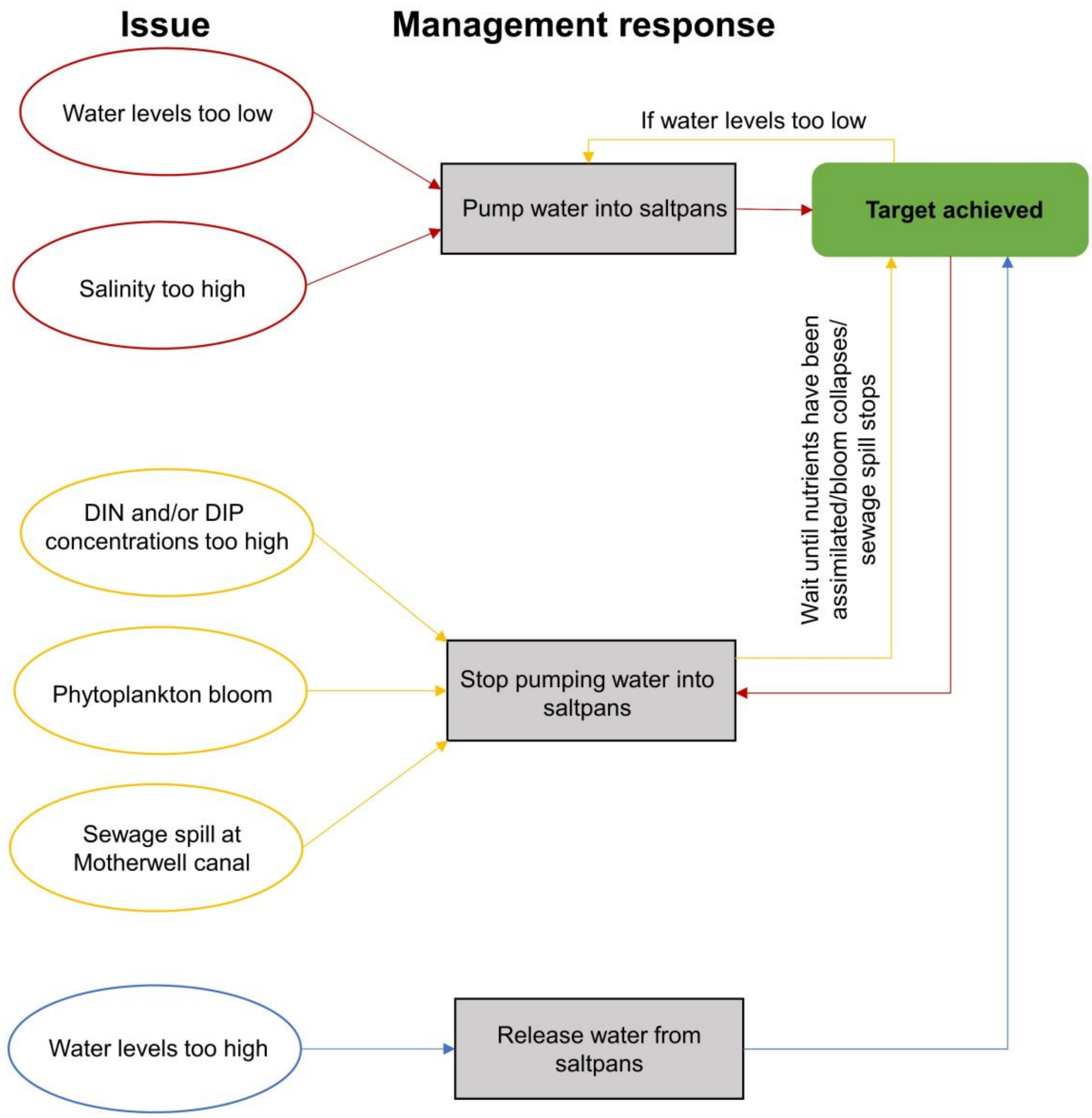

Figure 7.2 Recommended management responses to issues that may arise at the rehabilitated saltworks. Specific targets are stipulated in Table 7.4.

In order to monitor the progress of the rehabilitation project over time, the Five-Star System (Table 7.5) and the Ecological Recovery Wheel from the Society for Ecological Restoration's International Principles and Standards for the Practice of Ecological Restoration (Gann et al., 2019) can be applied. The saltworks is a semi-natural ecosystem and would thus fall into the "urban conservation areas and green space" category in the Standards. As such, the minimum recommended performance standard is a 3-star level for biological features. However, the saltworks has acted as a functional ecosystem since the early 1960s, lies in a Critical Biodiversity Area and is an important component of a global IBA. Therefore, it is recommended that the project aims to achieve a minimum level of 4-stars for as many of the ecosystem attributes as possible. The Ecological Recovery Wheel as applied to the site in its current condition, as well as a blank version for future evaluations, has been supplied in Appendix 8.

Table 7.5 SER's Five-Star System adapted for the rehabilitation of the abandoned saltworks (modified from Gann et al., 2019).

Attribute	★	★★	★★★	★★★★	★★★★★
Absence of threats	Restoration of the hydrology of the saltworks has commenced	The saltpans have received some water and the threat of desiccation has decreased	The saltpans have been filled and there is no threat of desiccation	The saltpans have been filled and there is no threat of desiccation or HABs. Any additional threats have been identified.	The saltpans have been filled and there is no threat of desiccation or HABs. Any additional threats have been identified and successfully addressed.
Physical conditions	The hydrology of the saltworks has been restored	The saltworks has been filled with water and target water levels have been achieved (Table 7.4)	Water level targets achieved and physico-chemical properties (particularly salinity) and nutrient concentrations in the saltworks are beginning to stabilise	Water level targets achieved and physico-chemical properties and nutrient concentrations in the saltworks have stabilised	Water level targets achieved, physico-chemical properties and nutrient concentrations have remained stable and it is evident that this state can support desirable species and ecological processes
Species composition	Primary producers and invertebrates have begun establishing at the saltworks and some waterbirds (≤ 5%) have returned to the site	More functional groups of primary producers and invertebrates are present and waterbird numbers have increased to roughly 20% of historic levels	There is evidence that primary producer and invertebrate communities have continued to diversify, and waterbird numbers have returned to roughly 40% of historic levels	High diversity of primary producers and invertebrates (similar to low and medium salinity ponds of functional saltworks), and waterbird numbers have returned to roughly 60% of historic levels. There is little threat of harmful algal blooms	High diversity of primary producers and invertebrates (similar to low and medium salinity ponds of functional saltworks) and waterbirds (≥ 80% of historic levels) present at the saltworks. There is no threat of harmful algal blooms.

Attribute	★	★★	★★★	★★★★	★★★★★
Structural diversity	The hydrology of the saltworks has been restored, providing some opportunity for aquatic primary producers to establish and waterbirds to utilise the area.	Primary producers begin establishing at the saltworks and have the potential to support higher trophic levels	Primary producers present in the water column and the benthos and salinity gradient is evident	A variety of functional groups of primary producers present in both the water column and the benthos and salinity gradient is evident	A number of habitats are present at the saltworks, ranging from brackish to hypersaline each with distinct primary producer communities
Ecosystem functionality	The restoration of the hydrology has commenced, providing some habitat value to waterbirds	The saltworks function as a waterbird habitat and show potential for effective nutrient cycling	Habitat provision has increased and some nutrient cycling is evident	The saltworks is successful at providing a habitat to fauna (particularly waterbirds) and is effective at removing nutrients from the water column	It is evident that the hydrology of the saltworks can be maintained, resulting in a resilient, self-sufficient ecosystem that is successful at both habitat provision and nutrient cycling
External exchanges	Exchanges of water between the saltworks and estuary has commenced, and the saltworks has become a suitable foraging habitat for migrant waterbirds	Potential for increased connectivity with the adjacent estuary evident and evidence of migrant waterbirds at the site	Increased connectivity with the estuary starting to become more evident and migrant waterbird numbers have increased	High level of connectivity with potential to avoid undesirable disturbances (e.g. HABs) and an abundance of migrant waterbirds present	High level of connectivity and migrant waterbird numbers that can evidently persist over the long term

8. Conclusion

This research investigated how to recreate a wetland at an abandoned saltworks. Since the saltpans are no longer actively being filled with water, the once rich and diverse waterbird communities have not returned. The saltworks is currently characterised by large areas of bare hypersaline sediment with little vegetation cover. Vegetation distribution is being limited by the high sediment salinity, which will likely not decrease if the site does not get flushed by water. However, vegetation would not cover the saltpans even if the hydrology of the saltworks is reinstated as the supratidal salt marsh species occurring there are sensitive to prolonged inundation. Nonetheless, the site holds little ecological value in its current state and the reinstatement of an artificial hydrological regime to recreate the important waterbird habitat is strongly recommended.

To reinstate the hydrology of the saltworks, estuary water can be pumped into all of the saltpans, or stormwater can be used to fill the Redhouse saltpan while the Bar None saltpans are filled with estuary water. Both water sources are rich in inorganic nutrients, but a nanocosm experiment showed that primary producers could effectively assimilate these nutrients. The use of both stormwater and estuary water resulted in an initial phytoplankton bloom that made the water column turbid. However, the bloom collapsed after rapidly depleting the water column of nutrients and a clear-water regime was reached with primary production driven by benthic communities (submerged macrophytes and MPB). The saltpans would have to be repeatedly filled with nutrient-rich water, so there may be intermittent transitions between turbid phytoplankton-driven regimes and clear water benthically-driven regimes, especially if the estuary water is used.

This research did not indicate that the use of stormwater could cause undesirable conditions (i.e. eutrophication and prolonged algal blooms) at the rehabilitated saltworks. Furthermore, no evidence suggested that HABs would proliferate if either water source is used. However, filling the Redhouse saltpan with stormwater would substantially cut running costs as estuary water would not have to be actively pumped into this large area. It is therefore recommended that the option of filling the Redhouse saltpan with stormwater and the Bar None saltpans with estuary water.

The findings of this research were used to develop a plan for implementing and monitoring the rehabilitation of the saltworks. Ecological targets have been stipulated, but the rehabilitation project will create a novel semi-natural ecosystem that may have unpredictable ecological responses. Regular monitoring will be necessary to establish a baseline for the rehabilitated site, which will allow for adaptive management and the adjustment of ecological targets if necessary.

This thesis did not cover aspects of stakeholder engagement and responsibilities, legal processes surrounding the rehabilitation project (such as lease of the site or any environmental authorisations that may be required) and an analysis of the logistics surrounding the rehabilitation activities. These details should be clarified prior to any rehabilitation actions take place. Further research is needed to provide accurate hydrological budget models for the two rehabilitation options (the use of stormwater and estuary water) and a plan to control access to the area after it has been rehabilitated. Once the rehabilitation activities have commenced, it will be important to keep comprehensive records of all relevant information and distribute these among the stakeholders. This will allow for effective adaptive management of the site to negate any unintended impacts and to maximise the benefits of this project.

Rehabilitating the saltworks will provide various opportunities for research and the generation of new knowledge. This would be the first recorded case of the rehabilitation of a saltworks in South Africa and possibly the first case of transforming a hypersaline environment into a stormwater retention pond. Filling the Redhouse saltpan with stormwater would result in the creation of a novel ecosystem and provide research opportunities, such as investigating the structure of biotic communities arising from salinised stormwater environments (Olding, 2000; Van Meter et al., 2011) and gaining a better understanding of biogeochemical cycling in hypersaline environments (Isaji et al., 2019). Furthermore, the creation of several wetlands with different salinity levels and nutrient concentrations will provide a study site suitable for further investigating thresholds for ecological regime shifts in wetlands in response to changes in salinity and/or nutrient availability (Sim et al., 2006; Herbert et al., 2015). Economic opportunities such as aquaculture, ecotourism, or using the site in a desalination scheme should also be considered as means to fund the running costs of managing the rehabilitated saltworks.

The rehabilitation of the saltworks, if implemented successfully, will highlight the potential to turn otherwise detrimental land use changes into opportunities for maximising the provision of ecosystem services. Restoring the wetland function of the site will recreate a highly important waterbird habitat with local benefits such as improving the overall health of the nationally important Swartkops Estuary and will help alleviate pressures on migratory waterbirds by provisioning a valuable stopover site for various such species. Furthermore, using stormwater to fill the Redhouse saltpan would be a demonstration of multipurpose restoration that addresses habitat provision as well as water quality regulation. Such innovative ecological restoration approaches will become increasingly relevant globally as the human population continues to grow and global change and the utilisation of natural resources escalate.

9. References

Abid O, Sellami-Kammoun A, Ayadi H, Drira Z, Bouain A, Aleya L. 2008. Biochemical adaptation of phytoplankton to salinity and nutrient gradients in a coastal solar saltern, Tunisia. *Estuarine, Coastal and Shelf Science* 80, 391 – 400.

Adams JB, Bate GC. 1994a. The ecological implications of tolerance to salinity by *Ruppia cirrhosa* (Petagna) Grande and *Zostera capensis* Setchell. *Botanica Marina* 37, 449 – 456.

Adams JB, Bate GC. 1994b. The tolerance to desiccation of the submerged macrophytes *Ruppia cirrhosa* (Petagna) Grande and *Zostera capensis* Setchell. *Journal of Experimental Marine Biology and Ecology* 183, 53 – 62.

Adams JB, Fernandes M, Riddin T. 2019a. Chapter 5: Estuarine habitat status and trends. South African National Biodiversity Assessment 2018: Technical Report. Volume 3: Estuarine Realm. South African National Biodiversity Institute, Pretoria. Report Number: SANBI/NAT/NBA2018/2019/Vol3/A. p. 52 – 74.

Adams JB, Pretorius L, Snow GC. 2019b. Deterioration in the water quality of an urbanised estuary with recommendations for improvement. *Water SA* 45, 86 – 96.

Adams JB, Veldkornet D, Tabot P. 2016. Distribution of macrophyte species and habitats in South African estuaries. *South African Journal of Botany* 107, 5 – 11.

Adams JB, Whitfield AK, van Niekerk L. 2020. A socio-ecological systems approach towards future research for the restoration, conservation and management of southern African estuaries. *African Journal of Aquatic Science* 45, 231 – 241.

Adams JB. 2020. Salt marsh at the tip of Africa: Patterns, processes and changes in response to climate change. *Estuarine, Coastal and Shelf Science* 237, 106650.

Allanson BR, Baird D (Eds.). 1999. *Estuaries of South Africa*. Cambridge University Press, UK. 356 pp.

Almeida D, Neto C, Esteves LS, Costa JC. 2014. The impacts of land-use changes on the recovery of saltmarshes in Portugal. *Ocean and Coastal Management* 92, 40 – 49.

Anderson DM, Glibert PM, Burkholder JM. 2002. Harmful algal blooms and eutrophication: nutrient sources, composition, and consequences. *Estuaries* 25, 704 – 726.

Anderson JT, Smith LM. 1999. Carrying capacity and diel use of managed playa wetlands by nonbreeding waterbirds. *Wildlife Society Bulletin* 27, 281 – 291.

Atia A, Rabhi M, Debez A, Barhoumi Z, Abdelly C, Smaoui A. 2011. Factors controlling germination and dormancy processes in dimorphic fruits of *Atriplex inflata* (Chenopodiaceae). *Plant Ecology and Evolution* 144, 307 – 312.

Ayadi H, Abid O, Elloumi J, Bouaïn A, Sime-Ngando T. 2004. Structure of the phytoplankton communities in two lagoons of different salinity in the Sfax saltern (Tunisia). *Journal of Plankton Research* 26, 669 – 679.

Baird D, Hanekom NM, Grindley JR. 1986. Estuaries of the Cape. Part II. Synopses of available information on individual systems – Report No. 23 Report Swartkops (CSE3). CSIR Research Report No. 422. CSIR, Stellenbosch, South Africa. 89 pp.

Baird D. 2001. Estuaries of South Africa. In: Dupra, V., Smith, S.V., Marshall Crossland, J.I. and Crossland, C.J. (Eds.). Estuarine Systems of Sub-Saharan Africa: Carbon, Nitrogen and Phosphorous Fluxes. LOICZ Reports and Studies No. 18. LOICZ, Texel. The Netherlands. 83 pp.

Baldwin DS, Rees GN, Mitchell AM, Watson G, Williams J. 2006. The short-term effects of salinization on anaerobic nutrient cycling and microbial community structure in sediment from a freshwater wetland. *Wetlands* 26, 455 – 464.

Barbier EB, Hacker SD, Kennedy C, Koch EW, Stier AC, Silliman BR. 2011. The value of estuarine and coastal ecosystem services. *Ecological Monographs* 81, 169 – 193.

Barnard RO. 1990. *Handbook of standard soil testing methods for advisory purposes*. Soil Science Society of South Africa, Pretoria. p 35.

Bate GC, Heelas BV. 1975. Studies on the nitrate nutrition of two indigenous Rhodesian grasses. *Journal of Applied Ecology* 12, 941 – 952.

Bate GC, Taylor RH. 2008. Sediment salt-load in the St Lucia Estuary during the severe drought of 2002–2006. *Environmental Geology* 55, 1089 – 1098.

Belanger SE. 1997. Literature review and analysis of biological complexity in model stream ecosystems: Influence of size and experimental design. *Ecotoxicology and Environmental Safety* 36, 1 – 16.

Benton TG, Solan M, Travis JM, Sait SM. 2007. Microcosm experiments can inform global ecological problems. *Trends in Ecology and Evolution* 22, 516 – 521.

Binning KA. 1999. An investigation into the relationship between water quality, sediment, and meiofaunal communities of the Swartkops River ecosystem. MSc Thesis. University of Port Elizabeth, South Africa.

BirdLife International. No date a. Important Bird Areas factsheet: Swartkops Estuary – Redhouse and Chatty Saltpans. Retrieved from: http://datazone.birdlife.org/site/factsheet/swartkops-estuary--redhouse-and-chatty-saltpans-iba-south-africa. Accessed 7 May 2019.

BirdLife International. No date b. Saving saltpans: A crucial habitat for migratory birds. Retrieved from: http://www.birdlife.org/sites/default/files/vbn_savingsaltpans_brochure_english_mail_0.pdf. Accessed 19 May 2020.

Borja A, Dauer DM, Elliott M, Simenstad CA. 2010. Medium-and long-term recovery of estuarine and coastal ecosystems: patterns, rates and restoration effectiveness. *Estuaries and Coasts* 33, 1249 – 1260.

Borja A. 2014. Grand challenges in marine ecosystems ecology. *Frontiers in Marine Science* 1, 1.

Bornman TG, Adams JB, Bate GC. 2004. The influence of floodplain geohydrology on the distribution of *Sarcocornia pillansii* in the Olifants Estuary on the West Coast, South Africa. *Journal of Arid Environments* 56, 603 – 625.

Bouzillé JB, Kernéis E, Bonis A, Touzard B. 2001. Vegetation and ecological gradients in abandoned salt pans in western France. *Journal of Vegetation Science* 12, 269 – 278.

Briggs D. 1977. *Sources and Methods in Geography*. Butterworths, London. 192 pp.

Brito A, Newton A, Tett P, Fernandes T. 2009. Development of an optimal methodology for the extraction of microphytobenthic chlorophyll. *Journal of International Environmental Application and Science* 4, 42 – 54.

Britton RH, Johnson AR. 1987. An ecological account of a Mediterranean salina: the Salin de Giraud, Camargue (S. France). *Biological Conservation* 42, 185 – 230.

Brooks ME, Kristensen K, van Benthem KJ, Magnusson A, Berg CW, Nielsen A, Skaug HJ, Maechler M, Bolker BM. 2017. glmmTMB balances speed and flexibility among packages for zero-inflated generalized linear mixed modelling. *The R Journal* 9, 378 – 400.

Brownlie SF. 1988. Restoration of degraded estuaries. Environmental Evaluation Unit, University of Cape Town. Report 8/88/32.

Bruschetti CM, Addino M, Luppi T, Iribarne O. 2018. Effects of nutrient enrichment and grazing by an invasive filter feeder on phytoplankton biomass in a South West Atlantic coastal lagoon. *Biological Invasions* 20, 2245 – 2256.

Bruschetti M, Bazterrica C, Luppi T, Iribarne O. 2009. An invasive intertidal reef-forming polychaete affect habitat use and feeding behaviour of migratory and local birds in a SW Atlantic coastal lagoon. *Journal of Experimental Marine Biology and Ecology* 375, 76 – 83.

Bruschetti M, Luppi T, Fanjul E, Rosenthal A, Iribarne O. 2008. Grazing effect of the invasive reef-forming polychaete *Ficopomatus enigmaticus* (Fauvel) on phytoplankton biomass in a SW Atlantic coastal lagoon. *Journal of Experimental Marine Biology and Ecology* 354, 212 – 219.

Cairns Jr J. 2000. Setting ecological restoration goals for technical feasibility and scientific validity. *Ecological Engineering* 15, 171-180.

Chefaoui RM, Chozas S. 2019. Abandonment of traditional saltworks facilitates degradation of halophytic plant communities and *Carpobrotus edulis* invasion. *Applied Vegetation Science* 22, 444 – 453.

Chien Y-H. 1992. Marine shrimp pond management: a review. In: Wybanm J. (Ed.) *Proceedings of the Special Session on Shrimp Farming*. World Aquaculture Society, Baton Rouge, Louisiana. pp. 100 – 137.

Choi YD, Temperton VM, Allen EB, Grootjans AP, Halassy M, Hobbs RJ, Naeth MA, Torok K. 2008. Ecological restoration for future sustainability in a changing environment. *Ecoscience* 15, 53 – 64.

Choi YD. 2004. Theories for ecological restoration in changing environment: Toward 'futuristic' restoration. *Ecological Research* 19, 75 – 81.

Choi YD. 2007. Restoration ecology to the future: a call for new paradigm. *Restoration Ecology* 15, 351 – 353.

Cilliers GJ, Adams JB. 2016. Development and implementation of a monitoring programme for South African estuaries. *Water SA* 42, 279 – 290.

Claassens L, Adams JB, Wasserman J, Whitfield AK. Under review. Estuary Restoration – A South African Perspective. Waltham N (Ed.). In: *Repairing the World's Estuaries: Knowledge, Challenges and Possible Way Forward*. Elsevier, Amsterdam.

Claassens L, Barnes RSK, Wasserman J, Lamberth SJ, Miranda NAF, van Niekerk L, Adams JB. 2020. Knysna Estuary health: ecological status, threats and options for the future. *African Journal of Aquatic Science* 45, 65 – 82.

Clavero V, García M, Fernández JA, Niell FX. 1993. Adsorption-desorption of phosphate and its availability in the sediment of a saline lake (Fuente de Piedra, southern Spain). *International Journal of Salt Lake Research* 2, 153 – 163.

Clewell A, Rieger JP. 1997. What practitioners need from restoration ecologists. *Restoration Ecology* 5, 350 – 354.

Cloern JE, Jassby AD. 2008. Complex seasonal patterns of primary producers at the land-sea interface. *Ecological Letters* 11, 1294 – 1303.

Cloern JE. 2001. Our evolving conceptual model of the coastal eutrophication problem. *Marine Ecology Progress Series* 210, 223 – 253.

Cochlan WP, Trainer VL, Trick CG, Wells ML, Bill BD, Eberhart BTL. 2013. *Heterosigma akashiwo* in the Salish Sea: defining growth and toxicity leading to fish kills. In *Proceedings of the 15th International Conference on Harmful Algae*.

Collins KA, Lawrence TJ, Stander EK, Jontos, RJ, Kaushal SS, Newcomer TA, Grimm NB, Ekberg MLC. 2010. Opportunities and challenges for managing nitrogen in urban stormwater: A review and synthesis. *Ecological Engineering* 36, 1507 – 1519.

Copeland BJ, Nixon SW. 1974. Hypersaline lagoons. *Coastal Ecological Systems of the United States* 1, 312 – 330.

Copeland BJ. 1967. Environmental characteristics of hypersaline lagoons. *University of Texas Contributions in Marine Science* 12, 207 – 218.

Costa DFDS, Barbosa JEDL, Soares AMVDM, Lillebø AI, Rocha RDM. 2015. Spatial modelling of limnological parameters in a solar saltwork of northeastern Brazil. *Acta Limnologica Brasiliensia* 27, 105 – 117.

Costanza R, d'Arge R, De Groot R, Farber S, Grasso M, Hannon B, Limburg K, Naeem S, O'neill RV, Paruelo J, Raskin RG. 1997. The value of the world's ecosystem services and natural capital. N*ature* 387, 253 – 260.

Cotner JB, Suplee MW, Chen NW, Shormann DE. 2004. Nutrient, sulfur and carbon dynamics in a hypersaline lagoon. *Estuarine, Coastal and Shelf Science* 59, 639 – 652.

Coulon C, Alexander V. 1972. A sliding-chamber phytoplankton settling technique for making permanent quantitative slides with applications in fluorescent microscopy and autoradiography. *Limnology and Oceanography* 17, 149 – 152.

Crawford RJM, Whittington PA, Martin AP, Tree AJ, Makhado AB. 2009. Population trends of seabirds breeding in South Africa’s Eastern Cape, and the possible influence of anthropogenic and environmental change. *Marine Ornithology* 37, 159–174.

Crisman TL, Takavakoglou V, Alexandridis T, Antonopoulos V, Zalidis G. 2009. Rehabilitation of abandoned saltworks to maximize conservation, ecotourism and water treatment potential. *Global NEST Journal* 11, 24 – 31.

CWAC (Coordinated Waterbird Counts). No date a. Bar None Saltpan (33502533). Retrieved from: http://cwac.birdmap.africa/cards_individual.php?card=501389. Accessed 18 November 2019.

CWAC (Coordinated Waterbird Counts). No date b. Redhouse saltpan (33502535). Retrieved from: http://cwac.birdmap.africa/sites.php?sitecode=33502535. Accessed 18 November 2019.

Davies BR, Stuart V, de Villiers M. 1989. The filtration activity of a serpulid polychaete population (*Ficopomatus enigmaticus* (Fauvel) and its effects on water quality in a coastal marina. *Estuarine, Coastal and Shelf Science* 29, 613 – 620.

Davis JA, McGuire M, Halse SA, Hamilton D, Horwitz P, McComb AJ, Froend R, Lyons M, Sim L. 2003. What happens when you add salt: predicting impacts of secondary salinisation on shallow aquatic ecosystems using an alternative states model. *Australian Journal of Botany* 51, 715 – 724.

Davis JS, Giordano M. 1996. Biological and physical events involved in the origin, effects, and control of organic matter in solar saltworks. *International Journal of Salt Lake Research* 4, 335 – 347.

Davis JS. 1978. Biological communities of a nutrient enriched salina. *Aquatic Botany* 4, 23 - 42.

Davis JS. 1990. Biological management for the production of salt from sea-water. Akutsuka I (Ed.). In: *Introduction to Applied Phycology*. SPB Academic Publishing. pp. 479 – 488.

Davis JS. 1993. Biological management for problem solving and biological concepts for a new generation of solar saltworks. *Seventh Symposium on Salt* 1, 611 – 616.

Davis JS. 1999. Solar Saltworks–An environmentally friendly industry. In Proceedings of the post conference symposium on saltworks: preserving saline coastal ecosystems. Global Network for Environmental Science and Technology, Pythagorion, Samos, Greece. pp. 31 – 37.

Davis JS. 2000a. Structure, function, and management of the biological system for seasonal solar saltworks. *Global NEST Journal* 2, 217 – 226.

Davis JS. 2009. Management of biological systems for continuously-operated solar saltworks. *Global NEST Journal* 11, 73 – 78.

Davis MA. 2000b. ‘‘Restoration’’ — a misnomer? *Science* 287, 1203.

Day JW, Hall AS, Kemp WM, Yáñez-Arancibia A (Eds.). 1989. *Estuarine Ecology*. Wiley, New York. 558 pp.

de Melo Soares RHR, de Assunção CA, de Oliveira Fernandes F, Marinho-Soriano E. 2018. Identification and analysis of ecosystem services associated with biodiversity of saltworks. *Ocean and Coastal Management* 163, 278 – 284.

Des Marais DJ. 1995. The biogeochemistry of hypersaline microbial mats. In: *Advances in microbial ecology*. Springer, Boston, MA. pp. 251 – 274.

Dethier MN, Hacker SD. 2005. Physical factors vs. biotic resistance in controlling the invasion of an estuarine marsh grass. *Ecological Applications* 15*,* 1273 – 1283.

Difford M. 2008. Towards the Bioremediation of the Hypertrophic Swartkops Solar Salt-works. PhD thesis. Nelson Mandela Metropolitan University, South Africa. 318 pp.

Dittmann S, Mosley L, Clanahan M, Quinn J, Crooks S, Emmer I, Min Thomson S, Seaman R. 2019. Proof of concept for tidal re-connection as a blue carbon offset project. *Goyder Institute for Water Research, Technical Report Series*, (19/29). 36 pp.

Dittmann S, Rolston A, Benger SN, Kupriyanova EK. 2009. Habitat requirements, distribution and colonisation of the tubeworm *Ficopomatus enigmaticus* in the Lower Lakes and Coorong. Report for the South Australian Murray-Darling Basin Natural Resources Management Board, Adelaide. 99 pp.

DMR (Department of Mineral Resources). 2011. Structure of the salt industry in the Republic of South Africa, 2011. Report R93/2011. 21 pp.

Dolapsakis NP, Tafas T, Abatzopoulos TJ, Ziller S, Economou-Amilli A. 2005. Abundance and growth response of microalgae at Megalon Embolon solar saltworks in northern Greece: An aquaculture prospect. *Journal of Applied Phycology* 17, 39 – 49.

Dos Reis-Neto AS, Meireles AJ de A, Cunha-Lignon M. 2013. Analyses of the mangrove's rehabilitation process in abandoned saltwork area, in the Ceará river, Northeast Brazil. *International Journal of Applied and Natural Sciences* 2, 27 – 36.

Dos Reis-Neto AS, Meireles AJ de A, Cunha-Lignon M. 2019. Natural regeneration of the mangrove vegetation on abandoned salt ponds in Ceará, in the semi-arid region of northeastern Brazil. *Diversity* 11, 27.

Du Toit SR, Campbell EE. 2002. An analysis of the performance of an artificial wetland for nutrient removal in solar saltworks. *South African Journal of Botany* 68, 451 – 456.

Du Toit SR. 2001. Biological management of South African solar saltworks. PhD thesis. University of Port Elizabeth, South Africa. 260 pp.

Duarte CM, Borja A, Carstensen J, Elliott M, Krause-Jensen D, Marbà N. 2015. Paradigms in the recovery of estuarine and coastal ecosystems. *Estuaries and Coasts* 38, 1202 – 1212.

Duarte CM, Conley DJ, Carstensen J, Sánchez-Camacho M. 2009. Return to Neverland: shifting baselines affect eutrophication restoration targets. *Estuaries and Coasts* 32, 29 – 36.

DWAF (Department of Water Affairs and Forestry). 1996. South African Water Quality Guidelines (first edition). Volume 8: Aquatic Ecosystems.

DWAF (Department of Water Affairs and Forestry, South Africa). 1999. Zwartkops River water resources management plan: Situation assessment and development of a catchment water quality monitoring programme. DWAF Draft Report No. N/M100/ REQ/0896. DWAF, Pretoria.

Ehrenfeld JG, Toth LA. 1997. Restoration ecology and the ecosystem perspective. *Restoration Ecology* 5, 307 – 317.

Ellis E. 2019. Eastern Cape: 'The worst drought in a thousand years'. Maverick Citizen. Retrieved from: https://www.dailymaverick.co.za/article/2019-10-10-eastern-cape-the-worst-drought-in-a-thousand-years/. Accessed 14 March 2020.

Elloumi J, Carrias JF, Ayadi H, Sime-Ngando T, Bouaïn A. 2009. Communities structure of the planktonic halophiles in the solar saltern of Sfax, Tunisia. *Estuarine, Coastal and Shelf Science* 81, 19 – 26.

Emmerson WD. 1985. The nutrient status of the Swartkops River estuary, Eastern Cape. *Water SA* 11, 189 – 198.

Enviro-Fish Africa. 2009. Swartkops Integrated Environmental Management Plan: Volume 1 – Situation Assessment. C.A.P.E. Estuaries Management Programme. 110 pp.

Enviro-Fish Africa. 2011. Integrated Management Plan: Swartkops Estuary and the Swartkops River valley and Aloes nature reserves. Volume 2 – Final Report. C.A.P.E. Estuaries Management Programme. 90 pp.

ESRI. 2018. ArcGIS Desktop: Release 10.6.1.9270. Redlands, CA: Environmental System Research Institute.

Evagelopoulos A, Spyrakos E, Koutsoubas D. 2009. Phytoplankton and macrofauna in the low salinity ponds of a productive solar saltworks: spatial variability of community structure and its major abiotic determinants. *Global NEST Journal* 11, 64 – 72.

Ferreira JG, Andersen JH, Borja A, Bricker SB, Camp J, Da Silva MC, Garcés E, Heiskanen AS, Humborg C, Ignatiades L, Lancelot C. 2011. Overview of eutrophication indicators to assess environmental status within the European Marine Strategy Framework Directive. *Estuarine, Coastal and Shelf Science* 93,117 – 131.

Fisher J, Acreman MC. 2004. Wetland nutrient removal: a review of the evidence. *Hydrology and Earth System Sciences* 8, 673 – 685.

Fong P, Donohoe RM, Zedler JB. 1993. Competition with macroalgae and benthic cyanobacterial mats limits phytoplankton abundance in experimental microcosms. *Marine Ecology Progress Series* 100, 97 – 102.

Galatowitsch SM. 2012. *Ecological restoration*. Sunderland: Sinauer associates. 630 pp.

Gann GD, McDonald T, Walder B, Aronson J, Nelson CR, Jonson J, Hallet JG, Eisenberg C, Guariguata MR, Liu J, Hua F, Echeverría C, Gonzales E, Shaw N, Decleer K, Dixon KW. 2019. International principles and standards for the practice of ecological restoration. Second edition. *Restoration Ecology* 27(S1), S1 – S46.

Gardner WH. 1965. Soil water content. In: Black, C.A., Evans, D.D., White JL, Ensminger LE, Clark FE, Dinauer RC (Eds.). *Methods of Soil Analysis. Part I. Agronomy* Vol. 9. American Society of Agronomy, Inc., Madison, Wisconsin. pp. 82 – 83.

Garret DE. 1966. Factors in the design of solar salt plants, Part 2. Optimum operation of solar ponds. In: Rau JL. (Ed.). *Symposium on salt*. pp. 176 – 181.

Gee GW, Bauder JW. 1986. Particle size analysis. In: Klut A. (Ed.). *Methods of soil analysis, Part I - Physical and mineralogical methods*. *Second edition*. Agronomy Number 9. Part1. Madison, Wisconsin, USA. pp. 383 – 411.

Glenn EP, Nelson SG, Ambrose B, Martinez R, Soliz D, Pabendinskas V, Hultine K. 2012. Comparison of salinity tolerance of three *Atriplex* spp. in well-watered and drying soils. *Environmental and Experimental Botany* 83, 62 – 72.

Gold AC, Thompson SP, Piehler MF. 2019. Nitrogen cycling processes within stormwater control measures: A review and call for research. *Water Research* 149, 578 – 587.

González-Alcaraz MN, Jiménez-Cárceles FJ, Álvarez Y, Álvarez-Rogel J. 2014. Gradients of soil salinity and moisture, and plant distribution, in a Mediterranean semiarid saline watershed: a model of soil–plant relationships for contributing to the management. *Catena* 115, 150 – 158.

Google. 2020. Google Earth Pro (version 7.3.3.7786). Available from: https://www.google.com/earth/.

Goschen WE, MacKay HM. 1992. Hydrodynamics of the Swartkops Estuary. SASAQS 1991 Conference on Aquatic Ecosystems, 7 - 10 July, Cape Town.

Haigh L. 2002. Catchment M, the Swartkops River. Report presented to the Department of Water Affairs and Forestry, Eastern Cape, South Africa.

Halpern BS, Frazier M, Potapenko J, Casey KS, Koenig K, Longo C, Lowndes JS, Rockwood RC, Selig ER, Selkoe KA, Walbridge S. 2015. Spatial and temporal changes in cumulative human impacts on the world’s ocean. *Nature Communications* 6, 1 – 7.

Halpern BS, Walbridge S, Selkoe KA, Kappel CV, Micheli F, D'agrosa C, Bruno JF, Casey KS, Ebert C, Fox HE, Fujita R. 2008. A global map of human impact on marine ecosystems. *Science* 319, 948 – 952.

Hanekom NM, Baird D, Erasmus T. 1988. A quantitative study to assess standing biomasses of macrobenthos in soft substrata of the Swartkops Estuary, South Africa. *South African Journal of Marine Science* 6, 163 – 174.

Harris JA, van Diggelen R. 2006. Ecological restoration as a project for global society. In: van Andel J, Aronson J (Eds.). *Restoration Ecology: The New Frontier*. Blackwell Science Ltd., Oxford, UK.

Heiman K, Vidargas N, Micheli F. 2008. Non-native habitat as home for non-native species: Comparison of communities associated with invasive tubeworm and native oyster reefs. *Aquatic Biology* 2, 47 – 56.

Herbert ER, Boon P, Burgin AJ, Neubauer SC, Franklin RB, Ardón M, Hopfensperger KN, Lamers LP, Gell P. 2015. A global perspective on wetland salinization: ecological consequences of a growing threat to freshwater wetlands. *Ecosphere* 6, 1 – 43.

Hewitson C. 2017. The Lion Salt Works, Northwich: a legacy of the Cheshire salt industry. *Industrial Archaeology Review* 39, 59 – 75.

HKS (Hill, Kaplan, Scott and Partners). 1974. Environmental Study, Swartkops River Basin. Part 2: Hydrology, hydraulics. Technical Data Report published by the Port Elizabeth City Engineer. 24 pp.

Hobbs RJ, Norton DA. 1996. Toward a conceptual framework for restoration ecology. *Restoration Ecology* 4, 93 – 110.

Hobbs RJ. 2007. Setting effective and realistic restoration goals: key directions for research. *Restoration Ecology* 15, 354 – 357.

Howell EA, Harrington JA, Glass SB. 2012. *Introduction to Restoration Ecology*. Island Press, Washington D.C. 418 pp.

Human LRD, Adams JB, Allanson BR. 2016. Insights into the cause of an *Ulva lactuca* Linnaeus bloom in the Knysna Estuary. *South African Journal of Botany* 107, 55 – 62.

Human LRD, Snow GC, Adams JB, Bate, GC. 2015. The benthic regeneration of N and P in the Great Brak estuary, South Africa. *Water SA* 41, 594 – 605.

Isaji Y, Kawahata H, Ogawa NO, Kuroda J, Yoshimura T, Jiménez-Espejo FJ, Makabe A, Shibuya T, Lugli S, Santulli A, Manzi V. 2019. Efficient recycling of nutrients in modern and past hypersaline environments. *Scientific Reports* 9, 1 – 12.

Javor BJ. 1983. Planktonic standing crop and nutrients in a saltern ecosystem. *Limnology and Oceanography* 28, 153 – 159.

Javor BJ. 1989. *Hypersaline environments. Microbiology and biogeochemistry*. Springer-Verlag, Berlin. 328 pp.

Jennerjahn TC, Mitchell SB. 2013. Pressures, stresses, shocks and trends in estuarine ecosystems – An introduction and synthesis. *Estuarine, Coastal and Shelf Science* 130, 1 – 8.

Joint I, Henriksen P, Garde K, Riemann B. 2002. Primary production, nutrient assimilation and microzooplankton grazing along a hypersaline gradient. *FEMS Microbiology Ecology* 39, 245 – 257.

Jolly ID, Walker GR, Thorburn PJ. 1993. Salt accumulation in semi-arid floodplain soils with implications for forest health. *Journal of Hydrology* 150, 589 – 614.

Jones HP, Jones PC, Barbier EB, Blackburn RC, Rey Benayas JM, Holl KD, McCrackin M, Meli P, Montoya D, Mateos DM. 2018. Restoration and repair of Earth's damaged ecosystems. *Proceedings of the Royal Society B: Biological Sciences* 285, 20172577.

Jones HP, Schmitz OJ. 2009. Rapid recovery of damaged ecosystems. *PLOS One* 4, e5653.

Khan S, Arakawa O, Onoue Y. 1997. Neurotoxins in a toxic red tide of *Heterosigma akashiwo* (Raphidophyceae) in Kagoshima Bay, Japan. *Aquaculture Research* 28, 9 – 14.

Khemakhem H, Elloumi J, Moussa M, Aleya L, Ayadi H. 2010. The concept of ecological succession applied to phytoplankton over four consecutive years in five ponds featuring a salinity gradient. *Estuarine, Coastal and Shelf Science* 88, 33 – 44.

Kingsford RT, Porter JL. 1994. Waterbirds on an adjacent freshwater lake and salt lake in arid Australia. *Biological Conservation* 69, 219 – 228.

Kirby JS, Stattersfield AJ, Butchart SH, Evans MI, Grimmett RF, Jones VR, O'Sullivan J, Tucker GM, Newton I. 2008. Key conservation issues for migratory land-and waterbird species on the world's major flyways. *Bird Conservation International* 18(S1), S49 – S73.

Kirst GO. 1990. Salinity tolerance of eukaryotic marine algae. *Annual Review of Plant Biology* 41, 21 – 53.

Klages N, Jegels J, Schovell I, Vosloo M. 2011. Nelson Mandela Bay Municipality State of the Environment Report, Repared by Arcuss GIBB (Pty) Ltd for Nelson Mandela Bay Municipality.

Koch BJ, Febria CM, Gevrey M, Wainger LA, Palmer MA. 2014. Nitrogen removal by stormwater management structures: A data synthesis. *Journal of the American Water Resources Association* 50, 1594 – 1607.

Korovessis NA, Lekkas TD. 2009. Solar saltworks' wetland function. *Global NEST Journal* 11, 49 – 57.

Kristoffersen AS, Hamre B, Frette Ø, Erga SR. 2016. Chlorophyll a fluorescence lifetime reveals reversible UV-induced photosynthetic activity in the green algae *Tetraselmis*. *European Biophysics Journal* 45, 259 – 268.

Lai F. 2013. Nature and the city: the salt-works park in the urban area of Cagliari (Sardinia, Italy). *Journal of Political Ecology* 20, 329 – 341.

Lakane CP. 2020. Investigating the nutrient removal of efficiency of the Motherwell Artificial Wetland before discharging to the Swartkops Estuary. Honours thesis. Nelson Mandela University, South Africa.

Laurance WF. 2001. Future shock: forecasting a grim fate for the Earth. *Trends in Ecology & Evolution* 16, 531 – 533.

Le HD, Smith C, Herbohn J, Harrison S. 2012. More than just trees: assessing reforestation success in tropical developing countries. *Journal of Rural Studies* 28, 5 – 19.

Lemley DA, Adams JB, Bornman TG, Campbell EE, Deyzel SH. 2019. Land-derived inorganic nutrient loading to coastal waters and potential implications for nearshore plankton dynamics. *Continental Shelf Research* 174, 1 – 11.

Lemley DA, Adams JB, Strydom NA. 2017. Testing the efficacy of an estuarine eutrophic condition index: Does it account for shifts in flow conditions? *Ecological Indicators 74*, 357 – 370.

Lemley DA, Adams JB, Taljaard S, Strydom NA. 2015. Towards the classification of eutrophic condition in estuaries. *Estuarine, Coastal and Shelf Science* 164, 221 – 232.

Lemley DA, Nunes M, Adams JB. 2018. Estuarine algal responses to increasing nitrate concentrations during closed mouth conditions of oligotrophic systems: A laboratory microcosm experiment. *Botanica Marina* 61, 559 – 572.

Lewitus AJ, Brock LM, Burke MK, DeMattio KA, Wilde SB. 2008. Lagoonal stormwater detention ponds as promoters of harmful algal blooms and eutrophication along the South Carolina coast. *Harmful Algae* 8, 60 – 65.

Litchfield CD, Irby A, Kis-Papo T, Oren A. 2000. Comparisons of the polar lipid and pigment profiles of two solar salterns located in Newark, California, USA, and Eilat, Israel. *Extremophiles* 4, 259 – 265.

López E, Aguilera PA, Schmitz MF, Castro H, Pineda FD. 2010. Selection of ecological indicators for the conservation, management and monitoring of Mediterranean coastal salinas. *Environmental Monitoring and Assessment* 166, 241 – 256.

Lotze HK, Lenihan HS, Bourque BJ, Bradbury RH, Cooke RG, Kay MC, Kidwell SM, Kirby MX, Peterson CH, Jackson JB. 2006. Depletion, degradation, and recovery potential of estuaries and coastal seas. *Science* 312, 1806 – 1809.

Luppi TA, Bas CC. 2002. The role of the invasive polychaete *Ficopomatus enigmaticus* Fauvel 1923 (Polychaeta: Serpulidae) reefs in the recruitment of *Cyrtograpsus angulatus* Dana 1851 (Brachyura: Grapsidae), in the Mar Chiquita coastal lagoon, Argentina. *Ciencias Marinas* 28, 319 – 330.

Ma Z, Cai Y, Li B, Chen J. 2010. Managing wetland habitats for waterbirds: an international perspective. *Wetlands* 30, 15 – 27.

MacKay HM. 1993. The impact of urban runoff on the water quality of the Swartkops estuary: Implications for water quality management. Stellenbosch, South Africa: Talbot and Talbot. Industrial and Environmental Management. Report to the Water Research Commission, WRC Report Nr. 324/1/93. 30 pp.

Margalef R. 1968. *Perspectives in Ecological Theory*. Chicago Series in Biology. University of Chicago. 111 pp.

Marneweck GC, Batchelor AL, Uys AC. 2004. Guidelines for Estuarine Rehabilitation. Eastern Cape Estuaries Management Research Programme Volume 2 Report F. WRC Report No. 1246/1/04. 60 pp.

Marnewick MD, Retief EF, Theron NT, Wright DR, Anderson TA. 2015. Important Bird and Biodiversity Areas of South Africa. BirdLife South Africa, Johannesburg. 130 pp.

Martin AP, Baird D. 1987. Seasonal abundance and distribution of birds on the Swartkops Estuary, Port Elizabeth. *Ostrich* 58, 122 – 134.

Martin AP, Randall RM. 1987. Numbers of waterbirds at a commercial saltpan, and suggestions for management. *South African Journal of Wildlife Research* 17, 75 – 81.

Martin P. 2020. Biannual waterbird counts at the Redhouse and Bar None saltpans as part of the Coordinated Waterbird Counts (CWAC) programme. Unpublished raw data.

McQuaid KA, Griffiths CL. 2014. Alien reef-building polychaete drives long-term changes in invertebrate biomass and diversity in a small, urban estuary. *Estuarine, Coastal and Shelf Science* 138, 101 – 106.

Millennium Ecosystem Assessment (MEA), 2005. Ecosystems and human well-being: current state and trends. 47 pp.

Mitchell SB, Jennerjahn TC, Vizzini S, Zhang W. 2015. Changes to processes in estuaries and coastal waters due to intense multiple pressures – an introduction and synthesis. *Estuarine, Coastal and Shelf Science* 156, 1 – 6.

Morris JT, Sundareshwar PV, Nietch CT, Kjerfve B, Cahoon DR. 2002. Responses of coastal wetlands to rising sea level. *Ecology* 83, 2869 – 2877.

Moss B, Stansfield J, Irvine K, Perrow M, Phillips G. 1996. Progressive restoration of a shallow lake: a 12 year experiment in isolation, sediment removal and biomanipulation. *Journal of Applied Ecology* 28, 586 – 602.

Mulderij G, Van Nes EH, Van Donk E. 2007. Macrophyte–phytoplankton interactions: the relative importance of allelopathy versus other factors. *Ecological Modelling* 204, 85 – 92.

Neill C. 1993. Seasonal flooding, soil salinity and primary production in northern prairie marshes. *Oecologia* 95, 499 – 505.

Neumann B, Vafeidis AT, Zimmermann J, Nicholls RJ. 2015. Future coastal population growth and exposure to sea-level rise and coastal flooding-a global assessment. *PLOS One* 10, e0118571.

Neves R. 2002. Interregional study on restoration of saltworks - ALAS Project. *ALAS (All about salt) — Salt and salinas in the Mediterranean*. 51 pp.

Nusch EA. 1980. Comparison of different methods for chlorophyll and phaeopigment determination. *Archiv für Hydrobiologie* 14, 14–36.

Oksanen J, Blanchet FG, Friendly M, Kidnt R, Legendre P, Minchin PR, O'hara RB, Simpson GL, Solymos P, Stevens MHM, Szoecs E, Wagner H. 2020. vegan: community ecology package. R package version 2.5-6.

Olding DD. 2000. Algal communities as a biological Indicator of Stormwater management pond performance and function. *Water Quality Research Journal* 35, 489 – 503.

Oren A, Dubinsky Z. 1994. On the red coloration of saltern crystallizer ponds. II. Additional evidence for the contribution of halobacterial pigments. *International Journal of Salt Lake Research* 3, 9 – 13.

Oren A. 2000. Salts and brines. In: Whitton BA, Potts M (Eds.). *The ecology of cyanobacteria: their diversity in time and space*. Kluwer Academic Publishers, Dordrecht, London/Boston. pp 281 – 306.

Oren A. 2002a. Diversity of halophilic microorganisms: environments, phylogeny, physiology, and applications. *Journal of Industrial Microbiology and Biotechnology* 28, 56 – 63.

Oren A. 2002b. Solar Salterns. Oren A (Ed.). *Halophilic Microorganisms and their Environments*. Kluwer Academic Publishers, Dordrecht. pp. 441 – 469.

Oren A. 2009. Saltern evaporation ponds as model systems for the study of primary production processes under hypersaline conditions. *Aquatic Microbial Ecology* 56, 193 – 204.

Oren A. 2013. Life in magnesium-and calcium-rich hypersaline environments: salt stress by chaotropic ions. Sebach J, Oren A, Stan-Lotter H (Eds.). *Polyextremophiles: Life under multiple forms of stress*. Springer, Dordrecht. pp. 215 – 232.

Oren A. 2014. The ecology of *Dunaliella* in high-salt environments. *Journal of Biological Research-Thessaloniki* 21, 23 30.

Oren A., 2015. Cyanobacteria in hypersaline environments: biodiversity and physiological properties. *Biodiversity and Conservation* 24, 781 – 798.

Ozimek T, Gulati RD, van Donk E. 1990. Can macrophytes be useful in biomanipulation of lakes? The Lake Zwemlust example. *Hydrobiologia* 200, 399 – 407.

Pannard A, Bormans M, Lefebvre S, Claquin P, Lagadeuc Y. 2007. Phytoplankton size distribution and community structure: influence of nutrient input and sedimentary loss. *Journal of Plankton Research* 29, 583 – 598.

Parsons TR, Maita Y, Lalli CM. 1984. *A manual of chemical and biological methods for seawater analysis*. Pergamon Press, New York. 173 pp.

Pedrós-Alió C, Calderón-Paz JI, MacLean MH, Medina G, Marrasé C, Gasol JM, Guixa-Boixereu, N. 2000. The microbial food web along salinity gradients. *FEMS Microbiology Ecology* 32, 143 – 155.

Pérez-Ruzafa A, Campillo S, Fernández-Palacios JM, García-Lacunza A, García-Oliva M, Ibañez H, Navarro-Martínez PC, Pérez-Marcos M, Pérez-Ruzafa IM, Quispe-Becerra JI, Sala-Mirete A. 2019. Long-term dynamic in nutrients, chlorophyll a, and water quality parameters in a coastal lagoon during a process of eutrophication for decades, a sudden break and a relatively rapid recovery. *Frontiers in Marine Science* 6, 26.

Pickett STA, Parker VT. 1994. Avoiding the old pitfalls: opportunities in a new discipline. *Restoration Ecology* 2, 75 – 79.

Pohlert T. 2014. The pairwise multiple comparison of mean ranks package (PMCMR). R package version 4.1.

Porter ET, Franz H, Lacouture R. 2018. Impact of Eastern oyster *Crassostrea virginica* biodeposit resuspension on the seston, nutrient, phytoplankton, and zooplankton dynamics: a mesocosm experiment. *Marine Ecology Progress Series* 586, 21 – 40.

Prach K, Durigan G, Fennessy S, Overbeck GE, Torezan .JM, Murphy SD. 2019. A primer on choosing goals and indicators to evaluate ecological restoration success. *Restoration Ecology* 27, 917 – 923.

Pretorius L. 2015. Spatial and temporal variability in water quality characteristics of the Swartkops Estuary. MSc thesis. Nelson Mandela Metropolitan University, South Africa. 217 pp.

R Core Team. 2020. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Rahaman AA, Ambikadevi M, Esso S. 1993. Biological management of Indian solar saltworks. *Seventh Symposium on Salt*. Elsevier, Amsterdam. 1, 633 – 643.

Raimondo DC, Adams JB, Lamberth JS, MacKay CF, Turpie JK, Sink K, van Niekerk L. 2019. Chapter 9: Indigenous species assessment: South African National Biodiversity Assessment 2018: Technical Report. Volume 3: Estuarine Realm. South African National Biodiversity Institute, Pretoria. Report Number: SANBI/NAT/NBA2018/2019/Vol3/A. p. 175 – 200.

Reddering JSV, Esterhuizen K. 1981. The Sedimentary Ecology of the Swartkops Estuary. Rosie Report 1, Geology Department, University of Port Elizabeth. 89 pp.

Rieger J, Stanley J, Traynor R. 2014. *Project planning and management for ecological restoration*.: Island Press, Washington, D.C. 320 pp.

Rogers G. 2020. Exotic algae spreads across Swartkops Estuary. *DispatchLIVE*. Retrieved from: https://www.dispatchlive.co.za/news/2020-06-02-exotic-algae-spreads-across-swartkops-estuary/#:~:text=The%20distinctive%20black%20streaks%20of,the%20Wylde%20Bridge%20at%20Swartkops.&text=A%20swathe%20of%20exotic%20black,red%20flag%20for%20human%20health. Accessed 8 July 2020.

Rohr, J.R., Bernhardt, E.S., Cadotte, M.W. and Clements, W.H., 2018. The ecology and economics of restoration. *Ecology and Society*, *23*(2).

Rubega MA, Robinson JA. 1997. Water salinization and shorebirds: emerging issues. *International Wader Studies* 9, 45 – 54.

Ruiz-Jaen MC, Aide TM. 2005. Restoration success: how is it being measured? *Restoration Ecology* 13, 569 – 577.

Rutherford ID, Jerie K, Marsh N. 2000. A Rehabilitation Manual for Australian Streams, Volume I. Co-operative Research Centre for Catchment Hydrology and Land and Water Resources Research and Development Corporation.

SAHRIS (South African Heritage Resources Information System). No date. Proposed Demolition of the unused Saltworks Facility and associated structures, Port of Ngqura. Retrieved from: https://sahris.sahra.org.za/cases/proposed-demolition-unused-saltworks-facility-and-associated-structures-port-ngqura. Accessed 17 July 2020.

Salgado J, Sayer CD, Brooks SJ, Davidson TA, Okamura B. 2018. Eutrophication erodes inter-basin variation in macrophytes and co-occurring invertebrates in a shallow lake: combining ecology and palaeoecology. *Journal of Paleolimnology* 60, 311 – 328.

Sammy N. 1983. Biological systems in north-western Australian solar salt fields. In: Schreiber BC, Harner HL (Eds.) *Sixth Symposium on Salt Vol. 1*. The Salt Institute. Alexandria, Virginia. pp. 207 – 215.

Sánchez MI, Green AJ, Castellanos EM. 2006. Temporal and spatial variation of an aquatic invertebrate community subjected to avian predation at the Odiel salt pans (SW Spain). *Archiv für Hydrobiologie* 166, 199 – 223.

Sanderson FJ, Donald PF, Pain DJ, Burfield IJ, Van Bommel FP. 2006. Long-term population declines in Afro-Palearctic migrant birds. *Biological Conservation* 131, 93 – 105.

Scharler UM, Baird D, Winter PED. 1997. Diversity and productivity of biotic communities in relation to freshwater inputs in three Eastern Cape estuaries. WRC Report No. 463/1/98. Water Research Commission, Pretoria.

Scheffer M, Carpenter SR. 2003. Catastrophic regime shifts in ecosystems: linking theory to observation. *Trends in Ecology and Evolution* 18, 648 – 656.

Scheffer M. 1989. Alternative stable states in eutrophic, shallow freshwater systems: a minimal model. *Hydrological Bulletin* 23, 73 – 83.

Schmidt J. 2013. Influence of predicted sea-level rise on the salt marsh of the Swartkops, Kromme and Knysna estuaries. MSc thesis. Nelson Mandela Metropolitan University, South Africa. 167 pp.

Segal RD, Waite AM, Hamilton DP. 2006. Transition from planktonic to benthic algal dominance along a salinity gradient. *Hydrobiologia* 556, 119 – 135.

Segal RD, Waite AM, Hamilton DP. 2009. Nutrient limitation of phytoplankton in solar salt ponds in Shark Bay, Western Australia. *Hydrobiologia* 626, 97 – 109.

Seitzinger SP, Styles RM, Pilling IE. 2001. Benthic microalgal and phytoplankton production in Barnegat Bay, New Jersey (USA): microcosm experiments and data synthesis. *Journal of Coastal Research* 32, 144 – 162.

Shackelford N, Hobbs RJ, Burgar JM, Erickson TE, Fontaine JB, Laliberté E, Ramalho CE, Perring MP, Standish RJ. 2013. Primed for change: developing ecological restoration for the 21st century. *Restoration Ecology* 21, 297 – 304.

Shumway SE, Allen SM, Boersma PD. 2003. Marine birds and harmful algal blooms: sporadic victims or under-reported events? *Harmful Algae* 2, 1 – 17.

Sim LL, Davis JA, Chambers JM, Strehlow K. 2006. What evidence exists for alternative ecological regimes in salinising wetlands?. *Freshwater Biology* 51, 1229 – 1248.

Simenstad C, Reed D, Ford M. 2006. When is restoration not?: Incorporating landscape-scale processes to restore self-sustaining ecosystems in coastal wetland restoration. *Ecological Engineering* 26, 27 – 39.

Simenstad C, Warren RS, 2002. Introduction to the special issue on dike/levee breach restoration of coastal marshes. *Restoration Ecology* 10, i.

Smyth K, Elliott M. 2016. Effects of changing salinity on the ecology of the marine environment. Solan M, Whiteley N (Eds.). In: *Stressors in the Marine Environment: Physiological and ecological responses; societal implications*. Oxford University Press. pp. 161 – 174.

Snow GC, Bate GC, Adams JB. 2000. The effects of a single freshwater release into the Kromme Estuary. 2: Microalgal response. *Water SA* 26, 301–310.

Solomon KR, Hanson M. 2014. Mesocosms and Microcosms (Aquatic). In: Wexler, P. (Ed.) *Encyclopedia of Toxicology: Third Edition.* Elsevier, Oxford. pp. 223-226.

Sousa AI, Martins I, Lillebø AI, Flindt MR, Pardal MA. 2007. Influence of salinity, nutrients and light on the germination and growth of *Enteromorpha* sp. spores. *Journal of Experimental Marine Biology and Ecology* 341, 142 – 150.

South Bay Salt Pond Restoration Project. No date. The restoration project. Retrieved from: https://www.southbayrestoration.org/page/restoration-project. Accessed 29 August 2019.

SRK Consulting (Pty) Ltd. 2010. Design, construction, operation and effectiveness of a pilot artificial wetland system to remove stormwater and sewage pollutants entering the Swartkops River Estuary via the Motherwell Canal, Nelson Mandela Bay, South Africa.

Steffen S, Mucina L, Kadereit G. 2010. Revision of *Sarcocornia* (Chenopodiaceae) in South Africa, Namibia and Mozambique. *Systematic Botany* 35, 390 – 408.

Strehlow K, Davis J, Sim L, Chambers J, Halse S, Hamilton D, Horwitz P, McComb A, Froend R. 2005. Temporal changes between ecological regimes in a range of primary and secondary salinised wetlands. *Hydrobiologia* 552, 17 – 31.

Stroebel T. 2019. Efficiency of the Motherwell artificial wetland at removing excess nutrients and *Escherichia coli* from storm water runoff before entering the Swartkops estuary. Honours thesis. Nelson Mandela University.

Suding KN. 2011. Toward an era of restoration in ecology: successes, failures, and opportunities ahead. *Annual Review of Ecology, Evolution, and Systematics* 42, 465 – 487.

Takekawa JY, Miles AK, Schoellhamer DH, Athearn ND, Saiki MK, Duffy WD, Kleinschmidt S, Shellenbarger GG, Jannusch CA. 2006. Trophic structure and avian communities across a salinity gradient in evaporation ponds of the San Francisco Bay estuary. *Hydrobiologia* 567, 307 – 327.

Taljaard S, van Niekerk L, Weerts SP. 2019. The legal landscape governing South Africa's coastal marine environment–Helping with the 'horrendogram'. *Ocean and Coastal Management* 178, 104801.

Taylor GD, Fletcher TD, Wong TH, Breen PF, Duncan HP. 2005. Nitrogen composition in urban runoff — implications for stormwater management. *Water Research* 39, 1982 – 1989.

Thrush SF, Townsend M, Hewitt JE, Davies K, Lohrer AM, Lundquist C, Cartner K, Dymond, J. 2013. The many uses and values of estuarine ecosystems. In: Dymond J (Ed.). *Ecosystem services in New Zealand–conditions and trends.* Manaaki Whenua Press, Lincoln, New Zealand. P. 226 – 237.

Toumi N, Ayadi H, Abid O, Carrias JF, Sime-Ngando T, Boukhris M, Bouain A. 2005. Zooplankton distribution in four ponds of different salinity: a seasonal study in the solar salterns of Sfax (Tunisia). *Hydrobiologia* 534, 1 – 9.

Tran, Huyen-Trang, Hao-Chu Wang, Tsai-Wen Hsu, Rakesh Sarkar, Chao-Li Huang, and Tzen-Yuh Chiang. 2019. Revegetation on abandoned salt ponds relieves the seasonal fluctuation of soil microbiomes. *BMC Genomics* 20, 478.

Turpie JK, Adams JB, Joubert A, Harrison TD, Colloty BM, Maree RC, Whitfield AK, Wooldrdige TH, Lamberth SJ, Taljaard S and van Niekerk L. 2002. Assessment of the conservation priority status of South African estuaries for use in management and water allocation. *Water SA* 28, 191 – 206.

Turpie JK, Clark B. 2007. Development of a conservation plan for temperate South African estuaries on the basis of biodiversity importance, ecosystem health and economic costs and benefits. Final Report by Anchor Environmental Consultants CC to C.A.P.E. Regional Estuarine Management Programme, Cape Town.

van Diggelen R, Grootjans AP, Wierda A. 1995. Hydro-ecological landscape analysis: a tool for wetland restoration. *Zeitschrift für Kulturtechnik und Landentwicklung* 36, 125 – 31.

Van Meter RJ, Swan CM, Snodgrass JW. 2011. Salinization alters ecosystem structure in urban stormwater detention ponds. *Urban Ecosystems* 14, 723 – 736.

van Niekerk L, Adams AB, Lamberth SJ, Taljaard S, MacKay CF, Bachoo S, Parak O, Murison G, Weerts SP. 2019a. Chapter 6: Pressures on the Estuarine Realm' in South African National Biodiversity Assessment 2018: Technical Report. Volume 3: Estuarine Realm. South African National Biodiversity Institute, Pretoria. Report Number: SANBI/NAT/NBA2018/2019/Vol3/A. p. 76 – 131.

van Niekerk L, Adams JB, Bate GC, Forbes AT, Forbes NT, Huizinga P, Lamberth SJ, MacKay CF, Petersen C, Taljaard S, Weerts SP. 2013. Country-wide assessment of estuary health: An approach for integrating pressures and ecosystem response in a data limited environment. *Estuarine, Coastal and Shelf Science* 130, 239 – 251.

van Niekerk L, Adams JB, James NC, Lamberth SJ, MacKay CF, Turpie JK, Rajkaran A, Weerts SP, Whitfield AK. 2020. An Estuary Ecosystem Classification that encompasses biogeography and a high diversity of types in support of protection and management. *African Journal of Aquatic Science* 45, 199 – 216.

van Niekerk L, Adams JB, Lamberth SJ, MacKay CF, Taljaard S, Turpie JK, Weerts SP, Raimondo DC (Eds.). 2019b. South African National Biodiversity Assessment 2018: Technical Report. Volume 3: Estuarine Realm. South African National Biodiversity Institute, Pretoria. Report Number: SANBI/NAT/NBA2018/2019/Vol3/A. 423 pp.

van Niekerk L, Skowno A, Adams JB, Lamberth SJ, Turpie J, MacKay CF, Sink K. 2019c. Chapter 8: Ecosystem threat status and protection levels. South African National Biodiversity Assessment 2018: Technical Report. Volume 3: Estuarine Realm. South African National

Biodiversity Institute, Pretoria. Report Number: SANBI/NAT/NBA2018/2019/Vol3/A. p 152 – 166.

van Niekerk L, Taljaard S, Adams JB, Bachoo S, George C, De Villiers P, Lamberth SJ, Sink K, MacKay FC, Parak P, Murison G. 2019d. Chapter 11: Government's response to declining estuarine ecosystems. South African National Biodiversity Assessment 2018: Technical Report. Volume 3: Estuarine Realm. South African National Biodiversity Institute, Pretoria. Report Number: SANBI/NAT/NBA2018/2019/Vol3/A. 234 – 268.

van Niekerk L, Taljaard S, Adams JB, Clark B, Lamberth SJ, MacKay CF, Weerts SP. 2019e. Chapter 7: Condition of South Africa's estuarine ecosystems. South African National Biodiversity Assessment 2018: Technical Report. Volume 3: Estuarine Realm. South African National Biodiversity Institute, Pretoria. Report Number: SANBI/NAT/NBA2018/2019/Vol3/A. 136 – 150.

van Niekerk L, Taljaard S, Adams JB, Fundisi D, Huizinga P, Lamberth SJ, Mallory S, Snow GC, Turpie JK, Whitfield AK, Wooldridge TH. 2015. Desktop Estuary EcoClassification and recommended health status of the temperate estuaries of South Africa. Draft CSIR Report to the Water Research Commission, WRC Project No. K5/2187.

van Niekerk L, Taljaard S. 2003. A framework for cooperative governance of South African estuaries. Water Research Commission. CSIR Report ENV-S-C 2003-077. CSIR, Stellenbosch.

Velasquez CR. 1992. Managing artificial saltpans as a waterbird habitat: species' responses to water level manipulation. *Colonial Waterbirds* 15, 43 – 55.

Veldkornet D. 2016. Estuarine species and habitats: Distribution and connectivity. PhD thesis. Nelson Mandela Metropolitan University, South Africa. 166 pp.

Veldkornet DA, Adams JB, van Niekerk L. 2015. Characteristics and landcover of estuarine boundaries: implications for the delineation of the South African estuarine functional zone. *African Journal of Marine Science* 37, 313 – 323.

Veldkornet DA, Potts AJ, Adams JB. 2016. The distribution of salt marsh macrophyte species in relation to physicochemical variables. *South African Journal of Botany* 107, 84 – 90.

Verdonschot PFM, Spears BM, Feld CK, Brucet S, Keizer-Vlek H, Borja A, Elliott M, Kernan M, Johnson RK. 2013. A comparative review of recovery processes in rivers, lakes, estuarine and coastal waters. *Hydrobiologia* 704, 453 – 474.

Vymazal, J., 2007. Removal of nutrients in various types of constructed wetlands. *Science of the Total Environment 380*, 48 – 65.

Warnock N, Page GW, Ruhlen TD, Nur N, Takekawa JY, Hanson JT. 2002. Management and conservation of San Francisco Bay salt ponds: effects of pond salinity, area, tide, and season on Pacific flyway waterbirds. *Waterbirds* 25(S2), 79 – 92.

Wetlands International. 2012. Waterbird population estimates, fifth edition – Summary report. Wetlands International, Wageningen, The Netherlands. 27 pp.

Whitfield AK, Buliwe NG. 2013. A century of science in South African estuaries: Bibliography and review of research trends. SANCOR Occasional Report 7. 289 pp.

Who Owns Whom. 2019. Manufacture of Edible Salt in South Africa. Report. 50 pp.

Williams WD. 1998. Salinity as a determinant of the structure of biological communities in salt lakes. *Hydrobiologia* 381, 191 – 201.

Wiseman KA, Sowman MR. 1992. An evaluation of the potential for restoring degraded estuaries in South Africa. *Water SA* 18, 13 – 19.

Wolanski E, Brinson MM, Cahoon, DR, Perillo GEM. 2009. Coastal wetlands: A synthesis. Perillo GEM, Wolanski E, Cahoon DR, Brinson MM (Eds.). In: *Coastal Wetlands: An Integrated Ecosystem Approach*. Elsevier. Amsterdam, The Netherlands. p. 1 – 64.

Wolfram G, Donabaum K, Schagerl M, Kowark VA. 1999. The zoobenthic community of shallow salt pans in Austria – preliminary results on phenology and the impact of salinity on benthic invertebrates. *Hydrobiologia* 408/409, 193 – 202.

Yih W, Kim HS, Myung G, Park JW, Du Yoo Y, Jeong HJ. 2013. The red-tide ciliate *Mesodinium rubrum* in Korean coastal waters. *Harmful Algae* 30, 53 – 61.

Zedler JB, Callaway JC. 1999. Tracking wetland restoration: Do mitigation sites follow desired trajectories? *Restoration Ecology* 7, 69 – 73.

Zhang J, Ma G, Deng Y, Dong J, Van Stappen G, Sui, L. 2016. Bacterial diversity in Bohai Bay solar saltworks, China. *Current Microbiology* 72, 55 – 63.

Zuur AF, Ieno EN, Elphick CS. 2010. A protocol for data exploration to avoid common statistical problems. *Methods in Ecology and Evolution* 1, 3 – 14.

Zuur AF, Ieno EN, Walker NJ, Saveliev AA, Smith GM. 2009. *Mixed effects models and extensions in ecology with R*. Springer, New York. 579 pp.

10. Appendices

Appendix 1: Site map of the abandoned saltworks.

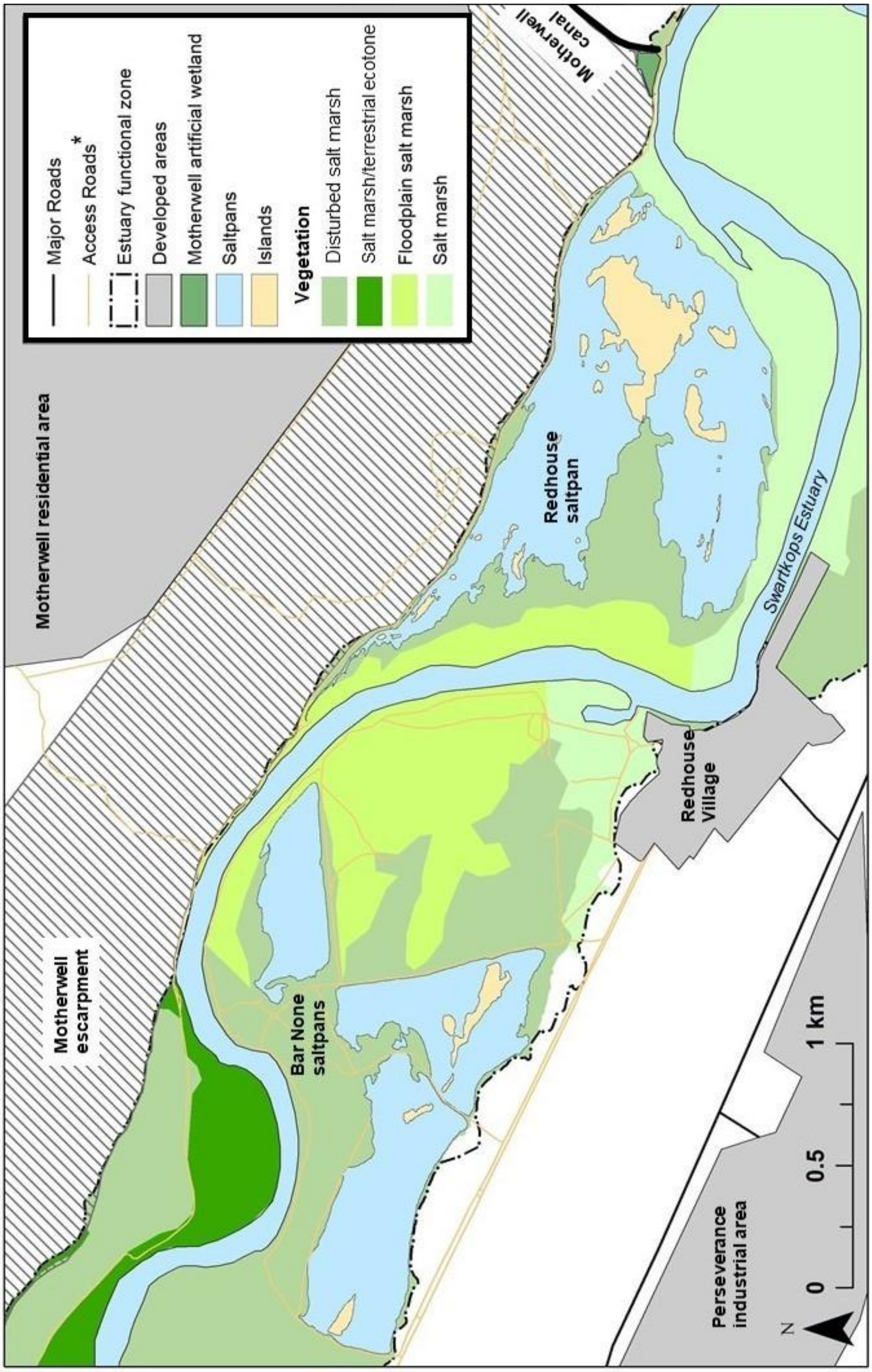

Figure 10.1 Comprehensive map of the abandoned saltworks at Swartkops Estuary.

* Note that the access roads were mapped via a desktop assessment and may not currently be accessible to vehicles.

Appendix 2: Location of transects and sampling points in Chapter 4

Table 10.1 Coordinates for the sampling points for Chapter 4.

Pan	Transect	Point	Latitude	Longitude
1	1	A	33° 50' 4.53" S	25° 35' 16.18" E
1	1	B	33° 50' 7.18" S	25° 35' 13.64" E
1	1	C	33° 50' 10.83" S	25° 35' 10.7" E
1	1	D	33° 50' 15.25" S	25° 35' 6.61" E
1	1	E	33° 50' 21.25" S	25° 35' 1.97" E
1	2	A	33° 50' 1.25" S	25° 35' 5.53" E
1	2	B	33° 50' 5.18" S	25° 35' 1.61" E
1	2	C	33° 50' 9.04" S	25° 34' 57.85" E
1	2	D	33° 50' 10.98" S	25° 34' 55.51" E
1	2	E	33° 50' 18.17" S	25° 34' 47.63" E
1	3	A	33° 50' 0.1" S	25° 34' 57.29" E
1	3	B	33° 50' 4.85" S	25° 34' 53.89" E
1	3	C	33° 50' 7.76" S	25° 34' 50.59" E
1	3	D	33° 50' 12.25" S	25° 34' 46.01" E
1	3	E	33° 50' 18.51" S	25° 34' 39.1" E
1	4	A	33° 49' 52.71" S	25° 34' 38.99" E
1	4	B	33° 49' 57.37" S	25° 34' 36.59" E
1	4	C	33° 50' 0.81" S	25° 34' 34.88" E
1	4	D	33° 50' 6.45" S	25° 34' 31.3" E
1	4	E	33° 50' 10.14" S	25° 34' 28.41" E
2	5	A	33° 49' 26.46" S	25° 33' 41.84" E
2	5	B	33° 49' 25.53" S	25° 33' 42.42" E
2	5	C	33° 49' 22.09" S	25° 33' 43.94" E
2	6	A	33° 49' 25.09" S	25° 33' 52.64" E
2	6	B	33° 49' 23.76" S	25° 33' 53.42" E
2	6	C	33° 49' 20.37" S	25° 33' 55.11" E
3	7	A	33° 49' 45.2" S	25° 33' 30.15" E
3	7	B	33° 49' 41.34" S	25° 33' 31.42" E
3	7	C	33° 49' 38.52" S	25° 33' 32.24" E
3	7	D	33° 49' 36.2" S	25° 33' 33.14" E
4	7	E	33° 49' 34.04" S	25° 33' 34" E
4	8	A	33° 49' 39.5" S	25° 33' 19.6" E
4	8	B	33° 49' 38.11" S	25° 33' 20.38" E
4	8	C	33° 49' 34.56" S	25° 33' 22.66" E
4	8	D	33° 49' 31.56" S	25° 33' 24.69" E
4	9	A	33° 49' 30.93" S	25° 32' 52.47" E
4	9	B	33° 49' 25.43" S	25° 32' 56.02" E
4	9	C	33° 49' 28.66" S	25° 32' 53.96" E
4	9	D	33° 49' 27.63" S	25° 32' 54.63" E

Appendix 3: Digital terrain model of Swartkops Estuary.

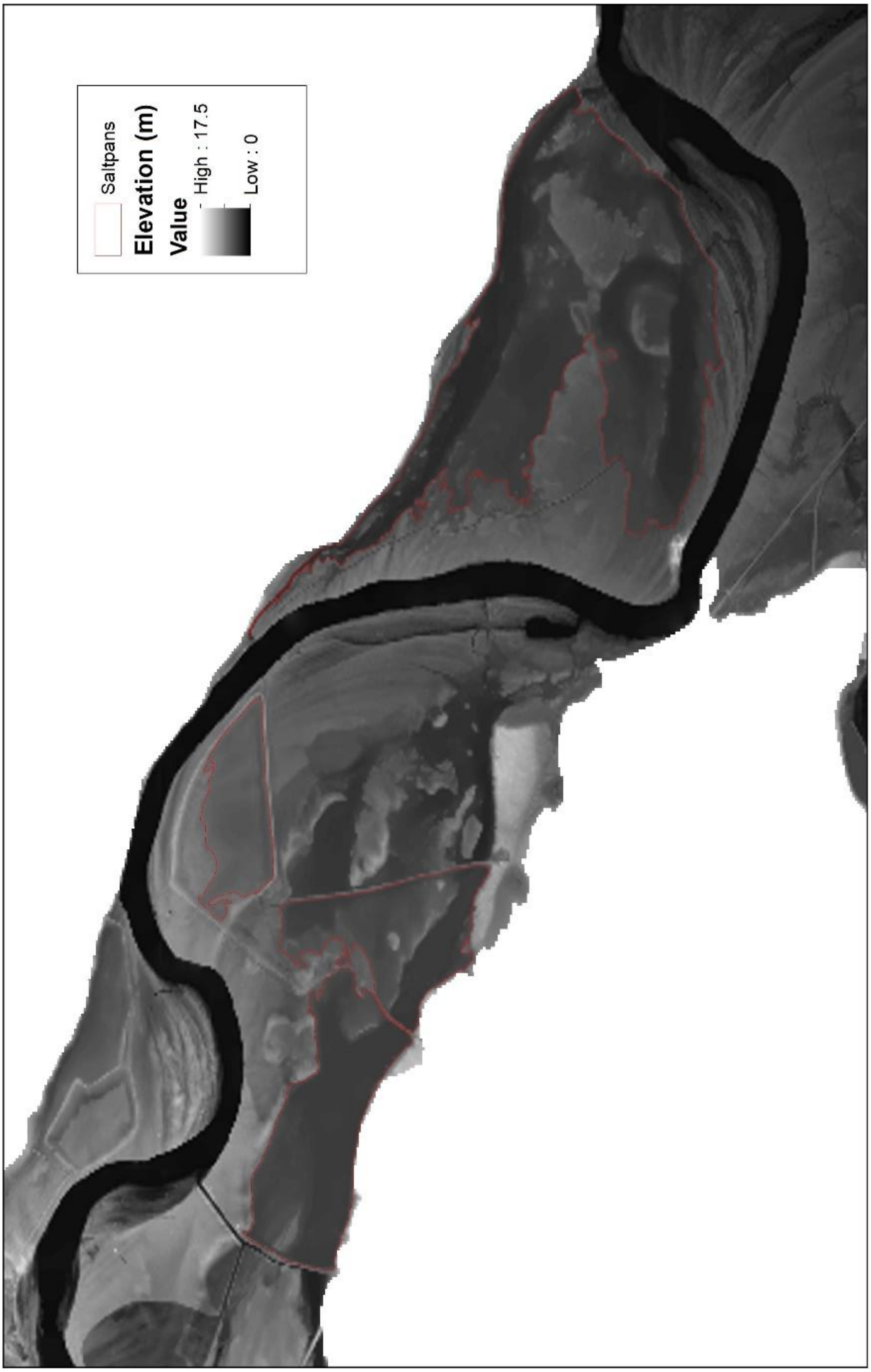

Figure 10.2 Digital terrain model of Swartkops Estuary derived from a 2019 LiDAR survey (at 1 m spatial resolution) by Nelson Mandela Bay Municipality (NMBM).

Appendix 4: Details of methods used in this thesis

A. Sediment moisture content

Sediment moisture content was determined using the method presented in Gardner (1965). 10 g of sediment was weighed out (to 0.01 g accuracy) from each sample and oven dried at 60°C for 48 hours. The dried sediment was weighed again, and the following equation was used to determine the percentage moisture content:

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

Where: Mw is the initial wet mass of the sediment and Md is the mass of the sediment after drying.

B. Sediment organic content

Sediment organic content was determined using the method presented in Briggs (1977). The dried sediment samples from the moisture content analysis were placed in an ashing furnace at 550°C for 8 hours. The samples were then weighed again, and the following formula was used to determine the percentage organic content:

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

Where: Ma is the mass of the sediment after ashing.

C. Sediment salinity

Sediment salinity was determined using the 'saturated paste' method as described in Barnard (1990). From each sample, 250 g of air-dried soil was placed in a beaker, and deionised water added and mixed until a saturated paste was formed. The paste was left overnight, then filtered through a Buchner funnel with a Munktell © MGC filter paper. The salinity of the filtrate was measured using a handheld refractometer.

D. Sediment particle size

Sediment particle size was determined using the hydrometer method (Gee and Bauder, 1986). A 50 g L^{-1} sodium hexametaphosphate ($(NaPO_3)_6$) solution was prepared with distilled water. In a beaker, 100 ml of this solution and 250 ml of distilled water was added to 40 g of air-dried sediment from each sample. The beakers were placed on a mechanical shaker overnight for thorough dispersal of the suspension. The suspension was transferred to 1 L measuring cylinder, which was then filled to the 1 L mark with distilled water and the cylinder was subsequently shaken by hand for a minute. A blank solution of sodium hexametaphosphate with no soil was also prepared in the same manner for the correction explained below. A soil hydrometer (AMH-152) was placed in the suspension and readings were taken after 30 s, 1 min, 3 mins, 1.5 h and 24 h. The percentage of silt (particle size > 2 μm), clay (particle size 2 – 50 μm) and sand (particle size > 50 μm) were determined using the following equations:

Firstly, the concentration of soil in suspension (C) in g L^{-1} was determined for each sample at each time interval:

$$C = R - R_L$$

Where: R is the uncorrected hydrometer reading in g L^{-1}; and
R_L is the hydrometer reading of a blank solution in g L^{-1}

Then, the clay fraction (P_{clay}) was determined using the hydrometer readings at 1.5 h and 24 h. First, the effective particle diameter *X* at 1.5 h and 24 h was calculated as:

$$X = \theta t^{-1/2}$$

Where θ is the sedimentation parameter calculated as:

$$\theta = 1000(Bh')^{-1/2}$$

Where B is calculated as:

$$B = \frac{30\eta}{g(p_s - p_l)}$$

And h' is calculated as:

$$h' = -0.164R + 16.3$$

Where: θ is the sedimentation parameter (µm min$^{1/2}$);
h' is the effective hydrometer depth (cm);
η is fluid viscosity in poise (g cm^{-1}s^{-1});
g is the gravitational constant (cm s^{-2});
p_s is the soil particle density (g cm^{-3}); and
p_l is the solution density (g cm^{-3})

The percentage of clay (P_{clay}) was then calculated as:

$$P_{clay} = m \ln\left(\frac{2}{X_{24}}\right) + P_{24}$$

Where: X_{24} is the mean particle diameter in suspension at 24 h;
P_{24} is the summation percentage at 24 h calculated as the corrected hydrometer reading at 24 h (C_{24}) divided by the dry weight (in g) of the soil sample (C_O); and
m is the slope of the summation percentage curve between X_{24} and $X_{1.5}$, calculated as:

$$m = \frac{P_{1.5} - P_{24}}{\ln\left(\frac{X_{1.5}}{X_{24}}\right)}$$

Where: $X_{1.5}$ is the mean particle diameter in suspension at 24 h; and
$P_{1.5}$ is the summation percentage at 24 h ($\frac{C_{1.5}}{C_O}$).

The sand fraction (P_{sand}) was calculated using the same procedure used to calculate the clay fraction, but the 30 s and 60 s hydrometer readings were used instead of the 1.5 and 24 h readings. The calculated P_{sand} values were subtracted from 100 to determine the summation percentage of sand.

Lastly, the silt fraction (P_{silt}) was determined as:

$$P_{silt} = 100 - (P_{\text{clay}} + P_{\text{sand}})$$

E. V notch weir discharge

The head of water above the V notch weir was measured at the outflow of the MWAW. Discharge was calculated using determined using the Kindsvater-Shen equation (USBR, 2001), which is:

$$Q = C_e \tan\left(\frac{\theta}{2}\right)(h+k)^{\frac{5}{2}}$$

Where: Q_{actual} is the discharge, or flow rate, over the weir in $m^3\ s^{-1}$;
C_e is the discharge coefficient (0.576);
θ is the notch angle (70°);
k is the head correction factor (0.001).

F. Determination of inorganic nutrient concentrations in the water column

Water sample (50 ml) were collected for the determination of the concentrations of dissolved inorganic phosphorus (DIP = PO_4^{3-}-P), ammonium (NH_4^+-N) and total oxidised nitrogen (NO_x = NO_3^- + NO_2^-). The water samples were first gravity-filtered through Whatman© GF/C glass-fibre filters and then through hydrophilic polyvinylidene difluoride (PVDF) 0.47 µM pore-size syringe filters. The filtrates were frozen in dark bottles until the analyses proceeded. DIP and NH_4^+ concentrations were determined using standard spectrophotometric methods (oxidation method: Parsons et al., 1984) and NO_x concentrations were determined using the reduced copper cadmium method (Bate and Heelas, 1975). Reverse osmosis water was used for all analyses.

For determination of DIP concentrations, a mixed reagent and standard series were first made up. The mixed reagent was made up of the following solutions: 5 ml ammonium molybdate (15 g in 500 ml H_2O), 12.5 ml sulphuric acid (140 ml concentrated sulphuric acid in 900 ml H_2O), 5 ml ascorbic acid (1.35 g in 25 ml H_2O) and 2.5 ml potassium antimony tartrate (0.34 g in 250 ml H_2O). To make the standard series, a 6 mM stock solution was made with 0.816 g anhydrous potassium dihydrogen phosphate in 1 L of H_2O. Then, a 100 ml diluted stock solution (72 µM) was made up by putting 1.2 ml of the stock solution in H_2O. The following volumes were taken out from the diluted stock and further diluted to 100 ml with H_2O: 1.5, 3, 6, 12, 24, 48 and 96 ml. The concentrations of the resultant standard series were: 0 (blank), 1.08, 2.16, 4.32, 8.64, 17.28, 34.56 and 69.12 µM. Once the standards and mixed reagent were made up, 2.5 ml of each collected water sample (as well as each standard) was rapidly mixed with 0.25 ml of the mixed reagent. After 5 minutes, the absorbances of the samples and standards were read using a GENESYS™ 10S UV-Vis spectrophotometer at a wavelength of 885 nm.

For the determination of NH_4^+ concentrations, a standard series was first made up by making up a 10 mM stock solution (0.535 g of ammonium chloride in 1 L H_2O). A 100 ml diluted stock solution (73 μM) was made up from this by putting 0.73 ml of the stock solution in H_2O. The following volumes were taken out from the diluted stock and further diluted to 100 ml with H_2O: 1.5, 3, 6, 12, 24, 48 and 96 ml. The concentrations of the resultant standard series were: 0 (blank), 1.095, 2.19, 4.38, 8.76, 17.52, 35.04 and 70.08 μM. Once the standard series was made up, 0.1 ml of phenol solution (20 g crystalline analytical grade phenol in 200 ml 95% ethanol) was added to 2.5 ml of each of the samples (as well as the standards). Then 0.1 ml of sodium nitroprusside solution (1 g in 200 ml H_2O) was added to this. Lastly, 0.25 ml of oxidising solution (an alkaline reagent solution of 20 g sodium citrate with 1 g sodium hydroxide in 100 ml H_2O which was mixed with a sodium hypochlorite solution [10 – 14%]) was added to the samples and standards, which were left in the dark to allow for colour to develop for between 1 and 24 hrs. The spectrophotometer was then used to read the absorbances of the samples and standards at 640 nm.

For the determination of NO_x concentrations, a standard series was first made up by making up a 5 mM stock solution (0.51 g of potassium nitrate in 1 L H_2O), from which 100 ml diluted stock solution (144 μM) was made by putting 2.88 ml of the stock solution into H_2O. The following volumes were taken out from the diluted stock and further diluted to 100 ml with H_2O: 1.5, 3, 6, 12, 24, 48 and 96 ml. The concentrations of the resultant standard series were: 0 (blank), 2.16, 4.32, 8.64, 17.28 34.56, 69.12 and 138.24 μM. Once the standard series was made up, 2 ml of a buffer solution (21.4 g of ammonium chloride in 1 L H_2O, adjusted to pH 9.6 with ammonium hydroxide) was added to 3 ml of each sample and standard. Then, approximately 2 g of copper cadmium (stored under weak acid-ethylenediamine tetraacetic acid solution in an airtight flask) was added to the samples and mixed for 10 minutes. Next, 1 ml of sulfanilimide solution (1 g in 100 ml 1.5 N HCl) was added to 1 ml of each sample. Lastly, 1 ml of diamine hydrochloride solution (0.02 g in 100 ml H_2O) was added to the samples, which were then left in the dark for 5 minutes for colour to develop. The spectrophotometer was then used to read the absorbances of the samples and standards at 540 nm.

G. Determination of phytoplankton and microphytobenthos biomass

Water samples were collected for determining phytoplankton biomass, which was measured as chlorophyll-*a* concentration. The samples were gravity-filtered through Whatman© GF/C glass-fibre filters which were subsequently frozen until the analysis was carried out. To extract the to extract the chlorophyll-*a*, the frozen filters were placed into glass vials with 10 ml of 95% ethanol for 24 h in a cold (1 – 2 °C) and dark room. After the extraction, chlorophyll-*a* concentrations were determined using spectrophotometric methods described by Nusch (1980). Absorbance was measured before and after acidification with 1N HCl were read using a GENESYS™ 10S UV-Vis spectrophotometer. The following equation was then used to calculate phytoplankton biomass:

$$Chl - a\ (\mu g\ l^{-1}) = (E_{b665} - E_{a665}) \times\ 29.6\ \times\ \left(\frac{v}{V \times I}\right)$$

Where: E_{b665} is absorbance at 665 nm before acidification;

E_{a665} is absorbance at 665 nm after acidification;

29.6 is the constant calculated from the maximum acid ratio (1.7) and the specific absorption coefficient of chlorophyll-*a* in ethanol (82 g l^{-1} 10 mm^{-1});

v is the volume of solvent used for extraction (ml);

V is the volume of the sample filtered (L); and

I is the path of the spectrophotometer cuvette (cm).

Sediment samples were collected to determine microphytobenthos (MPB) biomass, which was also measured as chlorophyll-*a* concentration. A Perspex corer (with an internal diameter of 20 mm) was used to collect the top 1 cm of sediment. The samples were kept frozen in the dark prior to analysis. At the onset of the analysis, 15 ml of 95% ethanol was added to the sediment samples, which were then left in a fridge for 6 hrs for the extraction of chlorophyll-*a* (Brito et al., 2009). The extract was then filtered from the sediment through Whatman© GF/C glass-fibre filters. The chlorophyll-*a* concentrations in the extract were then determined using the spectrophotometric method described by Nusch (1980) (the same method as described above for phytoplankton) and applying the following formula:

$$Chl - a\ (\mathrm{mg}\ m^{-2}) = (E_{b665} - E_{a665}) \times\ 29.6\ \times\ \left(\frac{v}{A \times I}\right) \times 10$$

Where: E_{b665} is absorbance at 665 nm before acidification;

E_{a665} is absorbance at 665 nm after acidification;

29.6 is the constant calculated from the maximum acid ratio (1.7) and the specific absorption coefficient of chlorophyll-*a* in ethanol (82 g l^{-1} 10 mm^{-1});

v is the volume of solvent used for extraction (ml);

A is the area of the corer (cm^2); and

I is the path of the spectrophotometer cuvette (cm).

H. Identification and enumeration of plankton

Water samples (100 ml) for the identification and enumeration of plankton were collected from each tank and preserved with 1 ml of 25% glutaraldehyde solution. Next, 25 ml of preserved sample was stained with Rose Bengal and allowed to settle overnight in an Utermöhl settling chamber (26.5 mm in diameter) (Coulon and Alexander, 1972). After settling, cell counts and microalgal identification were achieved using an inverted Leica DMIL phase contrast microscope at 630X magnification with a minimum of either 200 frames or 200 cells counted. Cells were identified and classified to a class level (i.e. diatoms, dinoflagellates, cyanophytes, cryptophytes, chlorodendrophytes and chlorophytes) and to the smallest taxon where possible. Microzooplankton were also counted and identified to the smallest taxon if possible. Cell abundance (per ml) of each functional group was calculated using the equation presented below (Snow et al., 2000):

$$Cells\ ml^{-1} = \left(\frac{(\pi r^2)}{A}\right) \times \left(\frac{C}{V}\right)$$

Where: r is the radius of the settling chamber (mm);

A is the area of each frame (mm^2);

C is the number of cells in each frame; and

V is the volume of the water sample in the settling chamber (ml).

Appendix 5: Plant species recorded at the abandoned saltworks

Table 10.2 List of plant species recorded at the abandoned saltworks.

Species	**Authority**
Atriplex lindleyi subsp. *inflata*	(F. Muell.) Paul G. Wilson
Bassia diffusa	(Thunb.) Kuntze
Disphyma crassifolium	(L.) L. Bolus
Drosanthemum fourcadei	(L. Bolus) Schwantes
Psilocaulon dinteri	(Engl.) Schwantes
Salicornia meyeriana	Moss
Salicornia pillansii	(Moss) Piirainen and G. Kadereit
Senecio sp.	(L.)
Spergularia media	(L.) C.Presl ex Griseb.
Sporobolus virginicus	(L.) Kunth
Suaeda fruticosa	(L.) Forssk

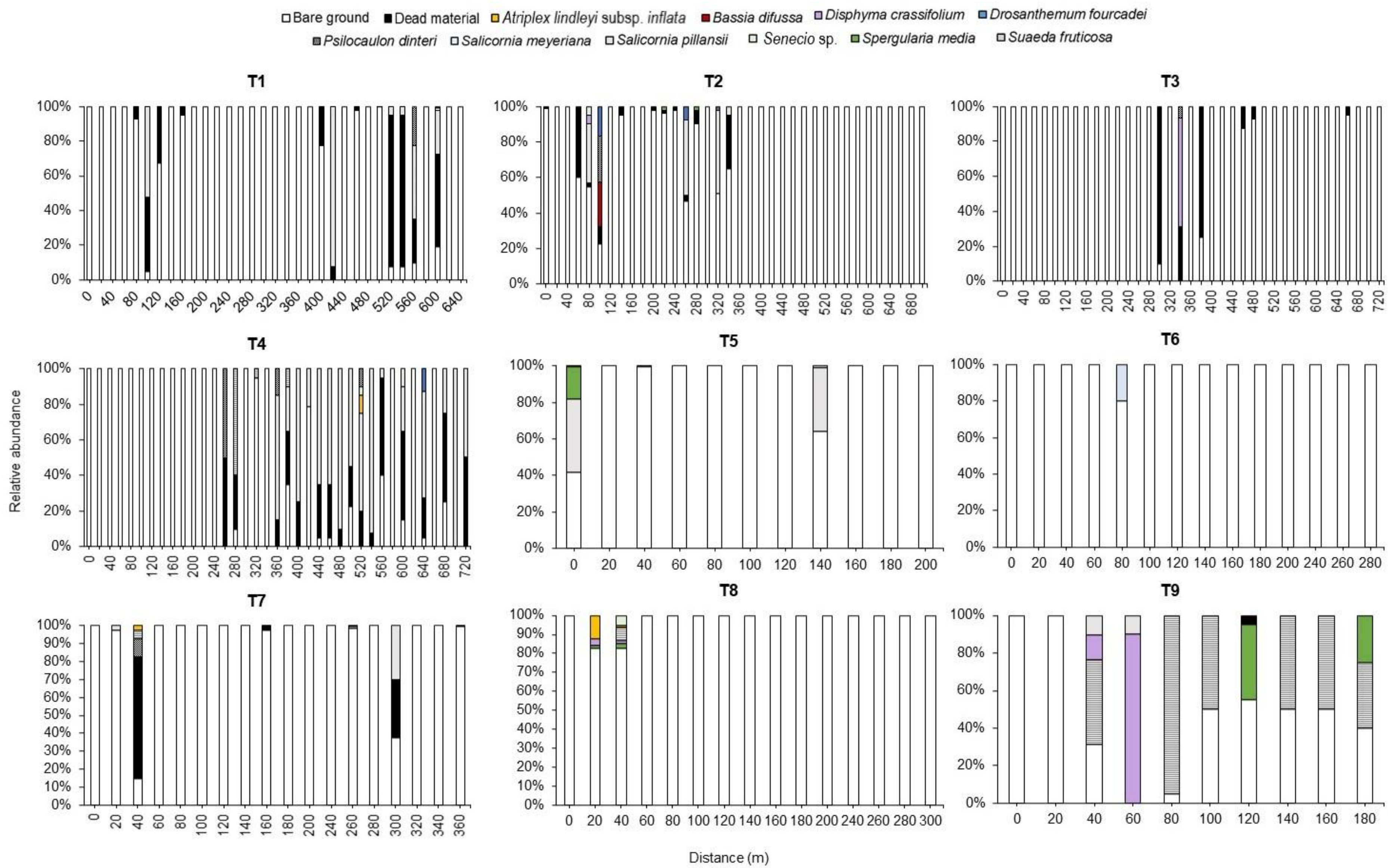

Figure 10.3 Plant species cover along the transects at the abandoned saltworks. The location of the transects are shown in Figure 4.1.

Appendix 6: List of waterbird species recorded at the saltworks

Table 10.3 Bird species recorded at the Redhouse and Bar None saltpans since 1994 and 1995, respectively, including maximum counts and comments. Species triggering the criteria for Ramsar Sites and Important Bird and Biodiversity Areas (IBAs) are listed first, followed by breeding and non-breeding species recorded at the site, respectively, in order of decreasing maximum counts. Data retrieved from Martin and Randall (1987), Crawford et al. (2009) and P. Martin (unpubl. data, 2020).

Species	**Redhouse saltpan**	**Bar None saltpans**	**Comments**
Curlew sandpiper *Calidris ferruginea*	**+** 4187	398	Frequently occur in summer in very high abundance at the Redhouse saltpan and less so at the Bar None saltpans
Greater flamingo *Phoenicopterus roseus*	**#+** 836	420	Hundreds often seen at the saltworks. Near-threatened.
Kelp gull [B] *Larus dominicanus*	***+** 879	**+** 181	Very common at the saltworks, higher abundance in summer. One of the largest breeding colonies in southern Africa. Has been recorded breeding from September to January.
Pied avocet *Recurvirostra avosetta*	**+** 248	***+** 348	Relatively common year-round
South African shelduck *Tadorna cana*	***+** 422	142	Frequently found at the saltworks; more abundant in summer
Cape shoveler [B] *Spatula smithii*	**+** 315	58	Frequently found at the saltworks; more abundant at the Redhouse saltpan. Recorded breeding around October.
White-breasted cormorant [B] *Phalacrocorax lucidus*	# 183	93	Relatively common year-round. Second largest breeding colony in southern Africa. Recorded breeding around February to September.

Species	Redhouse saltpan	Bar None saltpans	Comments
Black-necked grebe *Podiceps nigricollis*	37	**#+** 159	Frequently occur at the Bar None saltpans and rarely at the Redhouse saltpan
African spoonbill [B] *Platalea alba*	57	**+** 82	Frequently found at the saltworks; more abundant in summer
Caspian tern [B] *Hydroprogne caspia*	# 64	# 32	Frequently found at the saltworks; more common in summer. Second largest colony in South Africa. Recorded breeding around February to August and October. Near-threatened.
Cape teal [B] *Anas capensis*	199	300	Frequently found year-round at the saltworks. Has been recorded breeding around March to May and August to December.
Grey-headed gull [B]	149	228	Common at the saltworks, particularly in summer. Regionally important breeding colony; has been recorded breeding around May, July to August and October.
African sacred ibis [B] *Threskiornis aethiopicus*	326	20	Regularly seen in summer at the saltworks; sometimes hundreds occurring at the Redhouse saltpan. Generally breeds around October.
Yellow-billed duck [B] *Anas undulata*	212	92	Frequently occur at the saltworks year-round. Generally breeds around June.
Common ringed plover [B] *Charadrius hiaticula*	207	95	Regularly seen at the saltworks in summer
Grey plover [B] *Charadrius hiaticula*	242	11	Frequently seen at the Redhouse saltpan and rarely at the Bar None saltpans in summer. Generally breeds around January.

Species	Redhouse saltpan	Bar None saltpans	Comments
Little grebe [B] *Tachybaptus ruficollis*	168	59	Frequently occur at the saltworks in the winter with a small number sometimes recorded in summer; more common at the Rehouse saltpan. Generally breeds around June.
Egyptian goose [B] *Alopochen aegyptiaca*	25	13	Frequently seen at the saltpans, but in low numbers. Generally breeds around September to December.
Three-banded plover [B] *Charadrius tricollaris*	12	10	Rarely seen at the saltworks. Recorded breeding around March, September and December.
Little stint *Calidris minuta*	1320	1353	Occur in high abundance in summer
Ruff *Philomachus pugnax*	1296	200	Relatively abundant at the saltworks in summer
Cape cormorant *Phalacrocorax capensis*	300	105	Infrequently seen at the pans, mostly in winter. Near-threatened.
Red-knobbed coot *Fulica cristata*	376	17	Have been seen at the saltpans some years
Red-billed teal *Anas erythrorhyncha*	108	273	Occasionally occur at the saltworks
Black-winged stilt *Himantopus himantopus*	165	223	Frequently found year-round at the saltworks
Grey-headed gull [B] *Chroicocephalus cirrocephalus*	149	228	Common at the saltworks, particularly in summer
Reed cormorant *Microcarbo africanus*	350	23	Occur at the saltpans year-round, higher abundance at Rehouse saltpan

Species	**Redhouse saltpan**	**Bar None saltpans**	**Comments**
Lesser flamingo *Phoenicopterus minor*	268	39	Frequently seen at the Redhouse saltpan, mostly in winter, and rarely at the Bar None saltpans. Near-threatened.
Kittlitz's plover *Charadrius pecuarius*	107	125	Relatively common at the saltworks year-round
Marsh sandpiper *Tringa stagnatilis*	201	18	Frequently occur in summer at the saltworks
Blacksmith lapwing *Vanellus armatus*	121	33	Frequently occur at the saltworks
Common greenshank *Tringa nebularia*	122	17	Frequently occur year-round at the saltworks, especially the Redhouse saltpan; more abundant in summer
Little tern *Sternula albifrons*	74	56	Occasionally occur at the saltworks in summer; more common at the Redhouse saltpan
Grey heron *Ardea cinereal*	79	50	Frequently seen at the saltworks year-round
White-winged tern *Chlidonias leucopterus*	193	16	Have occurred at the saltworks on a few occasions in summer
Common tern *Sterna hirundo*	34	69	Occasionally occur at the saltworks; more common at the Redhouse saltpan
Cattle egret *Bubulcus ibis*	5	50	Have been seen at the saltpans a few times
Cape wagtail *Motacilla capensis*	30	24	Common at the saltworks year-round
Eurasian curlew *Numenius arquata*	21	-	A few individuals have been seen at the Redhouse saltpan in previous years
Bar-tailed godwit *Limosa lapponica*	19	2	Rare sightings have been recorded at the Redhouse saltpan

Species	Redhouse saltpan	Bar None saltpans	Comments
Common whimbrel *Numenius phaeopus*	19	2	Frequently occur at the Redhouse saltpan in summer; have been recorded at the Bar None saltpans
Ruddy turnstone *Arenaria interpres*	18	2	Occur at the saltworks infrequently; low abundance
Black-headed heron *Ardea melanocephala*	12	2	A few individuals have been recorded at the saltworks relatively frequently
African darter *Anhinga rufa*	7	2	A few individuals have been recorded in winter
Water thick-knee *Burhinus vermiculatus*	-	9	Very rarely seen at the saltworks
Black stork *Ciconia nigra*	5	1	Has been sighted on a few occasions. Near-threatened.
Hartlaub's gull *Chroicocephalus hartlaubii*	3	2	A few individuals have been recorded at the saltworks on rare occasions
Southern pochard *Netta erythrophthalma*	-	5	Have been seen at the Bar None saltpans on one occasion
Wood sandpiper *Tringa glareola*	1	4	Rarely seen at the saltworks
African fish-eagle *Haliaeetus vocifer*	2	2	Very rarely seen at the saltpans
Goliath heron *Ardea goliath*	3	1	Has been recorded at the saltworks on rare occasions
Pied kingfisher *Ceryle rudis*	2	2	Rarely seen at the saltworks
White-fronted plover *Charadrius marginatus*	3	1	Very rarely seen at the saltworks
Great egret *Ardea alba*	3	-	A few individuals have been seen at the Redhouse saltpan on rare occasions
African marsh-harrier *Circus ranivorus*	2	1	Sometimes seen at the saltworks; more often in summer. Vulnerable

Species	Redhouse saltpan	Bar None saltpans	Comments
Osprey *Pandion haliaetus*	1	2	Rarely seen at the saltworks
African black oystercatcher *Haematopus moquini*	3	-	Very rarely seen at the saltworks. Near-threatened.
Yellow-billed egret *Ardea intermedia*	1	1	Very rarely seen at the saltpans
Hadeda ibis *Bostrychia hagedash*		2	Very rarely seen at the saltworks
Red knot *Calidris canutus*	2		Very rarely seen at the saltworks
Brown-throated martin *Riparia paludicola*	1	1	A single bird has been recorded on rare occasions
Common sandpiper *Actitis hypoleucos*	-	2	Have been seen at the Bar None saltpans on a few occasions
Glossy ibis *Plegadis falcinellus*	-	1	Very rarely seen at the saltworks
Great white pelican *Pelecanus onocrotalus*	1	-	A vagrant has been recorded on rare occasions. Near-threatened.
Yellow-billed stork *Mycteria ibis*	-	1	A single bird has been recorded on rare occasions
Common moorhen *Gallinula chloropus*	-	1	Very rarely seen at the saltworks
Little egret *Egretta garzetta*	76	27	Frequently seen at the Redhouse saltpan and occasionally at the Bar None saltpans
[B] Species recorded breeding at the saltpans **+** Numbers meet the criteria for a sub-Regional Important Bird Area ***** Numbers meet the criteria for a Globally Important Bird Area # Numbers meet the criteria for a Ramsar site			

Appendix 7: Sampling details for Chapter 5

Table 10.4 Sampling occasion at the Motherwell artificial wetland (MWAW) and Swartkops Estuary for Chapter 5.

Date	Comments
Motherwell artificial wetland (MWAW)	
May 2017	Exact date not certain; pH not recorded
28 July 2017	pH not recorded
September 2017	Exact date not certain; no physico-chemical parameters recorded
26 July 2018	pH not recorded
23 April 2019	
23 May 2019	
25 June 2019	
2 August 2019	
10 September 2019	
24 February 2019	
2 March 2020	
9 March 2020	
16 March 2020	
23 March 2020	
Swartkops Estuary	
25 April 2019	Outgoing tide
25 July 2019	Outgoing tide
4 November 2019	Outgoing tide
4 February 2020	Outgoing tide
5 June 2020	Incoming tide
25 June 2020	Outgoing tide

Appendix 8: Ecological Recovery Wheel for assessing the success of the rehabilitation project

The Ecological Recovery Wheel (to go along with the Five-Star System: Table 7.5) was modified from the SER Standards and applied to the saltworks in its abandoned state (Figure 10.4). As the saltworks is currently lying dry, the threats of eutrophication and HABs are not relevant, hence the five-star rating. The 'waterbird habitat' category received one star as ≥ 5% of the mean annual waterbird numbers were recorded at the site in 2019 and 2020, but it should be noted that species richness was far lower than historically. The 'no undesirable species' category is mainly aimed to address potentially HAB-forming species which are currently irrelevant (until the hydrology of the site is restored) but received only four stars as the invasive halophyte *Atriplex lindleyi* subsp. *inflata* is present. The categories 'desirable animals', 'desirable plants' and 'primary producer diversity' have received just one star as some waterbirds and salt marsh vegetation are present. These attributes will change once the rehabilitation commences and periodic evaluations will be necessary to determine the success of rehabilitation measures. A blank Ecological Recovery Wheel has been provided in Figure 10.5 for this purpose.

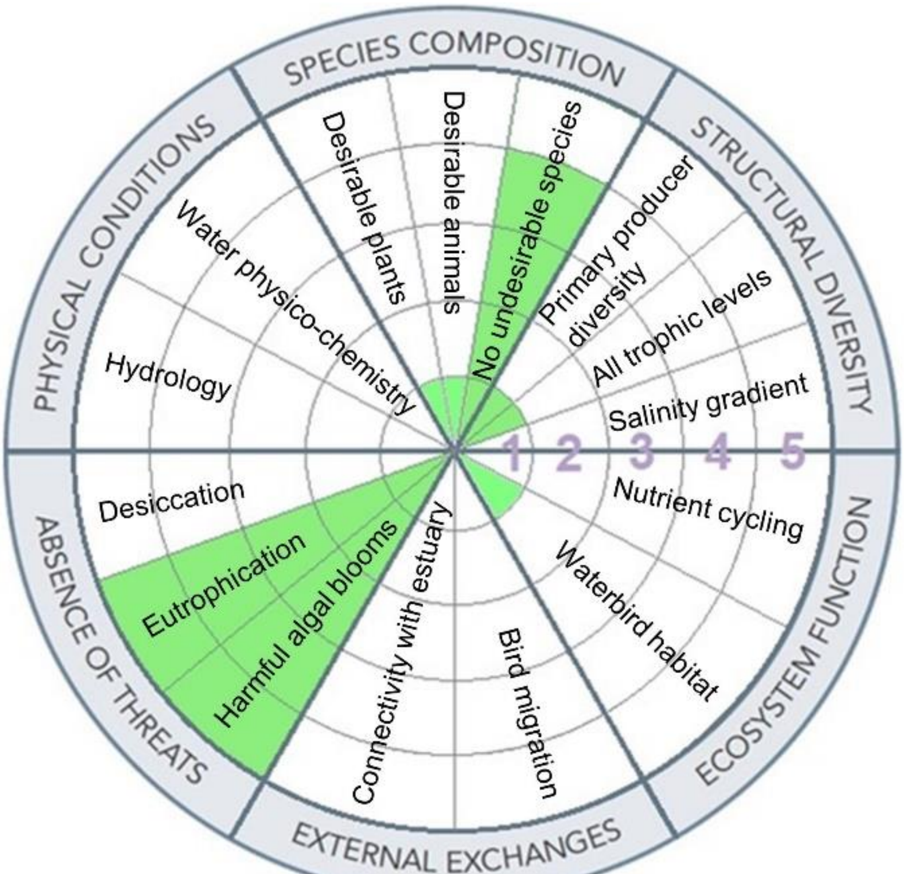

Figure 10.4 The Ecological Recovery Wheel adapted to the abandoned saltworks in its current condition (modified from Gann et al., 2019).

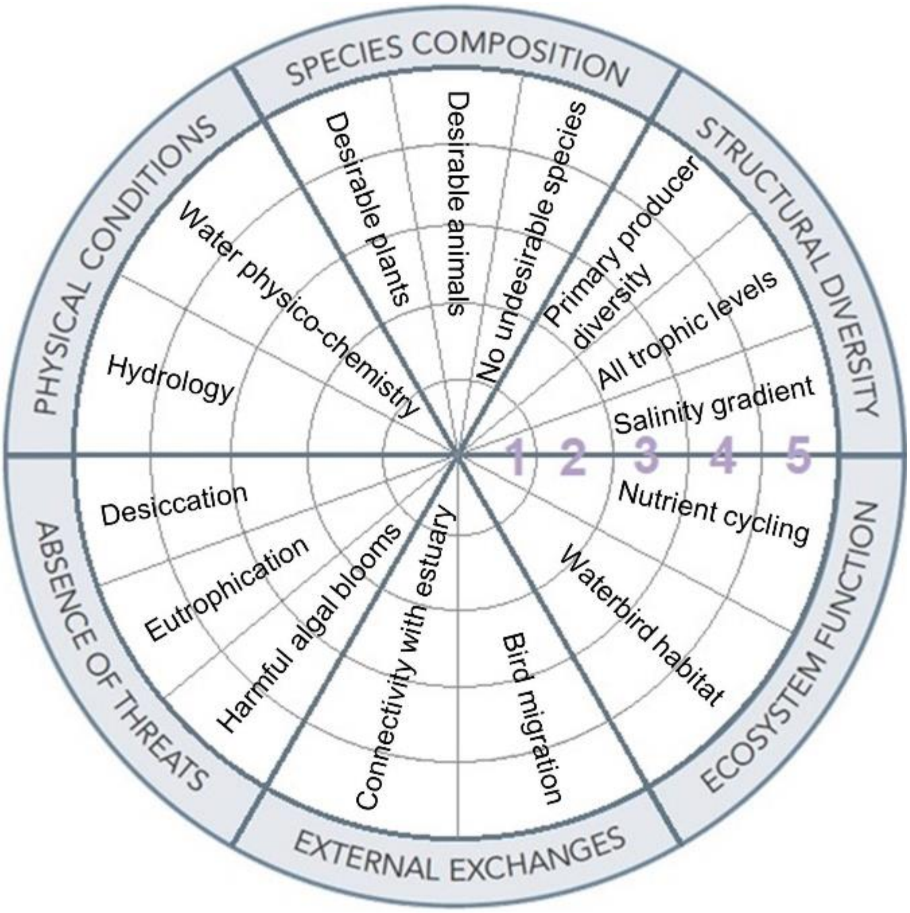

Figure 10.5 A blank Ecological Recovery Wheel for monitoring the success of rehabilitation activities at the saltworks (modified from Gann et al., 2019).