Investigating the potential for saltpan restoration for the provision of multiple ecosystem services

Johan Wasserman, Janine B Adams and Daniel A Lemley

Department of Botany and DSI/NRF Research Chair in Shallow Water Ecosystems, Institute for Coastal and Marine Research Nelson Mandela University, Gqeberha, South Africa

*Correspondence: johanwasserman7@gmail.com

Saltpans are increasingly being abandoned around the world, leading to the loss of ecosystem services provided by these unique semi-natural wetlands. The desertion of a saltpan at the Swartkops Estuary, South Africa has left behind a large area of desiccated hypersaline sediment and a sharp decrease in waterbird abundance and diversity. Here, we explore the potential for restoring this saltpan’s wetland function using stormwater inflow to support multiple ecosystem services and improve estuary health. Stormwater will be able to flow into the saltpan as a passive restoration approach that can maintain the site as a wetland habitat. This will contribute to improving the health of a nationally important, yet highly degraded, estuary by retaining 758 ML of stormwater to achieve full capacity, containing an estimated 2908 kg of dissolved inorganic nitrogen and 68 kg of dissolved inorganic nitrogen. Additional ecosystem services such as biodiversity maintenance, carbon storage and societal values can be expected. However, this would create a novel hypersaline stormwater wetland and a strategic adaptive management approach will be required. Management must be guided by monitoring, which should comprise collection of basic environmental data to establish a baseline condition, and against which, long term changes and responses resulting from stochastic events can be assessed and mitigated through the use of achievable ecological restoration targets.

Keywords: DIN, DIP, ecological restoration, hypersaline environments, stormwater management, waterbird conservation, water quality

Supplementary material: available online at https://doi.org/10.2989/16085914.2022.2067823

Introduction

As wetlands are in decline globally, investigating the potential for man-made wetlands to mitigate against the associated biodiversity losses has become increasingly relevant (Zhang et al. 2020). Saltpans represent semi-natural wetlands that have been created for salt production, although they provide unique habitats that support rich biota, particularly waterbirds, and supply a variety of ecosystem services such as biodiversity maintenance and water quality regulation (Dias et al. 2014; de Melo Soares et al. 2018). However, saltpans have been abandoned around the world (Crisman et al. 2009; Chefaoui and Chozas 2019; dos Reis-Neto et al. 2019; Tran et al. 2019), leaving behind large desiccated and hypersaline areas that culminate in a loss of ecosystem service provision. For example, the desiccation of saltpans leads to declines in biodiversity, including waterbirds (Crisman et al. 2009; Chefaoui 2021). Saltpan restoration has provided opportunities for reinstating ecosystem services (e.g., Crisman et al. 2009; Hewitson 2017; Dittman et al. 2019), but there remain opportunities to explore such restoration initiatives to reclaim otherwise highly disturbed coastal areas.

A saltpan at the Swartkops Estuary, South Africa, was abandoned in 2018 after nearly 60 years of salt production. The saltpan hosted thousands of waterbirds each year, including regionally important breeding colonies, and was a fundamental component of the Swartkops Estuary — Redhouse and Chatty Saltpans Globally Important Bird and Biodiversity Area (IBA) (Martin and Randall 1987). The site has now been left to dry, leaving behind a large (1.04 km²) barren area apparently devoid of birdlife. BirdLife International’s most recent assessment (in 2012) of the IBA found it to be in a very poor condition and under high threat with low management response (BirdLife International 2021). Habitat degradation and biodiversity loss has also been identified as a pressure on the health of the nationally important Swartkops Estuary (Adams et al. 2021). The process of gaining a Ramsar site designation for the Swartkops Estuary has been planned, with birdlife constituting a major component of the criteria to be met. However, with the loss of saltpan wetland functionality, the IBA status and potential Ramsar status for the area is under threat. To safeguard these designations and to improve the health of the Swartkops Estuary, it is necessary to restore the wetland habitat provided by the saltpan by reinstating its hydrological functioning. The restoration of the saltpan is currently being planned by the Zwartkops Conservancy, a local NGO, with the vision of creating a waterbird sanctuary.

Ideally, the entire saltpan should remain inundated throughout most of the year as this provided islands utilised by some of the largest breeding colonies of several shorebird species in the region, while water levels could be allowed to decrease over summer to provide foraging...
areas for large influxes of Palearctic migrant waders (Martin and Randall 1987). While the salt production process was underway, water from the adjacent Swartkops Estuary was actively pumped into the Redhouse saltpan. This approach will not be feasible in the restoration of the site due to the high running costs as well as the threat of vandalism and theft of the pumphouse infrastructure, which has historically been an issue while the saltpan was operational. Alternatively, a tidal connection can be created to convert the barren saltpan area into intertidal salt marsh, although this would require the entire site to be excavated to lower the elevation enough for the area to be inundated (Raw et al. 2021). This approach would also be beneficial in providing the additional ecosystem service of carbon storage.

An alternative approach that has been identified is to allow stormwater from a nearby canal to passively flow into the saltpan. Urban stormwater runoff is a major contributor to the deterioration of the health of the nationally important Swartkops Estuary (Adams et al. 2019). The estuary now persists in a highly degraded and eutrophic state, and experiences near-constant phytoplankton bloom conditions (Lemley et al. 2017; Adams et al. 2021). Concerningly, harmful algal blooms of the raphidophyte Heterosigma akashiwo have recently been recurring at the estuary (Adams et al. 2021). Diverting stormwater runoff into the Redhouse saltpan would be a more cost-effective approach for restoring this ecosystem, which has the potential to deliver multiple ecosystem services and improve the health of the estuary. In particular, it would help alleviate the severe pressure that anthropogenic nutrient loading currently exerts on the estuary, while also creating a wetland that functions as a waterbird habitat.

Baseline assessments of degraded sites are necessary to inform their restoration (Hobbs and Norton 1996; Gann et al. 2019). Such assessments allow for realistic goals to be set, provides insight into the required restoration actions, and guides monitoring the results of any actions taken. Excluding an avifaunal survey by Martin and Randall (1987), no information is available on the ecology of the abandoned saltworks at Swartkops Estuary, particularly in its current desiccated state. This study aimed to provide a baseline assessment of the vegetation and waterbirds at the abandoned saltpan, and to investigate the feasibility of restoring the site’s hydrological functioning using stormwater in order to inform the implementation and management of this restoration project. In doing so, this study highlights the potential of innovative restoration approaches that can address the global neglect of abandoned salt pans and other disturbed coastal environments for the provision of ecosystem services.

Materials and methods

Environmental characteristics and vegetation

The Redhouse saltpan lies on the floodplain of the Swartkops Estuary, near Gqeberha in the Eastern Cape of South Africa (Figure 1). Within the saltpan, environmental conditions were investigated to understand vegetation patterns at the saltpan in its abandoned state. Forty points were sampled within the saltpan across two consecutive days in September 2019. Sampling points were subjectively chosen to represent vegetated and unvegetated areas, with areas with vegetation cover exceeding 10% being classified as vegetated. At each sampling point, vegetation composition and cover were also recorded within 1 m² quadrats. Additionally, surface sediment (top 10 cm) samples were collected, and holes were augured to the top of the groundwater table. The depth to groundwater was recorded, as well as groundwater salinity using a Hanna HI98194 multiparameter meter. Sediment redox potential was recorded in situ using a Hanna HI918121 meter, while the collected sediment samples were taken to the laboratory for the determination of moisture content (Gardner 1965), organic content (Briggs 1977), particle size using the hydrometer method (Gee and Bauder 1986), and salinity using the ‘saturated paste’ method (Barnard 1990). Using the ‘vegan’ package in R (R Core Team 2020), non-metric multidimensional scaling (NMDS) analysis was applied to the vegetation abundance data with the recorded environmental characteristics fitted to the plot as vectors. This was done to assess which variables best explained the variability in vegetation cover at the saltpan. The predictor variables (i.e., environmental characteristics) were tested for collinearity using the ‘usdm’ package, with all variables returning variance inflation factors (VIF) ≤ 1.7. Furthermore, the annual change in vegetation cover since the abandonment of the saltpan was analysed through a desktop assessment of satellite imagery. Orthorectified Google Earth™ images were used to digitise vegetation as polygons in ArcGIS Desktop® 10.8, which were used to determine the area of vegetation cover at the saltpan.

Waterbirds

Waterbird abundance and species diversity data for the saltpan were obtained from the Coordinated Waterbird Counts (CWAC) programme. Counts at the Redhouse saltpan have been carried out biannually since 1994 — once in summer (January/February) and once in winter (June/July) by a single observer. The bird counts were conducted on foot with the observer traversing the saltpan from east to west. Each count was carried out over approximately an hour.

Stormwater volume and quality

The Motherwell stormwater canal and artificial wetland lie approximately 400 m east of the saltpan. The canal drains a large residential area and is intermittently subjected to sewage spills. The quality of the stormwater exiting the wetland was investigated over ten sampling occasions between April 2019 and March 2020. Salinity was recorded in situ using a YSI ProDSS multiparameter water quality meter. Water samples were collected for analysis of inorganic nutrients. Ammonium (NH₄⁺–N) and dissolved inorganic phosphorus (DIP = PO₄³⁻–P) concentrations were determined using the oxidation method (Parsons et al. 1984), while total oxidised nitrogen (NO₃⁻–N = NO₂⁻ + NO₃⁻) concentrations were determined using the reduced copper cadmium method (Bate and Heelas 1975). For these analyses, a ThermoScientific GENESYS 105 UV–Vis spectrophotometer was used. One-way ANOVAs were performed to test if there were significant differences in physico-chemical parameters and inorganic nutrient concentrations between sampling occasions. To determine the discharge of stormwater from
the artificial wetland, the head of water above a V notch weir was measured, and these measurements were applied to the Kindsvater–Shen equation (Kulin and Compton 1975). 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.

The volume of water required to fill the Redhouse saltpan was determined using the Surface Volume tool in ArcGIS Desktop® 10.8. The estimated daily inorganic nutrient loads that the saltpan could receive were subsequently calculated using the volume and the determined nutrient concentrations of the stormwater. The salinity that can be expected at the saltpan should it be filled with stormwater was estimated using a method adapted from Bate and Taylor (2008). This estimation considers sediment salinity, the surface area of sediment to be inundated, and salinity and total volume of stormwater that is expected to flow into the saltpan. The method was modified in that sediment salinity was measured using the previously mentioned ‘saturated paste’ method (Barnard 1990).

**Results**

**Environmental characteristics and vegetation**

The abandonment of the saltworks has left behind large expanses of desiccated hypersaline sediment (mean salinity $65.5 \pm 6.6$) overlying a hypersaline groundwater table (mean salinity $64.7 \pm 4.7$). Vegetation at the defunct saltpan was sparse and patchy and predominantly comprised of highly

---

**Figure 1:** The abandoned Redhouse saltpan at the Swartkops Estuary, South Africa with an inlay showing the full saltpan while it was functioning (photograph: Paul Martin)
monospecific stands of Salicornia pillansii, mostly limited to hummocks within the saltpan (Figure 2). Various other halophytic salt marsh species such as Suaeda fruticosa, Chenolea diffusa and Disphyma crassifolium were also recorded but in far lower abundance. These species typically inhabited the periphery of the salt pans and the most elevated areas and some of the hummocks.

Less than 18% (0.193 km$^2$) of the saltpan area was vegetated in the first year of abandonment. Within a year of desertion, vegetation cover decreased by 5%, but subsequently increased to 0.196 km$^2$ in the following year. Non-metric multidimensional scaling analysis revealed that groundwater salinity ($p < 0.01, r^2 = 0.28$) and elevation ($p < 0.01, r^2 = 0.24$) best explained variability in vegetation cover at the abandoned saltpan (Figure 3). All other recorded environmental parameters were found to be insignificant predictors ($p > 0.05$) of vegetation cover, although sediment salinity ($p = 0.07, r^2 = 0.13$) and groundwater depth ($p = 0.09, r^2 = 0.12$) also moderately contributed to the variability.

Waterbirds
While the saltworks was operational, the site regularly hosted thousands of waterbirds each year (Figure 4). Most waterbirds used the site for feeding, with most species being invertebrate-feeding waders that would overwinter at the site during the Austral summer. These large influxes of Palearctic migrants accounted for the generally greater waterbird abundance (mean 3 336 ± 373) and species richness (mean 31 ± 1) relative to winter. In winter, the saltpan hosted large breeding colonies of several resident shorebird species (Table 1; Martin and Randall 1987), with a mean waterbird abundance of 1 835 ± 161 and species richness of 28 ± 1. Several islands within the saltpan provided nesting areas protected from mammalian predators and egg poachers. The lowest waterbird numbers were recorded during periods of maintenance or disrepair when water levels in the saltpans were low (P Martin, Zwartkops Conservancy, pers. comm.).

Several of the species previously recorded at the Redhouse saltpan triggered the criteria necessary for the site to meet IBA and/or Ramsar site criteria (Table 1, also refer to Table S1). These species have all but disappeared from the saltpan since the site has been abandoned — just a few kelp gulls have been recorded. To safeguard the IBA and Ramsar designations, it is important that similar numbers of these species return to the site. Other important waterbirds previously recorded at the saltpan included large breeding colonies of Grey-headed Gull (Chroicocephalus cirrocephalus), Cape Teal (Anas capensis), African Sacred Ibis (Threskiornis aethiopicus), Common Ringed Plover (Charadrius hiaticula), Grey Plover (Pluvialis squatarola) and Little Grebe (Tachybaptus ruficollis). Only the colony of sacred ibis has been recorded roosting at the saltpan since its abandonment. None of the other large breeding colonies, nor the large influxes of Palearctic migrants have returned since the saltpan has dried up.

Stormwater volume and quality
In order to fill the Redhouse saltpan with stormwater, an approximately 400 m long pipe would have to be constructed from the outlet of the artificial wetland to the saltpan’s north-eastern corner (Figure 5). This point lies 1 m lower than the outlet of the artificial wetland, so stormwater will be able to flow into the saltpan without the use of pumps. Stormwater discharge from the artificial wetland was temporally stochastic and was correlated to rainfall, although the relationship was not strong ($p = 0.08, r = 0.57$). Stormwater has been observed to persistently flow into

![Figure 2: A hummock with highly monospecific (Salicornia pillansii dominant) vegetation adjacent to a large area of bare hypersaline sediment](image)

![Figure 3: Non-metric multidimensional scaling (NMDS) plot showing the relationship between environmental parameters and vegetation cover abundance among bare and vegetated sites at the abandoned Redhouse saltpan (k = 2; r = 0.61)](image)
the artificial wetland, even during periods of dry weather (Adams et al. 2019).

To fill the Redhouse saltpan to maximum capacity, a total of 758 ML of stormwater is needed. At this volume, the water surface area would be 0.926 km² and the mean depth would be 0.9 m. These water levels are the same as when the saltpan was full while salt production was still taking place and would therefore present a similar amount of potential foraging habitat once utilised by waterbirds here. The mean stormwater discharge from the artificial wetland was 0.08 m³ s⁻¹, which equates to 6.51 ML day⁻¹, or 2 375.07 ML year⁻¹. It would thus take approximately 116 days for the Redhouse saltpan to fill to maximum capacity. However, evaporative losses of 1 m can be expected as mean annual precipitation at the Swartkops Estuary is 636 mm, while annual evaporation is 1 650 mm (Reddering et al. 1981; Maclear 2001). Thus, there is a sufficient supply of stormwater to keep the saltpan inundated throughout the year despite evaporative losses.

Once the saltpan is inundated with stormwater, the salinity of the water column will be a function of the sediment salinity and surface area of the saltpan that would be inundated and the water volume and salinity. Should the saltpan be filled to maximum capacity, an initial salinity of 21 is estimated for the saltpan. It should be noted that this estimation does not account for the concentration of salts over time due to evaporation, nor dilution due to rainfall.

The stormwater exiting the Motherwell artificial wetland contained high dissolved inorganic nitrogen (DIN) concentrations (3.9 ± 0.3 mg l⁻¹) and relatively low dissolved inorganic phosphorus (DIP) concentrations (0.1 ± 0.01 mg l⁻¹). Inorganic nutrient concentrations also had significant temporal variability ($F_{9,1} = 5.64$, $p < 0.01$ for DIN and $F_{9,1} = 4.58$, $p = 0.01$ for DIP). No clear temporal trend was discerned and neither DIN nor DIP concentrations correlated with the stormwater discharge although loads were higher under higher flows, with a mean DIN load of $25.1 ± 2.05$ kg day⁻¹ and DIP load of $0.59 ± 0.12$ kg day⁻¹ (Figure 6). If filled to maximum capacity, it is estimated that the Redhouse saltpan could contain 27.55 to 133.51 kg DIP (mean = 68.48 ± 13.37 kg) and between 1 790.10 to 3 890.45 kg DIN (mean = 2 908.12 ± 237.76 kg).

**Figure 4:** Waterbird abundance and species richness at the Redhouse saltpan as recorded through the Coordinated Waterbird Counts (CWAC) programme (P Martin, unpub. data). Data for 2010 to 2013 were not available.
Important ecosystem services expected to arise from this restoration project include biodiversity maintenance through increasing the extent of aquatic habitat and improving the health of the degraded Swartkops Estuary by the creation of a stormwater retention pond (i.e. a nutrient sink). Additionally, various other ecosystem services are expected that coincide with the improved health of the estuary (Table 2). These include regulating services with ecological benefits like oxygen production and carbon storage, as well as numerous cultural services that would improve the societal value of the site. The societal (e.g. spiritual and cultural) value of the Swartkops Estuary could also be improved. For example, a popular site for baptisms lies near the outlet of the Motherwell stormwater canal, potentially putting the health of participants at risk. Furthermore, the simultaneous improvement of water quality and habitat restoration would improve the value of the site for education (e.g. the Zwartkops Conservancy’s programmes with local schools) and recreational activities. The estuary’s poor water quality has already led to the loss of a popular swimming event due to health concerns (Anon. 2015).

### Table 1: Maximum count of IBA and Ramsar criteria waterbird species recorded at the Redhouse saltpan since 1994. Data retrieved from Martin and Randall (1987), Crawford et al. (2009). A complete species list has been provided in Table S1

| Species                          | Redhouse saltpan | Criteria met                                      |
|---------------------------------|------------------|--------------------------------------------------|
| Caspian Tern \( ^b \)           | 64               | Ramsar site (> 8 individuals)                     |
| *Hydroprogne caspia*            |                  |                                                  |
| Curlew Sandpiper \( ^b \)      | 4187             | Sub-regional IBA (> 3 750 individuals)            |
| *Calidris ferruginea*           |                  |                                                  |
| Greater Flamingo \( ^b \)      | 836              | Ramsar site (> 760 individuals) and sub-regional IBA (> 625 individuals) |
| *Phoenicopterus roseus*         |                  |                                                  |
| Kelp Gull \( ^b \)             | 879              | Globally IBA (> 700 individuals)                  |
| *Larus dominicanus*             |                  |                                                  |
| Pied Avocet \( ^b \)            | 248              | Sub-regional IBA (125 individuals)                |
| *Recurvirostra avosetta*        |                  |                                                  |
| South African Shelduck \( ^b \) | 422              | Globally IBA (> 420 individuals)                  |
| *Tadorna cana*                  |                  |                                                  |
| Cape Shoveler \( ^b \)          | 315              | sub-regional IBA (> 175 individuals)              |
| *Spatula smithii*               |                  |                                                  |
| White-breasted Cormorant \( ^b \) | 183              | Ramsar site (> 130 individuals)                   |
| *Phalacrocorax lucidus*         |                  |                                                  |

\( ^b \) Species recorded breeding at the saltpan

![Figure 5: The Redhouse saltpan at maximum capacity, with the proposed pipeline from the artificial wetland indicated by the red line](image-url)
Discussion

**Baseline condition of the abandoned Redhouse saltpan**

The Redhouse saltpan, in its abandoned state, is depauperate relative to its previous condition while salt production was active. Waterbird abundance and diversity, in particular, have declined drastically. Without any water, the site no longer presents the once-important breeding sites for resident waterbirds and foraging sites for Palearctic migrant species. The desiccation of saltpans, or even just decreases in water levels, have also been shown to lead to substantial decreases in waterbird abundance and diversity elsewhere (Velasquez 1992; Múrias et al. 2002; Dias 2009; Chefaoui 2021). If the hydrological function of the saltpan is not reinstated, the site will not provide any value as a habitat for waterbirds and they will not return.

Vegetation at the saltpan is currently mostly limited to the elevated hummocks. The lower lying areas were kept almost permanently inundated while the saltpan was operational; as such, vegetation could not establish in the waterlogged soils. The vegetation recorded at the saltpan were typical supratidal salt marsh species that are
particularly sensitive to prolonged waterlogging (Tabot and Adams 2013). In addition to elevation, vegetation was also limited by groundwater salinity and to a lesser extent sediment salinity (which result from the previous management of the saltpan) and groundwater depth. These factors appear to explain why vegetation was highly monospecific with Salicornia pillansii as the dominant species as it well adapted to hypersaline conditions, can extend its roots to the water table and is able to tolerate hypersaline groundwater (Bornman et al. 2004; Steffen et al. 2010). Similarly, Bouzillé et al. (2001), González-Alcaraz et al. (2014) and Chefaoui and Chozas (2019) found that hydrological regimes implemented for salt production as well as sediment salinity are generally the drivers of vegetation patterns at abandoned saltworks.

Vegetation cover may continue to increase should the saltpan remain in its current desiccated state (i.e. if no restoration action is taken). This would help offset the 67% loss of supratidal salt marsh that has already occurred at the Swartkops Estuary (Adams 2020). Should the entire saltpan become vegetated, approximately 0.84 km² could be added to the existing 3.38 km² of supratidal salt marsh at the estuary. However, vegetation at the saltpan will likely remain highly monospecific unless the salts are flushed from the sediment, creating conditions more conducive for a variety of less salt-tolerant species to establish (Jolly et al. 1993; Neill 1993). This may happen naturally with rainfall, although it will likely take a long time. Furthermore, rainfall has been observed to create expansive long-standing pools of water in depressions within the saltpan, which would also limit the spread of vegetation throughout much of the site due to waterlogging of the soil. Also of concern is that the stressful conditions at disturbed environments often facilitate invasion by exotic plants (Almeida et al. 2014; Chefaoui and Chozas 2019). The invasive Atriplex lindleyi subsp. inflata, which is highly adapted to persist and reproduce in hypersaline environments (Atia et al. 2011) was noted at the Redhouse saltpan and is likely to spread throughout the site.

Implementing, managing and monitoring the restoration

The estimated volume of stormwater exiting the Motherwell artificial wetland appears to be sufficient to fill the Redhouse saltpan and to keep the saltpan permanently inundated despite the evaporative losses that will occur. It is anticipated that reinstating the hydrological function of the saltpan will re-establish the site as a waterbird habitat. This is not only important for the Swartkops Estuary, but also at a national scale, as waterbird populations have been in decline throughout South Africa (Raimondo et al. 2019). The saltpan will simultaneously act as an important nutrient sink for the Swartkops Estuary as it can, at maximum capacity, retain approximately 758 ML of stormwater containing an estimated 2 908 kg of DIN and 68 kg of DIP. This restoration project is therefore a promising intervention for improving the estuary’s water quality (particularly by decreasing DIN loading), which has experienced long-term deterioration with detrimental impacts such as harmful algal blooms (Adams et al. 2019, 2021; Lemley et al. 2019). Furthermore, this restoration project is expected to result in the provision of other ecosystem services such as carbon storage and increasing the cultural value of the area.

Restoring the Redhouse saltpan requires the construction of a pipeline that will continuously feed the saltpan with stormwater. Achieving the vision of restoring the saltpan as a waterbird sanctuary will require the management of water levels within the saltpan. Water depth and area are an important factor influencing the abundance and distribution of waterbirds at saltworks (Velasquez 1992; Dias 2009; López et al. 2010). Ideally, water levels should remain high from February (late summer) until the end of October (spring), when most of the important waterbird breeding colonies would be active at the saltpan (Martin and Randall 1987). High water levels allow for the formation of islands safe from small mammalian predators and egg poachers (Martin and Randall 1987). During summer, water levels can be left to decrease when Palearctic migrant waders utilise exposed mudflats for foraging. However, this restoration approach will not allow for active management of water levels in the saltpan, particularly increasing water depth and surface area. This should not be a major limitation as an excess of stormwater is expected each year. The constructed pipeline would require a mechanism to cut off flow to avoid overflow during high rainfall events, which would increase the water levels through increased stormwater runoff as well as rainfall directly into the saltpan. This will decrease the area of the islands used by waterbirds for breeding and possibly lead to the abandonment of nests (Ma et al. 2010).

The physico-chemical environment at the restored saltpan will be a challenge to manage due to the stochastic nature of stormwater. Furthermore, the discharge of stormwater into a hypersaline saltpan is expected to create a novel ecosystem within the Swartkops Estuary — one which has few comparable ecosystems. The only similar case found by the authors was the use of two former salt ponds being used as photosynthetic oxidation ponds in San Francisco Bay (Detweiler et al. 2014), although no information regarding water quality in these ponds was found. As water levels are no longer being managed by actively pumping estuary water into the saltpan, it will not be possible to manage water column salinity as is done in functional saltworks. Should the saltpan be filled to maximum capacity, the expected salinity of 21 is notably less saline than the salinity range (30–50) at which the saltpan was previously maintained (Martin and Randall 1987) but should allow for diverse primary producer communities to develop as is typical of low salinity saltponds (Britton and Johnson 1987; Davis 2000). The dominant vegetation noted in the saltpan while it was functional included the submerged macrophyte Ruppia cirrhosa and the macroalga Enteromorpha sp. (Martin and Randall 1987). The establishment of submerged macrophyte beds will be particularly desirable for rehabilitating the saltpan as a waterbird sanctuary as they support abundant and diverse invertebrate fauna that could in turn support rich avifaunal communities (Sánchez et al. 2006).

Over time, it can be expected that salinity will decrease at the saltpan as salts in the sediment are flushed away by the fresh stormwater, but it was not possible to quantify the degree to which this will occur nor the potential salinity variation over time. Initially, salinity variation is unlikely to significantly impact the growth of submerged macrophytes, particularly R. cirrhosa, which has a wide salinity tolerance.
range (0–70) although it thrives in less saline conditions (Adams and Bate 1994). However, should the salinity decrease continually, it is possible that reeds and sedges would become prevalent in the saltpan (Adams et al. 1992). The common reed Phragmites australis, which is widespread in South African estuaries, would grow favourably in the fresh, eutrophic conditions (Human and Adams 2011). The bulrush Typha capensis is currently planted in the artificial wetland that the stormwater passes through and can easily disperse to the saltpan. Salinity changes and the associated changes in macrophyte habitats will impact invertebrate communities, which in turn will influence waterbird abundance and diversity (Velasquez 1992; Evagelopoulos et al. 2008; Dias 2009). However, as it is not yet possible to model temporal salinity variation at the saltpan, it is difficult to speculate ecosystem trajectory and the implications for waterbirds.

Nutrient concentrations entering the saltpan, although highly variable, can be expected to be high overall. Diverting the nutrient-rich stormwater into the saltpan does raise the concern of eutrophication as salt pans are contained waterbodies with high water residence times. Eutrophication may be detrimental to the vision of restoring the site as a waterbird sanctuary as it may impact higher trophic levels through depleting oxygen from the water column or by having toxic effects on the waterbirds or their prey (Salgado et al. 2003; Shumway et al. 2003). However, it should be noted that the estuary water previously pumped into the saltpan was also highly eutrophic (Adams et al. 2019), yet waterbird abundance remained high.

As the ecosystem trajectory that will result from this restoration project is uncertain, it will be important to adopt a strategic adaptive management approach that is guided by monitoring. Following the onset of the restoration, frequent monitoring will be necessary to establish a baseline for the new wetland ecosystem. A similar approach to the South African National Estuarine Monitoring Programme (Colliers and Adams 2016) can be adopted, which is comprised of three tiers. The first tier would collect basic environmental data to develop the necessary baseline to inform future management actions. Data collection should cover physico-chemical parameters and environmental characteristics such as stormwater inflow, water depth and surface area, as well as the composition and abundance of the microalgae that arise. The second tier could consist of responses that are expected to take place over longer timescales, such as macrophyte abundance and composition and waterbird counts. The third tier is limited to monitoring events such as heavy rainfall and sewage spills in the stormwater canal, when rapid and distinct ecological responses may occur. Once the baseline has been established (first tier), specific ecological processes can be set to guide the management of the restored saltpan. These targets can be adjusted accordingly using information regarding long-term ecological responses to restoration efforts and those resulting from stochastic events (second and third tiers, respectively). While this study’s focus on water quality was limited to inorganic nutrients and salinity, it is recommended that other contaminants associated with stormwater runoff (e.g., heavy metals, microplastics and pharmaceuticals) are also considered in the design of a monitoring plan for the site.

This study has provided an initial scoping assessment for a novel saltpan restoration project that aims to recreate an important waterbird habitat and improve the health of an urban estuary. This project will be the first recorded case of saltpan restoration in South Africa and one of the first cases of transforming an abandoned saltpan into a stormwater pond. It presents a case study that can show how multiple ecosystem services could arise from a single restoration intervention. Furthermore, various research opportunities may arise including investigating the structure of biotic communities arising from saline stormwater environments (Olding, 2000; Van Meter et al. 2011) and 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).

Conclusion

The Redhouse saltpan is currently in a desiccated state characterised by hypersaline sediment and sparse, monospecific vegetation. If no restoration action is taken, the area may become revegetated and somewhat offset past losses of supratidal salt marsh at the Swartkops Estuary. However, the site also presents an environment susceptible to invasion by exotic plants such as A. lindleyi subsp. inflata. The most achievable vision of reclaiming the site as a waterbird sanctuary is to reinstate the hydrological function of the saltpan. The use of stormwater to re-establish a wetland habitat presents a passive restoration approach to do so, while simultaneously improving estuary health and deliver multiple ecosystem services including biodiversity maintenance, improving water quality and carbon storage. This restoration action will result in a novel stormwater wetland ecosystem and monitoring will be required to inform future management. Monitoring should generate basic environmental data, longer term responses and responses resulting from stochastic events. This study has identified a new potential use for an otherwise barren area that can improve estuary health and provide multiple ecosystem services. As coastal ecosystems are under pressure globally, it is becoming increasingly necessary to identify and investigate such innovative opportunities.

Acknowledgements — Dr Paul Martin is thanked for sharing data and knowledge of the study site. Members of the Zwartkops Conservancy are thanked for their valuable input and conversations (F Collier, D Clayton, J Rump and A Rump). This research is supported by the DSI/NRF Research Chair in Shallow Water Ecosystems (UID: 84375) and the Water Research Commission (project C2019/2020-0076) that provided the funding necessary to carry out field surveys and laboratory analyses. Nelson Mandela University is thanked for providing postgraduate research funding to Johan Wasserman. The National Research Foundation (NRF) of South Africa is also thanked for providing postdoctoral fellowship funding to co-author Dr Daniel Lemley (Grant Number: 120709).

ORCIDs

Johan Wasserman: https://orcid.org/0000-0003-0449-6042
Janine B Adams: https://orcid.org/0000-0001-7204-123X
Daniel A Lemley: https://orcid.org/0000-0003-0325-8499
Hewitson C. 2017. The Lion Salt Works, Northwich: a legacy of the Cheshire salt industry. *Industrial Archaeology Review* 39: 59–75. https://doi.org/10.1080/03090728.2017.1317141.

Hobbs RJ, Norton DA. 1996. Toward a conceptual framework for restoration ecology. *Restoration Ecology* 4: 93–110. https://doi.org/10.1111/j.1526-100X.1996.tb00112.x.

Human LRD, Adams JB. 2011. Reeds as indicators of nutrient enrichment in a small temporarily open/closed South African estuary. *African Journal of Aquatic Science* 36: 167–179. https://doi.org/10.2989/16085914.2011.589114.

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. https://doi.org/10.1016/0022-1694(93)90127-U.

Kulin G, Compton PR. 1975. A guide to methods and standards of water flow (Vol. 13). Gaithersburg: US Department of Commerce, National Bureau of Standards. 97 pp.

Lemley DA, Adams JB, Bornman TG, Campbell EE, Deyzel SH. 2021. Ecophysiology of salt marsh plants and potential implications for nearshore plankton dynamics. *Continental Shelf Research* 174: 1–11. https://doi.org/10.1016/j csr.2019.01.003.

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. https://doi.org/10.1016/j.ecolind.2016.11.034.

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. https://doi.org/10.1007/s10661-009-0998-2.

Ma Z, Cai Y, Li B, Chen J. 2010. Managing wetland habitats for waterbirds: an international perspective. *Wetlands* 30: 15–27. https://doi.org/10.1007/s13157-009-0001-6.

Maclean LGA. 2001. The hydrogeology of the Uitenhage Artesian Basin with reference to the Table Mountain Group aquifer. *Water SA* 27: 499–506. https://doi.org/10.4314/wsa.v27i4.4963.

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.

Múrias T, Cabral JA, Lopes R, Marques JC, Goss-Custard J. 2002. Eutrophication erodes inter-basin variation in macrophytes and co-occurring invertebrates in a shallow lake: combining ecology and palaeoecology. *Journal of Paleolimnology* 30: 311–328. https://doi.org/10.1007/s10933-017-9950-6.

Sánchez MI, Green AJ, Castellanos EM. 2006. Temporal and spatial variation of an aquatic invertebrate community subjected to avian predation at the Odíel salt pans (SW Spain). *Archiv für Hydrobiologie* 166: 199–223. https://doi.org/10.1127/0003-9136/2006/0166-0199.

Shumway SE, Allen SM, Dee Boersma P. 2003. Marine birds and harmful algal blooms: sporadic victims or under-reported events? *Harmful Algae* 2: 1–17. https://doi.org/10.1016/S1568-9883(03)00002-7.

Sim LL, Davis JA, Chambers JM, Streichow K. 2006. What evidence exists for alternative ecological regimes in salinizing wetlands? *Freshwater Biology* 51: 1229–1248. https://doi.org/10.1111/j.1365-2427.2006.01544.x.

Steffen S, Mucina L, Kadereit G. 2010. Revision of *Sarcocornia* (Chenopodiaceae) in South Africa, Namibia and Mozambique. *Systematic Botany* 35: 390–408. https://doi.org/10.1600/036364410791638379.

Tabot PT, Adams JB. 2013. Ecophysiology of salt marsh plants and predicted responses to climate change in South Africa. *Ocean & Coastal Management* 80: 89–99. https://doi.org/10.1016/j.ocecoaman.2013.04.003.

Tran HT, Wang HC, Hsu TW, Sarkar R, Huang CL, Chiang TY. 2019. Revetement on abandoned salt ponds relieves the seasonal fluctuation of soil microbiomes. *BMC Genomics* 20: 478. https://doi.org/10.1186/s12864-019-5875-y.

Van Meter RJ, Swan CM, Snodgrass JW. 2011. Salinization alters ecosystem structure in urban stormwater detention ponds. *Urban Ecosystems* 14: 723–736. https://doi.org/10.1007/s11252-011-0180-9.

Velasquez CR. 1992. Managing artificial saltpans as a waterbird habitat: species’ responses to water level manipulation. *Colonial Waterbirds* 15: 43–55. https://doi.org/10.2307/1521353.

Zhang C, Wen L, Wang Y, Liu C, Zhou Y, Lei G. 2020. Can constructed wetlands be wildlife refuges? A review of their potential biodiversity conservation value. *Sustainability* (Basel) 12: 1442. https://doi.org/10.3390/su12041442.

Manuscript received: 26 August 2021, revised: 30 March 2022, accepted: 13 April 2022

Associate Editor: W Froneman