THE STATE OF

SALDANHA BAY AND LANGEBAAN LAGOON

2020

Technical Report

October 2020

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Cover Photo: Steve Benjamin 2020
Diver Featured: Dr. Jessica Dawson

Citation: Clark BM, Hutchings K, Biccard A, Brown E, Dawson J, Laird M, Gihwala K, Swart C, Makhosonke A, Sedick S, Turpie J. and Mostert B. 2020. The State of Saldanha Bay and Langebaan Lagoon 2020, Technical Report. Report No. AEC 1876/1 prepared by Anchor Environmental Consultants (Pty) Ltd for the Saldanha Bay Water Quality Forum Trust, October 2020.

FOREWORD

The residents living in and around Saldanha Bay and Langebaan Lagoon are truly blessed to have such a unique ecological wonder on their doorstep. Visitors to our region continually confirm this view. It has taken millennia of natural processes to provide this phenomenon. The advent of man and his need to develop, almost at all costs, has the potential to destroy this gift within a short time. The question is - how do we balance the need to conserve our natural heritage with the requirement to develop and prosper economically? There is no simple answer to this very basic question.

The conservationists have shouted their 'green' messages from the treetops whilst the industrialists have simply argued the need to 'provide jobs and grow'. "Never the twain shall meet". We will all have to change our attitudes and work together to find the balance. This is a team effort. The government has taken the first steps in providing legal guidance with the proclamation of the National Environmental Management Act and the Integrated Coastal Management Act. These Acts still have a way to go before they have the required impact to provide the answer to our question.

Saldanha Bay has been identified as an economic development node by national government and the establishment of an Industrial Development Zone is well under way. The Bay hosts a major natural harbour and is actively exporting iron ore, lead, copper and manganese. To date, most environmental impact studies have been localized and the entire Saldanha Bay and Langebaan Lagoon ecological system has not been considered. The Saldanha Bay Water Quality Forum Trust has been instrumental in the establishment of the Integrated Governmental Task Team (IGTT) that has been given the mandate to address this problem and provide environmental guidance for all future development in and around our region and Saldanha Bay. The above-mentioned legislation plus the IGTT Environmental Guidelines will form the cornerstone to a balanced approach in terms of environmental sustainability, social wellbeing and economic growth in the future. The advent of "Climate Change" brings forth sea level rise and storm events. Thus, beach erosion and sediment movement are going to pose major challenges in the years ahead.

None of the above can take place without scientifically based information on the 'State of the Bay'. The Saldanha Bay Water Quality Forum Trust has been the pioneer in this regard and has conducted a series of all-encompassing scientific tests with minimal resources over the last 21 years. The report is once again a perfect example of the wonderful work that they perform. The report further comes at a critical time in answering our question of balancing conservation and development.

The Trust, like the rest of the World, has had to deal with the COV ID-19 Pandemic in this past year. Notwithstanding the challenges we were able to perform our monitoring function as an essential service during the "Lockdown". This has led to the innovation of Virtual Meetings and, for the first time, presenting the "State of the Bay" report as a webinar and in so doing expanding our reach to the various sectors of the community.

The Trust has also played an active role in the alignment of our monitoring programs within the Aquaculture Development Zone (ADZ) this past year. The Trust as a member of the ADZ Community Forum, has direct impact on the development of the ADZ where we attempt to ensure minimal impact on the Bay or the Lagoon.

The Trust is financially sound notwithstanding the closure of Arcellor Mittal Steel and PPC Cement factories. We have however identified and are busy negotiating with potential new contributors such as the Industrial Development Zone and 4 Special Forces Regiment. We are also in the process of building a financial reserve to carry the Trust through the difficult economic times that lie ahead.

In conclusion let us all, National, Provincial and Local Government with the Private Sector and Non-Governmental Organizations, as partners, take hands and make a difference in conserving Saldanha Bay and Langebaan Lagoon for future generations whilst ensuring responsible development.

Alderman André Kruger

Portfolio Chairperson: Community and Operational Services
Saldanha Bay Municipality
Chairperson Saldanha Bay Water Quality Forum Trust

Figure I. **SBWQFT Trustees. From left, Ethel Coetzee, Pierre Nel (SANParks), André Kruger (Saldanha Bay Municipality Councillor), Elmien de Bruyn (Duferco), and Christo van Wijk (Metsal).**

Acknowledgement and thanks go to SANP for assisting with data and services as well as a special thanks to all the contributing institutions that assist the SBWQFT throughout the year.

EXECUTIVE SUMMARY

Regular, long-term environmental monitoring is essential to identify and to enable adaptive management and mitigation of negative human impacts on the environment (e.g. pollution), and in so doing maintain the beneficial value of an area for all users. This is particularly pertinent for an area such as Saldanha Bay and Langebaan Lagoon, which serves as a major industrial node and port while at the same time supporting important biodiversity, tourism and fishing industries. The development of the Saldanha Bay port has significantly altered the physical structure and hydrodynamics of the Bay, whilst all developments within the area (industrial, residential, tourism etc.) have the potential to negatively impact on ecosystem health.

Saldanha Bay and Langebaan Lagoon have long been the focus of scientific study and interest, owing to its conservation importance as well as its many unique features. The establishment of the Saldanha Bay Water Quality Forum Trust (SBWQFT) in 1996, a voluntary organization representing various organs of State, local industry and other relevant stakeholders and interest groups, gave much impetus to the monitoring and understanding of changes in the health and ecosystem functioning of this unique bay-lagoon ecosystem. Direct monitoring of a number of important ecosystem indicators was initiated by the SBWQFT in 1999, including water quality (faecal coliform, temperature, oxygen and pH), sediment quality (trace metals, hydrocarbons, total organic carbon (TOC) and nitrogen) and benthic macrofauna. The range of parameters monitored has expanded since then to include surf zone fish and rocky intertidal macrofauna (both initiated in 2005) and led to the commissioning of a "State of the Bay" technical report series in 2006. This report has been produced annually since 2008, presenting data on parameters monitored directly by the SBWQFT as well as those monitored by others (government, private industry, academic establishments, and NGOs).

In this 2020 State of the Bay report, available data on a variety of physical and biological topics are covered, including activities and discharges affecting the health of the Bay (e.g. residential and industrial development, dredging, coastal erosion, shipping, mariculture, fishing and fish processing, sewage and other wastewaters), groundwater inflows, water quality in the Bay itself (temperature, oxygen, salinity, nutrients, and pH), sediment quality (particle size, trace metal and hydrocarbon contaminants, total organic carbon and nitrogen), and ecological indicators (aquatic macrophytes, benthic macrofauna, fish, birds and seals). Where possible, trends and areas of concern have been identified and recommendations for future monitoring are presented, with a view to further improving the environmental management and monitoring in the area. Key findings for each of the major components of the State of the Bay monitoring programme are summarised below.

Activities and discharges affecting the Saldanha Bay and Langebaan Lagoon

Major developments in the Bay itself over the last 50 years include the development of the Port of Saldanha (Marcus Island causeway and the iron ore terminal and associated infrastructure), the establishment of the small craft harbour, several marinas, mariculture farms and several fish processing factories. Extensive industrial and residential development has also become established on the periphery of the Bay. Anthropogenic pollutants and wastes find their way into the Bay from a range of activities and developments. These include port operations, shipping, ballast water discharges and oil spills, export of metal ores, municipal (sewage) and industrial discharges, biological waste associated with mariculture and storm water runoff. Urban and industrial developments encroaching into coastal areas have resulted in the loss of coastal habitats and have affected natural coastal processes, such as sand movement. Development of the port is expected to increase dramatically with the establishment of the Saldanha Bay Industrial Development Zone (SBIDZ), a process that was initiated in 2013.

Human settlements surrounding Saldanha Bay and Langebaan Lagoon have expanded tremendously in recent years. This is brought home very strongly by population growth rates of 2.7% per annum in Saldanha and 9.24% in Langebaan over the period 2001 to 2011. Numbers of tourists visiting the Saldanha Bay and Langebaan Lagoon area increased constantly, especially those visiting the West Coast National Park (WCNP) (average increase of 12% per annum since 2005). The limitations implemented due to COVID 19 restrictions resulted in lower visitor numbers for the 2019/2020 year, however, overall numbers were still similar to those seen between 2009-2014, and will hopefully increase with the reopening of national tourism. This rapid population and tourism growth translates to corresponding increases in the amount of infrastructure required to house and accommodate these people, and in the amount of waste and wastewater that is produced which must be treated and disposed of.

Metal ores exported from the Port of Saldanha Bay include iron, lead, copper, zinc, and manganese. The Port of Saldanha currently has the capacity to export up to 60 million tonnes of iron ore per year but is in the process of upgrading the infrastructure to support an annual export of 80 million tonnes. However, the Transnet Port Terminals (TPT) have thus far been unsuccessful in obtaining a variation to their existing Air Emission License (AEL) applicable to the Iron Ore Terminal for the storage and handling of the ore. The latest application was for the increase of handling and storage of coal and ore to 67 million tonnes per annum and was accompanied by an impact assessment and public participation process. The competent authority denied TPT the amendment concluding that environmental impacts at the current production level are already too high. In addition, the Provisional Air Emission License (PAEL) granted for the storage and handling Manganese at the Multi-Purpose Terminal (MPT) has been appealed and set aside. The competent authority decided that the activities required for the storage of Manganese in the port, specifically the need for the expansion of current storage facilities, should have triggered an EIA prior to the issuing of the PAEL, therefore the appeal was upheld and the PAEL set aside. Manganese volumes being stored at the MPT have subsequently been significantly reduced. Note that the TPT Iron Ore Terminal and the TPT Multipurpose Terminal are registered as two separate entities that hold separate port licences.

Disposal of wastewater is a major problem in the region, and much of it finds its way into the Bay as partially treated sewage, storm water, industrial effluent (brine, cooling water discharges and fish factory effluent) and ballast water. Until recently sewage discharge was arguably the most significant waste product that is discharged into Saldanha Bay in terms of its continuous environmental impact. Sewage is harmful to biota due to its high concentrations of nutrients which stimulate primary production that in turn leads to changes in species composition, decreased biodiversity, increased dominance, and toxicity effects. The Saldanha Bay Municipality (SBM) has, however, made a number of improvements both to existing wastewater treatment facilities and in the management of the effluent discharged from the plants. The majority of the wastewater is now being used for irrigation, and current water users receiving treated effluent include: the Weskus School, Saldanha Sports Ground (Stadium and practise field), Blue Bay Lodge (all three from Saldanha WWTW) and the Langebaan Country Estate (Water received from Langebaan plant). It is reported that no effluent from the Langebaan Wastewater treatment works is entering the Bay. However, subsequent to the closure of ArcelorMittal Steel Works the Saldanha plant no longer receives industrial effluent from the plant and no longer supplies treated effluent to the ArcelorMittal Reverse Osmosis plant. Therefore, the balance of treated effluent from Saldanha Bay WWTW that is not used for irrigation is currently discharged into the Bok river and ultimately ends up in the ocean. However, SMB has identified a future user for the treated effluent and an allocation has been made available to them. Additionally, SBM in collaboration with Sea Harvest have initiated a project to install litter traps on stormwater drains to minimize pollution entering the bay via these waterways.

Ballast water discharge volumes are increasing over time as shipping traffic and the overall size of ships visiting Saldanha Bay increases. The total number of ships entering the Port of Saldanha has increased substantially over the years from 262 ships in 1994/5 to peak at 616 in 2018/19 rolling year, before dropping slightly this past year to 571 (a decrease that is likely related to the COVID 19 Pandemic). As a result, the volume of ballast water discharged more than tripled between 1994 and its peak volume recorded in 2017/18, from 8.2 to 25.1 million tonnes. Vessels docking at the iron ore terminal have a higher average volume of ballast water discharge than other vessel types, with volumes increasing from 54.4 thousand tonnes per vessel in 2003/4, peaking at 78.6 thousand tonnes in 2015/16 and dropping to 71.2 thousand tonnes in 2018/19. Only total ballast water volumes for the entire port were available for 2019/20, however, when comparing the average discharge for all vessels combined in 2019/20 to similar data for the period 1994/5 to 2001/2, we see that volumes have increased by more than one third over historic volumes. Ballast water can and often does include high levels of contaminants such as trace metals and hydrocarbons, and, along with the vessels that carry the ballast water, serves to transport alien species from other parts of the world into Saldanha Bay. Ballast water discharges can, however, be effectively managed and the remit of the International Maritime Organisation (IMO) is to reduce the risks posed by ballast water to a minimum through the direct treatment of the water while on board the ship, as well as by regulating the way in which ballast water is managed while the ship is at sea. Although no domestic legislation is currently in place to regulate ballast water discharge (the Ballast Water Management Bill remains in draft format), the Transnet National Port Authority in Saldanha Bay have a well-recognised management plan in place and have implemented several mechanisms to track and control the release of ballast water into the harbour.

Dredging in Saldanha Bay has had tremendous immediate impact on benthic micro and macrofauna, as extensive dredging has been conducted in the bay in the past, and particles suspended in the water column during dredging can kill suspension feeding taxa including zooplankton, benthic macrofauna, and in severe cases fish. It also limits the penetration of sunlight in the water column and can cause die offs of seaweeds and phytoplankton. Furthermore, fine sediment can drift into the Langebaan Lagoon, changing the sediment composition, which in turn can directly and indirectly affect birds in the lagoon. The damage caused by dredging is generally reversible in the long term, and although the particle size composition of the settled material is likely to be different, ecological functioning as well as major species groups generally return in time. The most recent dredging occurred in September and October 2019 and February 2020, whereby 20 668 m3 of sediment was dredged for the upkeep and maintenance of the OSSB quay and channel, and the Mossgas channel.

Saldanha Bay is a highly productive marine environment and constitutes the only natural sheltered embayment in South Africa. These favourable conditions have facilitated the establishment of an aquaculture industry in the Bay. In January 2018 the then Department of Agriculture, Forestry and Fisheries (DAFF now DEFF) was granted Environmental Authorisation to establish a sea-based Aquaculture Development Zone (ADZ) in Saldanha Bay and expand the total area available for aquaculture in the Bay to 884 ha, which is located within four precincts (Small Bay, Big Bay North, Outer Bay North and South). In 2018, it was reported that of the new established area, 151 ha was being actively farmed. By the end of December 2019, approximately 36% of the ADZ had been leased, but less than 60% of the actively leased area was being utilised and this value is constantly changing as new leases are being granted, new farms start, current lease holders expand their areas, or alternatively shrink in size, based on economic factors. As of March 2020, 28 companies within the Saldanha Bay ADZ were registered on the Marine Aquaculture Right Register, of which only 15 companies were actively operational.

Historic studies as well as the State of the Bay surveys have shown that these culture operations can lead to organic enrichment and anoxia in sediments under the culture rafts, and ropes. DEFF were required to appoint an Environmental Control Officer and to set up two committees - a Consultative Forum (CF) that includes representatives of public and industry and an Aquaculture Management Committee (AMC) (government representatives only) - to ensure that the implementation of the ADZ is in line with the requirements specified in the Environmental Authorisations and Environmental Management Programme. Additionally, a Baseline Benthic Monitoring Report and the Annual Redox Survey Report for the Saldanha Bay ADZ have recently been made public, summaries of which are included in this report. DEFF are also continuing with in-house environmental monitoring which includes a rapid synoptic survey of oxygen and nutrient levels in the Bay (results of which are also included here – Chapter 6).

The source of the contamination from aquaculture operations comprises mainly of faeces, decaying mussels and fouling species. The scale of the proposed ADZ is significant and we believe that the scale of environmental monitoring in the Bay needs be intensified to allow for adaptive management of any significant ecological impacts, as well as loss to the mariculture sector itself.

Management and policy development

Continuously accelerating urban and industrial development is a major cause of fragmentation and loss of ecological integrity of remaining marine and coastal habitats in Saldanha Bay and Langebaan. The challenge of addressing cumulative impacts in an area such as Saldanha is immense. The current and future desired state of the greater Saldanha Bay area is polarised, where industrial development (Saldanha Bay Industrial Development Zone (IDZ) and associated industrial development) and conservation areas (Ramsar Site, Marine Protected Areas (MPAs) and National Parks) are immediately adjacent to one another. Furthermore, the Saldanha Bay environment supports conflicting uses including industry, fisheries, mariculture, recreation, and the natural environment itself. This situation necessitates sustainable development that is steered towards environmentally more resilient locations and away from sensitive areas.

Concerns have been raised that cumulative impacts on the marine environment in Saldanha Bay have not been adequately addressed for many recent development proposals. This applies especially to the cumulative impacts that are anticipated from future development within the Saldanha Bay IDZ and Aquaculture Development Zone (ADZ). Furthermore, the impact on the Saldanha Bay marine environment from projects that are primarily land-based, such as storage facilities for crude oil and liquid petroleum gas, has often been underestimated or even ignored. It has been proposed that a more holistic management strategy is needed to deal with these piece-meal Environmental Impact Assessments (EIAs). Various environmental management instruments have been proposed for the Greater Saldanha Bay Area, including (1) a generic Environmental Management Programme (EMPr), (2) an Environmental Management Framework (EMF), (3) a Strategic Environmental Assessment (SEA), and (4) declaration of Saldanha Bay as a Special Management Area. An Intergovernmental Task Team (IGTT) has been established to consider these and other proposals. If these management instruments are indeed implemented, we are confident that measures for the conservation alongside rapid development of the Saldanha Bay area will be addressed more effectively.

Groundwater

Within the Greater Saldanha Bay (GSB) area on the Cape West Coast, groundwater is a key component of the natural capital within the area. It plays a crucial role in sustaining critical and unique ecosystems and is also a backup source (and possible future primary source) of water supply to the municipality as well as providing support to the agricultural sector. The geological setting is complex and highly variable within the GSB area and for this reason, the groundwater is also highly variable across the study area in terms of flow rates, volumes and quality. A lot of geohydrological work has been completed in the area, dating back to the 1970s. There has also been a lot of recent and on-going geohydrological work with the establishment of the Elandsfontein Phosphate Mine; the extension of the Langebaan Road Aquifer wellfield and the establishment of the new Hopefield wellfield. In places, the aquifer/s are very high yielding with good groundwater quality yet in other areas there is essentially no groundwater and if present it is very saline. The potential for groundwater use within the GSB has been recognised for a long time and for this reason the Department of Water Affairs (as it was known at the time) declared a Subterranean Government Water Control Area within the Greater Saldanha Bay area in the 1970s essentially reserving groundwater for municipal use.

Freshwater flows occur into the southern end of the Langebaan Lagoon, in the Geelbek area, and these inflows play a key role in causing and maintaining the biodiversity of that area. The area is a declared Ramsar site. These freshwater inflows are groundwater driven and have to be maintained. For the entire GSB area, this is where the ecological role of groundwater' is the most crucial. Ten kilometres up-gradient of this area is the Elandsfontein phosphate mine. The mine is dewatering the groundwater (in order to access the ore body), however, the abstracted groundwater is recharged two kilometres down-gradient of the open pit with essentially no nett abstraction of groundwater occurring. A comprehensive groundwater monitoring network is in place to track the mine's activities. There are also monitoring boreholes at the Langebaan Road Aquifer and Hopefield wellfields and the monitoring thereof is being managed by the Saldanha Bay Municipality. There are also many boreholes across the area that the Department of Water and Sanitation (DWS) monitors on a regular basis.

The potential bigger impacts on groundwater are likely to be from (1) the agricultural sector (1.6 Mm3/a) (this registered quantity is groundwater abstraction for agriculture as at 2016 and probably increased significantly during the drought of 2015 to 2018); (2) abstraction from the Langebaan Road

Aquifer wellfield (intermittently operational since 1999, however, frequently non-operational due to regular and persistent vandalism); and (3) the Hopefield wellfield (not yet operational) where it is planned to abstract 5.1 Mm3/a and 1.8 Mm3/a, respectively. As mentioned, the Langebaan Road Aquifer has been operational for 20 years but only intermittently and the long-term monitoring trend shows only slight groundwater level drawdowns. The total utilisable groundwater exploitation potential (UGEP) under normal conditions is 15.2 Mm^3/a from the SBM area, so it is important to try and reduce the impact of this nett abstraction by using Managed Aquifer Recharge methodologies and it is possible that the wellfields will only be used in times of severe drought, so they need to be kept as "full" as possible in non-drought times. If the UGEP is adhered to, there is unlikely to be an impact on the outflow to the marine environment, however, the positioning of the abstraction is crucial to ensure there is no impact on these outflows. Comprehensive groundwater monitoring and associated database maintenance within the entire region is also essential for the long-term management and preservation of the aquifers and freshwater inflows into the Langebaan Lagoon. Within the Greater Saldanha Bay area, it is imperative to ensure all groundwater abstraction above the General Authorisation limit is authorised and that the associated compliance conditions are adhered to.

Elandsfontein Exploration and Mining (Pty) Ltd/Kropz recently started mining phosphate deposits in the area of the Elandsfontein Aquifer System on the eastern side of the R27. Mining is being conducted using an open-pit strip mining method which requires that groundwater levels around the mining pit be lowered to prevent the mine pit from being flooded. Groundwater is being abstracted from a series of boreholes surrounding the mine pit but is reinjected downstream (towards the lagoon), in an effort to ensure that surrounding ecosystems (including the Lagoon) are not affected. Available evidence suggests that this activity is unlikely to impact on the lagoon, however, Kropz Elandsfontein in conjunction with the Saldanha Bay Water Quality Forum Trust (SBWQFT) elected to initiate monitoring a range of biological and physico-chemical variables associated with Langebaan Lagoon to establish an appropriate baseline against which any potential future changes in the Lagoon can be benchmarked. This includes monitoring of temperature and salinity (see below) and biota (benthic macrofauna) as well as macrophytes (vegetation) around the top end of the lagoon.

Monitoring of temperature and salinity at the head of the lagoon was initiated in September 2016 using a Star ODDI Salinity, Conductivity, Temperature and Depth Logger. The Star ODDI was subsequently replaced with an Aqua TROLL 200 data logger (August 2019) which has been yielding considerably better and more useful data. These data records show clear diurnal/tidal and seasonal trends in water temperature and salinity. The diurnal fluctuations in temperature are similar across all seasons, with temperatures increasing over the course of the day, peaking in the early afternoon, then declining through the afternoon and night, reaching a minimum at the time of sunrise each day. The trend in salinity is more interesting though, exhibiting a similar diurnal oscillation to that for temperature, but this oscillation is linked to the state of the tide (not the time of day) and changes through the year could become much more pervasive if freshwater outflow from the aquifer were to drop in future. Also of interest in these data, there appears to be no link between rainfall and salinity levels in the lagoon which strongly suggests that variations in salinity in the lagoon are linked with groundwater inflow as opposed to surface water inflow, which is consistent with observations made by others

Currently, the groundwater status in the area surrounding Saldanha Bay and Langebaan Lagoon is stable, and no significant nett abstraction is occurring within the region. In the industrial areas which overlie saline groundwater, it is still imperative that groundwater quality monitoring takes place, especially for metals, nutrients and hydrocarbons, as the groundwater flows towards the Bay and Lagoon and groundwater will be a transport medium for nutrients and contaminants. Groundwater level and quality monitoring must continue at the two municipal wellfields, even though the wellfields are not in use; and monitoring must also continue at the Elandsfontein Mine and down-gradient right up to Langebaan Lagoon. The latter facility is being monitored and the results indicate there are no issues of concern currently. All groundwater monitoring data should be made publicly available.

Water quality

Aspects of water quality (temperature, salinity, dissolved oxygen, nutrients and chlorophyll concentrations) are currently, or have in the past been studied in Saldanha Bay, to better understand changes in the health of the environment. Regional oceanographic processes appear to be driving much of the variation in water temperature, salinity, dissolved oxygen, nutrients and chlorophyll concentrations observed in Saldanha Bay. However, there is clear evidence of altered current strengths, circulation patterns and wave energy within the Bay, which are ascribed to the construction of the ore terminal and causeway. These changes have also contributed to the deterioration in water quality in Small Bay in particular.

The water entering Small Bay appears to remain within the confines of the Bay for longer periods than was historically the case. There is also an enhanced clockwise circulation and increased current strength flowing alongside unnatural obstacles (i.e. enhanced boundary flow, for example alongside the ore terminal). The wave exposure patterns in Small Bay and Big Bay have also been altered as a result of harbour developments in Saldanha Bay. The extent of sheltered and semi-sheltered areas has increased in Small Bay, while wave exposure has increased in some areas of Big Bay leading to coastal erosion.

The increase in the frequency of Small Bay hypoxic events (periods of low dissolved oxygen <2 ml/l) occurred after the major harbour development in the 1970s, and the situation does not appear to have changed much since with similar data collected by continuous dissolved oxygen measurements around the turn of the century to those collected during the autumn-winter period in 2020. New data show that hypoxic and near anoxic conditions in the lower part of the water column are frequent occurrences during the summer-autumn season in Big and Small Bay (pointing to an external upwelled source of low oxygen water); whilst in Small Bay, anthropogenic organic loading appears to exacerbate the situation with decreased dissolved oxygen measured at sites under mariculture farms compared to control sites.

Regular monitoring of microbiological indicators at 20 stations in the Bay (ten in Small Bay, five in Big Bay and five in Langebaan Lagoon) was initiated by the Saldanha Bay Water Quality Forum Trust (SBWQFT) in 1999 and has continued with the assistance of the West Coast District Municipality. These data indicate that chronic faecal coliform pollution was present in the early parts of the record but that conditions have improved considerably over time since then. Two stations have “Poor” water quality, the remaining 18 monitoring stations in the Bay and Lagoon are rated as having “Fair” (5 stations), “Good” (2 stations) or “Excellent” (13 stations) water quality. The two beach sites in the vicinity of the Bok River Mouth were rated as ‘Fair’ and “Good” in 2019/2020, representing a sustained improvement over most earlier samples collected at these sites. This likely reflects improved treatment at the wastewater treatment works after upgrades in 2018 and high levels of wastewater reuse for industrial and irrigation purposes. It is a concerning that faecal coliform levels at the Hoedjiesbaai Beach remain elevated with "Poor" water quality recorded at this site for the last three years; as is the decline in water quality at the Pepper Bay Big Quay station over the 2019-20 period. Local authorities are advised to try to determine and remedy the sources of this pollution (probably stormwater run-off).

Four of the ten monitoring sites in Small Bay did not meet the 80th percentile faecal coliform limits for mariculture in 2020. Faecal coliform counts at all four sites in Big Bay were, however, within the 80th percentile limits for mariculture in 2020. Given the current importance and likely future growth of both the mariculture and tourism industries within Saldanha Bay, it is imperative that whatever efforts have been taken in recent years to combat pollution by faecal coliforms in Small Bay (e.g. upgrading of sewage and storm water facilities to keep pace with development and population growth) should be increased and applied more widely. Continued monitoring of bacterial indicators (with the inclusion of intestinal Enterococci), to assess the effectiveness of adopted measures, is also recommended and should be undertaken at all sites on a bimonthly basis.

Concentrations of trace metals in marine organisms (mostly mussels and oysters) in Saldanha Bay have historically been routinely monitored by the Department of Environmental Affairs (DEA) and by mariculture farm owners. DEA discontinued their Mussel Watch Programme in Saldanha Bay in 2007, but this has now been incorporated into the annual State of the Bay surveys. Data show that concentrations of trace metals are higher along the north eastern shore of Small Bay (particularly for lead and manganese). Concentrations of trace metals in cultured mussels offshore are typically lower; although concentrations of lead and cadmium in mussels farmed in Small Bay were on occasion above the limit for foodstuff prior to 2016, and cadmium exceeded guidelines in outer bay samples in recent years (2018-2020), which is concerning. The reasons for the lower concentrations of lead and cadmium in farmed mussels compared with those on the shore may be linked with higher growth rates for the farmed mussels, and the fact that the cultured mussels are feeding on phytoplankton blooms in freshly upwelled water that has only recently been advected into the Bay from outside and is thus relatively uncontaminated.

The high concentrations of lead and cadmium in mussels sampled from the shore in Small Bay points to the need for management interventions to address this issue, as metal contamination poses a serious risk to the health of people consuming mussels. It is vitally important that this monitoring continues in the future and that data are made available to the public for their own safety.

Sediments - shoreline stability

Beach erosion in Saldanha Bay, particularly at Langebaan Beach, has been the subject of some concern in recent years as coastal developments in Langebaan and Saldanha extend right to the water's edge and are at risk from a retreating shoreline. The origins of this erosion are not clear but most experts feel that it is likely due to natural causes that may have been exacerbated as a result of the construction of the iron ore terminal and associated infrastructure (Marcus Island causeway). Sediment used to build the causeway to Marcus Island was sourced from the historic ebb tide delta that existed at the mouth of Langebaan (an area where sediment derived from Langebaan Lagoon had been deposited over many thousands of years). Removal of sediment from this area resulted in a reduction in the extent to which incoming waves are refracted and a concomitant increase in the wave energy density along the shoreline by around 50%. It is thought that this, in turn, is what has caused the observed erosion of the shoreline.

A number of interventions have been introduced over the years in an effort to control the erosion, and to limit the loss of sediment from these beaches. This includes construction of rock revetments along Langebaan Beach (1997-2002), construction of groynes extending perpendicularly out from the shore at Langebaan Beach (2004-2008) and construction of gabion walls on Paradise Beach.

The Saldanha Bay Municipality initiated an erosion monitoring programme in 1994, designed to monitor change (erosion/accretion) in the beaches between Leentjiesklip 1 (Strandloper restaurant) and Alabama street. This entails undertaking beach surveys bi-annually - at the end of summer (Apr/May) and the end of winter (Oct/Nov) during spring low tide. This programme continues but has now been taken over by the SBWQFT. Data from this monitoring programme indicates that the seasonal patterns of erosion and accretion are complex and are to some extent reversed for the northern and southern portions of Langebaan Beach. For most of the monitoring period (1994-2020), Langebaan North Beach (the section between the Strandloper restaurant and Groyne 1), erodes in winter and accretes in summer with some reversal evident in the middle part of the record. The opposite is true for Langebaan South Beach, which typically erodes in summer and accretes in winter, again with some evidence of reversal in the middle part of the record. It is believed that this seasonal reversal in these erosion and accretion patterns is linked to the seasonal reversal of the wave climate experienced at these two sites, while the reversal of these processes in the middle of the record is linked with beach nourishment that was undertaken when the groynes were being constructed. Wave energy at Langebaan North Beach is typically much more intense in winter (waves striking the shore here approach from offshore and are generated by storms passing the Cape in winter) while wave energy at Langebaan South Beach is more intense in summer (and is derived from the southerly winds blowing across the Lagoon at this time of year).

Also, very clear in the long-term data, is the impact of the various interventions that were introduced to mitigate or control erosion on these beaches, particularly the two groynes. On Langebaan North Beach, data reveal a progressive loss of sand from Langebaan North Beach between 1994 and 2003, followed by a period of accretion between 2004 and 2007, and then a period from 2008 onwards where an equilibrium has been reached and the amount of sand on the beach remains more or less constant from year to year, albeit much reduced relative to the starting point in 1994. At Langebaan South Beach, there is some evidence of accretion across the first part of the record (1994-2003) prior to the construction of Groyne 1, followed by a period of erosion from this time up to 2015, following which the beach width has remained more or less static at a point corresponding with that observed at the start of monitoring in 1994.

Overall, these data suggest that the construction of the groynes was a very necessary and successful intervention and that it is very important that monitoring continue in future to confirm that this pattern continues going forwards. Additional interventions to enable reestablishment of beach habitat along the shoreline on the section north of the area currently being monitored is probably also warranted, as the shoreline here is currently made up of a rock revetment which makes access to the sea for people living in this area very difficult and dangerous.

Sediments -sediment quality

The distribution of mud, sand and gravel within Saldanha Bay is influenced by wave action, currents and mechanical disturbance (e.g. dredging). Under natural circumstances, the prevailing high wave energy and strong currents would have flushed fine sediment and mud particles out of the Bay, leaving behind the heavier, coarser sand and gravel fractions. However, obstructions to current flow and wave energy can result in increased deposition of finer sediment (mud). Large-scale disturbances of sediments (e.g. dredging) also re-suspends fine particles that were buried beneath the sand and gravel and these later settle in areas where water movement is reduced. Contaminants (trace metals and toxic pollutants) associate with fine sediment (silt and mud) and can have a negative impact on the environment when they are re-suspended. Accumulation of organic matter in benthic sediments can also give rise to problems as it depletes oxygen both in the sediments and surrounding water column as it decomposes. Prior to large scale development in the Bay, it was reported that the proportion of fine material (silt and mud) in the sediments of Saldanha Bay was very low. Reduced water circulation in the Bay and dredging activities have resulted in an overall increase in fine material in sediments in the Bay. The most significant increases have been observed following dredging events. Data collected as part of the State of the Bay surveys since 1999 has shown a progressive decline in the amount of fine sediment (mud) to levels similar to those last seen in 1974. Despite these overall encouraging trends, the sediment at several deeper or more sheltered sites within Small and Big Bay still have elevated mud fractions. Areas most significantly affected in this way are all located in the vicinity of the iron ore terminal, the mussel rafts and the Yacht Club Basin, however, the mud fraction at these sites decreased again in the latest (2020) survey.

Levels of total organic carbon (TOC) and total organic nitrogen (TON) are also elevated in the more sheltered and deeper areas of the bay, notably near the Yacht Club Basin and Iron Ore Terminal. Phytoplankton production is still considered to be the dominant natural source of organic matter in sediments in the Bay but has almost certainly been greatly augmented by anthropogenic inputs of TOC and TON associated with waste discharge from the fish factories, faecal waste from the mussel rafts, sewage effluent and storm water runoff. In the past, accumulation of organic waste, especially in sheltered areas where there is limited water flushing, has led to hypoxia (reduced oxygen) with negative impacts on benthic communities (e.g. the Saldanha Yacht Club and under the mussel rafts). Prior to any major development, TOC levels in Saldanha Bay were mostly very low (between 0.2 and 0.5%) throughout the Bay and Lagoon. Data collected in 1989 and 1999 indicated considerably elevated levels of TOC in the vicinity of the Iron Ore Terminal (particularly in the shipping channels) and in Small Bay. Data from subsequent surveys (2000, 2001, 2004) and those undertaken between 2008 and 2020 suggest that TOC levels have remained high throughout this period, with highest levels being recorded at the Yacht Club Basin and at the Multi-Purpose Terminal.

Levels of TON were first recorded in 1999 and were low at most sites in the Bay (≤0.2%) except for those in the Yacht Club Basin and near the mussel rafts in Small Bay. Levels were slightly or even considerably elevated at all sites that were monitored again in 2000, 2001 and 2004. Results from the State of the Bay surveys conducted between 2008 and 2020 suggest that levels dropped off slightly at many of the key sites in Small and Big Bay, however, have remained more or less steady in other parts of the Bay and in the Lagoon.

In areas of the Bay where muddy sediments tend to accumulate, trace metals and other contaminants often exceed acceptable threshold levels. This is believed to be due either to naturally-occurring high levels of the contaminants in the environment (e.g. in the case of cadmium) or due to impacts of human activities (e.g. lead, copper, manganese and nickel associated with ore exports). While trace metals are generally biologically inactive when buried in the sediment, they can become toxic to biota when re-suspended as a result of mechanical disturbance. On average, the concentrations of all metals were highest in Small Bay, lower in Big Bay and lowest (mostly below detection limits) in Langebaan Lagoon. Following a major dredging event in 1999, cadmium concentrations in certain areas in Small Bay exceeded internationally accepted safety levels, while concentrations of other trace metals (e.g. lead, copper and nickel) approached threshold levels. After this time, there have been numerous smaller spikes in trace metal levels, mostly as a result of dredging operations. For example, trace metals in the entrance to Langebaan Lagoon were significantly elevated in 2011 following dredging operations that were conducted as part of the expansion of the Naval Boat Yard in Salamander Bay. Currently, trace metal levels are mostly well within safety thresholds except for a few sites in Small Bay, where thresholds have been exceeded on a number of occasions between 2016 and 2020. In this year's survey, cadmium concentrations were noticeably high in Langebaan Lagoon of which one site even exceeded the environmental quality guideline. Key areas of concern regarding trace metal pollution within Small Bay include the Yacht Club Basin, where cadmium and copper have exceeded recommended thresholds for five years in a row and enrichment factors (EF) continue to be high, as well as adjacent to the Multi-Purpose Terminal where levels of cadmium and lead are below internationally accepted guidelines, but still remain highly enriched relative to historic levels. Recent increases in the concentration of manganese around the Iron Ore Terminal are also concerning. Regular monitoring of trace metal concentrations is thus strongly recommended to provide an early warning of any future changes.

Poly-aromatic hydrocarbon (PAH) contamination measured in the sediments of Saldanha Bay since 1999 has always been well below internationally accepted risk levels and not considered an environmental risk. Total petroleum hydrocarbon (TPH) levels, however, have fluctuated considerably in the vicinity of the ore terminal in recent years. In 2014, TPH Levels were found to be exceptionally high at some sites indicating heavily polluted conditions. The most likely explanation for the high observed TPH contamination levels is that a pollution incident associated with shipping activities took place. Alternatively, a pollution incident or routine operational activities on the jetty itself could be the cause of this contamination. While TPH and PAH findings in 2020 remain unchanged from 2019 and as such present no major concern, it is recommended that TPH monitoring within the vicinity of the ore terminal is continued to identify the occurrence of pollution incidents.

Macrophytes (eelgrass and saltmarshes)

Three distinct intertidal habitats exist within Langebaan Lagoon: seagrass beds, such as those of the eelgrass *Zostera capensis* (a type of seagrass); saltmarsh dominated by cordgrass *Spartina maritime* and *Sarcocornia perennis* and the dune slack rush *Juncus kraussi*; and unvegetated sandflats dominated by the sand prawn, *Callichirus kraussi* and the mudprawn *Upogebia capensis*. The other major vegetation type present in the upper lagoon area, particularly where groundwater inflow occurs, are reed beds dominated by *Phragmites australis.* Eelgrass and saltmarsh beds are extremely important as they increase habitat diversity in the lagoon, provide an important food source, increase sediment stability, provide protection to juvenile fish and invertebrates from predators and generally support higher species richness, diversity, abundance and biomass of invertebrate fauna compared to unvegetated areas. Eelgrass and saltmarsh beds are also important for waterbirds which feed directly on the shoots and rhizomes, forage amongst the leaves or use them as roosting areas at high tide. The primary physical factors influencing salt marsh distributions are salinity and water availability. Recent studies show that the aerial extent of seagrass beds in Langebaan Lagoon has declined by an estimated 38% since the 1960s, this being more dramatic in some areas than others (e.g. seagrass beds at Klein Oesterwal have declined by almost 99% over this period). Corresponding changes have been observed in densities of benthic macrofauna. At sites where eelgrass cover has declined, species commonly associated with eelgrass have declined in abundance, while those that burrow predominantly in unvegetated sand have increased in density. Fluctuations in the abundance of wading birds such as Terek Sandpiper, which feeds exclusively in *Zostera* beds, have also been linked to changes in eelgrass, with population crashes in this species coinciding with periods of lowest seagrass. The loss of eelgrass beds from Langebaan Lagoon is a strong indicator that the ecosystem is undergoing a shift, most likely due to anthropogenic disturbances. It is critical that this habitat and the communities associated with it be monitored in future as further reductions are certain to have long term implications, not only for the invertebrate fauna but also for species of higher trophic levels. In contrast, little change has been reported in the extent of saltmarshes in Langebaan Lagoon, these having declined by no more than 8% since the 1960s.

More recently, as part of the 2020 State of the Bay assessment, we assessed change in reed and sedge communities surrounding the head of Langebaan Lagoon using Landsat 5,7 and 8 and Sentinel-2 satellite imagery covering the period 1989 to 2020 and the open-source geospatial platform called Google Earth Engine (GEE). Results of this analysis indicate that variation in reed cover over time is relatively modest and that this has remained more or less constant over the last 31 years (1989-2020). The biggest perturbations in reed cover correspond with the two largest droughts that have been experienced in the region in this period (a 1:20 year event that occurred in the period 2002-2003) and an even bigger drought that occurred recently (a 1:100 year event in the period 2015-2017).

Future efforts in this field will entail expanding this assessment to other vegetation classes (specifically seagrass and salt marsh), assessing the level of change in each vegetation class over time, and ground truthing of each mapped vegetation class.

Benthic macrofauna

Soft-bottom benthic macrofauna (animals living in the sediment that are larger than 1 mm) are frequently used as a measure to detect changes in the health of the marine environment resulting from anthropogenic impacts. This is largely because these species are short lived and, consequently, their community composition responds rapidly to environmental changes. Monitoring of benthic macrofaunal communities over the period 1999-2020 has revealed a relatively stable community in most parts of the Bay and Lagoon except for 2008 when a dramatic shift in benthic community composition occurred at all sites. This shift involved a decrease in the abundance and biomass of filter feeders and an increase in shorter lived opportunistic detritivores. This was attributed to the extensive dredging that took place during 2007-2008. Filter feeding species are typically more sensitive to changes in water quality than detritivores or scavengers and account for much of the variation in overall abundance and biomass in the Bay.

Aside from this Bay-wide phenomenon, localised impact on and subsequent improvements in health have been detected in the Yacht Club Basin and at the monitoring sites adjacent to the Sea Harvest discharge pipe. At one point (2008) benthic fauna had been almost entirely eliminated from the Yacht Club Basin in Small Bay, owing to very high levels of trace metals and organic contaminants at this site (TOC, Cu, Cd and Ni). A similar scenario was evident in 2017 (high organic loading from discharge of fish waste) at Sea Harvest nearby. Benthic macrofauna communities in this area have, however, recovered steadily year-on-year since this time and are now almost on a par with other sites in Small Bay. Other notable changes in the health of benthic communities include the return of the suspension feeding sea-pen *Virgularia schultzei* to Big Bay and Langebaan Lagoon since 2004, as well as an increase in the percentage biomass of large, long lived species such as the tongue worm *Ochetostoma capense*, and several gastropods. Certain areas of Small Bay that experience reduced water circulation patterns in (e.g. base of the iron ore terminal, near the Small Craft Harbour and near mussel rafts) which results in the accumulation of fine sediment, organic material and trace metals (aggravated by anthropogenic inputs) still have impoverished macrofauna communities. Further to this, disturbance at the LPG site in Big Bay following installation of the Single Point Mooring (SPM) has resulted in reduced indices of abundance, biomass, and diversity in this area. Although highly localised, the negative impact of this development on the benthic macrofaunal community is significant. The latest results from this location do, however, indicate some improvement – further monitoring is required to determine whether full recovery has been achieved or not.

Rocky intertidal

As a component of the ongoing State of the Bay evaluation, monitoring of rocky intertidal communities in Saldanha Bay was initiated in 2005. Eight rocky shores spanning a wave exposure gradient from very sheltered to exposed, are sampled in Small Bay, Big Bay and Outer Bay. These surveys have been repeated annually from 2008 to 2020, however, due to financial constrains no survey was conducted in 2016. In the 2020 survey, a total of 100 taxa were recorded from the eight study sites, most of which had been found in previous surveys. The faunal component was represented by 23 species of filter-feeders, 23 species of grazers, and 14 species of predators/scavengers. The algal component comprised 24 corticated (foliose) seaweeds, 9 ephemerals, 5 species of encrusting algae, and 2 species of kelp. These species are common along much of the South African west coast and many have been recorded by other studies conducted in the Saldanha Bay area. Rocky shore species found included three alien invasive species, the Mediterranean mussel *Mytilus galloprovincialis,* and three introduced barnacle species *Balanus glandula*, *Perforatus perforatus* and *Amphibalanus amphitrite*.

The most important factor responsible for community differences among sites remains exposure to wave action and to a lesser extent shoreline topography. Within a site, the vertical emersion gradient of increasing exposure to air leads to a clear zonation of flora and fauna from low shore to high shore. Species composition and abundance has remained similar between years and any differences that are evident are considered to be natural seasonal and inter-annual phenomena, rather than anthropogenically-driven changes. Exceptions are the alien species introduced by hull fouling, ballast water or mariculture.

Fish

Recent seine net surveys have documented ongoing concerning trends in juvenile fish populations within the Saldanha Bay and Langebaan Lagoon system. The encouraging signs of recovery of white stumpnose and blacktail in Small Bay in 2016 did not continue through to 2017-2020, and white stump abundance remains very low throughout the system. The abundance of gobies in Small Bay also has also remained low since the 2007 survey. The decline in gobies cannot be attributed to fishery impacts but may be related to changes in water quality or habitat. The absence of gurnards, blacktail, elf and pipefish from 2020 Small Bay samples meant that diversity (just 10 species) was the lowest in the 16-year survey history. Overall, fish abundance in Small Bay was similar to levels seen in previous years, but it must be acknowledged that overall abundance in this area is dominated by harders, which appear resilient to any change in water quality. Despite the strong elf recruitment in Big Bay evident in the 2016 and 2017 sampling, none were caught during the following three sampling events (2018-2020). Notably for the first time in the history of the 16 seine net surveys undertaken since 1994, not a single elf was captured at any of the 16 sites surveyed in 2020. This is concerning as much of the remaining boat-based, recreational fishing effort that previously focused on white stump, appears to have shifted to elf, and three years of apparent poor recruitment to the surf zone nursery habitats does not bode well for future catches. After their scarcity in recent surveys, silversides were again abundant in Big Bay in 2020 (this species was absent or scarce in 2018 and 2019 samples), but three species - super klipvis, elf and pipefish - that are usually present in Big Bay surveys, were absent from the 2020 samples. The first record of galjoen and presence of the other common species including False Bay klipvis and sandsharks that were absent in 2019, however, resulted in a typical Big bay fish species count for the 2020 survey (13 species). Harders, gobies and silversides were present in Langebaan Lagoon samples in similar numbers to previous surveys, with only white stump catches remaining low.

For most of the seine net survey history (1986-), fish abundance at sites within or near the Langebaan MPA appeared to be stable within some modest inter-annual variability. This reflects natural and human induced impacts on the adult population size, recruitment success and use of the near shore habitat by juveniles; but may also be a result of the benefits of protection from exploitation and reduced disturbance at some sites due to the presence of the Langebaan MPA. Certainly, the studies by Kerwath *et al.* (2009), Hedger *et al.* (2010) and da Silva *et al.* (2013) demonstrated the benefits of the MPA for white stumpnose, elf and smooth hound sharks; and the protection of harders from net fishing in the MPA undoubtedly benefits this stock in the larger Bay area. The pressure to reduce this protection by allowing access to Zone B for commercial gill net permit holders should be resisted. This not only poses a threat to the productivity of the harder stock but also to other fish species that will be caught as bycatch. Harder recruitment to nearshore nursery areas appears to have not changed significantly over the monitoring period since 1994. A recent stock assessment, however, indicates that the Saldanha-Langebaan harder stock is overexploited, and effort reductions and commercial net gear changes are recommended to rebuild the stock.

The 2018 discovery of alien rainbow trout in Kraalbaai (almost certainly escapees from the pilot fish cage farming in Big Bay) is another threat to the indigenous fish fauna in the region. No further reports of fish cage escapees have been received, nor have any been caught during the seine net surveys undertaken since then. Nonetheless, the capture of these fish proved that escapees from fish cages can survive for a period and colonize different areas of the system. These predatory fish will prey on indigenous invertebrates and fish and could cause ecosystem level impacts. These alien fish are, however, highly unlikely to establish self-sustaining populations in the bay and lagoon due to the lack of suitable spawning habitat (cool, clear freshwater rivers) in the region. At the current experimental scale of fish farming, the number of escapees is not expected to be having highly significant impacts on indigenous fauna. However, at the proposed commercial scale finfish cage farming the number of alien salmonids introduced into the Bay and the Lagoon via ongoing escapes will probably have significant negative effects on indigenous fauna. Given the importance of the nearshore waters of Saldanha Bay and Langebaan lagoon as nursery areas for a number of vulnerable indigenous fishery species, we recommend that finfish cage farming be restricted to the outer Bay, and mitigation measures to minimise escapes from cages should be strictly enforced.

The significant declines in juvenile white stumpnose abundance at all sites throughout the system in over the last decade have shown that the protection afforded by the Langebaan MPA was not enough to sustain the fishery at the historical high effort levels. Arendse (2011) found the adult stock to be overexploited using data collected during 2006-2008 already, and the evidence from seine net surveys conducted since then, certainly suggests that recruitment overfishing has occurred (adult spawner stock has been reduced to a level where recruitment begin to drop). The annual seine net surveys did act as an early warning system that detected poor recruitment and should have allowed for timeous adjustments in fishing regulations to reduce fishing mortality on weak cohorts and preserve sufficient spawner biomass. Unfortunately, despite repeatedly expressing concern about the decline of white stump recruitment in State of the Bay Reports since at least 2013, and supporting the implementation of the harvest control measures recommended by Arendse (2011); namely a reduction in bag limit from 10 to 5 fish per person per day and an increase in size limit from 25 cm TL to 30 cm TL, the warning calls were not heeded. A statistically comprehensive analysis of fishery and survey data by Parker *et al.* (2017) confirmed the collapse of the Saldanha -Langebaan white stump stock and the fact that the fishery yield in recent years is a fraction of its historical peak or potential. The last three surveys have revealed some concerning declines in elf recruitment to surf zone nurseries, and it is recommended that this should be carefully monitored in the future.

There is now compelling scientific evidence that the stocks of the two most commercially important fish in the Saldanha–Langebaan system, namely white stump and harders, are overexploited. If the Saldanha Bay fisheries are to remain sustainable, fishing mortality will need to be reduced. We think that point arrived at least seven years ago for the white stumpnose fishery and recommend that resource users lobby the authorities to implement additional harvest control measures. Regional species-specific fishery management has been implemented elsewhere in South Africa (e.g. Breede River night fishing ban to protect dusky kob). White stumpnose in Saldanha Bay appear to be an isolated stock and there is good on-site management presence in the form of SAN Parks and DEFF and we think regionally specific management measures would work well in Saldanha-Langebaan. We again recommend the reduction of the daily bag limit and an increase in the minimum size limit for white stumpnose caught in the Saldanha Bay-Langebaan system. Although recruitment overfishing appears to have been taking place for several years now, the stock is not extirpated, and the situation is reversible. Reductions in fishing mortality can be achieved by effective implementation of more conservative catch limits and have an excellent chance of improving the stock status, catch rates and the size of white stumpnose in the future fishery. We also support the recommendation of Horton et al (2019) for a reduction in harder fishing effort and gear changes (increase in minimum mesh size) to facilitate stock recovery which will have significant socio-economic and ecological benefits in the long term.

The economic value of the recreational fishery in Saldanha-Langebaan should not be regarded as regionally insignificant as a lot of the expenditure associated with recreational angling is taking place within Langebaan and Saldanha itself. Furthermore, the historically popular white stumpnose fishery used to be a major draw card to the area and has probably contributed significantly to the growth in the residential property market the region has experienced. These benefits should be quantified by an economic study of the recreational fisheries. The value of Small Bay as a fish nursery and the economic value of the resultant fisheries could then be quantitatively considered when the environmental impacts of the proposed future industrial developments within Small Bay are assessed. The monitoring record from the annual seine net surveys will prove increasingly valuable in assessing and mitigating the impacts of future developments on the region's ichthyofauna.

Birds and seals

Together with the five islands within the Bay and Vondeling Island slightly to the South, Saldanha Bay and Langebaan Lagoon provide extensive and varied habitat for waterbirds and seals. This includes sheltered deep-water marine habitats associated with Saldanha Bay itself, sheltered beaches in the Bay, islands that serve as breeding refuges for seabirds and seals, rocky shoreline surrounding the islands and at the mouth of the Bay, and the extensive intertidal salt marshes, mud- and sandflats of the sheltered Langebaan Lagoon.

Saldanha Bay, and particularly Langebaan Lagoon, are of tremendous importance in terms of the diversity and abundance of waterbird populations species they support. At least 56 non-passerine waterbird species commonly use the area for feeding or breeding, and eleven species breed on the islands of Malgas, Marcus, Jutten, Schaapen and Vondeling alone. These islands support nationally important populations of African Penguin, Cape Gannet, Swift Tern, Kelp and Hartlaub's Gull, four species of marine cormorant, and important populations of the endemic African Oystercatcher. The lagoon is an important area for migratory waders and terns, as well as for numerous resident waterbird species. Waterbirds are counted annually on all the islands (DEFF: Oceans and Coasts), and bi-annually in Langebaan Lagoon (Avian Demography Unit of the University of Cape Town).

Except for bank cormorants, the populations of the seabirds breeding on the islands of Saldanha Bay were on an increasing trajectory from the start of monitoring in the 1980s and 90s until around 2000. Factors that probably contributed to this include the reduction and eventual cessation of guano collecting in 1991, banning of egg collecting, increases in the biomass of small pelagic fish (particularly sardines) over this period, and in the case of the African Oystercatcher, the increase in mussel biomass as a result of the spread of the Mediterranean mussel.

On the islands of Saldanha Bay, populations of all these species then started to decline, particularly the penguins, gannets, crowned cormorants and kelp gulls, which dropped to 7%, 40%, 23% and 22% of their populations at the turn of the century, respectively. Declines in the numbers of seabirds breeding on the Saldanha Bay Islands can be attributed to several causes. These include (1) emigration of birds to colonies further south and east along the South African coast in response to changes in the distribution and biomass of small pelagic fish stocks, (2) starvation as a result of a decline in the biomass of sardines nationally, and particularly along the west coast over the last decade, (3) competition for food with the small pelagic fisheries within the foraging range of affected bird species, (4) predation of eggs, young and fledglings by Great White Pelicans, Kelp Gulls and Cape Fur Seals, and (5) collapse of the West Coast Rock Lobster stock upon which Bank Cormorants feed.

However, because populations are so depressed, conditions at the islands in Saldanha, particularly predation by Cape Fur Seals, Pelicans and Kelp Gulls, have now become the major factors in driving current population decreases for many seabird species. Direct amelioration actions to decrease these impacts at the islands (*Pelican Watch*, problem seal culling) have had mixed results, with the former proving more effective than the latter. Cape Fur Seal and Kelp Gull predation continue to pose a major threat to seabird survival at the Saldanha Bay Island colonies.

Decreasing numbers of migrant waders utilising Langebaan Lagoon reflects a global trend, which can be attributed to loss of breeding habitat and hunting along their migration routes as well as human disturbance and habitat loss on their wintering grounds. In Langebaan Lagoon, drastic population declines in four species, including the Ruddy Turnstone, Red Knot, Grey Plover, and Curlew Sandpiper, signified this downward trend in summer migratory bird numbers. Most importantly, Curlew Sandpiper numbers have dropped from a pre-1990 average of just over 20 000 birds to 1 335 birds in 2019. Prior to 1990, this species accounted for almost two thirds of the total summer migratory wader numbers in the lagoon. Shrinking wader populations at Langebaan Lagoon are primarily signified by declining populations of migratory species. Conservation research and efforts should be prioritised for these species and conducted on an international scale.

Locally, unfavourable conditions persisting in Langebaan Lagoon as a result of anthropogenic impacts should also be managed more effectively to protect resident and migratory waders that do arrive in the lagoon. The fact that numbers of resident waders are also declining suggests that unfavourable conditions persisting in Langebaan Lagoon because of anthropogenic disturbance may be partly to blame. Resident wader numbers in the winter of 2019 dropped to the lowest recorded in the 40-year count record, a continuation of the declining trend over the last decade. Migratory wader counts in summer seem to have stabilized at around 3 000-5 000 birds over the last five years, a fraction of their former abundance. It is highly recommended that the status of coastal and wading bird species continues to be monitored and that these data are used to inform and assesses the efficacy of management interventions aimed at halting the observed declines and supporting recovery of the regions birds.

Cape Fur Seals are amongst the largest marine top predators found in and around Saldanha Bay. They are opportunistic, generalist feeders that have been shown to benefit from human activities including utilisation of discards from fishing boats or taking fish directly from fisherman. In addition, seals compete with seabirds, such as penguins and gannets, as well as with commercial fisheries, for small pelagic fish which form a key part of their diets. It has been suggested that the increasing numbers of seals on Vondeling island may lead to increased pressure to cull seals both from a fisheries perspective as well as to protect important seabird species on which seals are known to prey. Concerns have also been raised that, with the increased number of seals along the shores surrounding Saldanha Bay and with the addition of finfish aquaculture in the Bay, seal numbers within the Bay will likely increase, along with the occurrence of problem seals. Although seals are likely attracted to the aquaculture infrastructure within Saldanha Bay that provides floating resting sites, it is unlikely that their numbers will continue to increase significantly as they are restricted to sub-adult males. Additionally, the carrying capacity of Vondeling Island appears to have been reached and the overall population within Southern Africa has remained stable over the last 30 years.

Alien and invasive species

Human induced biological invasions have become a major cause for concern worldwide. The life history characteristics of the alien species, the ecological resilience of the affected area, the presence of suitable predators, biotic resistance and propagule pressure are a few of the many factors that can determine whether an alien species becomes a successful invader. Biological invasions can negatively impact biodiversity and can result in local or even global extinctions of indigenous species. Furthermore, alien species invasions can have tangible and quantifiable socio-economic impacts. Most of the introduced species in this country have been found in sheltered areas such as harbours and are believed to have been introduced through shipping activities, mostly ballast water and hull fouling. As ballast water tends to be loaded in sheltered harbours, the species that are transported originate from these habitats and have a difficult time adapting to South Africa's exposed coast.

A recent paper by Robinson *et al.* (2020) reviewing marine invasion in South Africa, reports 95 species as being alien to South Africa, of which 56 are considered invasive, i.e. populations are expanding and are consequently displacing indigenous species. With the recent addition of five new alien species – the barnacle *Perforatus perforatus*, the Japanese skeleton shrimp *Caprella mutica*, the North West African porcelain crab *Porcellana africana*, the Chilean stone crab *Homalaspis plana* and the South American sunstar *Heliaster helianthus* – 29 alien species are confirmed to be present in Saldanha Bay and/or Langebaan Lagoon. All of these, except *H. helianthus*, *H. plana*, *P. perforatus* and the previously reported anemone *Sagartia ornata*, are considered invasive. It should be noted that *P. africana* was previously misidentified as the European porcelain crab, *P. platycheles*.

Other noteworthy invasive alien species that are present in Saldanha Bay include the Mediterranean mussel *Mytilus galloprovincialis*, the barnacle *Balanus glandula*, the Pacific mussel *Semimytilus algosus* and the Western pea crab *Pinnixa occidentalis.* The abundance of *M. galloprovincialis* on rocky shores in Saldanha Bay has been decreasing over the last few years, even to such an extent that no mussels were detected at certain sites. The reason behind this decline is, however, still not clear although numerous factors might be at play. No trend in the abundance of *B. glandula* over time is evident, although this species has shown a decrease in percentage cover at almost all sites, with no barnacles being reported from some sites during the 2020 survey. No conclusive trend in the spread and site preference of the Western pea crab *P. occidentalis* could be established, although it does seem to flourish in deeper water habitats and occurs at lower densities close to the iron ore and multi-purpose terminals. Despite abundance and dominance of the alien crab fluctuating quite substantially at certain sites in Big and Small Bay over time, evidence from the 2020 survey suggests that *P. occidentalis* is well established in the Bay. Pea crab populations have also shown an increase in Langebaan Lagoon, although it is restricted to only one site. The status of this crab within Danger Bay is currently not confirmed and more sampling efforts are needed.

The discovery of five new alien species over the past six years raises concern and highlights the need for management action. This is further exacerbated by the fact that alien species are considered to represent one of the greatest threats to rocky shore communities in Saldanha Bay, owing to their potential to become invasive, thereby displacing naturally occurring indigenous species. An additional

19 species are currently regarded as cryptogenic in Saldanha Bay and/or Langebaan Lagoon and comprehensive genetic analyses are urgently required to determine the definite status of these cryptogenic species.

Management actions should firstly be focused on managing invasive species already present in Saldanha Bay by rating species based on their impacts. Secondly, efforts should be focused on preventing further invasions. Watchlists have been identified as a useful preventative measure and are created based on selecting species with an invasion history, pathways to the area of concern, occurring in similar climatic regions or those with biological traits that could predispose them to becoming successful invaders. Another vital aspect includes identifying and managing important pathways of introduction. This should be done in combination with port control to monitor vessels entering harbours, treatment of hull fouling and ballast water before entrance and the regular monitoring of harbours. All these efforts require in depth research. Such research will not only contribute towards our understanding of the drivers and traits governing successful invasions, but also give insight into associated impacts. This, in turn, could be used to support directed management actions for successfully controlling invasions and mitigating impacts. However, the knowledge gained from research needs to be shared with stakeholders and policy makers to implement appropriate management strategies and inform action.

Summary

In summary, developments in Saldanha Bay and Langebaan Lagoon during the past thirty years have inevitably impacted on the environment. Most parameters investigated in this study suggest a considerable degree of negative impact having occurred over the last few decades. Long-term decreases in populations of fish (e.g. white stumpnose) and many bird species in Saldanha Bay and Langebaan Lagoon are of particular concern. These most likely reflect long term changes in exploitation levels (fish) and habitat quality (sediment and water quality and increasing levels of disturbance) and in important forage species (e.g. benthic macrofauna). Recent improvements in some of these underlying indicators (e.g. sediment quality and macrofauna abundance and composition) are very encouraging, though, and will hopefully translate into improvements in the higher order taxa as well. There remains considerable work to be done in maintaining and restoring the health of the Bay, especially in respect of the large volumes of effluent that are discharged to the Bay, very little of which is compliant with the existing effluent quality standards. Reclaiming industry-grade or even potable water from effluent will play an important role in improving water quality in Saldanha Bay. A holistic approach in monitoring and assessing the overall health status of the Bay is essential, and regular (in some cases increased) monitoring of all parameters reported on here is strongly recommended, particularly in the face of increased development in the Bay.

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GLOSSARY

Alien species	Species whose presence in a region is attributable to human actions that enabled them to overcome fundamental biogeographical barriers (i.e. human-mediated extra-range dispersal) (synonyms: Introduced, non-indigenous, non-native, exotic).
Articulated coralline algae	Branching, tree-like plants which are attached to the substratum by crustose or calcified, root-like holdfasts.
Aquaculture	The sea-based or land-based rearing of aquatic animals or the cultivation of aquatic plants for food
Aquifer	Underground layer of water-bearing permeable rock, rock fractures or unconsolidated materials (gravel, sand, or silt) from which groundwater can be extracted using a water well.
Biodiversity	The variability among living organisms from all terrestrial, marine, and other aquatic ecosystems, and the ecological complexes of which they are part: this includes diversity within species, between species and of ecosystems.
Biota	All the plant and animal life of a particular region.
Colony-forming unit	A colony-forming unit (CFU) is a unit used to estimate the number of viable bacteria or fungal cells in a sample.
Community structure	Taxonomic and quantitative attributes of a community of plants and animals inhabiting a particular habitat, including species richness and relative abundance structurally and functionally.
Coralline algae	Coralline algae are red algae in the Family Corallinaceae of the order Corallinales characterized by a thallus that is hard as a result of calcareous deposits contained within the cell walls.
Corticated algae	Algae that have a secondarily formed outer cellular covering over part or all of an algal thallus. Usually relatively large and long-lived.
Crustose coralline algae	Slow growing crusts of varying thickness that can occur on rock, shells, or other algae.
Cryptogenic	Species of unknown origin.
Ephemeral algae	Opportunistic algae with a short life cycle that are usually the first settlers on a rocky shore.
Extralimital	Species whose native range falls within the political boundaries of a country, but whose presence in another part of the same country is attributable to human transport across fundamental biogeographical barriers.
Fauna	General term for all the animals found in a particular location.

Flora	General term for all the plant life found in a particular location.
Foliose algae	Leaf-like, broad and flat; having the texture or shape of a leaf.
Filter-feeders	Animals that feed by straining suspended matter and food particles from water.
Functional group	A collection of organisms of specific morphological, physiological, and/or behavioural properties.
Grazer	An herbivore that feeds on plants/algae by abrasion from the surface.
Groundwater	Water held underground in the soil or in pores and crevices in rock.
Indigenous	Species within the limits of their native range (Synonyms: native).
Intertidal	The shore area between the high- and the low-tide levels.
Invasive	Alien species that have self-replacing populations over several generations and that have spread from their point of introduction.
Invertebrate	Animals that do not have a backbone. Invertebrates either have an exoskeleton (e.g. crabs) or no skeleton at all (worms).
Kelp	A member of the order Laminariales, the more massive brown algae.
Macrophyte	An aquatic plant large enough to be seen by the naked eye.
Native	Species within the limits of their native range (Synonyms: indigenous).
Naturalised	Alien species that have self-replacing populations over several generations outside of captivity or culture, but that have not spread from their point of introduction.
Opportunistic	Capable of rapidly occupying newly available space.
Paleo-channel	Old or ancient river channels often infilled with course fluvial deposits which can store and transmit appreciable quantities of water.
Polychromatic	Having various or changing colours; multicoloured.
Rhizome	A modified subterranean plant stem that sends out roots and shoots from its nodes.
Rocky shore community	A group of interdependent organisms inhabiting the same rocky shore region and interacting with each other.
Scavenger	An animal that eats already dead or decaying animals.
Shore height zone	Zone on the intertidal shore recognizable by its community.
Thallus	General form of an alga that, unlike a plant, is not differentiated into stems, roots, or leaves.
Topography	The relief features or surface configuration of an area

LIST OF ABBREVIATIONS

ADZ	Aquaculture Development Zone
AOU	Apparent Oxygen Utilization
BA	Basic Assessment
BCLME	Benguela Current Large Marine Ecosystem
CBA	Critical Biodiversity Area
CGS	Council for Geoscience
CMP	Coastal Management Programme
COD	Chemical Oxygen Demand
CNG	Compressed Natural Gas
CSIR	Council for Scientific and Industrial Research
CWAC	Co-ordinated Waterbird Counts
CWDP	Coastal Water Discharge Permit
DAFF	Department of Agriculture, Forestry and Fisheries
DD	Decimal Degrees
DEA	Department of Environmental Affairs
DEA&DP	Western Cape Department of Environmental Affairs & Development Planning
DoE	Department of Energy
DWA	Department of Water Affairs
DWS	Department of Water and Sanitation
EA	Environmental Authorisation
EAS	Elandsfontein Aquifer System
EC	Electrical Conductivity
EEM	Elandsfontein Exploration and Mining (Pty) Ltd
EIA	Environmental Impact Assessment
EICAT	The Environmental Impact Classification for Alien Taxa
EMF	Environmental Management Framework
EMMP	Environmental Management and Maintenance Plan
EMPr	Environmental Management Programme
ET	Evapotranspiration

FEPA	Freshwater Ecosystem Priority Area
FPP	Floating Power Plant
GA	General Authorisation
ha	Hectare
ICMA	National Environmental Management: Integrated Coastal Management Act (No. 24 of 2008)
GIS	Geographical Information System
GRAII	Groundwater Resource Assessment (Phase II)
GRU	Groundwater Response (or Resource) Unit
GSB	Greater Saldanha Bay
IDZ	Industrial Development Zone
LRA	Langebaan Road Aquifer
LRAS	Langebaan Road Aquifer System
LNG	Liquefied Natural Gas
LPG	Liquid Petroleum Gas
MAE	Mean Annual Evaporation
MLRA	Marine Living Resources Act (No. 18 of 1998)
MPA	Marine Protected Area
Mamsl	Metres above mean sea level
MAP	Mean Annual Precipitation
MAR	Mean Annual Run-off
mbgl	Metres below ground level
Mtpa	Million tons per annum
NEMA	National Environmental Management Act (No. 107 of 1998)
NEMBA	National Environmental Management: Biodiversity Act (No. 10 of 2004)
NFEPA	National Freshwater Ecosystem Priority Area
NGA	National Groundwater Archive
NOAA	National Oceanic and Atmospheric Administration
NWA	National Water Act (No. 36 of 1998)
PAH	Poly-Aromatic Hydrocarbons
PSU	Practical Salinity Unit
Q	Discharge/Yield
RWL	Rest Water Level

RWQO	Receiving Water Quality Objectives approach
RQO	Resource Quality Objectives
SBIDZ	Saldanha Bay Industrial Development Zone
SBM	Saldanha Bay Municipality
SBWQFT	Saldanha Bay Water Quality Forum Trust
SWSA-gw	Strategic Water Source Areas - groundwater
TNPA	Transnet National Ports Authority
TOC	Total Organic Carbon
TON	Total Organic Nitrogen
TPH	Total Petroleum Hydrocarbon
TSS	Total Suspended Solids
UGEP	Utilisable Groundwater Exploitation Potential
VRF	Vessel Repair Facility
WARMS	Water Authorisation and Registration Management System
WCDM	West Coast District Municipality
WGS84	Since the 1st January 1999, the official co-ordinate system for South Africa
WL	Water Level
WRC	Water research Commission
WWTW	Wastewater Treatment Works

1 INTRODUCTION

1.1 Background

Saldanha Bay is situated on the west coast of South Africa, approximately 100 km north of Cape Town, and is directly linked to the shallow, tidal Langebaan Lagoon. Saldanha Bay and Langebaan Lagoon are areas of exceptional beauty and are considered South African biodiversity "hot spots". A number of marine protected areas have been proclaimed in and around the Bay, while Langebaan Lagoon and much of the surrounding land falls within the West Coast National Park (Figure 1.1). Langebaan Lagoon was also declared a Ramsar Site in 1988, along with a series of islands within Saldanha Bay (Schaapen, Marcus, Malgas, Jutten and Vondeling). As such, Saldanha Bay and Langebaan Lagoon have long been the focus of scientific interest.

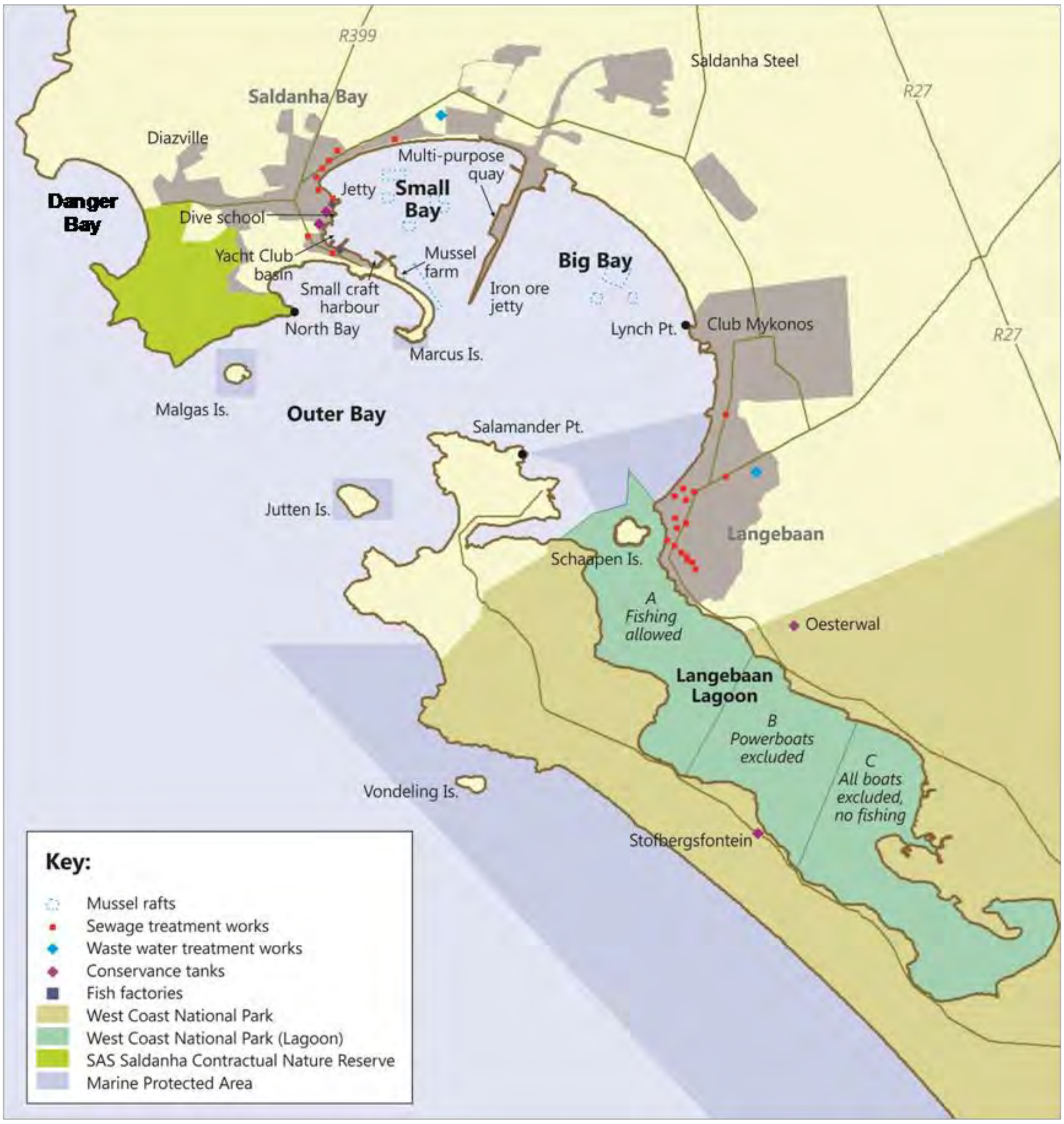

Figure 1.1. Regional map of Saldanha Bay and Langebaan Lagoon and Danger Bay showing development (grey shading) and conservation areas.

Saldanha Bay and Langebaan Lagoon have long been the focus of scientific study and interest largely owing to the conservation importance and its many unique features. A symposium on research in the natural sciences of Saldanha Bay and Langebaan Lagoon was hosted by the Royal Society of South Africa in 1976 in an attempt to draw together information from the various research studies that had been and were being conducted in the area. The symposium served to focus the attention of scientific researchers from a wide range of disciplines on the Bay and resulted in the development of a large body of data and information on the status of the Bay and Lagoon at a time prior to any major developments in the Bay.

More recently (in 1996), the Saldanha Bay Water Quality Forum Trust (SBWQFT), a voluntary organization representing various organs of State, local industry and other relevant stakeholders and interest groups, was inaugurated with the aim of promoting an integrated approach to the management, conservation and development of the waters of Saldanha Bay and the Langebaan Lagoon, and the land areas adjacent to, and influencing it. Since its inauguration the SBWQFT has played an important role in guiding and influencing management of the Bay and in commissioning scientific research aimed at supporting informed decision making and sustainable management of the Saldanha Bay/Langebaan Lagoon ecosystem. Monitoring of a number of important ecosystem indicators was initiated by the SBWQFT in 1999 including water quality (faecal coliform, temperature, oxygen and pH), sediment quality (trace metals, hydrocarbons, Total organic carbon (TOC) and nitrogen) and benthic macrofauna. The range of parameters monitored has since increased to include surf zone fish and rocky intertidal macrofauna (both initiated in 2005) and has culminated in the commissioning of a "State of the Bay" report series that has been produced annually since 2008. Despite these noteworthy successes in environmental monitoring, the history of the area has been tainted with overexploitation and lack of care for the environment, the environment generally being the loser in both instances.

The first State of the Bay report was produced in 2006 by Anchor Environmental Consultants (Pty) Ltd and served to draw together all available information on the health status and trends in a wide range of parameters that provide insights into the health of the Saldanha Bay and Langebaan Lagoon ecosystem. The 2006 report incorporated information on trends in a full range of physico-chemical indicators including water quality (temperature, oxygen, salinity, nutrients, and pH), sediment quality (particle size, trace metal and hydrocarbon contaminants, TOC and nitrogen) and ecological indicators (chlorophyll a, benthic macrofauna, fish and birds). This information was drawn from work commissioned by the SBWQFT as well as a range of other scientific monitoring programmes and studies. The 2006 report was presented in two formats – one data rich form that was designed to provide detailed technical information in trends in each of the monitored parameters and the second in an easy to read form that was accessible to all stakeholders.

The success of the first State of the Bay report and the ever-increasing pace of development in and around the Saldanha Bay encouraged the SBWQFT to produce the second Sate of the Bay report in 2008, and then annually from this time onwards. This (2020) report is the 13th in the series and provides an update on the health of all monitored parameters in Saldanha Bay, Langebaan Lagoon and Danger Bay in the time since the last State of the Bay assessment (2018). It includes information on trends in all of the parameters reported on in the previous reports (2006, 2008, 2009, 2010, 2011, 2012, 2013-4, 2015, 2016, 2017, 2018 and 2019).

This edition also incorporates a number of additional indicators not previously covered by the State of the Bay reports (focussing mostly on activities and discharges that affect the health of the system). Readers that are familiar with the State of Saldanha Bay and Langebaan Lagoon report series are encouraged to consult Section 1.3 of this report, which highlights new and updated information that has been included in this edition.

1.2 Structure of this report

This report draws together all available information on water quality and aquatic ecosystem health of Saldanha Bay and Langebaan Lagoon, and on activities and discharges affecting the health of the Bay. The emphasis has been on using data from as wide a range of parameters as possible that are comparable in both space and time and cover extended periods which provide a good reflection of the long term environmental health in the Bay as well as recent changes in the health status of the system. The report is composed of twelve chapters each of which addresses different aspects of the health of the system.

Chapter One introduces the State of the Bay Reporting programme and explains the origin of and rationale for the programme, and provides the report outline.

Chapter Two provides background information to anthropogenic impacts on the environment and the range of different approaches to monitoring these impacts, which captures the differences in the nature and temporal and spatial scale of these impacts.

Chapter Three provides a summary of available information on historic and on-going activities, discharges and other anthropogenic impacts to the Bay that are likely to have had or are having some impact on environmental health.

Chapter Four outlines the coastal and environmental management measures in the greater Saldanha Bay area developed/implemented to facilitate sustainable development in an area where industrial development (Saldanha Bay IDZ and associate industrial development), residential and conservation areas (Ramsar Site, MPAs and National Parks) are immediately adjacent to one another.

Chapter Five summarises available information on the importance of groundwater for Saldanha Bay and Langebaan Lagoon and presents information on the use of groundwater in this region and potential concerns this use poses for the ecology of the Bay.

Chapter Six summarises available information on water quality parameters that have historically been monitored in the Bay and Lagoon and reflects on what can be deduced from these parameters regarding the health of the Bay.

Chapter Seven summarises available information on sediment monitoring, including shoreline stability, that has been conducted in Saldanha Bay, Danger Bay and Langebaan Lagoon with further interpretation of erosion monitoring results along Langebaan Beach, which was initiated by the Saldanha Bay Local Municipality in 1996, and changes in sediment composition (particle size distribution) and quality (trace metal, organic carbon and nitrogen content) over time and/or related to dredging events. .

Chapter Eight presents data on the current status and historical changes in aquatic vegetation communities (macrophytes) associated with Langebaan Lagoon and Saldanha Bay and also highlights the importance of groundwater discharge to the Bay and Lagoon for these communities.

Chapter Nine presents data on changes in benthic macrofauna in Saldanha Bay and Langebaan Lagoon from the 1970's to the present day.

Chapter Ten addresses changes that have occurred in the rocky intertidal zones in and around Saldanha Bay over the past 20 years and presents results from a rocky intertidal monitoring survey initiated in 2005.

Chapter Eleven summarises all available information on the fish community and composition in the Bay and Lagoon, as deduced from both seine and gill net surveys, and presents results from a surf zone fish monitoring survey initiated in 2005. In 2014 this survey was expanded to include Danger Bay.

Chapter Twelve provides detailed information on the status of key bird species utilising the offshore islands around Saldanha Bay as well as providing an indication of the national importance of the area for birds. Additionally, a brief summary of the current status of the Saldanha Bay seal population is provided.

Chapter Thirteen summarises available information of marine alien species known to be present in Saldanha Bay and Langebaan Lagoon as well as trends in their distribution and abundance.

Chapter Fourteen provides a tabulated summary of the key changes detected in each parameter covered in this report and assigns a health status rank to each. This chapter also provides recommendations for future environmental monitoring for the Bay and of management measures that ought to be adopted in the future.

1.3 What's new in the 2020 edition of the State of Saldanha Bay and Langebaan Lagoon report

Readers who are familiar with the State of the Bay report series will know that while the various chapters of this report are updated each year with new data and information that has been collected during the course of the preceding twelve months, either through dedicated surveys commissioned by the SBWQFT or other dedicated individuals and agencies, much of the background or contextual information pertinent to the State of the Bay remains the same. While this background and contextual information is important, it can be a little tedious to wade through for those who have seen it all before. This section of the report thus serves to highlight what new data and information has been included in each of the chapters of this report to make it easier for those readers to home in on the material that is of greatest interest to them.

Chapter 3: Activities and discharges affecting the health of the Bay

Only developments and activities which have experienced changes since the last State of the Bay report (2019) are retained in this chapter. Completed, stagnated or pending developments are briefly summarised in the relevant section and the reader is referred to the previous report of 2019 for more details. Additional and updated information included in the sections of this chapter are listed below:

- Numbers of visitors to the West Coast National Park;
- Metal exports from the Saldanha Bay Multipurpose and iron ore terminals;
- Information on new and existing development proposals for Saldanha (Zandheuvel phosphate mine, and the development of additional vessel repair facilities in the Port of Saldanha);
- Shipping traffic and ballast water discharges;
- Effluent discharges into Saldanha Bay:
  - the volumes and quality of wastewater discharged into the Bay from the Saldanha and Langebaan Water Treatment Works, including the details on the effort of the Saldanha Bay Municipality to reclaim freshwater from treated wastewater
  - fish processing establishments in Saldanha (new information on environmental monitoring data for Sea Harvest)
- Mariculture industry in Saldanha, including an update on the development of the Aquaculture Development Zone.

Chapter 4: Coastal and environmental management

Little has happened in respect of new policy and management interventions in Saldanha Bay so this chapter simply provides a brief summary of key interventions that have been introduced in the last few years.

Chapter 5: Groundwater

This is the fourth year that this new addition appears in the State of the Bay report and the chapter has been extensively revised by the team at GEOSS Groundwater and Earth Sciences. This chapter serves to highlight the importance of groundwater for Saldanha Bay and Langebaan Lagoon and presents information on the use of groundwater in this region, potential concerns that this use poses for the ecology of the Bay, and highlights current data from relevant groundwater modelling and literature as well as the Trust's own continuous water quality monitoring data collected at the head of Langebaan Lagoon.

Chapter 6: Water quality

This chapter presents new information on variations in temperature, salinity, dissolved oxygen, and dissolved inorganic nitrogen at various locations in the Bay, as well as new updated information on levels of microbial indicators (faecal coliforms and *E. coli.*) in the Bay collected by the SBWQFT, Saldanha Bay Municipality, DEFF (for the Aquaculture Development Zone), Transnet NPA and Sea Harvest. It also includes new updated information on levels of trace metals in mussels on the shoreline and offshore mariculture facilities.

Chapter 7: Sediments

This chapter has been extensively updated this year and now includes data on erosion monitoring along Langebaan Beach, which was initiated by the Saldanha Bay Local Municipality in 1996, later picked up by the SBWQFT. The chapter also includes new information on grain size composition and health of benthic sediment in Saldanha Bay (Total Organic Carbon and Nitrogen, trace metals and hydrocarbons).

Chapter 8: Aquatic macrophytes: This chapter makes a comeback this year after having been dropped for several years and presents new data on the current status and historic variations in aquatic vegetation (macrophyte) communities that surround Langebaan Lagoon. It highlights the important contribution these communities make to the ecology of the bay (as a feeding, breeding and roosting area, for invertebrates, fish and birds) and their dependence on groundwater inflow.

Chapter 9: Benthic macrofauna

This chapter presents new information on species composition, abundance, biomass and health of benthic macrofauna communities in Saldanha Bay and Langebaan Lagoon as revealed by dedicated monitoring activities undertaken by the SBQWFT.

Chapter 10: Intertidal invertebrates (rocky shores)

This chapter presents new information on species composition, abundance, biomass and health of rocky intertidal invertebrate communities in Saldanha Bay and Langebaan Lagoon as revealed by dedicated monitoring activities undertaken by the SBQWFT.

Chapter 11: Fish

This chapter presents new information on species composition, abundance, biomass and health of fish communities in Saldanha Bay and Langebaan Lagoon and provides updates on the current population status of key line-fish, net-fish and angling species in the Bay.

Chapter 12: Birds and seals

This chapter provides an update on the species threat status, and presents new data on the composition, abundance and health of birds breeding and feeding in Langebaan Lagoon and on the Islands in Saldanha Bay. It also provides a snapshot synopsis of the local seal population in the Bay.

Chapter 13: Alien invasive species

This chapter presents new information and data on the number, distribution and abundance of alien invasive marine species in Saldanha Bay and Langebaan Lagoon, and provides a summary of new information on the ecology, spread, abundance of alien marine species that occur in Saldanha Bay and Langebaan Lagoon and on their ability to impact biodiversity as ecosystem engineers.

Chapter 14: Management and monitoring recommendations

This chapter includes updated recommendations on management actions that need to be implemented to mitigate key threats highlighted in the previous chapters.

2 BACKGROUND TO ENVIRONMENTAL MONITORING AND WATER QUALITY MANAGEMENT

2.1 Introduction

Pollution is defined by the United Nations Convention on the Law of the Sea as 'the introduction by man, directly or indirectly, of substances or energy into the marine environment, including estuaries, which results in such deleterious effects as harm to living resources and marine life, hazards to human health, hindrance to marine activities, including fishing and other legitimate uses of the sea, impairment of quality for use of the sea water and reduction of amenities'. A wide variety of pollutants are generated by man, many of which are discharged to the environment in one form or another. Pollutants or contaminants can broadly be grouped into five different types: trace metals, hydrocarbons, organochlorines, radionuclides, and nutrients. Certain metals normally found in very low concentrations in the environment (hence referred to as trace metals) are highly toxic to aquatic organisms. These include for example Mercury, Cadmium, Arsenic, Lead, Chromium, Zinc and Copper. These metals occur naturally in the earth's crust, but mining of metals by man is increasing the rate at which these are being mobilised which is enormously over that achieved by geological weathering. Many of these metals are also used as catalysts in industrial processes and are discharged to the environment together with industrial effluent and wastewater. Hydrocarbons discharged to the marine environment include mostly oil (crude oil and bunker oil) and various types of fuel (diesel and petrol). Sources of hydrocarbons include spills from tankers, other vessels, refineries, storage tanks, and various industrial and domestic sources. Hydrocarbons are lethal to most marine organisms due to their toxicity, but particularly to marine mammals and birds due to their propensity to float on the surface of the water where they come into contact with seabirds and marine mammals. Organochlorines do not occur naturally in the environment and are manufactured entirely by man. A wide variety of these chemicals exists, the most commonly known ones being plastics (e.g. polyvinylchloride or PVC), solvents and insecticides (e.g. DDT). Most organochlorines are toxic to marine life and have a propensity to accumulate up the food chain. Nutrients are derived from several sources, the major one being sewage, industrial effluent, and agricultural runoff. They are of concern owing to the vast quantities discharged to the environment each year which has the propensity to cause eutrophication of coastal and inland waters. Eutrophication in turn can result in proliferation of algae, phytoplankton (red tide) blooms, and deoxygenation of the water (black tides).

It is important to monitor both the concentration of these contaminants in the environment and their effects on biota such that negative effects on the environment can be detected at an early stage before they begin to pose a major risk to environmental and/or human health.

2.2 Mechanisms for monitoring contaminants and their effects on the environment

The effects of pollutants on the environment can be detected in a variety of ways as can the concentrations of the pollutants themselves in the environment. Three principal ways exists for assessing the concentration of pollutants in aquatic ecosystems - through the analysis of pollutant concentrations in the water itself, in sediments or in living organisms. Each has their advantages and disadvantages. For example, the analysis of pollutant concentrations in water samples is often problematic owing to the fact that even at concentrations lethal to living organisms, they are difficult to detect without highly sophisticated sampling and analytical techniques. Pollutant concentrations in natural waters may vary with factors such as season, state of the tide, currents, extent of freshwater runoff, sampling depth, and the intermittent flow of industrial effluents, which complicates matters even further. In order to accurately elucidate the degree of contamination of a particular environment, many water samples usually have to be collected and analysed over a long period of time. The biological availability of pollutants in water also presents a problem in itself. It must be understood that some pollutants present in a water sample may be bound chemically to other compounds that renders them unavailable or non-toxic to biota (this is common in the case of trace metals).

Another way of examining the degree of contamination of a particular environment is through the analysis of pollutant concentrations in sediments. This has several advantages over the analysis of water samples. Most contaminants of concern found in aquatic ecosystems tend to associate preferentially with (i.e. adhere to) suspended particulate material rather than being maintained in solution. This behaviour leads to pollutants becoming concentrated in sediments over time. By analysing their concentrations in the sediments (as opposed to in the water) one can eliminate many of the problems associated with short-term variability in contaminant concentrations (as they reflect conditions prevailing over several weeks or months) and concentrations tend to be much higher which makes detection much easier. The use of sediments for ascertaining the degree of contamination of a particular system or environment is thus often preferred over the analysis of water samples. However, several problems still exist with inferring the degree of contamination of a particular environment from the analysis of sediment samples.

Some contaminants (e.g. bacteria and other pathogens) do not accumulate in sediments and can only be detected reliably through other means (e.g. through the analysis of water samples). Concentrations of contaminants in sediments can also be affected by sedimentation rates (i.e. the rate at which sediment is settling out of the water column) and the sediment grain size and organic content. As a general rule, contaminant concentrations usually increase with decreasing particle size, and increase with increasing organic content, independent of their concentration in the overlying water. Reasons for this are believed to be due to increases in overall sediment particle surface area and the greater affinity of most contaminants for organic as opposed to inorganic particles (Phillips 1980, Phillips & Rainbow 1994). The issue of contaminant bioavailability remains a problem as well, as it is not possible to determine the biologically available portion of any contaminant present in sediments using chemical methods of analysis alone.

One final way of assessing the degree of contamination of a particular environment is by analysing concentrations of contaminants in the biota themselves. There are several practical and theoretical advantages with this approach. Firstly, it eliminates any uncertainty regarding the bioavailability of the contaminant in question as it is by nature 'bio-available'. Secondly, biological organisms tend to concentrate contaminants within their tissues several hundred or even thousands of times above the concentrations in the environment and hence eliminate many of the problems associated with detecting and measuring low levels of contaminants. Biota also integrates concentrations over time and can reflect concentrations in the environment over periods of days, weeks, or months depending on the type of organism selected. Not all pollutants accumulate in the tissues of living organisms, including for example nutrients and particulate organic matter. Thus, while it is advantageous to monitor contaminant concentrations in biota, monitoring of sediment and water quality is often also necessary.

Different types of organisms tend to concentrate contaminants at different rates and to different extents. In selecting what type of organism to use for bio monitoring it is generally recommended that it should be sedentary (to ensure that it is not able to move in and out of the contaminated area), should accumulate contaminants in direct proportion with their concentration in the environment, and should be able to accumulate the contaminant in question without lethal impact (such that organisms available in the environment reflect prevailing conditions and do not simply die after a period of exposure). Giving cognisance to these criteria, the most commonly selected organisms for bio monitoring purposes include bivalves (e.g. mussels and oysters) and algae (i.e. seaweed).

Aside from monitoring concentrations of contaminant levels in water, sediments, and biota, it is also possible, and often more instructive, to examine the species composition of the biota at a particular site or in a particular environment to ascertain the level of health of the system. Some species are more tolerant of certain types of pollution than others. Indeed, some organisms are extremely sensitive to disturbance and disappear before contaminant concentrations can even be detected reliably whereas others proliferate even under the most noxious conditions. Such highly tolerant and intolerant organisms are often termed biological indicators as they indicate the existence or concentration of a particular contaminant or contaminants simply by their presence or absence in a particular site, especially if this changes over time. Changes in community composition (defined as the relative abundance or biomass of all species) at a particular site can thus indicate a change in environmental conditions. This may be reflected simply as: (a) an overall increase/decrease in biomass or abundance of all species, (b) as a change in community structure and/or overall biomass/abundance but where the suite of species present remain unchanged, or (c) as a change in species and community structure and/or a change in overall biomass/abundance (Figure 2.1.). Monitoring abundance or biomass of a range of different organisms from different environments and taxonomic groups with different longevities, including for example invertebrates, fish and birds, offers the most comprehensive perspective on change in environmental health spanning months, years and decades.

The various methods for monitoring environmental health all have advantages and disadvantages. A comprehensive monitoring programme typically requires that a variety of parameters be monitored covering water, sediment, biota and community health indices.

2.3 Indicators of environmental health and status in Saldanha Bay and Langebaan Lagoon

For the requirements of the Saldanha Bay and Langebaan Lagoon State of the Bay monitoring programme a ranking system has been devised that incorporates both the drivers of changes (i.e. activities and discharges that affect environmental health) and a range of different measures of ecosystem health from contaminant concentrations in seawater to change in species composition of a range of different organisms (Figure 2.1. and Table 2.1.). Collectively these parameters provide a comprehensive picture of the State of the Bay and also a baseline against which future environmental change can be measured. Each of the threats and environmental parameters incorporated within the ranking system was allocated a health category depending on the ecological status and management requirements in particular areas of Saldanha Bay and Langebaan Lagoon. An overall Desired Health category is also proposed for each environmental parameter in each area, which should serve as a target to be achieved or maintained through management intervention.

Various physical, chemical and biological factors influence the overall health of the environment. Environmental parameters or indices were selected that can be used to represent the broader health of the environment and are feasible to measure, both temporally and spatially. The following environmental parameters or indices are reported on:

Activities and discharges affecting the environment: Certain activities (e.g. shipping and small vessel traffic, the mere presence of people and their pets, trampling) can cause disturbance in the environment especially to sensitive species, that, along with discharges to the marine environment (e.g. effluent from fish factories, treated sewage, and ballast water discharged by ships) can lead to degradation of the environment through loss of species (i.e. loss of biodiversity), or increases in the abundance of pest species (e.g. red tides), or the introduction of alien species. Monitoring activity patterns and levels of discharges can provide insight into the reasons for any observed deterioration in ecosystem health and can help in formulating solutions for addressing negative trends.

Water uality: Water quality is a measure of the suitability of water for supporting aquatic life and the extent to which key parameters (temperature, salinity, dissolved oxygen, nutrients and chlorophyll a, faecal coliforms and trace metal concentrations) have been altered from their natural state. Water quality parameters can vary widely over short time periods and are principally affected by the origin of the water, physical and biological processes and effluent discharge. Water quality parameters provide only an immediate (very short term – hours to days) perspective on changes in the environment and do not integrate changes over time.

Sediment quality: Sediment quality is a measure of the extent to which the nature of benthic sediments (particle size composition, organic content and contaminant concentrations) has been altered from its natural state. This is important as it influences the types and numbers of organisms inhabiting the sediments and is in turn, strongly affected by the extent of water movement (wave action and current speeds), mechanical disturbance (e.g. dredging) and quality of the overlying water. Sediment parameters respond quickly to changes in the environment but are also able to integrate changes over short periods of time (weeks to months) and are thus good indicators or short to very short-term changes in environmental health.

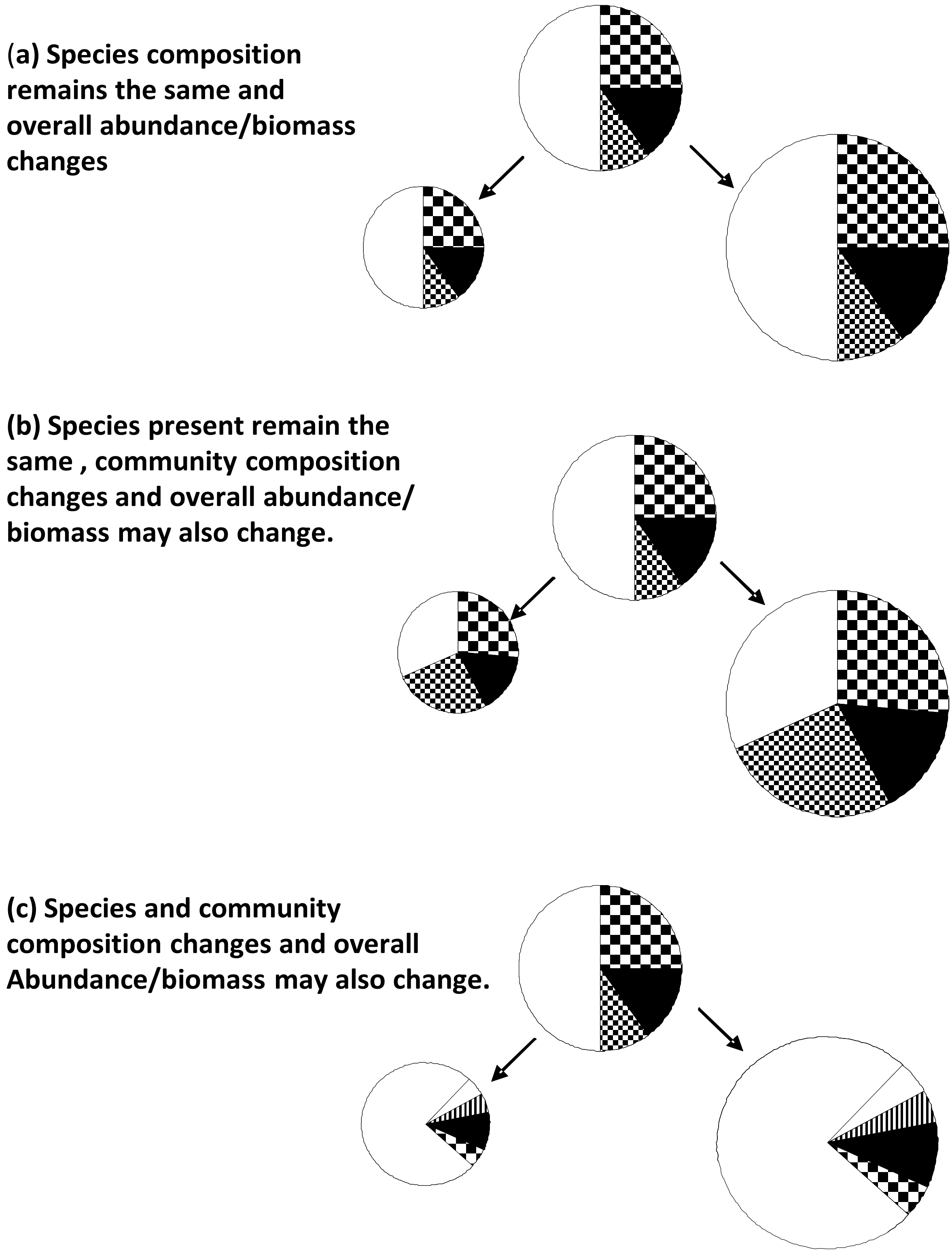

Figure 2.1. **Possible alterations in abundance/biomass and community composition. Overall abundance/biomass is represented by the size of the circles and community composition by the various types of shading. After Hellawell (1986).**

Coastal development: Coastal development includes development activities such as infrastructure (harbours and launch sites, cities, towns, housing, roads and tourism), as well as dredging and the disposal of dredge spoil. Coastal developments pose a major threat to many components of marine and coastal environments, owing to their cumulative effects, which are often not taken into account by impact assessments. Associated impacts include organic pollution of runoff and sewerage, transformation of the supratidal environment, alteration of dune movement, increased access to the coast and sea, and the negative impacts on estuaries.

Shoreline erosion: Anthropogenic activities, particularly structures erected in the coastal zone (e.g. harbours, breakwaters, buildings) and dredging activities, can also profoundly influence shorelines composed of soft sediment (i.e. sandy beaches) leading to erosion of the coast in some areas and the accumulation of sediment in others. Many of the beaches in Saldanha Bay have experienced severe erosion in recent decades to the extent that valuable infrastructure is severely threatened in some areas.

Macrophytes: Estuarine macrophytes are good indicators of ecological health and condition due to their temperature and salinity tolerance range, sensitivity to nutrient loads and extent in response to anthropogenic and climatic changes. They can be monitored relatively infrequently (1x per year) as well as at low costs once the initial ground truthing assessment has been captured. With advancements in remote sensing and spatial analytics a long-term monitoring framework can be easily maintained.

Macrofauna: Benthic macrofauna are mostly short-lived organisms (1-3 years) and hence are good indicators of short to medium term (months to years) changes in the health of the environment. They are particularly sensitive to changes in sediment composition (e.g. particle size, organic content and trace metal concentrations) and water quality.

Rocky intertidal: Rocky intertidal invertebrates are also mostly short-lived organisms (1-3 years) and as such are good indicators of short to medium term changes in the environment (months to years). Rocky intertidal communities are susceptible to invasion by exotic species (e.g. Mediterranean mussel), deterioration in water quality (e.g. nutrient enrichment), structural modification of the intertidal zone (e.g. causeway construction) and human disturbance resulting from trampling and harvesting (e.g. bait collecting).

Fish: Fish are mostly longer-lived animals (3-10 years +) and as such are good indicators of medium to long term changes in the health of the environment. They are particularly sensitive to changes in water quality, changes in their food supply (e.g. benthic macrofauna) and fishing pressure.

Birds: Birds are mostly long-lived animals (6-15 years +) and as such are good indicators of long-term changes in the health of the environment. They are particularly susceptible to disturbance by human presence and infrastructural development (e.g. housing development), and changes in food supply (e.g. pelagic fish and intertidal invertebrates).

Alien species: A large number of alien marine species have been recorded as introduced to southern African waters. South Africa has at least 85 confirmed alien species, some of which are considered invasive, including the Mediterranean mussel *Mytilus galloprovincialis*, the European green crab *Carcinus maenas*, and the barnacle *Balanus glandula*. Most of the introduced species in South Africa have been found in sheltered areas such as harbours and are believed to have been introduced through shipping activities, mostly ballast water. Ballast water tends to be loaded in sheltered harbours, thus the species that are transported often originate from these habitats and have a difficult time adapting to the more exposed sections of the southern African coastline, but are easily able to gain a foothold in sheltered bays such as Saldanha Bay.

Table 2.1. Ranking categories and classification thereof as applied to Saldanha Bay and Langebaan Lagoon for the purposes of this report.

Health category	Ecological perspective	Management perspective
Natural	No or negligible modification from the natural state	Relatively little human impact
Good	Some alteration to the physical environment. Small to moderate loss of biodiversity and ecosystem integrity.	Some human-related disturbance, but ecosystems essentially in a good state, however, continued regular monitoring is strongly suggested
Fair	Significant change evident in the physical environment and associated biological communities.	Moderate human-related disturbance with good ability to recover. Regular ecosystem monitoring to be initiated to ensure no further deterioration takes place.
Poor	Extensive changes evident in the physical environment and associated biological communities.	High levels of human related disturbance. Urgent management intervention is required to avoid permanent damage to the environment or human health.

3 ACTIVITIES AND DISCHARGES AFFECTING THE HEALTH OF THE BAY

3.1 Introduction

Industrial development of Saldanha Bay dates back to the early 1900s with the establishment of a commercial fishing and rock lobster industry in the Bay. By the mid-1900s, Southern Seas Fishing Enterprises and Sea Harvest Corporation had been formed, with Sea Harvest becoming the largest fishing operation in Saldanha Bay to date. Human settlement and urbanization grew from village status in 1916, to an important city with a population of more than 40 000 today. With increasing numbers of fishing vessels operating in Saldanha Bay, and to facilitate the export of iron ore from the Northern Cape, the bay was targeted for extensive development in the early 1970s. The most significant developments introduced at this time were the causeway linking Marcus Island to the mainland, to provide shelter for ore-carriers, and the construction of the iron ore terminal. These two developments effectively separated the Bay into two compartments – Small Bay and Big Bay. By the end of the 1970s Saldanha Bay harbour was an international port able to accommodate large ore-carriers.

Port facilities in Saldanha Bay now include the main Transnet iron ore terminal with berths for three ore carriers, an oil jetty, a multi-purpose terminal, and a general maintenance quay, a fishing harbour which is administered by the Department of Environmental Affairs, a Small Craft Harbour which is used by fishing vessels and tugs, three yacht marinas (Saldanha, Mykonos and Yachtport SA), a Naval boat yard at Salamander Bay and numerous slipways for launching and retrieval of smaller craft. Development of the port and fishing industry have served to attract other industry to the area, including oil and gas, ship repair and steel industries, and also resulted in a rapid expansion in urban development in Saldanha and Langebaan. Urban and industrial developments encroaching into coastal areas have caused the loss of coastal habitats and affect natural coastal processes, such as sand movement. Development of the port is expected to increase dramatically with the establishment of the Saldanha Bay Industrial Development Zone (SBIDZ), a process that was initiated in 2013.

Metal ores exported from the Port of Saldanha Bay include iron, lead, copper, zinc, and manganese. The Port of Saldanha currently has the capacity to export up to 60 million tonnes of iron ore per year but is in the process of upgrading the infrastructure to support an annual export of 80 million tonnes. However, the Transnet Port Terminals have thus far been unsuccessful in obtaining a variation to their existing Air Emission License (AEL) applicable to the Iron Ore Terminal for the storage and handling of coal and ore. The latest application was for the increase of handling and storage of coal and ore to 67 million tonnes per annum and was accompanied by an impact assessment and public participation process. The competent authority denied TPT the amendment concluding that environmental impacts at the current production level are already too high.

Disposal of wastewater is a major problem in the region, and much of it finds its way into the Bay as partially treated sewage, storm water, industrial effluent (brine, cooling water discharges and fish factory effluent) and ballast water. Until recently sewage discharge was arguably the most important waste product that is discharged into Saldanha Bay in terms of its continuous environmental impact. Sewage is harmful to biota due to its high concentrations of nutrients which stimulate primary production that in turn leads to changes in species composition, decreased biodiversity, increased dominance, and toxicity effects. The changes to the surrounding biota are likely to be permanent depending on distance to outlets and are also likely to continue increasing in future given the growth in industrial development and urbanisation in the area. During the recent drought in the Western Cape, however, industry and local municipalities came together to investigate the feasibility of reclaiming industrial grade and potable freshwater from treated sewage in Saldanha Bay. Major infrastructural changes are required for the re-cycling of treated sewage and are associated with significant initial as well as ongoing fiscal investments. Budgetary constraints experienced by local municipalities were overcome by means of a public-private partnership. The majority of the wastewater is being used for irrigation, and current water users receiving treated effluent include: the Weskus School, Saldanha Sports Ground (Stadium and practise field), Blue Bay Lodge and the Langebaan Country Estate. It is reported that no effluent from The Langebaan Wastewater treatment works is entering the Bay. In contrast, the balance of treated effluent from Saldanha WWTW not used for irrigation is currently discharged into the Bok river and ultimately ends up in the ocean, however, SMB has identified a future user for the treated effluent and an allocation has been made available to them.

Ballast water discharges are by far the highest in terms of volume and have been increasing year on year due to constant and increasing shipping traffic. Ballast water often includes high levels of contaminants such as trace metals and hydrocarbons, and, along with the vessels that carry the ballast water, serves to transport alien species from other parts of the world into Saldanha Bay. Ballast water discharges can, however, be effectively managed and the remit of the International Maritime Organisation (IMO) is to reduce the risks posed by ballast water to a minimum through the direct treatment of the water while on board the ship, as well as by regulating the way in which ballast water is managed while the ship is at sea. Although no domestic legislation is currently in place to regulate ballast water discharge, the Transnet National Port Authority in Saldanha Bay has implemented a number of mechanisms to track and control the release of ballast water into the harbour.

Dredging in Saldanha Bay has had tremendous immediate impact on benthic micro- and macrofauna, as particles suspended in the water column kill suspension feeders like fish and zooplankton. It also limits the penetration of sunlight in the water column and causes die offs of algae and phytoplankton. Furthermore, fine sediment can drift into the Langebaan Lagoon, changing the sediment composition, which in turn can directly and indirectly (through their food supply) affect wader birds in the lagoon. The damage caused by dredging is generally reversible in the long term, and although the particle composition of the settled material is likely to be different, ecological functions as well as major species groups generally return in time. Transnet intends to construct new port infrastructure to support the Industrial Development Zone (IDZ) and dredging activities are likely to commence in the near future.

Saldanha Bay is a highly productive marine environment and constitutes the only natural sheltered embayment in South Africa (Stenton-Dozey *et al.* 2001). These favourable conditions have facilitated the establishment of an aquaculture industry in the Bay. A combined 464 ha of sea space are currently available for aquaculture production in Outer Bay, Big Bay and Small Bay. With the support of finances and capacity allocated to the Operation Phakisa Delivery Unit, the Department of Agriculture Forestry and Fisheries established a sea-based Aquaculture Development Zone (ADZ) in Saldanha Bay. The ADZ areas comprise four precincts totalling 420 ha of new aquaculture areas in Saldanha Bay. Previously allocated areas have been incorporated into the ADZ which now comprised a total area of 884 ha set aside for mariculture.

Historic studies as well as the State of the Bay surveys have shown that these culture operations can lead to organic enrichment and anoxia in sediments under the culture rafts and ropes. The source of the contamination is believed to be mainly faeces, decaying mussels and fouling species. The scale of the proposed ADZ is significant and environmental monitoring of the Bay should be intensified to prevent significant ecological impacts, as well as loss to the mariculture sector itself.

Each of the aspects summarised above are addressed in more detail in the various sections of this Chapter. The impacts of these various activities and discharges are evaluated against their potential threat to the ecological integrity of Saldanha Bay and Langebaan Lagoon. In some instances, proposed developments (including environmental impacts and proposed mitigation measures) detailed in previous reports have been omitted and the reader is referred to earlier State of Saldanha Bay and Langebaan Lagoon Reports for further information on these development proposals. This only applies to those developments and activities that have not changed significantly in the past year.

Concerns have been raised that cumulative impacts on the marine environment in Saldanha Bay have not been adequately addressed by many of recent development proposals. This applies especially to the cumulative impacts that will arise from future development within the Saldanha Bay IDZ and Aquaculture Development Zone (ADZ). Furthermore, the impact on the Saldanha Bay marine environment from projects that are primarily land-based, such as storage facilities for crude oil and liquid petroleum gas, has generally been underestimated or even ignored. It has been proposed that a more holistic management strategy is needed to deal with the piece meal Environmental Impact Assessments (EIA). Various environmental management instruments have been proposed for the Greater Saldanha Bay Area, including (1) a generic Environmental Management Programme (EMPr), (2) an Environmental Management Framework (EMF), (3) a Strategic Environmental Assessment (SEA), and (4) the declaration of a Special Management Area (Refer to Chapter 4 for more details on this). An Intergovernmental Task Team (IGTT) has been set-up to consider these and other proposals. If these management instruments are indeed implemented, measures for the conservation alongside rapid development of the Saldanha Bay area will be addressed more effectively.

3.2 Urban and industrial development

Saldanha grew from a small fishing village into a town that supports multiple industries largely as a result of the benefits it accrues from being a sheltered bay on an otherwise exposed coastline. The development of a large-scale industrial port in Saldanha Bay commenced with the construction of an iron ore export facility in the 1970s. The primary purpose of the port at that stage was to facilitate the export of iron ore as part of the Sishen-Saldanha Bay Ore Export Project. The first major development in the Bay towards the realisation of these goals was the construction of the iron ore terminal and a causeway, built in 1975, that linked Marcus Island to the mainland, providing shelter for ore-carriers. The construction of the iron ore terminal essentially divided Saldanha Bay into two sections: a smaller area bounded by the causeway, the northern shore and the ore terminal (called Small Bay); and a larger, more exposed area adjacent called Big Bay, leading into Langebaan lagoon (Figure 3.3.).

In the late 1990s, a multi-purpose terminal (MPT) was completed, which was followed by an offshore fabrication facility. Existing facilities now include an oil import berth, three small craft harbours, a loading quay and a tug quay. Mariculture farms and several fish processing factories also make use of the Bay. Approximately 400 ha of Saldanha Bay were zoned for mariculture operations in 1997, the majority of which farm mussels and oysters. Development of the causeway and iron ore terminal in Saldanha Bay greatly modified the natural water circulation and current patterns (Weeks et al. 1991b) in the Bay. Combined with increasing land-based effluent discharges into the bay, these developments have led to reduced water exchange and increased nutrient loading of water within the Bay. More recently, Henrico and Bezuidenhout (2020) illustrate how the construction of the harbour altered the bathymetry within the Bay leading to an increased water depth of roughly 1.4 m, steeper surf zone slopes (as a result of erosion of the north eastern edge of the Bay) and a generally smoother, steeper bottom profile in the Bay (Chapter 7).

Aerial photographs taken in 1960 (Figure 3.1), 1989 (Figure 3.2) and in 2007 (Figure 3.3.) clearly show the extent of development that has taken place within Saldanha By over the last 50 years. The current layout of the Port of Saldanha is shown in Figure 3.4. Future plans, including short term (2028) and long-term (Beyond 2048) goals for the development of the bay are shown in Figure 3.5 and Figure 3.6, these were taken from the National Port Plans 2019 update (TNPA 2019).

Future industrial development of Saldanha Bay will be strongly driven by Operation Phakisa, which was launched in July 2014 by the South African Government with the goal of boosting economic growth and creating employment opportunities. Operation Phakisa is an initiative that was highlighted in the National Development Plan (NDP) 2030 to address issues such as poverty, unemployment and inequality in South Africa. "Phakisa" means "hurry up" in Sesotho emphasising the government's urgency to deliver. Operation Phakisa is a cross-sectoral programme, one of which is focused on unlocking the economic potential of South Africa's oceans through innovative programmes. Four critical areas were identified to further explore and unlock the potential of South Africa's oceans:

1. Marine transport and manufacturing
2. Offshore oil and gas exploration
3. Marine aquaculture
4. Marine protection services and ocean governance

In line with this development, Transnet and Transnet National Ports Authority (TNPA) have thus far initiated three developments in the Port of Saldanha Bay related to oil and gas services as well as marine infrastructure repair and fabrication. These developments are described in more detail in the sections below. Furthermore, the established Saldanha Bay aquaculture industry will be expanded through the Saldanha Bay Aquaculture Development Zone (ADZ) under the auspices of Operation Phakisa (Section 3.8).

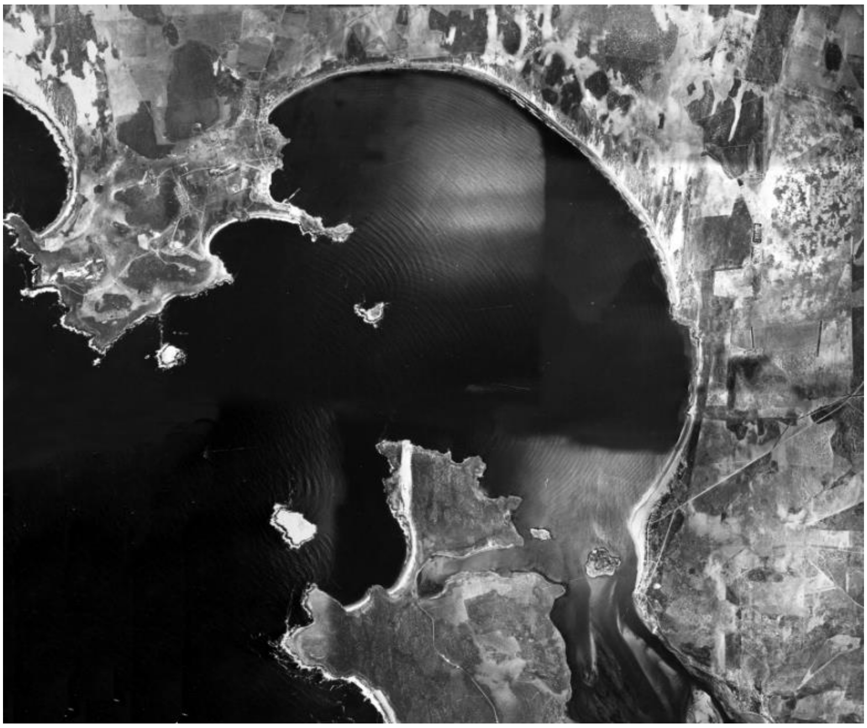

Figure 3.1. Composite aerial photo of Saldanha Bay and Langebaan Lagoon taken in 1960. (Source Department of Surveys and Mapping). Note the absence of the ore terminal and causeway and limited development at Saldanha and Langebaan.

Figure 3.2. **Composite aerial photo of Saldanha Bay and Langebaan Lagoon taken in 1989 (Source Department of Surveys and Mapping). Note the presence of the ore terminal, the causeway linking Marcus Island with the mainland, and expansion of settlements at Saldanha and Langebaan.**

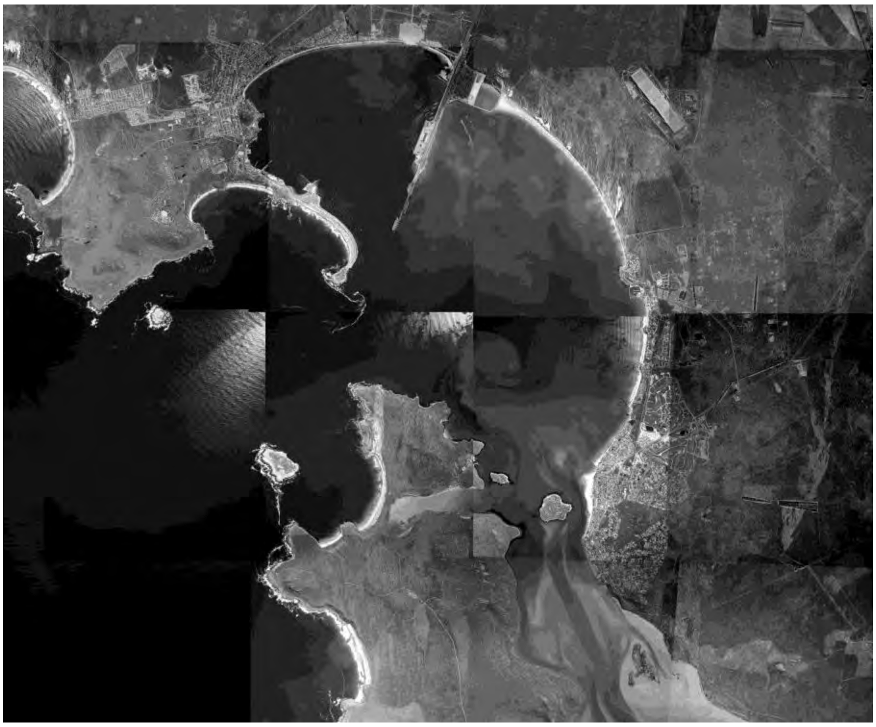

Figure 3.3. **Composite aerial photo of Saldanha Bay and Langebaan Lagoon taken in 2007. (Source Department of Surveys and Mapping). Note expansion in residential settlements particularly around the town of Langebaan.**

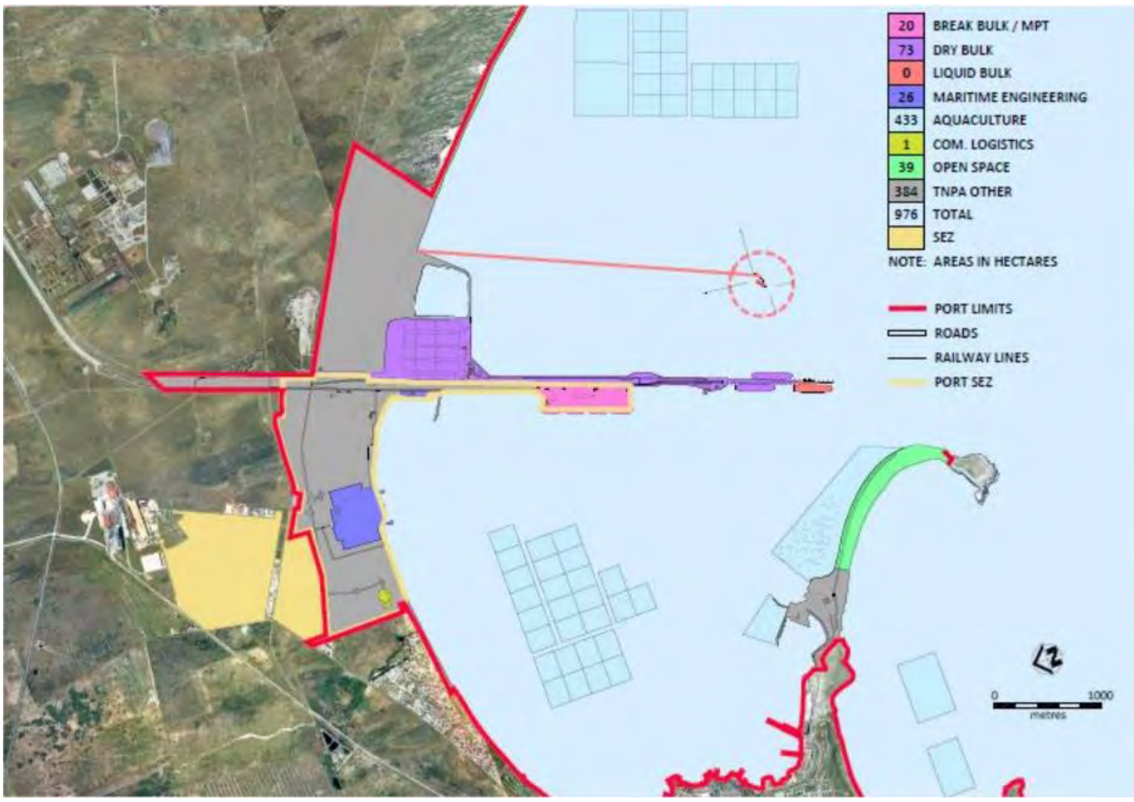

Figure 3.4. Current layout of Transnet Saldanha Bay Port (Source: Transnet National Port Authority 2019, National Port Plans 2019 update).

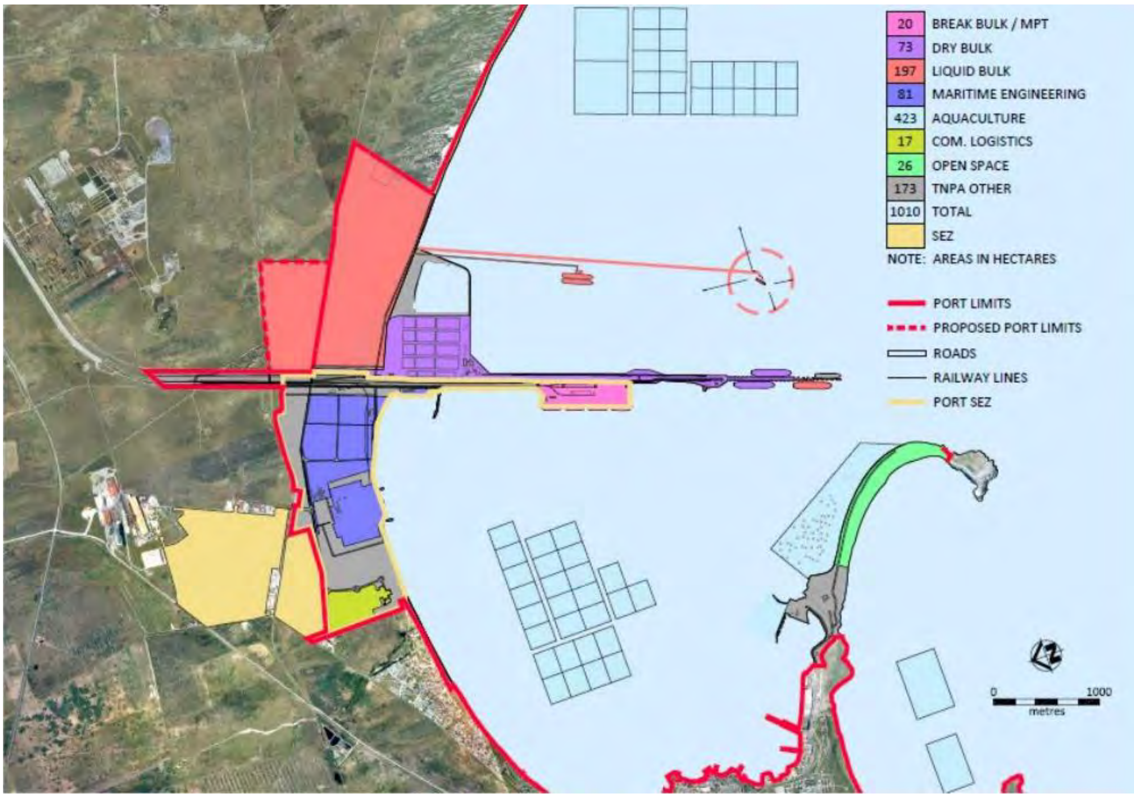

Figure 3.5. Short term layout (2028) of Transnet Saldanha Bay Port (Source: Transnet National Port Authority 2019, National Port Plans 2019 update).

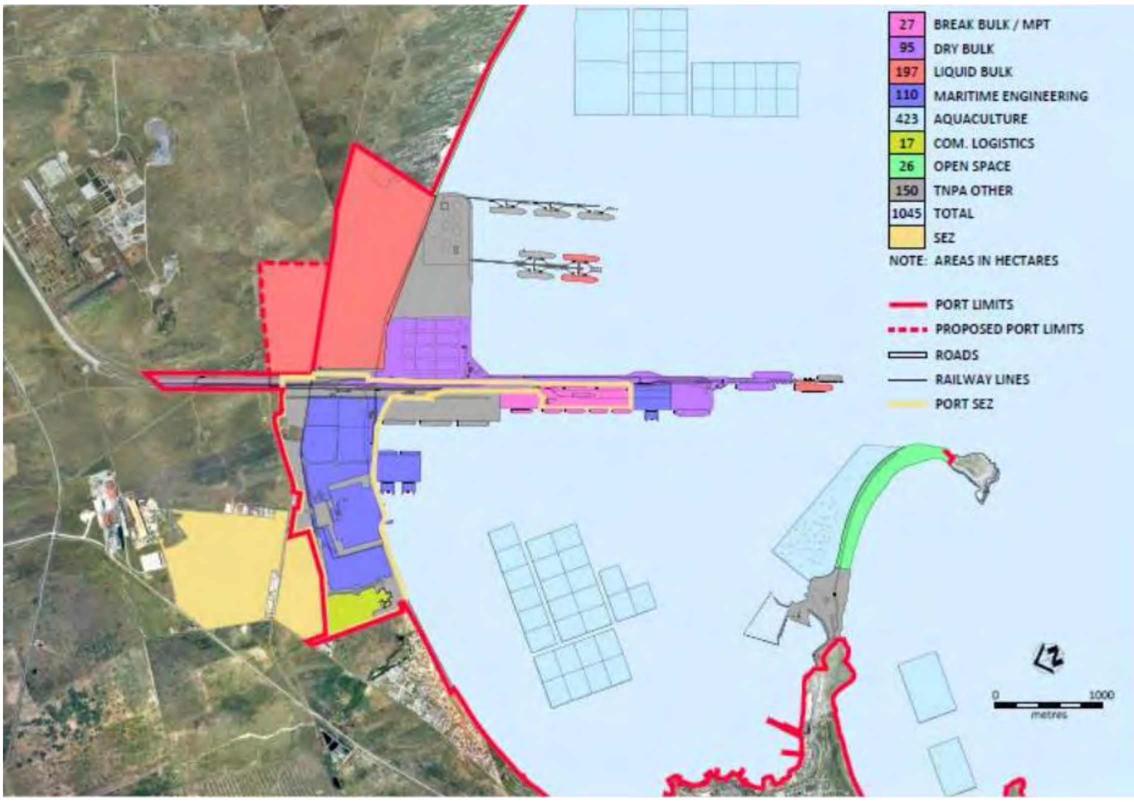

Figure 3.6. Long term layout (2044) of Transnet Saldanha Bay Port (Source: Transnet National Port Authority 2019, National Port Plans 2019 update).

Data on population growth in the town of Saldanha and Langebaan Lagoon are available from the 1996, 2001 and 2011 census data. The population of Saldanha increased from 16 820 in 1996 to 21 636 in 2001 and to 28 135 in 2011, growth slowing from an initial rate of 5.7% per year in the first period to just 2.7% per year in the second (Statistics South Africa 2014). In contrast, the Langebaan population increased from 2 735 to 3 428 between 1996 and 2001 (2.5% per year), and rapidly from there up to 8 294 in 2011 (a growth rate of 9.24%/year) (Table 3.1.) (Statistics South Africa 2014). The human population in Saldanha Bay, particularly that in Langebaan Village, is thus expanding rapidly, which has been attributed to the immigration of people from surrounding municipalities in search of real or perceived jobs (Saldanha Bay Municipality 2011). These population increases are no doubt increasing pressure on the marine environment and the health of the Bay through increased demand for resources, trampling of the shore and coastal environments, increased municipal (sewage) and household discharges (which are ultimately disposed of in Saldanha Bay) and increased storm water runoff due to expansion of tarred and concreted areas.

Urban development around Langebaan Lagoon has encroached right up to the coastal margin, leaving little or no coastal buffer zone (Figure 3.7. and Figure 3.8.). Allowing an urban core to extend to the waters' edge places the marine environment under considerable stress due to trampling and habitat loss. It also increases the risks of erosion due to removal of vegetation and interferes with certain coastal processes such as sand deposition and migration.

Expansion of tarred areas also increases the volumes of storm water entering the marine environment, which ultimately can have a detrimental effect on ecosystem health via the input of various contaminants and nutrients (See Section 3.6).

Table 3.1. Total human population and population growth rates for the towns of Saldanha and Langebaan from 2001 to 2011 (Statistics South Africa, 2014).

Location	Total Population 1996	Total Population 2001	Total Population 2011	Growth 2001-2011 (%/yr.)
Saldanha	16 820	21 363	28 135	2.66
Langebaan	2 735	3 428	8 294	9.24

Figure 3.7. Satellite image of Saldanha (Small Bay) showing little or no set-back zone between the town and the Bay. Source: Google Earth.

Figure 3.8. Satellite image of Langebaan showing absence of development set-back zone between the town and the lagoon. Source: Google Earth.

Industrial and urban development in and around Saldanha Bay has been matched with increasing tourism development in the area, specifically with the declaration of the West Coast National Park, Langebaan Lagoon being declared a National Wetland RAMSAR site and establishment of holiday resorts like Club Mykonos and Blue Water Bay. The increased capacity for tourism results in higher levels of impact on the environment in the form of increased pollution, traffic, fishing and disturbance. Long term data (2005-2020) on numbers of visitors to the West Coast National Park (WCNP) indicate strong seasonal trends in numbers of people entering the park, peaking in the summer months and during the flower season in August and September (Figure 3.9). Paying day guests (excluding international visitors) and free guests[1] contribute the most to this seasonal pattern, while international guests and overnight guest numbers are relatively constant throughout the year. International and overnight guest numbers are considerably lower than the other visitor categories.

Visitor numbers have been increasing at an average rate of 13% per annum[2] up until the 2016/2017 period which had a total of over 338 thousand visitors. Since then, the total number of visitors to the park has been decreasing steadily until 2018/2019 and then dropped by 23% in the last rolling year to just under 216 thousand visitors in 2019/2020, this is due in part to the fact that as a result of COVID-19 restrictions the park was closed for just over two months (Figure 3.10). The number of free guests has been increasing steadily over time and now equals the proportion of day guests, however, the

[1] These include Wild Card, school class, military personnel, official visit, staff, residents and 'other' entries.

[2] The average annual growth rate was calculated from the data reflecting the total numbers of tourists entering the West Coast National park in a rolling 12-month period.

recent data format shows that this category is likely made up of equal parts complimentary guests and guests with memberships, such as Wild Card holders. The number of international visitors has stayed relatively constant over time while popularity of overnight stays inside the park decreased substantially after 2009, reaching its lowest level in 2015/2016 with 2 041 guests. However, overnight visitor numbers have increased over the last two years, reaching an all-time low of 215 visitors in 2019/2020. It should be noted that SANParks tourism data is now managed by national head office and the reporting structure has been standardised across all national parks. Only in total number of guests and wild card holder numbers were available in 2018/19. Wild card holders comprised 28 and 25% of the total number of visitors to the West Coast National Park in 2018/2019 and 2019/2020 respectively.

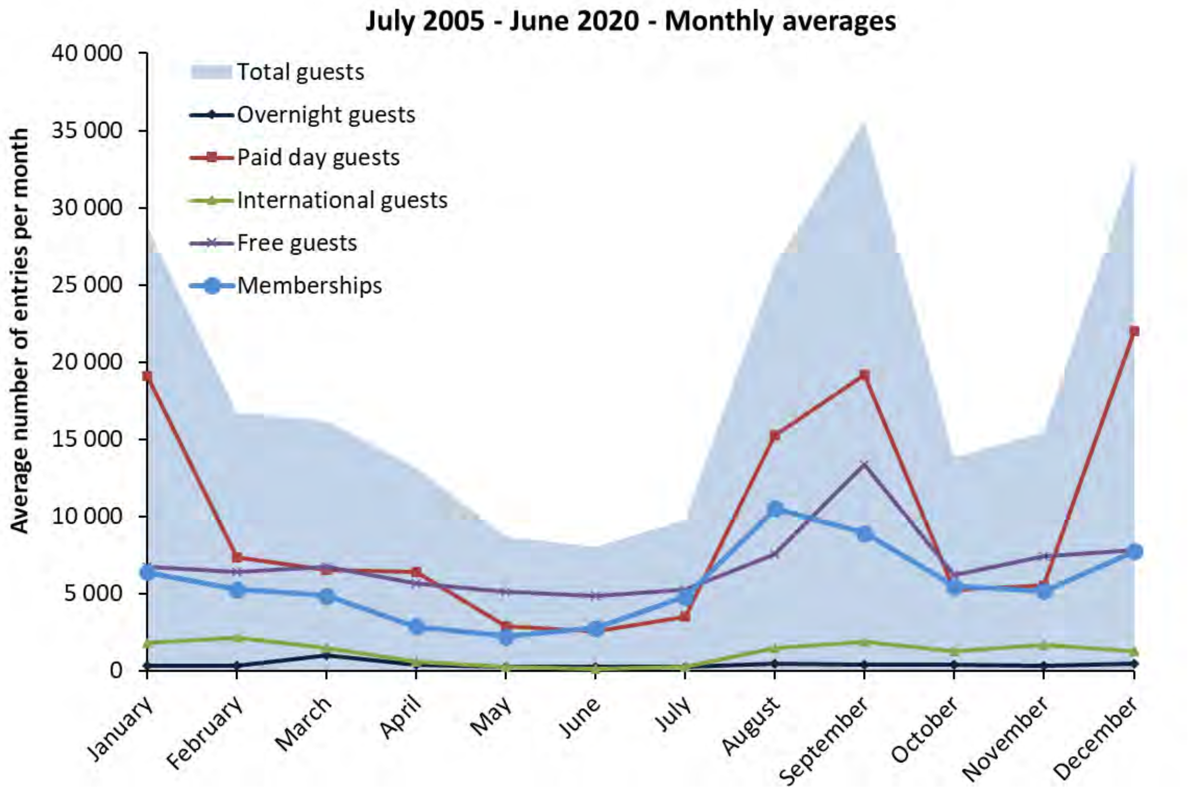

Figure 3.9. Monthly average numbers of entries into the West Coast National Park between July 2005 and June 2020. Paid day guests include all South African visitors (adults and children), while Overnight guests refer to those staying in SANParks accommodation. International guests include all SADC and non-African day visitors (adults and children) while the category 'Free guests' includes residents, staff, military, school visits, wild cards etc. Note that SANParks tourism data is now managed by national head office and the reporting structure has changed. In 2018/2019 only total number of guests and wild card holders were recorded, however, this has changed with recent data again being categorised. 2019/2020 'Free guests' now divided into 'Free' and 'Memberships' (Wild cards) (Source: Pierre Nel, WCNP).

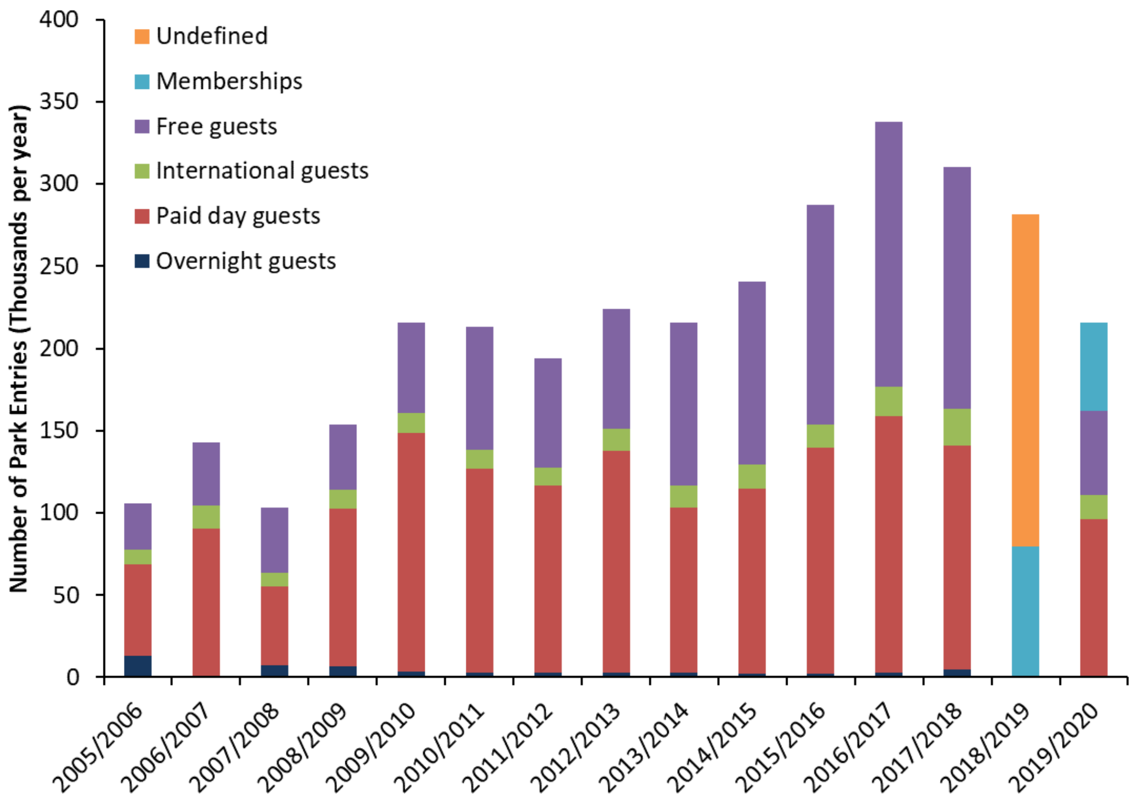

Figure 3.10. Numbers of entries into the West Coast National Park in a rolling 12-month periods from July 2005 until June 2020. Paid day guests include all South African visitors (adults and children) while Overnight guests refer to those staying in SANParks accommodation. International guests include all SADC and non-African day visitors (adults and children) while the category 'Free guests' includes residents, staff, military, school visits, wild cards etc. Note that SANParks tourism data is now managed by national head office and the reporting structure has changed. In 2018/2019 only in total number of guests and wild card holders (Memberships) were recorded, however, this has changed with recent data again being categorised. 2019/2020 'Free guests' now divided into 'Complimentary' and 'Memberships' (Wild cards) (Source: Pierre Nel, WCNP).

In terms of the Municipal Systems Act 2000 (Act 32 of 2000) every local municipality must prepare an Integrated Development Plan (IDP) to guide development, planning and management over the five-year period in which a municipality is in power. A core component of an IDP is the Spatial Development Framework (SDF) which is meant to relate the development priorities and the objectives of geographic areas of the municipality and indicate how the development strategies will be co-ordinated. An SDF aims to guide decision making on an on-going basis such that changes, needs and growth in the area can be managed to the benefit of the environment and its inhabitants. The latest version of the Saldanha Municipality IDP covers the period 2017-2022 IDP. The latest SDF for the Saldanha Bay Municipality (SBM) was produced in 2011, reviewed in 2018 and is available on the municipality website. This document advocates a holistic approach to the development of the municipality, ensuring that the municipal spatial planning of the rural and urban areas is integrated for the first time since the establishment of the municipality. The Spatial Development Framework 2020 of the West Coast District Municipality was adopted at a Council meeting held on 27 May 2020.

A study by Van der Merwe *et al.* (2005) assessing the growth potential of towns in the Western Cape (as part of the provincial SDF) identified Langebaan and Saldanha as towns with high growth potential.

It was estimated that, given the projected population figures, there would be a future residential demand of 9 132 units in Saldanha and 3 781 units in Langebaan. The SDF proposes addressing these demands by increasing the residential density in specified nodes in both towns and by extending the urban edge of Saldanha in a northerly direction towards Vredenberg, and that of Langebaan inland towards the North-East.

3.2.1 The Saldanha Bay Industrial Development Zone

Saldanha Bay has long been recognised as a strategically important industrial centre in the Western Cape. This provided a strong foundation for the establishment of an Industrial Development Zone (IDZ) in October 2013. The Saldanha Bay IDZ (SBIDZ) is the first Special Economic Zone (SEZ) to be located within a port and is the only sector specific SEZ in South Africa catering specifically to the oil and gas, maritime fabrication and repair industries and related support services (SBIDZ 2019). The SBIDZ is managed by the SBIDZ Licensing Company (LiCo). The SBIDZ LiCo is the holder of an Environmental Authorisation (EA) for the development of an oil and gas offshore service complex (EA was granted on 16 November 2015). More information on the on the SBIDZ can be found in previous versions of the State of Saldanha Bay and Langebaan Lagoon report (AEC 2018).

At the time of the initial application for EA, it was not known which future operations and specific industries would be established within the SBIDZ. It was thus not possible to account for all possible activities in terms of the NEMA EIA Regulations that might be triggered by future developments or operations within the SBIDZ (SLR 2019). Recently, EA for the storage of dangerous goods/hazardous substances within the IDZ was granted on 2 August 2019. The appeal period was concluded on 26 August 2019.

The SBIDZ has the potential to impact on the marine environment in Saldanha Bay in numerous ways, including increased vessel traffic, which cumulatively contributes to underwater noise and invasive alien species transfer (via ballast water release); increased pollution of the Saldanha Bay through maintenance and repair activities, and storm water runoff. Although a detailed marine ecological specialist study was not conducted as part of the EIA process, mitigation measures for these direct and indirect marine ecological impacts were included in the Final Environmental Impact Report (SLR 2016). Potential impacts that may occur as a result of the construction and operation of marine infrastructure associated with the Offshore Service Complex (OSC) is to be investigated in a separate EIA process undertaken by the TNPA at a later stage.

3.2.2 The Sishen-Saldanha oreline expansion project

Currently, iron ore is mined in Hotazel, Postmasburg and Sishen before being transported on a freight train 861 km to Saldanha Bay. From the train, it is loaded onto conveyor belts and then placed in stockpiles to be loaded into the holds of cargo ships. Transnet is currently installing a third iron ore tippler to ensure that 60 million tonnes per annum of iron ore can continue to be exported (GIBB 2013b) (refer to the 2014 State of Saldanha Bay and Langebaan Lagoon report for more information on this project).

Transnet in conjunction with six mining companies (Aquila Steel, Assmang, Kumba Iron Ore, PMG, Tshipi é Ntle and UMK) are now proposing an oreline expansion project. This would increase the capacity of the current Sishen-Saldanha railway and port from 60 to 88 million tonnes per annum in order to satisfy the global demand for iron ore (GIBB 2013). The Sishen-Saldanha oreline expansion project has three major components, namely a facility for emerging miners (mine-side ore loading), iron ore rail and a port iron ore terminal (GIBB 2013). The three components of this project are currently still in the planning phase (refer to the 2014 State of Saldanha Bay and Langebaan Lagoon report for more information on this project). Transnet recently

Transnet recently increased the length of the Manganese ore train from 312 to 375 carriages, thereby increasing the volume of manganese transported during individual trips from roughly 19.5 thousand tonnes to roughly 23.5 thousand tonnes.

An increase in rail capacity will result in a greater volume of ore arriving in Saldanha and accordingly an increase in ship traffic will be necessary to transport this product globally. In 2020, 291 iron ore ships arrived and departed from the iron ore terminal in the Port of Saldanha, exporting 56.9 million tonnes of iron ore (Section 3.3). The slight drop in ship numbers and overall export volumes as compared to 2019 (314 ships and 57.2 million tonnes) could be the result of restrictions places on the ports during the country wide COVID 19 lock down, as well as heavy storms and winds experienced during the winter months which resulted in a loss of 182 hours of load time (Mining Weekly August 2020).

3.2.3 Development of liquid petroleum gas facilities in Saldanha Bay

Liquid Petroleum Gas (LPG) is a fuel mix of propane and butane which is in a gaseous form at ambient temperature but is liquefied under increased pressure or by a temperature decrease. The LPG industry is currently expanding to provide an alternative energy source in South Africa and to reduce the pressure on South Africa's electricity grid. In line with the National LPG Strategy (DEA&DP 2014), 1.5 million households are aimed to convert to LPG over the next five years. These new developments will contribute cumulatively to existing impacts in Saldanha Bay such as stormwater runoff and increased vessel traffic. The offloading of imported LPG in the harbour poses an additional pollution risk to ecosystems in Saldanha Bay.

Sunrise Energy (Pty) Ltd is currently building an LPG import facility in the Saldanha Bay Harbour and was scheduled to be completed in mid-2016 (Sunrise Energy (Pty) Ltd, Janet Barker, *pers. comm.* 2014*)*. This development aims to supplement current LPG refineries and distributors in the Western Cape and ensure that industries dependant on LPG can remain in operation. An EIA process in terms of section 24 of the NEMA was initiated by ERM Southern Africa in 2012 and EA was granted on 13 May 2013 by the DEA&DP (refer to AEC 2014 for more information). The Draft EMPr for the project required that environmental/sediment monitoring be undertaken prior to and during installation of marine infrastructure to monitor effects on the surrounding environment, and that annual monitoring of environment/sediment in the vicinity of the marine facilities to assess any potential operational impacts on water quality. It was recommended that such monitoring be undertaken as part of the Saldanha Bay Water Quality Forum Trust`s monitoring program, and this is currently underway. The bulk earthworks and construction commenced in January 2014, and installation of the marine infrastructure commenced in September 2017 (Sunrise Energy (Pty) Ltd, Janet Barker, *pers. comm.* 2015*)*. Delays in this project have occurred due to unforeseen difficulties and legal issues with competitors Avedia Energy.

Avedia Energy developed a land based liquid petroleum gas storage facility on Portion 13 of Farm Yzervarkensrug No. 127 in Saldanha. The storage facility was designed to include 16 mounded bullet tanks with a storage capacity of 250 metric tonnes each (Frans Lesch, ILF Consulting Engineers, Project Manager at Avedia Energy Saldanha LPG plant, *Pers. Comm.* 2015) (refer to AEC 2014 for more information). Avedia Energy completed construction of their LPG storage facility in 2017 and upgraded the facilities to support an additional 6000 tonnes of LPG monthly in the second half of 2018.

3.2.4 Liquefied Natural Gas Import Facilities

The proposed Liquefied Natural Gas (LNG) Import Facilities aim to secure gas supplies to supplement land-based gas power plants, other industrial users and FPPs (ERM 2015b). This project constitutes phase two in national efforts to contribute towards meeting South Africa's electricity requirements. Phase two will allow for the development of medium- to long-term gas power plants outside of the port boundaries (Section 3.2.5) (ERM 2015a and 2015b). ERM provided stakeholders with a Background Information Document in October 2015 of which excerpts and illustrations are provided in previous AEC reports (ERM 2015a). The facilities will provide for the importation, storage, regasification and the transmission of natural gas to a distribution hub and will include both land-based (terrestrial) and marine-based components. Both, floating and land-based regasification technologies are currently considered for this project (refer to AEC 2017 and 2018 for more information on the infrastructure). A feasibility study for the Integrated Liquefied Natural Gas Importation and Gas-to-Power Project was completed in 2019 which demonstrated that that there is a demand for natural gas for industrial processes as well as a likelihood of the development of new industries in Saldanha that would require natural gas (Delphos International 2019).

3.2.5 Gas fired independent power plant

The International Power Consortium South Africa (Pty) Ltd (" IPCSA") have proposed the construction of a Combined Cycle Gas Turbine (CCGT) power plant (1507 MW net capacity) as a solution to medium to long-term sustainability of Arcelor Mittal's Saldanha Steel and surrounding economy (ERM 2015c). The project is primarily a Liquefied Natural Gas (LNG) power supply project to the Saldanha Steel Plant (ERM 2015c). LNG will be supplied by ship to the Port of Saldanha, where it will be re-gased and then offloaded via a submersible pipeline either from a mooring area located offshore or a berthing location in the Port of Saldanha. ArcelorMittal South Africa obtained Environmental Authorisation (EA) from the National Department of Environmental Affairs (DEA) under the National Environmental Management Act (NEMA) (Act No. 107 of 1998) (as amended) through a Scoping and Environmental Impact Assessment (EIA) process on 24 February 2017.

It is anticipated that this project will connect to the Department of Energy's (DoE's) planned LNG import terminal in the Port of Saldanha (Section 3.2.4). Should this not occur, a separate EIA will be undertaken to permit the marine component of the import of LNG. Given the closure of the

ArcelorMittal Steel Plant the marine component of the LNG import facility will likely require a separate EIA.

3.2.6 Crude oil storage facility

The Port of Saldanha reportedly represents an excellent strategic location to receive, store process and distribute crude oil from West Africa and South America (SouthAfrica.info 2013). Oil tanking MOGS Saldanha (RF) (Pty) Ltd (OTMS), a joint venture between MOGS (Pty) Ltd and OTGC Holdings (Pty) Ltd, are in the process of constructing a commercial crude oil blending and storage terminal with a total capacity of 13.2 million barrels, comprising twelve 1.1 million barrel in-ground concrete tanks in Saldanha Bay. The construction phase commenced at the beginning of 2015, but it is currently unknown when this project will be completed (refer to the 2014 State of Saldanha Bay and Langebaan Report for more information).

3.2.7 Elandsfontein phosphate mine

The Elandsfontein phosphate deposit is currently the second biggest known resource in South Africa. The deposit is located on the farm Elandsfontein 349, approximately 12 km to the east of Langebaan (Braaf 2014). The proposed mining area is located on the Elandsfontyn Aquifer System (EAS) and in close proximity to the Langebaan Road Aquifer System (LRAS). These aquifer systems are defined by palaeo-channels that have been filled with gravels of the Elandsfontyn Formation and represent preferred groundwater flow paths that feed into Langebaan Lagoon and Saldanha Bay, respectively (Braaf 2014). Consequently, the phosphate deposits underlie the groundwater table (i.e. within the saturated zone) (GEOSS, Julian Conrad, *pers. comm.* 2016).

The dominant application of phosphorus is in fertilisers and the demand in the agricultural sector is growing (Braaf 2014). Kropz Elandsfontein, previously known as Elandsfontein Exploration and Mining (Pty) Ltd. (EEM) commissioned Braaf Environmental Practitioners to facilitate the environmental authorisation process for the proposed Elandsfontein Phosphate project. Environmental Authorisation (EA) was granted in February 2015 and a water use license in April 2017 (refer to the 2016 State of Saldanha Bay and Langebaan Lagoon Report for details on the project description, potential impacts on Langebaan Lagoon, and ongoing environmental monitoring).

The commissioning of the mine has been halted for an extended period due to a long delay in the issuing of the mine's water use license (Furlong 2017). An environmental non-governmental organisation, the West Coast Environmental Protection Association (WCEPA), lodged an appeal with the Water Tribunal, which found in November 2017 that there was a "prima facie basis" to challenge the licence. In addition, the tribunal found that temporary permission granted by the Department of Water and Sanitation in December 2017 was "questionable". The temporary permission referred to by Kropz as having been granted by the responsible authority appears to be questionable as only a water use licence or general authorisation allows a person to use water according to the National Water Act. The hearing was set for September 2019 and there is still question around the validity of the WUL as well as direct opposition, by local activists, environmental lawyers and the community, to the sustainability of the water use in a water-poor. The general opinion remains that Kropz requires environmental authorisation under the National Environmental Management Act (NEMA) before any further work can be done.

Additionally, phosphate prices have reached a ten-year low, decreasing by almost 30% since the mining company was issued its mining right in January 2015. This, together with technical problems identified during the commissioning phase, has resulted in the temporary suspension of mining activities in Elandsfontein. Kropz intends to recommence operations in the 4th quarter of 2020 provided their WUL is granted/ re-instated.

Kropz Elandsfontein has adopted a precautionary approach and is carefully monitoring any potential impacts on Langebaan Lagoon in association with the Saldanha Bay Water Quality Forum Trust (SBWQFT). The State of the Bay monitoring activities undertaken by the SBWQFT have thus been expanded to incorporate monitoring of various biological and physico-chemical variables to establish an appropriate baseline against which any potential future changes in the Lagoon can be benchmarked. This includes monitoring of salinity and biota (benthic macrofauna) at the top of the lagoon. The results are presented in Chapter 5 (Groundwater), Chapter 8 (Aquatic macrophytes and Chapter 9 (Benthic macrofauna). Additionally, research into the impacts on groundwater are still underway by local hydrology experts, geohydrologists, the CSIR, independent researchers and consultants.

3.2.8 Zandheuvel phosphate mine

Adelaide Ruiters Mining & Exploration intends to develop a new phosphate mine 3 km outside of Louwville and 4.5 km north of Bluewater Bay. The intention is to mine phosphate on the Zandheuvel farm Portions 126 and 124, as well as on Witteklip and Yzervarksrug farms. The Mining Right application also includes apatite, quartz, calcite, feldspar, hematite/goethite, ilmenite, rutile, zircon, monazite, schorl (tourmaline), garnet, titanium oxide, limestone, sandstone, rare Earth elements and aggregates. These minerals are likely to be found on site as they are associated with the phosphate deposit in this area. The proposed mining methods are conventional truck and shovel open pit mining and will not include blasting. Backfilling and rehabilitation will decrease the overall environmental footprint of the project.

Water requirements will be met by municipal treated wastewater to reduce the impact of the mining activities on availability of potable water in the area. The mine will require approximately 2 ML per day. The proposed project will include the mine itself, offices, a processing plant and an upgrade to the existing access road to the R79.

The Draft Scoping Report was submitted to the Department of Mineral Resources (DMR) in August 2018. Stakeholders had until 19 September 2018 to comment on the proposed development. Mining engineering consultants VBKOM were appointed in 2019 to conduct the feasibility study for the Mine and despite COVID-19 restrictions a drilling campaign for a pilot plant test was successfully conducted in July 2020. VBKOM are also supplying engineering support for the Environmental Impact Assessment which has not yet been published.

It was reported in 2019 that according to an Intergovernmental Task Team the probability of prospecting for this project occurring prior to 2022 was high (as confirmed by the drilling campaign and pilot plant test mentioned above) however, the probability of extraction occurring before 2022 was reported as low (DEA&DP 2019).

3.2.9 TNPA projects under auspices of Operation Phakisa

Due to an increase in offshore activity in South Atlantic and West African waters, and the resulting demand for vessel repair facilities, the National Government and Transnet National Ports Authority (TNPA) proposed the development of new infrastructure at the Port of Saldanha in line with the objectives of Operation Phakisa. The new infrastructure is expected to include the following components:

1. A Vessel Repair Facility (VRF) for ships and oil rigs (Berth 205);
2. A 500 m long jetty at the Mossgas quay; and
3. A floating dry dock for inspection of Offshore Supply Vessels (OSV).

These three projects are described in more detail in Sections 3.2.9.1-3.2.9.3. The potential impacts on the marine environment associated with the VRF and the Mossgas Jetty are also summarised in Section 3.2.9.4. The development of Berth 205 and the Mossgas Jetty will require extensive dredging operations to allow large oil and gas vessels access to new berthing infrastructure. The total dredge area during construction for the long-term development scenarios for the Mossgas Jetty and Berth 205 was estimated by TNPA at approximately 2.6 million m^3. This equates to the second largest dredge event in the history of Small Bay and is comparable to the dredging which commenced in 1996 for the construction of the MPT (Refer Section 3.3 for more information about dredging in Saldanha Bay).

3.2.9.1 Vessel Repair Facility (VRF) at Berth 205

At present, Vessel Repair Facilities (VRFs) in Saldanha Bay are limited to minor repairs of fishing vessels, although a few offshore rigs have been repaired at Berths 203, 204 and the MPT. In order to harness opportunities that exist in the vessel repair business, dedicated and purpose-built quays with associated bulk services and onshore back of port services are required. The location study identified the site immediately to the south of Berth 204 of the MPT (referred to here as Berth 205) as the preferred location, with the alternative being to the north (ARUP 2014) (Figure 3.11). According to ARUP (2014), the southern location has several engineering and logistical advantages over the other sites considered:

- Berth 205 is adjacent to the navigation channel to the MPT and to the dredge channel to the Iron Ore Expansion berth, which will keep dredging to a minimum.
- The location is within the Port security boundary simplifying access.
- In the event of the market failing to materialise, the facility could be incorporated into the MPT or could serve as an additional bulk export facility.

Possible disadvantages are as follows:

- Future expansion would be prevented if the Iron Ore Expansion Project were to proceed, although it would be possible to expand into the MPT.

- Vessels under repair could be impacted by vessels travelling to and from the MPT.
- High airborne dust concentrations at this site may damage vessels unless regularly washed down.

3.2.9.2 Mossgas Jetty

In 2009, a study was undertaken to identify the options and costs for the extension of the Mossgas yard in order to provide a 500-metre-long quay to form an offshore vessel repair facility (ZLH 2009). More recently, a pre-feasibility study reported an increasing demand for semi-submersibles, Floating Production Storage Offload Vessels (FPSOs) and jack-up platforms (ARUP 2016). This sparked the proposal of a complimentary offshore supply vessel repair facility adjacent to Mossgas Quay.

The pre-feasibility study considered three possible locations for the jetty (Figure 3.11):

- The eastern side of Mossgas Quay (preferred site)
- The western side of Mossgas Quay (alternative site)
- At the existing Mossgas Quay (not feasible)

The existing Mossgas Quay option was eliminated due to current port operations and existing lease agreements. The western side of the Mossgas Quay was not preferred due to cost limitations and the current location of the marina. As sediment transportation adjacent to Mossgas is predominantly from west to east, more frequent maintenance dredging and a longer groyne would be necessary if the jetty is constructed to the west (ARUP 2016). A jetty positioned to the east is preferable to developers as costs are projected to be lower, while activity will be further away from designated aquaculture areas and the Bluewater Bay residential area (Figure 3.11).

3.2.9.3 Floating dry dock for the inspection of Offshore Supply Vessels

A floating dry dock is essentially a semi-submersible vessel that can adjust its ballasting to increase its draft to allow a vessel to manoeuvre into the main dock barrel. The floating dry dock is then de-ballasted to raise the vessel out of the water. The floating dry dock may be manoeuvred into deeper water to service larger vessels, therefore reducing the depth of dredging required at the ship maintenance site.

3.2.9.4 Marine Environmental Impact Assessment

The proposed impact sites are already moderately disturbed by shipping, pollution (including iron ore dust) and maintenance dredging. Despite these existing impacts and pressures, Small Bay should not be regarded solely as an industrial port. This area still provides valuable goods and services to the Saldanha Bay-Langebaan Lagoon system as a whole and is essential for the healthy functioning of the area.

Anchor Environmental Consultants (Pty) Ltd. were appointed by CCA Environmental (Pty) Ltd. (CCA) to conduct a marine environmental screening study for the construction of the VRF at Berth 205 and a 500 m long jetty in the vicinity of the existing Mossgas Quay in the Port of Saldanha (Laird and Clark 2016).

The study found that based on data reviewed from the Saldanha State of the Bay Report (Anchor 2015) and from hydrological and sediment modelling (ZAA 2016), impacts from construction at the 'preferred' and 'alternative' sites are unlikely to differ within a development option (i.e. Mossgas Jetty east no different from Mossgas Jetty west and VRF north no different from VRF south) when viewed from a marine environmental perspective. In contrast, differences in the severity of some impacts are expected between the two projects (i.e. between Mossgas and the VRF at Berth 205).

For example, despite the fact that the proposed construction footprint at the Mossgas Jetty is 150% smaller than that at Berth 205, impacts were rated higher at the Mossgas Jetty due to the ecological importance of the intertidal and shallow subtidal area in the northern part of Small Bay and the relative scarcity of this habitat. Planned annual maintenance dredging at the Mossgas Jetty also elevated significance ratings by increasing the impact duration from short/medium-term to long-term. The shallow intertidal beach area in the northern section of Small Bay is crucially important for fish recruitment. If construction of the Mossgas Jetty is approved, up to 15% of the total nursery area in Small Bay will be lost. Although fish can potentially utilise similar habitat west of the proposed jetty, it is not clear whether this area will be sufficient to sustain increased densities of juvenile fish during a prosperous recruitment year. With the intention of preventing collapse of commercially important fish stocks such as white stumpnose (which are already declining in the Saldanha Bay-Langebaan Lagoon system), it is recommended that no further net loss of shallow intertidal beach habitat in Small Bay should be permitted after the completion of the Mossgas Jetty.

Other impacts that are considered as important include turbidity plumes created by dredging. The effects of increased Total Suspended Solids (TSS) in the water column during dredging can have severe impacts on the marine environment through the mobilisation of fine sediments, contaminants, nutrients and increased turbidity (Refer to Section 3.3 for more information). ZAA reported on the likely severity of an increased concentration of TSS at the dredge sites based on a settling rate of 0.45 mm/s (ZAA 2016). Due to the combination of mud and fine calcrete dust (which creates extensive white plumes when removed) known to be present in Small Bay, previous modelling studies applied settling rates of 0.1 and 0.2 mm/s for very fine (< 2 µm) and fine material respectively (Anderson 2008). The substantially higher settling rate applied for the Berth 205 and Mossgas project is likely to result in an underestimation of the extent of the turbidity plume. Although modelled dredge volume was elevated to anticipated 'worst case scenario' by ZAA, the settling rate may not have been conservative enough considering the presence of the calcrete layer between 3 and 17 m in subsurface marine substrata in the construction footprint (ARUP 2014 and 2016). Although deep sediments are unlikely to contain toxic levels of trace metals, excess fine sediments will intensify the impacts of smothering and increased turbidity. The study by Anchor Environmental therefore recommended that the sediment particle size included in the model is revised to take the estimated dredge volume of calcrete into account. For the construction phase, standard mitigation measures (i.e. real-time monitoring and installation of a silt curtain) for minimising the impact of turbidity plumes were recommended.

Figure 3.11. **The iron ore terminal (IOT), the multi-purpose terminal (MPT), the Dry Bulk Terminal (DBT) and the Liquid Bulk Terminal (LBT) separating Big Bay and Small Bay. The preferred (green) and alternative (orange) position of the Berth 205 VRF and the preferred (yellow) and alternative (blue) options for the proposed Mossgas Jetty are indicated (Adapted from: ARUP 2016).**

3.3 Export of metal ores from the Port of Saldanha

Metal ores exported from the Port of Saldanha Bay include iron, lead, copper, zinc, and manganese. Most of the iron ore is exported from the iron ore terminal (IOT) (Figure 3.12), while more recently a very small proportion has been exported from the *Multi-Purpose Terminal* (MPT) (Figure 3.13). The Port of Saldanha currently has the capacity to export up to 60 million tonnes of iron ore per year but is in the process of upgrading the infrastructure to support an annual export of 80 million tonnes (Section 3.2.2). Iron ore exports increased steadily from 20.7 to 53.7 million tonnes between 2003 and 2013, after which the rate on increase slowed and values fluctuated around 55 million, peaking at 57.2 million tonnes in 2019, but dropped again to 56.8 million tonnes in 2020 (Figure 3.12, note that annual metal export is calculated based on the fiscal year, i.e. April-March).

South Africa accounts for approximately 78% of the world's identified manganese resources, with Ukraine accounting for 10%, in second place. South Africa's manganese production increased from 4.2 million tonnes in 2004 to 13.7 million tonnes in 2016. Most of the locally produced manganese is exported (Chamber of Mines 2017). Manganese exports started in the 2013/14 fiscal year with a mass of 95 thousand tonnes exported (roughly 17.3% of the total MPT exports for that year), export masses increased substantially (by more than one third of the previous year) until 2017/18 after which they stabilised, averaging roughly 4300 thousand tonnes in the past 3 years and comprising roughly 70% of the total exports from the MPT (Figure 3.13).

Lead, copper and zinc metal exports from the MPT increased steadily from 2007/8, peaking in 2012/13 before stabilising at an average of roughly 138.1 million tonnes between 2013/14 and 2018/19. In the 2019/20 fiscal year the Lead and copper exports appear similar to previous years, however, the mass of Zinc exported has increased dramatically to roughly 4.5 times that of previous years (Figure 3.14 – note this may be an artifact of variability in the format of data provided). Initially only lead, copper and zinc were exported from the MPT, with lead and zinc exported in similar quantities, and copper the smallest proportion of the exported material until in 2019/20 when zinc export proportion increased dramatically (Figure 3.14). The export of combined lead, copper and zinc increased from 74 thousand tonnes in 2007/8 to 183 thousand tonnes in March 2013 and has since fluctuated around 141 thousand tonnes (Figure 3.14). Individual annual export volumes for lead, copper and zinc are only available since 2010/11 (Figure 3.14). Lead exports remained stable around 80 thousand tonnes between 2010 and 2013 before dropping by nearly half in 2014-2016. Lead exports have since recovered to approximately 60 thousand tonnes per annum. Zinc exports picked up in 2011, roughly equalling lead exports with an average of 60 thousand tonnes per annum between 2013/14 and 2018/19 before increasing dramatically by the end of Mach 2020 (Figure 3.14). Copper is exported in small quantities compared to other metal ores with exports averaging 21.6 thousand tonnes in the last 5 years. Since then zinc exports have averaged around 22 thousand tonnes. In 2011, Transnet started the export of iron from the MTP. Up until 2016, iron ore comprised on average 58% of the total exports from the MPT, although thereafter the MPT has been primarily used for Manganese exports (Figure 3.13).

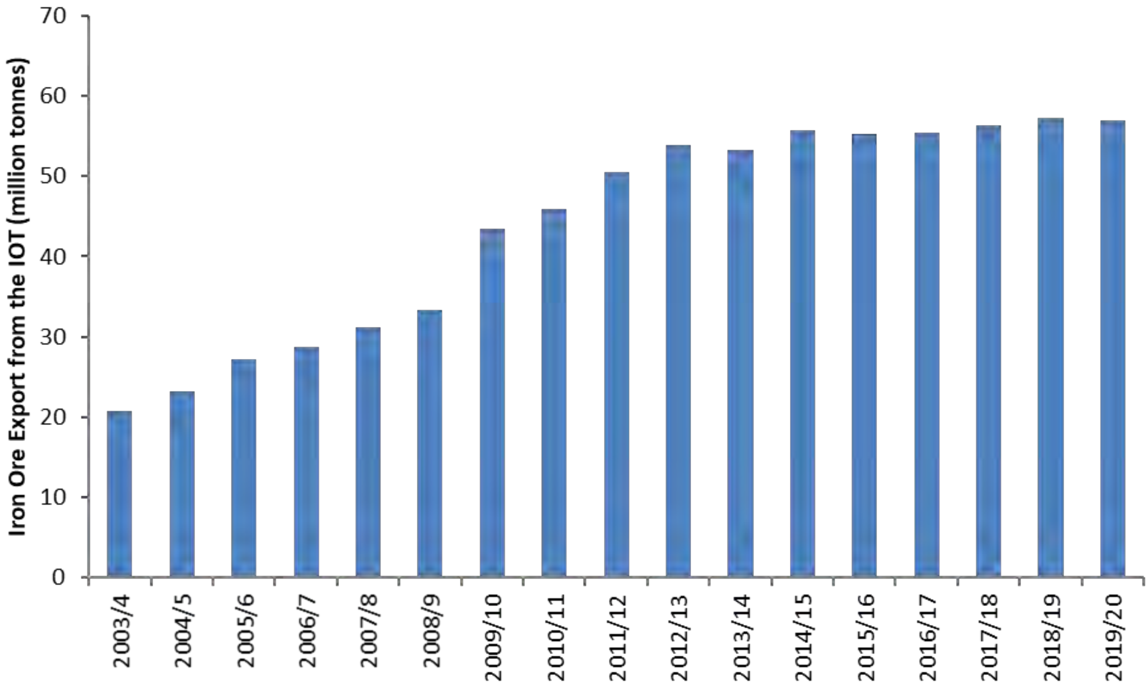

Figure 3.12. Annual exports of iron ore from the iron ore terminal at the Port of Saldanha between April 2003 and March 2020. (Data provided by Transnet National Ports Authority 2020).

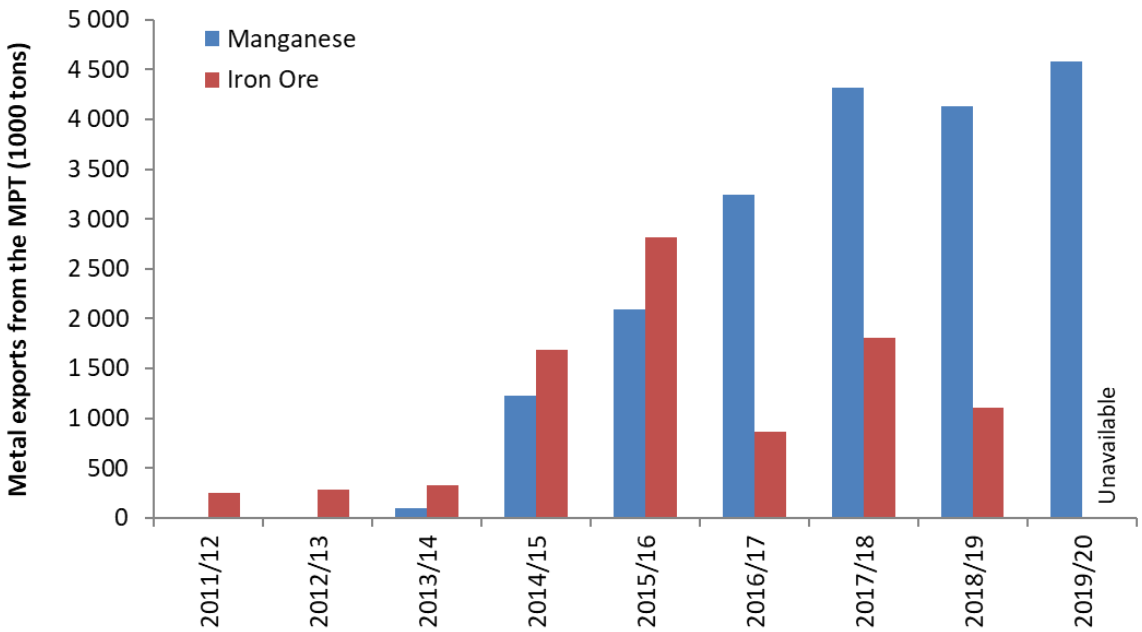

Figure 3.13. Annual exports (April 2011 – March 2020) of manganese and iron ore from the *multi-purpose terminal* at the Port of Saldanha Bay (Data provided by Transnet National Ports Authority 2020).

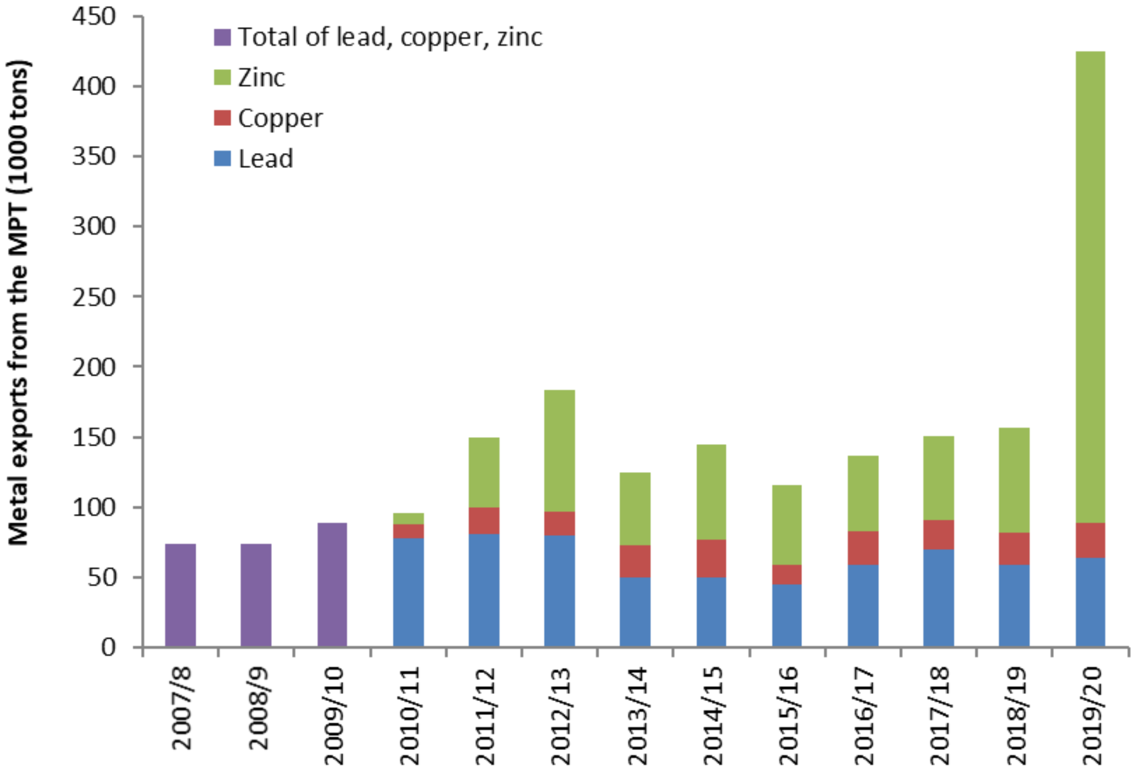

Figure 3.14. Annual exports (April 2007 – March 2020) of lead, copper and zinc from the Multi-Purpose Terminal at the Port of Saldanha Bay. Note that separate data for these commodities was only available for April 2010-March 2020 and due to changes in data format 2019/20 data is for the entire port (Data provided by Transnet National Ports Authority 2020).

3.3.1 Air quality management in Saldanha Bay

Suspended particles in the atmosphere eventually settle and result in pollution of the marine environment of Saldanha Bay and Langebaan Lagoon (direct settlement and stormwater runoff). Chemical processes in the water column facilitate the uptake of metals into the tissue of mariculture organisms destined for human consumption. Effective air quality management in Saldanha Bay is therefore considered an important component of water quality management in the study area.

The West Coast District Municipality acknowledged and accepted its responsibility in terms of Chapter 5 of the National Environmental Management: Air Quality Act, 2004 (Act 39 of 2004) (NEM: AQA) and fulfils the function of licensing authority in the area of jurisdiction of the West Coast District. Since the promulgation of NEM: AQA on 01 April 2010 the majority of atmospheric emission licences were issued within the Saldanha Bay Municipality.

Listing notice GN No. 893 of 22 November 2013 (as amended) published in terms of section 21 of NEM: AQA identifies certain categories of activities requiring an atmospheric emission licence and which must be compliant with minimum emission standards in terms of Part 3 of the Regulations. The storing, processing and handling of minerals is listed as a Category 5 activity and includes the storage of handling of ore and coal not situated on the premises of a mine or works as defined in the Mines Health and Safety Act 29 of 1996 (Subcategory 5.1). Licensing is, however, only required if the location is designed to hold more than 100 000 tonnes.

The main atmospheric emissions originate from the Iron Ore Terminal and the TPT currently holds a license for the storage and handling of 60 million tonnes of iron ore per annum which was issued on 5 February 2016. In line with the planned expansions of the iron ore export business, the TPT submitted an application for a variation to the existing AEL to increase the throughput from 60 to 67 million tonnes on 12 June 2018. As part of this application, TPT was required to submit an Air Quality Assessment Report (dated February 2018) and to conduct a public participation process. The application was denied by the competent authority on 12 September 2018 for a number of reasons. Most importantly, the impact assessment report demonstrated that during the monitoring period, National Dust Control Regulations for residential and non-residential fallout dust rates of 600 and 1200 mg/m^2 per day respectively were exceeded. It was concluded that cumulative impacts going forward would be unacceptable considering the current impact of dust emissions. Furthermore, a total of approximately 400 complaints relating to property staining and 11 complaints regarding spillages were lodged between 2016 and 2018.

A Provisional Air Emissions Licence (PAEL) for the storage and handling of ore and coal, specifically Manganese (MN), at the Multi-Purpose Terminal (MPT) was issued by the air quality officer of the Department of Environmental Affairs on 26 September 2018 (Reference: AEL/WCP/TPT/26/06/2018-2387). The air quality impact assessment for the MPT conducted by WSP in December 2017 indicated that the annual average and 99th percentile of PM_{10} (coarse particles smaller than 10 micrometres in diameter) and PM_{25} concentrations associated with the storage of manganese were well below the relevant National Ambient Air Quality Standard in Saldanha Bay. However, the study also found that annual average manganese concentrations are predicted to exceed the annual World Health Organisation manganese guidelines at Bluewater Bay and the Saldanha Caravan Park, with annual average concentrations remaining below the guideline for other sites in the Bay. The PAEL was appealed in November 2018 by 15 appellants, the main concern being the "Harmful and health effects of manganese to people, water, aqua farms, tourism and businesses including the efforts to develop Saldanha Bay as a Green City and that a EIA should have been conducted'. In her appeal decision, dated January 2020 (Reference LSA 177442), the Minister of Environment, Forestry and Fisheries (Ms BD Creecy) decided that the appeal should be upheld and therefore the PAEL for the storage of Manganese ore should be set aside. Ms Creecy indicated that the activities required for the storage of Manganese in the port, specifically the need for the expansion of current storage facilities, should have triggered an EIA prior to the issuing of the PAEL. Given this, the quantity of Manganese being stored at the MPT has been significantly reduced.

The establishment of several small operations not requiring an Atmospheric Emissions License in the Saldanha Bay Municipality resulted in significant cumulative impacts on air quality. Users of the bay and regulating authorities raised concerns, including but not limited to the uncovered transportation of materials through residential areas by rail or road.

To protect the consumers of mariculture organisms and the industry itself, the transportation, storage, handling and exporting of ore (more specifically, manganese ore) were investigated and discussed with role players in July 2016 at the Greater Saldanha Bay (GSB) Intergovernmental Task Team (IGTT).

It was concluded that a guideline document be compiled in fulfilment of duty of care obligations specified in NEMA section 28.

The draft guideline document requires that all operators storing and handling ore below the 100 000-tonne threshold should inform authorities of the (i) transport mode (ii) frequency of incoming ore/coal and how much, (iii) average offloading frequency and (iv) storage capacities per month. The operator should also inform the authorities of increases in handling capacities or relevant infrastructural changes. The guideline further specifies that transportation, loading and offloading, storage and further distribution of ores, coal, concentrates and other dusty materials must be done in such a manner to avoid the spread of particulate matter:

- **Transportation:** Material transported by rail or road must be suitably covered to prevent the spread of windblown dust. The use of alternative methods to effectively contain material whilst in transit may be considered, on condition that the transporter provides documentation confirming that the alternative method ensures reliable and equivalent containment of the material to prevent windblown dust. In many instances existing transport corridors i.e. railway lines run through residential developments with the effect that the environment and human health and wellbeing are impacted on. The transportation of material through these corridors must be discouraged and if unavoidable, more stringent conditions such as containerisation should be considered. A suitably designed road vehicle washing facility to effectively remove particulate matter from wheels, wheel arches, mud flaps and undercarriages must be provided on the storage and handling site. Effluent from washing facility must be drained to a sump for re-use or safe disposal;
- **Storage:** Manganese and other potentially hazardous ores, and concentrates must be stored within an enclosed building on a hard, impervious surface graded and drained to a sump from where the effluent will be re-used or safely disposed of;
- **Handling:** Loading and offloading of materials can also be a significant source of dust emissions. Materials can be reclaimed by underfeed conveyor, grab crane or front-end loader with totally enclosed conveyors used to transport dust-forming material. Transfer by pneumatic, dense phase systems may also be used. The loading and offloading of material must as far as practically possible be done inside the enclosed storage facility. In instances where this is not practically possible, material must be offloaded into containers or onto trucks for direct transportation into the enclosed storage facility. The double handling of material must be avoided. The storage of potentially hazardous material (concentrates e.g. manganese and zinc) in open air stockpiles is not allowed. Approved dust suppression methods that result in zero visible emissions must be applied and the area used for this purpose must be provided with a suitably drained, hard and impervious surface such as concrete. Material spillages must be removed immediately and contained for re-use or safe disposal. Emergency spillage incidents must be reported to the relevant authorities in terms of section 30 of the National Environmental Management Act, 1998 (Act 107 of 1998). Excess contaminated water used for dust suppression must be drained to a sump from where it is collected for re-use or safe disposal.

The guideline also requires that dust fallout monitoring be conducted at the storage and handling location, the transport corridor, as well as within residential areas that are in close proximity to the transport corridor. Dust monitoring must be conducted as prescribed in the National Dust Control Regulations No. R. 827 of 1 November 2013 (as amended).

The draft guideline was presented on 5 April 2017 and stakeholders were given until the 18th April 2017 to provide written comment. The WCDM intends to promulgate the guideline as a policy document under Section 30 of the WCDM Bylaw. The WCDM will be the competent authority once the guideline has been promulgated as a policy. The adoption and successful implementation of this guideline document will hopefully reduce metal contamination of the Saldanha Bay and Langebaan Lagoon marine environment with a positive impact on the existing and future mariculture sector. The 2nd generation West Coast District Municipality air quality management plan was published in June 2019 and is available on their website.

3.4 Dredging and port expansion

Dredging of the seabed is performed worldwide in order to expand and deepen existing harbours/ports or to maintain navigation channels and harbour entrances (Erftemeijer & Lewis 2006) and has thus been touted as one of the most common anthropogenic disturbance of the marine environment (Bonvicini Pagliai *et al.* 1985). The potential impacts of dredging on the marine environment can stem from both the removal of substratum from the seafloor and the disposal of dredged sediments, and include:

- Direct destruction of benthic fauna populations due to substrate removal;
- Burial of organisms due to disposal of dredged sediments;
- Alterations in sediment composition which changes nature and diversity of benthic communities (e.g. decline in species density, abundance and biomass);
- Enhanced sedimentation;
- Changes in bathymetry which alters current velocities and wave action; and
- Increase in concentration of suspended matter and turbidity due to suspension of sediments. The re-suspension of sediments may give rise to:
    - Decrease in water transparency
    - Release in nutrients and hence eutrophication
    - Release of toxic metals and hydrocarbons due to changes in physical/chemical equilibria
    - Decrease in oxygen concentrations in the water column
    - Bioaccumulation of toxic pollutants
    - Transport of fine sediments to adjacent areas, and hence transport of pollutants
    - Decreased primary production due to decreased light penetration to water column

Aside from dredging itself, dredged material may be suspended during transport to the surface, overflow from barges or leaking pipelines, during transport to dump sites and during disposal of dredged material (Jensen & Mogensen 2000 in Erftemeijer & Lewis 2006).

Saldanha Bay is South Africa's largest and deepest natural port and as a result has undergone extensive harbour development and has been subjected to several bouts of dredging and marine blasting as listed below (refer to AEC 2014 for more detailed information on the dredging events):

- 1974-1976: 25 million m^3 of sediment was dredged during the establishment of the ore terminal;

- 1996-1997: 2 million m^3 of sediment was removed for the expansion of the multi-purpose terminal;
- 2005-2007: 380 000 m^3 sediment removed from Big Bay for the nourishment of Langebaan Beach
- 2007-2008: 50 000 m^3 of sediment was removed for maintenance of the Mossgas quay and multi-purpose terminal; and
- 2009-2010: 7300 m^3 of sediment was removed to allow for the establishment of a new ore-loading berth.
- 2009-2010: Maintenance dredging (unknown quantity) conducted by the South African National Defence Force (SANDF) at the Salamander Bay boatyard.
- 2015-2016: 25 000 m^3 Expansion of the General Maintenance Quay
- 2019: 14 265 m^3 of sediment was removed for maintenance of the OSSB quay
- 2019-2020: 6403 m^3 of sediment was removed for maintenance of the Mossgas channel
- 2020: 13 433 m^3 of sediment was removed for maintenance of the OSSB channel.

The most recent dredging occurred in September and October 2019 and February 2020, 20 668 m^3 of sediment was dredged for the upkeep and maintenance of the OSSB quay and channel and the Mossgas channel.

3.5 Shipping, ballast water discharges, and oil spills

Shipping traffic comes with a number of associated risks, especially in a port environment, where the risks of collisions and breakdowns increase owing to the fact that shipping traffic is concentrated, vessels are required to perform difficult manoeuvres, and are required to discharge or take up ballast water in lieu of cargo that has been loaded or unloaded. Saldanha Bay is home to the Port of Saldanha, which is one of the largest ports in South Africa receiving more than 500 ships per annum. The Port is comprised of an iron ore terminal for export of iron ore, an oil terminal for import of crude oil, a multi-purpose terminal dedicated mostly for export of lead, copper, zinc and manganese concentrates, and the Sea Harvest/Cold Store terminal that is dedicated to frozen fish products (Figure 3.4). There are also facilities for small vessels within the Port of Saldanha including the Government jetty used mostly by fishing vessels, the Transnet-NPA small boat harbour used mainly for the berthing and maintenance of Transnet-NPA workboats and tugs, and the Mossgas quay. Discharge of ballast by vessels visiting the iron ore terminal in particular poses a significant risk to the health of Saldanha Bay and Langebaan Lagoon.

3.5.1 Shipping and ballast water

Ships carrying ballast water have been recorded since the late nineteenth century and by the 1950s had completely phased out the older practice of carrying dry ballast. Ballast is essential for the efficient handling and stability of ships during ocean crossings and when entering a port. Ballast water is either freshwater or seawater taken up at ports of departure and discharged on arrival where new water can be pumped aboard, the volume dependant on the cargo load. The conversion to ballast water caused a new wave of marine invasions, as species with a larval or planktonic phase in their life cycle were now able to be transported long distances between ports on board ships. Furthermore, because ballast water is usually loaded in shallow and often turbid port areas, sediment is also loaded along with the water and this can support a host of infaunal species (Hewitt *et al.* 2009). The global nature of the shipping industry makes it inevitable that many ships must load ballast water in one area and discharge it in another, which has an increasing potential to transport non-indigenous species to new areas. It has been estimated that major cargo vessels annually transport nearly 10 billion tonnes of ballast water worldwide, indicating the global dimension of the problem (Gollasch *et al.* 2002). It is estimated that on average, 3 000-4 000 species are transported between continents by ships each day (Carlton & Geller 1993). Once released into ports, these non-indigenous species have the potential to establish in a new environment which is potentially free of predators, parasites and diseases, and thereby out compete and impact on native species and ecosystem functions, fishing and aquaculture industries, as well as public health (Gollasch *et al.* 2002). Invasive species include planktonic dinoflagellates and copepods, nektonic Scyphozoa, Ctenophora, Mysidacea, benthos such as annelid oligochaeta and polychaeta, crustacean brachyura and molluscan bivalves, and fish (Carlton & Geller 1993). Carlton & Geller (1993) record 45 'invasions' attributable to ballast water discharges in various coastal states around the world. In view of the recorded negative effects of alien species transfers, the International Maritime Organisation (IMO) considers the introduction of harmful aquatic organisms and pathogens to new environments via ships ballast water as one of the four greatest threats to the world’s oceans (Awad *et al.* 2003). In addition, it has been emphasised that the link between marine alien species and harbours (just over half of the alien species in South African marine waters are located in harbours) highlights the role of shipping as a method by which these alien species are introduced (Robinson *et al*. 2020).

A recent update on the number of alien marine species present in South Africa lists 95 alien species as being present in this country, of which 56 species have spread outside of their original point of introduction and are considered invasive i.e. population are expanding and are consequently displacing indigenous species (Robinson *et al*. 2020). The West Coast of South Africa is the most invaded region with 67 recorded alien species. The presence of five new alien species – the barnacle *Perforatus*, the Japanese skeleton shrimp *Caprella mutica*, the North West African porcelain crab *Porcellana africana*, the Chilean stone crab *Homalaspis plana* and the South American sunstar *Heliaster helianthus* – have been confirmed in Saldanha Bay and Langebaan Lagoon since 2014. With these recent additions, the list of alien species present in Saldanha Bay and/or Langebaan Lagoon, is updated to a total of 28. All of these except three are considered to be invasive. It should be noted that *P. africana* was previously misidentified as the European porcelain crab, *P. platycheles*. Other noteworthy invasive alien species that are present in Saldanha Bay include the Mediterranean mussel *Mytilus galloprovincialis*, the barnacle *Balanus glandula*, the Pacific mussel *Semimytilus algosus* and the *Western pea crab Pinnixa occidentalis.*

Recently, Peters *et al.* (2014) established that the brachiopod *Discinisca tenuis,* previously only known to occur in aquaculture facilities, has spread into the port of Saldanha and on the leeward side of Schaapen Island (Peters *et al.* 2014). Most of the introduced species are found in sheltered areas such as harbours and because ballast water is normally loaded in sheltered harbours, the species that are transported also originate from these habitats and thus have a difficult time adapting to South Africa’s exposed coast. This might, in part, explain the low number of introduced species that have become invasive along the coast (Griffiths *et al.* 2008). Most introduced species in South Africa occur along the west and south coasts, very few having been recorded east of Port Elizabeth. This corresponds with the predominant trade routes being between South Africa and the cooler temperate regions of Europe, from where most of the marine introductions in South Africa originate (Awad *et al.* 2003). More detail on alien invasive species in Saldanha Bay is provided in Chapter 13.

Other potential negative effects of ballast water discharges are contaminants that may be transported with the water. Carter (1996) reported on concentrations of trace metals such as cadmium, copper, zinc and lead amongst others that have been detected in ballast water and ballast tank sediments from ships deballasting in Saldanha Bay. All parameters measured in 1996 exceeded the current South African Water Quality Guidelines for the Marine Environment (DEA 2018) (Table 3.2.). These discharges are almost certainly contributing to trace metal loading in the water column and are indicated by their concentration in filter-feeding organisms in the Bay (refer to Chapter 6).

Table 3.2. **Mean trace metal concentrations in ballast water (µg/l) and ballast tank sediments from ships deballasting in Saldanha Bay (Source: Carter 1996) and SA Water Quality Guideline limits (DEA 2018). Those measurements in red denote exceedance of these guidelines.**

	Water (µg/L)	SA WQ Guideline limit (µg/L)	Sediment	ERL Guideline (mg/kg)
Cd	5	0.12	0.040	1.2
Cu	5	3	0.057	34
Zn	130	20	0.800	150
Pb	15	2	0.003	46.7
Cr	25	2	0.056	-
Ni	10	5	0.160	20.9

To address the above environmental impacts and risks, the International Convention for the Control and Management of Ship's Ballast Water and Sediments of 2004 (BWM Convention) was ratified by 30 states representing 35% of the world merchant shipping tonnage (IMO 2015). The BWM Convention provides for standards and procedures for the management and control of ballast water and sediments carried by ships, which are aimed at preventing the spread of harmful aquatic organisms from one region to another.

Under the BWM Convention all vessels travelling in international waters must manage their ballast water and sediment in accordance with a ship-specific ballast water management plan. It is required that every ship maintains a ballast water record book and holds an international ballast water management certificate. Ballast water management standards and treatment technology are slowly being implemented, but in the interim ships are required to exchange ballast water mid-ocean. Parties to the BWM Convention are given the option to take additional measures to those described above and which are subject to criteria set out in the BWM Convention and to the guidelines that have been developed to facilitate implementation of the Convention.

South Africa ratified to this Convention, but it took almost a decade until the Draft Ballast Water Management Bill was published in the *Government Gazette* in April 2013 (Notice 340 of 2013) aimed to implement the BWM Convention. The public comments period for the bill was extended in February 2017 (Notice 111 of 2017), however the Draft Bill has not yet been promulgated. The

Department of Transport is the authority responsible for administration of this Act. Detailed information on the Draft Bill can be found in previous versions of the State of Saldanha Bay and Langebaan Lagoon report (AEC 2018).

A study examining the status of ballast water management in South African ports discovered that there was a lack of publicly accessible documentation which detailed the requirements for ballast water management in numerous South African ports with the exception of Saldanha Bay (Calitz 2012). This documentation was prepared for the Port of Saldanha during a 2002 pilot study called GloBallast (Global Ballast Water Management Programme, Calitz 2012).

In the absence of domestic legislation regulating ballast water discharge, the Transnet National Port Authority in Saldanha Bay implements the following measures to control the release of alien species into the harbour:

Procedure to follow when granting permission for international vessels to enter the Port of Saldanha:

1. The agent shall request, 72 hours in advance, permission for de-ballasting operations.
2. The TNPA Pollution Officer or the Marine Safety Specialist shall grant or declined permission after scrutinizing the Ballast Water Reporting Form, Ship Particulars & Port of Call list.
3. The TNPA must confirm the ballast water intake location.
4. The Pollution Officer shall board the vessel and check the relevant documentation and seal all overboard valves with a unique TNPA seal.
5. TNPA may board the vessel and check the running hours of the ballast water pump against the ballast water logbook should there be any concern regarding the ballast water of the vessel.
6. Should the vessel not comply with the Harbour Master's written Instructions or the IMO requirements, the TNPA shall request the Captain of the vessel to comply before permission is granted to conduct de-ballasting operations at the Port of Saldanha.

Ballast water carried by ships visiting the Port of Saldanha is released in two stages - a first release is made upon entering Saldanha Bay (i.e. Big Bay) and the second once the ship is berthed and loading (Awad *et al.* 2003). As a result, as much as 50% of the ballast water is released in the vicinity of the iron ore quay on either the Small Bay side or Big Bay side of the quay depending on which side the ship is berthed.

The total number of ships entering the Port of Saldanha nearly doubled between 1994 and 2011 from 261 to 487 vessels, after which ship numbers remained fairly constant until 2017/18 when numbers increased by 25% from 474 to 591 vessels per annum (Figure 3.15). While vessel numbers in 2018/19 increased further to 616, the total number of ships for the 2019/20 year dropped slightly to 571 vessels, it is possible that this decline in numbers is as a result of the global COVID 19 pandemic. Overall, iron ore tankers contributed 51% to the observed vessel traffic in 2019/20 and 91% to the total water discharged between July 2018 and June 2019 (specific vessel discharge volumes are not available for 2019/20, see Figure 3.15 and Figure 3.16). Iron ore tankers are large vessels and hold the highest quantities of ballast water.

Average vessel size increased over the years (Figure 3.17) and as a result, the volume of ballast water discharged annually almost tripled between 1994/5 and 2010/11 from 8.2 to 21.1 million tonnes

(Figure 3.16). Since 2011, ballast water discharge has remained fairly stable averaging around 23 million tonnes per annum, peaking in 2017/18 (25.1 million tonnes) and then declining in the two subsequent years (Figure 3.16). Vessels docking at the Iron ore terminal have a higher average volume of ballast water discharge than other vessel types, with volumes increasing from 54.4 thousand tonnes per vessel in 2003/4, peaking at 78.6 thousand tonnes in 2015/16 and dropping to 71.2 thousand tonnes in 2018/19 (Figure 3.17). While discharge volumes for Tankers fluctuates irregularly ranging between zero and 21.8 thousand tonnes, vessels docking at the Multipurpose Terminal showed low average ballast discharge volumes between 2003/4 and 2010/11 (less than three thousand tonnes) before increasing in size until 2016/2017 (12.6 thousand tonnes) and dropped in the two subsequent years to 9.6 thousand tonnes in 2018/19 (Figure 3.17). Only total ballast water volumes for the entire port were available for 2019/20 however, when comparing the average discharge for all vessels combined in 2019/20 to similar data for the period 1994/5 to 2001/2, we see that volumes have increased by more than one third of historic volumes (Figure 3.17).

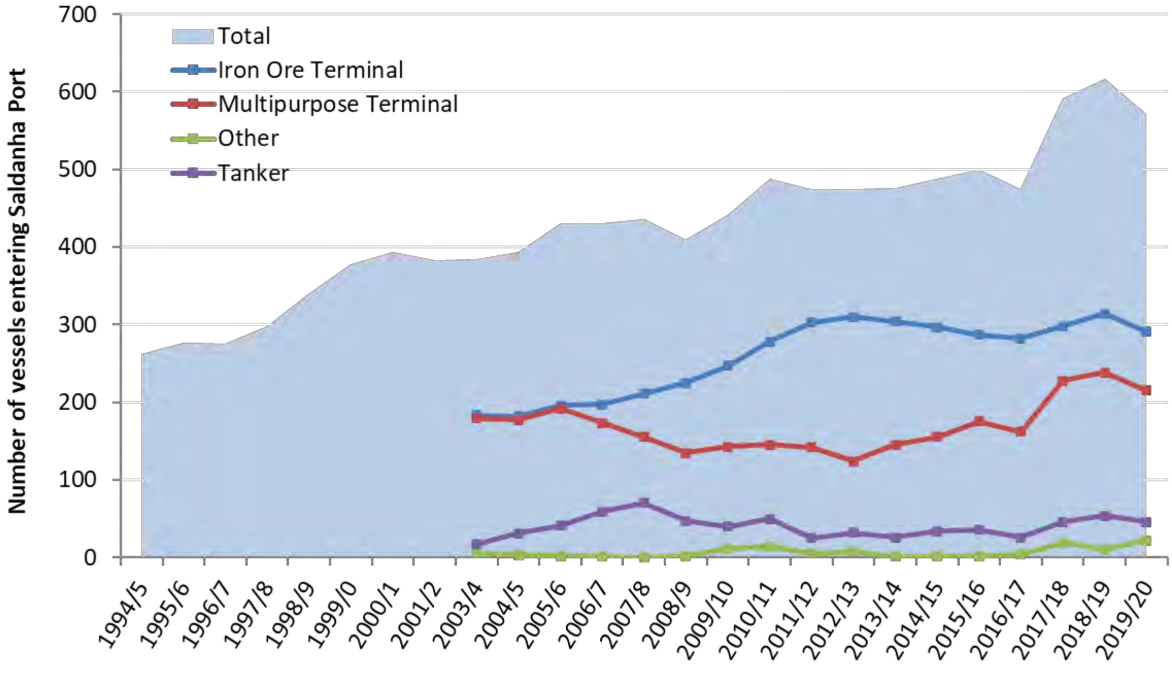

Figure 3.15. The numbers and types of vessels entering Saldanha Port per year. The total number of vessels entering Saldanha Port between July 1994 and June 2020 is shown as the blue area. The numbers of vessels docking at the iron ore terminal, the *multi-purpose terminal*, tankers and other vessels are shown in blue, red, green and purple respectively. Data for the different types of vessels is only available from 2003 onward (Sources: Marangoni 1998, Awad *et al.* 2003, Transnet-NPA unpublished data 2003-2020).

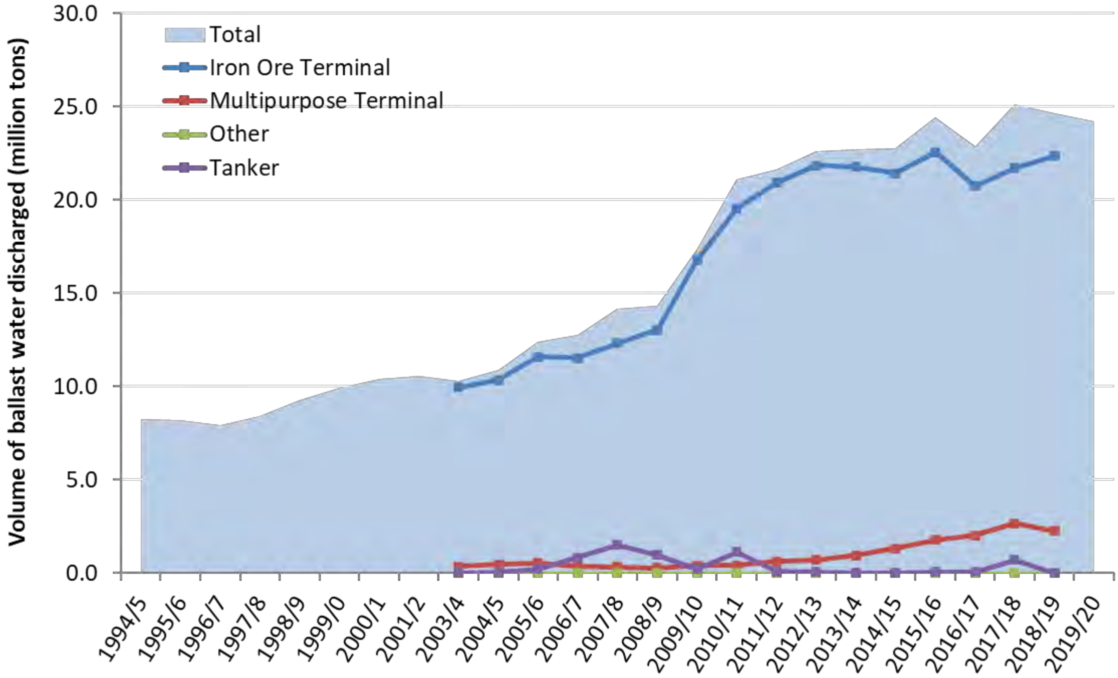

Figure 3.16. **Volumes of ballast water discharged into Saldanha Port per year. The total amount of ballast water discharged in Saldanha Port between the years 1994 and June 2019 as well as for 2019/20 is shown as the blue area. Ballast water discharged by vessels docking at the iron ore terminal, the *multi-purpose terminal*, tankers and other vessels are shown in blue, red, green and purple respectively. Data for the different types of vessels is only available from 2003 to June 2019 (Sources: Marangoni 1998, Awad *et al.* 2003, Transnet-NPA unpublished data 2003-2020).**

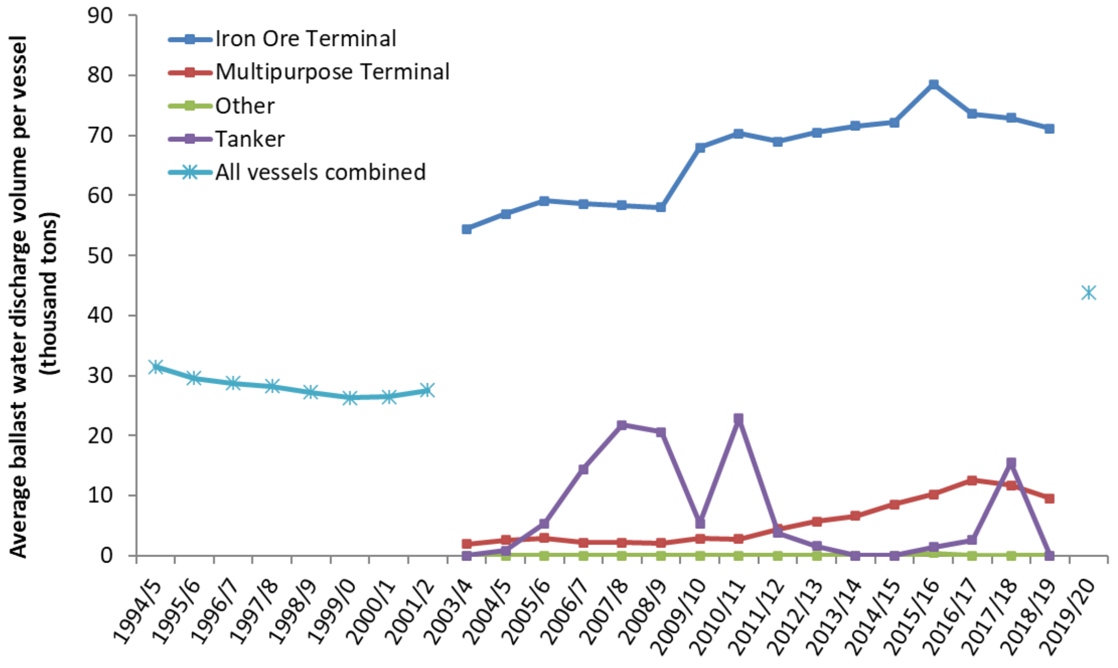

Figure 3.17. **Average ballast water volumes discharged per vessel into Saldanha Port per year. The total amount of ballast water discharged in Saldanha Port between the years 1994 and June 2002, and for 2019/20 is shown as the blue line. Ballast water discharged by vessels docking at the iron ore terminal, the *multi-purpose terminal*, tankers and other vessels are shown in blue, red, green and purple respectively. Data for the different types of vessels is only available from 2003 to 2019, (Sources: Marangoni 1998, Awad *et al.* 2003, Transnet-NPA unpublished data 2003-2020).**

3.5.2 Oil spills

Also associated with this increase in shipping traffic, is an increase in the incidence and risk of oil spills. In South Africa there have been a total of five major oil spills, two off Cape Town (1983 and 2000), one in the vicinity of Dassen Island (1994), one close to the St. Lucia estuary in KwaZulu-Natal (2002) and one in the Goukamma Nature Reserve (2013). No comparable oil spills have occurred in Saldanha Bay to date (SAMSA, Martin Slabber *pers. comm*.). Minor spills do occur however, which have the potential to severely impact the surrounding environment. In April 2002, about 10 tonnes of oil spilled into the sea in Saldanha Bay when a relief valve malfunctioned on a super-tanker. Booms were immediately placed around the tanker and the spill was contained. More recently in July 2007, a Sea Harvest ship spilled oil into the harbour while re-fuelling, the spill was managed but left oil on rocks and probably affected small invertebrates living on the rocks and in the surrounding sand.

In 2007 Transnet National Ports Authority and Oil Pollution Control South Africa (OPC), a subsidiary of CEF (Central Energy Fund) signed an agreement which substantially improved procedures in the event of oil spills and put in place measures to effectively help prevent spills in the Port of Saldanha. These are laid out in detail in the “Port of Saldanha oil spill contingency plan” (Transnet National Ports Authority 2007). The plan is intended to ensure a rapid response to oil spills within the port itself and by approaching vessels. The plan interfaces with the “National oil spill contingency plan” and with the “Terminal oil spill contingency plan” and has a three-tiered response to oils spills:

Tier 1: Spill of less than approximately 7 tonnes
Response where the containment, clean up and rescue of contaminated fauna can be dealt with within the boundaries of the vessel, berth or a small geographical area. The incident has no impact outside the operational area but poses a potential emergency condition.

Tier 2: Spill between 7-300 tonnes
Response where the nature of the incident puts it beyond the containment, clean up and rescue of contaminated fauna capabilities of the ship or terminal operator. The containment of clean up requires the use of some of or the government and industry resources.

Tier 3: Spill in excess of 300 tonnes.
Response where the nature of the incident puts it beyond containment, clean up and rescue of contaminated fauna capabilities of a national or regional response. This is a large spill which has the probability of causing severe environmental and human health problems.

Upon entry to the port, all vessels undergo an inspection by the Pollution Control Officer to minimise risks of pollution in the port through checking overboard valves and ensuring the master and crew of the vessel are familiar with the Port’s environmental requirements. Every tanker is contained by booms while oil is being pumped. Immediate containment of any minor spills is thereby ensured (SAMSA, Martin Sabber, *pers. comm*.). The OPC has facilities and equipment to effectively secure an oil spill as well as for the handling of shore contamination including oiled sea birds and beach-cleaning equipment. However, given the environmental sensitivity of the Saldanha Bay area, particularly Langebaan Lagoon, prevention is the most important focus (CEF 2008). The implementation of Floating Power Plants (FPPs) (Section 3.2.5) will increase the risk of oil spills (frequency and magnitude) unless the Environmental Management Programme contains effective mitigation measures and implementation is ensured.

Pollution incident at government jetty slipway reported by Sea Harvest

In early July of 2020 is was reported that a pollution incident had occurred on the Government jetty slipway. According to reports, high tidal intrusion caused an over-full bin (located adjacent to the ship repair facility) to be knocked over. As a result, the contents of the bin which included pieces of sponge and domestic waste were observed floating in the adjacent water. In addition, an oily-fatty solution coated the slipway, and was seen running down into the water of the Bay and forming an oily slick on the water surface towards the quay side of the Sea Harvest Operations plant. Although the harbour master sent out a small boat to retrieve the sponges and domestic waste, the presence of the oily slick on the water surface reduces water quality and can prevent the abstraction of water from the Bay for use in the Sea Harvest Desalination plant.

Incidents such as this are not uncommon and poor operational management practices at the ship repair facility (based on the slipway next to Sea Harvest) have previously been reported. The facility appears to lack appropriate structures or procedures to contain waste and other materials (fouling material, paint, oil etc.) generated during ship maintenance operations. This is of great concern as continuous or repetitive pollution input at the slipway could threaten the quality of the oysters and mussels in adjacent aquaculture farms as well as having negative impacts for the Sea Harvest desalination plant. Therefore, it is strongly suggested that this matter be investigated, and that the facility is better managed in future. Potential mitigation measures include: 1) the retention and storage of oily/contaminated run-off from the vessels on the slipway in steel drums in an appropriate storage facility with retaining bund walls and 2) the installation of separate bins for domestic waste located at a suitable distance from the water to prevent tidal disruption and with lids to prevent the distribution of waste via wind.

3.5.3 Noise

A variety of noises are produced in the coastal underwater world, including short and high intensity sounds that are generated by underwater construction activities (for example pile driving) (Popper & Hastings 2009) as well as noise produced by shipping vessels which is characterised in wide spread and prolonged low frequency noise (Slabberkorn *et al.* in press).

Impacts of noises in the coastal environment on fish behaviour and physiology have received a good deal of attention in recent years. For example, Bregman (1990) described the 'auditory scene' of fishes which provides information from great distances or information at night for navigation, predator avoidance and prey detection. Consequences of a disturbance in the 'auditory scene' of fishes have been shown in captive three-spined sticklebacks (*Gasterosteus aculeatusl*) (Purser & Radford 2011). Foraging efficiency was significantly reduced when subjected to brief as well as prolonged noise, as more time was spent on attacking their prey due to a shift in attention. Several published studies have demonstrated the importance of sound in predator avoidance and prey detection (Knudsen *et al.* 1997, Konings 2001). Reproductive efficiency can also be affected as more than 800 fish species are known to produce sounds when spawning (Aalbers 2008) and during courtship (McKibben & Bass 1998). It has been suggested that entire fish assemblages in very noisy environments might be impacted by noise through reduced reproductive efficiency, thereby affecting number of individuals. For example, roach (*Rutilus rutilus*) and rudd (*Scardinius erythrophthalmus*) showed an interruption of spawning in the presence of noise produced by speed boats (Boussard 1981). Impacts of sound waves on fish physiology were investigated in controlled experiments where pile driving was lethal to some fish species (Caltrans 2001) but not for others (Abbot *et al.* 2005). The examination of dead and fatally injured fish revealed damaged and bleeding swim bladders (Caltrans 2001).

It appears that not all fish species respond to noise in the same way (Voellmy *et al.* 2014) and current research is insufficient to successfully predict the effects of noise on fish in the marine environment. It is recommended that a precautionary approach be adopted and that impacts of sound, especially future construction of infrastructure in the Port of Saldanha are mitigated. An air bubble curtain around piling operations is commonly cited as an effective mitigation measure to reduce the sound transmission (Abbott & Bing-Sawyer, 2002, Bellmann & Remmers 2013). Producing bubbles around the noise source prevents transmission of sound due to the reflection and absorption of sound waves (Würsig *et al.* 2000).

3.6 Effluent discharges into the Bay

Contemporary coastal water management strategies around the world focus on maintaining or achieving receiving water quality such that the water body remains or becomes fit for other designated uses. Designated uses of the marine environment include aquaculture, recreational use, industrial use, as well as the protection of biodiversity and ecosystem functioning. This goal oriented management approach arose from the recognition that enforcing end of the pipe effluent limits in the absence of an established context, i.e. not recognising the assimilative capacity and requirements of receiving environments, would reach a point where water bodies would only be marginally fit for their recognised uses. This management approach is referred to as the receiving water quality (RWQ) framework (AEC 2015) and it appears that most countries have adopted this framework and have developed water quality guidelines for a variety of uses, which include target values for a range of contaminants that must be met in the receiving environment. Furthermore, in most countries water quality guidelines are legislated standards and are thus a legal requirement to be met by every user/outfall. Although the importance of managing water quality through the RWQ framework is undisputed, the degree to which this is implemented differs widely between countries.

There are a wide variety of legal instruments that are utilised by countries to maintain and/or achieve water quality guidelines in the receiving environment. These include setting appropriate contaminant limits, the banning or restricting of certain types of discharges in specified areas, prohibiting or restricting discharge of certain substances, as well as providing financial incentives to reduce pollution at the source alongside the implementation of cleaner treatment technology. The only effective method however, that ensures compliance of an effluent with water quality guidelines/standards is to determine site-specific effluent limits which are calculated based on the water quality guidelines/standards of a given water body, the effluent volume and concentration, as well as the site-specific assimilative capacity of the receiving environment. This method is also identified as the water quality-based effluent limits (WQBEL) approach (AEC 2015) and recognises that effluent (and its associated contaminants) is rapidly diluted by the receiving waters as it enters the environment. In order to take advantage of this beneficial effect, allowance is generally made for a "mixing zone" which extends a short distance from the outfall point (or pipe end) and is an area in which contaminant levels are "allowed" to exceed the established water quality standards (or guidelines) for the receiving environment. The magnitude of the "mixing zone" should, in theory, vary in accordance with the sensitivity and significance of the receiving environment and the location of the outfall point in the environment, but in practice is usually set at a distance of around 100 m from the pipe end for marine systems. The WQBEL approach differs from the Uniform Effluent Standard (UES) approach in which fixed maximum concentrations or loads are applicable for contaminants in wastewater discharges for all users or outfalls, irrespective of where they are located (AEC 2015).

3.6.1 Legislative context for pollution control in South Africa

South Africa has adopted the RWQ framework for the management of water quality in both inland (freshwater) and marine water bodies and uses both, the WQBEL and the UES approaches to implement the framework. Receiving water quality guidelines were thus published in 1995 for the full range of beneficial uses for inland water (human consumption, aquaculture, irrigation, recreational use, industrial use, and protection of biodiversity and ecosystem functioning) and also for the marine environment (natural environment, recreational use, industrial use and mariculture). Revised Water Quality Guidelines for the Natural Environment and Mariculture Use were recently published by the DEA: O&C (DEA 2018), replacing Volumes 1 (Natural Environment) 4 (Mariculture) of the 1995 Guidelines.

The 2018 Water Quality Guidelines for Coastal Marine Waters contain narrative statements and guideline values along with relevant background information (e.g. description, source, fate in the environment, occurrence in South African marine waters etc.) for seawater properties (temperature, salinity, dissolved oxygen etc.) and constituents (nutrients, toxic substances, pathogens).

In the case of Saldanha Bay, which is extremely important for biodiversity conservation (there are several Marine Protected Areas (MPAs) in the Bay), is also an important regional centre for aquaculture (mussels, oysters, finfish), is important for recreation (swimming, kite surfing, windsurfing, etc.), and an area from where water is abstracted for industrial purposes (cooling water and desalination), the most stringent receiving environment water quality guidelines should be applicable (see Chapter 6 for more details on this).

Effluent discharges into the coastal waters were previously regulated in terms of the National Water Act (Act No 36 of 1998) (NWA). The NWA categorised the discharging of waste or water containing waste into a "water resource through a sea outfall or other conduit" as a "water use" for which a "licence" was required, unless such use was authorised through a "general authorisation" indicated by a notice published in the *Government Gazette*.

With the promulgation of the National Environmental Management: Integrated Coastal Management Act (No. 24 of 2008) (ICMA) (as amended[3]), responsibility for regulating land-derived effluent discharges into coastal waters was transferred to the Department of Environmental Affairs (DEA). In terms of Section 69 of ICMA, no person is permitted to discharge effluent originating from a source on land into coastal waters except in terms of a General Discharge Authorisation (GDA) or a Coastal

[3] ICMA was amended by the National Environmental Management: Integrated Coastal Management Amendment Act, 2014 (Act No. 36 of 2014) (ICMAA).

Waters Discharge Permit (CWDP). Exemptions were issued to proponents who, at the time of promulgation, were discharging effluent into coastal waters in terms of permits issued under the NWA, provided that the effluent was treated to meet the *General and Special Standard* (Government Gazette No. 20526, 8 October 1999[4]), and required that they applied for a CWDP within three years of promulgation of the ICMA. In practice though, not all operations that discharge wastewater into the Bay have applied for a CWDPs even though many years have elapsed since the promulgation of the ICMA. New operators wishing to discharge effluent to coastal waters are required to apply for a CWDP before commencing and are also required to comply with the applicable water quality guidelines for the receiving environment. Applications for CWDP are expected to include data on contaminant levels in the effluent to be discharged, as well as results of dilution and dispersion model studies indicated maximum expected levels for the same contaminants at the edge of the defined mixing zone. These levels are of course expected to comply with published guideline levels as defined by other existing, or potential, beneficial uses of the receiving environment.

The DEA is currently in the process of implementing a permitting system for such effluent discharges. The Assessment Framework for the Management of Effluent from Land Based Sources Discharged to the Marine Environment (AEC 2015) provided a road map for the development of regulations for the permitting system. This framework recognises that discharges differ in effluent characteristics (volume and quality) and discharge locality (i.e. biophysical conditions, use of the receiving environment), which ultimately determines the risk a discharge poses to the receiving environment. It was recommended that the potential scope of a General Discharge Authorisation, the level of assessment during the application process for a CWDP, as well as licensing conditions should be based entirely on the environmental risk posed by an effluent. Accordingly, the guidelines provide a framework within which an effluent can be characterised (effluent components and properties) and its potential impacts be assessed within the context of the receiving environment (i.e. sensitive versus robust receiving environments).

In March 2019 the DEA:O&C published the Coastal Waters Discharge Permit Regulations (GNR. 382, *Government Gazette* 42304). The new regulations seek to provide an administrative framework to implement Section 69 of the ICMA and stipulate timeframes, renewal application processes, applicable fees and information to be submitted as part of an application for a CWDP. The DEA:O&C are still in the process of finalising regulations for General Discharge Authorisations discussed above.

To date, seven CWDPs have been issued to companies discharging effluent into Saldanha Bay and two applications are currently pending. A list of these and other relevant information has been included in Table 3.3.

[4] The latest revision of the General Authorisation was promulgated on 6 September 2013 (Government Gazette No. 36820).

Table 3.3. Pending applications for Coastal Waters Discharge Permit and issued permits for effluent discharges into Saldanha Bay (Source: Department of Environment, Forestry and Fishories Branch: Oceans and Coasts).

Applicant/permit holder	Status	Type of discharge	Impact level	Compliance
OTMS Mogs Saldanha	Permit granted	Hydrostatic testing	Low	N/A
ArcelorMittal Saldanha Steel	Permit granted	Reverse Osmosis	Low	Quarterly monitoring
Sea Harvest Corporation (Pty) Ltd	Permit granted	Fish processing effluent and brine	Medium	Quarterly monitoring
Sunrise Energy (Pty) Ltd	Permit granted	Once off discharge	Low	Monitoring occurred after discharge
Transnet State Owned Company (SOC) Ltd	Permit granted	Desalination (brine)	Medium	Quarterly monitoring
Saldanha Oyster	Permit granted	Holding facility	Low	N/A
Oceana Lobster Saldanha	Decision pending	Unknown (processing/holding facility?)	Unknown	N/A
Transnet Port Terminals	Permit granted	Industrial Storm Water	Medium	Quarterly monitoring

Table 3.4. General Limit as specified in the revised general limit for general authorisation (6 September 2013) under the National Water Act (No. 36 of 1998)

Substance/parameter	General limit as specified in the Revision of General Authorisations in terms of Section 39 of the National Water Act (Government Gazette No. 36820, 6 September 2013)
Temperature	-
Faecal Coliforms (per 100 ml)	1000
Electrical Conductivity measured in milliSiemens per meter (mS/m)	70 above intake to a maximum of 150*
pH	5.5-9.5
Chemical oxygen demand (mg/L)	75 (after removal of algae)
Suspended Solids (mg/L)	25
Soap, oil or grease (mg/L)	2.5
Ortho-Phosphate as P (mg/L)	10
Nitrate/Nitrite as Nitrogen (mg/L)	15
Ammonia (ionised and un-ionised) as N (mg/L)	6
Fluoride (mg/L)	1
Chlorine as Free Chlorine (mg/L)	0.25
Dissolved Cyanide (mg/L)	0.02
Dissolved Arsenic (mg/L)	0.02
Dissolved Cadmium(mg/L)	0.005
Dissolved Chromium (VI) (mg/L)	0.05
Dissolved Copper (mg/L)	0.01
Dissolved Iron (mg/L)	0.3
Dissolved Lead (mg/L)	0.01
Dissolved Manganese (mg/L)	0.1
Mercury and its compounds (mg/L)	0.005
Dissolved Selenium (mg/L)	0.02
Dissolved Zinc (mg/L)	0.1
Boron (mg/L)	1
Phenolic compounds as phenol (mg/L)	-

*Electrical conductivity is only applicable to wastewater discharges into freshwater.

3.6.2 Reverse osmosis plants

Reverse Osmosis is used to re-claim potable water from fresh, brackish or saline water. Desalination specifically refers to a water treatment process whereby salts are removed from saline water to produce fresh water. Reverse Osmosis involves forcing water through a semi-permeable membrane under high pressure, leaving the dissolved salts and other solutes behind on the surface of the membrane. Water is relatively scarce in the West Coast District Municipality (WCDM) and the rapidly developing industry in Saldanha Bay requires vast quantities of potable water for their operations. Construction of reverse osmosis desalination plants has been identified as a potential solution to reduce dependency of industry on municipal water supplies.

RO plants can have severe impacts on the receiving marine environment if potable water is reclaimed from seawater due to the highly saline and negatively buoyant brine water that is discharged by these plants, which often contains biocides that serve to limit marine growth in their intake pipe work. Potential environmental impacts associated with the operation of RO plants are listed below:

- Altered flows at the discharge resulting in ecological impacts (*e.g*. flow distortion/changes at the discharge, and effects on natural sediment dynamics);
- The effect of elevated salinities in the brine water discharged to the bay;
- Biocidal action of non-oxidising biocides such as dibromonitrilopropionamide in the effluent;
- The effects of co-discharged wastewater constituents, including possible tainting effects affecting both mariculture activities and fish factory processing in the bay;
- The effect of the discharged effluent having a higher temperature than the receiving environment;
- Direct changes in dissolved oxygen content due to the difference between the ambient dissolved oxygen concentrations and those in the discharged effluent; and
- Indirect changes in dissolved oxygen content of the water column and sediments due to changes in phytoplankton production as a result of altered nutrient dynamics (both in terms of changes in nutrient inflows and vertical mixing of nutrients) and altered remineralisation rates (with related changes in nutrient concentrations in near bottom waters) associated with near bottom changes in seawater temperature due to the brine discharge plume.

3.6.2.1 Transnet NPA Desalination Plant

Transnet NPA recently built a RO plant in Saldanha Bay to produce freshwater for dust mitigation during the loading and offloading of iron ore. The RO plant has been operational since obtaining a water use license from the DWA and subsequent performance tests in 2012 (Membrane Technology 2013) (refer to AEC 2014 for more details on the project design and EIA). The RO plant was recently granted a CWDP in terms of ICMA (DEA: O&C, *pers. comm.*, 2017).

A marine baseline monitoring study was conducted by Anchor Environmental Consultants prior to the commissioning of the RO plant to ensure that impacts in the marine environment are such that the beneficial uses of the potentially impacted area are considered (Hutchings and Clark 2011). Monitoring of the physical and chemical characteristics of the receiving environment were also conducted during the period June 2010 to March 2011 in order to establish a baseline prior to the RO plant coming into operation (van Ballegooyen *et al.* 2012).

The monitoring requirements as specified by the Water Use License and the Record of Decision issued by the Department of Environmental Affairs for the RO plant (these are also reflected in the Transnet Specification No. 1243487-SP-0001) were as follows:

(a) Monthly monitoring of temperature, salinity, dissolved oxygen, turbidity, concentrations in the brine basin;
(b) Continuous (hourly) monitoring of temperature, salinity, dissolved oxygen, and turbidity at representative outfall monitoring station and a reference station for at least 1 year; and
(c) Surveys of trace metals and benthic macrofauna to be conducted bi-annually for an unspecified period.

The monitoring of the marine environment in fulfilment of the Environmental Monitoring Programme was being conducted by the Council for Scientific and Industrial Research (CSIR) (Refer to the 2016 State of Saldanha Bay and Langebaan Lagoon Report for details on the methods and results of the first two surveys conducted in 2014 and 2015) but this has since passed on to Cellozyme Environmental in 2018.

3.6.2.2 West Coast District Municipality Desalination Plant

The West Coast District Municipality (WCDM) has proposed the construction of an additional RO plant in the Saldanha Bay area, intended as a long-term sustainable alternative water source. The WCDM has limited water resources (semi-arid climate) and yet is required to supply 22 towns and 876 farms across the region with potable water. Currently water is supplied by the Voëlvlei and Misverstand dams on the Berg River, and the Langebaan road aquifer, however, the volume allocated from these sources for this is close to the maximum possible. In the financial year 2012/2013, abstractions for the WCDM exceeded allocation by 3.6 million m^3 (DWA 2013). A feasibility study conducted in 2007 to assess the most viable solution to the water scarcity issue in the WCDM identified the following potential additional water resources:

- The Twenty-four Rivers Scheme
- Lowlift pumps at the Misverstand Dam
- The Michel's pass Diversion
- Groundwater potential
- Water Quality Management
- Alien vegetation clearing

The most cost-effective solution was identified as a 25 500 m^3/day sea water desalination plant. EA was granted on 13 August 2013 for the preferred location for the RO plant, which will be situated on the farm Klipdrift at Danger Bay on a portion of municipal owned land (Please refer to the 2013/2014 State of Saldanha Bay and Langebaan Lagoon Report for SOB report more information).

Subsequent costs estimates suggest, however, that the proposed desalination plant and bulk infrastructure will cost R500 million, which is more than double the initial estimated cost. As a result, funding is currently a major challenge for the WCDM as they did not receive Grant funding for the construction of the desalination plant and therefore the project has been put on hold (SBM, David Wright, *pers. comm.* 2020). Should funds become available, construction of this RO plant is planned to be executed in three phases, with an initial capacity of 8.5 million litres later building up to a final capacity of 25.5 million litres. Alternatively, an application for additional allocation of water sourced from the Berg River was submitted by the WCDM. In October 2017, SBM received an increased allocation of 20 427 000m³ per annum from the WCWSS which equates to 56 ML/day and in higher than the current water demand. This in conjunction with the Langebaan Road Well field and Hopefield Well field development has reduced the need for the Desalination plant at this stage however SMB with retain the project in their future augmentation projects planning (SBM, David Wright, *pers. comm.* 2020).

3.6.2.3 ArcelorMittal RO plant

ArcelorMittal was a largely export-focussed steel plant, producing high quality ultra-thin Hot Rolled Coil (UTHRC) and located close to the deep-sea port of Saldanha. It was announced in November 2019 that due to financial losses the plant would be closed, with the immediate winding down of operations and ultimate closure of the plant in the first quarter of 2020. Prior to this, ArcelorMittal Saldanha operations required approximately 6 500 m^3/day of freshwater to operate, which represented approximately 25% of Saldanha Bay municipality's potable water total usage. ArcelorMittal Saldanha modified its water treatment infrastructure to partially replace its fresh water supply with treated municipal sewage wastewater (from the Saldanha Wastewater Treatment Works (WWTW). Please refer to AEC 2019 for details on the initiation of the RO plant).

Since the closure of the ArcelorMittal Saldanha Works, all operations have stopped which include the operations of the RO plant and therefore no industrial effluent is sent to the Saldanha WWTW and no treated effluent is being pumped from Saldanha WWTW to ArcelorMittal for use in the RO Plant.

3.6.3 Sewage and associated wastewaters

3.6.3.1 Environmental impacts

Sewage is by far the most important waste product discharged into rivers, estuaries and coastal waters worldwide. However, sewage is not the only organic constituent of wastewater, received by sewage treatment plants, other degradable organic wastes, which can result in nutrient loading, include:

- Agricultural waste
- Food processing wastes (e.g. from fish factories and slaughterhouses)
- Brewing and distillery wastes
- Paper pulp mill wastes
- Chemical industry wastes
- Oil spillages

Our present knowledge of the impacts of wastewaters on water systems has, until recently, largely been based on lake-river eutrophication studies. However, recent focus on how anthropogenic nutrient enrichment is affecting near-shore coastal ecosystems is emerging (for a review see Cloern

2001, Howarth *et al.* 2011). In general, the primarily organic discharge in wastewater effluents contains high concentrations of nutrients such as nitrates and phosphates (essentially the ingredients in fertilizers). Existing records provide compelling evidence of a rapid increase in the availability of nitrogen and phosphorus to coastal ecosystems since the mid-1950s (Cloern 2001). These nutrients stimulate the growth and primary production of fast-growing algae such as phytoplankton and ephemeral macroalgae, at the expense of slower-growing vascular plants and perennial macroalgae (seagrasses) which are better adapted to low-nutrient environments. This process requires oxygen, and with high nutrient inputs, oxygen concentrations in the water can become reduced which can lead to deoxygenation or hypoxia in the receiving water (Cloern 2001).

When phytoplankton die and settle to the bottom, aerobic and anaerobic bacteria continue the process of degradation. However, if the supply rate of organic material continues for an extended period, sediments can become depleted of oxygen leaving only anaerobic bacteria to process the organic matter. This then generates chemical by-products such as hydrogen sulphide and methane, which are toxic to most marine organisms (Clark 1986). The sediments and the benthic communities they support are thus amongst the most sensitive components of coastal ecosystems to hypoxia and eutrophication (Cloern 2001). The ecological responses associated with decreasing oxygen saturation in shallow coastal systems include the initial escape of sensitive demersal fish, followed by mortality of bivalves and crustaceans, and finally mortality of other molluscs, with extreme loss of benthic diversity (Vaquer-Sunyer & Duarte 2008, Howarth *et al.* 2011). Vaquer-Sunyer & Duarte (2008) propose a precautionary limit for oxygen concentrations at 4.6 mg O_2/litre equivalent to the 90th percentile of mean lethal concentrations, to avoid catastrophic mortality events, except for the most sensitive crab species, and effectively conserve marine biodiversity.

Some of the indirect consequences of an increase in phytoplankton biomass and high levels of nutrient loading are a decrease in water transparency and an increase in epiphyte growth, both of which have been shown to limit the habitat of benthic plants such as seagrasses (Orth & Moore 1983). Furthermore, there are several studies documenting the effects that shifts in natural marine concentrations and ratios of nitrates, phosphates and elements such ammonia and silica, have on marine organisms (Herman *et al.* 1996, van Katwijk *et al.* 1997, Hodgkiss & Ho 1997, Howarth *et al.* 2011). For instance, the depletion of dissolved Silica in coastal systems, as a result of nutrient enrichment, water management and the building of dams, is believed to be linked to worldwide increases in flagellate/ dinoflagellate species which are associated with harmful algal blooms, and are toxic to other biota (Hodgkiss & Ho 1997, Howarth *et al.* 2011). The toxic effect that elevated concentrations of ammonia have on plants has been documented for *Zostera marina* and shows that plants held for two weeks in concentrations as low as 125 µmol start to become necrotic and die (van Katwijk *et al.* 1997).

The effects of organic enrichment, on benthic macrofauna in Saldanha Bay, have been well documented (Jackson & McGibbon 1991, Stenton-Dozey *et al.* 2001, Kruger 2002, Kruger *et al.* 2005). Tourism and mariculture are both important growth industries in and around Saldanha Bay, and both are dependent on good water quality (Jackson & McGibbon 1991). The growth of attached algae such as *Ulva sp.* and *Enteromorpha sp.* on beaches is a common sign of sewage pollution (Clark 1986). Nitrogen loading in Langebaan Lagoon associated with leakage of conservancy/septic tanks and storm water runoff has resulted in localised blooms of *Ulva sp*. in the past. In the summer 1993-94, a bloom of *Ulva lactuca* in Saldanha Bay was linked to discharge of nitrogen from pelagic fish processing plants

(Monteiro *et al.* 1997). Dense patches of *Ulva sp.* are also occasionally found in the shallow embayment of Oudepos (CSIR 2002). Organic loading is a particular problem in Small Bay due to reduced wave action and water movement in this part of the Bay caused by harbour structures such as the iron ore terminal and the Causeway, as well as the multitude of organic pollution sources within this area (e.g. fish factories, mariculture farms, sewage outfalls, sewage overflow from pump stations, and storm water runoff). Langebaan Lagoon is also sheltered from wave action, but strong tidal action and the shallow nature of the lagoon make it less susceptible to the long-term deposition of pollutants and organic matter (Monteiro & Largier 1999).

Treatment of effluent is pivotal in reducing the environmental impacts described above. However, the side effects of treating effluent with chlorine have been well established in the literature. Chlorine gas, generated through a process of electrolysis, is toxic to most organisms and is used to sterilise the final effluent (i.e. kill bacteria and other pathogens present in the effluent) before it is released into settling ponds or the environment. Chlorine breaks down naturally through reaction with organic matter and in the presence of sunlight but should not exceed a concentration 0.25 mg/L at the end of pipe terms of the revised General and Special Standard (Government Notice No. 36820 –6 September 2013) promulgated under the NWA (Table 3.4). Furthermore, chlorine, while disinfecting the effluent, produces a range of toxic disinfection by-products (DBPs) through its reactions with organic compounds (Richardson *et al.* 2007, la Farré *et al.* 2008, Sedlak & von Gunten, 2011).

3.6.3.2 Management of treated effluent in Saldanha Bay and Langebaan

There are two wastewater treatment works (WWTW) that produce treated effluent which used to enter the Saldanha/Langebaan marine environment, namely the Saldanha WWTW and the Langebaan WWTW. Twenty-seven sewage pump stations in Langebaan are situated throughout the town, many of which are near the edge of the lagoon and 16 sewage pump stations are located in Saldanha Bay (Figure 3.18). To prevent raw sewage being released directly into Saldanha Bay due to malfunction or during power failures, mechanical and electrical equipment upgrades to the pump stations in Saldanha and Langebaan were undertaken in 2012 and implementation of upgrades will continue as and when required. Fifteen million Rand was made available on the 2016-2017 Capital Budget for the implementation of various interventions that prevent overflow of raw sewage that were completed in 2017 (SBM, Gavin Williams, *pers. comm.* 2016) (Figure 3.19). It is hoped that all these interventions will prevent future spills such as the one experienced in September 2016 (Refer to 2016 State of Saldanha Bay and Langebaan Lagoon Report).

Figure 3.18. **Location of wastewater treatment works, sewage pump stations and sewer pipes in the Saldanha and Langebaan area in 2014 (Source: Saldanha Bay Municipality, Elmi Pretorius 2014).**

Figure 3.19. Emergency generators that have been installed at various pump stations in Saldanha Bay and Langebaan Lagoon (Source: SBM, Gavin Williams, 2016).

There are approximately 200 conservancy tanks in Langebaan, east of Club Mykonos (SBM, Elmi Pretorius, *pers. comm.* 2014). Overflow of these tanks is considered an unlikely event today, as the municipality empties these tanks on a regular basis (SBM, Gavin Williams, *pers. comm.* 2014).

Details on the two WWTW are provided in Sections 3.6.3.3 and 3.6.3.4, which present data on monthly trends in the effluent produced by the WWTWs. Data was provided by the SBM and water quality parameters recorded as "trace", "less than" or "greater than" was adjusted in accordance with the following standard international convention:

- "trace" = half the detection limit
- "less than" = half the detection limit
- "greater than" = detection limit multiplied by a factor of three

In the case of the Saldanha Bay WWTW, concentrations of contaminants in the effluent are compared with the General Discharge Limits of the revised General and Special Standard (Government Notice No. 36820 –6 September 2013) promulgated under the NWA (Table 3.4.).

As the global climate pattern termed El Niño Southern Oscillation[5] weakens, most of the country has been able to recover from the worst drought since 1904. The Western Cape, however, continues to struggle to meet water demands in the province. Water shortages will be a reality for many years to come, as several years of above-average rainfall conditions and continued conservative use of drinking water are required to fill the dams to pre-drought levels. As of 14 September 2020 the overall dam water levels for the Western Cape Water Supply System (WCWSS) were at 96.4%, while this is significantly higher than at the same time during the peak of the drought (roughly 37.4% in September 2017), it is still less than the >100% seen in winter of 2012-2014 despite a drop in average water use within the province of over 30% (https://www.capetown.gov.za). Additionally, long-term climate models predict that global warming will result in drier conditions in the Western Cape and it is very well possible that water shortages must be understood as the 'new normal'. Not only climate patterns must be considered in this scenario, but also the growing demand by industry, especially in the Saldanha Bay Municipality (SBM). This critical situation brought industry and local municipalities together to investigate the feasibility of re-using treated wastewater and/or reclaiming industrial grade or even potable freshwater from treated sewage by means of further treatment. Initially wastewater was supplied without further treatment to be used for dust suppression at various construction sites (total allocation of 540 m^3/day), the Blouwaterbaai Lodge (60 m^3 per day), and Saldanha Sports Grounds (300 m^3 per day).

Industry in Saldanha Bay also expressed the need for high quality recycled water and motivated for the supply of free treated wastewater by the SBM, which would then be treated by means of Reverse Osmosis to suit the needs of industry. Similar projects implemented elsewhere in South Africa demonstrated that major infrastructural changes were required for the re-cycling of treated sewage and were associated with significant initial as well as ongoing fiscal investments (Refer to AEC 2017 for more detail on the water reclamation project implemented by Veola Water Services in Durban). Local municipalities experience significant budgetary constraints, and a public-private partnership has been the key for successful implementation in Saldanha Bay. Considering the water shortage and the environmental impacts associated with the discharge of WWTW effluent, this was conceived as an attractive opportunity.

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5 El Niño is the warm phase of the El Niño Southern Oscillation (commonly called ENSO) and is associated with a band of warm ocean water that develops in the central and east-central equatorial Pacific (between approximately the International Date Line and 120°W), including off the Pacific coast of South America. El Niño Southern Oscillation refers to the cycle of warm and cold temperatures, as measured by sea surface temperature, SST, of the tropical central and eastern Pacific Ocean. El Niño is accompanied by high air pressure in the western Pacific and low air pressure in the eastern Pacific. The cool phase of ENSO is called "La Niña" with SST in the eastern Pacific below average and air pressures high in the eastern and low in western Pacific. The ENSO cycle, both El Niño and La Niña, cause global changes of both temperatures and rainfall.

3.6.3.3 Saldanha Wastewater Treatment Works

The Saldanha Bay WWTW treats raw sewage by means of activated sludge with mechanical aeration and drying beds. In addition to sewage waste, the WWTW in Saldanha also receives and treats industrial wastewater from a range of industries in Saldanha:

- Sea Harvest
- Hoedtjiesbaai Hotel
- Protea Hotel
- Bongolethu Fishing Enterprises
- SA Lobster
- Transnet Port Authority
- Arcelor Mittal (until its closure in the first quarter of 2020)
- Abattoir
- Duferco

The effective functioning of WWTW is largely dependent on the quality of contributor effluent and sewage that is directed into the plant. Local by-laws control to which extent industries must treat their effluent before it is directed into municipal wastewater treatment works. New by-laws have been put in place, which require contributors to agree on the amount and quality of effluent to be discharged into the municipal stream. Strict monitoring of effluent volumes and quality has been implemented and penalties are levied for transgression of the signed agreement (Gavin Williams *pers. comm.* 2018).

The capacity of the Saldanha Bay WWTW was increased to 5 ML to accommodate the projected increase wastewater production, especially with the establishment of the Saldanha Bay Industrial Development Zone (IDZ). Various other improvements to the plant were also implemented to ensure that the treated wastewater is of acceptable quality (refer to AEC 2017 for more details). The IDZ funded and managed this project.

The plant now requires an updated Water Use License (WUL) to ensure compliance with the NWA. Originally, the Saldanha WWTW was issued an exemption under the NWA section 21(f) and (g), provided that the effluent volume does not exceed 958 000 m^3 per year and that the water quality of the treated effluent is compliant with the General Discharge Limits of the revised General and Special Standard (Government Notice No. 36820 –6 September 2013) promulgated under the NWA (Table 3.4.). The SBM has applied for a new Water Use License for the upgrades required to accommodate the Industrial Development Zone. A decision is pending as a delineation study by DWS is still outstanding (Quintin Williams, SBM, *pers. comm.* 2020).

The WWTW in Saldanha originally disposed of all their treated effluent into the Bok River which drains into Small Bay adjacent to the Blouwaterbaai Resort and has been dry for at least the last ten years. However, in response to the serious drought that the Western Cape has been experiencing since 2014, the SBM has made the treated wastewater available for irrigation, dust suppression, water features, and industrial cooling processes.

Before 2008, the average daily volume discharged never exceeded the average daily limit of 2625 m^3, but volumes of effluent released increased steadily over time (Figure 3.20.). Between the years 2008-

2012, the Saldanha WWTW was non-compliant only during the winter months. Between January 2013 and June 2020 however, the average daily limit was exceeded 69% of the time, reaching unprecedented levels of 6317 m^3 effluent in November 2019. It is important to note though that the WWTW plant capacity was 5000 m^3 and then upgraded to 7500 m^3 in December 2019, which means that the effluent quality was not compromised despite regular exceedance of the legal limit and during this time frame the plant capacity limit was only exceeded twice. Finally, wastewater volumes treated at the Saldanha Bay WWTW decreased in 2017/2018 due to the water restrictions implemented by the SBM however these has increased with the lessoning on restriction in 2019/2020.

The annual State of the Bay Report normally reports on the amount of effluent produced and therefore discharged into the Bay. Together with the effluent volumes, the report also shows a long-term trend in effluent quality and compliance with the GA. However, SBM allocates treated effluent to multiple different users for re-use, thereby dramatically reducing the amount of effluent that is discharged into the otherwise dry Bok River. Since the closure of the ArcelorMittal Saldanha Steel Works and subsequent cessation of the RO plant operations no treated effluent is being pumped from Saldanha WWTW to ArcelorMittal, however the closure of all Steel Works operations also means that the Saldanha WWTW no longer receives any industrial effluent from ArcelorMittal. The current water users receiving treated effluent from the WWTW include: the Weskus School, Saldanha Sports Ground (Stadium and practise field) and Blue Bay Lodge (Quintin Williams, SBM, *pers. comm*. 2020). The balance of treated effluent not used is currently discharged into the Bok river and ultimately ends up in the ocean, however SMB has identified a future user for the treated effluent and an allocation has been made available to them. In addition, a flow meter has been installed at the Bok River discharge point; although, it is not known whether the discharge volume is recorded (this would likely be a requirement of the new water use license if it is issued).

The Bok River has been dry for over a decade and as a result any effluent discharged would reach the shore undiluted. However, it is noteworthy that with the new wastewater management scheme, the amount of wastewater entering the marine environment is likely to be negligible (Gavin Williams, SBM, *pers. comm*.) and that contribution to pollution would likely be insignificant. The changes implemented by the SBM are therefore significantly positive and future interpretation of water quality results must consider the volume of effluent entering the marine environment via the Bok river.

The annual State of the Bay report will continue to report on the effluent quality of the WWTW over time. This year's results in relation to historic data are shown in the graphs below.

Concentrations of faecal coliforms in the effluent from the WWTW exceeded the allowable limit of 1000 org/100 ml on 43 occasions since 2003 (21% of the time) (Figure 3.21). The frequency of non-compliance increased dramatically in 2008, although at a lower concentration (3000 org/100 ml) than previously recorded. Allowable limits for faecal coliforms in the effluent were exceeded on 26 occasions since January 2013, frequently reaching the maximum detectable limit (the maximum detectable limit = 2419 org/100ml, which is multiplied by a safety factor of three = 7257 org/100ml). Some improvement was evident for the period July 2016-June 2017, and for the 2019/2020 rolling year faecal Coliform counts did not exceed the limit once. Making the 2019/20 compliance 100% of the time, an improvement from 2018/19 where compliance was 60% (Figure 3.21).

Allowable limits for total suspended solids (TSS) of 25 mg/L have been exceeded 18% of the time since April 2003 (Figure 3.22). While compliance clearly improved between 2008 and 2014, the allowable limit has been exceeded 38% of the time since December 2014. However, it is positive to note that the TSS limit has not been exceeded since May 2019, which shows a significant improvement to previous years.

Chemical oxygen demand (COD) in filtered effluent exceeded the allowable limit of 75 mg/L 24% of the time since April 2003 (Figure 3.23). COD is commonly used to indirectly measure the amount of organic material in water. COD was highest from June-October 2008 peaking at 260 mg/L in July 2008. This trend coincided with the high faecal coliform counts in the effluent over the same period. Overall, compliance improved substantially between January 2009 and June 2017 where the allowable limit was only exceeded on ten occasions at a much lower magnitude than in 2008. However, the COD was consistently above the legal limit between November 2017 and May 2019, achieving only 17% compliance. These observations are congruent with high ammonia nitrogen, faecal coliform and free chlorine levels during the same period. It is therefore positive to note that the COD limit was only exceeded once (89 mg/L in November 2019) between June 2019 and June 2020.

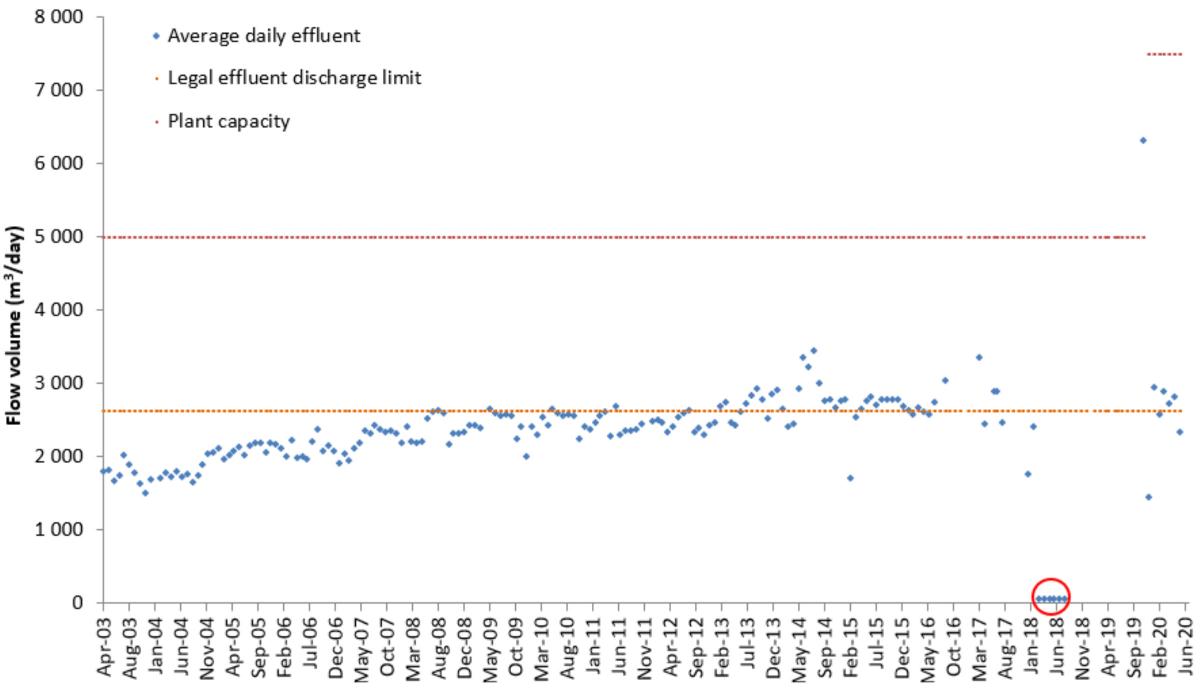


Figure 3.20. Trend in average daily effluent (m^3/month) released from the Saldanha Wastewater Treatment Works, April 2003-June 2020. Allowable discharge limits in terms of the exemption issued by DWS under the National Water Act (No. 36 of 1998) are represented by the dashed orange line and the design capacity of the plant by the red line (Source: Saldanha Bay Municipality). The data points circled in red represent the estimated effluent discharged into the Bok River (60 m^3 per day) (*pers. comm.* Gavin Williams 2018).

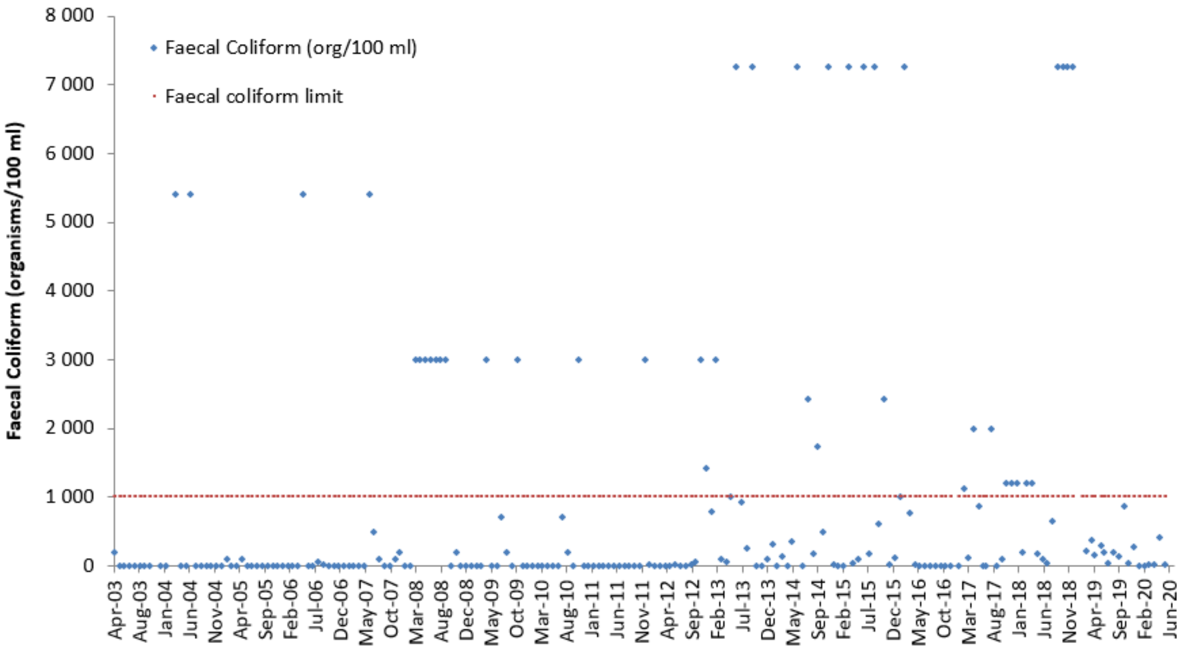


Figure 3.21. **Monthly trend in Faecal Coliforms (org/100ml) in effluent released from the Saldanha Wastewater Treatment Works, April 2003-June 2020. Allowable limits in terms of a General Authorisation under the National Water Act (No. 36 of 1998) are represented by the dashed red line (Source: Saldanha Bay Municipality).**

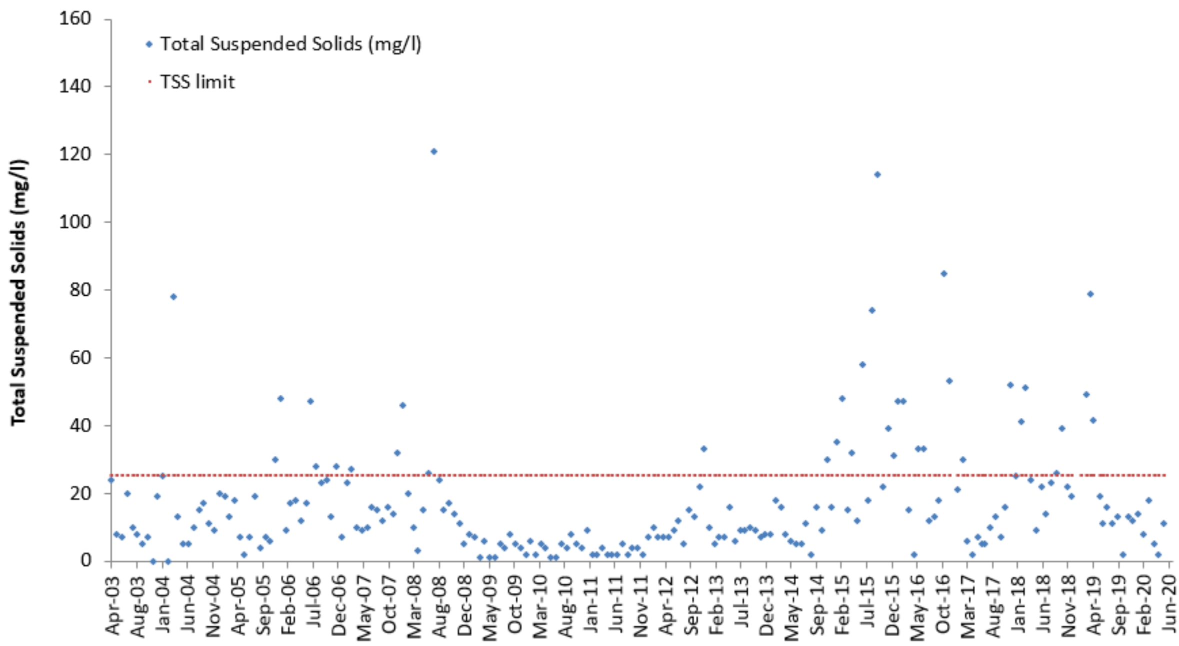


Figure 3.22. **Monthly trend in total suspended solids (mg/L) in effluent released from the Saldanha Wastewater Treatment Works, April 2003 – June 2020. Allowable limits as specified in terms of a General Authorisation under the National Water Act (No. 36 of 1998) are represented by the dashed red line (Source: Saldanha Bay Municipality).**

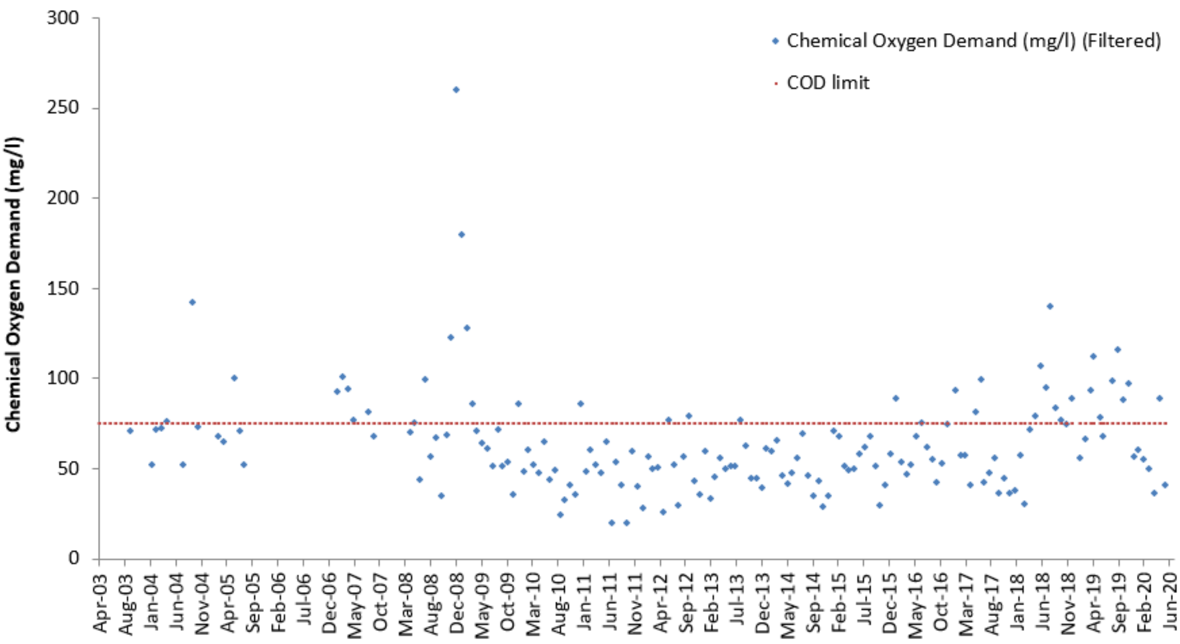


Figure 3.23. Monthly trends in chemical oxygen demand (mg/L filtered) in effluent released from the Saldanha Wastewater Treatment Works, April 2003-June 2020. Allowable limits in terms of a General Authorisation under the National Water Act (No. 36 of 1998) are represented by the dashed red line (Source: Saldanha Bay Municipality).

Levels of Ammonia-Nitrogen (mg/L as N) are of great concern in the treated wastewater of the Saldanha WWTW as readings have exceed the allowable limit of 6 mg/L, 82% of the time since April 2003 (Figure 3.24.). Ammonia levels in the effluent have only been compliant once since November 2017, measuring 91.5 mg/L in October 2018, the highest concentration ever recorded. While the average concentration during the period June 2018 to June 2019 was 58.2±28.4 mg/L, concentrations dropped in 2019/20 to an average of 10.0±3.4 mg/L. Although only very little effluent is released, ammonia is toxic to aquatic organisms and such high concentrations should not be permitted to be released into the Bok River.

The Nitrate-Nitrogen limit of 15 mg/L was exceeded 14% of time since 2003. Nitrate-Nitrogen levels have been fluctuating over time, reaching levels exceeding the legal limit in 2005, 2009/2010, 2013, and 2016/2017 (Figure 3.25). It is possible that generally higher Nitrate-Nitrogen levels in 2017 can be attributed to more effective treatment of effluent in the new aeration basins, where more Ammonia-Nitrogen is converted into non-toxic Nitrate-Nitrogen by means of bacterial treatment processes. Conversely, low nitrate nitrogen levels (<0.3 mg/L) between November 2017 and April 2019 complement extremely high levels of ammonia nitrogen indicating the lack of bacterial treatment. In 2019/20 the Nitrate-Nitrogen concentration averaged 2.5 ± 3.5 mg/L and the limit has not been exceeded since June 2017.

The concentration of orthophosphate in the effluent has only been measured since October 2007 showing a relatively low level of exceedance of 20%. Orthophosphate levels remained below the limit between July 2013 and October 2016 however, since then the limit has been exceeded on 10 occasions, 5 of which have been in the 2019/2020 rolling year. In addition, the highest reading on record to date (16 mg/L) occurred in December 2019 (Figure 3.26).

Permissible chlorine levels of 0.25 mg/L have been exceeded 61% of the time (Figure 3.27) since 2003. In 2018/2019 chlorine levels improved dramatically compared to previous years, with legal limits only exceeded on three occasions (70% compliance) and concentrations were generally low with an average of 0.25 ± 0.4 mg/L (Figure 3.27) however, 2019/2020 saw compliance drop to only 25% with an average of 0.72 ± 0.7 mg/L.

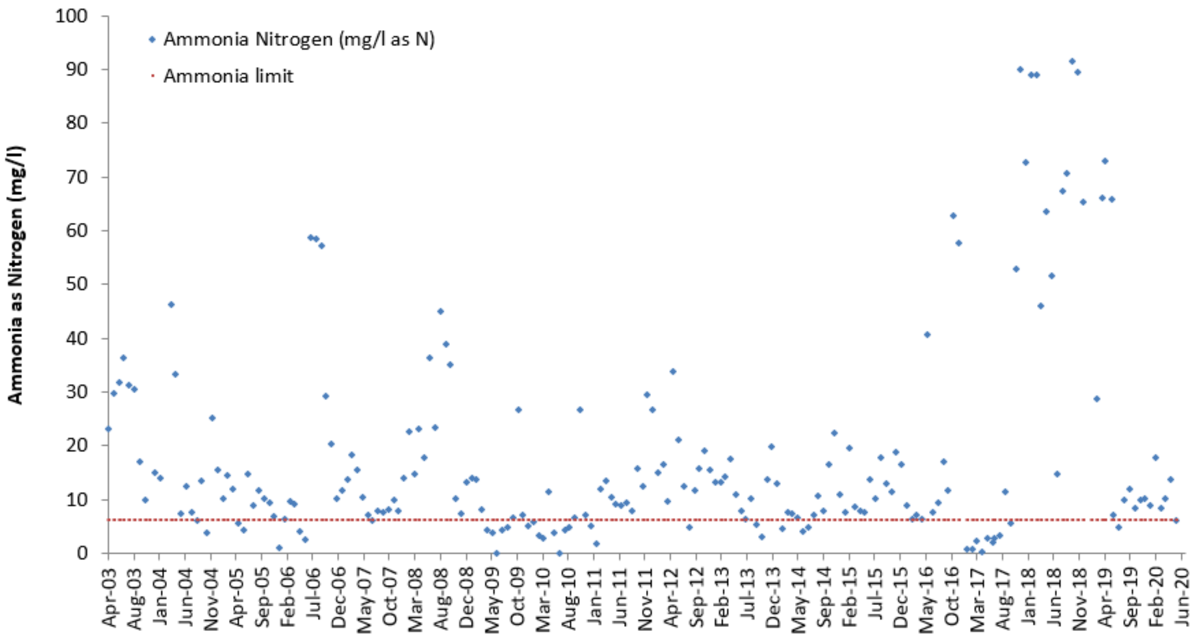


Figure 3.24. Monthly trends in Ammonia Nitrogen (mg/L as N) in effluent released from the Saldanha Wastewater Treatment Works April 2003-June 2020. Allowable limits in terms of a General Authorisation under the National Water Act (No. 36 of 1998) are represented by the dashed red line (Source: Saldanha Bay Municipality).

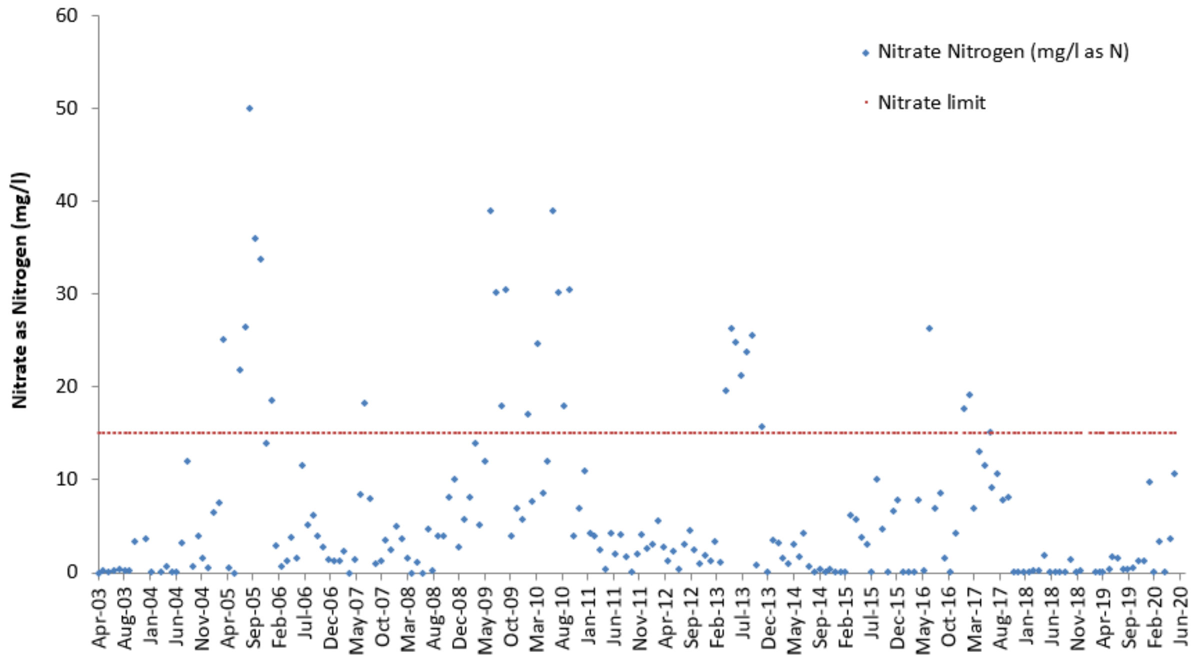


Figure 3.25. **Monthly trends in Nitrate Nitrogen (mg/L as N) in effluent released from the Saldanha Wastewater Treatment Works April 2003-June 2020. Allowable limits in terms of a General Authorisation under the National Water Act (No. 36 of 1998) are represented by the dashed red line (Source: Saldanha Bay Municipality).**

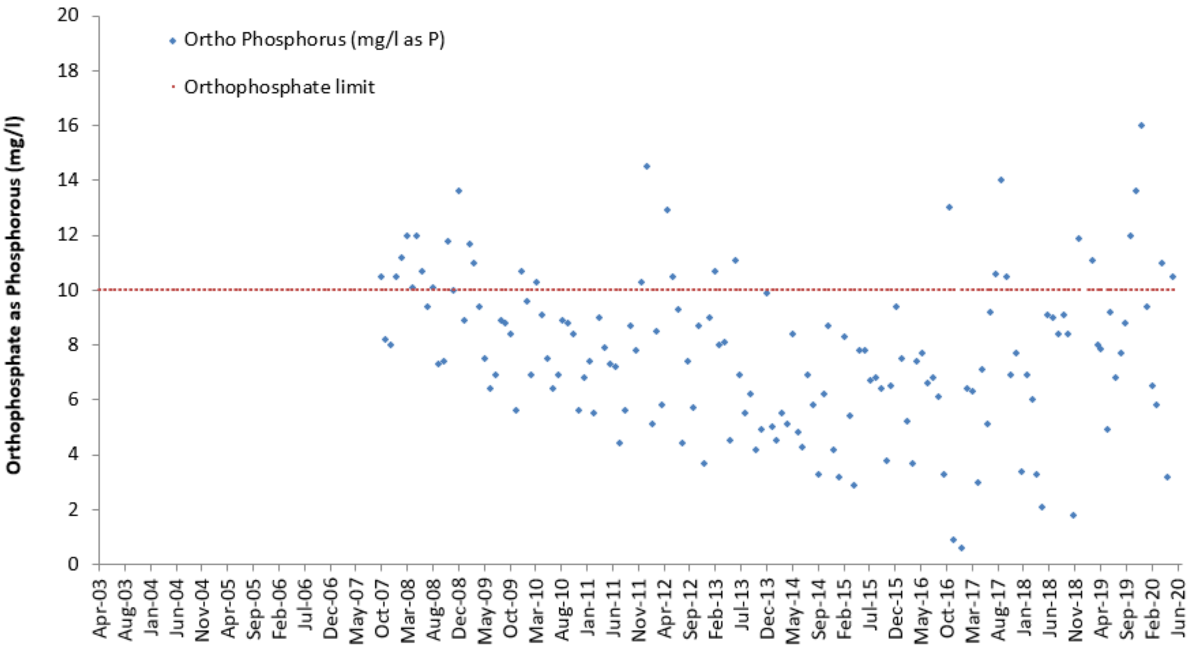


Figure 3.26. **Monthly trends in Orthophosphate (mg/L as P) in effluent released from the Saldanha Wastewater Treatment Works April 2003-June 2020. Allowable limits in terms of a General Authorisation under the National Water Act (No. 36 of 1998) are represented by the dashed red line (Source: Saldanha Bay Municipality).**

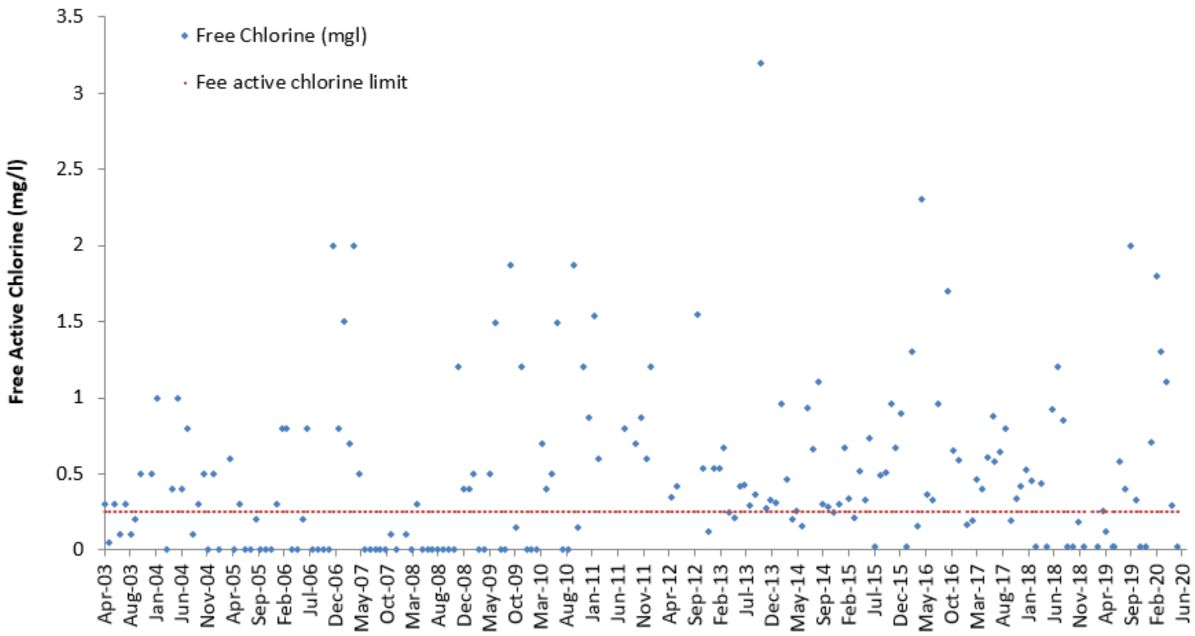


Figure 3.27. Monthly trends in Free Active Chlorine (mg/L) in effluent released from the Saldanha Wastewater Treatment Works April 2003-June 2020. Allowable limits in terms of a General Authorisation under the National Water Act (No. 36 of 1998) are represented by the dashed red line. An outlier of 12 mg/L measured for January 2008 was removed to show the trend more clearly (Source: Saldanha Bay Municipality).

3.6.3.4 Langebaan Wastewater Treatment Works

The Langebaan WWTW treats sewage by means of activated sludge with BNR and drying ponds. However, as is the case with effluent from the Saldanha WWTW, SBM has for quite some time been favouring alternative uses of wastewater from the Langebaan WWTW over discharge to the marine environment. Most recently, the SBM obtained permission from the Department of Water and Sanitation (DWS) to use a maximum of 200 m^3 for the irrigation of lawn on the WWTW premises as well as the flower beds along Oosterwal Rd leading into Langebaan. Furthermore, the majority of wastewater produced by the Langebaan WWTW is diverted to the Langebaan Country Estate for the irrigation of the golf course. Prior to irrigation, the wastewater is further treated by means of 11 polishing ponds. Wastewater is exposed to UV radiation in these ponds, reducing harmful pathogen populations.

While at first all the wastewater was used for irrigation, increasing volumes of effluent received by the Langebaan WWTW was yielding more water than required for irrigation of the golf course, especially during winter. Consequently, more and more excess wastewater was discharged into the Langebaan Lagoon Marine Protected Area (MPA). However, with the implementation of water restrictions, wastewater produced by the Langebaan WWTW has been decreasing considerably, which means that only very small quantities of wastewater overflowed into the MPA during the winter months in 2018 (SBM, *pers. comm.* 2018). In addition, SBM has undergone an agreement with Langebaan Country estate to irrigate effluent within its boundaries during the Winter months. Subsequently, it is believed that through this agreement all discharge of effluent into the MPA has been eliminated (Quintin Williams, SBM, *pers. comm.* 2020).

The overflow from the storage dams was noticed by the Department of Environmental Affairs: Branch Oceans & Coasts, which identified this as an illegal activity in terms of the National Environmental Management: Protected Areas Amendment (Act No 21 of 2014) (NEMPAAA).

Section 48A *(d)* prohibits the discharging or depositing of waste or any other polluting matter into an MPA, unless a CWDP is granted by the Minister of Environmental Affairs in terms of the ICMA. A directive was issued to the SBM to stop releasing effluent into the Langebaan Lagoon MPA. The DEA: O&C made it clear to the SBM that a CWDP would not be issued for this discharge and that alternative measures should be implemented instead to prevent overflow. The SBM is experiencing a high demand for wastewater, especially during summer for irrigation purposes. The SBM therefore conducted a comprehensive study regarding the re-use of treated effluent from the Langebaan WWTW and other WWTW. Options that emerged from this study included storage of surplus effluent during the winter months for use in summer, supply of wastewater to industry throughout the year and reclamation of potable water by means of reverse osmosis. Alternative options will be investigated for their feasibility and implemented once upgrades to the Langebaan WWTW have been completed (see more detail below).

While the SBM is responsible for ensuring that an appropriate amount of treated sewage is supplied to the Langebaan Country Estate to prevent non-compliance with the ICMA, the Langebaan Country Estate must ensure compliance with the National Water Act (Act 36 of 1998 as amended) NWA in terms of the storage and irrigation of wastewater. The Langebaan Country Estate is currently in the process of registering as a water user for these very water uses.

Legislative requirements applicable to the Saldanha Bay Municipality

The Department of Water and Sanitation (DWS) confirmed in January 2018 that the SBM was successfully registered as a water user in terms of Section 22(1)(a)(iii), which prescribes that "*A person may only use water without a licence if that water use is permissible in terms of a general authorisation issued under Section 39."* (Refer to AEC 2017 for more information on previous authorisations/exemptions). The Langebaan WWTW is permitted to irrigate up to 73 000 m^3 (daily maximum of 200 m^3 per day) of wastewater per annum on 12.68 ha (water use as prescribed in NWA Section 21(e)). Furthermore, the SBM is permitted to store treated effluent for irrigation purposes in ponds with a maximum storage capacity of 4 485 m^3 (water use as prescribed in NWA Section 21(g): "disposing of waste in a manner which may detrimentally impact on a water resource."). The conditions of the General Authorisation applicable to the above described water uses are prescribed in Regulations 1 and 3 of the *GN 665 Government Gazette 36820* dated 6 September 2013. Regulation 1 prescribes that specific wastewater quality limit values are applicable depending on the volume of wastewater irrigated. The SBM intends to irrigate more than 50 m^3 but less than 500 m^3 per day. The applicable limits are shown in Table 3.5. The General Authorisation also specifies that:

1) Water user must follow acceptable construction, maintenance and operational practices to ensure the consistent, effective and safe performance of the wastewater irrigation system, including the prevention of -
    a. waterlogging of the soil and pooling of wastewater on the surface of the soil;
    b. nuisance conditions such as flies or mosquitoes, odour or secondary pollution;
    c. waste, wastewater or contaminated stormwater entering into a water resource;

d. the contamination of run-off water or stormwater;
e. the unreasonable chemical or physical deterioration of, or any other damage to, the soil of the irrigation site;
f. the unauthorised use of the wastewater by members of the public; and
g. people being exposed to the mist originating from the irrigation of the wastewater.

2) Suspended solids must be removed from any wastewater, and the resulting sludge disposed of according to the requirements of any relevant law or regulation, including the document Guidelines for the Utilisation and Disposal of Wastewater Sludge, Volumes 1-5, Water Research Commission Reports TT 261/06, 262/06, 349/09, 350/09, 351/09, as amended from time to time (obtainable from the responsible authority upon written request).
3) All reasonable measures must be taken to provide for mechanical, electrical, operational, or process failures and malfunctions of the wastewater irrigation system.
4) All reasonable measures must be taken for storage of the wastewater used for irrigation when irrigation cannot be undertaken, of which the storage must be in accordance with general authorisation in section 3 of this Notice.
5) All reasonable measures must be taken to collect contaminated stormwater or runoff emanating from the area under irrigation and to retain it for disposal of which the disposal must be in accordance with general authorisation in section 3 of this Notice.
6) Upon the written request of the responsible authority the registered user must ensure the implementation of any additional construction, maintenance and operational practices that may be required in the opinion of the responsible authority to ensure the consistent, effective, safe and sustainable performance of the wastewater irrigation system.

The SBM is also obligated to establish monitoring programmes for the quantity and quality of wastewater to be used for irrigation prior to commencement and thereafter, in the following manner:

a. The quantity must be metered and the total recorded weekly; and
b. the quality of water irrigated must be monitored once every month by taking a grab sample at the point at which the wastewater enters the irrigation system for all parameters listed in paragraph 1.7(1)(i), (ii) and (iii) and results submitted to the responsible authority.

More detailed information can be requested by the DWS from the SBM.

Table 3.5. Wastewater limit values applicable to the irrigation of any land or property up to 500 cubic metres (National Water Act 36 of 1998, *GN 665 Government Gazette 36820* dated 6 September 2013).

Variables	Limits
pH	Not less than 6 of more than 9 pH units
Electrical conductivity	Not exceed 200 milliSiemens per metre (mS/m)
Chemical Oxygen Demand (COD)	Does not exceed 400 mg/L after removal of algae
Faecal coliforms	Do not exceed 100 000 per 100 mL
Sodium Adsorption Ratio (SAR)	Does not exceed 5 for biodegradable industrial wastewater

Regulation 3.14 prescribes the conditions applicable with regards to record-keeping and disclosure of information for the storage of wastewater. The SBM is required to conduct monthly monitoring of water quantity and quality. Water quality parameters are not specified in Regulation 3 and it is therefore assumed that the parameters as specified in Table 3.5 are applicable (the wastewater is not discharged into a water resource and those limits are therefore not applicable in terms of the GA).

Regulation 3 of the General Authorisation also specifies that:

1) The water user must follow acceptable design, construction, maintenance and operational practices to ensure the consistent, effective and safe performance of the wastewater discharge system, including the prevention of -
    h. nuisance conditions such as flies or mosquitoes, odour or secondary pollution;
    i. the contamination of run-off water or stormwater;
    j. contaminated stormwater entering into a water resource; and
    k. the unauthorised use of the wastewater by members of the public.
2) Suspended solids must be removed from any wastewater, and the resulting sludge disposed of according to the requirements of any relevant law or regulation.
3) All reasonable measures must be taken to prevent wastewater overflowing from any wastewater disposal system or wastewater storage dam.
4) All reasonable measures must be taken to provide for mechanical, electrical, or operational failures and malfunctions of any wastewater disposal system or wastewater storage dam.
5) Sewage sludge must be removed from any wastewater and the resulting sludge disposed of according to the requirements of any relevant law and regulation, including –
6) Guidelines for the Utilisation and Disposal of Wastewater Sludge, Volumes 1-5, Water Research Commission Reports TT 261/06, 262/06, 349/09, 350/09, 351/09, as amended from time to time; and
7) "Guide: Permissible utilisation and disposal of treated sewage effluent", 1978, Department of National Health and Population Development Report No. 11/2/5/3, as amended from time to time (obtainable from the Department upon written request).

Planned upgrades to the Langebaan WWTW

Various upgrades are required to improve the overall performance of the treatment plant (SBM, Gavin Williams, *pers. comm*. 2016) and have been ongoing in the form of a phased approach. The first phase included the construction of a new reactor basin, installation of new aeration equipment and new sludge drying beds and was completed in 2017/18 financial year. These upgrades increased the plant capacity to 3.5 ML and included an additional aeration basin, a new clarifier and drying beds as well as new inlet works and screens with a total budget of R17 million (SBM, Gavin Williams, *pers. comm.* 2019). The ongoing phased approach and installation of new infrastructure will increase the capacity of the plant to 5-7 ML. Phase 2 of these upgrades is in the final stages and is set to be completed by the end of September 2020 and tender processes for phase 3 are currently being implemented. This phase will include the refurbishment of the old bio reactor, mechanical dewatering facility and power supply to the works (SBM, Quintin Williams, pers. comm. 2020). An aerial view of the Langebaan WWTW is shown in Figure 3.28.

Over time more effluent than currently absorbed by the Langebaan Country Club will be produced. The SBM intends to appoint a consultant to design proposals on how to use or discharge excess effluent (SBM, Gavin Williams, *pers. comm.* 2019). For example, the municipality is planning to use excess effluent to irrigate the lawn at the Langebaan Sports Complex. It appears that the demand for wastewater is high enough to absorb the excess effluent. Most importantly, however, water users would have to be identified prior to the expansion of the plant to prevent non-compliance with the ICMA as described above.

Figure 3.28. Construction activities for the upgrade of the Langebaan Waste Water Treatment Plant to increase treatment capacity and improve treatment processes (Source: Saldanha Bay Municipality).

Treated wastewater quality monitoring

The annual State of the Bay Report has been reporting water quality parameters measured prior to the transfer of the effluent to the Langebaan Country Club. It is noteworthy that the effluent is further treated prior to irrigation by means of 11 polishing ponds. However, water quality is currently not monitored prior to irrigation and although, according to SBM no effluent has entered the MPA in over a year, the actual water quality of the treated wastewater that could enter the MPA via the illegal overflow is currently unknown. This report therefore continues to describe the water quality trend over time as measured at the end of pipe at the Langebaan WWTW. Note that the legal water quality limits as per GA in terms of Section 21(f): "*Discharging waste or water containing waste into a water resource through a pipe, canal, sewer, **sea outfall** or other conduit"* are no longer applicable as the **sea outfall** is now regulated by the ICMA by means of CWDPs. Accordingly, the GA of 2013 (*GN 665 Government Gazette 36820* dated 6 September 2013) specified that the GA is no longer applicable to sea outfalls.

Trends of water quality parameters in the effluent released into the Langebaan Lagoon MPA between 2009 and 2019 are therefore no longer compared to the GA limits for **wastewater discharge**. Instead, where monitoring information is available, the results have been compared to GA limits for **irrigation** as shown in Table 3.5. These parameters include pH, electrical conductivity, Chemical Oxygen Demand, and Faecal Coliforms. No data is currently available for Sodium Adsorption Ration (SAR).

In addition to the above, due to occasional discharges of effluent into the MPA, the effluent monitoring results will be compared to a limit that is more relevant to the inshore marine environment. As part of the Assessment Framework for the Management of Effluent from Land Based Sources Discharged to the Marine Environment that was recently developed by Anchor for the DEA: O&C (AEC 2015), recommendations were made regarding the applicability of General Discharge Authorisations and what type of effluents should qualify. The overflow into the MPA would not be considered to fall under a GDA (and the DEA: O&C indicated that a CWDP would not be issued for a new outfall in an MPA), however, the GDA special limits as recommended in the Assessment Framework are more applicable to the marine environment than limits derived for irrigation or wastewater discharges into freshwater resources. Wastewater monitoring results have therefore been compared to the recommended special limits purely to provide context.

Long-term trends in water quality are shown in Figure 3.29- Figure 3.39. It is noteworthy that for quite some time, the amount of wastewater entering the marine environment has been very low and is unlikely to have contributed significantly to pollution of the receiving environment (although due to the lack of water quality and quantity data this is impossible to say with confidence). The changes implemented by the SBM are therefore mainly positive and interpretation of water quality results must consider that volumes are likely to be low and of better quality than indicated in the graphs below.

The previous exemption permitted the irrigation of the local golf course with 1 611 m^3 treated effluent per day, which was exceeded 92% of the time between 2009 and December 2017 (Figure 3.29.). Overall, effluent volumes peak over the December holidays when plant capacity is often reached and in some instances exceeded (e.g. December 2016, average daily effluent volumes were 2 840 m^3 with a maximum daily flow of 5 545 m^3) (Figure 3.30). The legal limit for effluent production increased to 4 485 m^3/day in January 2018 when the SBM was issued with a new General Authorisation permission. Shortly thereafter, plant capacity was increased to 3 500 m^3. Since then the Langebaan WWTW has been compliant in terms of the legal effluent volume limit. Hydraulic design capacity (3500 m^3) was exceeded in January 2019 with an average daily flow of 4167 m^3 per day (i.e. 119% capacity) and marginally in March 2020 (3529 m^3 per day or 100.8%).

The Langebaan WWTW has been recording pH since 2009. The monitoring data shows that the wastewater always falls within the pH range to be met in terms of the GA for the irrigation of <500 m^3 wastewater (i.e. between pH of 6 and 9, Figure 3.31). Values of pH recommended for the protection of the inshore marine environment are range between 7.3 and 8.2 (AEC 2015). Since 2009 the pH of the wastewater effluent has fallen outside of these limits 46% of the time, with more basic (pH > 8.2) values recorded four times and more acidic (pH < 7.3) values recorded on 57 occasions. However, in the past year 2019/2020 pH levels have been more consistent, only dropping slightly below recommended values for the marine environment in August 2019 (7.25) and May 2020 (7.29).

In the 11 years since electrical conductivity (in mS/m) was first recorded at Langebaan WWTW. Conductivity has been declining steadily (Figure 3.32). With a peak reading of 625 mS/m recorded in September 2011 and the lowest recording to date occurring in March of this year (22 mS/m). Values have been fluctuating around the prescribed limit (200 mS/m) since December 2014, with this limit exceeded on 16 occasions (24% of the time), however no exceedance has occurred since November 2018.

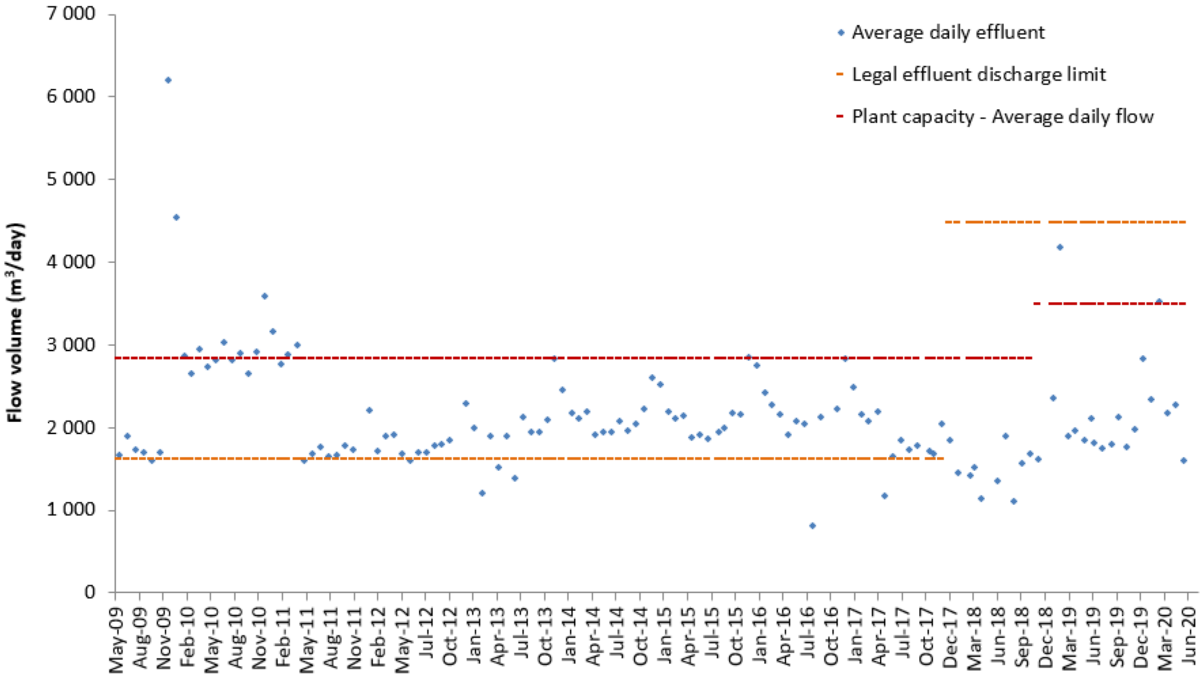


Figure 3.29. Trends in average daily effluent volume (m^3/month) released from the Langebaan Wastewater Treatment Works, June 2009 - June 2020. Allowable discharge limits in terms of the exemption issued by DWAF under the National Water Act (No. 36 of 1998) are represented by the dashed orange line and the design capacity of the plant by the red line (Source: Saldanha Bay Municipality).

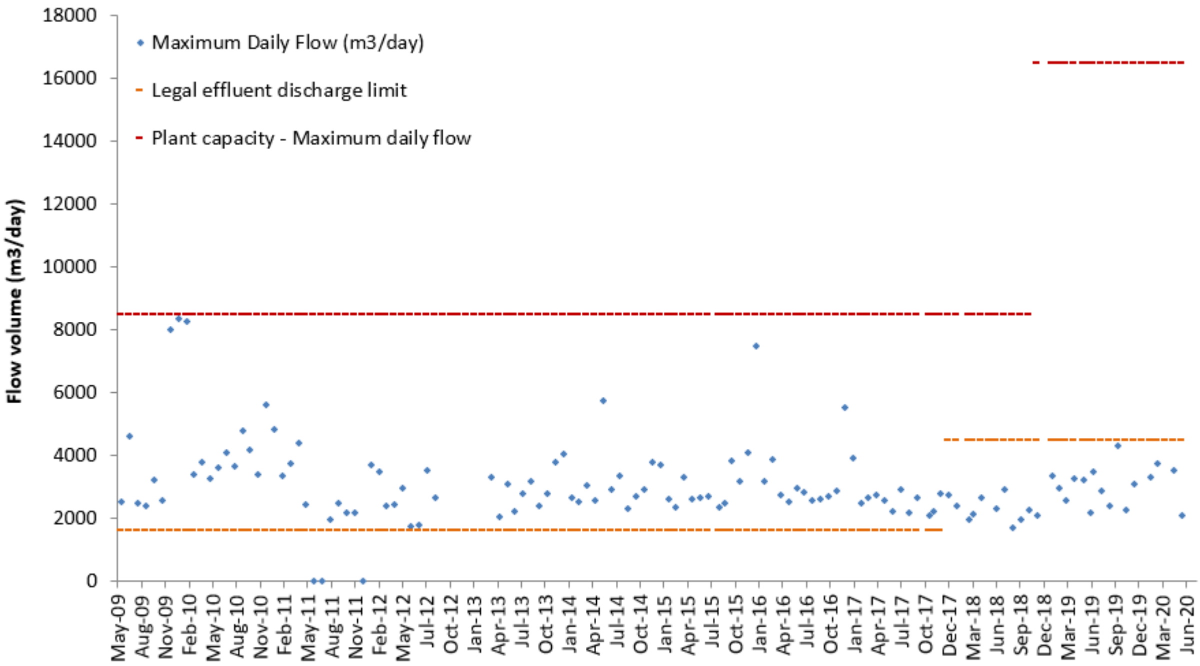


Figure 3.30. Trends in maximum daily effluent volume (m^3/month) released from the Langebaan Wastewater Treatment Works, June 2009 - June 2020. Allowable discharge limits in terms of the exemption issued by DWAF under the National Water Act (No. 36 of 1998) are represented by the dashed orange line and the design capacity of the plant by the red line (Source: Saldanha Bay Municipality).

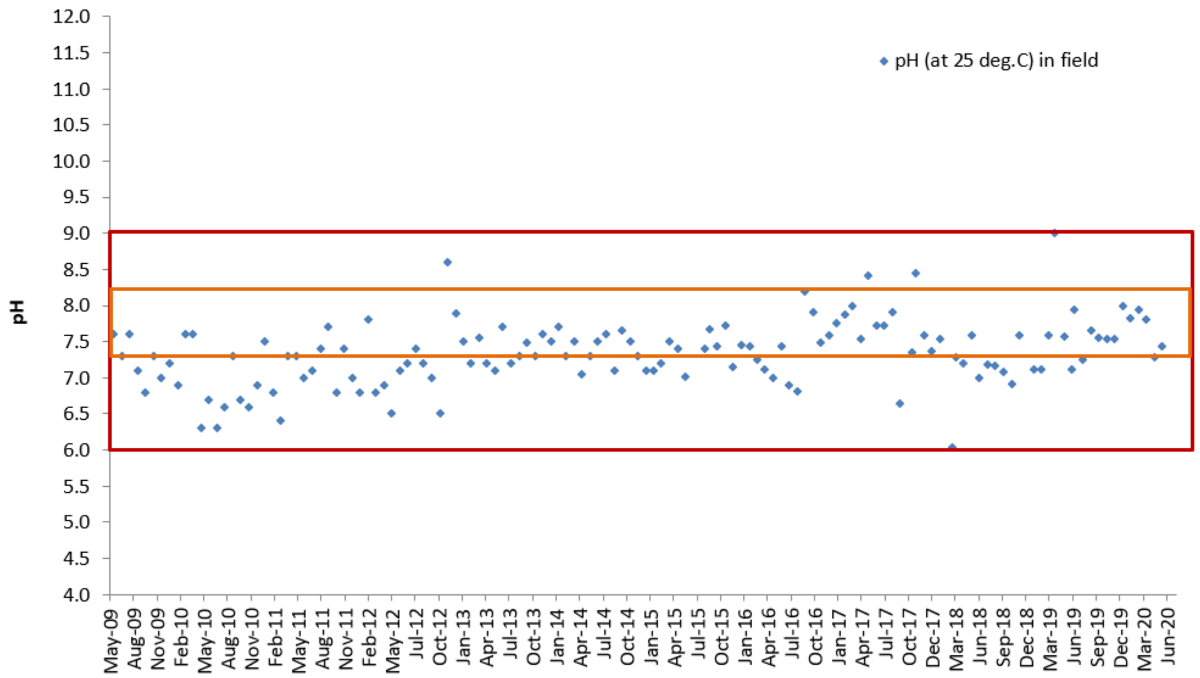


Figure 3.31. Monthly trends in pH of effluent from the Langebaan Wastewater Treatment Works, June 2009 - June 2020. The allowable range in terms of the General Authorisation for irrigation purposes under the National Water Act (No. 36 of 1998) is 6-9 and is depicted by the red square. The recommended range to protect marine inshore environments is 7.3-8.2 and is depicted by the orange square (AEC 2015) (Source: Saldanha Bay Municipality).

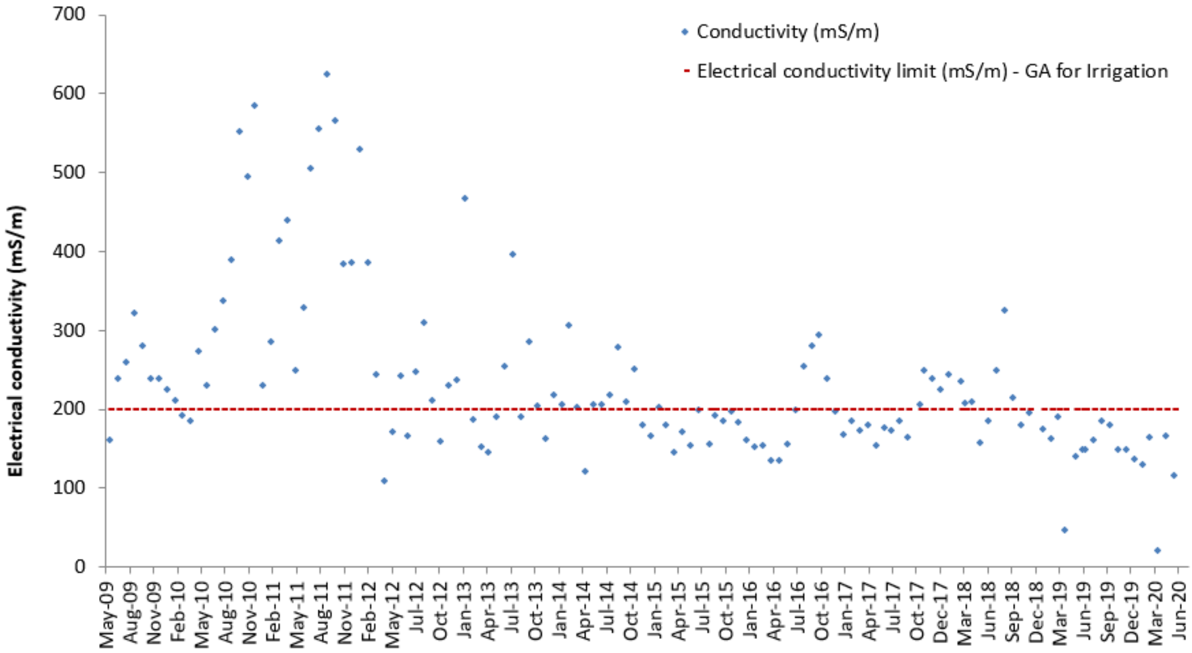


Figure 3.32. Monthly trends in conductivity of effluent from the Langebaan Wastewater Treatment Works, June 2009 - June 2020. The allowable limit in terms of the General Authorisation for irrigation purposes under the National Water Act (No. 36 of 1998) is 200 mS/m and is depicted by the red line (Source: Saldanha Bay Municipality).

COD in filtered effluent exceeded the allowable limit for the protection of marine organisms of 75 mg/L 30% of the time since June 2009, reaching an all-time maximum of 235 mg/L in January 2018 (Figure 3.33). However, it appears that there has been an improvement in recent years with the limit for the protection of the marine environment only being exceeded three times since June 2018 (12% of the time). In terms of the limit imposed by the GA applicable for irrigation, the SBM is compliant as COD is always lower than 400 mg/L (Figure 3.33).

To date concentrations of faecal coliforms in the effluent from the Langebaan WWTW have not exceeded the limit of 100 000 organisms per 100 mL imposed by the GA applicable to irrigation (Figure 3.34). In terms of recreational and mariculture concerns, 100 000 org/100 mL in the overflow would be unacceptable. The wastewater has stayed well below this limit however, the frequency of readings greater than the detection limit, and therefore multiplied by 3 to reach the conservative maximum value of 7258 org/100 mL, has increased in the last few years. In 2019/20 as many as eight of the 12 readings recorded were greater than the detection limit. Overall, it would be desirable for faecal coliform readings to stay below 1 000 org/100 mL as prescribed in the GA applicable to the discharge of wastewater into freshwater resources.

No Total Suspended Solids (TSS) limit is prescribed by the GA applicable to irrigation of wastewater. Overall, the water user is required to remove all suspended solids prior to irrigation of the wastewater. Therefore, the SBM will be required to remove TSS prior to the irrigation of their own premises and the flower beds on Oosterwal Road. The polishing ponds on the Langebaan Country Estate are likely to act as settlement ponds and TSS is likely to be lower than shown here. TSS values exceeded the recommended special limit for the protection of the inshore marine environment of 10 mg/L on 78 occasions since 2009 (59% of the time) (Figure 3.35). Overall, TSS levels appear to be steadily increasing since December 2014 with the majority of readings exceeding the recommended limit, the maximum TSS value of 198 mg/L occurred in March 2015. Similarly, annual peaks in the concentration occur at the end of the summer or early autumn each year with reading of 50 mg/L recorded in March 2017, 93 mg/L in April 2018, 55 mg/L in April 2019 and most recently 101 mg/L in March 2020.

No ammonia nitrogen limit is prescribed by the GA applicable to irrigation of wastewater, however ammonia is very toxic to marine life as it acts as a biocide. The recommended ammonia nitrogen limit for the inshore marine environment is 3 mg/L (Figure 3.36.). The water quality guidelines for the coastal environment specify a target of 0.6 mg/L to prevent chronic toxicity. Ammonia levels increased steeply between November 2012 and March 2018 from <10 mg/L to nearly 100 mg/L. Subsequent to that the ammonia nitrogen concentrations dropped significantly but were, however still grossly exceeding the recommended limit for the protection of the marine environment (3 mg/L). Readings have again started to increase in the past year with two readings measuring above 55 mg/L in 2020. Considering the above, the levels of ammonia in the Langebaan WWTW effluent is alarming and any amount of effluent released into the nearshore marine environment is likely to have a significant negative effect on marine biota.

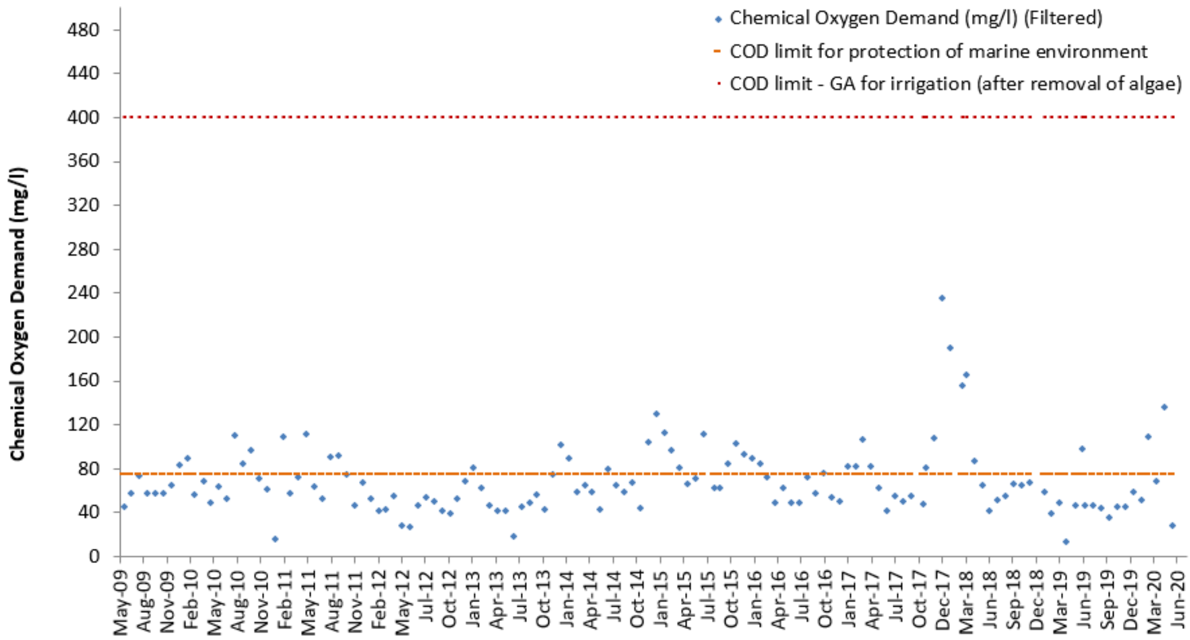


Figure 3.33. Monthly trends in chemical oxygen demand (mg/L filtered) in effluent released from the Langebaan Wastewater Treatment Works, June 2009 - June 2020. Allowable limits as specified in terms of a General Authorisation under the National Water Act (No. 36 of 1998) are represented by the dashed red line. The recommended limit to protect marine inshore environments is shown by the orange dashed line (AEC 2015) (Source: Saldanha Bay Municipality).

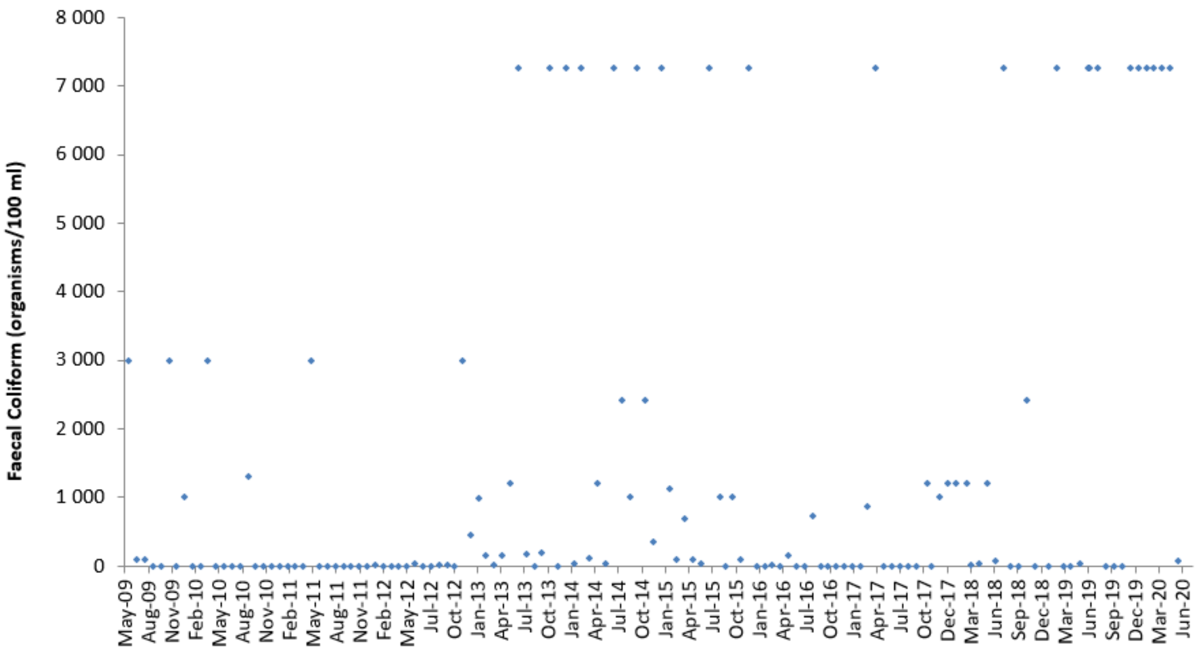


Figure 3.34. Monthly trends in Faecal Coliforms (org/100ml) in effluent released from the Langebaan Wastewater Treatment Works, June 2009 - June 2020. The allowable limit in terms of a General Authorisation for irrigation purposes under the National Water Act (No. 36 of 1998) is 100 000 organisms per 100 mL. (Source: Saldanha Bay Municipality).

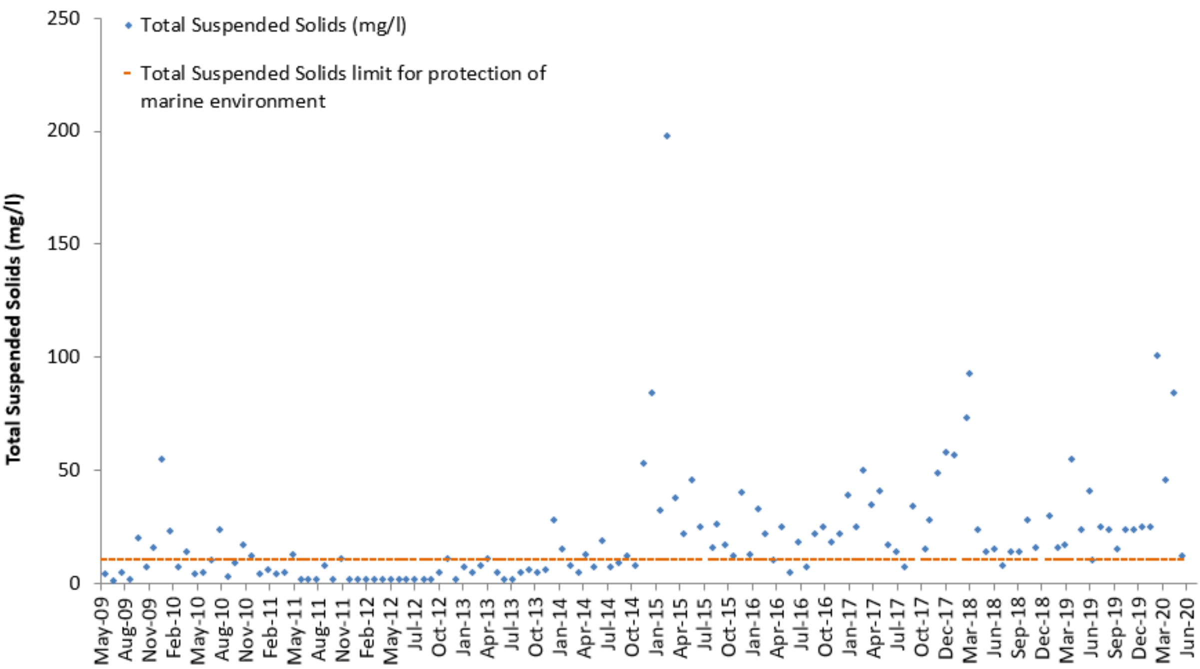


Figure 3.35. Monthly trends in total suspended solids (mg/L) in effluent released from the Langebaan Wastewater Treatment Works, June 2009 - June 2020. The recommended limit to protect marine inshore environments is shown by the orange dashed line (AEC 2015) (Source: Saldanha Bay Municipality).

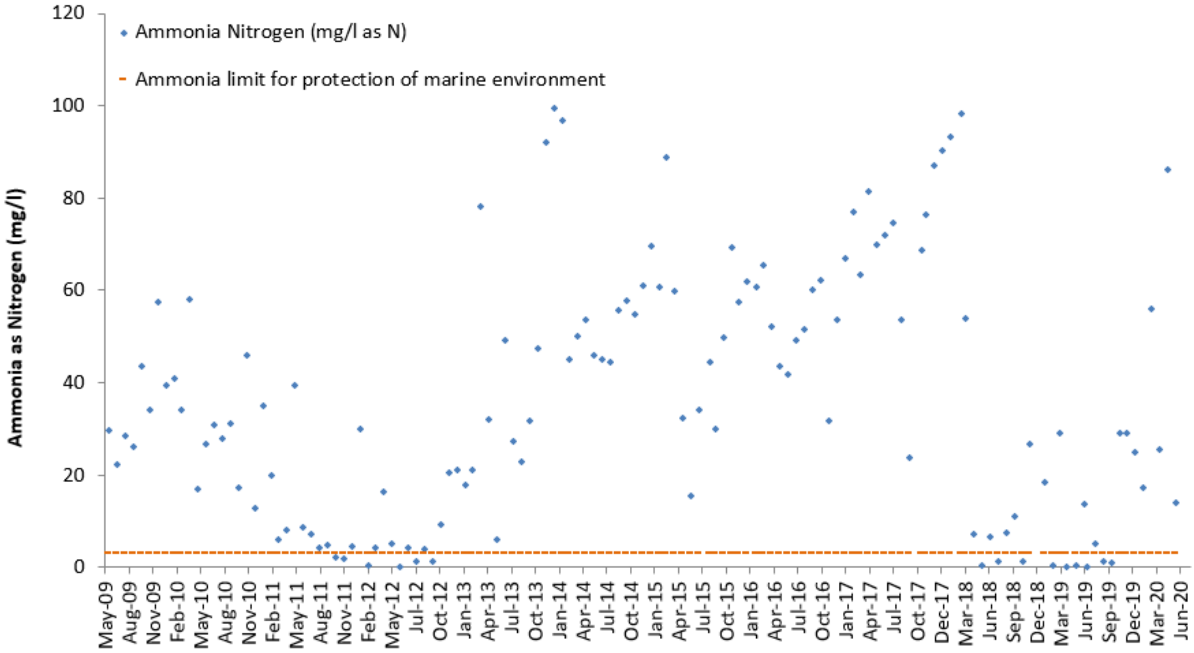


Figure 3.36. Monthly trends in Ammonia Nitrogen (mg/L as N) in effluent released from the Langebaan Wastewater Treatment Works June 2009 - June 2020. The recommended limit to protect marine inshore environments is shown by the orange dashed line (AEC 2015) (Source: Saldanha Bay Municipality).

Nitrate Nitrogen is not toxic to marine life but is a primary nutrient (usually marine systems are nitrogen limited) and could stimulate nuisance algae growth near the outfall point and its surrounds. No nitrate nitrogen limit is prescribed by the GA applicable to irrigation of wastewater. The recommended nitrate nitrogen limit for the inshore marine environment is 1.5 mg/L. This limit has been exceeded on 57 occasions since June 2009 (43% of the time) (Figure 3.37). Lower concentrations were recorded between April 2016 and March 2018 with all values less than the limit. The 2018/19 rolling year saw only 3 readings below the limit however this appears to have improved significantly in 2019/20 with 8 of the 12 readings falling below the limit. Toxic ammonia nitrogen is converted to non-toxic nitrate nitrogen by means of bacterial treatment in WWTWs. It is likely that the 2018/2019 observed higher levels are congruent with lower ammonia levels in the effluent. This suggests that the bacterial treatment is currently more effective than in years preceding 2018/2019.

Orthophosphate is usually not the limiting nutrient for primary production in the marine environment. The recommended limit applicable for discharges into the inshore marine environment is 1 mg/L. No orthophosphate limit is prescribed by the GA applicable to irrigation of wastewater. Orthophosphate levels have steadily increased since 2013, reaching the highest value recorded to date at 19 mg/L in May 2018 shortly thereafter which levels dropped significantly and remained fairly low (<4 mg/L) until April of this year when they increased to reach 14.4 mg/L in June 2020 (Figure 3.38). Overall, the orthophosphate concentration in the Langebaan WWTW effluent is considerably higher than 1 mg/L (89% exceedance). However, as observed with several other effluent parameters, orthophosphate levels improved significantly since November 2018, with an average of 4.3 ± 6.7 mg/L (45% of readings <1 mg/L).

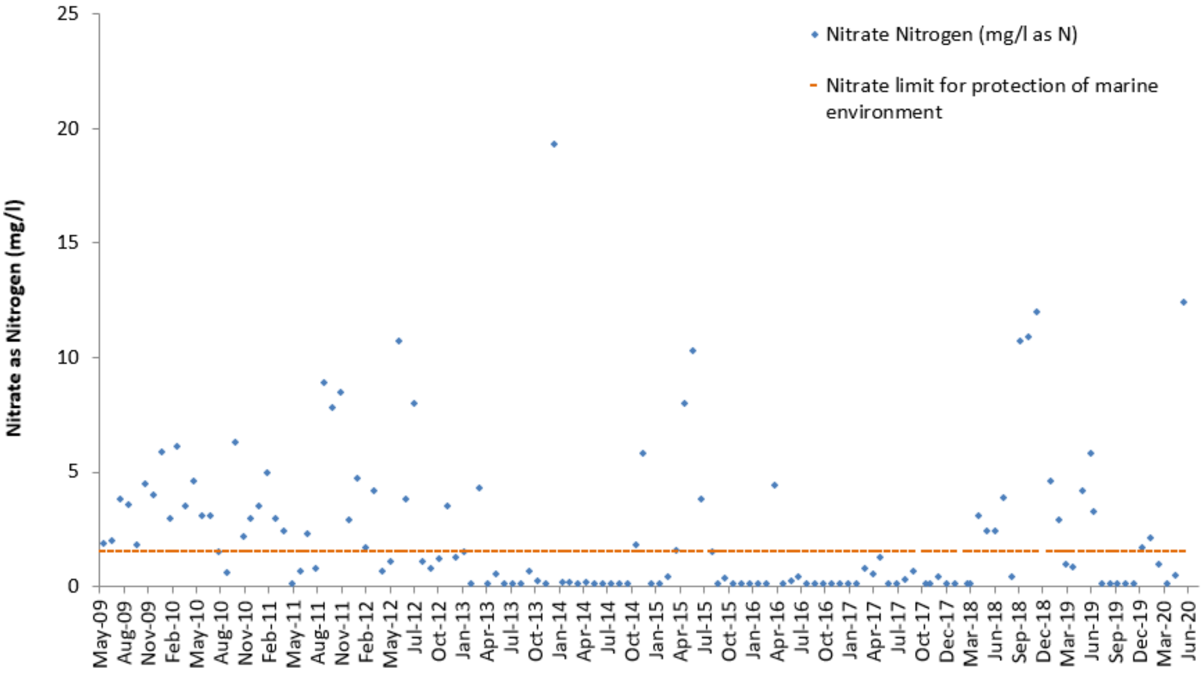


Figure 3.37. Monthly trends in Nitrate Nitrogen (mg/L as N) in effluent released from the Langebaan Wastewater Treatment Works June 2009 - June 2020. The recommended limit to protect marine inshore environments is shown by the orange dashed line (AEC 2015) (Source: Saldanha Bay Municipality).

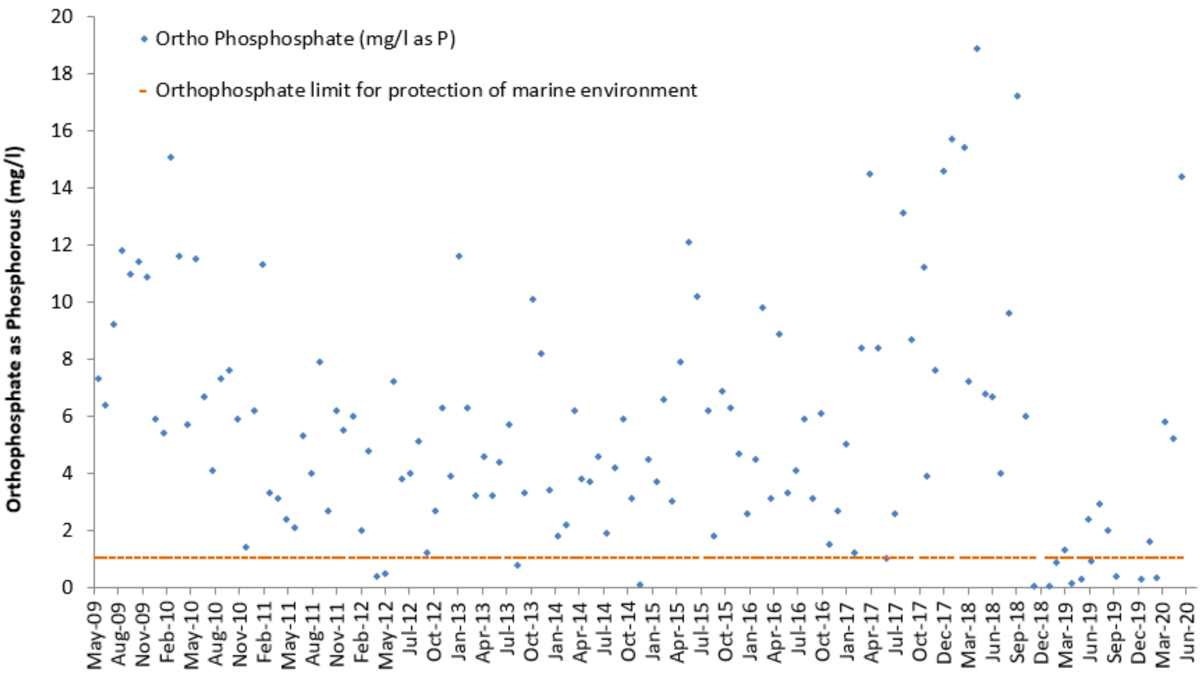


Figure 3.38. Monthly trends in Orthophosphate (mg/L as P) in effluent released from the Langebaan Wastewater Treatment Works June 2009 - June 2020. The recommended limit to protect marine inshore environments is shown by the orange dashed line (AEC 2015) (Source: Saldanha Bay Municipality).

No free active chlorine limit is prescribed by the GA applicable to irrigation of wastewater, however, free active chlorine is highly toxic to marine life as it acts as a biocide. The recommended limit to protect the inshore marine environment is 0.5 mg/L. Concentrations have exceeded this limit 47% of the time since 2009 and fluctuated around 1.2 ±1.2 mg/L between July 2016 and June 2019 (Figure 3.39. Monthly values for 2019/20 appeared to improve with most values below 1.2 mg/L and six of the 12 readings not exceeding the limit, however two high exceptions exist, recorded in February and March 2020, where both readings are above the measurable detection limit. These levels are significantly higher than what would be considered acceptable if discharged into the nearshore environment and more careful dosing of chlorine should be implemented.

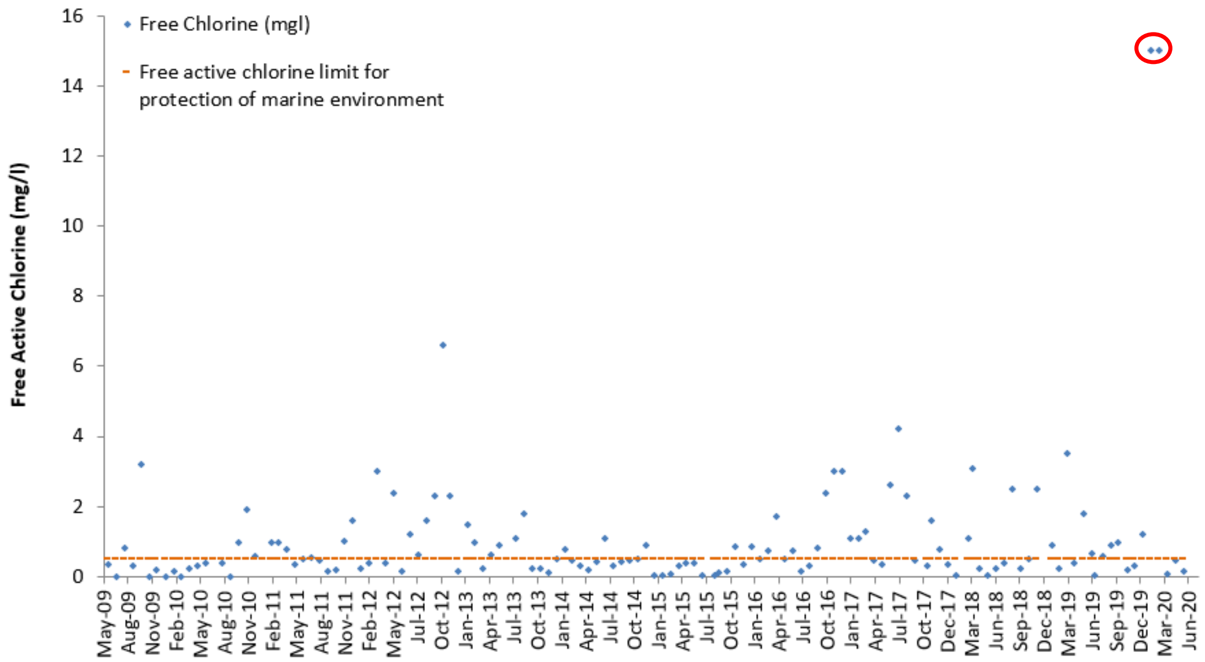


Figure 3.39. **Monthly trends in Free Active Chlorine (mg/L) in effluent released from the Langebaan Wastewater Treatment Works June 2009 - June 2020. The recommended limit to protect marine inshore environments is shown by the orange dashed line (AEC 2015) (Source: Saldanha Bay Municipality).**

3.6.3.5 Summary

The Saldanha Bay Municipality (SBM) has made a considerable effort over the last few years to re-use treated wastewater to save precious potable water where possible. Treated wastewater has been supplied for irrigation, industrial use (e.g. cooling processes) and dust suppression at construction sites. Overall, it appears that, especially in summer, the demand for treated wastewater is very high and the SBM is unable to meet the demand at current wastewater treatment capacity. Very small volumes of effluent have entered the marine environment from both WWTWs since early 2018, which is expected to continue in the foreseeable future. Despite this new effluent discharge pattern, effluent quality monitoring results will continue to be compared to relevant legal and/or recommended limits. When interpreting these results, the reader must remain cognisant of the fact that very small volumes are entering the marine environment and impacts are likely to be limited (over time, extent and magnitude).

Overall, the data shows that the majority of Saldanha Bay WWTW water quality parameters have improved in the past year although they are not always within allowable limits and conditions as set out in the NWA (Government Gazette No. 36820, 6 September 2013). In 2019/20 the Nitrate-Nitrogen concentration averaged 2.5 ± 3.5 mg/L and the limit has not been exceeded since June 2017. Similarly, Faecal coliform and TSS had a 100% compliance rate and the limit for COD was only exceeded once in 2019/20. In contrast, despite a 5-fold reduction in the average levels of Ammonia Nitrate in 2019/20 compared to those in 2018/19, this average was still above the allowable limit. The average Orthophosphate levels were below the limit however, the limit was exceeded 5 out of 12 times and it is somewhat concerning that the highest reading on record to date (1.6 times the limit) occurred in December 2019. In addition, the permissible chlorine limit was exceeded 75% of the time in 2019/20. Therefore, while many parameters showed an improvement there are still some that need to be better managed, especially because since the closure of ArcelorMittal a portion of effluent is being discharged into the Bok which ultimately exits into the Bay.

Improved effluent quality was recorded at the Langebaan WWTW for some parameters. Orthophosphate levels have improved significantly since November 2018, with 45% of readings being below the recommended limit. Conductivity has also been consistently decreasing and has been compliant with the General Authorisation for irrigation since November 2018. Additionally, the Chemical Oxygen Demand compliance improved with levels only exceeding the recommended limit for the protection of the marine environments on three occasions in the past two years. Conversely, although faecal coliforms in the effluent from the Langebaan WWTW have not exceeded the limit imposed by the GA applicable to irrigation values appear to be increasing with time and eight of the 12 readings recorded in 2019/20 were greater than the measured detection limit. Similarly, TSS levels appear to be steadily increasing since December 2014 with the vast majority of readings exceeding the recommended limit for the protection of marine environments, values appear to peak at the end of summer or early autumn. While generally improved in the past 3 years, the ammonia Nitrogen levels have increased in the last year with the majority above the recommended limit for the protection of marine environments. Despite this, it is important to remember that according to SBM no effluent is reaching the marine environment as it is all been allocated for reuse.

The data shows that the Saldanha WWTW is receiving greater volumes of effluent for treatment than permitted, despite the cessation of industrial effluent from ArcelorMittal. However, it should be noted that the SBM is currently in the process of amending their Water Use License and that effluent volumes rarely exceed the plant capacity (nearly double that of the legal limit). The Langebaan WWTW was recently upgraded to 3 500 m^3 and was issued permission to store 4 485 m^3 in January 2018. The legal limit has not been exceeded since and plant capacity has only been exceeded twice.

3.6.4 Storm water

Storm water runoff, which occurs when rain flows over impervious surfaces into waterways, is one of the major non-point sources of pollution in Saldanha Bay (CSIR 2002). Sealed surfaces such as driveways, streets and pavements prevent rainwater from soaking into the ground and the runoff typically flows directly into rivers, estuaries or coastal waters. Storm water running over these surfaces accumulates debris and chemical contaminants, which then enters water bodies untreated and may eventually lead to environmental degradation. Contaminants that are commonly introduced into coastal areas via storm water runoff include metals (Lead and Zinc in particular), fertilizers, hydrocarbons (oil and petrol from motor vehicles), debris (especially plastics), bacteria and pathogens and hazardous household wastes such as insecticides, pesticides and solvents (EPA 2003).

It is very difficult to characterise and treat storm water runoff prior to discharge, and this is due to the varying composition of the discharge as well as the large number of discharge points. The best way of dealing with contaminants in storm water runoff is to target the source of the problem by finding ways that prevent contaminants from entering storm water systems. This involves public education as well as effort from town planning and municipalities to implement storm water management programmes.

The volume of storm water runoff entering waterways is directly related to the catchment characteristics and rainfall. The larger the urban footprint and the higher rainfall, the greater the runoff will be. At the beginning of a storm a "first flush effect" is observed, in which accumulated contaminants are washed from surfaces resulting in a peak in the concentrations of contaminants in the waterways (CSIR 2002). Several studies have shown degradation in aquatic environments in response to an increase in the volume of storm water runoff (Booth & Jackson 1997, Bay *et al.* 2003).

Typical concentrations of various storm water constituents (metals, nutrients, bacteriological) for industrial and residential storm water from South Africa and elsewhere were extracted from the literature by the CSIR in 2002 (Table 3.6.). These values are rough estimates as site specific activities will have a strong influence on storm water composition and ideally more accurate data should be acquired by monitoring of contaminants in the storm water systems of Saldanha and Langebaan. It is clear that the estimated concentrations of many of the potentially toxic compounds are above the South African 1998 water quality guidelines for coastal and marine waters (values indicated in red). It is likely that introduction of contaminants via storm water runoff negatively impact the health of the marine environment, especially during the "first flush" period as winter rains arrive.

Table 3.6. Typical concentrations of water quality constituents in storm water runoff (residential and Industrial) (from CSIR 2002) and South Africa 1998 Water Quality Guidelines for the Natural Environment (*) and Recreational Use (**). Values that exceed guideline limits are indicated in red.

Parameter	Residential	Industrial	Water Quality Guidelines
Total suspended solids (mg/L)	500	600	-
Chemical oxygen demand (mg/L)	60	170	-
Nitrate-N (mg/L)	1.2	1.4	0.015*
Total Ammonia-N (mg/L)	0.3	0.4	0.6*
Orthophosphate-P (mg/L)	0.07	0.1	-
Cadmium (mg/L)	0.006	0.005	0.004*
Copper (mg/L)	0.05	0.05	0.005*
Lead (mg/L)	0.3	0.1	0.012*
Zinc (mg/L)	0.4	1.1	0.025*
Faecal coliform counts (counts/100 ml)	48 000	48 000	100**

Storm water runoff that could potentially impact the marine environment in Saldanha and Langebaan originates from industrial areas (490 ha), the Saldanha Bay residential area (475 ha), industrial sites surrounding the Port of Saldanha (281 ha), and Langebaan to Club Mykonos (827 ha) (Figure 3.40.). All residential and industrial storm water outlets drain into the sea.

The CSIR (2002) estimated the monthly flow of storm water entering Saldanha Bay and Langebaan Lagoon using rainfall data and runoff coefficients for residential and industrial areas. In this report, these estimates have been updated by obtaining more recent area estimates of industrial and residential developments surrounding Saldanha Bay and Langebaan Lagoon using Google Earth and by acquiring longer term rainfall data (Figure 3.40. and Table 3.7.). Runoff coefficients used to calculate storm water runoff from rainfall data were 0.3 for residential areas and 0.45 for industrial areas (CSIR 2002). Note that runoff from the Port of Saldanha and ore terminal have been excluded from these calculations. Storm water runoff is highly seasonal and peaks in the wet months of May to August. Due to the rapid pace of holiday and retail development in the area, Langebaan residential area produces the greatest volumes of storm water runoff, followed by the industrial areas, with lower volumes arising from the Saldanha residential area. The actual load of pollutants entering the Bay and Lagoon via this storm water can only be accurately estimated when measurements of storm water contaminants in the storm water systems of these areas are made.

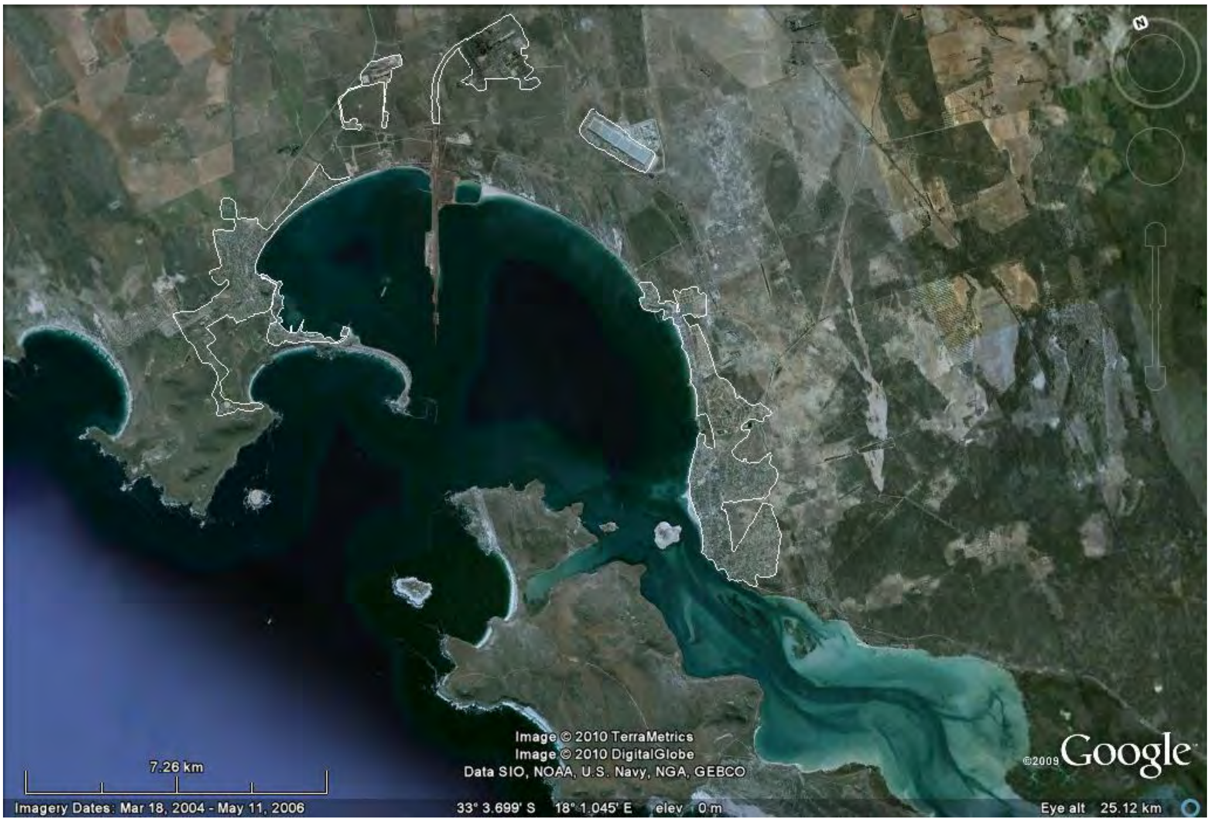


Figure 3.40. Spatial extent of residential and industrial areas surrounding Saldanha Bay and Langebaan Lagoon from which storm water runoff is likely to enter the sea (areas outlined in white). Note that runoff from the Port of Saldanha and ore terminal have been excluded as this is now reportedly all diverted to storm water evaporation ponds.

Table 3.7. Monthly rainfall data (mm) for Saldanha Bay over the period 1895-1999 (source Visser *et al.* 2007). MAP = mean annual precipitation.

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
MAP	6	8	11	25	47	61	64	46	25	18	13	8	332
Ave. rain days	1.4	1.4	2.2	3.8	6.2	7.1	7.5	6.4	4.8	3.0	1.9	1.8	47.5
Ave./day	4.1	5.5	5.1	6.6	7.6	8.5	8.5	7.3	5.2	6.0	6.6	4.6	7.0

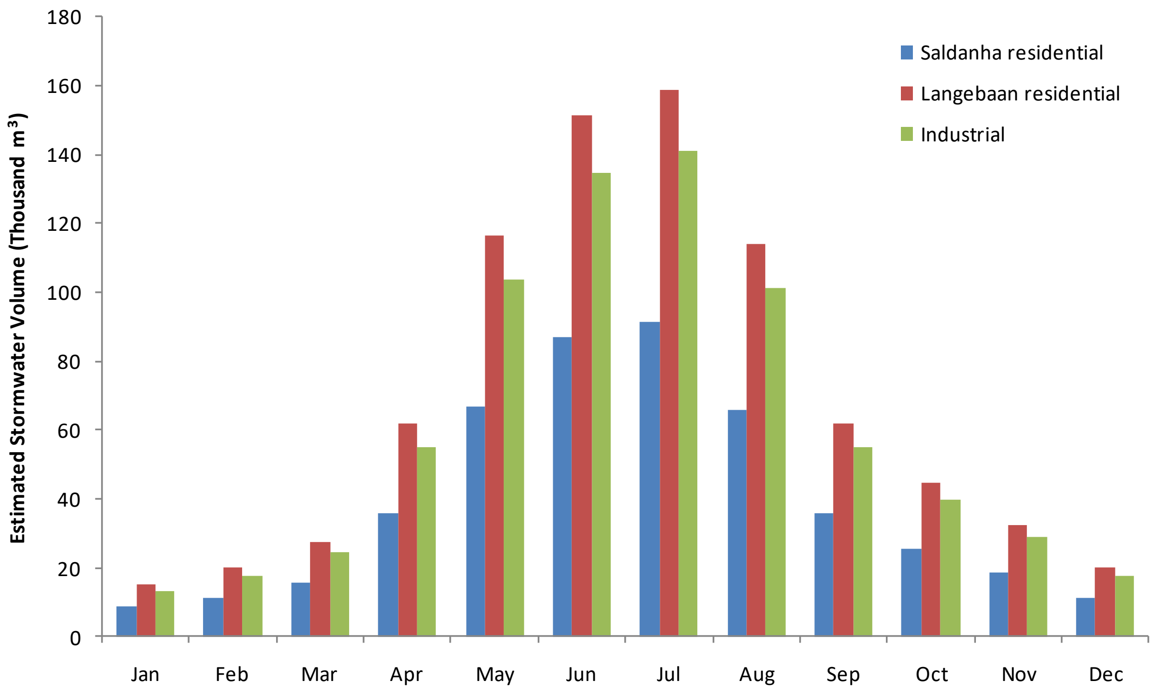


Figure 3.41. Monthly estimated storm water volume (m³) for Saldanha and Langebaan residential areas and industrial area. Note that runoff from the Port of Saldanha and ore terminal have been excluded as this is now reportedly all diverted to storm water evaporation ponds.

3.6.4.1 Stormwater management in Saldanha

There are approximately 15 outlets in the Saldanha Bay residential area. Historically, storm water from the Port of Saldanha and ore terminal was allowed to overflow into the Bay but most of this is now diverted to storm water evaporation ponds and any material settling in these ponds is trucked to a landfill site. The Saldanha Bay Municipality (SBM) intends to upgrade the existing stormwater infrastructure in the operational and non-operational areas within the boundaries of the Port of Saldanha. These upgrades include:

- Development of three new storm water retention ponds;
- Expansion and reshaping of existing storm water retention ponds;
- Development of a wastewater treatment facility,

- Upgrade of the storm water management infrastructure as well as maintenance of existing ones; and
- Associated activities.

These upgrades require Environmental Authorisation from the Western Cape Department of Environmental Affairs and Development Planning and the SBM commissioned NSOVO Environmental Consulting to conduct the Basic Assessment Process (NSOVO Environmental Consulting 2017). The draft BAR was published in January 2019 and the process is still ongoing.

Despite the efforts by the iron ore industry to reduce dust emission (refer to Section 3.3.1) and to divert and store stormwater in evaporation ponds, Saldanha Bay experiences frequent and considerable pollution, especially when the terminals are washed down with hosepipes (Figure 3.42). A report on the impacts of iron on the marine environment in Saldanha Bay was produced by Anchor Environmental Consultants in 2012 (Anchor Environmental Consultants 2012c). This report distinguished between the impacts of iron on the marine environment in its solid and hydrated state. Iron in the solid state affects organism by either smothering or through physical damage, thereby reducing the survival fitness of the affected organism. For example, high concentration of iron dust is known to inhibit photosynthesis in primary producers (Woolsey & Wilkinson 2007) and reduce fitness of intertidal organisms by changing the rate of heat absorption and reflective properties of their shells (Erasmus & De Villiers 1982). If iron is dissolved through chemical reactions with organic matter and oxygen, it becomes available to organisms in the marine environment. Dissolved iron is a micronutrient and shortage of this element can limit primary productivity in certain areas, while excess dissolved iron can result in unusual phytoplankton blooms. It has been shown that toxin levels in phytoplankton responsible for red tides also increase as a response to enhanced dissolved iron levels (He *et al.* 2009). Furthermore, accumulation of iron in tissue of bivalves can be harmful to humans when ingested and high levels of iron in tissue is recognised as an indicator for readily bioavailable iron (Rainbow 2002).

Figure 3.42. Pollution of Saldanha Bay by particulate iron carried by stormwater runoff (Source: Jaco Kotze, September 2014, Langebaan Rate Payers Association).

3.6.4.2 Stormwater management in Langebaan

Concerns and complaints have been publicly raised by the residents of Langebaan with regard to the poor stormwater management in Langebaan. Some parts of Langebaan are situated below the sea level and in the winter months, water becomes trapped on the roads in these areas. As a result, residents struggle to access their properties and to commute on flooded roads (Saldanha Bay Municipality 2014). Furthermore, the following concerns have been registered by the SBM:

- Deterioration/destructions of wetlands as well as canalisation of streams and rivers reduce the assimilative and dissipative capacity of the natural environment.
- Inadequate capacity of stormwater retention facilities east of Oosterwal Street.
- Impact of stormwater effluent containing pollutants from roads, private properties and businesses discharging into the Langebaan Lagoon.
- Lack of maintenance of conveyance systems with large sediment deposits.
- Impact on tourism market due to deteriorating aesthetic value.

As a result of these concerns, a Stormwater Management Master Plan was drafted and is amended as new issues arise (living document) (Saldanha Bay Municipality 2014). A Stormwater Management Plan is a necessary precursor to an action plan for improving stormwater management in Saldanha. However, the importance of drafting and implementing a policy for the maintenance of existing and future stormwater management structures has also been recognised. Langebaan currently has approximately 30 existing ponds of various sizes for the collection of stormwater and three additional large ponds are proposed (Note that these numbers may change as the Stormwater Master Plan is amended). There are about 20 outlets for stormwater that drain directly into the Langebaan Lagoon. Three types of structural stormwater controls are proposed for Langebaan, namely stormwater wet extended detention ponds, enhanced swale and litter/silt traps. The former will control the volume and quality of stormwater to be released into the Lagoon. The enhanced swale will encourage groundwater recharge and litter/silt traps will enable separation of refuse and larger debris at the entrance to chosen stormwater structures.

Stormwater litter traps

A collaboration between Sea Harvest and Saldanha Bay Municipality (SBM) saw the installation of a pilot stormwater litter trap in Saldanha Bay. This net, attached to the end of a stormwater outlet, traps any litter and debris suspended in the stormwater thereby preventing it from entering and polluting the Bay. The example shown in Figure 3.43 was the first installation of its type in within the Bay and unfortunately due to COVID 19 restrictions the project has not yet been expanded although, Sea Harvest and SBM hope to resume the initiative in the future (Sea Harvest Group Sustainability Manager, Kirshni Naidoo, *pers. comm.* 2020).

Figure 3.43. A stormwater litter trap installed in Saldanha Bay to prevent litter entering the Bay (Source: SADSTIA annual report 2019.

3.6.5 Fish processing plants

Three fishing companies currently discharge land-derived wastewater into Saldanha Bay: SA Lobster Exporters (Marine Products), Live Fish Tanks (West Coast) – Lusitania (CSIR 2002) and Sea Harvest. The latter is dealt with in more detail in below. The locations of the fish factory intake and discharge points are shown in Figure 3.44. Premier Fishing is currently in the process of re-commissioning and upgrading their fish processing plant.

SA Lobster Exporters (Oceana Lobster Saldanha) discharges seawater from their operations into Pepper Bay. The average monthly effluent volumes range from 40 to 60 000 m^3, and this water cycles through tanks where live lobsters are kept prior to packing (CSIR 2002). It was not possible to obtain more updated information or data for effluent volume and quality. No CWDP has been issued (Source: DEA: OC) and it is unknown whether this organisation is compliant with the revised General Discharge Limit.

Live Fish Tanks (West Coast)-Lusitania take up and release wash water from Pepper Bay. Neither discharge volume nor water quality is being monitored on a routine basis (CSIR 2002), but it is reported to be not markedly different from ambient seawater, as it basically cycles through tanks where live lobsters are kept prior to packaging (CSIR 2002). It is therefore unknown if this organisation is compliant with the revised General Discharge Limit and no CWDP has been issued (Source: DEA: OC). Furthermore, municipal water is released on a regular basis into the sea after cleaning of concrete slabs without cleaning agents (Live Fish Tanks, *pers. comm.* 2014). It must be determined how much freshwater is released into Small Bay by Live Fish Tanks (West Coast)-Lusitania in order to assess whether it significantly impacts the receiving environment.

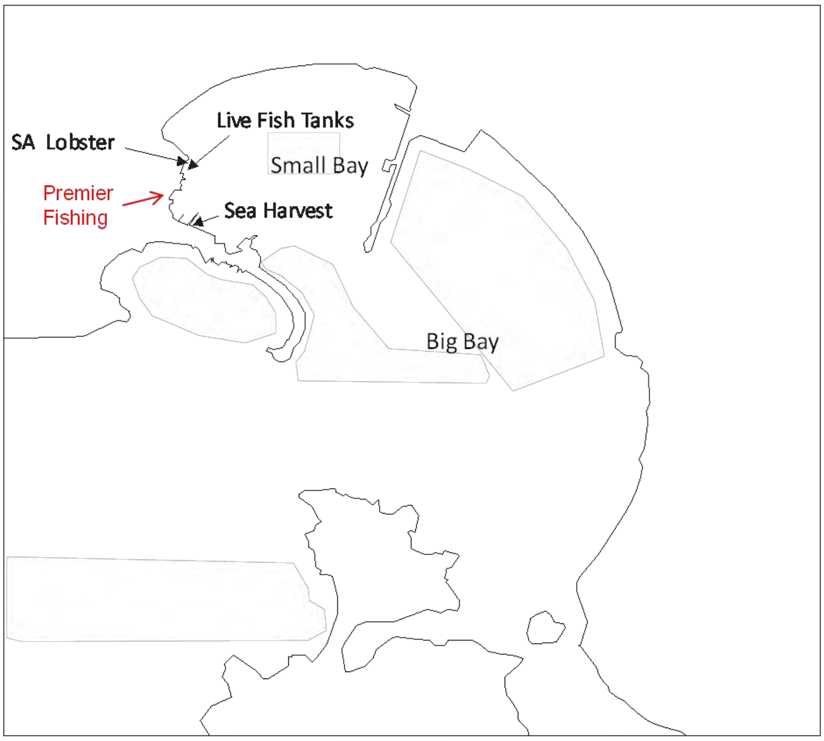


Figure 3.44. Location of seawater intakes and discharges for current and proposed seafood processing factories in Saldanha Bay. Current factories are indicated in black while the proposed Premier Fishing Fish Processing Plant is indicated in red.

3.6.5.1 Sea Harvest Fish Processing Plant

Sea Harvest is a predominantly demersal trawl fishing company which was established in 1964. The fish processing factory is situated near the base of the causeway to Marcus Island in Saldanha Bay and processes mostly hake (*Merlucius paradoxus* and *M. capensis*) into a variety of primary fish products including fillets, cutlets, steaks and loins.

Sea Harvest discharges large volumes of brackish effluent from the fish processing plant (FFP) into the sea. This includes seawater that has been used as wash-water as well as freshwater effluent originating from the fish processing. The effluent contains sea water, potable water, brine, food safe hygiene chemicals, annatto food dyes, sanitizers and fish particles smaller than 200 microns. In 2014, the plant was upgraded to ensure continuous operation and better solids handling capabilities (Sea Harvest, Site Engineer Nico Van Houwelingen, *pers. comm.* 2014) (Refer to AEC 2017 for a detailed description of the improvements made). The plant underwent further upgrading towards the end of 2018 in which the Offcuts and Trimmings plant was upgraded.

Sea Harvest requires high volumes of potable water for the processing of fish. With the implementation of water restrictions, Sea Harvest implemented a Reverse Osmosis (RO) plant (commissioned in April 2018) for the reclamation of potable water from seawater and potentially fish processing wastewater. The RO plant is expected to produce 42 m^3 per hour of potable water. The effluent consisting of RO brine, FFP factory effluent (i.e. process seawater is used to keep the floor drains flowing, to save potable water, to rinse ice off fish and to hose down floors etc.) and Added Value factory effluent from the DAF plant (10 m^3/h) will be diluted with sea water before discharge. The RO plant also requires Sea Harvest to abstract more seawater than before, thus the current abstraction quantity reported is approximately 6000 m^3.

<u>Coastal Waters Discharge Permit</u>

Sea Harvest Corporation (Pty) Ltd was issued with a Coastal Waters Discharge Permit (CWDP) in terms of Section 69 of the Integrated Coastal Management Act (2009) for discharge of effluent into Saldanha Bay on 26 June 2017. The effluent from the RO plant as described above was incorporated into the CWDP by means of an amendment issued by the DEA: O&C on 9 March 2018.

This amended CWDP authorised the disposal of industrial effluent into the Saldanha Bay harbour through an existing marine outfall and authorised Sea Harvest to dispose a maximum quantity of 420 480 m^3 per annum at a maximum daily discharge volume of 1152 m^3. Unfortunately, the Saldanha Bay Municipal Water Treatment Works does not have the capacity to process the effluent volume and type generated by this operation and therefore the effluent is disposed directly into the sea. Additionally, the CWDP stipulates that an independent external auditor should conduct sampling of the effluent bi-annually to verify the results obtained (measured at the end of pipe).

Anchor Environmental Consultants Pty (Ltd) was appointed by Sea Harvest to undertake scientific assessments required to meet the requirements of the permit conditions in 2018. The marine specialist study covered the following aspects:

1. Design of a monitoring programme to address the requirements of the CWDP;
2. Water column profile sampling;

3. Collection of sediment and macrofauna samples from all monitoring stations plus one control station (n = 8) and analysis of these samples for grain size, composition, percentage organic carbon and nitrogen, macrofauna species composition, abundance and biomass;
4. Dispersion modelling to establish the plume behaviour, assimilative capacity of the receiving environment and confirm a reasonable mixing zone;
5. Assessment of potential impact resulting from the effluent discharges on the receiving environment, the effectiveness of management strategies and actions to ensure compliance with the permit conditions, trends, status and changes in the environment related to the ecological health and designated beneficial uses of the system and whether the environmental quality limits are complied with in the area from the end of the mixing zone
6. Provision of recommendations on an effluent improvement plan to reduce the impacts of effluent in the marine environment.

The dispersion modelling study was completed by Anchor Environmental Consultants (Pty) in November 2018. It was discovered that the RO plant, was unexpectedly unable to process effluent from the Fish Processing Plant. Consequently, the CWDP needed to be amended to include the discharge of three effluent streams from the fish processing plant, the RO plant and added value factory. Sea Harvest Submitted an application for the amended CWDP in July 2018. The dispersion modelling study recommended that the effluent outfall be moved further offshore along the Government Jetty to facilitate effective mixing of the effluent (Figure 3.45).

Figure 3.45. Proposed outfall position at the end of the Government Jetty (33° 1'17.00"S; 17°57'6.76"E) for effluent originating at the fish processing plant, the reverse osmosis plant and the added value factory of Sea Harvest in Saldanha Bay.

On 11 June 2019 the Department of Public Works authorised Sea Harvest to proceed with the installation of the outfall pipeline on the Government Jetty and commenced with the installation of the pipeline on 15 August 2019. Sea Harvest received a draft Permit from the DEA in respect of the amendment applications made in 2018 on 16 August 2019 and the final amended permit was granted on 7 November 2019. The permit required that a new monitoring plan be developed and implemented for the new outfall location. Anchor Environmental launched monitoring equipment, a AquaTROLL 200 Conductivity-Temperature-Depth (CTD) instrument in the bay on 24 January 2020, to commence 12-month continuous monitoring at the edge of the recommended Mixing Zone (RMZ). Please refer to the 2018 annual monitoring report (AEC 2018) for details on the effluent quality and monitoring requirements of the 2017 CWDP, the outcomes of the preliminary environmental monitoring study as described above and recommendations of the dispersion modelling study.

The conditions of the amended 2019 authorisation include a daily discharge volume of 6000 m^3 per day and maximum prescribed limits of effluent constituents are given in Table 3.8. in addition, effluent discharge quantity must be metered by a continuous recording device or pump capacity prior to the discharge of effluent into the Bay. On 18 March 2020, on behalf of Sea Harvest, an Environmental Monitoring Programme and a Standard Operating Procedure for composite sampling was submitted to DEA by Anchor Environmental.

Table 3.8. Effluent Emissions Limits for constituents and Physico-Chemical Properties and the frequency of monitoring prior to discharge of into coastal waters.

Substance/Parameter	Limits from the date of issue of this permit	Limits from 1 December 2022 until expiry date of this permit	Frequency
Temperature	38°C	38°C	Weekly
pH	5.5-9.5	5.5-9.5	Weekly
Salinity	47 PSU	47 PSU	Weekly
Chemical Oxygen Demand	250 mg/l	250 mg/l	Monthly
Total Suspended solids	230 mg/l	63 mg/l	Monthly
Ammonia Nitrogen	100mg/l	20mg/l	Monthly

Effluent quantity and quality monitoring results

Effluent is discharged seven days a week with the exception of weekends extended by a public holiday on Monday and/or Friday. Effluent is also released on public holidays that fall on a Tuesday, Wednesday or Thursday in the early morning hours and after 8pm for sanitation purposes. No effluent volume monitoring data is available between January 2008 and 14 July 2013. Prior to 2015 effluent meter readings were not taken on public holidays and weekends. Although meter readings are now supposed to be taken daily, effluent volumes are most commonly not recorded on weekends. Furthermore, the flow metre malfunctioned relatively frequently and as a result fewer measurements were taken prior to the meter replacement in December 2018 (Table 3.9). Sea Harvest had 2066 operational days since 15 July 2013 and effluent readings were only taken 42% of the time. Between July 2018 and June 2019 effluent meter readings were recorded 48% of the time due to upgrades to the plant and occasionally faulty meter (Table 3.9).

In the most recent year, 2019/20, the effluent reading percentage showed a marked improvement increasing to 67% of the time. Despite this improvement, a higher compliance level would be desirable to ensure accurate monitoring of effluent volumes discharge. Sea Harvest discharge volumes only exceeded the allowable limit 22% of the time between July 2019 and June 2020, this is a significant improvement from the 70% exceedance that occurred in 2018/19 (Table 3.9)[6]. This high percentage of exceedance was anticipated due to the inability of the RO plant to process fish processing plant effluent upon installation. The reduced percentage of exceedance can be attributed to the change in daily effluent flow limit from 1152 m^3 per day to 6000 m^3 per day in the amended CWDP (7 November 2019).

It is noteworthy that DEA has issued a Draft CWDP to accommodate the changes that have occurred as a result of the severe drought in the region.

Table 3.9. Effluent volume monitoring efforts by Sea Harvest for various periods between 2004 and 2019. Note that no data is available for January 2008 – 14 July 2013 and this time period has been omitted from the calculations.

	January 2004 – December 2007	Since 15 July 2013	Since 26 June 2017	July 2018-June 2019	July 2019-June 2020
Number of operational days	1424	2066	659	321	342
Number of readings	704	859	316	153	229
Readings taken relative to number of operational days (%)	49%	42%	48%	48%	67%
Number of days where effluent volume was calculated [A]	571	780	305	146	227
Effluent volume calculated relative to number of operational days (%)	40%	38%	46%	45%	66%
Legal daily effluent volume limit (m^3) [B]	2000	3546	1152	1152	1152/6000
Exceedance of legal effluent volume limit (count)	225	137	134	102	49
Exceedance of legal effluent volume limit relative to number of operational days (%)	39%	18%	44%	70%	22%

A Note that effluent volume is calculated by subtracting the previous day's reading. This means that whenever there is a larger gap between readings or the meter has been malfunctioning, the effluent volume cannot be calculated.

B Note the amended CWDP was finalised on 7 November 2019 and therefore there are 2 legal limits for the 2019/2020 time period.

6 Effluent volume is calculated by subtracting the previous day's continuous meter reading. The first reading after a gap (public holiday or weekend) cannot be used to calculate an effluent volume for the day as the volume represents several days of effluent discharge. These data gaps do not occur in a reliable pattern throughout the dataset and are therefore not conducive for automated data processing. Average values for these gaps could therefore not be calculated. Non-compliance with the maximum daily discharge limit of 1152 m^3 may therefore be over-estimated. The compliance rating would become more reliable if meter reading is conducted over the weekends.

The highest calculated average daily effluent discharge volume[7] reached was 5 108 m^3 in 2006/7, the lowest average daily volume recorded was 248 m^3 in 2016/17, however it is also noteworthy that the latter time period had the lowest percentage of operational days for which effluent volumes were monitored. Due to the additional effluent produced by the RO plant, average daily discharge volume increased to be slightly above the legal limit (1162 m^3) for 2018/19. The daily average for 2019/20 was 767 m^3 which is well below the daily maximum volume for both the old (1152 m^3/day) and the newly amended CWDP (6000 m^3/day), in addition this period had the highest recorded percentage of operational days for which effluent volumes were monitored (93%, Figure 3.46).

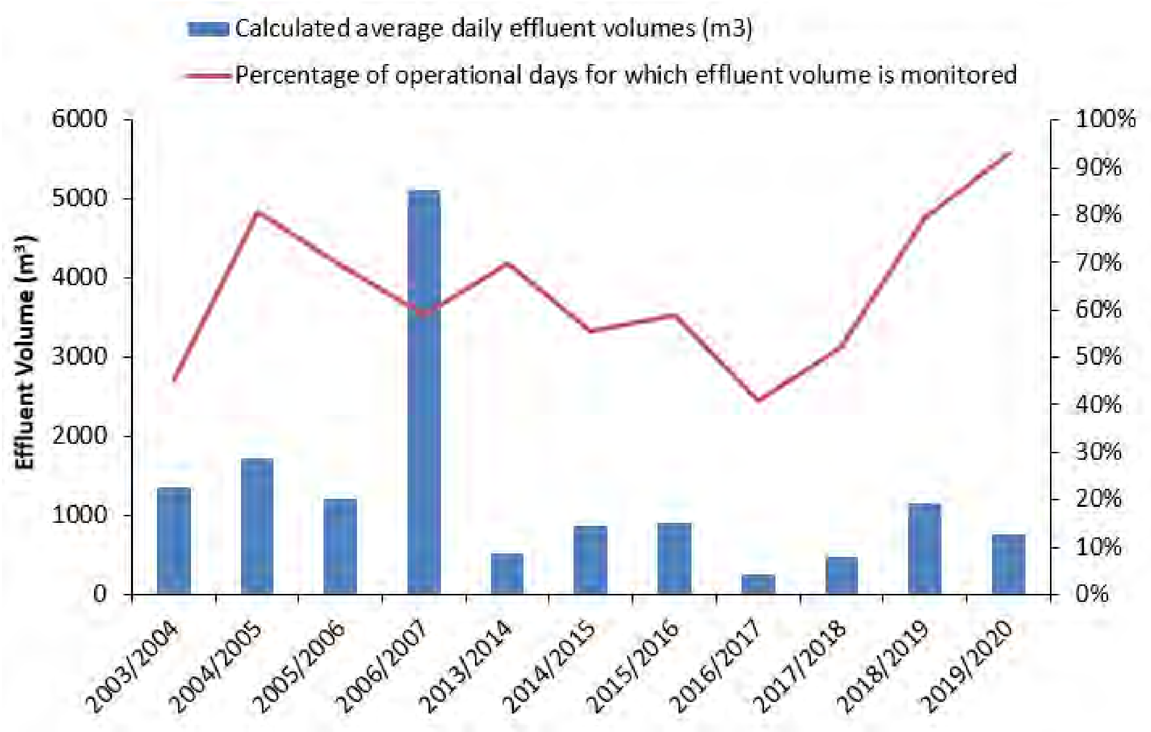


Figure 3.46. **Calculated average daily effluent volumes discharged into Small Bay per year by Sea Harvest from July 2004 - June 2020 and the percentage of operational days for which effluent volume is monitored. Data was not available for the period May 2007 – August 2013. (Source: Meryl Lee Edwards, Environmental Intern at Sea Harvest fish Processing Plant).**

Estimated annual fish processing effluent volumes discharged into Small Bay between July 2003 and June 2020 by Sea Harvest is shown in Figure 3.47 and is compared to the prescribed annual effluent limits over time. No data is available for the period April 2007 to December 2012. Overall, measurements show that effluent volumes discharged into Small Bay have fluctuated substantially since 2004. During the period of August 2006 to November 2007, the volume of effluent disposed by Sea Harvest increased peaking at unusually high levels. It is not clear why this increase occurred, but data reporting and environmental monitoring at Sea Harvest had suffered irregularities at the time

[7] Average daily effluent volume was calculated by dividing the measured annual volume by the number of measurements taken, with a correction applied to account for weekends.

due to high staff turnover (Sea Harvest, F. Hickley *pers. comm.*). It can be concluded with reasonable confidence that the annual effluent volume has not exceeded the prescribed limit since 2013. Prior to the issuance of the first CWDP (June 2017) effluent quality at the pipe end was compared to the General Discharge Limits of the General and Special Standard (most recent amendment constitutes Government Notice No. 36820 –6 September 2013) promulgated under the NWA. The 2018/2019 data shows that Sea Harvest was able to meet the annual limit of 420 480 m^3 as specified in the CWDP conditions, despite exceedance of the daily limit of 1152 m^3 70% of the time. Similarly, data for 2019/2020 show that the annual effluent readings are well below the prescribed limit[8] of 157 0608 m^3 per year (Figure 3.47).

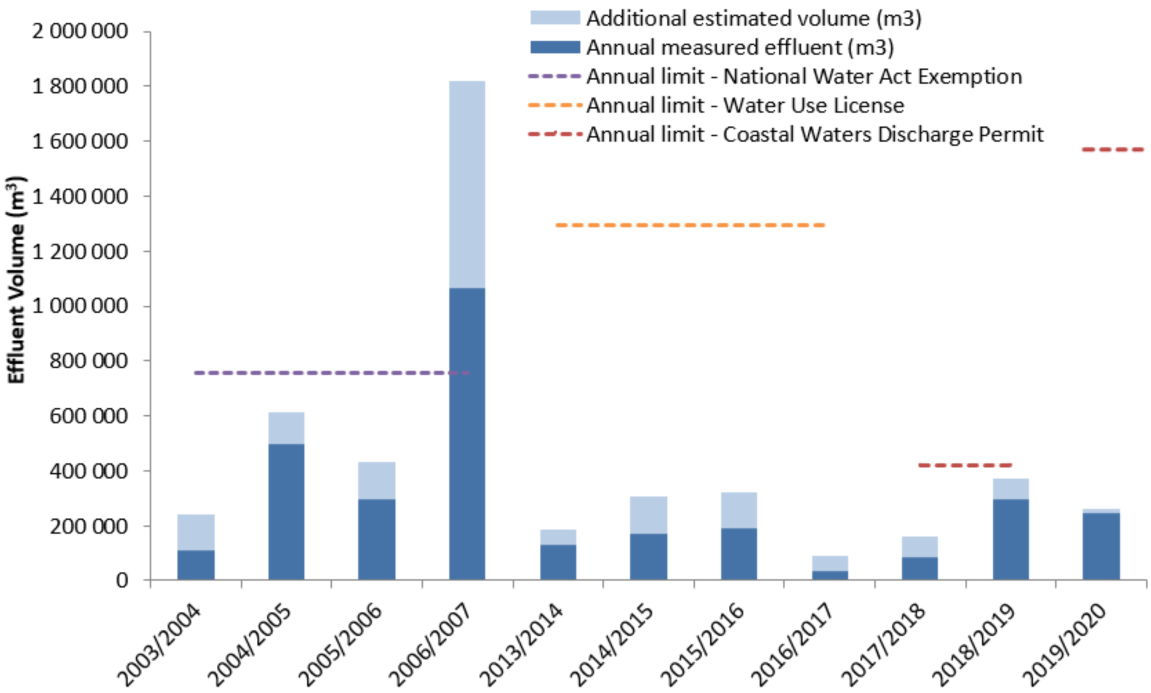


Figure 3.47. Estimated Fresh fish processing effluent volume discharged into Small Bay per year by Sea Harvest from July 2004 - June 2020. Data was not available for the period May 2007 – August 2013. The legal annual effluent limits are indicated as dashed lines. (Source: Meryl Lee Edwards, Environmental Intern at Sea Harvest fish Processing Plant).

Trends in TSS since 2010 suggest that concentrations fluctuate over time, although it appears that peak concentrations are decreasing in magnitude with the exception of one extreme value in June 2018 (870 mg/L, Figure 3.48). Prior to the issuance of the first CWDP, TSS concentrations were significantly higher than the limit prescribed by the revised General Discharge Limit of 25 mg/L, with compliance only reached once in October 2013 (14 mg/L, Figure 3.48) however, the CWDP issued on 26 June 2017, and subsequent amendments, specifies a legal limit of 230 mg/L. Since then, TSS concentration in the effluent only exceeded the legal limit eight times, which means that Sea Harvest

[8] This value is a combination of the number of days during which daily discharge limit was at 1152 m^3 and the number of days at the newly amended daily limit (6000 m^3)

is compliant 79% of the time. The average TSS levels for 2019/2020 were 56.5 ± 96.8 mg/L, an improvement over the previous year (2018/2019) during which the average was 91.1 ± 71.3 mg/L and the limit was only exceeded once in each year reaching 421 mg/L in April 2019 and peaking at 355 mg/L in August 2019.

Sea Harvest was required to comply with the revised General Discharge Limit for ammonia nitrogen of 6 mg/L until the CWDP was issued on 26 June 2017. This limit was very conservative considering that the water quality guidelines for the coastal environment specified a target of the same value (DAFF 1995) (note that since then revised guidelines for the marine environment have been published by DEA, refer to Section 3.6.1 for more details). This limit was therefore exceeded 95% of the time (Figure 3.49). Notwithstanding, ammonia levels have been unacceptably high in the past, reaching a maximum of 474 mg/L in September 2012. Overall, ammonia nitrogen has been decreasing since then due to a change in sanitising protocols. The CWDP, and subsequent amendments, specifies a legal limit of 100 mg/L, which has not been exceeded since the permit was issued.

Ammonia nitrogen concentration averaged 17 ± 14 mg/L in 2018/19, which is a significant improvement when compared to the period 2017/18 (34 ± 27 mg/L), levels in 2019/2020 further improved with average concentrations of 7 ± 8 mg/L. Changes in cleaning protocols at the fish processing facility were implemented in 2018/19 where screens are sprayed every 30 minutes to ensure that no rotting occurs on the screens. This improved effluent management practice at the FFP Offcuts and Trimmings Plant could have contributed to the decreased ammonia nitrogen levels. Additional effluent from the RO plant would also dilute the ammonia nitrogen concentrations in the effluent.

Fish processing involves the use of freshwater and sea water and therefore effluent salinity (ppt) is lower than what is expected in the receiving environment as seen prior to July 2017 (Figure 3.50). It is, however, evident that salinity increased between January 2015 and June 2017 (see the 2015 State of Saldanha Bay and Langebaan Lagoon for conductivity (mS/m) trends prior to January 2015), approaching levels expected in the marine receiving environment (35 ppt). This is likely due to the increasing use of seawater for fish processing over time. Since the implementation of the RO plant in April 2018, and the subsequent addition of brine to the discharge effluent, salinity readings exceeded the limit specified in the CWDP (37 ppt) 56% of the time. The average salinity was 40 ± 17 ppt with a maximum salinity of 93.8 ppt measured on 27 August 2019. Since the revision of the salinity limit to 47 ppt in the amended CWDP of 7 November 2019, the salinity readings have improved substantial and have only been non-compliant on two occasions (9% of the time), with an average of 29 ± 15 ppt and a maximum of 49.5 ppt recording on 3 December 2019.

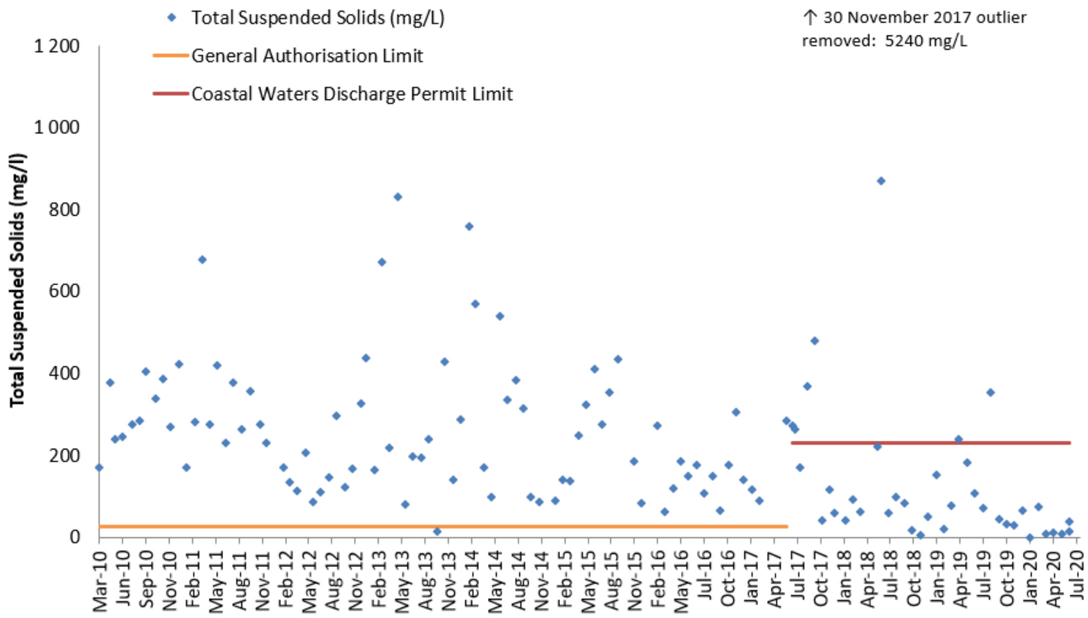


Figure 3.48. **Monthly trends in total suspended solids (TSS) (mg/L) in the effluent discharged from the Sea Harvest fresh fish processing (FFP) plant into Small Bay in the period March 2010 to June 2020 (concentration measured at the end of pipe). No data is available between April and June 2017. The lines indicates the limit prescribed by the General Discharge Limit of the revised General and Special Standard (orange line, 25 mg/L) (Government Notice No.36820 – 6 September 2013) and subsequent Coastal Waters Discharge Permits (depicted as the red line, 230 mg/L). (Source: Meryl Lee Edwards, Environmental Intern at Sea Harvest fish Processing Plant).**

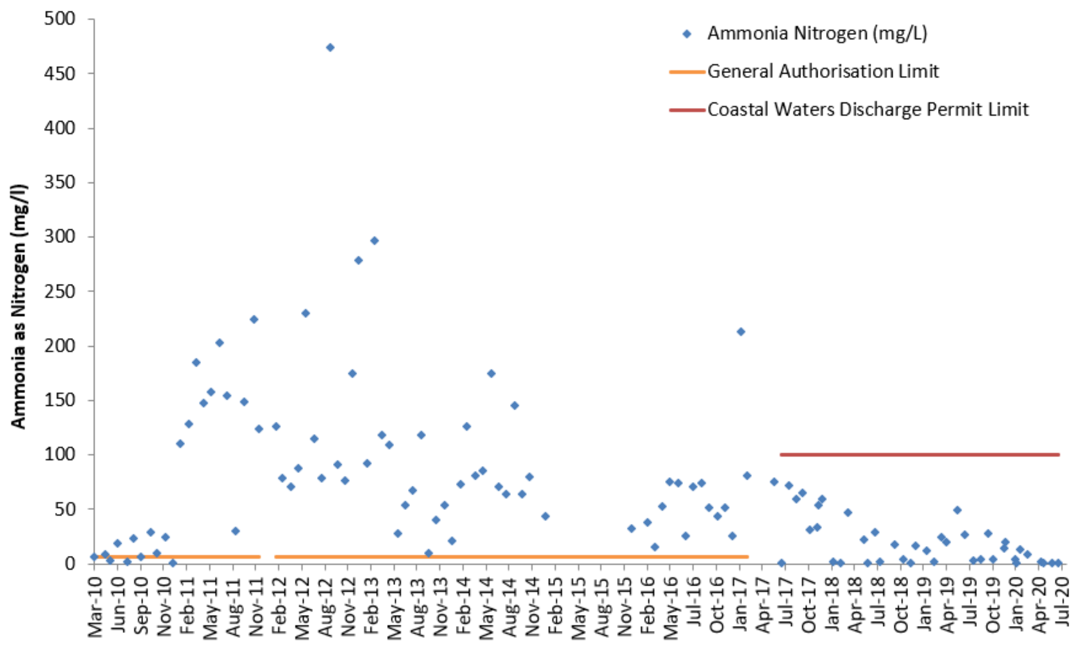


Figure 3.49. **Monthly trends in ammonia nitrogen (mg/L) in the effluent discharged from the Sea Harvest fresh fish processing (FFP) plant into Small Bay in the period March 2010 to June 2020 (concentration measured at the end of pipe). No data is available between April and June 2017. The lines indicates the limit prescribed by the General Discharge Limit of the revised General and Special Standard (orange line, 25 mg/L) (Government Notice No.36820 – 6 September 2013) and subsequent Coastal Waters Discharge Permits (depicted as the red line, 230 mg/L). (Source: Meryl Lee Edwards, Environmental Intern at Sea Harvest fish Processing Plant).**

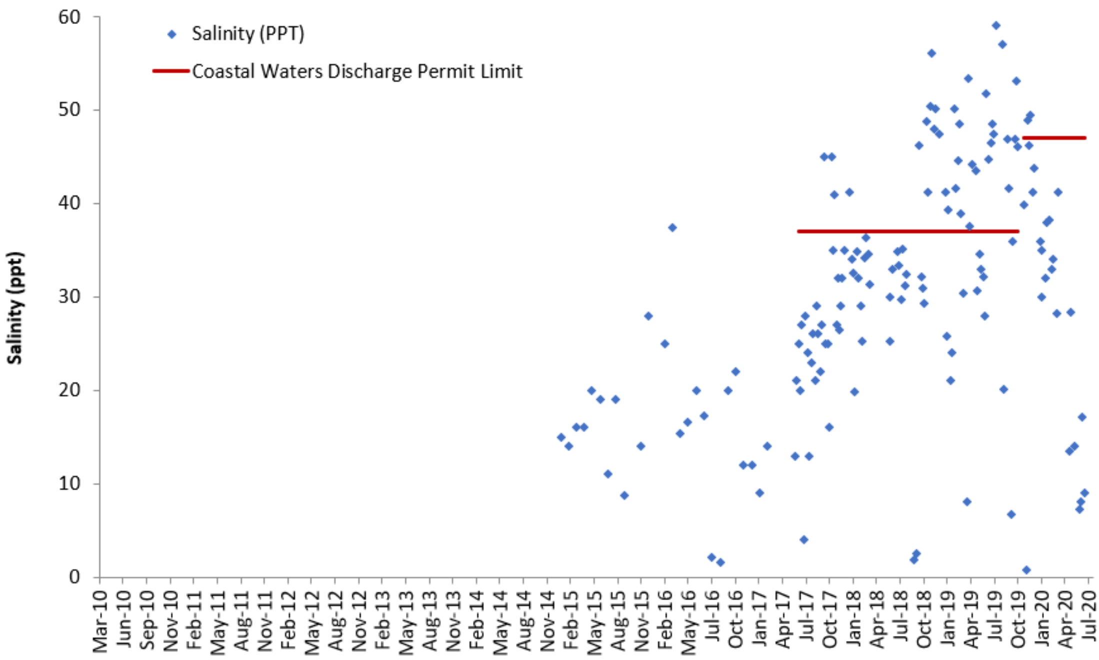


Figure 3.50. Monthly salinity (ppt) trends in the effluent discharged from the Sea Harvest fresh fish processing (FFP) plant into Small Bay in the period January 2015 to June 2020 (concentration measured at the end of pipe). No data is available between April and June 2017. Sea Harvest was granted a Coastal Waters Discharge Permit on 26 June 2017, which prescribes a limit of 37 ppt (depicted as the red line), this was amended to 47 ppt. (Source: Meryl Lee Edwards, Environmental Intern at Sea Harvest fish Processing Plant).

Sea Harvest has been measuring Chemical Oxygen Demand (COD) since November 2015. Although COD has consistently been significantly higher than the prescribed limit there appears to be a trend of general improvement. In 2017/2028 COD exceeded the limit 100% of the time with an average of 1 121 ± 601 mg/L, in 2018/2019 non-compliance dropped to 75% with an average of 545 ± 289 mg/L and since July 2019 non-compliance is down to 57% of the time with an average of 383 ± 297 mg/L. The highest value was recorded in June 2018 with 2 957 mg/L. The results suggest that a large amount of oxygen is required to breakdown the organic waste in the effluent. Despite the overall improvement, Sea Harvest has not been able to meet the requirements of the CWDP (<250 mg/L) under current effluent treatment methods (Figure 3.51). Improving COD to acceptable levels will reduce risks of anoxic conditions developing in the receiving marine environment, especially in Small Bay which is considered a sheltered environment with limited mixing capacity. Sea Harvest is currently investigating the installation of an AFM filtration system to reduce the COD in the effluent being discharged to sea.

Oil and grease were monitored monthly between March and December 2015 (Figure 3.52). Values always exceeded the General Authorisation limit of 2.5 mg/L, averaging 27 ± 25 mg/L and reaching a maximum of 91 mg/L in September 2015. The CWDP requires that Sea Harvest's effluent contains less than 10 mg/L of oil and grease and effluent monitoring was therefore reinstated in June 2017. In 2017/2028 Oil and grease values exceeded the limit 58% of the time with an average of 98 ± 247 mg/L, in 2018/2019 non-compliance increased to 67% with however the average values improved to 36 ± 35 mg/L. Furthermore, a reading taken in July 2018 measured 17 472 mg of oil and grease per litre. This result is not considered reliable and was removed from the monitoring results. In contrast, during the past year (2019/20) discharge concentrations have improved substantially, with non-compliance down to only 9% and an average well below the required 10 mg/L limit (2 ± 5 mg/L).

Sea Harvest monitored pH between March 2010 and December 2014. The current CWDP requires the monitoring of pH, which was resumed in July 2017. The results from 2017-20 demonstrate that the effluent has generally been compliant with the legal limit bracket (5.5 - 9.5) with the exception of one occasion when pH measured 4.8 in July 2017 and on four occasions in June and July 2020 where readings were greater than 9.5.

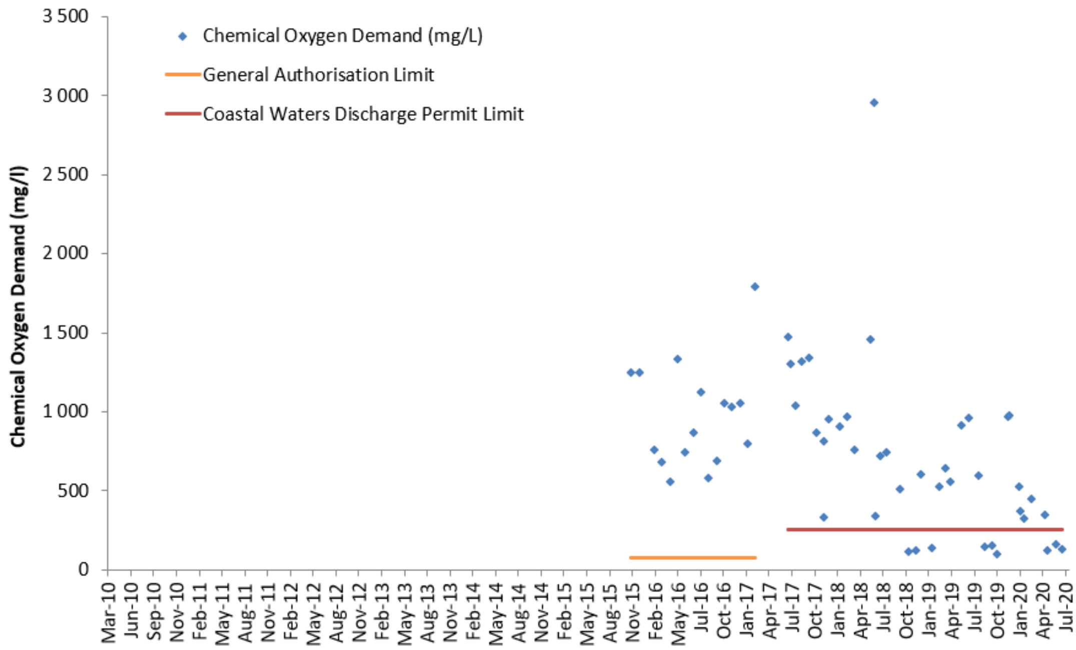


Figure 3.51. **Monthly chemical oxygen demand (COD) trends in the effluent discharged from the Sea Harvest fresh fish processing (FFP) plant into Small Bay in the period November 2015 to June 2020 (concentration measured at the end of pipe). The lines indicates the limit prescribed by the General Discharge Limit of the revised General and Special Standard (orange line, 25 mg/L) (Government Notice No.36820 – 6 September 2013) and subsequent Coastal Waters Discharge Permits (depicted as the red line, 230 mg/L). (Source: Meryl Lee Edwards, Environmental Intern at Sea Harvest fish Processing Plant).**

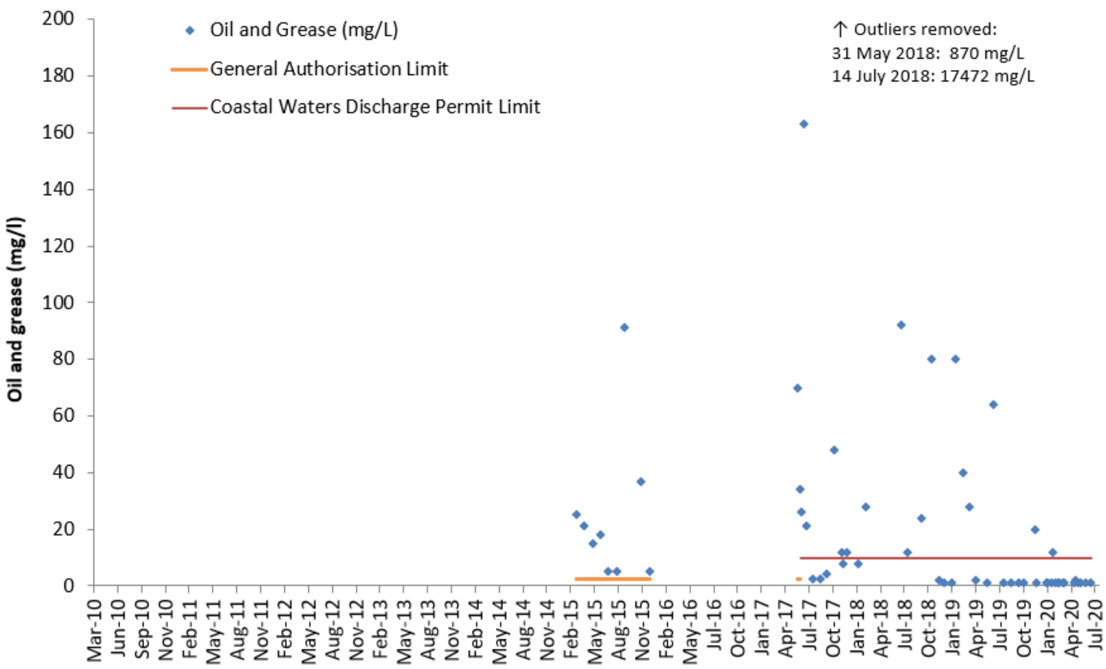


Figure 3.52 Monthly trends of oil and grease (mg/L) in the effluent discharged from the Sea Harvest fresh fish processing (FFP) plant into Small Bay in the period March to December 2015 and from June 2017 to June 2020 (concentration measured at the end of pipe). The lines indicates the limit prescribed by the General Discharge Limit of the revised General and Special Standard (orange line, 25 mg/L) (Government Notice No.36820 – 6 September 2013) and subsequent Coastal Waters Discharge Permits (depicted as the red line, 230 mg/L). (Source: Meryl Lee Edwards, Environmental Intern at Sea Harvest fish Processing Plant).

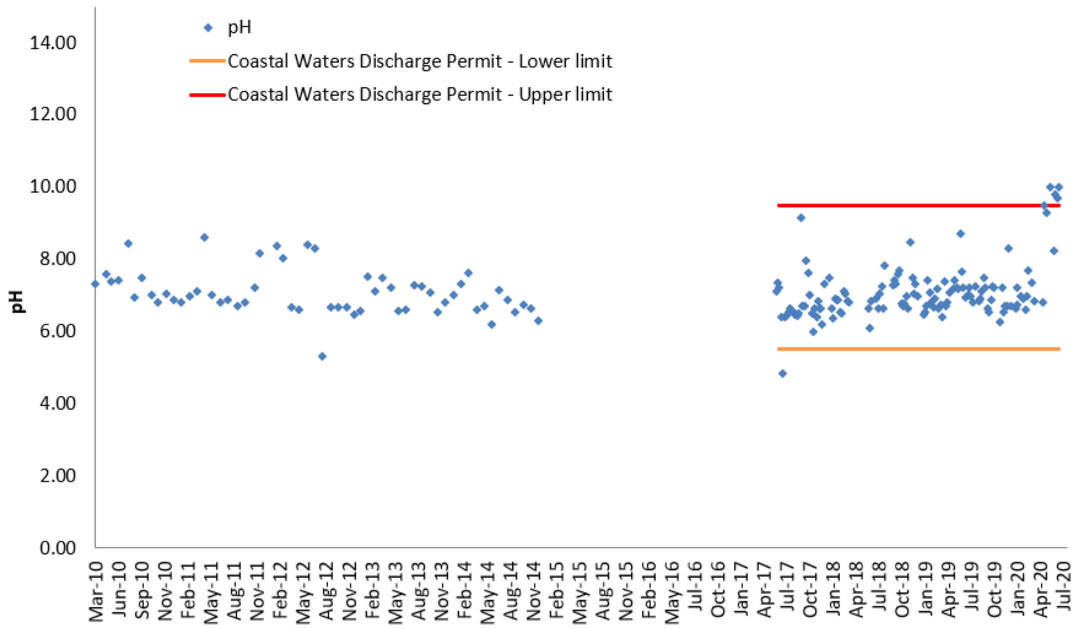


Figure 3.53. Monthly trends of the pH measured in the effluent discharged from the Sea Harvest fresh fish processing (FFP) plant into Small Bay in the period March 2010 to December 2014 and from June 2017 to June 2020. The red dashed lines indicate the lower (5.5) and upper (9.5) limits prescribed by the Coastal Waters Discharge Permit dated 26 June 2017. (Source: Meryl Lee Edwards, Environmental Intern at Sea Harvest fish Processing Plant).

With the ongoing drought in the Western Cape, Sea Harvest reclaims potable water by means of a Reverse Osmosis plant with the intention to save municipal water and to improve effluent quality (Frank Hickley, Sea Harvest *pers. comm*., 2018). Sea Harvest is committed to meeting effluent quality thresholds and environmental monitoring requirements as stipulated in the CWDP. However, the effluent at the Sea Harvest Fish Processing Plant is currently not treated adequately to ensure minimum impact to the receiving environment. The fish processing facility is still failing to comply with the chemical oxygen demand and oil and grease concentrations prescribed in the CWDP, despite the fact that both of these have improved in 2019/2020 relative to the two preceding years. Sea Harvest has been meeting the pH range prescribed in the CWDP though recent high readings recorded in June and July 2020 may warrant consideration if they continue. The effluent produced by the RO plant has increased the salinity of the overall effluent dramatically and CWDP requirements were exceeded roughly 50% of the time until the limit was increased in the amended CWDP subsequently salinity readings are compliant 91% of the time. In addition, significant improvements have been observed in terms of the ammonia nitrogen and total suspended solids concentration and the current CWDP limits are being met.

3.6.5.2 Re-commissioning of the Premier Fishing fish processing plant

Southern Seas Fishing (now trading as Premier Fishing) previously discharged wastewater into the Bay but closed its factories in 2008 after being operational for 50 years. Premier Fishing is in the process of re-commissioning and upgrading the existing fishmeal and fish oil processing plant situated in Pepper Bay, the western side of Saldanha Bay. EA was granted in June 2013 and the Atmospheric Emission Licence was also approved in April 2014 but has been appealed. An application for a CWDP in terms of ICMA has been submitted to the Department of Environmental Affairs: Oceans and Coasts Branch (DEA: OC) for the discharge of cooling water containing condensate from the plant’s scrubber to the sea. The permit application was provided for public review in Appendix H of the Revised Final EIA Report for the project (SRK Report 431676/10). On 24 April 2014 DEA: OC requested additional information for the CWDP application and that the application is subjected to another round of public participation. No Coastal Waters Discharge Permit has since been issued and construction/operation has not commenced (Department of Environmental Affairs, Branch Oceans and Coast 2017).

3.7 Fisheries

There is a long history of fishing within the Bay and Lagoon, with commercial exploitation beginning in the 1600s (Thompson 1913). Presently, there is a traditional net fishery that targets mullet (or harders), while white stumpnose, white steenbras, silver kob, elf, steentjie, yellowtail and smooth hound shark support large shore angling, as well as recreational and commercial boat line-fisheries. These fisheries contribute significantly to the tourism appeal and regional economy of Saldanha Bay and Langebaan.

The two most important species in the fisheries in Saldanha Langebaan are white stumpnose that are caught by commercial and recreational line fishers, and harders that are commercially harvested by approximately 16 gill net permit holders. The total annual catch of white stumpnose by commercial

(31% of total) and recreational line fishers (boat: 56% and shore 13%) was estimated at 125.3 tonnes for the 2006-2008 period (Parker et al. 2017).

Assuming a selling price of R40/kg, the landed catch value of the commercial sector's catch of 39 tonnes is approximately R 1.6 million; the value of the recreational fisheries in the region has not yet been quantified, but undoubtedly exceeds the landed catch value of the commercial fisheries. Commercial white stumpnose catch-per-unit-effort has declined considerably in the last 15 years, whilst recruitment has also crashed (Figure 3.54). This Saldanha - Langebaan white stumpnose stock is clearly under threat and more stringent catch control measures are required.

The commercial gill net fishery in Saldanha Langebaan reports an average of approximately 20 tonnes per year with a landed catch value of around R 200 000 (DAFF, unpublished data). This stock also appears to be under pressure with a notable decline in the average size of harders landed in both Saldanha and Langebaan between 1999 and 2012 (see Chapter 11 for more details on this). The observed shift towards a smaller size class of harders in catches does suggest that growth overfishing is occurring and further increases in fishing pressure will probably lead to declines in overall yield (catch in terms of mass) from the fishery. There has been considerable pressure to open the restricted Zone B within the Langebaan MPA to all commercial gill net fisher's resident in Saldanha and Langebaan. Permitting increased fishing effort within Zone B would drive further declines in average harder size which has a disproportionate negative impact on the reproductive output of the stock, as large female fish spawn exponentially more eggs as the grow. This would negatively impact the productivity of the harder stock in the Saldanha-Langebaan system and may lead to further long-term declines in the overall fishery catch (See Chapter 11).

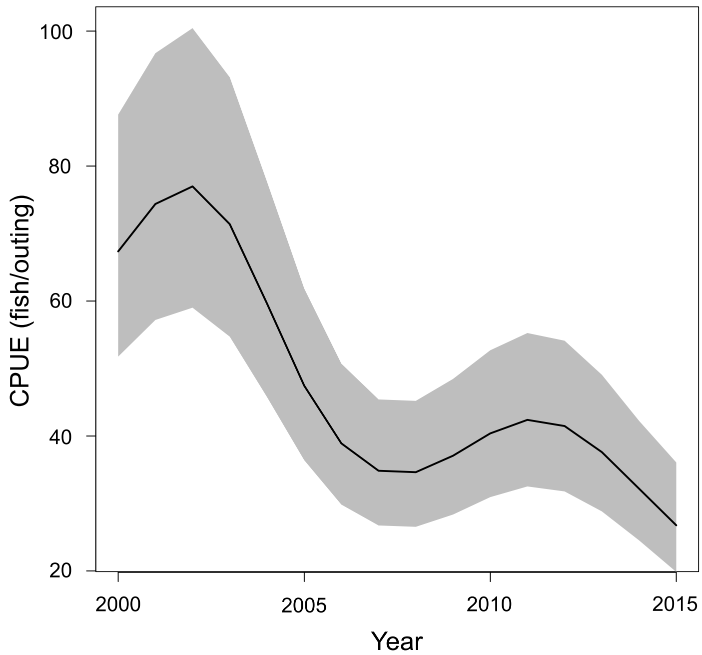


Figure 3.54. Annual Catch Per Unit Effort (CPUE) estimates (±95% Confidence Interval) of white stumpnose derived from commercial boat catches logged in the National Marine Linefish System (NMLS) database (Source: Parker et al. 2017).

3.8 Marine aquaculture

The Department of Environment, Forestry and Fisheries (DEFF, previously known as DAFF) is driving accelerated development of the aquaculture sector in South Africa with the aim to create jobs for marginalised coastal communities and to contribute towards food security and national income. The development of the aquacultures sector is considered an important opportunity that can contribute to job creation and the local economy, and was therefore identified as a key priority of Operation Phakisa (Section 3.2).

Saldanha Bay is a highly productive marine environment and constitutes the only natural sheltered embayment in South Africa (Stenton-Dozey et al. 2001). These favourable conditions have facilitated the establishment of an aquaculture industry in the Bay. In January 2018 the then Department of Agriculture, Forestry and Fisheries was granted Environmental Authorisation to establish a sea-based Aquaculture Development Zone (ADZ) in Saldanha Bay and expand the total area available for aquaculture in the Bay to 884 ha, which is located within four precincts (Small Bay, Big Bay North, Outer Bay North and South, Figure 3.55). In 2018, it was reported that of the new established area, 151 ha was being actively farmed. By the end of December 2019, approximately 36% of the ADZ had been leased, but less than 60% of the actively leased area was being utilised and this value is constantly changing as new leases are being granted, new farms start, current lease holders expand their areas, or alternatively shrink in size, based on economic factors. As of March 2020, 28 companies within the Saldanha Bay ADZ were registered on the Marine Aquaculture Right Register, of which only 15 companies were actively operational. More details on progress made in establishing the ADZ is included in Section 3.8.1 below.

3.8.1 Saldanha Bay Aquaculture Development Zone

The aim of establishing the ADZ by DEFF is to (a) encourage investor and consumer confidence (b) create incentives for industry development (c) provide marine aquaculture services, (d) manage the risks associated with aquaculture; and to provide skills development and employment for coastal communities.

The ADZ project triggered activities listed in terms of Listing Notice 1 of the EIA Regulations, 2014, required a Basic Assessment. SRK Consulting (Pty) Ltd. (SRK) was appointed as the independent consultant to develop a framework for the Saldanha Bay ADZ and undertake the Basic Assessment. The competent authority (the then DEA) granted three separate Environmental Authorisations (EAs) for aquaculture in the Bay to the then DAFF, Southern Cross Farm (combined application but separate EAs) and the Molapong Aquaculture farm on 8 January 2018. Four appeals to the EA were received from interested and affected parties. The Appeal Decision by the then DEA was issued on 7 June 2018 and stated that three appeals against the ADZ were overturned by the then Minister and that the EA was upheld. Subsequently, the DEFF appointed an Environmental Control Officer and set up two committees: (1) a Consultative Forum (public and industry forum), which has approximately 137 members and meets every three months, and (2) the Aquaculture Management Committee (AMC) (government committee) which meets every two months, to ensure that the implementation of the ADZ is in line with the requirements specified in the EA and EMPr. DEFF published a "Guideline for Bivalve Production Estimates for the Saldanha Bay Aquaculture Development Zone". This document is designed to ensure that the production per annum, as specified in the EA, is upheld by the ADZ operators. Coupled with environmental monitoring, adherence to the authorised tonnages should facilitate adaptive environmental management in the ADZ as a whole. DEFF outsourced the development of a marine monitoring programme referred to as the Sampling Plan, as well as a dispersion model for the finfish farming (more details on these two studies are provided in Section 3.8.1.1 below in in the 2019 SOB Report). The baseline sample collection was undertaken in the beginning of 2019 and the annual redox survey of the Saldanha Bay ADZ was conducted during the annual Saldanha State of the Bay survey (April 2020) and are summarised below. Over and above, DEFF specialist scientists are conducting environmental monitoring which includes a rapid synoptic survey of oxygen and nutrient levels in the Bay as well as long term monitoring undertaken by an independent service provider.

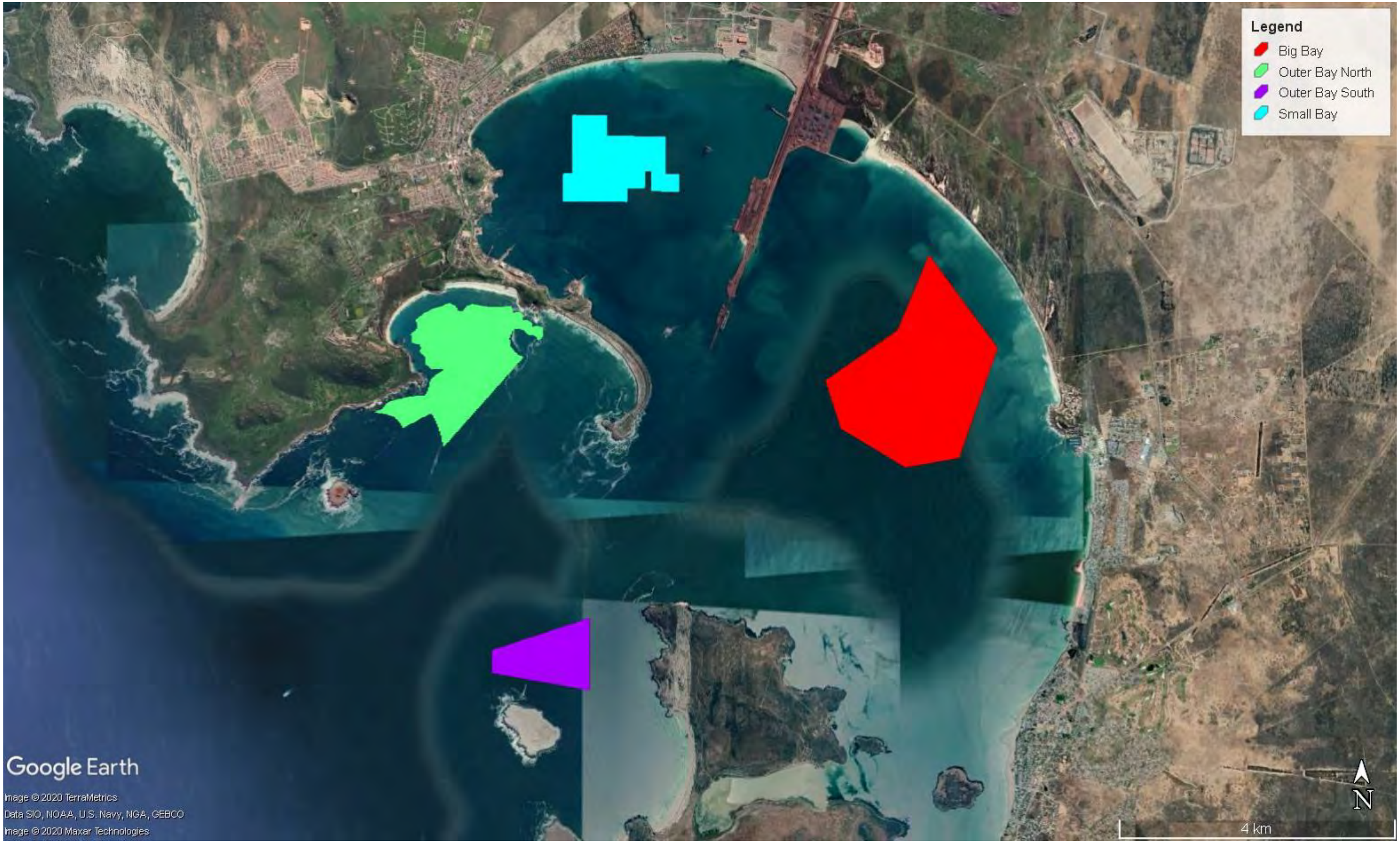


Figure 3.55. **Map showing the four precincts that make up the entire ADZ area (884 ha). Approximately 36% of this area has been leased to the aquaculture sector with less than 60% of the actively leased area currently being utilised (DEFF, *pers. comm.* 2020).**

Various guidelines and protocols have been developed for managing the ADZ, these include an ADZ Entanglement Guideline (April 2020), Compliance Strategy (April 2020) and Specific Emergency Response Protocol for potential incidents (May 2020). Additionally, DEFF has engaged with the Saldanha Bay Water Quality Forum Trust to combine sampling efforts. The mussel industry is in the process of undertaking a Fisheries Improvement programme to improve overall fishing practices in the Rope Grown Mussel Industry, enhance the management of the fishery, establish critical partnerships, generate community support to inspire change in other fisheries in South Africa, and improve the accessibility of the Marine Stewardship Council standard in South Africa and other countries in the global South Africa.

3.8.1.1 Impacts modelling and monitoring for finfish cage culture

The impacts of fish farming on the marine environment are generally well studied globally. One of the primary impacts of mariculture cage farming is that untreated wastes resulting mainly from uneaten food and faeces of fish in sea cages are discharged directly into the sea and represent a potentially significant source of nutrients (Brooks *et al.* 2002, Staniford 2002a). Studies have documented increased dissolved nutrients and particular components (POC and PON) both below, and in plumes downstream, of fish cages (Pitta *et al.* 2005). These wastes impact both on the benthic environment and on the water column. Sediments and benthic invertebrate communities under fish farms usually show chemical, physical and biological changes attributable to nutrient loading. Nutrient enrichment and resulting eutrophication of sediments under fish cages is regarded as a serious issue in some areas (Staniford 2002b). Nutrient loading of the water column, along with the reduction of dissolved O_2 concentrations, as a result of fish cages has been implicated in conditions that stimulate harmful algal blooms, which pose a threat human health and shellfish mariculture operations (Gowen & Ezzi 1992, Navarro 2000, Ruiz 2001, all cited in Staniford 2002a).

As such, the then DAFF commissioned a far-field dispersion modelling study of the proposed finfish production as part of the Environmental Impact Assessment (EIA) for the Saldanha ADZ. The aim was to provide data to inform future monitoring and monitoring stations for further aquaculture expansion in the bay, specifically for finfish. This work was carried out by PRDW in 2017 in association with Lwandle Technologies. Far-field modelling was considered prudent to assess cumulative impacts of the whole aquaculture development zone, and to differentiate between various sources of nutrient input within the bay (PRDW 2017).

A 3-dimesional coupled wave, hydrodynamic, dispersion and ecological Mike-21 model assessed three forms of nutrient (dissolved nitrogen compounds) loading typical of finish cages (fish excretion, fish faeces particulates and uneaten food pellet particulates), using environmental thresholds that were developed for both dissolved nitrogen in the water column and sedimented particulate organic matter (PRDW 2017). A total of 5 000 t/year production across the three farm areas in Saldanha Bay was modelled (modelled annual productions of 2 000 t at the Big Bay precinct and 1 500 t each of the Outer Bay sites). A limitation of this study was the lack of near-field assessment (near-field impacts closer than 500 m from the farms were not identified or quantified) as they require validation from field data.

PRDW (2017) far-field modelling results showed that:

- dissolved nitrogen concentrations attributable to the fish farms will be low compared to the identified effect threshold of 0.021 mg/l, have no toxicity effects on biota and only possibly minor effects on nitrate-nitrogen based phytoplankton productivity in the immediate vicinity of the Big Bay precinct;
- while nitrogen rich particulate matter will accumulate at various locations within Saldanha Bay and in Langebaan Lagoon, the thickness of the deposited layers will be well below the 5 mm threshold set, and there is no evidence in the model results of systematic build-up over time; and,
- based on a Food Conversion Ratio (FCR) of 1.4, which was less conservative than that applied in the Basic Assessment Report (BAR) assessment, the nitrate nitrogen-based production capacity of Saldanha Bay was 6 748 t/yr (assuming that the waste nitrogen load did not exceed 15% of the flux of nitrate nitrogen into Saldanha Bay).

The study concluded that finfish production at these increased levels should not generate adverse environmental effects on the Saldanha Bay and Langebaan Lagoon system.

These model results were used in the design of an *in-situ* monitoring programme, to identify potential changes (impacts) to Saldanha Bay in the long-term. Based on this, DAFF (2018) published "*Protocols for environmental monitoring of the Aquaculture Development Zone in Saldanha Bay, South Africa"*, often referred to as the "Sampling plan", the stated purpose of which is "a sampling/monitoring plan to address the concerns related to impacts on the marine ecology of the Saldanha Bay/Langebaan Lagoon system during the operational phase of the ADZ" (DAFF 2018). This monitoring data has been designed to validate the results of the modelling work undertaken by PRDW (2017) and to inform the proposed phased implementation and expansion of the ADZ.

The monitoring plan highlighted a number of potential impacts identified during the EIA process, including the modification of seabed by bio-deposition, and of the water column by dissolved oxygen and inorganic nitrogen; removal of seston[9] by shellfish; creation of habitat by farm structures; alteration of behaviour and entanglement of seabirds and marine fauna at finfish sites; introduction of aliens and spread of pests; transmission of diseases to wild population; genetic interaction with wild populations by shellfish and finfish; and pollution by therapeutants and trace metals.

DAFF (2018) identifies key indicators that need to be monitored (and in most cases thresholds against which these can be evaluated) as follows:

- benthic macrofaunal community species richness and biomass;
- sediment geochemical variables (total sulphides and/or redox);
- visual and odour characteristics;
- surficial sediment geochemical characteristics (total organic carbon and nitrogen (TOC/N), Al, Cu and Zn);
- sediment geotechnical characteristics (size structure, porosity);
- near-bottom oxygen concentration; and
- upper water column chlorophyll concentration (fluorometer and discrete samples).

[9] Seston are the organisms and non-living matter swimming or floating in a water body.

Proposed sampling sites are shown in Figure 3.56.

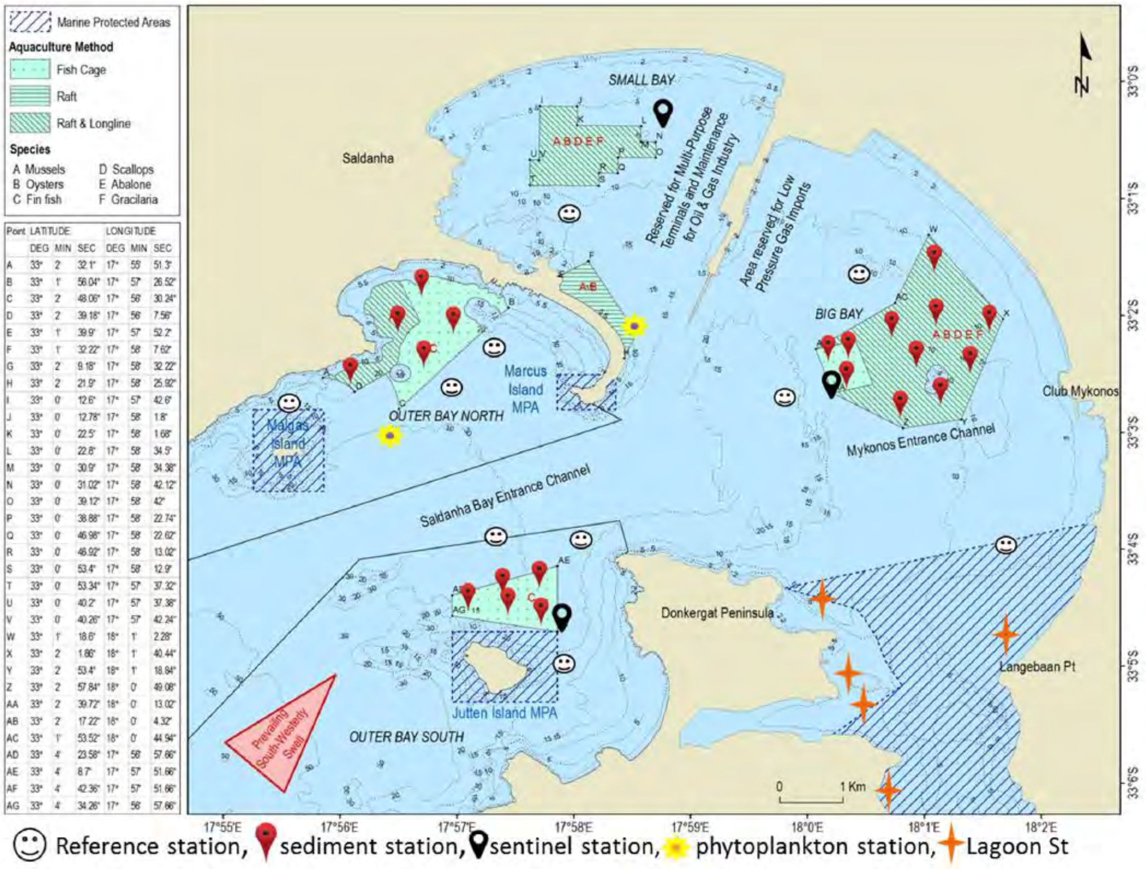


Figure 3.56. **Map of baseline sampling stations for the Saldanha Bay Aquaculture Development Zones from DAFF (2018). The monitoring program notes that these stations positions are to be finalised prior to sampling, and that the lagoon stations are from this State of the Bay monitoring programme.**

Monitoring protocols listed in the programme that are addressed in detail are divided into two components, namely baseline (1) and operational monitoring (2).

Baseline monitoring (1) is further subdivided into (a) seabed and (b) water column. Specifications are detailed and it is stated that "protocols should be aligned with the State of the Bay Programme where 3 replicate samples of 0.08 m^2 and 30 cm deep, where possible, are taken by divers at each station and pooled for subsequent taxonomic analysis of macrofauna in the >1 mm size fraction". There is also no specification on when (time of year) the samples should be collected so this could complicate comparisons between these data sets. It must be noted that the sites in the lagoon are to be from the annual SOB monitoring programme, and therefore, sensible comparability between the data is an imperative.

A comparison of indicators to be monitored with the list of identified significant impacts suggests that the DAFF (2018) monitoring protocol may come up short in the following areas:

- impacts on marine megafauna (due to entanglement, interaction with farm infrastructure),

- transmission of diseases to wild population,
- genetic interaction with wild populations by shellfish and finfish (only relevant when indigenous species are farmed in the Bay); and
- pollution by therapeutants.

Interaction with megafauna is collected by the farms and reported on in their monthly farm monitoring reports and recorded by the ECO in the summary reports for the ADZ. In addition, any instances of diseases are recorded and reported on by an independent vet. Genetic impact depends on which species are being farmed, currently no indigenous fish species that could have genetic impact on the natural population are being farmed. All the species are alien and as there are no naturally occurring salmonids in the Bay. The protocol does list what are referred to as “basic requirements for an effective biosecurity plan” (p33) but no specific actions are proposed. Risks from pollution by therapeutants are expected to be addressed through the South African Live Molluscan Shellfish Monitoring and Control Programme (SAMSM&CP) and the South African Aquaculture Marine Fish Monitoring and Control Programme. Shellfish farms do not use thereapeutants but antibiotic residues are tested routinely for both farmed finfish and shellfish in terms of the National Residue Control Programme which is in line with internal standards and regulations. In addition, heavy metals are being measured in the sediment to monitor any impacts from the use of antifoulants on cages.

Protocols for water column monitoring (1b) include requirements for installation of a fluorometer at the head of the lagoon (SANParks jetty) and collection of “calibration samples” for size-fractionated chlorophyll analysis which we support. This monitoring is underway, as the fluorometer has been situated at the head of the lagoon and the phytoplankton technician for the food safety programme has been trained to measure size fractionation.

Specifications for operational monitoring (2) recommend regular chemical surveys to ascertain if there are any impacts. Instrumentation has been deployed within the Bay for about 6 months now for the continuous monitoring of nitrate and oxygen, as well as annual redox surveys. If the thresholds are exceeded additional microbenthic monitoring is suggested otherwise, full macrobenthic surveys should be conducted every 3-5 years which is in line with international monitoring standards. There is concern that the interval at which the macrobenthic survey is to be repeated (every 3-5 years) is too long, given the high level of variation that is inherent amongst macrofaunal communities in the Bay as has been very well demonstrated through the SOB monitoring programme. For example, year to year variation in the average number of invertebrates per square meter ranges from 64-1 139 in Small Bay and from 88-1 403 in Big Bay (See Chapter 9 for more details on this). As such, we recommend that the macrobenthic monitoring surveys should be repeated at least on an annual basis.

There is a requirement for “Annual, non-quantitative samples should be taken of fouling organisms on farm infrastructures, infrastructures, preferably in conjunction with the State of the Bay Programme”. This is a potentially valuable addition to this monitoring work, as is the requirement for establishing sentinel and reference stations for monitoring temperature and oxygen in Small Bay and for monitoring of sulphide levels in Small Bay. This monitoring is to form part of the new service provider appointment (DEFF *pers. comm*. Michelle Pretorius)

Also noted are several ecosystem indicators that are currently monitored as part of the State of the Bay programme that also need to be considered in the context of expansion of aquaculture in the bay, including fish abundance, bird breeding success and alien species occurrence. The sampling plan indicates that future reporting should take these into consideration.

The benthic macrofauna baseline sampling campaign was undertaken from January to April 2019, with 27 stations sampled, of which nine were control stations (Heinecken 2019). Sampling was undertaken in Big Bay, North Bay and North Bay (Jutten Island) (Figure 3.57). In addition, the annual redox survey of the Saldanha Bay ADZ was conducted during the annual Saldanha State of the Bay survey (April 2020). Both the Baseline Macrofauna Report and the Annual Benthic Redox Survey report have been completed and were made public in September 2020.

3.8.1.2 Baseline benthic monitoring

The Saldanha Bay ADZ operators appointed an independent service provider (Capricorn Fisheries Monitoring) to collect baseline samples for all the ADZ precincts from January to April 2019 in line with the requirements of the Environmental Authorisation and EMPR. This baseline survey included the collection of sediment samples that were analysed for particle size distribution, total organic carbon (TOC), total organic nitrogen (TON) and trace metals at the identified finfish sites. Additionally, sediment samples were collected for the identification and enumeration of macrofaunal species. Sampling was undertaken in the three lease areas namely; Big Bay, Outer Bay North, and Outer Bay South (Jutten Island, Figure 3.57) with a total of nine 'Control' stations and 18 'Impact" stations selected, of which all but three were successfully sampled (three sites were identified as comprising "hard" or rocky substrata where grab samples could not be collected). DEFF appointed an independent service provider, Anchor Research and Monitoring (Pty) Ltd, to compile the baseline technical report using the data derived from the baseline sample collection. A brief summary of the findings is provided below.

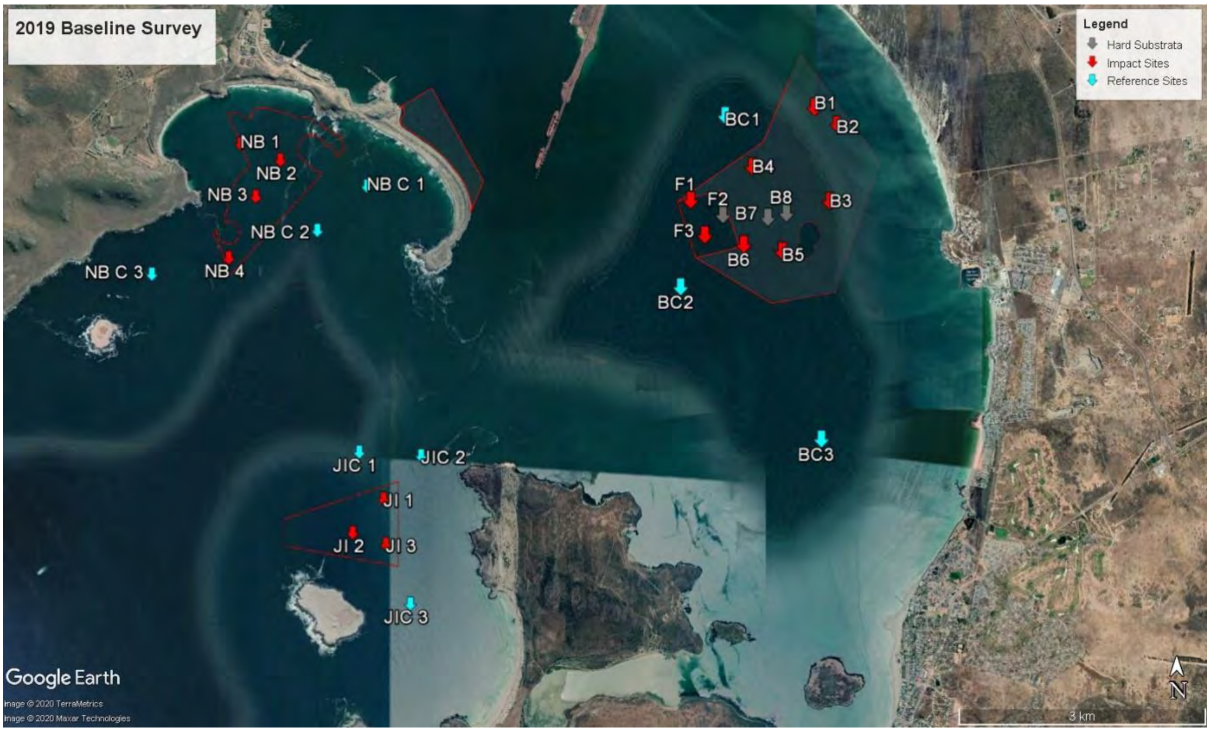


Figure 3.57. Map of Saldanha Bay showing the stations sampled during the baseline survey of the Saldanha ADZ, control sites are indicated with blue arrows while impact sites are indicated with red arrows, grey arrows indicate hard substrata.

Sediment granulometry revealed that all three lease areas were dominated by sand with minimal quantities of finer (mud) and coarser sediment (gravel) present. In addition, total sediment organic carbon and nitrogen at all three lease areas were similar to those recorded in the State of the Bay 2019 survey and heavy metal concentrations (Copper and Zinc) tested at the finfish sites were significantly lower than the concentration at which toxicity may begin to be observed in sensitive species. Overall, the baseline data for sediment quality was comparable between impact and reference sites, as well as sites sampled elsewhere in the Bay during the State of the Bay 2019 survey (SOB 2019), and no anthropogenic disturbances to the physico-chemical nature of the sediments were detected.

Macrofaunal baseline survey data indicate that the current aquaculture operations are having a negligible effect on benthic macrofauna present in the lease areas. Analyses of four community indices (Shannon Weiner Diversity, Total number of species, Abundance per sample and Pielou's Evenness) between reference and impact sites of all three lease areas were not significantly different for most comparisons. Outer Bay North showed a significant increase in species abundance at the impact sites, suggesting some impact of shellfish aquaculture operations was occurring in this lease area. Additionally, the total number of species in Outer Bay South reference sites was significantly lower than at sites marked out for future aquaculture development, despite the lack of any aquaculture operations currently. Two biological indices assessed showed a reasonable high level of agreement and generally indicate that current aquaculture operations are having a limited effect on benthic macrofauna in the three lease areas. Suggestions for future studies include; (1) the creation and sharing of taxonomic reference collections from previous surveys to be compared with SOB reference collections to facilitate accurate comparisons of macrofaunal communities, (2) the capture of macrofaunal biomass to facilitate the use of additionally advantageous statistical analyses such as Cumulative abundance-biomass plots (ABC curves) of macrobenthic communities (Warwick 1993) and (3) the possible shift from using macrofaunal indices created for use in the Northern hemisphere to one created locally.

During the baseline surveys and subsequent deployments of monitoring instruments in the finfish lease area, it was noted that low-profile reef was present in the form of calcrete rock roughly < 1m in height, additionally some outcrops of reef exceeding 1 m in height were also found. The extent of the abrasion platform present in Big Bay is currently unquantified and the proportion of this habitat type impacted by current and future mariculture activities unknown. Additionally, the effects of aquaculture on patches of this habitat type and its associated epifaunal communities has not previously been assessed in Big Bay with the exception of a known outcropping called Lynch Blinder. It is therefore suggested that further studies be conducted to address this gap in the information base. These should include conducting surveys of suitable reef impact sites in the finfish area along with corresponding reference sites, as well as a detailed side-scan sonar (and/or bathymetry) survey of the whole of Big Bay using side scan sonar or multibeam echosounder, data from which can be used to delineate the full extent of reef in this area. These data should enable the formulation of provide mitigation and management measures that will support the ongoing adaptive management of the Big Bay ADZ precinct.

3.8.1.3 Annual benthic redox survey

Monitoring of benthic impacts below mariculture installations is international best practice and is being undertaken in Saldanha Bay by DEFF to validate dispersion model predictions of minimal impact. Two independent service providers, Capricorn Fisheries Monitoring (Pty) Ltd and Anchor Research and Monitoring (Pty) Ltd (AR&M) were appointed, to undertake specialist monitoring including a baseline and the first annual redox survey of the Saldanha Bay ADZ in 2019 and 2020, respectively.

Organic deposition and the subsequent decomposition by sediment bacteria increases oxygen demand which can lead to anaerobic (without air) conditions in the seabed beneath both finfish and shellfish farms. Ammonification and sulphate reduction to sulphides occur as typical responses to lowering of the oxygen reduction (Redox) potential. Sediment organic carbon, redox potential (Eh) and total sulphides (S^{2-}) have effectively been used in describing adverse impacts below finfish aquaculture and can be used to classify sediments associated with fish farming into five organic enrichment groups: two oxic (oxygen present), two hypoxic (oxygen deprived), and one anoxic (without oxygen) group. The Aquaculture Stewardship Council (ASC 2017) provides threshold limits directly below the aquaculture structures, as well as at the edge of the Acceptable Zone of Effect (AZE), a prescribed distance from the aquaculture structures - defined as 30 m from a fish cage array or shellfish longlines. Failure to meet the prescribed thresholds at the AZE limit or at finfish cages or directly below shellfish longlines will require management intervention and/or additional sampling (DAFF 2018). Non-compliance is dependent on the farm or AZE station being significantly greater than levels measured at the reference/control stations.

Sediment was successfully collected by grab sampling at 33 sites within the ADZ during April and May 2020 (14 Sites in Big Bay, 7 in Outer Bay North, 6 within the Outer Bay South precinct and 6 within Small Bay; Figure 3.58, Figure 3.59). Triplicate redox measurements were taken at each site. Difficulties encountered during grab sampling at some sites in Big Bay confirmed the presence of the abrasion platform (and in some areas, exposed reef). Historic data on the extent of the abrasion platform (based on that collected by Flemming 1977) was overlaid on a map of Big Bay and the ADZ boundaries and this suggests that the finfish area falls over a section of the abrasion platform that is predominantly exposed. Sediment sulphur concentrations, although known to be a critical tool for determining the impacts of aquaculture on benthic environments, were not measured during the survey due to lack of a suitable instrument. Redox values were used as proxy for sulphide concentrations and it was acknowledged that measuring sulphide concentrations in the future would provide additional valuable information, as well as validation for previous data collected.

Redox measurements were highly variable among sites. Values recoded at the finfish precinct were all within the specified thresholds and will serve as a baseline for future monitoring as there are no finfish currently on site. Although closer to the threshold than in other lease areas, all sites within Big Bay except one, were within the specified thresholds. Similarly, all readings but one in Outer Bay North, were within the threshold limits. The single site that exceeded the threshold was statistically different from Outer Bay reference sites in 2020, however, was not significantly different in the 2019 survey. The location of the site in a relatively sheltered area of Outer Bay North could allow the current direction to result in deposition of organic matter from the Outer Bay North ADZ precinct, which facilitates the reduction of oxygen levels in the area. It was suggested that the site be monitored closely and should the redox value progress further towards the Anoxic category management interventions should be undertaken. The sites at the Outer Bay South precinct were significantly below the threshold levels. Although readings of a single site in Small Bay exceeded the stipulated threshold, it had statistically similar redox values to two of the reference stations and therefore does not trigger management action. It was however noted that the two reference sites were not at a comparable depth to the impact sites and suggested that were the reference sites located at a similar depth to the impact sites it would provide a more accurate reference to measure redox and sulphide impacts against.

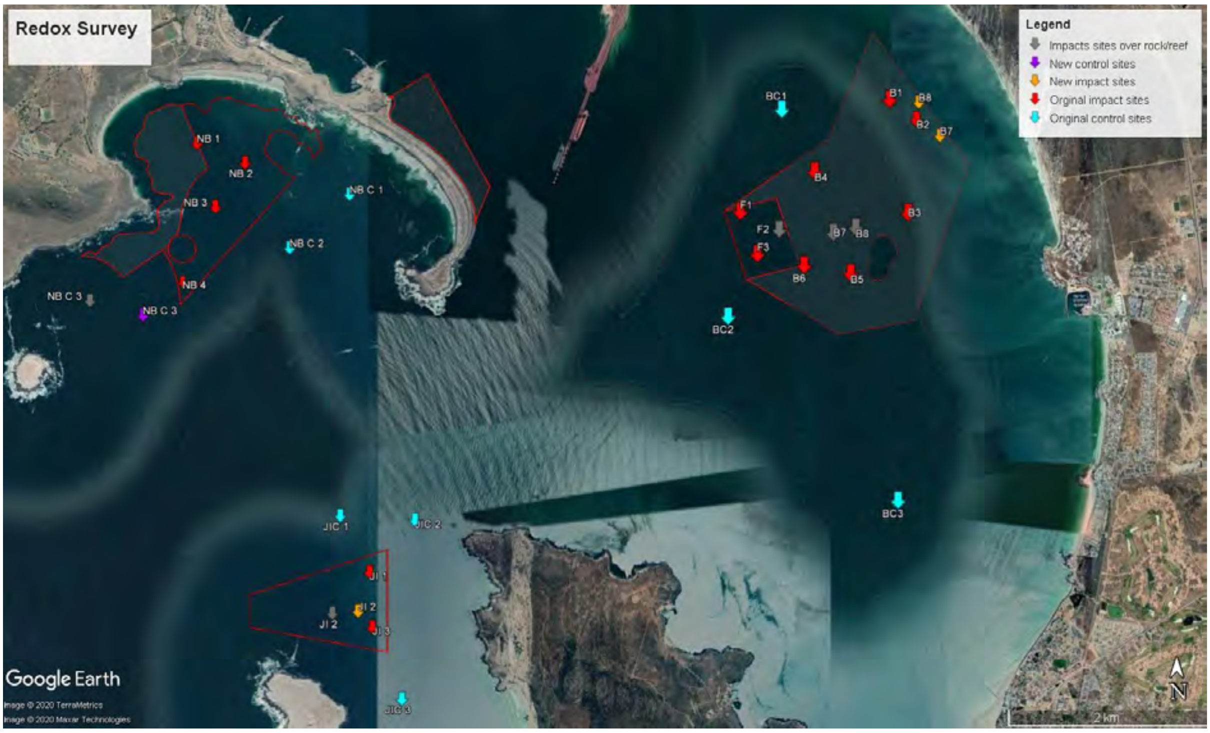


Figure 3.58. Map of Saldanha Bay showing the stations sampled during the annual redox survey of the Saldanha ADZ, control sites are indicated with blue arrows while impact sites are indicated with red arrows, grey arrows indicate sites over rock/reef. Replacement impact sites are indicated with orange arrows and replacement control sites are indicated with purple arrows.

Additional suggestions based on the statistical analyses of the data collected during the 2020 survey included:

- Redox measurements yielded highly variable readings among sites. Several factors (e.g. sediment granulometry and organic content) may influence redox values in sediment and, as an additional measure, these should be analysed in the future in conjunction with the chemical surveys as they are measured during the macrobenthos surveys. These sediment characteristics (granulometry and organic content) can also be used to monitor potential impacts of ADZ development and will allow better use to be made of the sediment samples collected in future.
- The presence of the abrasion platform in Big Bay prevented the collection of sediment samples at certain sites and may cause the concentration of organic matter in depressions at others. Determining the extent and nature of platform would help in interpreting finding from future surveys and in the assessment of impacts of aquaculture development in Saldanha Bay as a whole.

- In instances where farming structures fall over hard substrata, redox and sulphide measurements are not considered suitable tools for monitoring the health of the benthic environment as sediment cannot be collected and these analyses require sediment. Alternative means for monitoring the health of the benthic environment in these areas (e.g. assessment of visual or photo-quadrats) needs to be identified and implemented in the future.

Figure 3.59. Map showing the sites sampled for the Small Bay redox survey, yellow arrows indicate impact sites and blue arrows indicate control sites.

Based on the 2019 and 2020 redox surveys the following key findings are noted:

- The majority of the impact sites surveyed within the four ADZ precincts in Saldanha Bay (Big Bay, Outer Bay North, Outer Bay South and Small Bay) fall within the stipulated thresholds, and it is recommended that these sites be surveyed again in April 2021 in accordance with the ADZ Sampling Plan requirements.
- The same applies to the sites in Big Bay (B4) and Outer Bay North (NB1) where, in 2020, measured redox values exceeded stipulated thresholds and were significantly different to their respective reference stations, although no aquaculture activity was present in the immediate vicinity.
- Similarly, while the redox values recorded at SB2 in Small Bay exceeded the stipulated threshold, measured values were not significantly different from the two reference stations in this area, and thus should not trigger any management action. This precinct should be surveyed again along with the new recommended reference stations during the 2021 annual redox and sulphide survey.

3.8.2 Aquaculture sub-sectors

Most established operators hold rights to farm mussels (*Mytilus galloprovincialis* and *Choromytilus meridionalis*) and the pacific oyster *Crassostrea gigas*, while fin fish rights (*Salmo salar* and *Oncorhynchus mykiss*) have only been issued to two farms since 2014 (**Error! Reference source not found.**). Abalone, scallops, red bait and seaweed are currently not cultured on any of these farms, although some of the farms have the right to do so (Refer to the 2014 and 2015 State of Saldanha Bay and Langebaan Lagoon Reports for details on individual farms). Most of the farming occurs in Small Bay, however, oysters and mussels are cultured in Big Bay by the Saldanha Bay Oyster Company and West Coast Oyster Growers, and mussels are being grown in Outer Bay North.

Overall, the drive is to farm indigenous species as they do not require comprehensive risk assessments and are likely to have a lower impact on the marine ecology of Saldanha Bay and Langebaan Lagoon. However, in some cases indigenous species may be economically less viable. The DEFF therefore included alien trout species in their application for EA. Consequently, the Environmental Authorisation issued to DEFF for the ADZ includes the following alien finfish:

- Atlantic salmon (*Salmo salar*)
- Coho salmon (*Oncorhynchus kisutch*)
- King/Chinook salmon (*Oncorhynchus tshawytscha*)
- Rainbow trout (*Oncorhynchus mykiss*)
- Brown trout (*Salmo trutta*)

Biodiversity Risk and Benefit Assessments have been conducted for all five salmon and trout species and generally the risk for establishment of this species is considered low due to the fact that these species will be farmed in the sea and rivers in this region are not suitable for successful reproduction of salmonids. Arguably the greatest risk of salmonid cage culture is the transfer of diseases and parasites to indigenous fish species.

Other new indigenous species include Abalone (*Haliotis midae*), South African scallop (*Pecten sulcicostatus*), white stumpnose (*Rhabdosargus globiceps*), kabeljou (*Argyrosomus inodorus*) and yellow tail (*Seriola lalandi*).

3.8.2.1 Shellfish marine aquaculture

Raft culture of mussels has taken place in Saldanha Bay since 1985 (Stenton-Dozey *et al.* 2001). Larvae of the mussels *Mytilus galloprovincialis* and *Choromytilus meridionalis* attach themselves to ropes hanging from rafts and are harvested when mature. Mussels are graded, washed and harvested on board a boat. Overall mussel productivity has been increasing exponentially since 2007, peaking in 2019 at 3053 tonnes (Figure 3.60.). Mussel production has more than tripled since 2012, which can be attributed to the establishment of a new mussel farm and the conversion of an oyster farm to a mussel farm (DAFF 2015). In 2015 the mussel sub-sector (based in Saldanha Bay) contributed 48.83% to the total mariculture production and is highest contributor to the overall mariculture productivity for the country (DAFF 2016). Oyster production has fluctuated around 250 tonnes per annum since 2000. Oyster production reached a peak in 2016 at 357 tonnes per annum but has since decreased to 288 tonnes in 2019 (Figure 3.60.).

A study conducted between 1997 and 1998 found that the culture of mussels in Saldanha Bay created organic enrichment and anoxia in sediments under mussel rafts (Stenton-Dozey *et al.* 2001). The ratios of carbon to nitrogen indicated that the source of the contamination was mainly pseudofaeces, decaying mussels and fouling species. In addition, it was found that the biomass of macrofauna was reduced under the rafts and the community structure and composition had been altered (Stenton-Dozey *et al.* 2001).

Ongoing environmental impact monitoring surveys undertaken in Saldanha Bay by the Department of Environment, Forestry and Fisheries (DEFF) will provide an indication of the environmental impact of oyster culture (DEFF unpublished data). However, visual observations of the benthos underneath oyster rafts and preliminary data show minimal impact in this area when compared to other sites within the Bay.

A recent study by Olivier *et al.* (2013) investigated the ecological carrying capacity of Saldanha Bay with regards to bivalve (in particular mussels and oysters) farming. The findings indicate that the sector could increase 10 to 28-fold, potentially creating an additional 940 to 2500 jobs for the region without compromising the environment.

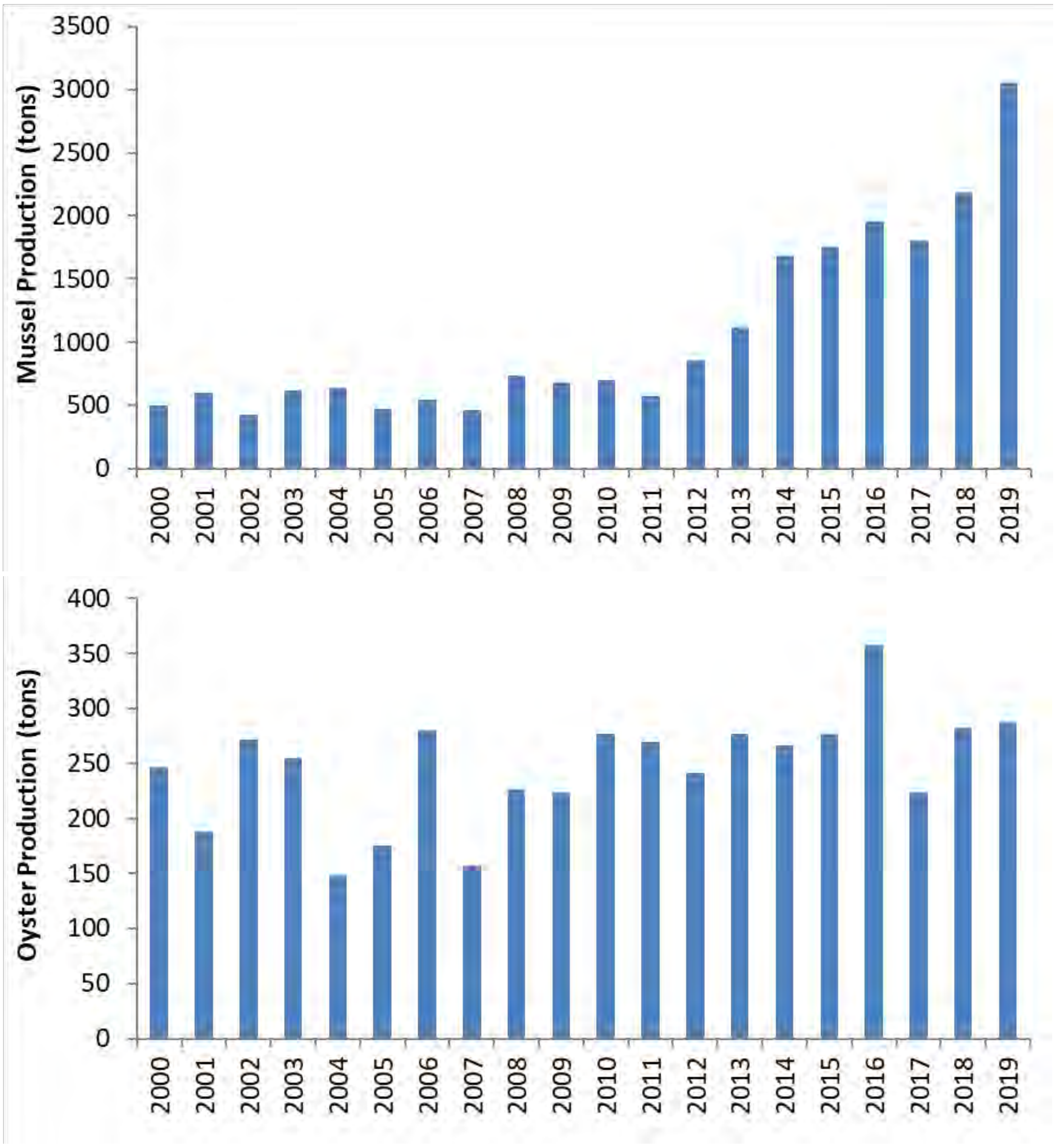


Figure 3.60. Annual mussel (top) and oyster (bottom) production (tonnes) in Saldanha Bay between 2000 and 2019 (source: Department of Environment, Forestry and Fisheries 2020 unpublished data, which may be subject to change).

3.8.2.2 Finfish cage farming

Marine cage culture of Atlantic salmon was piloted in Gansbaai several years ago, however, this reportedly failed when the heavily fouled cages sank in strong seas. The biofouling accumulated on the cage mesh due to a lack of suitable cleaning equipment (specifically a suitable size work boat equipped with a crane) (Hutchings *et al.* 2011). The identification of marine aquaculture sites is a complex process that must take into consideration a number of factors. These include physical (e.g. sea surface temperatures, currents), biophysical (e.g. harmful algal blooms, optimal culture temperatures), infrastructural (e.g. road access, airports), and existing resource-use issues (e.g. urbanisation, parks and recreational areas) (FAO 2015).

Saldanha Bay is protected when compared to the exposed west coast of South Africa and has been identified as one of very few areas where finfish cages can be installed successfully (Ecosense CC 2017). Finfish cage culture is currently being pioneered in Saldanha Bay and is largely focused on the farming of salmonid species, including Atlantic salmon (*Salmo salar*) and rainbow trout (*Oncorhynchus mykiss*). Both species are non-native to South Africa; however, *O. mykiss* is farmed in many parts of the country in ponds and raceways but this has been severely impacted by the drought and is limited in terms of seasonality of rain and temperatures.

Southern Atlantic Sea Farms attempted to pioneer Atlantic salmon in Saldanha Bay. During the pilot phase of this project, however, it was found that Outer Bay North is not suitable for Atlantic salmon due to the susceptibility of this species to amoebic gill disease, which combined with frequent low dissolved oxygen events led to high mortality rates. The project was therefore terminated in 2015 (Southern Atlantic Seafarms, Director Gregory Stubbs, *pers. comm.,* 2015).

Molapong Aquaculture (Pty) Ltd (Molapong) piloted under 50 tonnes of finfish per annum in Big Bay within Saldanha Bay. This experimental phase was successful and Molapong appointed Ecosense CC to conduct a Basic Assessment process to obtain Environmental Authorisation for the phased installation of sea cages on 40 ha in Big Bay and 15 ha near Jutten Island for the production of finfish, mussels and seaweed in Saldanha Bay up to a total of 2000 tonnes per year over both lease areas. Environmental Authorisation was issued on 8 January 2018 for the following project phases:

- Phase 1 (Experimental) – *The current level of finfish project* (50 tonnes/annum – duration 12 -14 months).
- Phase 2 – early commercial phase finfish project (100 t/annum 12 -14 months). Establish seaweed lines. Establishment of mussel settlement lines.
- Phase 3 – 500 t/annum finfish project (12/14 months). Seeding mussel production lines.
- Phase 4 – 1200 t/annum finfish project (12-14 months. Harvesting mussels and possibly reducing numbers.
- Phase 5 – 2000 t/annum finfish project (12-14 months). Harvesting mussels and possibly reducing numbers.

Molapong Aquaculture recently refined the draft Fish Escape and Stock Loss Action Plan which offers “insight into the operational and preventative measures employed by the farm and the subsequent contingency measures, should an escape occur.”

Southern Cross Salmon Farming (Pty) Ltd was also issued with an Environmental Authorisation on 8 January 2018 for the production of shellfish in the Outer Bay North Site (20 ha) to total production not exceeding 2500 tonnes (graded) on long line. Furthermore, permission was granted to produce 1000 tonnes of marine finfish per annum on 10 ha (at full production) within the Outer Bay South site by means of floating cages. Southern Cross Salmon Farming (Pty) Ltd is permitted to farm the same species that were authorised for the Aquaculture Development Zone. Southern Cross Salmon Farming has been focusing on the mussel production and have been farming mussels for roughly 2.5 years at the Outer Bay North Site, however, they have not yet commenced with finfish farming and given the restraints associated with COVID 19 are not likely to start in the foreseeable future (Barend Stander, *pers. comm.* 2020).

Operational phase environmental impacts of finfish cage culture have been well reported in international literature and are listed in Table 3.10 along with methods currently in use in the Saldanha ADZ to address and manage each.

Table 3.10. Possible environmental impacts of finfish cage culture and the methods currently in use in the Saldanha ADZ to mitigate and manage them.

Potential environmental impacts	Mitigation and management currently in place
Incubation and transmission of fish disease and parasites	Regular and continuous animal health monitoring of the farms are conducted by independent aquatic vets.
Pollution of coastal waters due to the discharge of organic wastes	Possible impacts were assessed through the modelling study and instruments have been installed to conduct continuous monitoring of bottom oxygen.
Escape of genetically distinct fish that compete and interbreed with wild stocks that are often already depleted	This is only relevant when indigenous species are being farmed, which is currently not the case within the Bay as only Salmonids are currently farmed. In addition, there is a "Fish Escape and Stock Loss Action Plan" already in place should the farming of indigenous species occur.
Chemical pollution of marine food chains (& potential risk to human health) due to the use of therapeutic chemicals in the treatment of cultured stock and antifouling treatment of infrastructure	Managed and addressed through the National Residue Control Programme.
Physical hazard to cetaceans and other marine species that may become entangled in ropes and nets	Cages are continuously monitored and inspected for possible entanglements.
Conflict between piscivorous marine animals (including mammals, sharks, bony fish and birds) and farmers and the temptation to kill problem predators or use acoustic deterrents	Regulations are in place and neither of these deterring methods are permitted.
User conflict due to exclusion from mariculture zones for security reasons	Possible conflicts were accessed during the original EIA process and the size of the ADZ was reduced to ensure that conflicts are limited or absent.

More information on the marine ecological impacts of finfish farming can be found in previous versions of this monitoring report (AEC 2018/17/16).

4 MANAGEMENT AND POLICY DEVELOPMENT

Continuously accelerating urban and industrial development poses a significant threat in the form of fragmentation, loss of natural habitat and loss of ecological integrity of remaining marine and coastal habitats in Saldanha Bay and Langebaan Lagoon. While many developments are ostensibly "land-based", several rely on ships to transport raw material and/or processed products to and from them. While the increase in vessel traffic associated with each of these individual developments may be minor, collectively they contribute to the ever-increasing number of vessels visiting the Bay each year as well as to the increasing volumes of ballast water that are discharged into the Bay. Similarly, each of the individual developments also contributes to the increases in the volume of wastewater and stormwater that is produced (and ultimately discharged into the Bay) each year. The challenge of addressing these cumulative impacts in an area such as Saldanha is immense.

The current and future desired state of the greater Saldanha Bay area is polarised, where industrial development (Saldanha Bay IDZ and associated industrial development) and conservation areas (Ramsar Site, MPAs and National Parks) are immediately adjacent to one another. Furthermore, the Saldanha Bay environment is utilised by a variety of, what could be considered, conflicting activities including industry, fishery, mariculture, recreation and the natural environment itself. This situation necessitates sustainable development that is steered towards environmentally more resilient locations and away from sensitive areas (Thérivel *et al.* 1994). Several environmental management tools are considered in developing this region:

1. Coastal Management Programme (ICMA)
2. Strategic Environmental Assessment (NEMA)
3. Environmental Management Framework (NEMA)
4. Environmental Management Programme (NEMA)
5. Establishment of a Special Management Area (ICMA)
6. Erosion management

These management tools are described in more detail in this chapter.

4.1 Coastal Management Programme

The National Environmental Management: Integrated Coastal Management Act (No. 24 of 2008) (ICMA) provides for the integrated management of South Africa's coastline to ensure the sustainable development of the coast. The ICMA mandates all three spheres of Government (local, provincial and national) to develop and implement Coastal Management Programmes (CMPs). CMPs contain principles and objectives to guide decisions and successful coastal management. These policy tools consist of three core components: 1) a situational analysis or status quo assessment; 2) a vision, priority and objectives setting component; and, 3) a five-year implementation programme, which includes specific coastal management objectives and implementation strategies for each identified priority area.

The Saldanha Bay Municipality (SBM) compiled its first CMP in 2013, which was recently reviewed and updated (SBLM 2019). Ten objectives for coastal management have been identified in this updated

CMP, which will be implemented by defined coastal management strategies. Objectives relevant to this monitoring report have been extracted from the CMP 2019-2024 document (Table 4.1). The implementation of this five-year plan will be monitored, and implementation success will be measured by indicators identified in the CMP.

Table 4.1. Selected objectives of the Second Generation Saldanha Bay Local Municipality Coastal Management Programme.

Coastal Management Objective	Coastal Management Strategy
1. Improve cooperative governance and clarify institutional arrangements	• Clarification of institutional arrangements for coastal management and the facilitation of the generation of capacity • The continued implementation and update of the Coastal Management Programme • The promotion of cooperative governance through engagement with all relevant coastal stakeholders
3. To ensure that coastal planning and development is conducted in a manner that ensures the protection and rehabilitation of the coastal zone.	• Incorporation of biodiversity, environmental and climate change policies into town planning processes • Addressing Coastal Erosion within the coastal zone • To address the high percentage of vacant plots and the low occupancy levels of residential dwellings
4. To enhance compliance monitoring and enforcement efforts in the district	• Developing Local Authority Environmental Management Inspectorate and Honorary Marine Conservation Capacity • Facilitating and encouraging public reporting of illegal activities • Facilitating the development and enforcement of Municipal by-laws • Addressing the increase in illegal Off-Road Vehicle activity
5. To ensure effective management of estuarine resources in the West Coast District Municipality	• Facilitating the designation of Responsible Managing Authorities (RMA) • Supporting the development of Estuarine Management Plans for smaller estuaries in the WCDM • Facilitating the implementation of Estuarine Management Plans in the District
6. The protection, management and sustainable use of natural resources	• The effective control of invasive alien plants • Cooperative management of Protected Areas • Monitoring mining activities in the coastal zone • Facilitating the coordinated management of Marine Living Resources
8. The effective management and control of pollution in the coastal zone	• Managing the discharge of effluent, stormwater and other industrial-based pollutants into coastal waters • Continue to plan, install, alter, operate, maintain, repair, replace, protect and monitor municipal WWTWs in coastal towns • To promote the effective management of Air Quality • To ensure the effective management of solid waste in the coastal zone • Encouraging the Reinstatement of the Blue Flag Beach Programme
9. Ensuring the socio-economic development of coastal communities	• Promotion of the Small Harbours: Spatial and Economic Development Framework • Development of marine aquaculture within the District • Supporting the Small-Scale Fisheries Industry • The facilitation of coastal tourism development • Preparing for the growth of the renewable energy sector

4.2 Strategic Environmental Assessments for the Greater Saldanha Bay Area

Shortcomings that limit the role project-level EIA's as a tool for achieving sustainable development are widely documented. These are often linked to the reactive and piecemeal focus of project level EIAs which have limited capacity for anticipating and assessing changes to affected ecosystems beyond property boundaries. Project level EIAs are also not effective in addressing cumulative impacts from multiple developments or activities (Thérivel *et al*. 1994; Brown and Hill 1995; Glasson *et al*. 1999; Dalal-Clayton and Sadler 2005). Inefficiencies arising from fragmented, activity-based EIA procedures can be countered by means of a strategic environmental management approach, which places a proposed activity within the environmental context of a particular geographical area. Accordingly, NEMA Section 24(3) provides that:

> *The Minister, or an MEC with the concurrence of the Minister, may compile information and maps that specify the attributes of the environment in particular geographical areas, including the sensitivity, extent, interrelationship and significance of such attributes which must be taken into account by every competent authority.*

A task team has been set up by the Department of Environmental Affairs and Development Planning (DEADP) with the objective to conduct a Strategic Environmental Assessment (SEA) for the Greater Saldanha Bay Area (DEADP 2016). SEAs are effective environmental management instruments that are designed to ensure that environmental and other sustainability aspects are considered effectively and holistically in policy, plan and programme making within an area such as Saldanha Bay. The development of a SEA typically involves formulating a desired environmental state for the area under consideration and the identification and evaluation of limiting environmental attributes against a set of thresholds beyond which the realisation of the desired environmental state would be compromised. Any proposed development can then be evaluated against the SEA to ascertain whether the activities are congruent with the desired environmental state.

4.3 Environmental Management Framework

Environmental Management Frameworks (EMFs) are one of several prescribed environmental management instruments that give effect to NEMA Section 24(3) through the Environmental Management Framework Regulations of 2010 (Figure 4.1). These regulations take cognisance of the fact that important natural resources must be retained to provide for the needs and ensure the health and well-being of citizens in a particular area in the long-term. The EMF Regulations of 2010 state that an EMF should aim to promote sustainability, secure environmental protection and promote cooperative governance and may be adopted by the competent authority. If adopted by the competent authority, EMFs must be considered in all EIAs and must be considered by every competent authority during the decision-making process. The burden of proof to demonstrate that a proposed development is aligned to the EMF lies with the project proponent. The EMF provides applicants with a preliminary indication of the areas in which it would be potentially inappropriate to undertake an activity listed in terms of the NEMA EIA regulations by:

1. Specifying the sensitivity or conservation status of environmental attributes in a particular area;
2. Stating the environmental management priorities of the area; and
3. Indicating which activities would be compatible or incompatible with the specified area.

Chand Environmental Consultants were appointed in 2010 by the Western Cape Department of Environmental Affairs and Development Planning (DEA&DP) to compile a Draft EMF in 2013 for Saldanha Bay (for more information on the original EMF refer to AEC 2016). The original Draft EMF was recently reviewed as part of the Greater Saldanha Regional Spatial Implementation Framework (DEA&DP 2018). The original extent of the Saldanha Bay EMF was expanded to include the Berg River and its estuary, and a Draft Environmental Management Framework was completed by the Western Cape Government in April 2017. No final EMF is available, and it is unknown whether the EMF has been adopted yet.

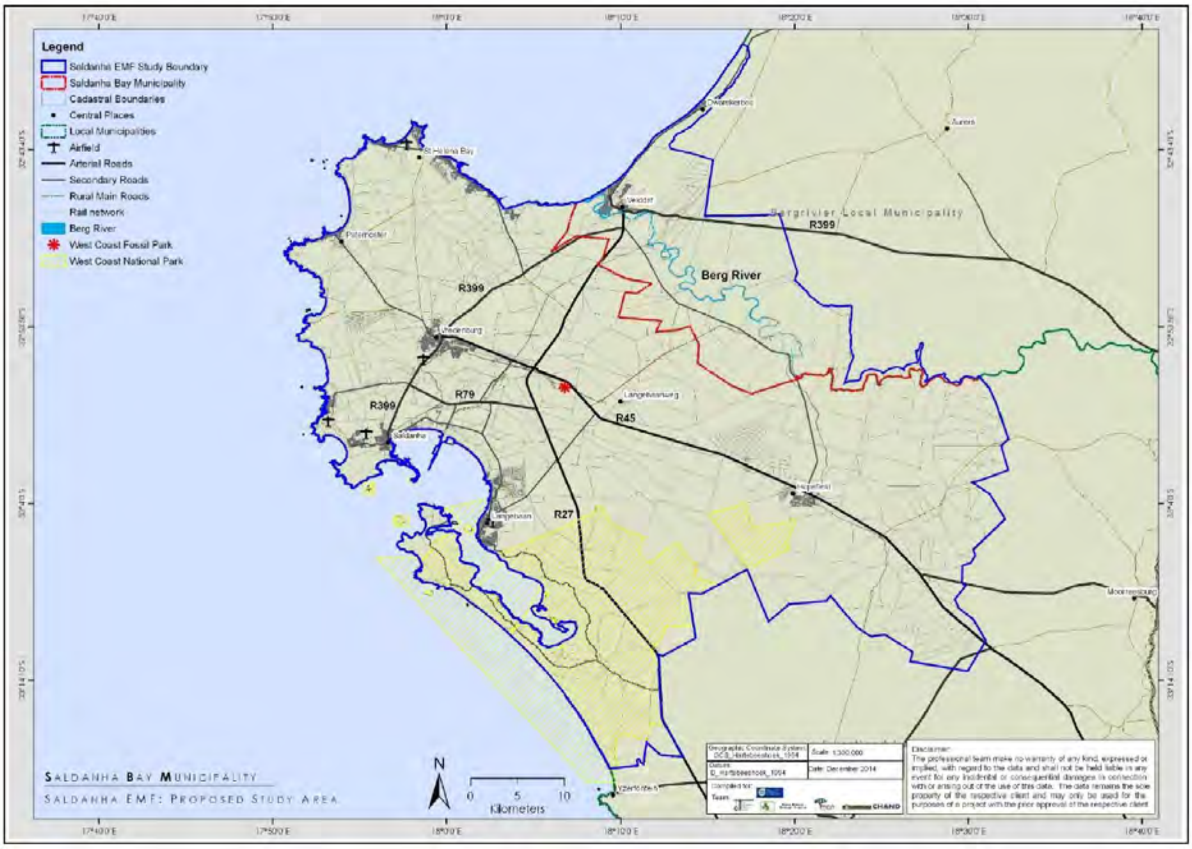


Figure 4.1. **Study Area for the Greater Saldanha Bay Environmental Management Framework (DEA&DP 2017).**

4.4 Generic Environmental Management Programme

DEA&DP compiled an Environmental Management Programme (EMPr) in collaboration with the National Department of Environmental Affairs (Directorates Oceans and Coast and Environmental Impact Assessment), the Saldanha Bay Municipality and the Saldanha Bay Water Quality Forum Trust (DEA&DP 2016). The EMPr Key contains mitigation measures and other interventions appropriate for a range of developments and associated impacts on the coastal and marine environment of Saldanha

Bay. This document was implemented in 2019 and allows government officials involved in the environmental authorisation process to compare the EMPr submitted by the applicant against a definite set of criteria applicable to the environmental challenges faced in the Greater Saldanha Bay Area.

4.5 Special Management Area

An initiative for the establishment of a Special Management Area in Saldanha Bay is gathering momentum and has the potential to improve environmental management in Saldanha Bay and Langebaan Lagoon. A Special Management Area under the ICMA may be declared in terms of section 23 (1) (a) of the Act, if environmental, cultural or socio-economic conditions require the introduction of measures which are necessary to more effectively conserve, protect or enhance coastal ecosystems and biodiversity in the area of question. The Minister may declare any area that is wholly or partially within the coastal zone to be a special management area and has the power to prohibit certain activities should these activities be considered contrary to the objectives of the Special Management Area (ICMA Section 23 (4)).

5 GROUND WATER

5.1 Introduction

The Greater Saldanha Bay (GSB) region is experiencing significant growth from many sectors as it is close to a port, is well suited for industrial development, has good infrastructure, is particularly scenic in places and is located within a unique natural setting. These attributes have resulted in rapid growth of the region and thus any future growth needs to be managed strategically with a long-term perspective and with the aim of satisfying as many role players as possible. At a policy-level, the GSB region has been proposed as a hub for:

- the 'blue' economy (marine opportunities),
- the 'green' economy (sustainable development with reduced environmental risks), and
- a responsible and transition away from a 'brown' economy (associated with non-sustainable resource utilisation).

The geographical extent of the project area is the administrative boundary of the Saldanha Bay Municipality (Figure 5.1) with the addition of a marine component comprising 12 nautical miles from the High-Water Mark, Langebaan Lagoon including the islands in the bay (Schaapen and Marcus) and just outside (Malgas, Jutten and Vondeling).

Groundwater is a key component of the natural capital within the area. It plays a crucial role in sustaining critical and unique ecosystems and is also a backup source (and possible future source) of water supply to the municipality as well as providing support to the agricultural sector. The geological setting is highly variable within the GSB area and for this reason, the groundwater is also highly variable across the study area in terms of flow rates, volumes and quality. This variability has meant that a lot of geohydrological work has been completed in the area, dating back to the 1970s. There has also been a lot of recent and on-going geohydrological work with the establishment of the Elandsfontein Phosphate Mine, the Langebaan Road Aquifer wellfield extension, and the Hopefield wellfield. Nonetheless, despite all this geohydrological research and work, there is still some uncertainty on the geohydrological characteristics of the area, including aquifer dimensions, parameters and recharge characteristics.

This chapter provides a broad outline of the geohydrology of the area with a little more detail on the areas of actual and planned groundwater abstraction and managed aquifer recharge. It highlights the areas where crucial ecosystem support is received from groundwater.

5.2 Regional Setting

The Saldanha Bay area is located on the West Coast approximately 100 km north of Cape Town, in the Western Cape. The site is located across a number of quaternary catchments which form part of the Berg River Water Management Area. The study area, within a regional context, is shown in Figure 5.3. For this report the G10M quaternary catchment within the GSB is the main area of concern.

5.2.1 Topography

The topography of the study area is variable and encompasses a variety of landscape zones, i.e., floodplains, coastline, lagoon, etc. The regional topography is mainly flat to slightly undulating with most elevations approximately at 120 mamsl (Timmerman, 1985). The site displays elevated topography at

Vredenburg and directly north of it, reaching a maximum elevation of 261 mamsl. In the south, gentler topography occurs from Langebaan Lagoon to Hopefield in the east.

Figure 5.4 shows more detailed views of the study site with relevant information superimposed on a topographical background.

5.2.2 Climate

The Saldanha area is characterised by low rainfall and experiences a Mediterranean Climate with cold wet winters and hot dry summers. Precipitation in the study area is in the form of coastal fog and low variable rainfall. Figure 5.1 shows the monthly average air temperature distribution and Figure 5.2 shows the monthly median rainfall and evaporation distribution for Saldanha (Schulze, 2009). The long term (1950-2000) mean annual precipitation for Saldanha is 337 mm/a. The rainfall exceeds evaporation in the winter months (June and July) and the peak groundwater recharge period will thus be in the winter. During the summer months groundwater plays an important role in meeting water requirements for nearby industry, local ecology and other land uses.

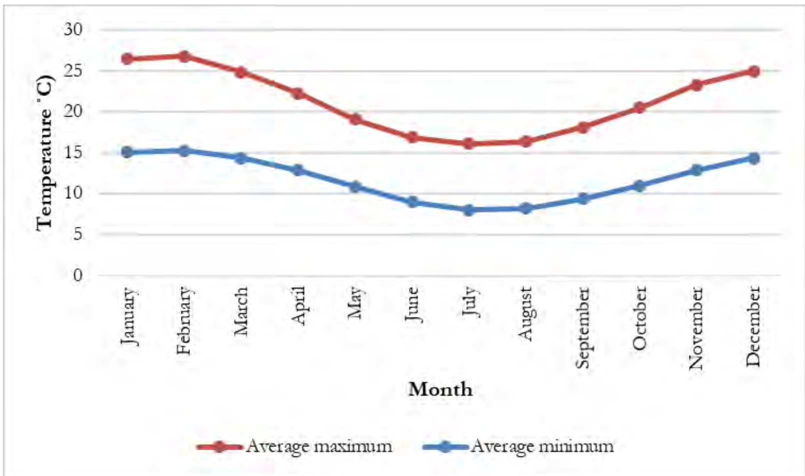


Figure 5.1. Monthly average air temperature distribution for Saldanha area (Schulze, 2009).

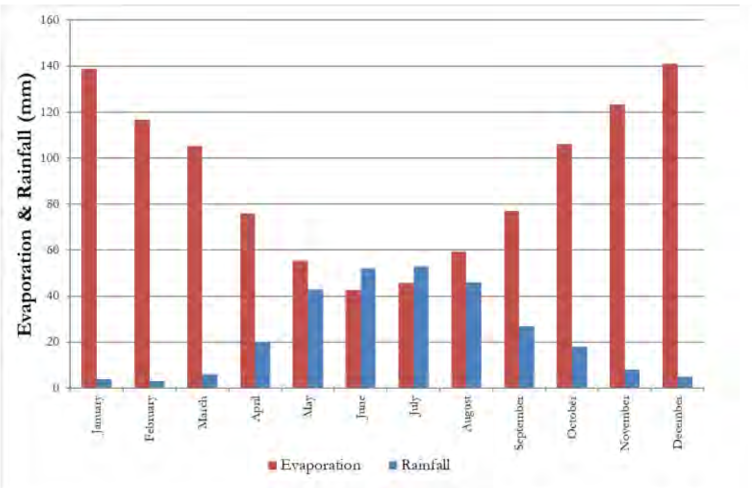


Figure 5.2. Monthly average rainfall and evaporation distribution for Saldanha area (Schulze, 2009).

Figure 5.3. **Location of the study area (Greater Saldanha Bay) within a regional setting.**

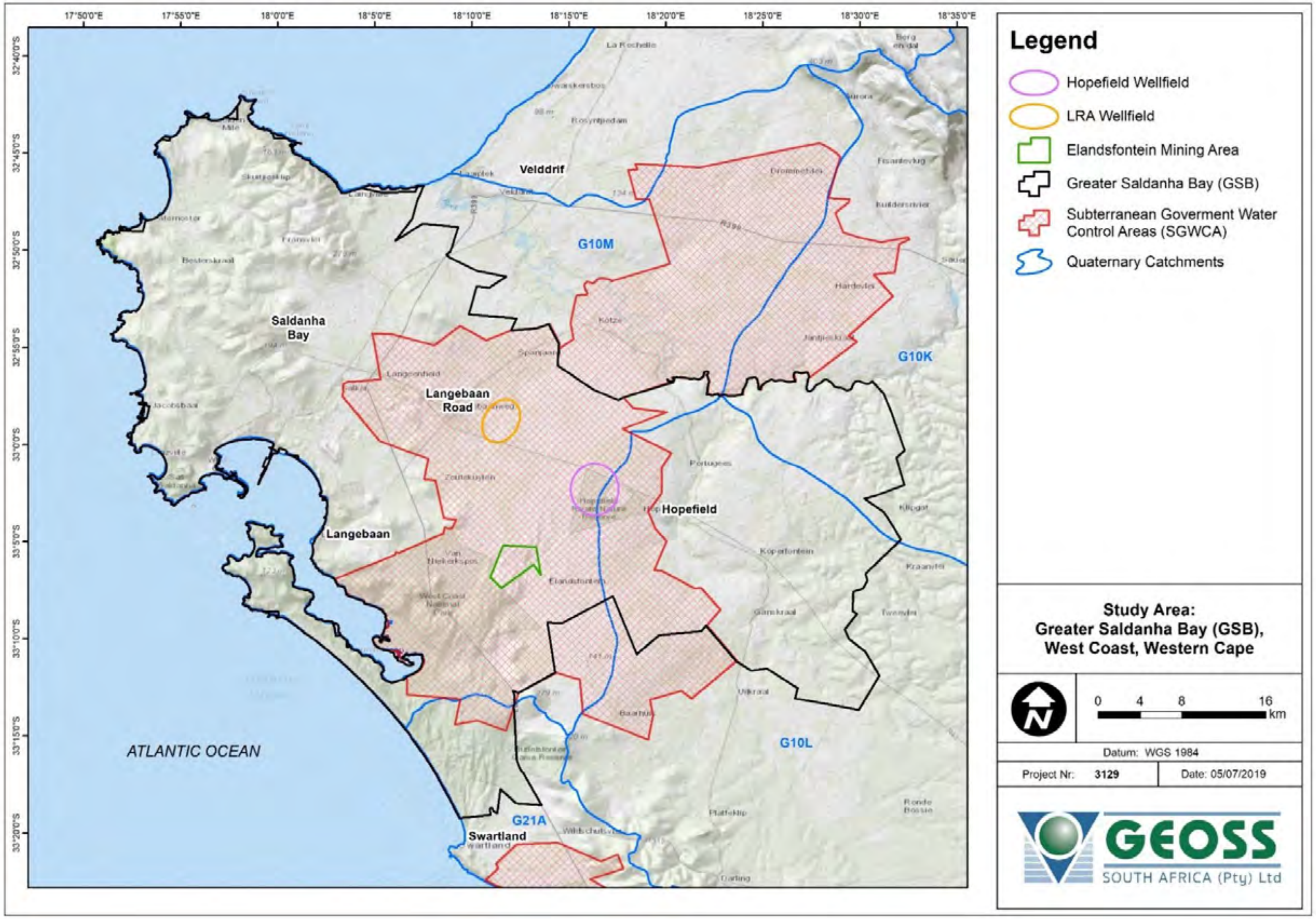


Figure 5.4. **The study area showing relevant features and topography.**

5.3 Regional Geology

The Geological Survey of South Africa (now the Council for Geoscience) has mapped the area at 1:125 000 scale. The important geological features in the area are listed in Table 5.1 and the geological setting is shown in Figure 5.5.

Table 5.1. Geological formations within the study area.

Code	Formation	Group	Description
	-	Quaternary to Tertiary Deposits/ Sandveld Group	Alluvium
Q1	Springfontein Formation		Sandy Soil
Q2	-		Sand and sandy loam from the hillocky veld
Q5	Witzand Formation		Dune sand, highly calcareous in places
QB 1			Consolidated to unconsolidated sand and gravel with marine shells
QC	Langebaan limestone		Consolidated and unconsolidated limestone and lime-rich sand
QP	Varswater Formation		Consolidated and unconsolidated phosphatic sand, clay and shelly gravel
C1Q1	Peninsula Formation	Table Mountain Group	Quartzitic sandstone with minor shale and conglomerate lenses
C1S1	Graafwater Formation		Reddish brown shale, sandy shale and siltstone
C1Q1R	Piekenierskloof Formation		Quartzitic sandstone and conglomerate
Kl	Klipheuwel Formation	Malmesbury Group	Brightly coloured shale, sandstone, greywacke and conglomerate
G2	Post-Nama		

* The nomenclature associated with the Sandveld Group has changed with time and also changes geographically

The geology of the area consists of basement Cape Granite and Malmesbury Group rocks that underlie the sediments of the Sandveld Group. The Langebaan area is dominated by unconsolidated to semi-consolidated sediments that overlay the basement rock of the Malmesbury Group unconformity (Nel, 2018).

5.4 Regional Hydrogeology

The aquifer yield and aquifer quality classifications are based on regional datasets, and therefore, only provide an indication of conditions to be expected.

5.4.1 Aquifer types and yield

According to the 1:500 000 scale groundwater map of Cape Town (3317) the area does host a range of aquifer types with varying associated yields. The western area of the site is characterised by intergranular aquifers with increasing yields towards the east, while most of the inland areas overlie fractured aquifers.

In the south, around the West Coast National Park, there are transitioning intergranular and fractured aquifers. The aquifer types include: intergranular (0.0 L/s to 0.1 L/s), intergranular (0.1 L/s to 0.5 L/s), intergranular (2.0 L/s to 5.0 L/s), intergranular and fractured (0.0 L/s to 0.1 L/s), intergranular and fractured (0.1 L/s to 0.5 L/s), fractured (0.1 L/s to 0.5 L/s), and fractured (0.5 L/s to 2 L/s) (Figure 5.6) (Meyer, 2001). Please note these classifications are based on a regional scale and boreholes do occur within the GSB with yields > 5 L/s. Borehole yields from the National Groundwater Archive (NGA) have also been included in Figure 5.6.

5.4.2 Groundwater Quality

Groundwater quality is an important aspect of groundwater and is most easily represented and understood by the parameter called "electrical conductivity" (EC). EC is a measure of the ability of the groundwater to conduct electricity and this is directly related to the concentration of ions in the water. This parameter is used as an indication of the quality of the groundwater. Conductivity in water is dependent on the presence of inorganic dissolved solids such as chloride, nitrate, phosphate, sodium, calcium, magnesium, iron, etc., i.e., anions and cations. Lower EC values indicate better quality water and elevated EC values indicate brackish and saline quality water. Drinking water typically has an EC between 0 and 70 mS/m (see Table 5.2 on drinking water standards by DWAF 1998).

Table 5.2. Classification table for the EC of water per DWAF (1998) standards for domestic consumption

<70 mS/m	Blue	(Class 0)	**Ideal water quality** - suitable for lifetime use.
70-150 mS/m	Green	(Class I)	**Good water quality** - suitable for use, rare instances of negative effects.
150-370 mS/m	Yellow	(Class II)	**Marginal water quality** - conditionally acceptable. Negative effects may occur.
370-520 mS/m	**Red**	(Class III)	**Poor water quality** - unsuitable for use without treatment. Chronic effects may occur.
>520 mS/m	Purple	(Class IV)	**Dangerous water quality** - totally unsuitable for use. Acute effects may occur.

The regional groundwater quality maps (DWAF, 1998), developed by the Department of Water Affairs and Forestry (as they were known at the time of developing the maps) do not have exactly the same categories as Table 5.2, however the classes are similar (i.e., <70 mS/m, 70-300 mS/m, 300-1000 mS/m and >1000 mS/m) and are still relevant to this report. The groundwater quality, as indicated by EC for the study area, is shown in Figure 5.7. The groundwater quality at Vredenburg and the surrounding area, as well as the far east of the study area, is characterised by saline water (300-1 000 mS/m and >1000 mS/m), while the central area (west of Hopefield) has good to marginal water quality (0-70 mS/m and 70-300 mS/m). Borehole quality (EC) from the NGA has also been included in Figure 5.7.

Based on field data from various studies in the area over the last 20 years, the groundwater quality of the Langebaan Road Aquifer (LRA) aquifer (central to the study area) is good (<100 mS/m) (Nel, 2018). The groundwater quality from the Elandsfontein Aquifer is also good.

5.4.3 Aquifer Vulnerability

The national scale groundwater vulnerability map for South Africa (Conrad and Munch, 2007), which was developed according to the DRASTIC methodology (Aller et al, 1987), shows that the area has a very-high vulnerability to surface based contaminants (Figure 5.8).

The DRASTIC method takes into account the following factors (with the weighting (i.e. importance) factor indicated in parenthesis):

D = depth to groundwater (5)
R = recharge (4)
A = aquifer media (3)
S = soil type (2)
T = topography (1)
I = impact of the vadose zone (5)
C = conductivity (hydraulic) (3)

For the study area, the high to very high vulnerability rating relates to the shallow groundwater level and the unconsolidated aquifer material. The risk of these aquifers being impacted by surface-based contaminants is very high and the groundwater quality needs to be very carefully managed.

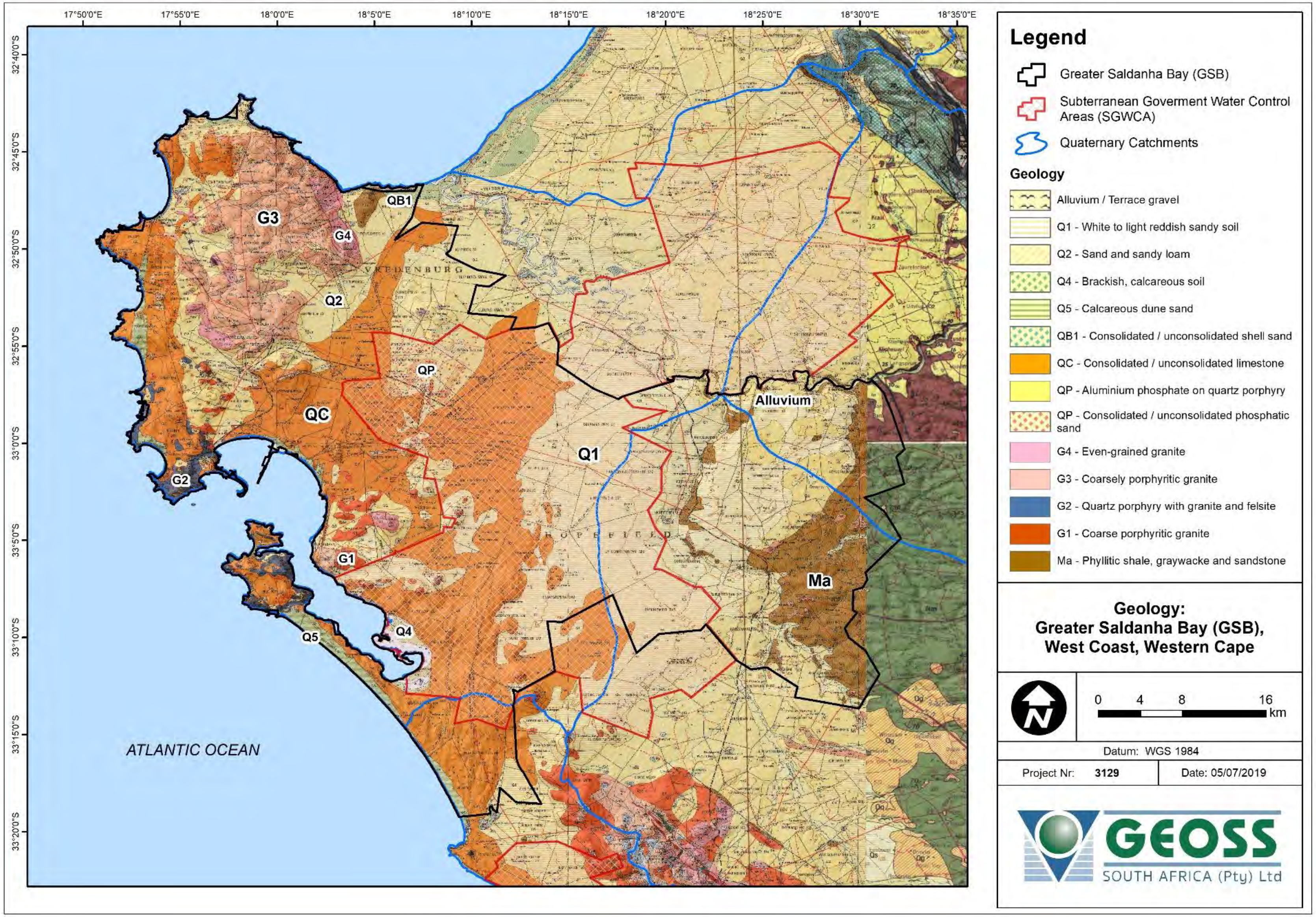


Figure 5.5. **Geological setting of the study area, (Cape Town, 3318) (CGS, 1990).**

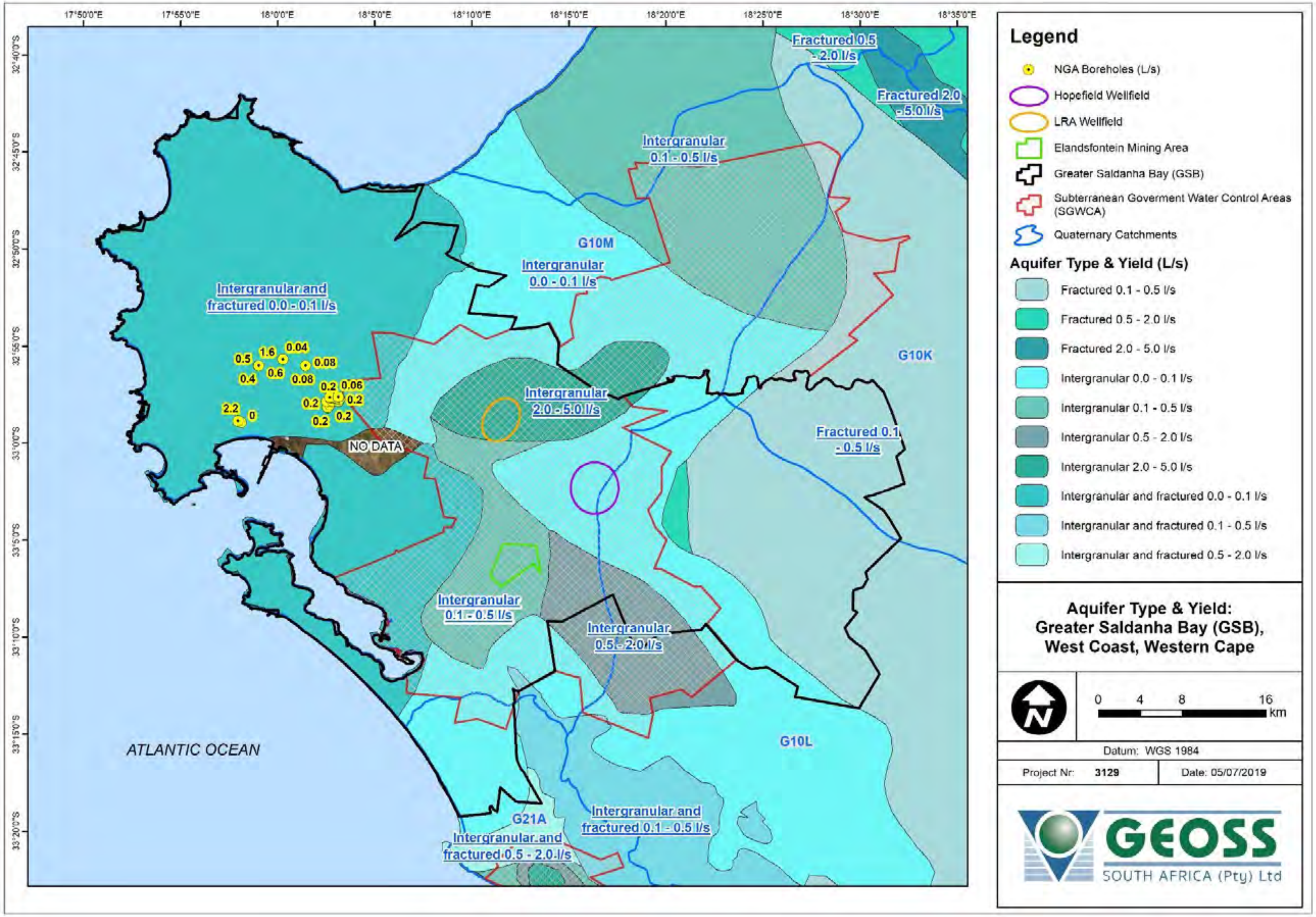


Figure 5.6. **Regional aquifer yield from the 1:500 000 scale groundwater map (3317 –Cape Town) (DWAF, 2000).**

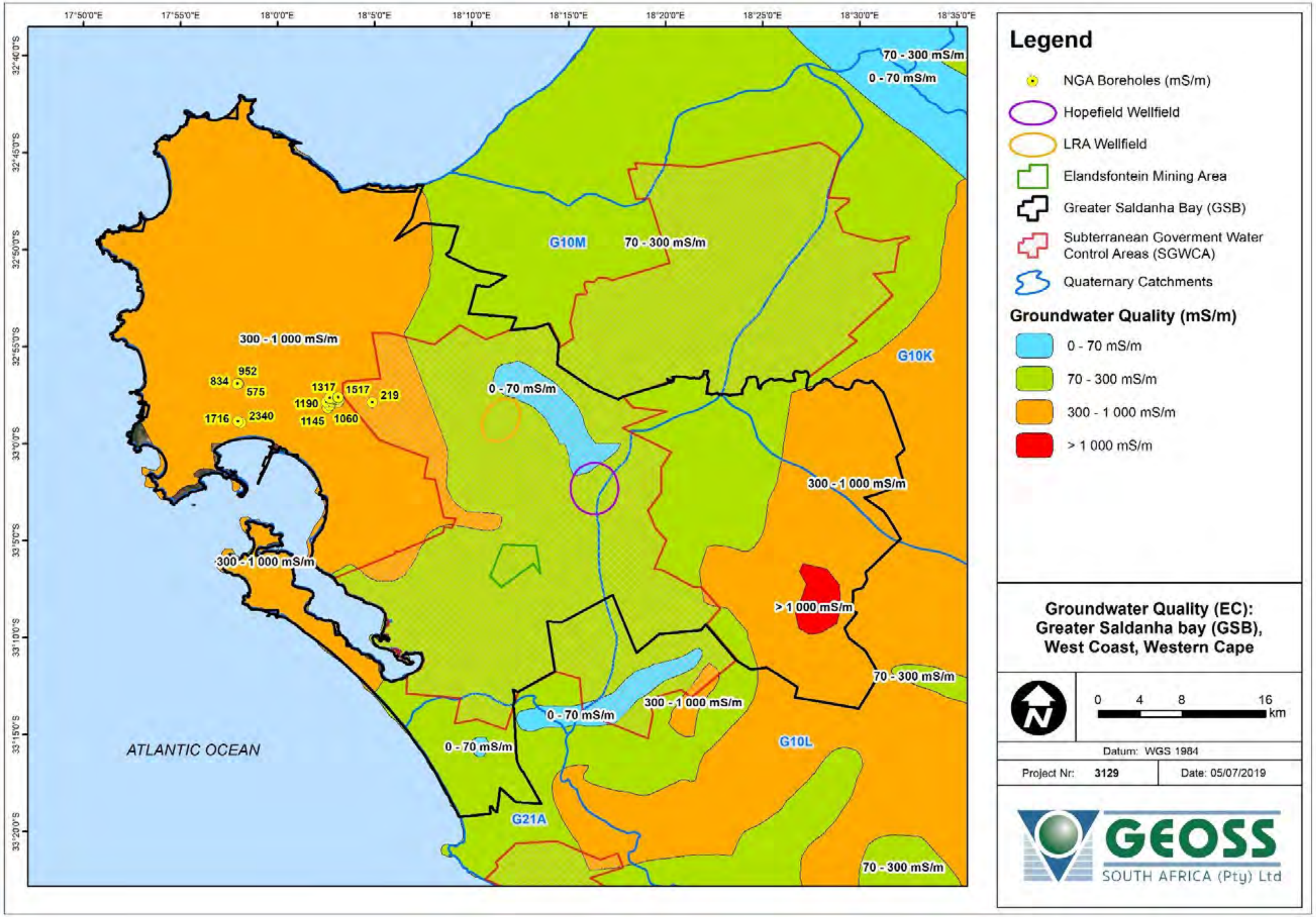


Figure 5.7. **Regional groundwater quality (EC in mS/m) from WRC (2012).**

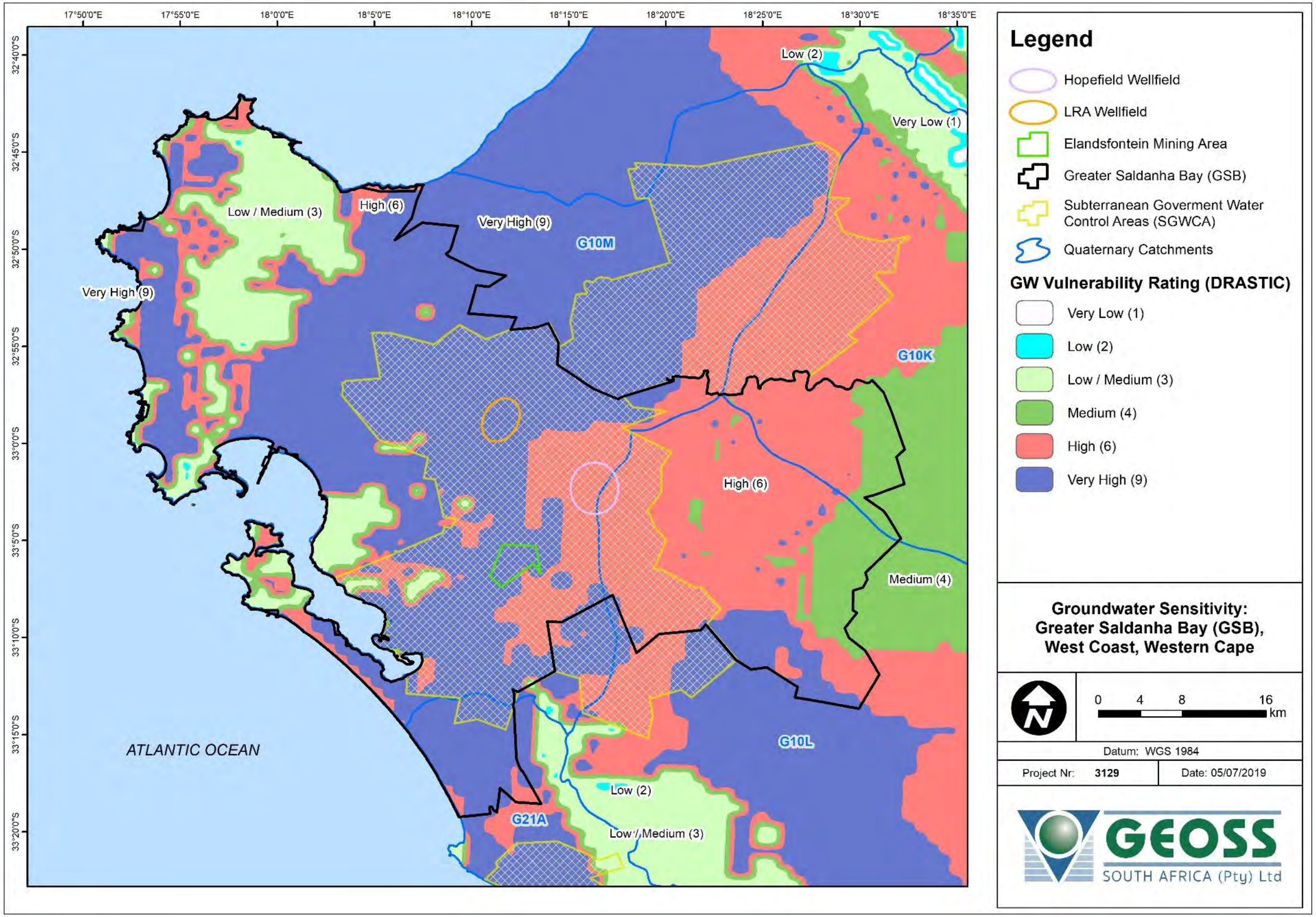


Figure 5.8. **Regional groundwater vulnerability for the study area (DWAF, 2005).**

5.4.4 Aquifer yield for the study area

GIS modelling was completed on 1 km x 1 km cell size and included the study area. This modelling resulted in a Utilisable Groundwater Exploitation Potential (UGEP). The UGEP represents a management restriction on the volumes that may be abstracted based on the defined 'maximum allowable water level drawdown'. It is likely that, with an adequate and even distribution of production boreholes in accessible portions of most catchments or aquifer systems, these volumes of groundwater may be annually abstracted on a sustainable basis. For more information on the calculations used to derive UGEP please refer to DWAF (2005). The UGEP distribution is shown in Figure 5.9. The UGEP calculated for the study area 15 237 107 m3/a. This value is based on normal rainfall conditions. Under drought conditions this value may reduce by approximately 30%. If the UGEP is adhered to, there is unlikely to be an impact on the outflow to the marine environment, however, the positioning of the abstraction is crucial to ensure there is no impact on these outflows. Essentially the drawdown zones from the abstraction must not reach the shoreline. Numerical modelling will enable the optimal location of the abstraction boreholes to ensure there is no impact in the groundwater flows into the lagoon. Woodford (2003, cited in WCDM, 2005) determined a combined exploitation potential for the LRAS and EAS of approximately 14.0 million m3/a. These estimates appear in line with estimates generated in the Seyler et al (2016) study and listed in this report.

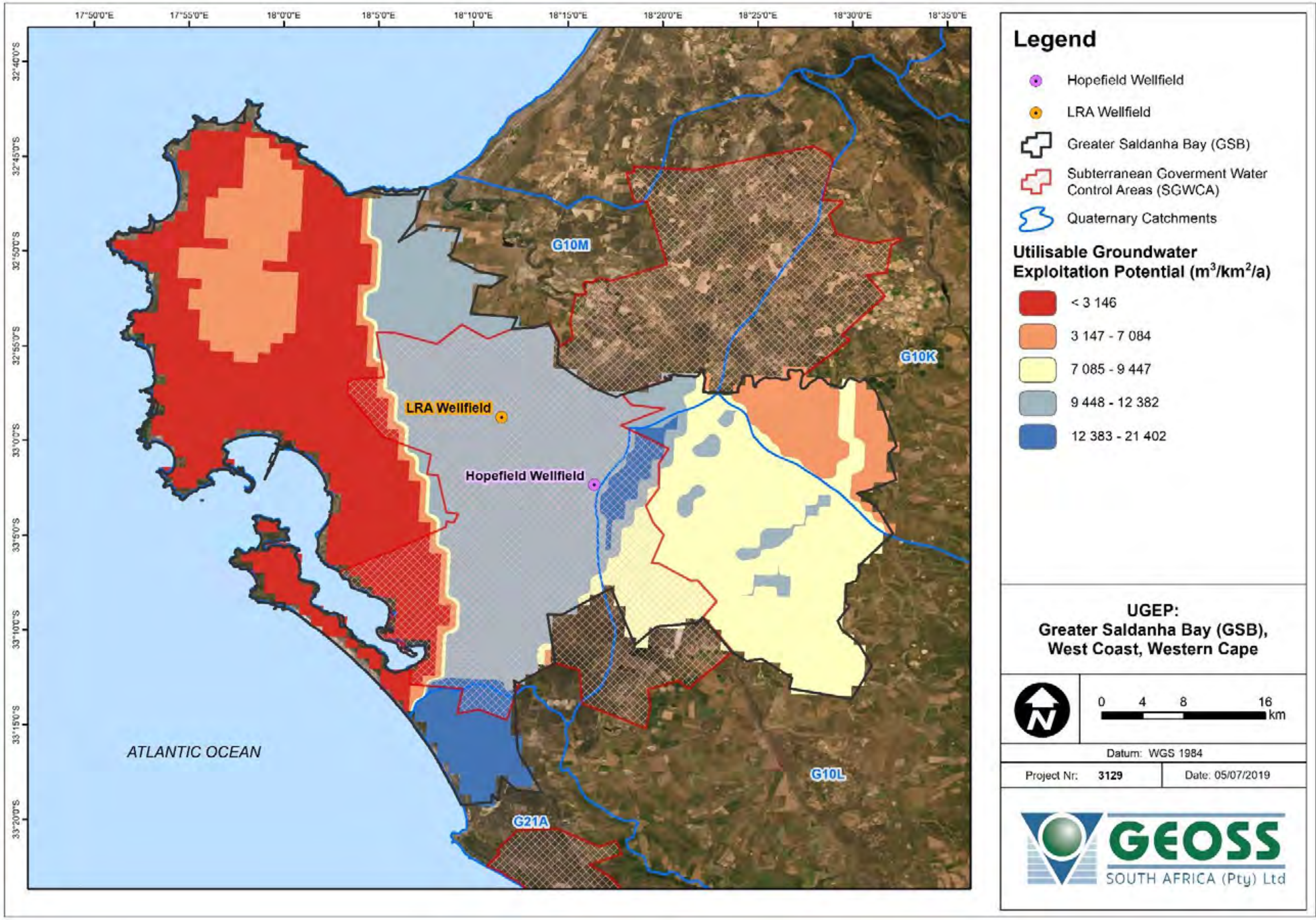


Figure 5.9. Utilisable Groundwater Exploitation Potential map.

5.5 Groundwater, aquifers and wellfields

5.5.1 Agricultural use – non-aquifer specific

The main existing groundwater use in the study area is for agriculture in the form of irrigation. Due to the average to poor groundwater quality characteristic of the area, especially along the coastline, irrigation use is limited to some degree. However, many production farms in the study area are reliant on groundwater. A search of available data was conducted, and the information indicates that the authorised volumes relate to non-municipal abstractions. The total registered groundwater use in the area is 1 583 994 m3/a. It is important to note that the data is limited to registered volumes up to 2016 and the number of groundwater authorisations in the area has since increased. The registered agricultural usage is 1 529 744 m3/a (96.6%).

5.5.2 Langebaan Road Aquifer (LRA)

Aquifers in the area were studied many years ago – large scale drilling and exploration work was carried out by DWAF from the mid-1980s for a period of 10 years. The Langebaan Road Aquifer (LRA) (Figure 5.10) is located between the lower Berg River and Saldanha Bay. The LRA was developed as part of potential emergency supply, but also as a long-term water supply to Saldanha Bay Municipality (Nel, 2018). The existing wellfield at the LRA was officially established for the DWS in 1998. The Langebaan Road wellfield was then expanded for the purposes of emergency measures.

The LRA wellfield forms part of the Berg River hydrogeological unit of the Sandveld Group Aquifer. The geological depositional environment has a direct bearing on the most significant groundwater occurrences (Nel, 2018). The Langebaan Road Aquifer System (LRAS), is situated above the northern paleo-channel.

The LRAS is formed by the thick sand and gravel layers of the Elandsfontein Formation. The confining layer is formed by the Elandsfontein clay and peat. A shallow, unconfined aquifer of lesser extent exists in the top Bredasdorp calcerous layers.

The clay layer that separates the Elandsfontein Formation and the Varswater Formation creates (semi) confined conditions in the basal gravels (GEOSS 2010). The confined Lower Aquifer Unit (LAU) is the most significant aquifer in the Langebaan Road area and is mostly restricted to paleo-channels, which defines the LRAS and the EAS. For the Langebaan Road area, the geology and hydrogeology are summarized in Table 5.3 (Weaver 1998a).

Table 5.3. Summary of the geology and hydrogeology of the Langebaan Road Wellfield

Lithostratigraphic Unit		Aquifer Type	Approximate thickness (m)	Transmissivity (m^2/day)
Formation	**Member**			
Bredasdorp	Sand, calcrete	Unconfined	10 to 20	< 100
Elandsfontein	1) Clay 2) Sand and gravel	1) Aquitard 2) confined	20 to 30 40 to 60	<1 Approx. 1 000
Granite	-	Bedrock	-	-

The LRA wellfield became operational in 1999, providing an additional 1 460 000 m3/a to the water supply for Saldanha Local Municipality (Nel, 2018). In 2008 and 2009, artificial recharge was tested at the wellfield, showing that it could be a viable option for the long-term sustainable management of the groundwater resources. The scheme has been hampered by vandalism. It is intended to abstract more groundwater from the LRA. Based on field data from various studies in the area over the last 20 years, the water quality of the aquifer is good (less than 100 mS/m). Saldanha Bay Municipality will be responsible for the monitoring of LRA as the License holder once the wellfield-extension license is issued. SBM will however outsource the monitoring to a specialist geohydrologist on tender or quotation basis (*pers. comm*., D. Wright, 2 July 2019).

The LRA has been intermittently in operation for 20 years. The Resource Quality Objective's (RQOs) (Department of Water and Sanitation, South Africa 2018) state that there should be no long term "mining" of groundwater and that groundwater levels must recover outside of drought periods. This clear long-term objective needs to be clearly highlighted. The Department of Water and Sanitation (DWS) have been monitoring groundwater levels in the LRA area as well as much wider. The distribution of the monitoring boreholes listed is shown in Figure 5.11. The monitoring data is from both the lower and upper aquifer systems. From the graphs (in close proximity to the older production boreholes) the stability of the groundwater levels can be seen after the initial drop in water levels associated with the start-up of abstraction, however, the groundwater abstraction has not been continuous and this is reflected in the groundwater levels. Managed aquifer recharge, or MAR, refers to the intentional recharge of water to aquifers for subsequent use or environmental benefit. MAR offers numerous benefits, including storage to improve security of water supply and natural treatment. The success of the Managed Artificial Recharge can be seen in the increase in water levels in the period 2013/2014/2015. Managed Artificial Recharge needs to be operationalised again. The LRA wellfield is to be started up again soon and on-going monitoring is critical. At the time of writing this chapter, an exact breakdown of when the LRA wellfield was in operation or dormant was not available.

5.5.3 Hopefield wellfield

The recently developed Hopefield wellfield occurs to the south-east of the Langebaan Road Aquifer wellfield (Figure 5.11) and is considered to be part of the LRA.

The following is based on all the field work and interpretation of the data at Hopefield, as per Nel (2019):

- The groundwater in the aquifer appears to be of good quality, with EC-values between 54 and 63 mS/m.
- A total of 12 production boreholes were drilled based on the geophysical profile results. A combination of thick sand and deep weathering areas were targeted. High conductive clayey areas were avoided.
- An additional 6 monitoring boreholes were drilled to provide data for local management of the aquifer.

The Hopefield wellfield was developed for groundwater supply for the Saldanha Bay Municipality for a supply of 1 642 500 m3/a. Monitoring boreholes have also been set up to serve as pilot boreholes to characterise the aquifer conditions, to monitor aquifer performance, as well as assess the impact on the surrounding users. DWS provided data for the region, and the data from a few DWS boreholes in close proximity to the Hopefield Wellfield (Figure 5.11) has been evaluated. A declining groundwater level trend is evident from the boreholes and the largest decline is approximately 2.5 m over a period of 10 years. In the near future, the results will be comprehensively analysed by a specialist hydrogeologist appointed by SBM. Once again, the SBM will ultimately be responsible for the overall management and monitoring of groundwater abstraction. Decisions will be made by SBM in conjunction with the inputs from the appointed geohydrologist (*pers. comm*., D. Wright, 2 July 2019).

5.5.4 Elandsfontein Aquifer System

The Elandsfontein Aquifer System has the following boundaries: the Langebaan Lagoon in the west, the Darling batholith in the south, the Brak and Groen Rivers to the south-east and east, respectively, and the zero-flow boundary to the north. The Elandsfontein Aquifer System thus occurs to the south of the Hopefield Wellfield (Figure 5.10) and comprises an Upper Aquifer System and a Lower Aquifer System. The shallow, unconfined aquifer is made up of the more calcareous layers of the Bredasdorp Formation. The Elandsfontein Aquifer System also has a confined aquifer formed by the Elandsfontyn sand and gravel underneath a thick confining sequence of clay and peat. The groundwater quality and yields are good within both Upper and Lower Aquifer Systems. The Elandsfontein Aquifer System (EAS) is situated above the southern palaeo-channel.

The Elandsfontein Phosphate Mine is situated within this aquifer, however, the nett abstraction of groundwater is minimal as the groundwater abstracted is recharged within a closed system. In the vicinity of the open pit active dewatering is taking place and then re-injection of the abstracted groundwater 2 km down-gradient of the open-pit. The anticipated impacts of the drawdown and injection have been numerically modelled, and the modelled results were included with the Elandsfontein Mine Water Use License Application. The License was approved based on the modelled and anticipated abstraction and recharge impacts. Thus, the actual monitoring the mine is doing of 28 dedicated monitoring boreholes is closely compared to the modelled results. At this stage there is close correlation between modelled and actual results.

As mentioned above, to date, the groundwater level decline and rise (down-gradient of the injection boreholes) is encouragingly very similar to the modelled results. The Water Use Licence compliance conditions for the mine stipulate comprehensive groundwater level and quality monitoring and the mine is adhering to these conditions.

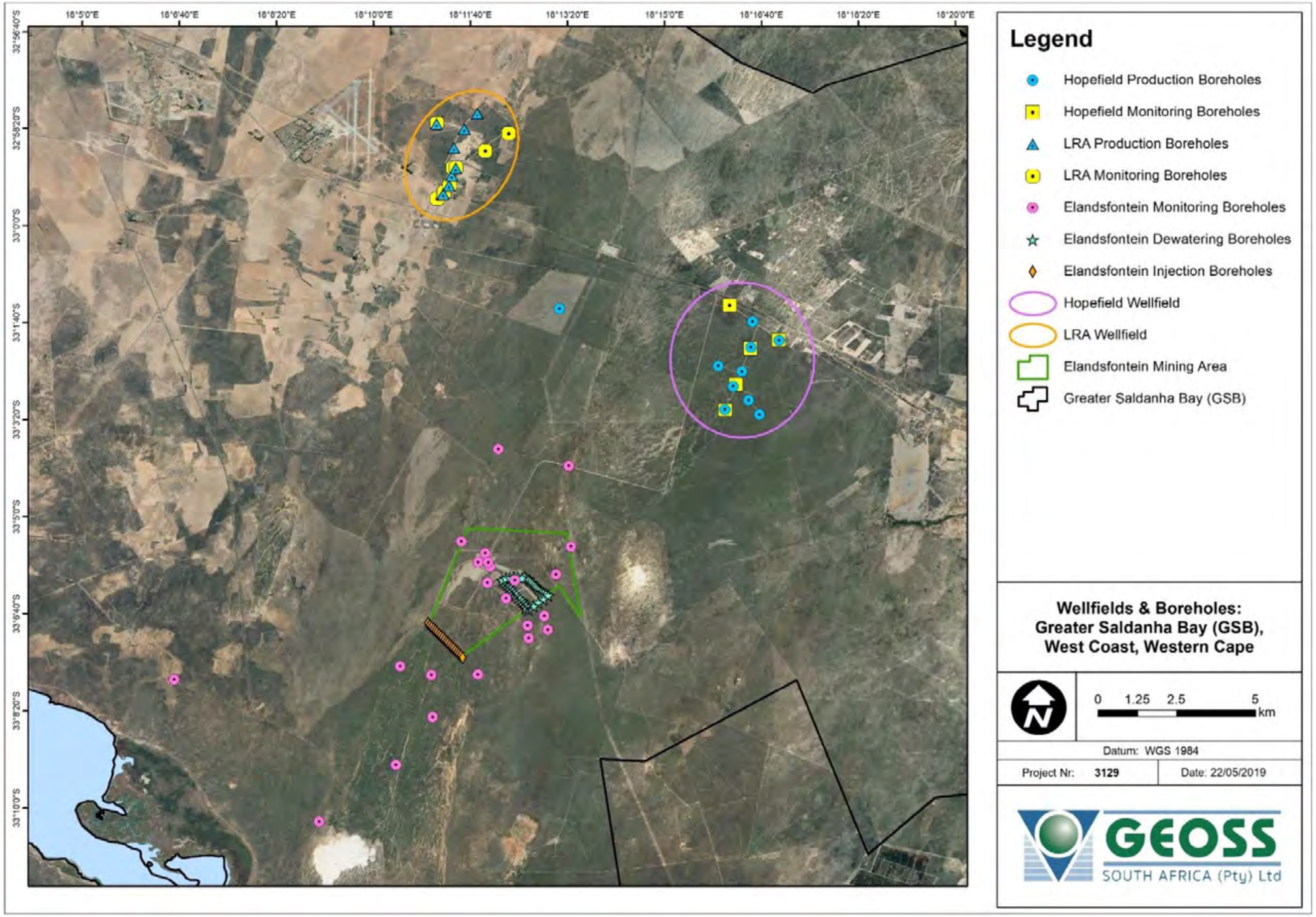


Figure 5.10. Map outlining the approximate locations of the Elandsfontein property, Hopefield wellfield and Langebaan Road Aquifer wellfield.

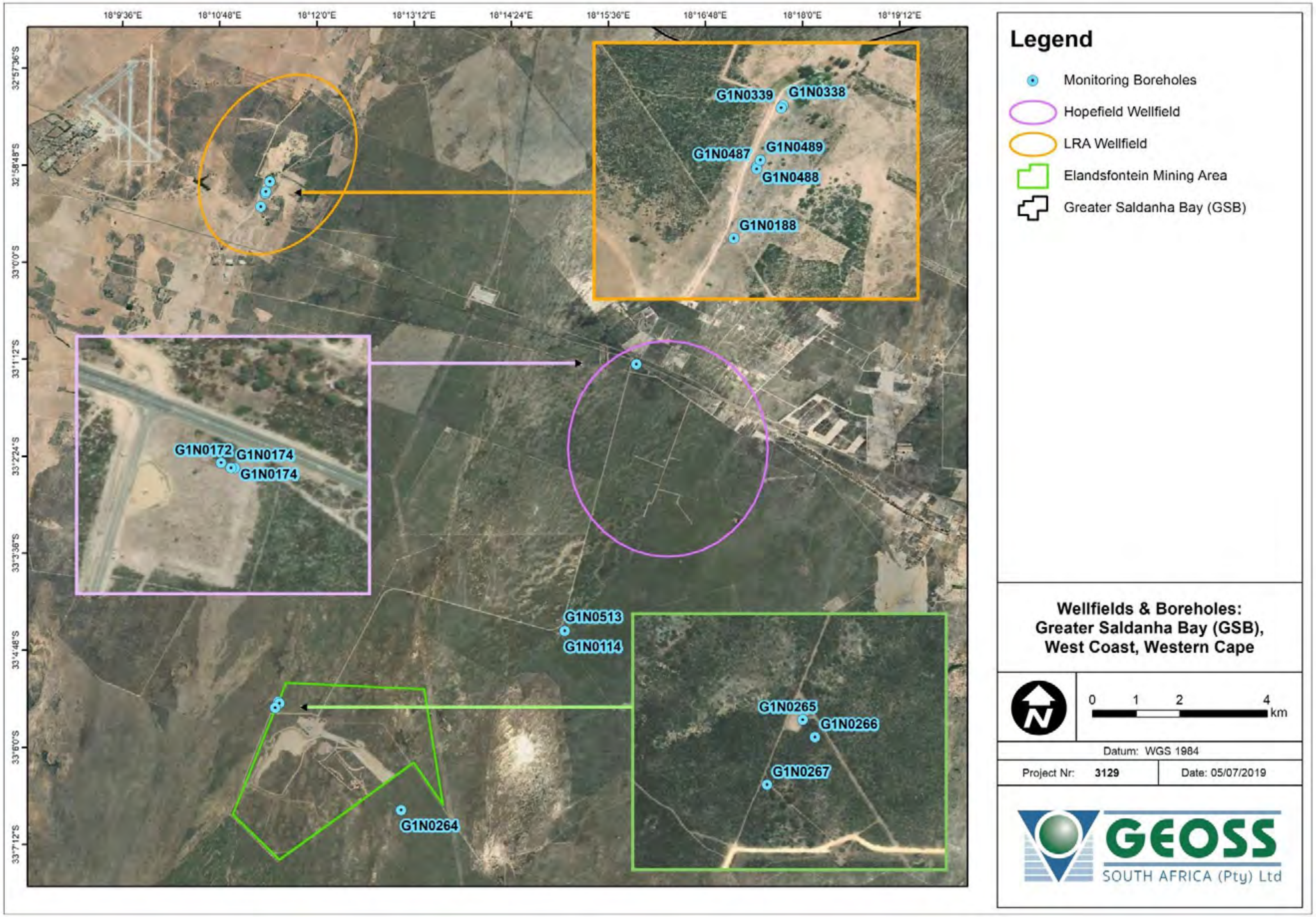


Figure 5.11. **DWS monitoring boreholes near the Langebaan Road Aquifer and Hopefield wellfields and the Elandsfontein mine.**

5.6 Groundwater- surface water interaction

It is noted in the Berg River study by Clark and Radcliffe (2007), that groundwater contributes to base flow in the headwaters of the catchment and along its lower reaches. It is estimated that the contribution of groundwater to river base flow is in the order of 10-20% of mean annual rainfall. Similarly, the work undertaken by GEOSS (2006) indicates the importance of groundwater in maintaining surface water systems.

The shallow depth to groundwater indicates that it is a source of water to river ecosystems in low-lying areas. Nodes of importance were defined by GEOSS which indicate locations where there is a high probability of groundwater contributing to surface water flow. Groundwater discharge zones were also identified, which show areas where surface water is supported by groundwater.

In the GEOSS (2006) study the following point is made (pg. 15): "…*the health and maintenance of surface water systems are largely dependent on the protection of groundwater, both in terms of its depth below surface and quality. Within the study area there are also a number of non-riverine wetland systems that have important interactions with groundwater. Wetlands are regulators of water flow and water quality. When water moves from most types of wetlands into an aquifer, water is filtered and cleaned. Water is then transferred to alternate wetlands with more stable biological communities through the rising of the groundwater to the surface (Scialabba, 1999). Thus, groundwater at discharge sites and discharge zones needs to be managed and protected to ensure the continued viability of freshwater systems*."

However, please note for this study area, the lower reaches of the Berg River occur to the north of the northern study area boundary, thus the groundwater contribution to base flow within the study area is negligible.

5.7 Legal status of groundwater use

The existing LRA is licensed to abstract 6 ML/d (2.19 Mm3/a) and it is possible that for the LRA extension will be licensed for an additional 8 ML/d (2.92 Mm3/a). Thus, it is possible that 14 ML/d (5.11 Mm3/a) can be abstracted from the LRA. The licence for the Hopefield wellfield has not yet been finalised, however, an abstraction of 4 to 5 ML/d (1.46 to 1.83 Mm3/a) has been applied for.

Private groundwater users abstract in the region of 1.58 Mm3/a for agriculture mainly; as of 2016, however, this is likely to have increased since the drought. Farmers and other landowners in the GSB area have invested in groundwater development to supplement municipal water supply given the uncertainty of water resources in light of the drought. Further to large-scale abstractions, many commercial industries in particular, and farms, have registered water use on their respective properties.

5.8 Groundwater importance and provision of ecosystem services

This section provides an overview of the importance of groundwater for the region and the ecosystem services it provides. As previously mentioned, the main existing use of groundwater in the area is for agricultural use. It should be noted that groundwater does play a role within the urban and environmental sectors of the area.

5.8.1 Urban/Industrial

Because of the overall poor quality of the groundwater in the study area, it is usually not the preferred source of water for the urban and industrial activities around the Saldana Town, Vredenburg and Hopefield. Most of the boreholes located in the industrial and urban areas are only for monitoring of groundwater quality. Because of the general shallow depths of the primary aquifer (10-20 mbgl) and the permeability of sandy soil, contamination is always a high threat, thus explaining why the industries need to monitor the groundwater quality.

Although the groundwater was not outlined as an important water resource for urban and industrial areas, it is important to note that there might be some industries that have started abstracting groundwater. It can also be assumed that low volumes of groundwater could be abstracted in the residential areas for private use.

5.8.2 Agriculture

A large economic sector in the GSB area is agriculture, in the form of livestock farming, and to a larger extent – crop irrigation. However, the West Coast is generally not regarded as being of high agricultural potential in relation to the rest of the Western Cape region.

Water availability in particular, combined with soils of relatively low fertility are the limiting factors in this regard. There is predominately grazing and mixed farming between Hopefield and Langebaanweg, and between the West Coast National Park and Berg River in the north (GIBB, 2017). Grain and mixed farming take place in the Saldanha, Vredenburg and Velddrif areas. Areas of highest agricultural activity are located in a band between Saldanha Bay and St Helena Bay, along the Berg River and in the vicinity of Hopefield and southwards from this town.

Most crop farming in the study area comprises dryland agriculture. Irrigated lands occur in the vicinity of Hopefield, where farmers obtain water from the Berg River. Irrigated agricultural land is limited to this vicinity of the study area (GIBB, 2017).

5.8.3 Mineral – Elandsfontein Phosphate Mine

The Elandsfontein Phosphate Mine (620 ha footprint) is situated midway between the towns of Hopefield and Langebaan. Dewatering began in February of 2017. It is located in an area where essentially no development (be it agricultural or otherwise) has occurred adjacent to SANParks property. The mine is located on two aquifers, the Upper Aquifer Unit and Lower Aquifer Unit of the Elandsfontein Aquifer. In this area the aquifers are high yielding and of good quality. The groundwater level is in the region of 20 metres below ground level. The phosphate is within the Muishond Fontein Pelletal Phosphorite Member of the Varswater Formation, however, this member is within the aquifer (GEOSS, 2014). The open cast mine is thus surrounded by a ring of 36 dewatering boreholes and these boreholes have thus lowered the groundwater level to enable mining of the phosphate layer. An essential stipulation of the Water Use License (WUL) issued for the mine, is that the abstracted groundwater has minimal contact with the atmosphere and is returned back into the ground. There are 20 injection boreholes 2 km down-gradient of the open pit mine and the abstracted groundwater is being successfully recharged into the Elandsfontein aquifer. This statement is based on the results of groundwater monitoring that occurs in the area.

The mine is in a state of "care and maintenance" as the processing plant has had difficulties in processing the ore excavated from the edge of the ore body coupled with a sub-economic phosphate price. Modifications to the plant are currently underway and it is expected that the mine will resume production in the 4th quarter of 2021.

Interestingly, the numerical modelling of the aquifer prior to the approval of the WUL indicated that the Elandsfontein Aquifer in the area of the mine is so high yielding that the groundwater does not need to be recharged and could be used (for municipal supply). The modelling showed that even if the groundwater was used, the freshwater inflows into the southern end of the Langebaan Lagoon would not be affected (GEOSS, 2015 and SRK, 2016). Nonetheless, the groundwater has to be recharged as stipulated in the WUL. Thus, the potential threat of the mining operation on the freshwater inflows into the lagoon is considered to be very low.

A total of 28 dedicated monitoring boreholes are situated up- and down- gradient of the open pit and are illustrated in Figure 5.12 and Figure 5.13. These boreholes constitute a continuous monitoring network since the onset of mine dewatering in 2017, and regular groundwater monitoring takes place from the boreholes. The purpose of monitoring is to ensure that the abstraction and injection of groundwater do not result in deterioration of groundwater quality and quantity, effectively altering groundwater supported ecosystems. Monitoring data is regularly compared to projections for initial numerical model and long-term data will be used to recalibrate the model in future.

A selected number of these boreholes are equipped with continuous water level sensors and the data is transmitted to an open access database. At all boreholes, water levels are measured monthly on a quarterly basis, groundwater samples are also collected for chemical analysis. Monthly reports based on the findings of these analyses are generated. Potential thresholds of concern have been identified for the boreholes and are continuously compared to physical and chemical analyses to ensure that thresholds are not exceeded.

Long-term monitoring illustrates the initial steady decline of water levels in the dewatering boreholes and increase in water levels at the injection boreholes, which have now stabilised. There is a decrease in water level decline with increasing distance from the mine, and water levels indicate close correlation to initial projections.

In addition to monthly groundwater levels, groundwater temperature and EC values are recorded. Long-term baseline EC monitoring provides insight into natural fluctuations of the groundwater chemistry as well as any effects imposed by mining activity. Changes in EC are largely due to seasonal variation, as well as the mobilisation of salts due to artificial fluctuations of the water level (i.e. dewatering and injection) whereas temperature across all the monitoring boreholes have remained stable. Water quality in terms of inorganic (major anions and cations) and organic (microbiologic and hydrocarbons) chemistry, is analysed on a quarterly basis. Mining related activities and water use have not resulted in any microbiological or hydrocarbon contamination of the aquifer. Quality fluctuations are largely recorded in major ion concentrations, particularly sodium, chloride and occasionally nitrate. Salt concentrations remain relatively low and like EC, are attributed to natural variation and dewatering activities. Chemical monitoring reveals that previously established thresholds of concern are not continuously exceeded. Both water level and chemical monitoring further indicates that fresh flow into the lagoon is not affected by the dewatering activities.

All data collected to date at Geelbek boreholes, BH1; BH2 and BH2a are displayed in Figure 5.14, Figure 5.15 and Figure 5.16. These boreholes are considered most relevant to this study. The borehole details are provided in Table 5.4. There is considerable additional information about the boreholes as they were drilled as part of a large research project into groundwater / surface water interaction and the reader can access the relevant references for additional information on the project and borehole details (Saayman, 2005; GEOSS, 2019).

Table 5.4. Selected Geelbek boreholes and associated details.

Borehole ID	Latitude (WGS84)	Longitude (WGS84)	Borehole Depth (m)
GB_BH_1	-33.186025	18.13190	81
GB_BH_2	-33.19332	18.12690	45
GB_BH_2A	-33.18601	18.13195	13

The relative stability of the groundwater is clearly evident from the graphs, while the data from boreholes located in close proximity to the Lagoon provides crucial data for assessing whether these conditions change with time. In recent times, water quality at all three boreholes has improved slightly (i.e. the salinity levels are dropping). This is attributed to the better rainfall conditions that have been experienced in recent years compared to the drought period of 2015, 2016 and 2017. The on-going monitoring of these sites (and all the dedicated Elandsfontein Mine monitoring sites) is important.

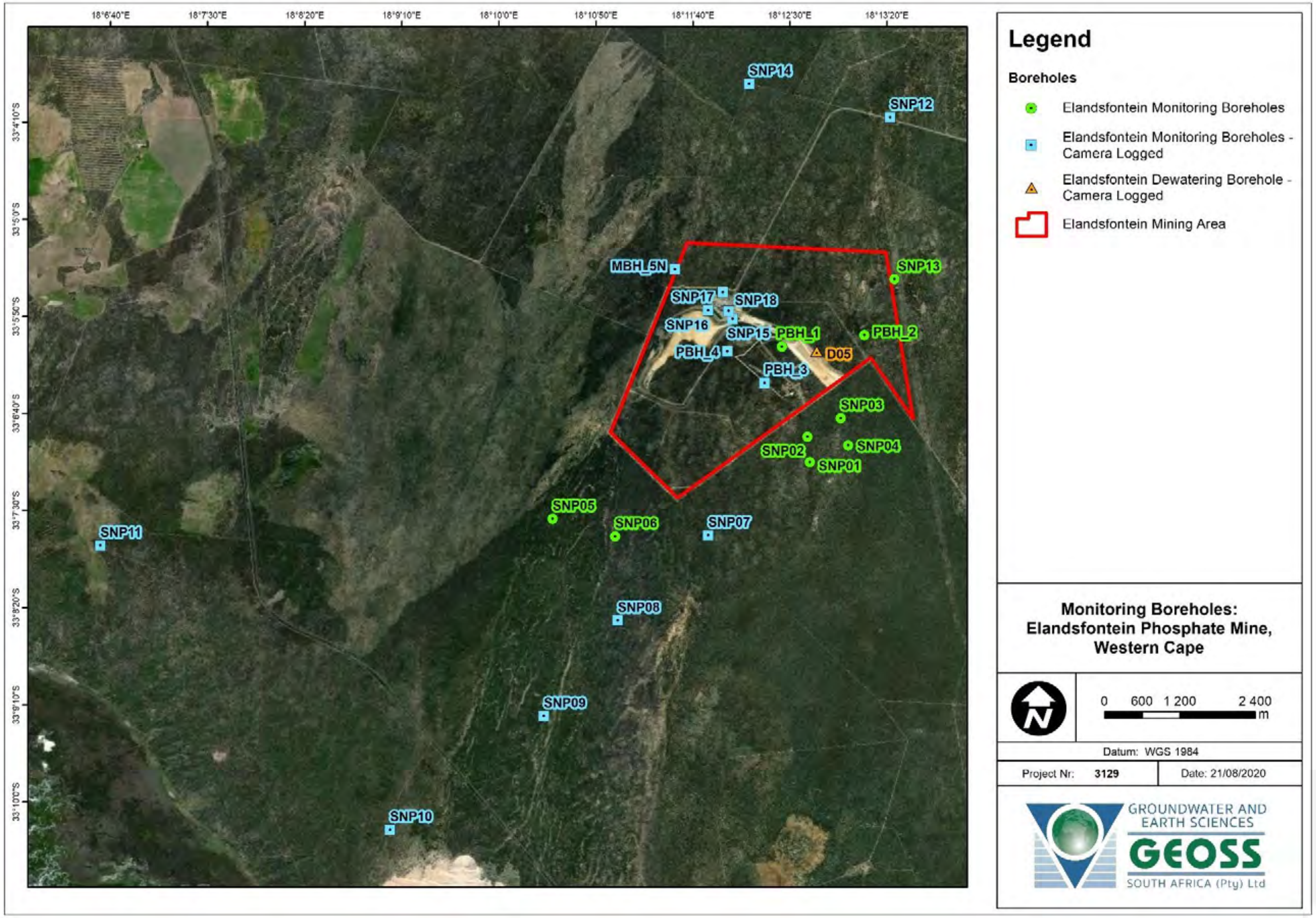


Figure 5.12. Elandsfontein monitoring boreholes, Langebaan Road Aquifer and Hopefield wellfields.

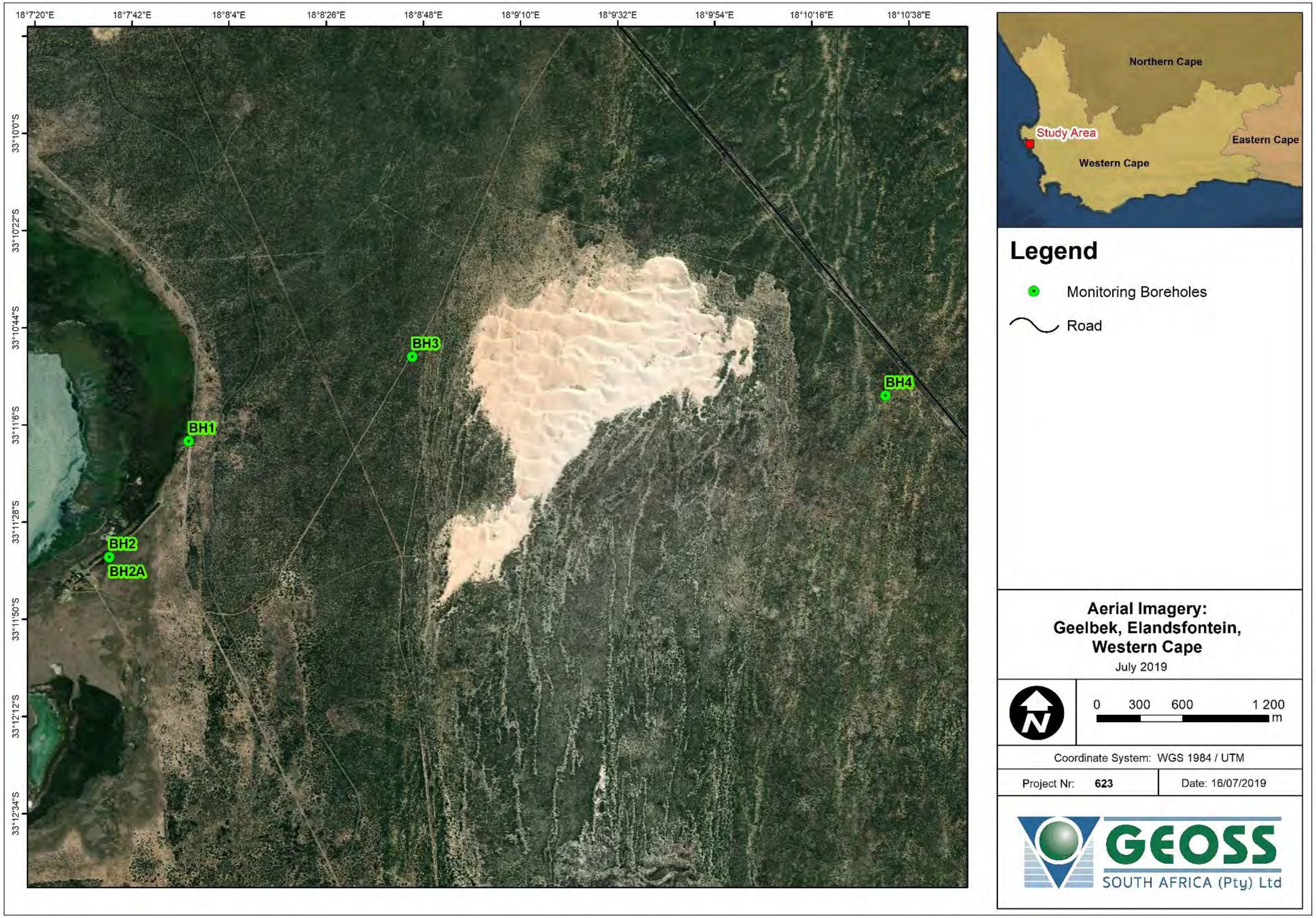


Figure 5.13. **Geelbek monitoring boreholes, Langebaan Road Aquifer and Hopefield wellfields, near the Elandsfontein mine.**

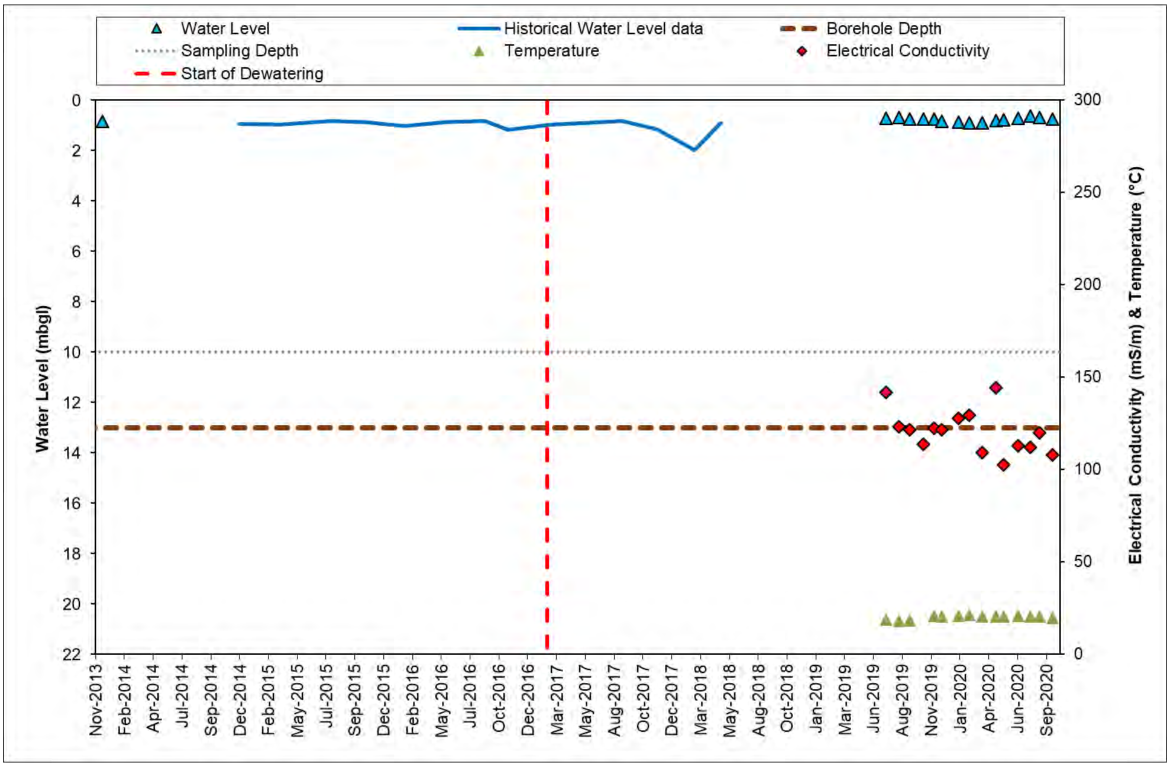


Figure 5.14. Geelbek BH1 Long-term water level, temperature and EC monitoring.

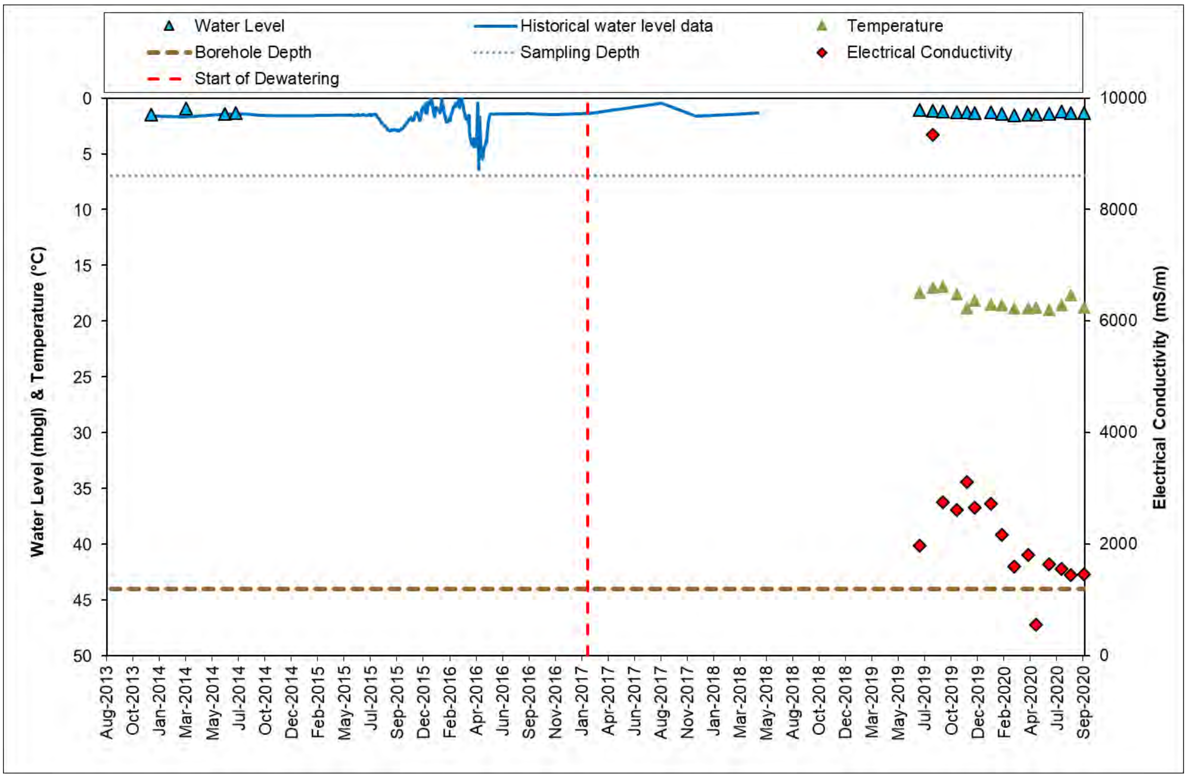


Figure 5.15. Geelbek BH2 Long-term water level, temperature and EC monitoring.

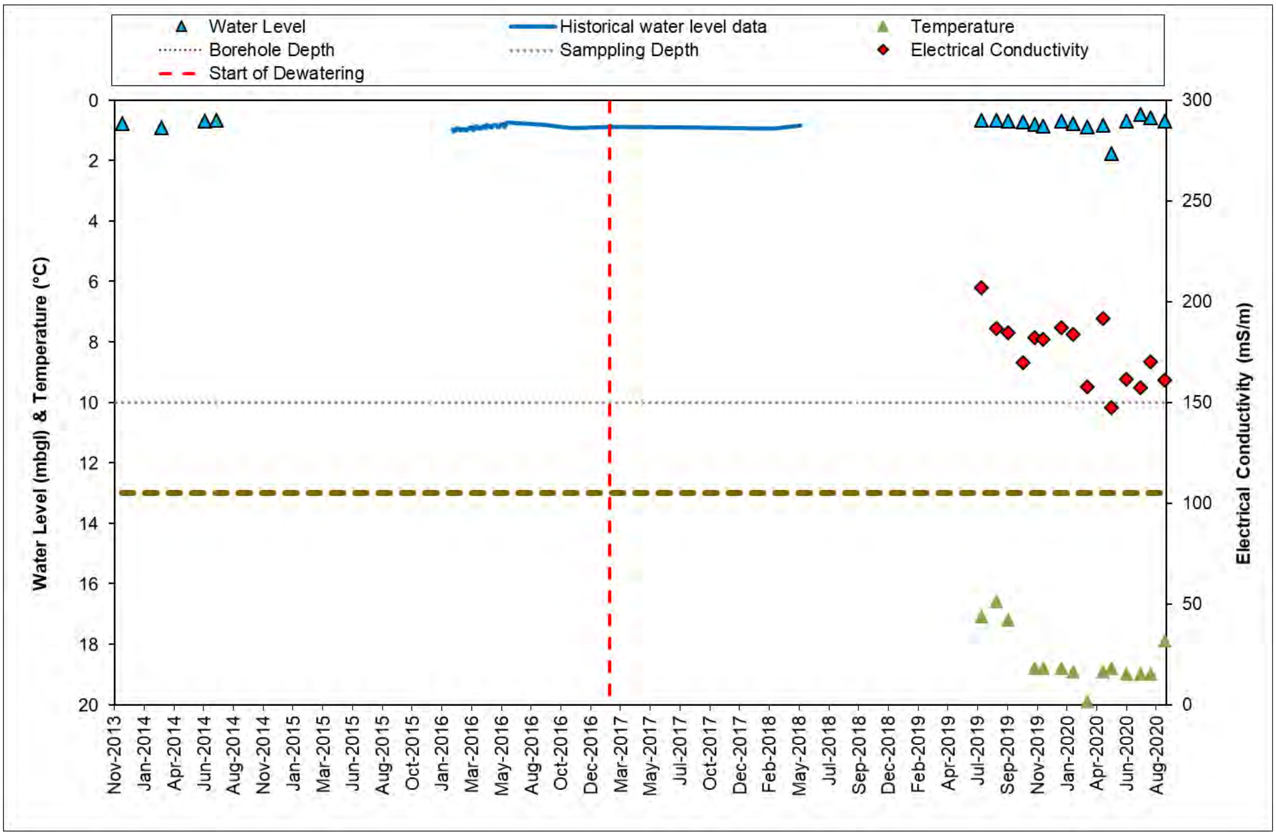


Figure 5.16. Geelbek BH2A Long-term water level, temperature and EC monitoring.

Given the sensitivity of the Langebaan Lagoon, and in spite confidence expressed by a range of specialists that groundwater use by Elandsfontein mine is unlikely to impact on the lagoon, Kropz Elandsfontein in conjunction with the Saldanha Bay Water Quality Forum Trust (SBWQFT) has also been monitoring various biological and physico-chemical variables associated with Langebaan Lagoon to establish an appropriate baseline against which any potential future changes in the Lagoon can be benchmarked. This includes monitoring of temperature and salinity (see below) and biota (see Chapter 9) as well as macrophytes (see Chapter 8) around the top end of the lagoon.

Monitoring of temperature and salinity at the head of the lagoon was initiated in September 2016 using a Star ODDI Salinity, Conductivity, Temperature and Depth Logger. This instrument was configured to take measurements of temperature, salinity and depth at ten-minute intervals. The instrument was retrieved, data downloaded and redeployed at approximately 3-month intervals following this time. Some modifications to the mooring and its deployment site were necessary over the course of the monitoring undertaken to date. The instrument was initially fixed to a mooring block on the edge of the channel near Geelbek at the head of the lagoon during September 2016. The instrument was retrieved, and data downloaded for the first time in December 2016. Data records from this period up until last years' report (August 2019) have been presented in previous State of the Bay reports and are not repeated here. The Star ODDI was replaced with an Aqua TROLL 200 data logger at the end of August 2019 and quality of data collected has improved dramatically since then. The instrument was unfortunately removed (stolen) at some point between January and April 2020 (theft was discovered when we went to service the instrument in April 2020) but has since been replaced, with the new instrument having been deployed in August 2020 (the delay in securing an new instrument being related to the global COVID-19 pandemic). Data collected over the period August 2020 to January 2020 and August – September 2020 are presented on Figure 5.17 to Figure 5.23 below, along with data on tidal variation in Saldanha Bay obtained from the South African Navy Hydrographer (based on measurements made at the Port of Saldanha) and precipitation data provided by South African Weather Service from their weather station at Langebaanweg Airforce Base.

The data record for the period August 2019 to January 2020 show a clear seasonal trend (Figure 5.17), with water temperature gradually increasing across the entire period, while records for individual months (Aug 2019 to January 2020 and August to September 2020) show clearly the day to day oscillations in temperature, and salinity and water level (Figure 5.18 -Figure 5.23). The diurnal fluctuations in temperature are similar across all seasons, with temperatures increasing over the course of the day, peaking in the early afternoon, then declining through the afternoon and night, reaching a minimum at the time of sunrise each day. The trend in salinity is more interesting though, exhibiting a similar diurnal oscillation to that for temperature, but this oscillation is linked to the state of the tide (not the time of day) and changes through the year. In winter (August, Figure 5.18 and Figure 5.23), salinity oscillates between that of normal seawater (around 35.0 PSU) at high tide and a slightly fresher state (between 32.0 and 34.0 PSU) at low tide. Salinity appears to drop as the tide recedes and is most likely linked with outflow of freshwater from the aquifer at this time. In summer, the pattern reverses with salinity increasing from that of normal seawater (35.0 PSU) at the peak of the high tide becoming hyper-saline (39 – 40 PSU) as the tide recedes (Figure 5.21 and Figure 5.22). It is likely that this is a function of increased evaporation at this time of year (linked to higher prevailing air temperatures) and that the water emerging from the marshes at the head of the lagoon becomes severely hypersaline as a result, and even though it is diluted by freshwater flowing out of the aquifer, this is not sufficient to bring the level below that of normal seawater. It is likely that this effect (development of hypersaline conditions) is quite localised at present (i.e. restricted to the extreme upper reaches of the lagoon only) but could become much more pervasive if freshwater outflow from the aquifer were to drop in future.

Also of interest in these data, there appears to be no link between rainfall and salinity levels in the lagoon which strongly suggests that variations in salinity in the lagoon are linked with groundwater inflow as opposed to surface water inflow, which is consistent with observations made by others (Smith 2017, Conrad & Naicker 2019, Nel 2019).

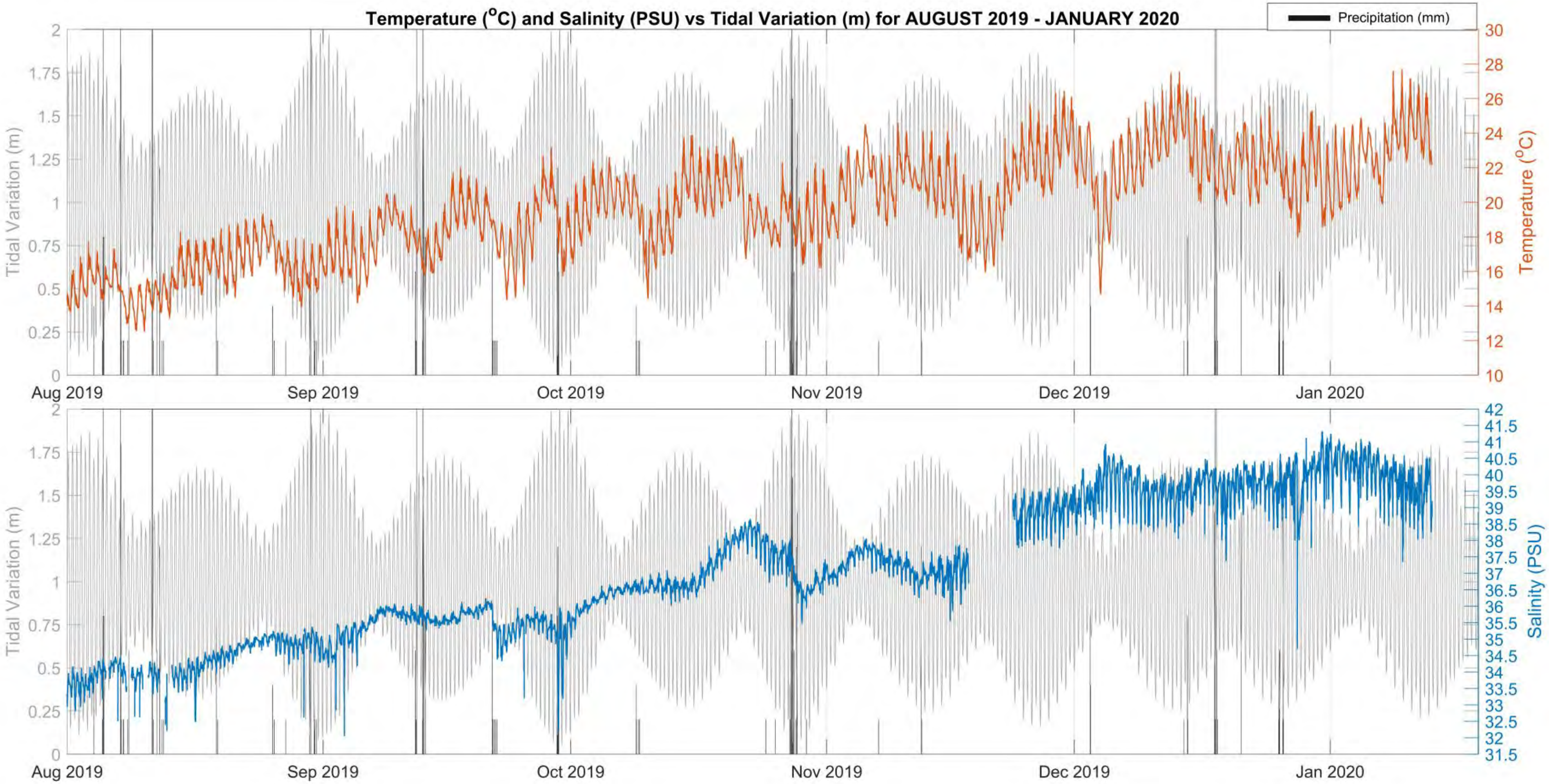


Figure 5.17. **Seasonal trend in salinity (PSU), temperature (°C), water level (m) and precipitation (mm/day) over a six month period from August 2019 through January 2020. Salinity (blue line, bottom, right axis), temperature (red line, top, right axis) recorded at ten minute intervals, tidal variation in hourly intervals (grey lines, left axis, top and bottom), precipitation (black bars). Tidal data provided by hydrographer, SA Navy, precipitation data provided by South African Weather Service.**

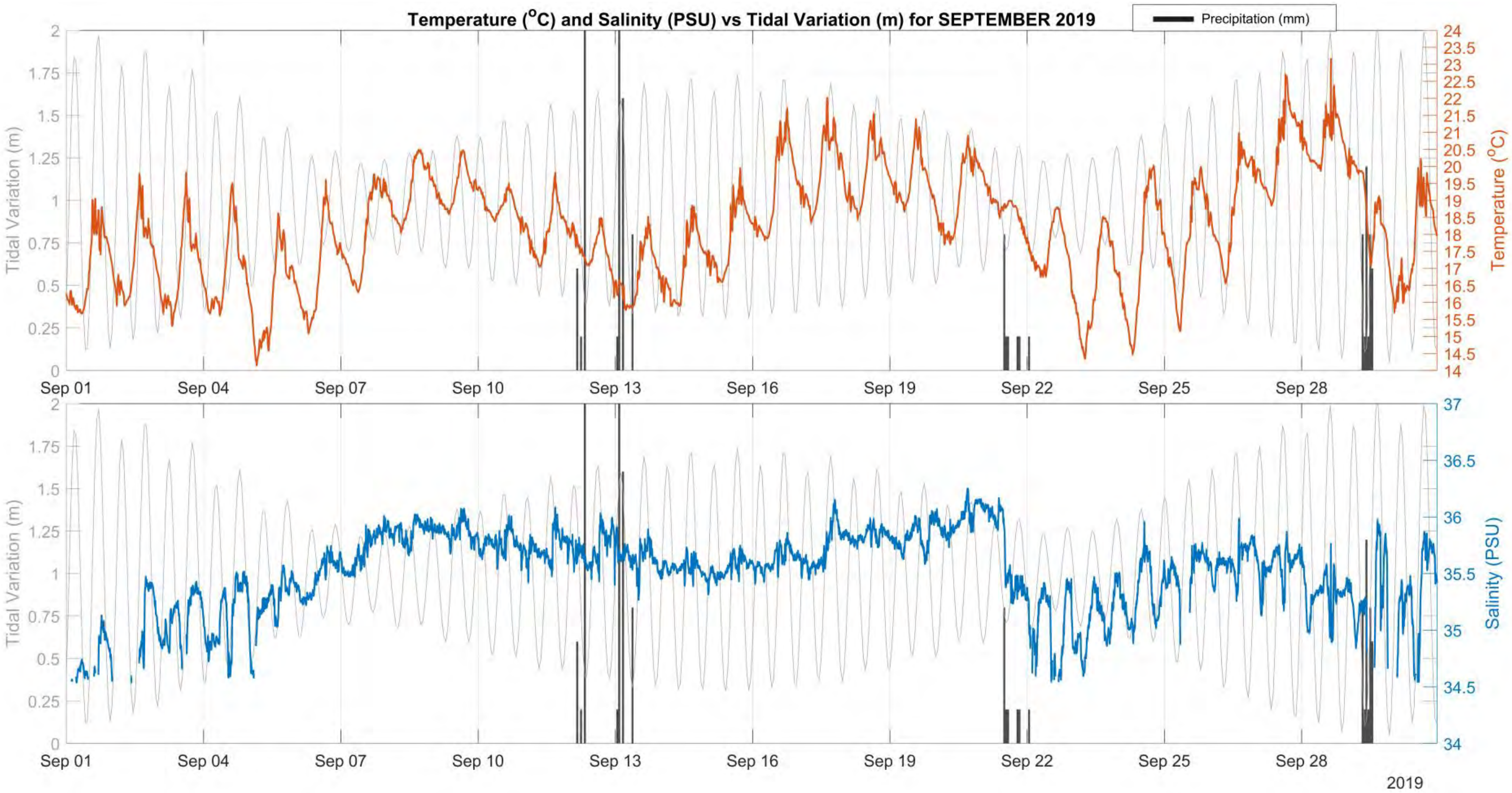


Figure 5.18. **Diurnal pattern in salinity (PSU) and temperature (°C) over a one-month period, 1-30 September 2019. Salinity (blue line, bottom, right axis), temperature (red line, top, right axis) recorded at ten minute intervals, tidal variation in hourly intervals (grey lines, left axis, top and bottom), precipitation (black bars). Tidal data provided by hydrographer, SA Navy, precipitation data provided by South African Weather Service.**

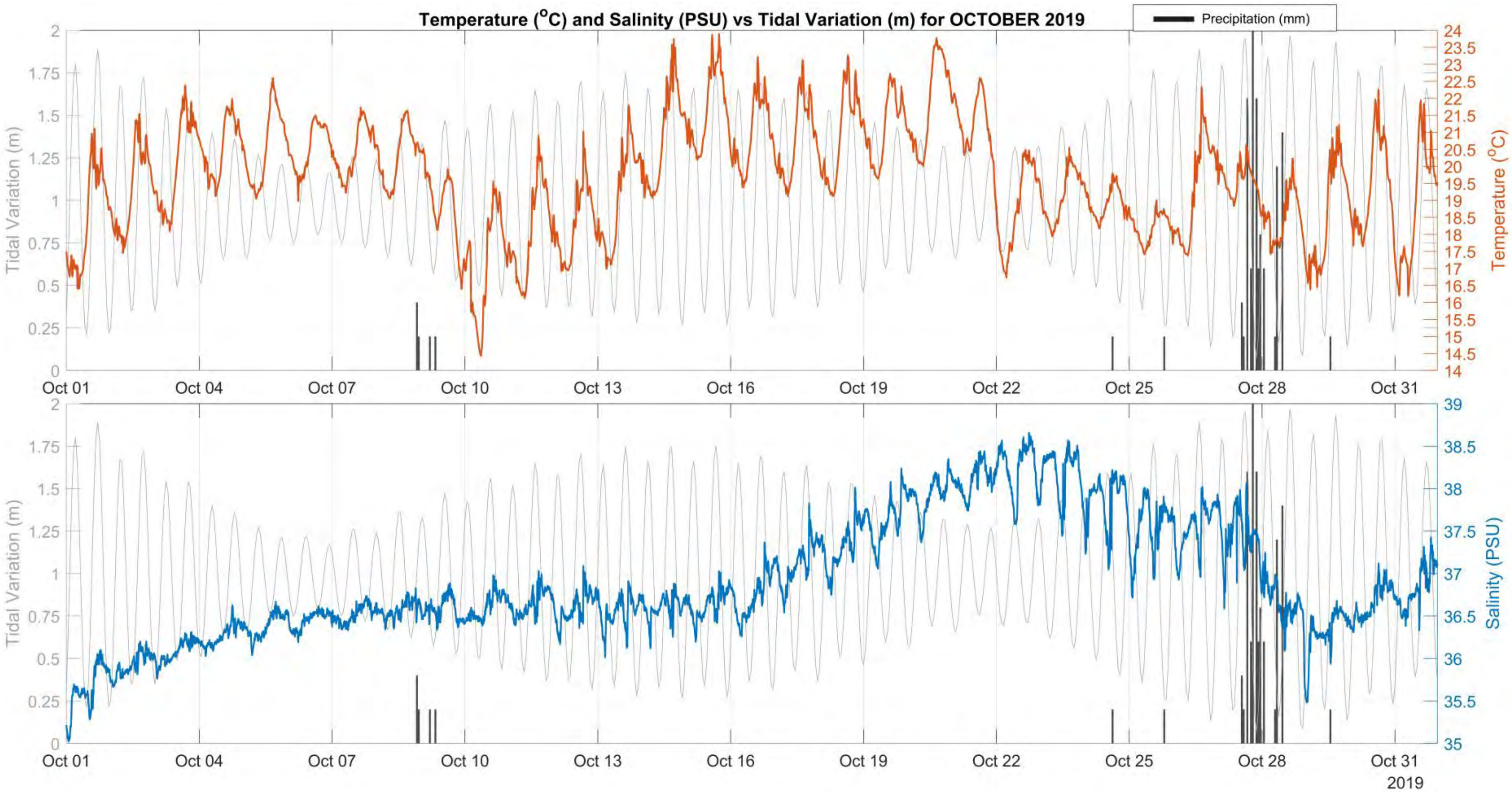


Figure 5.19. **Diurnal pattern in salinity (PSU) and temperature (°C) over a one month period, 1-31 October 2019. Salinity (blue line, bottom, right axis), temperature (red line, top, right axis) recorded at ten minute intervals, tidal variation in hourly intervals (grey lines, left axis, top and bottom), precipitation (black bars). Tidal data provided by hydrographer, SA Navy, precipitation data provided by South African Weather Service.**

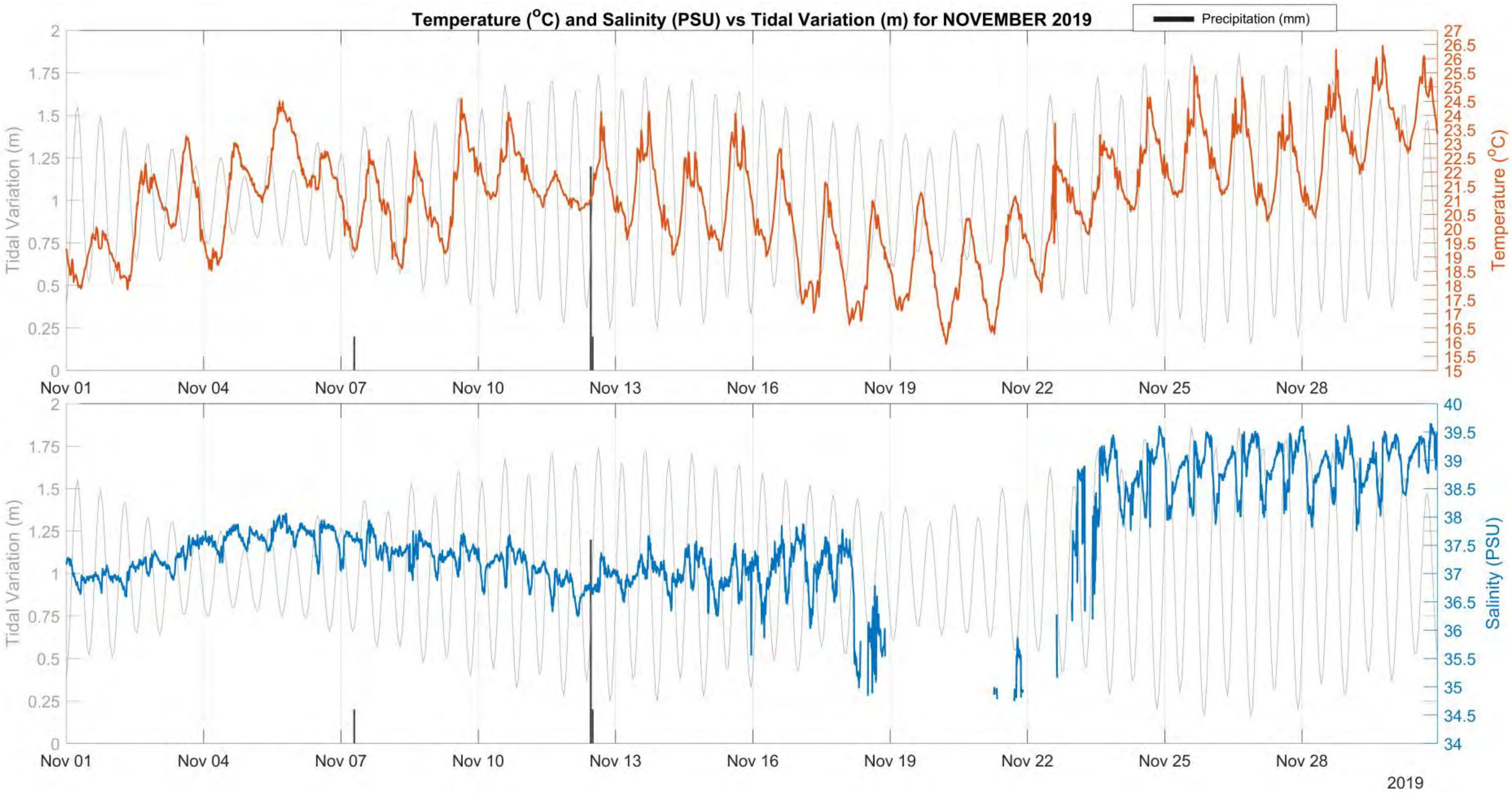


Figure 5.20. **Diurnal pattern in salinity (PSU) and temperature (°C) over a one month period, 1-30 November 2019. Salinity (blue line, bottom, right axis), temperature (red line, top, right axis) recorded at ten minute intervals, tidal variation in hourly intervals (grey lines, left axis, top and bottom), precipitation (black bars). Tidal data provided by hydrographer, SA Navy, precipitation data provided by South African Weather Service.**

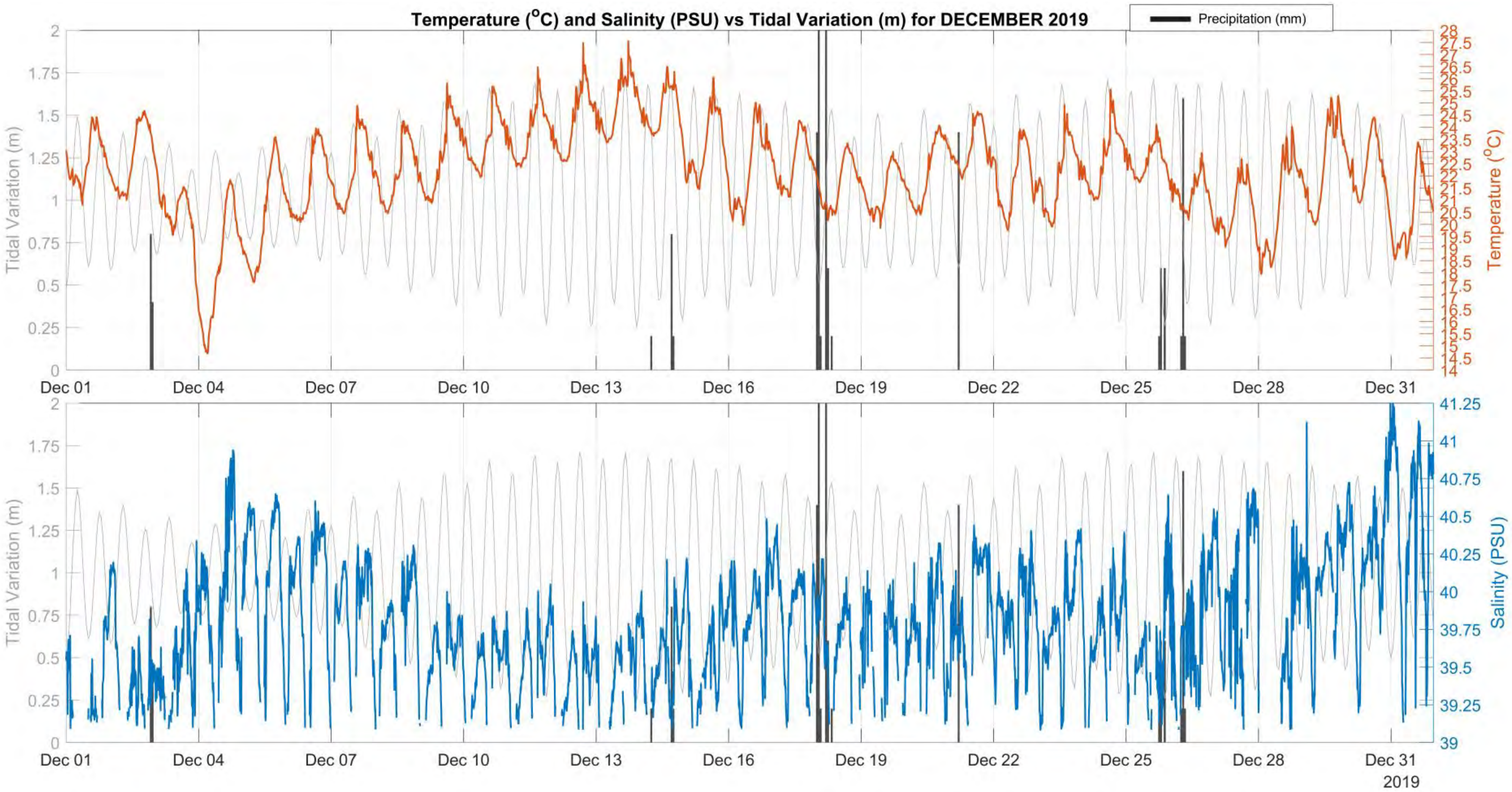


Figure 5.21. **Diurnal pattern in salinity (PSU) and temperature (°C) over a one month period, 10-31 December 2019. Salinity (blue line, bottom, right axis), temperature (red line, top, right axis) recorded at ten minute intervals, tidal variation in hourly intervals (grey lines, left axis, top and bottom), precipitation (black bars). Tidal data provided by hydrographer, SA Navy, precipitation data provided by South African Weather Service.**

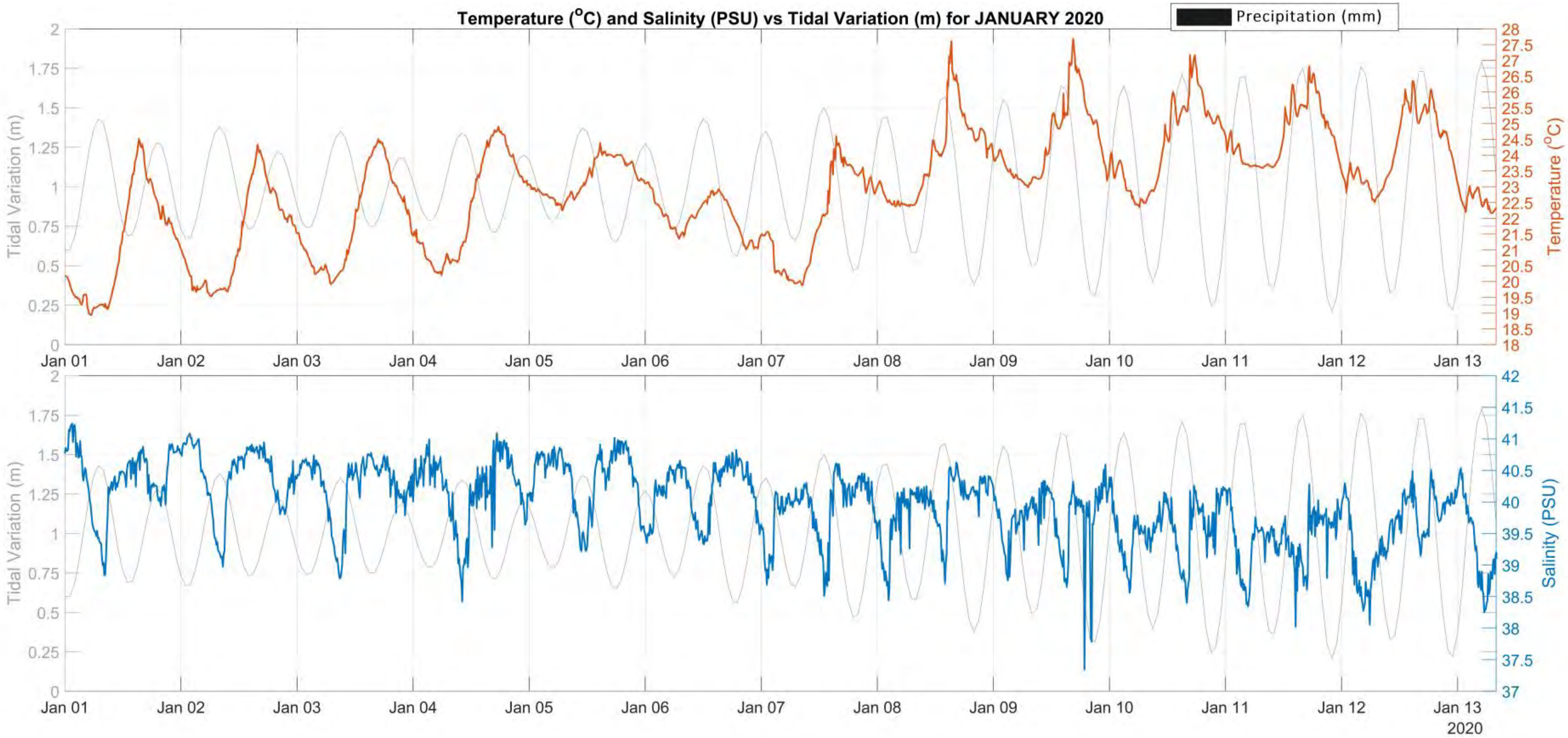


Figure 5.22. Diurnal pattern in salinity (PSU) and temperature (°C) over a 14 day period 1-14 January 2020. Salinity (blue line, bottom, right axis), temperature (red line, top, right axis) recorded at ten minute intervals, tidal variation in hourly intervals (grey lines, left axis, top and bottom), precipitation (black bars). Tidal data provided by hydrographer, SA Navy, precipitation data provided by South African Weather Service.

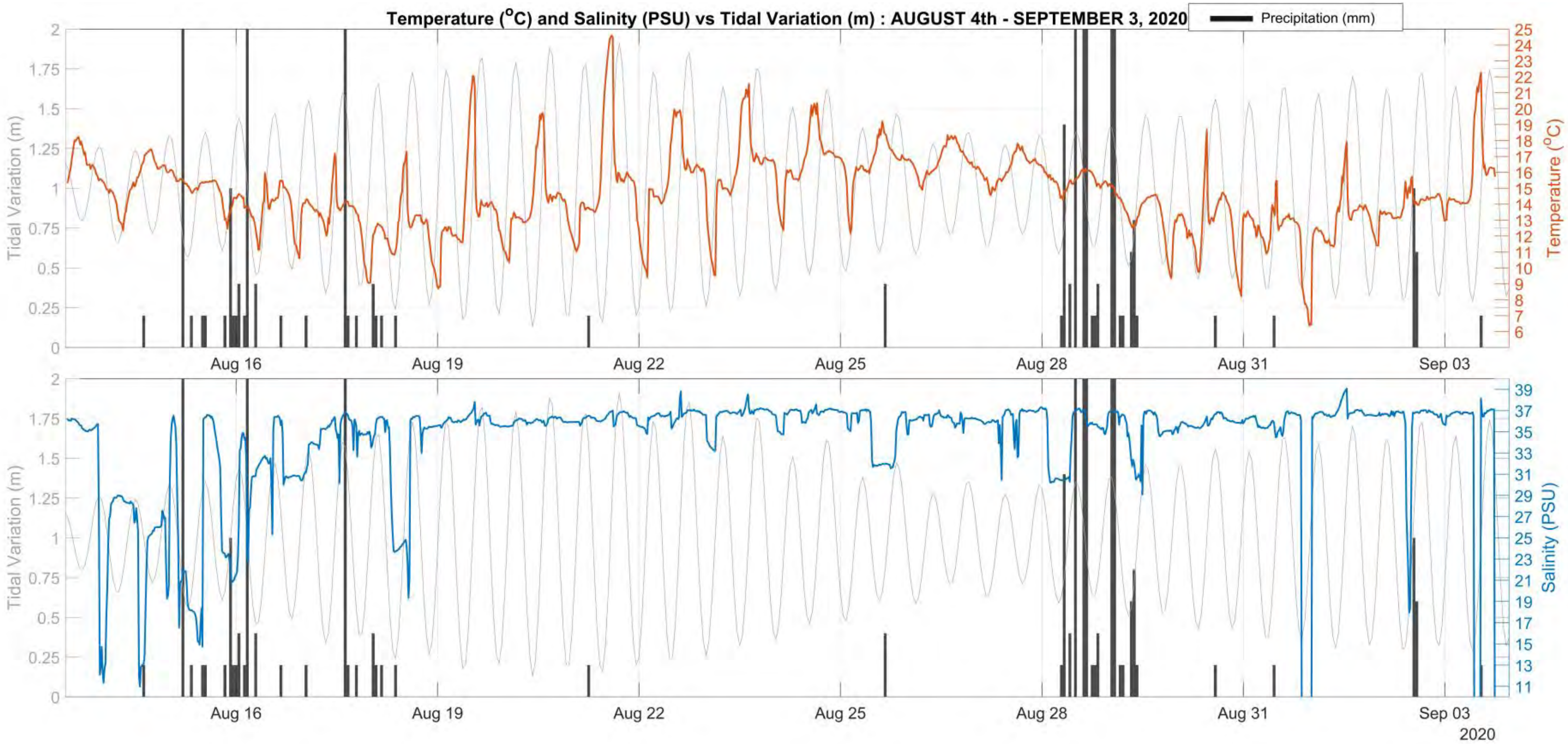


Figure 5.23. **Diurnal pattern in salinity (PSU) and temperature (°C) over an 18 day period, 14 August -3 September 2020. Salinity (blue line, bottom, right axis), temperature (red line, top, right axis) recorded at ten minute intervals, tidal variation in hourly intervals (grey lines, left axis, top and bottom), precipitation (black bars). Tidal data provided by hydrographer, SA Navy, precipitation data provided by South African Weather Service.**

5.8.4 Discussion

An overview of the geological and geohydrological setting of the Greater Saldanha Bay area has been provided. Groundwater is a key component of the natural capital within the area. Additional information has been provided on the existing and planned usage of groundwater. The main existing abstraction of groundwater is for the agricultural sector (as obtained from WARMS). Further to private groundwater use in the GSB area, there are additional proposed abstractions that will take place in the study area. These are in the form of municipal town supply from the Langebaan Road Aquifer (LRA) and Hopefield wellfield. Given that the planned LRA and Hopefield wellfield abstraction is in the region of 5.11 Mm3/a and 1.64 M m3/a, respectively, and taking the registered volumes through WARMS into account (1.58 Mm3/a), the estimated utilisable groundwater exploitation potential still available in the study area is 6.91 Mm3/a. In summary then the groundwater balance for the entire study area is:

Utilisable Groundwater Exploitation Potential	+15.24 Mm3/a (normal conditions)
Total registered groundwater use	-1.58 Mm3/a
Total planned Langebaan Road Aquifer use	-5.11 Mm3/a
Total planned Hopefield wellfield use	-1.64 Mm3/a
Balance	+6.91 Mm3/a

Please note that within the study area the groundwater contribution to river baseflow and springs is negligible. In addition, the calculations above are based on normal climatic conditions and the WARMS data (i.e. total groundwater usage) is 3 years old. In times of drought when the groundwater supply is critical to the area, there will be obviously less groundwater inflow to the system. However, there is enough available groundwater within the area to support the proposed abstraction volumes from the Langebaan Road Aquifer and Hopefield wellfields. It is anticipated that this groundwater abstraction will not impact the freshwater inflows at Geelbek. However, any additional allocation of groundwater needs to be carefully considered within the above context. Due to the complexity of the geohydrological setting within the area, and the critical groundwater dependent ecosystems within the area, as well as the valuable role groundwater plays in meeting water requirements during times of drought, regional monitoring of groundwater levels and quality is crucial. Figure 5.24 shows the distribution of existing monitoring boreholes within the study area and it is crucial that all this monitoring data is collated and analysed regularly to ensure that the groundwater use within in the area is carried out sustainably and responsibly. As custodians of groundwater the ultimate responsibility of groundwater monitoring resides with the DWS, however, this responsibility also resides with groundwater users. For any authorised groundwater use, depending on the volumes abstracted annually, groundwater quantities, levels and quality have to be reported to DWS. So regional monitoring (i.e. not at production / injection boreholes), currently resides with DWS. The missing element is a database that integrates all the groundwater information (from DWS and private authorised users), including the analysis of the data. This situation is in the preliminary stages of being addressed by groundwater consultants; the DWS; the Saldanha Bay Municipality and the Saldanha Bay Water Quality Forum Trust. However significant progress is still required in this regard, especially the funding model.

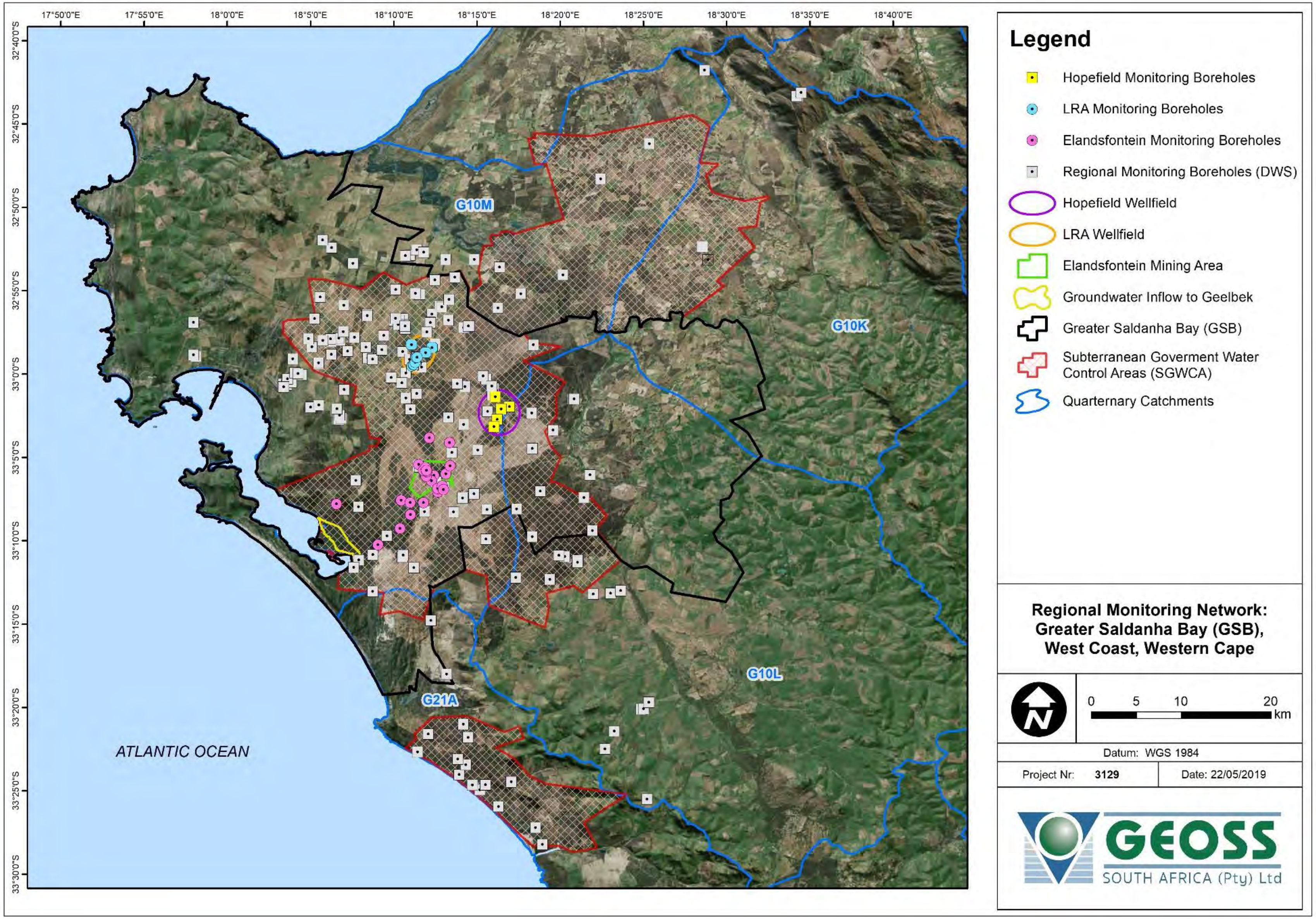


Figure 5.24. **Groundwater monitoring network in the study area.**

5.9 Ecosystem services provided by groundwater

The Saldanha municipality includes the Langebaan lagoon. The lagoon is a unique wetland in that no river feeds into it. (These salt marshes constitute approximately 32% of the entire saltmarsh habitat in South Africa.) The lagoon is groundwater fed, has Ramsar status, and falls within the West Coast National Park. Because of this, the area is afforded high ecological importance and conservation status. There is also little doubt of the high level of sensitivity of the ecological system. Because the wetland is dependent on groundwater inflows, any activity that could alter the flow volumes and velocity of groundwater has the potential to impact the wetland. Thus, the ecosystem is critically dependent on groundwater. Longer term mapping and monitoring of the groundwater dependant ecosystems within Langebaan Lagoon should be undertaken to evaluate the longer term and cumulative impacts of development of the Langebaan and Saldanha area (Belcher and Grobler, 2015). The groundwater inflow mainly occurs in the Geelbek area, making it a point of interest for future monitoring. Figure 5.25 shows the direction of groundwater inflow into the area.

Even when moving away from the lagoon, it is important to note that the SBM area still hosts large areas with natural vegetation in place. The ecosystem is specialized and well adapted to the very arid climate. With shallow groundwater aquifer (< 10mbgl) systems, one has to take into account that the ecosystems may also be dependent on the groundwater.

Figure 5.25. Groundwater flow paths of fresh water into the Geelbek area.

6 WATER QUALITY

6.1 Introduction

The temperature, salinity (salt content) and dissolved oxygen concentration occurring in marine waters are the variables most frequently measured by oceanographers in order to understand the physical and biological processes impacting on or occurring within a body of seawater. Historical long-term data series exist for these three variables for Saldanha Bay spanning the period 1974-2000 and have been augmented by monitoring studies undertaken by the Council for Scientific and Industrial Research (CSIR) (van Ballegooyen *et al.* 2012) on behalf of Transnet for their Reverse Osmosis (RO) desalination plant (data for the period 2010-2011) and various other projects. A trial deployment of a conductivity temperature and depth (CTD) instrument by Anchor Environmental from 3 April to 13 May 2017 provided six weeks of recent data in this area. A thermistor string comprising four underwater temperature meters (UTMs), used for continuous monitoring of water temperature in the Bay, was deployed at North Buoy in Small Bay in April 2014 by Anchor Environmental Consultants on behalf of the SBWQFT. This array is retrieved and maintained during the annual field survey and data up until April 2020 are included in this report. Current data were collected by an Acoustic Doppler Current Profiler (ADCP) from 7 to 10 April 2017 at a site adjacent to the Sea Harvest processing factory in Small Bay as well as at Club Mykonos Beach in Big Bay from 14 February to 28 February 2018. Recently, from the 24 April 2020, the Department of Environmental Affairs, Forestry and Fisheries (DEFF) initiated dissolved oxygen monitoring at Aquaculture Development Zones in Small Bay and Big Bay with instruments moored 0.5 m from the sea floor at Farm (within the ADZ precincts) and control stations. The Big Bay control station also included a nitrate (NO_3) sensor. Data are recorded hourly and the instruments are being serviced every 6-8 weeks. Data for the period 24 April- 8 July 2020 are presented below, but this monitoring is scheduled to be ongoing and should reveal interesting seasonal and long-term trends in the bay's water quality. Some data is also available on other physico-chemical parameters from the Bay including turbidity and bromide, as well as for faecal coliforms and trace metals (introduced to the Bay through wastewater discharges).

Figure 6.1. Saldanha ADZ precincts (red border), MPA (white border) and location of moored oxygen and nitrate sensors.

6.2 Circulation and current patterns

Circulation patterns and current strengths in Saldanha Bay prior to development in 1974/1975 were investigated using various techniques (drogues, dye-tracing, drift cards and sea-bed drifters, Shannon & Stander 1977). Surface currents within the upper five meters were found to be complex and appeared to be dependent on wind strength and direction as well as tidal state. Within Small Bay, currents were typically weak (5-15 cm.s^{-1}) and tended to be clockwise (towards the NE) irrespective of the tidal state or the wind (Figure 6.2). Greater current strengths were observed within Big Bay (10-20 cm.s^{-1}) and current directions within the main channels were dependent on tidal state. The strongest tidal currents were recorded at the mouth of Langebaan Lagoon (50-100 cm.s^{-1}), these being either enhanced or retarded by the prevailing wind direction. Currents within the main channels in Langebaan Lagoon were also relatively strong (20-25 cm.s^{-1}). Outside of the main tidal channels, surface currents tended to flow in the approximate direction of the prevailing wind with velocities of 2-3% of the wind speed (Shannon & Stander 1977). Current strengths and direction at 5 m depth were similar to those at the surface but were less dependent on wind direction and velocity and appeared to be more influenced by tidal state. Currents at 10 m depth at the mouth of the Bay were found to be tidal (up to 10 cm.s^{-1}, either eastwards or westwards) and in the remainder of the Bay, a slow (5 cm.s^{-1}) southward or eastward movement, irrespective of the tidal state, was recorded.

The currents and circulation of Saldanha Bay subsequent to the construction of the Marcus Island causeway and the iron ore/oil terminal were described by Weeks *et al.* (1991a). Historical data of drogue tracking collected by the Sea Fisheries Research Institute during 1976-1979 were analysed in this paper. This study confirmed that wind is the primary determinant of surface currents in both Small Bay and Big Bay; although tidal flows do influence currents below the thermocline and are the dominant forcing factor in the proximity of Langebaan Lagoon. Weeks *et al.* (1991a) noted that because much of the drogue tracking was conducted under conditions of weak or moderate wind speeds, the surface current velocities measured (5-20cm.s^{-1}), were probably underestimated. The authors concluded that the harbour construction had constrained water circulation within Small Bay, enhancing the general clockwise pattern and increasing current speeds along the boundaries, particularly the south-westward current flow along the iron ore terminal (Figure 6.2).

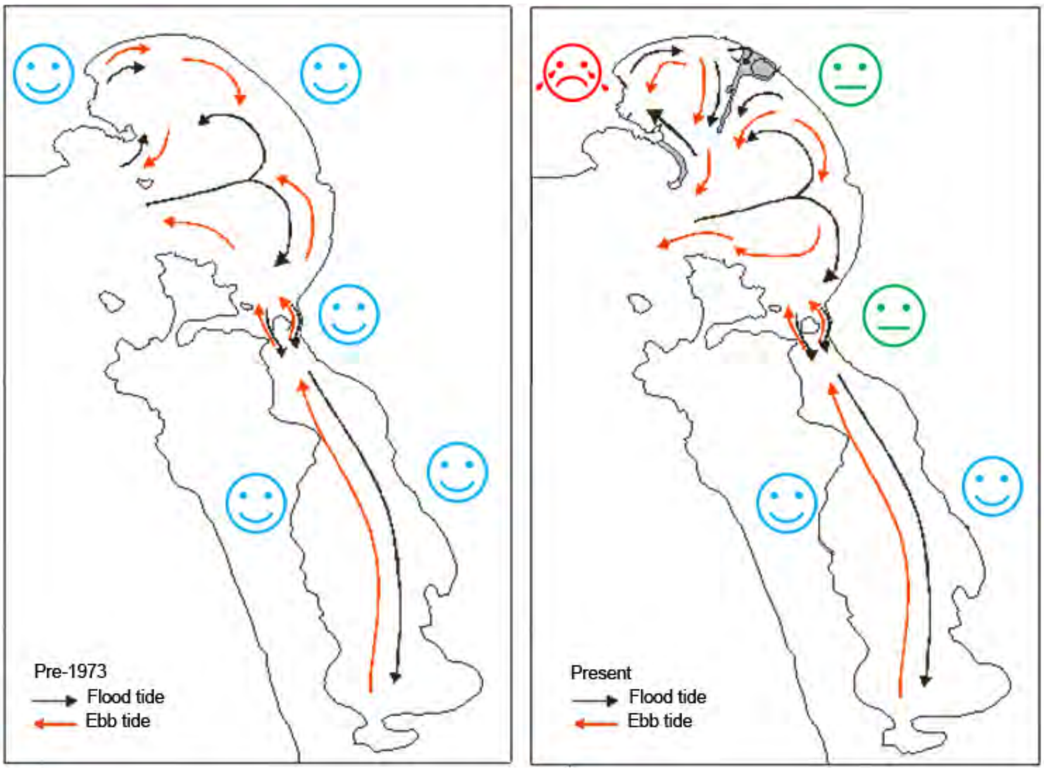


Figure 6.2. Schematic representation of the surface currents and circulation of Saldanha Bay prior to harbour development (pre-1973) and after construction of the causeway and iron ore terminal (present) (Adapted from: Shannon & Stander 1977 and Weeks *et al.* 1991a).

More recent data collected during strong NNE wind conditions in August 1990 revealed that greater wind velocities do indeed influence current strength and direction throughout the water column (Weeks *et al.* 1991b). These strong NNE winds were observed to enhance the surface flowing SSW currents along the ore terminal in Small Bay (out of the Bay), but resulted in a northward replacement flow (into the Bay) along the bottom, during both ebb and flood tides. The importance of wind as the dominant forcing factor of bottom, as well as surface, waters was further confirmed by Monteiro & Largier (1999) who described the density driven inflow-outflow of cold bottom water into Saldanha Bay during summer conditions when prevailing SSW winds cause regional scale upwelling.

An ADCP was deployed from 7 to 10 April 2017 at Sea Harvest in Small Bay (see Figure 6.3 - left) to inform a Coastal Waters Discharge Permit (CWDP) application for a proposed RO Plant outfall. The data were analysed as a dynamic cell, moving with the tide, in 7 m water depth. This enabled quantification of typical current velocities and directions at this location under the prevailing wind conditions. The data are summarised in a current rose that shows the prevailing current moving alongshore in a SSE direction (Figure 6.3 - right). Current velocities recorded at the deployment site over the sampling period indicated that calms were measured 29.9% of the time and current velocities of 1-5 cm/s were measured 64.6% of the of the time (Figure 6.3 - right). The maximum current speed recorded was 15.14 cm/s (Wright *et al*. 2018a).

Currents were found to be primarily wind driven, rather than tidally driven (Figure 6.4). A positive correlation (r^2 = 0.3) was found between current speed and wind speed - a period of strong wind resulted in a corresponding peak in current speed, while a relaxation of the wind forcing led to a decrease in current speed (Figure 6.4). A wide range of wind speeds was experienced during the four-day deployment period, ranging from 3-16 knots (1.5 to 8 m/s) with winds consistently blowing from the south.

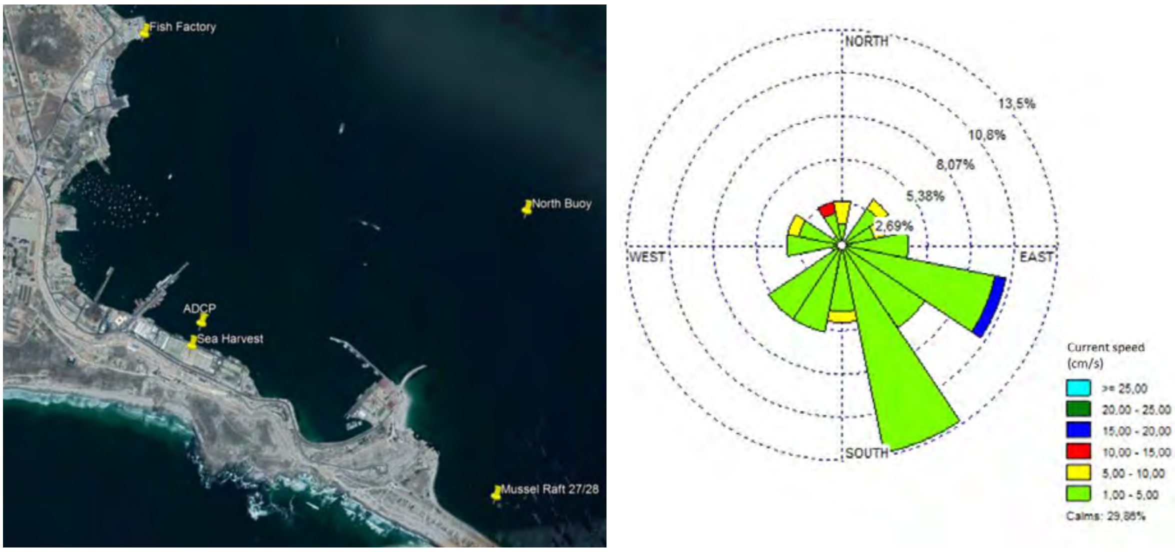


Figure 6.3. Location of the Sea Harvest ADCP (left) and current rose depicting current direction (flowing to) and strength at -7 m water depth (right). (Source: Wright *et al*. 2018a).

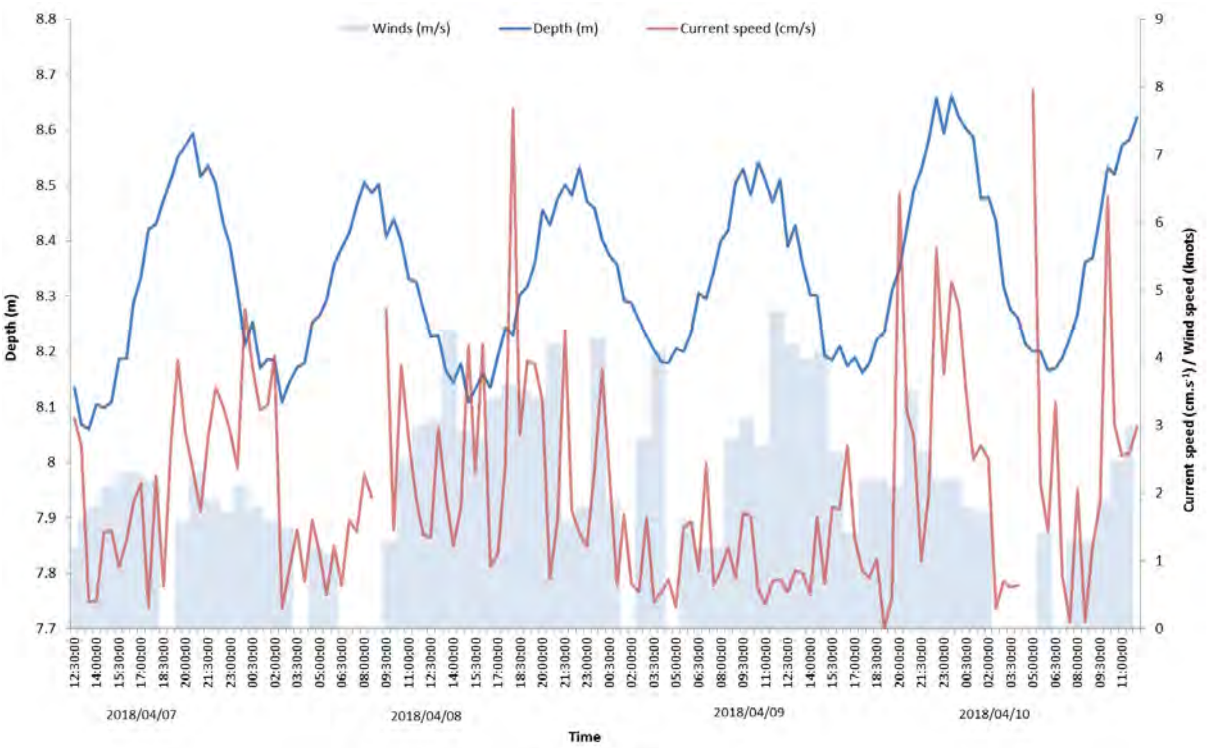


Figure 6.4. **ADCP data collected every 30 minutes for depth (m), indicating the tidal cycle current speed (cm/s), and wind speed (m/s) over the four days of ADCP deployment at Sea Harvest in Small Bay in April 2018 (Source: Wright *et al*. 2018a).**

A current rose depicting the strength, frequency and direction of currents was constructed from ADCP data collected from 14 February to 28 February 2018 at the proposed Club Mykonos RO discharge site in Big Bay (33°2'50.48"S; 18°1'59.71"E) (Figure 6.5). The data for the dynamic cell that recorded currents at 8.5 m water depth show the prevailing current moving alongshore in a north-easterly direction (Figure 6.5). Less frequently, currents were recorded flowing in a northerly direction. Again, currents appeared to be primarily wind driven rather than tidally driven. Of the current velocities measured, 35.6% fell between 10 and 15 cm/s, while current speeds between 5 and 10 cm/s were recorded 20.4% of the time (Figure 6.5). Maximum and average current speeds were recorded as 27.7 cm/s and 11.23 cm/s respectively (Wright *et al*. 2018b). Wind speeds during the deployment period ranged from 2 to 22 knots (1 to 11 m/s) and were consistently from the south.

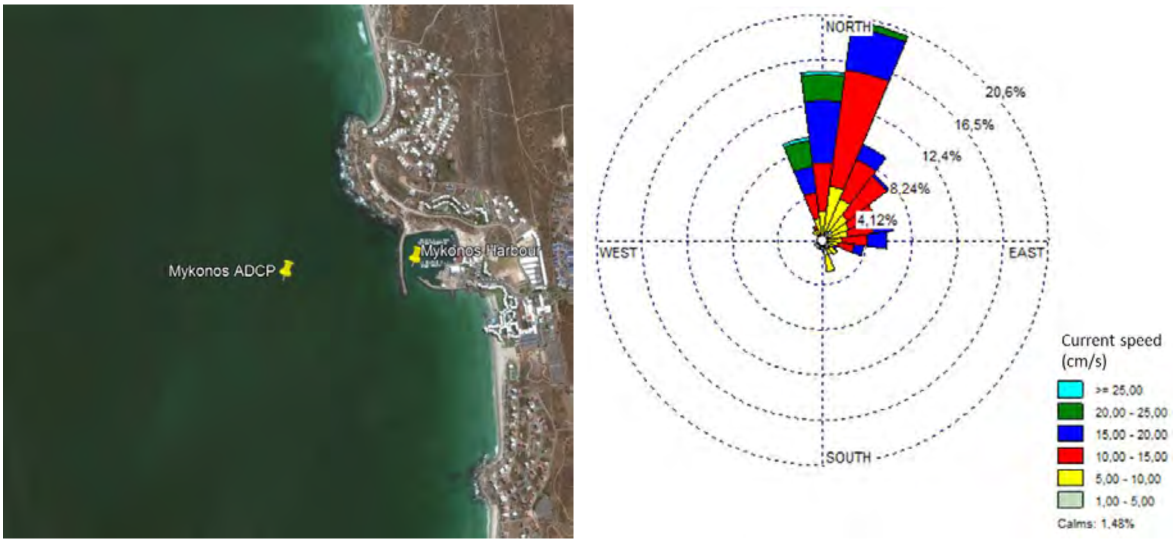


Figure 6.5. Location of the Mykonos ADCP (left) and the resulting current rose showing current direction (flowing to) and strength data at -8.5 m water depth (Wright *et al*. 2018b).

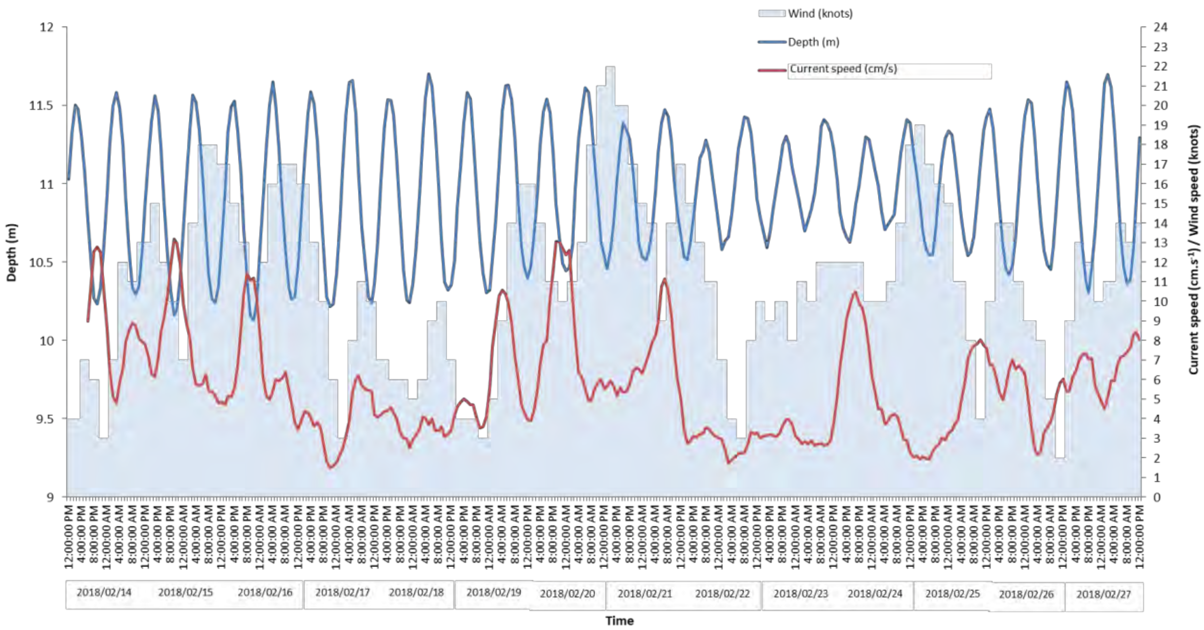


Figure 6.6. ADCP data collected every hour over a 14-day ADCP deployment period in February 2018 showing current speed (cm.s^{-1}) and wind speed (knots). Depth (m) indicates the tidal cycle (blue line). (Source: Wright *et al*. 2018b).

Krug (1999) studied current speeds at the mouth of Langebaan lagoon and circulation into and out of the lagoon. A two-weeks field survey was conducted in Langebaan Lagoon and Saldanha Bay In March 1997 with the aim of furthering understanding of the processes driving the mixing and the exchange between Saldanha Bay and Langebaan Lagoon. Flows into and out of the lagoon were found to be predominantly tidally driven and that there existed an asymmetry between the ebb and the flood flows at both of the lagoon's inlets. When tidal forcing was strong, water particles released at the lagoon inlets during the ebb were subject to long drifts. Outflow from the east inlet appeared to take the form of a turbulent jet, while at the west inlet, strong frictional interactions between the flow and land boundaries was observed which caused the flow to rapidly expand and lose momentum and was thus prevented from forming a jet. It was also found that buoyancy forcing on lagoon outflow was generally small and that water issuing from the lagoon during the ebb remained attached to the sea-bed as it propagated into Saldanha· Bay. However, when Saldanha Bay was strongly stratified, the east inlet ebb jet lifted off from the bottom as it reached the 8 m depth contour. Outflow from the lagoon made an important contribution to vertical mixing in Saldanha Bay, specially near the lagoon entrance. Southerly winds contributed to the overall residual circulation by driving water out of the Lagoon.

6.3 Wave action

Construction of the iron ore terminal and the Marcus Island causeway had a major impact on the distribution of wave energy in Saldanha Bay, particularly in Small Bay. Prior to port development in Saldanha Bay, Flemming (1977) distinguished four wave-energy zones in the Bay, defined as being a centrally exposed zone in the area directly opposite the entrance to the Bay, two adjacent semi-exposed zones on either side, and a sheltered zone in the far northern corner of the Bay (Figure 6.7 left). The iron ore terminal essentially divided the Bay into two parts, eliminating much, if not all, the semi-exposed area in Small Bay, greatly increasing the extent and degree of shelter in the north-western part of Small Bay, and subtly altering wave exposure patterns in Big Bay (Figure 6.7 right). Wave exposure in Big Bay was altered less dramatically; however, the extent of sheltered and semi-sheltered wave exposure areas increased after harbour development (Luger *et al.* 1999).

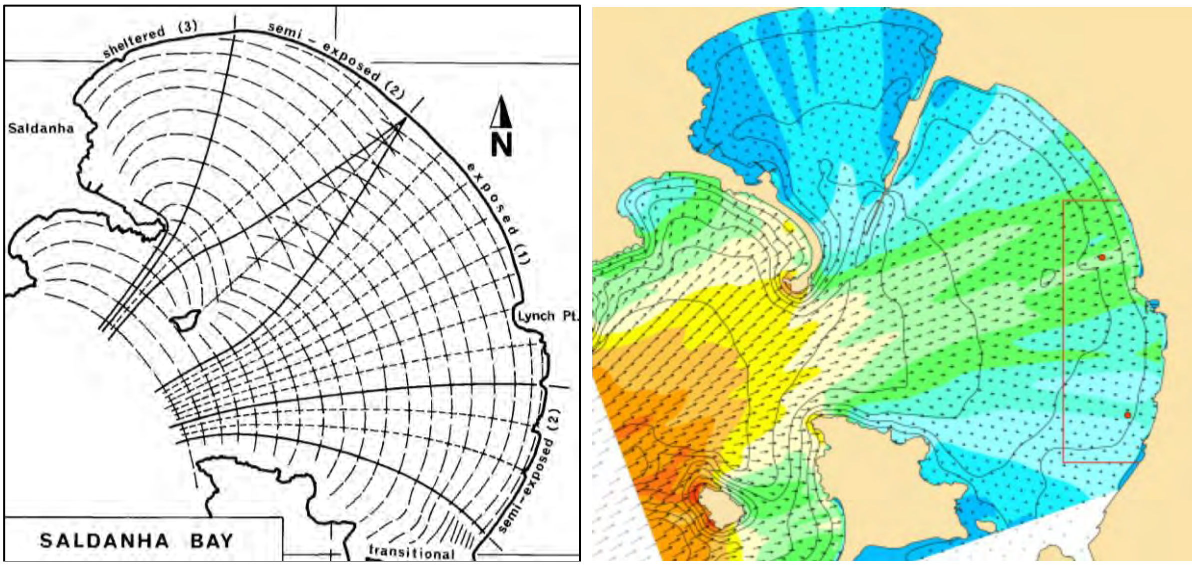


Figure 6.7. Predicted wave fields in Saldanha Bay showing wave height and direction prior to (left) and post (right) harbour development. Orange shading indicates wave heights >1.4 m, while blue shading indicates wave heights of <0.6 m (Sources: Flemming 1977 and WSP Africa Coastal Engineers 2010).

6.4 Water temperature

Water temperature records for Saldanha Bay and Langebaan Lagoon were first collected during 1974/1975 as part of a detailed survey by the then Department of Industries - Sea Fisheries branch, later renamed the Department of Environmental Affairs and Tourism (DEAT) - Marine and Coastal Management (MCM), Department of Environmental Affairs - Oceans and Coasts (DEA-O&C) and currently DEFF. The survey was initiated to collect baseline data of the physical and chemical water characteristics prior to the development of the Bay as an industrial port. The findings of this survey were published in a paper by Shannon & Stander (1977). Surface water temperatures prior to the construction of the iron ore/oil terminal and Marcus Island causeway varied from 16 to 18.5°C during summer (January 1975) and 14.5 to 16°C during winter (July 1975). For the duration of sampling, higher temperatures were measured in the northern part of Small Bay and within Langebaan Lagoon, whilst cooler temperatures were measured at sampling stations in Outer Bay and Big Bay.

The water column was found to be fairly uniform in temperature during winter and spring (i.e. temperature did not change dramatically with depth) and the absence of a thermocline (a clear boundary layer separating warm and cool water) was interpreted as evidence of wind driven vertical mixing of the shallow waters in the Bay. A clear shallow thermocline was observed at about 5 m depth during the summer and autumn months at some deeper stations and was thought to be the result of warm lagoon water flowing over cooler sea water. The absence of a thermocline at other shallow sampling stations was once again considered evidence of strong wind driven vertical mixing. Shannon & Stander (1977) suggested that there was little interchange between the relatively sun-warmed Saldanha Bay water and the cooler coastal water through the mouth of the Bay, but rather a "slopping backwards and forwards tidal motion".

The Sea Fisheries Research Institute continued regular quarterly monitoring of water temperature and other variables in Saldanha Bay until October 1982. These data were presented and discussed in papers by Monteiro *et al.* (1990) and Monteiro & Brundrit (1990). The temperature time series for Small Bay and Big Bay is shown in Figure 6.8. This expanded data series allowed for a better understanding of the oceanography of Saldanha Bay. The temperature of the surface waters was observed to fluctuate seasonally with surface sun warming in summer and cooling in winter, whilst the temperature of deeper (10 m depth) water shows a smaller magnitude, non-seasonal variation, with summer and winter temperatures being similar (Figure 6.8). In most years, a strong thermocline separating the sun warmed surface layer from the cooler deeper water was present during the summer months at between 5-10 m depth. During the winter months, the thermocline breaks down due to surface cooling and increased turbulent mixing, and the water column becomes nearly isothermal (surface and deeper water similar in temperature) (Figure 6.8). Unusually warm, deeper water was observed during December 1974 and December 1976 and was attributed to the unusual influx of warm oceanic water during these months (Figure 6.8).

Warm oceanic water is typically more saline and nutrient-deficient than the cool upwelled water that usually occurs below the thermocline in Saldanha Bay. This was reflected in the high salinity (Figure 6.13), and low nitrate and chlorophyll concentration (a measure of phytoplankton production) measurements taken at the same time (Monteiro & Brundrit 1990).

Monteiro *et al.* (1990) suggested that the construction of the Marcus Island causeway and the iron ore/oil terminal in 1975 had physically impeded water movement into and out of Small Bay, thus increasing the residence time and leading to systematically increasing surface water temperatures when compared with Big Bay. There appears to be little support for this in the long-term temperature time series (Figure 6.8) and although the pre-construction data record is limited to only one year, Shannon & Stander (1977) show Small Bay surface water being 2˚C warmer than Big Bay during summer, prior to any harbour development. It is likely that the predominant southerly winds during summer concentrate sun warmed surface water in Small Bay, whilst much of the warm surface layer is driven out of Big Bay into Outer Bay.

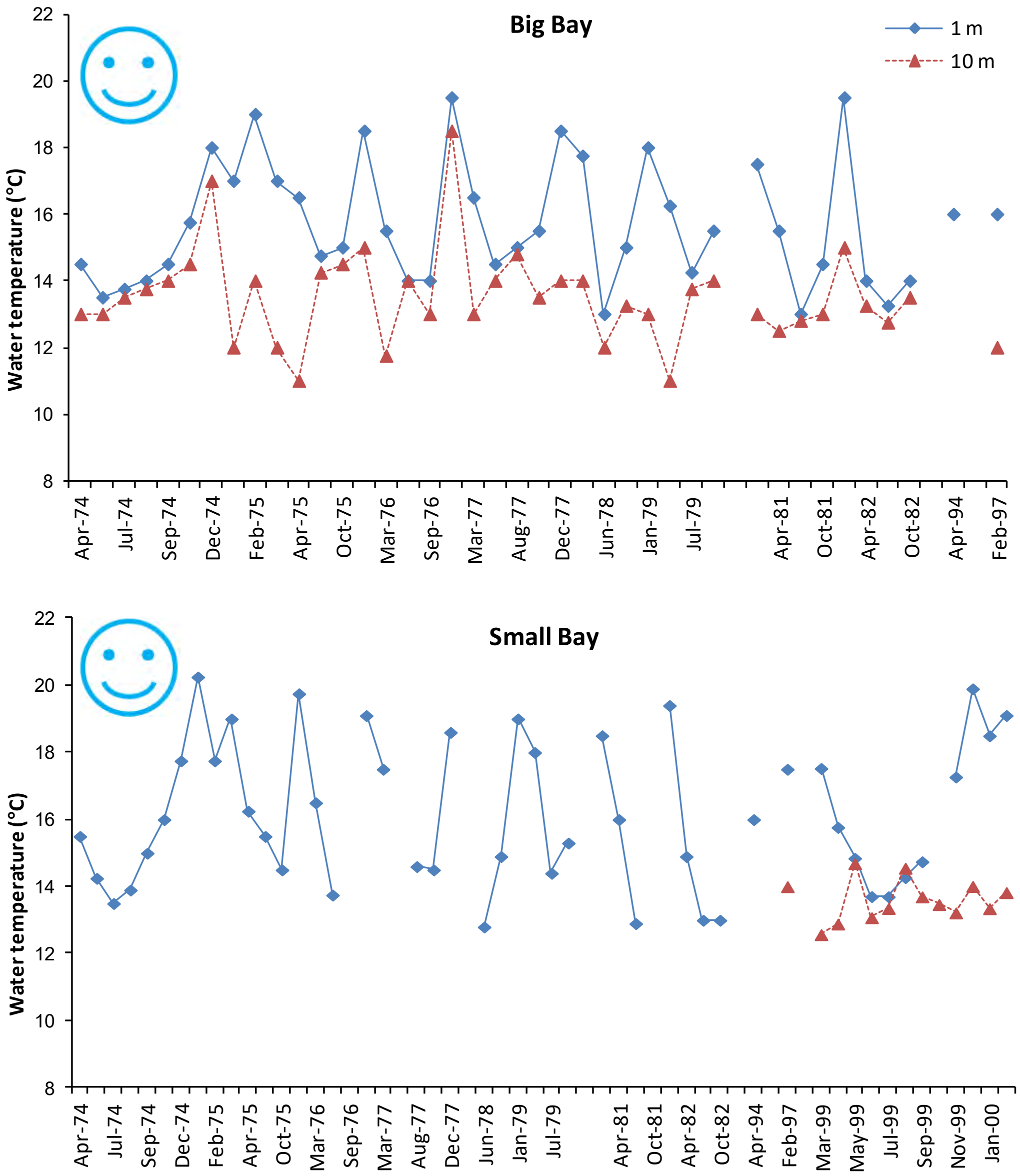


Figure 6.8. **Water temperature time series at the surface and at 10 m depth for Big Bay and Small Bay in Saldanha Bay (Data: Monteiro *et al.* 1990, Monteiro & Brundrit 1990, Monteiro *et al.* 2000 and Shannon & Stander 1977).**

Detailed continuous monitoring of temperature throughout the water column at various sites in Outer Bay, Small Bay and Big Bay during a two-week period in February-March 1997 allowed better understanding of the mechanisms causing the observed differences in the temperature layering of the water column. It revealed that the summer thermocline is not a long-term feature but has a six to eight-day cycle. Cold water, being denser than warmer water, flows into Saldanha Bay from the adjacent coast when wind driven upwelling brings this cold water close to the surface. The inflow of cold, upwelled water into the Bay results in a thermocline, which is then broken down when the cooler bottom water flows out the Bay again. This density driven exchange between Saldanha Bay and coastal waters is estimated to be capable of flushing the Bay within six to eight days, substantially less than the approximately 20 day flushing time calculated based on tidal exchange alone by Shannon & Stander (1977). The influx of nutrient rich upwelled water into Saldanha Bay is critical in sustaining primary productivity within the Bay, with implications for human activities such as fishing and mariculture. The fact that the thermocline is seldom shallower than 5 m depth means that the shallower parts of Saldanha Bay, particularly Langebaan Lagoon, are not exposed to the nutrient (mainly nitrate) import from the Benguela upwelling system. As a result, these shallow water areas do not support large plankton blooms and are usually clear.

Water temperature in Saldanha Bay was intensively monitored by the CSIR over the period March 1999 to February 2000 (Monteiro *et al.* 2000). At the time, this was the most detailed long-term temperature record available, with continuous measurements (every 30 minutes) taken at one-meter depth intervals over the 11 m depth range of the water column where the monitoring station was situated in Small Bay. The average monthly temperature at the surface (1 m) and bottom (10 m) for this period is shown in Figure 6.8. These data confirmed the pattern evident in earlier data, showing a stratified (layered) water column from spring to summer caused by wind driven upwelling, with the water column being more or less isothermal (of equal temperatures) during the winter (Figure 6.8). The continuous monitoring of temperature also identified a three-week break in the usual upwelling cycle during December 1999, with a consequent gradual warming of the bottom water. This "warm water event" was associated with a decrease in phytoplankton production due to reduced import of nitrate, which in turn, impacted negatively on local mussel mariculture yields (Monteiro *et al.* 2000). However, since the water column remained stratified, the magnitude of this event was not as great as the December 1974 and 1976 events.

The CSIR also undertook baseline monitoring in Saldanha Bay on behalf of Transnet before the implementation and operation of the Transnet Reverse Osmosis (RO) desalination plant in 2012 (van Ballegooyen *et al.* 2012). Monitoring of sea water temperature, salinity and dissolved oxygen took place over a period of 10 months (July 2010 to March 2011) at one site immediately adjacent to the proposed desalination plant outfall (an underwater mooring). Water column profiling was also undertaken at nine stations at discrete intervals during the year. Locations of the sampling stations are listed in Table 6.1 and indicated on Figure 6.9. The combination of continuous monitoring and discrete profiling measurements was designed to address seasonal (every 3 months), event (3 to 10 days), and diurnal (daily) scales of temporal variability in the Bay.

Sites were selected in an effort to address the following issues/aspects:

- Brine Discharge Site (BDS) to provide a measure of brine plume impacts in the immediate vicinity of the proposed brine discharge at Caisson 3.
- WRO3 and WRO4 to measure the brine plume extent moving seawards along the dredged shipping channel.
- WRO1 and WRO2 to monitor potential plume excursions out of the dredge channel and towards Small and Big Bay.
- Mussel Farm (MF) and Intermediate Dredge Site (IDS) to couple WRO1 and WRO2 to data measured previously. The MF site was also considered to be a sensitive location, while the IDS lies roughly on a line between the proposed RO Plant discharge and the MF.
- North Buoy (NB) to create a baseline to complement both past and potential future long-term mooring at North buoy.
- Big Bay (BB) to provide a baseline station in Big Bay to act as a control site.

Table 6.1. Location and details of sites sampled during the water column profiling surveys undertaken by the CSIR between July 2010 and March 2011.

Site	Latitude	Longitude	Depth (m)	Distance from discharge (m)	Location
North Buoy (NB)	33° 1.114'S	17°58.130'E	12.5	1 875	Outside channel
Mussel Farm (MF)	33° 1.794'S	17° 58.247'E	16.0	1 400	Outside channel
Intermediate Dredge site (IDS)	33° 1.889'S	17° 58.642'E	16.0	880	Outside channel
WRO3	33° 1.935'S	17° 59.030'E	26.5	525	Inside channel
WRO4	33° 1.721'S	17° 59.127'E	28.5	105	Inside channel
WRO2	33° 1.651'S	17° 59.094'E	23.0	85	On slope
Brine Discharge Site (BDS)	33° 1.679'S	17° 59.147'E	17.3	30	On slope between dredge channel berthing areas
WRO1	33° 1.688'S	17° 59.215'E	18.0	85	Outside channel
East Buoy (Big Bay)	33° 3.188'S	18° 0.433'E	15.5	3450	Outside channel

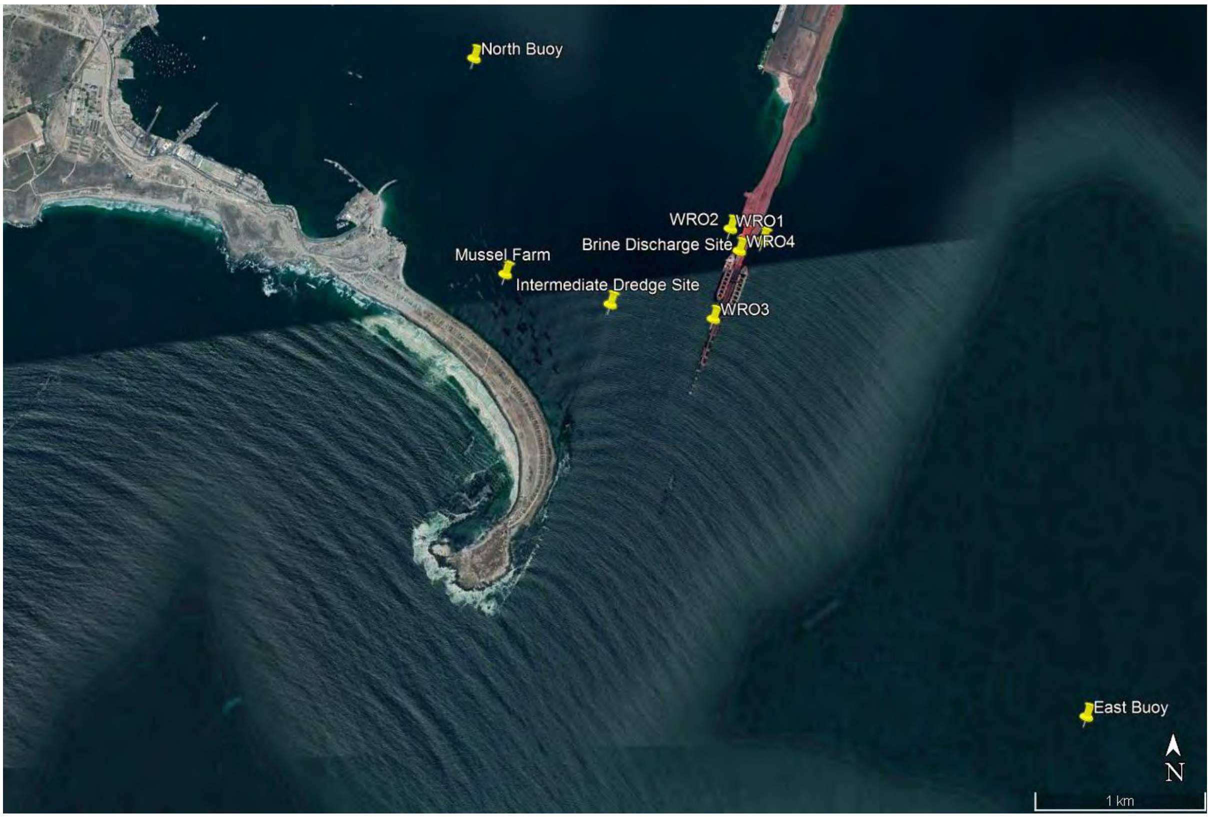


Figure 6.9. **Water quality monitoring stations adopted for the RO plant baseline survey undertaken by the CSIR (Source: van Ballegooyen *et al.* 2012).**

Examples of the temperature data from the water column profiling exercises undertaken at North Buoy are shown in Figure 6.10. In general, the profiles at all sites indicated a well-mixed column in winter, becoming increasingly stratified in spring and early summer, and highly stratified in late summer/autumn. The temperature variability in the lower water column was very high during spring and early summer when strong wind events change the water column from being moderately to highly stratified to a well-mixed water column under strong wind conditions. This variability was much lower in summer due to the presence of cold upwelled waters that help to stratify the water column and in so doing, increase the resistance of the water column to vertical mixing. Stratification was less pronounced at East Buoy in Big Bay than at the more sheltered stations in and around Small Bay (van Ballegooyen *et al.* 2012). This was ascribed to more turbulent conditions in Big Bay compared to Small Bay. A strong thermocline was also evident in the shipping channel, which is more accessible to the cold bottom waters associated with upwelling that enters the Bay.

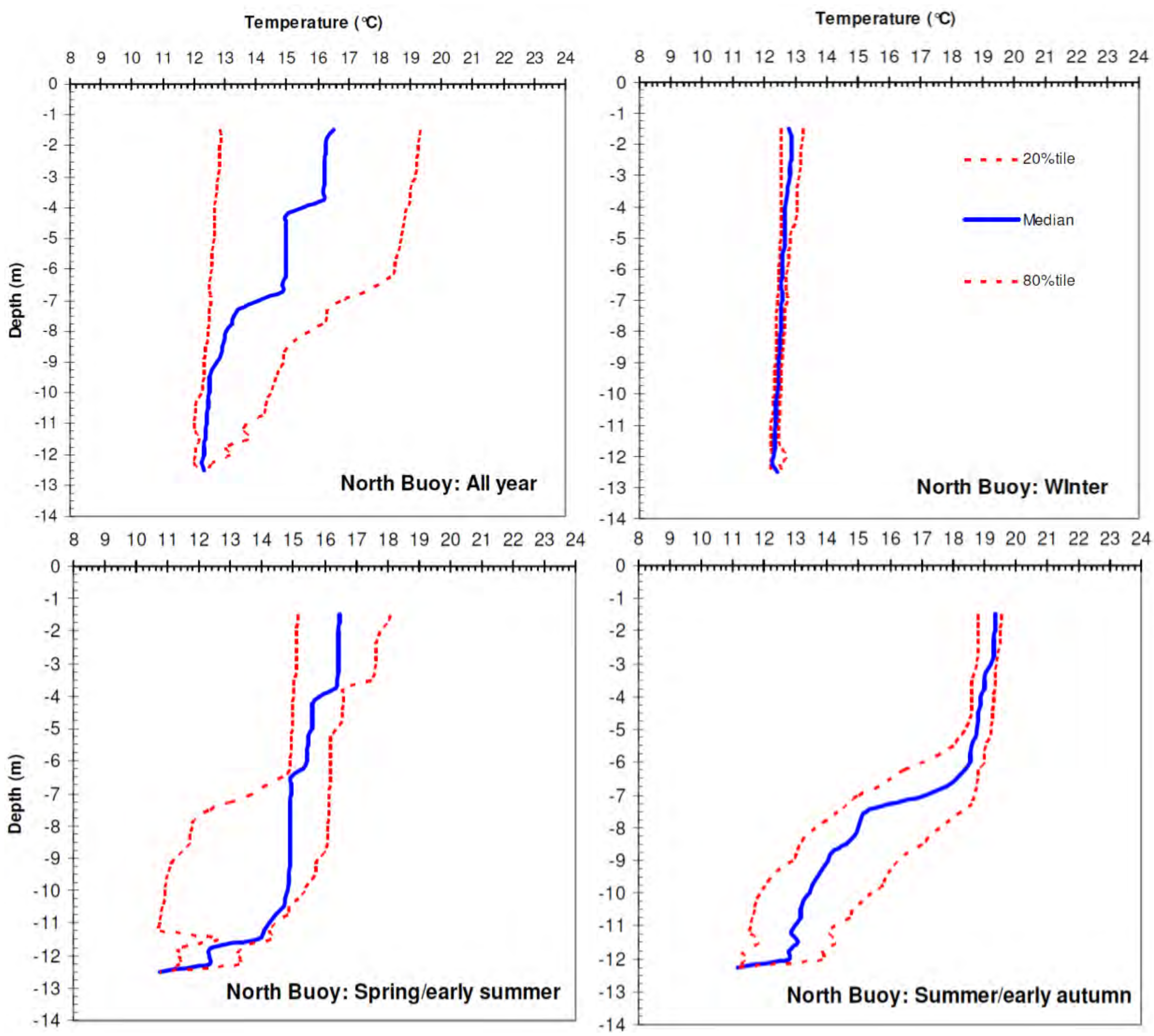


Figure 6.10. Seawater temperature median profiles at North Buoy for all four seasons. The 20th and 80th percentile limits of the profiles are indicated by the dotted red lines (Source: van Ballegooyen *et al.* 2012).

With a view to continuing the long-term temperature data set at North Buoy, five underwater temperature recorders (UTRs), programmed to record temperature every hour were deployed at 2 m, 4.5 m, 7 m, 9.5 m and 12 m depth on the 12 April 2014. These thermistors are retrieved and serviced annually, and average daily temperature data for the period April 2014 to April 2020 are shown in Figure 6.11.

Unfortunately, three of the UTRs reached the end of their battery life during the April 2019-April 2020 deployment period and data could only be retrieved from the bottom two loggers at 9.5 m and 12 m depths. The data from 12 April 2014 to 20 April 2020 shows a similar pattern to historical data, with high variability and water column stratification evident from September to May (i.e. from spring through to autumn) and a well-mixed, isothermal water column in the winter months in most years (Figure 6.11). Variation in bottom water temperature is greater than in the surface waters and appears to happen over synoptic time scales as noted by van Ballegooyen *et al.* (2012). Relaxation of upwelling and the down mixing of warmer surface waters, or the intrusion of warm oceanic waters that results in warming of the bottom water is most frequently observed in spring to early summer and again in late summer to early autumn.

Notable inter-annual variation in the water column temperature profile is evident in the data series with the period April 2016 to February 2018 appearing anomalous compared to the other data collected in the period between April 2014 and March 2016 and subsequently from March 2018 to April 2020. This coincides with the extreme drought experienced in the Western Cape over this period. During this period, maximum summer water temperatures are reduced (below 20°C) and the stratification of the water column appears much more limited, only becoming properly established for a short period from December 2016 to February 2017. Although some stratification is evident in the spring of 2017, a complete breakdown of the thermocline occurred for an extended period during January 2018, when cool (approximately 12°C) water persists throughout the water column. This stands in marked contrast to historical data when thermocline breakdown typically occurred only during winter, or when it did occur in summer, it was associated with a “warm water” event. Winter water temperatures during 2018 (average of 13.9°C for the period June to August) were also elevated compared to the previous three winters when the average for winter was ~12°C, albeit not noticeably different from the 2014 winter data (average of 13.73°C). The winter 2020 data is only available for the bottom of the water column and the average (12.9°C) was closer to the bottom water temperatures recorded during 2014 and 2018 winters (~13°C) rather than those recorded during the drought years (11.7-12.3 °C). This inter annual variation is not unusual and may be linked with *El Nino-La Nina* climatic cycles. The anomalous data collected over the period December 2016 to February 2017 during the drought is almost certainly linked to the dominance of the South Atlantic High Pressure system during this period. Persistent southerly winds throughout most of the year would have promoted coastal upwelling, resulting in reduced summer water maxima (in extreme cases decreasing temperatures throughout the water column) and causing cooler than average winter water temperatures.

The monthly average bottom (12-14°C) or surface (13-20°C) water temperatures in the period 2014 to 2020 are, however, similar to those recorded in earlier monitoring (since 1974) (Figure 6.8). There also appears to be no clear trend of seawater warming or cooling over time, but rather anomalous, seasonal scale events are being detected. Establishment of continuous, high temporal resolution water temperature monitoring will prove valuable in analysing long-term trends. This is an economically viable way of detecting changes in the frequency of anomalous conditions such as the intrusion of warm oceanic water events that would have significant impacts on ecosystem productivity and health.

The nitrate sensor deployed for the recent ADZ monitoring provides a temperature record for the Control Station in Big Bay at 18 m water depth (S 33.05, E 17.99). During the autumn-winter deployment period for which data are available to date (24 April - 8 July 2020) temperature varied between 10.2-15.5°C with a gradual warming trend attributed to decreasing frequency of upwelling and vertical mixing of the water column associated with the seasonal transition from autumn to winter (Figure 6.12). The temperature readings were within the observed historical range for 10 m water depth in Big Bay, and were within the range of water temperatures recorded at North Buoy in Small Bay (Figure 6.8).

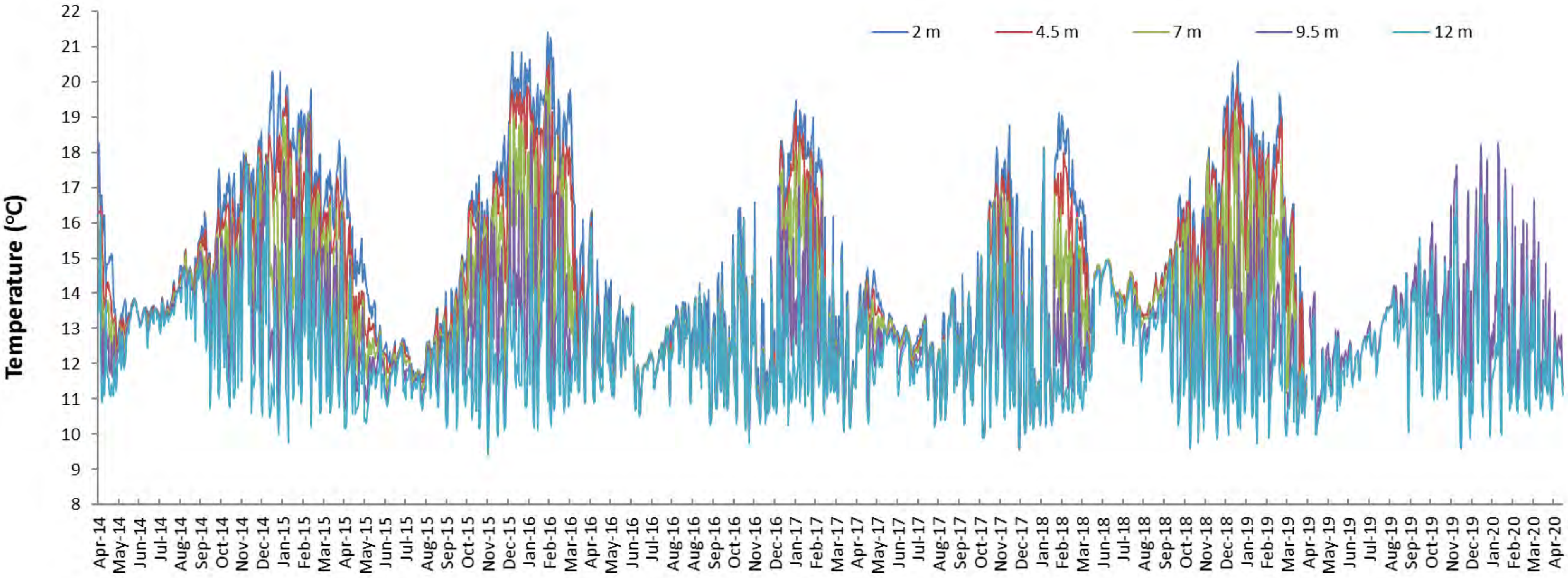


Figure 6.11. North buoy temperature time series for the period 12 April 2014 - 20 April 2020. Temperature was recorded every hour and the average daily temperature is shown here. Note that the no data are shown for the period 10 April- 5 June 2018, and only the bottom two sensors provided data over the last 12 months.

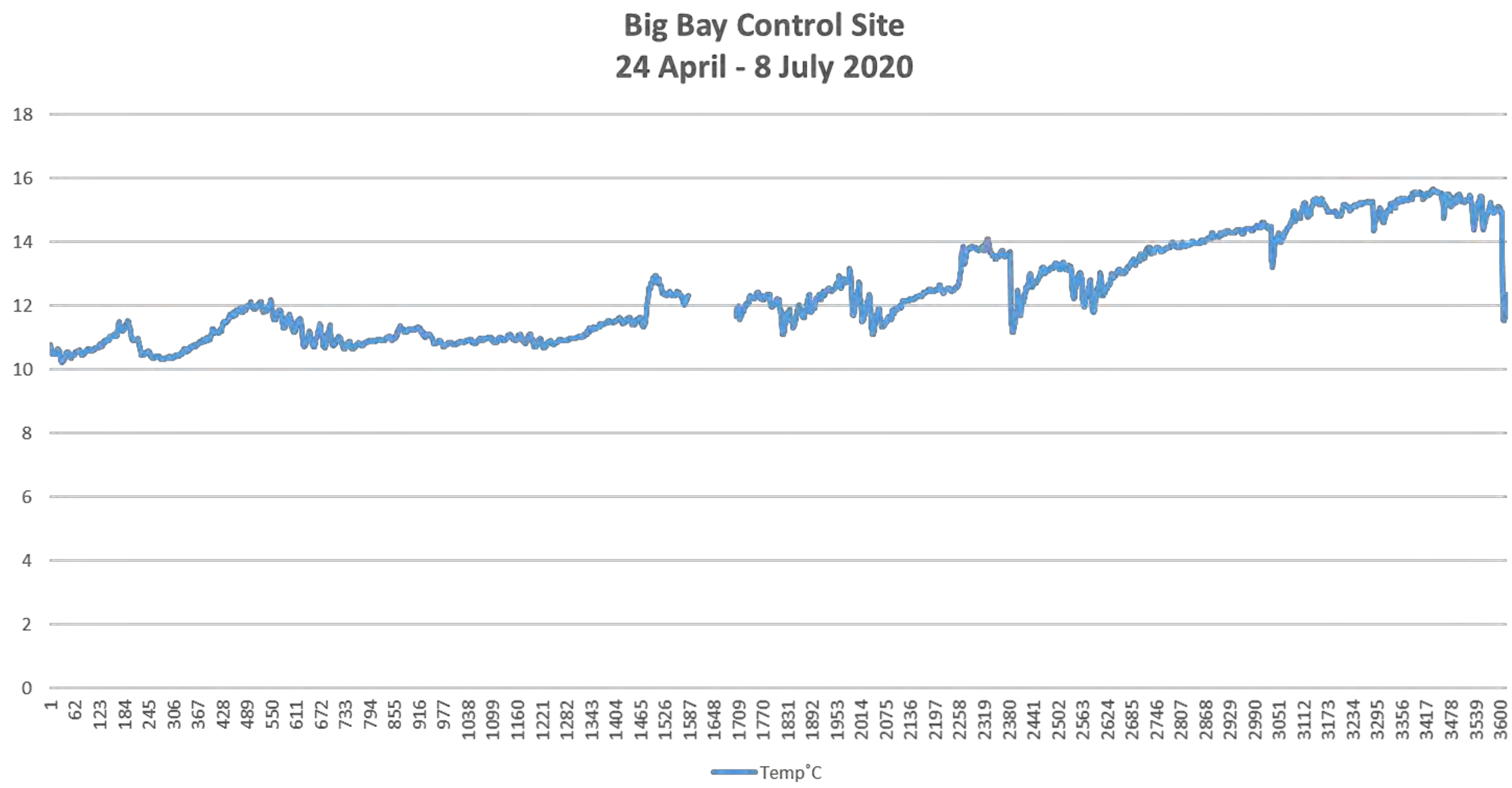


Figure 6.12. Hourly temperature readings recorded at the Big Bay ADZ control site (18m depth) over the period 24 April – 8 July 2020. Graph courtesy of DEFF.

6.5 Salinity

Salinities of the inshore waters along the West Coast of South Africa typically vary between 34.6 and 34.9 Practical Salinity Units (PSU) (Shannon 1966), and the salinity values recorded for Saldanha Bay usually fall within this range. During summer months when wind driven coastal upwelling within the Benguela region brings cooler South Atlantic central water to the surface, salinities are usually lower than during the winter months when the upwelling front breaks down and South Atlantic surface waters move against the coast (warm surface waters are more saline due to evaporation).

The historic salinity data time series for Sadhana Bay covers much of the same period as that for water temperature. Salinity data at 10 m depth were extracted from the studies of Shannon & Stander (1977), Monteiro & Brundrit (1990), Monteiro *et al.* (1990) and Monteiro *et al.* (2000) and are presented in Figure 6.13. There was little variation in salinity with depth. Under summer conditions when the water column is stratified, surface salinities may be slightly elevated due to evaporation, therefore, salinity measurements from deeper water more accurately reflect those of the source water.

The salinity time series at 10 m depth shows salinity peaks in December 1974 and 1976 which reflect an influx of warm water that occurred at this time (Figure 6.13). Higher than normal salinity values were also recorded in August 1977 and July 1979. Although this was not reflected in the temperature time series, probably due to rapid heat loss and mixing during winter, the salinity peaks do indicate periodic inflows of surface oceanic water into Saldanha Bay.

Oceanic surface waters tend to be low in nutrients, limiting primary production (i.e. phytoplankton growth). The oceanic water intrusions into Saldanha Bay that were identified from the temperature and salinity measurements corresponded to low levels of nitrate and chlorophyll concentrations measured at the same time as salinity and temperature peaks (Figure 6.14). This highlights the impacts of the changes in physical oceanography (water temperature and salinity) in the immediate area on the biological processes (nitrate and chlorophyll) occurring within Saldanha Bay (Monteiro & Brundrit 1990). Data concerning these parameters cover a short period only (1974 to 1979) and are little use in examining effects of human development on the Bay.

Examples of the salinity data from the water column profiling exercises undertaken at North Buoy by the CSIR in 2010/2011 are shown in Figure 6.15 (van Ballegooyen *et al.* 2012). In general, the profiles at all sites were found to be consistent with the notion that lower salinity bottom waters enter the Bay during the upwelling season (summer), and higher salinity surface waters are present in late summer/autumn. The low salinity "spikes" observed in the profile data are reportedly spurious (instrument error) and can be ignored (van Ballegooyen *et al.* 2012).

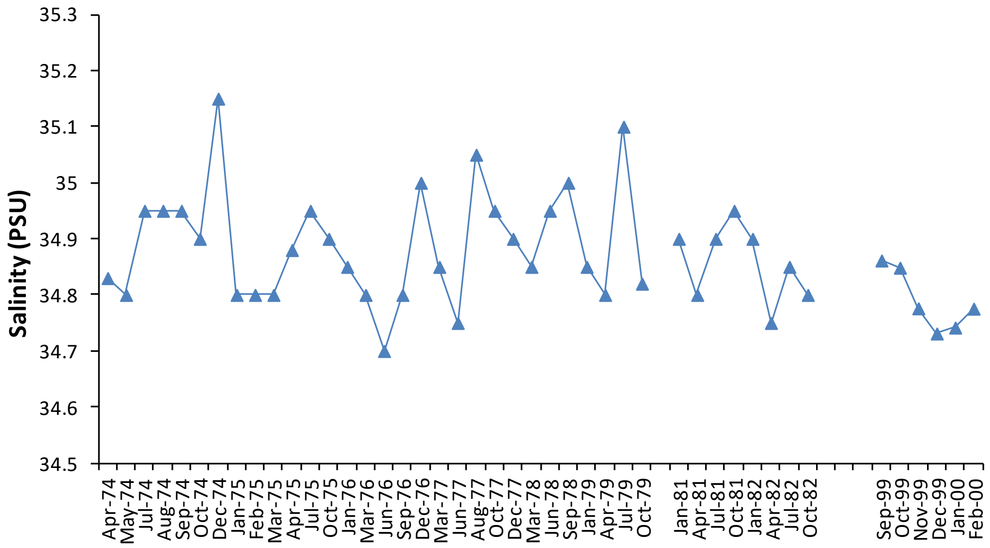
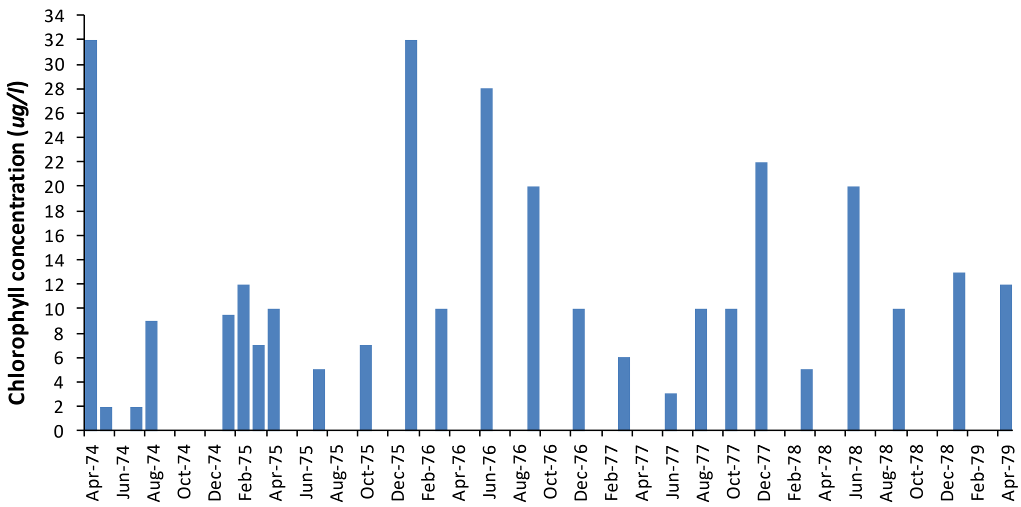


Figure 6.13. Time series of salinity records for Saldanha Bay. (Data sources: Shannon & Stander 1977, Monteiro & Brundrit 1990, Monteiro *et al*. 1990 and Monteiro *et al*. 2000).



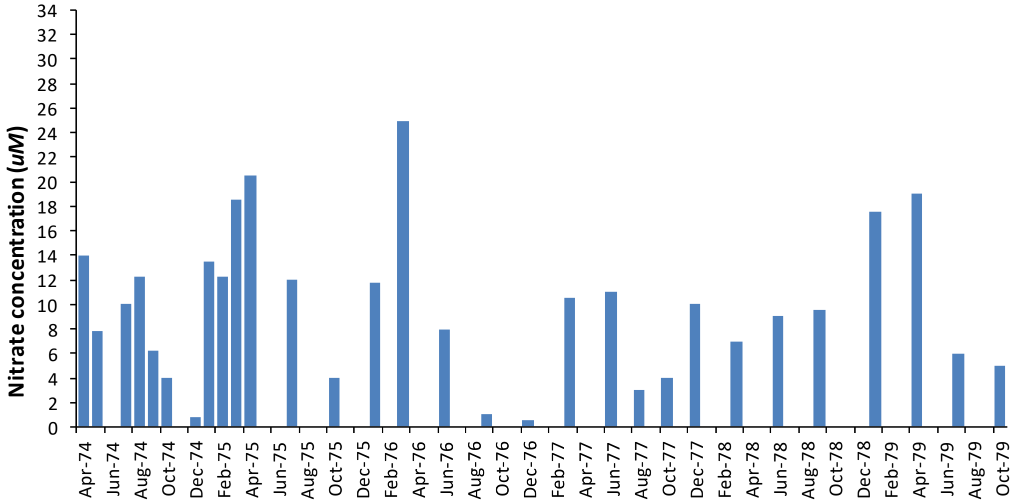


Figure 6.14. Time series of chlorophyll and nitrate concentration measurements for Saldanha Bay. (Data source: Monteiro & Brundrit 1990).

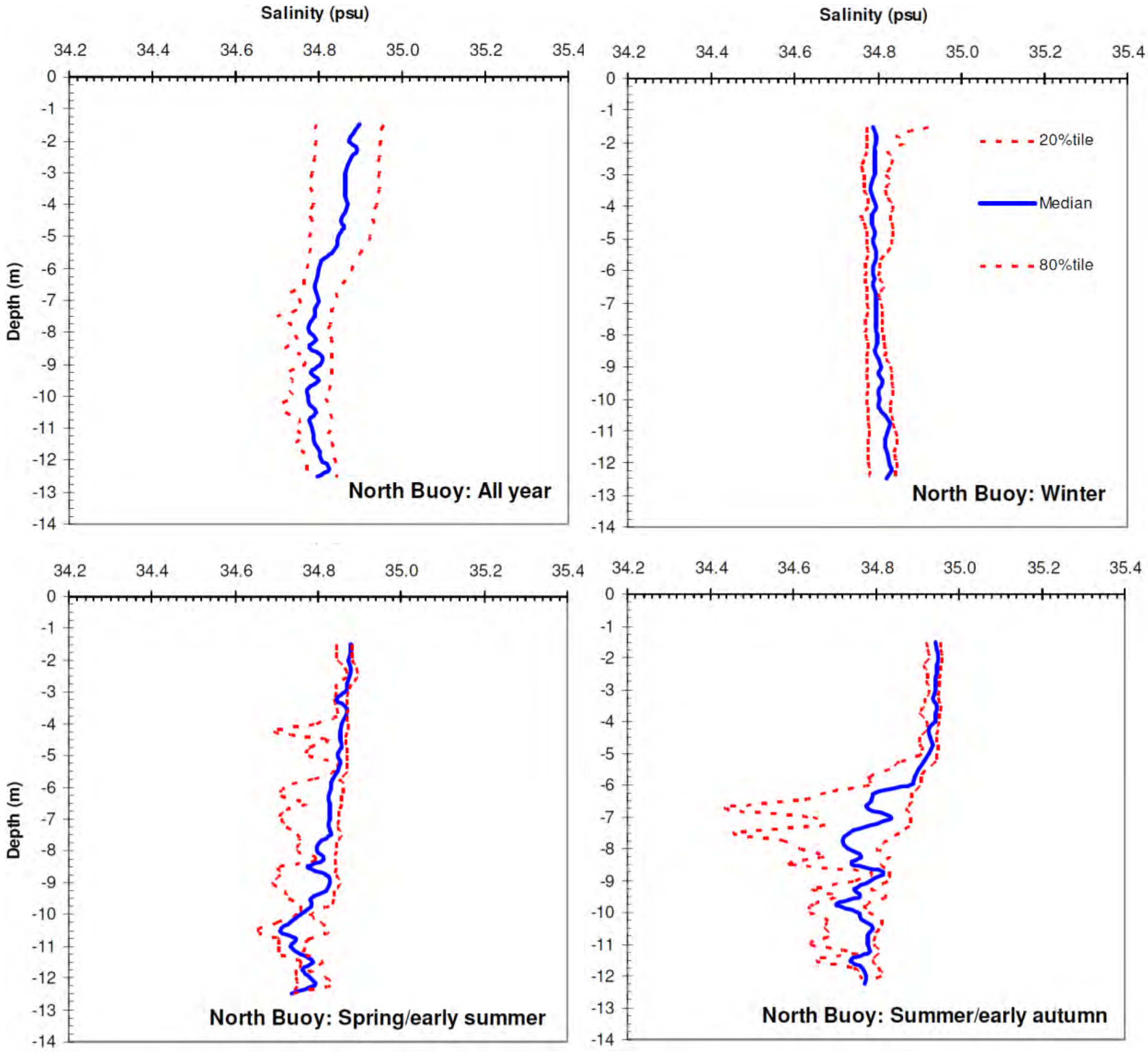


Figure 6.15. Salinity median profiles at North Buoy in Small Bay for all seasons (winter, spring/early summer and summer/early autumn). The 20th and 80th percentile limits of the profiles are indicated by the dotted red lines. (Source: van Ballegooyen *et al.* 2012).

6.6 Dissolved oxygen

Sufficient dissolved oxygen in sea water is essential for the survival of nearly all marine organisms. Low oxygen (or anoxic conditions) can be caused by excessive discharge of organic effluents (from for example, fish factory waste or municipal sewage) as microbial breakdown of this excessive organic matter depletes oxygen in the water. The well-known “black tides” and associated mass mortalities of marine species that occasionally occur along the west coast results from the decay of large plankton blooms under calm conditions. Once all the oxygen in the water is depleted, anaerobic bacteria (not requiring oxygen) continue the decay process, causing the characteristic sulphurous smell.

Apparent Oxygen Utilization (AOU) is a measure of the potential available oxygen in the water that has been used by biological processes. Values for Small and Big Bay over the period April 1974 to October 1982 and July 1988 are given in Monteiro *et al.* (1990). AOU is defined as the difference between the saturated oxygen concentration (the highest oxygen concentration that could occur at a given water temperature e.g. 5 ml/l) and the measured value (e.g. 1 ml/l). Hence positive AOU (5 ml/l – 1 ml/l = 4 ml/l) values indicate an oxygen deficit (highlighted red in Figure 6.16). More recent data on oxygen concentrations in Small Bay (covering the period September 1999 to February 2000) were provided by Monteiro *et al.* (2000). During this study, oxygen concentration at 10 m depth was recorded hourly by an instrument moored in Small Bay. These values were converted to AOU and monthly averages are plotted in Figure 6.14.

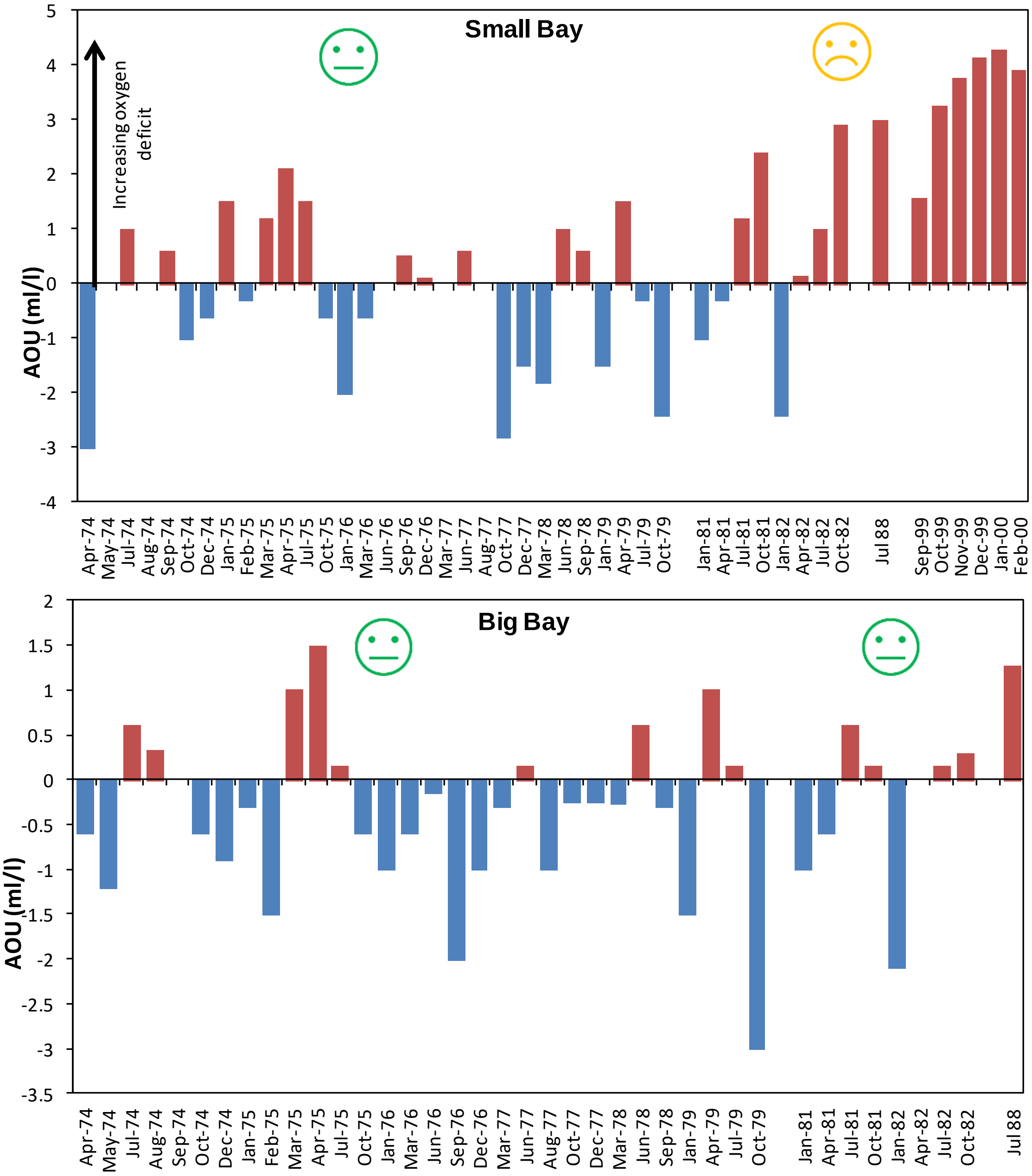


Figure 6.16. Apparent oxygen utilization time series for Small Bay and Big Bay in Saldanha Bay. Positive values in red indicate an oxygen deficit (Data sources: Monteiro *et al.* 1990 and 2000).

There is no clear trend evident in the AOU time series, as low oxygen concentrations (high AOU values) occur during both winter and summer months (Figure 6.16). Small Bay does experience a fairly regular oxygen deficit during the winter months, whilst Big Bay experiences less frequent and lower magnitude oxygen deficits. Monteiro *et al.* (1990) attributed the oxygen deficit in Small Bay largely to anthropogenic causes, namely reduced flushing rates (due to the causeway and ore terminal construction) and discharges of organic rich effluents. The most recent data (September 1999 to February 2000) indicate a persistent and increasing oxygen deficit as summer progresses (Figure 6.16). It is clear that oxygen levels within Small Bay are very low during the late summer months, likely as a result of naturally occurring conditions; however, the ecological functioning of the system could be further compromised by organic pollutants entering the Bay. There is evidence of anoxia in localised areas of Small Bay (e.g. under the mussel rafts and within the yacht basin) that is caused by excessive organic inputs. Monteiro *et al.* (1997) identified the effluent from a pelagic fish processing factory as the source of nitrogen that resulted in an *Ulva* seaweed bloom in Small Bay.

Examples of the dissolved oxygen data from the water column profiling exercises undertaken by the CSIR at North Buoy in 2010/2011 are shown in Figure 6.17 (van Ballegooyen *et al.* 2012). The profiles indicated that dissolved oxygen concentrations are high in winter but very low in the bottom waters and near the seabed in summer, late summer and early autumn. These low oxygen concentrations in the near bottom waters are considerably lower than those reported by Shannon & Stander (1977) for the period prior to the development of the port, but those in the upper water column are similar. Shannon & Stander's results for dissolved oxygen concentrations for the period April 1974 to October 1975 are as follows:

- 8.60 ± 1.86 (standard deviation) mg/l at the surface
- 7.96 ± 1.63 mg/l at -5m
- 6.85 ± 1.54 mg/l at -10 m
- 5.13 ± 1.80 mg/l at -20m

The *in situ* mooring installed by the CSIR in 2010/2011 as part of the baseline monitoring for the RO plant yielded temperature, salinity and dissolved oxygen times series for the period 9 July 2012 to 23 March 2012 at a temporal resolution of 10 minutes (Figure 6.18).

Observations highlighted by the CSIR (van Ballegooyen *et al.* 2012) from these data are as follows:

- The most obvious variability in the Bay is that which occurs over synoptic (weather) time scales.
- South-easterly to southerly winds result in upwelling that advects cold, lower salinity and oxygen deficient waters into the Bay.
- If the winds continue to blow, then a degree of vertical mixing takes place, resulting in a slow increase in temperature, salinity and dissolved oxygen in the bottom waters.
- When the wind drops or reverses to NW, then the water column develops a high degree of stratification shortly followed by a relaxation of upwelling that leads to the colder, less saline and low oxygen bottom waters exiting the Bay. Coupled with vertical mixing, this results in the warmer more oxygenated surface waters being mixed downwards, sometimes to the depth of the mooring.

- As summer progresses, the bottom waters are more insulated from the surface waters and the variability in temperature, salinity and dissolved oxygen of the bottom waters decreases compared to spring and early summer.
- The dissolved oxygen in the bottom waters decreases throughout summer to early autumn when the winter storms and vertical mixing of the water column alleviated these low oxygen conditions.

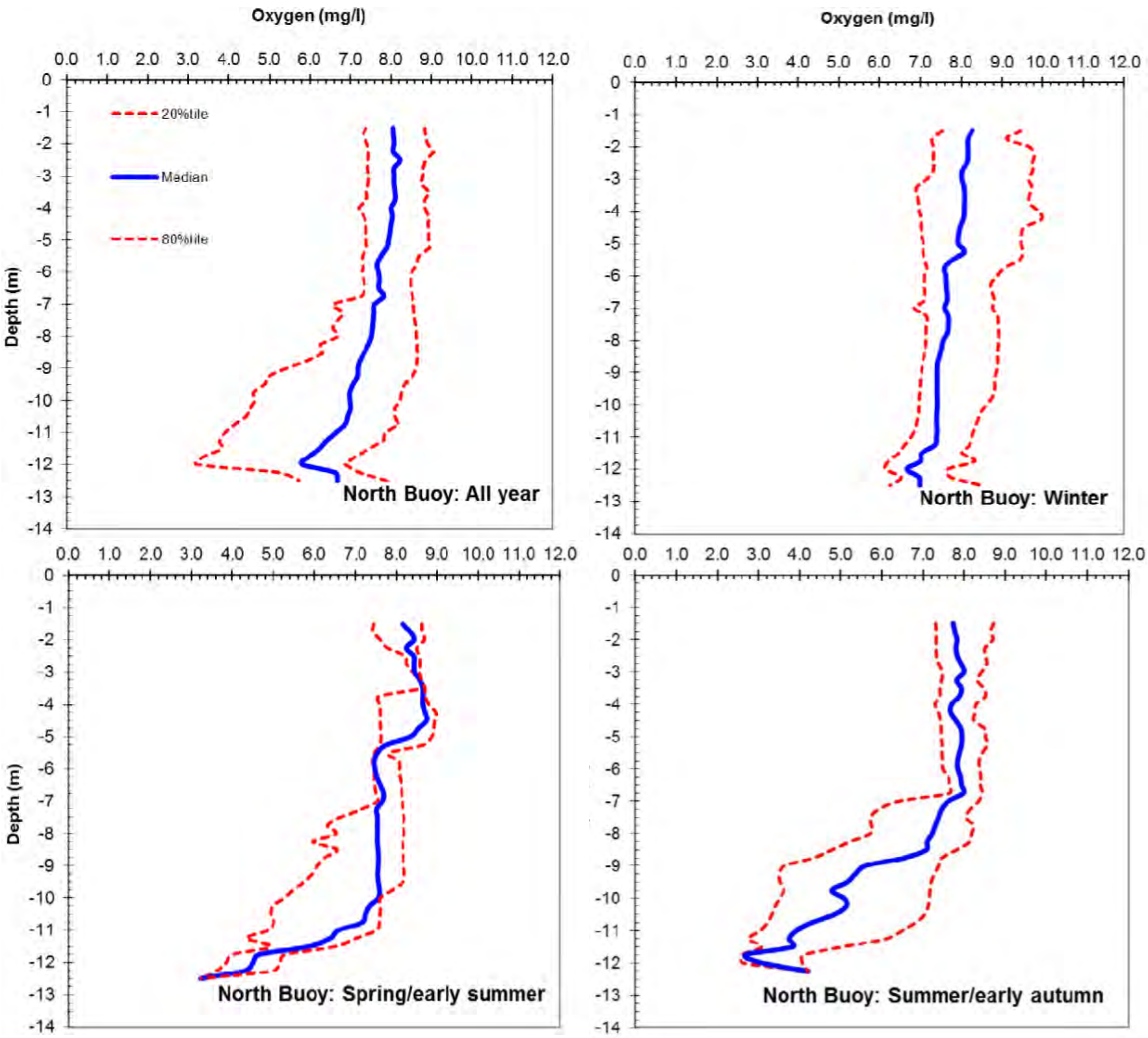


Figure 6.17. Dissolved oxygen concentration median profiles at North Buoy for all seasons (winter, spring/early summer and summer/early autumn). The 20th and 80th percentile limits of the profiles are indicated by the dotted red lines (Source: van Ballegooyen *et al.* 2012).

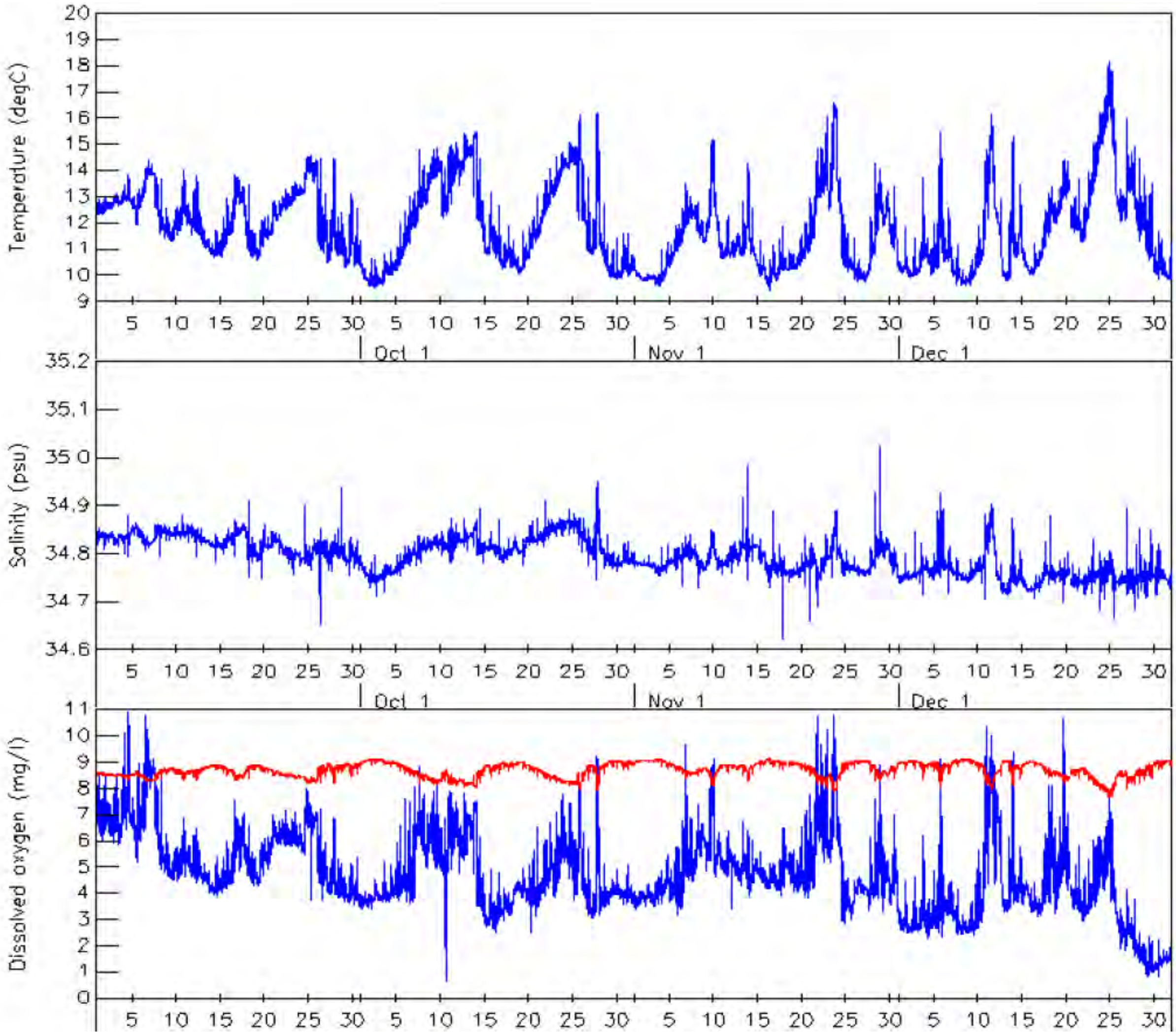


Figure 6.18. **Time series of water temperature, salinity and dissolved oxygen concentrations from the mooring site (33° 01.679'S; 17° 59.143'E) for spring/early summer (Source: van Ballegooyen *et al.* 2012).**

The CTD deployment during April/May 2017 in 22 m water depth on the Big Bay side of the RO Plant discharge was very close to the mooring deployed by the CSIR in 2010/2011. The instrument recorded depth, temperature, pH, salinity and dissolved oxygen at 20-minute intervals (Figure 6.19). The data show the same synoptic scale variability in temperature and dissolved oxygen as reported by van Ballegooyen *et al.* (2012), with a positive correlation between dissolved oxygen and temperature reflecting alternate stratification and water column mixing associated with upwelling and relaxation phases over 3-10 day periods. During this late autumn deployment, dissolved oxygen levels were noticeably lower than those recorded by the CSIR mooring that was in shallower water (18 m vs 23 m) and during the spring/early summer period. The very low dissolved oxygen values recorded for a short period in early May (1 to 2 $mg.l^{-1}$) are below the level that is tolerable for many invertebrates and most fish species. This low oxygen event was associated with an influx of cold water from the adjacent coast where low oxygen water is known to occur during autumn. Salinity remained constant within a narrow range for most of the deployment period except for two sharp drops to just below 33.5 ppt (these are probably anomalous readings due to instrument error). No salinity spikes were detected in the data series indicating that discharges of brine from the RO plant were not detected at the mooring site during the deployment, but it is not known if the RO Plant was operational during this period.

The most recent dissolved oxygen measurements come from ongoing deployments of instruments at mariculture Farm and Control sites in both Small Bay and Big Bay. The Small Bay control site is situated at North Buoy and is directly comparable to the dissolved oxygen recorded at 10 m depth at this location covering the period September 1999 to February 2000 as reported by Monteiro *et al.* (2000). The data from both these periods is presented in Figure 6.20. The clear decline in dissolved oxygen with the onset of summer upwelling is evident in the early 1999-2000 data set, with levels declining quite abruptly from 4-7 $ml.l^{-1}$ in mid-September to between 1-4 $ml.l^{-1}$ thereafter (Figure 6.20). The more recent data set shows that these frequently hypoxic conditions (<2 $ml.l^{-1}$) continue through until autumn (late May) when, as described above, the decline in upwelling and turbulent mixing of the water column with the onset of winter leads to higher dissolved oxygen levels in the bottom waters. It is also clear that there have not been major changes in dissolved oxygen levels in Small Bay over the last two decades, with regular hypoxic and even anoxic events recorded during summer and autumn both the 1999-2000 and 2020 data sets. The major increase in the frequency of Small Bay hypoxic events occurred after the major harbour development in the 1970s and the situation does not appear to have changed much since (Figure 6.16).

The 2020 data reveal that Big Bay also experiences regular hypoxic events during autumn, reflecting the external source of low oxygen upwelled water, but these are of a lower magnitude that seen in Small Bay (Figure 6.20). This is attributed to better mixing of the water column in Big Bay, lower retention times (enhanced flushing) and lower organic loading from anthropogenic sources (e.g. mariculture, fish factory effluent, wastewater treatment works). Note that there is currently no mariculture activity on the Big Bay Farm site that is earmarked for future fin fish cage farming. Within Small Bay oxygen concentration at the farm site are distinctly lower than the control site despite being substantially shallower (as dissolved oxygen typically decreases with depth, the deeper control site is expected to have lower dissolved oxygen under natural conditions) (Figure 6.20). The assumption it that this is a result of organic loading and higher microbial respiration, i.e. a consequence of bivalve mariculture. Conditions very close to anoxia are observed on the Small Bay farm site when control site dissolved oxygen concentrations, although also hypoxic, remained consistently higher (Figure 6.20). Near anoxic conditions have, however, occurred historically at the North buoy site in late December 1991 (Figure 6.20). Hypoxic and near anoxic conditions in the lower part of the water column are frequent occurrences during summer-autumn seasons in Small Bay and anthropogenic organic loading, appears to exacerbate the situation.

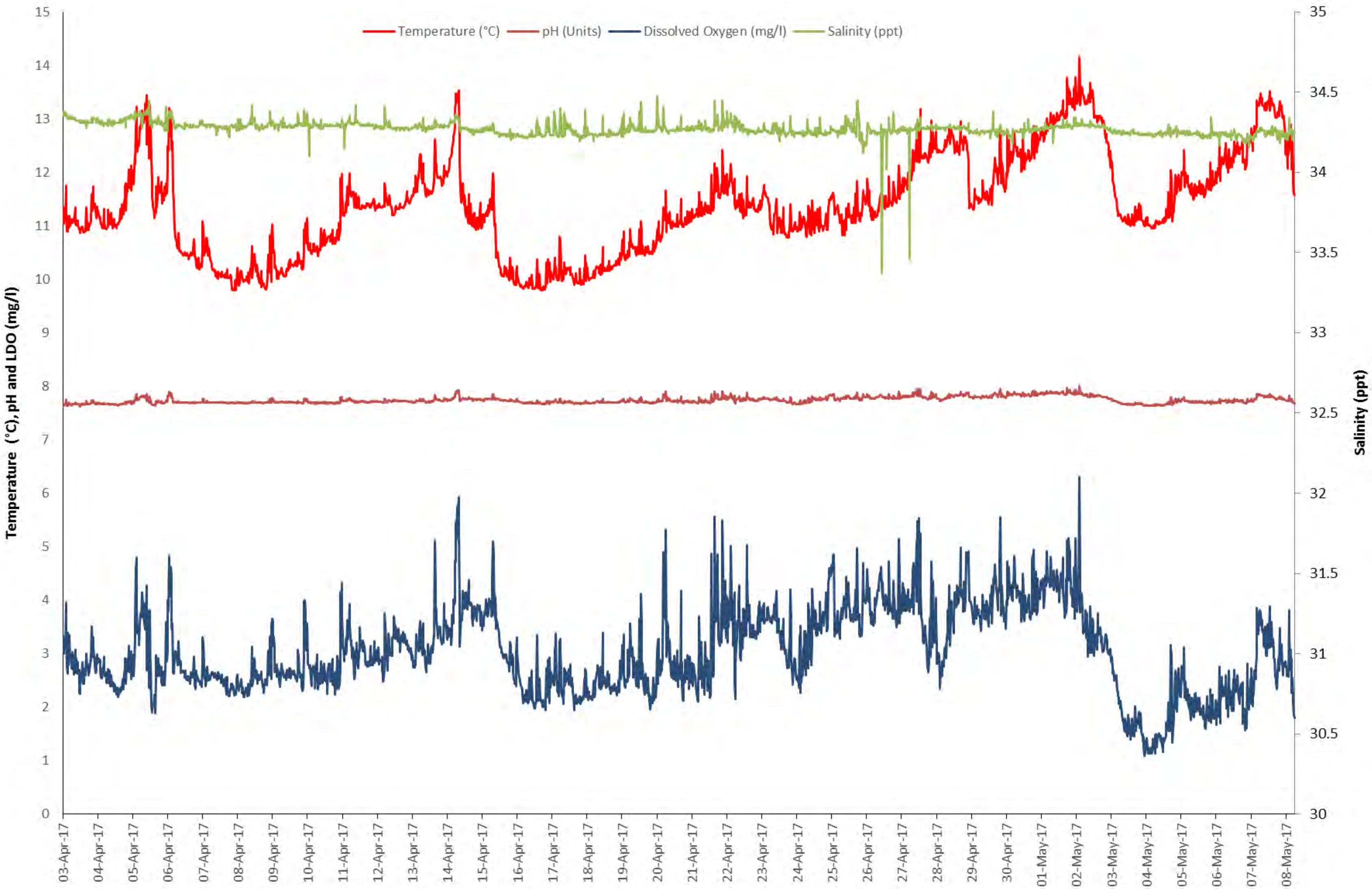
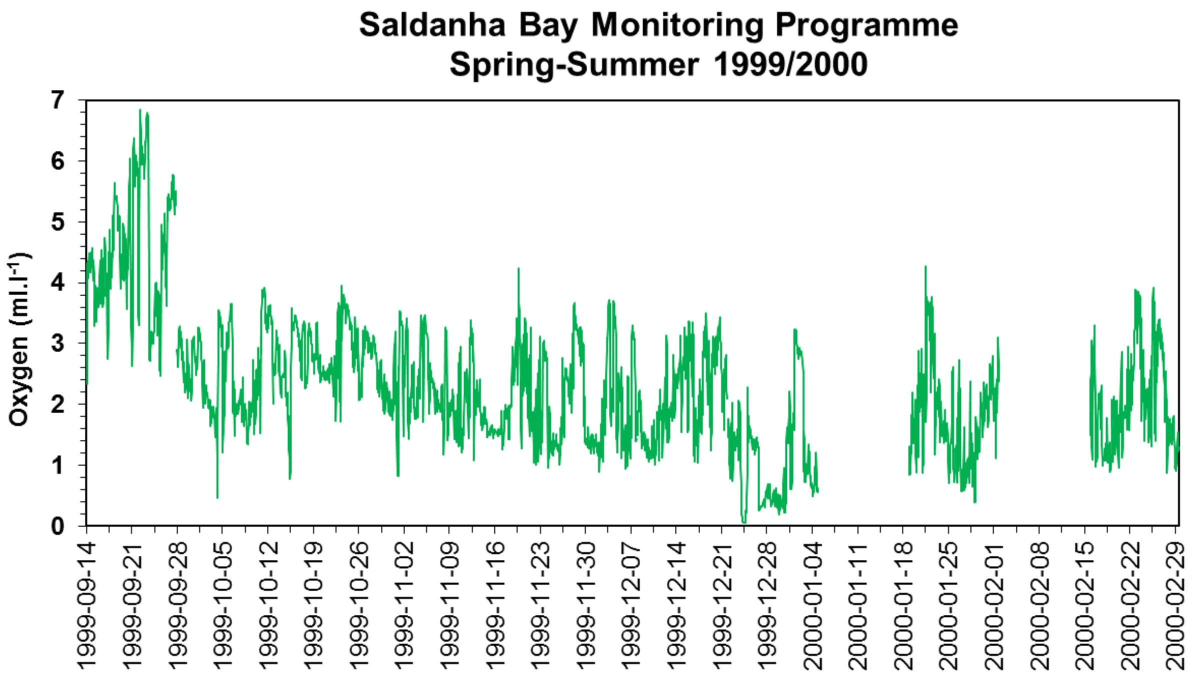


Figure 6.19. **Temperature, salinity, pH and dissolved oxygen (DO) recorded by the CTD deployed in 23 m water depth adjacent to the RO plant discharge at the base of the iron ore terminal.**



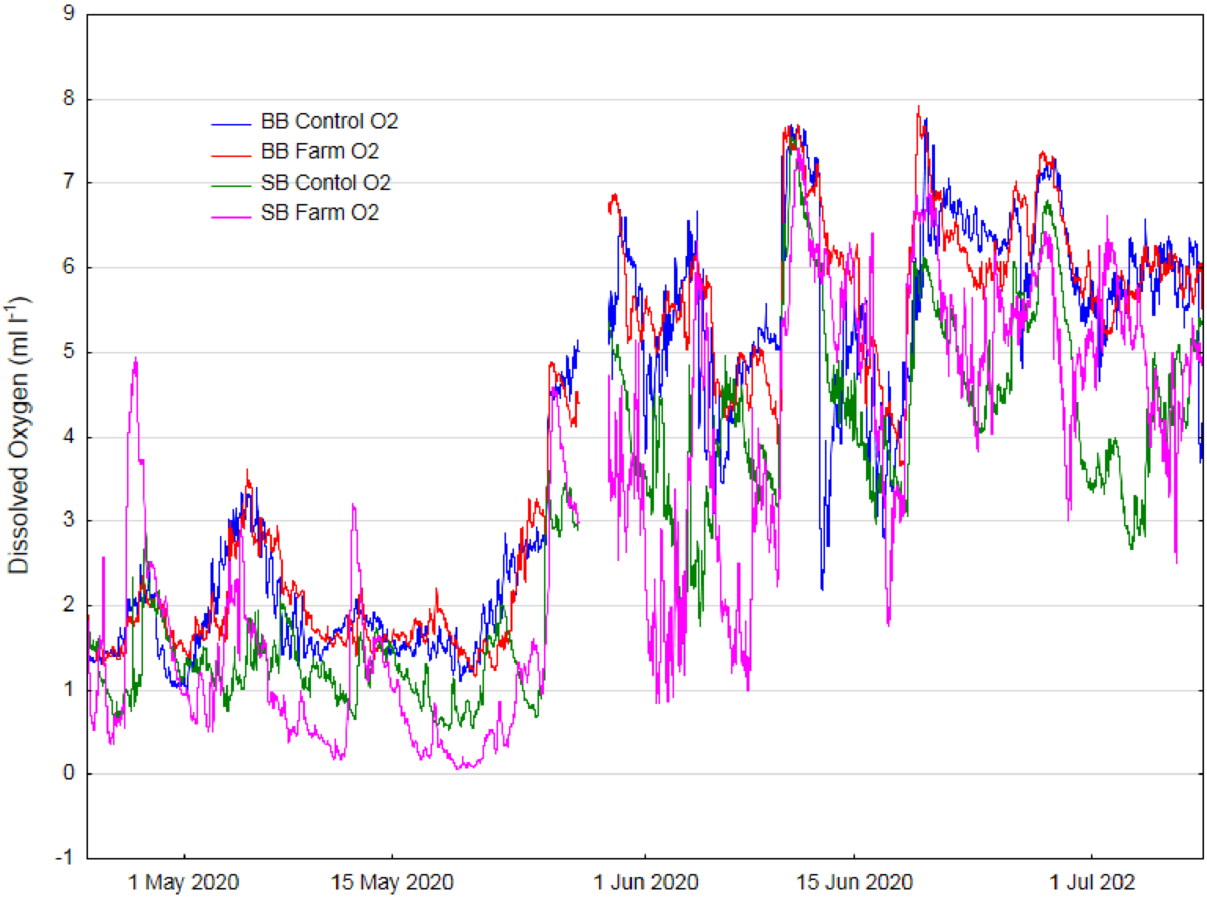


Figure 6.20. **Comparison of Dissolved Oxygen concentration recorded at 12 m depth at North Buoy (Small Bay Control) in 1999-2000 (top) and 2020 (bottom), as well as three other sites in 2020 Small Bay Farm (7 m depth), Big Bay Farm (16 m depth) and Big Bay Control (18 m depth). 2020 graph courtesy of DEFF.**

6.7 Turbidity

The CSIR describe the water of Saldanha Bay as being "fairly turbid", the turbidity comprising both organic and inorganic particulates that are suspended in the water column (van Ballegooyen *et al.* 2012). Turbidity in the Bay generally peaks under strong wind conditions (due to wind and wave action that suspends particulate matter in the water column, particularly Big Bay). Langebaan Lagoon, however, typically remains very clear even when the winds are very strong. This is likely due to the coarse nature of the sediment in the Lagoon when compared to the finer sediment in Saldanha Bay. Phytoplankton blooms and shipping movements have also been observed to cause significant increases in turbidity in the Bay. Historic measurements (n = 90) made by Carter and Coles (1998) indicate that average levels of Total Suspended Solids (TSS) in the Bay are in the order of 4.08 mg/l (± 2.69 mg/l SD) and peak at around 15.33 mg/l. Higher values caused by shipping movements (162 mg/l) have, however, been recorded by the CSIR (1996). Variations in turbidity caused by these different driving forces are clearly demonstrated in Google Earth images presented by CSIR (van Ballegooyen *et al.* 2012).

Data on turbidity (a measure of light conditions in the water column) and TSS (a measure of the mass per unit volume of particulates in the water column) were collected at water column profiling stations sampled for the RO plant baseline in 2010/2011 (van Ballegooyen *et al.* 2012). Turbidity data for the North Buoy site in Small Bay are shown here (Figure 6.21). In general, TSS concentrations are greatest near the seabed, particularly at the shallower sites in and around Small Bay. Concentrations generally did not exceed 10 mg/l, except for a few occasions where higher TSS of between 10 mg/l and 40 mg/l were observed (typically in the near bottom waters at the Mussel Farm site, at East Buoy in Big Bay, and in the immediate vicinity of the berths along the iron ore terminal). A few values above 100 mg/l were recorded in the vicinity of the iron ore terminal, reportedly related to shipping activities. The water column turbidity data reflected the same general trends as the TSS data, with turbidity in winter generally in the range of 5-12 NTU while in the other seasons the turbidity typically lay between 5 and 8 NTU (van Ballegooyen *et al.* 2012).

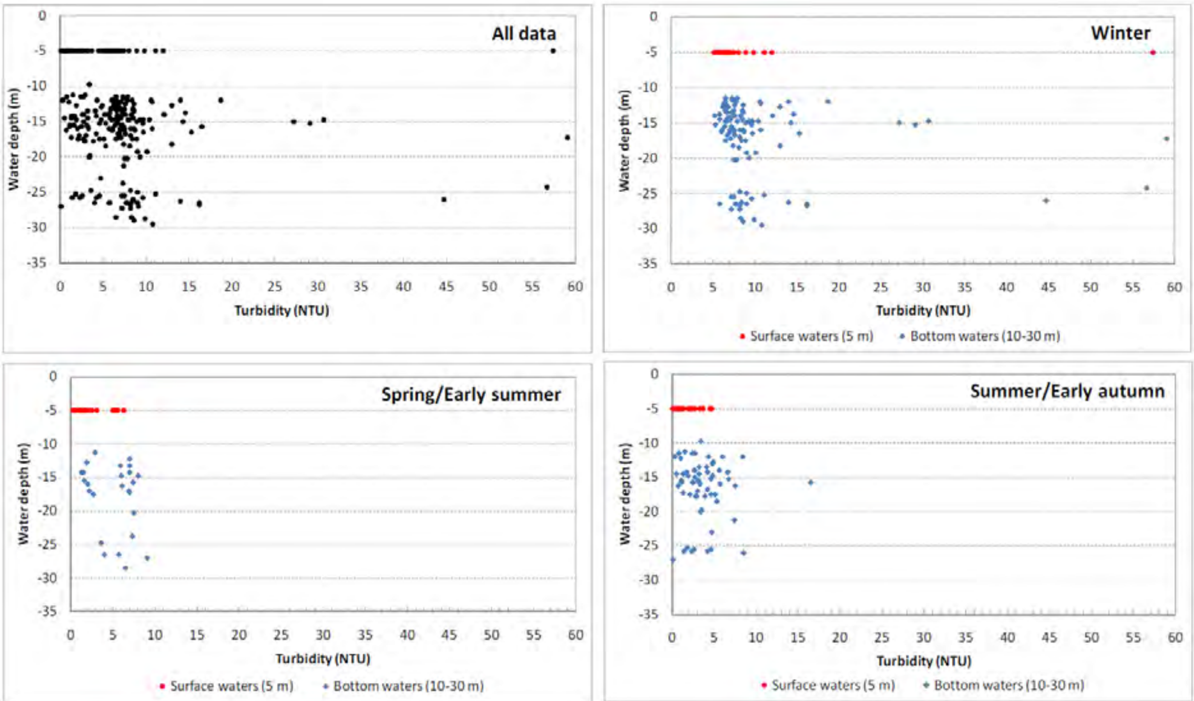


Figure 6.21. Turbidity (NTU) plotted as a function of depth and season (Source: van Ballegooyen *et al.* 2012).

Figure 6.22. Turbidity generated under high wind conditions (top) and by propeller wash (bottom) in Saldanha Bay (Source: van Ballegooyen *et al.* 2012).

6.8 Bromide

Measurements of bromide concentrations were collected at water column profiling stations sampled for the RO plant baseline in 2010/2011 (van Ballegooyen *et al.* 2012). Measurements were taken at the surface and near the bottom to determine natural occurrence in Saldanha Bay. The purpose was to ensure that the biocide proposed for the RO plant (2,2-dibromo-3-nitrilopropionamide or its break-down products) do not change natural distributions of bromide. Bromide concentrations in seawater are generally in the range of 65 mg/l to well over 80 mg/l in some confined sea areas. Data presented by the CSIR were consistent with these observations (between 40 and 95 mg/l, Figure 6.23), with variability higher in summer than in winter (van Ballegooyen *et al.* 2012). Variability was particularly high in spring/early summer and it was suggested that this may be related to maintenance dredging that occurred close to the sample sites around the iron ore terminal at the time.

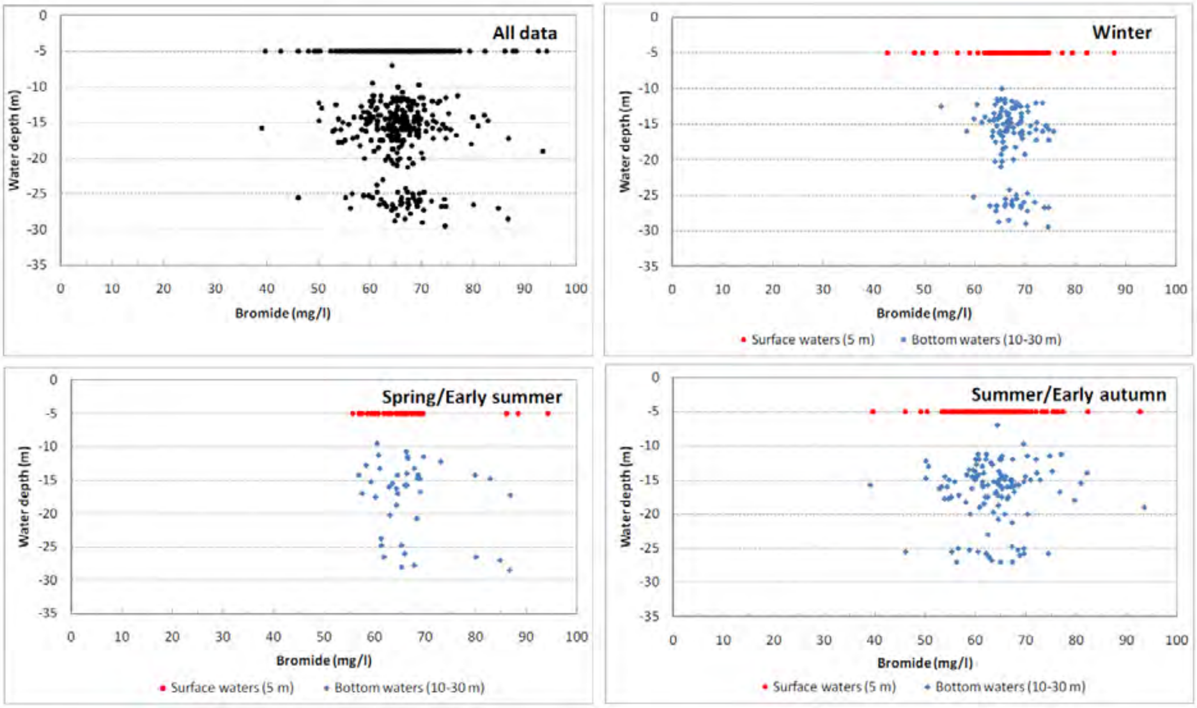


Figure 6.23. Bromide concentrations measured at all stations in winter, spring/early summer, and summer/early autumn (Source: van Ballegooyen *et al.* 2012).

6.9 Microbial indicators

Untreated sewage or storm water runoff may introduce disease-causing micro-organisms into coastal waters through faecal pollution. These pathogenic micro-organisms constitute a threat to recreational water users and consumers of seafood. Although faecal coliforms and *Escherichia coli* are used to detect the presence of faecal pollution, they provide indirect evidence of the possible presence of water borne pathogens and may not accurately represent the actual risk to water users (Monteiro *et al.* 2000). These organisms are less resilient than *Enterococci* (and other pathogenic bacteria), which can lead to risks being underestimated due to mortality occurring in the time taken between collection and analysis. To improve monitoring results, the enumeration of *Enterococci* should be included in water quality sampling programmes (DEA 2012).

6.9.1 Water quality guidelines

Marine water quality is assessed according to the most sensitive water use applicable to the specific area (e.g. mariculture vs. industrial use). For this study, WQGs for the natural environment (DWAF 1995a), industrial use (DWAF 1995c), and mariculture (DWAF 1995d) were used to assess water bodies not designated as recreational areas, while the evaluation of microbial data collected from Saldanha Bay and Langebaan Lagoon was undertaken in accordance with the revised guidelines for recreational use (DEA 2012) as described below.

6.9.1.1 Recreational Use

In the past, the DWAF (1995b) Water Quality Guidelines (WQGs) for coastal marine waters were used to assess compliance in respect of human health criteria for recreational use; however, these WQGs were replaced in 2012 by the revised *South African Water Quality Guidelines for Coastal Marine Waters Volume 2: Guidelines for Recreational Waters* (DEA 2012). The revised WQGs do not distinguish between different levels of contact recreation but rather evaluate aesthetics (bad odours, discolouration of water and presence of objectionable matter), human health and safety (gastrointestinal problems, skin, eye, ear and respiratory irritations, physical injuries and hypothermia), and mechanical interference. Measurable indicators commonly monitored include 'objectionable matter', water temperature and pH as well as the levels of intestinal *Enterococci* (or less ideally concentrations of *E. coli* or faecal coliforms). Guidelines state that samples should be collected 15 to 30 cm below the water surface on the seaward side of a recently broken wave in order to minimise contamination and reduce sediment content (DEA 2012). Samples to be tested for *E. coli* counts should be analysed within six to eight hours of collection, and those to be tested for intestinal *Enterococci*, within 24 hours.

The Hazen non-parametric statistical method is recommended for dealing with long-term microbiological data that do not typically fit a normal (bell shaped) distribution. The data are ranked into ascending order and percentile values are calculated using formulae incorporated in the Hazen Percentile Calculator (McBride and Payne 2009). In order to calculate 95th percentiles, a minimum of ten data points is required, while the calculation of the 90th percentile estimates require only five data points. Rather than using a measure of actual bacterial concentrations, a compliance index is used to determine deviation from a fixed limit (DEA 2012). This method is being increasingly used globally to determine compliance in meeting stringent water quality targets within specified time frames (e.g. Carr & Rickwood 2008). Compliance data are usually grouped into broad categories, indicating the relative acceptability of different levels of compliance. For example, a low count of bacteria would be 'Excellent', while a 'Poor' rating would indicate high levels of bacteria. Target limits, based on counts of intestinal *Enterococci* sp. and/or *E. coli*, for recreational water use in South Africa are indicated in Table 6.2.

Table 6.2. Target limits for *Enterococci* sp. and *E. coli* based on the revised guidelines for recreational waters of South Africa's coastal marine environment (DEA 2012). The probability of contracting a gastrointestinal illness (GI) is also listed.

Category	Estimated risk per exposure	Enterococci (count/100 ml)	*E. coli.* (count/100ml)
Excellent	2.9% GI risk	≤ 100 (95 percentile)	≤ 250 (95 percentile)
Good	5% GI risk	≤ 200 (95 percentile)	≤ 500 (95 percentile)
Sufficient/Fair (min. requirement)	8.5% GI risk	≤ 185 (90 percentile)	≤ 500 (90 percentile)
Poor (unacceptable)	>8.5 % GI risk	>185 (90 percentile)	>500 (90 percentile)

6.9.1.2 Mariculture Use

Filter feeding organisms, such as shellfish, can accumulate pathogenic organisms in their bodies and thereby infect the people that consume them. The *Guidelines for Inland and Coastal Waters: Volume 4 Mariculture* (DWAF 1995d) provides target levels for faecal coliforms in water bodies used for mariculture as outlined in Table 6.3. These guidelines aim to protect consumers of shellfish from bacterial contamination. For mariculture, faecal coliform concentrations for the 80th and 95th percentiles were calculated.

Table 6.3 Maximum acceptable count of faecal coliforms (per 100 ml sample) for mariculture according to the DWAF 1995 guidelines (DWAF 1995d).

Purpose/Use	Guideline value
Mariculture	20 faecal coliforms in 80% of samples 60 faecal coliforms in 95% of samples

6.9.2 Microbial monitoring in Saldanha Bay and Langebaan Lagoon

In 1998 the CSIR were contracted by the Saldanha Bay Water Quality Forum Trust (SBWQFT) to undertake fortnightly sampling of microbiological indicators at 15 stations within Saldanha Bay. The initial report by the CSIR, covering the period February 1999 to March 2000, revealed that within Small Bay, faecal coliform counts frequently exceeded the guidelines for both mariculture and recreational use (the 1995 guidelines of 100 faecal coliforms occurring in 80% of samples analysed) at nine of the 10 sampling stations. These results indicated that there was indeed a health risk associated with the collection and consumption of filter-feeding shellfish (mussels) in Small Bay. Much lower faecal coliform counts were recorded at stations within Big Bay, except for the 80th percentile guideline for mariculture being exceeded at one station (Paradise Beach). All other stations ranged within the guidelines for mariculture and recreational use (Monteiro *et al.* 2000).

Regular monitoring of microbiological indicators within Saldanha Bay has continued to the present day and is now undertaken by the West Coast District Municipality (WCDM). The available data cover the period February 1999 to August 2020 for 20 stations (ten in Small Bay, five in Big Bay and five in Langebaan Lagoon). Data during this period has, for the most part, been collected on a monthly or bimonthly basis since 1999 at 14 stations within Small and Big Bay in Saldanha, with the exception of Station 11 (Seafarm – Transnet National Ports Authority). where no data were collected during 2003, 2004, 2008, 2010 and 2011. Regular data collection was initiated at some of the Langebaan sites in 2004. Samples were collected at Stations 19 and 20 (Kraalbaai North and South respectively) for the first time in 2012. In previous SOB reports, data were presented cover a complete calendar year to account for seasonal differences, the 2019 and 2020 reports, however, includes data up until end July which includes both summer and winter data. Compliance with mariculture guidelines were assessed by comparing faecal coliform counts to the DWAF 1995 guidelines (DWAF 1995d), whilst recreational use compliance was assessed by comparing *E. coli* count data to the revised recreational guidelines (DEA 2012).

6.9.2.1 Water quality for recreational use

Recreational water quality rankings for all sampled sites throughout Saldanha Bay and Langebaan Lagoon are shown in Table 6.4, whilst Figure 6.24 and Figure 6.25 graphically depict these data for Langebaan Lagoon. Data from the microbial monitoring programme suggest that nearshore coastal waters in the system have improved considerably for recreational use since 2005 (Table 6.4). Based on the 2020 *E. coli* data, 11 of the 20 sampled stations were categorized as having excellent water quality and only two stations had poor water quality. The Bok River beach site that frequently had poor water quality for most of the monitoring record has shown consistent improvement over the last 3 years, ranked "Fair" in 2018 and 2019 and "good" in 2020. Water quality at the Hoedtjiesbaai site, however, has deteriorated and has been ranked as "Poor" for the last three years. Water quality at Pepper Bay (Site 5) also deteriorated considerably in 2020 and was ranked as "poor" for the first time since 2006. Three sites within Langebaan lagoon were rated as "Fair" water quality (Paradise beach, Leentjiesklip and Kraalbaai South), although in all three cases this was a result of insufficient data to calculate the Hazen 95th percentile values (precluding the allocation of a score better than "Fair"), and the 90th percentile values remained well below the guideline limit of 500 for "Poor" (Figure 6.25).

It is encouraging that "Fair" to "Excellent" water quality is being maintained at the popular swimming and water sport sites close to Langebaan (i.e. Mykonos Beach and Langebaan Main Beach). It is also encouraging that there have been improvements in water quality at beaches along the northern shore of Small Bay that are also popular swimming sites. It appears that the reuse of the majority of treated wastewater from the Langebaan Wastewater Treatment Works (WWTW) for other uses (including industrial, construction and irrigation) that was historically discharged via the Bok River Mouth is having a positive effect. Infrastructure upgrades on the treatment plant that were completed recently also appear to have had the desired effect of improved effluent quality. The fact that water quality has improved at sites near the Bok River mouth but have deteriorated at Hoedtjiesbaai suggests that the contamination may be from other sources (e.g. storm water, sewage leaks etc.). Similarly, the cause of the decline in water quality at the Pepper Bay-Big Quay site is not known, but potential sources should be investigated. Both these sites have buildings in close proximity to the waters edge and the integrity of any waste disposal infrastructure should be examined. See Chapter **3** for further information regarding activities and discharges in the Saldanha Bay-Langebaan Lagoon System.

Table 6.4. Sampling site compliance for <u>recreational use</u> based on <u>*E. coli* counts</u> for 10 sites in Small Bay, 5 sites in Big Bay and 5 sites in Langebaan Lagoon. Ratings are calculated using Hazen percentiles with the 90th and 95th percentile results grouped together to give an overall rating per annum. 'ND' indicates that there was insufficient data for the calculation of Hazen percentiles in that year and 'Ex.' indicates excellent water quality.

	Site	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020
Small Bay	1. Beach at Mussel Rafts	Fair	Fair	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Fair
	2. Small Craft Harbour	Ex.	Fair	Good	Ex.	Ex.	Ex.	Good	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Good	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.
	3. Sea Harvest - Small Quay	Fair	Fair	Ex.	Ex.	Fair	Ex.	Fair	Ex.	Ex.	Ex.	Good	Ex.	Fair	Ex.	Ex.	Ex.	Ex.	Ex.	Fair	Ex.	Ex.	Ex.
	4. Saldanha Yacht Club	Poor	Poor	Poor	Fair	Poor	Poor	Poor	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.
	5. Pepper Bay - Big Quay	Poor	Fair	Poor	Fair	Fair	Fair	Fair	Poor	Ex.	Ex.	Fair	Ex.	Ex.	Good	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Poor
	6. Pepper Bay - Small Quay	Poor	Fair	Fair	Good	Ex.	Good	Ex.	Ex.	Good	Ex.	Good	Good	Ex.	Good	Fair	Fair	Ex.	Ex.	Ex.	Ex.	ND	Ex.
	7. Hoedjies Bay Hotel - Beach	Fair	Fair	Poor	Fair	Good	Poor	Poor	Good	Fair	Ex.	Fair	Fair	Poor	Poor	Fair	Good	Fair	Good	Fair	Poor	Poor	Poor
	8. Beach at Caravan Park	Fair	Fair	Fair	Poor	Ex.	Fair	Poor	Ex.	Good	Poor	Fair	Fair	Fair	Poor	Good	Fair	Ex.	Fair	Fair	Fair	Fair	Fair
	9. Bok River Mouth - Beach	Poor	Fair	Poor	Poor	Poor	Poor	Poor	Ex.	Fair	Poor	Poor	Good	Ex.	Poor	Fair	Good	Ex.	Poor	Poor	Fair	Fair	Good
	10. General Cargo Quay - TNPA	Ex.	Fair	Ex.	Ex.	Ex.	Ex.	Good	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.
Big Bay	11. Seafarm - TNPA	Ex.	Fair	Ex.	Ex.	ND	ND	Ex.	Ex.	Ex.	ND	Ex.	ND	ND	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.
	12. Mykonos - Paradise Beach	Ex.	Fair	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Fair
	13. Mykonos - Harbour	Fair	Fair	Ex.	Ex.	Fair	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Fair	Ex.	Ex.	Good	Fair	Ex.	Ex.	Ex.	Ex.	Ex.
	14. Leentjiesklip	ND	ND	Good	Fair	Good	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Fair	Ex.	Good	Ex.	Ex.	Ex.	ND	Ex.	Ex.
Langebaan La	15. Langebaan North - Leentjiesklip	Ex.	Fair	Good	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Poor	Good	Ex.	Good	Ex.	Good	Ex.	Ex.	Fair
	16. Langebaan - Main Beach	ND	ND	Fair	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Good	Ex.	Ex.	Ex.	Ex.	Ex.	Fair	Ex.	Ex.	ND	Ex.	Good
	17. Langebaan Yacht Club	ND	ND	ND	ND	ND	Poor	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Good	Ex.	Ex.	Fair	Good	ND	Ex.	Ex.
	18. Tooth Rock	ND	ND	ND	ND	ND	Fair	Ex.	Ex.	Ex.	Ex.	Fair	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	ND	Ex.	Ex.
	19. Kraalbaai North	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	ND	Fair	Ex.
	20. Kraalbaai South	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	ND	Ex.	Fair

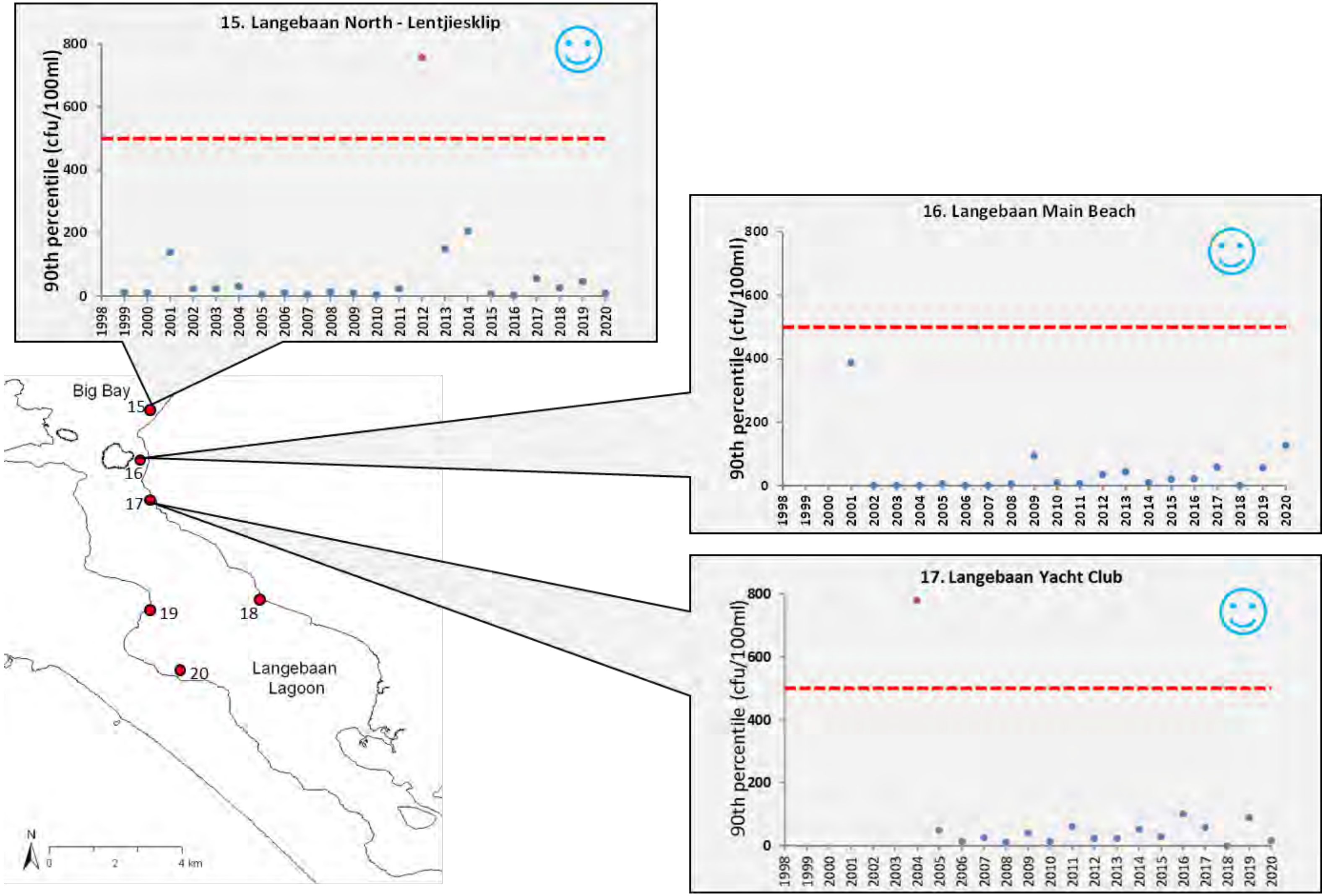


Figure 6.24. **Hazen method 90th percentile values of *E.coli* counts at three of the six sampling stations within Langebaan Lagoon (Feb 1999 – Aug 2020). The red line indicates the Hazen method <u>90th percentile contact recreation limit of *E. coli* counts</u> (500 colony-forming units/100 ml) above which water quality is ranked as 'Poor/Unacceptable'. Red data points indicate 90th percentile values exceeding the guideline, whilst blue data points fall within the recommended guideline. The faces correspond to changes water quality over time.**

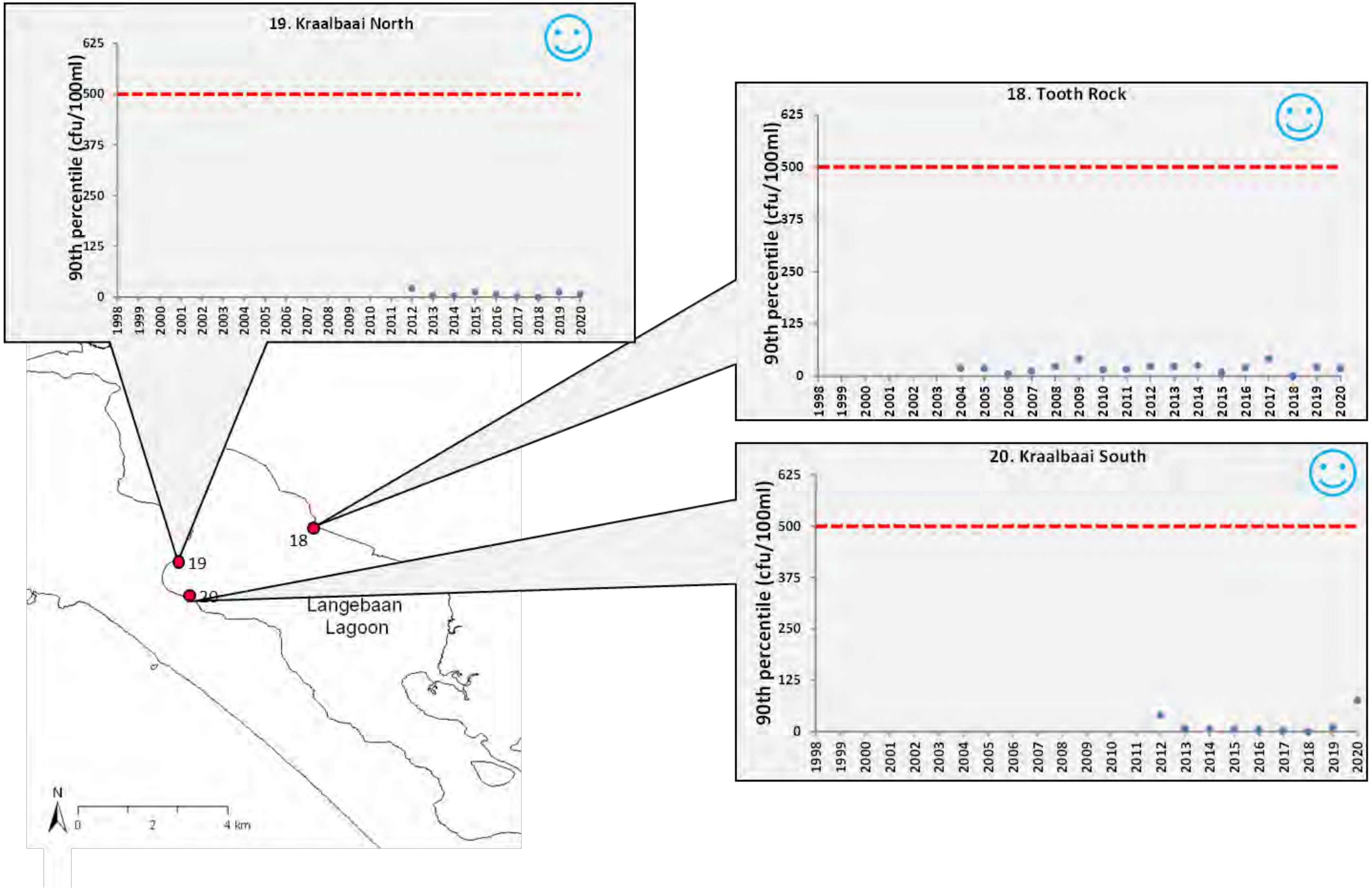


Figure 6.25. **Hazen method 90th percentile values of *E.coli* counts at three of the six sampling stations within Langebaan Lagoon (Feb 1999 –August 2020). The red line indicates the Hazen method <u>90th percentile contact recreation limit of *E. coli* counts</u> (500 colony-forming units/100 ml) above which water quality is ranked as 'Poor/Unacceptable'. The faces correspond to water quality over time.**

6.9.3 Water quality for mariculture

Guideline limits for mariculture are much more stringent than recreational guideline limits and levels of compliance for mariculture are much lower than for recreational use. Concentrations of microbiological indicators in samples collected from shallow coastal waters close to sources of contamination (storm water drains etc.) were found to be higher than those further away from populated areas. At the start of the monitoring in 1999, nine out of the 10 sites in Small Bay (Sites 1-9) were non-compliant in respect of the 80th percentile mariculture guideline limits for faecal coliforms (Figure 6.26, Figure 6.27 and Figure 6.28). There has been considerable improvement over time, particularly at sites near the entrance to Small Bay (the beach at the Mussel Rafts, the Small Craft Harbour and the Saldanha Bay Yacht Club) that have met standards every year since 2000. More recent improvement is seen at the small quay at Pepper Bay, but the big quay was again non-compliant in 2020. In 2019, the General Cargo Quay didn't meet the mariculture standard for the first time in the 20-year sampling history, but was again compliant in 2020. The remaining three sites within Small Bay, however, continue to exceed the mariculture guidelines (i.e. Hoedjies Bay Beach, the beach at Caravan park and the Bok River Mouth). The areas of particular concern are Hoedjies Bay and the Bok River Mouth that historically have exceeded the guideline by orders of magnitude. However, at both these sites, some improvement has been seen over the last three years with year-on-year declines and cfu counts that are approaching the guideline (Figure 6.27 and Figure 6.28).

Although a sustained improvement in levels of compliance with mariculture WQGs has occurred since the 1999-2005 period at most sites (Figure 6.26, Figure 6.27), these data indicate that there remains a serious issue of water quality with respect to mariculture operations within Small Bay, particularly in light of the proposed additional mariculture development in the area. The prevailing poor water quality in the near-shore waters of Small Bay may force sea water abstraction further offshore at an increased cost for any land-based mariculture facilities wishing to develop within the Industrial Development Zone (IDZ).

Faecal coliform counts at three of the four sites sampled within Big Bay in 2018 were within the 80th percentile limit for mariculture, whilst all four sites were within the limit in 2019 and 2020 (data to beginning August 2020) (Figure 6.29). There has been no discernible trend over time at these four sites with the exception of a dramatic decrease in faecal coliform counts after the first three (2001-2003) sampling events at Leentjiesklip. The water quality in Big Bay has met mariculture guidelines nearly every year since 2004, with the exception of the Mykonos Harbour site when levels were marginally exceeded in 2009, 2011 and recently in 2017 and 2018.

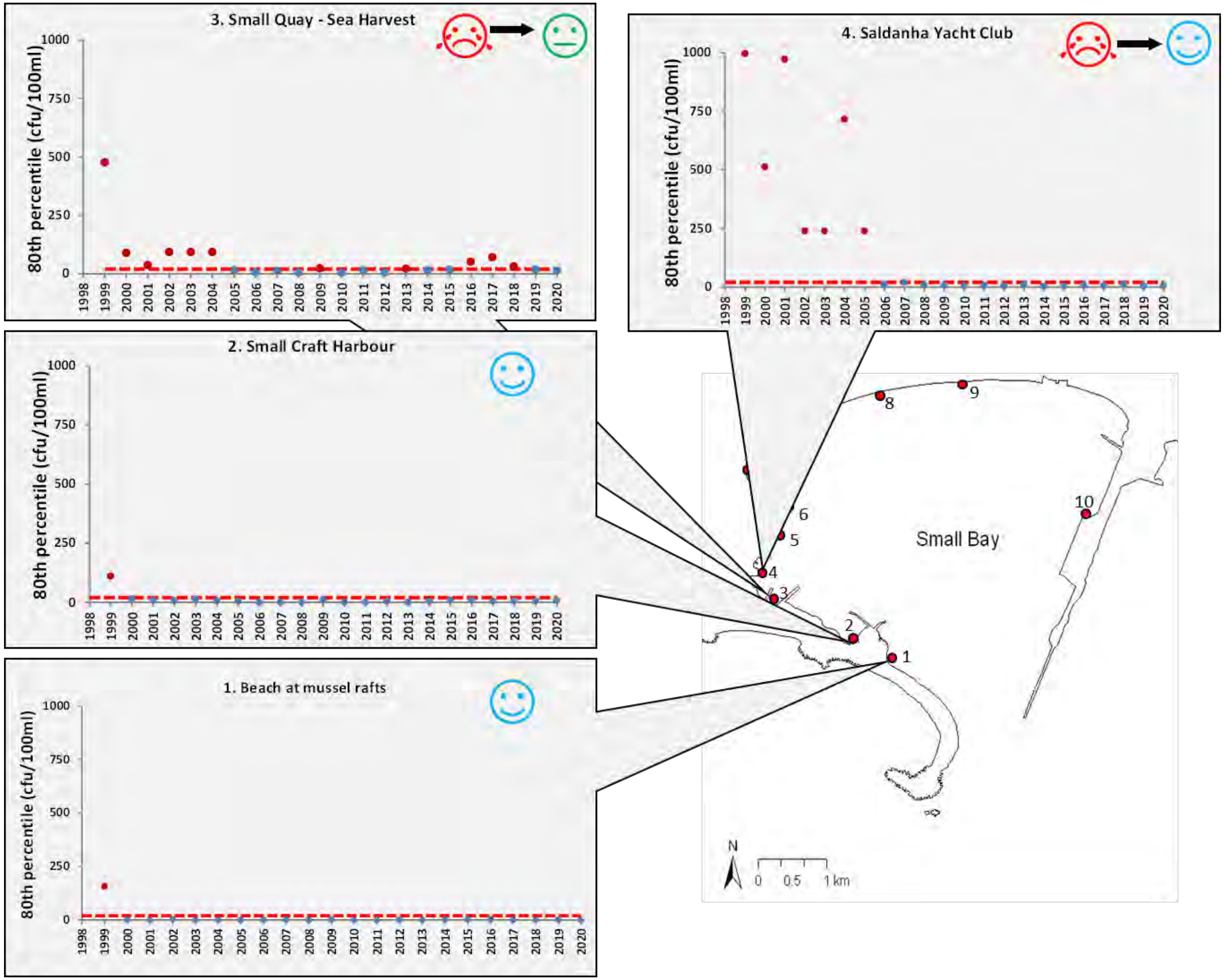


Figure 6.26. **80th percentile values of faecal coliform counts at four of the 10 sampling stations within Small Bay (Feb 1999 – August 2020). The red line indicates the <u>80th percentile mariculture limit of faecal coliforms</u> (20 colony-forming units/100 ml). Red data points indicate 80th percentile values exceeding the guideline, whilst blue data points fall within the recommended guideline. The faces correspond to changes in water quality over time.**

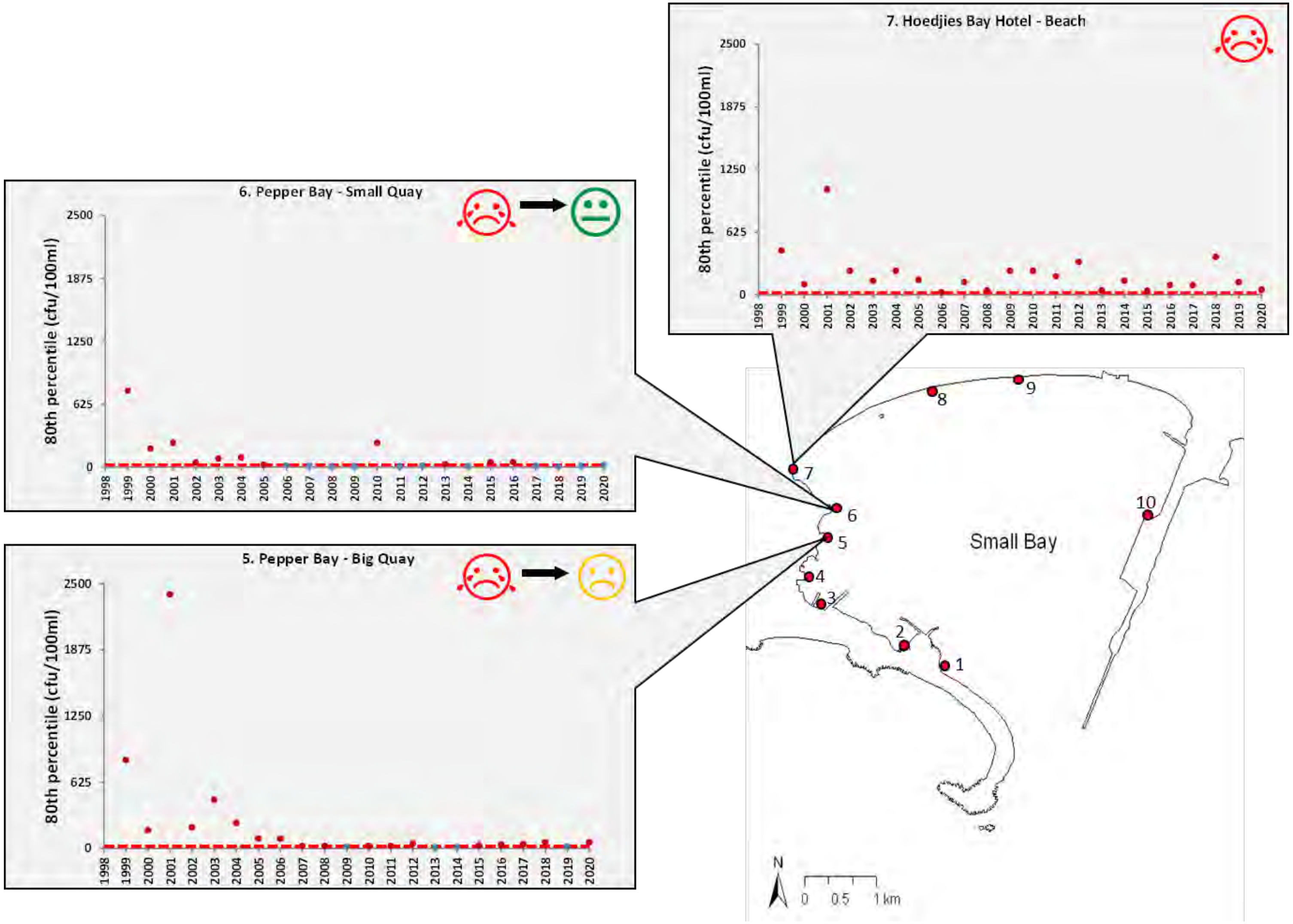


Figure 6.27. **80^{th} percentile values of faecal coliform counts at three of the 10 sampling stations within Small Bay (Feb 1999 – Aug 2020). The red line indicates the <u>80^{th} percentile mariculture limit of faecal coliforms</u> (20 colony-forming units/100 ml). Red data points indicate 80^{th} percentile values exceeding the guideline, whilst blue data points fall within the recommended guideline. The faces correspond to changes in water quality over time.**

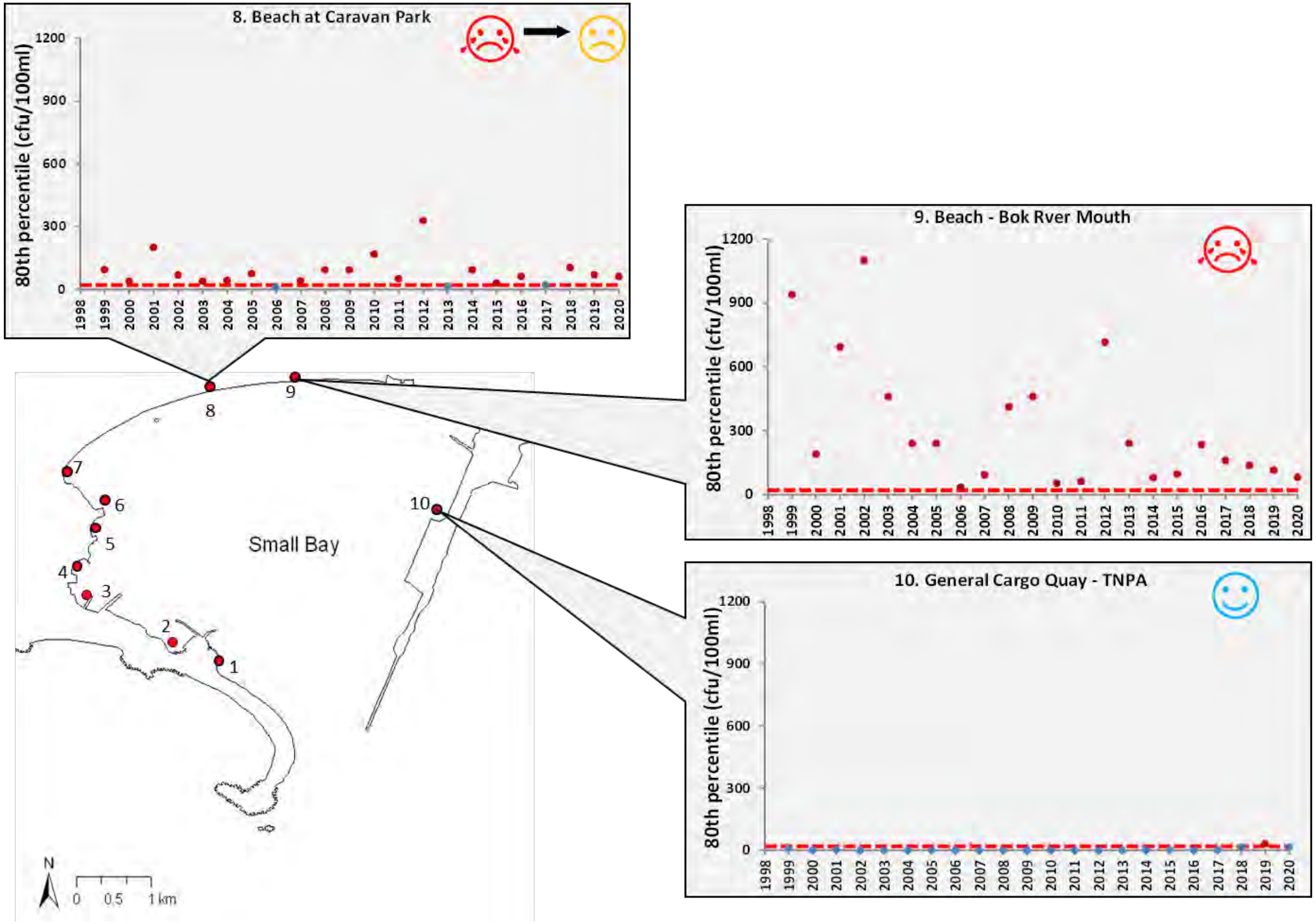


Figure 6.28. **80th percentile values of faecal coliform counts at three of the 10 sampling stations within Small Bay (Feb 1999 – Aug 2020). The red line indicates the <u>80th percentile mariculture limit of faecal coliforms</u> (20 colony-forming units/100 ml). Red data points indicate 80th percentile values exceeding the guideline, whilst blue data points fall within the recommended guideline. The faces correspond to changes in water quality over time.**

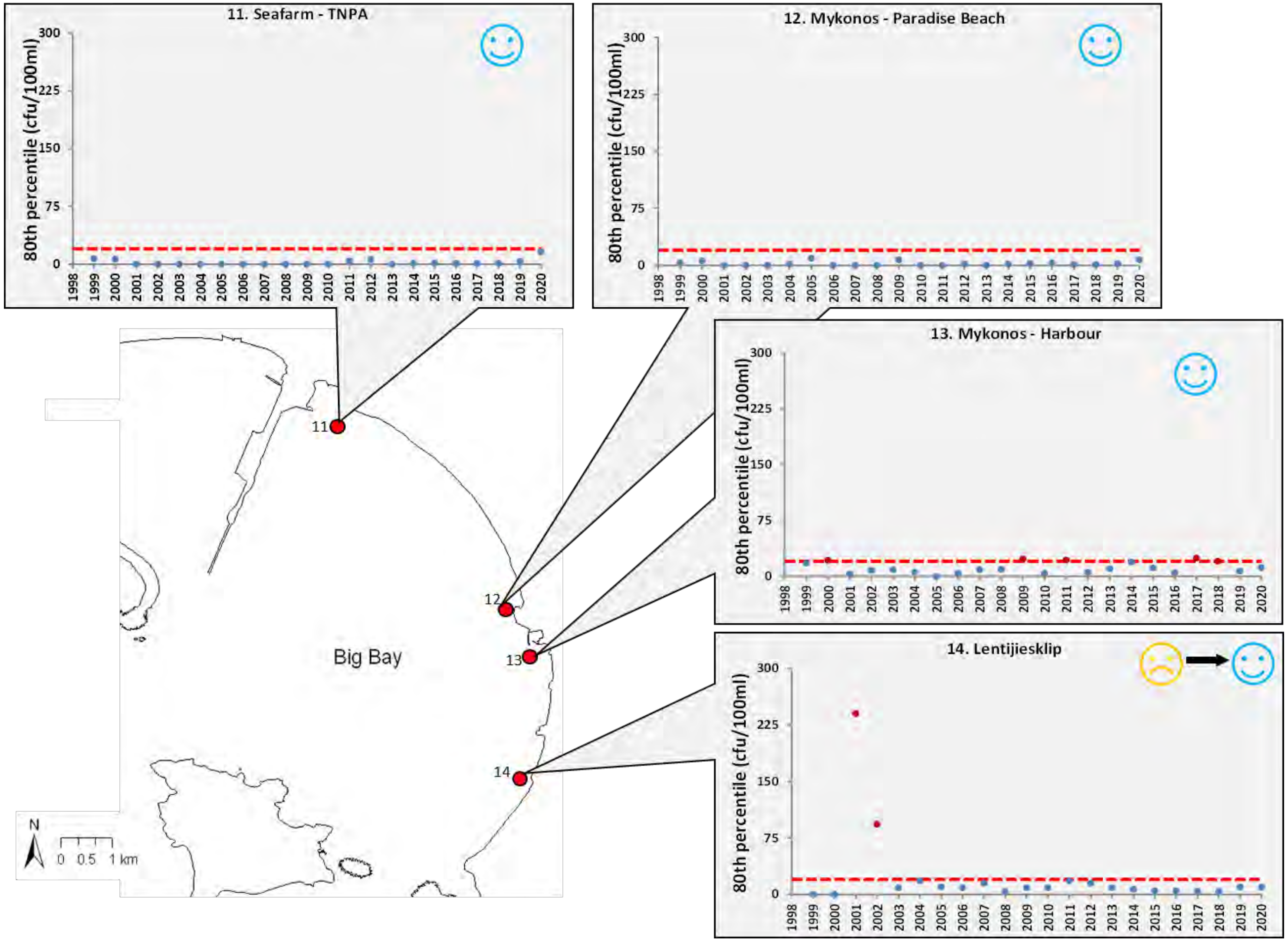


Figure 6.29. 80th percentile values of faecal coliform counts at the four sampling stations within Big Bay (Feb 1999 – Aug 2020). The red line indicates the 80th percentile mariculture limit of faecal coliforms (20 colony-forming units/100 ml). Red data points indicate 80th percentile values exceeding the guideline, whilst blue data points fall within the recommended guideline. The faces correspond to changes in water quality over time.

6.10 Heavy metal contaminants in the water column

It is common practise globally in countries like Canada, Australia, New Zealand and South Africa to monitor the long-term effects of pollution in water bodies by analysing levels in the tissues of specific marine species or species assemblages. Sessile bivalves (e.g. mussels and oysters) are good indicator species for monitoring water quality as these filter feeding organisms tend to accumulate trace metals, hydrocarbons and pesticides in their flesh. These sessile molluscs (anchored in one place for their entire life) are affected by both short-term and long-term trends in water quality. Monitoring contaminant levels in mussels or oysters can provide an early warning of poor water quality and detect changes in contaminant levels in the water column.

Trace/heavy metals are often regarded as pollutants of aquatic ecosystems; however, they are also naturally occurring elements, some of which (e.g. copper and zinc) are required by organisms in considerable quantities (Phillips 1980). Aquatic organisms accumulate essential trace metals that occur naturally in water as a result of, for example, geological weathering. All these metals have the potential to be toxic to living organisms at elevated concentrations (Rainbow 1995). High levels of cadmium, for example, reduces the ability of bivalves to efficiently filter water and extract nutrients, thereby impeding successful metabolism of food. Cadmium can also lead to injury of the gills of bivalves further reducing the effectiveness of nutrient extraction. Similarly, elevated levels of lead result in damage to mussel gills, increased growth deficiencies and possibly mortality. High levels of zinc are known to suppress the growth of bivalves at levels between 470 to 860 mg/l and can result in mortality of the mussels (DWAF 1995d).

Human activities greatly increase the rates of mobilization of trace metals from the earth's crusts and this can lead to increases in their bioavailability in coastal waters via natural runoff and pipeline discharges (Phillips 1995). Analysing dissolved metals in water is challenging as concentrations are typically low and difficult to detect, they have high temporal and spatial variability (e.g. with tides, rainfall events etc.) and most importantly they reflect the total metal concentration rather than the portion that is available for uptake by aquatic organisms (Rainbow 1995). Measuring metal concentrations in benthic sediments resolves analytical and temporal variability problems as metals accumulate in sediments over time and typically occur at higher concentrations than dissolved levels, but this still does not reflect their bioavailability. Analysing metal concentrations in the tissues of aquatic organisms appears to be the most suitable method for assessing ecotoxicity as the metals are frequently accumulated to detectable concentrations and reflect a time-integrated measure of bioavailable metal levels (Rainbow 1995).

Filter feeding organisms such as mussels of the genus *Mytilus* have been successfully used as bio-indicator organisms in environmental monitoring programs throughout the world (Kljaković-Gašpić *et al.* 2010). These mussels are abundant, have a wide spatial distribution, are sessile, are able to tolerate changes in salinity, are resistant to stress, and have the ability to accumulate a wide range of contaminants (Phillips & Rainbow 1993, Desideri *et al.* 2009, Kljaković-Gašpić*et al.* 2010).

6.10.1 Mussel Watch Programme

In 1985 the Marine and Coastal Management (MCM) branch of the Department of Environmental Affairs (DEA) initiated the Mussel Watch Programme whereby brown mussels *Perna* or Mediterranean mussels *Mytilus galloprovincialis* were collected every six months from 26 coastal sites. Mussels were collected periodically from five stations in Saldanha Bay. According to DEA, challenges in processing the mussel samples have resulted in data from the Saldanha Bay Mussel Watch Programme only being available between 1997-2001 and 2005-2007. As the programme was discontinued in 2007, Anchor Environmental Consultants initiated sampling again in 2014 by collecting mussel samples from the same five sites during the annual 'State of the Bay' field survey. The most recent mussel samples were collected in April 2019 and analysed for the metals lead (Pb), cadmium (Cd), zinc (Zn), copper (Cu), iron (Fe), manganese (Mn) and mercury (Hg). Data from the Mussel Watch Programme and from the annual 'State of the Bay' field trips are represented in Figure 6.31 to Figure 6.36 below.

In July 2017 DAFF fisheries management branch published the South African live molluscan shellfish monitoring and control programme (DAFF 2017). This document states that "sampling for heavy metals, polychlorinated biphenyls (PCBs) and pesticides should be conducted annually, while tests for radionuclides should be conducted every three years or more frequently if there is reason to suspect contamination. Sampling for specific contaminants is recommended only when the sanitary survey reveals a potential problem, or if there is concern due to a paucity of data." Sampling remains the responsibility of aquaculture facilities (see Section 6.10.2).

The maximum legal limits prescribed for each contaminant in shellfish for human consumption in South Africa (as stipulated by the Regulation R.500 of 2004 published under the Foodstuffs, Cosmetics and Disinfectants Act, Act 54 of 1972) are listed in Table 6.5 and indicated in red text on each series of graphs. All limits refer to concentrations of contaminants analysed relative to the wet weight of the flesh of the organism. Where limits have not been specified in national legislation, those adopted by other countries have been used (Table 6.5). Regulation No. 588 was updated on 15 June 2018 (Government Gazette No. 41704) to reduce the acceptable concentration of cadmium in marine bivalve molluscs from 3 to 2 mg/kg or ppm. As concentrations of lead and arsenic in marine mollusc flesh were not mentioned, the 2004 regulations were applied for these metals.

Table 6.5. Regulations relating to maximum levels for metals in molluscs (wet weight) in different countries.

Country	Cu (ppm)	Pb (ppm)	Zn (ppm)	As (ppm)	Cd (ppm)	Hg (ppm)
South Africa[1]		0.5		3.0	2.0[11]	0.5
Canada[2]	70.0	2.5	150.0	1.0	2.0	
Australia & NZ[3]		2.0			2.0	0.5
European Union[4]		1.5			1.0	0.5
Japan[5]		10.0			2.0	0.2
Switzerland[2]		1.0			0.6	0.5
Russia[6]		10.0			2.0	
South Korea[2]		0.3				
USA[7, 8]		1.7			4.0	
China[9]					2.0	
Brazil[10]						0.5
Israel[10]						1.0

1. Regulation R.500 (2004) published under the Foodstuffs, Cosmetics and Disinfectants Act, 1972 (Act 54 of 1972)
2. Fish Products Standard Method Manual, Fisheries & Oceans, Canada (1995).
3. Food Standard Australia and New Zealand (website)
4. Commission Regulation (EC) No. 221/2002
5. Specifications and Standards for Foods. Food Additives, etc. Under the Food Sanitation Law JETRO (Dec 1999)
6. Food Journal of Thailand. National Food Institute (2002)
7. FDA Guidance Documents
8. Compliance Policy Guide 540.600
9. Food and Agricultural Import Regulations and Standards.
10. Fish Products Inspection Manual, Fisheries and Oceans, Canada, Chapter 10, Amend. No. 5 BR-1, 1995.
11. Regulation No. 588 on 15 June 2018 (Government Gazette No. 41704) published under the Foodstuffs, Cosmetics and Disinfectants Act, 1972 (Act 54 of 1972)

To facilitate comparison with food quality guidelines, trace metal levels in bivalves in the 'State of the Bay' are presented relative to wet weights of bivalve tissue. Mercury concentrations within mussel tissues were measured for the first time in 2016 when they peaked at around 0.2 ppm. To date, values have not exceeded the safe limit of 0.5 ppm (Figure 6.30). Lead concentrations were found to exceed the regulatory limit for foodstuffs of 0.5 ppm at Portnet, Fish factory and the Saldanha Bay North sites in 2020 (Figure 6.31). Lead concentration in mussel tissue collected from the Iron ore jetty and Mussel raft 27/28 near the mouth of Small Bay was below the guideline limit, presumably due to better flushing compared to sites further in Small Bay. Mussels collected at the Portnet site have historically had high concentrations of lead in their tissue and although values in the last six years have not been as high as historical peaks, they remain more than double the recommended level in the 2020 samples. The high levels of lead are almost certainly linked to the export of lead ore from the multipurpose quay, which is situated near the Portnet site. The average concentration of lead in the tissues of mussels collected at the five sites within Small Bay has fluctuated from 0.9 ppm to 1.7 ppm over the last five years with an average of 0.9 ppm in 2020. This indicates that the lead pollution situation in Small Bay overall has not improved much. The level of lead in mussels at the Portnet site and Saldanha Bay north sites was double to five times respectively the level considered safe for human consumption in 2020 and levels of exceedance greater than this have been frequent at these two sites. This remains extremely concerning considering that mussels farmed within Small Bay are sold for human consumption (although trace metals in farmed mussels is consistently below that found in wild mussels on the shore, see 6.10.2).

Average cadmium levels in mussels from all sites over the period 2015-2019 ranged between 1.0 and 1.6 ppm, with an average of 0.9 ppm recorded in 2020. Historically, a maximum value of 10.9 ppm was recorded in April 2007 at the Mussel Raft. This was also the only site where the recommended level of 2.0 ppm was exceeded in 2018, where a concentration of 3.7 ppm was measured. In 2020 cadmium concentration in mussel tissue from all five sites sampled within Small Bay fell below the limit (Figure 6.32).

Average zinc concentrations recorded in 2020, and historically at nearly all sites, were much lower than the 150 ppm regulatory limit listed by the Canadian Authorities (Figure 6.33). This metal only rose above the limit once at the Saldanha Bay north site (165 ppm in 2016), which was also elevated in 2019 & 2020 samples albeit not above the guideline Figure 6.33). Concentrations of copper has remained well below the specified level of 70 ppm at all sites over the entire sampling period. There appears to be no spatial or temporal trend in level of copper in mussel samples. No regulatory limits exist for manganese in mollusc flesh as elevated levels have not been shown to have an adverse effect on marine life. Manganese is an important micronutrient in the oceans and there is evidence that manganese deficiency may limit phytoplankton productivity in some oceanic upwelling systems (Sunda 1989, Brand *et al*. 1983). Historically concentrations were highest at the Portnet site, and this was again the case in 2020 where levels peaked at just over 3 ppm, matching the all-time high recorded in 2019 (Figure 6.35). Manganese export volume has been steadily increasing from 95 000 tonnes in 2013/2014 to just over 4.5 million tonnes in 2017/2018 (see Chapter 7). Manganese concentrations in mussel tissue appears to have matched that trend at the Portnet site. Although the manganese loading terminal is midway between the General Purpose Quay at the base of the iron ore jetty and the iron ore terminal, currents and onshore winds will cause manganese dust to move towards the base of the jetty and accumulate in this area. As this trend appears to be ongoing, measures should be put in place to prevent excessive amounts of manganese dust from entering the Bay. In 2020, the Minister of Environmental Affairs instructed Transnet NPA to address the issue of manganese storage and pollution at the export terminal.

Iron concentrations in mussel tissue appears to have increased over the period 2014-2018 compared to most historical values over the 1997-2007 period (Figure 6.36). This trend may reflect increases in iron ore export volumes, despite dust mitigation measures implemented over time. The data is, however, not equivocal with some years e.g. 2000, recording high concentrations at all sites. Iron concentrations are typically highest at the Fish Factory and Saldanha Bay North sites and lowest at the Mussel Raft site, which probably reflects the effects of the prevailing southerly wind and the more retentive (less flushed) nature of the former sites. In 2020, a historical high of 87 ppm was recorded in the tissue of mussels collected at the Saldanha Bay north site. As there are no official limits outlined for the safe concentration of iron present in foodstuffs, it is not possible to comment on the suitability of these mussels for consumption based on this trace metal. Iron poisoning may be associated with the ingestion of more than 10-20 mg/kg of human body weight, but no cases of acute toxicity from regular foodstuffs (excluding supplements) has been recorded. Large volumes of iron ore is shipped from Saldanha Bay and iron ore residue is apparent on all structures downwind of the ore jetty and in the vicinity of the Saldanha Steel processing plant, it is therefore recommended that the concentration of this metal in the flesh of bivalves continue to be monitored.

The high level of lead in bivalve flesh remains a human health concern in Small Bay. This was also reported by Firth *et al.* (2019) who collected mussel samples from a Small Bay aquaculture farm every two months over a two-year period. Lead concentrations in excess of the South African guideline (0.5 ppm) were detected four times. These authors do note that the South African guideline for lead is set for "fish", and not specifically for bivalves. The European Commission has set a higher limit for bivalves which implies that farmed mussels from Saldanha Bay are actually safe for human consumption (Table 6.5). Nonetheless Firth *et al*. (2019) do recommend that it is "imperative to better control and regulate sources of lead pollution within Saldanha Bay". Wild mussels harvested from the shore do appear to accumulate higher levels of trace metals than farmed mussels probably due to better flushing and faster growth rates at farms. Signboards warning of the health risks of consuming coastal mussels in this area and discouraging their collection should be posted in areas where these bivalves are easily accessible (e.g. Hoedjiesbaai).

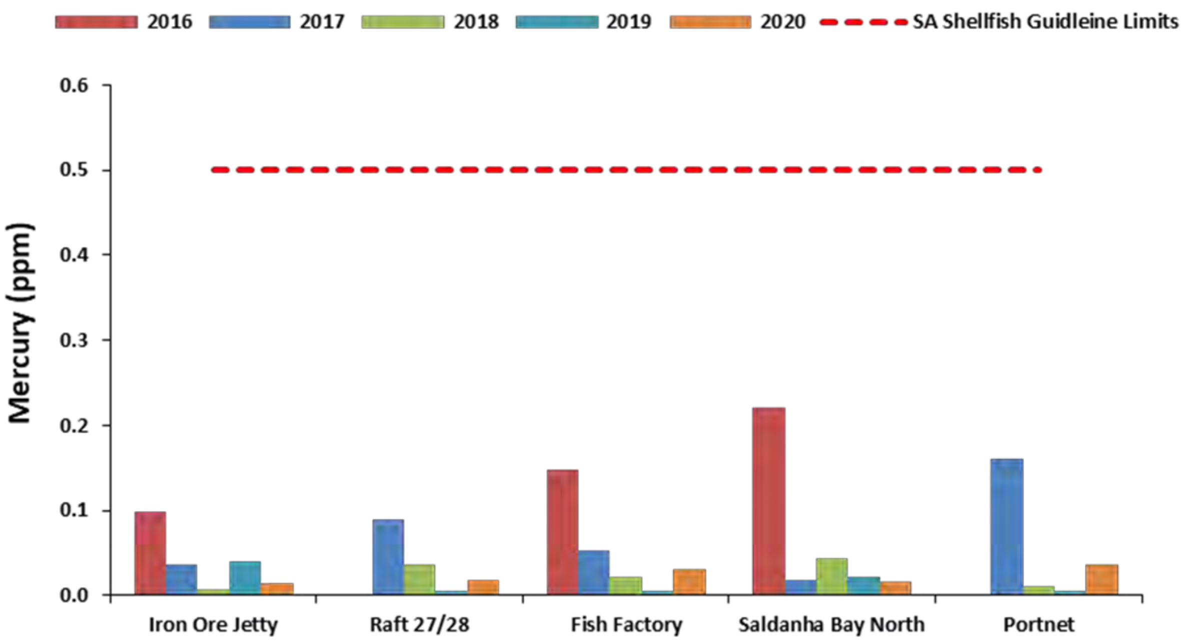


Figure 6.30. **Mercury concentrations in wet mussel flesh collected by Anchor from five sites in Saldanha Bay in autumn 2016 to 2020. The recommended maximum limit for mercury in seafood (0.5 ppm) is shown as a dotted red line.**

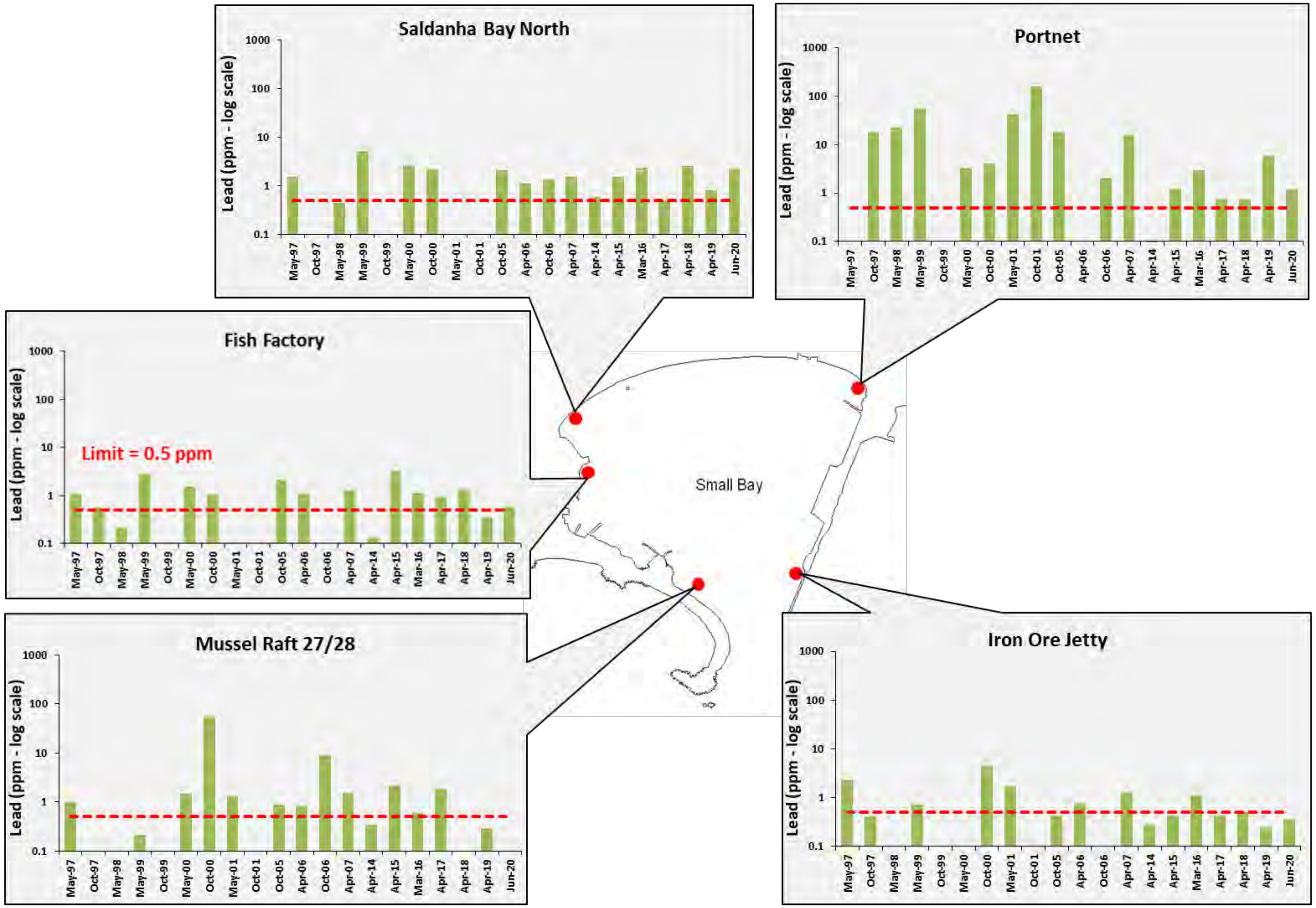


Figure 6.31. Lead concentrations in mussels (wet weight) collected from five sites in Saldanha Bay from 1997-2007 as part of the Mussel Watch Programme (Source: G. Kiviets, Department of Environmental Affairs) and by Anchor from 2014 to 2020. The recommended maximum limit for lead in seafood (0.5 ppm) is shown as a dotted red line. Note that data are plotted on a log scale.

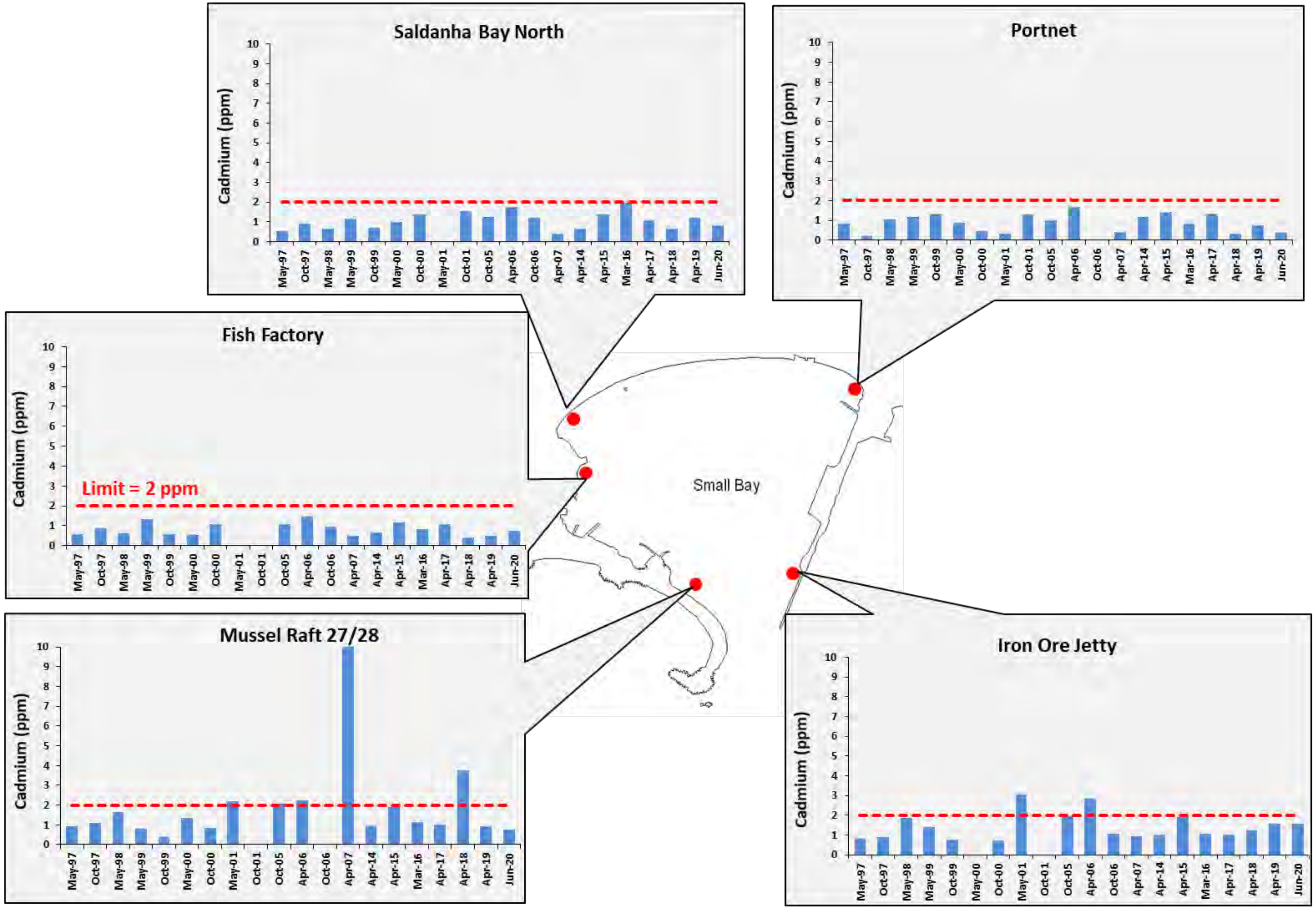


Figure 6.32. Cadmium concentrations in mussels (wet weight) collected from five sites in Saldanha Bay from 1997-2007 as part of the Mussel Watch Programme (Source: G. Kiviets, DEA) and by Anchor from 2014 to 2020. The recommended maximum limit for cadmium in seafood was reduced to 2 ppm (dotted red line) in 2018.

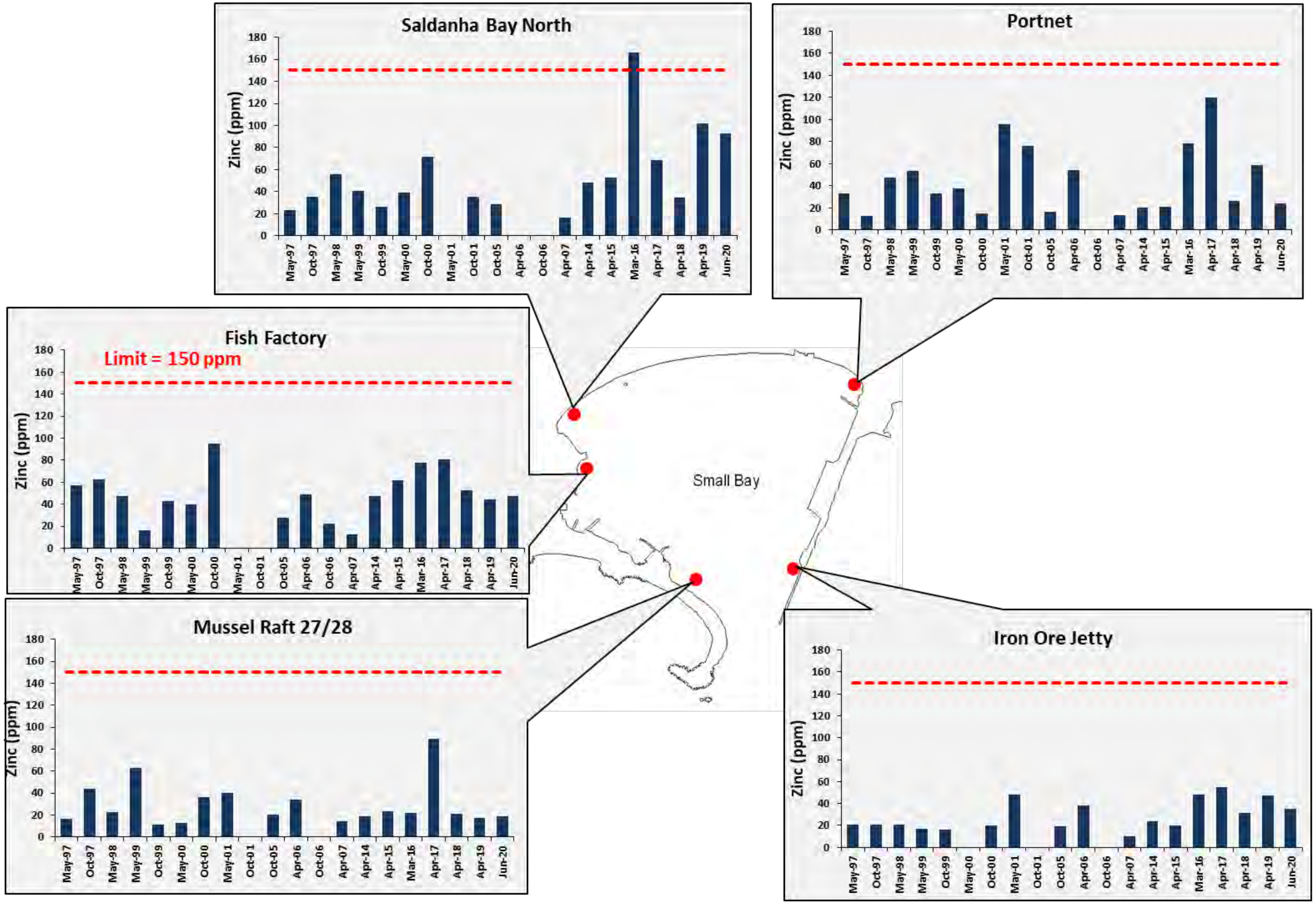


Figure 6.33. Zinc concentrations in mussels (wet weight) collected from five sites in Saldanha Bay from 1997-2007 as part of the Mussel Watch Programme (source: G. Kiviets, Department of Environmental Affairs) and by Anchor from 2014 to 2020. The recommended maximum limit for zinc in seafood (150 ppm) is shown as a dotted red line.

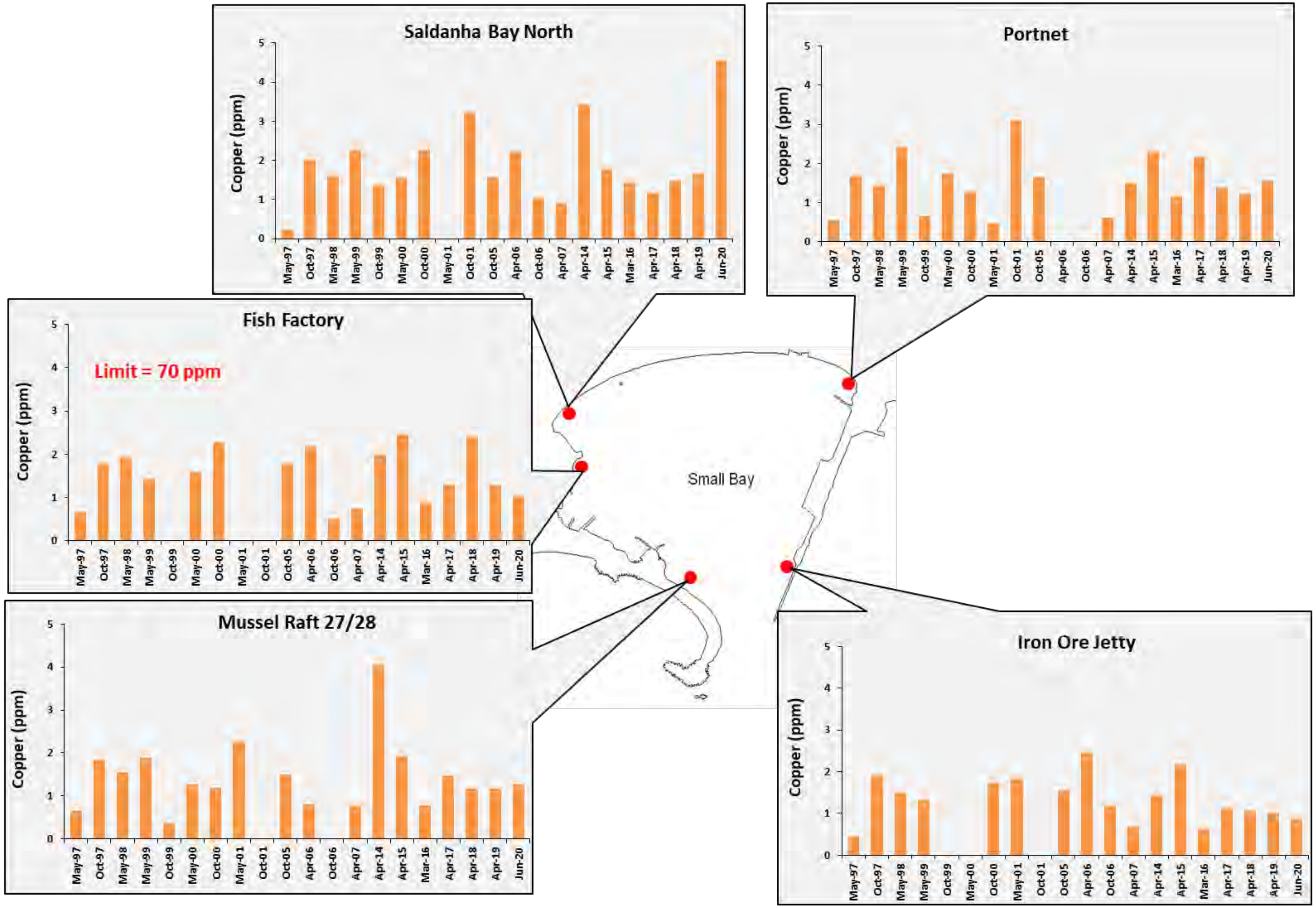


Figure 6.34. Copper concentrations in mussels (wet weight) collected from five sites in Saldanha Bay from 1997-2007 as part of the Mussel Watch Programme (Source: G. Kiviets, Department of Environmental Affairs) and by Anchor from 2014 to 2020. The recommended maximum limit for copper in seafood is 70 ppm (not indicated on graphs).

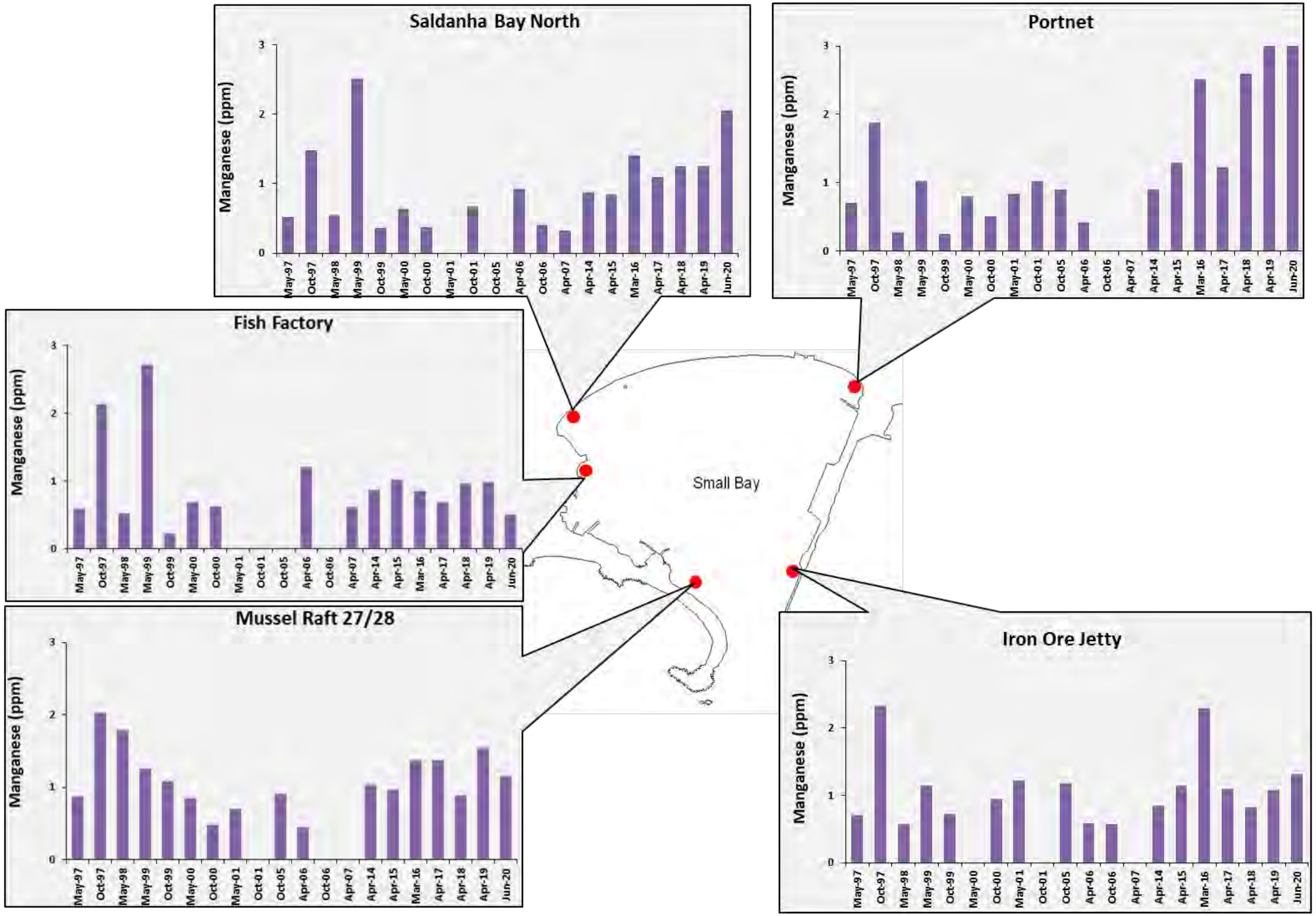


Figure 6.35. Manganese concentrations in mussels (wet weight) collected from five sites in Saldanha Bay from 1997-2007 as part of the Mussel Watch Programme (Source: G. Kiviets, Department of Environmental Affairs) and by Anchor from 2014 to 2020. No limits are specified for manganese in seafood.

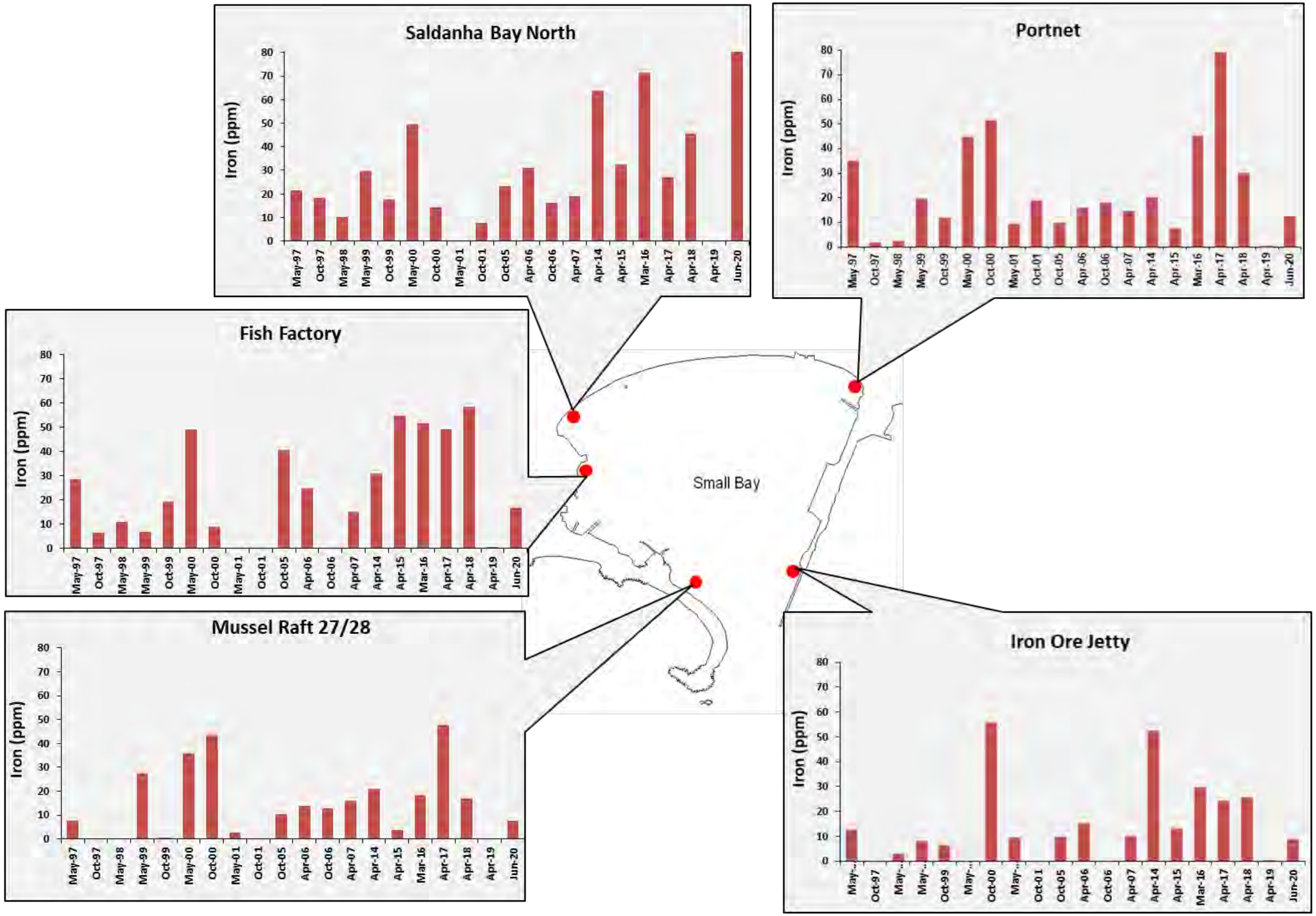


Figure 6.36. **Iron concentrations in mussels (wet weight) collected from five sites in Saldanha Bay from 1997-2007 as part of the Mussel Watch Programme (Source: G. Kiviets, Department of Environmental Affairs) and by Anchor from 2014 to 2020. No limits are specified for iron in seafood.**

6.10.2 Mariculture bivalve monitoring

A combined 884 ha of sea space are currently available for aquaculture production in Saldanha Bay, of which approximately 318.2 ha have been leased to 28 individual mariculture operators for mussels, oysters, finfish and algae (see Chapter 3 for the layout of concession areas). Rights holders engaged in bivalve culture of mussels and oysters in South Africa are required to report on trace metal concentrations and bacterial indicators in harvested organisms on an annual basis. Data were obtained for four trace metal indicators (lead, cadmium, mercury and arsenic) from aquaculture farms in Saldanha Bay Data for mussels for the period 2009 to 2020 are shown on Figure 6.37, while Figure 6.38 shows data for oysters for the period 2005 to 2020. Gaps in the data exist depending on the frequency of monitoring and the year each company was founded. For comparative purposes, independent research data from the Mussel Watch Programme (1997-2007) and SOB monitoring (2014-2020) and from research surveys (Bezuidenhout *et al*. 2015, Pavlov *et al*. 2015) are also displayed on the graphs. Data were also included from an oyster monitoring programme initiated by Transnet Port Terminals (TPT) in June 2018 and continued until June 2020, during which oyster baskets were deployed at channel marker buoys adjacent to the ore jetty and multipurpose quay. Oysters samples were harvested and stocked from these baskets at three-month intervals and analysed for trace metals. Triangles represent data recorded from aquaculture farms, whereas circles represent data recorded during research studies. Research samples were collected from a variety of locations including the shore, port (oil jetty, multipurpose quay, channel markers), and mariculture infrastructure (mussel rafts, oyster longlines).

6.10.2.1 Trace metals in mussels farmed in Saldanha Bay

Bezuidenhout *et al*. (2015) sampled the flesh of mussels in Saldanha Bay and Langebaan Lagoon on six occasions between March 2014 and March 2015. Distinct seasonal patterns were observed, with mussels accumulating higher metal concentrations in winter than in summer. This seasonal pattern has been reported in other studies and is thought to be associated with seasonal reproductive patterns, metabolic rate fluctuation and food type and availability associated with upwelling (increased bioavailability of cadmium is correlated with increased dissolved organic carbon which is elevated when diatom blooms decay) (Sparks *et al*. 2018). Bezuidenhout *et al.* (2015) suggested that the observed seasonal dynamics of trace metal concentrations could be a result of the spawning that takes place in summer; with the subsequent large release of gametes (Van Erkom Schurink & Griffiths 1991); effectively eliminating any trace metal accumulation in the gametes from the mussel's bodies. Iron and zinc were the most prevalent trace metals in mussel tissue. Wild mussels typically had higher concentrations of arsenic, iron, mercury and zinc than those that were farmed. Cadmium concentrations, in farmed mussels were also lower than wild mussels in 2014 samples from Small Bay, however, the opposite was true (higher in farmed mussels) in recent samples collected from Outer Bay North and in Small Bay (Figure 6.37). Concentrations of magnesium and lead were especially high close to the iron ore jetty where ores are loaded onto vessels in the Port (Bezuidenhout *et al*. 2015, Pavlov *et al*. 2015). This concurs with the results of the Mussel Watch and ongoing SOB monitoring reported above (see Section 6.10.1).

Prior to 2000, concentrations of lead in farmed mussels was generally above regulatory limits with especially high levels reported in 1988 when levels ranged between 4-14 ppm (Anchor 2016). From

2000 onwards, lead concentrations were mostly within the regulatory limit (i.e. less than 0.5 ppm); although mussels from some farms continued to exceed this limit on occasion. Lead concentrations in farmed mussels from Small Bay have not exceeded guideline limits in the last three years, with the reported concentration typically much lower than that measured in research samples collected from the nearshore (the "mussel watch" sample results described above). Both research and farm data do show lower lead concentration in mussel tissue samples collected from Outer Bay and North Bay than in mussel samples from Small Bay (Figure 6.31). The lower lead concentrations in mussels collected by researchers from Danger Bay when compared to the higher concentrations in Small Bay, does indicate higher lead pollution within Small Bay, particularly in nearshore environments that are not well flushed.

Data received from mussel farms showed that cadmium concentrations in Small Bay only exceeded the prescribed limit of 2 ppm once in 2015 and once in 2020 (Figure 6.37). Similarly, cadmium concentrations in mussel tissue collected from a Small bay farm on 10 occasions between March 2015 and February 2017 ranged between 0.57-1.4 ppm, remaining below the limit in all samples (Firth et al 2019). Mussels collected by researchers including DAFF and Anchor, from both the shore in Small Bay and off Mussel Raft 27/28 however, had concentrations that frequently exceeded this limit (Figure 6.37). This is confirmed by analyses run on mussels collected in 2014 and 2015 by Bezuidenhout *et al*. (2015). In recent samples collected since 2018, cadmium concentrations regularly exceeded the limit at aquaculture farms in Outer Bay North and in one sample each from Big Bay and Small Bay. Reasons for this discrepancy are still to be determined, although as described above, high levels exceeding prescribed limits have previously been recorded in research samples from Small Bay. Cadmium naturally occurs in high concentrations within the sediments of near-shore upwelling environments such as the southern Benguela (Griffiths *et al.* 2004, Summers 2012). High levels of cadmium within the mussels in previous studies have been attributed to disturbances such as dredging, causing trace metals buried in sediment to become re-suspended in the water. The link between cadmium concentration in mussel tissue and dissolved organic carbon associated with decaying diatom blooms described by Sparks *et al.* (2018), may also play a role in these periodic peaks in cadmium in mussel tissue collected from different localities.

Mercury concentrations submitted to DAFF have largely been within the regulatory limit of less than 0.5 ppm, apart from one elevated value in 2009. Since 2009, no exceedance has been recorded and all samples collected contained less than 0.02 ppm of mercury (Figure 6.37). Research samples have also all been below the prescribed limit, but as with the other trace metals, mercury concentrations have generally been higher than farm samples. Mussel samples were analysed for arsenic for the first time in 2012. Scant data exist for 2012 and 2013 and arsenic was dropped from the suite of aquaculture farm measurements in September 2013. All of the aquaculture farms assessed over this period met the regulatory requirements (<3 ppm), and mussel tissue collected at all sites sampled for research since 2013 have not exceeded the limit (Figure 6.37). Overall, data from the mussel farms discussed above indicates that trace metal contamination in the deeper parts of Saldanha Bay, where the aquaculture farms are located, is in most cases lower than in the nearshore coastal waters. Mussels are filter feeders which extract particulate matter out of the water column for food; thus, it is expected that organisms filtering clean water advected into the Bay from offshore will accumulate fewer toxins than mussels filtering potentially contaminated water close to the shore. The reasons for the lower concentrations of trace metals in farmed mussels compared with those on the shore may also be linked to higher growth rates experienced by the farmed mussels due the fact that the mussels are submerged and hence can feed 24 hours per day on the farms, whilst those inhabiting intertidal areas are exposed to air and other stressors for part of the tidal cycle. The availability of phytoplankton in deeper areas of the Bay may also facilitate faster growth rates. Faster growth results in less time for the accumulation of toxins within the mussel tissue over the lifetime of the animal.

6.10.2.2 Trace metals in oysters farmed in Saldanha Bay

Lead concentration in farmed oyster tissue from both Small Bay and Big bay occasionally exceeded the guideline value of 0.5 ppm, most recently in 2015 (Figure 6.38). Research samples collected as part of the Anchor Oyster Monitoring Programme over the period 2018 to 2020, from locations much closer to ore loading facilities than the mariculture farms, also largely show compliance with guideline levels (97%) with only two samples from Small Bay exceeding the limit (Figure 6.38). A large number of the samples collected prior to 2017 would not have met the revised 2018 guideline for cadmium (2 ppm), although nearly all did meet the previous 3 ppm guideline. Farm and research oyster samples collected from 2017 onwards have nearly all met the revised cadmium guideline value of 2 ppm, with just two samples exceeding the limit (Figure 6.38). Cadmium concentration in all 65 research samples collected from Small Bay over the period 2018 to 2020 fell below the 2 ppm guideline, whilst one of the eight research samples from Big Bay exceeded the guideline. Mercury concentrations in farm and research samples have largely been within the regulatory limit of less than 0.5 ppm, apart from two samples collected in 2007 and 2011 (Figure 6.38). Samples were analysed for arsenic for the first time in 2012. Arsenic concentration in farmed oyster tissue exceeded the regulatory requirements (<3 ppm) on three occasions between 2012 and 2013, whilst reported values since this time have met the guideline (Figure 6.38). All 72 samples analysed as part of the Anchor Oyster Monitoring Programme over the period June 2018- June 2020 fell below the regulatory limit for arsenic (Figure 6.38).

In general, trace metal concentrations in farmed oyster samples have largely met the regulatory limits for the four trace metals tested, with high levels of compliance in samples collected since 2016. This is also the case with samples collected as part of the TPT Oyster Monitoring Programme, with the exception of two samples where lead concentration and one where the cadmium exceeded the limit. Oysters in farmed in Saldanaha Bay accumulate trace metals in their tissues at lower levels than mussels, but where occasional exceedance is observed, it is for the same two trace metals, namely lead and cadmium, that are most problematic in mussels.

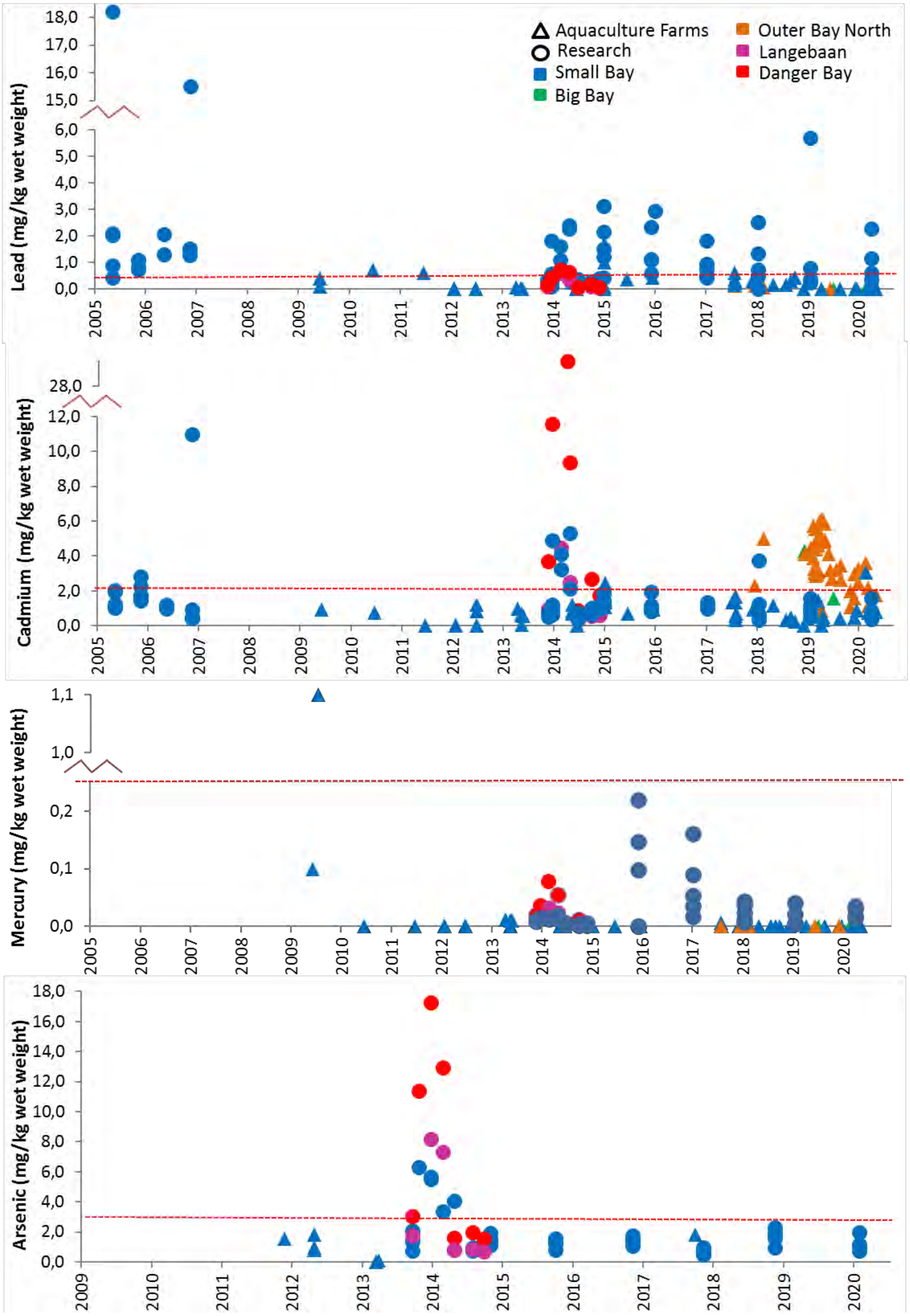


Figure 6.37. **Trace metal concentrations (wet weight) in mussel tissue provided by aquaculture facilities (triangles) and samples collected by researchers, primarily from the shore (circles).**

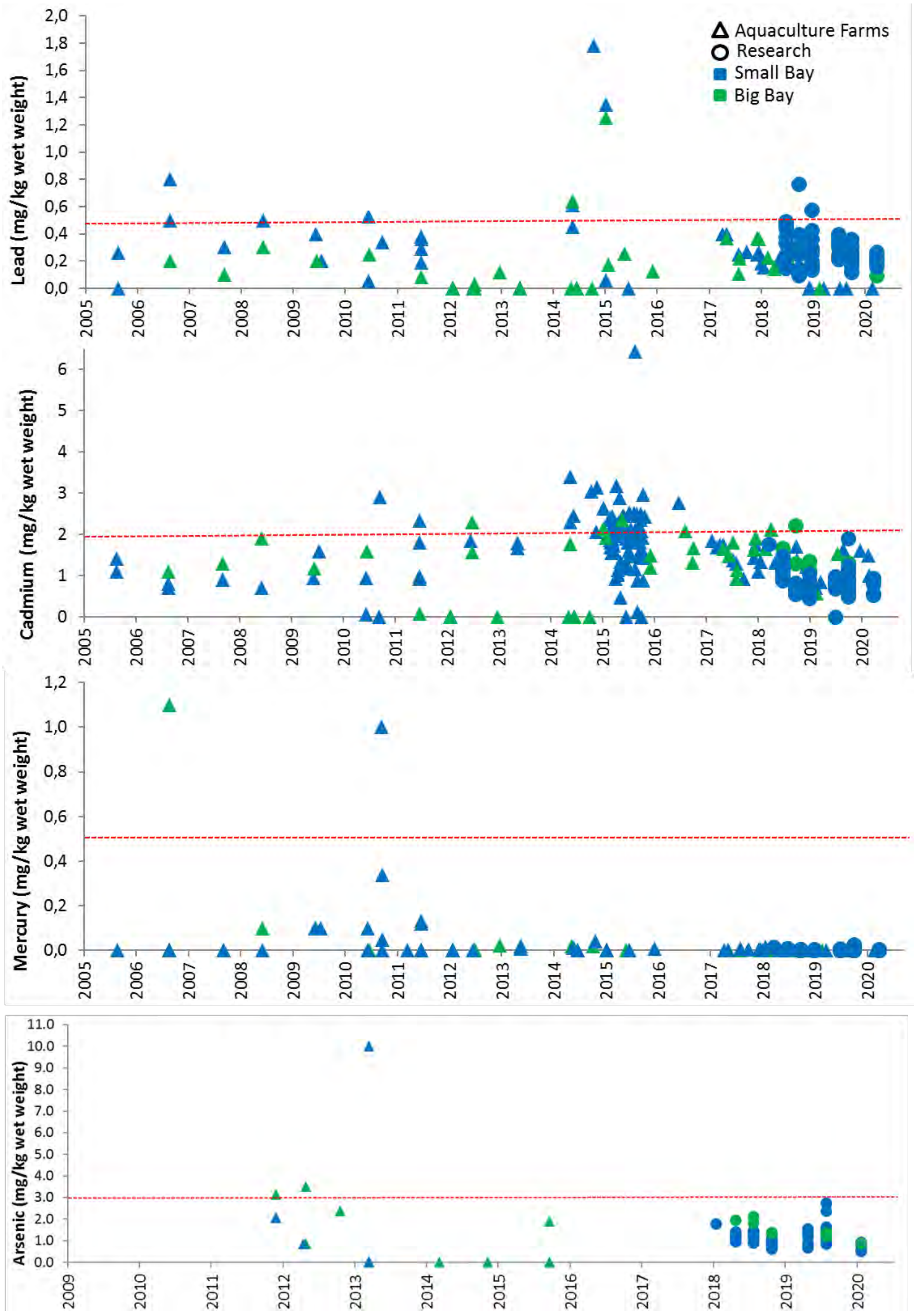


Figure 6.38. Trace metal concentrations (wet weight) in oyster tissue provided by aquaculture facilities and the Anchor Oyster Monitoring Programme (indicated by triangles and circles respectively).

6.11 Summary of water quality in Saldanha Bay and Langebaan Lagoon

There are no clear long-term trends evident in the water temperature, salinity and dissolved oxygen data series that solely indicate anthropogenic causes. In the absence of actual discharges of industrially heated sea water into Saldanha Bay, water temperature is unlikely to show any change that is discernible from that imposed by natural variability or long-term warming or cooling due to climate change (notoriously difficult to differentiate from natural variability). What may, however, be detected is an increase in frequency of “uncommon events” e.g. thermocline breakdown with cool water throughout the water column in summer, as observed in 2018. There is unfortunately limited pre-development data (pre-1975) against which to benchmark the prevailing oceanographic conditions. Although it is likely that construction of the causeway and ore/oil jetty has impeded water flow, increased residence time, increased water temperature, decreased salinity and decreased oxygen concentration (particularly in Small Bay); there is little data to support this. Decreased dissolved oxygen in Small Bay after the harbour development is perhaps the clearest signal. The increase in the frequency of Small Bay hypoxic events occurred after the major harbour development in the 1970s, and the situation does not appear to have changed much since with similar data collected by continuous dissolved oxygen measurements around the turn of the century to those collected during the autumn-winter period in 2020. Natural, regional oceanographic processes (wind driven upwelling or downwelling and extensive coast to bay exchange), rather than internal, anthropogenic causes, appear to remain the major factors affecting physical water characteristics in Saldanha Bay. New data show that hypoxic and near anoxic conditions in the lower part of the water column are frequent occurrences during summer-autumn season in Big and Small Bay (pointing to an external upwelled source of low oxygen water); whilst in Small Bay anthropogenic organic loading, appears to exacerbate the situation with decreased dissolved oxygen measured at sites under mariculture farms than at control sites. The construction of physical barriers (the iron ore/oil jetty and the Marcus Island causeway) has changed current strengths and circulation within Small Bay, resulting in increased residence time (decreased flushing rate), enhanced clockwise circulation and enhanced boundary flows. There has also been an increase in sheltered and semi-sheltered wave exposure zones in both Small and Big Bay subsequent to harbour development.

The microbial monitoring program provides evidence that while chronic problems with faecal coliform pollution were present in the early parts of the record; conditions have improved considerably since this time with the remaining areas of concern in the region of the Hoedtjies Bay Hotel and Pepper Bay. The remaining 18 monitoring stations in the Bay and Lagoon are rated as having “Fair” (5 stations), “Good” (2 stations) or “Excellent” (13 stations) water quality based on the 2019-2020 data. The two beach sites in the vicinity of the Bok River Mouth are rated as “Fair” and “Good” representing a sustained improvement over most earlier samples collected at these sites. This likely reflects improved treatment at the wastewater treatment works after upgrades in 2018 and high levels of wastewater reuse for industrial and irrigation purposes. It is a concerning that faecal coliform levels at the Hoedjiesbaai Beach remain elevated with “poor” water quality recorded at this site for the last three years; as is the decline in water quality at the Pepper Bay-Big Quay station over the 2019-20 period. Local authorities are advised to try determining the sources of this pollution. Faecal coliform counts at all four sites in Big Bay were within the 80th percentile limits for mariculture in 2020. In Small Bay however, the 80th percentile values for mariculture were still exceeded at most sites along the northern shore of Small Bay. Given the current importance and likely future growth of both the mariculture and tourism industries within Saldanha Bay, it is imperative that whatever efforts have been taken in recent years (e.g. upgrading and reuse of sewage and storm water facilities to keep pace with development and population growth) to combat pollution by harmful microbes, (for which *E. coli* and faecal coliforms are indicators), in Small Bay should continue to be implemented. Continued monitoring of bacterial indicators (intestinal *Enterococci* in particular), to assess the effectiveness of adopted measures, is also required and should be undertaken at all sites on a bimonthly basis.

Data supplied by the Mussel Watch Programme (DEA), data collected as part of the State of the Bay Monitoring Programme, and recent research suggests that concentrations of trace metals are elevated at sites along the shore within Small Bay and are frequently above published guidelines for foodstuffs(particularly for lead at the Portnet and Saldanha Bay-North sites). In comparison, data collected by mariculture operators in Saldanha Bay show that concentrations in deeper water are lower and are mostly within food safety guideline limits (with nearly all samples collected from farmed mussel and oyster tissue in Big Bay and Small Bay since 2016 meet the limits). Cadmium concentration in farmed mussels from Outer Bay North, however, exceeded the guidelines in 2018, 2019 & 2020 samples. Exceedance of food safety limits for lead in mussels collected for research from the shore and the aquaculture farms in Small Bay, however, points to the need for management interventions, as metal contamination poses a serious risk to the health of people harvesting mussels.

7 SEDIMENTS

7.1 Background

Sediment quality is a measure of the extent to which the nature of benthic sediments (particle size composition, organic content and contaminant concentrations) has been altered from its natural state. This is important as it influences the types and numbers of organisms inhabiting the sediments and is in turn, strongly affected by the extent of water movement (wave action and current speeds), mechanical disturbance (e.g. dredging) and quality of the overlying water. Sediment parameters respond quickly to changes in the environment but are able to integrate changes over short periods of time (weeks to months) and are thus good indicators for short to very short-term changes in environmental health.

Coastal erosion is one of the main contributing factors that largely influences shoreline stability (Lopez *et al.* 2017). It is often gradual, but occasionally, rapid removal of sediments from the shoreline. Coastal erosion can be caused by natural processes such as storm surges, extreme seasonal and tide changes, sediment morphology, as well as via anthropogenic factors in the form of harbour construction (i.e. dredging activities) (Lopez *et al.* 2017; Woodworth *et al.* 2019). Coastal erosion effectively reshapes the shoreline and directly impacts the fauna/flora inhabiting these areas and can potentially threaten coastal property. Coastal erosion is a major problem in Saldanha Bay, affecting beach mostly in Big Bay and at the entrance to Langebaan Lagoon.

The particle size composition of the sediments is strongly influenced by wave energy and circulation patterns in the Bay. Coarser or heavier sand and gravel particles are typically found in areas with high wave energy and strong currents as the movement of water in these areas suspends fine particles (mud and silt) and flushes these out of these areas. Disturbances to the wave action and current patterns, which reduce the movement of water, can result in the deposition of mud in areas where sediments were previously much coarser. The quantity and distribution of different sediment grain particle sizes (gravel, sand and mud) influences the status of biological communities and the extent of organic and contaminant loading that may occur.

Organic matter (TOC/TON) is one of the most universal pollutants affecting marine life and it can lead to significant changes in community composition and abundance, particularly in semi-enclosed or closed bays where water circulation is restricted, such as Saldanha Bay. High organic loading typically leads to eutrophication, which can lead to a range of different community responses amongst the benthic macrofauna. These include increased growth rates, disappearance of species due to anoxia, changes in community composition and reduction in the number of species following repeat hypoxia and even complete disappearance of benthic organisms in severely eutrophic and anoxic sediments (Warwick 1993).

Trace metals occur naturally in the marine environment and some are important in fulfilling key physiological roles. Disturbance to the natural environment by either anthropogenic or natural factors can lead to an increase in metal concentrations occurring in the environment. Contaminants are predominantly associated with fine sediment particles (mud and silt) as fine-grained particles have a relatively larger surface area for pollutants to adsorb and bind to. Higher proportions of mud, relative to sand or gravel, can thus lead to high trace metal contamination. Disturbance to the sediment (e.g.

dredging) can lead to re-suspension of the mud component from underlying sediments, along with the associated organic pollutants and metals.

An increase in metal concentrations above natural levels, or at least above established safety thresholds, can result in negative impacts on marine organisms, especially filter feeders like mussels that tend to accumulate metals in their flesh. High concentrations of metals can also render these species unsuitable for human consumption.

7.2 Shoreline stability

Beach erosion in Saldanha Bay, particularly at Langebaan Beach, has been the subject of concern for several decades. Erosion of the beaches just outside Langebaan Lagoon has been reportedly going on since the 1960s, with the loss of over 100 m of beach in some areas and up to 40 m of shoreline lost in places in just the last 5 years (McClarty *et al.* 2006, Gericke 2008). This issue has been addressed in some detail in previous versions of the State of the Bay report (see for example Anchor Environmental Consultants 2010, 2011 and 2013b).

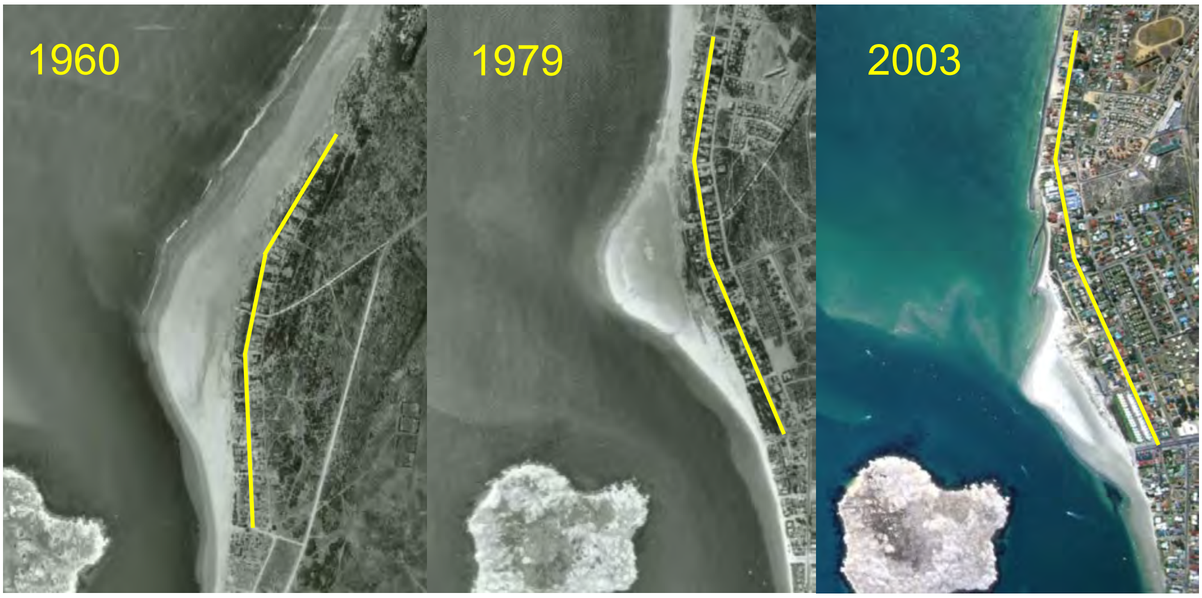


Figure 7.1. Shoreline erosion in Saldanha Bay at the entrance to Langebaan Lagoon. Source: McClarty *et al. (*2006)

The causes of this erosion is not clear but most experts feel that it is likely due to natural causes that may have been exacerbated as a result of the construction of the iron ore terminal and associated infrastructure (Marcus Island causeway) (McClarty *et al.* 2006, Flemming 2016). A recent report by Flemming (2016) identified dredging operations conducted during the Port construction programme as being a possible contributor to these problems (i.e. erosion of Langebaan Beach, Figure 7.2). Flemming (2016) highlighted the fact that much of the sediment used to build the causeway to Marcus Island was dredged from the historic ebb tide delta that existed at the mouth of Langebaan Lagoon (an area where sediment derived from the Lagoon had been deposited over many thousands of years, Figure 7.3, Figure 7.4). Removal of sediment from this area reduced the extent of the outwards refraction of incoming waves thereby increasing the wave energy density along the shoreline by around 50% (Figure 7.5), thereby contributing to erosion of the shoreline. Flemming (2016) has suggested that the most effective way to remedy this situation would be to refill the hole created by the dredging and subsequently nourish the beach with sand from another source.

Figure 7.2. Position of the original shoreline at Langebaan Beach in 1975 (Source: Flemming 2016).

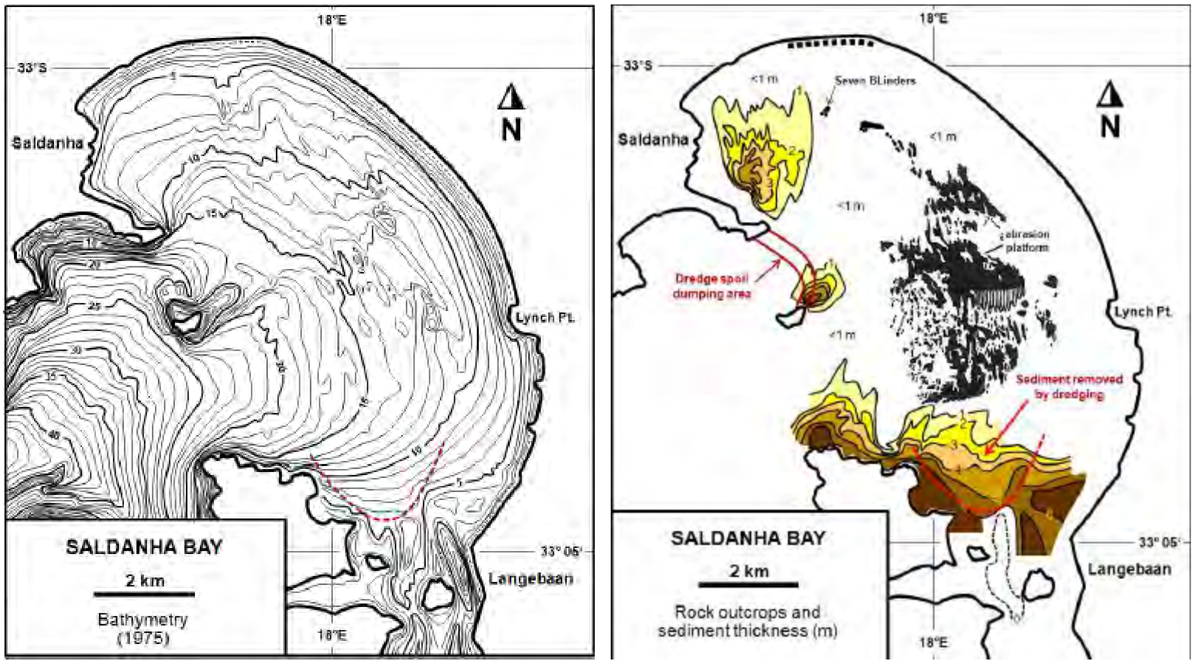
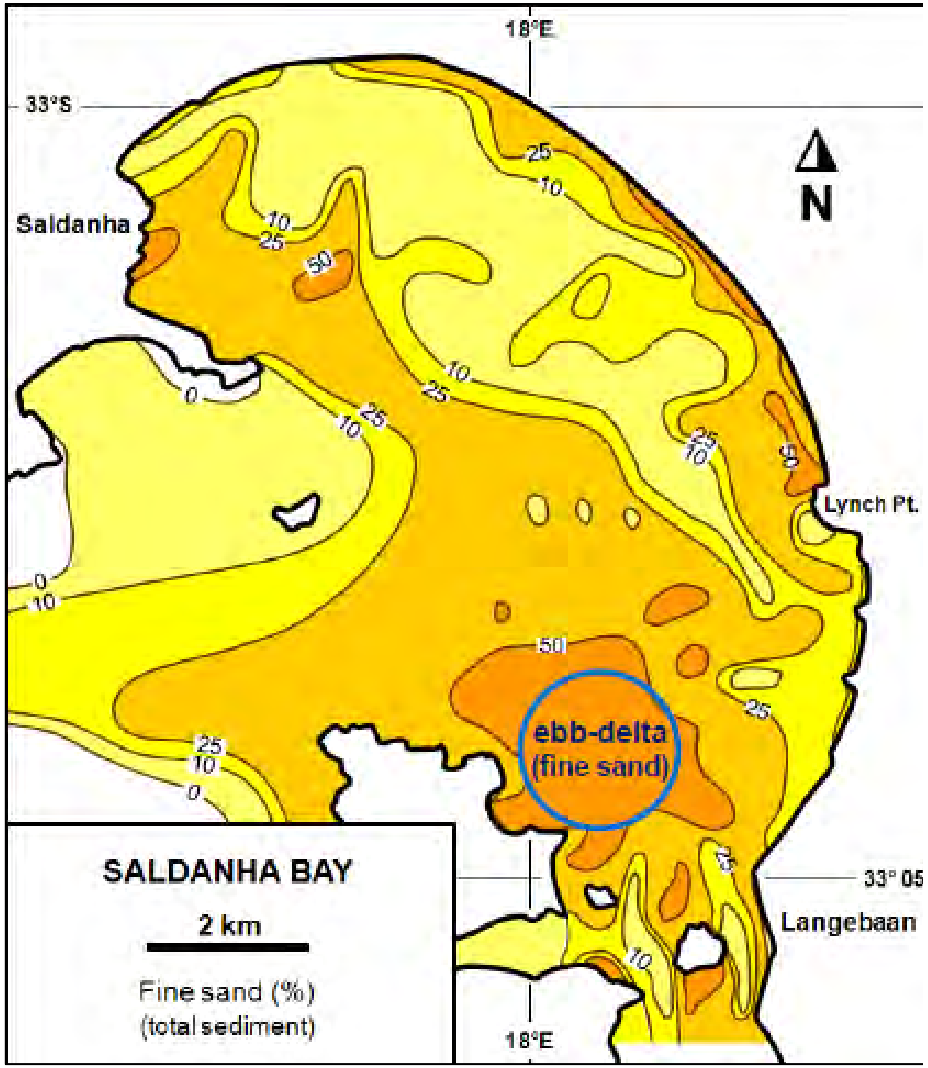


Figure 7.3. Ebb tide delta at the entrance to Langebaan Lagoon (left) where sediment was dredged for construction of the causeway between Marcus Island and the mainland in the late 1970s (right). Source: Flemming (2016).



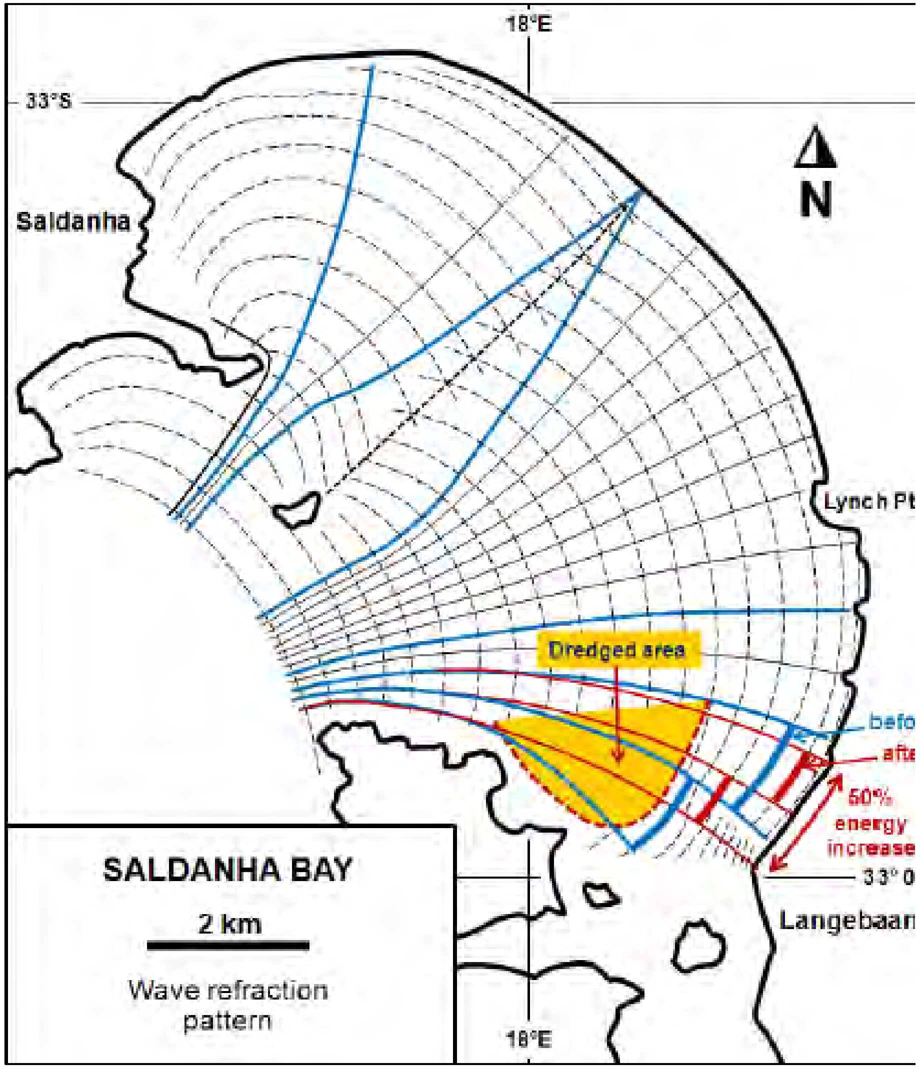


Figure 7.4. **Ebb tide delta at the entrance to Langebaan Lagoon where sediment was dredged for construction of the causeway between Marcus Island and the mainland in the late 1970s. Source: Flemming (2016).**

Figure 7.5. **Changes in wave refraction patterns and a consequent increase in wave energy density at the shoreline at Langebaan Beach - a result of sediment removal during the construction of the causeway linking Marcus Island with the mainland. Source: Flemming (2016).**

A number of interventions have been introduced over the years in an effort to control the erosion, and to limit the loss of sediment from these beaches, precipitated in part by extensive damage caused by a severe storm in June 1997 (Figure 7.1). This includes including construction of rock revetments along Langebaan Beach (1997-2002), construction of groynes extending perpendicularly out from the shore at Langebaan Beach (2004-2008) and construction of gabion walls on Paradise Beach (Figure 7.7).

Figure 7.6. Damage to houses in Langebaan caused by a severe storm in June 1997.

The Saldanha Bay Municipality also initiated an erosion monitoring programme in 1994, designed to monitor change (erosion/accretion) in the beaches between Leentjiesklip 1 (Strandloper restaurant) and Alabama street (Figure 7.8). This entails undertaking beach surveys bi-annually - at the end of summer (Apr/May) and the end of winter (Oct/Nov) during spring low tide. Measurements are taken between the high-water mark and approximately two meters below mean sea-level across 24 transects within the study area (Figure 7.8). Wave height and period are also being measured at the entrance of Saldanha Bay throughout the year, and measurements are analysed in relation to observed shoreline erosion (SBWQFT 2019). The Municipality of Saldanha Bay aborted the original monitoring programme at the end of 2017. In May 2019, the Saldanha Bay Water Quality Forum Trust (SBWQFT) restarted the monitoring programme and has produced two reports thus far, the first report presenting the survey results from November 2017 to October 2018 and the second report covering the period November 2018 to April 2019.

Data from this monitoring programme is presented on Figure 7.9 and Figure 7.10. It is clear from these data that the seasonal patterns of erosion and accretion are complex and are to some extent reversed for the northern and southern portions of Langebaan Beach. For most of the monitoring period (1994-2020), Langebaan North Beach (the section between the Strandloper restaurant and Groyne 1, Figure 7.8), generally eroded in winter and accreted in summer (top graph on Figure 7.9), with some reversal evident in the middle part of the record. The opposite is true for Langebaan South Beach, which typically eroded in summer and accreted in winter (bottom graph in Figure 7.9), again with some evidence of reversal in the middle part of the record. It is likely that this seasonal reversal in these erosion and accretion patterns is linked to the seasonal reversal of the wave climate experienced at these two sites. Wave energy at Langebaan North Beach is typically much more intense in winter (waves striking the shore here approach from offshore and are generated by storms passing the Cape in winter) while wave energy at Langebaan South Beach is more intense in summer (and is derived from the southerly winds blowing across the Lagoon at this time of year).

Figure 7.7. Langebaan Beach as it is at present (2020), with groynes extending out perpendicularly from the shore on the lower (southern) end of the image and rock revetment extending all along the shore from the uppermost groyne towards the upper end of the image.

Figure 7.8. **Erosion monitoring sampling sites in Langebaan between Leentjiesklip 1 (Strandloper restaurant) and Alabama street (SBWQFT 2019).**

Also very clear in the long-term data, is the impact of the various interventions that were introduced to mitigate or control erosion on these beaches. Construction of the shoreline revetment between 1997 and 2002 had an immediate positive impact on Langebaan North Beach in 1997, inasmuch as erosion during the winter of this year was essentially reduced to zero (Figure 7.9). However, levels of erosion increased dramatically again after this, suggesting that the problem had merely been shifted to another section of the beach and had not been eliminated. The effect at Langebaan South Beach was reversed, with no accretion being experienced in the winter of 1997 but some accretion evident in the summer of that year and again the following year, before patterns reverted back to normal.

Construction of the two groynes on the beach had a much more dramatic and longer lasting impact. The shoreline at Langebaan North Beach accreted in both winter and summer during the period from when construction of Groyne 1 started (2004) right through to the period when construction on Groyne 2 was completed in 2007. This is not surprising as beach nourishment (accomplished through dredging of sediment from the channel further offshore and depositing it on the beach) was undertaken through this entire period. After complete of the groynes in 2007, patterns of erosion and accretion on Langebaan North Beach settled back to their familiar pattern of erosion in winter and accretion in summer, but the overall magnitude of change from one year to the next was considerably reduced. This is also clearly evident on the graph showing cumulative change over the entire period (1994-2020, Figure 7.10), which show progressive loss of sand from Langebaan North Beach between 1994 and 2003, followed by a period of accretion between 2004 and 2007, and then a period from 2008 onwards where an equilibrium has been reached and the amount of sand on the beach remains more or less constant from year to year, albeit much reduced relative to the starting point in 1994 (Figure 7.10).

Patterns of accretion and erosion at Langebaan South Beach became a little erratic over the period when the two groynes were being constructed and has never really settled down again after that to any sort of clear seasonal pattern (Figure 7.9). For a period of around 10 years after completion of Groyne 2 (2007-1016), Langebaan South Beach experienced erosion in both summer and winter but since then has become somewhat erratic again. Cumulative change at this site has been very small for the full period (1994-2020), with some evidence of accretion across the first part of the record (1994-2003) prior to the construction of Groyne 1, followed by a period of erosion from this time up to 2015, following which the beach width has remained more or less static at a point corresponding with that observed at the start of monitoring in 1994 (Figure 7.10).

Overall, these patterns of erosion and accretion suggests that the construction of the groynes was a very necessary and successful intervention and that it is very important that monitoring continue in future to confirm that this pattern continues going forwards. Additional interventions to enable reestablishment of beach habitat along the shoreline on the section north of the area currently being monitored is probably also warranted, as the shoreline here is currently made up of a rock revetment which makes access to the sea for people living in this area very difficult and dangerous.

Deployment of gabions in front of the dunes at Paradise Beach (opposite Club Mykonos) has also reportedly been very successful, as is evident in Figure 7.11.

Figure 7.9. Long-term erosion and accretion monitoring of Langebaan Beach between Leentjiesklip 1 (Strandloper restaurant) and Alabama street. Net sand accretion and erosion on Langebaan Beach North (top, North of Groyne 1) and South (bottom, South of Groyne 1) are shown for summer (blue bars), winter (orange bars) and nett change for the year (grey line), between October 1994 and October 2019. Note that no data was collected in summer 2018 (Data Source: SBWQFT 2019).

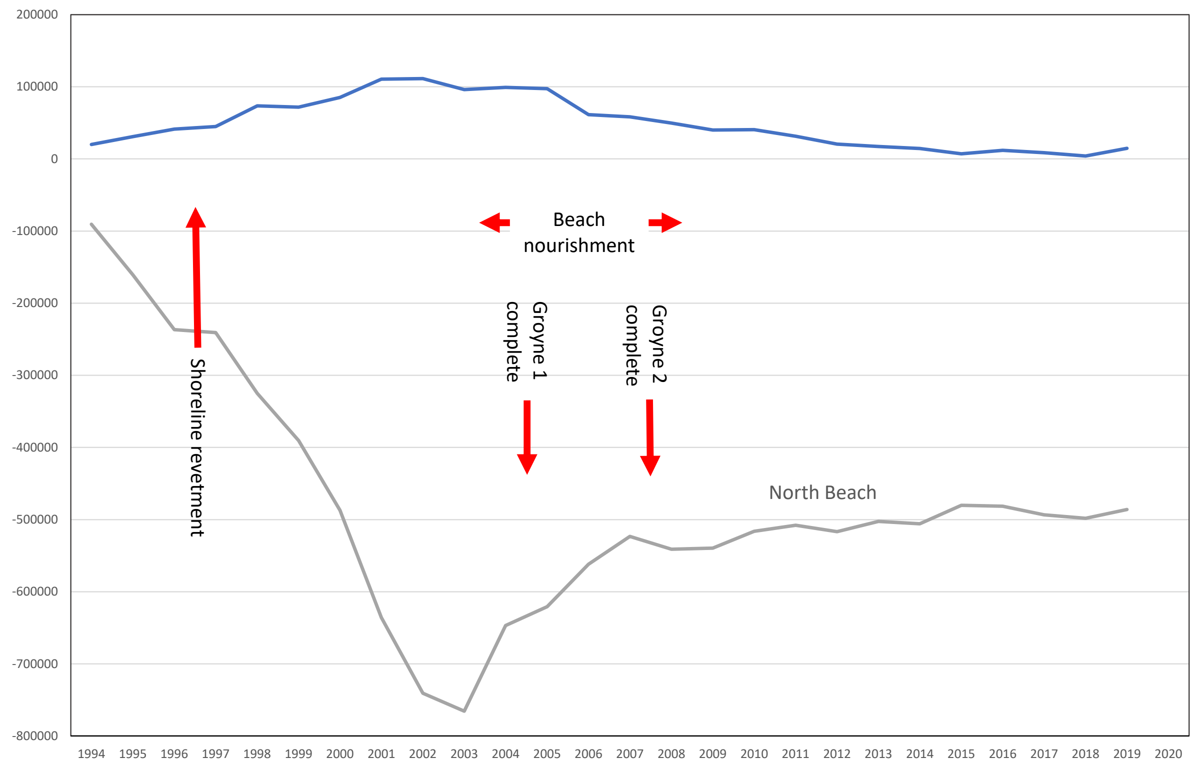


Figure 7.10. **Cumulative change in the amount of sand on Langebaan North and South beaches between 1994 and 2020.**

Figure 7.11. Conditions at Paradise Beach before (top) and after (bottom) deployment of gabions to protect the foredune. Source: Paradise Beach Homeowners Association.

7.3 Sediment particle size composition

The particle size composition of the sediments occurring in Saldanha Bay and Langebaan Lagoon are strongly influenced by wave energy and circulation patterns in the Bay. Coarser or heavier sand and gravel particles are typically found in areas with high wave energy and strong currents as the movement of water in these areas suspends fine particles (mud and silt) and flushes these out of these areas. Disturbances to the wave action and current patterns, which reduce the movement of water, can result in the deposition of mud in areas where sediments were previously much coarser. Since 1975, industrial developments in Saldanha Bay (Marcus Island causeway, iron ore terminal, multi-purpose terminal and establishment of a yacht harbour) have resulted in some changes to the natural patterns of wave action and current circulation prevailing in the Bay. The quantity and distribution of different sediment grain particle sizes (gravel, sand and mud) through Saldanha Bay influences the status of biological communities and the extent of contaminant loading that may occur in Saldanha Bay. The extent to which changes in wave exposure and current patterns has impacted on sediment deposition and consequently on benthic macrofauna (animals living in the sediments), has been an issue of concern for many years.

Contaminants such as metals and organic toxic pollutants are predominantly associated with fine sediment particles (mud and silt). This is because fine grained particles have a relatively larger surface area for pollutants to adsorb and bind to. Higher proportions of mud, relative to sand or gravel, can thus lead to high organic loading and trace metal contamination. It follows then that with a disturbance to natural wave action and current patterns, an increase in the proportion of mud in the sediments of Saldanha Bay, could result in higher organic loading and dangerous levels of metals retention (assuming that these pollutants continue to be introduced to the system). Furthermore, disturbance to the sediment (e.g. dredging) can lead to re-suspension of the mud component from underlying sediments, along with the associated organic pollutants and metals. It may take several months or years following a dredging event before the mud component that has settled on surface layers is scoured out of the Bay by prevailing wave and tidal action. Changes in sediment particle size in Saldanha Bay is therefore of particular interest and are summarised in this section.

A recent study by Henrico & Bezuidenhout (2020) compared a two-time bathymetric data series (before and after the harbour construction in the early seventies) and found that this major construction activity radically changed the hydrodynamic sedimentation process within Saldanha Bay (Figure 7.12). Results of the study revealed that due to the culmination of the harbour construction, the average depth of the Bay had increased by around 0.8 m between 1957 and 1977, the bottom of the Bay became more evenly smooth, and hydrological movement drastically changed as a result of physical structures and bathymetry changes. These apparent alterations have almost certainly influenced wave action and tidal movement in the Bay and by inference, sediment structure in the Bay as well. The "hole" created as a result of dredging sediment from the pro-delta at the entrance to Langebaan Lagoon highlighted by Flemming (2016, Section 7.2) is clearly evident on the baythmetry charts prepared by Henrico & Bezuidenhout (2020, Figure 7.12) and does lend some support to his hypothesis.

The earliest detailed study on the sediments of Saldanha Bay and Langebaan Lagoon was conducted by Flemming (1977a, b) based on a large number of samples (n = ~500) collected from the Bay and Lagoon in 1974, prior to large scale development of the area. He found that sediments in Saldanha

Bay were comprised mostly of fine (0.125-0.25 mm) or very fine sand (0.063-0.125 mm). Significant amount of medium and coarse sand were also present but coarse (0.5-1.0 mm) and very coarse sand (1-2 mm) was rare, as was mud (<0.063 mm) (Figure 7.14).

Sediments in Langebaan Lagoon were comprised mostly of medium, fine and very fine sand, with significant amounts of coarse and very coarse sand near the entrance of the lagoon, but again very low levels of mud (Figure 7.15).

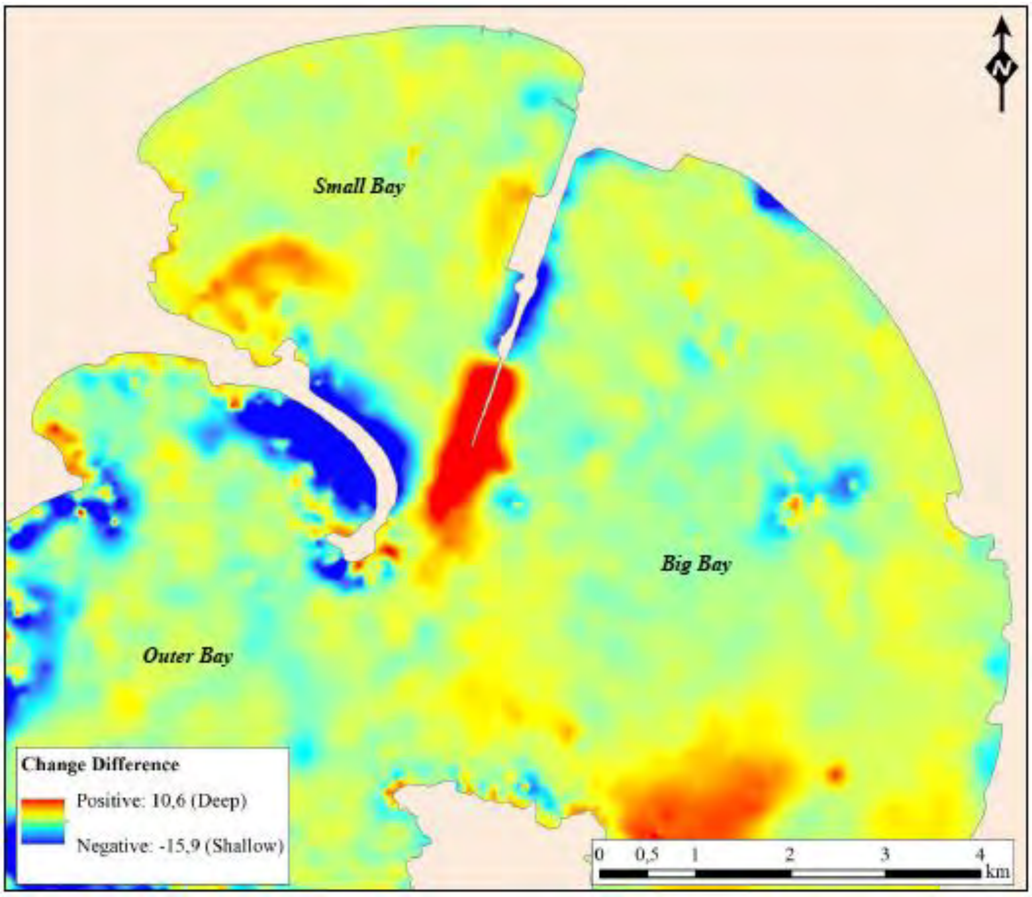


Figure 7.12. Changes in bathymetry in Saldanha Bay from before (1957) to after (1977) harbour construction. Source: Henrico & Bezuidenhout (2020).

Due to concern about deteriorating water quality in Saldanha Bay, sediment samples were collected again in 1989 and 1990 (Jackson & McGibbon 1991). At the time of the Jackson & McGibbon study, the iron ore terminal had been built, dividing the Bay into Small Bay and Big Bay, the multi-purpose terminal had been added to the ore terminal, various holiday complexes had been established on the periphery of the Bay and the mariculture industry had begun farming mussels in the sheltered waters of Small Bay. Sampling was only conducted at a limited number of stations in 1989 and 1990 but results suggested that sediments occurring in both Small Bay and Big Bay were still primarily comprised of sand particles but that mud now made up a noticeable, albeit small, component at most sites (Figure 7.17).

Sampling of sediment in Saldanha Bay as part of the State of the Bay monitoring programme commenced in 1999 (nearly a decade later) and was followed by two further sampling events in 2000 and 2001. However, immediately preceding this (in 1997/98) an extensive area adjacent to the ore terminal was dredged, resulting in a massive disturbance to the sediments of the Bay. Data from the 1999 study, where sampling was conducted in Small and Big Bay (Figure 7.17, Figure 7.19) suggested that there had been a substantial increase in the proportion of mud in sediments in the Bay, specifically at the multi-purpose terminal, the end of the ore terminal, the Yacht Club Basin and in the Mussel Farm area. Two sites least affected by the dredging event were the North Channel site in Small Bay and the site adjacent to the iron ore terminal in Big Bay. The North Channel site is located in shallow water where the influence of strong wave action and current velocities are expected to have facilitated in flushing out the fine sediment particles (mud) that are likely to have arisen from dredging activities. Big Bay remained largely unaffected by the dredging event that occurred in Small Bay and fine sediments appear to be removed to some extent by the scouring action of oceanic waves in this area. Subsequent studies conducted in 2000 and 2001, which were restricted to Small Bay only, indicated that the mud content of the sediment remained high but that there was an unexplained influx of coarse sediment (gravel) in 2000 followed by what appears to be some recovery over the 1999 situation (Figure 7.17).

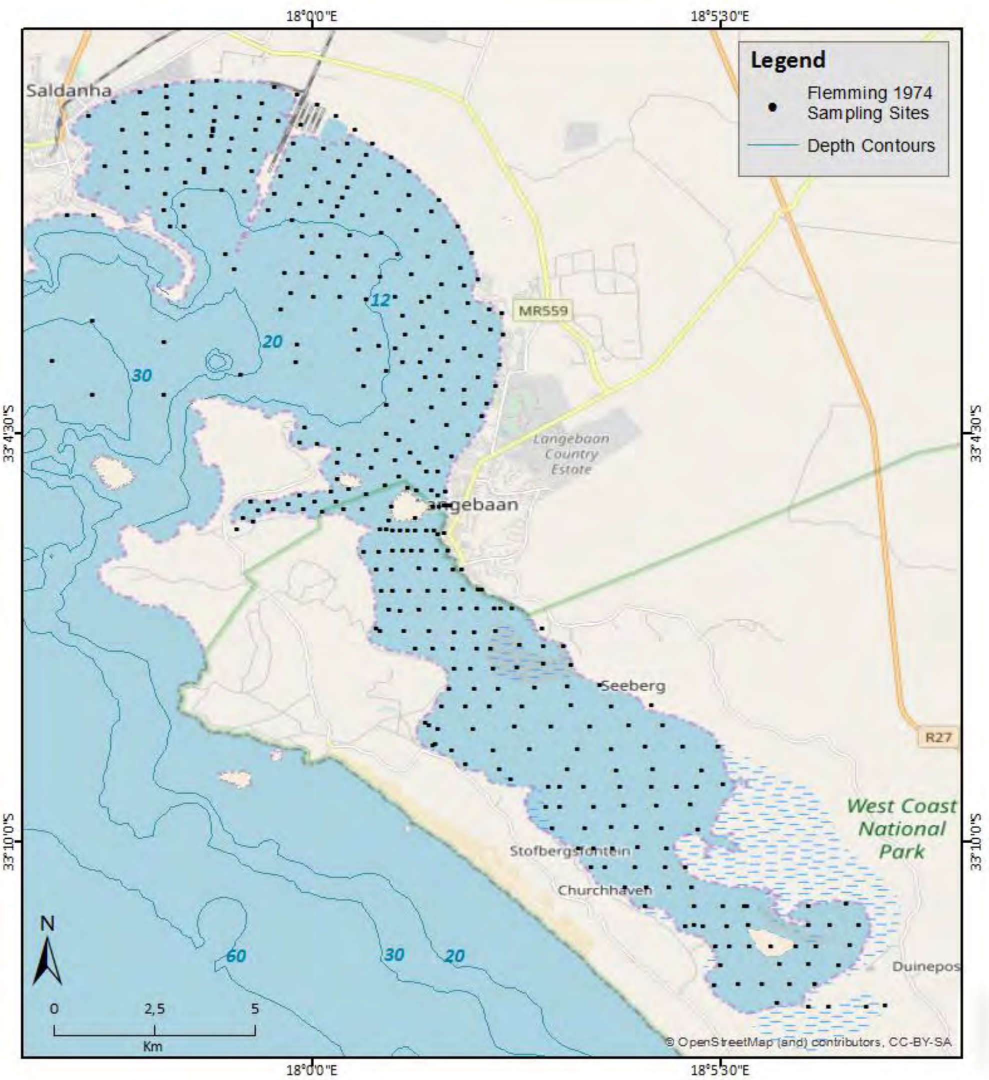


Figure 7.13. Stations sampled by Flemming (1977b) in Saldanha Bay and Langebaan Lagoon in 1974.

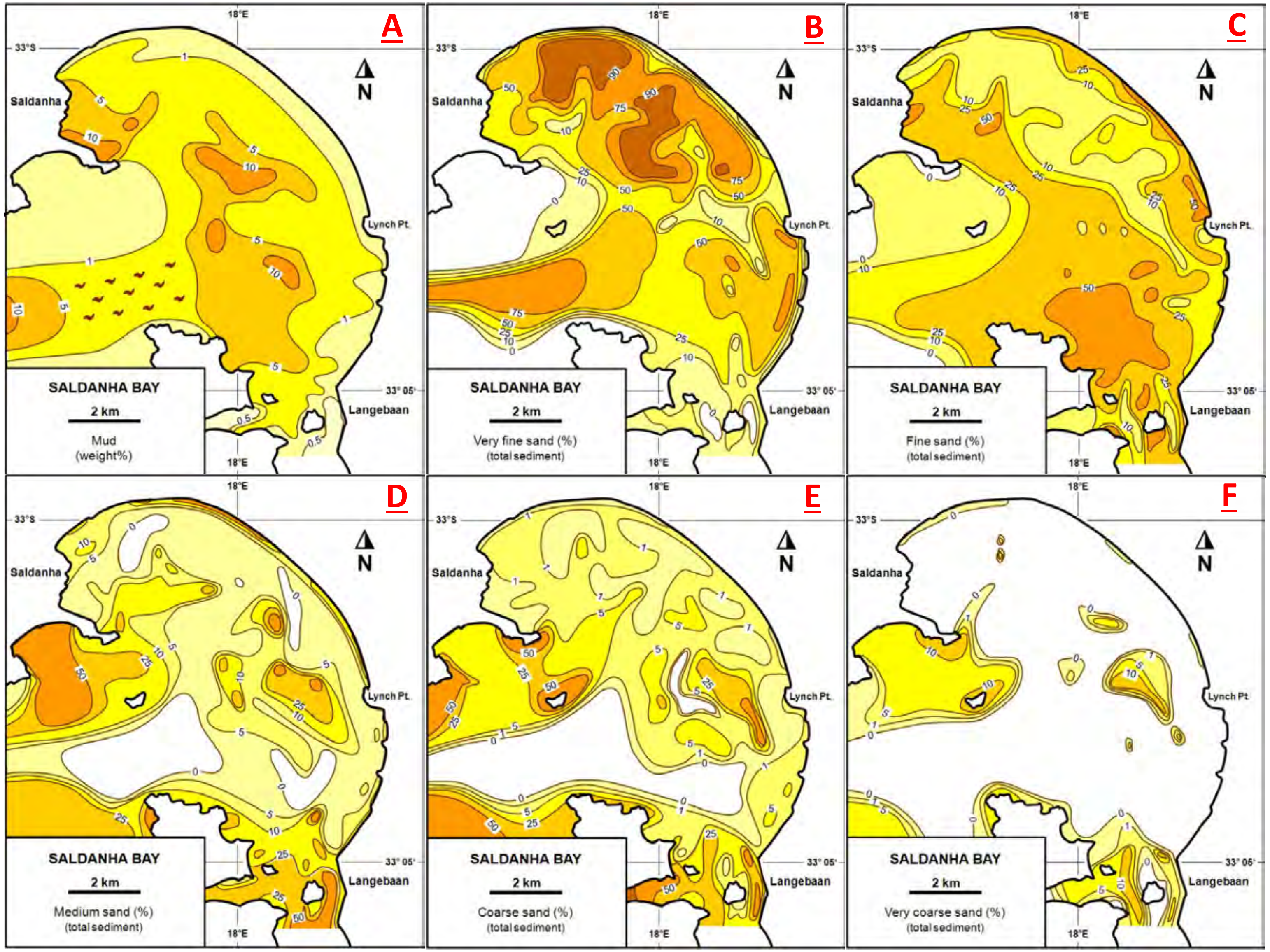


Figure 7.14. Distribution of different sediment types (% of total) in Saldanha Bay in 1975: (A) mud (<0.063 mm), (B) very fine sand (0.063-0.125 mm), (C) fine sand (0.125-0.25 mm), (D) medium sand (0.25-0.5 mm), (E) coarse sand (0.5-1.0 mm), (F) very coarse sand (1-2 mm). Source: Flemming (2015).

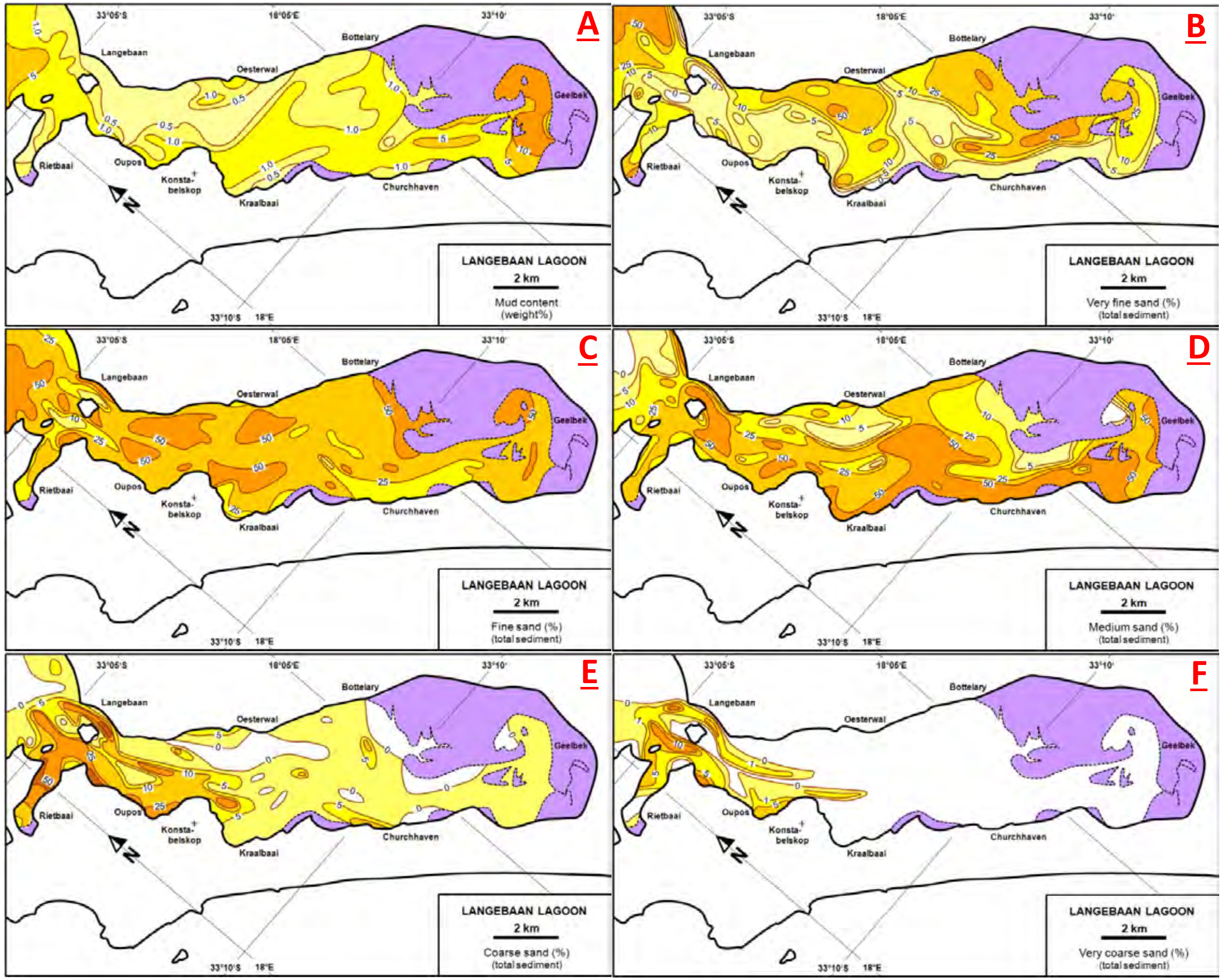


Figure 7.15. Distribution of different sediment types (% of total) in Langebaan Lagoon in 1975: (A) mud (<0.063 mm), (B), very fine sand (0.063-0.125 mm), (C) fine sand (0.125-0.25 mm), (D) medium sand (0.25-0.5 mm), (E) coarse sand (0.5-1.0 mm), (F) very coarse sand (1-2 mm). Source: Flemming (2015).

Sampling as part of the State of the Bay programme was conducted again in 2004 and encompassed the whole of the Bay and Lagoon for the first time since 1974. Data collected as part of this sampling event indicated an almost complete recovery of sediments over the 1999 situation, to a situation where sand (as opposed to mud) made up the bulk of the sediment at most of the six sites assessed in this study (Figure 7.17). The only site where a substantial mud component remained was at the multi-purpose terminal. The shipping channel adjacent to the terminal is the deepest section of Small Bay (artificially maintained to allow passage of vessels) and is expected to concentrate the denser (heavier) mud component of sediment occurring in the Bay.

The next survey, conducted in 2008, revealed that there had been an increase in the percentage of mud at most sites in Small and Big Bay, most notably in the Yacht Club Basin and at the multi-purpose terminal. This was probably due to the maintenance dredging that took place at the Mossgas and multi-purpose terminals at the end of 2007/beginning of 2008. The Yacht Club basin and the Small Bay side of the multi-purpose terminal are sheltered sites with reduced wave energy and are subject to long term deposition of fine-grained particles. The benthic macrofauna surveys conducted between 2008 and 2011 revealed that benthic health at both the Yacht Club basin and adjacent to the multi-purpose terminal was severely compromised, with benthic organisms being virtually absent from the former.

Smaller dredging programmes were also undertaken in the Bay 2009/10, when 7 300 m^3 of material was removed from an area of approximately 3 000 m^2 between Caisson 3 and 4 near the base of the Iron ore terminal on the Saldanha side, and a 275 m^2 area in Salamander Bay was dredged to accommodate an expanded SANDF Boat park. The former programme seems to have had a minimal impact on the Bay while the latter appears to have had a more significant impact and is discussed in detail below.

The percentage mud in sediments declined at most sites in Small Bay over the period 2008 to 2016[10]. This bay-wide progressive reduction in mud content suggested a shift in the balance between the rate at which fine sediments are suspended and deposited and the rate at which currents and wave activities flushed fine sediments from the Bay. This is certainly a positive development as it suggests that sediments in the Bay may be reverting back to a more natural condition where sediments were comprised of mostly sand with a very small mud fraction.

The paucity of data on variations in sediment grain size composition in Langebaan Lagoon do not allow for such a detailed comparison as for the Bay. Available data do suggest, however, that sediments in Langebaan Lagoon have changed little over time and continue to be dominated by medium to fine grained sands with a very small percentage of mud. It is important to note though that the absence of any data between 1974 and 2004 does not allow us to assess what happened during the period between 1999-2001 when levels of mud in sediments in the Bay rose to such critically high levels and may mask a corresponding spike in mud levels in the Lagoon as well.

Sediment samples were collected from a total of 31 sites in Saldanha Bay and Langebaan Lagoon in 2020, as part of the annual State of the Bay sampling programme (Figure 7.16). This included 10 sites

[10] Data for six key sites surrounding the iron ore terminal and in Small Bay are shown on Figure 7.17**Error! Reference source not found.**. T he reader is referred to individual State of the Bay reports for each year for more detail on this.

in Small Bay, 9 in Big Bay, and 12 in Langebaan Lagoon. Samples collected comprised predominantly of sand (particle size ranging between 63 µm and 2000 µm). Sites located in Big Bay had on average the highest proportion of mud (2.01%), followed by Small Bay (1.57%) (Table 7.1). Currently, there is an overall decrease in mud percentage in both Small and Big Bay sites compared to last year. No gravel (particles exceeding 2000 µm) was found across any of the sampling sites in 2019; however, small fractions of gravel were detected in the current survey, more so in Small Bay than in the other areas (Figure 7.17).

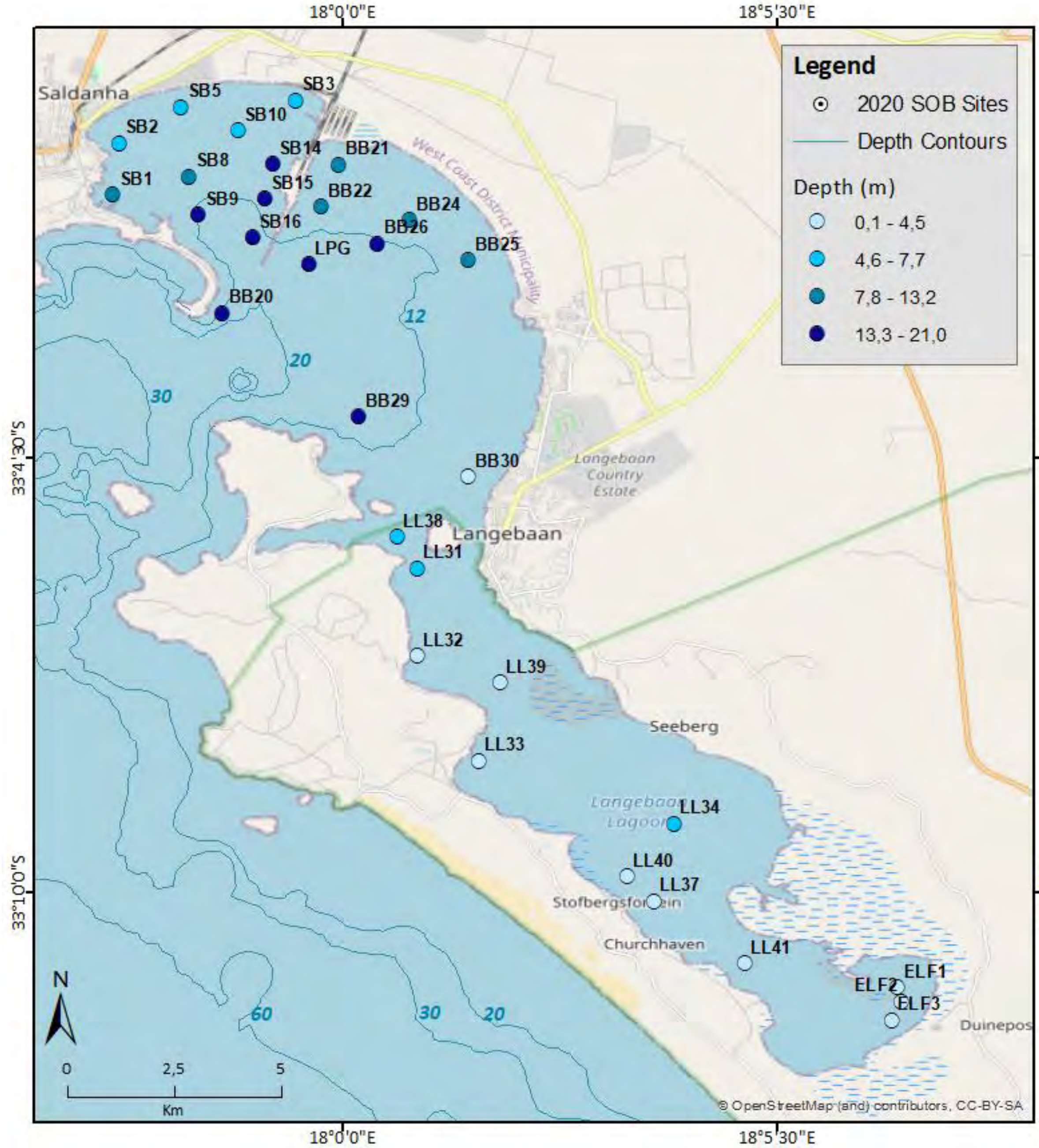


Figure 7.16. **Sediment sampling sites and respective depth ranges (m) in Saldanha Bay, Langebaan Lagoon and Elandsfontein for 2020.**

Mud is the most important particle size component to monitor given that fine grained particles provide a larger surface area to which contaminants bind. The sites beneath the mussel farm the lies adjacent to the causeway linking Marcus Island to the mainland, and in the shipping channels adjacent to the iron ore terminal, are the deepest and are expected to yield sediments with a higher mud fraction than elsewhere in the Bay. Long term sampling confirms these expectations, with the highest proportion of mud recorded in sediments in the vicinity of the iron ore terminal, multi-purpose terminal, the mussel farms and the Yacht Club Basin. The remainder of sites in Big Bay had a relatively moderate to low mud content and Langebaan Lagoon had very low mud content in all recent surveys (Table 7.1).

A 2-way crossed PERMANOVA design was performed using Year (eleven levels: 2009-2011 and 2013-2020) and Region (seven levels: Small Bay, Big Bay, Langebaan Lagoon, Elandsfontein, Sea Harvest, Danger Bay and Liquid Petroleum Gas) as fixed factors. The results confirm that both factors have a significant effect on sediment composition (Year: Pseudo-F_{10} = 10.06, $p < 0.001$; Region: Pseudo-F_6 = 38.41, $p < 0.001$). However, there was no significant interaction between Region and Year (Region × Year: Pseudo-F_{30} = 1.10, $p > 0.05$) which suggests that the extent of the differences in sediment composition does not vary with region from one year to the next and vice versa. The former results are illustrated in Multidimensional Scaling (MDS) plots (Figure 7.18) which depict the similarities/dissimilarities amongst sediment composition in each region for each year. What is striking, though, is that Langebaan Lagoon has consistently remained different in sediment composition (separate grouping) from the rest of the sites from 2009-2020. Sediments in Big Bay and Small Bay are mostly quite similar, but inter-sample variation in Small Bay is clearly much higher than Big Bay or Langebaan Lagoon. Furthermore, there is a clear deviation of the LPG site from its surrounding Big Bay sites (stations 21 and 22) in 2017 as compared to the 2016 survey. This is most likely linked to disturbance (mainly dredging) that occurred near this site at that time. However, sediments at this site have since reverted to a more natural profile in the recent surveys (Figure 7.18). Sampling for the Sea Harvest environmental monitoring programme was also undertaken at the same time as the SOB survey. The first survey conducted as part of this monitoring programme was undertaken in 2017 (Wright *et al.* 2018) and this current 2020 survey was conducted after the installation and use of the "new" discharge site (Wright *et al.* 2020). It is evident from these data that site SH1 was distinctly separated from its own SH grouping as well as the rest of Saldanha Bay regional areas (Figure 7.18) due to it being located in close proximity to the quayside (discharge site) and having left over construction material (stone and gravel) in the sediments.

Table 7.1. **Particle size composition and percentage total organic carbon (TOC) and total organic nitrogen (TON) in surface sediments collected from Small Bay, Big Bay, Langebaan Lagoon and Elandsfontein in 2020 (Particle size analysed by Scientific Services and TOC and TON analysed by the Council for Scientific and Industrial Research).**

	Sample	Gravel (%)	Sand (%)	Mud (%)	TOC (%)	TON (%)	C:N
Small Bay	SB1	0.13	94.73	5.15	3.67	0.42	10.18
	SB2	0.08	99.42	0.50	0.16	0.04	4.78
	SB3	3.08	95.76	1.16	0.38	0.08	5.47
	SB5	0.00	99.75	0.25	0.13	0.03	5.02
	SB8	0.00	99.25	0.75	0.12	0.04	3.56
	SB9	0.34	96.40	3.26	1.26	0.19	7.76
	SB10	0.08	99.75	0.17	0.09	0.02	5.43
	SB14	0.00	99.32	0.68	1.93	0.23	9.77
	SB15	5.20	92.44	2.36	0.99	0.11	10.48
	SB16	0.00	98.53	1.47	0.16	0.03	6.30
	Average	**0.89**	**97.53**	**1.57**	**0.89**	**0.12**	**6.87**
Big Bay	BB20	0.17	99.23	0.60	0.85	0.07	14.22
	BB21	0.00	99.13	0.87	0.16	0.03	6.38
	BB22	0.00	97.75	2.25	0.42	0.06	8.15
	LPG	1.59	96.01	2.40	1.45	0.20	8.47
	BB24	0.00	97.65	2.35	0.23	0.03	8.91
	BB25	0.00	99.22	0.78	0.34	0.03	13.26
	BB26	0.08	94.65	5.27	0.53	0.06	10.38
	BB29	0.08	96.38	3.53	0.74	0.09	9.53
	BB30	0.00	99.96	0.04	0.07	<0.01	-
	Average	**0.21**	**97.78**	**2.01**	**0.53**	**0.07**	**9.91**
Langebaan Lagoon	LL31	0.00	99.96	0.04	0.06	<0.01	-
	LL32	0.33	99.58	0.08	0.05	<0.01	-
	LL33	0.00	99.87	0.13	0.06	<0.01	-
	LL34	0.00	99.67	0.33	0.25	0.01	29.52
	LL37	0.00	99.83	0.17	0.10	<0.01	-
	LL38	3.01	95.65	1.34	0.45	0.06	8.69
	LL39	0.00	100.00	0.00	0.04	<0.01	-
	LL40	0.00	99.92	0.08	0.04	<0.01	-
	LL41	0.00	99.83	0.17	0.07	<0.01	-
	Average	**0.37**	**99.37**	**0.26**	**0.12**	**0.04**	**19.10**
Elandsfontein	Eland 1	0.08	99.33	0.59	0.43	0.06	8.36
	Eland 2	0.00	99.75	0.25	0.09	<0.01	-
	Eland 3	0.00	99.00	1.00	0.26	0.03	10.23
	Average	**0.03**	**99.36**	**0.61**	**0.26**	**0.05**	**9.29**

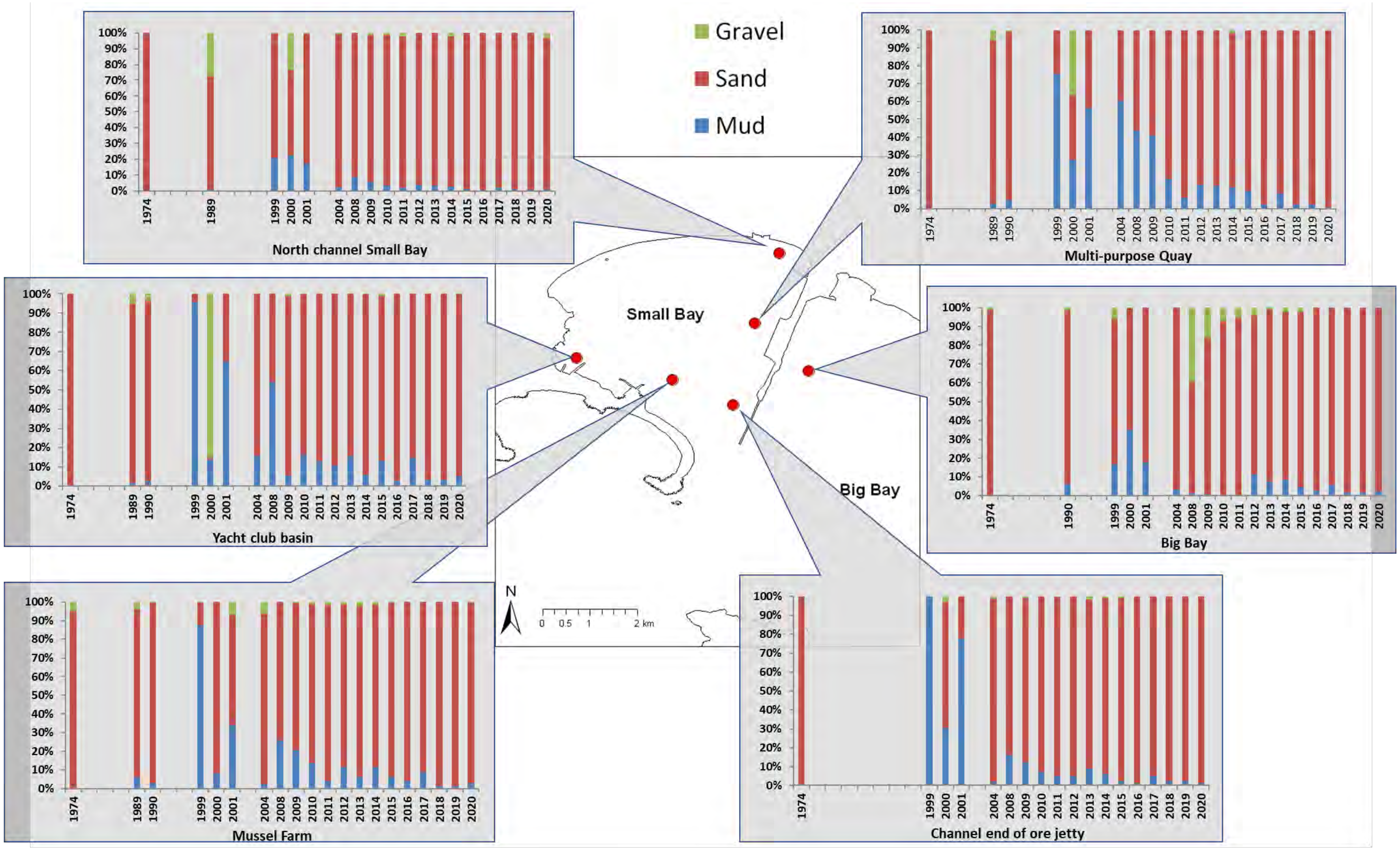


Figure 7.17. Particle size composition (percentage gravel, sand and mud) of sediments at six localities in the Small and Big Bay area of Saldanha Bay between 1974 and 2020. Data sources: 1974: Flemming (1977b). 1899-1990: Jackson & McGibbon (1991). 1999-2018: SBWQFT.

Figure 7.18. MDS plots of particle size distribution (PSD) from samples collected at sites from Saldanha Bay. Langebaan Lagoon and Elandsfontein from 2009-2020. Each region (SB: Small Bay, LL: Langebaan Lagoon, BB: Big Bay, SH: Sea Harvest, LPG: Liquid Petroleum Gas, EL: Elandsfontein and DB: Danger Bay) is represented by a unique symbol and colour.

In summary, the natural, pre-development state of sediment in Saldanha Bay comprised predominantly of sand particles; however, developments and activities in the bay (causeway, ore terminal, Yacht Club Harbour and mussel rafts) reduced the overall wave energy and altered the current circulation patterns. This compromised the capacity of the system to flush the bay of fine particles and led to the progressive accumulation of mud (cohesive sediment) in surface sediments in the Bay which peaked around 2000, and has been followed in more recent times by a reduction in the mud fraction to levels similar to those last seen in 1974. This pattern is very clearly evident in a comparison between the proportions of mud present in sediments in the Bay in 1974, 1999 and 2020 (Figure 7.19).

Dredge events, which re-suspended large amounts of mud from the deeper lying sediments, seem to be a dominant contributor to the elevated mud content in the Bay and results of surveys have shown a general pattern of an increase in mud content following dredge events followed by a recovery in subsequent years. Any future dredging or other such large-scale disturbance to the sediment in Saldanha Bay are likely to result in similar increases in the mud proportion as was evident in 1999, with accompanying increase in metal content.

7.4 Total organic carbon (TOC) and nitrogen (TON)

Total organic carbon (TOC) and total organic nitrogen (TON) accumulates in the same areas as mud as organic particulate matter is of a similar particle size range and density to that of mud particles (size <60 µm) and tends to settle out of the water column together with the mud. Hence, TOC and TON are most likely to accumulate in sheltered areas with low current strengths, where there is limited wave action and hence limited dispersal of organic matter. The accumulation of organic matter in the sediments doesn't necessarily directly impact the environment but the bacterial breakdown of the organic matter can (and often does) lead to hypoxic (low oxygen) or even anoxic (no oxygen) conditions. Under such conditions, anaerobic decomposition prevails, which results in the formation of sulphides such as hydrogen sulphide (H_2S). Sediments high in H_2S concentrations are characteristically black, foul smelling and toxic for living organisms.

The most likely sources of organic matter in Saldanha Bay are from phytoplankton production at sea and the associated detritus that forms from the decay thereof, fish factory waste discharged into the Bay, faecal waste concentrated beneath the mussel and oyster rafts in the Bay, treated sewage effluent discharged into the Bay from the wastewater treatment works (Saldanha & Langebaan) and stormwater. The molar ratios of carbon to nitrogen (C:N ratio) can be useful in determining the sources of organic contamination. Organic matter originating from marine algae typically has a C:N ratio ranging between 6 and 8, whereas matter originating from terrestrial plant sources exceeds this. Fish factory waste is nitrogen-rich and thus extremely low C:N ratios would be expected in the vicinity of a fish waste effluent outfall. However, nitrogen is typically the limiting nutrient for primary productivity in most upwelling systems including the Benguela, and the discharge of nitrogen-rich waste from fish factories has been linked to algal blooms using stable isotope studies (Monteiro *et al.* 1997). The excess nitrogen in the system is taken up by algae thereby allowing for bloom development. By consuming the nitrogen, the bloom effectively increases the C:N ratio. In addition, phytoplankton production and decomposition will then add to the levels of organic matter within the system.

Historical data on organic carbon levels in sediments in Saldanha Bay are available from 1974 (Flemming 1977), 1989 and 1990 (Jackson & McGibbon 1991), 1999, 2000 and 2001 (CSIR 1999a, 2000, 2001) and from 2004 and 2008-2017 from the State of the Bay sampling programme. According to data from Flemming (1977). TOC levels in Saldanha Bay were mostly very low (between 0.2 and 0.5%) throughout the Bay and Lagoon prior to any major development (Figure 7.20 and Figure 7.21).

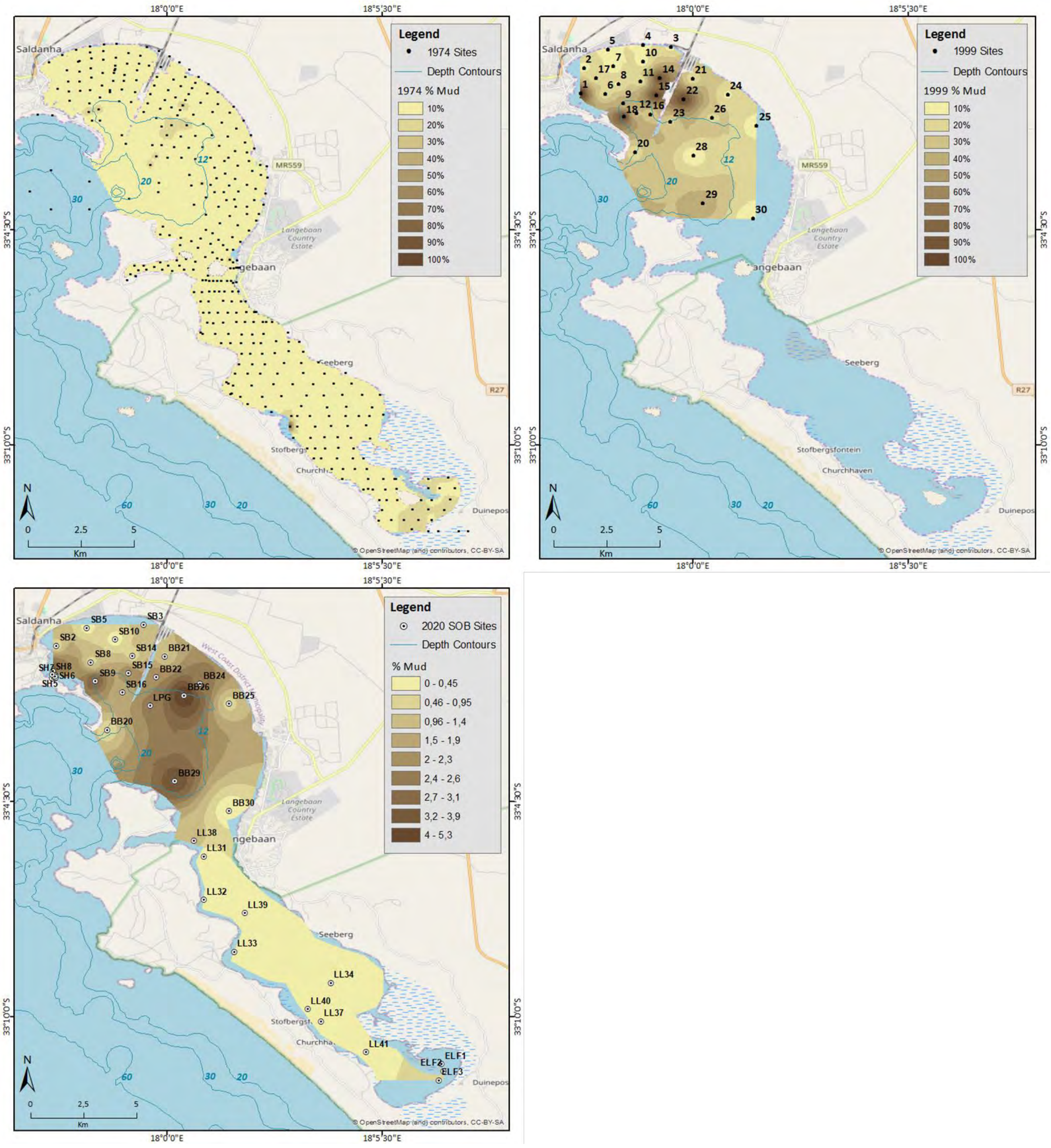


Figure 7.19. **Change in the percentage mud in sediments in Saldanha Bay and Langebaan Lagoon between 1974 (left), 1999 (right) and 2020 (bottom) survey results.**

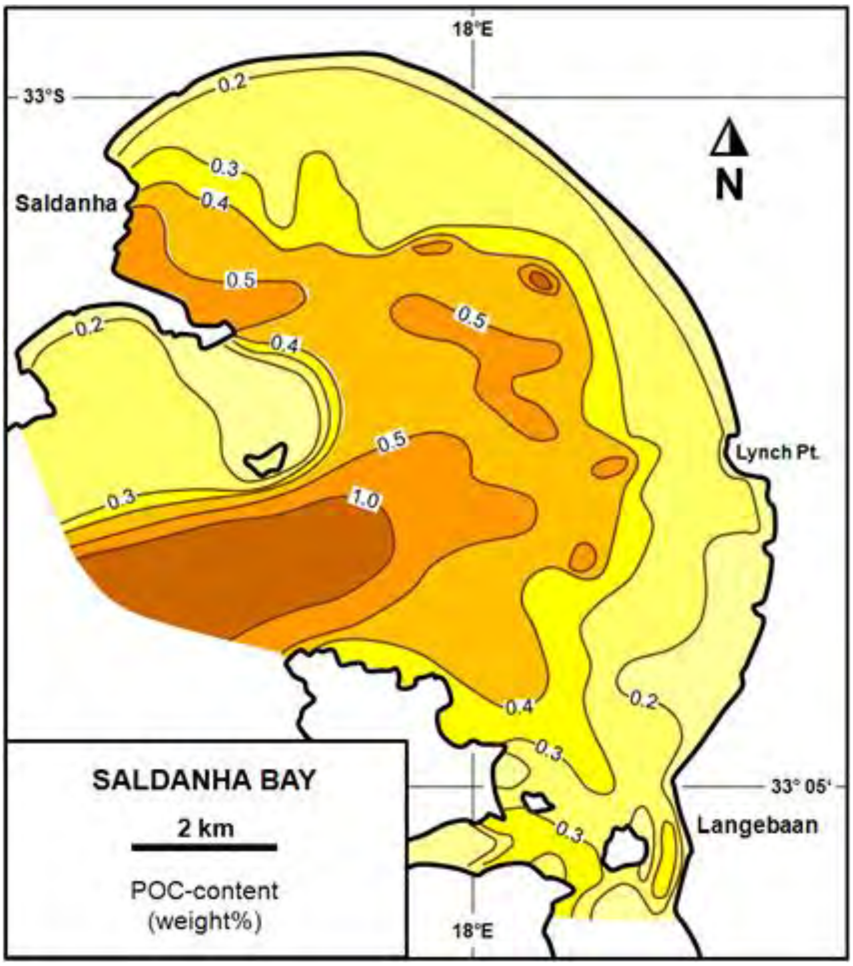


Figure 7.20. **Levels of organic carbon in sediments Saldanha Bay in 1974. Source: Flemming (2015).**

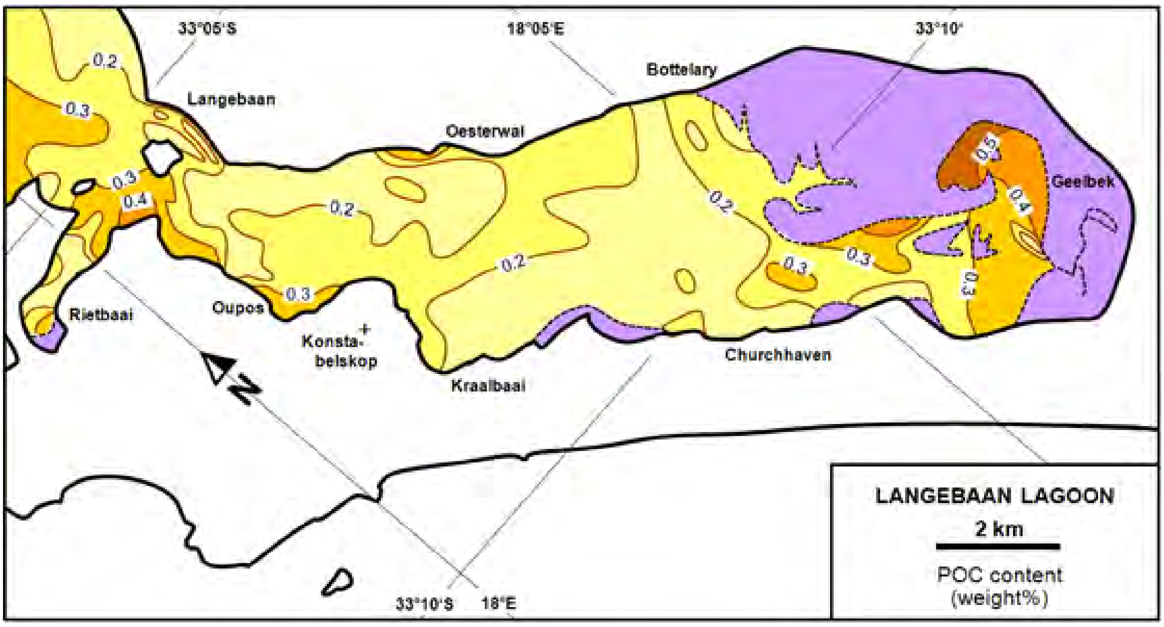


Figure 7.21. **Levels of organic carbon in sediments in Langebaan Lagoon in 1974. Source: Flemming (2015).**

The next available TOC data was collected in 1989 after the construction of the iron ore terminal and the establishment of the mussel farms in Small Bay. At this stage, all key monitoring sites in the vicinity of the iron ore terminal and in Small Bay had considerably elevated levels of TOC with the greatest increase occurring in the vicinity of the Mussel Farm (Figure 7.23). By the time the next surveys had been undertaken in 1999 (CSIR 1999a, Figure 7.22, Figure 7.23) levels of TOC had increased still further at most sites in the Bay. Results from 2000 and 2001, which were restricted to Small Bay, showed a similar pattern (Figure 7.23). Data from subsequent surveys undertaken in 2004 and between 2008 and 2018 are presented in the individual State of the Bay reports and are summarised in Figure 7.23. Data on the spatial distribution of TOC from 1999, 2019 and the most recent survey (2020) are shown in Figure 7.22. These data suggest that high TOC levels recorded in 1999 particularly at the Yacht Club Basin (SB1) and Multi-Purpose Terminal (SB14) have dropped slightly in the 2019 and recent 2020 survey. Although, TON levels have noticeably increased at the latter two sites in Small Bay over the years (present survey) in comparison to 1999 (Figure 7.22).

Levels of Total Organic Nitrogen (TON) in sediments in the Bay were first recorded in 1999 by the CSIR (CSIR 1999a) at the behest of the SBWQFT. Levels of TON in sediments were assessed again in 2000 and 2001 (CSIR 2000, 2001); and have been monitored annually from 2004 onwards as part of the State of the Bay monitoring programme. TON levels in 1999 were low at most sites (≤0.2%) except for those in the Yacht Club Basin and near the mussel rafts in Small Bay (Figure 7.23). Levels were slightly or even considerably elevated at all sites in 2000 and 2001 (Figure 7.23). Sampling conducted in 2004 spanned a large number of sites in Small Bay, Big Bay and Langebaan Lagoon and results indicated that levels remained elevated at sites near the Yacht Club Basin, Mussel Raft and Iron Ore Terminal in Small Bay, near the Iron Ore Terminal and in the deeper depositional areas in Big Bay; but were low elsewhere, especially in the Lagoon (Figure 7.23, see also the 2004 State of the Bay report). Results from the State of the Bay surveys conducted between 2008 and 2020 suggest that levels have dropped off slightly at many of the key sites in Small Bay but have remained more or less steady in other parts of the Bay and in the Lagoon (Figure 7.23). There was a clear increase in TON in 2018 compared to 2017 for Big Bay, but levels dropped again in 2019-2020 (Figure 7.23). Additionally, increased TON levels were observed in 2020 in Small Bay at the North Channel. Similar increases were also evident at the Yacht Club Basin and Mussel farm albeit to a lesser degree. Spatial variation in TON levels recorded in the sediments in Saldanha Bay and Langebaan Lagoon in 1999, 2019 and 2020 are presented in Figure 7.22. Once again, concentrations are generally higher in Small Bay, particularly at the Yacht Club Basin and along the Iron Ore Terminal. However, TON concentrations were similar in both the 2019 and 2020 surveys; mirroring the patterns observed for TOC levels in the Bay (Figure 7.22). Overall, levels of TON at remaining sites in Small Bay remain low relative to levels recorded in 1999 and this is certainly encouraging.

Sources of organic nitrogen in Small Bay include fish factory wastes, biogenic waste from mussel and oyster culture as well as sewage effluent from the wastewater treatment works. Elevated levels of TON in Small Bay are considerably linked to the discharge of waste from the fish processing plants in this area, along with faecal waste accumulating beneath the mussel rafts and dredging operations at the Multi-Purpose Terminal.

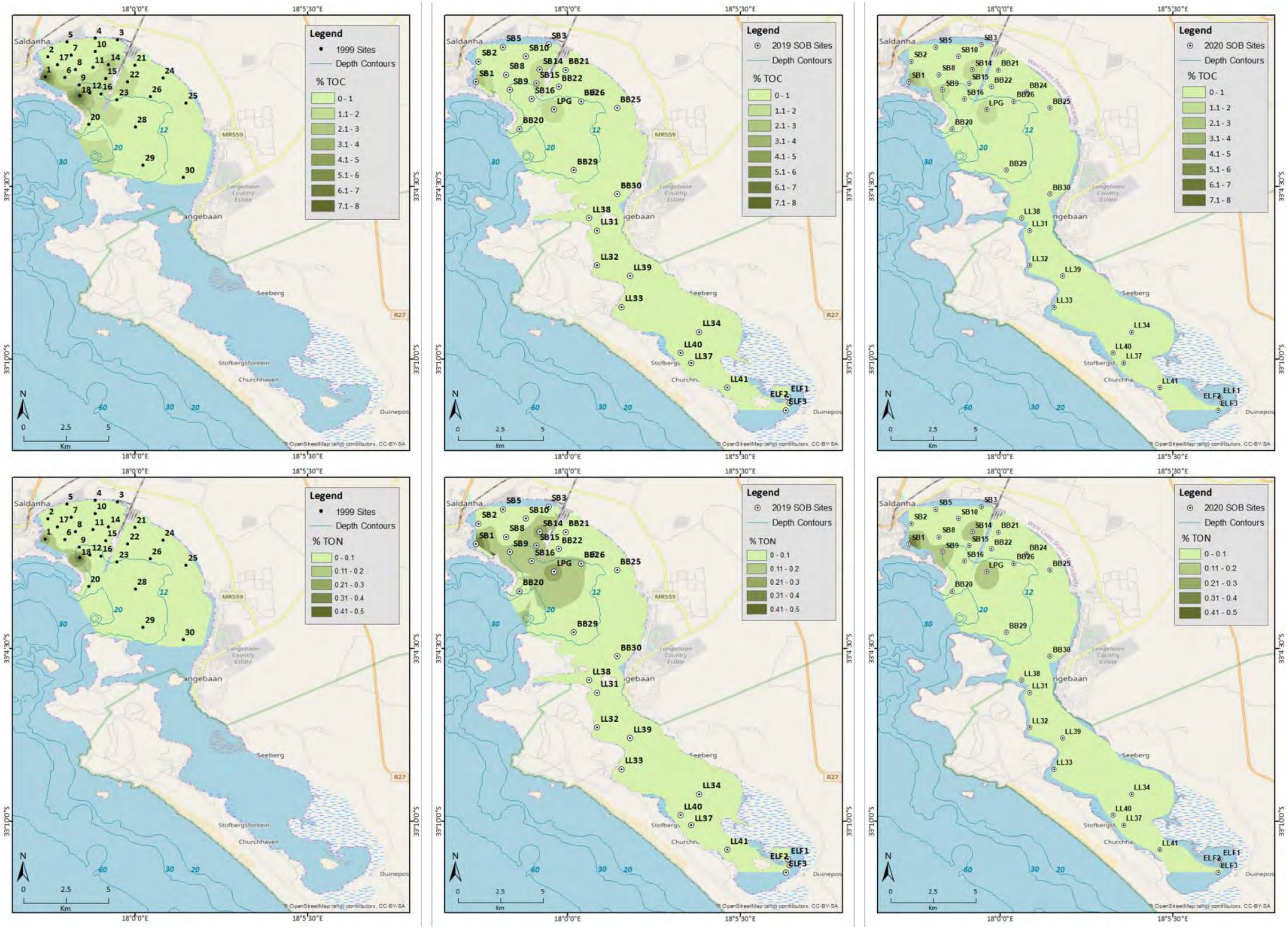


Figure 7.22. **Total organic carbon (TOC) and total organic nitrogen (TON) levels in sediments in Saldanha Bay in 1999 (Source: CSIR 1999a), 2019 and 2020.**

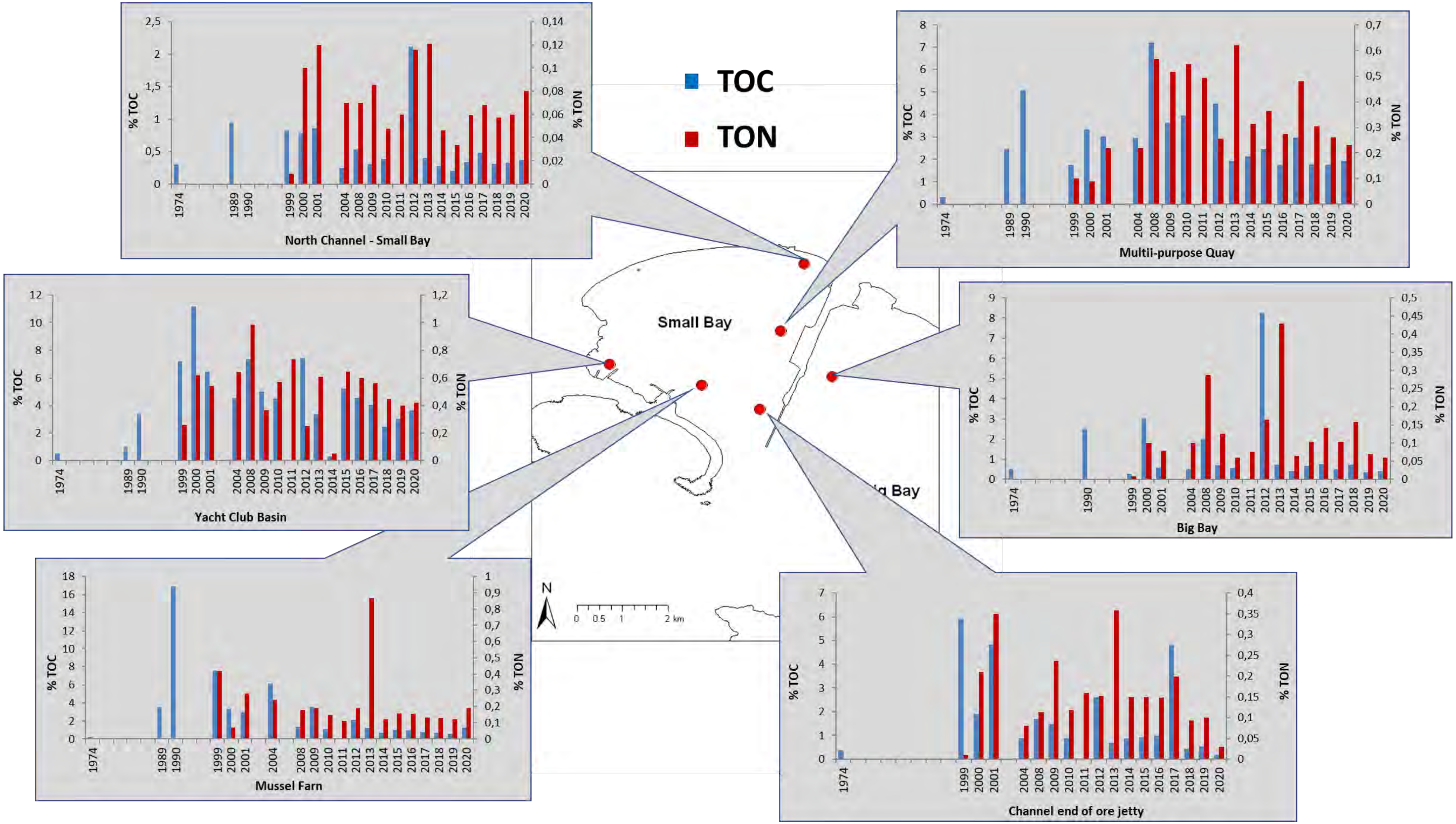


Figure 7.23. Total organic carbon and nitrogen in sediments of Saldanha Bay at six locations between 1974 and 2020. Data sources: 1974: Flemming (1977b), 1899-1990: Jackson & McGibbon (1991), 1999-2018: SBWQFT.

The ratio between TOC and TON in marine sediments is also important and provides an indication of the source of the organic matter present in sediment. The C:N ratio results from 2018-2019 were similar, where majority of sites were within the expected range of marine production. However, the Multi-Purpose Terminal (SB14) in 2018 and the Yacht Club Basin (SB1) in 2019 were above the expected range. It is likely that this is not associated with terrestrial inputs but rather with nitrogen depletion (denitrification) in these areas (Figure 7.24). Whilst majority of the sites within Small Bay as well as Big Bay and Langebaan Lagoon were below the range of marine production during the 2018-2019 survey; the latter pattern did not follow through to the recent 2020 survey (Figure 7.24). It is clearly evident that all of the sites within Big Bay (except BB21), the Yacht Club Basin along with other Small Bay sites surrounding the multipurpose terminal were above the expected C:N range in this year's survey. What is even more striking is that marine production in Langebaan Lagoon has also increased dramatically compared to the two previous surveys; of which one site LL34 is above the range (Figure 7.24).

There are two possible reasons for elevated C:N ratios observed in 2020; the first being that the organic matter found in these areas originated from terrestrial sources. The alternate explanation is that natural decomposition processes reduced the amount of nitrogen present thereby elevating the C:N ratio (see low levels of TON across the Bay in Table 7.1). This process is known as denitrification and it occurs in environments where oxygen levels have been depleted (anoxic or hypoxic) and nitrates are present. Under these conditions, denitrifying bacteria are likely to dominate, as they are able to substitute oxygen which are normally required for organic matter degradation through nitrate reduction (Knowles 1982, Tyrrell & Lucas 2002). In areas where photosynthetic rates are very high, such as in upwelling systems, or where there is a high degree of organic input; a high biological oxygen demand deeper in the water column and sediments can lead to complete oxygen utilisation. Denitrification may be responsible for the elevated C:N ratios in the deeper areas where a high TOC content was recorded, and stratification is possible. It is, however, highly unlikely that this process is responsible for the elevated C:N ratios at Langebaan sites in 2020; given that many of the sites with high C:N ratios are in highly exposed, shallow areas with low organic content. It thus seems likely the organic matter in many areas of the system originates from a terrestrial source. An alternative hypothesis is that enhanced productivity with selectively greater recycling of nitrogen-rich relative to carbon-rich organic matter can lead to elevated C:N ratios (Twichell *et al.* 2002).

The low C:N ratio values recorded in 2018 and 2019 for the northern sites within Small and Big Bay are most likely due to the shallow water and/or high wave action and current patterns experienced at these sites resulting in a considerable amount of organic carbon being flushed out (Atkinson *et al.* 2006). Another alternative explanation for the reduced C:N ratios in 2018-2019 compared to the 2020 survey is related to the low mud content present in the Bay. Previous studies have revealed that organic carbon content in terrestrial soils and marine sediments is often positively correlated with mud content (Baptista *et al.* 2000; Falco *et al.* 2004; Leipe *et al.* 2011; Serrano *et al.* 2016). Progressive reductions in the amount of fine material (mud) in the Bay in recent years may thus account for corresponding reductions in the C: N ratios. The observed temporal variability of C:N ratios in Saldanha Bay may well also reflect changes in upwelling intensity and benthic productivity over the summer period that precedes the annual surveys in April.

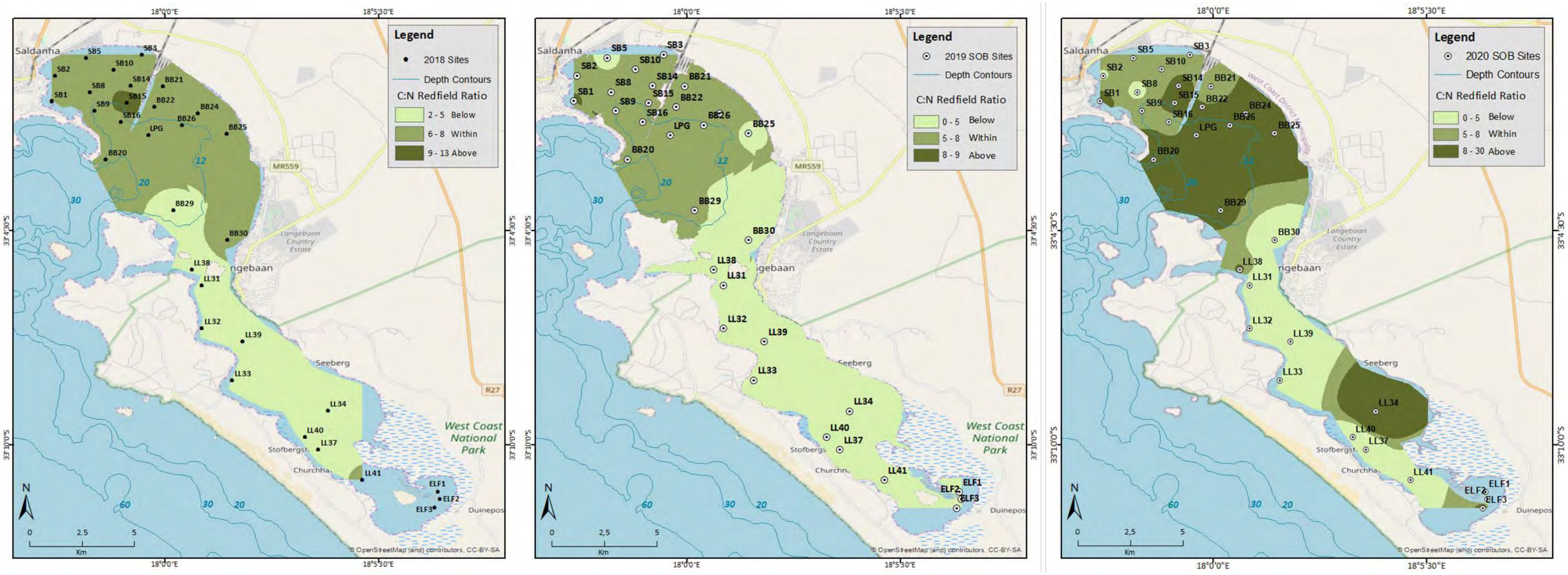


Figure 7.24. C:N ratios at different sites surveyed in Saldanha Bay, Langebaan Lagoon and Elandsfontein in 2018, 2019 & 2020 (dark green = exceeds the range expected for marine production; mild green = within the range expected for marine production and light green = below range expected for marine production).

7.5 Trace metals

Trace metals occur naturally in the marine environment and some are important in fulfilling key physiological roles. Disturbance to the natural environment by either anthropogenic or natural factors can lead to an increase in metal concentrations occurring in the environment, particularly sediments. An increase in metal concentrations above natural levels, or at least above established safety thresholds, can result in negative impacts on marine organisms, especially filter feeders like mussels that tend to accumulate metals in their flesh. High concentrations of metals can also render these species unsuitable for human consumption. Metals are strongly associated with the cohesive fraction of sediment (i.e. the mud component) and with TOC. Metals occurring in sediments are generally inert (non-threatening) when buried in the sediment but can become toxic to the environment when they are converted to the more soluble form of metal sulphides. Metal sulphides are known to form as a result of natural re-suspension of the sediment (strong wave action resulting from storms) and from anthropogenic induced disturbance events like dredging activities.

The Benguela Current Large Marine Ecosystem (BCLME) program reviewed international sediment quality guidelines in order to develop a common set of sediment quality guidelines for the coastal zone of the BCLME (Angola, Namibia and west coast of South Africa) (Table 7.2). The BCLME guidelines cover a broad concentration range and still need to be refined to meet the specific requirements of each country within the BCLME region (CSIR 2006). There are thus no official sediment quality guidelines that have been published for the South African marine environment as yet, and it is necessary to adopt international guidelines when screening sediment metal concentrations. The National Oceanic and Atmospheric Administration (NOAA) have published a series of sediment screening values which cover a broad spectrum of concentrations from toxic to non-toxic levels as shown in Table 7.2.

The Effects Range Low (ERL) represents the concentration at which toxicity may begin to be observed in sensitive species. The ERL is calculated as the lower 10th percentile of sediment concentrations reported in literature that co-occur with any biological effect. The Effects Range Median (ERM) is the median concentration of available toxicity data. It is calculated as the lower 50th percentile of sediment concentrations reported in literature that co-occur with a biological effect (Buchman 1999). The ERL values represent the most conservative screening concentrations for sediment toxicity proposed by the NOAA and ERL values have been used to screen the Saldanha Bay sediments.

Table 7.2. Summary of Benguela Current Large Marine Ecosystem and National Oceanic and Atmospheric Administration metal concentrations in sediment quality guidelines

Metal (mg/kg dry wt.)	BCLME region (South Africa. Namibia. Angola)		NOAA	
	Special care	Prohibited	ERL	ERM
Cd	1.5 – 10	> 10	1.2	9.6
Cu	50 – 500	>500	34.0	270.0
Pb	100 – 500	> 500	46.7	218.0
Ni	50 – 500	> 500	20.9	51.6
Zn	150 – 750	> 750	150.0	410.0
1(CSIR 2006). 2 (Long et al. 1995. Buchman 1999)				

Dramatic increases in trace metal concentrations, especially those of cadmium and lead after the start of the iron ore export from Saldanha Bay, raised concern for the safety and health of marine organisms, specifically those being farmed for human consumption (mussels and oysters). Of particular concern were the concentrations of cadmium which exceeded the lower toxic effect level published by NOAA. Both lead and copper concentrates are exported from Saldanha Bay and it was hypothesised that the overall increase of metal concentrations was directly associated with the export of these metals. The concentrations of twelve different metals have been evaluated on various occasions in Saldanha Bay; however, the overall fluctuations in concentrations are similarly reflected by several key metals throughout the time period. For the purposes of this report, five metals that have the greatest potential impact on the environment were selected from the group. These are cadmium (Cd), copper (Cu), lead (Pb), nickel (Ni) and zinc (Zn).

The earliest data on metal concentrations in Saldanha Bay were collected in 1980, prior to the time at which iron ore concentrate was first exported from the ore terminal. The sites sampled were 2 km north of the multi-purpose terminal (Small Bay) and 3 km south of the multi-purpose terminal (Big Bay) and metals reported on included lead (Pb), cadmium (Cd) and copper (Cu). Concentrations of these metals in 1980 were very low, well below the sediment toxicity thresholds. Subsequent sampling of metals in Saldanha Bay (for which data is available) only took place nearly 20 years later in 1999. During the period between these sampling events, a considerable volume of ore had been exported from the Bay, extensive dredging had been undertaken in the Bay (1997/98) along with the Mussel Farm and the small craft harbour (Yacht Club Basin) being established (1984). As a result of these activities, the concentrations of metals in 1999 were very much higher (up to 60-fold higher) at all stations monitored. This reflects the accumulation of metals in the intervening 20 years, much of which had recently been re-suspended during the dredging event and had settled in the surficial (surface) sediments in the Bay. Concentrations of most metals in Saldanha Bay were considerably lower in the period 2000-2010. This closely mirrors changes in the proportion of mud in the sediments and most likely reflects the removal of fine sediments together with the trace metal contaminants from the Bay, by wave and tidal action. Monitoring surveys between 2001 and 2019 indicates that with a few exceptions, metal concentrations have continued to decline over time which is encouraging.

Sediments were analysed for concentrations of aluminium (Al), iron (Fe), cadmium (Cd), copper (Cu), nickel (Ni), lead (Pb) and manganese (Mn). Metals in the sediments were analysed by Scientific Services using a nitric acid (HNO_3) / perchloric acid ($HClO_3$)/ hydrogen peroxide (H_2O_2)/ microwave digestion and JY Ultima Inductively Coupled Plasma Optical Emission Spectrometer. Trace metal concentrations recorded in the sediments of Saldanha Bay are shown in Table 7.3 and the sections dedicated to each of the trace metals below.

In 2020, only cadmium and copper concentrations were highest and exceeded ERL guidelines in the vicinity of the Yacht Club Basin (Table 7.3). Compared to previous years, cadmium concentrations had noticeably dropped to less than 1 mg/kg at the rest of the monitoring sites; except for site LL 34 in Langebaan Lagoon which in turn exceeded guideline levels. Levels of other trace metals were all lower than in previous years and did not exceed ERL guidelines. Although lead did not exceed ERL guidelines, concentrations were considerably elevated adjacent to the Multi-Purpose Terminal. The latter was also observed at the Yacht Club Basin for manganese. Comparing these results to the ERL guidelines provides a useful indication of areas in the Bay that may be toxic to living organisms. However, this comparison does not provide an indication of whether the build-up of a trace metal is due directly to anthropogenic contamination of the environment with that particular metal or whether it is an indirect result of other environmental influences - for high levels of mud or organic carbon.

Table 7.3. Concentrations (mg/kg) of metals in sediments collected from Saldanha Bay in 2020. Values that exceed sediment quality guidelines are highlighted in red font.

	Sample	Al	Fe	Cd	Cu	Ni	Pb	Mn
***ERL Guideline (mg/kg)**		-	-	1.2	34	20.9	46.7	56.50
Small Bay	SB1	11228	6662.07	2.46	35.26	11.09	20.12	53.74
	SB2	1818	1193.37	<1.0	1.71	8.77	6.41	23.78
	SB3	1778	1636.49	<1.0	2.98	5.90	18.82	38.55
	SB5	1421	825.37	<1.0	<1.0	16.68	3.33	19.53
	SB8	2249	1708.20	<1.0	1.68	4.51	7.61	23.87
	SB9	4826	4253.29	<1.0	3.16	7.41	7.33	41.25
	SB10	1795	1386.38	<1.0	1.25	9.38	7.23	23.95
	SB14	5389	4514.51	<1.0	9.48	8.42	41.81	54.97
	SB15	3242	3085.50	<1.0	4.45	4.78	14.29	50.77
	SB16	3116	2568.08	<1.0	3.07	4.69	5.80	28.43
Big Bay	BB20	1325	1230.73	<1.0	3.08	2.10	1.23	12.91
	BB21	2584	2068.40	<1.0	1.75	3.94	3.22	35.51
	BB22	3113	2858.90	<1.0	4.40	5.48	5.51	45.20
	LPG	4303	3976.64	<1.0	4.57	6.74	4.66	37.39
	BB24	2743	1957.17	<1.0	1.45	3.53	3.03	29.11
	BB25	1370	956.94	<1.0	<1.0	4.40	<1.0	23.77
	BB26	4004	3119.11	<1.0	2.59	5.45	4.10	39.76
	BB29	3123	2073.56	<1.0	3.63	6.03	1.32	22.18
	BB30	777	551.64	<1.0	1.07	5.02	<1.0	12.40
Langebaan Lagoon	LL31	1678	1206.43	<1.0	1.73	4.35	1.65	18.77
	LL32	1849	2009.45	<1.0	<1.0	8.03	2.25	17.37
	LL33	984	551.96	<1.0	<1.0	11.18	<1.0	7.57
	LL34	1800	1076.37	1.24	4.70	9.43	1.81	14.39
	LL37	801	476.83	<1.0	<1.0	9.53	<1.0	7.84
	LL38	4486	3054.82	<1.0	2.19	12.89	2.79	37.80
	LL39	897	716.91	<1.0	<1.0	3.79	<1.0	12.08
	LL40	1184	609.96	<1.0	1.45	12.49	<1.0	17.89
	LL41	1380	797.80	<1.0	<1.0	13.37	<1.0	9.07
Elandsfontein	Eland 1	2735	1956.93	<1.0	4.85	10.30	<1.0	16.26
	Eland 2	2185	1181.39	<1.0	<1.0	10.15	<1.0	14.45
	Eland 3	3944	2253.25	<1.0	2.39	15.41	<1.0	21.93

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The concentrations of metals in sediments are affected by grain size, total organic content and mineralogy. Since these factors vary in the environment, one cannot simply use high absolute concentrations of metals as an indicator for anthropogenic metal contamination. Metal concentrations are therefore commonly normalized to a grain-size parameter or a suitable substitute for grain size; and only then can the correct interpretation of sediment metal concentrations be made (Summers *et al.* 1996a). A variety of sediment parameters can be used to normalize metal concentrations, and these include aluminium (Al), iron (Fe) and total organic carbon. Aluminium or iron are commonly used as normalisers for trace metal content as they ubiquitously coat all sediments and occur in proportion to the surface area of the sediment (Gibbs 1994); they are abundant in the earth's crust and are not likely to have a significant anthropogenic source (Gibbs 1994. Summers *et al.* 1996a); and ratios of metal concentrations to Al or Fe concentrations are relatively constant in the earth's crust (Summers *et al.* 1996a). Normalized metal/aluminium ratios can be used to estimate the extent of metal contamination within the marine environment and to assess whether there has been enrichment of metals from anthropogenic activities. Due to the known anthropogenic input of iron from the iron ore quay and industrial activity in Saldanha Bay; metal concentrations were normalized against (divided by) aluminium and not iron.

Another means of evaluating the extent of contamination of sediments by metals is to calculate the extent to which the sediments have been enriched by such metals since development started. Metal enrichment factors were calculated for cadmium, lead and copper relative to the 1980 sediments (Table 7.4). Unfortunately, historic enrichment factors could not be calculated for nickel and manganese as no data were available for these elements in 1980. Enrichment factors equal to (or less than) 1 indicate no elevation relative to pre-development conditions, while enrichment factors greater than 1 indicate a degree of metal enrichment within the sediments over time. The extent of contamination for cadmium, copper, nickel and lead is discussed below using both metal concentrations and the metal enrichment factors.

7.5.1 Spatial variation in trace metals levels in Saldanha Bay

7.5.1.1 Cadmium

Sediments from sites located alongside the Iron Ore Terminal within Small Bay displayed low cadmium concentrations (< 1.0 mg/kg); whereas the area within the vicinity of the Yacht Club Basin revealed the highest concentration of cadmium. Furthermore, high concentrations of cadmium (exceeding the ERL limit) were also detected in Langebaan Lagoon, particularly at site LL34 (Figure 7.25; Table 7.3). Cadmium is a trace metal used in electroplating, in pigment for paints, in dyes and in photographical process. The likely sources of cadmium to the marine environment are in emissions from industrial combustion processes, from metallurgical industries, from road transport and waste streams (OSPAR 2010). A likely point source for cadmium contamination in the marine environment is that of storm water drains. Cadmium is toxic and liable to bioaccumulation and is thus a concern for both the marine environment and human consumption (OSPAR 2010). Given the spatial pattern it is unlikely that the contamination of cadmium in the Bay is a result of storm water drainage, but rather that the cadmium contamination is resulting from shipping and boating. The area where this is particularly concerning is site SB1 (near the Yacht Club Basin) as the level of contamination at this site frequently exceeds the

ERL limits. Furthermore, the enrichment values for this site since 1980 are high, indicating significant contamination of these areas with cadmium since 1980 (Table 7.4).

7.5.1.2 Copper

Copper concentrations were highest along the Iron Ore Terminal and near the Saldanha Bay Yacht Club within Small Bay (Figure 7.25 & Table 7.3). This suggests that there may be a source of copper pollution affecting the Small Bay region. Copper is used as a biocide in antifouling products as it is very effective for killing marine organisms that attach themselves to the surfaces of boats and ships. Anti-fouling paints release copper into the sea and can make a significant contribution to copper concentrations in the marine environment (Clark 1986). The areas with elevated, normalized copper values also correspond with those with high levels of boat traffic. It is thus likely that anti-fouling paints used on boats may have been contributing copper to the system. It must be noted that no sites are situated in close proximity to Mykonos and the yacht club in Langebaan Lagoon. It is possible that both these areas have also been contaminated by copper. The copper concentration at the Yacht Club Basin in Saldanha Bay exceeded the ERL guideline, the normalized value indicates the pollution source was anthropogenic and the enrichment factor was also alarmingly high in 2020 (Table 7.4).

7.5.1.3 Nickel

Nickel values measured in 2020 were elevated at the yacht club and alongside the iron ore terminal within Small Bay. However, Langebaan Lagoon appeared to have greater nickel concentrations as opposed to the rest of the Bay (Figure 7.25 & Table 7.3). Nickel is introduced to the environment by both natural and anthropogenic means. Natural means of contamination include windblown dust derived from the weathering of rocks and soils, fires and vegetation (Cempel & Nickel 2006). Common anthropogenic sources include the combustion of fossil fuels and the incineration of waste and sewerage (Cempel & Nickel 2006). Contamination of the Bay by nickel is not of great concern as concentrations are well below the ERL guideline limits.

7.5.1.4 Lead

Elevated lead concentrations were recorded in Small Bay particularly in the vicinity of the Multi-Purpose Terminal and the Saldanha Bay Yacht Club (Figure 7.25 & Table 7.3). Lead pollution is a worldwide problem and is generally associated with mining, smelting and the industrial use of lead (OSPAR 2010). Lead is a persistent compound which is toxic to aquatic organism and mammals and thus, the contamination is of concern for the marine environment and human consumption (OSPAR 2010). The area adjacent to the multi-purpose terminal had the highest lead values indicating that this area is subject to high levels of lead pollution. The enrichment factor for the site nearest to the multi-purpose terminal was very high (52.26), however, the concentration of lead was below recommended ERL toxicity limits (Table 7.3). Normalized metal/aluminium ratios revealed that lead contamination was high at numerous sites in Small Bay (Table 7.5). Areas of concern corresponded with sites where high metal concentrations and metal enrichment were indicated.

7.5.1.5 Manganese

Manganese concentrations were highest near the Yacht Club Basin and along the iron ore terminal within Small Bay; despite being below the ERL toxicity thresholds (Figure 7.25 & Table 7.3). This suggests that there may be a source of manganese pollution affecting these areas of the Small Bay region. Manganese is naturally ubiquitous in the marine environment, however, can become potentially harmful through its tendency to accumulate in certain organisms, such as shellfish. The concentration of manganese recorded is possibly associated with the recent start of manganese exports (Section 3.3).

Table 7.4. Enrichment factors for Cadmium, Copper and Lead in sediments collected from Saldanha Bay in 2020 relative to sediments from 1980. ND indicates no data.

	Sample	Cd	Cu	Pb
	1980 average	**0.075**	**0.41**	**0.8**
Small Bay	SB1	32.86	85.99	25.15
	SB2	ND	4.17	8.01
	SB3	ND	7.27	23.52
	SB5	ND	ND	4.16
	SB8	ND	4.10	9.52
	SB9	ND	7.71	9.17
	SB10	ND	3.04	9.04
	SB14	ND	23.13	52.26
	SB15	ND	10.86	17.87
	SB16	ND	7.48	7.25
Big Bay	BB20	ND	7.52	1.54
	BB21	ND	4.27	4.02
	BB22	ND	10.73	6.89
	LPG1	ND	11.14	5.83
	BB24	ND	3.53	3.79
	BB25	ND	ND	ND
	BB26	ND	6.32	5.12
	BB29	ND	8.86	1.65
	BB30	ND	2.62	ND

Table 7.5. Normalized values for Cadmium, Copper Nickel, Lead and Manganese in sediments collected from Saldanha Bay and Langebaan Lagoon in 2020. ND indicates no data.

	Sample	Cd:Al	Cu:Al	Mn: Al	Ni:Al	Pb:Al
Small Bay	SB1	2.19	31.40	47.86	9.88	17.92
	SB2	ND	9.41	130.76	48.23	35.26
	SB3	ND	16.77	216.79	33.20	105.83
	SB5	ND	ND	137.41	117.38	23.40
	SB8	ND	7.48	106.12	20.04	33.85
	SB9	ND	6.55	85.47	15.35	15.20
	SB10	ND	6.94	133.45	52.28	40.30
	SB14	ND	17.60	102.00	15.62	77.57
	SB15	ND	13.73	156.57	14.75	44.08
	SB16	ND	9.84	91.26	15.04	18.63
Big Bay	BB20	ND	23.26	97.47	15.85	9.30
	BB21	ND	6.78	137.45	15.26	12.46
	BB22	ND	14.13	145.22	17.59	17.70
	LPG	ND	10.61	86.89	15.67	10.83
	BB24	ND	5.27	106.12	12.88	11.06
	BB25	ND	ND	173.45	32.09	ND
	BB26	ND	6.47	99.28	13.60	10.23
	BB29	ND	11.64	71.02	19.32	4.22
	BB30	ND	13.81	159.56	64.53	ND
Langebaan Lagoon	LL31	ND	10.31	111.83	25.89	9.80
	LL32	ND	ND	93.97	43.41	12.18
	LL33	ND	ND	76.96	113.66	ND
	LL34	6.88	26.14	79.95	52.40	10.03
	LL37	ND	ND	97.85	118.94	ND
	LL38	ND	4.87	84.27	28.73	6.23
	LL39	ND	ND	134.76	42.32	ND
	LL40	ND	12.21	151.11	105.49	ND
	LL41	ND	ND	65.74	96.91	ND

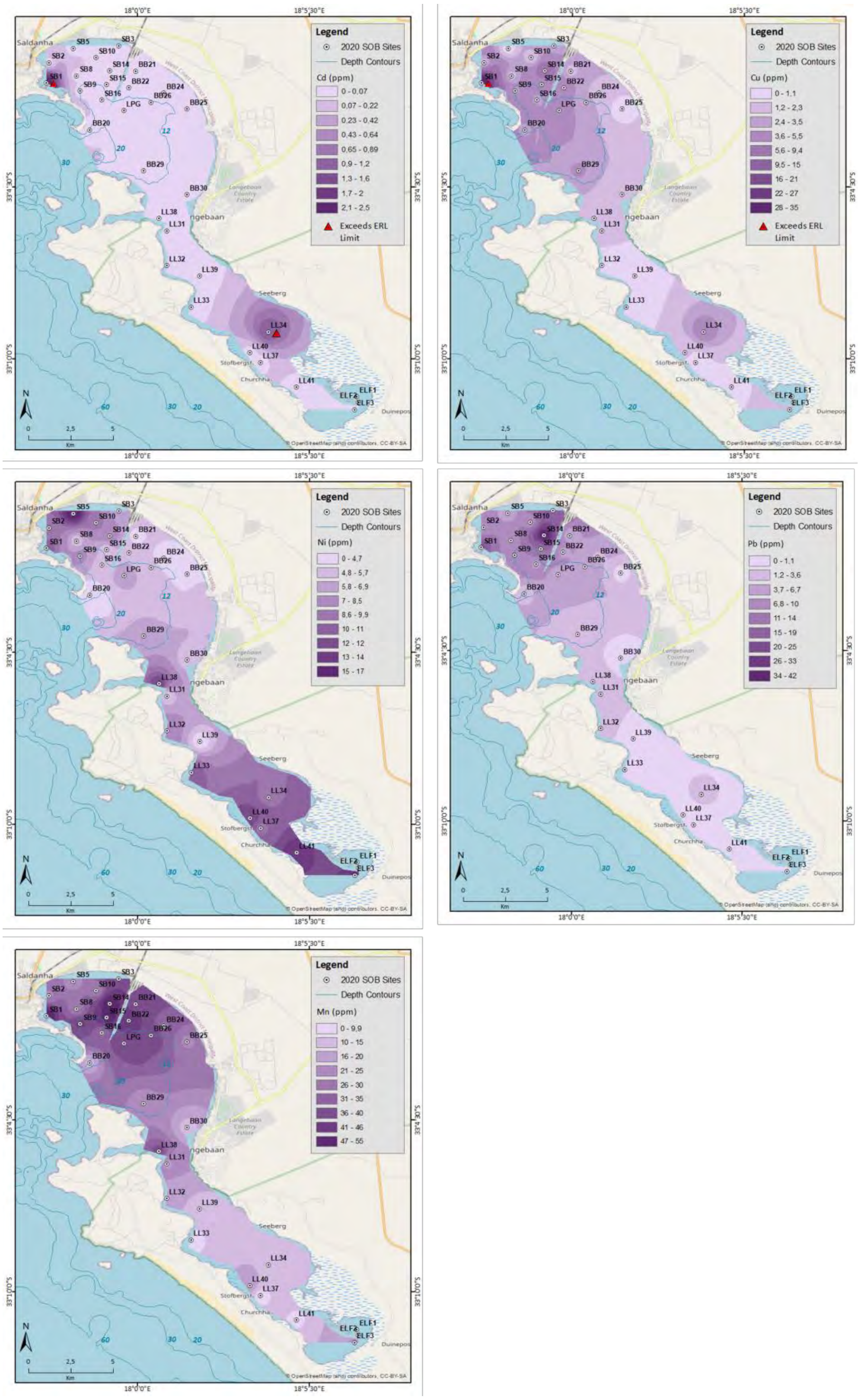


Figure 7.25. **Spatial interpolation of cadmium (Cd), copper (Cu), nickel (Ni), lead (Pb) and manganese (Mn) values measured in sediments in Saldanha Bay in 2020. Red triangles indicate sites that exceed the ERL limit.**

7.5.2 Temporal variation in trace metal levels in Saldanha Bay

The temporal variation in the concentration of trace metals in the most heavily contaminated areas (Small Bay and along the iron ore terminal in Big Bay) relative to the ERL guidelines is discussed below.

7.5.2.1 Cadmium

There was a considerable increase in the concentration of cadmium detected in the sediments of Saldana Bay between 1980 and 1999. In 1999, the levels of cadmium recorded at the Mussel Farm, the Yacht Club Basin and the Channel End of the iron ore terminal exceeded the ERL toxicity threshold of 1.2 mg/kg established by NOAA (Figure 7.26). Cadmium concentrations have shown a progressive and dramatic decrease in the period 1999-2010; however, the results between 2010 and 2013 indicated a steady increase again in the cadmium concentrations at the Yacht Club Basin and Multi-purpose Quay. At the time of the 2014 survey, cadmium concentrations had decreased to below the ERL toxicity threshold within the Yacht Club Basin, but since 2015, levels have remained high. Concentrations at the multi-purpose terminal have shown a steady decrease since 2014 and reached its lowest to < 1.0mg/kg in 2020. Low cadmium concentrations such as the latter were also detected across the rest of the key sites in the bay in the year 2020; with the exception at the Yacht Club Basin; where concentrations exceeded the ERL threshold but had decreased compared to the 2019 survey. The Channel end ore jetty and the north channel sites in Small Bay are the two sites where cadmium concentrations remained consistently low over the years (Figure 7.26).

7.5.2.2 Copper

The total concentration of copper in the sediments has remained well below the ERL threshold consistently since 1980; with the exception of the Yacht Club Basin which has exceeded the ERL in most years (Figure 7.27). Apart from the low levels recorded in 2014, copper concentrations at the Yacht Club Basin have remained high (above the ERL guideline) over the past ten years. In the recent 2020 survey, there has been a noticeable decrease in copper concentrations across all sites, particularly at the Yacht Club Basin (Figure 7.27).

7.5.2.3 Nickel

The concentration of nickel was the highest at the Yacht Club Basin and the Mussel Farm in 1999 where it exceeded the ERL threshold (Figure 7.28). Since 1999, nickel concentrations have declined markedly at both sites, never again exceeding the ERL threshold. Peak nickel concentration at the remaining four sites were observed in 2000; though concentrations did not exceed the ERL threshold. Since 2000, levels of nickel have declined at all four of these sites and remained relatively constant to present date up to the 2018 survey. From 2019 onwards, all six localities had an increase in nickel concentration; except for the Yacht Club Basin having a decline in the current 2020 survey (Figure 7.28).

7.5.2.4 Lead

The concentration of lead peaked and exceeded the ERL threshold at the Yacht Club Basin and Mussel farm site in 1999 (Figure 7.29). The concentration of lead at these sites has not exceeded the ERL level since this time. Lead concentrations in sediments adjacent to the multi-purpose terminal have frequently exceeded the ERL threshold over the last 16 years. This result suggests that industrial and shipping activities taking place at the multi-purpose terminal continue to contaminate the adjacent marine environment with lead. The 2020 survey indicated elevated levels in lead concentrations at all six localities; of which some were subtle (Figure 7.29).

7.5.2.5 Manganese

The temporal variation in manganese concentrations in sediments around the ore terminal in Saldanha Bay is shown in Figure 7.30. Manganese concentrations at sites located along the ore terminal within Small Bay have fluctuated over recent years. High concentrations of manganese were recorded at the Small Bay sites in 2014 but have gradually decreased over the last three years; however, the manganese concentrations are greatly elevated in 2019 for all three sites within Small Bay. The latter pattern was also evident for the two sites located along the ore terminal within Big Bay for 2019. However, in the current survey, all manganese concentrations have relatively declined across all key sites in the Bay, with the exception of the multi-purpose terminal in Small Bay (Figure 7.30).

7.5.2.6 Iron

The temporal variation in the concentration of iron in sediments around the ore terminal in Saldanha Bay is shown in Figure 7.31. The concentration of iron increased between 1999 and 2004 at sites SB14 and SB15 which are in closest proximity to and on the downwind side (of the predominant southerly winds) of the multi-purpose terminal. This may have been due to increases in volumes of ore handled or increases in losses into the sea over this period, or simply reflects accumulation of iron in the sediments over time. There was a reduction in the concentration of iron in the sediments at most sites on the Small Bay side of the ore terminal between 2004 and 2010. Dredging took place at the multi-purpose terminal in 2007 and the removal of iron rich sediment at SB15 is probably the reason for the dramatic decrease in iron concentration recorded at this station between 2008 and 2009 sampling. Sediment iron concentration at this site did increase in 2009; but decreased again in 2010 samples. The 2011 survey revealed that iron concentrations had increased at most sites around the ore terminal despite reductions in the mud contents at all sites. This suggests that fluctuations in iron content are a result of iron inputs rather than the flushing experienced at the sites.

Transnet has implemented a number of new dust suppression measures in recent years (SRK 2009, Viljoen *et al.* 2010). Dust suppression mitigation measures implemented since mid-2007 include conveyer covers, a moisture management system, chemical dust suppression and surfacing of roads and improved housekeeping (road sweeper, conveyor belt cleaning, vacuum system, dust dispersal modelling and monitoring) amongst others. The volume of ore handled at the bulk quay has increased from around 4.5 million tonnes per month during 2007-2008 to around 6.5 million tonnes during 2009-2010 (~50% increase); yet the concentration of iron in the sediments at sites adjacent to the ore terminal remained fairly stable or decreased between 2009 and 2010. Relatively small fluctuations in the concentration of iron were seen at five of the six sites between 2010 and 2020 (Figure 7.31). However, the concentration of iron at SB15 has fluctuated dramatically since 2012; but has shown an overall decrease in the last eight years. This does suggest that the improved dust control methods implemented since 2007 have been successful in reducing the input to the marine environment. Although in 2019, there was a significant increase in lead concentration at SB15. Overall, iron concentrations have decreased greatly across all sites in the current 2020 survey. On-going monitoring of sediment iron concentration will reveal whether the decrease recorded across these sites will continue with the anticipated higher volumes of ore handling or if concentrations will continue to fluctuate.

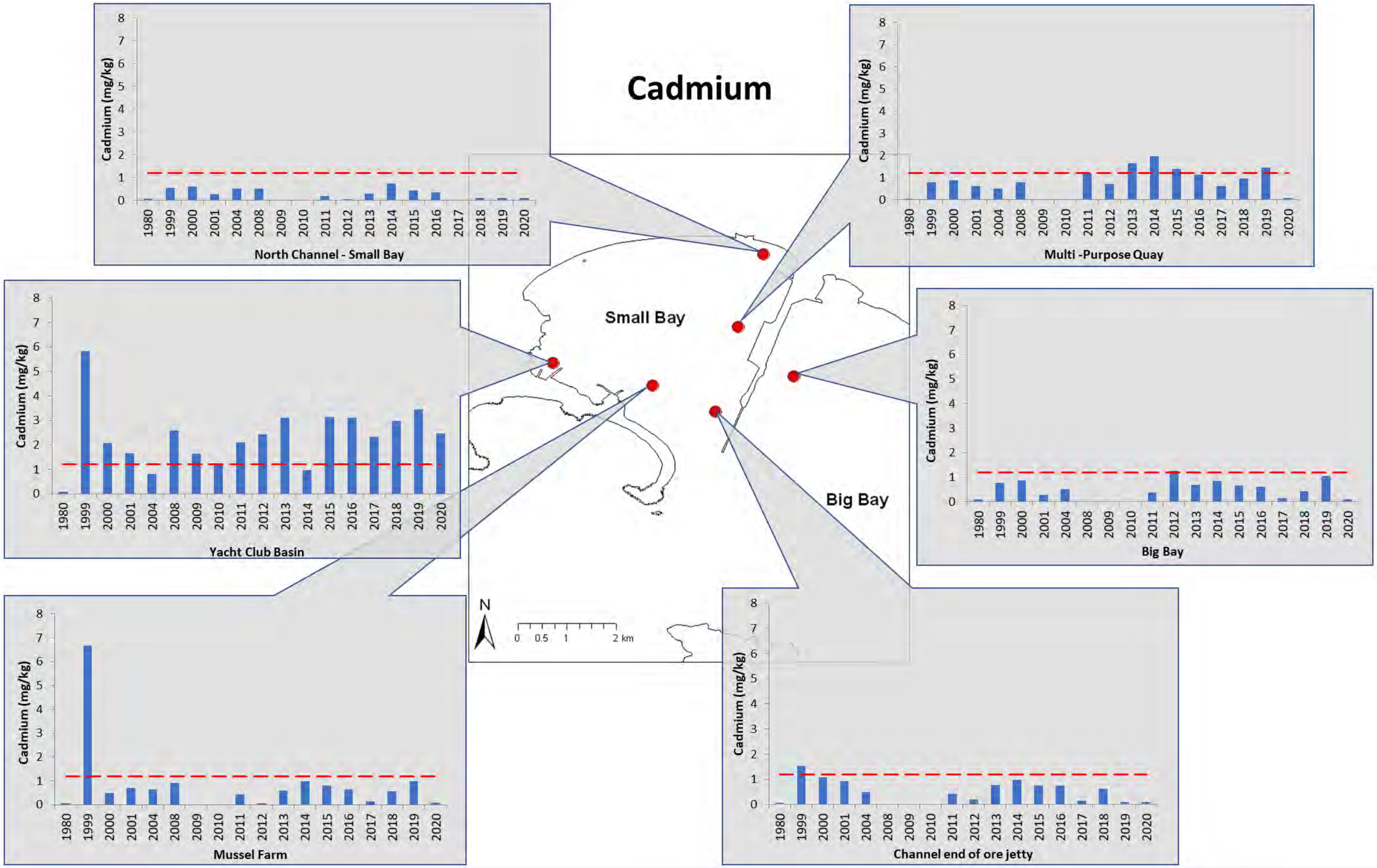


Figure 7.26. **Concentrations of Cadmium (Cd) in mg/kg recorded at six sites in Saldanha Bay between 1980 and 2020. Dotted lines indicate Effects Range Low values for sediments.**

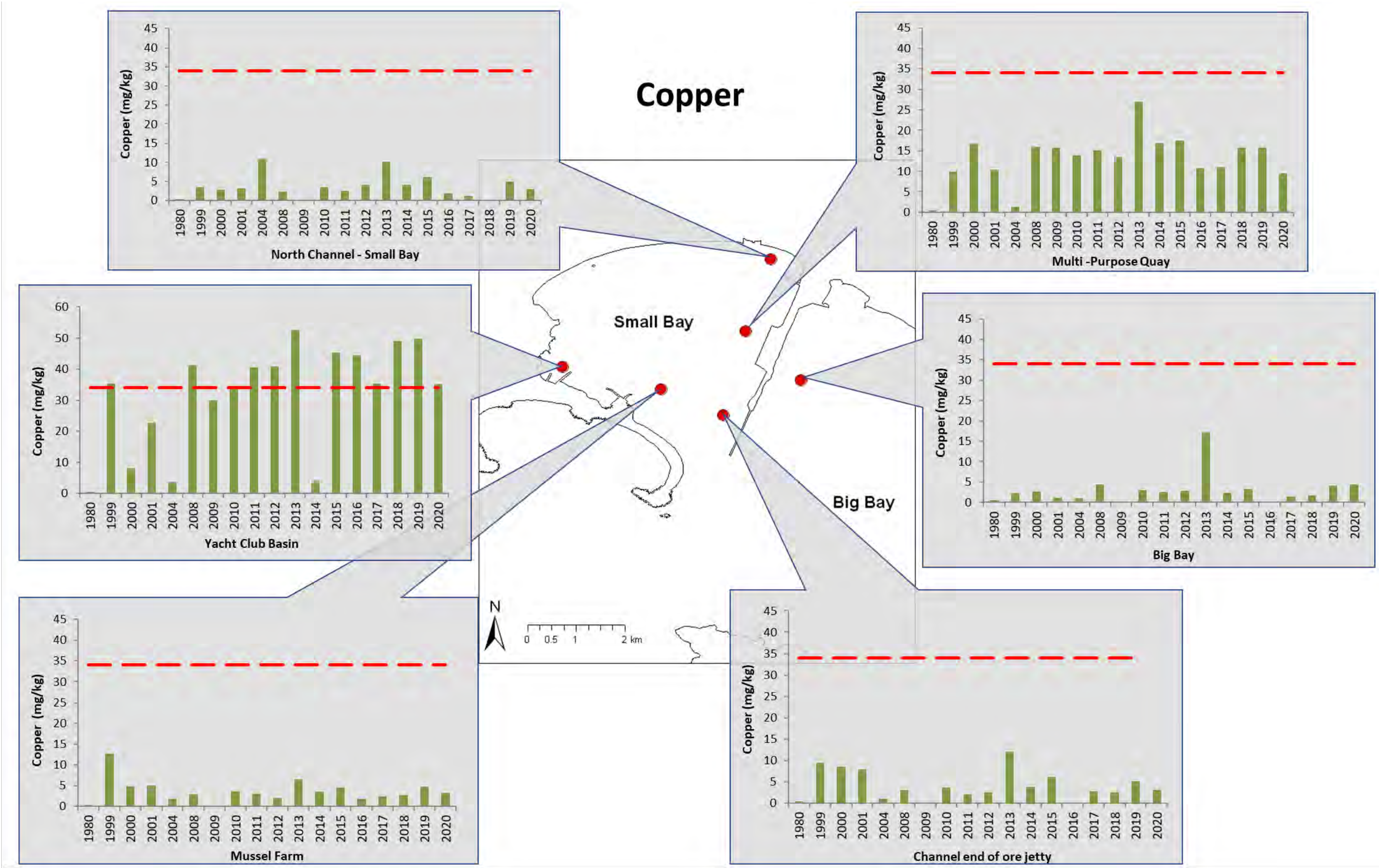


Figure 7.27. Concentrations of Copper (Cu) in mg/kg recorded at six sites in Saldanha Bay between 1980 and 2020. Dotted lines indicate Effects Range Low values for sediments.

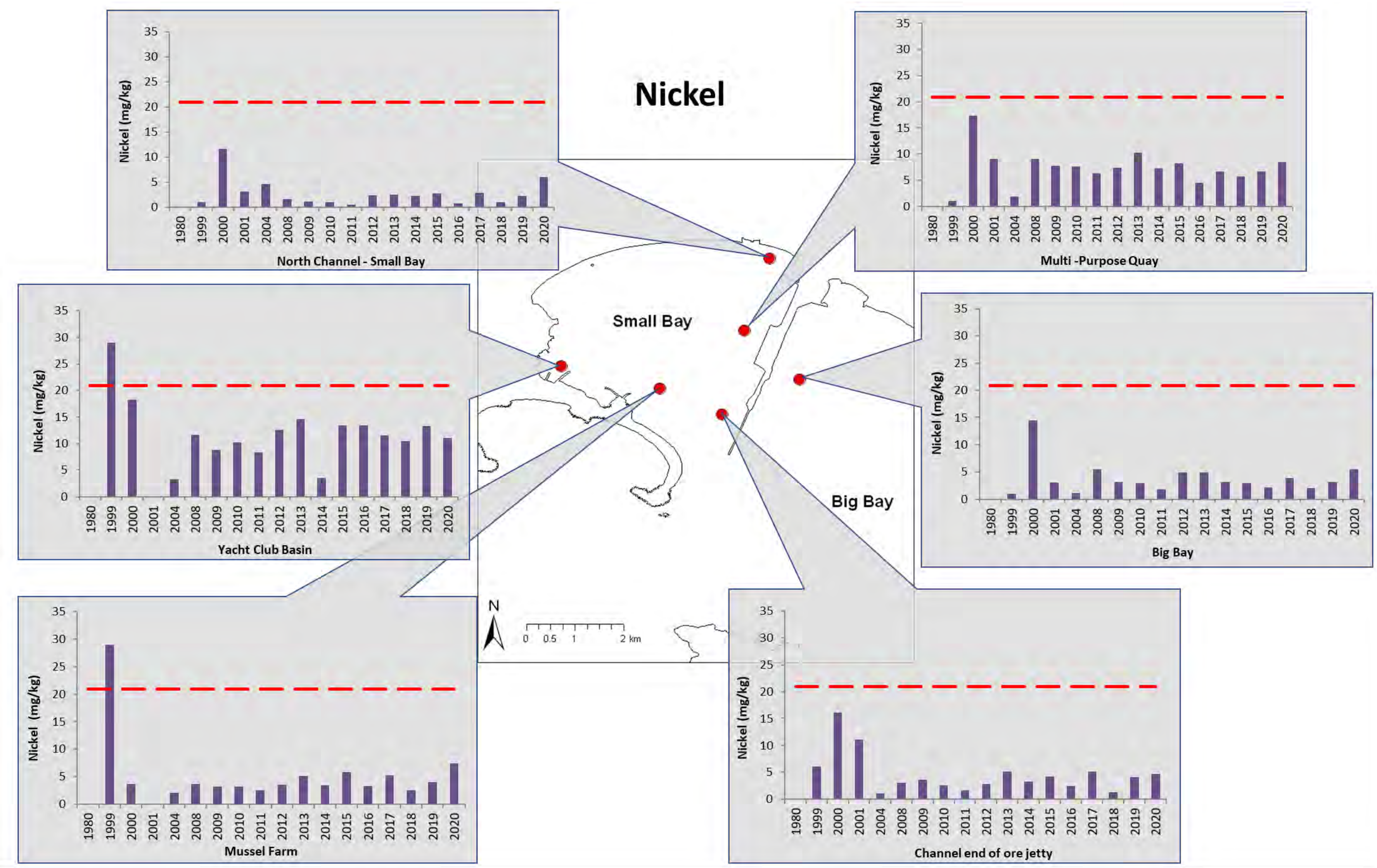


Figure 7.28. Concentrations of Nickel (Ni) in mg/kg recorded at six sites in Saldanha Bay between 1980 and 2020. Dotted lines indicate Effects Range Low values for sediments.

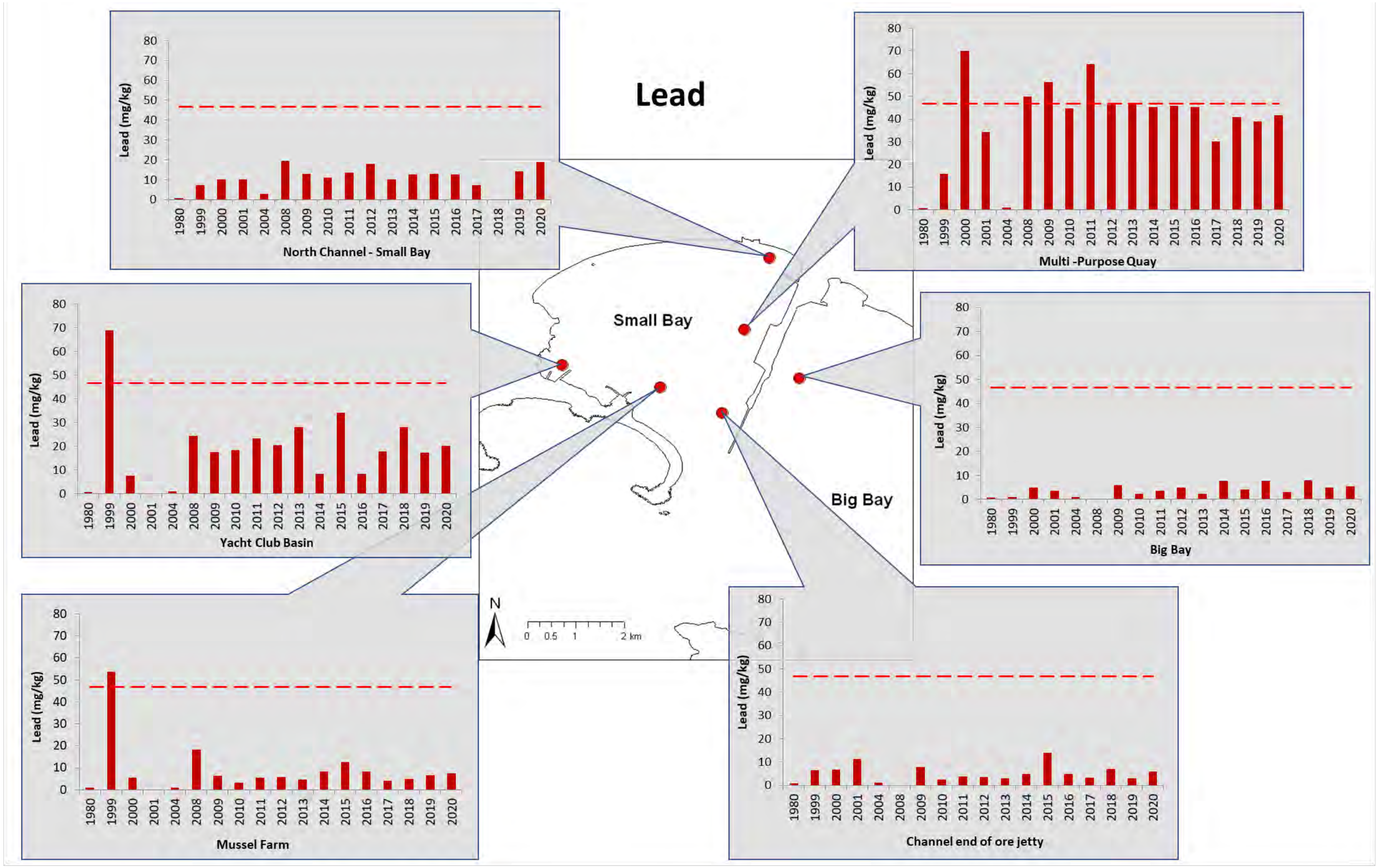


Figure 7.29. Concentrations of Lead (Pb) in mg/kg recorded at six sites in Saldanha Bay between 1980 and 2020. Dotted lines indicate Effects Range Low values for sediments.

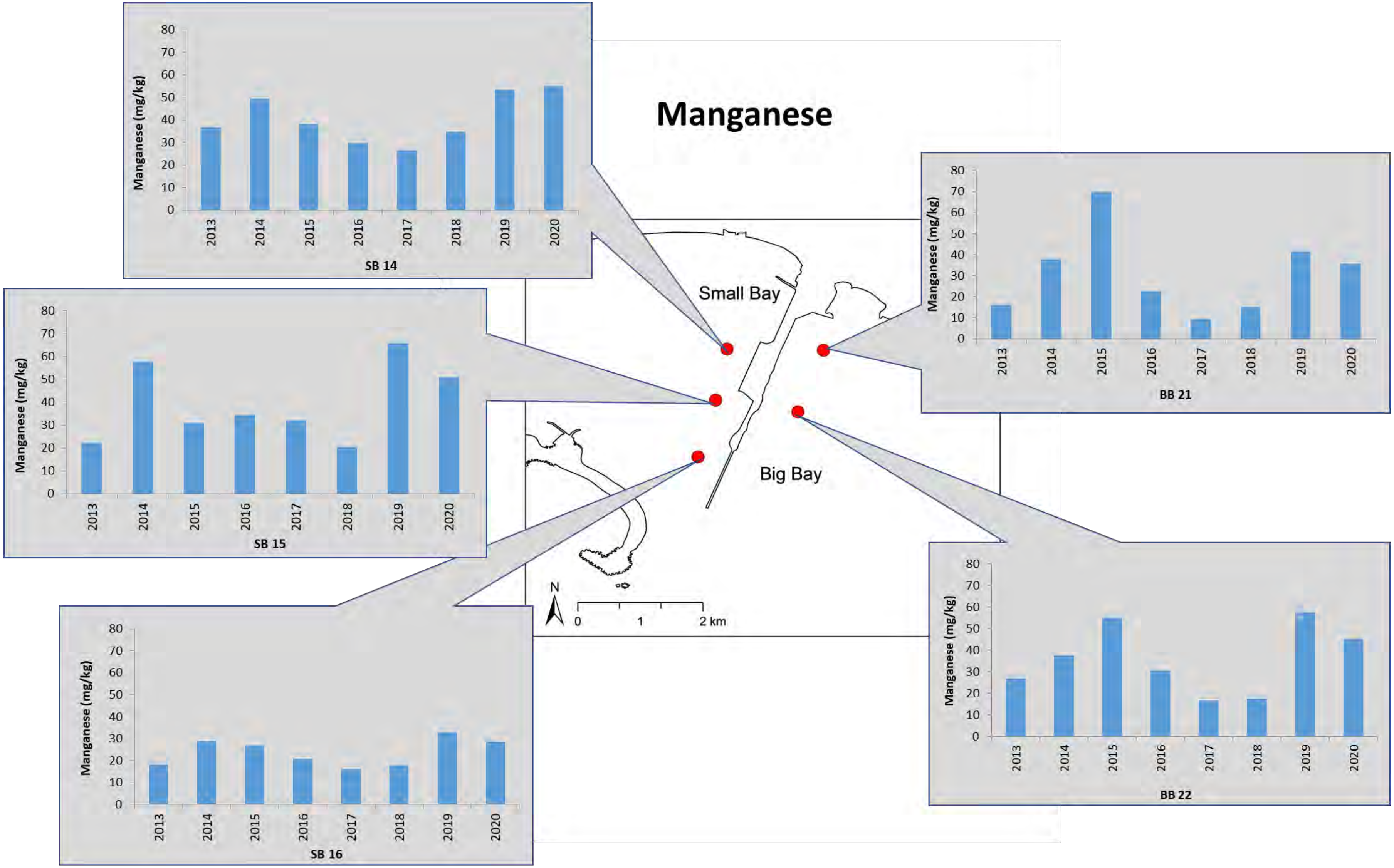


Figure 7.30. **Concentration of manganese (Mn) in mg/kg recorded at five sites in Saldanha Bay between 2013 and 2020.**

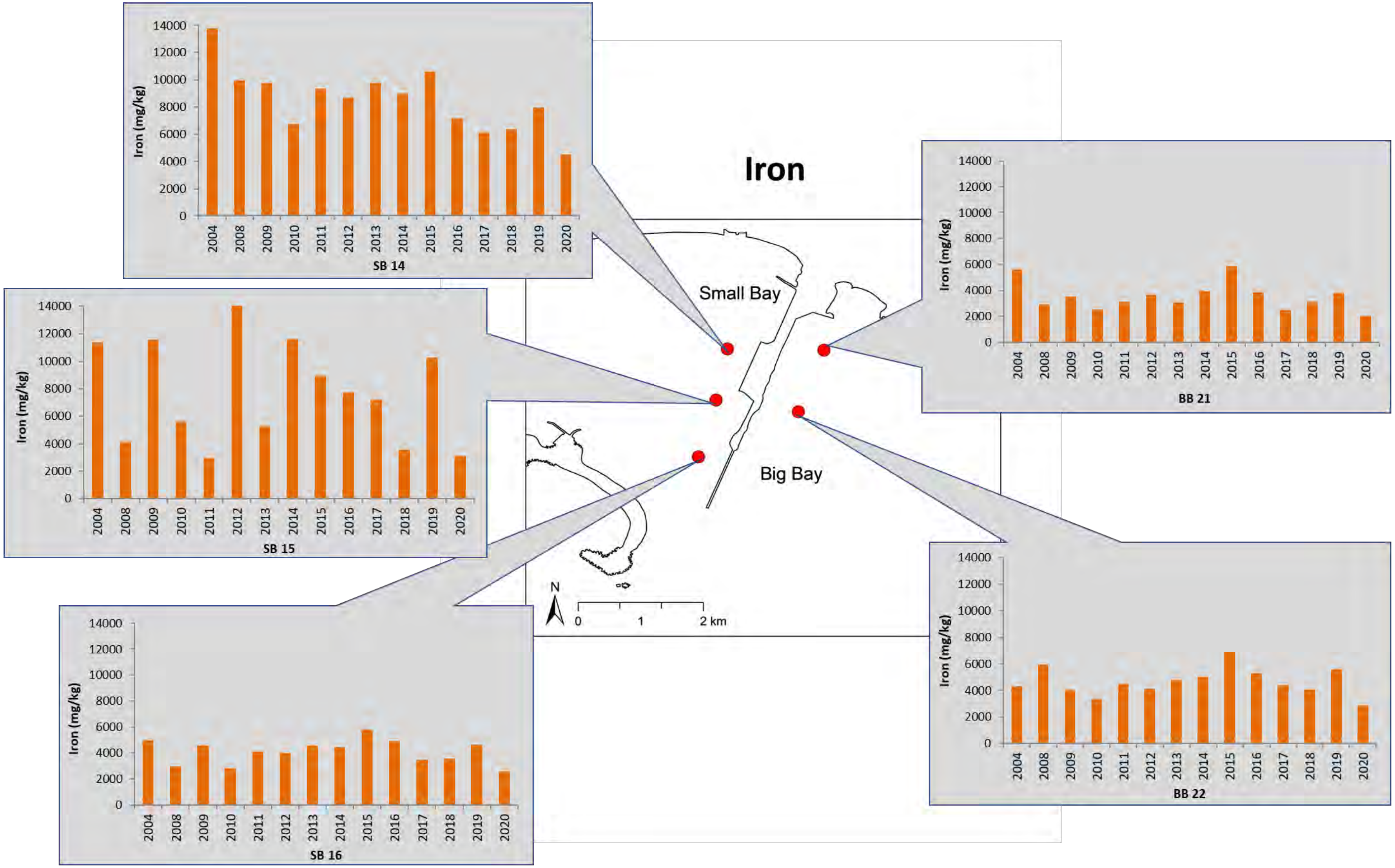


Figure 7.31. **Concentrations of Iron (Fe) in mg/kg recorded at five sites in Saldanha Bay between 2004 and 2020.**

7.6 Hydrocarbons

Poly-aromatic hydrocarbons (PAH) (also known as polynuclear or polycyclic-aromatic hydrocarbons) are present in significant amounts in fossil fuels (natural crude oil and coal deposits), tar and various edible oils. They are also formed through the incomplete combustion of carbon-containing fuels such as wood, fat and fossil fuels. PAHs are one of the most wide-spread organic pollutants and they are of particular concern as some of the compounds have been identified as carcinogenic for humans (Nikolaou *et al.* 2009). PAHs are introduced to the marine environment by anthropogenic (combustion of fuels) and natural means (oil welling up or products of biosynthesis) (Nikolaou *et al.* 2009). PAHs in the environment are found primarily in soil, sediment and oily substances, as opposed to in water or air, as they are lipophilic (mix more easily with oil than water) and the larger particles are less prone to evaporation. The highest values of PAHs recorded in the marine environment have been in estuaries and coastal areas as well as in areas with intense vessel traffic and oil treatment (Nikolaou *et al.* 2009).

Marine sediment samples from Saldanha Bay were analysed for the presence of hydrocarbons in 1999. No PAHs were detectable in the samples, but low levels of contamination by aliphatic (straight chain) molecules, which pose the lowest ecological risk, were detected. This suggested that the main source of contamination is the spilling and combustion of lighter fuels from fishing boats and recreational craft (Monteiro *et al.* 1999). Sediment samples from five sites in the vicinity of the oil terminal in Saldanha Bay were tested for PAH contamination in April 2010. PAH concentrations at all five sites were well below ERL values stipulated by NOAA. From 2011 to 2014 PAH levels were not tested due to the continual low levels. However, analysis of total petroleum hydrocarbon (TPH) concentrations was continued.

Table 7.6. Total petroleum hydrocarbons (mg/kg) in sediment samples collected over the period 2011-2020 from five stations in Saldanha Bay. Values in red indicate exceptionally high total petroleum hydrocarbon levels. ND indicates no data available.

	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020
SB14	<20	34	130	19	<38	<38	<38	<38	<38	<38
SB15	<20	35	ND	53	<38	<38	<38	<38	<38	<38
SB16	<20	24	28	14 649	<38	<38	<38	<38	<38	<38
BB21	<20	20	32	20	<38	<38	<38	<38	<38	<38
BB22	<20	17	27	<0.2	<38	<38	<38	<38	<38	<38

PAH levels have been well below the guideline limits and despite there being no guideline limits to determine the toxicological significance of TPH contamination there have been considerable fluctuations in contamination levels since 2011. TPH levels recorded in 2011 were below the detection limit of 20 mg/kg while slight increases were recorded at all sites in 2012 and 2013 (Table 7.6). TPH levels at site SB14 decreased from 130 mg/kg to 19 mg/kg in 2014, however, there was the extreme increase at site SB16 from 28 mg/kg to 14 649 mg/kg. The most likely explanation for the high TPH levels recorded is that a pollution incident associated with shipping activities took place. Alternatively, a pollution incident or routine operational activities on the jetty itself could be the root of this contamination. Since 2015, TPH concentrations have been below the detection limit of 38mg/kg and remained at this level at all five sites to present date.

Sediment samples collected in 2020 had low PAH levels across all sites (Table 7.7). While the TPH and PAH findings present no major concern, it is recommended that TPH monitoring within the vicinity of the ore terminal is continued annually in order to identify the frequency of occurrence of pollution incidents; like that recorded in 2014, and assess the ecological implications to the Bay.

Table 7.7. Sediment Quality guidelines and Poly-aromatic hydrocarbons concentrations measured in sediment samples collected from Saldanha Bay in April 2020.

Hydrocarbon (mg/kg)	ERL*	ERM**	SB14	SB15	SB16	BB21	SB22
Acenaphthene	**0.016**	**0.5**	<0.004	<0.004	<0.004	<0.004	<0.004
Acenaphthylene	**0.044**	**0.64**	<0.004	<0.004	<0.004	<0.004	<0.004
Anthracene	**0.0853**	**1.1**	<0.004	<0.004	<0.004	<0.004	<0.004
Benzo(a) anthracene	**0.261**	**1.6**	<0.004	<0.004	<0.004	<0.004	<0.004
Benzo(a) pyrene	**0.43**	**1.6**	<0.004	<0.004	<0.004	<0.004	<0.004
Benzo(b+k) flouranthene	-	-	<0.0008	<0.0008	<0.0008	<0.0008	<0.0008
Benzo (g.h.i) perylene	-	-	<0.0008	<0.0008	<0.0008	<0.0008	<0.0008
Crysene	**0.384**	**2.8**	<0.004	<0.004	<0.004	<0.004	<0.004
Dibenzo (a.h) anthracene	**0.0634**	**0.26**	<0.0008	<0.0008	<0.0008	<0.0008	<0.0008
Flouranthene	**0.6**	**5.1**	<0.004	<0.004	<0.004	<0.004	<0.004
Flourene	**0.019**	**0.54**	<0.004	<0.004	<0.004	<0.004	<0.004
Indeno (1.2.3-c.d) pyrene	-	-	<0.0008	<0.0008	<0.0008	<0.0008	<0.0008
Naphthalene	**0.16**	**2.1**	<0.004	<0.004	<0.004	<0.004	<0.004
Phenanthrene	**0.24**	**1.5**	<0.004	<0.004	<0.004	<0.004	<0.004
Pyrene	**0.665**	**2.6**	<0.004	<0.004	<0.004	<0.004	<0.004
Total PAH	**4**	**44.7**	-	-	-	-	-

*Effects Range Low guideline stipulated by NOAA below which toxic effects rarely occur in sensitive marine species.

**Effects Range Median guideline stipulated by NOAA above which toxic effects frequently occur in sensitive marine species.

8 AQUATIC MACROPHYTES IN LANGEBAAN LAGOON

8.1 Community composition and distribution

Three distinct intertidal habitats exist within Langebaan Lagoon: seagrass beds, such as those of the eelgrass *Zostera capensis;* salt marsh dominated by cordgrass *Spartina maritime* and *Sarcocornia perennis* and the dune slack rush *Juncus kraussi*, and unvegetated sandflats dominated by the sand prawn, *Callianassa krausii* and the mudprawn *Upogebia capensis* (Siebert & Branch 2005). The other major vegetation type present in the upper lagoon area, particularly where groundwater inflow occurs, are reed beds dominated by *Phragmites australis.* The most recent, detailed vegetation map of the area surrounding Langebaan Lagoon dates to 2013 (Figure 8.1) (van der Lindern 2013). In this map, eelgrass *Zostera capensis* falls within the submerged macrophyte category.

Salt marsh communities are generally comprised of herbs, shrubs and grasses within areas that are tidally inundated (Nybakken 2001). Within traditional salt marshes, plant communities occur along distinct zones following a tidal gradient and elevation pattern (Hughes and Paramor 2004; Perry and Atkinson 2009). Salt marsh species occur in a hostile environment, and as few species are able to cope in such environments, species diversity is low. Salt marshes tend to be associated with euhaline (30 to 35 ppt) conditions that many salt marsh species are able to cope with, however, growth rates tend to decrease as salinity increases and germination occurs only when the surrounding water salinity decreases (Smart & Barko 1980, Price *et al*. 1988).

The primary abiotic factors influencing salt marsh distributions are salinity and water availability (Pan *et al*. 1998). Salt marshes growing in areas with high water availability (high rainfall and intertidal zones) are influenced by sediment salinity more than by water availability in terms of zonation patterns (Krüger and Peinemann 1996). Sediment moisture limits the growth of xerohalophytes (those that occur in drier soils, Zedler *et al.* 1986), which in turn is dependent on the depth of the water table (Bornman *et al.* 2008). Salt marsh communities often show a distinct zonation pattern along tidal inundation and salinity gradients, whereby different plant species and different vegetation colours are seen (Adams and Ngesi 2002). Salt marshes are often separated into three zones, subtidal, intertidal and supratidal (Figure 8.1). Zonation is influenced by biotic interactions and by spatial and temporal gradients in physical variable such as salinity and soil moisture (Noe & Zedler 2001; Rogel *et al.* 2001). Subtidal and intertidal zones are generally structure by stress tolerances, especially by high salt gradients, while the supratidal zone may be characterised by competition (Emery *et al.* 2001).

Sand and mud pawns are considered ecosystem engineers as their feeding and burrowing activities modify the local environmental conditions, which in turn modify the composition of the faunal communities (Rhoads & Young 1970, Woodin 1976, Wynberg & Branch 1991, Siebert & Branch 2006). Seagrass beds and salt marshes perform an opposite and antagonistic engineering role to that of the sand and mud prawns as the root-rhizome networks of the seagrass and saltmarsh plants stabilize the sediments (Siebert & Branch 2005). In addition, the three dimensional leaf canopies of the seagrass and saltmarsh plants reduce the local current velocities thereby trapping nutrients and increasing sediment accretion (Kikuchi & Perez 1977, Whitfield *et al.* 1989, Hemmingra & Duarte 2000). The importance of seagrass and saltmarsh beds as ecosystem engineers has been widely recognized. The increased food abundance, sediment stability, protection from predation and habitat complexity offered by seagrass and saltmarsh beds provide nursery areas for many species of fish and invertebrates. These habitats support, in many cases, a higher species richness, diversity, abundance and biomass of invertebrate fauna compared to unvegetated areas (Kikuchi & Peres 1977, Whitfield *et al.* 1989, Hemmingra & Duarte 2000, Heck *et al.* 2003, Orth *et al.* 2006, Siebert & Branch 2007). It is therefore surprising that recent research in the Langebaan Lagoon (Pillay *et al.* 2011) showed that the opposite was true when comparing sediment penetrability and species richness between habitats dominated by the sandprawn *Calianassa kraussi* and cordgrass *Spartina maritime.* Bioturbation by the sandprawn loosened the sediment, resulting in less anoxic conditions, enhanced organic content and colonisation of burrowing species. It was speculated that the sandprawn may aid in increasing food availability to higher trophic levels. Seagrass and saltmarsh beds are also important for waterbirds some of which feed directly on the shoots and rhizomes, forage amongst the leaves or use them as roosting areas at high tide (Baldwin & Lovvorn 1994, Ganter 2000, Orth *et al.* 2006).

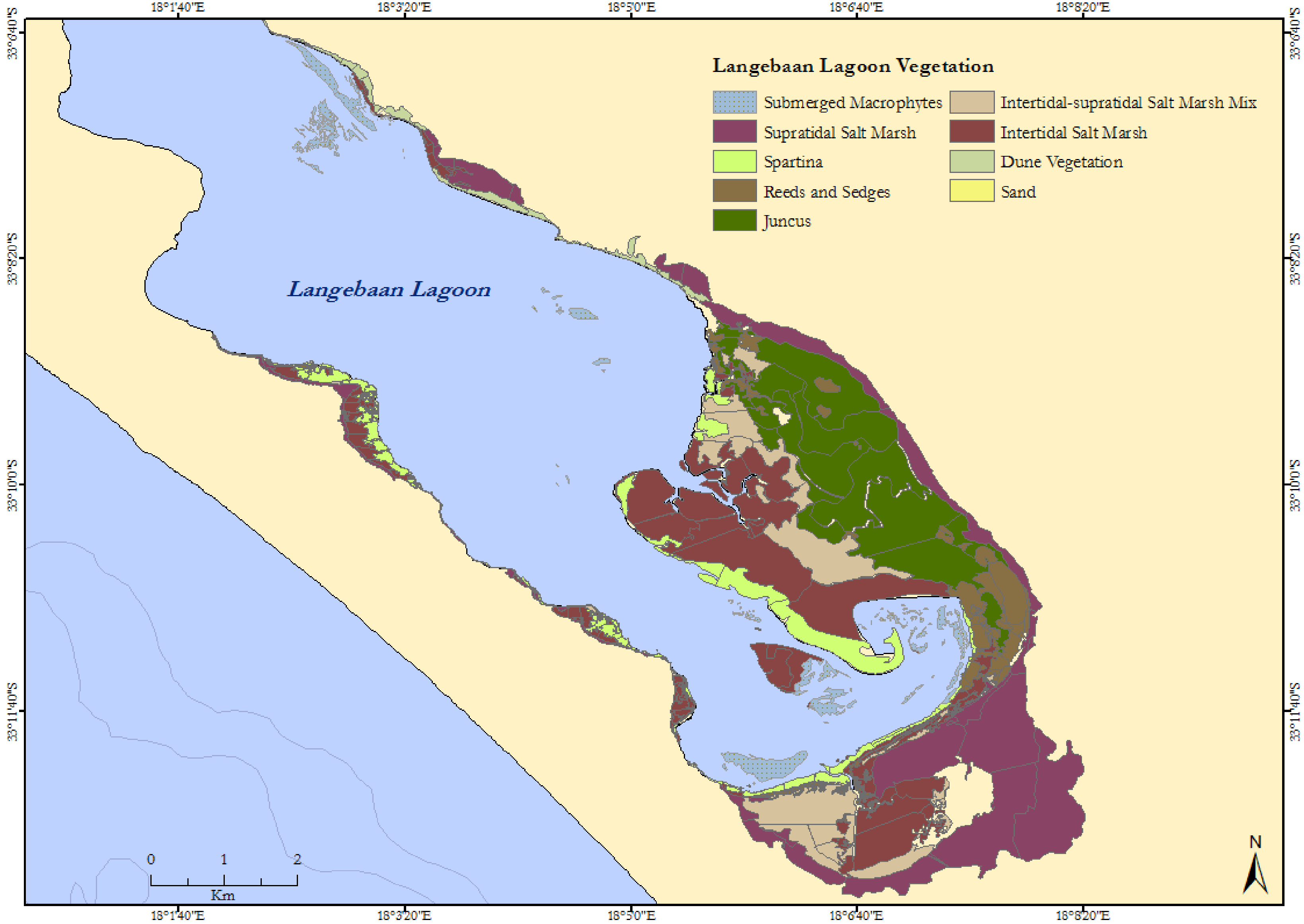


Figure 8.1. **Vegetation and habitat structure at Langebaan Lagoon (Source: Shapefiles provided by van der Linden 2013).**

8.2 Long term changes in seagrass in Langebaan Lagoon

Seagrass beds are particularly sensitive to disturbance and are declining around the world at rates comparable to the loss of tropical rainforests, placing them amongst the most threatened ecosystems on the planet (Waycott *et al.* 2009). The loss of seagrass beds is attributed primarily to anthropogenic impacts such as deterioration in water quality through nutrient enrichment or eutrophication, alterations to food webs caused by the overexploitation of predatory fish, modified sediment dynamics associated with coastal and harbour development and direct physical damage through bait collection (Waycott *et al.* 2009, Pillay *et al.* 2010). Most recently, research has shown that warmer temperatures and longer exposure to air resulted in significantly lower biomass of seagrass in the Langebaan Lagoon (University of Cape Town, Cloverly Lawrence, *pers. comm.* 2014).

The loss of seagrass meadows has been shown to have profound implications for the biodiversity associated with them, including loss of invertebrate diversity, fish populations that use the sheltered habitat as nurseries, and waterbirds that use the seagrass meadows as foraging grounds during their non-breeding period (Hughes *et al.* 2002). Loss of seagrass is also associated with increased fragmentation of large seagrass beds, which leads to the reduced species diversity. For example, Källén *et al.* (2012) demonstrated that large seagrass beds were home to significantly greater epifaunal richness and abundance of *Assiminea globules. A. globules* is a gastropod which favours seagrass bed edges. Species composition was found to differ between the edges and the interior of seagrass beds and interestingly, it was shown that species composition was more homogenous in more fragmented seagrass beds (Källén *et al.* 2012).

Long-term changes in seagrass beds in Langebaan Lagoon have been investigated by Angel *et al.* 2006 and Pillay *et al.* (2010). Angel *et al.* (2006) focused on long term trends at Klein Oesterwal and Bottelary, and was able to show that the width of the *Z. capensis* bed changed substantially between 1972 and 2004, with three major declines evident in this period (Figure 8.2.). The first occurred in the late 1970s, and was followed by a slow recovery in the early 1980's, the second occurred between 1988 and 1993 and the third between 2002 and 2004 (Angel *et al.* 2006). Mirroring this decline were substantial fluctuations in the abundance of the small endemic limpet *Siphonaria compressa*, which lives on the leaves of *Z. capensis* and is completely dependent on the seagrass for its survival. The densities of *S. compressa* collapsed twice in this period to the point of local extinction, corresponding with periods of reduced seagrass abundance (Figure 8.2.). At Bottelary, the width of the seagrass bed and densities of *S. compressa* followed the same pattern as at Klein Oesterwal, with a dramatic collapse of the population between 2002 and 2004, followed by a rapid recovery in 2005 (Angel *et al.* 2006). The first decline in seagrass cover coincided with blasting and dredging operations in the adjacent Saldanha Bay, but there is no obvious explanation for the second decline (Angel *et al.* 2006).

Pillay *et al.* (2010) documents changes in seagrass *Zostera capensis* abundance at four sites in the Lagoon – Klein Oesterwal, Oesterwal, Bottelary and the Centre banks using a series of aerial photographs covering the period 1960 to 2007. During this time, the total loss of *Z. capensis* amounted to 38% or a total of 0.22 km^2 across these sites. The declines were most dramatic at Klein Oesterwal where close to 99% of the seagrass beds were lost during this period, but were equally concerning at Oesterwal (82% loss), Bottelary (45% loss) and Centre Bank (18% loss) (Pillay *et al.* 2010). Corresponding changes were also observed in densities of benthic macrofauna at these sites, with species that were commonly associated with Zostera beds such as the starfish *Parvulastra exigua*, the limpets *Siphoneria compressa* and *Fisurella mutabilis* and general surface dwellers such as the gastropods *Assiminea globules*, *Littorina saxatilis*, and *Hydrobia* sp. declining in abundance. Species that burrowed predominantly in unvegetated sand, such as amphipods *Urothoe grimaldi* and the polychaetes *Scoloplos johnstonei* and *Orbinia angrapequensis* increased in density over that same period. Pillay *et al.* (2010) was also able to show that the abundance of at least one species of wading bird, the Terek Sandpiper which feeds exclusively in Zostera beds was linked to changes in the size of these beds, with population crashes in this species coinciding with periods of lowest seagrass abundance at Klein Oesterwal. By contrast, they were able to show that populations of wader species that do not feed in seagrass beds were more stable over time.

While the precise reasons for the loss of *Z. capensis* beds remain speculative, the impact of human disturbance cannot be discounted, particularly at Klein Oesterwal where bait collection and in the last decade, kite surfing, has become very popular (Pillay *et al.* 2010). Most recent research in the Langebaan Lagoon shows that seagrass morphometric growth patterns are mainly controlled by temperature, followed closely by turbidity as a proxy for light levels. It was found that cooler temperatures and less tidal exposure time favour higher seagrass biomass than warmer more exposed areas. This finding could partly explain the distribution patterns in the lagoon as determined from aerial photography (University of Cape Town, Cloverly Lawrence, *pers. comm.* 2014).

By 2007 the intertidal habitat at Klein Oesterwal had been transformed from a seagrass bed community to an unvegetated sand flat which was colonized by the burrowing sandprawn *Callinassa kraussi* and other sandflat species that cannot live in the stabilized sediments promoted by the seagrass (Pillay *et al.* 2010). The burrowing sandprawn turns over massive quantities of sediment and once established effectively prevents the re-colonization of seagrass and the species associated with it (Siebert & Branch 2005, Angel *et al.* 2006). The long-term effects of the loss of seagrass at Klein Oesterwal, and to a lesser degree at Bottelary and the Central banks, are not yet fully understood. However, studies suggest that the reduced seagrass bed coverage and the associated changes to macro-invertebrates may have cascading effects on higher trophic levels (Whitfield *et al.* 1989, Orth *et al.* 2006). Alterations to fish species diversity and abundance, and changes in the numbers of water birds that forage or are closely linked to seagrass beds may be seen in Langebaan Lagoon as a result of seagrass bed decline (Whitfield *et al.* 1989, Orth *et al.* 2006). To date, however, despite more than a decade of monitoring, changes in fish and bird communities (with the exception of the Terek Sandpiper) in Langebaan that can be attributed to sea grass loss have not been detected. This may be due to several reasons; certainly the timing of sea grass loss predated the State of the Bay monitoring that started in 2005 and any significant changes in the community compositions of fish and birds had already occured. The relatively modest scale of seagrass loss throughout the lagoon may also explain the undetected impacts on higher trophic level species, despite Pillay *et al.* (2010) recording a reduction to nearly 25 ha, Van Der Linden (2014) mapped the area of submerged macrophytes (*Zostra*) at 85.8 ha indicating that substantial *Zostra* habitat remains in the Lagoon (Adams 2016). Alternatively, more severe impacts on fish and bird populations (e.g. fishing and hunting) may be masking the effect of sea grass loss on higher trophic level species. This does not imply that the loss of sea grass beds in Langebaan is not concerning, as site specific changes in associated macrofauna and at least one wader species were clearly documented by Pillay *et al.* (2010). Also important to note, is the fact that the Terek Sandpiper is a summer migratory bird and its decline is occurring globally (see Chapter 12). However, continued loss of sea grasses could cause a “tipping point” beyond which major ecosystem changes would occur throughout the lagoon.

The loss of seagrass beds from Langebaan Lagoon is a strong indicator that the ecosystem is undergoing a shift, most likely due to anthropogenic disturbances. Additionally, several studies have highlighted the potential for climate driven changes in water temperature and pH to alter seagrass physiology and possibly their distribution and abundance (Duarte 2002, Mead *et al.* 2013). However, information on the temperature and pH tolerance of South African seagrasses is currently lacking and warrants investigation. It is critical that this habitat and the communities associated with it be monitored in future as further reductions are certain to have long term implications, not only for the invertebrate fauna but also for species of higher trophic levels.

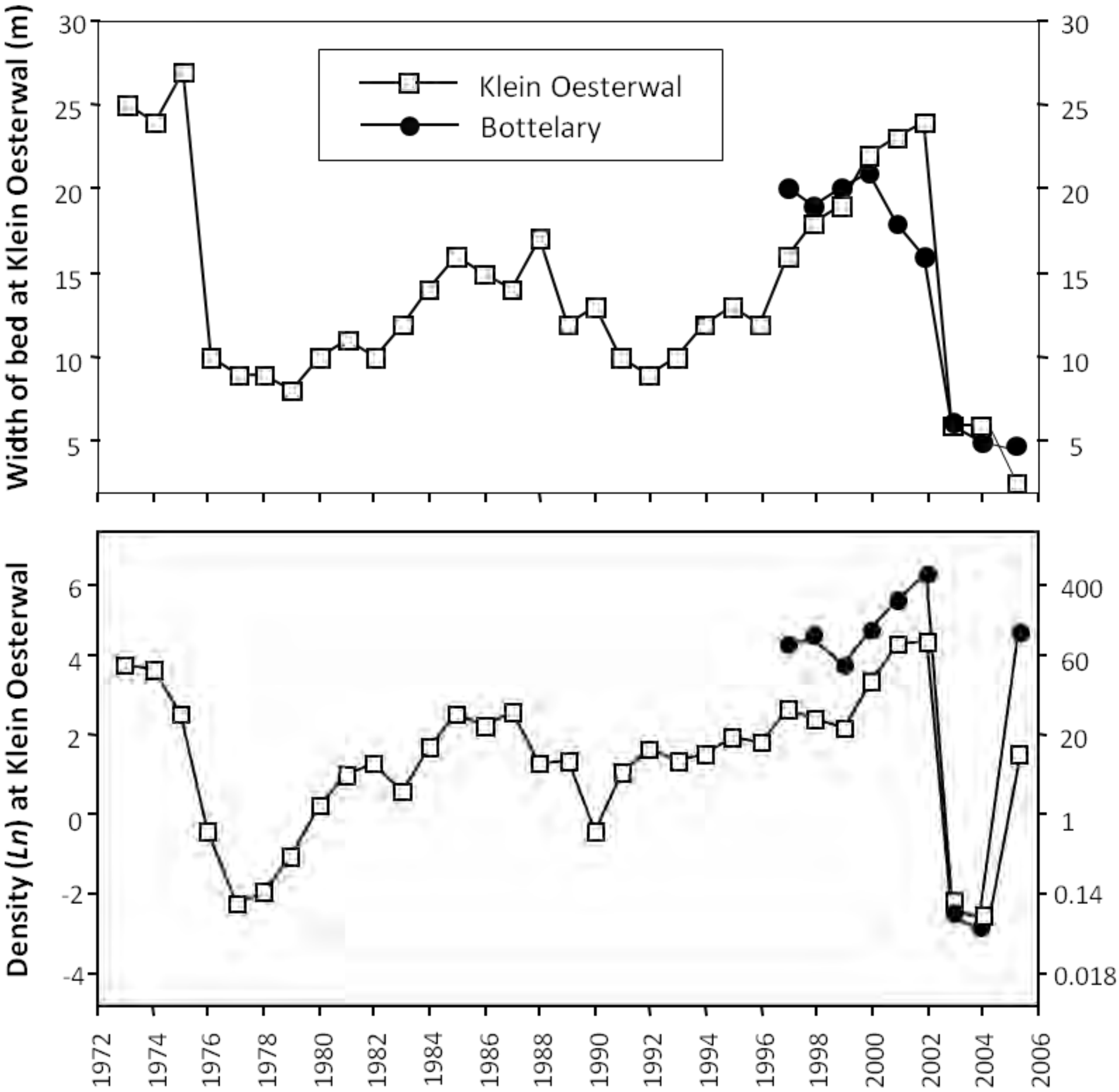


Figure 8.2. **Width of the *Zostera* beds and density of *Siphonia* at Klein Oesterwal and Bottelary in Langebaan Lagoon, 1972-2006. Source: Angel *et al. (*2006).**

8.3 Long term changes in saltmarshes in Langebaan Lagoon

Saltmarshes in Langebaan are an important habitat and breeding ground for a range of fish, bird and invertebrate species (Christie 1981, Day 1981, Gericke 2008). Langebaan Lagoon incorporates the second largest salt marsh area in South Africa, accounting for approximately 30% of this habitat type in the country, being second only to that in the Knysna estuary (Adams *et al.* 1999).

Long term changes in salt marshes in Langebaan Lagoon were investigated by Gericke (2008) using aerial photographs taken in 1960, 1968, 1977, 1988 and 2000. He found that overall saltmarsh area had shrunk by only a small amount between 1960 and 2000, losing on average 8 000 m^2 per annum. Total loss during this period was estimated at 325 000 m^2, or 8% of the total (Figure 8.3.). Most of this loss has been from the smaller patches of salt marsh that existed on the seaward edge of the main marsh. This is clearly evident from the change in the number of saltmarsh patches in the lagoon over time, which has declined from between 20 and 30 in the 1960s and 70s, to less than 10 in 2000. Gericke (2008) attributed the observed change over time to increases in sea level that would have drown the seaward edges of the marshes or possibly reduced sediment inputs from the terrestrial edge (i.e. reduced input of windblown sand due to stabilization by alien vegetation and development).

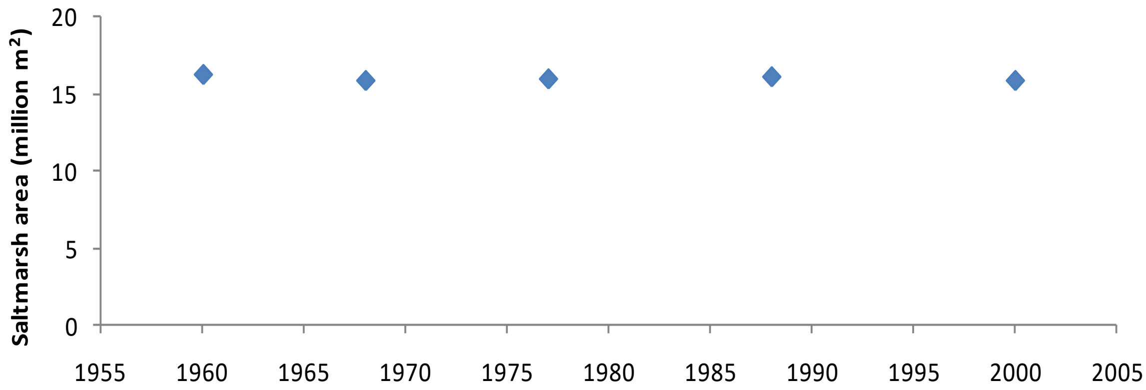


Figure 8.3. **Change in saltmarsh area over time in Langebaan Lagoon. (Data from Gerricke 2008).**

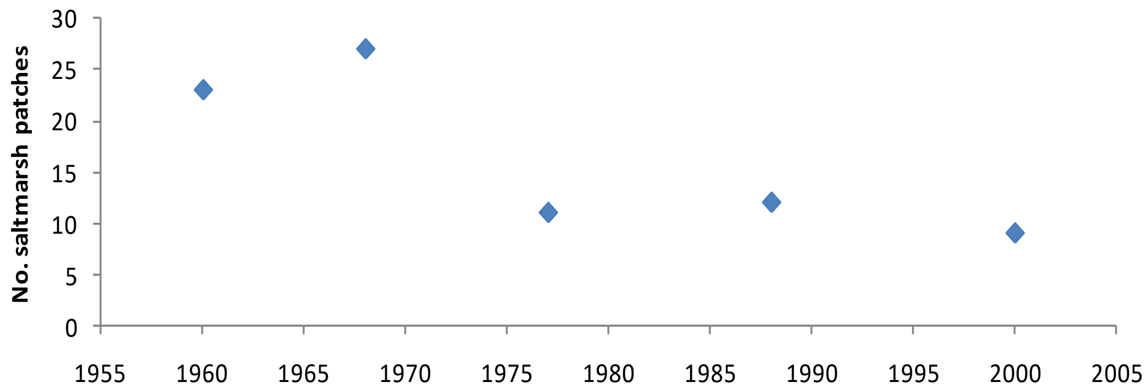


Figure 8.4. **Change in the number of discrete saltmarsh patches over time in Langebaan Lagoon. (Data from Gerricke 2008).**

8.4 Long term changes in reed and sedge communities surrounding Langebaan Lagoon

Recently, concern has been voiced about potential impacts that the Elandsfontein Exploration and Mining (Pty) Ltd (EMM) phosphate mine at Elandsfontein may have on groundwater quality and flows to Langebaan Lagoon. Due to the porous nature of the surrounding sediments and the arid conditions, Langebaan lagoon is not fed by overland streams or rivers and it has been suggested that groundwater plays a significant role in sustaining the marsh ecosystems at the head of the lagoon (Valiela et al. 1990; Burnett et al. 2001). Diagnostic plants indicate significant contributions of groundwater (Adams & Bate 1999). For example, reeds (*Phragmites australis* and *Typha capensis*) occur at discrete points on the shoreline surrounding Langebaan lagoon (Figure 8.1.) These plants can only survive in water or at least damp soil and are only able to tolerate salinity levels up to a maximum of 20-25 ppt (Adams & Bate 1999, Nondoda 2012). The salinity of the water in the lagoon is generally the same (or occasionally higher) than that of seawater – i.e. 35 ppt, and these species' are only found at sites where freshwater is seeping into the lagoon (i.e. the main groundwater input sites in the south east of the lagoon along the shoreline at Geelbek). The fauna and flora in the Lagoon are mostly marine and estuarine in nature, and while some are euryhaline and are able to tolerate salinity (salt) levels anywhere between fresh water (i.e. 0 parts per thousand) and normal seawater (35 parts per thousand), most species are not tolerant of salinities in excess 35ppt.

Reducing freshwater inflow into Langebaan Lagoon that may result from the mining activities could result in the development of more extreme hypersaline conditions in the upper lagoon, killing flora and fauna sensitive to salinities in excess of normal seawater. To mitigate impacts on groundwater flow, it was suggested that the extracted water is injected back into the aquifer system via boreholes downstream of the mining site. This mining method is predicted to use only a small proportion of the extracted water for mining and processing and thus have little to no impact on the marsh habitat at Geelbek (Conrad 2014).

While it has been established from a groundwater assessment undertaken by Conrad (2014) that the proposed mining operations are highly unlikely to have any impact on the groundwater quality and flow (see Chapter 5 for more details on this), Kropz Elandsfontein have opted to take a precautionary approach and carefully monitor any potential impacts on Langebaan Lagoon in association with the Saldanha Bay Water Quality Forum Trust (SBWQFT). The State of the Bay monitoring activities undertaken by the SBWQFT have thus been expanded to incorporate monitoring of various biological and physico-chemical variables to establish an appropriate baseline against which any potential future changes in the Lagoon can be benchmarked. This includes monitoring of water level, temperature, salinity (Chapter 5), reeds and sedges (this chapter) and biota (Chapter 9) at the top of the lagoon.

Satellite and aerial image data offer a unique opportunity to identify areas where change in surface properties can be mapped and linked to land condition variability. Within the last decade, these efforts have increased as resources become more readily available through open-source databases and catalogues. Taking advantage of these new developments, a framework has been created as part of the State of the Bay monitoring programme to assess and visualize spatial variability in vegetation communities surrounding Langebaan Lagoon using an open-source geospatial platform called Google Earth Engine (GEE). GEE is a cloud-based geospatial processing platform centred on processing satellite imagery and derivates. The platform is often applied in global- or regional-scale environmental monitoring and analysis efforts, especially where large quantities of data and/or over long time periods are required. This web-based platform provides access to publicly available remote sensing imagery, as well as high-speed parallel processing and machine learning algorithms using Google's computational infrastructure. Within this infrastructure, a library of Application Programming Interfaces (APIs) can be utilized and modified for a multitude of environmental analysis (Tamiminia *et al.*, 2020).

In this study, we present a novel approach to assess changes in the common reed *P. australis* cover variability at the pixel level, at a nominal spatial resolution of 10-30 m (high to moderate). We illustrate our approach using over three decades of Landsat 5,7 and 8 as well as Sentinel-2 satellite imagery (1989 to 2020). In future it is our intention to follow this up with ground-truthing exercises using field observations to validate the classification results, make refinements and to develop an accuracy assessment.

The common reed *Phragmites australis* dominates the flora of the reedbeds where groundwater inflow occurs. At Langebaan Lagoon this is predominantly along the southern and south-eastern shores, near Geelbek. We needed to refine our study area in which to conduct analyses and thus used the vegetation cover shapefile developed by van der Linden (2013) of nine habitat types in the area around Langebaan Lagoon prepared using aerial photography. We excluded permanent water (and derived transition classes) from this by eliminating permanent water using an existing method for dynamic reference cover with the JRC the Global Surface Water (GSW) Metadata v1.2(Pekel *et al.*, 2016) from 1984-2015. Once a stable permanent water class (i.e. permanent water throughout the period) was removed we could focus effort only on near-shore vegetation cover changes. Sentinel-2 MSI: Multispectral Instrument Level -2A (Sentinel-2) imagery for 2015-2020 were temporally aggregated into seasonal composites of Summer (September-March), and Winter (March-September) using the colour infrared band combination (8,4,3) which emphasises different vegetation spectral signatures. Landsat 8 (2013 and 2014), 7 (1999 to 2012) and 5 (1989 to 1998) imagery were temporally aggregated into annual composites using the same band combination. All temporal aggregation was done by determining the medoid of a season or a year of the imagery, creating a specific data point instead of an averaged or blended value (Flood, 2013), utilising only images with cloud cover of 10% or less. The end result creates a regular temporal sequence by minimizing missing data and cloud contaminations.

Once the temporal imagery sequence was generated, we undertook an unsupervised (computer-based automated rather than user-influenced and defined) classification approach. We used a method known as "clustering" which falls under the category of statistical machine learning. This can be defined as a case where a statistical relationship is established between the spectral bands or frequencies used and the variable measured (field-based) without there necessarily being a causal relationship (Holloway & Mengersen, 2018). Clustering is an unsupervised learning method that attempts to combine objects into clusters based on similarity criteria of input variables without training data (Holloway & Mengersen, 2018). We specified ten groups (clusters) and assume that a permanent water class is excluded from this cluster assignment due to the methodology mentioned previously. Per temporal aggregation, each clustering effort was then exported from GEE into GeoTIFF (.tiff) format and overlaid on the respective imagery composites within ArcGIS.

Figure 8.5. Reeds and sedges as classified by van der Linden 2013 at Langebaan Lagoon (Source: Shapefiles provided by van der Linden 2013).

As the classification was unsupervised, we needed to ascertain whether the method was reliable for determining reeds as a vegetation class that was sufficiently distinctive from other vegetation and land cover types. For the post-processing classification, we extracted the 'Reeds and Sedges' multi-polygon (hereafter reeds shapefile) from the habitat structure shapefile as a bounding area for reeds. Since the "reeds" shapefile was published in 2013, we used the 2013 Landsat image to gauge the most appropriate value for reeds. All .tiffs were reprojected into a projected co-ordinate system. Value 2 had the highest overall percentage match to the reeds shapefile with a mean 71% match for the polygons comprising 75% of the total area (including a mean of 93% for the three largest polygons making up 31.2% of the total area, Figure 8.6). We can also presume that since some of the shapefile is comprised of sedges, that the remaining percentage may reflect these as they could have a different spectral classes, however, ground-truthing is needed to verify this. For the full dissolved shapefile, value 2 comprised 70% of the total area and was thus deemed suitable to classify this value as reeds. This value did not stay consistent across each year and we needed to manually verify the corresponding value for other years. Where not value 2, the corresponding values were 3 (in 2014), 5 (in 2000) and 1 (1997, 2002, 2003, 2005 and 2006).

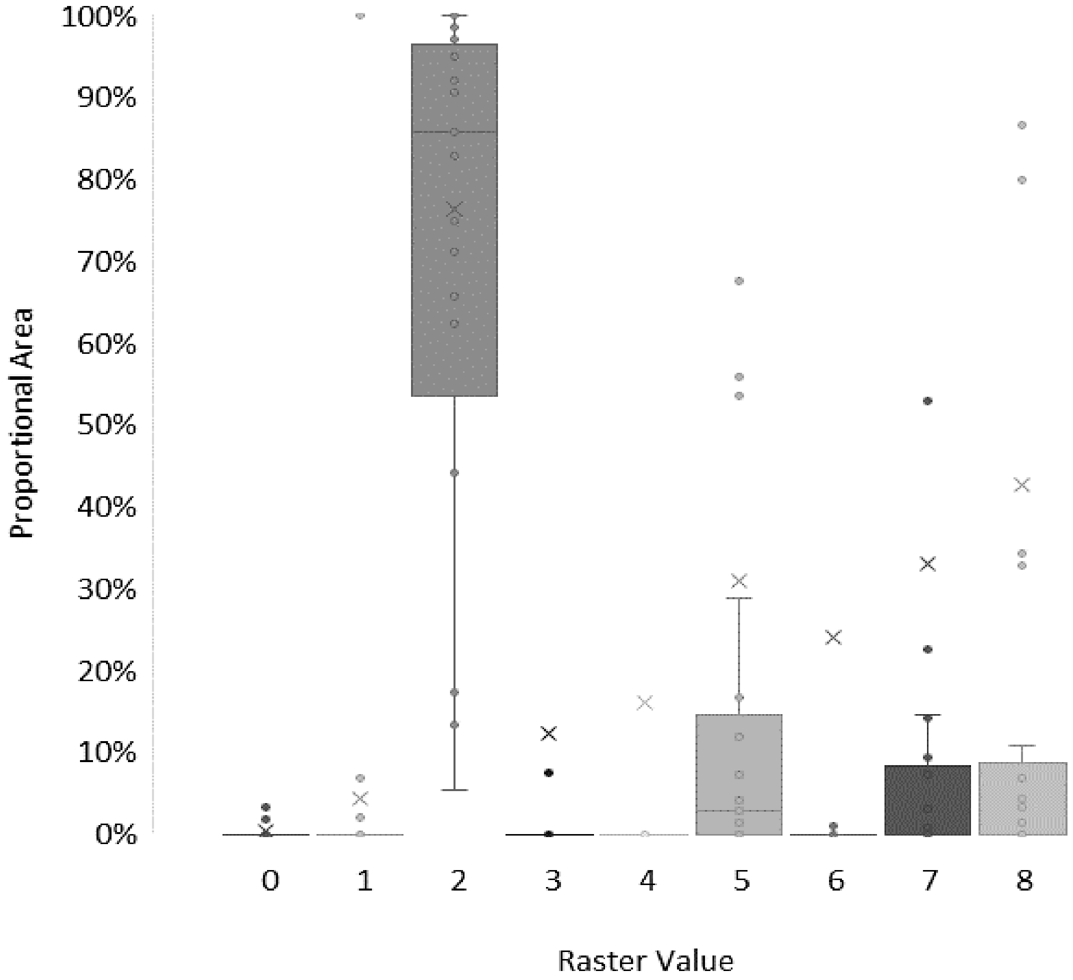


Figure 8.6. **Box plot showing the spread of proportional areas per raster value for the 2013 Landsat image within the reeds shapefile, for the largest polygons comprising the top 75% of the total reeds shapefile area.**

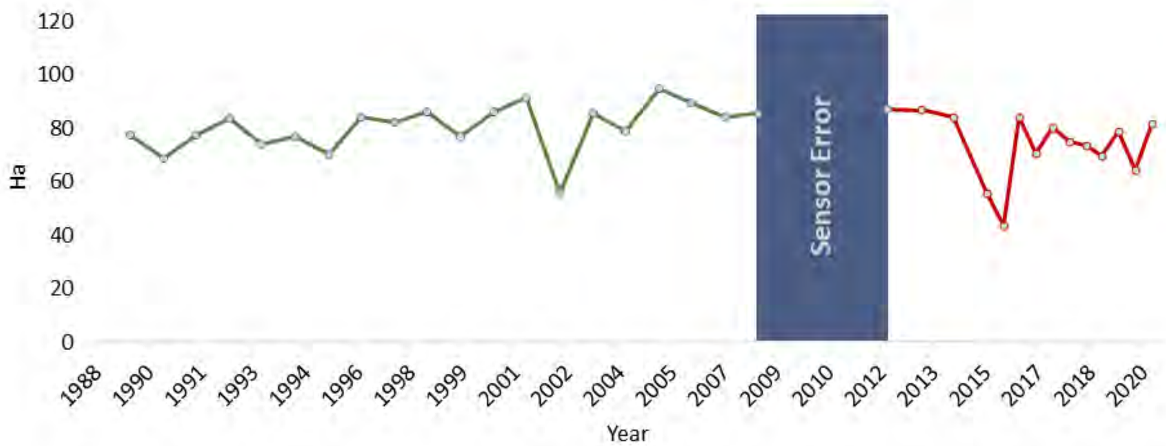


Figure 8.7. ***Phragmites australis*** **trends between 1989 and 2020 as per the unsupervised classification. Green indicates annual Landsat imagery values, red indicates biannual Sentinel-2 imagery.**

Results of our analysis suggests that variation in reed cover over time is relatively modest and that this has remained more or less constant over the last 31 years (1989-2020, Figure 8.7). The biggest perturbations in reed cover correspond with the two largest droughts that have been experienced in the region in this period (a 1:20 year event that occurred in the period 2002-2003) and an even bigger drought that occurred recently (a 1:100 year event in the period 2015-2017). Also of interest is the fact that the extent of reed cover around Langebaan Lagoon as indicated by the satellite data is significantly greater than that inferred from aerial photography (i.e. as delineated by van der Linden 2013, Figure 8.8 - Figure 8.10).

Future efforts in this field will entail expanding this assessment to other vegetation classes (specifically seagrass and salt marsh), assessing the level of change in each vegetation class over time, and also ground truthing of each mapped vegetation class.

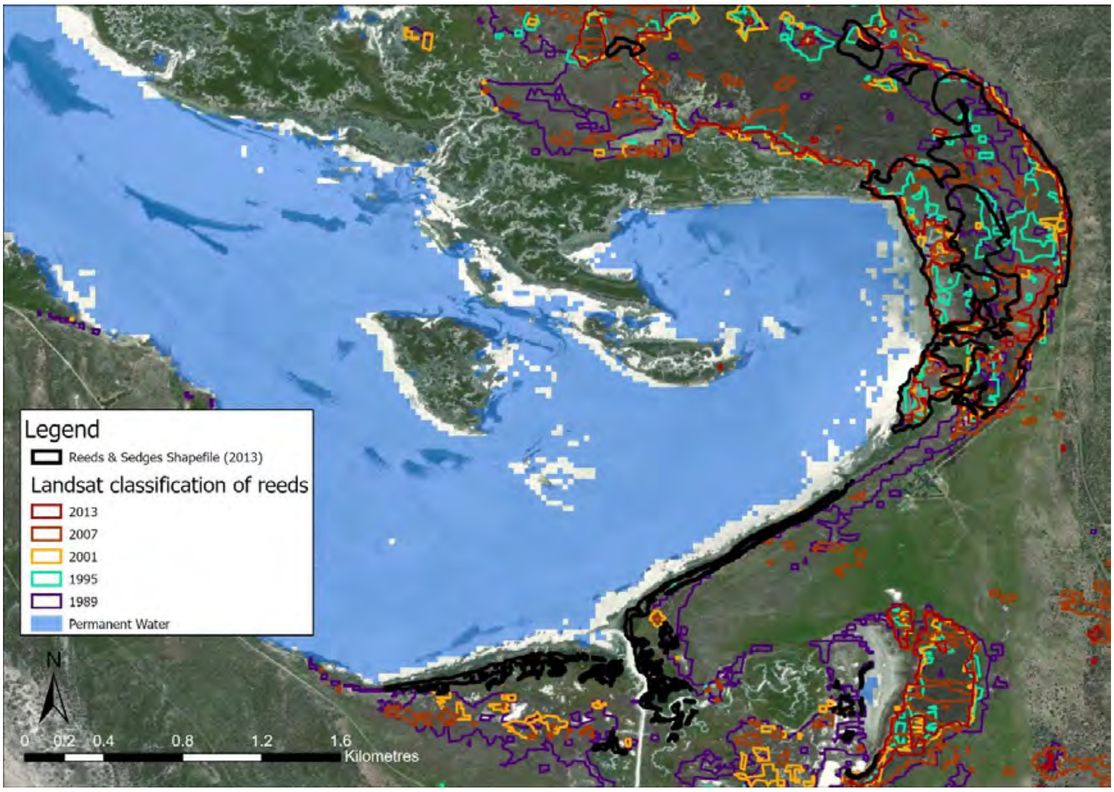


Figure 8.8. ***Phragmites australis*** **delineated bounds between 1989 and 2013 as per the unsupervised classification effort on Landsat imagery from GEE.**

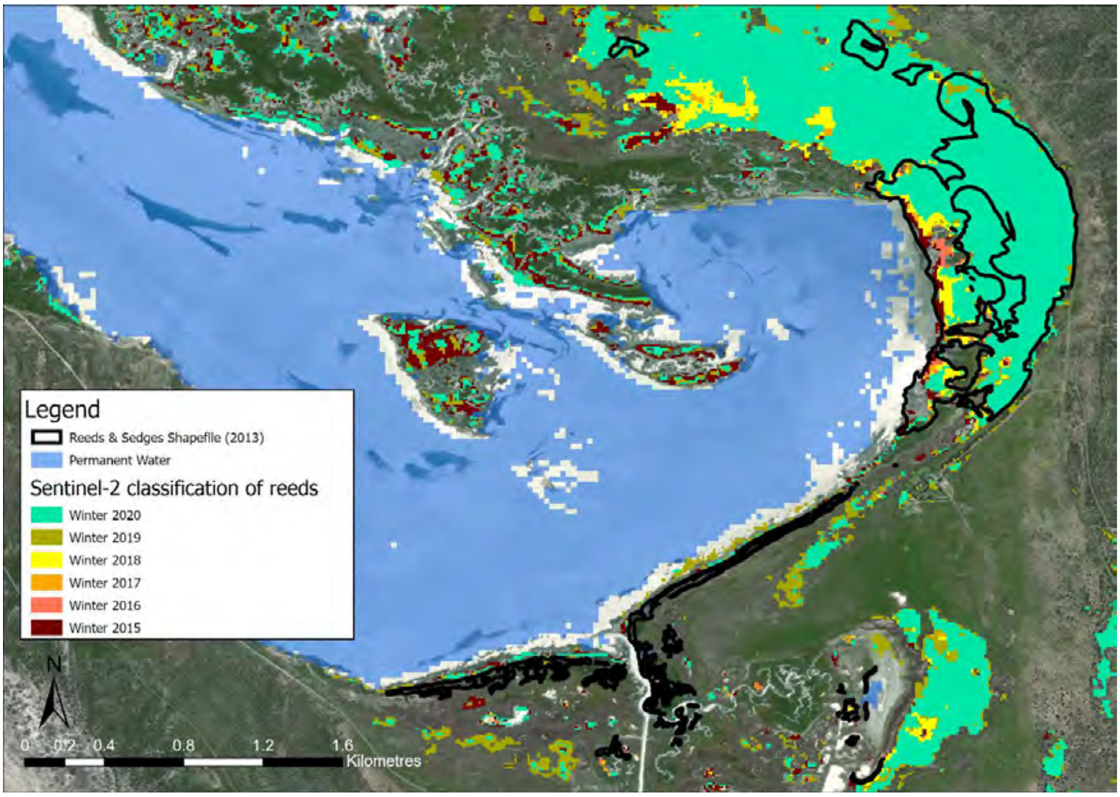


Figure 8.9. ***Phragmites australis*** **delineated area between 2015 and 2020 as per the unsupervised classification effort on Sentinel-2 imagery from GEE. The areas are overlapped with the most recent showing on top.**

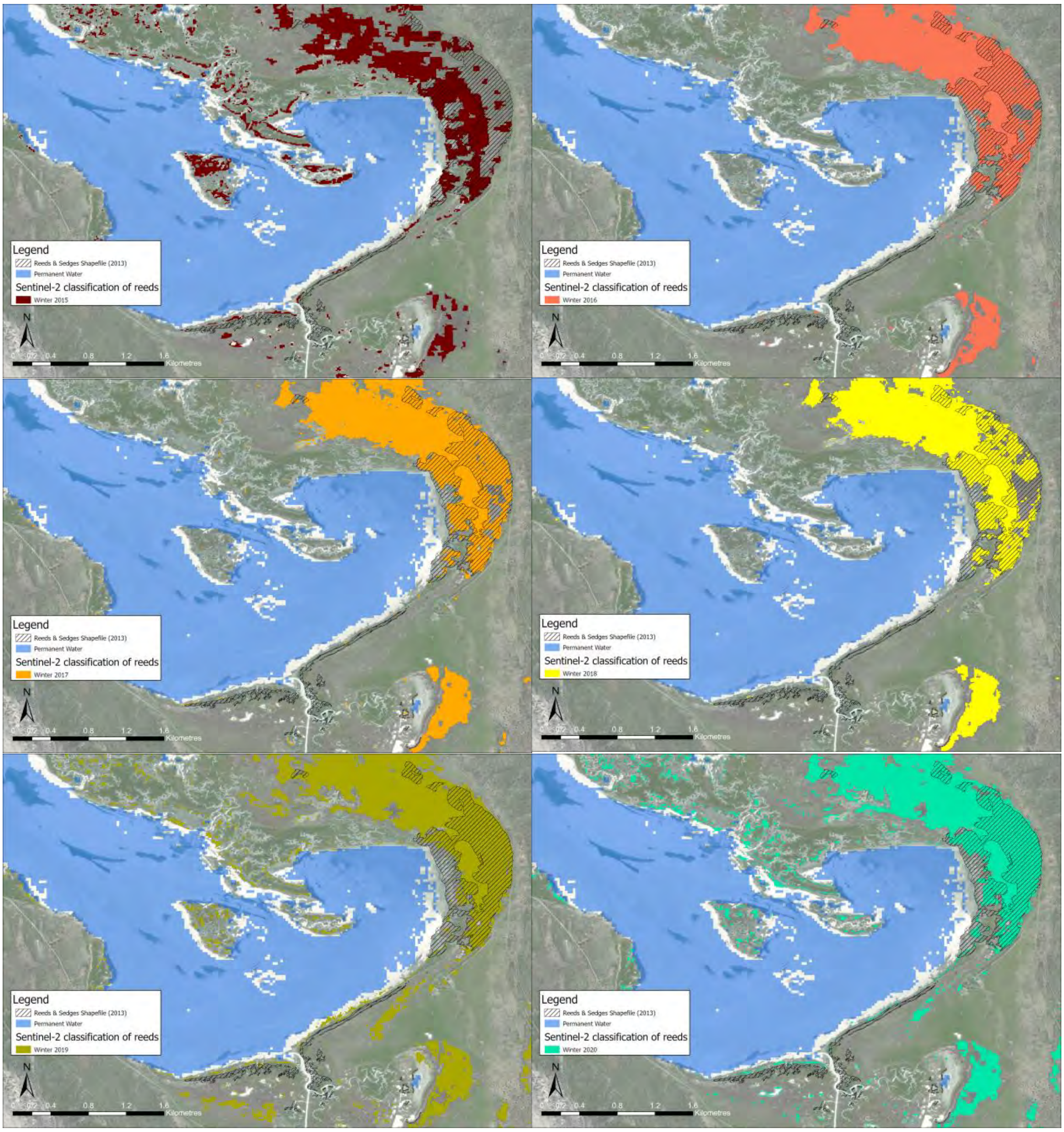


Figure 8.10. Area of the value representing the highest match with the reeds on the 2013 shapefile based on the classification of the spectral signature using Sentinel-2 imagery.

9 BENTHIC MACROFAUNA

9.1 Background

It is important to monitor biological components of the ecosystem in addition to physico-chemical and eco-toxicological variables, as biological indicators provide a direct measure of the state of the ecosystem at a selected point in space and time. Benthic macrofauna are the biotic component most frequently monitored to detect changes in the health of the marine environment. This is largely because these species are short-lived and, as a consequence, their community composition responds rapidly to environmental changes (Warwick 1993). Given that they are also relatively non-mobile (as compared with fish and birds) they tend to be directly affected by pollution and they are easy to sample quantitatively (Warwick 1993). Furthermore, they are scientifically well-studied compared with other sediment-dwelling components (e.g. meiofauna and microfauna), and taxonomic keys are available for most groups. In addition, benthic community responses to a number of anthropogenic influences have been well documented.

Organic matter is one of the most universal pollutants affecting marine life and it can lead to significant changes in community composition and abundance, particularly in semi-enclosed or closed bays where water circulation is restricted, such as Saldanha Bay. High organic loading typically leads to eutrophication, which can lead to a range of different community responses amongst the benthic macrofauna. These include increased growth rates, disappearance of species due to anoxia, changes in community composition and reduction in the number of species following repeat hypoxia and even complete disappearance of benthic organisms in severely eutrophic and anoxic sediments (Warwick 1993). The community composition of benthic macrofauna is also likely to be impacted by increased levels of other contaminants such as trace metals and hydrocarbons in the sediments. Furthermore, areas that are frequently disturbed by mechanical means (e.g. through dredging) are likely to be inhabited by a greater proportion of opportunistic pioneer species as opposed to larger, longer lived species.

The main aim of monitoring the health of an area is to detect the effects of stress, as well as to monitor recovery after an environmental perturbation. There are numerous indices, based on benthic invertebrate fauna information, which can be used to reveal conditions and trends in the state of ecosystems. These indices include those based on community composition, diversity and species abundance and biomass. Given the complexity inherent in environmental assessment it is recommended that several indices be used (Salas *et al.* 2006).

The community composition, diversity, abundance and biomass of soft bottom benthic macrofauna samples, collected in Saldanha Bay from 1999 to 2020 (with additional sites at Elandsfontein), are considered in this report.

9.2 Historic data on benthic macrofauna communities in Saldanha Bay

The oldest records of benthic macrofauna species occurring in Saldanha Bay date back to the 1940s, prior to the construction of the iron ore terminal and Marcus Island causeway. Due to differences in sampling methodology, data from these past studies are not directly comparable with subsequent studies and as such cannot be used for establishing conditions in the environment prior to any of the major developments that occurred in the Bay. Moldan (1978) conducted a study in 1975 where the effects of dredging in Saldanha Bay on the benthic macrofauna were evaluated. Unfortunately, this study only provided benthic macrofauna data after the majority of Saldanha Bay (Small Bay and Big Bay) had been dredged. A similar study conducted by Christie and Moldan (1977) in 1975 examined the benthic macrofauna in Langebaan Lagoon, using a diver-operated suction hose, and the results thereof provide a useful description of baseline conditions present in the Lagoon from this time.

Studies conducted in the period 1975-1990, examined the benthic macrofauna communities of Saldanha Bay and/or Langebaan Lagoon, but are also, regrettably not comparable with any of the earlier or even the more recent studies. Recent studies conducted by the Council for Scientific and Industrial Research (CSIR) in 1999 (Bickerton 1999) and Anchor Environmental Consultants in 2004 and 2008-2019 do, however, provide benthic macrofauna data from Saldanha Bay and Langebaan Lagoon that are comparable with those collected in recent years. Direct comparisons to earlier studies are complicated owing to the fact that different equipment was used in the earlier surveys than those undertaken from 1999 to present. The 1975 study, for example, made use of a modified van Veen grab weighted to 20 kg which sampled an area of 0.2 m^2 from the surface fraction of sediment. Subsequent surveys, from 1999 to present, made use of a diver-operated suction sampler with a sampling area of 0.24 m^2 to a depth of 30 cm. The former sampling technique (van Veen grab) would be expected to sample a smaller proportion of benthic macrofauna due to its limited ability to penetrate the sediment beyond the surface layers. The suction sampler is effective in penetrating to a depth of 30 cm, which is within range of larger species such as prawns and crabs. The study conducted in 1975 in Langebaan Lagoon (Christie and Moldan 1977), and those conducted for all State of the Bay surveys have all made use of a diver-operated suction sampler which sampled an area of 0.24 m^2. However, in 1975 a depth of 60 cm was sampled while in surveys since 2004 a depth of only 30 cm has been sampled. Thus, considering the differences in sampling techniques employed, it is likely that the changes reflected by the data between the 1975 and 1999-2008 in Saldanha Bay and Langebaan Lagoon are a function both of real changes that occurred in the Bay and an artefact of differences in sampling methodology. The location of sites sampled during 1975 and the 1999-2019 studies also differed (refer to previous versions of this report), however, the broad distribution of sites throughout the sampling area ensures that the data collected are representative of the study areas concerned and as such, can be compared with one another.

9.3 Approach and methods used in monitoring benthic macrofauna in 2020

9.3.1 Sampling

Benthic macrofauna have been sampled at more than 30 sites in Big Bay (nine sites), Small Bay (ten sites) and Langebaan Lagoon (nine sites) since the inception of the State of the Bay monitoring programme in 2004. This year, a further eight sites were sampled in the vicinity of the Sea Harvest Corporation discharge pipe – a requirement in terms of their updated Coastal Waters Discharge Permit (CWDP - Permit Reference Number: 2012/025/WC/Sea Harvest) issued by the Department of Environmental Affairs: Oceans and Coasts. The permit conditions stipulate that the Sea Harvest environmental monitoring samples be collected at the same time as the State of the Bay samples (using the same technique) such that the latter can serve as control points against which potential impacts of the Sea Harvest outfall can be benchmarked. The findings from these additional Sea Harvest sites are presented here together with the "traditional" suite of State of the Bay monitoring sites.

The localities and water depth ranges of the 2020 sampling sites are illustrated in Chapter 7. Samples are, by convention, collected using a diver-operated suction sampler, which sampled an area of 0.08 m^2 to a depth of 30 cm and retained benthic macrofauna in a 1 mm mesh sieve bag. Three samples are taken at each site and pooled, resulting in a total sampling surface area of 0.24 m^2 per site. Three hand-core samples were taken at sites less than 2 m deep, totalling a sampling surface area of 0.08 m^2. In 2016 and 2017 Elandsfontein samples were collected using the same hand-core. It was later agreed that the use of a Van Veen grab to collect samples at Elandsfontein was most appropriate and from 2018 onwards a Van Veen grab with a bite size of 0.092 m^2 was used. It was noted that the grab was more effective at sampling benthic macrofauna in this area, and we recommend this be continued for future monitoring. All macrofauna abundance and biomass data were ultimately standardised per unit area (m^2). Samples were stored in plastic bottles and preserved with 5% formalin.

In the laboratory, samples were rinsed of formalin and stained with Rose Bengal to aid sorting of biological from non-biological matter. All fauna were removed and preserved in 1% phenoxetol (Ethyleneglycolmonophenylether) solution. The macrofauna were then identified to species level where possible, but at least to family level in all instances. The validity of each species was then checked on The World Register of Marine Species (WoRMS, www.marinespecies.org). The biomass (blotted wet mass to four decimal places) and abundance of each species was recorded for each sample.

9.3.2 Statistical analysis

The data collected from this survey were used for two purposes 1) to assess spatial variability in the benthic macrofauna community structure and composition between sites in 2020 and 2) to assess changes in benthic community structure over time (i.e. in relation to past surveys). Both the spatial and temporal assessments are necessary to provide a good indication of the current state of health of the Bay.

9.3.2.1 Community structure and composition

Changes in benthic species composition can be the first indicator of disturbance, as certain species are more sensitive (i.e. likely to decrease in abundance in response to stress) while others are more tolerant of adverse conditions (and may increase in abundance in response to stress, taking up space or resources vacated by the more sensitive species). Monitoring the temporal variation in community composition also provides an indication of the rate of recovery of the ecosystem following disturbances in different areas of the system. This allows one to more accurately predict the impacts of proposed activities. “Recovery” following environmental disturbance is generally defined as the establishment of a successional community of species which progresses towards a community that is similar in species composition, density and biomass to that previously present (C-CORE 1996 and Newell 1998). The rate of recovery is dependent on environmental conditions and the communities supported by such conditions. Given the spatial variation in environmental conditions (largely influenced by depth and exposure) and anthropogenic disturbance throughout Saldanha Bay and Langebaan Lagoon, it is expected that recovery will vary throughout system.

It has been shown that species with a high fecundity, rapid growth rates and short life-cycles are able to rapidly invade and colonise disturbed areas (Newell 1998). These species are known as “r-strategists”, pioneer or opportunistic species and their presence generally indicates unpredictable short-term variations in environmental conditions as a result of either natural factors or anthropogenic activities. In stable environments, the community composition is controlled predominantly by biological interactions rather than by fluctuations in environmental conditions. Species found in these conditions are known as “K-strategists” and are selected for their competitive ability. K-strategists are characterised by long life-spans, larger body sizes, delayed reproduction and low natural mortality rates. Intermediate communities with different relative proportions of opportunistic species and K-strategists are likely to exist between the extremes of stable and unstable environments.

The statistical program, PRIMER 6 (Clarke and Warwick 1993), was used to analyse benthic macrofauna abundance data. Data were root-root (fourth root) transformed and converted to a similarity matrix using the Bray-Curtis similarity coefficient. Multidimensional Scaling (MDS) plots were constructed in order to find ‘natural groupings’ between sites for the spatial assessment and between years for the temporal assessment. SIMPER analysis was used to identify species principally responsible for the clustering of samples. These results were used to characterise different regions of the system based on the communities present at the sites. It is important to remember that the community composition is a reflection of not only the physico-chemical health of the environment but also the ability of communities to recover from disturbance.

9.3.2.2 Diversity indices

Diversity indices provide a measure of diversity, i.e. the way in which the total number of individuals is divided up among different species. Understanding changes in benthic diversity is important because increasing levels of environmental stress generally decreases diversity. Two different aspects of community structure contribute to community diversity, namely species richness and equability (evenness). Species richness refers to the total number of species present while equability or evenness expresses how evenly the individuals are distributed among different species. A sample with greater evenness is considered to be more diverse. It is important to note when interpreting diversity values that predation, competition and disturbance all play a role in shaping a community. For this reason, it is important to consider physical parameters as well as other biotic indices when drawing a conclusion from a diversity index.

The *Shannon-Weiner diversity index* (H') was calculated for each sampling location using PRIMER V 6:

H' = - Σipi(log pi) [11]

The diversity (H') value for each site was plotted geographically and this was used to interpolate vales for the entire system using ArcGIS in order to reveal any spatial patterns. Alpha diversity (total number of species) was also then calculated for the pre-designated locations for past surveys from 1999 to present: Small Bay, Big Bay, Langebaan Lagoon and Elandsfontein.

9.4 Benthic macrofauna 2020 survey results

9.4.1 Species diversity

Variation in species diversity (represented by the Shannon Weiner Index, H') is presented in Figure 9.1. Diversity was highest in Langebaan Lagoon and Big Bay (at sites LL 31, LL 37 and BB 30) and was lowest in Small Bay near the Sea Harvest discharge pipe (SH 2) where a value of zero was recorded. Other sites where low diversity was recorded include BB 26, SH 8 and SB 10. These findings are most likely attributable to the high levels of anthropogenic disturbance (e.g. dredging) and the presence of elevated levels of contaminants (trace metals, organic material, etc.) in the sediment (mud) collected at these sites. It is well known that high levels of disturbance associated with pollution can allow a small number of opportunistic, short-lived or r-selected species to colonize the affected area and prevent a more diverse community comprising longer living k-strategist species from becoming established.

9.4.2 Community structure

An ordination plot, prepared from 2020 macrofaunal abundance data, is presented in Figure 9.2. These data show a very similar pattern as for the diversity data, with the macrofaunal communities present at the Langebaan Lagoon and particularly the Elandsfontein sites (near the head of the lagoon) standing out as being clearly different to those in Big Bay and Small Bay. The sampling sites in Big Bay and Small Bay are also distinct from one another, but to a lesser extent than those in the lagoon. The Sea Harvest sites are grouped together with the Small Bay sites. Upon closer inspection, sites within Small Bay itself also show some spatial grouping of their own with sites in the northern reaches of the bay forming a separate cluster that was separate to those further south. This observation is a function of differences in community structure (i.e. the abundance or presence/absence of different species at each site) and not just the total number of species present at a particular site. "Sensitive" species that

[11] Where p_i is the proportion of the total count arising from the *i*th species. This is the most commonly used diversity measure and it incorporates both species richness and equability.

cannot tolerate high levels of anthropogenic disturbance are present in abundance at Elandsfontein and in Langebaan Lagoon but are largely absent from the Big Bay sites, Sea Harvest sites and the southern Small Bay sites in proximity the iron ore terminal. It should be noted that differences in macrofaunal community structure are also partly explained by the physical and environmental parameters present at each site (i.e. freshwater ingress, tidal currents, sediment granulometry and depth).

The "hardier" filter feeders such as *Upogebia capensis* are, for example, abundant in both Big Bay and Small Bay samples, but the "more sensitive" filter feeders such as the amphipods *Ampelisca spinimana* and *A. anomala*, the mollusc *Macoma odinaria* and the polychaete *Sabellides luderitzi* were notably more abundant in Big Bay than Small Bay. Similarly, the sea pen *Virgularia schultzei,* widely regarded *as a "*sensitive species*"* was found only in Big Bay.

The relationship between 2020 macrofaunal abundance data and abiotic data (sediment grain size fractions, TOC, TON and trace metals) was investigated using a Distance Based Linear Model (DistLM) (Anderson *et al.* 2008). A sequential test revealed that a combination of all input variables explained ~64% of the variation observed in the macrofaunal abundance data, with Manganese (16%) explaining the greatest amount, followed by mud (7%), Nickel (7%), gravel (5%) and sand (5%). Results from the analysis of sediments for trace heavy metals (Chapter 7) indicate that the concentration of manganese is elevated at sites surrounding the iron ore jetty (SB 9, SB 14, SB 15 and BB 22) and yacht club basin (SB 1). This is coupled with the initiation and subsequent year-on-year increase of manganese exports since 2014 (Chapter 3 and Chapter 6) and may serve to explain the observed influence of manganese on benthic macrofaunal community structure.

The full model can be visualised by examining the distance-based redundancy analysis (dbRDA) ordination (Figure 9.3). The first two axes capture 54% of the variability in the fitted model, and 35% of the total variation in the data cloud. The blue lines in the dbRDA plot are category vectors, whereby the length of the vectors is a measure of the strength of the relationship between that category and the axes. The Manganese concentration and mud fraction clearly separated the Langebaan Lagoon and Elandsfontein sites from the Big Bay and Small Bay sites.

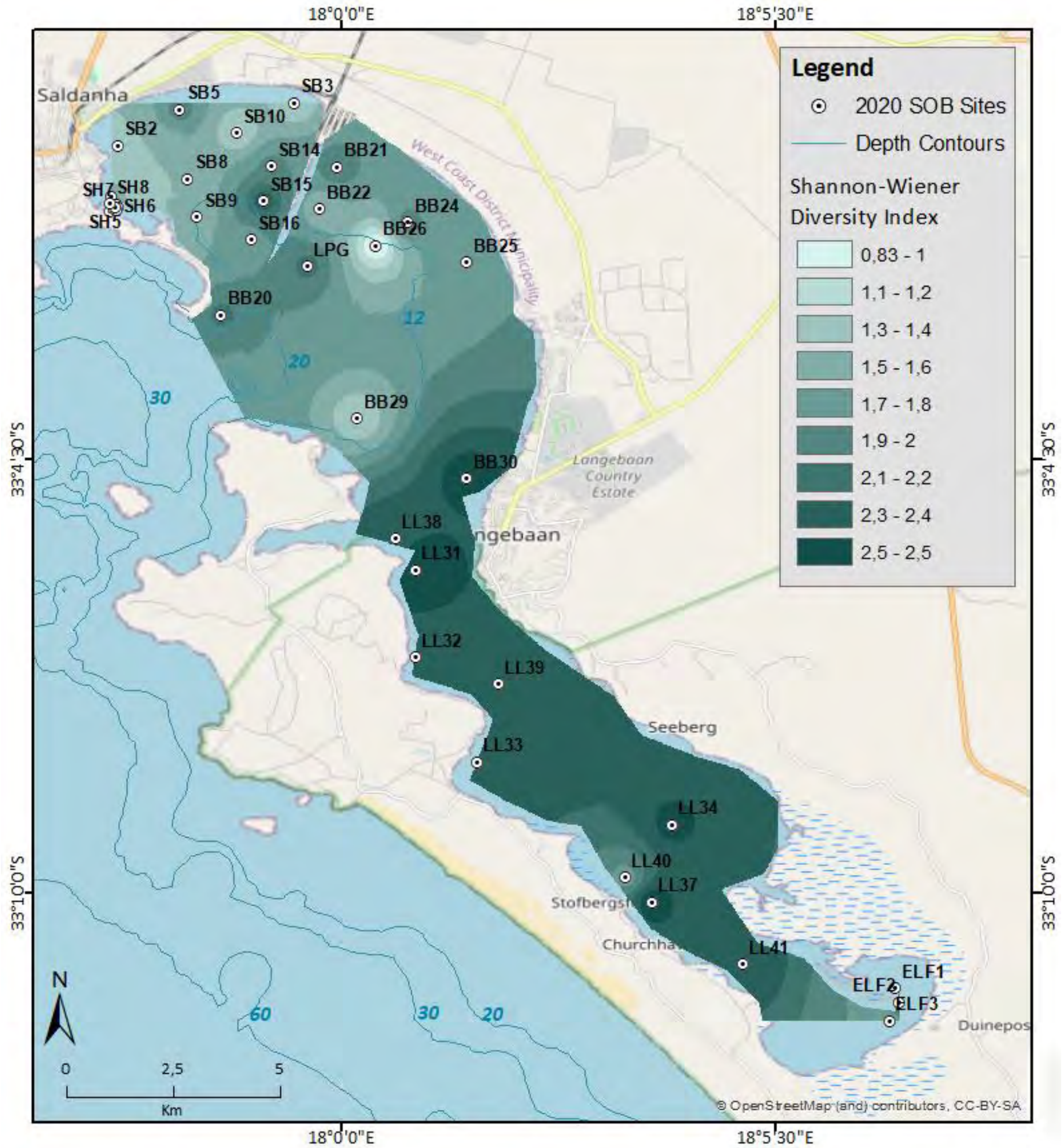


Figure 9.1. **Variation in the diversity of the benthic macrofauna in Saldanha Bay and Langebaan Lagoon as indicated by the 2020 survey results (H' = 0 indicates low diversity, H' = 2.8 indicates high diversity).**

Species that contributed significanty to the dissimilarity between the Saldanha Bay and Langebaan Lagoon samples include the filter feeding amphipods *Ampelisca* sp. and the predatory whelks *Nassarius* sp. that were relatively abundant in Small Bay and Big Bay, but either rare or absent from lagoon samples. Other species such as the sand prawn *Callichirus kraussi*, the isopod *Natatolana hirtipes*, the crown crab *Hymenosoma orbiculare* and polychaete *Orbinia angrapequensis* (detritivores, scavengers or predators) were more abundant in the lagoon samples.

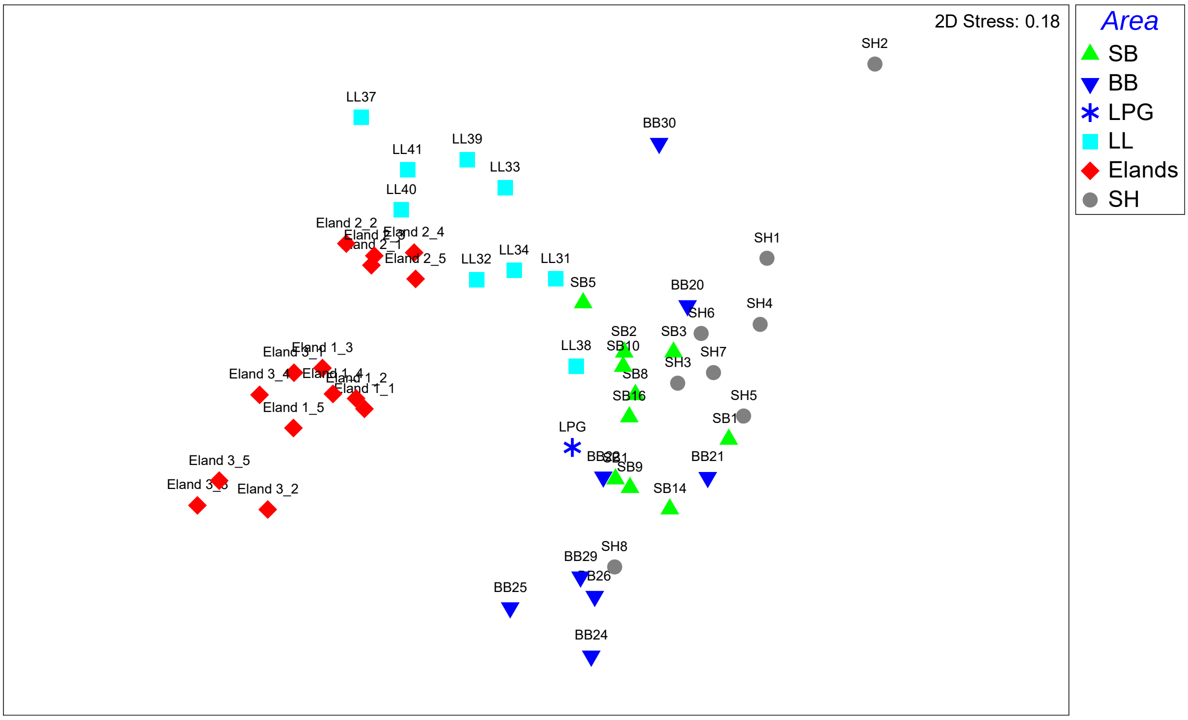


Figure 9.2. **Ordination showing similarity amongst sample sites based on benthic macrofauna abundance in 2020. Symbols on the ordination plots are as follows: Small Bay (SB), Big Bay (BB), Langebaan Lagoon (LL), Elandsfontein (Elands) and Sea Harvest (SH).**

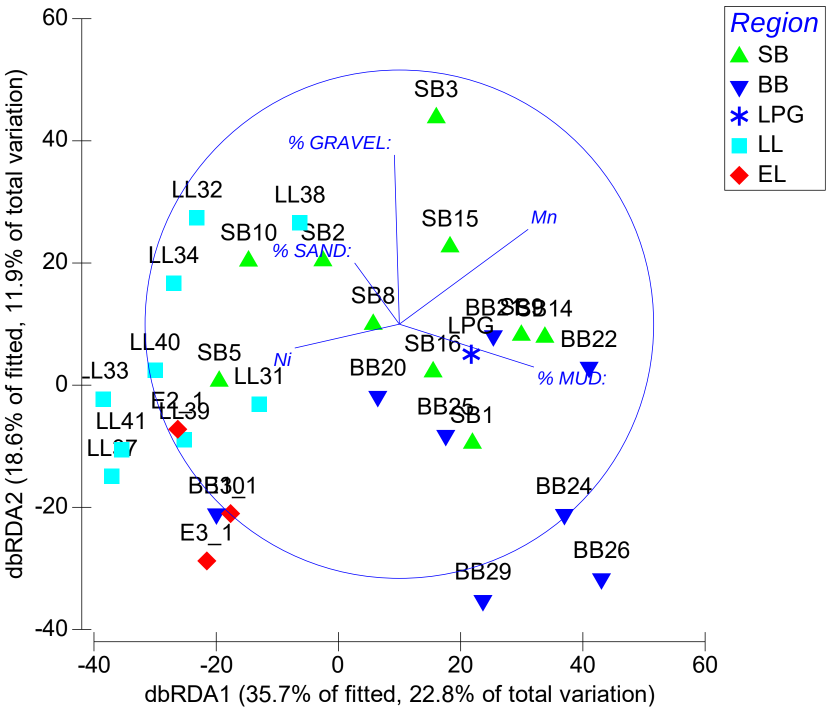


Figure 9.3. **dbRDA plot of 2020 macrofaunal abundance data. Sediment fractions, TOC, TON, C:N and trace metal concentrations were included as categorical predictors in this design. Sediment fractions were arcsine transformed prior to analysis. The blue lines are category vectors, whereby the length of the vector is a measure of the strength of the relationship between that category and the axes.**

The community structure of benthic macrofauna at Elandsfontein was dominated by small crustaceans (mostly amphipods), and polychaetes. The presence of unique species such as the sandflat crab, *Danielella edwardsii* and the abundance of the sand prawn, *Callichirus kraussi,* the mud prawn, *Upogebia africana*, and small sand-dwelling amphipod, *Urothoe coxalis*, were the main causes of dissimilarity in community structure between Elandsfontein and the Saldanha Bay and Langebaan Lagoon samples.

Species composition can sometimes be more easily understood at higher taxonomic or functional group (essentially feeding mode) levels. Macrofaunal abundance and biomass results for each of the areas sampled in Small Bay, Sea Harvest, Big Bay, Langebaan Lagoon and Elandsfontein are shown in Figure 9.4. Crustaceans (this diverse group includes prawns, shrimps, mysids, crabs, amphipods and isopods) were the dominant taxonomic group in all areas. The next most abundant taxonomic group were polychaetes (bristle worms), and a relatively greater abundance of these worms were found in Langebaan Lagoon, Elandsfontein and Sea Harvest than in Small Bay and Big Bay (Figure 9.4). Filter feeders were by far the dominant functional group in Small bay and Big bay with a greater average abundance in the latter area (Figure 9.4). Detritivores were numerically the most abundant group on the mudflats at Elandsfontein and in Langebaan Lagoon (Figure 9.4). Interestingly, echinoderms (consisting mostly of the sea urchin, *Parechinus anglulosus*) contributed significantly to the overall biomass of samples collected at the Sea Harvest sites. These differences are attributable to physical habitat differences between the benthic environments found in the different areas which in turn are linked to past and present anthropogenic activities e.g. port construction, dredging and organic pollution.

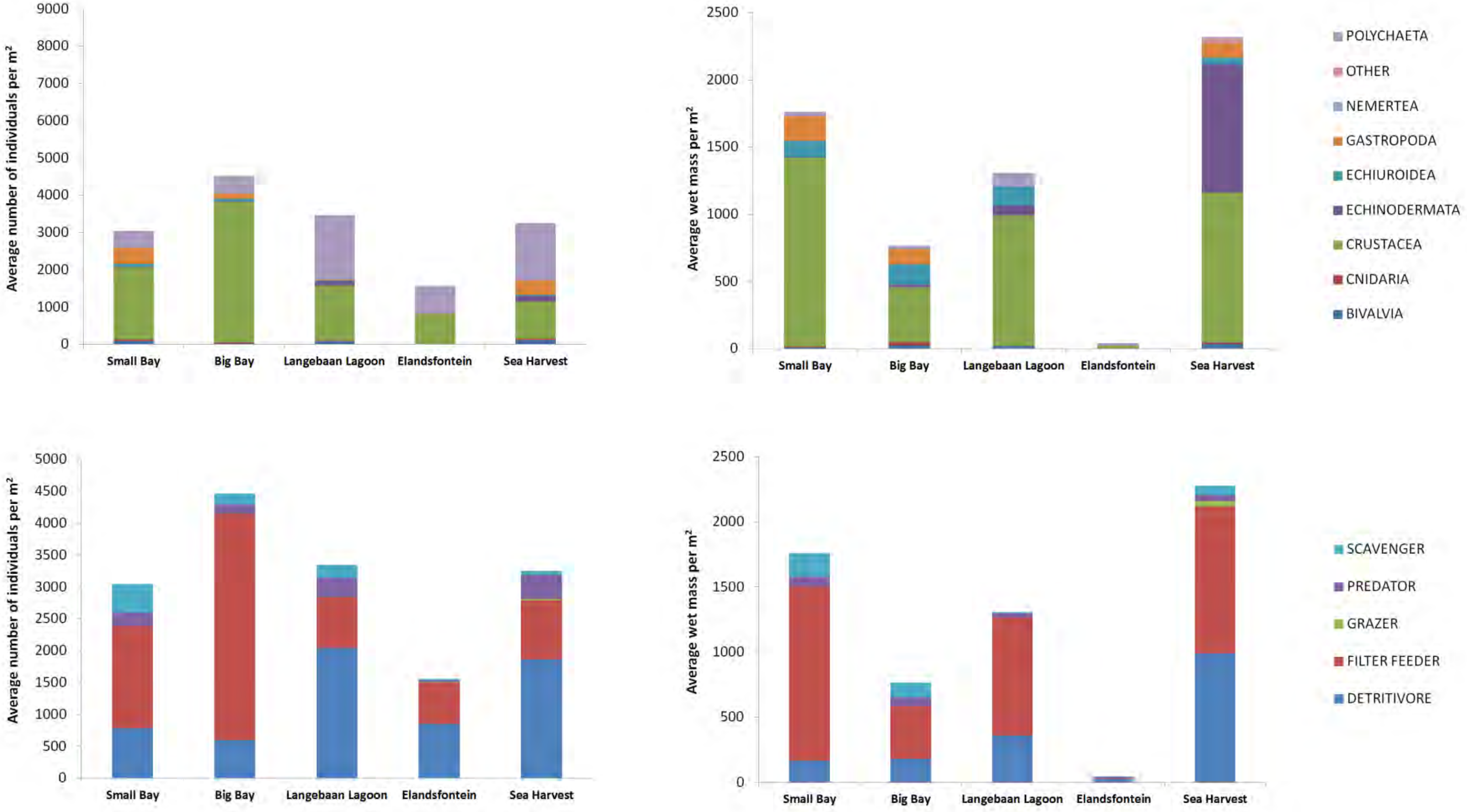


Figure 9.4. **Average abundance and biomass (g/m²) of benthic macrofauna by functional and taxonomic group in Big Bay, Small Bay, Langebaan Lagoon, Elandsfontein and Sea Harvest in 2020.**

9.5 Changes in abundance, biomass and community structure over time

9.5.1 Species richness

Variation in the total number of macrofauna species recorded in Small Bay, Big Bay, Langebaan Lagoon and Elandsfontein during each annual survey from 1999 to 2020 is shown in Figure 9.5. While there appears to be a slight increase in the numbers of species recorded over time, this is more than likely related to improvements in taxonomic resolution rather than a real increase with time. In Small Bay and Big Bay species richness was lowest in 1999, 2008 and 2012, while in Langebaan Lagoon the lowest richness was recorded in 2004, 2008 and 2012 (note that no samples were collected from the Lagoon in 1999). If one considers these data in the light of recent developments in the Bay, it is immediately clear that these changes may be linked to major dredging events in the Bay. Following construction of the original port in 1973, the most significant dredging events were implemented in 1996/7 (when 2 million m^3 of material was removed from the Small Bay side of the iron ore terminal for the construction of the *multi-purpose terminal*), the second in 2007/2008 (when approximately 50 000 m^3 of seabed material was removed from the area of the Mossgas quay and the multi-purpose terminal) and the third in 2009/2010, (when 7 300 m^3 of material was removed from the Saldanha side of the iron ore terminal). Species richness tends to drop (or starts off very low) immediately following these events (1999, 2008 and 2012) but tends to be higher (or even increase with time) in the intervening periods (2004, 2009-2011, 2013-2019) before declining again at Small Bay and Langebaan Lagoon in 2020.

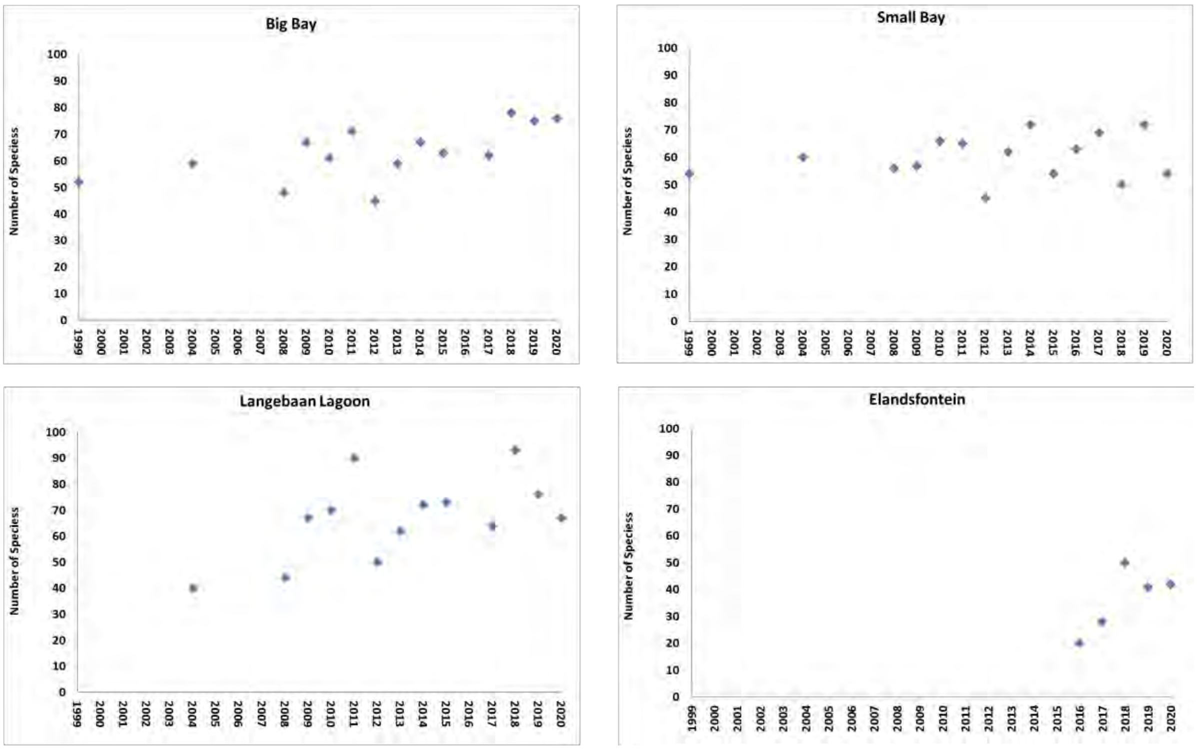


Figure 9.5. Variation in the number of species recorded at Small Bay, Big Bay, Langebaan Lagoon (1999 – 2020) and Elandsfontein (2016 – 2020).

The low species richness in Langebaan Lagoon recorded during the 2004 sampling event may be related to an entirely different phenomenon. During the mid-1990s the alien invasive mussel *M. galloprovincialis* began establishing dense intertidal beds on two intertidal sand flats close to the mouth of Langebaan Lagoon (Hanekom and Nel 2002). The mussel beds reached an estimated biomass of close to eight tonnes in 1999, and gave rise to concerns that the invasion could spread to the rest of the lagoon and other sandy substrata (Hanekom and Nel 2002). In early 2001, however, the mussels started to die off and by mid-2001 only dead shells and anoxic sands remained. In an effort to prevent the re-settlement of the mussel, South African National Parks began to remove dead mussel shells in late 2001 (Robinson *et al.* 2007b). The precise causes of the die off have not been established but siltation and lowered food availability are suggested as possible reasons behind the declines (Hanekom and Nel 2002). There is a high probability that the reduced macrofauna species richness in the 2004 State of the Bay samples may thus have been linked to a residual impact of the mussel invasion.

Species richness at Elandsfontein is low in comparison to rest of the system and although this is likely a result of high natural disturbance (variation in temperature and salinity), it may also be an artefact of low cumulative sampling effort, this being only the fourth survey conducted in this area to date. Additional species are likely to be detected with subsequent surveys (albeit at a decreasing rate) until a point is reached where adequate cumulative sampling effort has resulted in the detection of most species present. Significantly more species were recorded here in 2018-2020 – this is most certainly attributable to the change in sampling gear used (from suction sampling to Van Veen grab). The Van Veen grab does appear to be more effective at sampling benthic macrofauna in this area, and we recommend this be continued for future monitoring.

9.6 Abundance, biomass and community composition

Changes in the abundance and biomass of benthic macrofauna in Small Bay, Big Bay and Langebaan Lagoon and Sea Harvest are presented in Figure 9.6 - Figure 9.9. The relative importance of different feeding groups (i.e. trophic functioning which reflects changes in food availability) and taxonomic groups (i.e. different species which differ in size, growth rates and other characteristics) in each year are also shown on the same graphs. In all three areas (Small Bay, Big Bay and Langebaan Lagoon), there is a suggestion that both abundance and biomass of benthic macrofauna has been increasing over time up until 2014, aside from a number of major perturbations (troughs) that are evident at the start of the monitoring period (1999, Small and Big Bay only), and 2008/2009 (all three areas) and 2012 (all three areas). However, in 2015 both abundance and biomass decreased slightly in all areas apart from the lagoon where biomass remained more or less constant. There are some clear changes in the relative contribution of major taxonomic groups (Bivalvia, Crustacea, Gastropoda, etc.) in the periods of reduced abundance/biomass but the changes in the relative contributions by the different feeding groups is much more pronounced. The significant increase in the overall biomass and abundance indices recorded in Langebaan Lagoon this year can be attributed to a high number of sand prawns, *Callichirus kraussi*, in the samples collected during the 2020 survey. This species is a harvested extensively for bait and it is encouraging to see an increase in both abundance and biomass in Langebaan Lagoon. At this stage we can only speculate as to why this might be the case.

The relative contribution by the group known as filter feeders (i.e. those that feed by filtering particulate matter out of the water column) dropped dramatically during these perturbations in all three areas of the Bay while the contribution by the group known as detritivores (those that feed on particulate organic matter in or on the surface of the sediment) tended to increase. Filter feeders tend to be more sensitive to levels of suspended sediment that the other feeding groups, and this certainly lends weight to the argument that these period of reduced abundance and/biomass may be linked to major dredging events that have taken place in the Bay.

These filter feeders consist mostly of the mud prawn (*Upogebia capensis*) and smaller amphipod species belonging to the genus *Ampelisca*. The Sea pen, *Virgularia schultzei,* is another important filter feeding species in the Bay. This species was reportedly “very abundant” in the period prior to port development and was present throughout Big Bay and Small Bay. It is now completely absent from Small Bay but still present in Big Bay albeit in small numbers only. Detritivores, the second most important group of benthic macrofauna in Small Bay, comprise mostly of tongue worms (*Ochaetostoma capense*) and polychaetes belonging to the genera *Polydora* and *Euclymene*. These species are less sensitive to water quality and changes in wave movement patterns and hence tend to increase in abundance or even dominate when conditions deteriorate.

The higher proportion of detritivores at Sea Harvest sites compared to Small Bay sites correlates with the levels of TOC measured across these same areas (Chapter 7). It is known that decaying particulate organic matter (detritus) provides sustenance to detritivorous invertebrates. Thus, one can infer that TOC (a proxy for detritus) is largely correlated with the presence of detritivores. Tongue worms (*Ochaetostoma capense*) are a relatively large species that form a major a component of detritivores elsewhere in Saldanha Bay and thus explain the high biomass of detritivores recorded in Small Bay (Anchor 2020).

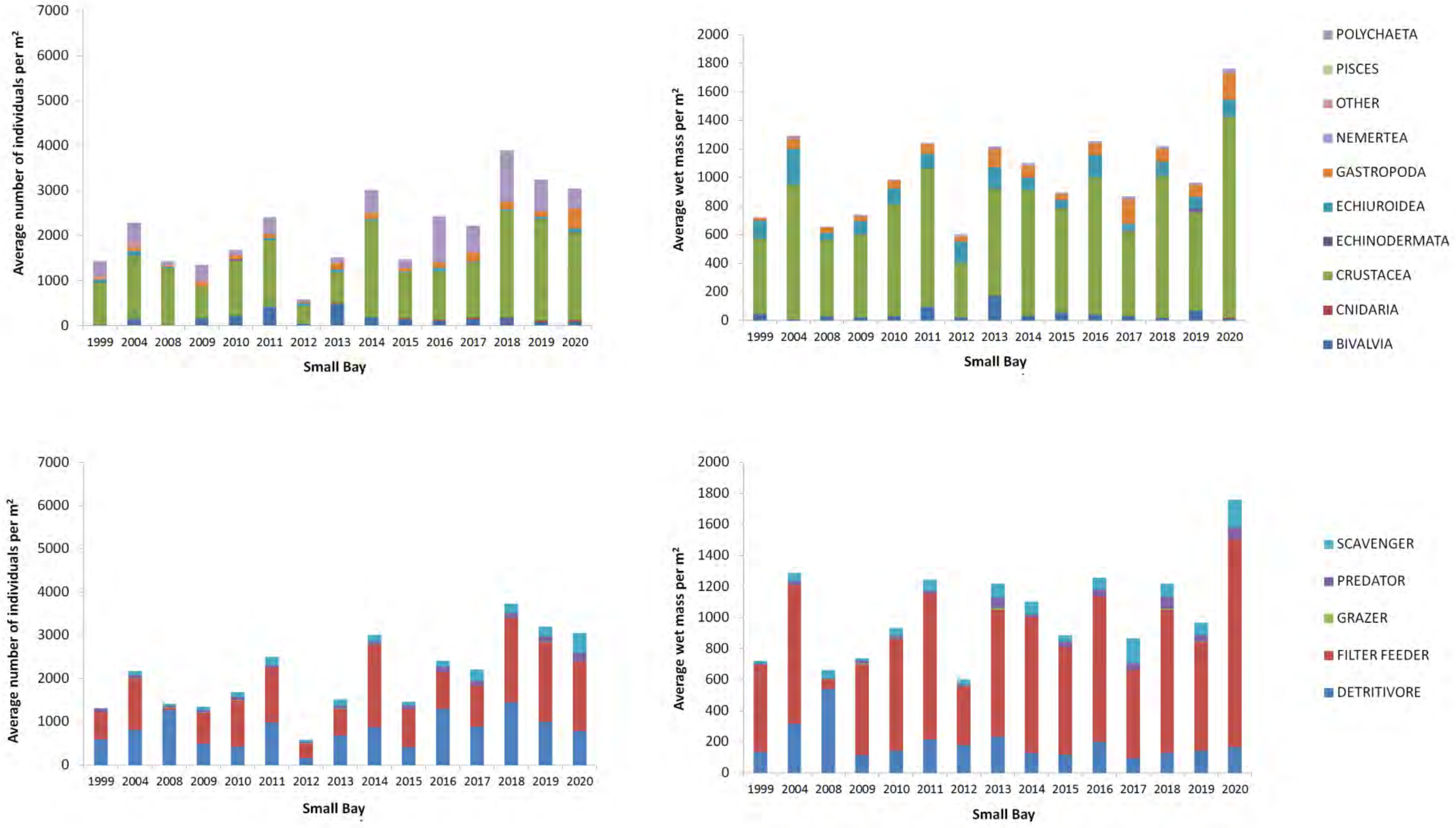


Figure 9.6. Overall trends in the abundance and biomass (g/m²) of benthic macrofauna in Small Bay as shown by taxonomic and functional groups.

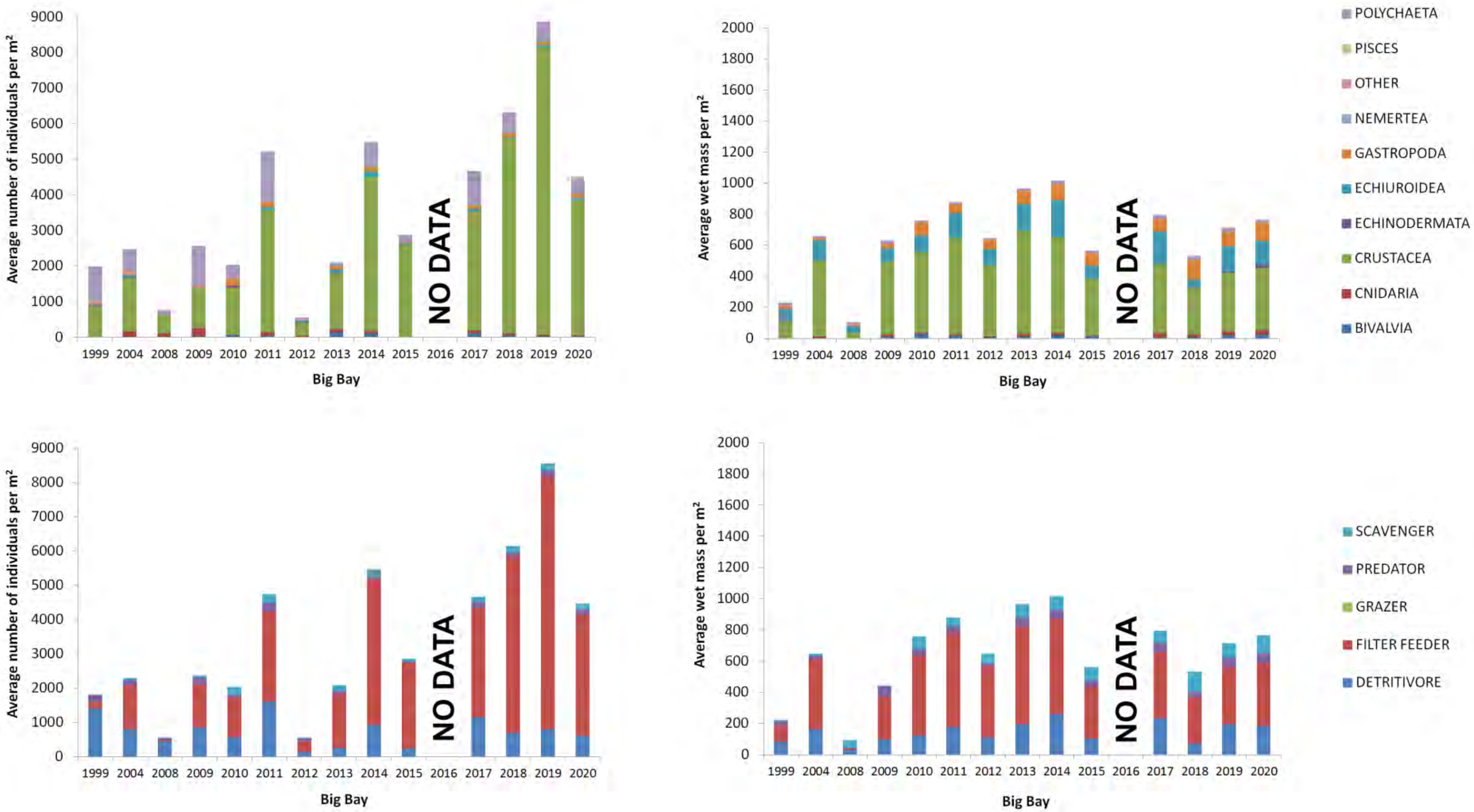


Figure 9.7. **Overall trends in the abundance and biomass (g/m²) of benthic macrofauna in Big Bay as shown by taxonomic and functional groups.**

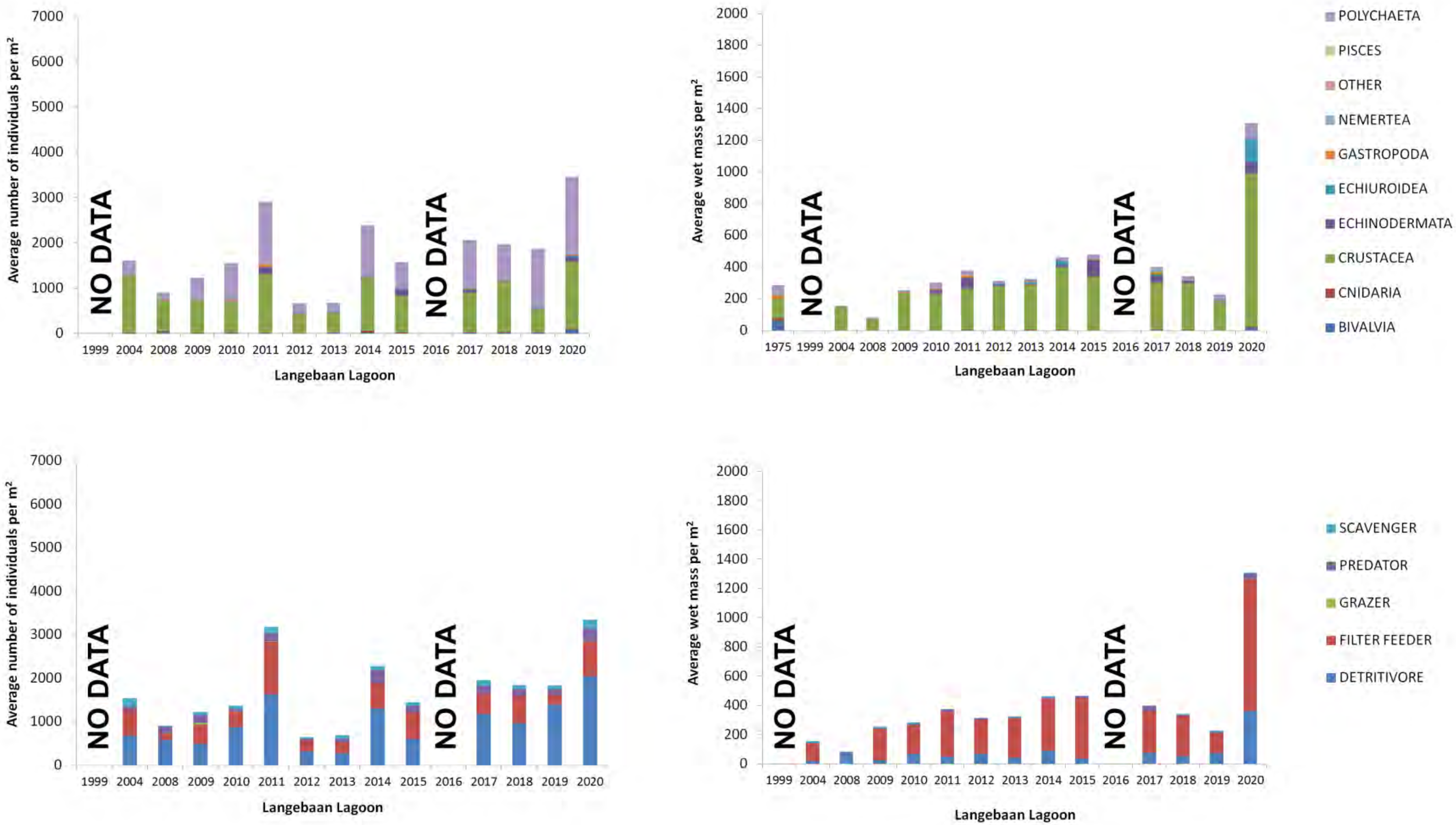


Figure 9.8. **Overall trends in the abundance and biomass (g/m²) of benthic macrofauna in Langebaan Lagoon as shown by taxonomic and functional groups.**

Previously, tongue worms were absent at the Sea Harvest effluent monitoring sites in 2017; but are now found at some of the monitoring sites following the relocation of the discharge pipe to a more suitable location. Hardier filter feeders such as the prawn *Upogebia capensis* were abundant at both the Sea Harvest outfall and Small Bay sites in 2020 which would explain the high biomass of filter feeders observed (Figure 9.9). Conversely, the previous 2017 Sea Harvest survey showed that the proportion of filter feeders was very low and the absence of sensitive filter feeders (e.g. the amphipods *Ampelisca spinimana* and polychaete *Sabellides luderitzi*) was likely the result of poor water quality (Wright *et al.* 2018). In the current survey, the proportion of filter feeders has grown significantly, similar to that observed in Small Bay, indicating an improved and healthier water and sediment quality, facilitating the presence and growth (biomass) of various taxonomic functional groups.

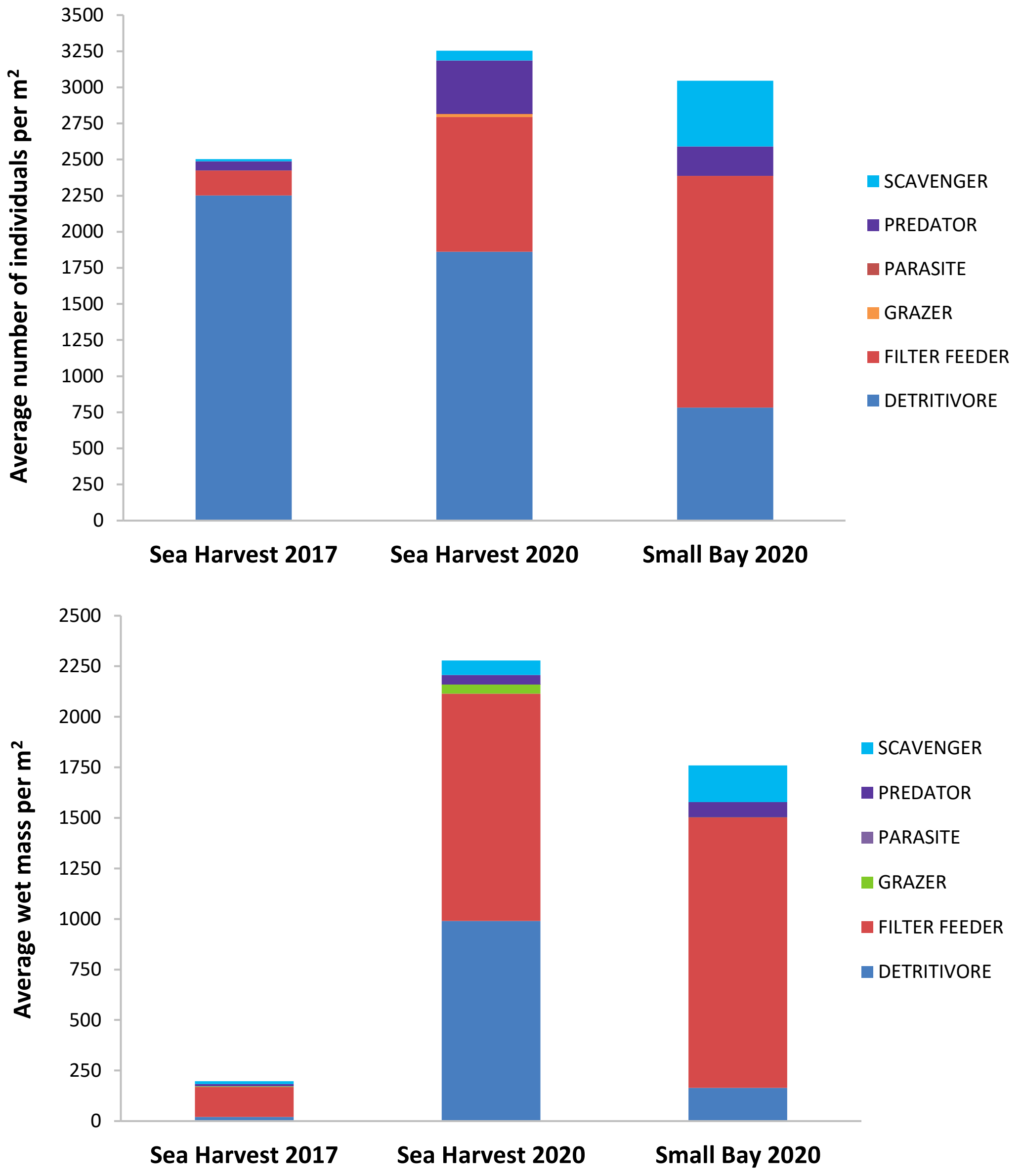


Figure 9.9. **Trends in average abundance/m² and biomass (g/m²) of benthic macrofauna in Small Bay and the Sea Harvest monitoring sites as shown by functional groups.**

9.7 Community structure

In this and previous reports, multivariate analysis has revealed clear differences in the macrofaunal communities inhabiting Small Bay, Big Bay and Langebaan Lagoon that are largely driven by physical habitat characteristics of each area. Investigation of any changes in macrofaunal communities over time, however, is useful as an ecosystem health monitoring tool as community scale perturbations outside of natural variability can indicate anthropogenic impacts on habitat quality. In order to do this without the confounding effects of the documented spatial structure, multivariate analysis of macrofaunal abundance data collected in all years since 2004 was undertaken separately for Small Bay, Big Bay and Langebaan Lagoon. A similar exercise was performed for the Sea Harvest sites where macrofaunal community structure at the Sea Harvest monitoring sites in 2017 was compared to that observed in 2020 – data from Small Bay were included as reference against which recovery could be measured.

9.7.1 Small Bay

The Small Bay ordination plot (a technique that groups samples with similar macrofaunal communities close together and separates dissimilar samples), shows clear separation of all samples collected during 2008 from samples collected in all other years (Figure 9.10). Overall abundance in Small Bay was not notably low in 2008, but the macrobenthic community was different in that there were a high abundance of detritivores such as the shrimp *Betaeus jucundus*, the polychaetes *Mediomastus capensis* and *Maldanidae* sp., and crustaceans of the Family Cumacea that were not common in samples collected during other years. Conversely, detritivorous crustaceans such *Spiroplax spiralis,* polychaetes *Polydora* sp. and *Orbinia angrapequensis*, the tongue worm *Ochetostoma capense,* predatory whelks of the genus *Nassarius* and filter feeding amphipods *Ampelisca* sp*.* and the mud prawn *Upogebia capensis*, were common in samples collected in other years, but were rare or absent in 2008 samples.

As mentioned above, these changes in macrobenthic community structure are thought to be related to the extensive dredging activities undertaken during 2007 and early 2008 that appeared to have had Bay-wide impacts, resulting a temporary loss of less tolerant species and a shift in community composition to one dominated by more tolerant species. Multivariate analysis of the macrobenthic samples collected over the period 2009-2020 suggests that the smaller 2009 dredging event had a limited impact with little change in macrobenthic community structure over the last ten years.

9.7.2 Big Bay

The 2008 Big Bay macrobenthos samples also clustered separately from all other years on the ordination plot indicting that they were dissimilar to the others in some way (Figure 9.10). Species primarily responsible for the dissimilarity of 2008 samples from all other years include very low abundance or absence of detritivores*, Orbinia angrapequensis and Ochetostoma capense*, filter feeders such as *Upogebia capensis, Ampelisca* sp. and *Virgularia schultzei* and predators such as *Nassarius* sp. whelks in 2008 samples. The same resilient species that were abundant in Small Bay

2008 samples also dominated the macrofauna in Big Bay, e.g. *Betaeus jucundus*, *Mediomastus capensis* and *Platynereis australis*.

9.7.3 Langebaan Lagoon

The 2008 samples were also outliers in the Langebaan Lagoon ordination plot (Figure 9.10). Low abundance or absence of filter feeding mud prawns *Upogebia capensis*, the polychaete *Notomastus latericeus* and the isopod *Natatolana hirtipes*; and high abundance of *Betaeus jucundus* and the polychaetes *Marphysa sanguine* and *Eteone foliosa* in 2008 samples were the species consistently responsible for the dissimilarity of 2008 Lagoon samples from those collected in other years

As mentioned above, these changes in macrobenthic community structure are thought to be related to the extensive dredging activities undertaken during 2007 and early 2008 that appeared to have had Bay-wide impacts, resulting a temporary loss of less tolerant species and a shift in community composition to one dominated by more tolerant species. Multivariate analysis of the macrobenthic samples collected over the period 2009-2020 suggests that the smaller 2009 dredging event had a limited impact with little change in macrobenthic community structure over the last ten years.

9.7.4 Sea Harvest Sites

Multivariate results indicate that there has been a substantial improvement in macrofaunal community structure and functioning adjacent to the Sea Harvest discharge (Figure 9.11). The macrofauna sampled in 2017 at sites surrounding the old discharge location were dominated by small detritivores and represented a barely functioning community with very low overall biomass. SIMPER analyses revealed that Sea Harvest monitoring sites in 2017 and 2020 had a dissimilarity of 72%; of which the polychaetes, *Polydora sp., Caulleriella acicula* and *Mediomastus capensis* were the three species contributing most to this dissimilarity. All three polychaetes are small detritivores which would explain the low biomass observed at the Sea Harvest sites in 2017. The 2020 samples from sites surrounding the new discharge locality were indicative of a fully functioning macrofaunal community with good representation of various functional groups similar to communities sampled elsewhere in Small Bay (Figure 9.11). The greater proportion of detritivores and slightly higher average abundance and biomass (Figure 9.9) at Sea Harvest sites compared to Small Bay sites in 2020 does suggest some organic enrichment and retention at the former but the community composition is much closer to "natural". This constitutes a notable improvement in benthic ecosystem health adjacent to the Sea Harvest discharge when compared with findings from 2017. Ongoing monitoring is, however, recommended to ensure that the situation does not revert through accumulation of organic material at the new discharge location over time.

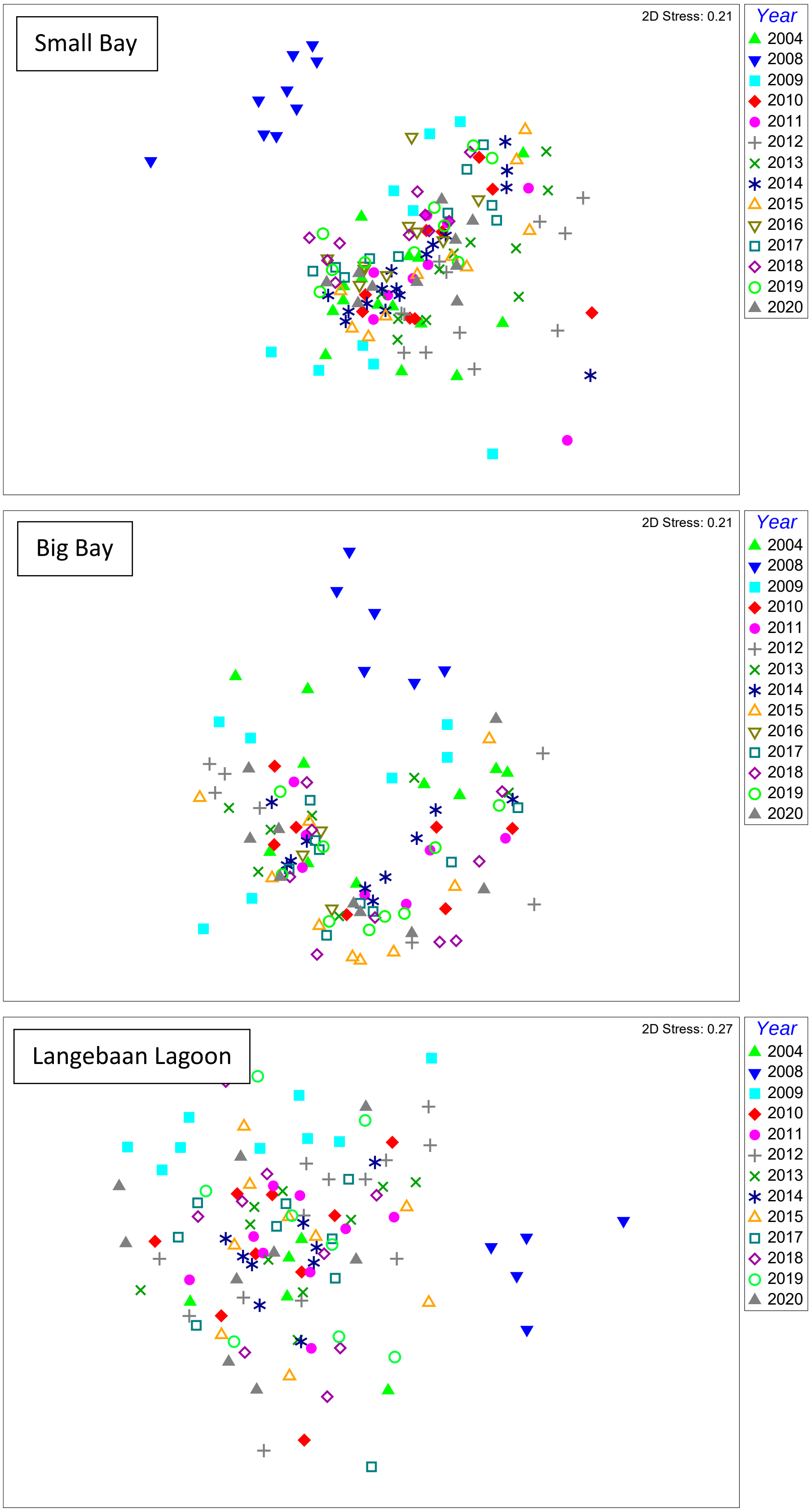


Figure 9.10. **MDS plots based on macrofaunal abundance data from samples collected in Small Bay (top), Big Bay (middle) and Langebaan Lagoon (bottom) during the period 2004-2020.**

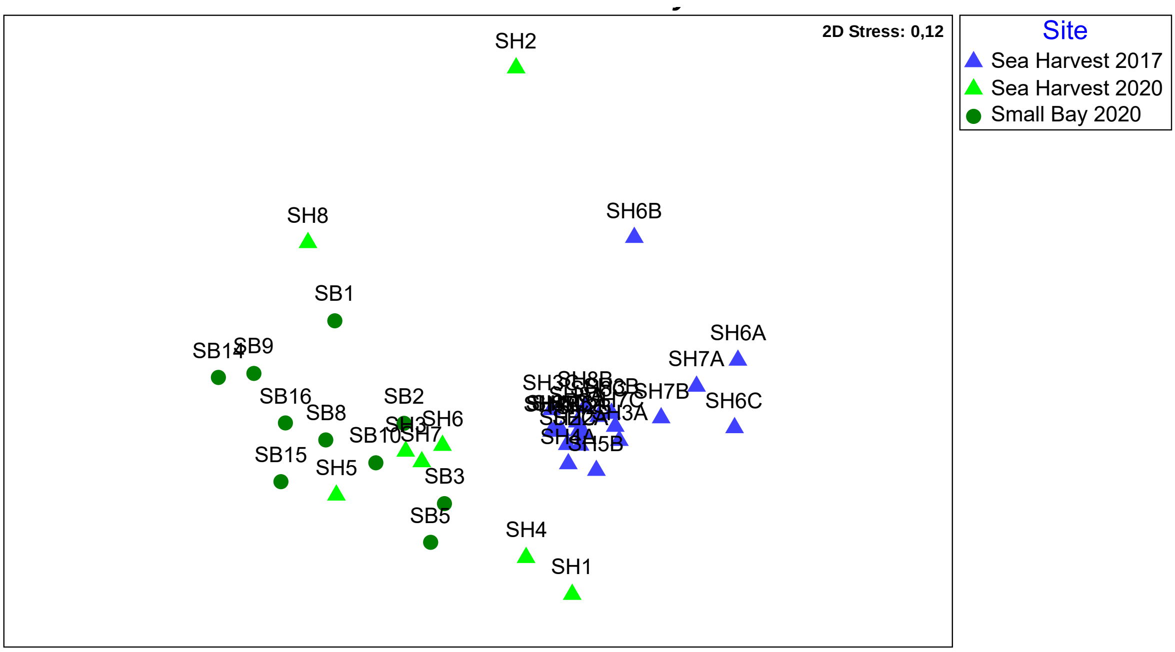


Figure 9.11. MDS plot of macrofaunal abundance, based on the Bray-Curtis resemblance matrix, collected at Sea Harvest sites in 2017, 2020 and Small Bay in 2020.

9.8 Elandsfontein 2020 survey results

The State of the Bay monitoring activities were expanded to include monitoring of benthic macrofauna at three new sampling sites near the head of the Lagoon at Elandsfontein in 2016. Concern had been raised around potential impacts that the proposed phosphate mine at Elandsfontein might have on groundwater quality and flows to Langebaan Lagoon; hence the objective to establish an appropriate baseline of the present benthic macrofauna community structure against which any potential future changes in the Lagoon can be benchmarked. The fourth set of baseline results are presented here and are assessed in context of the entire Saldanha Bay/Langebaan Lagoon system.

The ordination plot prepared from the 2020 macrofauna abundance data, are presented in Figure 9.2. It is evident that significant spatial dissimilarities in macrofaunal community composition exist between samples from Saldanha Bay (Small Bay and Big Bay), Langebaan Lagoon and Elandsfontein with each area forming a distinct cluster. The Langebaan Lagoon cluster falls directly between the Saldanha Bay and Elandsfontein clusters which implies that the macrofaunal community composition at the Elandsfontein sites are most similar to that present in Langebaan Lagoon (76.8% dissimilarity) and in turn are most dissimilar to those in Small Bay (86.8%) and Big Bay (91%). This suggests that a spatial trend in macrofaunal communities exists from the marine dominated Saldanha Bay through the sheltered lagoon to the very sheltered, shallow, sun-warmed and possibly freshwater/estuarine influenced Elandsfontein sites.

To date, a total of 65 species (consisting of polychaetes, crustaceans, gastropods, bivalves, a nemertean and a cnidarian - Figure 9.12) have been recorded at Elandsfontein. Six of these are found nowhere else in the system namely the polychaetes *Ancistrosyllis rigida* and *Scoloplos johnstonei*; the crabs *Danielella edwardsii* and *Paratylodiplax algoensis*; the gastropod *Nassarius kraussianus*; and an isopod belonging to the Sphaeromatidae.

Macrofaunal abundance and biomass results from 2016 to 2020 (broken down into taxonomic and functional feeding groups) are shown in Figure 9.12. There does not appear to be any significant difference in mean abundance over the years. On a community composition level, the samples collected in 2018, 2019 and 2020, group separately to those collected in 2016 and 2017 (Figure 9.13). In addition there are further differences in macrofaunal community structure between the different sites at Elandsfontein with sites Eland_1 and Eland_2 grouping together and site Eland_3 forming its own cluster (Figure 9.13). This is likely to be explained by the difference in physical conditions present at each of the sites. From Figure 9.1, it can be seen that Eland_3 is situated directly opposite the "mouth" of the channel from Langebaan Lagoon and appears to be mostly marine, whereas Eland_1 and Eland_2 are located further east, closer to the source of freshwater in what appears to be a more estuarine habitat. Interpretation of water quality data from a conductivity, temperature and depth (CTD) instrument deployed in the vicinity and further sampling in years to come would provide further insight into our findings thus far.

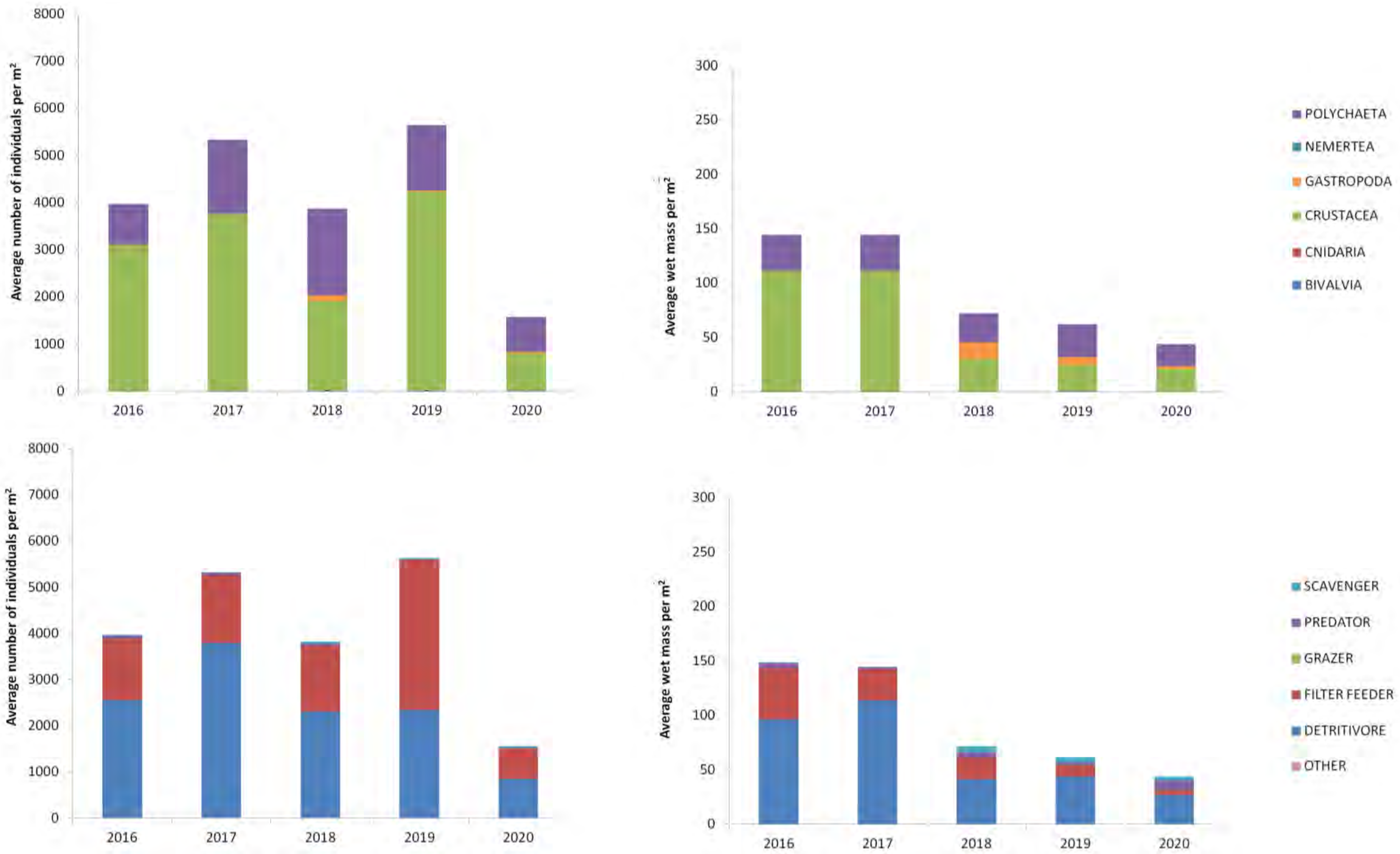


Figure 9.12. **Average abundance and biomass (g/m2) of benthic macrofauna by functional and taxonomic group from sampling sites at Elandsfontein from 2016 to 2020.**

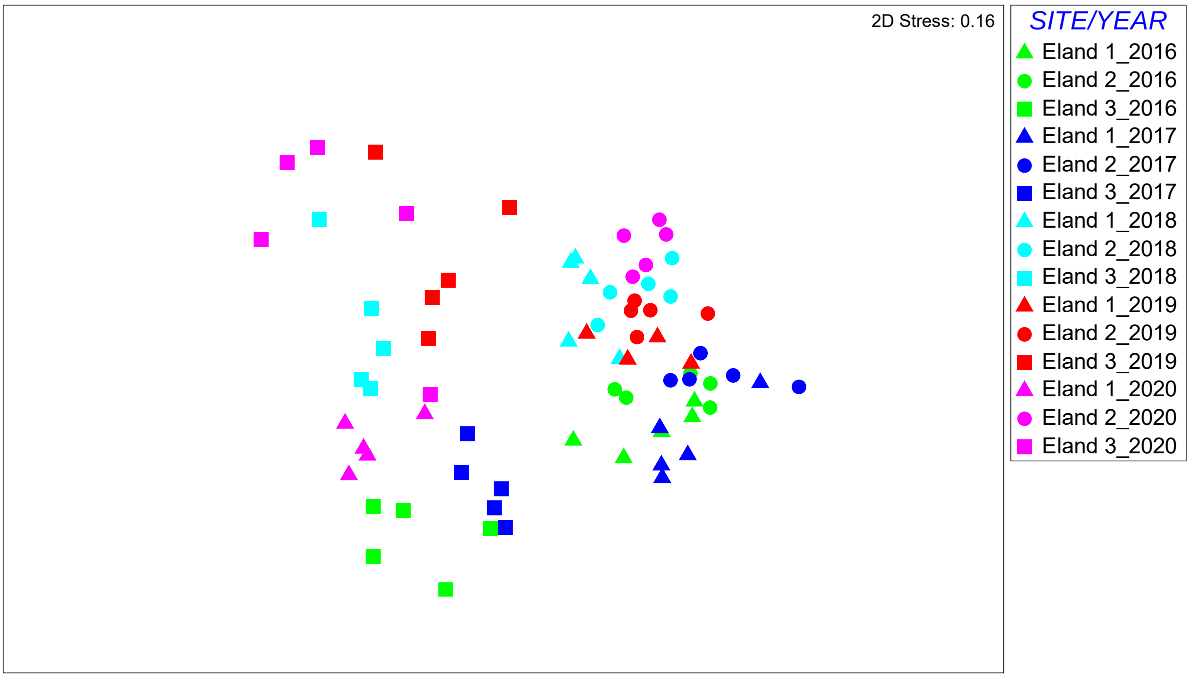


Figure 9.13. MDS plot based on macrofaunal abundance data from samples collected at Elandsfontein from 2016 to 2020.

9.9 Summary of benthic macrofauna findings

Macrofaunal community structure within Saldanha Bay has been the subject of several studies in the past, most of which focus on anthropogenic impacts to benthic health. These earlier studies showed very clearly that there was a substantial change in benthic communities before and after harbour development in the early 1970s. At this time, approximately 25 million cubic meters of sediment were dredged from the Bay, and the dredge spill was used to construct the new harbour wall (Moldan 1978). Severe declines in a number of species were reported, along with a change in the relative abundance of different trophic (feeding) groups, with a reduction in the number of suspension feeders in particular and an increase in the numbers of opportunistic scavengers and predators (Moldan 1978, Kruger *et al.* 2005). Within Saldanha Bay, many species disappeared completely after dredging (most notably the sea-pen, *Virgularia schultzei*) and were replaced by opportunistic species such as crabs and polychaetes (Moldan 1978). Dredging reportedly directly impacts benthic community structure in a variety of ways: many organisms are either directly removed or buried, there is an increase in turbidity and suspended solids, organic matter and toxic pollutants are released and anoxia occurs from the decomposition of organic matter (Moldan 1978). Indeed, reduced indices of abundance, biomass and diversity observed at the LPG site in 2019 appear to be linked with increased disturbance at this site since the SPM was installed in this area. Harbours are known to be some of the most highly altered coastal areas that characteristically suffer poor water circulation, low oxygen concentrations and high concentrations of pollutants in the sediment (Guerra-Garcia and Garcia-Gomez 2004). Beckley (1981) found that the marine benthos near the iron-ore loading terminal in Saldanha Bay was dominated by pollution-tolerant, hardy polychaetes.

This is not surprising since sediments below the iron ore terminal were found to be anoxic and high in hydrogen sulphide (characteristically foul-smelling black sludge).

Methods for collecting macrofauna samples for the State of the Bay surveys, which commenced in 1999, are unfortunately very different to those that were employed for the earlier surveys, and thus data from these studies cannot be compared directly. Analysis of the data from these studies as reported in this chapter is thus focussed on changes that have occurred in this latter period only. Variations in species richness, abundance biomass, and community composition and community structure all show very similar patterns over this period. Starting off at modest levels in 1999, both abundance and biomass rose to fairly high levels in Small Bay and Big Bay in 2004 before dropping down to low levels again in 2008 (regrettably no data are available to show what happened in the intervening years between 1999 and 2004 and between 2004 and 2008). Thereafter both overall abundance and biomass in all three parts of the Bay (Langebaan Lagoon included) increased steadily year-on-year until 2011, before dropping dramatically again in 2012, rising again in 2013 and 2014 and then remaining fairly stable up to the present 2020 survey. These changes in abundance and biomass were, to a large extent, driven by the loss of filter feeding species during period of low abundance (1999, 2008 and 2012). Filter feeding species are thought to be highly sensitive to changes in water quality (more so than detritivores or scavengers) and it is thought that reductions in abundance and biomass of these species may also be linked to a sequence of dredging events that have occurred in recent years (1996/, 2007/2008 and 2009/2010). The latest suite of results indicates a significant increase in the prevalence of the sand prawn, *Callichirus kraussi*, in Langebaan Lagoon. This is a popular bait species and it is encouraging to see an increase in both abundance and biomass in Langebaan Lagoon. At this stage we can only speculate as to why this might be the case.

Other more localised factors are also clearly important in structuring benthic macrofauna communities in the Bay and the Lagoon (see previous versions of the State of the Bay Report – Anchor Environmental 2010-2019 for more details on this). For example, reduced water circulation patterns in parts of Small Bay (e.g. near the Small Craft Harbour) and localised discharges of effluent from fish processing establishment in this area, contribute to the accumulation of fine sediment, organic material and trace metals, and results in macrofauna communities in this area being highly impoverished. Similarly, the impacts of dredging required for the expansion and refurbishment of the Salamander Bay boatyard at the entrance of the lagoon in 2010 had a very clear impact on macrofaunal communities in this area (Anchor Environmental 2012, 2013). Invasion of Langebaan Lagoon by the European mussel *Mytilus galloprovincialis* also had a major impact on the fauna in the affected areas of the Lagoon (Hanekom and Nel 2002, Robinson and Griffiths 2002, Robinson *et al.* 2007b) and presumably on the results of the earliest 2004 State of the Bay survey as well.

Overall, increases in abundance, biomass and diversity of macrofauna across all parts of the Bay (Small Bay, Big Bay and Langebaan Lagoon) in 2013 and 2014 was taken as a very positive sign and points to an overall increase in the health of the Bay. The slight fluctuations observed in abundance and biomass data from 2016 to 2020 are not of major concern as overall community structure remains largely unchanged. Results from the Elandsfontein baseline survey show that the macrofaunal community present at these sites are most similar to that present in Langebaan Lagoon. A spatial comparative analysis revealed a clear trend in macrofaunal communities from the marine dominated Saldanha Bay through the sheltered Lagoon to the very sheltered, shallow and possibly freshwater/estuarine influenced Elandsfontein habitat. Furthermore, physical habitat and associated macrobenthic biota appear to be driving dissimilarity among the Elandsfontein sites themselves. In terms of the concerns raised around potential impacts that the proposed phosphate mine at Elandsfontein may have on groundwater quality and flows to Langebaan Lagoon, ongoing collection of baseline data on macrobenthic communities in Elandsfontein to capture natural variability, is essential for objective and quantitative assessment of any impacts should they occur. Results from samples collected in the vicinity of the Sea Harvest discharge pipe indicate a marked improvement on those from 2017. This would suggest that the relocation of the discharge outfall was justified and has resulted in a notable improvement in benthic ecosystem health.

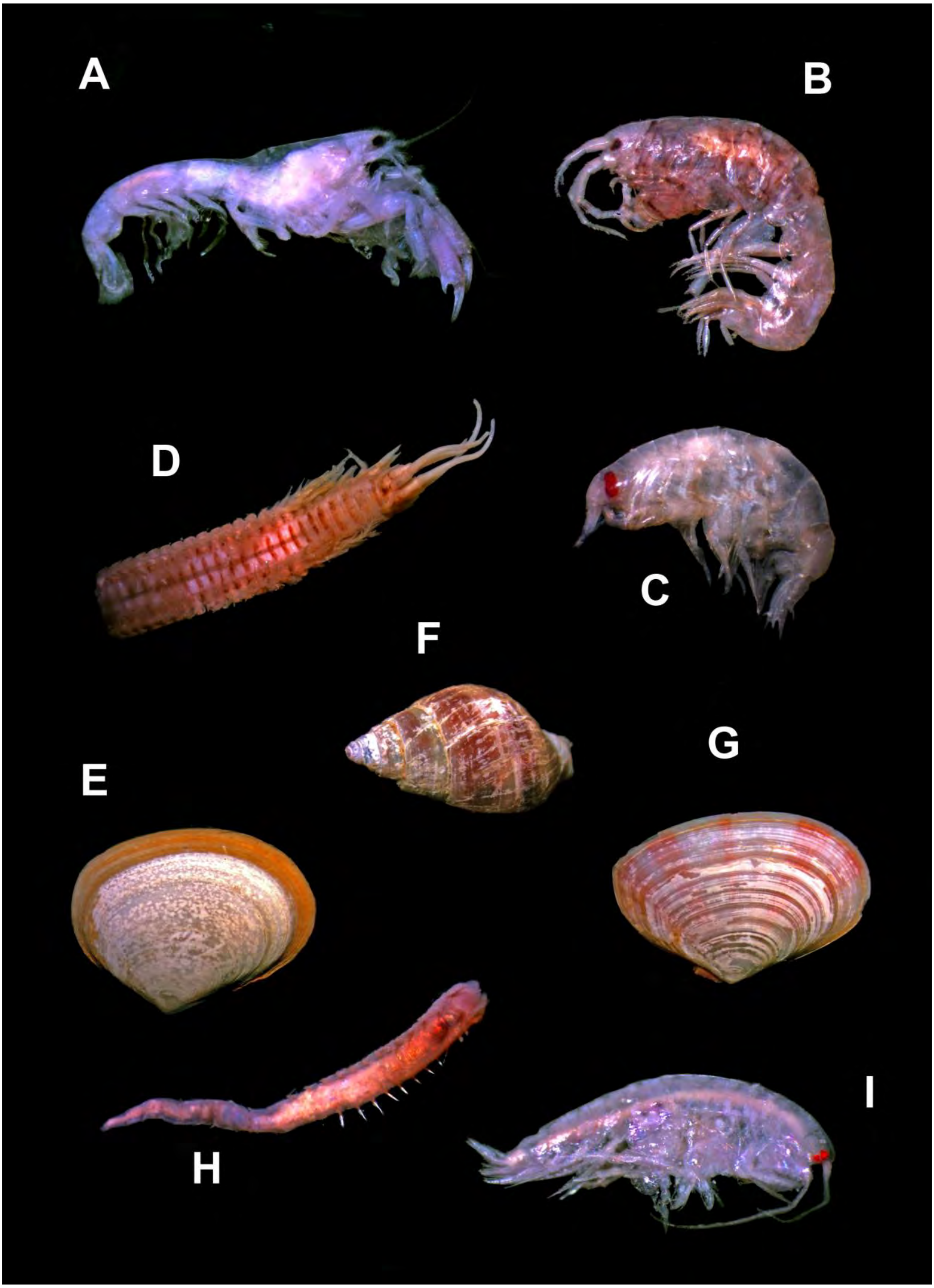


Figure 9.14. Benthic macrofauna species frequently found to occur in Saldanha Bay and Langebaan Lagoon, photographs by: Aiden Biccard. A – *Upogebia capensis*, B – *Idunella lindae*, C – *Hippomedon normalis*, D – *Diopatra monroi*, E – *Macoma c. ordinaria*, F – *Nassarius vinctus*, G – *Tellina gilchristi,* H – *Sabellides luderitzi*, I – *Ampelisca anomola.*

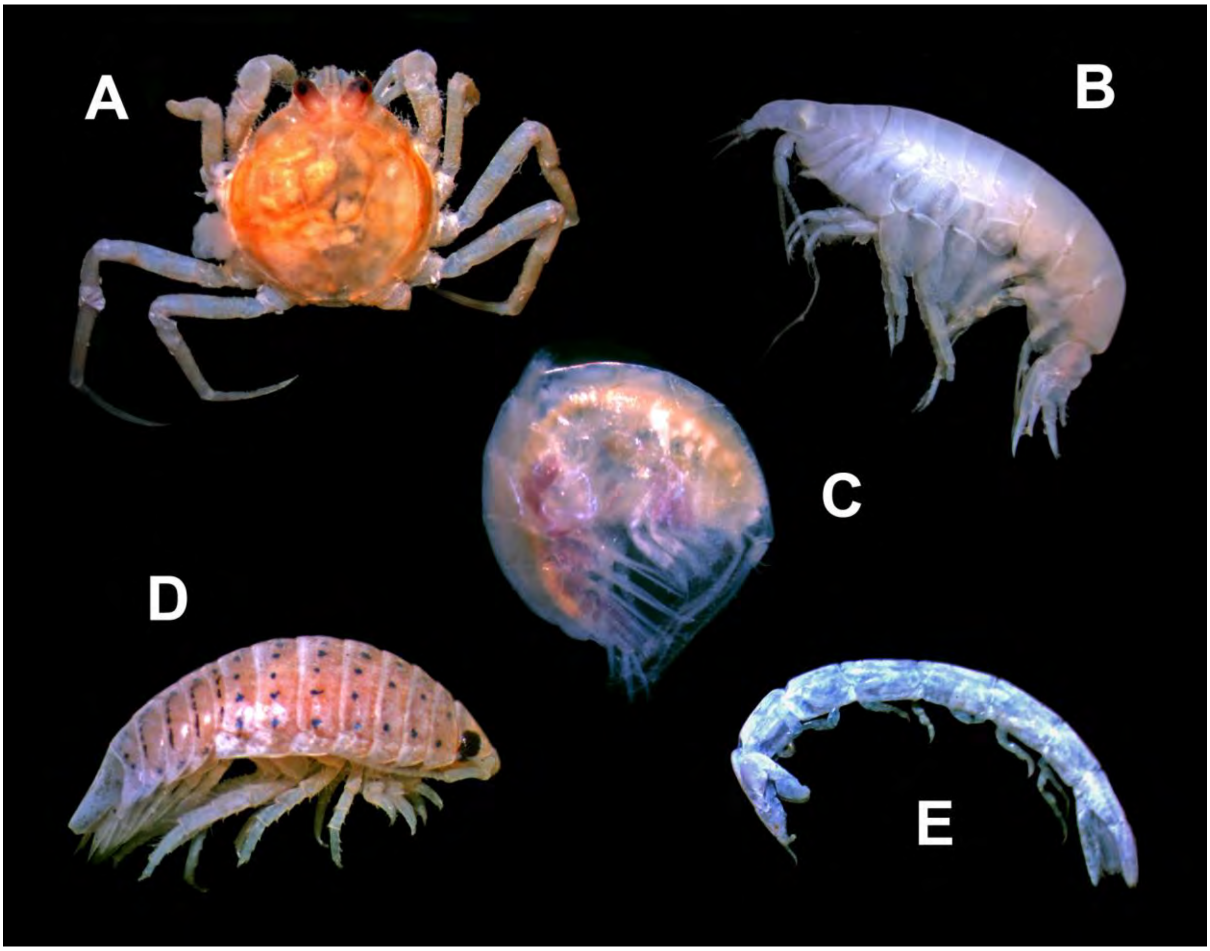


Figure 9.15. **Benthic macrofauna species frequently found to occur in Saldanha Bay and Langebaan Lagoon, photographs by: Aiden Biccard. A – *Hymenosoma obiculare,* B – *Socarnes septimus*, C – *Ampelisca palmata,* D – *Eurydice longicornis,* E – *Centrathura caeca*.**

10 ROCKY INTERTIDAL COMMUNITIES

10.1 Background

Limited historical data exists on the state of the rocky-shore habitats within the Saldanha Bay system. Species presence/absence data was collected by undergraduate students of the University of Cape Town at Lynch Point and Schaapen Island between 1965 and 1974 (University of Cape Town, Prof. C. Griffith, *pers. comm.*); however, the accuracy and reliability of these data is questionable and they provide limited value for monitoring changes in the health of the Saldanha Bay ecosystem. Simons (1977) and Schils *et al.* (2001) reported on the algal species assemblages in the Bay, while Robinson *et al.* (2007b) examined the species composition of rocky intertidal communities on Marcus Island between 1980 and 2001, focusing on the impact of the alien invasive Mediterranean mussel, *Mytilus galloprovincialis* (see Chapter 13).

Monitoring of rocky intertidal communities in Saldanha Bay was initiated as part of the State of the Bay Monitoring Programme in 2005 in an effort to fill the gap in knowledge relating to rocky intertidal communities in Saldanha Bay and Langebaan Lagoon. The first rocky shore survey for this programme was conducted in 2005, the results of which are presented in the first 'State of the Bay' report (Anchor Environmental Consultants 2006). Eight rocky shores spanning a wave exposure gradient from very sheltered to exposed were sampled in Small Bay, Big Bay and Outer Bay as part of this baseline. These surveys have been repeated more or less annually from 2008 to 2020.

The baseline survey report concluded that wave exposure was the primary physical driver shaping intertidal rocky shore communities across the study area. More sheltered shores were dominated by seaweeds, while sites exposed to higher wave energy were dominated by filter-feeders. It was suggested that the construction of the Marcus Island causeway and the iron ore terminal had reduced wave energy reaching rocky shores across much of Small Bay and led to a change in community structure. The lack of historical data from these shores precludes confirmation of this hypothesis, however.

The results further indicated that the topography and substratum type of the shore influences community structure as, for example, sites consisting of rocky boulders had different biotic cover to shores with a flatter profile. Geographic location was also considered to be important, for example, sampling stations on Schaapen Island are situated in a transitional zone between the Saldanha Bay and the Langebaan Lagoon system. These same sites are also affected by high nutrient input from seabird guano that favours algal growth. Generally, the Saldanha Bay communities were healthy, although the presence of a number of alien invasive species including the Mediterranean mussel *Mytilus galloprovincialis* and the three barnacles *Balanus glandula*, *Perforatus perforatus* and *Amphibalanus amphitrite amphitrite* were noted.

This chapter presents results from the thirteenth annual monitoring survey conducted in March 2020.

10.2 Approach and methodology

10.2.1 Study sites

The locations of the eight rocky shore sampling sites are shown in Figure 10.1. The Dive School and Jetty sites are situated along the northern shore in Small Bay. The Marcus Island, Iron Ore Terminal and Lynch Point sites are in Big Bay, while the Schaapen Island East and West sites are located at the entrance to Langebaan Lagoon. The North Bay site is situated in Outer Bay at the outlet of Saldanha Bay.

Figure 10.1. The location of the eight rocky shore study sites in Saldanha Bay are indicated by red dots.

The sampling sites were specifically chosen to cover the different rocky shore habitats found in the Saldanha Bay system and incorporate the full range of wave exposure and topographical heterogeneity (type of rock surface and slope). Dive School (DS) and Jetty (J) are very sheltered sites with gentle slopes, consisting of boulders and rubble interspersed with sandy gravel (Figure 10.2). Schaapen Island East (SE) is situated in a little baylet and is relatively sheltered and mostly flattish with some ragged rock sections. Schaapen Island West (SW) is a little less sheltered and mostly flat with some elevated topography.

The site at the Iron Ore Terminal (IO) is semi-exposed with a very steep slope resulting in a very narrow total shore width (distance from low-water to high-water mark). The rock surface at this site comprises medium-sized broken boulders that are piled up to support a side arm of the Iron Ore Terminal, which encircles a small area that was previously used for aquaculture purposes. The semi-exposed site Lynch Point (L) has a relatively smooth surface with occasional deep crevices. North Bay (NB) is exposed with a relatively flat high and mid shore. The low shore consists of large unmovable square boulders separated by channels. The rocky intertidal site on Marcus Island (M) is flat and openly exposed to the prevailing south-westerly swell.

Figure 10.2. Rocky shore study sites in Saldanha Bay. Dive School and Jetty are situated in Small Bay, Schaapen Island East and West are in Langebaan Lagoon, Iron Ore Jetty and Lynch Point are in Big Bay, and North Bay and Marcus Island are in Outer Bay.

10.2.2 Methods

At each study site, the rocky intertidal was divided into three shore height zones: the high, mid and low shore. In each of these zones, six 100x50 cm quadrats were randomly placed on the shore and the percentage cover of all visible species recorded as primary (occurring on the rock) and secondary (occurring on other benthic fauna or flora) cover. The quadrat was subdivided into 171 smaller squares with 231 points to aid in the estimation of the percentage cover. Individual mobile organisms were counted to calculate densities within the quadrat area (0.5 m^2). Finally, the primary and secondary cover data for both mobile and sessile organisms were combined and down-scaled to 100%. Percentage cover refers to the space that organisms occupy on the rock surface, while abundance refers to the number of organisms present. The survey protocol has remained consistent for all surveys.

Sampling is non-destructive, *i.e.* the biota were not removed from the shore, and smaller infaunal species (e.g. polychaetes, amphipods, isopods) that live in the complex matrix of mussel beds or dense stands of algae were not recorded by this survey protocol. Some algae and invertebrates that could not be easily identified to genus or species level in the field were recorded under a general heading (e.g. crustose and articulate corallines, red turfs, sponge, colonial ascidian). For further analysis, intertidal species were categorized into seven functional groups: grazers (mostly limpet species), filter-feeders (including sessile suspension feeders such as mussels and barnacles), predators and scavengers (such as carnivorous whelks and anemones), encrusting algae (crustose and articulated coralline algae), corticated algae, ephemeral foliose algae and kelps.

10.2.3 Data analysis

The rocky shore biota from the eight study sites were analysed with multivariate statistical techniques employing the software package PRIMER 6. These methods provide a graphical presentation of the results obtained from the typically large data sets collected during ecological sampling. The principle aim of these techniques is to discern the most conspicuous patterns in the community data. Comparisons between intertidal communities are based on the extent to which they share particular species at similar levels of occurrence. Patterns in the data are represented graphically through hierarchical clustering (dendrogram) and multi-dimensional scaling (MDS) ordination techniques. The former produces a dendrogram in which samples with the greatest similarity are fused into groups, and are successively clustered as the similarity criteria defining the groups are gradually reduced. MDS techniques compliment hierarchical clustering methods by more accurately 'mapping' the sample groupings two-dimensionally in such a way that the distances between samples represent their relative similarities or dissimilarities. All percentage cover data were 4th-root transformed and a Bray-Curtis resemblance matrix was used.

Statistical comparisons of *a priori* defined groups of samples (e.g. sites, years) were analysed by means of PERMANOVA. PERMANOVA is a routine for testing the simultaneous response of one or more variables to one or more factors in an analysis of variance (ANOVA) experimental design on the basis of any resemblance measure, using permutation methods (Anderson *et al.* 2008). In essence, the routine performs a partitioning of the total sum of squares according to the specified experimental design, including appropriate treatment of factors that are fixed or random, crossed or nested, and all interaction terms. A distance-based pseudo-*F* statistic is calculated in a fashion that is analogous to the construction of the *F* statistic for multi-factorial ANOVA models. P-values are subsequently obtained using an appropriate permutation procedure for each term. Following the main overall test, pair-wise comparisons are conducted. Significance level for the PERMANOVA routine is $p < 0.05$ (i.e. a 95% probability that the finding is not due to chance).

The contributions of each species to the average dissimilarity between two sites, and to the average similarity within a site, were assessed using a SIMPER (Similarity Percentages) analysis. The taxa principally responsible for differences detected in community structure between sites or groups were identified.

A variety of diversity indices were determined that are used as measures of community structure. Diversity indices include:

- *Species number (S)* - total number of species present.
- *Percentage/biotic cover* - the percentage of intertidal rocky surface that is covered by biota (fauna and flora).
- *Evenness (J')* - expresses how evenly the individuals are distributed among the different species, in other words, whether a shore is dominated by individuals of one or few species (low evenness) or whether all species contribute evenly to the abundance on the shore (high evenness). The index is constrained between 0 and 1 where the index increases towards 1 with less variation in communities.
- *Shannon-Wiener diversity index (H'[loge] or d)* - a measurement of biodiversity taking into account the number of species and the evenness of the species. The index is increased either by having additional unique species, or by having greater species evenness.

10.3 Results and discussion

10.3.1 Spatial variation in community composition

In 2020, a total of 100 taxa were recorded from all rocky shore sites, of which 60 taxa were invertebrates (60%) and 40 (40%) algae. The faunal component was represented by 23 filter feeding taxa, 23 grazers, and 14 predators/scavengers. The algal component comprised 24 corticated (foliose) seaweeds, 9 ephemerals, 5 encrusting algae, and 2 kelp species. Coralline algae taxa are likely underestimated as most species are not identifiable in the field and are thus lumped into larger groups. The total number of taxa recorded at the study sites has remained relatively constant over the years (Anchor Environmental Consultants 2009, 2010, 2011, 2012b, 2013b, 2014, 2015, 2017, 2018, 2019). Most of the species have already been recorded during one or more of the previous monitoring years, and many are listed by other studies conducted in the Saldanha Bay area (e.g. Simons 1977, Schils *et al.* 2001, Robinson *et al.* 2007b). The species are generally common to the South African west coast (e.g. Day 1974, Branch *et al.* 2010).

Figure 10.3. Photographs of a typical high, mid and low rocky shore site in Saldanha Bay (from left to right).

Intertidal rocky shores are alternately submerged underwater and exposed to air by tidal action. This creates a steep vertical environmental gradient for the biota that inhabit these shores resulting in biota lower on the shore being mostly submerged and biota higher on the shore mostly exposed. Rocky shores can thus be partitioned into different zones according to shore height level, whereby each zone is distinguishable by their different biological communities (Menge & Branch 2001). This is indeed true for all sites over the survey years (Anchor Environmental Consultants 2009, 2010, 2011, 2012b, 2013b, 2014, 2015, 2017, 2018, 2019).

10.3.1.1 High shore

The composition and distribution of the rocky intertidal biota is strongly influenced by the prevailing wave exposure at a shore, as well as substratum topography (McQuaid & Branch 1984). Within a site, shore height is a critical factor as a result of the increasing exposure to air from low to high shore, whereby the existence of distinct patterns of zonation of flora and fauna has been well described (Stephenson & Stephenson 1972). The effects of wave action are generally attenuated up-shore and superseded by the uniformly severe desiccation stress experienced high on the shore.

In agreement with the above, previous 'State of the Bay' reports showed that very few mobile species occurred on the high shore at all Saldanha Bay sites (Anchor Environmental Consultants 2015). It was also found that at the very sheltered boulder shores (Dive School and Jetty), considerable amounts of sand and gravel accumulated amongst the boulders (Anchor Environmental Consultants 2015). A typical species found at the high shore sheltered sites was the winkle *Oxystele antoni*, while at the exposed sites the anemone *Bunodactis reynaudi* and, in larger numbers, the tiny periwinkle *Afrolittorina knysnaensis* dominated this zone (Anchor Environmental Consultants 2015). The latter typically accumulated in moist cracks and crevices at Lynch Point, Marcus Island and North Bay.

Field data collected in 2020 showed that the Dive School was only 17.35% similar to the other high shore sites due to the periwinkle *Oxystele antoni* being relatively more abundant. The alien barnacle *Balanus glandula* occurred in the high shore zone at Dive School, Jetty and Lynch Point with less than 1% average cover. The Iron Ore Terminal had slightly more than 1% average cover of *B. glandula*, although densities were much higher on Marcus Island with an average of ±4% cover with almost 23% of one of the quadrats surveyed in the high shore zone on Marcus Island covered by this alien. *B. glandula* was absent from the high shore at Schaapen Island. On average, barren rock accounted for >80% at some sites on the high shore and algal cover was extremely sparse. Encrusting algae such as diatoms and *Hildenbrandia spp*. made up on average >20% at Marcus Island and Schaapen West, respectively. Furthermore, Marcus Island and Schaapen West had an average 3.1% cover of *Porphyra capensis,* and Schaapen West >8%. North Bay had slightly less than 2% average cover of *Ulva* spp. while Schaapen East and Schaapen West had <1%. *Nothogenia erinacea* was also found in the high shore at Schaapen East and Schaapen West with less than 1% coverage.

10.3.1.2 Mid shore

The mid shores at the sheltered sites were also relatively barren, while the exposed sites had higher biotic cover (Anchor Environmental Consultants 2015). The dwarf cushion starfish *Parvulastra exigua* was typically found in moist rock-depressions and small pools, while the whelk *Burnupena* spp. and the periwinkle *Oxystele antoni* were frequently observed sheltering in depressions created by mussel beds. In previous years, *Gunnarea gaimardi*, a tube-building polychaete living deeply cemented in a compact matrix of sand was common at sheltered sites (Anchor Environmental Consultants 2011), but in 2012 the worm had declined at the mid shore and was only recorded from lower down the shore, albeit with low cover. Field data collected in 2020 showed that encrusting coralline, the ephemeral alga *Ulva* spp., the corticated alga *Gigartina polycarpa*, encrusting alga *Ralfsia verrucosa*, the barnacle *Balanus glandula*, the encrusting alga *Hildenbrandia* spp., the periwinkle *Oxystele tigrina*, the whelk *Burnupena* spp., the mussels *Mytilus galloprovincialis* and *Aulacomya atra* together accounted for ~50% of the similarity between mid-shore sites. Algal presence was generally low in the mid shore, but *Ulva spp*. was more abundant at Marcus Island with an average cover of 30%.

With increasing wave force across sites, the mid shores were dominated by filter feeders, particularly the alien invasive barnacle *Balanus glandula*. The latter species was most abundant at the semi-exposed Lynch Point where ~40% of the quadrats surveyed on the mid shore was occupied by this barnacle. Neither of the remaining filter feeders were present in substantial numbers on the mid shore at the remaining sites. The tiny periwinkle *A. knysnaensis* was found nestling in amongst the barnacles at sites inundated with *B. glandula*. This snail is normally abundant primarily in the upper intertidal where it congregates in crevices to escape the heat of the day, emerging at night or on moist days to feed (Branch *et al.* 2010b). In the high shore where wave stress is minimal, *A. knysnaensis* is naturally abundant but in the mid-shore, where wave stress is greater, the periwinkle normally declines in abundance without shelter (Laird & Griffiths 2008, Griffiths *et al.* 2011).

10.3.1.3 Low shore

Reflecting known zonation patterns, total biotic cover generally increased from high to low shore from an average of 23% to 51% cover. At the very sheltered sites (Dive School and Jetty), average faunal cover was low in comparison with the exposed sites (North Bay and Marcus Island). Algal cover at sheltered sites was much lower than that at exposed sites, and consisted primarily of the green alga *Ulva* spp*.,* the encrusting alga *Hildenbrandia* spp., a variety of encrusting coralline species and the corticated alga *Gigartina polycarpa*. At the sheltered Schaapen Island sites, the ground cover was dominated by a diverse array of algal species, with encrusting coralline species being more prominent.

The following species together accounted for 53% of the similarity attributed between low shore sites: encrusting coralline; the algae *Gigartina polycarpa*, *Ulva* spp., *Ralfsia verrucosa*; the mussels *M. galloprovincialis* and *Aulacomya atra along* with the whelk *Burnupena* spp. The indigenous mussel *A. atra* can be found living deep within the *Mytilus* beds where they take advantage of the moisture trapped within the overlaying dense mussel matrix. In 2011, *A. atra* was fairly prominent at the low shore at Marcus Island, and could locally supersede the alien mussel *M. galloprovincialis* (Anchor Environmental Consultants 2012b). However, during the 2020 survey, the ribbed mussel contributed <1% to the low shore cover. As these populations cannot be seen without destructive sampling, it is possible that the changes in *A. atra* cover recorded between survey years are at least partly due to the *Mytilus* layers being ripped off from the rocks by waves, exposing the indigenous mussel beneath.

10.3.1.4 Spatial analysis of diversity indices

Diversity indices provide insight into the way in which the total number of individuals in a community is divided up among different species. Understanding changes in benthic diversity is important because increasing levels of environmental stress generally decreases diversity. Two different aspects of community structure contribute to community diversity, namely species richness (calculated using the Shannon-Weiner diversity index) and equability (evenness). Species richness refers to the total number of species present, while evenness expresses how uniformly the individuals are distributed among different species. A sample with greater evenness is considered to be more diverse. It is important to note when interpreting diversity values that predation, competition and disturbance all play a role in shaping a community. For this reason, it is important to consider physical parameters as well as other biotic indices when drawing a conclusion from a diversity index.

Temporal biotic cover data were averaged across years from 2005 to 2020 at each site (Figure 10.4). Sites were sorted from left to right according to increase in wave force and the indices are calculated for the whole shore across all zones. North Bay had the highest average number of species over this period, while Jetty had the lowest, although there was no clear trend across the wave exposure gradient. In contrast, average biotic cover increased among the shores with intensifying wave force from ~15% cover at Dive School to >50% cover at North Bay and Marcus Island, although dips in biotic cover were observed for Lynch Point and Marcus Island. This trend was not evident for evenness and Shannon-Wiener diversity, although the site at Marcus Island had the lowest values for evenness and the site at Iron Ore terminal had the lowest Shannon-Weiner Diversity. This indicated low overall diversity but higher variation in communities over the years, which may be an indication of disturbance.

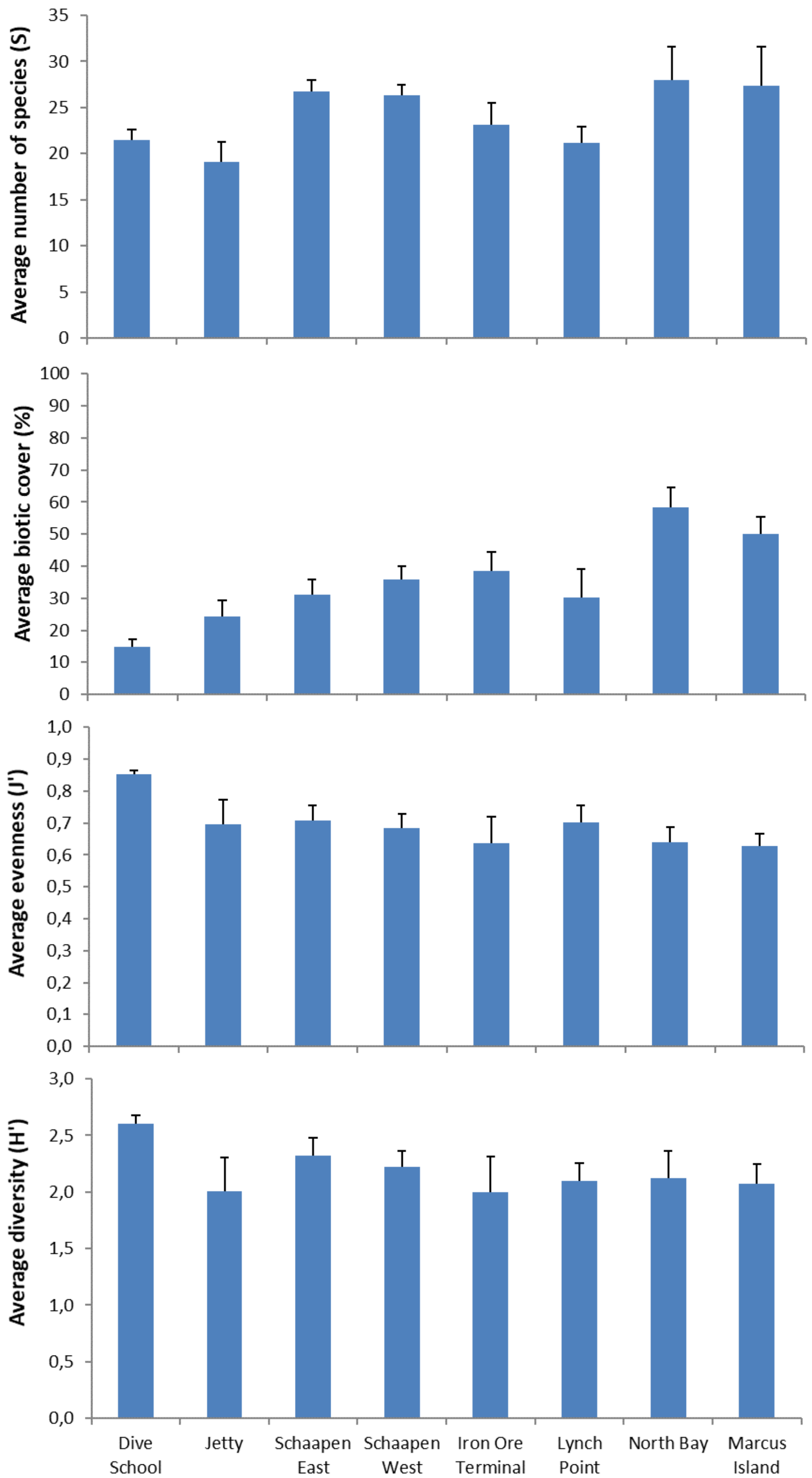


Figure 10.4. Spatial biotic cover data from 2005 – 2020 averaged across years and displayed as biotic indices of 'species number' (S), 'biotic cover' (N), 'diversity' (d) and 'evenness' (J'). Error bars indicate standard error.

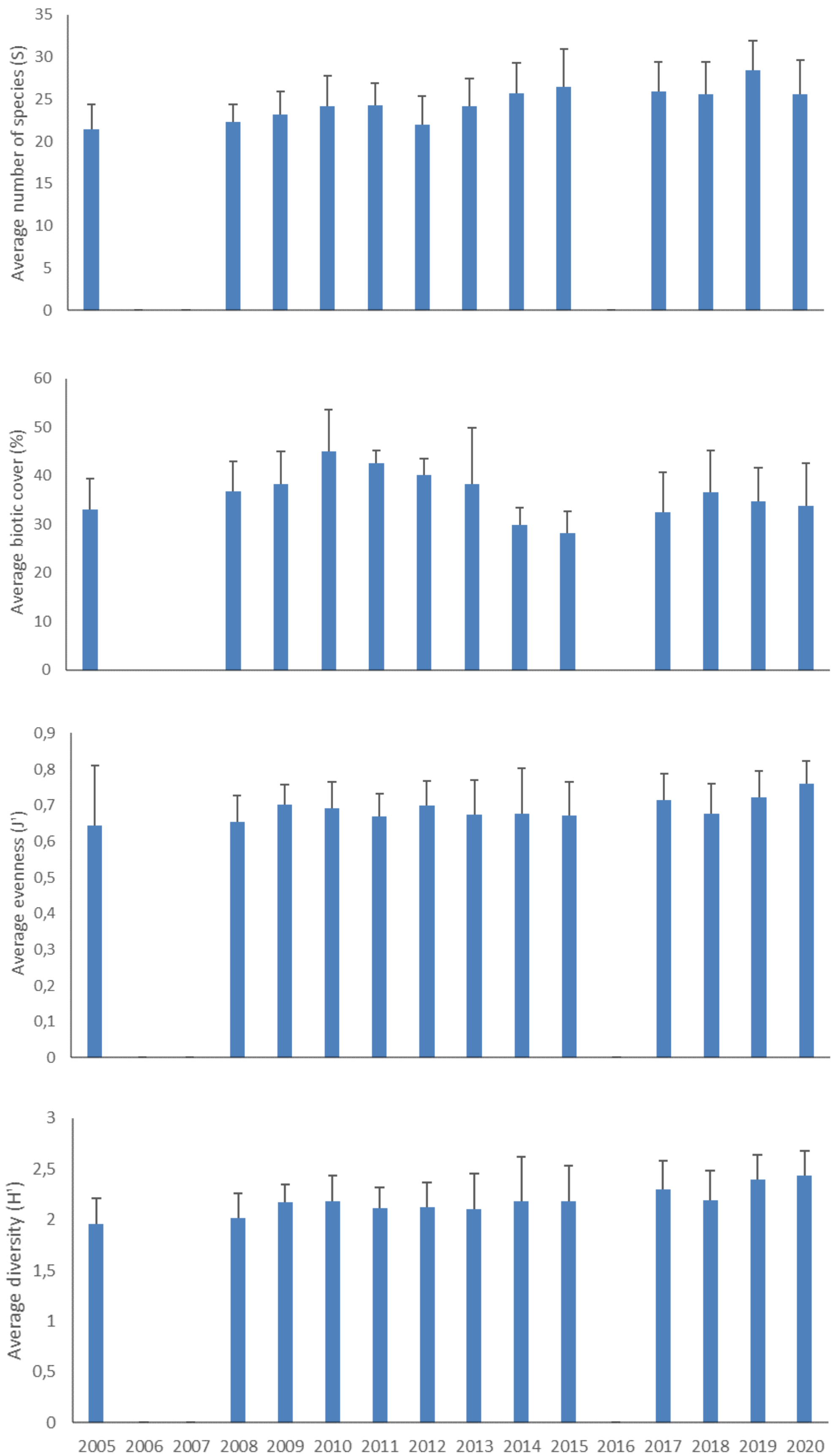


Figure 10.5. Temporal biotic cover data from 2005 – 2020 averaged across sites and displayed as biotic indices of 'species number' (S), 'biotic cover' (N), 'diversity' (d) and 'evenness' (J'). Error bars indicate standard error.

10.3.2 Temporal analysis

10.3.2.1 Temporal analysis of diversity indices

Diversity indices were averaged across sites from 2005-2020 (Figure 9.5). No clear trend in diversity indices can be seen with averages remaining somewhat consistent over time. Although, biotic cover dipped most notably in 2014 and 2015 but with higher average numbers of species than previous years and similar evenness and diversity values. Thus, even though communities were diverse over these years, the total number of individuals across species was low. In 2020, higher numbers of diversity and evenness are observed with a dip in the average number of species and an average biotic cover that is similar to previous years. Jetty, Schaapen West and North Bay have a greater total number of species compared to the previous survey and the remining sites experienced a decrease in the total number of recorded species. This may be due to several factors such as settlement rate, substratum availability and wave exposure where planktonic larvae are more likely to settle on spatially complex surfaces so as not to be washed away by waves (Guarnieri *et al*. 2009). However, other influences such as predation and competition should also be considered in conjunction with the aforementioned factors.

10.3.2.2 Temporal trends in rocky shore community patterns

PERMANOVA tests conducted for each site confirmed significant differences among the years ($p = 0.001$ for all tests). Pair-wise tests further reveal that for every site-by-year combination tested, inter-annual changes in community composition were significant ($p>0.5$).

Temporal trends in rocky shore community patterns are illustrated in the MDS plot (Figure 10.5). Consistent for all years is the grouping according to wave exposure, with the cluster on the left of the MDS plot grouping all samples from the more exposed sites (Iron Ore Terminal, Lynch Point, North Bay, and Marcus Island), a cluster in the centre grouping the semi-exposed sites (Schaapen Island East and West) and a cluster on the right grouping samples from the sheltered sites (Dive School and Jetty). Within the exposed cluster, a separation of Iron Ore Terminal from the other three sites is apparent.

Inter-annual variability within each site is also evident, but this is more pronounced for some of the sites than for others. At Dive School, samples from 2013 tend to be on the right of the cluster, while those from 2005 are on the left (Figure 10.5). The greatest within-site variability (or patchiness) occurs at the boulder site Jetty where the replicates per year often disperse widely. Due to the high stress level of 0.23, the MDS plot needs to be interpreted with caution, but there is good agreement with the pattern observed between years, suggesting that the representation is fairly reasonable.

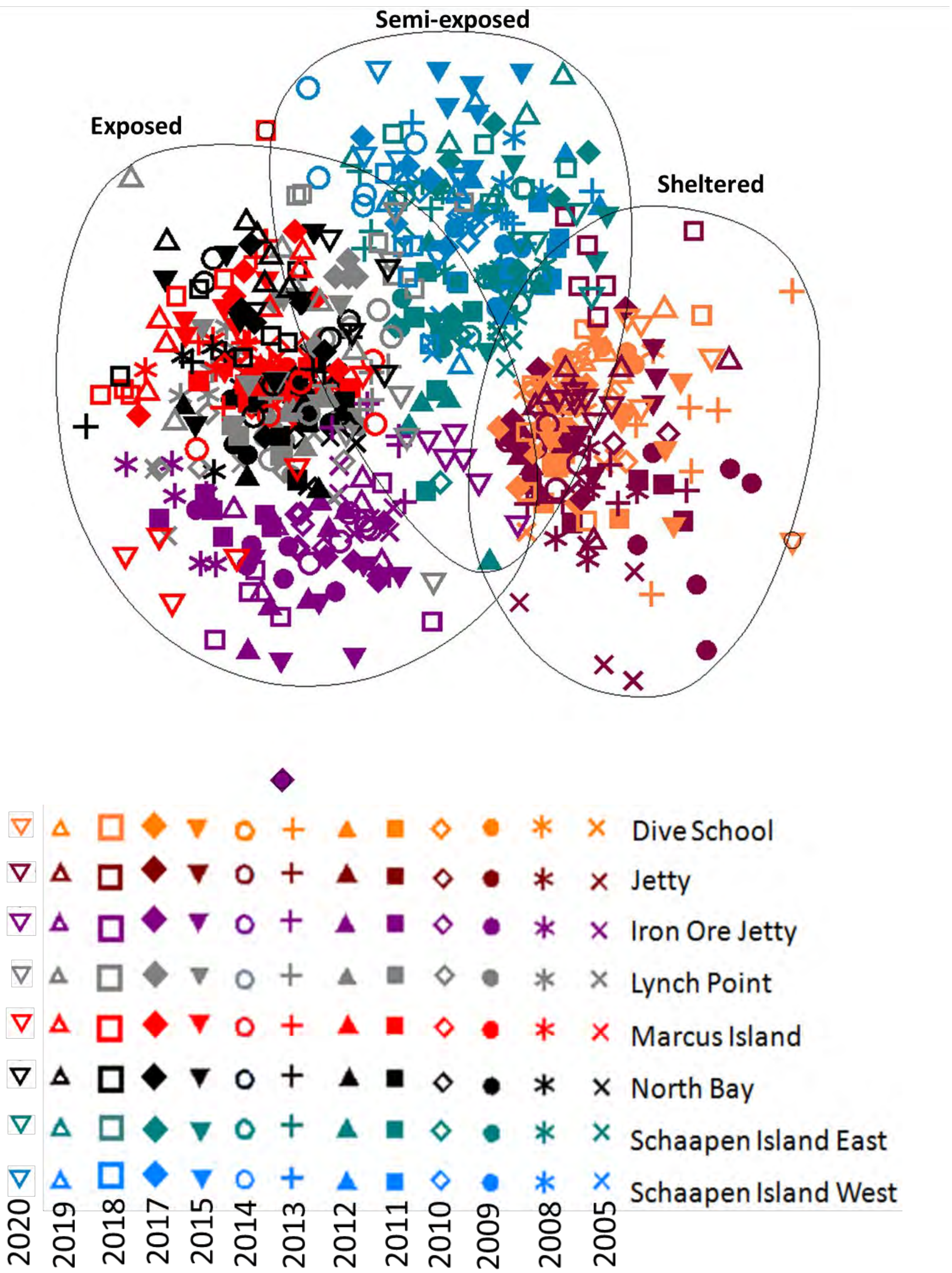


Figure 10.5. Multi-dimensional scaling (MDS) plot of the rocky shore communities at the eight study sites from 2005 to 2020. The circles delineate a 38% similarity level and the plot has with a 2D stress of 0.23.

10.3.2.3 Species responsible for temporal trends

The species that are primarily responsible for the observed differences in community structure among the years are identified by the SIMPER routine. For brevity, only species contributing >4% to the dissimilarity at any specific site and only comparisons between 2019 and the current dataset from 2020 are presented (Table 10.1). At half of the sites, more than one species contributed largely (>4%) to the differences in community structure between 2019 and 2020. Dive school and Schaapen East had two largely contributing species; Jetty and Marcus Island had five and no single species contributed >4% for the remaining sites (Schaapen West, Iron Ore terminal, Lynch Point, North Bay, Table 10.1). For the latter sites, the species contributing the most to the dissimilarity was listed.

Notable changes in species composition included the appearance of three species (*Choromytilus meridionalis, Spirorbis* spp *and Corynactis annulata)* that were not present at these specific sites the previous year; the disappearance of five species (*Scutellastra tabularis,*, *Mytilus galloprovincialis,* Sandy tube worm*,* Diatoms and *Notomegabalanus algicola),* the increase in abundance of four species (*Balanus glandula, Porphyra capensis, Ralfsia verrucosa* and *Hildenbrandia spp.)* and the decrease in abundance of a further two species (*Aulacomya atra and Coralline (crustose)*)(Table 10.1).

The common contributor species to differences observed between years were invertebrates, of which nine were listed. Most are filter feeders and the remaining one (*S. tabularis*) a grazer. This is the first appearance of the indigenous mussel *C. meridionalis* at Dive School and Jetty since being recorded in previous years, while the tubeworm *Spirorbis spp*. and the strawberry anemone *C. annulata* have been recorded at Jetty and Schaapen West in 2018 but not 2019. Percentage cover of the alien barnacle *B. glandula* increased from 2019 to 2020 at Jetty and North Bay while the alien mussel *M. galloprovincialis* was absent from Jetty (Table 10.1). *S. tabularis* was recorded at Dive School in 2019 but was absent in 2020. Fluctuations in the abundance of larval species on the rocky intertidal are not unexpected as the success of larval supply and settlement varies naturally both seasonally and inter-annually.

Of the five algal species listed, three were ephemeral algae, one was corticated algae and one was encrusting algae. Diatoms, which often temporarily cover high shore rocks until succeeded by macroalgae (Robles 1982, Cubit 1984, Maneveldt *et al.* 2009), contributed to differences between the years at Schaapen East and Marcus Island where both were absent; while the increase in cover of *P. capensis* contributed to the difference between the years at Schaapen East. Encrusting *coralline* cover decreased at Marcus Island but the ephemeral alga *Hildenbrandia spp*. increased in abundance at this site.

Table 10.1. SIMPER results listing the species that contribute >4% to the dissimilarity between 2019 and 2020 at each site. The percentage cover data presented are averages across the six replicates per site and are on the fourth-root transformed scale.

Site	Species	2019 %cover	2020 %cover	Ave. dissimilarity	% Contribution	Ave. dissimilarity between years
Dive School	*Choromytilus meridionalis*	0	0.85	2.47	5.07	48.64
	Scutellastra tabularis	0.77	0	2.25	4.62	
Jetty	*Mytilus galloprovincialis*	0.75	0	2.29	4.90	46.68
	Balanus glandula	0.33	1.06	2.23	4.77	
	Spirorbis spp	0	0.69	2.11	4.52	
	Sandy tube worm	0.66	0	2.04	4.37	
	Choromytilus meridionalis	0	0.65	1.97	4.22	
Schaapen East	Diatoms	1.17	0	2.52	4.65	54.24
	Porphyra capensis	0.21	1.23	2.23	4.12	
Schaapen West*	*Corynactis annulata*	0	0.81	1.80	3.47	51.85
Iron ore terminal*	Ralfsia verrucosa	0.22	1.08	1.68	2.99	56.11
Lynch Point*	*Hildenbrandia spp.*	0.77	1.85	2.29	3.97	57.63
North Bay*	*Balanus glandula*	0.82	1.13	1.63	3.73	43.72
Marcus Island	*Diatoms*	1.69	0	3.57	5.88	60.81
	Aulacomya atra	1.47	0.12	2.94	4.83	
	Coralline (crustose)	1.50	0.25	2.74	4.50	
	Hildenbrandia spp.	0.62	1.64	2.67	4.38	
	Notomegabalanus algicola	1.08	0	2.44	4.02	

* Note that at sites marked with an asterisk none of the species contributed >4% to the dissimilarity. The species with the highest contribution is thus listed.

10.3.2.4 Temporal variations in abundance of functional groups

Many studies have been conducted worldwide focusing on the effect of wave action on the distribution of organisms on rocky shores (Lewis 1964, McQuaid & Branch 1984, Raffaelli & Hawkins 1996, Bustamante *et al.* 1997, Menge & Branch 2001, Denny & Gaines 2007). Increasing exposure reduces siltation and increases the supply of dissolved oxygen and particulate food, favouring certain sessile, filter-feeding species and leading to an elevation of overall biomass (McQuaid & Branch 1985, Bustamante & Branch 1996, Bustamante *et al.* 1995, Steffani & Branch 2003a). Although increasing exposure carries an increased risk of dislodgement and physical damage thus limiting the range of susceptible and physically fragile species, Pfaff *et al.* (2011) showed that wave exposure has an overall positive effect on the recruitment of mussels and barnacles on the southern African west coast. In contrast, sheltered shores are typically dominated by algae (McQuaid & Branch 1985) as species richness of most algal groups decrease with increasing exposure. The effect of wave exposure, however, varies with phyla and functional form group as some forms can better withstand hydrodynamic forces than others (Denny & Gaylord 2002, Nishihara & Terada 2010).

Despite adaptations evolved as a result of different wave exposures, hydrodynamic forces can at times cause massive damage to rocky shore communities, fundamentally altering the structure and function of exposed rocky habitats and creating changes that may persist for many years. The magnitude and frequency of physical disturbance is not as severe on protected shores as on exposed shores; thus the structure of protected communities is often more stable than that of exposed assemblages. The rocky shores at Saldanha Bay are subject to a range of wave forces from very sheltered to exposed.

While wave force is clearly the main factor for differences among the shores, shore topography is also of importance. The roughness of the substratum or generally termed habitat structure can be a crucial factor driving species richness, abundance and even body size (Kostylev *et al.* 2005). According to McCoy and Bell (1991), habitat structure is generally thought to have two independent components: complexity (the physical architecture of a habitat) and heterogeneity (the relative abundance of different structural features such as boulders or crevices within a habitat). Several studies have shown that many mobile animals exhibit preferential movement from smooth surfaces into habitats with more structural complexity (e.g. crevices) where they are more protected from hydrodynamic forces (McGuinness & Underwood 1986, Kostylev *et al.* 2005, O'Donnell & Denny 2008). This does not apply only to physical complexity, but also microhabitats offered by biota (e.g. the barnacle *Balanus glandula*). Mobile invertebrates can respond to environmental extremes by moving between microhabitats to ameliorate thermal and desiccation stress (Meager *et al.* 2011).

The distribution of sessile species is largely driven by the longer-term processes of settlement, growth and mortality; whereby substratum availability, micro-topography and surface smoothness can be limiting factors at local scales (Guarnieri *et al.* 2009). Topographic complexity influences the settlement of benthic organisms as planktonic larvae are more likely to be retained on rough surfaces, while water movement may wash them off smooth surfaces (Eckman 1990, Archambault & Bourget 1996, Skinner & Coutinho 2005, Guarnieri *et al.* 2009).

Boulder shores also have greater microhabitat diversity compared to more level shores. One of the reasons for this is because the tops of larger boulders stay exposed for a significantly longer period than smaller boulders (or flat platforms), with each boulder essentially having its own shore height zonation. During low tide, the top of the boulder provides the lower section with shade, thus maintaining lower temperatures and higher moisture content (Takada 1999). This arrangement increases the surface area for the attachment of organisms but may reduce water movement, which may cause detritus to accumulate, possibly resulting in low oxygen conditions. Large boulders can considerably reduce the water flow velocity, thus invertebrate biomass is expected to decrease significantly downstream of boulders. Smaller boulders may be unstable and often have a more impoverished community than larger rocks (McGuinness 1987, Guichard & Bourget 1998, Londoño-Cruz & Tokeshi 2007, McClintock *et al.* 2007). All these factors result in boulder fields supporting different species assemblages in comparison to those of flatter shores (Sousa 1979a, McGuinness 1984, McQuaid *et al.* 1985, McGuinness & Underwood 1986, Takada 1999, Cruz-Motta *et al.* 2003, Davidson *et al.* 2004, Hir & Hily 2005).

Shore topography is a likely reason for differences in community structure between the rocky shores on Schaapen Island and the other two sheltered sites, although it may also be related to the fact that Schaapen Island lies in the transition zone between Saldanha Bay and Langebaan Lagoon.

The water in the Lagoon has slight differences in water quality (e.g. temperature) compared to the water in the Bay, which in turn leads to differences in their biological communities (Day 1959, Robinson *et al.* 2007b). For example, Schils *et al.* (2001) report a distinct separation in algal composition between the Bay and the Lagoon as the Lagoon contains a significant number of south coast species due to its warmer waters. Perlemoen Punt, located less than one kilometre from Schaapen Island at the entrance to Langebaan Lagoon, is described as the transition area between the Bay and the Lagoon, but with a marked Lagoon affinity in its overall algal composition. Clear differences in community composition between the Bay and the Lagoon are also described for zooplankton and sandy substrate assemblages (Grindley 1977, Anchor Environmental Consultants 2012b).

The biotic cover of the various functional groups across the shores with regard to exposure is depicted in Figure 10.7 with sites arranged from very sheltered to exposed. Very sheltered shores had generally low biotic cover consisting primarily of grazers, corticated algae and encrusting algae, with the exception of Schaapen Island that had high biotic cover and was clearly dominated by algae. With an increase in wave force, the dominance of sessile filter feeders (e.g. barnacles) was evident.

At the two sheltered sites (Dive School and Jetty), filter feeders and ephemeral algae slightly decreased over time, while corticated algae, encrusting algae and grazers increased slightly. At both Schaapen Island sites, the abundance of ephemerals and encrusting algae varied considerably over the years but without a consistent trend. In 2010 and 2011, filter feeders at the Schaapen Island sites had increased in cover to >10% averaged across the whole shore but declined again from 2012 onwards. Iron Ore Terminal and Lynch Point remained relatively constant over time, with only minor variations in encrusting algae and ephemeral cover, although biotic cover was high at Iron Ore Jetty in 2018 due to an increase in filter feeders. At North Bay, filter feeders increased slightly over time with a slight drop in cover in 2012. Ephemerals again showed slight temporal fluctuations, with encrusting algae increasing noticeably in 2014 but decreasing again in 2015. At Marcus Island, ephemeral algae had greatly increased from 2008 to 2009, while at the same time corticated algae, encrusting algae and filter feeders declined. This substantial increase in ephemeral cover resulted in greater biotic cover overall in 2009. In 2010, ephemerals had somewhat reduced but returned again in 2011. There was no noteworthy change in functional groups in 2012 but encrusting algae and kelp increased substantially in 2013, decreasing again in 2014. Ephemeral algae increased substantially at Schaapen Island West, Iron Ore Terminal and Marcus Island in 2017 but decreased at Schaapen Island West in 2018. In 2019, densities of biota were slightly lower than the previous year at Lynch Point and North Bay, while more encrusting alga was recorded at Marcus Island. In 2020, there was a decrease in overall biota abundance except for Lynch Point and Schaapen West which had a substantial increase in encrusting and corticated algae. The exposed and semi-exposed sites; North Bay, Marcus Island and Iron Ore Jetty had a noticeable decrease in filter feeders but these sites, as well as Schaapen East, also had an increase in ephemeral algae which usually displace filter feeders.

Overall, none of the sites indicated a temporal change in their rocky shore communities that would suggest a dramatic alteration such as the arrival or loss of a key species. Instead, the intertidal communities show temporal fluctuations that reflect mostly the dominance of ephemerals over one or more years, often with a concomitant decline in filter feeders (e.g. Schaapen West in 2008). Ephemeral algae are usually the first to colonize rock space denuded of biota due to physical (e.g. wave action) or biological (e.g. grazing) disturbance. In the ecological succession that follows, ephemerals are then replaced by longer-lived late successional species (Sousa 1979b, 1984). No major pollution events or point sources of pollution are apparent in these data and the slight fluctuations of functional groups over the years are a natural seasonal and inter-annual phenomenon.

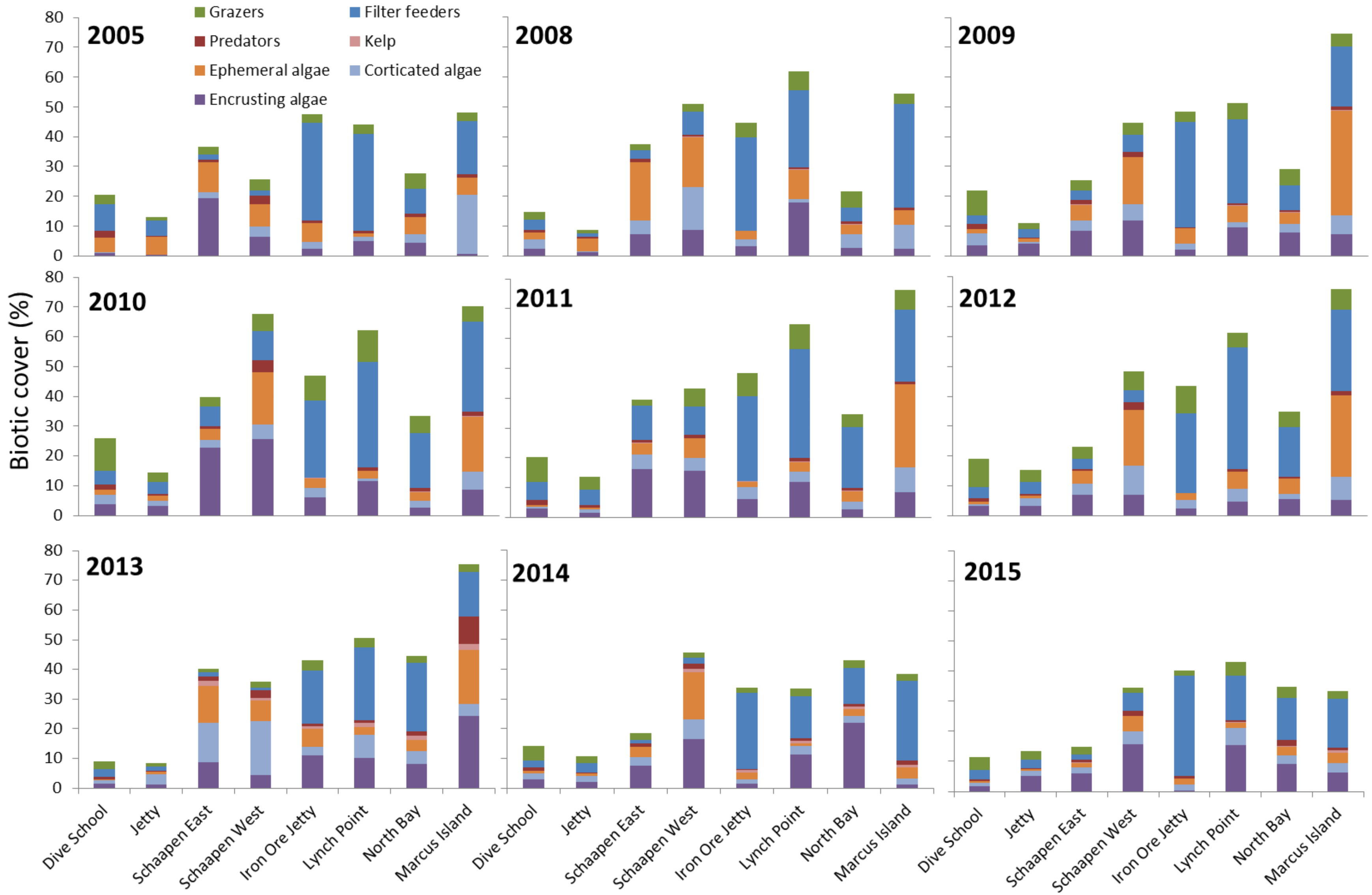


Figure 10.6. **Total percentage cover (averaged across the whole shore) of the seven functional groups at the eight study sites from 2005 to 2015.**

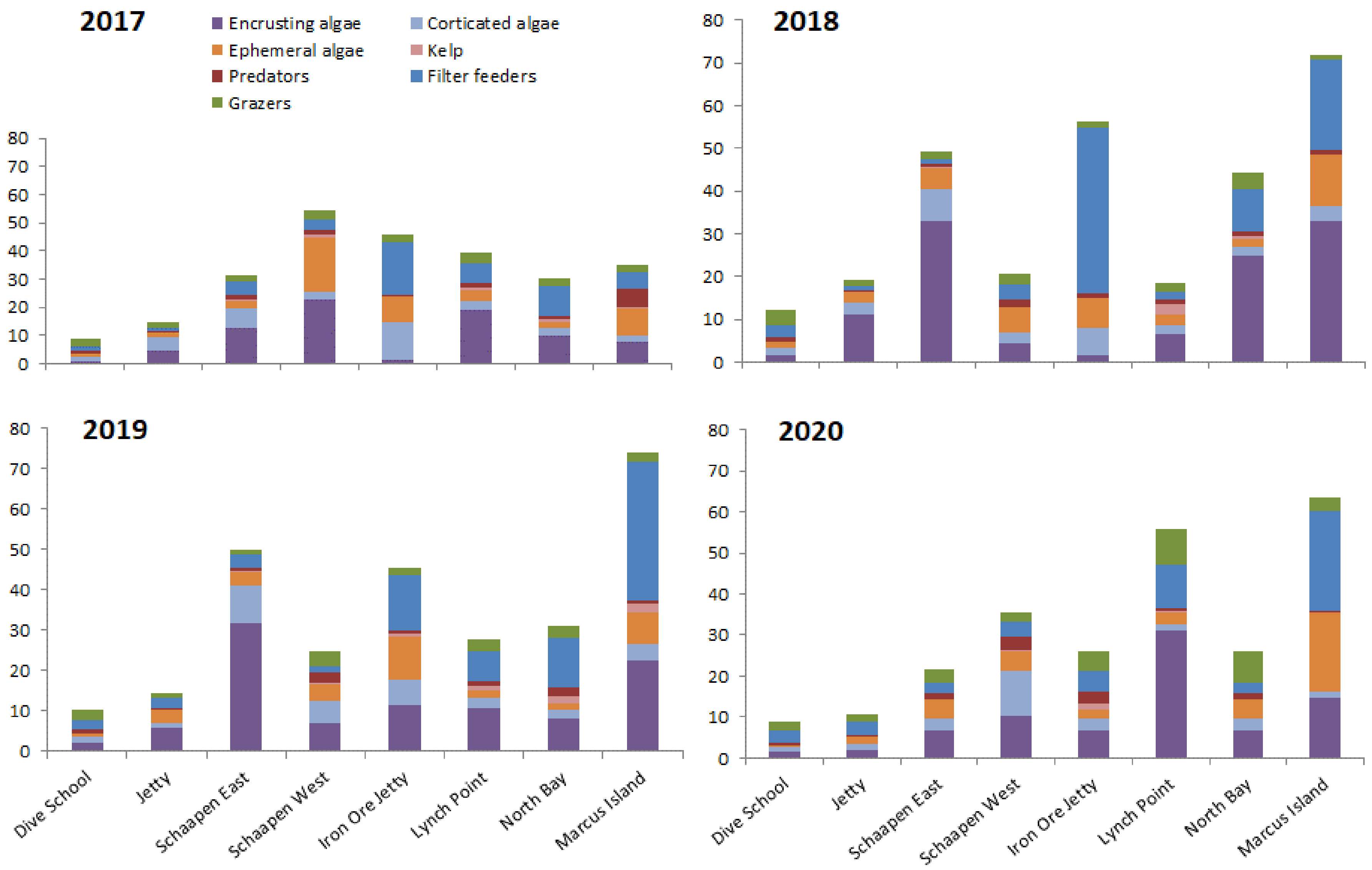


Figure 10.7. **Total percentage cover (averaged across the whole shore) of the seven functional groups at the eight study sites from 2017 to 2020.**

10.3.3 Summary of findings

In 2020, a total of 100 taxa were recorded from the eight study sites, most of which had been found in previous survey years. The faunal component was represented by 23 species of filter-feeders, 23 species of grazers, and 14 species of predators and scavengers combined. The algal component comprised 24 corticated (foliose) seaweeds, 9 ephemerals, 5 species of encrusting algae and 2 species of kelp. The species recorded in this report are generally common to the South African West Coast and many, including two alien invasive species the Mediterranean mussel *Mytilus galloprovincialis* and the North American acorn barnacle *Balanus glandula*, and two alien barnacle species *Perforatus perforatus and Amphibalanus amphitrite* , are listed by other studies conducted in the Saldanha Bay area.

Within a site, the vertical emersion gradient of increasing exposure to air resulted in a clear zonation of flora and fauna from low shore to high shore. Differences among the rocky shores, however, were strongly influenced by the prevailing wave exposure as well as substratum topography. Very sheltered shores had generally low biotic cover consisting primarily of grazers, with minor cover of sessile filter feeders and algae. Sheltered shores were dominated by seaweeds and encrusting corallines. With increasing wave exposure, filter feeders were clearly the most important group. The two very sheltered sites in Small Bay separate out from the flat Schaapen Island sites, a result which may be related to geographic location as Schaapen Island lies in a transitional zone between the Bay and the Lagoon. Another contributing factor may be the substantial nutrient input in the form of seabird guano that enters the sea via runoff from Schaapen and Marcus Islands favouring algal growth in these areas. The steep boulder beach at the Iron Ore Terminal has high biotic cover, most likely due to the complex artificial habitat with many cracks and crevices available for shelter when compared to the more flattish semi-exposed sites of natural bedrock.

From the temporal variation evident in the rocky shore communities, it appears that there is no directional shift in community composition that would indicate a persistent change, such as the permanent loss of a species. Instead the communities demonstrate temporal fluctuations, reflecting the temporary dominance of short-lived ephemeral species and/or inter-annual variation in larval supply or recruitment success. In general, rocky shore communities were relatively stable with only minor changes over the years.

The two most important filter feeders were the aliens *M. galloprovincialis* and *B. glandula.* These were the characteristic species at most shores and zones, although the barnacle appeared to be declining in abundance over time with only empty shells and base plate scars left on rocks at some sites but has slightly increased in the present survey. The latter species is most abundant in the mid shore zone of semi-exposed sites, but rarer at exposed sites and low shores. *M. galloprovincialis*, on the other hand, is most abundant at wave-exposed sites and lower down the shore. One of the greatest threats to rocky shore communities in Saldanha Bay is the introduction of alien species via shipping, and their potential to become invasive (see Chapter 13 for detailed information on invasive species).

11 FISH COMMUNITY COMPOSITION AND ABUNDANCE

11.1 Introduction

The waters of Saldanha Bay and Langebaan Lagoon support an abundant and diverse fish fauna. Commercial exploitation of the fish within the Bay and lagoon began in the 1600s by which time the Dutch colonists had established beach-seine fishing operations in the region (Poggenpoel 1996). These fishers targeted harders *Chelon richardsonii* and other shoaling species such as white steenbras *Lithognathus lithognathus* and white stumpnose *Rhabdosargus globiceps*. Most of the catch was dried and salted for supply to the Dutch East India Company boats, troops and slaves at the Castle in Cape Town (Griffiths *et al.* 2004). Commercial netfishing continues in the area today, and although beach-seines are no longer used, gill-net permits holders continue to target harders. Species such as white stumpnose, white steenbras, silver kob *Argyrosomus inodorus*, elf *Pomatomus saltatrix*, steentjie *Spodyliosoma emarginatum*, yellowtail *Seriola lalandi* and smooth hound shark *Mustelus mustelus* support large shore angling, recreational and commercial boat line-fisheries which contribute significantly to the tourism appeal and regional economy of Saldanha Bay and Langebaan. In addition to the importance of the area for commercial and recreational fisheries, the sheltered, nutrient rich and sun warmed waters of the Bay provide a refuge from the cold, rough seas of the adjacent coast and constitute an important nursery area for the juveniles of many fish species that are integral to ecosystem functioning.

The importance and long history of fisheries in the Bay and Lagoon, has led to an increasing amount of scientific data on the fish resources and fisheries in the area. Early studies, mostly by students and staff of the University of Cape Town, investigated fish remains in archaeological middens surrounding Langebaan Lagoon (Poggenpoel 1996), whilst many UCT Zoology Department field camps sampled fish within the lagoon (unpublished data). Gill net sampling with the aim of quantifying bycatch in the commercial and illegal gill net fishery was undertaken during 1998-99 (Hutchings & Lamberth 2002b). A once-of survey for small cryptic species utilizing rotenone, a fish specific, biodegradable toxin that prevents the uptake oxygen by small fish, was conducted by Anchor Environmental Consultants during April 2001 (Awad *et al.* 2003). The data from the earlier gill netting and rotenone sampling survey was presented in the “State of the Bay 2006” report (Anchor Environmental Consultants 2006). Seine-net sampling of near-shore, sandy beach fish assemblages was conducted over short periods during 1986-1987 (UCT Zoology Department, unpublished data), in 1994 (Clark 1997), and 2007 (Anchor Environmental Consultants, UCT Zoology Department). Monthly seine-net hauls at a number of sites throughout Saldanha Bay-Langebaan over the period November 2007 - November 2008 were also conducted by UCT M.Sc. student Clement Arendse who was investigating white stumpnose recruitment.

Other recent research on the fish fauna of the area includes acoustic tracking and research on the biology of white stumpnose, hound sharks and elf within Langebaan lagoon and Saldanha Bay; monitoring of recreational shore and boat angler catches and research on the taxonomy and life history of steentjies and sand sharks and (Næsje *et al.* 2008, Kerwath *et al.* 2009, Tunley *et al.* 2009, Attwood *et al.* 2010, Hedger *et al.* 2010, da Silva *et al.* 2013). Key findings of these studies include evidence that the Langebaan lagoon Marine Protected Area (MPA) provides some protection for white stumpnose, during the summer months that coincides with both peak spawning and peak recreational fishing effort (Kerwath *et al.* 2009). Elf and smooth hound sharks were also shown to derive protection from the MPA, with tagged individuals of both species spending the majority of the study period (up to 2 years) within the MPA boundaries, and indeed a high degree of residency within Saldanha Bay as a whole (Hedger *et al.* 2010, da Silva *et al.* 2013). Tagged elf did show a long-term movement out of the lagoon into the Bay and one individual was recaptured in Durban confirming that long distance migration does take place (Hedger *et al.* 2010). However, the fact that nearly all fish within the Bay were resident for the one to two years after tagging and the presence of young of the year juveniles in the surf zone, suggests that elf within Saldanha Bay exhibit a mixed evolutionary strategy with migratory and resident spawning components (Hedger *et al.* 2010). Out of the 24 hound sharks acoustically tagged within Langebaan lagoon, 15 were monitored for more than 12 months and two of these did not leave the MPA at all. Six of these tagged hound sharks left the Saldanha embayment for the open coast, during spring and winter for periods of between two to 156 days, but all returned during the study period. These acoustic telemetry studies have clearly demonstrated that these three priority fishery species all derive protection from the Langebaan MPA. The research on sand sharks suggests that the common sand shark species in Bay and Lagoon is actually *Rhinobatos blockii*, not *R. annulatus* as previously thought (Dunn & Schultz, UCT Zoology Department *pers. comm.*).

White stumpnose within the Saldanha-Langebaan system grow more rapidly and mature earlier than populations elsewhere on the South African coast (Attwood *et al.* 2010). Male white stumpnose in Saldanha Bay reach maturity in their second year at around 19 cm fork length (FL) and females in their third year at around 22 cm FL (Attwood *et al.* 2010). Similar differences in growth rate and the onset of maturity for steentjies between Saldanha Bay and south coast populations were reported by Tunley *et al.* (2009). These life history strategies (relatively rapid growth and early maturity) in combination with the protection afforded by the MPA are probably part of the reason that stocks fishery species in Saldanha and Langebaan had until recently been resilient to rapidly increasing recreational fishing pressure (but see paragraph below on stock status). Results from angler surveys undertaken during the early 2000's indicated that approximately 92 tonnes of white stumpnose was landed by anglers each year (Næsje *et al.* 2008).

Recent studies on the stock status of white stumpnose, the most important angling species within Saldanha-Langebaan, however, shows that the stock is fully exploited or overexploited, suggesting that the Langebaan MPA alone may not be enough to prevent stock collapse with the observed increases in fishing pressure (Arendse 2011, Parker et al 2017). Arendse (2011) used catch-at-age data from the boat fishery and per-recruit modelling to estimate that spawner biomass at the time (2006-2008) was less than 25% of pristine. The target reference point for optimally exploited stocks is 40-50% of pristine biomass, and Arendse (2011) calculated that a 20% reduction in fishing mortality was required to achieve this target. It was recommended by Arendse (2011) that a reduction in bag limit from 10 to 5 fish per person per day, or an increase in size limit to 29 cm Total Length (TL) be implemented. These management measures were modelled to rebuild spawner biomass to the 40-50% target, but unfortunately, have not been implemented to date. Parker *et al.* (2017) provide an updated analysis of angler survey data, commercial linefish catch returns and the juvenile white stumpnose catch in the seine net surveys, which conclusively demonstrate substantial declines in both adult and juvenile abundance estimates over the last decade. These authors also urge that a reduction in bag limit and increase in size limit are required to sustain the Saldanha Bay white stumpnose fishery.

The most recent published research on fish of the Saldanha Bay system was an investigation of the age, growth and stock assessment of the harder *Chelon* (previously *Liza*) *richardsonii* stock (Horton et al 2019). Results of this study show that gill net fishers have seen substantial declines in harder catches and Catch-Per-Unit-Effort (CPUE) over the last two decades (Horton *et al*. 2019). By comparing monitored landings with reported catches, Hutchings & Lamberth (2002a); estimated that 590 tonnes of harders, valued at approximately R1.8 million was landed during 1998-1999. The reported catch has declined from around 130 tonnes per year over the period 2008-2012 to about 90 tonnes per year over the period 2013-2016, whilst effort remained fairly constant (Horton *et al*. 2019). The average size of harders in catches has declined significantly over the same period and a stock assessment indicated that the stock was at risk of recruitment failure (the current spawner biomass was estimated at less than 25% of the pristine level).. A reduction in fishing mortality (approximately 30%) and an increase in mesh size to 51mm were recommended to help rebuild the stock (Horton *et al*. 2019).

SANPARKS and UCT researchers are currently undertaking research on the movement patterns of broad nosed seven gill cow sharks *Notorynchus cepedianus* and have acoustically tagged animals within Saldanha Bay and deployed listening stations (acoustic receivers) near the mouth of the bay to record the passage of any tagged individuals into and out of the Bay (A. Kock SANParks personal communication). In February 2019, SANParks also implemented a fish, shark and ray monitoring project in the West Coast National Park MPA using Baited Remote Underwater Video cameras (BRUVs). BRUVs are a non-destructive and cost-effective monitoring tool which is popularly used along the South African coastline. The aim of this monitoring programme is to describe relative abundance and diversity of fish and sharks across different management zones of the WCNP MPA. To date, SANParks have deployed 72 BRUVs across all three MPA zones (and in Saldanha Bay) and across summer and winter seasons. In total, 17 different species have been detected on the BRUVs, including vulnerable and commercial species, like white stumpnose, blacktail, elf and smooth hound sharks. The deployments to date have demonstrated that BRUVs are a valuable monitoring tool in the lagoon for key fisheries species. These video surveys that attract adult fish in deeper water to the baited cameras are a great complement to the annual seine net surveys that largely capture juvenile fish in surf zone nursery habitats. Analysis of the data is underway to compare diversity and abundance across the MPA zones. When results from these analyses becomes available, we hope to be able to share some findings in the annual SOB report

The Saldanha Bay Water Quality Forum Trust (SBWQFT) commissioned Anchor Environmental to undertake experimental seine-net sampling of near shore fish assemblages at a number of sites throughout the Saldanha-Langebaan system during 2005, and annually over the period 2008-2020 as part of the monitoring of ecosystem health “State of the Bay” programme. Seine-net surveys were conducted during late summer to early autumn, as this was the timing of peak recruitment of juveniles to the near-shore environment, as well as the timing of most of the earlier surveys. Since 2008, seine-net surveys have therefore been conducted during March-April of each year. These studies have made a valuable contribution to the understanding of the fish and fisheries of the region. This chapter presents and summarises the data for the 2020 seine-net survey and investigates trends in the fish communities by comparing this with data from previous seine-net surveys (1986/87, 1994, 2005, 2008-2019) in the Saldanha- Langebaan system.

11.2 Methods

11.2.1 Field sampling

Experimental seine netting for all surveys was conducted using a beach-seine net, 30 m long, 2 m deep, with a stretched mesh size of 12 mm. Replicate hauls (3-5) were conducted approximately 50 m apart at each site during daylight hours. The net was deployed from a small inflatable boat 30-50 m from the shore. Areas swept by the net were calculated as the distance offshore multiplied by the mean width of the haul. Sampling during 1986-87 was only conducted within the lagoon where 30 hauls were made, whilst 39 and 33 replicate hauls were made at 8 and 11 different sites during 1994 and 2005 surveys respectively in both the Bay and Lagoon. During 2007, 21 hauls were made at seven sites in the Bay and Lagoon and over the period 2008-2012, 2-3 hauls have been made at each of 15 sites every April. Since the 2013 survey, a sixteenth site was added in the lagoon at Rietbaai (Figure 11.1). Large hauls were sub-sampled on site, the size of the sub-sample estimated visually, and the remainder of the catch released alive.

11.2.1.1 Data analysis

Numbers of fish caught were corrected for any sub-sampling of large hauls that took place in the field prior to data analysis. All fish captured were identified to species level (where possible, larval fish to Family level) and abundance calculated as the number of fish per square meter sampled. The resulting fish abundance data were used for analysis of spatial and temporal patterns.

The number of species caught and average abundance of fish (all species combined) during each survey were calculated and graphed. The average abundance of the most common fish species caught in the three main areas of the system, namely Small Bay, Big Bay and Langebaan lagoon during each survey, were similarly calculated and presented graphically. The average abundance of the five most ubiquitous species in the system over all survey years was calculated and plotted for each sampling site.

Trends in the abundance of key species that are of importance in local fisheries were analysed using a one-way ANOVA and post-hoc unequal N HSD tests in the software package STATISTICA 18. Abundance data for all sites throughout the Bay were log (x + 1) transformed to account for heteroscedacity (unequal variance) prior to analysis.

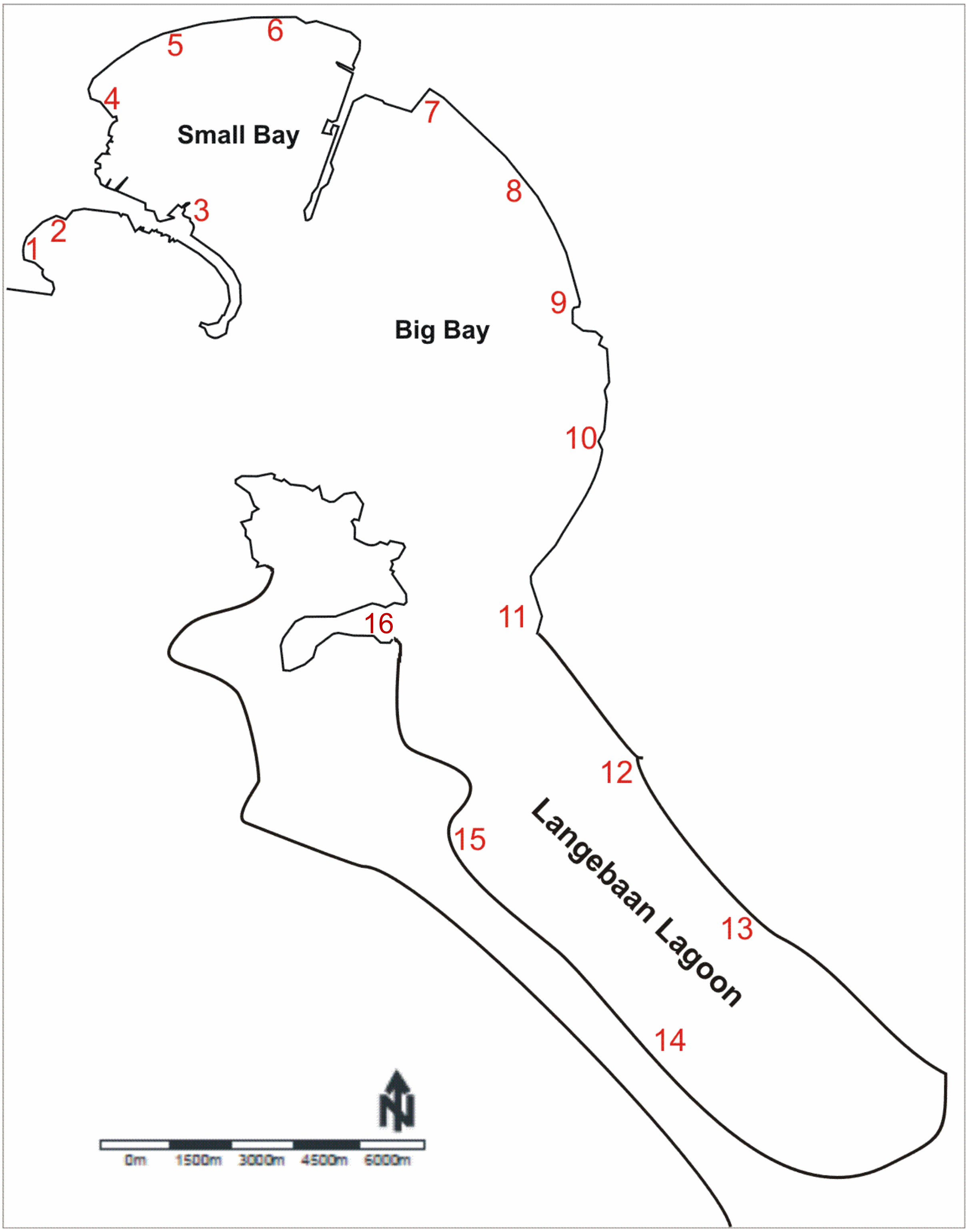


Figure 11.1. **Sampling sites within Saldanha Bay and Langebaan lagoon where seine net hauls were conducted during the 2005 and 2007-2020 annual sampling events. 1: North Bay west, 2: North Bay east, 3:Small craft harbour, 4: Hoedjiesbaai, 5: Caravan site, 6: Blue water Bay, 7: Sea farm dam, 8: Spreeuwalle, 9: Lynch point, 10: Strandloper, 11: Schaapen Island, 12: Klein Oesterwal, 13: Bottlery, 14: Churchaven, 15: Kraalbaai, 16: Rietbaai.**

11.3 Results

11.3.1 Description of inter annual trends in fish species diversity

For the first time in the 2020 survey, a juvenile galjoen *Dichistius capensis* (South Africa's national fish) was caught at the North Bay west site, taking the total species count in all surveys to date to 51. Although galjoen was historically common along the west coast, it is not surprizing that they have not been caught before in the seine net surveys which are largely conducted in the relatively calm waters inside the Bay and lagoon, whilst galjoen juveniles (and adults) are typically found in turbulent surf zones or kelp beds. Fish diversity (total number of species caught) across all surveys remains highest and similar in Big Bay (39) and in Small Bay (37), compared to the Lagoon (26). Species richness is typically similar in Small Bay and Big Bay, although the number of species sampled has been less variable over time in Small Bay (Figure 11.2.). Slightly more variation in the number of species caught over the period of sampling is apparent for Langebaan Lagoon and Big Bay, with the most diverse samples collected from Big Bay during 2012 (Figure 11.2.). In the 2020 samples, fish diversity in Big Bay was up from the historical low in 2019 and equal to the long-term average (13 species). Diversity in Langebaan Lagoon samples was also similar to the long-term average), but was the lowest recorded in Small Bay to date, with just 10 species caught (Figure 11.2.).

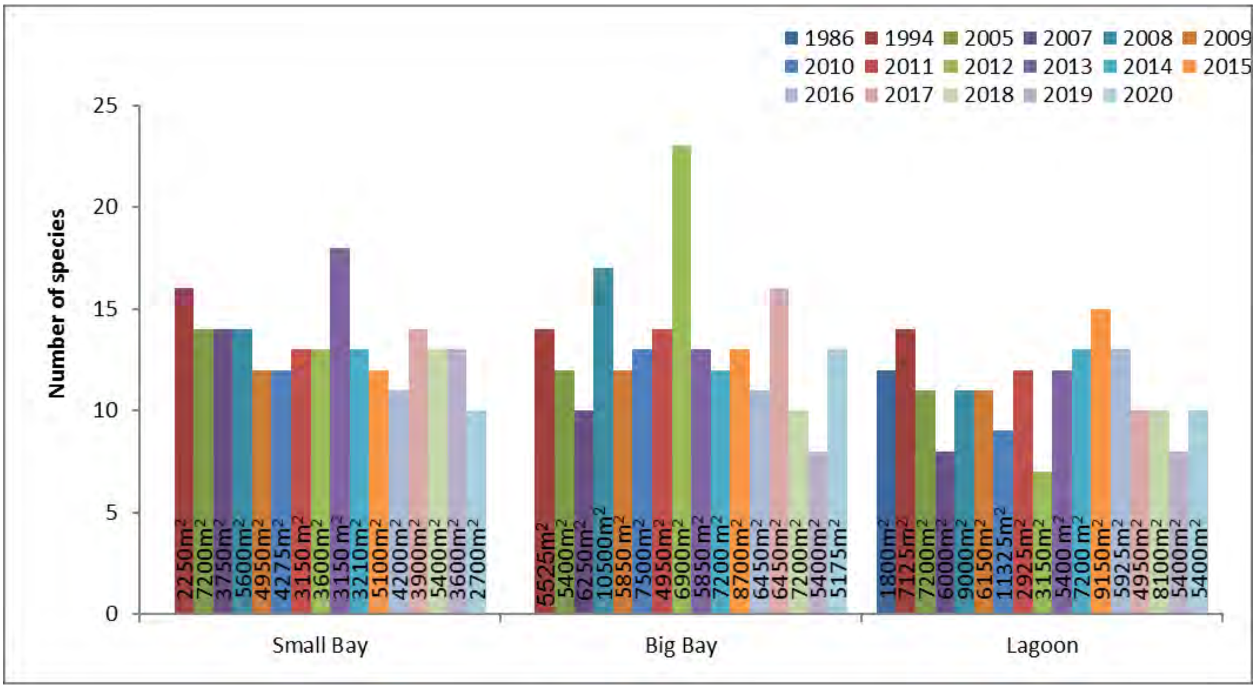


Figure 11.2. **Number of fish species caught during 17 seine-net surveys in Saldanha Bay and Langebaan lagoon conducted over the period 1986-2020. The total area netted in each area and survey is shown.**

Catch composition and abundance of more common species caught in Small Bay, Big Bay and the Lagoon during each of the different surveys are shown in Figure 11.4 - Figure 11.7. The actual species composition in the different areas does change substantially between years, but the same ubiquitous species occur in nearly all surveys. Within Small Bay, seven species (Cape silverside, *Caffrogobius*, super klipvis, Cape sole, harders, white stumpnose and bluntnose guitar fish) have occurred in all surveys to date , with two additional species, namely blacktail only absent in the 2015, 2017 and 2020 samples, and pipefish absent in the 2005, 2015 and 2020 samples. Gurnard captured in all of the first six surveys, but not over the period 2011-2014 and was again absent in the three most recent surveys (2018-2020).

Four of the 37 species recorded in Big Bay occurred in all surveys (gurnard, Cape sole, harders and white stumpnose). False Bay klipvis were only absent in the earliest 1994 survey (possibly not correctly identified), and in 2019, whilst Cape silverside has been absent from two surveys (2007 and 2018). Elf was only absent in one of the 13 surveys conducted over the period 1994-2017 and it is concerning that they have not been recorded during the last three surveys (2018-2020). Notably for the first time in the history of the 16 seine net surveys undertaken since 1994, not a single elf was captured at any of the 16 sites surveyed in 2020. This is concerning as much of the remaining boat-based, recreational fishing effort that previously focused on white stump, appears to have shifted to elf (personal observation) and three years of apparent poor recruitment to the surf zone nursery habitats does not bode well for future catches. Sand sharks were not caught in Big Bay during the 2014, 2016 and 2019 surveys, but were caught in every other survey conducted to date, including 2020. Six of the 26 species found in the lagoon (silversides, commafin gobies, Cape sole, harders, Knysna sand gobies and white stump) occurred in all surveys and these species were all present in 2020 samples. It appears that Small Bay has the highest proportion of “resident” species that are there consistently, whilst a larger proportion of the Big Bay and Langebaan Lagoon ichthyofauna occur seasonally or sporadically in these areas. Short term fluctuations in diversity and abundance of near shore sandy beach fish communities with changes in oceanographic conditions are the norm rather than the exception (see for e.g. Clark 1994). Over the past 16 sampling events average species richness has been similar in Small Bay and Big Bay (13 species) and slightly lower in the lagoon (11 species) (Figure 11.2.).

11.3.2 Description of inter-annual trends in fish abundance in Small Bay, Big Bay and Langebaan lagoon

The overall fish abundance (all species combined) shows high inter annual variability in all three areas of the Bay (Figure 11.3.). Harders and silversides numerically dominated the catches for all surveys and large variation in the catches of these abundant shoaling species is the main cause of the observed variability between years. Overall, the catches made during the 2012 survey were the lowest on record for all three areas. Over the last seven years, 2014-2020, the overall abundance of fish has compared favourably with earlier surveys, but as mentioned above, this largely reflects the abundance of harders and silversides (Figure 11.3.).

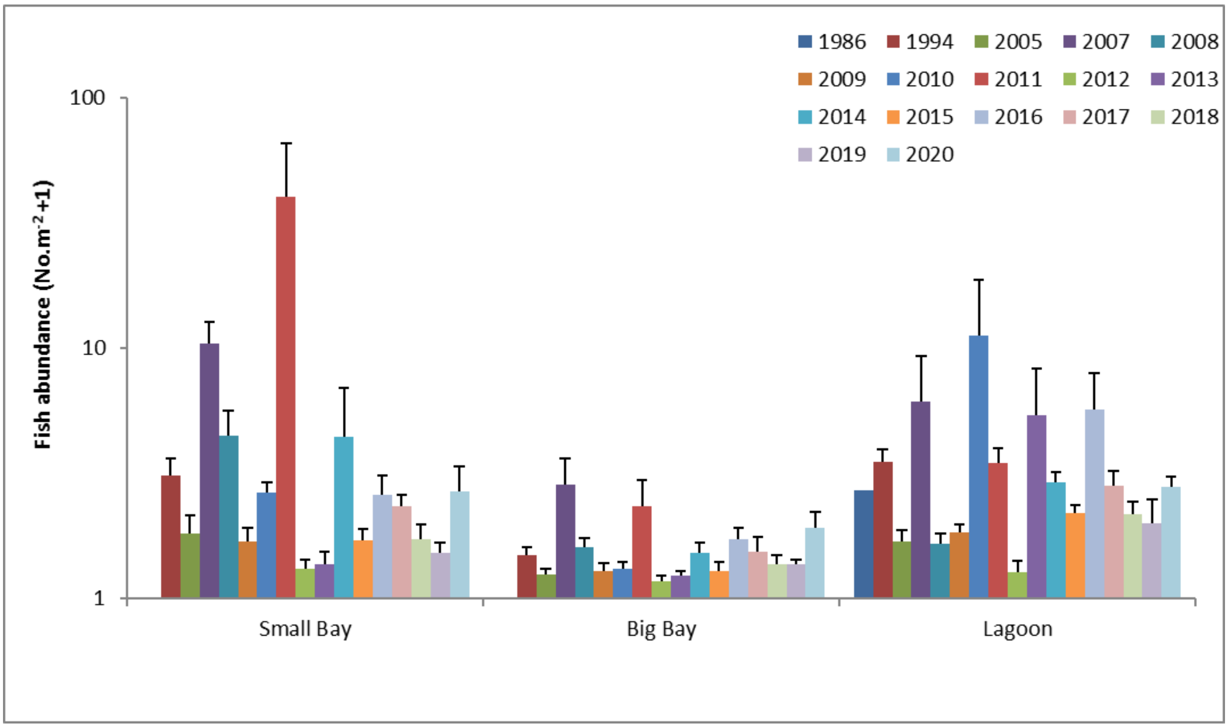


Figure 11.3. **Average fish abundance (all species combined) during 17 seine-net surveys conducted in Saldanha Bay and Langebaan lagoon. (Error bars show one Standard Error of the mean). The data are transformed (x + 1) and displayed on a logarithmic axis.**

Abundance of white stumpnose, nude goby and blacktail in seine net hauls that was above average in Small Bay during the 2007 and 2008 surveys, but have remained below these maxima since 2009 (Figure 11.4.). It may be that the peak densities attained by these species during 2007-2008 were the exception, and the lower densities recorded before and after this period, represent the more typical situation. The concerning declining trend in white stumpnose and blacktail abundance over the 2012-2015 period in Small Bay was thought to have reversed with the third highest white stumpnose abundance and second highest blacktail abundance recorded in 2016 samples, but unfortunately this moderate "recovery" was not sustained. Blacktail juveniles were entirely absent from Small Bay catches in 2017, remained scarce in 2018 samples with only five individuals caught, recovered somewhat in 2019 with 61 individuals caught, but were again entirely absent in 2020 samples. White stumpnose abundance in Small Bay remained significantly down from that recorded in earlier years with just 55 individuals caught (cf.600 in 2016, 1 566 in 2008, and 12 331 in 2007).

Within Big Bay too, average harder density observed during the 2013-2020 sampling was comparable to earlier surveys, but the abundance of the four next most common species remains low compared to earlier sampling events (Figure 11.4.). White stumpnose abundance within Big Bay over the period 2015-2018 had recovered somewhat from the very low 2013 and 2014 results, crashed again in 2019 and then recovered slightly in 2020 to levels similar to the 2013-14 period (Figure 11.4.). The strong elf recruitment in Big Bay evident in the 2016 and 2017 sampling has not reoccurred, with the species entirely absent from Big Bay catches for the last three surveys (three of the four times this has occurred in the 16-year survey history). Elf start to become sexually mature at one year (Maggs & Mann 2013), but as larger and older fish spawn exponentially more eggs, it will likely be several years before the strong 2016 and 2017 cohorts will be able contribute significantly to recruitment in the Bay (presuming the cohort survives beyond maturity and doesn't emigrate from the Bay). In the 2019 survey, elf were only caught at the Bluewater Bay site in Small Bay (33 fish) and a single individual at Kraalbaai in the lagoon, in 2020 no elf were caught at any of the 16 sampled sites.

The abundance of harders, silversides and Knysna sand goby in Langebaan lagoon during 2020 survey was comparable with earlier surveys but the abundance of *Caffrogobius* sp. and white stumpnose remained low (Figure 11.4.). The recovery in Knysna sand goby abundance is encouraging and hopefully *Caffrogobius sp.* abundance also recovers in the future. White stumpnose abundance is likely to remain depressed throughout the system until (if) recovery of the adult stock occurs.

Naturally high variability in recruitment strength is common for marine fish species and it is probably, at least partly natural environmental fluctuations rather than anthropogenic factors that caused the poor recruitment of most species in 2009 and 2012 as abundance was low throughout the system. The lower than average recruitment into the surf zones suggests that these were "poor" years for egg, larval and juvenile survival within the Bay as a whole. Either the environmental conditions were not suitable for the survival of eggs and larvae, or it was not good for the survival of young juveniles. The improved recruitment of most species seen during the 2016 and 2017 survey suggested improved environmental conditions that facilitated survival of eggs, larvae and juveniles during the preceding summer. The continued low abundance estimates of juvenile white stumpnose and recently elf throughout this period, however, indicates that the spawning capacity of the adult stock remains compromised. These two species are the main targets of the line-fishery operating in the system

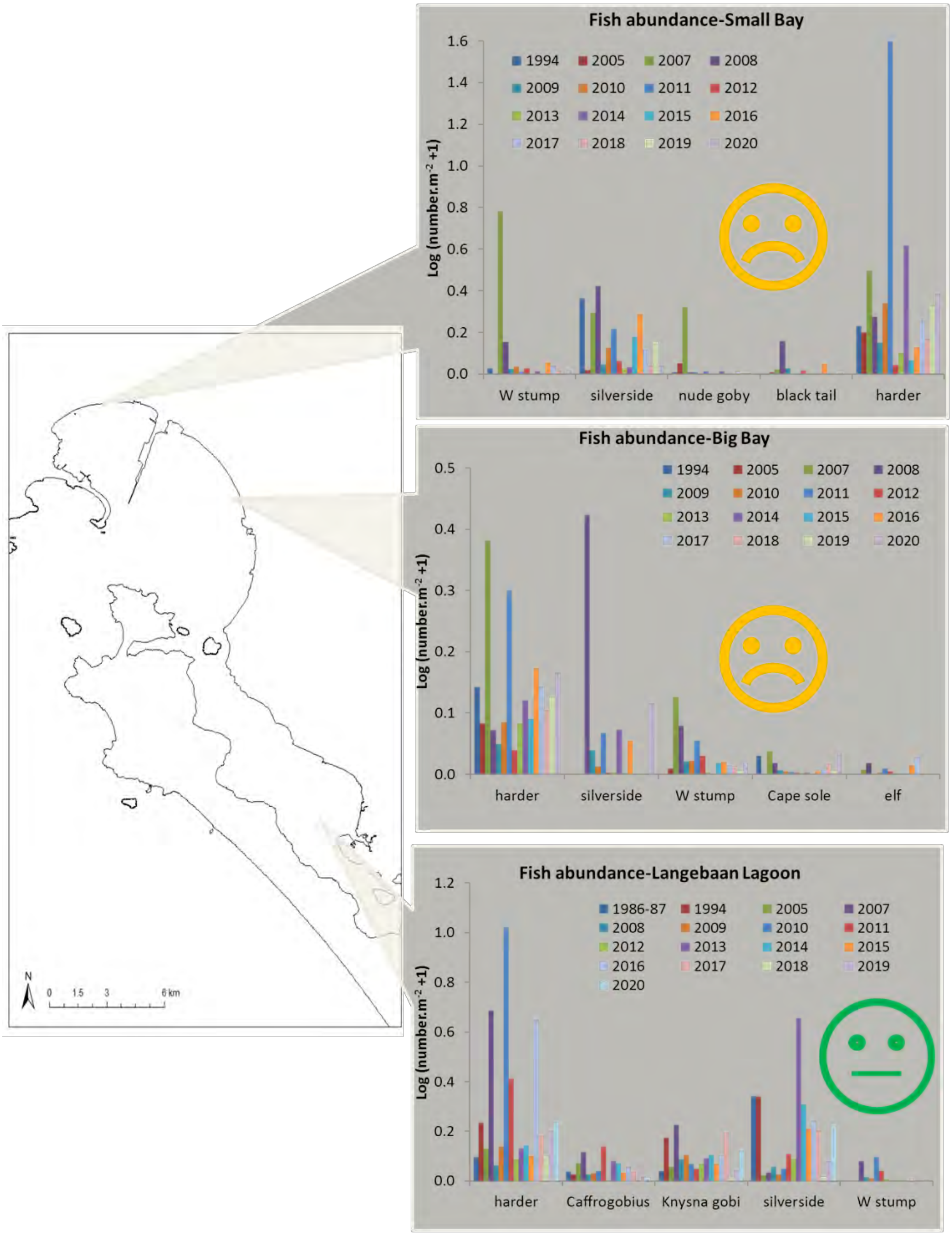


Figure 11.4. **Abundance of the most common fish species recorded in annual seine-net surveys within Saldanha Bay and Langebaan Lagoon (1986/87, 1994, 2005 & 2007-2020).**

11.3.3 Status of fish populations at individual sites sampled in 2020

The average abundance of the four most common species in catches made during all earlier surveys and the most recent 2020 survey at each of the sites sampled is shown in Figure 11.5., Figure 11.6. and Figure 11.7. These common fish species include two commercially important species (white stumpnose and harders), benthic gobies of the genus *Caffrogobius,* and the ubiquitous shoaling silverside (an important forage fish species).

The average abundance of gobies, silversides and white stump at Small Bay sites in the 2020 survey was lower than the long term average recorded in earlier surveys, whilst 2020 harder catches were greater (Bluewater bay and Hoedtjiesbaai) or less (Campsite and Small craft harbour) (Figure 11.5.). At four of the six Big Bay sites, catches of harders and silversides during the 2020 survey were greater than the historical average, whilst they were less than the historical average at the remaining two sites with no clear spatial trend (Figure 11.6.). White stumpnose catches at half the Big Bay sites in 2020 were less than the long-term average, whilst they were similar or greater at the other three sites (Figure 11.6.). Catches of harders at Lagoon sites during 2020 were variable, ; greater than the historical average at some sites and less at others, but overall similar to the long term average (Figure 11.7). White stumpnose density was lower at five of the six sites and greater than the long-term average only at Churchaven (Figure 11.7).

In summary, there have been significant, ongoing declines in white stumpnose density at most sites throughout the system and declines in gobies at nearly all sites in Small Bay and the Lagoon. At most sites, harder and silverside abundance was highly variable between survey years, but on balance is similar to the historical averages. Observed declines occurred across many sites and indicate larger system-wide impacts (i.e. fishing pressure in the case of white stumpnose) or natural fluctuations in abundance and distribution (gobies), rather than point source impacts. Indeed, in the more than a decade of data analysis investigating trends in the abundance of the most common fish at individual sites, we have failed to detect a consistent trend at any specific site (even at sites close to point pollutant sources such as the Bok River or potentially enriched, retentive sites such as Hoedtjiesbaai). This is partly a result of the high natural, temporal variability in nearshore fish assemblages (due to variations in recruitment success, water and sea conditions etc.). Indeed, high inter-sample variability within surveys is recorded at individual sites (resulting in relatively large standard errors associated with average values). However, this also indicates that to date, there do not appear to have been acute anthropogenic impacts on fish at any of the specific sampling sites. The most abundant species (harders), are also known to be resilient to organic enrichment and tend to thrive in enriched environments, provided that dissolved oxygen levels remain sufficient. For these reasons, we recommend that future analysis focus on trends in fish abundance either at an area level spatial scale (i.e. Small Bay, Big Bay, Lagoon) as in Section 11.3.2 above, or at a system wide spatial scale as in Section 11.4 below.

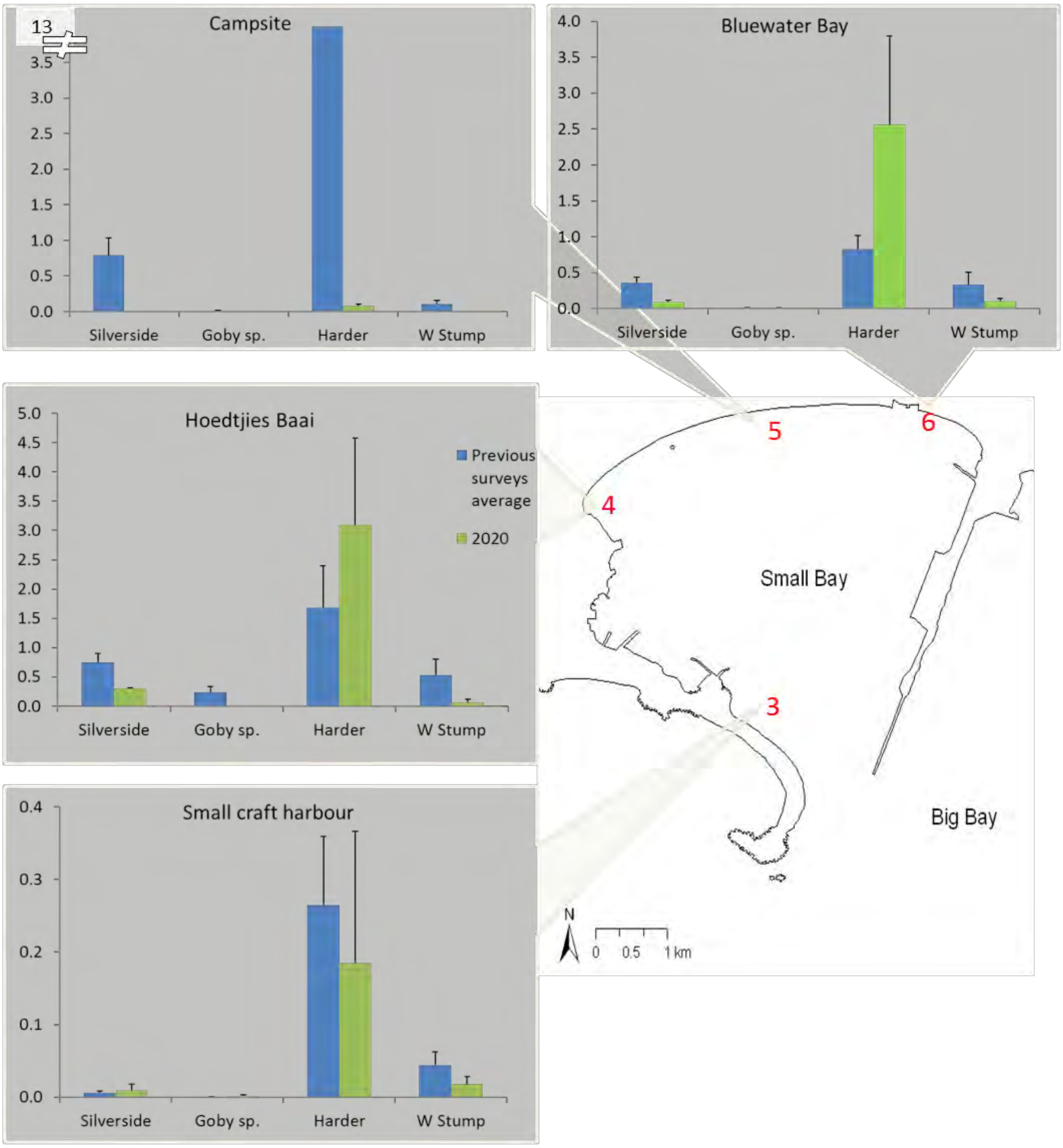


Figure 11.5. **Average abundance (No. fish.m^{-2}) of the four most common fish species at each of the sites sampled within Small Bay during the earlier surveys (1994, 2005, 2007-2019) and during the 2020 survey. Errors bars show plus 1 Standard Error.**

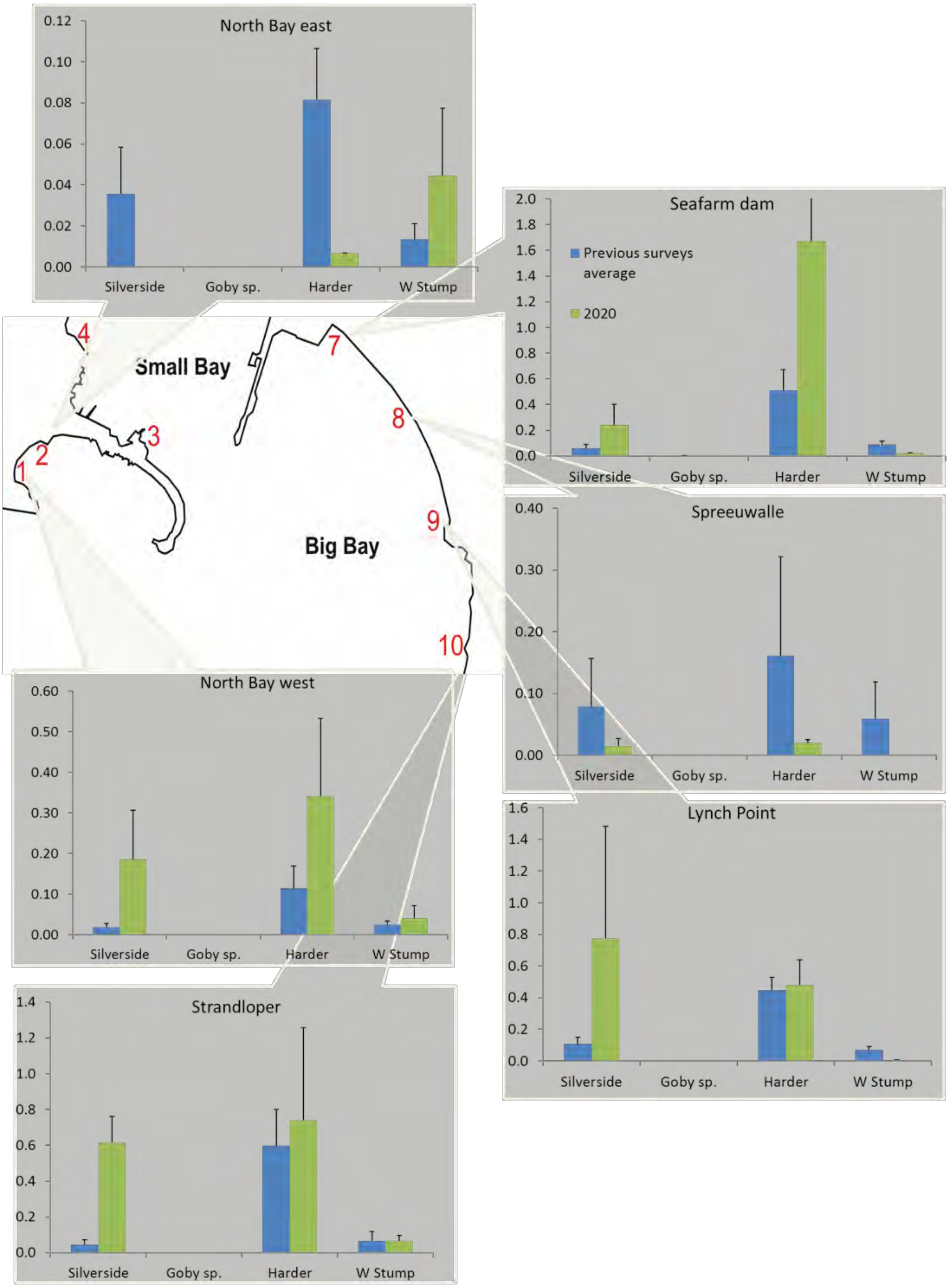


Figure 11.6. Average abundance (No.fish.m^{-2}) of the four most common fish species at each of the sites sampled within Big Bay during the earlier surveys (1994, 2005, 2007-2019) and during the 2020 survey. Errors bars show plus 1 Standard Error.

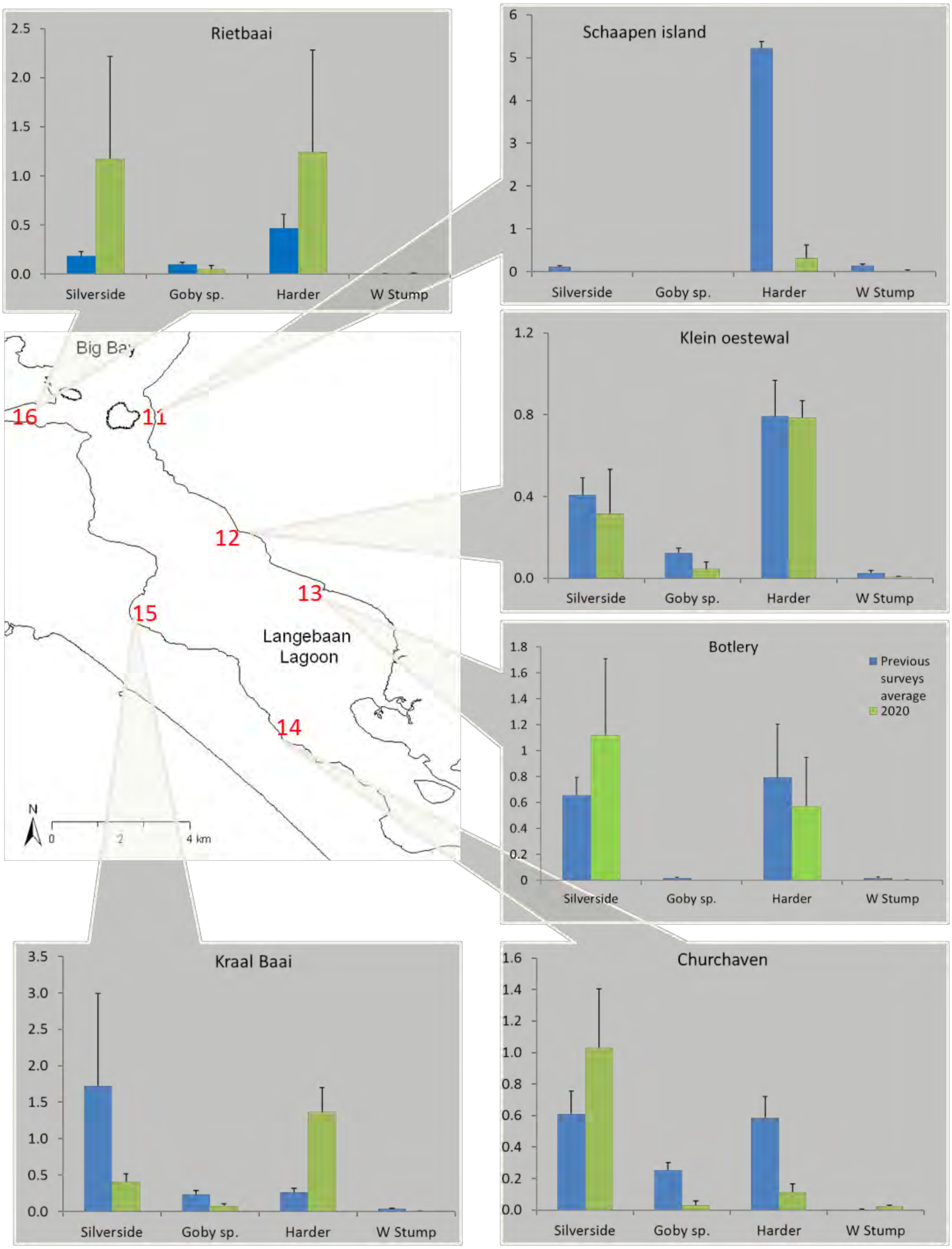


Figure 11.7. **Average abundance (no. fish.m^{-2}) of the four most common fish species at sites sampled within Langebaan lagoon during the earlier surveys (1994, 2005, 2007-2019) and during the 2020 survey. Error bars show plus 1 Standard Error.**

11.4 Temporal trends in key fishery species

The spatially separate analysis of fish survey data by site or embayment (Big Bay, Small Bay and Langebaan Lagoon) is a valid approach for the purposes of ecosystem health monitoring whereby sites or areas of concern need to be identified. The analyses presented above have identified a concerning decrease in abundance of most of the dominant species in Small Bay in surveys over the period 2008-2015 and a notable decrease in white stumpnose abundance throughout the system over this same period. The 2016 survey revealed some encouraging signs of increased white stumpnose recruitment in Small Bay, but 2017-2020 catches were again much lower than average. The inter-annual variation in recruitment of white stumpnose could be due to natural variability in spawning success and survival (poor and good year classes are normal), but given the sustained declines throughout the system, and the findings of Arendse (2011) and Parker *et al*. (2018), it appears that recruitment overfishing is the cause. Recruitment overfishing can be defined as overfishing of the adult population so that the number and size of mature fish (spawning biomass) is reduced to the point that it did not have the reproductive capacity to replenish itself. Recent research suggests that the Saldanha bay harder stock is also overexploited with changes to management measures (increased mesh size, reduced fishing mortality) required to rebuild stocks (Horton et al 2019).

To further investigate temporal variation in recruitment of species important in the Bay's fisheries (harders, blacktail, elf, steentjies and white stumpnose) univariate statistical analysis (ANOVA) was used to test for significant differences in abundance between survey years. Saldanha Bay and Langebaan Lagoon appears to function as a semi-closed system with respect to the demographics of many of the key fishery species (based of tagging and life history studies, e.g. harders: Horton *et al.* 2019, white stumpnose: Kerwath *et al.* 2009 & Attwood *et al.* 2010, steentjies: Tunley *et al.* 2009, elf: Hedger *et al.* 2010, and smooth hound sharks: da Silva *et al.* 2013). Furthermore, different sites may be more intensively utilized by juvenile fish in different years depending on prevailing weather and local oceanographic conditions. For these reasons, to assess recruitment of key fishery species to surf zone habitats for the Bay as a whole, abundance data from all sites were combined for this analysis. These analyses revealed statistically significant inter-annual variation in the abundance of blacktail, harders and white stumpnose, but not in the average density of elf and steentjies (Figure 11.8, Figure 11.9, Figure 11.10).

The density of blacktail juveniles in sampled habitats was significantly higher in 2008 than in all other years, there was an absence of blacktail recruits in the 2015 and 2017 samples and only five were caught in 2018, 61 in 2019 and just three in 2020 (Figure 11.8). Inter annual variation in the abundance of harders was greater than the other species, with estimated abundance in 2007, 2010 and 2011 being significantly greater than most other sampling events. The abundance of juvenile harders in 2020 hauls was the fourth highest on record and only significantly lower than that recorded in 2011. Estimated white stumpnose abundance in 2007 and 2008 was significantly greater than nearly all other years, whilst the estimated abundance from the 2020 survey remained lower than average and not significantly different from the abundance estimates recorded in other surveys conducted after 2008. Steentjie and elf abundance also showed high inter-annual variation with relatively high average abundance of steentjie juveniles recorded in 2005 and 2011 and relatively high average abundance of elf juveniles in 2007, 2008, 2011, 2012, 2016 and 2017 (the highest recorded to date) which was followed by zero catches in 2018 and 2020. The intra annual variability in abundance of these two species, a result of a zero catches at many sites, however, means that these differences are not statistically significant.

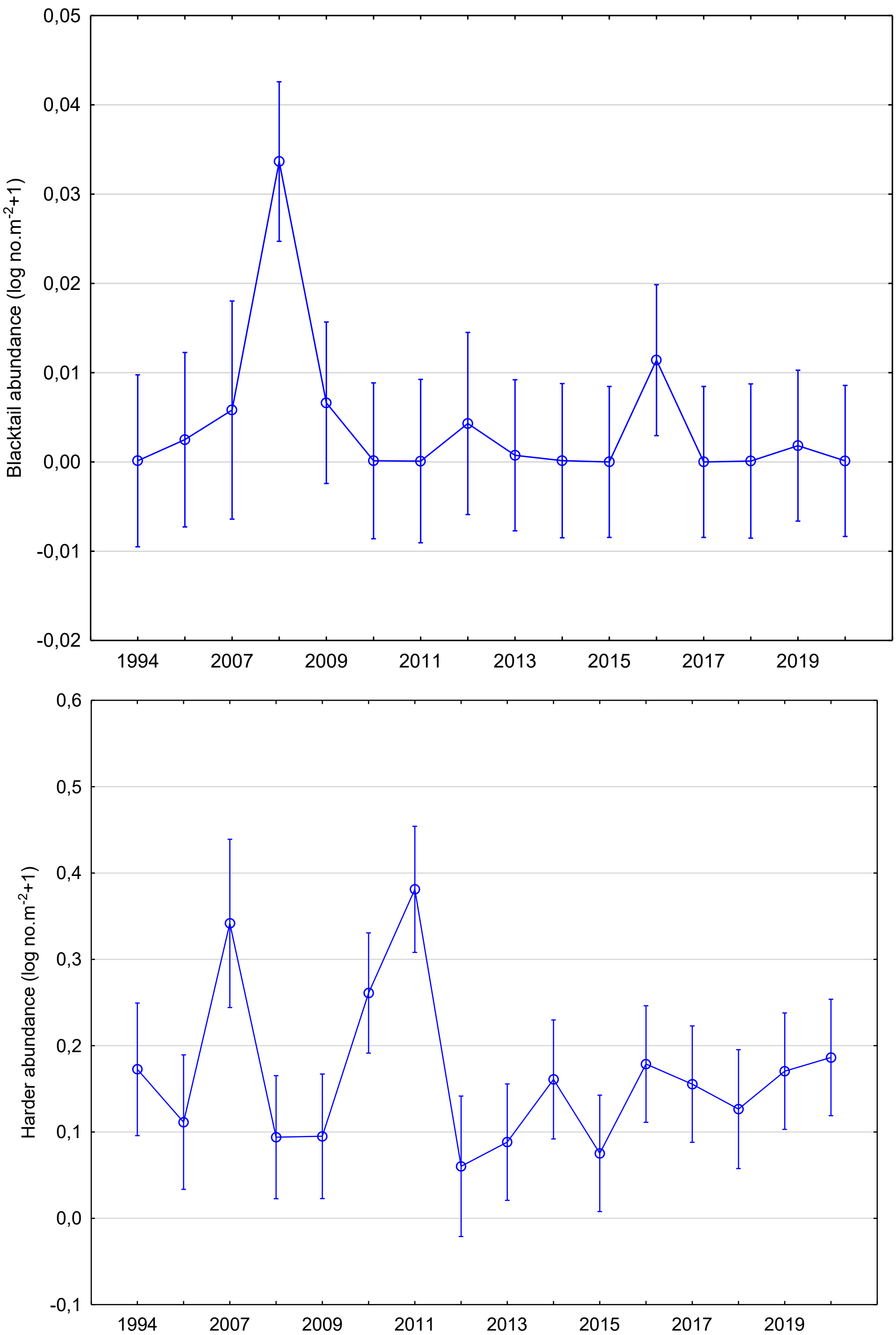


Figure 11.8. ANOVA results comparing the average annual density of blacktail (top) and harders (bottom) at all sites sampled in all surveys (1994-2020).

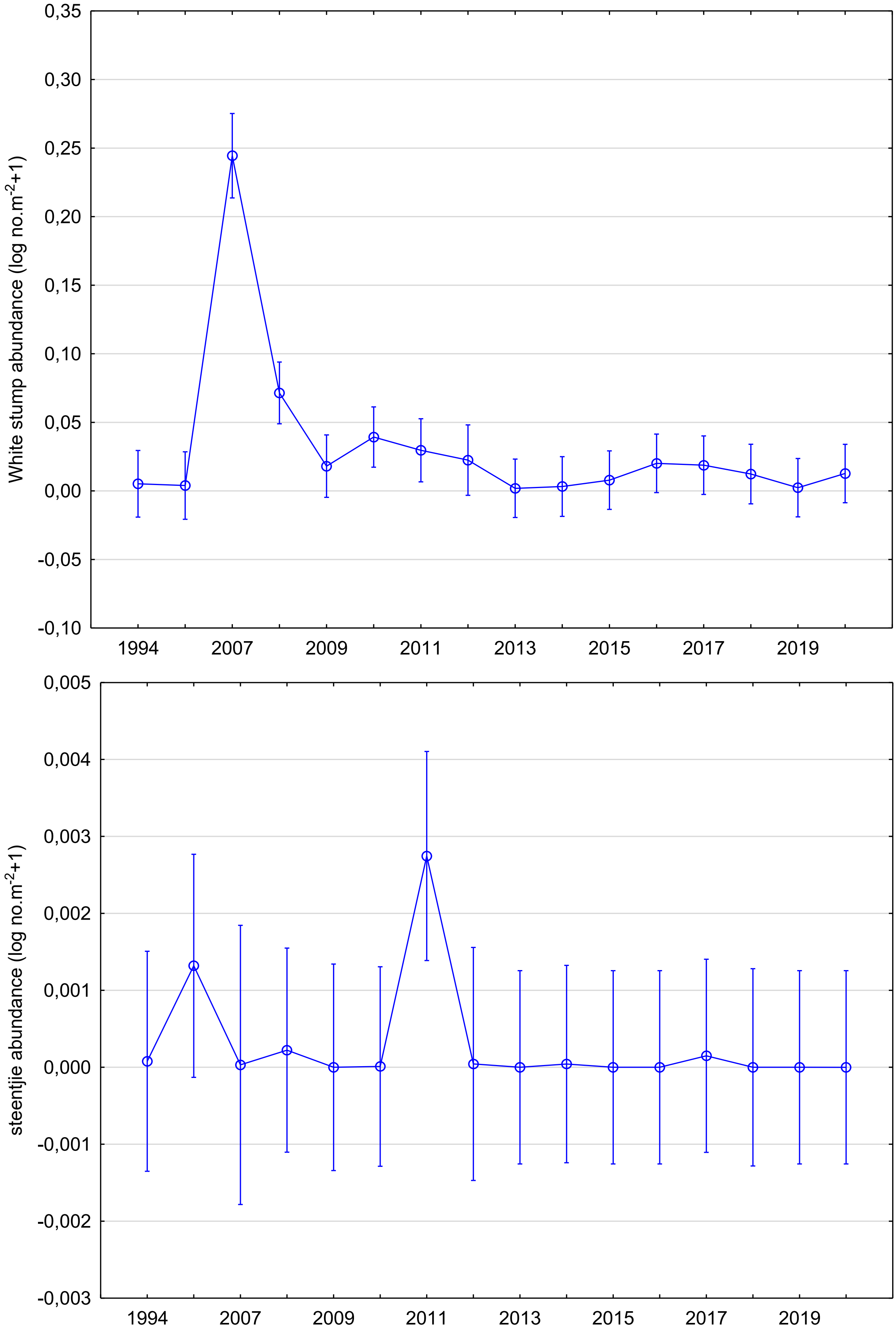


Figure 11.9. **ANOVA results comparing the average annual density of white stumpnose (top) and steentjies (bottom) at all sites sampled in all surveys (1994-2020). Error bars indicate 95% confidence intervals.**

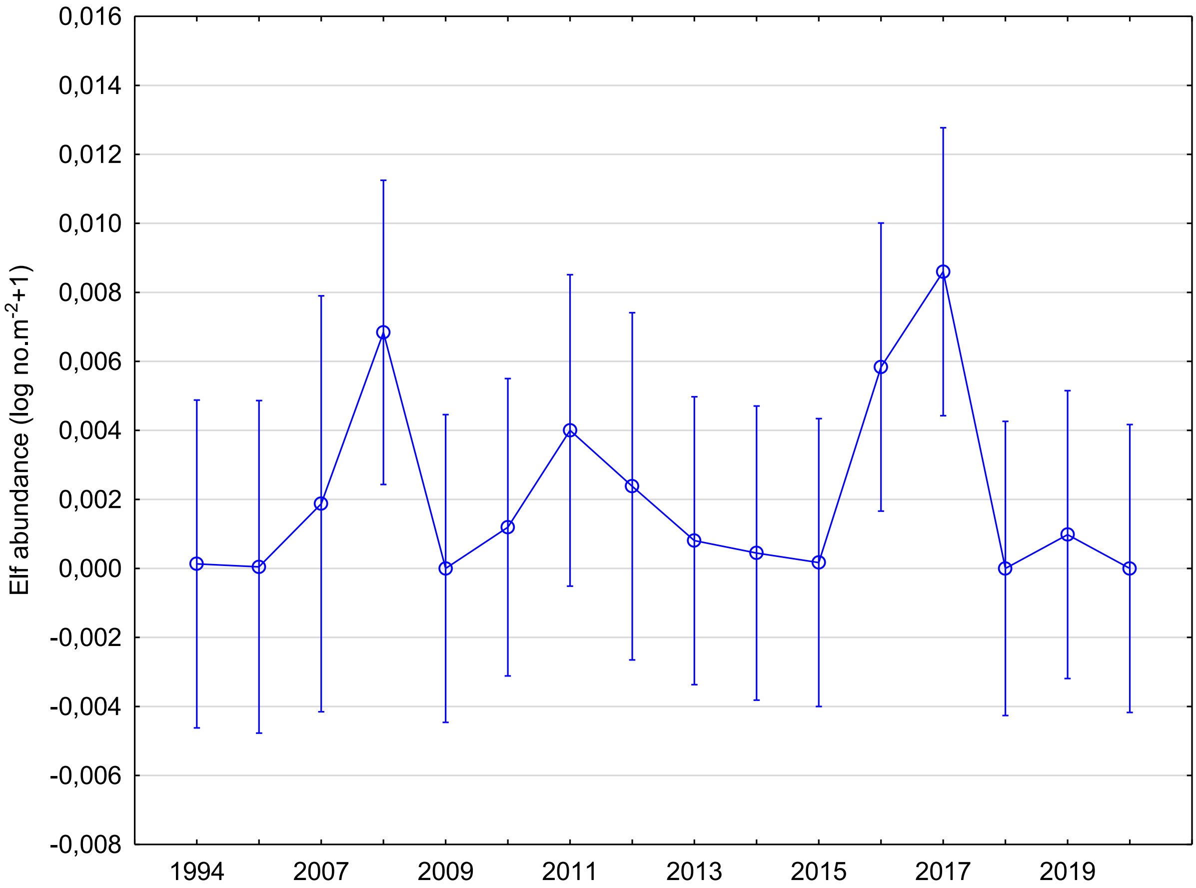


Figure 11.10. ANOVA results comparing the average annual density of elf at all sites sampled in all surveys. Error bars indicate 95% confidence intervals.

11.5 Conclusion

Recent seine net surveys have documented ongoing concerning trends in juvenile fish populations within the Saldanha Bay and Langebaan Lagoon system. The encouraging signs of recovery of white stumpnose and blacktail in Small Bay in 2016 did not continue through to 2017-2020, and white stump abundance remains very low throughout the system. The abundance of gobies in Small Bay also has also remained low since the 2007 survey. The decline in gobies cannot be attributed to fishery impacts but may be related to changes in water quality or habitat. The absence of gurnards, blacktail, elf and pipefish from 2020 Small Bay samples meant that diversity (just 10 species) was the lowest in the 16-year survey history. Overall fish abundance does not, however, show a declining trend in Small Bay, but it must be acknowledged that overall abundance is dominated by harders, which appear resilient to decreases in water quality. Despite the strong elf recruitment in Big Bay evident in the 2016 and 2017 sampling, none were caught during the following three sampling events (2018-2020). Notably for the first time in the history of the 16 seine net surveys undertaken since 1994, not a single elf was captured at any of the 16 sites surveyed in 2020. This is concerning as much of the remaining boat-based, recreational fishing effort that previously focused on white stump, appears to have shifted to elf (personal observation) and three years of apparent poor recruitment to the surf zone nursery habitats does not bode well for future catches. After their scarcity in 2018 and 2019 samples, silversides were again abundant in Big Bay in 2020, but three species (super klipvis, elf and pipefish) that are usually present in Big Bay surveys, were absent from 2020 samples. The first capture of galjoen, and presence of the other common species (including False Bay klipvis and sandsharks that were absent in 2019), however resulted in a typical Big bay fish species count (13 species). Harders, gobies and silversides were present in Langebaan Lagoon samples in similar numbers to previous surveys, with only white stump catches remained low.

For most of the seine net survey history, fish abundance at sites within or in close proximity to the Langebaan MPA appeared to be stable within the observed inter-annual variability. This reflects natural and human induced impacts on the adult population size, recruitment success and use of the near-shore habitat by fish species; but may also be a result of the benefits of protection from exploitation and reduced disturbance at some sites due to the presence of the Langebaan MPA. Certainly, the studies by Kerwath *et al.* (2009), Hedger *et al.* (2010) and da Silva *et al.* (2013) demonstrated the benefits of the MPA for white stumpnose, elf and smooth hound sharks; and the protection of harders from net fishing in the MPA undoubtedly benefits this stock in the larger Bay area. The pressure to reduce this protection by allowing access to Zone B for commercial gill net permit holders should be resisted. This not only poses a threat to the productivity of the harder stock but also to other fish species that will be caught as bycatch.

The 2018 discovery of alien rainbow trout in Kraalbaai (almost certainly escapees from the pilot fish cage farming in Big Bay) is another threat to the indigenous fish fauna in the region. No further reports of fish cage escapees have been received, nor have any been caught during the seine net surveys undertaken since then. Nonetheless, the capture of these three fish proved that escapees from fish cages can survive for a period and colonize different areas of the system. Trout are predators that, like many other fish species, exhibit a high degree of dietary plasticity. They will therefore consume indigenous invertebrates and small fish. Given the relatively small number of trout that are thought to have escaped (this is an assumption as the fish cage operators have yet to report on the magnitude of the breakout) compared to the abundance of indigenous predatory fish such as elf, it is considered unlikely that any escapees from the experimental cages to date, have had a significant impact on juvenile fish abundance in the Lagoon (or elsewhere in Saldanha Bay). Trout, and other salmonids earmarked for fish cage farming in the Saldanha area require cool, fast flowing mountain streams for successful spawning. Given the paucity of clear river systems entering the sea along the west coast it is considered a low risk that escaped salmonids will become naturalised (i.e. form self-sustaining natural populations) and invasive. Although the possibility of escapees establishing a spawning population in a South Western Cape River cannot be rejected with absolute certainty, we believe that the probability of this occurring is low. However, escapes from fish cages moored in the sea are inevitable and may occur due to storm sea damage to cage infrastructure. Should finfish cage farming in Saldanha Bay be expanded to a commercial scale of several thousand tonnes per annum (as proposed in the recent Aquaculture Development Zone EIA), however, then the impact of ongoing escapees of alien salmonids on indigenous biota (including juvenile fish) are expected to be more severe. The environmental impacts caused by the introduction of salmonids into pristine ecosystems with no native salmonid species are severe and interlinked with oftentimes unpredictable knock-on effects at the ecosystem scale. Salmonid invasions present a strong top-down control on community structure and ecosystem functioning by inducing change in individual behaviour, distributions of populations and abundance within functional groups. Given the importance of the nearshore waters of Saldanha Bay and Langebaan lagoon as nursery areas for a number of vulnerable indigenous fishery species, finfish cage farming should be restricted to the outer Bay, and mitigation measures to minimise escapes from cages should be strictly enforced.

The significant declines in juvenile white stumpnose abundance at all sites throughout the system in over the last decade have shown that the protection afforded by the Langebaan MPA was not enough to sustain the fishery at the historical high effort levels. Arendse (2011) found the adult stock to be overexploited using data collected during 2006-2008 already, and the evidence from the seine net surveys conducted since then certainly suggests that recruitment overfishing has occurred. The annual seine net surveys did act as an early warning system that detected poor recruitment and should have allowed for timeous adjustments in fishing regulations to reduce fishing mortality on weak cohorts and preserve sufficient spawner biomass. Unfortunately, despite repeatedly expressing concern about the collapse of white stump recruitment in State of the Bay Reports since at least 2013 and supporting the implementation of the harvest control measures recommended by Arendse (2011); namely a reduction in bag limit from 10 to 5 fish per person per day and an increase in size limit from 25 cm TL to 30 cm TL, the warning calls were not heeded. A statistically comprehensive analysis of fishery dependent and survey data confirmed the collapse of the Saldanha -Langebaan white stump stock and the fishery yield in recent years is a fraction of its historical peak or potential (Parker *et al.* 2017). The last three surveys have revealed some concern about elf recruitment to surf zone nurseries, and this should be carefully monitored in the future. Harder recruitment to nearshore nursery areas appears to have not changed significantly over the monitoring period since 1994. A recent stock assessment, however, does indicate that the Saldanha-Langebaan harder stock is overexploited and effort reductions and commercial net gear restrictions are recommended to rebuild the stock (Horton 2018).

The monetary value of the recreational fishery in Saldanha-Langebaan should not be regarded as regionally insignificant as a lot of the expenditure associated with recreational angling is taking place within Langebaan and Saldanha itself. Furthermore, the once popular white stumpnose fishery was undoubtedly a major draw card to the area and has probably contributed significantly to the residential property market growth the region has experienced. These benefits should be quantified by an economic study of the recreational fisheries. The value of the Bay and Lagoon as a fish nursery and the economic value of the resultant fisheries could then be quantitatively considered when the environmental impacts of the proposed future developments in the region are assessed.

The monitoring record from the annual seine net surveys will prove increasingly valuable in assessing and mitigating the impacts of future developments on the region's ichthyofauna. Extending the seine net monitoring record would also facilitate analysis of the relationship between recruitment to the near shore nursery habitat and future catches in the commercial and recreational fisheries in the Bay. A preliminary investigation of this relationship was undertaken for white stumpnose and harders in the 2011 and 2012 reports, respectively, and investigated again in the 2015 report for the commercial white stumpnose fishery. Should this relationship prove robust and quantifiable as more years of data become available, this will allow for adaptive management of the fisheries in the future as fishing effort continues to increase and at some point fishing mortality will need to be contained, if the fisheries are to remain sustainable. We think that point arrived at least seven years ago for the Saldanha-Langebaan white stumpnose fishery and recommended that resource users lobby the authorities to implement the recommended harvest control measures. Regional species-specific fishery management has been implemented elsewhere in South Africa (e.g. Breede River night fishing ban to protect dusky kob). White stumpnose in Saldanha Bay appear to be an isolated stock and there is good on-site management presence in the form of SANParks and DAFF, and we think this approach would work well in Saldanha-Langebaan. We again recommend the reduction of bag limit and an increase in size limit for white stumpnose in the Saldanha Bay Langebaan region. Although recruitment overfishing appears to have been taking place for several years now, the stock is not extirpated, and the situation is reversible. Reductions in fishing mortality can be achieved by effective implementation of more conservative catch limits and have an excellent chance of improving the stock status, catch rates and the size of white stumpnose in the future fishery. We also support the recommendation of Horton *et al.* (2019) for a reduction in harder fishing effort and gear changes (increase in minimum mesh size) to facilitate stock recovery. Short term reductions in fishing mortality will have an economic cost, but will yield substantially greater socio-economic benefits for fishers in the medium to longer term as the sustainable catch from an optimally exploited fish stock greatly exceeds that from a collapsed stock.

12 BIRDS AND SEALS

12.1 Introduction

Saldanha Bay and Langebaan Lagoon provide extensive and varied habitat for waterbirds and seals. This includes sheltered deep-water marine habitats associated with Saldanha Bay itself, sheltered beaches in the Bay, islands that serve as breeding refuges for seabirds and seals, rocky shoreline surrounding the islands and at the mouth of the Bay, and the extensive intertidal salt marshes, mud and sandflats of the sheltered Langebaan Lagoon. Langebaan Lagoon has 1 750 ha of intertidal mud- and sandflats and 600 ha of salt marshes (Summers 1977). Extensive sea grass *Zostera capensis* beds are present in the upper parts of Langebaan Lagoon, while beds of the red seaweed *Gracilaria verrucosa* are mainly found at the mouth and patchily distributed over the sandflats in the lagoon. Drainage channels also contribute to habitat diversity around the lagoon. Most of the plant communities bordering the lagoon belong to the West Coast Strandveld, a vegetation type which is seriously threatened by agricultural activities and urban development. Twelve percent of this vegetation type is conserved within the West Coast National Park which surrounds much of the lagoon (Boucher & Jarman 1977, Jarman 1986). Although there are no rivers flowing into the Lagoon, it has some estuarine characteristics due to the input of fresh groundwater in the southern portion of the lagoon.

Saldanha Bay and Langebaan Lagoon are not only extensive in area but provide much of the sheltered habitat along the otherwise very exposed West Coast of South Africa. There are only four other large estuarine systems which provide sheltered habitat comparable to Langebaan Lagoon for birds along the West Coast – the Orange, Olifants, Berg and Rietvlei/Diep. There are no comparable sheltered bays and relatively few offshore islands. Indeed, these habitats are even of significance at a national scale. While South Africa's coastline has numerous estuaries (about 300), it has few very large sheltered coastal habitats such as bays, lagoons or estuaries. The Langebaan-Saldanha area is comparable in its conservation value to systems such as Kosi, St Lucia and Knysna.

A total of 283 bird species have been recorded within the boundaries of the West Coast National Park (Birdlife International 2011). At least 56 non-passerine waterbird species commonly use the area for feeding or breeding (University of Cape Town, Animal Demography Unit Coordinated Waterbird Counts); 11 breed on the islands of Malgas, Marcus, Jutten, Schaapen and Vondeling alone. These islands support nationally important populations of African Penguin, Cape Gannet, Swift Tern, Kelp and Hartlaub's Gull, and four species of marine cormorant, as well as important populations of the endemic African Oystercatcher. The lagoon is an important area for migratory waders and terns, as well as for numerous resident waterbird species. Seabirds are counted annually on all the islands by Department of Environment, Fisheries and Forestry staff; and bi-annually in Langebaan Lagoon as part of the Coordinated Water Counts (CWAC) Programme conducted by the Animal Demography Unit (ADU) at the University of Cape Town (UCT).

In contrast, the Cape fur seal *Arctocephalus pusillus pusillus* is the only seal species that breeds in Southern Africa and will be discussed in the last section of this chapter.

12.2 Birds of Saldanha Bay and the islands

12.2.1 National importance of Saldanha Bay and the islands for birds

Saldanha Bay and the islands are important not so much for the diversity of birds they support, but for the sheer numbers of birds of a few species in particular. The islands of Vondeling (21 ha), Schaapen (29 ha), Malgas (18 ha), Jutten (43 ha), Meeuw (7 ha), Caspian (25 ha) and Marcus (17 ha), support important seabird breeding colonies and make up one of only a few such breeding areas along the West Coast of South Africa. They support nationally-important breeding populations of African Penguin (recently up-listed to Endangered under IUCN's red data list criteria), Cape Gannet (Vulnerable), Cape Cormorant (recently up-listed to Endangered under IUCN's red data list criteria), White-breasted Cormorant, Crowned Cormorant (Near Threatened), Bank Cormorant (Endangered), Kelp and Hartlaub's gulls, Caspian Tern and Swift Tern.

In addition to seabird breeding colonies, the islands also support important populations of the rare and endemic African Oystercatcher (Near-threatened). These birds are resident on the islands but are thought to form a source population for mainland coastal populations through dispersal of young birds.

The Department of Environmental Affairs (DEA, now DEFF) conducts ongoing bird counts on all islands to track population trends of each of these species over time. Each island is visited several times a year to ensure that each species is counted during its peak breeding season. The maximum counts for each species obtained in a calendar year are then used to estimate population sizes. All islands are visited roughly three times per calendar year with the exception of Malgas (nine times) and Vondeling (less than three times due to accessibility) (Rob Crawford, Department of Environmental Affairs, *pers. comm.* 2016). Section 12.2.1.1 provides data on long-term trends of each of these important seabirds and the African Oystercatcher, using the data collated by the DEA.

12.2.1.1 Ecology and status of the principle bird species

The African Penguin *Spheniscus demersus* is endemic to southern Africa, and breeds in three regions: central to southern Namibia, Western Cape and Eastern Cape in South Africa (Whittington *et al.* 2005a). The species has recently been up-listed to Endangered, under IUCN's 'red data list' due to recent data revealing rapid population declines as a result of numerous factors including pollution (from oil spills), changes in the abundance and distribution of small pelagic fish populations, competition with commercial fisheries and seals for food and predation pressure from Kelp Gulls and Cape Fur Seals, as well as potential exposure to conservation-significant pathogens (David *et al.* 2003, Pichegru *et al.* 2009, Crawford 2009, Birdlife International 2011, Crawford *et al.* 2011, 2014, Weller *et al.* 2014, 2016, De Moor & Butterworth 2015, Gremillet *et al.* 2016, Parsons *et al.*2016). The Namibian population collapsed in tandem with the collapse of its main prey species, the sardine *Sardinops sagax*; (Ludynia *et al.* 2010). In South Africa the penguins breed mainly on offshore islands in the Western and Eastern Cape with strongly downward trends at all major colonies (Whittington *et al.* 2005b).

Throughout South Africa, the African Penguin population declined from an average of 48 000 pairs over the period 1979-2004 to just 17 000 pairs in 2013 (Crawford *et al.* 2014). The number of African penguins breeding in the Western Cape decreased in a similar fashion from some 92 000 pairs in 1956, to 18 000 pairs in 1996. There was a slight recovery to a maximum of 38 000 pairs in 2004, before another dramatic collapse to 11 000 pairs in 2009, equating to a total decline of 60.5% in 28 years (Crawford *et al.* 2008a, b, R. Crawford unpubl. data). In Saldanha Bay the population initially grew from 552 breeding pairs in 1987 to a peak of 2 156 breeding pairs in 2001 and then underwent a severe and continuous decline to just 144 breeding pairs in 2019 (Figure 12.1.). This reduction in numbers is consistent with the overall downward trend evident since 2002 and strongly reinforces the argument that immediate conservation action is required to prevent further losses of these birds. In light of the ongoing decline in African Penguin numbers nationally, a Biodiversity Management Plan for the African Penguin was gazetted in 2013, with aims: "*To halt the decline of the African Penguin population in South Africa within two years of the implementation of the management plan and thereafter achieve a population growth which will result in a downlisting of the species in terms of its status in the IUCN Red List of Threatened Specie*s". Despite the successful implementation of many of the actions listed in the plan, these aims were not attained, and African Penguins in South Africa have continued to decline. This has led to the recent gazetting of a second revised draft Biodiversity Management Plan for implementation over the period 2019-2024 (Government Gazette No. 42775 18 October 2019). This draft plan attributes population declines mostly to a scarcity of prey and recommends pelagic fishery exclusion zones around colonies, seasonal closures at penguin feeding grounds before and post moult, oil spill risk management and colony specific management such as predator control.

The changes in African Penguin population size at the islands in Saldanha is believed to be partially linked to patterns of immigration and emigration by young birds recruiting to colonies other than where they fledged, with birds tending to move to Robben and Dassen Islands in recent years

(Whittington *et al.* 2005b). However, once they start breeding at an island, they will not breed anywhere else.

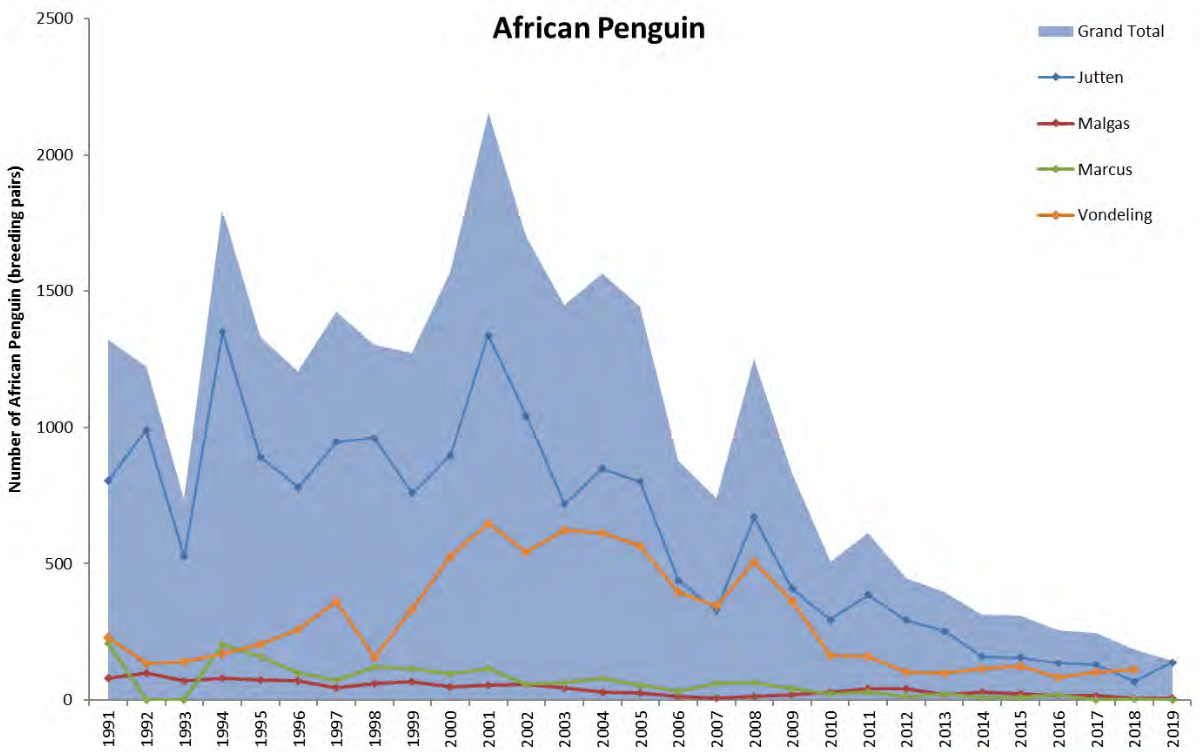
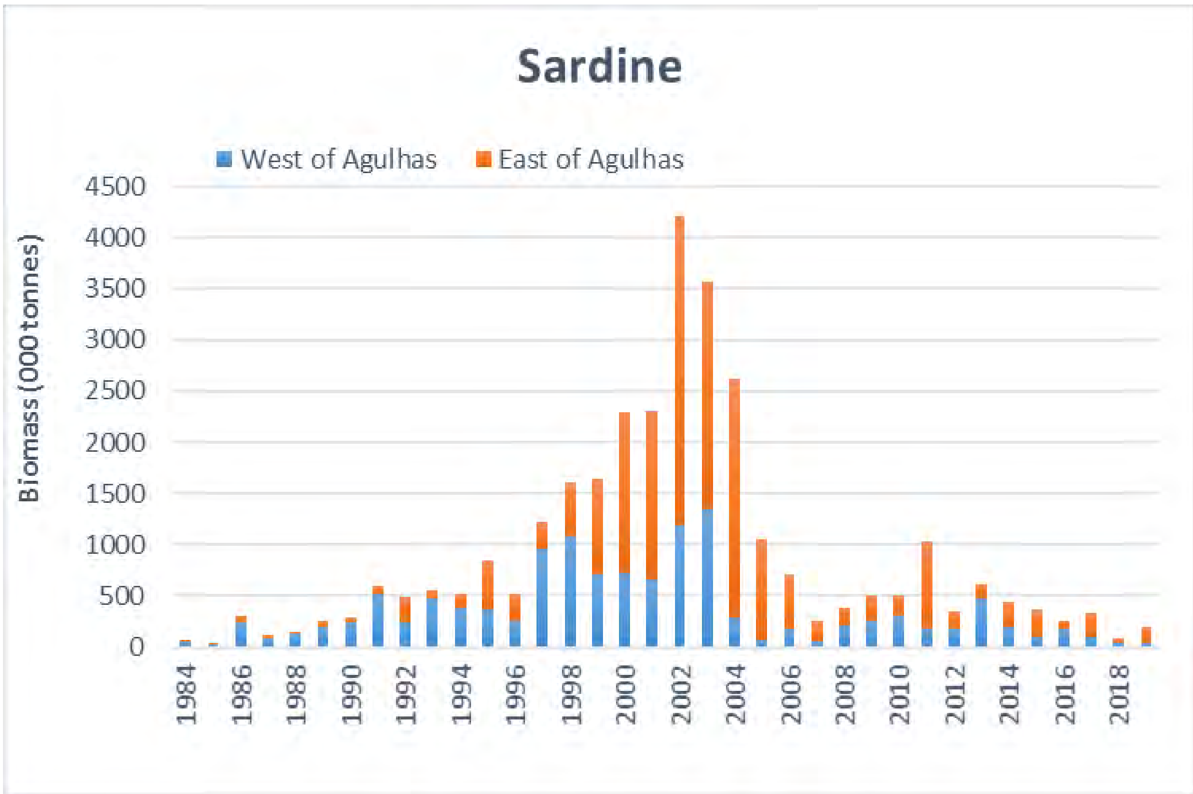


Figure 12.1. Trends in African Penguin populations at Jutten, Malgas, Marcus and Vondeling islands in Saldanha Bay from 1991-2019 measured in number of breeding pairs (Data source: Department of Environmental Affairs: Oceans & Coasts, 2020).

Penguin survival and breeding success has been linked to the availability of pelagic sardines *S. sagax* and anchovies *Engraulis encrasicolus* within 20-30 km of their breeding sites (Pichegru *et al.* 2009). Diet samples taken from penguins at Marcus and Jutten Islands showed that the diet of African penguins in the Southern Benguela from 1984 to 1993 was dominated by anchovy (Laugksch & Adams 1993). During periods when anchovies are abundant, food is more consistently available to penguins on the western Agulhas Bank than at other times (older anchovy remain there throughout the year and sardines are available in the region in the early part of the year). The reduced abundance of anchovy in the 1980s may partly explain the decrease in the African penguin population evident from 1987 to 1993 clearly reflected in the Saldanha data (Figure 12.1.). Subsequently the penguin population at Saldanha bay increased in tandem with a “boom” period for the South African sardine stock that increased from less than 250 000 tonnes in 1990 to over four million tons in 2002 (Figure 12.2). Anchovy biomass also increased from the late 1990s, peaked at over 4 million tonnes in 2001, remained relatively high (compared to the 1980s and 1990s) at between 2-4 million tonnes in most years until 2014 (Figure 12.2). Although both anchovy and sardine were still abundant along the west coast during the “boom” period around the turn of the century, much of the growth in biomass in these small pelagic stocks occurred to the east of Cape Agulhas benefiting seabirds at colonies along the south and east coast. Subsequently, the sardine stock crashed over the period 2004-2007 and the proportion of the sardine stock along the west coast declined dramatically at this time. The numbers of African Penguins on the Saldanha Bay Islands followed a similar trajectory, despite anchovy remaining abundant off the West Coast and an increase in the proportion of the sardine stock west of Cape Agulhas up until 2013 (Figure 1.1, Figure 12.2). In the last six years however, the estimated sardine biomass along the west coast has declined dramatically, with almost none detected in the 2018 and 2019 acoustic surveys (Figure 12.2). Anchovy biomass too has recently declined to about 800 000 tonnes in 2019, the second lowest estimate in the last two decades and the estimated biomass on the west coast was at its fourth lowest level since the turn of the century (Figure 12.2).

Several studies have identified addittional drivers of African Penguin populations at the colony level; these include oiling and predation by seals and kelp gulls, with the importance fishing and food availability decreasing at small colony size (<3 500 breeding pairs) (Ludynia *et al.* 2014, Weller *et al.* 2014, 2016). There is considerable uncertainty around the causes of African penguin population decreases which is a result of multiple pressures, some operating throughout the species range and others operating at different intensities at different colonies. One of the measures currently being employed to curb these declines is the use of no-take zones for purse-seine fishing. This strategy, recently tested at St Croix Island in the Eastern Cape, was effective in decreasing breeding penguins' foraging efforts by 30% within three months of closing a 20 km zone to purse-seine fisheries (Pichegru *et al.* 2010). In this case, the use of small no-take zones presented immediate benefits for the African penguin population dependent on pelagic prey, with minimum cost to the fishing industry, while protecting ecosystems within these habitats and important species. However, experimental fishing closures at Dassen and Robben Islands have not delivered such positive results, resulting in published rebuttals labelling the findings of Pichegru *et al.* (2010) premature. The revised, Draft Biodiversity Management Plan for the African penguin does consider a decline of food availability as a major driver of African Penguin population decline and recommends fishery closures around colonies (Government Gazette No. 42775 18 October 2019).

The reduction in colony sizes at most of the islands in Saldanha Bay will have had severe negative consequences for penguins. When Penguins breed in large colonies, packed close to one another, they are better able to defend themselves against egg and chick predation by Kelp gulls. Also, these losses are trivial at the colony level. However, the fragmented colonies and the rise in gull numbers associated with the rapidly expanding human settlements in the area during the 1980s, meant that gull predation became problematic. Kelp gull numbers in Saldanha Bay have decreased dramatically in recent years (see below), but the population remains at more than 2 000 pairs and gull predation on penguin eggs almost certainly remains problematic. Research has indicated that the provision of correctly designed artificial nest sites that provide protection both from gull predation and extreme temperatures (half concrete pipes were found to be superior to fibreglass artificial burrows) can be effective in enhancing fledging success (Pichegru 2012). Similarly, predation by seals (on land and around colonies) is having an increasingly negative impact on these dwindling colonies (Makhado *et al.* 2006, 2009). Additional stress, such as turbidity and increased vessel traffic, will not only impact penguins directly, but is likely to influence the location of schooling fish that the penguins are targeting and their ability to locate these schools. There are also concerns that toxin loads influence individual birds' health, reducing their breeding success and/or longevity (Game *et al.* 2009).



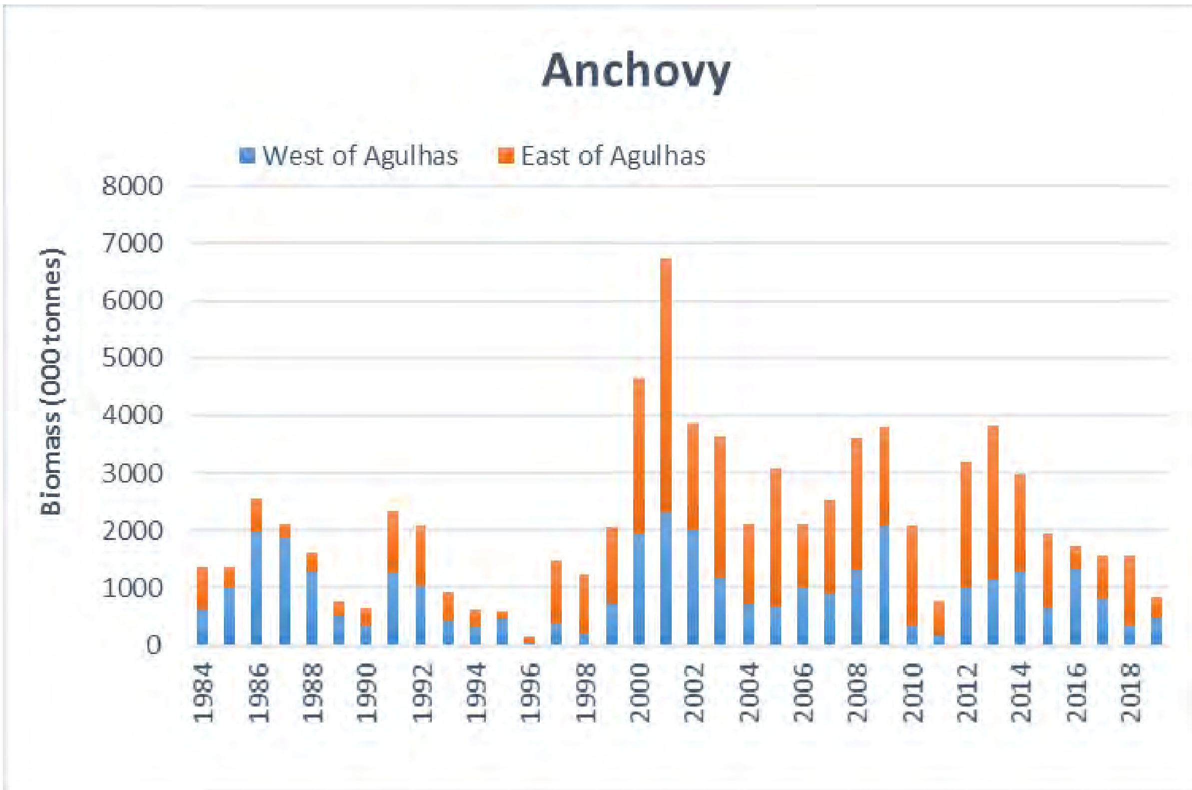


Figure 12.2. **Long term trends in the biomass of small pelagic fish (sardine and anchovy) to the west and east of Cape Agulhas based on hydro acoustic surveys conducted bi-annually from 1984-2019 (Data source: Department of Environment, Forestry and Fisheries).**

Parsons *et al.* (2016) conducted a large-scale health assessment on the African Penguin and found that this species is potentially exposed to conservation-significant pathogens. Disease constitutes a major ecological force and has been shown to play an even greater role in threated populations (Friend, McLean & Dein 2001 in Parsons *et al.* 2016). The effect of diseases on seabird population dynamics is currently poorly understood. Both, disease outbreaks as well as chronic diseases should both be considered as potential threats to the African Penguin and should be investigated further as part of the conservation efforts (Parsons *et al.* 2016).

In summary, the initial collapse of the penguin colonies in the area is probably related to food availability around breeding islands and in areas where birds not engaged in breeding are foraging, particularly before and after moulting. However, now that colonies have shrunk so dramatically, the net effect of local conditions at Saldanha Bay are believed to be an increasingly important factor in the continued demise of African penguin colonies at the islands. Concerningly, numbers of breeding pairs recorded in 2019 are the lowest on record for the ninth year in a row, whist the biomass of their small pelagic fish prey (particularly sardines) along the west coast is also at a historically low level.

The Kelp Gull *Larus dominicanus* breeds primarily on offshore islands, as well as a small number of mainland sites. The Islands in Saldanha Bay support a significant proportion of South Africa's breeding population. Within this area, the majority breed on Schaapen, Meeuw and Jutten Islands, with additional small but consistent breeding populations on Vondeling and Malgas islands. Small numbers of breeding kelp gulls were recorded on Marcus Island in 1978, 1985 and 1990-92, but breeding has since ceased, probably due to the causeway connecting the island to the mainland allowing access to mammal predators (Hockey *et al.* 2005). Kelp Gulls are known to eat the eggs of several other bird species (e.g. African penguins, Cape Cormorants and Hartlaub's Gulls). Prior to the 1960s, numbers of Kelp Gulls on offshore islands were controlled to protect the guano and egg producing species (Crawford *et al.* 1982).

Post 1970, Kelp Gull populations were no longer controlled, which, together with the supplementary food provided by fisheries and landfill sites resulted in the doubling of breeding pairs in South Africa by 2002 (Whittington *et al.* 2016). The introduction and spread of the invasive alien mussel species *Mytilus galloprovincialis* could also have contributed toward the increased availability of food. Consequently, pressures on guano-producing seabird populations shifted from guano exploitation to egg predation by increasing Kelp Gull numbers.

Since 2000, the populations on the islands have been steadily decreasing following large-scale predation by Great White Pelicans *Pelecanus onocrotalus* that was first observed in the mid-1990s (Crawford *et al.* 1997). During 2005 and 2006 pelicans caused total breeding failure of Kelp Gulls at Jutten and Schaapen Islands (de Ponte Machado 2007) the effects of which are still apparent (Figure 12.3.). Recent counts show that Kelp Gull numbers remain below those at the start of the comprehensive counting period although the decline appears to have abated with numbers stabilizing in the last four years (Figure 12.3.). The loss of breeding pairs at the Saldanha Bay Islands since 2000 were to some degree offset by an increase in numbers breeding on mainland sites, especially around greater Cape Town and along the south coast (Whittington *et al.* 2016).

Witteveen *et al.* (2017) found anthropogenic debris in Kelp Gull nests, especially in colonies located near landfill sites and coastal sites where there was a limited vegetation available for construction. Debris in nests can lead to injury or death as a result of entanglement of chicks and adults. Often ropes and straps are used by Kelp Gulls to construct nests. Plastic bags and food wrappers mostly appear to accumulate during the chick rearing period as those items were mostly regurgitated. Whether anthropogenic debris is playing an important role in the steady decreasing trend of Kelp Gull populations off the Saldanha Islands is however, unknown.

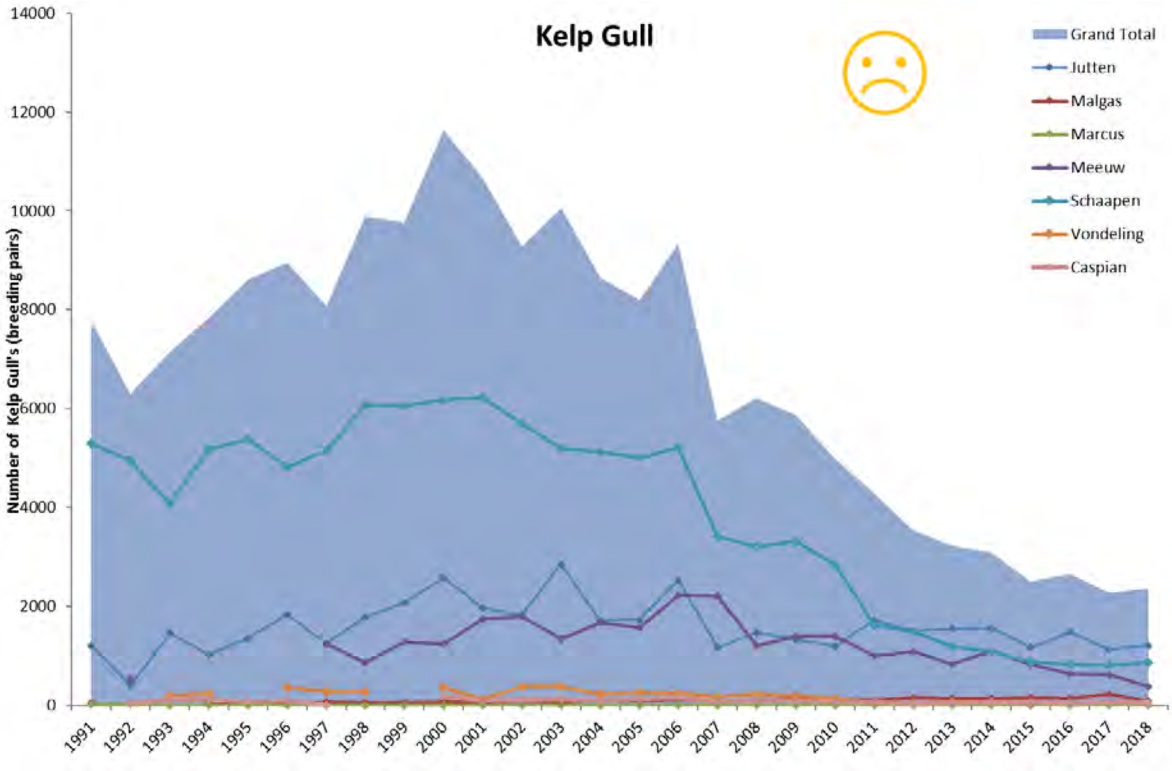


Figure 12.3. **Trends in breeding population of Kelp gulls at Jutten, Malgas, Marcus, Meeuw, Schaapen, Vondeling and Caspian Islands in Saldanha Bay from 1985-2018 measured in number of breeding pairs (Data source: Department of Environmental Affairs: Oceans & Coasts, 2019).**

Hartlaub's Gull, *Larus hartlaubii,* is about the 10th rarest of the world's roughly 50 gull species. It is endemic to southern Africa, occurring along the West Coast from Swakopmund to Cape Agulhas. It breeds mainly on protected islands but has also been found to breed in sheltered inland waters. Hartlaub's Gulls are relatively nomadic and can alter breeding localities from one year to the next (Crawford *et al.* 2003). The numbers breeding on the different islands are highly erratic, as are the total numbers in the Bay. The highest and most consistent numbers of breeding birds are found on Malgas, Jutten and Schaapen islands, with a few birds breeding Vondeling Island between 1991 and 1998 and last in 2006 when 30 pairs were recorded. They have also been recorded breeding on Meeuw Island in 1996, from 2002 to 2004 and again during 2012-2014 and in 2017 There are substantial inter-annual fluctuations in numbers of birds breeding, suggesting that in some years an appreciable proportion of the adults do not breed (Crawford *et al.* 2003). Natural predators of this gull are the Kelp Gull, African Sacred Ibis and Cattle Egret, which eat eggs, chicks and occasionally adults (Williams *et al.* 1990). In Saldanha Bay there is no discernible upward or downward trend over time. The total number of breeding pairs recorded in 2018 was just 36 pairs on Jutten and Malgas Islands, whilst in 2019, 996 breeding pairs were recorded with the majority on Schaapen Island.

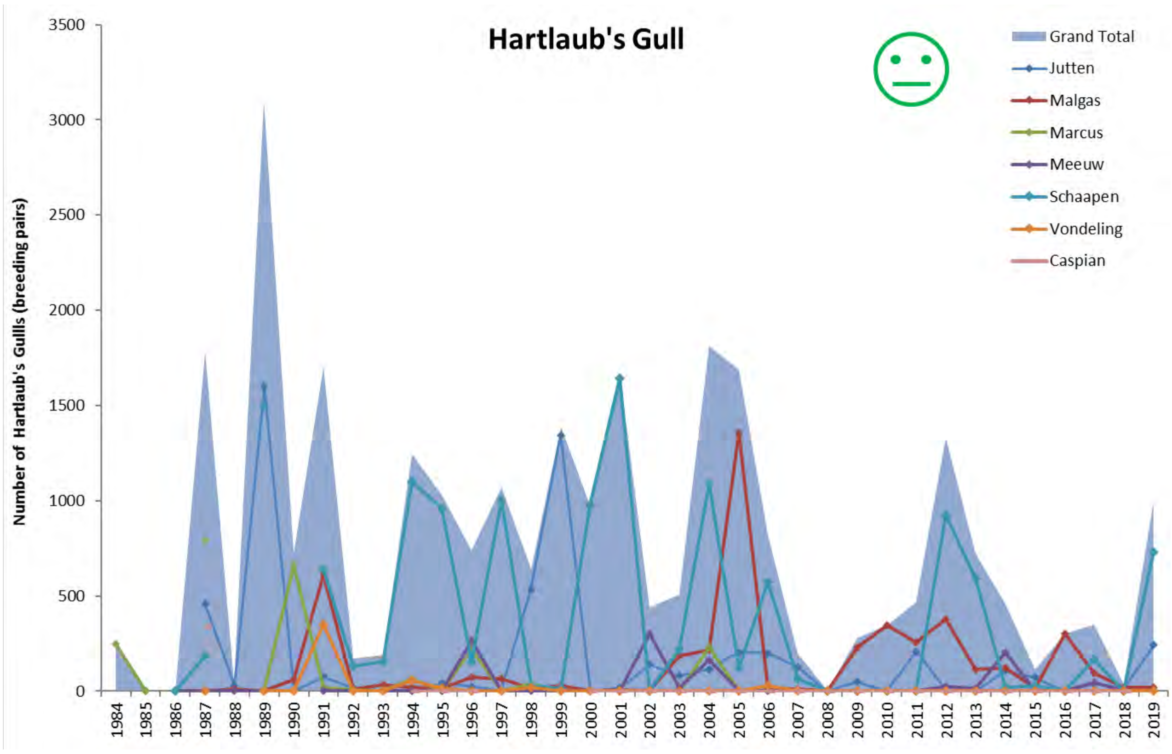


Figure 12.4. Trends in breeding population of Hartlaub's Gulls at Jutten, Malgas, Marcus, Meeuw, Schaapen, Vondeling and Caspian Islands in Saldanha Bay from 1984-2019 measured in number of breeding pairs (Data source: Department of Environmental Affairs: Oceans & Coasts 2020).

The Swift Tern, *Thalasseus bergii,* is a widespread species that occurs as a common resident in southern Africa. Swift Terns breed synchronously in colonies, usually on protected islands, and often in association with Hartlaub's Gulls. Sensitive to human disturbance, their nests easily fall prey to Kelp Gulls, Hartlaub's Gulls and Sacred Ibis (Le Roux 2002). During the breeding season, fish form 86% of all prey items taken, particularly pelagic shoaling fish, of which the Cape Anchovy (*Engraulis encrasicolus*) is the most important prey species. The steady increase in Swift Tern numbers between 2002 and 2005 coincided with a greater abundance of two of their main prey species, sardines and anchovies (Figure 12.2). However, since 2005, the population in the Western Cape has shifted south and eastward, coinciding with a similar shift of their prey species (Crawford 2009). In southern Africa, Swift Terns show low fidelity to breeding localities, unlike the African Penguin, Cape Gannet and Cape Cormorant, which enables them to rapidly adjust to changes in prey availability (Crawford 2009, 2014).

In Saldanha Bay, Jutten Island has been the most important island for breeding Swift Terns over the past 30 or more years, but breeding numbers are erratic at all the islands. The breeding population shifted to Schaapen Island in 2007, but no swift terns were reported breeding on islands in the Bay for the three years following this, the longest absence on record. Subsequent to this, Swift Terns have bred on the Saldanha Islands every year, albeit with the typical, erratic variability. In the last two years the numbers recorded have fluctuated from amongst the highest on record in 2018, to amongst the lowest in 2019, when just two pairs were recorded (Figure 12.5.).

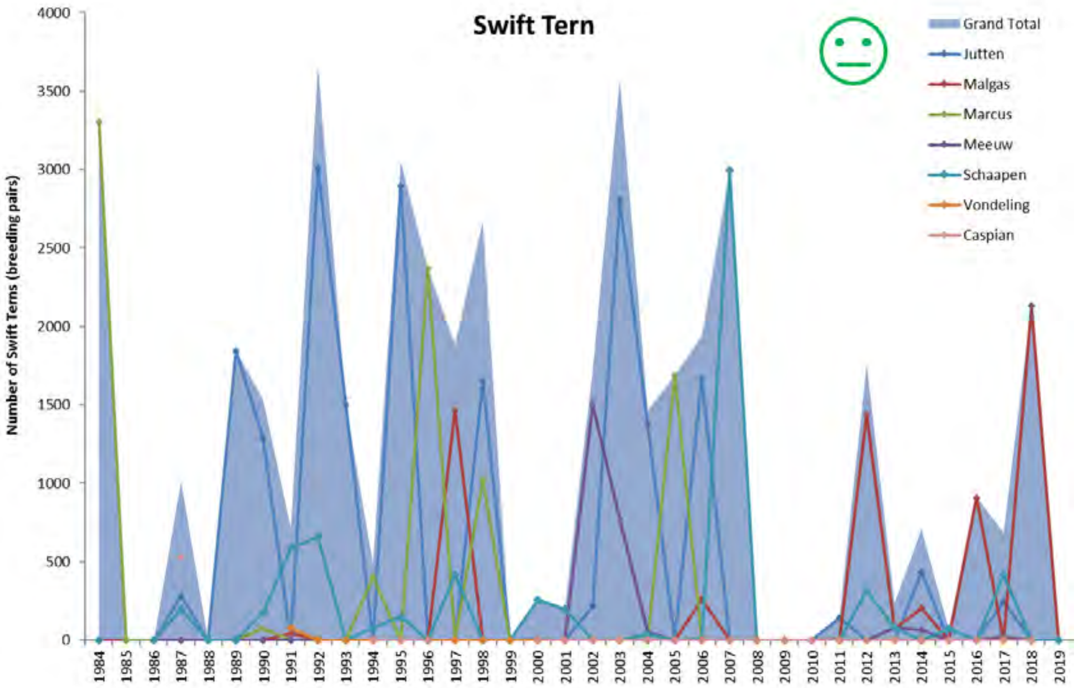


Figure 12.5. **Trends in breeding population of Swift Terns at Jutten, Malgas, Marcus, Meeuw, Schaapen, Vondeling and Caspian Islands in Saldanha Bay from 1984-2019 measured in number of breeding pairs (Data source: Department of Environmental Affairs: Oceans & Coasts 2020).**

Cape Gannets *Morus capensis* are restricted to the coast of Africa, from the Western Sahara, around Cape Agulhas to the Kenyan coast. In southern Africa they breed on six offshore islands, three off the Namibian coast, and two off the west coast of South Africa (Bird Island in Lambert's Bay and Malgas Island in Saldanha Bay), and one (Bird Island) at Port Elizabeth. The Cape Gannet is listed as Endangered on the IUCN's global Red Data List, due to its restricted range and population declines (Birdlife International 2018).

Cape Gannets breed on islands which afford them protection from predators. They feed out at sea and will often forage more than a hundred kilometres away from their nesting sites (Adams & Navarro 2005). This means that only a small proportion of foraging takes place within Saldanha Bay. The quality of water and fish stocks in Saldanha Bay should therefore not have a significant effect on the Cape Gannet population.

The bird colony at Malgas Island has shown substantial population fluctuation since the early 1990's and an erratic decline since 1996 (Figure 12.6.). The 2001-2014 data revealed a dramatic decline in the breeding population on Malgas Island. Numbers of breeding pairs recovered somewhat in 2017 but then declined steeply again to low levels in 2019 that were less than half the peak recorded in the mid-1990s. The decline in numbers at Malgas Island contrasts with population figures for Bird Island, off Port Elizabeth, where numbers have increased. The total South African gannet population appears to respond to the population dynamics of small pelagic fish (particularly sardines), with the number of breeding pairs averaging at 123 thousand pairs since 1995 (Crawford *et al.* 2014). A study suggested that Cape Gannet population trends are driven by food availability during their breeding season (Lewis *et al.* 2006). Pichegru *et al.* (2007) showed that Cape Gannets on the west coast have been declining since the start of the eastward shift of the pelagic fish in the late 1990s. This has resulted in west coast gannets having to increase their foraging efforts. During the breeding season, they forage in areas with very low abundance of their preferred prey, and feed primarily on low-energy hake trawl fishery discards which constituted 93% of total prey intake (Crawford *et al.* 2006, Pichegru *et al.* 2007). A bioenergetics model showed that enhanced availability of low-energy hake fishery discards does not seem to compensate for the absence of natural prey and a study of foraging energetics suggested that Gannets tracked from Malgas Island were not maintaining their energy budget during feeding flights (Pichegru *et al.* 2007, Gremillet *et al.* 2016). Despite only a small documented overlap (13%) in Cape Gannet foraging zones from Malgas Island with the purse-seine fishery, the total fishery catch was estimated at 41% of the food requirements of the colony (Okes *et al.* 2009). Some of these studies have called for increased restrictions on purse-seine fishing in the vicinity of bird colonies, but these conclusions have been challenged by fishery scientists who point out that small pelagic fish biomass was actually increasing in the area at the time the Cape Gannet numbers started declining (Figure 12.2). Gannets with their extensive foraging range and diverse diets have proved adaptable to the changes in pelagic fish distribution and nationally numbers have not declined (Crawford 2014).

Possibly of greater significance for the Malgas Island Cape Gannet Colony and of more concern at a local level, are high rates of predation by Cape fur seals *Arctocephalus pusilus pusillus*, Kelp Gulls and until recently, the Great White Pelican *Pelecanus onocrotalus* (Makhado *et al.* 2006, Pichegru *et al.*

2007). Kelp Gull predation accounts for between one and two thousand gannet breeding failures per season in average years (Pelican Watch *pers. comm.* 2017). Furthermore, Cape Fur Seals prey on fledgling sea birds that land in the waters around their home islands for the first time (David *et al.* 2003, Makhado *et al.* 2009). Seal numbers nationally increased at an average of 3.5% per annum since 1971 until 1993 when aerial census of seal colonies was undertaken (David *et al.* 2003). In Saldanha waters, seal numbers have increased dramatically since 2000 when they started re-colonising Vondeling Island. A census in 2014 recorded over 23 000 seal pups on the island and the consequent increase in competition for already depleted food resources has led groups of young male seals to augment their normal diet by hunting cormorant and gannet fledgling on their first forays from the islands (Pelican Watch *pers. comm*. 2017).

Estimates of Cape Gannet mortality caused by Cape Fur Seals were 6 000 fledglings around Malgas Island in the 2000/01 breeding season, 11 000 in 2003/04 and 10 000 in 2005/06 (Makhado *et al.* 2006). This amounted to about 29%, 83% and 57% of the overall production of fledglings at the island in these breeding seasons respectively, despite an ongoing "problem" seal culling programme around Malgas Island that was initiated in 1993 (David *et al.* 2003, Makhado *et al.* 2009). These seal predation rates were considered unsustainable and largely responsible for the 25% decline in the Malgas Island Cape Gannett population between 2001 and 2006 (Makhado *et al.* 2006). Seal predation of seabirds is ongoing and it was estimated by the Department of Environmental Affairs seal culling team that in January 2016 "… all young gannets landing on the waters around Malgas were taken by seals…" (Pelican Watch *pers. comm.* 2017). These recent findings have changed the overall health of the Gannet population on Malgas Island from Fair to Poor based on the ongoing predation by fur seals. Management measures were implemented between 1993 and 2001, and 153 fur seals seen to kill Gannets were shot (Makhado *et al.* 2006). This practice has continued in an effort to improve breeding success (Makhado *et al.* 2009). The effects of this may be manifest in the slight recovery in Gannet numbers between 2006 and 2009, but numbers have declined further since then suggesting that predation and other pressures such as food availability remain problematic (particularly in light of ongoing declines in small pelagic fish biomass along the west coast).

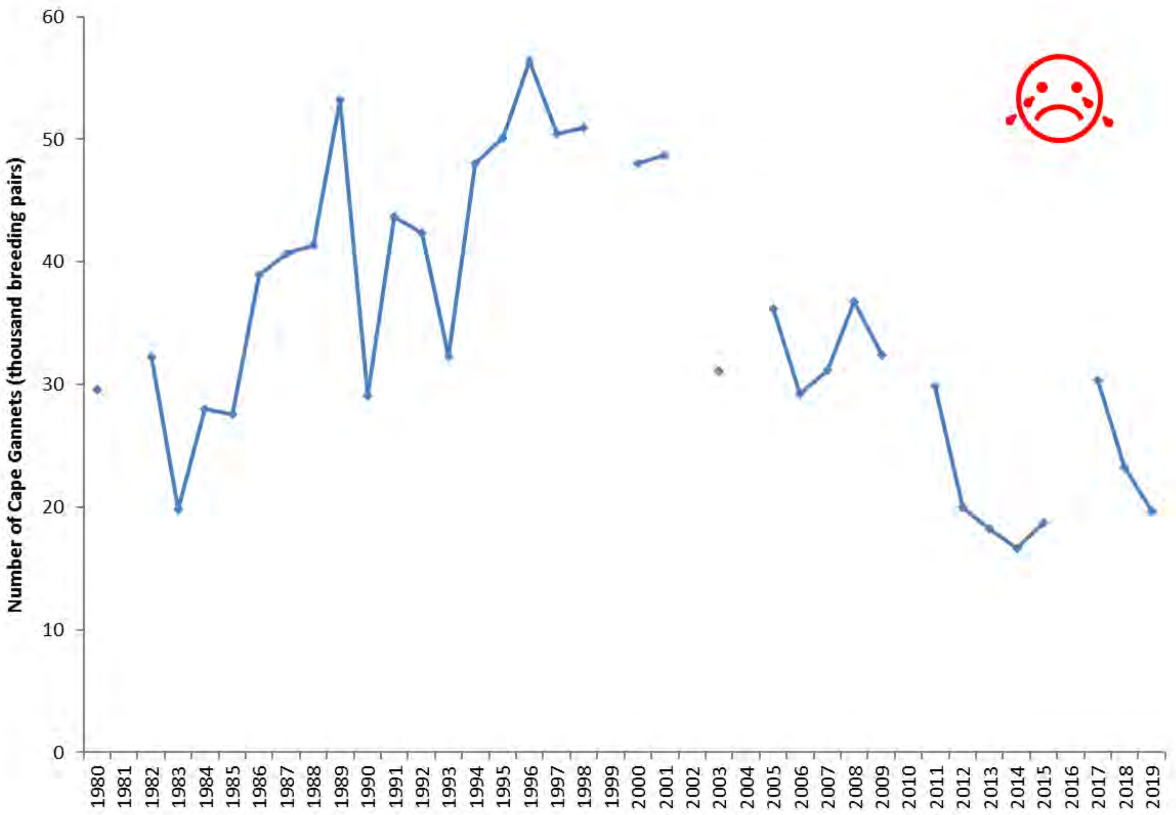


Figure 12.6. Trends in breeding population of Cape Gannets at Malgas Island, Saldanha Bay from 1980-2019 measured in number of breeding pairs (Data source: Department of Environmental Affairs: Oceans & Coasts).

Cape Cormorants *Phalacrocorax capensis* are endemic to southern Africa, where they are abundant on the west coast but less common on the east coast, occurring as far east as Seal Island in Algoa Bay.

They breed between Ilha dos Tigres, Angola, and Seal Island in Algoa Bay, South Africa. They generally feed within 10-15 km of the shore, preying on pelagic goby *Sufflogobius bibarbatus*, Cape anchovy *Engraulis capensis*, pilchard *Sardinops sagax* and Cape horse mackerel *Trachurus trachurus* (du Toit 2004).

Key colonies of the Cape Cormorant in South Africa and Namibia have undergone very rapid population declines over the past three generations and the Cape Cormorant has therefore been up listed to Endangered (BirdLife International 2018). Declines are primarily believed to have been driven by collapsing pelagic fish stocks (BirdLife International 2015). However, pelagic fish stocks increased greatly in the late 1990s and early 2000s, and although sardine biomass subsequently crashed, anchovy biomass remained relatively high until recently (Figure 12.2). This suggests that other factors are also involved in declining Cape Cormorant numbers. The species is susceptible to oiling and avian cholera outbreaks. This trend currently shows no sign of reversing, and immediate conservation action is required to prevent further declines (Crawford *et al.* 2013, 2015).

In South Africa, numbers decreased during the early 1990s following an outbreak of avian cholera, predation by Cape fur seals and White Pelicans as well as the eastward displacement of sardines off South Africa (Crawford *et al.* 2007). A semi-systematic count by the Pelican Watch on Jutten in December 2015, suggests that about 3 000 young Cape Cormorants were taken by seals during the fledging period. There are large inter-annual fluctuations in breeding numbers due to breeding failure, nest desertion and mass mortality related to the availability of prey, for which they compete with commercial fisheries. This makes it difficult to accurately determine population trends on the Saldanha Islands. In addition, during outbreaks of avian cholera, tens of thousands of birds die. Cape Cormorants are also vulnerable to oiling and are difficult to catch and clean. Discarded fishing gear and marine debris also entangles and kills many birds. Kelp Gulls prey on Cape Cormorant eggs and chicks and this is exacerbated by human disturbance, especially during the early stages of breeding, as well as the increase in gull numbers (du Toit 2004).

The Saldanha Bay population has been quite variable since the start of monitoring in 1988, with the bulk of the population residing on Malgas and Jutten Island in recent years, until 2019 when the 2 089 breeding pairs recorded were found exclusively on Malgas Island (Figure 12.7.). Overall, the number of breeding pairs declined gradually since the 1990s, and rather dramatically since 2015. In 2013, a total of only 801 breeding pairs were recorded, representing the lowest level recorded to date (Figure 12.7.). Between 2013 and 2016, a short-lived recovery of breeding pairs to 9 273 was linked to an increase in the number of breeding pairs on Malgas Island. Since then, numbers of breeding pairs have dropped once again to a total of 2089 in 2019, which is the second lowest this century.

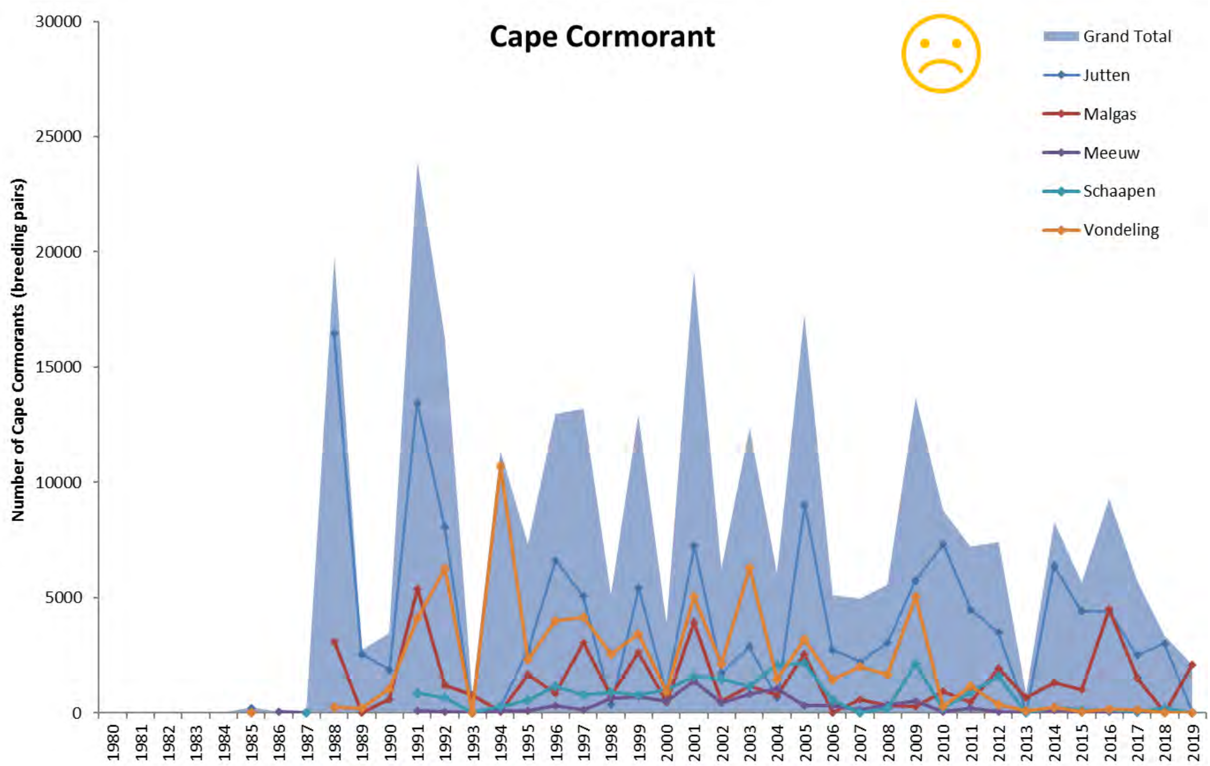


Figure 12.7. Trends in breeding population of Cape Cormorants at Jutten, Malgas, Meeuw Schaapen, and Vondeling islands in Saldanha Bay from 1980 – 2019 measured in number of breeding pairs (Data source: Oceans & Coasts, Department of Environmental Affairs 2020).

Bank Cormorants *Phalacrocorax neglectus* are endemic to the Benguela upwelling region of southern Africa, breeding from Hollamsbird Island, Namibia, to Quoin Rock, South Africa. They seldom range farther than 10 km offshore. Their distribution roughly matches that of kelp *Ecklonia maxima* beds. They prey on various species of fish such as the pelagic goby *Sufflogobius bibarbatus*, crustaceans and cephalopods, feeding mainly amongst kelp where they catch West Coast rock lobster, *Jasus lalandii* (du Toit 2004). The total population decreased from about 9 000 breeding pairs in 1975 to less than 5 000 pairs in 1991-1997, to 2 800 pairs in 2006 (Kemper *et al.* 2007). The South African population approximately halved from 1. 500 pairs in 1978-1980 to 800 pairs in 2011-2013 (Crawford *et al.* 2015). One of the main contributing factors to the decrease in the North and Western Cape colonies was a major shift in the availability of the West Coast rock lobster from the West Coast to the more southern regions, observed between the late 1980s and early 1990s to the turn of the century (Cockcroft *et al.* 2008). The abundance of lobsters was further severely affected by an increase in the number and severity of mass lobster strandings (walkouts) during the 1990s and increases in illegal fishing, with the national stock rock lobster status now estimated at just 2% of pristine biomass (Cockcroft *et al.* 2008, DAFF 2017). Ongoing population declines led to the Bank Cormorant's status being changed from Vulnerable to Endangered (Birdlife International 2011).

Breeding pair count data from the Saldanha Bay area shows the dramatic decrease in the population at Malgas Island, which was previously the most important island for this species. The number of breeding pairs on Jutten, Marcus and Vondeling has declined steadily since 2003 on all the islands. Overall, the population in Saldanha Bay has declined drastically by approximately 98% since 1990 (Figure 12.8.). Currently numbers of breeding pairs are the lowest on record, with just five pairs recorded in 2019. These declines are mainly attributed to scarcity of their main prey, the rock lobster which in turn has reduced recruitment to the colonies (Crawford 2007, Crawford *et al.* 2008c). Bank Cormorants are also very susceptible to human disturbance and eggs and chicks are taken by Kelp Gulls and Great White Pelicans. Increased predation has been attributed to the loss of four colonies in other parts of South Africa and Namibia (Hockey *et al.* 2005). Smaller breeding colonies are more vulnerable to predation which would further accelerate their decline. Birds are also known to occasionally drown in rock-lobster traps, and nests are often lost to rough seas.

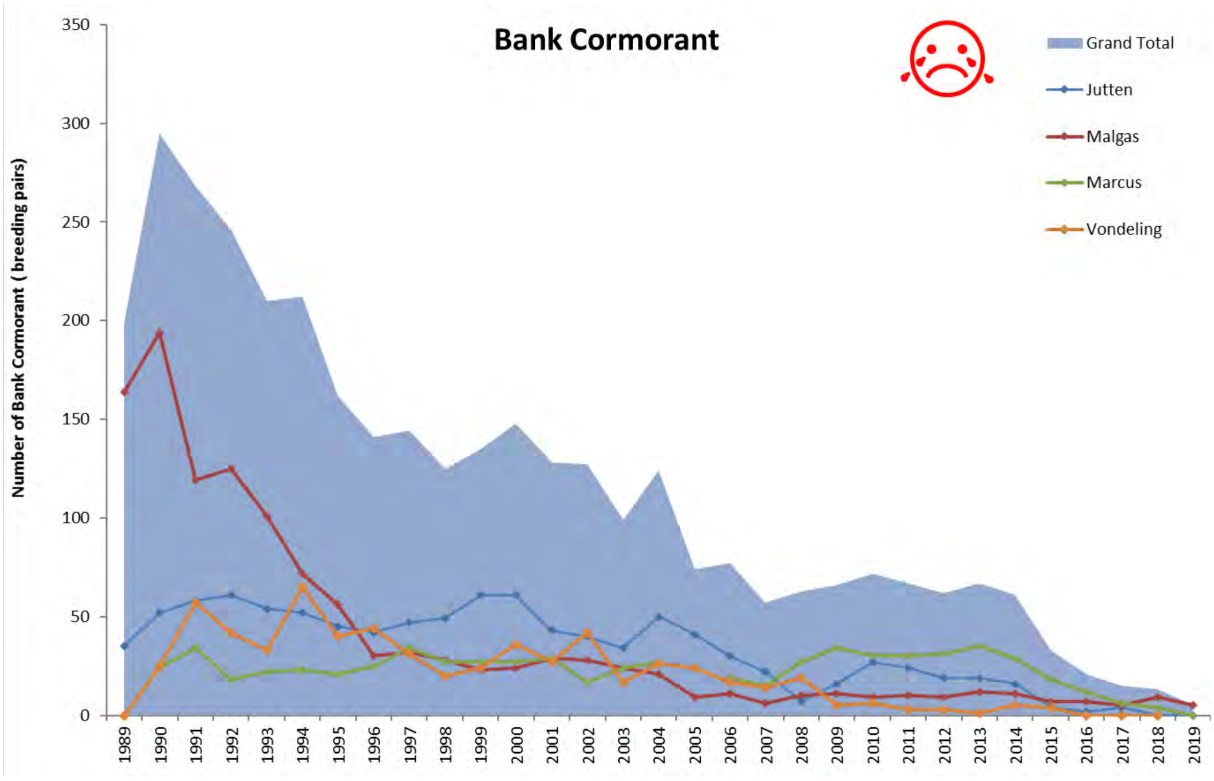


Figure 12.8. **Trends in breeding population of Bank Cormorants at Jutten, Malgas, Marcus and Vondeling islands in Saldanha Bay from 1980 – 2019 measured in number of breeding pairs (Data source: Oceans & Coasts, Department of Environmental Affairs 2020).**

The **White-breasted Cormorant** *Phalacrocorax lucidus,* also known as Great Cormorant, occurs along the entire southern African coastline, and is common in the eastern and southern interior, but occurs only along major river systems and wetlands in the arid western interior. The coastal population breeds from Ilha dos Tigres in southern Angola, to Morgan Bay in the Eastern Cape. Along the coast, White-breasted Cormorants forage offshore, mainly within 10 km of the coast, and often near reefs. White-breasted Cormorants that forage in the marine environment feed on bottom-living, mid-water and surface-dwelling prey, such as sparid and mugillid fishes e.g. Steentjies, white stumpnose and harders (du Toit 2004). This species forages in Saldanha Bay and Langebaan Lagoon, making it susceptible to local water quality and fishing activities (Hockey *et al.* 2005).

Within Saldanha Bay, breeding effort has occasionally shifted between islands. White-breasted Cormorants bred on Malgas Island in the 1920's, and low numbers of breeding pairs were counted on Marcus and Jutten Islands intermittently between 1973 and 1987 when they stopped breeding there and colonized Schaapen, Meeuw and Vondeling islands (Crawford *et al.* 1994). Most of the breeding population was on Meeuw in the early 1990s but shifted to Schaapen in about 1995. By 2000, the breeding numbers at Schaapen had started to decline and the breeding population had shifted entirely back to Meeuw by 2004, where it has remained since (Figure 12.9.). Overall, numbers of breeding pairs were more or less stable until 2012 but have declined steeply since then. The last five annual counts (2015-2019) have, been the lowest on record and substantially down from the 100-150 breeding pairs recorded in most years prior to 2012. The 16 pairs recorded in 2019 represent the lowest number recorded in the past 31 years.

Human disturbance poses a threat at breeding sites. These cormorants are more susceptible to disturbance than the other marine cormorants, and leave their nests for extended periods if disturbed, exposing eggs and chicks to Kelp Gull predation. Other mortality factors include Avian Cholera, oil pollution, discarded fishing line and hunting inland (du Toit 2004). White Breasted Cormorants also predate on fish caught in gill nets utilized in the harder fishery and risk becoming entangled in the gear and drowning. Effort in the harder fishery has increased in recent years and the average size of harders in the Saldanha- Langebaan fishery has decreased (see fish chapter), potentially negatively affecting foraging opportunities for White Breasted Cormorants in the Bay. Due to Schaapen Islands' close proximity to the town of Langebaan, the high boating, kite-boarding and other recreational uses of the area may have been an important source of disturbance to these birds. The substantial growth in participation in recreational water sports (particularly kite boarding) over the last two decades could have been a contributing factor to the shift in breeding location from Schaapen to Meeuw Island in 2004, but this appears unlikely given that the opposite shift happened ten years previously.

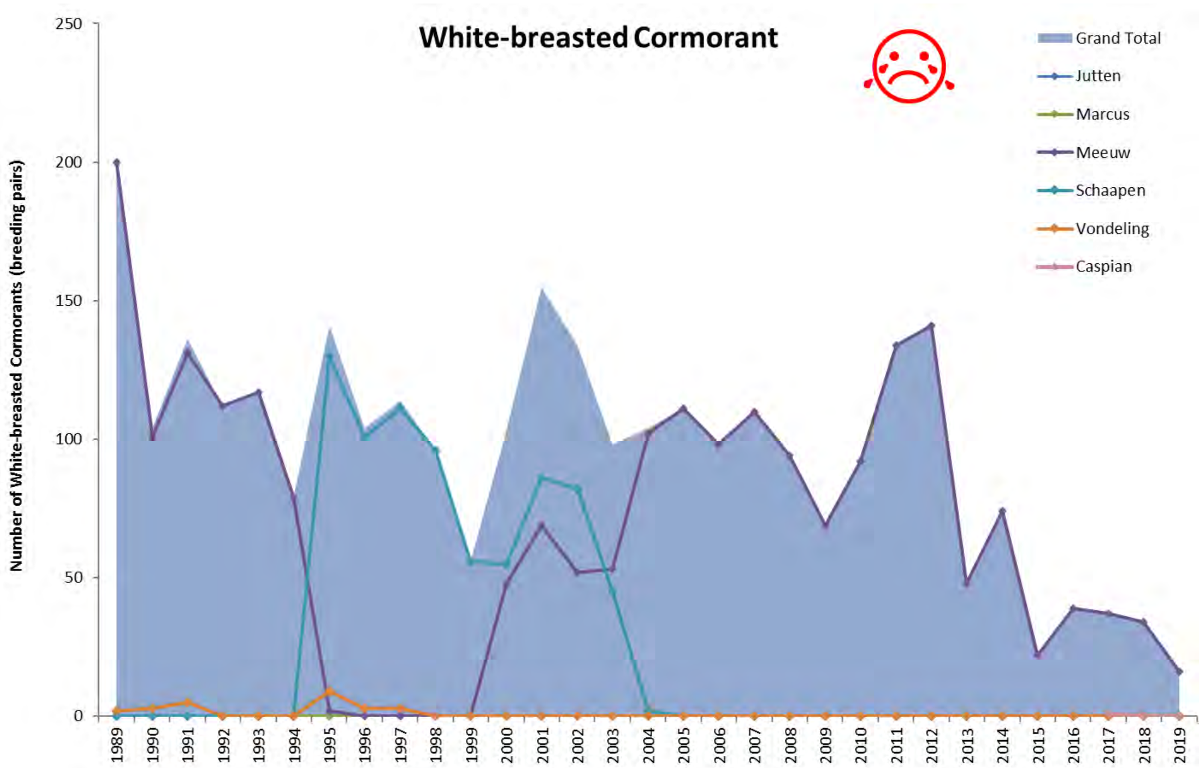


Figure 12.9. **Trends in breeding population of White-breasted Cormorants at Jutten, Marcus, Meeuw, Schaapen, Vondeling and Caspian islands in Saldanha Bay from 1980 – 2019 measured in number of breeding pairs (Data source: Department of Environmental Affairs: Oceans & Coasts 2020).**

The **Crowned Cormorant** *Microcarbo coronatus* is endemic to Namibia and South Africa, occurring between the Bird Rock Guano Platform in southern Namibia and Quoin Rock, South Africa. It is listed as Near Threatened on the IUCN's Red Data List due to its small and range restricted population, making it very vulnerable to threats at their breeding colonies (Birdlife International 2018). This species is highly susceptible to human disturbance and predation by fur seals, particularly of fledglings. Crowned Cormorants generally occur within 10 km from the coastline and occasionally in estuaries and sewage works up to 500 m from the sea. They feed on slow-moving benthic fish and invertebrates, which they forage for in shallow coastal waters and among kelp beds (du Toit 2004). Populations of this species have been comprehensively counted since 1991 (Figure 12.10.). Since then, numbers have shown considerable interannual variations with an overall decreasing trend (Figure 12.10.). Currently, numbers are well below average, and the lowest in the last three decades. Furthermore, the trajectory in population size has been downwards for the last decade.

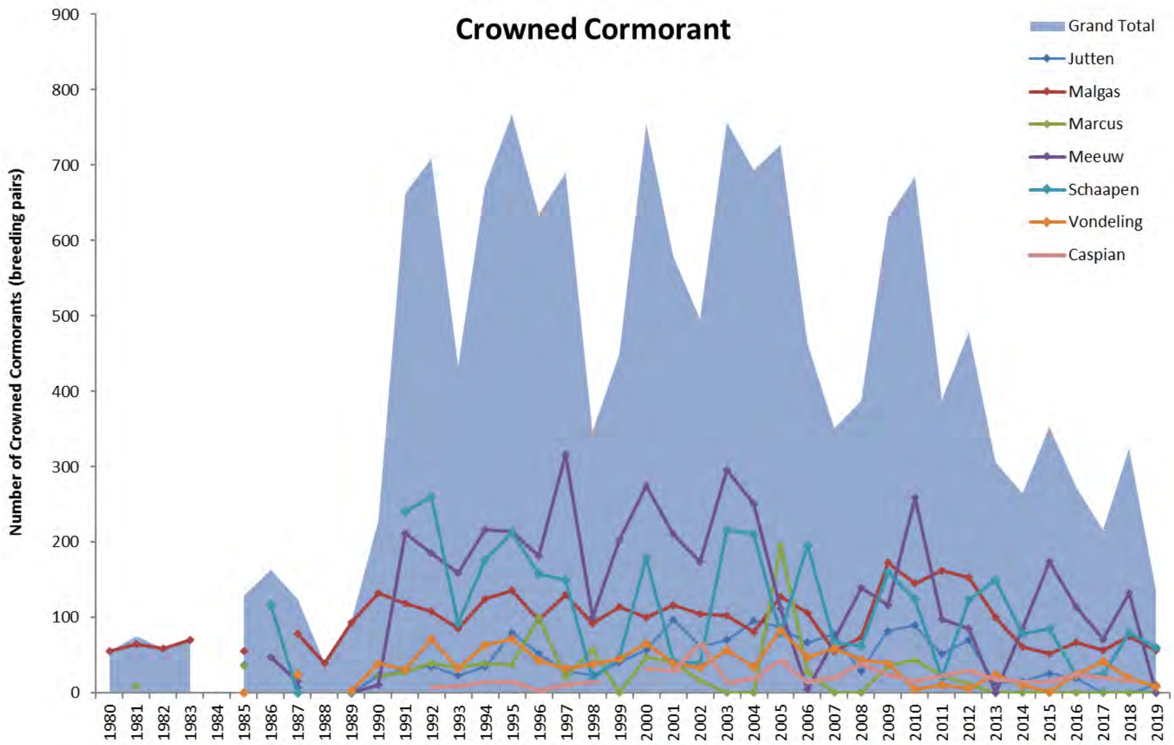


Figure 12.10. Trends in breeding population of Crowned Cormorants at the Jutten, Malgas, Marcus, Meeuw, Schaapen, Vondeling, and Caspian islands in Saldanha Bay from 1980 – 2019 measured in number of breeding pairs (Data source: Department of Environmental Affairs: Oceans & Coasts 2020).

The **African Oystercatcher** *Haematopus moquini* is endemic to southern Africa and is currently listed as Least Concern on the IUCN Red Data Species List (Birdlife International 2017). Their global numbers increased dramatically from the 1980s, which was attributed primarily to the introduction and proliferation of the alien mussel *Mytilus galloprovincialis,* as well as due to the enhanced protection of the Oystercatcher throughout much of its range (Hockey & van Erkom Schurink 1992, Loewenthal 2007). This population growth lead to the revision of the original Endangered status in 2017 (Birdlife International 2017). The African Oystercatcher breeds in rocky intertidal and sandy beach areas from Namibia to southern KwaZulu-Natal.

African Oystercatchers are resident on the islands, where highest numbers are encountered at Marcus, Malgas and Jutten Islands (Figure 12.11.). The islands in Saldanha Bay contribute a fair proportion to the global population that was estimated at 6 670 in 2007 (Loewenthal 2007). The population stabilised in the early 2000s (Figure 12.11.). This possibly reflects stabilisation in the alien Mediterranean mussel biomass as the island rocky shore ecosystems settle into their new equilibrium. Oystercatchers could be affected by water quality in Saldanha Bay in as much as it affects intertidal invertebrate abundance. Like most of the birds described above, they are, however, vulnerable to catastrophic events such as oil spills. Threats to the breeding success of these birds include human-induced habitat degradation, uncontrolled dogs predating on chicks and the drowning of chicks hiding from humans and their associated pets (Loewenthal 2007). Due to the sad passing of the two champions of the Oystercatcher Conservation Project (Prof. Phil Hockey and Dr Douglas Loewenthal) the regular censuses of oystercatchers in Saldanha Bay have stopped and no data has been collected since 2016.

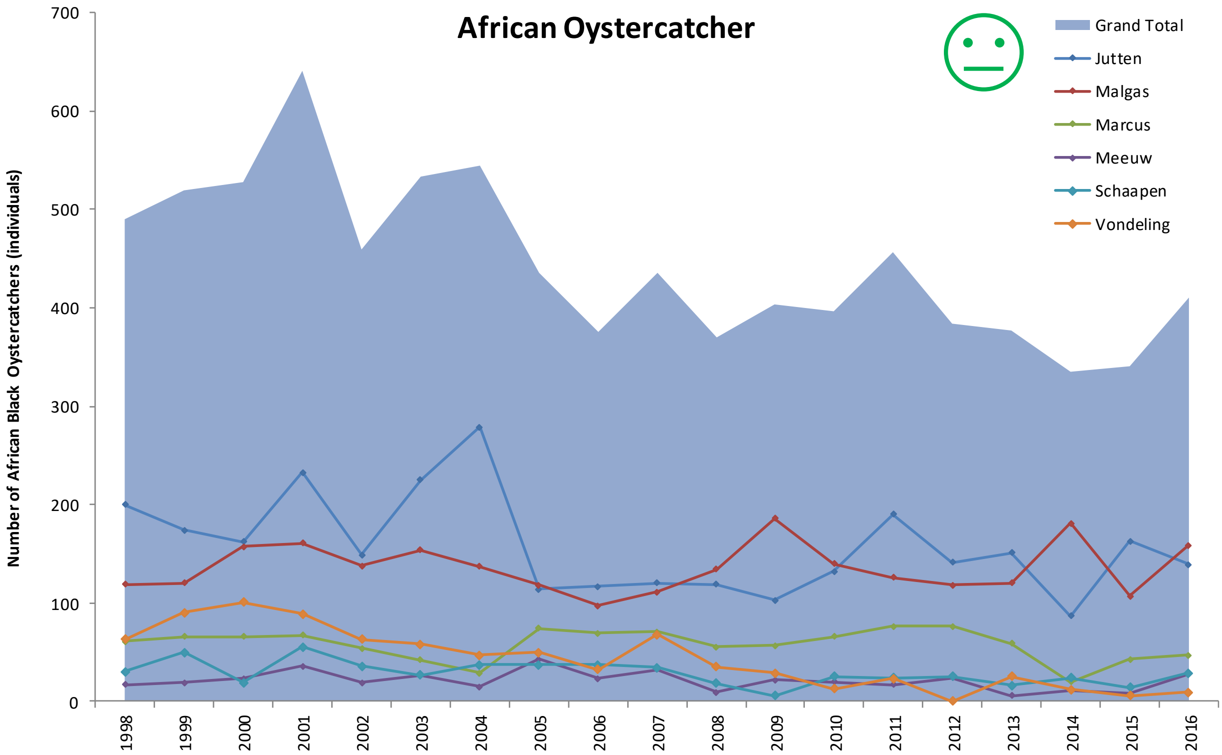


Figure 12.11. **Trends in breeding population of African Oystercatchers on Jutten, Malgas, Marcus, Meeuw, Schaapen, and Vondeling Islands from 1988 - 2016. (Data source: Department of Environmental Affairs: Oceans & Coasts 2019).**

12.3 Birds of Langebaan Lagoon

12.3.1 National importance of Langebaan Lagoon for waterbirds

Langebaan Lagoon, with its warm, sheltered waters and abundance of prey, supports a high diversity and abundance of waterbirds, especially in summer when it is visited by thousands of migratory waders from the northern hemisphere. A number of commonly found migratory waders are globally recognised as Near Threatened and include Red Knot *Calidris canutus,* Curlew Sandpiper *Calidris ferruginea,* Bar-tailed Godwit *Limosa lapponica* and Eurasian Curlew *Numenius arquata*. Langebaan Lagoon represents a critical 'wintering' area for migratory waterbirds in South Africa (Underhill 1987) and is recognised as an internationally important site under the Ramsar Convention on Wetlands of International Importance, to which South Africa is a signatory.

The true importance of Langebaan Lagoon for waders cannot be assessed without recourse to a comparison with wader populations at other wetlands in southern Africa. During the summer of 1976 to 1977, wader populations at all coastal wetlands in the south-western Cape were counted (Siegfried 1977). The total population was estimated at 119 000 birds of which 37 000 occurred at Langebaan. Only one other coastal wetland, the Berg River estuary, contained more than 10 000 waders. Thus, Langebaan Lagoon held approximately one third of all the waders in the south-western Cape (Siegfried 1977). Studies were extended to Namibia (then South West Africa) in the summer of 1976-77. Walvis Bay Lagoon contained up to 29 000 waders and Sandvis had approximately 12 000 waders. Therefore, it was determined that Langebaan Lagoon was the most important wetland for waders on the west coast of southern Africa (Siegfried 1977).

Taking species rarity and abundance into account, Langebaan Lagoon has been ranked fourth of all South African coastal lagoons and estuaries in terms of its conservation importance for waterbirds (Turpie 1995). With regard to density and biomass of waders, Langebaan Lagoon compared favourably to other internationally important coastal wetlands in West Africa and Europe.

Waterbird numbers on Langebaan Lagoon have, however, declined dramatically since monitoring began in the 1970s. Decreases in both migratory and resident wader numbers are a common trend around the South African coast. Decreases in numbers of migrants can be attributed to loss of breeding habitat and hunting along their migration routes as well as human disturbance and habitat loss on their wintering grounds. The fact that numbers of resident waders may also be declining suggests that local human disturbance is also to blame at Langebaan Lagoon. In 1985, Langebaan Lagoon was declared a National Park (West Coast National Park), and recreational activities such as boating, angling and swimming have since been controlled within the Lagoon through zonation. Nevertheless, the dramatic increases in visitor numbers to the area over the last three decades and the more recent increases in sporting activities on the lagoon impact on some of the important feeding areas in the lagoon. The area most impacted by increased visitor numbers and water sports activities are the sandflats near Oesterwal, that in the 1970's, were identified as one of the most important feeding areas for waders.

12.3.2 The main groups of birds and their use of habitats and food

The waterbirds of Langebaan Lagoon can be grouped into seven categories, namely (1) Cormorants, darters, and pelicans; (2) wading birds; (3) waterfowl; (4) waders (5) gulls and terns (6) kingfishers; and (7) birds of prey (Table 12.1). The relative contribution of the various bird groups to the bird numbers in the lagoon differs substantially in summer and winter, due to the prevalence of migratory birds in summer (Figure 12.12). Currently, waders account for about 60% of the birds in Langebaan Lagoon during summer, nearly all of these being migratory. In winter, the contribution by resident waders increases to around 8%, and numbers of wading birds increase from 31 to 61% of total bird numbers. The influx of waders into the area during summer accounts for most of the seasonal change in community composition. Most of the Palaearctic migrants depart synchronously in early April, but the immature birds of many of these species remain behind, accounting approximately 13% of the total waterbird numbers. The resident species take advantage of relief in competition for resources and use this period to breed. The migrants return over a longer period in spring, with birds beginning to filter in from August, rising rapidly in numbers during September and November. In the 1970s, it was determined that the most important sandflats, in terms of the density of waders they support, were in Rietbaai, in the upper section of Langebaan Lagoon, and at the mouth, near Oesterwal. The important roosting sites were the salt marshes, particularly between Bottelary and Geelbek (Summers 1977).

Table 12.1. Major waterbird groups found in Langebaan Lagoon, and their defining features.

Bird group	Defining features, typical/dominant species
Cormorants, darters & pelicans	Cormorants, darters and pelicans are common as a group, but are dominated by the marine cormorants which breed on the Saldanha Bay islands. Great White Pelicans visit the bay and lagoon to feed, but they breed beyond the area at Dassen Island. African Darters *Anhinga rufa* are uncommon and are more typical of lower salinities and habitats with emergent vegetation which are not common in the study area.
Wading birds	This group comprises the egrets, herons, ibises, flamingos and spoonbills. Loosely termed piscivores, their diet varies, with fish usually dominating, but often also includes other vertebrates, such as frogs, and invertebrates. The ibises were included in this group, though their diet mainly comprises invertebrates and is fairly plastic. They tend to be tolerant of a wide range of salinities. Wading piscivores prefer shallow water up to a certain species dependant wading depth.
Waterfowl	This group includes waterfowl in the orders Podicipediformes (grebes), Anseriformes (ducks, geese) and Gruiformes (rails, crakes, gallinules, and coots). Waterfowl occur in fairly large numbers because of the sheer size of the study area, but they are not as dense as they might be in freshwater wetland habitats or nearby areas such as the Berg River floodplain. Piscivorous waterfowl comprises the Grebes; herbivorous waterfowl are dominated by species that tend to occur in lower salinity or freshwater habitats, such as the Southern Pochard and the rallids, and are therefore not common in the lagoon. The omnivorous waterfowl comprises ducks which eat a mixture of plant material and invertebrate food such as small crustaceans. Species include the Yellow-billed Duck, Cape Teal, Red-billed Teal and Cape Shoveller. Although varying in tolerance, these species are tolerant of more saline conditions.
Waders	This group includes all the waders in the order Charadriiformes (e.g. Greenshank, Curlew Sandpiper). Waders feed on invertebrates that mainly live in intertidal areas, at low tide, both by day and night (Turpie & Hockey 1993). They feed on a whole range of crustaceans, polychaete worms and gastropods, and adapt their foraging techniques to suit the type of prey available. Among the waders, plovers stand apart from the rest in that they have insensitive, robust bills and rely on their large eyes for locating prey visually. Oystercatchers have similar characteristics, using their strong bills to prise open shellfish. Most other waders have soft, highly sensitive bills and can locate prey by touch as well as visually. Those feeding by sight tend to defend feeding territories, whereas tactile foragers often forage in dense flocks. The influx of waders into the area during summer accounts for most of the seasonal change in community composition. Most of the Palaearctic migrants depart quite synchronously around early April, but the immature birds of many of these species remain behind and do not don the breeding plumage of the rest of the flock. The resident species take advantage of relief in competition for resources and use this period to breed. The migrants return more gradually in spring, with birds beginning to trickle in from August, and numbers rising rapidly during September to November. Waders require undisturbed sandflats in order to feed at low tide and undisturbed roosting sites at high tide. In the 1970's it was determined that the most important sandflats, in terms of the density of waders they support, were in Rietbaai, in the upper section of Langebaan Lagoon, and at the mouth, near Oesterwal. The important roosting sites were the salt marshes, particularly between Bottelary and Geelbek (Summers 1977).
Gulls and terns	This group comprises the rest of the Charadriiformes and includes all the gull and tern species occurring in the lagoon. These species are primarily piscivorous, but also feed on invertebrates. Gulls and terns are common throughout the area. Although their diversity is relatively low, they make up for this in overall biomass, and form an important group. Both Kelp Gulls and Hartlaub's Gulls occur commonly in the lagoon.
Kingfisher	Kingfishers prefer areas of open water with overhanging vegetation. They are largely piscivorous but also take other small prey. Common species to the lagoon include the Pied Kingfisher.
Birds of prey	This group are not confined to a diet of fish, but also take other vertebrates and invertebrates. Species in this group include African Fish Eagle, Osprey and African Marsh Harrier.

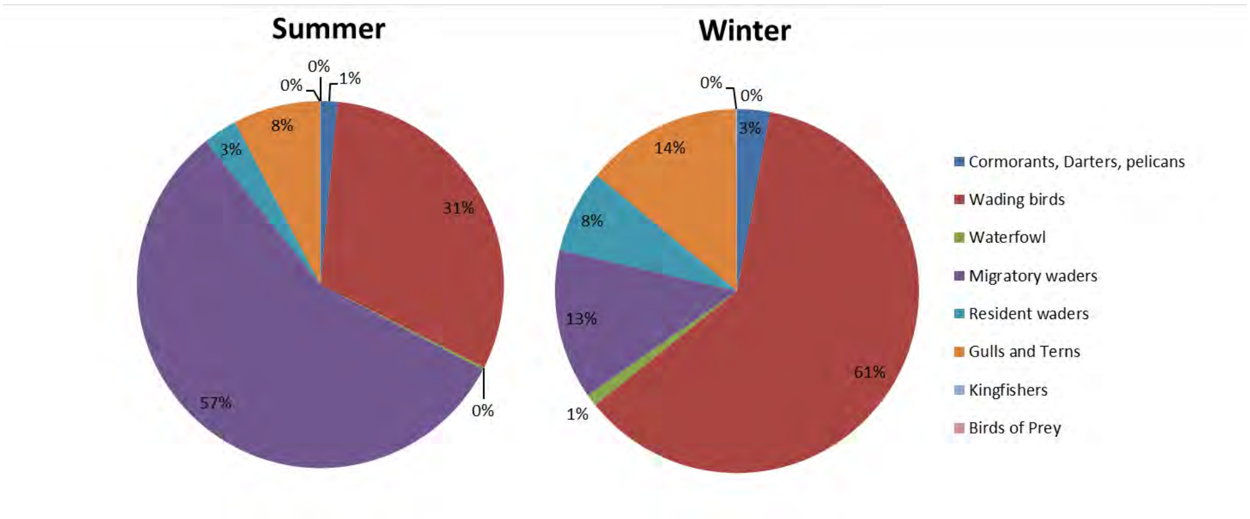


Figure 12.12. Present average numerical composition of the waterbirds on Langebaan Lagoon during summer (left) and winter (right) and winter (2014-2020) (Data source: CWAC data, Animal Demography Unit at the University of Cape Town).

Approximately 56 non-passerine waterbird species are regularly recorded at Langebaan Lagoon (species recorded more than 20% of the time). About two thirds of these waterbird species are waders, of which 20 species are regular migrants from the Palaearctic region of Eurasia. Important non-waders which utilise the system are Kelp and Hartlaub's Gulls, Greater Flamingo, Sacred Ibis and Common Tern. Resident waterbird species which utilise the rocky and sandy coastlines include the African Oystercatcher and the White-fronted Plover, both of which breed in the area. The waterbirds of Langebaan Lagoon are comprised of five different taxonomic orders (Table 12.2). A total of 117 bird species (i.e. including rare vagrants, terrestrial bird species, and passerines) have been recorded at Langebaan Lagoon as part of the CWAC surveys, of which 60 are South African non-passerine resident waterbird species and 26 are migrant waterbird species. The most species-rich order, the Charadriiformes, include a total of 31 wader species, three gull species and eight tern species (note the Antarctic Tern was recorded for the first time in August 2018) (Table 12.2). There are 14 resident wading bird species which include flamingos, herons, egrets, ibises and spoonbills.

Other birds that commonly occur on the lagoon include passerine species such as the Cape Wagtail *Motacilla capensis* and the Brown-throated Martin *Riparia paludicola*, as well as the Hadeda *Bostrychia hagedashn* (order Ciconiiformes). These species have been excluded from the waterbird categories due to their widespread distribution in non-coastal habitats.

Table 12.2. Taxonomic composition of non-passerine waterbirds in Langebaan Lagoon (excluding rare vagrants) (Data source: CWAC data, Animal Demography Unit at the University of Cape Town. Orders are listed in line with the 7th Edition of the Roberts Birds of South Africa).

Common groupings	Order	No. of SA resident species	No. of migrant species
Cormorants, darters, pelicans	Ciconiiformes (Cormorants, darters, pelicans)	8	
Wading birds	Ciconiiformes (Herons, egrets, ibises, spoonbill, flamingoes)	14	
Waterfowl	Ciconiiformes (Grebes)	2	
	Anseriformes (Ducks, geese)	8	
	Gruiformes (Rails, crakes, gallinules, coots)	5	
Waders	Charadriiformes	11	20
Gulls	Charadriiformes	3	
Terns	Charadriiformes	3	5
Kingfishers	Alcediniformes	3	
Birds of prey	Falconiformes	2	1
	Strigiformes	1	
	Total	**60**	**26**

12.3.3 Inter-annual variability in bird numbers

Irregular waterbird surveys were conducted at Langebaan Lagoon from 1934, but, due to the large size of the lagoon, these early counts were confined to small areas. It was not until 1975 that annual summer (January or February) and winter (June or July) surveys of the total population of waders at high tide, when waders congregate to roost on saltmarshes and sand spits, were conducted by members of the Western Cape Water Study Group (WCWSG) (Underhill 1987). The WCWSG monitored Langebaan continuously up to 1991, and since 1992 the Lagoon has been monitored bi-annually by the Co-ordinated Waterbird Counts (CWAC), organised by the Animal Demography Unity (ADU) at the University of Cape Town. These data sets provide the opportunity to examine the long-term trends in waterbird numbers at Langebaan Lagoon up to the present day.

Waterbird numbers on Langebaan Lagoon have declined dramatically since monitoring began in the 1970s. This is largely due to changes in the numbers of waders, which used to account for more than 90% of water bird numbers (Figure 12.13). In the 1970s and 1980s, migratory waders commonly numbered over 35 000 during summer, and over 10 000 in winter. Migratory wader bird numbers have since decreased significantly with only 2 483 individuals recorded in summer 2011. Since 2011, numbers have fluctuated around 5 000 (range: 2795-9539) individuals (Figure 12.13 and Figure 12.14). Since 2011, waders make up only 29-71% of summer water bird numbers (Figure 12.13). Total numbers of bird counted in the lagoon in the last two summers of 2019 and 2020 are now the lowest on record.

Migratory wader numbers crashed in summer 2009 and reached an all-time minimum in 2011 with just over 2 300 birds and have not recovered since then. The estimated population of 3634 migratory waders in summer 2020 is approximately 89% down from the pre-1990 average of ~33 000 birds. Drastic population declines in four species, including the Ruddy Turnstone, Red Knot, Grey Plover, and Curlew Sandpiper (Figure 12.15) typify this downward trend in summer migratory bird numbers. Most importantly, Curlew Sandpiper numbers have dropped from a pre-1990 average of just over 20 000 birds to 702 birds in 2020. Congruent with the overall temporal pattern described above, Curlew Sandpiper numbers reached a minimum in 2011 with only 413 individuals. Prior to 1990, this species accounted for almost two thirds of the total summer migratory wader numbers in the lagoon.

Resident wader numbers have fluctuated widely over time, reaching a near maximum only recently in the winter of 2013 with 1273 birds (Figure 12.14). This notwithstanding, resident bird numbers appear to be on a negative trajectory since 2007 with no signs of recovery in the last five years. The winter 2019 count was the lowest on record (Figure 12.14).

The reasons for these declines, particularly in migratory wader numbers, are diverse and poorly understood, but seem to be a combination of loss and degradation of their breeding sites as well as of their over-wintering grounds during their non-breeding period (Dias *et al.* 2006). Hunting of migratory waterbirds is a strong tradition in several European and North African countries and is thought to contribute towards global declines in migratory water birds (Bregnballe *et al.* 2006). The downward trend in migrant wader numbers seems to echo global trends in certain wader populations. Indeed, Ryan (2012) reports on similar declines in migrant waders throughout the Western Cape over the last three decades, irrespective of the protection status of the areas where counts were undertaken. This suggested that factors outside of the Western Cape were at least partially responsible for the observed trends and probably reflected global population declines (Ryan 2012). Conditions at Langebaan Lagoon could also have contributed to the decline in wader numbers over the last two decades. The most likely problems are that of siltation of the system reducing the area of suitable (e.g. muddy) intertidal foraging habitat, loss of seagrass beds with their associated invertebrate fauna (Pillay *et al.* 2010 see Chapter 8), and human disturbance, which has been shown to have a dramatic impact on bird numbers in other estuaries (Turpie & Love 2000). In 1985, Langebaan Lagoon was declared a National Park (West Coast National Park), and recreational activities such as boating, angling, kite surfing and swimming have since been controlled within the Lagoon through zonation. Nevertheless, some important feeding areas such as Oesterwal, lie within the zones that are highly utilised for recreation.

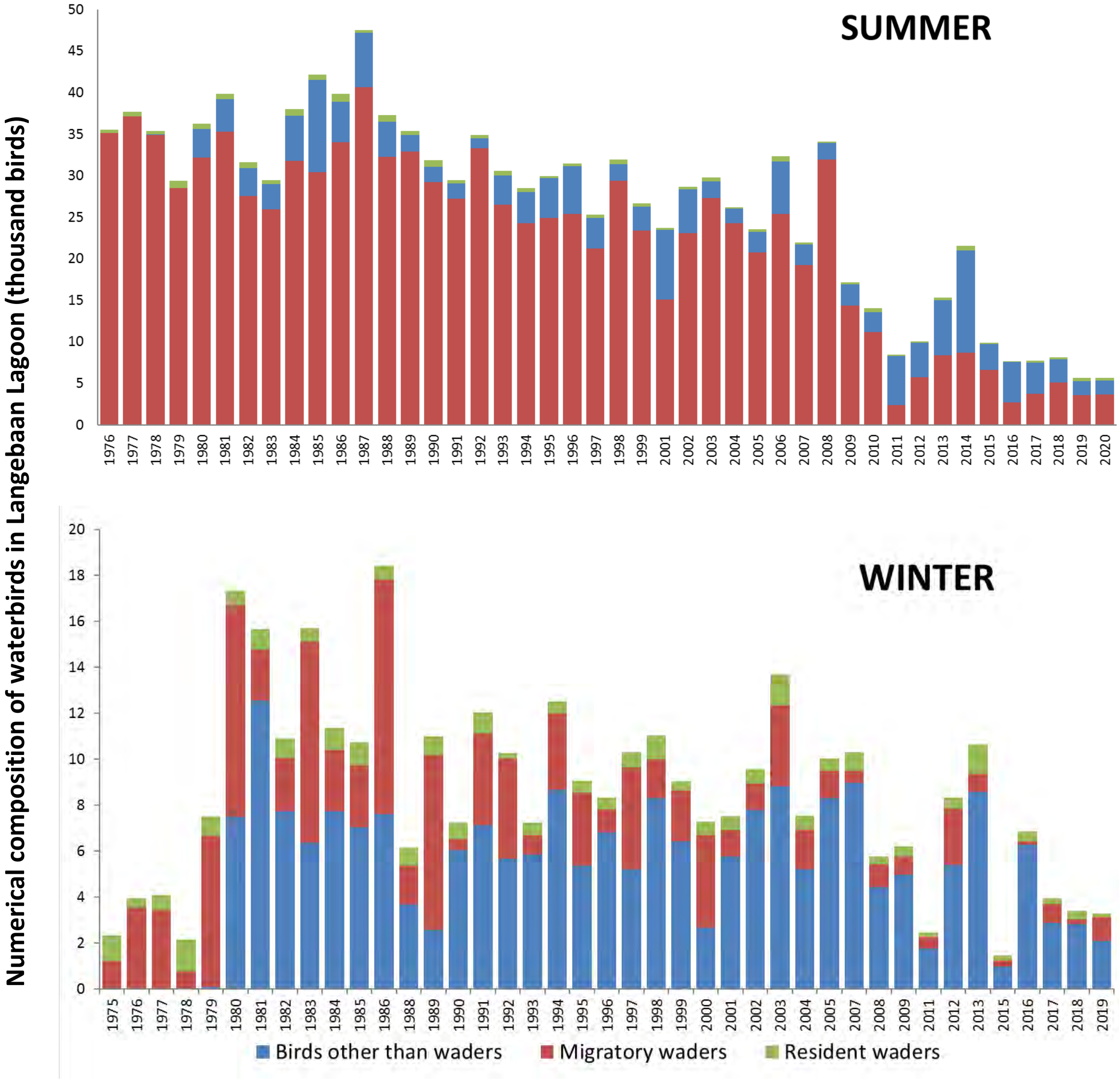
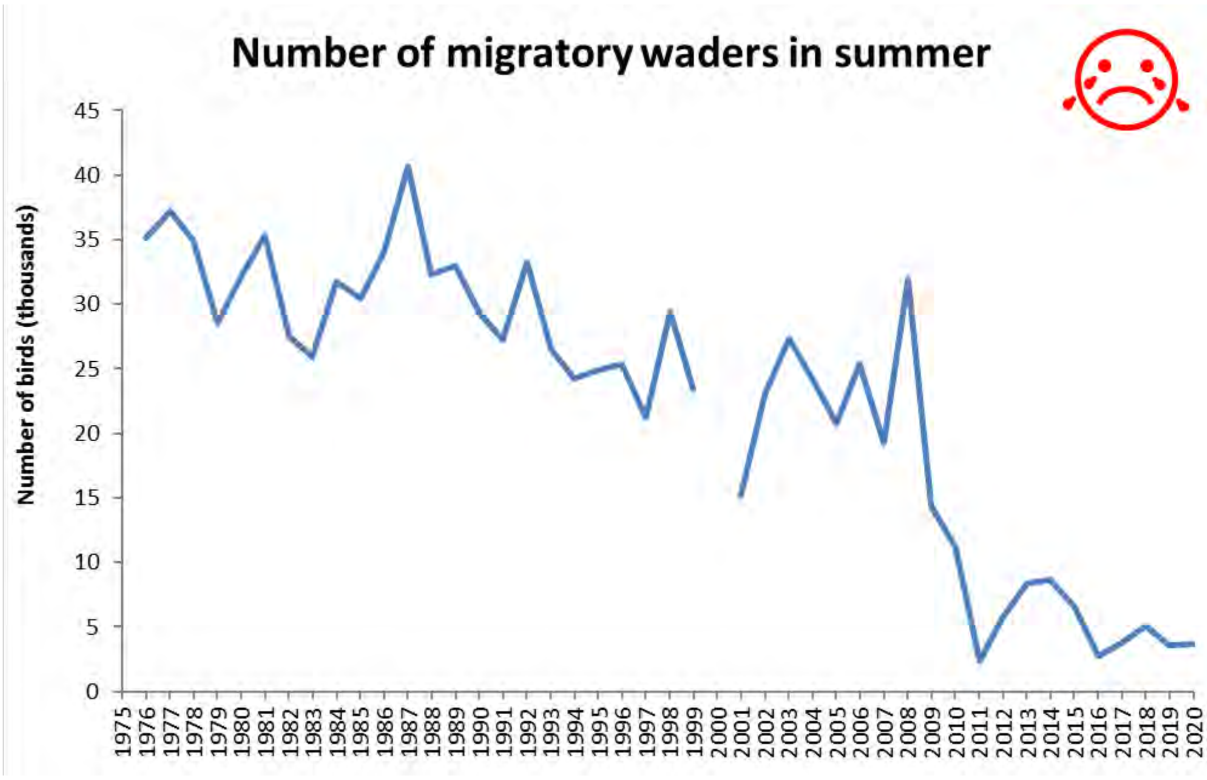


Figure 12.13. Long-term trend in the numerical composition of waterbirds in the Langebaan Lagoon during summer (top) and winter (bottom) (1976-summer 2019). Note that no data was collected in the summer of 1975, as well as in the winter of 1987, 2006, 2010, and 2014 (Data source: Coordinated Waterbird Count data, Animal Demography Unit at the University of Cape Town 2020).



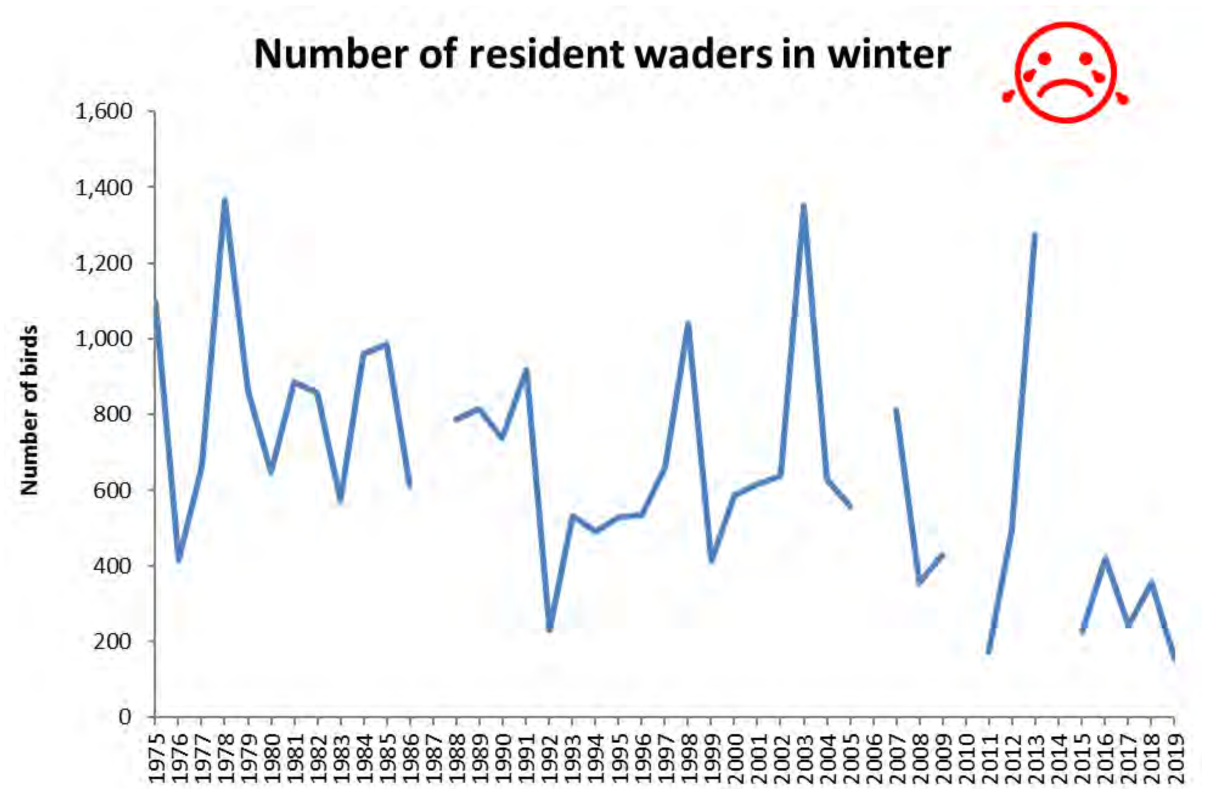


Figure 12.14. Long term trends in the numbers of summer migratory (top) and winter resident (bottom) waders on Langebaan Lagoon for the years 1976-summer 2020 (Data source: Coordinated Waterbird Count data, Animal Demography Unit at the University of Cape Town 2020).

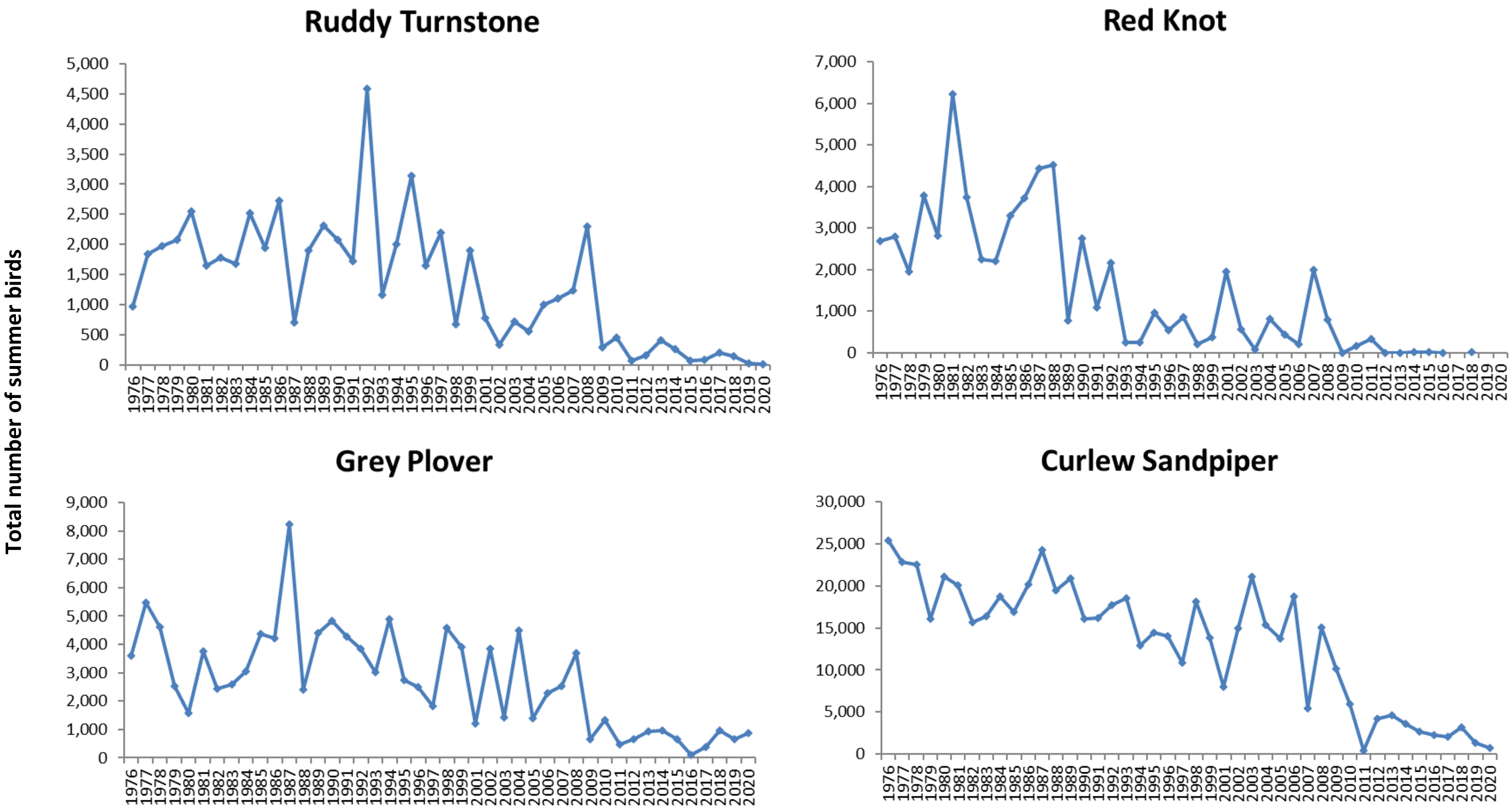


Figure 12.15. **Long-term trends in the numbers of four summer migratory waders (Ruddy Turnstone, Red Knot, Grey Plover and Curlew Sandpiper) on Langebaan Lagoon for the years 1976-2020. (Data source: Coordinated Waterbird Count data, Animal Demography Unit at the University of Cape Town 2020).**

12.4 Overall status of birds in Saldanha Bay and Langebaan Lagoon

Except for Bank Cormorants, the populations of the seabirds breeding on the islands of Saldanha Bay were on an increasing trajectory from the start of monitoring in the 1980s and 90s until around 2000. Factors that probably contributed to this include the reduction and eventual cessation of guano collecting in 1991, banning of egg collecting, increases in the biomass of small pelagic fish particularly sardines over this period, and in the case of the African Oystercatcher the increase in mussel biomass as a result of the spread of the Mediterranean mussel.

On the islands of Saldanha Bay, populations of all these species then started to decline, particularly, the penguins, gannets crowned cormorants and kelp gulls, which have declined to 7%, 40%, 23% and 22%, respectively of their populations at the turn of the century. Declines in the numbers of seabirds breeding on the Saldanha Bay Islands can be attributed to several causes. These include (1) emigration of birds to colonies further south and east along the South African coast in response to changes in the distribution and biomass of small pelagic fish stocks, (2) starvation as a result of a decline in the biomass of sardines nationally, and particularly along the west coast over the last decade, (3) competition for food with the small pelagic fisheries within the foraging range of affected bird species, (4) predation of eggs, young and fledglings by Great White Pelicans, Kelp Gulls and Cape Fur Seals, and (5) collapse of the West Coast Rock Lobster stock upon which Bank Cormorants feed.

However, because populations are so depressed, conditions at the islands in Saldanha, particularly predation by Cape Fur Seals, Pelicans and Kelp Gulls, have now become the major factors in driving current population decreases for many seabird species. Direct amelioration actions (*Pelican Watch*, problem seal culling) to decrease these impacts at the islands have had mixed results, with the former proving more effective than the latter. Cape Fur Seal and Kelp Gull predation continue to pose a major threat to seabird survival at the Saldanha Bay Island colonies.

Decreasing numbers of migrant waders utilising Langebaan Lagoon reflects a global trend, which can be attributed to loss of breeding habitat and hunting along their migration routes as well as human disturbance and habitat loss on their wintering grounds. In Langebaan Lagoon, drastic population declines in four species, including the Ruddy Turnstone, Red Knot, Grey Plover, and Curlew Sandpiper have signified this downward trend in summer migratory bird numbers. Most importantly, Curlew Sandpiper numbers have dropped from a pre-1990 average of just over 20 000 birds to 1 335 birds in 2019. Prior to 1990, this species accounted for almost two thirds of the total summer migratory wader numbers in the lagoon. The fact that numbers of resident waders are also declining, however, suggests that unfavourable conditions persisting in Langebaan Lagoon as a result of anthropogenic disturbance may be partly to blame. Resident wader numbers in the winter of 2019 dropped to the lowest recorded in the 40-year count record, a continuation of the declining trend over the last decade. Migratory wader counts in summer appear to be stabilizing at around 3 000-5 000 birds over the last five years, a fraction of their former abundance. It is highly recommended that the status of coastal and wading bird species continues to be monitored and that these data are used to inform and assesses the efficacy of management interventions aimed at halting the observed declines and supporting recovery of the regions birds.

12.5 SEALS

12.5.1 Cape Fur Seals

The Cape fur seal *Arctocephalus pusillus pusillus* is the only seal species that breeds in Southern Africa. Its range extends from the centre of Angola to the east coast of South Africa, with breeding colonies extending south from Baia dos Tigres on the southern border of Angola, through Namibia, down the west coast of South Africa and around to Algoa Bay in the Eastern Cape of South Africa (Figure 12.16). Historically (before 1900), it is likely that seals were present on most (if not all) islands off South Africa and Namibia, where they prefer to breed as they are protected from mainland predators. However, populations on many of the islands were significantly depleted or disappeared completely as a result of uncontrolled hunting, and human occupation of the islands for the collection of guano and other seabird related products (Kirkman *et al*. 2007). Subsequent to the ban on seal hunting, the Cape Fur seal population recovered, showing an almost 20-fold growth in numbers in the 20th century before stabilizing at about two million animals (Butterworth *et al.* 1995, Kirkman *et al*. 2007). In addition, the number of breeding colonies have increased since 1970 from 23 to 40 colonies (Kirkman *et al.* 2013). The overall population count has reportedly remained largely unchanged since 1993 and is estimated at 1.5-2.0 million, however, the distribution of these seals has been shown to vary in relation to prey distribution and shortages (Kirkman *et al*. 2007, Kirkman *et al*. 2013, Kirkman et al. 2019).

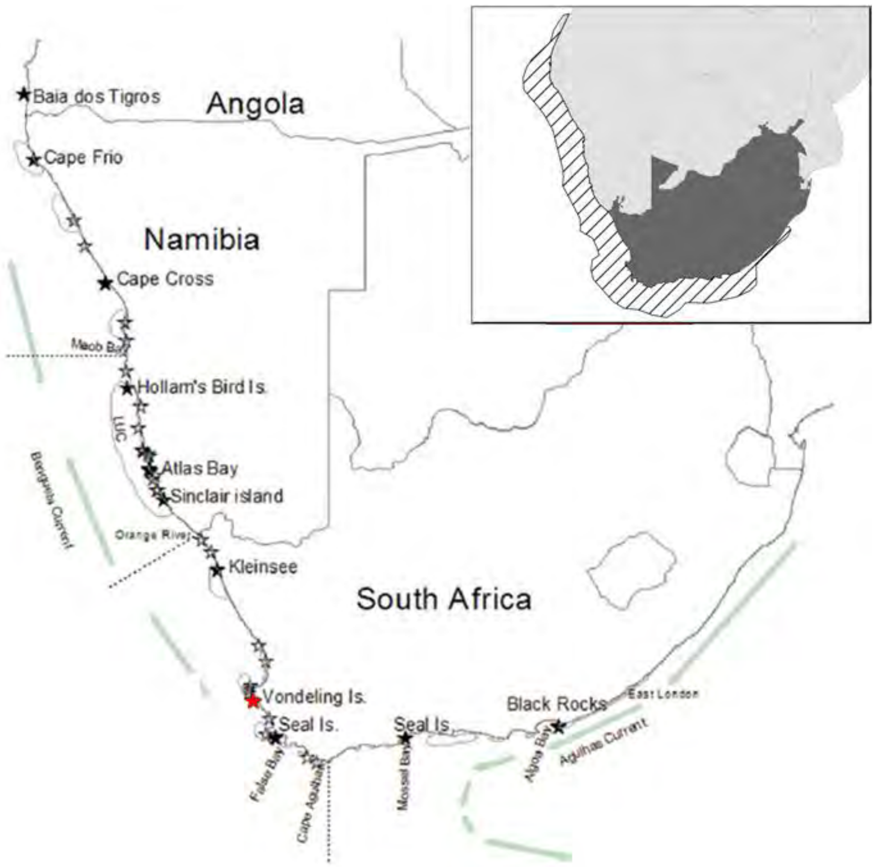


Figure 12.16. Distribution of selected Cape fur seal breeding colonies of Southern Africa with insert showing complete distribution of seal sitings in Southern Africa. The red star indicates the location of the recently established breeding colony at Vondeling Island just outside Saldanha Bay. (Adapted from Kirkman *et al.* 2013).

Seal populations along the west coast of South African exhibited a fairly stable distribution over the period 1976 to 2006, with the centre of the distribution of the breeding colonies remaining fixed within the region (Kirkman *et al.* 2013). However, more recently, it has been noted that there is a general shift southwards of St Helena Bay with new breeding colonies being established on Vondeling Island, at Cape Point and potentially on the south coast near Betty's Bay (DEFF, Mduduzi Seakamela, pers. comm. 2020). It is possible that this southward shift coincides with the eastern shift of small pelagic fish species which are a key food source for the seals (see Figure 12.2). Although seals historically would frequent the seabird islands (Jutten, Malgas and Vondeling) around Saldanha Bay, coming on land (hauling out) to rest or sun themselves, it is only since the turn of the century that a breeding colony has been established on Vondeling Island - south of the entrance to Saldanha Bay (Figure 12.17).

Figure 12.17. Map of Saldanha Bay showing the location of the Seabird Islands and highlighting the location of newly established seal breeding colony on Vondeling Island.

The Department of Environmental Affairs (DEA, now DEFF) monitors seal populations at 11 colonies in South Africa through aerial surveys which are undertaken to count pup numbers and hence to track seal population trends over time. Although these counts are normally only undertaken every three years, because it is a newly established breeding colony, aerial surveys have been conducted at Vondeling Island every year since 2006 (DEFF, Mduduzi Seakamela, *pers. comm*. 2020). Initially, the number of pups on the island increased dramatically up until 2010, thereafter (2010-2013), the rate of increase slowed and pup numbers on the island have fluctuated significantly in recent years - peaking at 23.4 thousand pups in 2014 and dropping to 16.7 thousand in 2018 (Figure 12.18). These fluctuations suggest that the island may have reached carrying capacity, with the annual changes linked to the availability of prey resources, and increases and decreases in the colony size mirroring those of sardines and anchovies on the West Coast (see Figure 12.2).

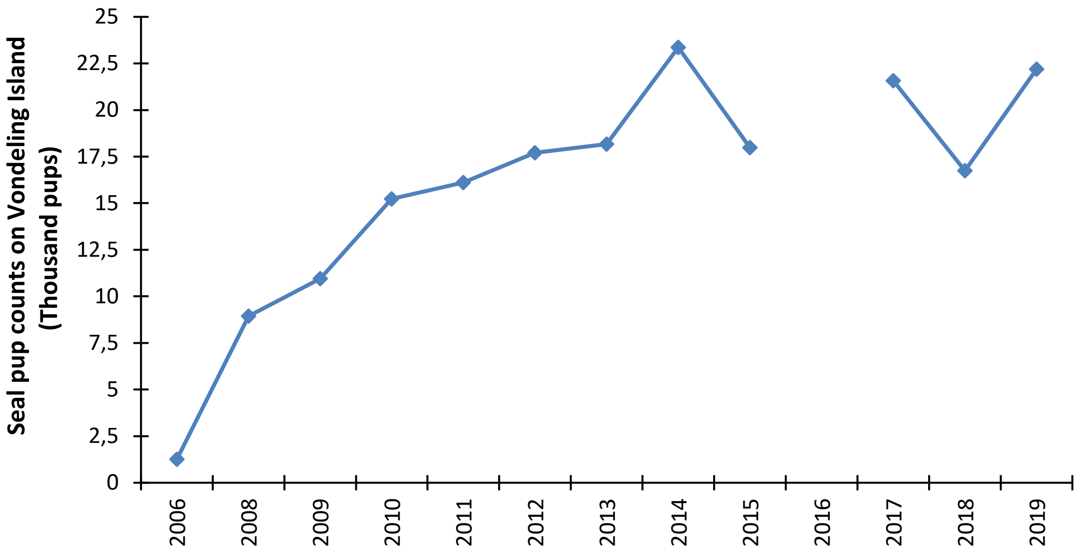


Figure 12.18. **Trends in seal pup counts collected during aerial surveys conducted at Vondeling Island, Saldanha Bay from 2006-2019. No data available for 2016 (Source: DEFF: Oceans and Coasts).**

Cape Fur Seals are amongst the largest marine top predators found in and around Saldanha Bay. They are opportunistic, generalist feeders that have been shown to benefit from human activities including utilisation of discards from fishing boats, or taking fish directly from fisherman (Wickens *et al.* 1992, Makhado *et al.* 2009). In addition, seals compete with seabirds, such as penguins and gannets, as well as with commercial fisheries, for small pelagic fish which form a key part of their diets (Crawford *et al.* 2011, De Moor & Butterworth 2015). Kirkman *et al*. (2013) suggested that the increasing numbers of seals on Vondeling island may lead to increased pressure to cull seals both from a fisheries perspective as well as to protect important seabird species on which seals are known to prey. In fact, some culling has been undertaken of seals off the west coast in recent years in an attempt to limit the mortality of seabirds that are of conservation importance. The culling of 'problem' seals seen killing Cape gannet fledglings at Malgas Island, located north west of the Vondeling breeding colony, resulted in a reduced mortality of gannet fledglings, however, seals learnt to avoid the boat used for culling, and the predation of seabirds around the island is ongoing (Makhado *et al.* 2009).

Concerns have also been raised that, with the increased number of seals along the shores surrounding Saldanha Bay and with the addition of finfish aquaculture in the Bay, seal numbers within the Bay will likely increase, along with the occurrence of problem seals. This is supported by the presence of groups of seals that can be seen hauling out onto the finfish cages currently located within the Bay (Figure 12.19). In a review paper, Callier *et al.* 2018, showed that numerous species of fish as well as benthic invertebrate species, such as crabs and starfish, are attracted to finfish cages. This attraction is a result of both the protection offered by the structure and the additional food in the form of fish feed and waste material surrounding these cages, as well as secondary attractions of predators drawn to smaller species accumulating around the cages. Globally, it has been shown that seals are attracted to, and may become more abundant in areas with fish farms, than areas without (Callier *et al.* 2018). Additionally, seals have been shown to consume mussels and associated benthic organisms in and around shellfish aquaculture farms and the fish and larger benthic invertebrates attracted to finfish cages (Roycroft *et al.* 2004, Callier *et al.* 2018). The floats or pontoons of the cages themselves also offer a solid structure above water on which the seals can haul out, thus automatically making them more visible in an area where they could previously not have been easily seen (Figure 12.19). It is standard practise to deter seals and reduce the impacts of seals attracted by finfish farms and associated fish, through the use of seal blinds and predator nets (Callier *et al.* 2018).

Figure 12.19. Image of Cape Fur Seals gathering on finfish cages within Saldanha Bay in April 2019. The cage on the left was empty, with no predator nets installed, while the cage on the right has stock in it and is surrounded by predator nets.

Studies investigating the predation of seabirds by seals at Malgas Island outside Saldanha Bay show that the 'problem' seals are restricted to sub-adult males which average less than 5 years old (Makhado *et al.* 2006, 2009). These individuals are not confined to the breeding colonies and are too young to be part of a breeding harem, and therefore tend to be more nomadic with inconsistent feeding areas. Conversely, data collected by DEFF using GPS tracks attached to female Cape Fur Seals tagged on Vondeling Island, indicates that these animals favour offshore feeding grounds and do not enter Saldanha Bay at all (Figure 12.20 and Mduduzi Seakamela *pers. comm.* 2020). It is likely that in order to maintain sufficient body fat and health to produce enough milk to support their pups, these females prefer the high quality food provided by the small pelagic fish species as opposed to irregular and limited food sources associated with fledgling birds and aquaculture. In addition, the females are more likely to be disturbed by human activities, selecting to avoid contact with humans to reduce the risk of conflict and therefore the risk of not returning to the breeding colony and their pups. This is supported by research showing that breeding and pupping harbour seals on the west coast of North America have been displaced by shellfish aquaculture activities (Becker *et al*. 2011). Therefore, although seals are likely attracted to the aquaculture sites within Saldanha Bay, chances are that their numbers will not continue to increase significantly as they are restricted to sub-adult males. Additionally, the carrying capacity of Vondeling Island appears to have been reached and the overall population within Southern Africa has remained stable over the last 30 years.

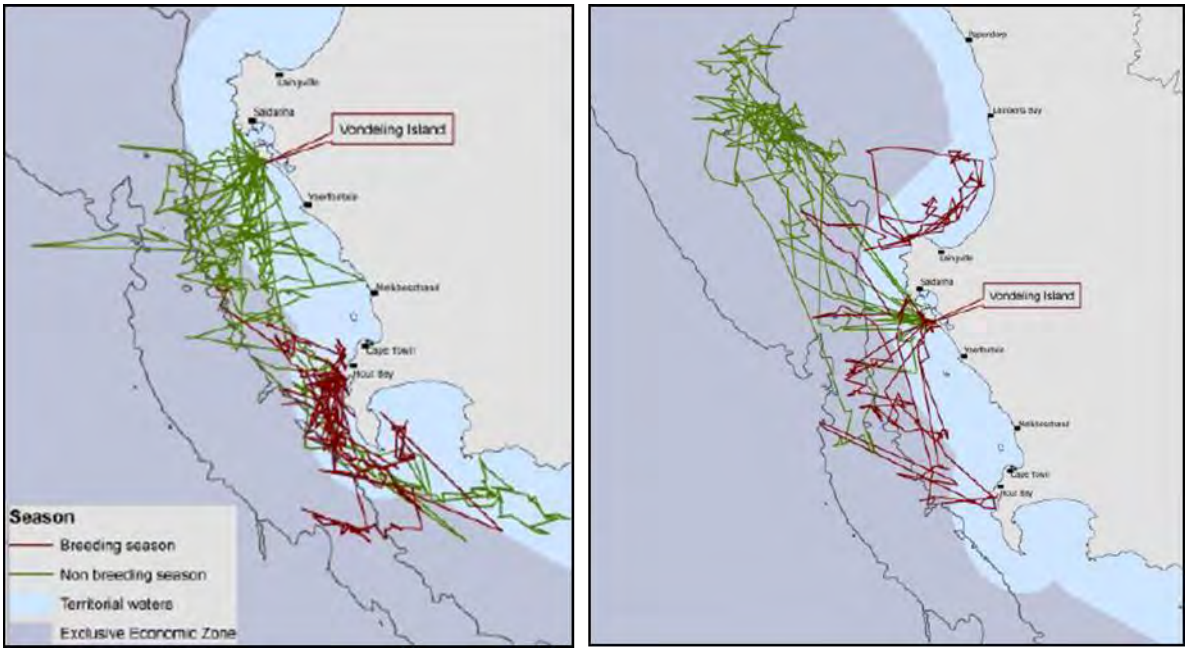


Figure 12.20. GPS tracks of female Cape fur seals tagged on Vondeling Island showing the routes travelled during the breeding and non-breeding season (Source: DEFF: Oceans and Coasts).

13 ALIEN AND INVASIVE SPECIES IN SALDANHA BAY AND LANGEBAAN LAGOON

13.1 Background information

13.1.1 General information and definitions

Human induced biological invasions have become a major cause for concern worldwide. The life history characteristics of the alien species, the ecological resilience of the affected area, the presence of suitable predators, biotic resistance and propagule pressure are only a few of the many factors that could determine whether an alien species becomes a successful invader or not. Biological invasions can negatively impact biodiversity and result in local or even global extinctions of indigenous species. Alien species invasions can furthermore have tangible and quantifiable socio-economic impacts and management of these species are thus vital. A pre-cautionary approach to prevent biological invasions is often considered the most efficient method of management and can include identifying and managing important pathways of introduction. If species are already present, however, regular monitoring and management protocols should be implemented to reduce the impacts of these invaders on the receiving environment and biota.

Until recently, alien species were recognised as invasive if they were found to have an environmental impact. However, much debate has occurred around the definition of environmental impacts in relation to an alien species (impact intensity, frequency, significance, positive versus negative, etc.) and consequently only few studies have attempted to determine whether an alien species can in fact be considered invasive (Robinson *et al.* 2016). The revised, internationally accepted approach recognises an alien species as invasive if the species has self-replacing populations over several generations and has expanded its range beyond the point of introduction (Wilson *et al.* 2009; Blackburn *et al.* 2011; Richardson *et al.* 2011). This approach has been proposed for South African marine invasion biology research going forward (Robinson *et al.* 2016).

13.1.2 Marine alien species in South Africa

A recent publication by Robinson *et al*. (2020) reviewing marine invasions in the South African context, reports a total of 95 alien marine species occurring in South African waters, of which 56 are considered invasive. An additional 39 (Mead *et al*. 2011a) species are currently regarded as cryptogenic (of unknown origin and potentially introduced), but very likely introduced to South Africa (Robinson *et al*. 2016).

The latest alien species list published by Robinson *et al*. (2016), was based on data collated up until 2014 and reported 36 alien and 53 invasive marine and estuarine species occurring in South African waters. The species list published five years before this by Mead *et al*. (2011) had identified 85 introduced species, without determining their status (i.e. alien versus invasive). The additions and changes since Mead *et al*. (2011) reported in Robinson *et al*. (2016), are considered below. Four species were removed from the 2011 alien species list. The polychaete *Hydroides elegans,* for example, was reassigned as cryptogenic (Çinar 2013), while the oyster *Ostrea edulis* and the urchin *Tetrapygus niger* were removed from the list as these populations no longer exist in mariculture dams previously surveyed, and were also absent from adjacent intertidal and subtidal areas of the coast (Mabin *et al.* 2015). Finally, the dune plant *Ammophila arenaria* was also removed as it is covered by the terrestrial alien plant list. Six species were added to the list, including the barnacle *Austrominius modestus* (Sandison 1950), the amphipod *Ericthonius difformis* (Peters *et al.* 2014), the crab *Pinnixa occidentalis* (Clark & Griffiths 2012), the polychaete *Polydora* cf. *websteri* (Simon 2015), and the red algae *Asparagopsis armata* and *A. taxiformis* (Bolton *et al.* 2011). Three name changes were also noted. First, the polychaete *Neanthes succinea,* which has been assigned to the genus *Alitta* (Read & Glasby 2017), and second, the hydrozoan *Moerisia maeotica,* which has been assigned to the genus *Odessia* (Schuchert 2017). Finally, the widespread tunicate *Ciona intestinalis* was found to represent two morphologically separate species, namely *C. intestinalis* and *C. robusta.* Of these two species, *C. robusta* is in fact the alien species that occurs in South Africa (Brunetti *et al.* 2015; Robinson *et al.* 2016).

With alien species being continually discovered and introduced, it comes as no surprise that, after 2014, there has been seven more additions to the list of known marine alien species in South Africa. Five of these are known to be introduced to Saldanha Bay. In 2015 and 2017, respectively, two species both native to Chile, were reported from Saldanha Bay. These were the South American sunstar *Heliaster helianthus* and the Chilean stone crab *Homalaspis plana* (Peters & Robinson 2018). It should be noted, however, that only one individual of each was found, despite extensive intertidal and subtidal surveys in 2018. Nevertheless, these species have been added to the alien species list of South Africa and should also be added to a watchlist, as even if these were isolated individuals recorded previously, reintroduction is probable. The presence of the barnacle *Perforatus perforatus* (Biccard & Griffiths *pers. comm*. 2017)*,* the Japanese skeleton shrimp *Caprella mutica* (Peters & Robinson 2017) and the South West African porcelain crab *Porcellana africana,* have been confirmed in Saldanha Bay and Langebaan Lagoon. This crab was previously incorrectly identified as the European porcelain crab, *P. platycheles* (Griffiths *et al.* 2018). The study by Griffiths *et al.* (2018) further revealed that *P. africana*, first discovered in 2012 on Schaapen Island (*Prof. George Branch pers. obs.*), has now been confirmed to occur in the study area, i.e. Saldanha Bay. The semi-terrestrial isopod, *Ligia exotica* was reported as alien to South Africa by Greenan *et al.* (2018) and officially reported only from Durban harbour by Barnard (1932). The native range of this species is unknown, although it lives in the upper intertidal and supralittoral zone (Roman 1977), where it grazes on diatoms and encrusting algae (Schultz 1977). Greenan *et al.* (2018) found that there is a lot of cryptic diversity within *Ligia* isopods in Southern Africa and recommends taxonomic re-evaluation of this group. The Maritime earwig *Anisolabis maritima,* was first discovered in 2015 on the east coast at Port Shepstone (Griffiths 2018). It is, however, thought to have been introduced more than a century ago (1880–1902). Although this species is technically an insect and not marine, it does occur and feed in the upper littoral zone (Bennett 1904; Griffiths 2018).

13.1.3 Marine alien and invasive species in Saldanha Bay

With the addition of the five new species, at least 67 alien species are present along the west coast (Robinson et al. 2020), 29 of which are now confirmed to be present in Saldanha Bay and/or Langebaan Lagoon. All of these, except *H. helianthus*, *H. plana*, *P. perforatus* and the previously reported anemone *Sagartia ornata*, are considered invasive (Table 16.3). Of the 39 cryptogenic species, 19 are likely to be found in Saldanha Bay and/or Langebaan Lagoon and six have already been identified from the Bay. Comprehensive genetic analyses are urgently required to determine the definite status of the cryptogenic species (Griffiths *et al.* 2008). With new species being discovered every year and with the status of existing species changing regularly as new information becomes available, the list of alien species present in South Africa is by no means complete and should be regularly reviewed.

Other noteworthy invasive species commonly found in the study area include the invasive Mediterranean mussel *Mytilus galloprovincialis* (Hockey & van Erkom Schurink 1992), the Western pea crab *Pinnixa occidentalis* (Clark & Griffiths 2012), the barnacle *Balanus glandula* (Laird & Griffiths 2008) and the Pacific South American mussel *Semimytilus algosus* (de Greef *et al.* 2013). Interestingly, the abundance of *M. galloprovincialis* on rocky shores in Saldanha Bay has been decreasing in the last few years (Section 13.3.8). The reason behind this decline is, at this stage, still unclear, although this trend has been noted for *M. galloprovincialis* in the past (Hanekom & Nel 2002; Robinson *et al.* 2007a) and might be due to numerous factors, including predation pressure and environmental factors. *B. glandula,* on the other hand, has shown a steady increase in abundance over time at most sites where it has been recorded in the Bay, and remains one of the more abundant species on the mid-shore in Saldanha Bay (Section 13.3.2). *P. occidentalis* is now well established and has slowly been increasing in number over time in both Big Bay and Small Bay (Section 13.3.5). It was also present again this year in Langebaan Lagoon. It may be in the process of expanding into more exposed and deeper habitats outside of the Bay, including Danger Bay. This notable increase in abundance of this crab raises concern and highlights the need for management actions.

13.1.4 Potential vectors of introduction to South Africa

Of the 95 marine alien species recognised in South Africa, a notable 91% of these introductions have been associated with shipping activities such as ballast water discharge and hull fouling. In addition, 50 of the reported alien species are confined to sheltered areas such as harbours. These findings emphasise the importance of shipping as a pathway of introduction (Robinson *et al*. 2020) and highlight the need for implementing more efficient protocols to monitor vessels entering South African harbours, the treatment of hull fouling before entering and the regular monitoring of harbours for alien species. As ballast water tends to be loaded in sheltered harbours, the species that are transported originate from these habitats and therefore have trouble adapting to South Africa's exposed coast. This might explain the low number of introduced species that have established along the coast (Griffiths *et al.* 2008) and the high number found in sheltered bays such as Saldanha.

Both land and sea-based mariculture have also been identified as important vectors for the introduction of alien marine species. For example, it has been shown that translocated oysters act as vectors for marine alien species all over the world. Oysters attach to rocks, walls and other surfaces and are colonised by fouling organisms, which can be exported into other countries on the oyster spat. Alien species imported on oyster shells may have significant ecological impacts in areas where they establish (Haupt *et al.* 2010).

13.1.5 Patterns related to invasion success of alien species

Marine scientists are trying to find new ways to predict invasion success and the spread of established invasive species to facilitate early detection and to inform focused management interventions. One method has been exploring the link between biological characteristics of invasive species in relation to their observed success. For example, invasive species are often more efficient at utilising resources when compared to native species. Recent research on the invasive *M. galloprovincialis* shows that the success of this species on the west coast of South Africa could be explained, at least partially, by the species' capability to utilise food resources more efficiently when compared to other mussel species (invasive *S. algosus* and native *Aulacomya atra*) (Alexander *et al.* 2015). Alexander *et al.* (2015) showed that *M. galloprovincialis* was the most efficient consumer of algal cells at colder temperatures when the resource was presented in both low and high starting densities. These results may explain the observed success of this species on the west coast of South Africa relative to the new invader *S. algosus,* which, based on the results of this study, is predicted to become established along the south coast of South Africa. This is linked to the finding that algae consumption was more efficient in warmer water. Conversely, results from a recent study exploring the relationship between invasion success of predatory crabs and their biological traits, could not identify any specific traits associated with their success. This was due to an unexpected gap in the basic biological knowledge for even this conspicuous alien group (Swart *et al.* 2018). Such a lack of knowledge makes it difficult to draw conclusions between traits and invasion success and emphasizes the need and importance of basic knowledge of species to explore drivers behind invasion success.

13.2 Study approach to monitor alien species within Saldanha Bay

Sampling and monitoring of alien species within Saldanha Bay, forms part of the State of the Bay monitoring programme. Data for this initiative is obtained from both the Benthic Macrofauna monitoring survey initiated in 2004 (Chapter 9) and the Rocky Intertidal monitoring survey initiated in 2005 (Chapter 10).

The locations of the eight rocky shore sampling sites include the Dive School and the Jetty (both sites situated along the northern shore in Small Bay), Schaapen Island East and West (located at the entrance to Langebaan Lagoon and considered sheltered sites), Marcus Island, the Iron Ore Terminal, and Lynch Point (all located in Big Bay), and North Bay (situated in Outer Bay at the entrance to Saldanha Bay; Figure 13.1) . The very sheltered sites include the Dive School and Jetty; the sheltered sites include Schaapen Island East and West; the semi-exposed sites include the Iron Ore Terminal and Lynch Point and the exposed sites include Marcus Island and North Bay (Figure 10.2 in Chapter 10.2.1).

Figure 13.1. The location of the eight rocky shore study sites in Saldanha Bay are indicated by red dots.

Sampling of benthic macrofauna is undertaken in four areas in the Bay every year since 2004 and includes Big Bay, Small Bay, Langebaan Lagoon and Danger Bay. In total, eleven sites within Big Bay, 30 within Small Bay, 18 within Langebaan Lagoon and 14 within Danger bay (Figure 13.2), have been sampled and monitored since 2004.

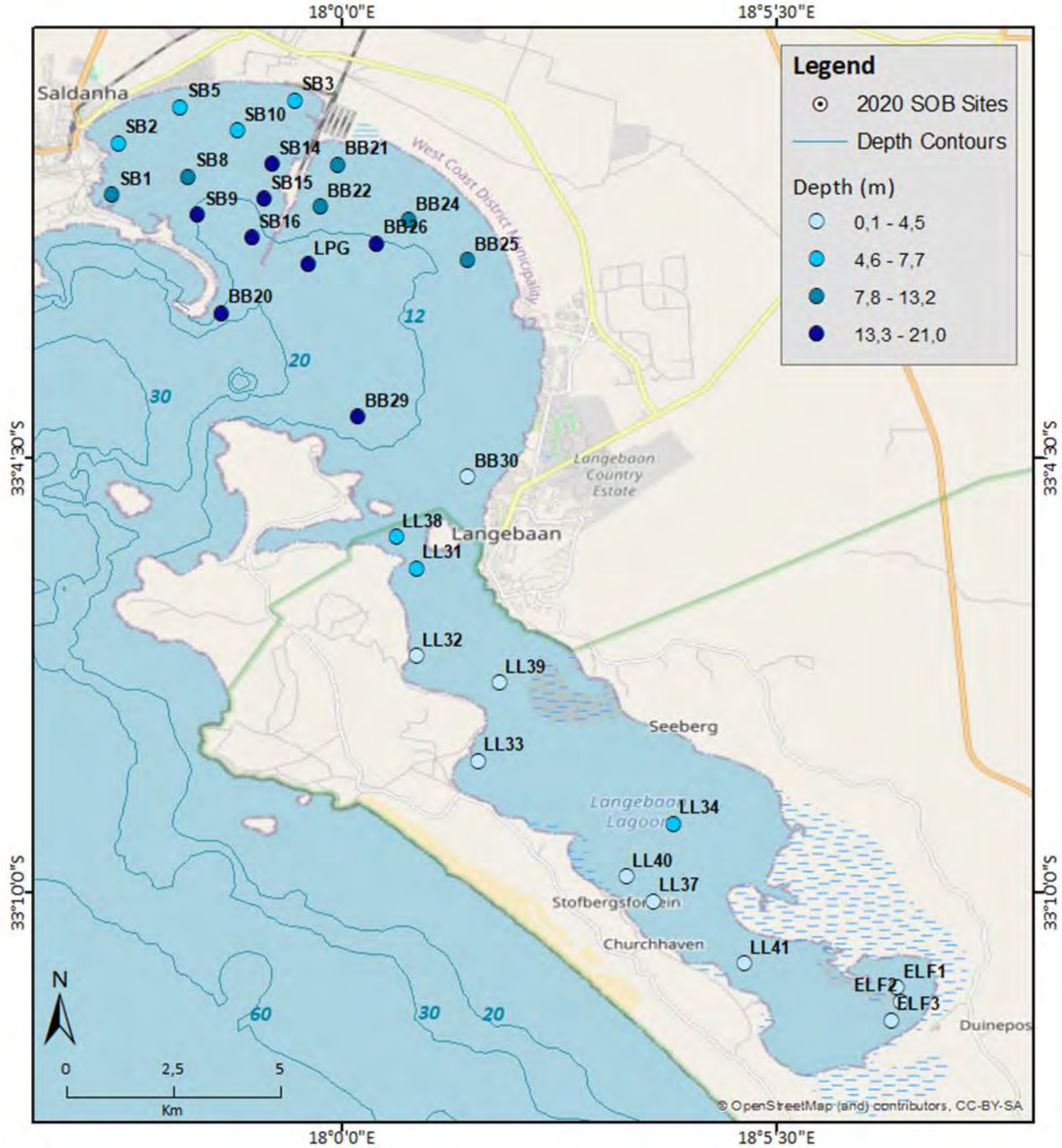


Figure 13.2. **Sampling sites and respective depth ranges (m) in Saldanha Bay and Langebaan Lagoon for 2020 macrofauna sampling.**

13.3 Alien and invasive species confirmed in Saldanha Bay and/ or Langebaan Lagoon

Below follows information for all the known alien and invasive species occurring in Saldanha Bay and/or Langebaan Lagoon. (Information on the cryptogenic barnacle, *Amphibalanus amphitrite amphitrite* is also presented below). Additional information for these and other cryptogenic species are presented in the Appendix (Table 16.3). Species occurrence is listed as either confirmed or likely (not confirmed from Saldanha Bay, but inferred from the regional distribution of the species). In addition to the general information presented below, abundance and biomass data is also presented for three of the invasive species present in the Bay, i.e. the Acorn barnacle *Balanus glandula*, the European mussel *Mytilus galloprovincialis* and the Western pea crab *Pinnixa occidentalis.* Data for both *B. glandula* and *M. galloprovincialis* were obtained from the rocky shore surveys, whereas data for *P. occidentalis* was obtained from the benthic macrofauna survey. Future surveys in Saldanha Bay will be used to confirm the presence of unconfirmed listed species and to ascertain if any additional or newly arrived introduced species are present.

13.3.1 Shell worm *Boccardia proboscidea*

Boccardia proboscidea is a small (20 mm long) tube-dwelling worm found in shallow sand-lined burrows on the surfaces of oysters, abalone and other shellfish (Figure 13.3). It occurs naturally on the Pacific coast of North America and Japan (Simon *et al.* 2009; Picker & Griffiths 2011). In South Africa, it is known to occur on a number of oyster and abalone farms and has also recently been recorded in Saldanha Bay outside aquaculture facilities (Haupt *et al.* 2010). Oceanographic modelling and population genetic approaches revealed that *B. proboscidea* has the potential to disperse and establish itself along the South African coast, despite biogeographic boundaries. Although this is partly attributed to its broad thermal tolerance and flexible reproductive strategy, it is believed that anthropogenic movement will be the primary factor governing its spread and establishment in southern Africa (David *et al.* 2016).

Figure 13.3. Shell worm *Boccardia proboscidea* (Photo: Geoffrey Read)

13.3.2 Acorn barnacle *Balanus glandula*

Figure 13.4. Acorn barnacle *Balanus glandula* (Photo: Prof. C.L. Griffiths)

The presence of *Balanus glandula*, which originates from the Pacific coast of North America, was first recognized in South Africa in 2008 (Laird & Griffiths 2008; Simon-Blecher *et al.* 2008). It seems, however, that this species has been in South Africa since at least the early 1990s. It is has been observed to be the most abundant intertidal barnacle in Saldanha Bay and indeed along much of the southern west coast (Laird & Griffiths 2008; Figure 13.4). The species has recently been reported to have spread east, past Cape Point, which was until now, thought of as a biogeographical barrier (Robinson *et al.* 2015). Recent research shows that when compared to the indigenous barnacle species *Notomegabalanus algicola, B. glandula* more efficiently takes up algae regardless of water temperature or cell concentration. Furthermore, warmer conditions on the south coast enhanced the uptake of algae cells, which could result in *B. glandula* spreading further east than currently observed (Pope *et al.* 2016). The State of the Bay surveys and studies conducted elsewhere suggest that this species competes directly with other alien species for space on the shore.

B. glandula looks very similar to the indigenous species, *Chthamalus dentatus,* which may account for the fact that it went undetected for so long (Figure 13.5). *B. glandula* has reportedly displaced populations of the indigenous and formerly abundant *C. dentatus* species which is now very rare on South African west coast shores (Laird & Griffiths 2008). *B. glandula* was first correctly identified in the State of the Bay surveys in Saldanha Bay in 2010 and as such, data for this species is only presented from 2010. It is very likely, however, that it had been present during the baseline surveys in 2005 and 2008-2009, but overlooked due to it being incorrectly identified as the indigenous barnacle species.

Data from the State of the Bay surveys suggest that *B. glandula* has, over the past decade, been most abundant on the mid shore, having the highest densities at the semi-exposed rocky shores sites (Iron Ore terminal and Lynch Point), followed by the exposed rocky shore sites (North Bay and Marcus Island). It was very abundant when it was first detected in 2010, reaching a maximum of 74% at the Iron ore terminal in 2011. This species also reached maximum densities at Lynch Point in 2010 (27%), North Bay in 2019 (21%) and Marcus Island in 2014 (14%). Although there was a notable increase in the percentage cover of *B. glandula* at both North Bay and Lynch Point from 2018 to 2019, densities decreased again in 2020. This is, however, not surprising as the abundance of this species has been fluctuating over time, most noticeably at the Iron ore terminal, Lynch Point and North Bay. *B. glandula* populations at the sheltered Schaapen Island East and West sites are generally sparse, only reaching a percentage cover of 0-1.5%. One exception includes 2011, where densities on Schaapen East reached 20%.

Except for the slight increase in percentage cover at the Jetty and Dive School in 2020, recent surveys revealed that densities of *B. glandula* have decreased at all the other sites. The abundance at the Iron Ore Jetty had decreased to such an extent that no barnacles were recorded there during the 2020 survey. The highest percentage cover recorded in 2020 was at Lynch Point where 9% cover was recorded. The total percentage cover for all sites in 2020 was the lowest ever recorded for the past decade. This trend may reflect a new ecosystem equilibrium as predator numbers have probably responded to the new food source and now exert some control over the abundance of this alien species. In addition, the maintenance and indeed any increases in population size for *B. glandula* have been reported to depend upon high densities of this species, or new propagules from elsewhere, as this barnacle cannot self-fertilise (Kado 2003). Small populations of this species are thus likely to further decline without sufficient influx of propagules. The presence of species such as limpets are also known to decrease population sizes by dislodging newly settled barnacles (Miller & Carefoot 1989). In light of recent findings that found no significant impact of *B. glandula* on community structure at Marcus Island (Sadchatheeswaran *et al.* 2018), *B. glandula* is not believed to have any significant impacts on communities in Saldanha Bay, and even less so with smaller population sizes. However, as densities could increase again in the future, monitoring of this species should continue, especially since it is one of the more abundant species on the shore in Saldanha Bay.

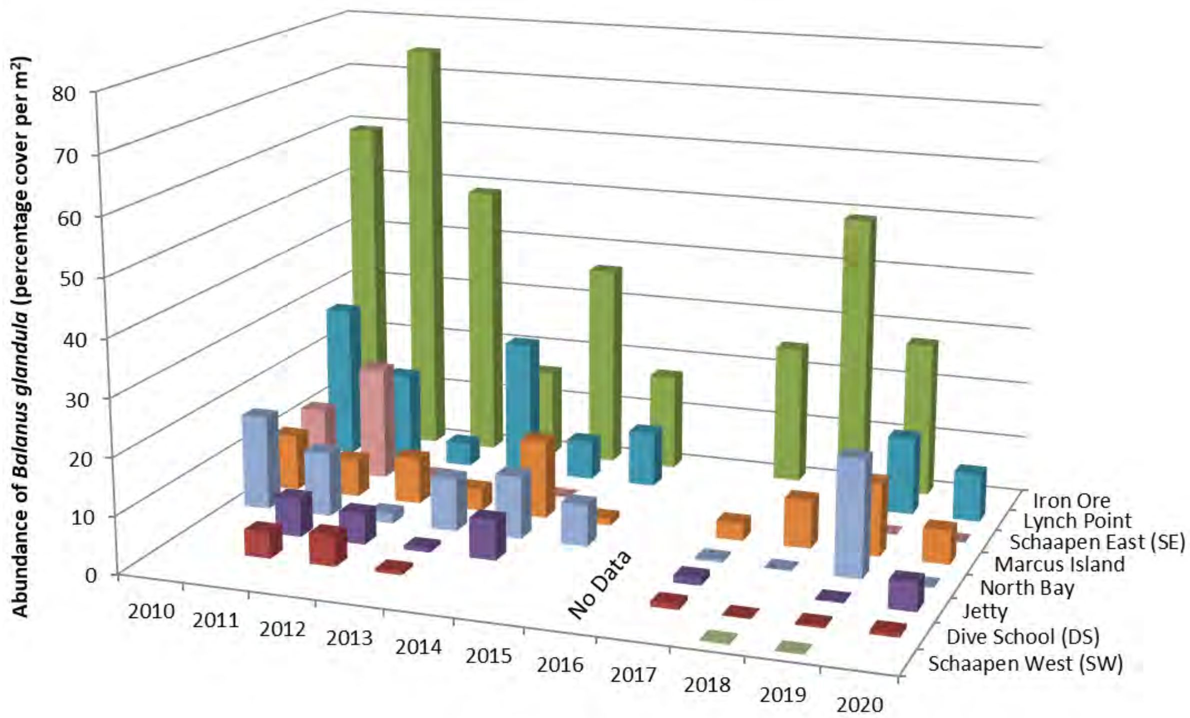


Figure 13.5. Changes in the abundance (% cover) of the acorn barnacle *Balanus glandula* at eight rocky intertidal sites on the mid shore in Saldanha Bay over the period 2010-2020. Data are shown as an average of percentage cover on the mid shore. No samples were collected 2016. See Figure 13.1 for locations of these sampling stations.

13.3.3 Hitchhiker amphipod *Jassa slatteryi*

Jassa slatteryi is a small (9 mm) inconspicuous amphipod that constructs tubes of soft mud or crawls around on seaweeds, hydroids and other marine growth (Colan 1990; Picker & Griffiths 2011; Figure 13.6). It is common on piers, buoys and other structures in Saldanha Bay. It was first collected in South Africa in the 1950s, but incorrectly classified as the South African species, *J. falcata*. It was only after the genus was revised, that it was correctly identified as *J. slatteryi* and classified as alien in South Africa. It is suspected that it was introduced directly via ship fouling or ballast water transfer from its native habitat in Pacific North America or another invaded temperate harbour. It is small and occurs in high densities and is probably a valuable food source for fish and other predators.

Figure 13.6. Hitchhiker amphipod *Jassa slatteryi* (Photo: Prof. C.L. Griffiths).

13.3.4 European shore crab *Carcinus maenas*

Carcinus maenas is a native European crab species that has been introduced on both the Atlantic and Pacific coasts of North America, in Australia, Argentina, Japan and South Africa (Carlton & Cohen 2003) (Figure 13.7). It is typically restricted to sheltered, coastal sites and appears thus far to have been unable to establish on the open wave-swept coastline in South Africa (Hampton & Griffiths 2007). In South Africa, it was first collected from Table Bay Docks in 1983 and later in Hout Bay Harbour. It has established dense populations in both harbours where it has reportedly decimated shellfish populations (Robinson *et al.* 2005). Surveys in Saldanha Bay have not turned up any live specimens of this species to date, but a single dead specimen was picked up by Robinson *et al.* (2004) in Small Bay at the Small Craft Harbour. Due to a lack of specimens, it is unlikely that there is an extant population in Saldanha Bay at present.

Figure 13.7. European shore crab *Carcinus maenas.* (Photo: Prof. C.L. Griffiths).

13.3.5 Western pea crab *Pinnixa occidentalis*

The Western Pea crab *Pinnixa occidentalis* (Figure 13.8) is a small Pinnotherid crab with a carapace width of <2.5 cm (Zmarzly 1992). It was originally described from California by MJ Rathbun in 1893 (Rathbun 1894), although its native range is presently reported to include North America's entire west coast, from Alaska to Mexico (Zmarzly 1992). *P. occidentalis* is a deep-water species and prefers depths ranging from 11-319m (Ocean Biogeographic Information System 2011). These crabs can be free-living, although they are commonly known to live in symbiosis with other animals. This usually includes living inside bivalves or ascidians or the burrows of polychaetes or spoon worms (echiurans; McDermott 2009). Mutualistic relationships, such as these, are known to facilitate the establishment and spread of introduced species.

Figure 13.8 Western pea-crab *Pinnixa occidentalis* (Photo: Anchor Environmental Consultants).

The Western pea crab appears to have established itself in Saldanha Bay in the period between 1999 (at which time no specimens were recorded in a comprehensive set of samples from Saldanha Bay) and 2004, when it was recorded at three sites in Big Bay and at one site in Small Bay (detection rate of 30% and 6% respectively). At his stage, it was still listed as unidentified. The vector of introduction is unclear, although this crab was potentially introduced via ship fouling or ballast water (Clark and Griffiths 2012). It was only identified as *P. occidentalis* in the collections from the Saldanha Bay State of the Bay surveys in 2010 (Anchor Environmental Consultants 2011). The rate of detection of this crab (i.e. the percentage of sites where the species was detected), has been determined within each of the four sites that are sampled each year (i.e. Big Bay, Small Bay, Langebaan Lagoon and Danger Bay; Figure 13.9). Furthermore, the average abundance and biomass of the Western pea crab was analysed within Big Bay and Small Bay (Figure 13.10) in addition to determining the abundance and biomass at selected sites within the latter two locations (Figure 13.11).

The rate of detection of *P. occidentalis* was the greatest in Big Bay, followed by Small Bay, fluctuating between 40-67% and 18-40%, respectively, over the last decade. Although the rate at which the pea crab has been detected within Small Bay has slightly decreased from the 2019 to the 2020 sampling, it remains unchanged in Big Bay at 67% (Figure 13.9). In 2019, the average abundance of *P. occidentalis* in Big Bay, peaked at 200 crabs/m^2, although the abundance has decreased again in 2020. These fluctuations in average abundance over time are not statistically significant (as indicated by the overlapping standard error bars), although this might be attributed to the highly variable abundance at the different sites within Big Bay (Figure 13.10).

During the 2019 and 2020 sampling, crabs were present at six of the nine Big Bay sites sampled, unlike most other years where it was only recorded at four or fewer sites. Although it was found to be most prevalent to the east of the iron ore and multi-purpose terminals in 2019 (site BB25), exceeding 1500 individuals/m^2, very few crabs were detected at this site during the 2020 sampling (Figure 13.11). The 2020 sampling did, however, reveal the crab to be most prevalent at one of the deep-water sites (BB29) where it exceeded 400 individuals/m^2. This is not surprising since this species is known to prefer deeper water (>10 m) within its native range in North America (Ocean Biogeographic Information System 2011).

P. occidentalis was present at only four of the 12 Small Bay sites sampled during the 2020 sampling campaign. In 2019, this crab demonstrated a noticeable threefold increase since the 2018 and exponential increase since the 2004 sampling at one of the sites (SB9) east of the terminal at the entrance of Small Bay. Here, it exceeded 670 individuals/m^2 (Figure 13.11). Although the 2020 sampling campaign revealed that the abundance of the pea crab at the aforementioned site had decreased to 245 individuals/m^2, it remained most prevalent at this site. The abundance of *P. occidentalis* increased at site SB1, located close to the Sea Harvest quay. The reason for the increase of this species at this site could be due to the higher biological pollution, contaminants and disturbance at this site due to the presence of the Sea Harvest processing factory. Large volumes of brackish effluent water containing suspended solids, organic material such as protein and oils, ammonia, nitrogen and phosphate are discharged in the surrounding area. Alien species are typically more tolerant of areas with brackish water, low water quality and pollution, which generally support lower natural species diversity. As such, alien species are better able to establish and thrive in these areas (Williamson & Fitter 1996; Streftaris *et al.* 2005). Indeed, the macrofauna study reported that species diversity was lower in Small Bay near the Sea Harvest discharge pipe (Chapter 9). Pea crabs generally feed on detritus and waste from their host, while many can also filter feed. The increased abundance of the pea crab at this site might thus be a combination of the reduced water quality, low natural species diversity and potential food source.

The crab remained present only in low densities (SB16 and SB14; <25 individuals/m^2) or absent (SB15) at sites close to the ore terminal (Figure 13.11). Fluctuations in the abundance and dominance of the pea crab at certain sites within Big and Small Bay is currently unknown, although fluctuations in the abundance of macrofauna community structure in sheltered inlets are not uncommon and could be attributed to a variety of physical and environmental factors (Nichols 1985), including upwelling currents, rate of sedimentation, substrate particle size, amount of organic matter present and variations in temperature, dissolved oxygen and salinity (Guevara-Fletcher *et al.* 2011) which in turn also fluctuate in time and space.

P. occidentalis has been sporadically present in low densities (four individuals/m^2) at three sites within Langebaan Lagoon (LL40, LL33 and LL31) over the past decade (Figure 13.9). Only the latter site, located close to the mouth of the lagoon, was found to support populations of this crab for three consecutive years in a row (2018-2020). Sampling over these three years revealed densities of 4, 20 and 60 individuals/m^2, respectively, suggesting that the crab population is increasing in size at this particular site in Langebaan Lagoon. This crab appears, to our knowledge, not to be spreading or increasing in density in the rest of Langebaan Lagoon. Its presence and increasing abundance at site LL31 could be explained by the fact that this site is slightly deeper (3.5-6 m) than the rest of the shallow lagoon (1.2-3.5 m) and is also located adjacent to the Big Bay sites known to support populations of this crab. Danger Bay was only sampled during the 2014 and 2015 sampling campaigns. No crabs were detected in the first survey, although it was found at one of the 13 sites sampled during the second survey, albeit in small numbers (eight individuals/m^2).

In conclusion, these data suggest that *P. occidentalis* is now well established in both Big Bay and Small Bay and potentially increasing in abundance within Langebaan Lagoon. The average abundance and biomass of the alien crab in Big and Small Bay in addition to the abundance and biomass at selected sites within the latter two locations, has fluctuated over time, with no upward, downward or recruitment trend being apparent thus far. The status of this crab within Danger Bay is currently not confirmed and more sampling effort is thus needed at this site. Although no conclusive trend in the spread and site preference of this species is apparent, the pea crab does seem to flourish in deeper water habitats and is generally absent from or occurs in low densities in locations close to the iron ore and multi-purpose terminals. The impact of *P. occidentalis* on, and its role in the benthic community of Saldanha Bay, remains undetermined. This gap in knowledge highlights the need for more in-depth studies and potential management action.

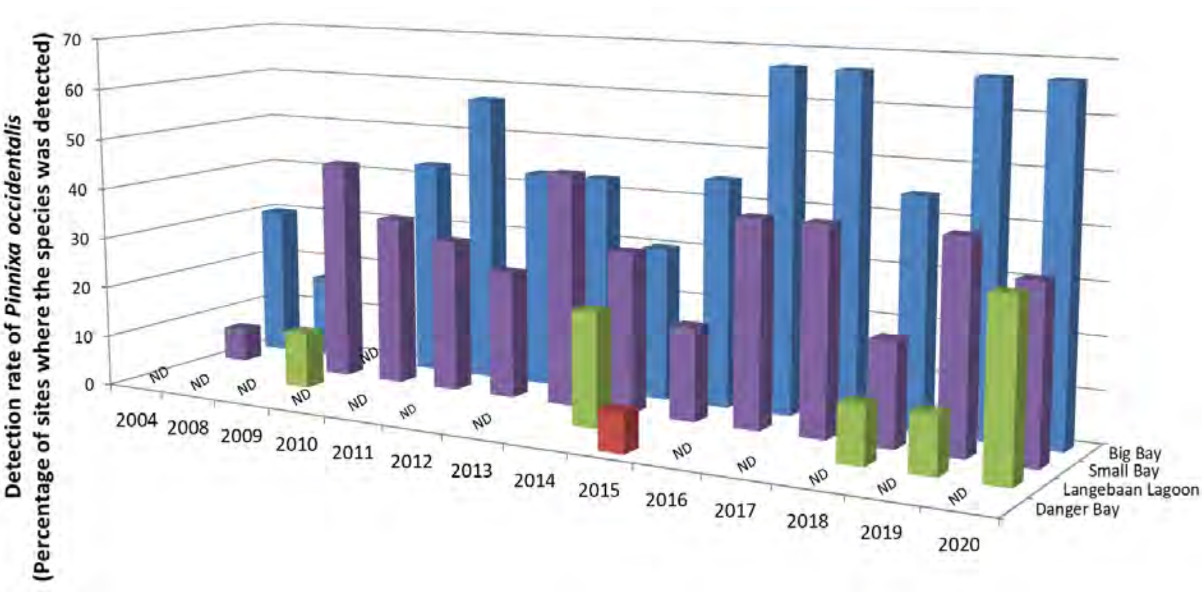


Figure 13.9. **The detection rate (percentage of sites where the species was detected) of the Western Pea crab *Pinnixa occidentalis* in Big Bay, Small Bay, Langebaan Lagoon and Danger Bay in the period 2004-2020. Note that Langebaan Lagoon and Danger Bay were first sampled in 2004 and 2014, respectively. No data were collected in the period 2005-2007. 'ND' denotes that no data was collected in the region for that year.**

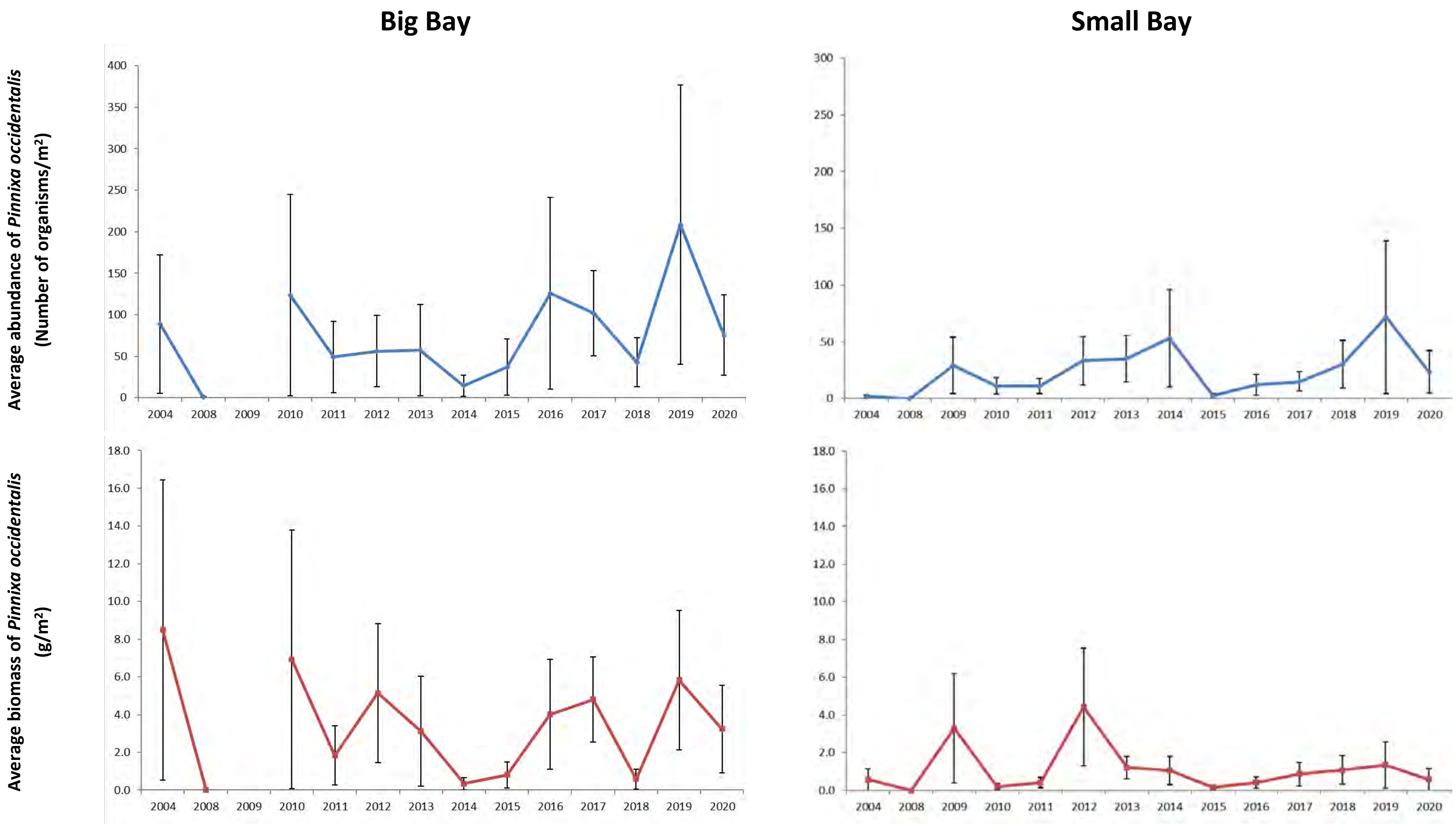
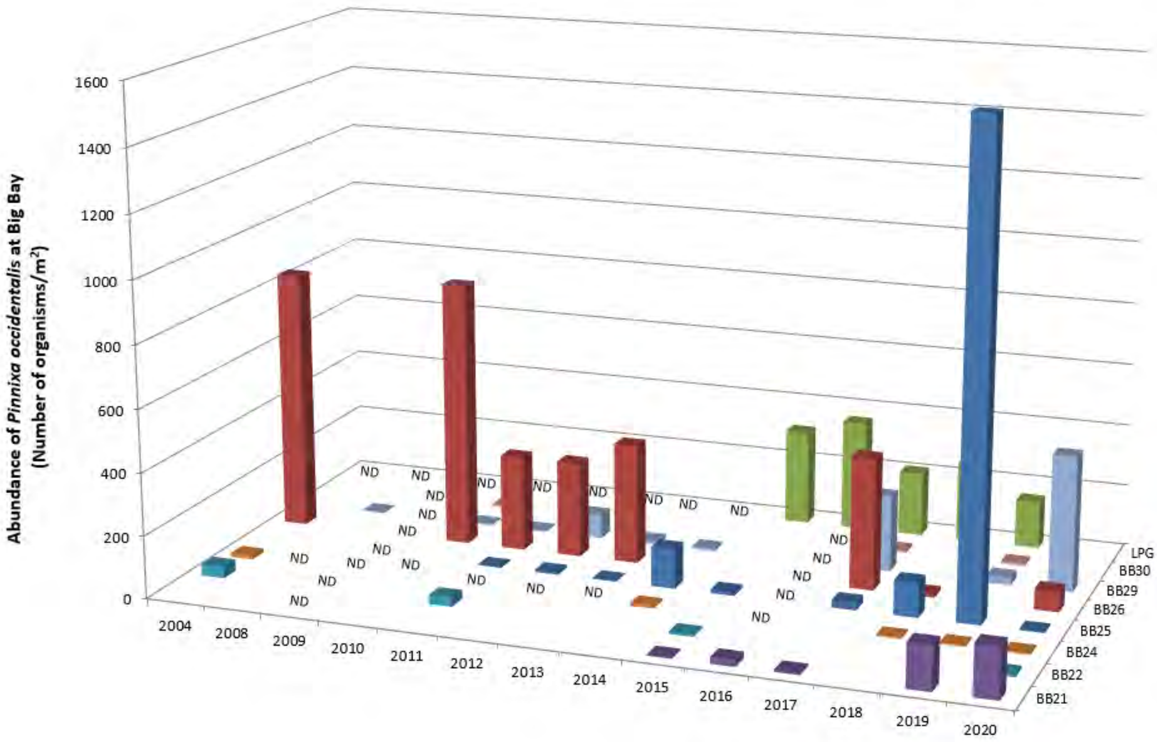


Figure 13.10. Average abundance (top) and biomass (bottom) of the Western Pea crab *Pinnixa occidentalis* in Saldanha Bay, Big Bay (left) and Small Bay (right) from 2004-2020. No data were collected from 2005-2007 and no data were collected for Big Bay in 2009.



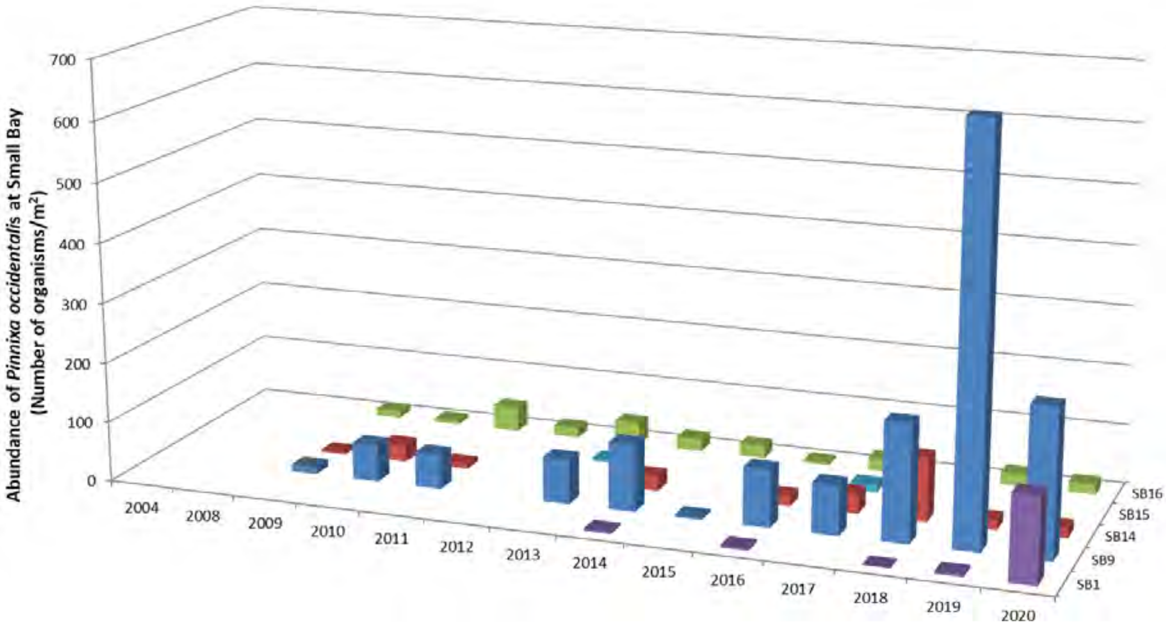


Figure 13.11. Abundance of the Western Pea crab *Pinnixa occidentalis* in Saldanha Bay at selected sites in Big Bay (top) and Small Bay (bottom) from 2004-2020. No data were collected from 2005-2007. 'ND' denotes that no data was collected in the region for that year.

13.3.6 Lagoon snail *Littorina saxatilis*

Littorina saxatilis was first recorded in South Africa in 1974 (Day 1974), and the only known populations are those in Langebaan and Knysna lagoons (Hughes 1979; Robinson *et al.* 2004; Picker & Griffiths 2011). In its home range in the North Atlantic, this species occurs in crevices on rocky shores (Gibson *et al.* 2001), but in South Africa, it is restricted to sheltered salt marshes and lagoons, where it occurs on the stems of the cord grass *Spartina maritima* (Hughes 1979; Figure 13.12). It occurs only in the upper reaches of Langebaan Lagoon, between Bottelary and Churchhaven, and has not spread further afield than this in at least 20 years (Robison *et al.* 2004). It is not considered to be a major threat to the Lagoon or Bay ecosystems.

Figure 13.12. Lagoon snail *Littorina saxatilis* (Photo: Prof. C.L. Griffiths)

13.3.7 Pacific oyster *Crassostrea gigas*

Crassostrea gigas is native to Japan and South East Asia (Figure 13.13). This oyster was introduced to the Knysna Estuary in South Africa in the 1950s with the intention to farm. The species has been farmed in the Kowie and Swartkops estuaries as well as at three marine locations, Algoa Bay, Saldanha Bay and Alexander Bay (Robinson *et al.* 2005).

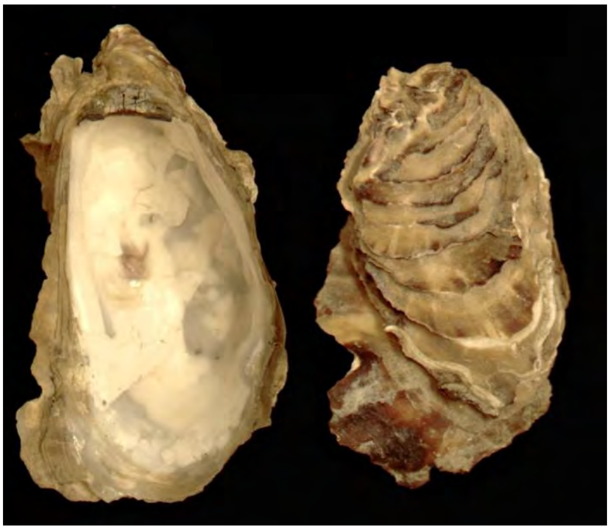


Figure 13.13. Pacific oyster Crassostrea gigas (Photo: Serge Gofas. Source: Marinespecies.org

Initially, the species was never considered an invasive threat as the oysters seemed unable to reproduce and settle successfully under the local environmental conditions which differ from its native habitat. However, the farmed populations have spread within the country. Using DNA sequencing, Robinson *et al.* (2005b) confirmed the presence of three naturalised populations of *C. gigas* in South Africa (specifically the Breede, Knysna and Goukou estuaries). The highest densities of individuals were found in the Breede Estuary (approximately 184 000 individuals). *Crassostrea gigas* were originally farmed in the Seafarm dam east of the iron ore terminal and are now farmed in baskets moored in the Bay. Feral populations of this oyster have established inside the dam, which is open to Big Bay. However, self-sustaining populations outside of the dam have not been noted to date.

Translocated oysters act as vectors for the introduction of marine alien species all over the world. Oysters attach to rocks, walls and other surfaces and are exposed to colonisation by fouling organisms, which can be transported to other countries. Marine alien species imported on oyster shells may have significant ecological impacts in areas where they establish (Haupt *et al.* 2010) (e.g. Disc lamp shell *Discinisca tenuis* – Section 13.3.10).

13.3.8 European mussel *Mytilus galloprovincialis*

Mytilus galloprovincialis was first detected in South Africa (in Saldanha Bay) in 1979 (Mead *et al.* 2011b) but was only confirmed in 1984 (Grant *et al.* 1984; Grant & Cherry 1985; Figure 13.14). At this stage, the population was already widespread in the country, being the most abundant mussel species on rocky shores between Cape Point and Lüderitz. This species has subsequently extended its distribution range as far as East London (Robinson *et al.* 2005). It is suspected that *M. galloprovincialis* was most likely first introduced to the country between the late 1970s and early 1980s (Griffiths *et al.* 1992) and the reason for the late detection is due to the fact that it is easily confused with the indigenous black mussel, *Choromytilus meridionalis. Mytilus* is, however, easily distinguished by the trained eye, being fatter, and having a pitted residual ridge. The preferred habitat of the two species also differs with *M. galloprovincialis* occurring higher on the shore and away from sand-inundated sites. The alien mussel is commercially cultured in Saldanha Bay and elsewhere, and is widely exploited by recreational and subsistence fishers (Robinson *et al.* 2005 & 2007a).

Figure 13.14. European mussel *Mytilus galloprovincialis.* (Photo: Prof. C.L. Griffiths.)

In Europe, *M. galloprovincialis* is known to form dense subtidal beds directly on sandy bottoms (Ceccherelli & Rossi 1984). While this species is typically found on exposed rocky shores in the intertidal in southern Africa, it was recently reported to grow subtidally on the kelp, *Ecklonia maxima* in False Bay. Implications could include negatively impacting kelp ecosystems and a decrease in useable kelp for economic purposes. It could also uproot kelp beds which in turn could lead to the spread of native and invasive species (Lindberg *et al*. 2020). *Mytilus* began establishing dense intertidal beds on the sandy centre banks of Langebaan Lagoon in the mid-1990s (Hockey & van Erkom Schurink 1992; Hanekom & Nel 2002; Robinson & Griffiths 2002; Robinson *et al.* 2007a), with biomass peaking at an estimated eight tonnes in 1998 (Robinson & Griffiths 2002). The population subsequently crashed, decreasing in size by 88% by early 2001 (Hanekom & Nel 2002) and had died off completely by mid-2001, leaving only empty shells and anoxic sand (Robinson *et al.* 2007a). The reason for the die off is still unclear and impacts on the macrobenthic infauna on the banks was evident for at least six months after most of the dead mussel shells had been removed by SANParks in late 2001.

At Marcus Island, a comparison of intertidal communities pre- and post-invasion of *M. galloprovincialis* (1980 vs 2001), *S. algosus* and *B. glandula* (1980 vs 2012), demonstrated that the indigenous mussels *C. meridionalis* disappeared by 2012, and *A. atra* decreased in abundance. While recruits of the limpet *Scutellastra granularis* initially benefited from the arrival of *M. galloprovincialis*, adults were adversely affected (Sadchatheeswaran *et al.* 2015). Although *M. galloprovincialis* did not alter habitat complexity when replacing *C. meridionalis* on the low shore at Marcus Island, it was responsible for diminishing habitat complexity when replacing *A. atra* on the mid shore. Here, *M. galloprovincialis* was responsible for a reduction in abundance and diversity of other species (Sadchatheeswaran *et al.* 2015). *Mytilus* has also been shown to overshadow interannual and seasonal changes of intertidal rocky shore communities on Marcus Island and was found to be the most important factor influencing community composition (Sadchatheeswaran *et al.* 2018). As a result, *M. galloprovincialis* is considered to be an alien ecosystem engineer within the intertidal zone of the South African west coast (Sadchatheeswaran *et al.* 2015).

Saldanha Bay State of the Bay Surveys recording *Mytilus galloprovincialis* were initiated in 2005 and included at eight rocky shore sites. These sites included the Dive School, the Jetty, Marcus Island, the Iron Ore Terminal, Lynch Point, North Bay and Schaapen Island East and West (Figure 13.15). No sampling was done in 2006, 2007 or 2016. Surveys indicated that *M. galloprovincialis* was generally predominant at exposed rocky shore sites (i.e. Marcus Island, Lynch Point, North Bay, Iron ore terminal), reaching higher densities than at the more sheltered sites (Dive School, Jetty and Schaapen Island East and West). Over the past 15 years, this mussel's percentage cover per m^2 at these sheltered sites, has never exceeded 5%, and was frequently recorded as 0% at Schaapen Island East and West. The density at the jetty slightly increased in 2020 and is higher than for previous years. Observations revealed that this invasive mussel is by far the most dominant faunal species on the rocky shores and can cover up to 100% of the available space across substantial portions of the shore. It reaches its highest densities low down on the shore, in areas exposed to high wave action.

Since the start of the surveys and up until 2015, *M. galloprovincialis* increased steeply in abundance at the exposed sites, reaching maximum densities at Marcus Island in 2008 (39%), Lynch Point (58%) and North Bay (23%) in 2012, and at the iron ore terminal in 2015 (40%) . Recent surveys have revealed that densities of *M. galloprovincialis* has since decreased in certain areas, and it is now present in lower densities at Marcus Island, North Bay and the Iron Ore terminal than in previous years. The abundance of this mussel at the latter site had decreased to such an extent that no mussels were recorded there during the 2020 survey. The reason for the decrease in abundance of *M. galloprovincialis* is not clear, although such a sudden, unexplained decrease in abundance has been noted for this species in the past and might be a result of numerous factors. It is, for example, hypothesised that this decreasing trend may reflect a new ecosystem equilibrium as predator numbers have probably responded to *Mytilus* as a new food source and now exert more control on the abundance of this invasive species. In addition, marine alien mussels such as *M. galloprovincialis* have been found to become parasitised by endolithic bacteria which can cause shell damage, reduced attachment ability and death. Such parasitism has been recorded in South Africa and could explain the reduction in the abundance of this mussel species (Zardi *et al.* 2009), although this is only speculation and has not been confirmed in Saldanha Bay. High trace metal concentrations could also affect the survival of these mussels. A lab study on *Mytilus edulis* found that even low concentrations of environmental trace metals such Pb, Mn and Cd can affect reproduction and survival (Fraser *et al*. 2017). High concentrations of trace metals have been reported in the mussels in Saldanha Bay in 2020 and in the past, sometimes even exceeding the recommended levels for foodstuffs (Chapter 6). These high concentrations could be responsible for a decrease in mussels in Saldanha Bay. Furthermore, the decrease at the iron ore site can be linked to the high levels of lead due to the export of lead ore from the multipurpose quay.

In light of the fact that *M. galloprovincialis* occurs subtidally in its native range and has recently been reported subtidally elsewhere in South Africa, the presence of this species should be monitored subtidally within Saldanha Bay.

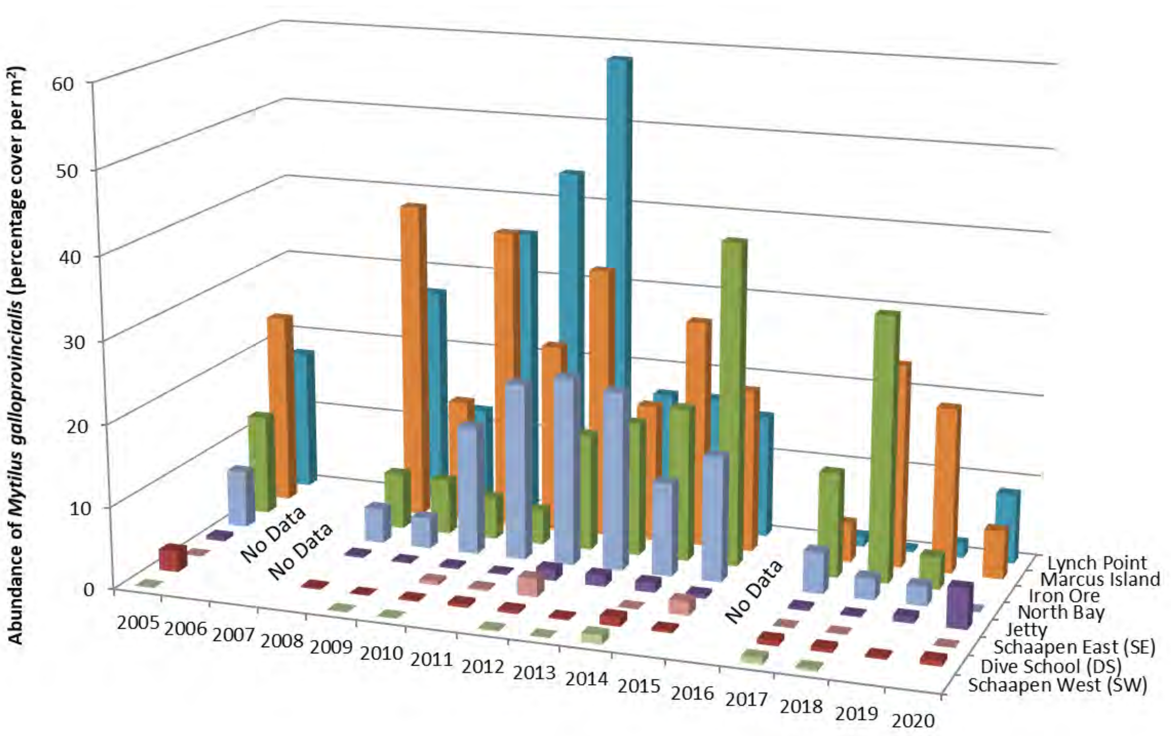


Figure 13.15. Changes in the abundance (% cover) of the Mediterranean mussel *Mytilus galloprovincialis* at eight rocky intertidal sites in Saldanha Bay over the period 2005-2020. Data are shown as an average of percentage cover on the mid and low shore. No samples were collected in 2006, 2007 and 2016. See Figure 13.1 for locations of these sampling stations.

13.3.9 Pacific South American mussel *Semimytilus algosus*

Figure 13.16. Pacific South American mussel *Semimytilus algosus* (Photo: Prof. C.L. Griffiths)

The Pacific South American mussel *Semimytilus algosus* is a small (up to 50 mm) elongated, relatively flat and smooth brown mussel, with a green tinged shell (Figure 13.16). This species originates from Chile and has been long known from Namibia (since the 1930s, Kensley & Penrith 1970) but was only recently (2010) found in South Africa. It is unknown when *S. algosus* arrived in South Africa. It is likely that it was transported southwards from Namibia either by shipping as a new invasion or through range expansion from the Namibian population (de Greef *et al.* 2013). The present geographic range of *S. algosus* in South Africa extends some 500 km, from Bloubergstrand in the south to Groenriviersmond in the north (de Greef *et al.* 2013).

At exposed sites, this species proliferates on the low shore, numerically dominating intertidal organism abundance, with extremely dense beds constituting a significant proportion of the total intertidal biomass (de Greef *et al.* 2013). A recent study addressed the lack of information available on subtidal mussel communities (Skein *et al.* 2018a). This study confirmed that *S. algosus* has a strong preference for wave exposed shores and forms dense intertidal beds along the west coast (de Greef *et al.* 2013; Skein *et al.* 2018a). However, the subtidal surveys found that *S. algosus* represents the dominant species at sheltered sites on the west coast and forms equally dense beds at exposed sites when compared to the indigenous species (Skein *et al.* 2018a). These findings may explain why *S. algosus* has previously been found on mussel farm ropes in Saldanha Bay. A subtidal reef survey to confirm or deny the presence and spread of *S. algosus* could provide more information on adaptability of this species. Furthermore, subtidal specimens were generally found to be considerably larger than those found in the intertidal zone. *S. algosus* attained maximum sizes larger than 120 mm, in contrast to 54 mm in the intertidal (Skein *et al.* 2018a). It has been proposed that mussels smaller than 60 mm could be vulnerable to predators which could potentially have implications for the future spread and success of the species (de Greef *et al.* 2013).

In a laboratory study conducted by Alexander *et al.* (2015) algae consumption exhibited by *S. algosus* was shown to be more efficient in warm water than in cold water, which led to the conclusion that this species may have the potential to establish along the south coast of South Africa (Alexander *et al.* 2015). In conclusion, the establishment of large individuals in the subtidal zone could have important implications for the future invasion of *S. algosus* as large mussels contribute proportionally more to the reproductive output of the population (van Erkom Schurink and Griffiths 1991; Skein *et al.* 2018a. Given these findings, it is suggested that this species be closely monitored to prevent future spread.

13.3.10 Disc lamp shell *Discinisca tenuis*

The disc lamp shell *Discinisca tenuis* is a small (20 mm diameter) disc shaped brachiopod with a semi-transparent, hairy, fringed shell (Figure 13.17). It was first recorded clinging on oysters grown in suspended culture in Saldanha Bay in 2008 (Haupt *et al.* 2010). More recently, it has been reported as living freely outside of the oyster culture operation on Schaapen Island (Peters *et al.* 2014). This species is endemic to Namibia and is thought to have been introduced to South Africa with cultured oyster imports from this country (Haupt *et al.* 2010). This species reportedly reaches very high densities in it home range and could become a significant fouling species in Saldanha Bay in the foreseeable future, although no previous history of invasion exists for this brachiopod.

Figure 13.17. Disc lamp shell *Discinisca tenuis* (Photo: Prof. C.L. Griffiths)

13.3.11 Vase tunicate *Ciona robusta*

C. robusta was initially misidentified as *C. intestinalis,* which was recently found to represent two morphologically separate species, namely *C. intestinalis* and *C. robusta.* Of these two species *C. robusta* is in fact the species that occurs in South Africa (Brunetti *et al.* 2015; Robinson *et al.* 2016). *C. robusta* is a tall (15 cm), cylindrical yellowish solitary ascidian with a soft floppy, transparent test. It forms large aggregations on submerged structures in harbours and lagoon from Saldanha Bay to Durban (Figure 13.18). It was originally introduced from the North Atlantic prior to 1955. It is an economically important pest as it rapidly fouls hard marine surfaces. It is known to smother and kill mussels on aquaculture facilities, especially mussel ropes.

Figure 13.18. A typical aggregation of *Ciona robusta* (Photo: National Museums Northern Ireland).

13.3.12 Jelly crust tunicate *Diplosoma listerianum*

Diplosoma listerianum is a colonial sea squirt that forms thin, fragile, yellow to dark grey jelly-like sheets up to 50 cm in diameter that grow over all types of substrata on sheltered shores between Alexander Bay and Durban (Monniot *et al.* 2001; Picker & Griffiths 2011; Figure 13.19). It is believed to have been accidentally introduced from Europe prior to the 1949, probably as a fouling organism.

Figure 13.19. Jelly crust tunicate *Diplosoma listerianum* (Photo: Prof. C.L. Griffiths).

13.3.13 Brooding anemone *Sagartia ornata*

The only known records of the brooding anemone *Sagartia ornata* in South Africa are from Langebaan Lagoon (West Coast National Park (WCNP)), where it occurs intertidally in seagrass beds, attached to rocks covered by sand, and in loose rocks resting on fossilized oyster beds (Acuña *et al.* 2004, Robinson *et al.* 2004; Picker & Griffiths 2011; Robinson & Swart 2015; Figure 13.20). *S. ornata* was first detected in 2001 (Acuña *et al.* 2004) and was probably introduced unintentionally through shipping via the Saldanha Bay harbour (Robinson *et al.* 2004). Its home range extends throughout Western Europe, Great Britain and the Mediterranean (Manuel 1981), where it occurs in crevices on rocky shores and on kelp holdfasts (Gibson *et al.* 2001). Introduced species commonly exploit novel habitats, which may reflect the adaptive ability of *S. ornata.*

Robinson & Swart (2015) recently established the status and distribution of this alien anemone, which represents the first comparison to the baseline data collected in 2001 (Robinson *et al.* 2004). The distribution of *S. ornata* has changed within the lagoon and the species is now found in *Nanozostera capensis* (Cape eelgrass) instead of in *Spartina maritima* (spiky cord grass) beds. No apparent reason explains the increase in *S. ornata* abundance compared to 2001 (increasing from 426±81 to 508±218 individuals/m^2).

Figure 13.20. Brooding anemone *Sagartia ornata* (Photo: Prof. C.L. Griffiths)

Sandy-shore areas invaded by this anemone, support a higher invertebrate abundance, biomass and diversity, as well as altered community structures. Communities thus appear to be impacted by *S. ornata,* less so through its role as a predator, but rather as a result of impacts on the habitat structure and associated indirect impacts on native biota (Robinson & Swart 2015). *S. ornata* consolidates sand and traps coarse sediment (*Dr Tammy Robinson pers. obs.)*, which has the potential to significantly change the soft sediment system by altering abiotic factors (e.g. water movement, sediment characteristics) (Ruiz *et al.* 1997; Berkman *et al.* 2000; McKinnon *et al.* 2009).

The habitat types currently preferred by *S. ornata* in South Africa are geographically restricted and limit the potential of this alien species to significantly affect indigenous biota within the WCNP. This species has been categorised as 'naturalised', which means that it has established self-sustaining populations at the point of introduction, but has failed to expand its range beyond Langebaan Lagoon. However, it has the potential to spread more widely into Saldanha Bay and along the South African west coast, where conditions and habitats are similar to that in its home range (Robinson & Swart 2015).

13.3.14 Alien barnacle *Perforatus perforatus*

This species is known only by its scientific name *Perforatus perforatus* (Note: previously misidentified and reported as *Minesiniella regalis*) and as yet has not been assigned a common name. The presence of *P. perforatus* in Saldanha Bay was first recognised in 2011 and was picked up as "an unfamiliar barnacle" at the Dive School in Saldanha Bay as part of the Rocky Shore Intertidal survey in that year. It constitutes the first known record of this barnacle species in South Africa. This species is included in the Sub-family, *Concavinae* (Pitombo 2004) – animals with an extended sheath and longitudinal abutment present on the inner surface of the radii and a bifid sutural edge present on the outer. Characters of the terga, a pronounced beak, closed spur-furrow and absence of longitudinal striations (Newman 1982; Zullo 1992) confirm the identification to species level (Figure 13.21).

Figure 13.21. ***Perforatus perforatus*** **(Pilsbry, 1916) (Photograph: Dr Nina Steffani)**

This species originates from the Pacific coast of North America, with live material recorded intertidally from Baja California, Mexico (Pilsbry 1916). It is difficult to tell when exactly it was introduced to Saldanha Bay in South Africa as, to the untrained eye, it can be easily confused with the local volcano barnacle, *Tetraclita serrata*. However, past reports from the annual State of the Bay monitoring programme have shown that *T. serrata* has never been recorded at the dive school in Saldanha Bay and that *P. perforatus* appeared for the first time in April 2011. It is likely that the introduction of this species occurred via shipping given the high amount of shipping traffic in Saldanha Bay much like the alien acorn barnacle, *B. glandula*, which was also introduced from the Pacific coast of North America (Laird & Griffiths 2008).

13.3.15 Acorn barnacle *Amphibalanus amphitrite amphitrite*

This cryptogenic barnacle species was recorded from Saldanha Bay in the baseline survey in 2005. Only in 2012 was this species recognised to be *Amphibalanus amphitrite amphitrite*, the cryptogenic barnacle which is a prolific fouling species worldwide. This species has longitudinal striations on the exterior shell, which is marked with thick, sparse, purple longitudinal stripes (Figure 13.22). *A. amphitrite amphitrite* is easily confused with another 'purple-pink striped' species which has not yet been identified (Biccard 2012).

Figure 13.22. *Amphibalanus amphitrite amphitrite* (Photo: Prof. C.L. Griffiths)

13.3.16 North West African porcelain crab *Porcellana africana*

The porcelain crab, *Porcellana africana* (Figure 13.23), was previously incorrectly identified as the European porcelain crab, *P. platycheles* (Griffiths *et al.* 2018). Up to date, *P. africana* is the first and only known alien porcelain crab in South Africa. *P. africana* is native to the region between Senegal and Western Sahara in North West Africa. Here, it occurs intertidally and subtidally to a depth of 22 m on rocky shores and boulder beaches (Chace 1956). Species within this genus are cryptic filter feeders and detritivores (Stevcic 1988). Due to the high shipping traffic in Saldanha Bay, *P. africana* was most likely introduced via shipping, by means of ballast water or hull fouling. It was first discovered in South Africa in relatively high numbers on Schaapen Island, Langebaan Lagoon in 2012 (*Prof. George Branch 2012, pers. obs.*). However, its date of introduction has been estimated to be between 2003 and 2009. It is now well established and abundant in Saldanha Bay on the northern, eastern and western shores. Here, it occurs across the intertidal zone under boulders and loose rocks as well as in beds of *M. galloprovincialis*. They are no longer present in Langebaan Lagoon and also absent from the mouth of the Bay. This has been attributed to the absence of rocks and boulders in this area (Griffiths *et al.* 2018). Based on numbers recorded in 2016, it is estimated that the population densities of this porcelain crab can range from anything between 15 to 976 crabs per linear metre of shoreline. Ecological impacts by this species in Saldanha Bay have not yet been quantified, although impacts on native benthic invertebrates are not anticipated. This species is not a typical prey item and due to its feeding habits, will not pose a major threat or compete with native species. This species

Figure 13.23. European porcelain crab *Porcellana africana* (Photo: Prof. C.L. Griffiths).

should, however, be monitored as it has demonstrated its ability to expand its range and increase in numbers in a short period of time (Griffiths *et al.* 2018). In addition, even though subtidal habitats have to date not yet been surveyed, it should be included in future surveys as this species is known to occur to depths of 22 m.

13.3.17 South American sunstar *Heliaster helianthus*

Heliaster helianthus (Lamarck, 1816) is commonly known as the South American multiradiate sunstar (Figure 13.24). It is native to southern Peru and northern and central Chile where it occurs in the intertidal and shallow subtidal (Castilla & Paine 1987). This species can grow to be up to 20 cm in diameter (Barahona & Navarrete 2010) with up to 40 arms (Madsen 1956). *H. helianthus* is a ferocious, generalist predator, its diet consisting mainly of the local mussels, *S. algosus* (Tokeshi 1989) and *Perumytilus purpuratus*. It occasionally shifts its diet to other prey species when mussels are scarce (Barahona & Navarrete 2010). It is mostly free from natural predators within the intertidal zone, although the seastar *Meyenaster gelatinosus* (Gaymer & Himmelman 2008) and to some degree the crab *Homalaspis plana* (Castilla 1981) and rockfish *Graus nigra* (Fuentes 1982), are known to predate upon this species in the subtidal zone. This species sexually reproduces via external fertilisation (Castilla *et al.* 2013) and has planktotrophic larvae with a high longevity. This allows for long distance dispersal (Navarrete & Manzur 2008), a trait that could facilitate invasion.

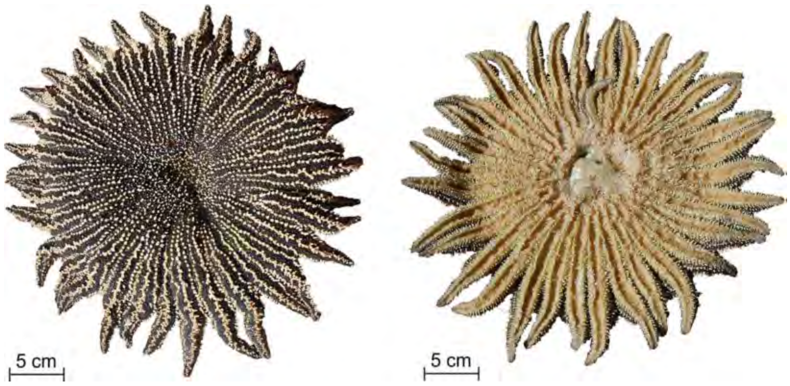


Figure 13.24. *Heliaster helianthus* (Lamarck, 1816) (Photo: Dr Tammy Robinson)

H. helianthus was first discovered in Saldanha Bay in 2015 on the seafloor under a pier within Small Bay, close to Hoedjiesbaai (Peters & Robinson 2018). The area is characterised by sand and rocks. The specimen was a large adult with 35 arms and measuring 33,42cm in diameter. Only a single individual was found and subsequent subtidal and intertidal surveys in the surrounding rocky shore habitats in 2016, revealed no other individuals (Peters & Robinson 2018). This species has the ability to spread and survive in both Saldanha Bay and along the west coast, as in its native range, it inhabits both subtidal and intertidal habitats (Gaymer & Himmelman 2008) and because the natural prey species of *H. helianthus,* i.e. *S. algosus*, is already abundant within Saldanha Bay. In its native range, *H helianthus*, is a keystone species, playing an important role in structuring intertidal and subtidal communities (Paine *et al.* 1985; Navarrete & Manzur 2008). Together with its ferocious, generalist predatory nature (Navarrete & Manzur 2008; Peters & Robinson 2018), this species is expected to greatly impact native biodiversity. In light of these facts, it is imperative that Saldanha Bay and the adjacent coastline be routinely monitored as reintroduction of this species is probable.

13.3.18 Chilean stone crab *Homalaspis plana*

The Chilean stone crab *Homalaspis plana* (H. Milne Edwards, 1834) is native to sheltered habitats along the Chilean coast (Morales & Antezana 1983). It is an important fishery species in the region (Fernández & Castilla 2000). Juveniles occur intertidally on boulder shores in shell fragments, sand and rock platforms and are polychromatic (multicoloured), a trait that might protect them from predation (Fernández and Castilla 2000).

Figure 13.25. *Homalaspis plana* (H. Milne Edwards, 1834) (Photo: Dr Koebraa Peters)

This crab is a generalist predator, feeding predominantly on the barnacle *Balanus laevis*, mussel *S. algosus*, porcelain crab *Petrolisthes tuberculatus,* gastropod *Tegula atra* as well as numerous other crustaceans (Morales & Antezana 1983). Not much is known about the habitat preference or life history of this species (Fernández & Castilla 2000), although it is known to have no invasion history. *H. plana* was first discovered in Saldanha Bay in 2017 in the same area as *H. helianthus*, under a pier within Small Bay (Peters & Robinson 2018; Figure 13.25). The specimen was a purple, adult male with distinctive markings on its carapace. Only a single individual was found and subsequent subtidal and intertidal surveys in the surrounding rocky shore habitats in 2018, revealed no other individuals (Peters & Robinson 2018). This species is not anticipated to survive along the open coast. In light of the fact that Saldanha Bay offers a suitable sheltered habitat with abundant prey species (i.e. S. a*lgosus*), it is important that the area be routinely monitored, as reintroduction of this species is probable.

13.3.19 Hydrozoan *Coryne eximia*

The hydrozoan *Coryne eximia* (Figure 13.26) was first discovered in South Africa in 1946 and occurs mainly along the West Coast where it has been found from Cape Town docks to Llandudno and also in Langebaan Lagoon. It is a fouling organism which commonly occurs in shallow water up to a depth of 25 m on anchoring chains of buoys, rafts, mussels, rocks and seaweed (Millard 1975; Schuchert 2005). The native region of *Coryne eximia* is assumed to be the North Atlantic or North Pacific region (Millard 1975).

Figure 13.26. Hydrozoan *Coryne eximia* (Photo: Peter Schuchert. http://www.ville-ge.ch/musinfo/ mhng/hydrozoan/ hydrozoa-directory.htm).

It has also been recorded as alien in the Pacific Ocean from California to Alaska, Chile, Brazil, Papua New Guinea, Western Australia and New Zealand; in the Atlantic Ocean from Norway to Galicia, the east coast of North America and Canada as well as in the Mediterranean (Schuchert 2001; Puce *et al.* 2003).

13.3.20 Tubeworm *Neodexiospira brasiliensis*

Neodexiospira brasiliensis is a small epifaunal tubeworm that forms a coil with a diameter of 2 mm (Figure 13.27). It is native to the Indo-Pacific, although its exact distribution is unknown (Knight-Jones *et al.* 1975). This polychaete is a filter feeder, and feeds mainly on phytoplankton (Fofonoff *et al.* 2019). They are hermaphroditic and self-fertilization can occur on rare occasions (Benkwitt 1982). Larvae tend to settle on algae and seagrass (Critchley & Thorp 1997) and these worms have been found to occur on boats, hulls of ships, floats, pilings, mussels, floating seaweeds, driftwood and snail shells (Critchley *et al.* 1997) and commonly settle in areas containing bacterial films (Kirchman *et al.* 1982). Due to their nature of settling on floating objects, there is potential for this species to spread once introduced.

Figure 13.27. Tubeworm *Neodexiospira brasiliensis* (Photo: CBG Photography Group, Centre for Biodiversity Genomics and Boldsystems.org).

This polychaete was first recorded in South Africa in 1953, but misidentified as *Spirorbis foraminosus*. It has been recorded as occurring in tide pools on the algae, *Ceramium planum* from Cape Town to Port Elizabeth in the past (Fofonoff *et al.* 2019), although an updated survey of its distribution is required. It is also alien to the East and West coasts of North America and Europe. Observations suggest that this species has the potential to impact eelgrass (Fofonoff *et al.* 2019).

13.3.21 Shell-boring spionid *Polydora hoplura*

Polydora hoplura is a shell-boring spionid polychaete native to Europe where it occurs from the Mediterranean to England (Figure 13.28). It bores into calcareous materials including mollusc shells, barnacles, sponges, coralline algae, and limestone (Fofonoff *et al.* 2019). It is commonly found on cultivated oyster beds and culture facilities for abalone and oysters. This polychaete has a wide alien distribution including California, Australia, New Zealand, Japan, Chile, Brazil, the Canary Islands and South Africa. It was first recorded in South Africa in 1947 in Table Bay (Millard 1952).

Figure 13.28. Shell-boring spionid *Polydora hoplura* (Photo: Prof. C.A. Simon).

Subsequently, Day (1967) reported it in the intertidal and shallow waters from Saldanha Bay to Plettenberg Bay. This polychaete commonly infests the commercially cultured oysters (Nel *et al.* 1996) and abalone *Haliotis midae* in Saldanha Bay (Simon *et al.* 2006; Boonzaaier *et al.* 2014). This can have negative economic implications as it decreases the survival and condition of cultured oysters and abalone (Fofonoff *et al.* 2019).

13.3.22 Tube-dwelling amphipod *Cerapus tubularis*

Cerapus tubularis is an intertidal, tube dwelling amphipod native to North America. It occurs on sandy substrates with shell fragments, large sand grains and among algae. Its alien range includes areas within the tropical and temperate oceans. This species was first recorded from South Africa, off the coast of KwaZulu-Natal, in 1901, but was incorrectly reported as *Cerapus abditus* (in Barnard 1916). It was most likely introduced via ballast water or ship fouling. The most recent publication reports this species' range as extending from Saldanha Bay to the east coast of South Africa (Mead *et al.* 2011).

Note: No photo available at this stage.

13.3.23 Wood-boring amphipod *Chelura terebrans*

Chelura terebrans is a wood-boring amphipod, easily distinguishable due to its enlarged third uropod, fused urosomites and reddish appearance (Figure 13.29). It is thought to be native to Europe and, in addition to South Africa, has a very broad alien distribution which includes New Zealand, the West and East coasts of North America and Hong Kong (Fofonoff *et al.* 2019). The most likely mode of introduction is hull fouling of the wooden ships used in the past (Kuhne & Becker 1964).

Figure 13.29. Wood-boring amphipod *Chelura terebrans* (Photo: Eric A. Lazo-Wasem)

C. terebrans is dependent upon wood-boring isopods of the genus *Limnoria* for shelter and food as it inhabits the burrows of these isopods and feeds on their faecal matter (Kuhne & Becker 1964; Borges 2010). Its diet also includes bacteria, protists and decaying wood. It is believed that this amphipod will, under certain circumstances, be able to create its own burrows (Green Extabe 2013). Some of the first specimens in South Africa were collected in 1888 and reported by Stebbing (1910). More recently, it has been reported as occurring in all harbours between Langebaan and Port Elizabeth, although further surveys are required to determine if it has spread to the open coast (Mead *et al.* 2011). Due to its wood-boring nature, it is considered a pest, but only in the presence of *Limnoria*. Damage by this species includes expanding the burrows and increasing the damage by *Limnoria* (Kuhne and Becker 1964). This species thus has the potential to negatively impact the economy by destroying wooden structures.

13.3.24 Sand-hopper *Orchestia gammarella*

Figure 13.30. The sand-hopper *Orchestia gammarella* (Photo: Auguste Le Roux [CC BY-SA 3.0 (https://creative commons.org/licenses/by-sa/3.0)].

Orchestia gammarella, or the sand-hopper, is a semi-terrestrial amphipod (Figure 13.30). Its native range includes Norway to the Mediterranean, as well as Madeira, Canary Islands and the Azores (Henzler & Ingolfsson 2008). It occurs in the upper intertidal of rocky shores, primarily in the drift-line, under, rocks, debris and vegetation (Mead *et al.* 2011; Fofonoff *et al.* 2019). *O. gammarella* is mainly herbivorous, but also known for its scavenging behaviour. It feeds on detritus, algae, seaweed, seagrasses and microorganisms (Persson 1999). It was first discovered in South Africa in Langebaan Lagoon during a UCT ecological survey, but incorrectly described as a new endemic species, *Talorchestia inaequalipes* by Barnard (1951). It was later correctly identified by Griffiths (1975). Alien populations have also been described from Knysna Estuary (Griffiths 1974) and Table Bay (Milnerton Lagoon; Mead *et al.* 2011). It was most likely introduced via solid ballast. Alien populations are also known from North America (Newfoundland to Maine), South America (Argentina and Chile) and Iceland (Fofonoff *et al.* 2019).

13.3.25 Bryozoan *Conopeum seurati*

Figure 13.31. The colonial bryozoan *Conopeum seurati* (Photo: De Blauwe 2009).

The bryozoan, *Conopeum seurati* is a fouling organism, native to brackish water, lagoons and estuaries in Europe (Poluzzi & Sabelli 1985), although its exact distribution is unknown. It has been introduced via ship fouling to the East coast of North America, New Zealand and Australia (Gordon & Mawatari 1992; Winston 1995; Wyatt *et al.* 2005; Rouse 2011) and South Africa (Awad *et al.* 2005). Overlooked alien populations are likely to occur in estuaries all around the globe. This species is a filter feeder of phytoplankton and tends to form small colonies on shells, seagrasses, seaweeds, and other hard surfaces, including man-made structures (Figure 13.31). It was first recorded in South Africa in Saldanha Bay in 2001 (Awad *et al.* 2005), although it has probably been present for decades, if not centuries. This species potentially also occurs in Zandvlei Lagoon (Falxse Bay), although proper identification is required (Mead *et al.* 2011).

13.3.26 Bryozoan *Cryptosula pallasiana*

Figure 13.32. The colonial bryozoan *Cryptosula pallasiana* (Photo: Cohen 2011).

The bryozoan *Cryptosula pallasiana* occurs in brackish waters, where it forms pink, white or orange encrusting colonies (Occhipinti Ambrogi & d'Hondt 1981; Figure 13.32) on eelgrass beds and hard structures including shells, oyster beds, rocks, hulls of ships and other man-made structures (Hayward & Ryland 1999; Fofonoff *et al.* 2019). It is native to Europe, specifically the Black sea and also suitable habitats ranging from the Mediterranean Sea to Norway. This species is a filter feeder, feeding mainly on phytoplankton (Barnes 1983).

It has been reported from numerous harbours around the globe (Gordon & Mawatari 1992). Cryptogenic populations occur along the East coast of North America and Northwest Pacific. Alien populations are known from South Africa, the Pacific coast of North America, Argentina, New Zealand and Australia (Fofonoff *et al.* 2019). It was first recorded in South Africa in Table Bay harbour, as *Lepralia pallasiana*, based on specimens collected during 1947-1949

(Millard 1952). It was later also discovered in Simon's Town (Henschel *et al.* 1990) and Saldanha Bay (Awad *et al.* 2005), although it is most likely widespread throughout South African estuaries (Mead *et al.* 2011).

13.3.27 Red-rust bryozoan *Watersipora subtorquata*

Also known as the red-rust bryozoan, *Watersipora subtorqua*ta is a shallow water fouling organism. It forms calcareous crusts on hard surfaces such as rocks, shells, pilings, hulls of ships, floating objects, fouling plates and oil platforms and creates secondary habitat for the settlement of other marine invertebrates (Mackie *et al.* 2006; Page *et al.* 2006; Cohen & Zabin 2009; Ryland *et al.* 2009; Figure 13.33). *W. subtorquata* is a suspension feeder, feeding predominantly on phytoplankton. The exact native range of this bryozoan is unknown, primarily because of taxonomic confusion and the notion that it might be a species complex (Fofonoff *et al.* 2019).

A recent taxonomic revision of the genus *Watersipora* revealed unexpected changes in the distribution and nomenclature (Vieira *et al.* 2014). Until further studies and genetic analysis can resolve the confusion, this bryozoan will retain its name where it has previously been identified (Florence *et al.* 2007; Fofonoff *et al.* 2019). It was first reported in South Africa in 1935 (although it has probably been present for longer than that) as *W. cucullata* (O'Donoghue & deWatteville 1935) and later synonymised with *W. subtorquata* (Florence *et al.* 2007). Its distribution has been reported as Saldanha Bay on the west coast to False Bay on the south coast (Florence *et al.* 2007). It has been widely distributed throughout the world via hull fouling and ballast water. Introduced populations have also been recorded from New Zealand, Australia, Hawaii, Europe and possibly the West coast of North America (Mead *et al.* 2011).

Figure 13.33. Red-rust bryozoan *Watersipora subtorquata* (Photo: Luis A. Solórzano in Cohen 2011).

13.3.28 Light bulb tunicate *Clavelina lepadiformis*

Figure 13.34. Light bulb tunicate *Clavelina lepadiformis* (Photo: Esculapio CC BY 3.0, https://commons.wikimedia.org /w/index.php?curid=4765030).

This species of tunicate, has transparent zooids with yellow, white or pink bands around the dorsal lamina and oral siphon, earning them the name the Light bulb tunicate (Figure 13.34). Colonial tunicates can reproduce both sexually and asexually through budding and feeds primarily on phytoplankton and detritus (Fofonoff *et al.* 2019). *C. lepadiformis* originates from Europe, where it ranges from the Mediterranean Sea to Southern Norway (Tarjuelo *et al.* 2001). It has most likely been introduced via ship fouling to South Africa, the east Coast of America, Azores and South Korea. They occur in rocky, shallow water areas and is commonly found in harbours, marinas and ports where they attach to the bottom and sides of jetties and boats (Mead *et al.* 2011). It was first reported in South Africa in Port Elizabeth and Knysna (Monniot *et al.* 2001) and subsequently from numerous areas around the coast, including Saldanha Bay (Rius *et al.* 2014). More in-depth surveys inspecting artificial substrata are required to confirm if this species has spread throughout Saldanha Bay and to Langebaan Lagoon.

13.3.29 Algae *Antithamnionella spirographidis*

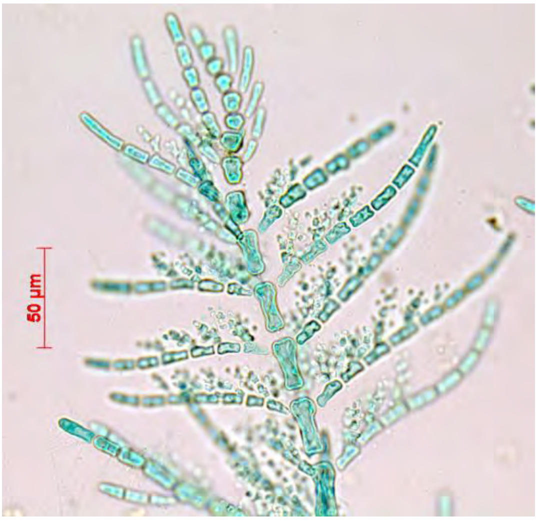


Figure 13.35. Stained slide of the algae *Antithamnionella spirographidis* (Photo: Anderson *et al.* 2016).

Antithamnionella spirographidis is a small algal species (Figure 13.35), most likely native to North Pacific regions (Lindstrom & Gabrielson 1989) and has been introduced, most likely via ship fouling or aquaculture activities. Introduced populations are reported from England, Wales, Ireland, Scotland, northern France, the Mediterranean and Australia (Wollaston 1968; Maggs & Hommersand 1993; Eno *et al.* 1997). It is commonly associated with harbours and docks (Wollaston 1986) and its success as an invader is attributed to its vegetative, rapid reproduction. It was first recorded in South Africa in sheltered areas of Saldanha Bay attached to jetties in 1989 (Stegenga *et al.* 1997). More in-depth surveys inspecting artificial substrata are required to confirm if this species has spread throughout Saldanha Bay and to Langebaan Lagoon.

13.3.30 Dead Man's Fingers *Codium fragile ssp. tomentosoides*

The invasive strain of this algae is known as *Codium fragile ssp. tomentosoides*. This intertidal green algae has thick spongy, "finger-like" branches, hence its common name 'Dead Man's Fingers' (Figure 13.36). There has been a lot of confusion regarding the date of first record and the presence of this species in South Africa, mostly as it can and has been confused with the native species, *C. fragile ssp. capense* (Mead *et al.* 2011) which is widespread in the sublittoral and intertidal areas from Namibia on the west coast to Plettenberg Bay on the east coast (Stegenga *et al.* 1997). However, it is believed that *C. fragile sp. tomentosoides* occurs interspersed among the native populations and as such, it remains on the list of species alien to South Africa and Saldanha Bay (Mead *et al.* 2011).

Figure 13.36. Dead Man's Fingers *Codium fragile* (Photo: Flyingdream and Wikipedia).

It is native to waters around Japan and introduced populations have also been recorded from North and South America, Europe, Greenland and New Zealand. It occurs in both rocky and sandy habitats, where it attaches to hard substrates including oyster and seagrass beds, shells, stones, seawalls, breakwaters, jetties, piers and docks (Ramus 1971; Gosner 1978; Bulleri and Airoldi 2005; Geraldi *et al.* 2014). *C. fragile* species also frequently invades kelp beds (Scheibling and Gagnon 2006) and *Zostera marina* eelgrass beds (Ramus 1971). *C. fragile* is considered euryhaline, tolerant of a wide range of temperatures (-2 to 30°C) and tolerant to desiccation, traits that could contribute towards its introduction and invasion success (Malinowski and Ramus 1973; Hanisak 1979; Schaffelke and Deane 2005). Although impacts in Saldanha Bay have not yet been quantified, this species is known for both its positive and negative impacts in other invaded locations, such as being an important food item and habitat for many invertebrates (Cruz-Rivera and Hay 2001; Scheibling and Anthony 2001; Harris and Jones 2005). More in-depth surveys and genetic analysis are required to confirm the identity and exact distribution of this species in Saldanha Bay and Langebaan Lagoon.

14 MANAGEMENT AND MONITORING RECOMMENDATIONS

Monitoring of aquatic health and activities and discharges potentially affecting health of Saldanha Bay and Langebaan Lagoon has escalated considerably in recent years owing to increases in the rate of development in the area surrounding the Bay and Lagoon and concerns over declining health of the Bay. This section provides a summary of the state of health of Saldanha Bay and Langebaan Lagoon as reflected by the various environmental parameters reported on in this study. It also briefly describes current monitoring efforts and provides recommendations as to management actions that need to be implemented in order to mitigate some of the threats that have been detected. It also provides recommendations on how existing monitoring activities may need to be modified in the future to accommodate changes in the state of the Bay.

14.1 The management of activities and discharges affecting the health of the Bay

Continuously accelerating urban and industrial development is a major cause of fragmentation and loss of ecological integrity of remaining marine and coastal habitats in Saldanha Bay and Langebaan Lagoon. The challenge of addressing cumulative impacts in an area such as Saldanha is immense. The current and future desired state of the greater Saldanha Bay area is polarised, where industrial development (Saldanha Bay IDZ and associated industrial development) and conservation areas (Ramsar Site, MPAs and National Park) are immediately adjacent to one another. Furthermore, the Saldanha Bay environment supports conflicting uses including industry, fishery, mariculture, recreation and the natural environment itself. This situation necessitates sustainable development that is steered towards environmentally more resilient locations and away from sensitive areas.

Concerns have been raised that cumulative impacts on the marine environment in Saldanha Bay have not been adequately addressed by many recent development proposals. This applies especially to the cumulative impacts that will arise from future development within the Saldanha Bay IDZ and the Aquaculture Development Zone (ADZ). Furthermore, the impact on the Saldanha Bay marine environment from projects that are primarily land-based, such as storage facilities for crude oil and liquid petroleum gas, has often been underestimated or even ignored. It has been proposed that a more holistic management strategy is needed to deal with piece meal Environmental Impact Assessments (EIAs). Various environmental management instruments have been proposed for the Greater Saldanha Bay Area, including (1) a generic Environmental Management Programme (EMPr), (2) an Environmental Management Framework (EMF), (3) a Strategic Environmental Assessment (SEA), and (4) the declaration of a Special Management Area. An Intergovernmental Task Team (IGTT) has been established to consider these and other proposals. If these management instruments are indeed implemented, we are confident that measures for the conservation alongside rapid development of the Saldanha Bay area will be addressed more effectively.

14.1.1 Human settlements, water and wastewater

Human settlements surrounding Saldanha Bay and Langebaan Lagoon have expanded tremendously in recent years. This is brought home very strongly by population growth rates of 9.24% per annum in Langebaan and nearly 2.7% in Saldanha over the period 2001 to 2011 (Statistics South Africa 2014). Numbers of tourists visiting the Saldanha Bay and Langebaan Lagoon area are constantly rising, especially those visiting the West Coast National Park (WCNP) (Average rate of 12% per annum since 2005). This rapid population and tourism growth translate to corresponding increases in the amounts of infrastructure required to house and accommodate these people and also in the amounts of waste and wastewater that is produced and must be treated and disposed of.

In an effort to reduce potable water consumption in the area, the Saldanha Bay Municipality (SBM) has come to an agreement with various types of water users (construction, irrigation, industry) to re-use treated wastewater. This has dramatically reduced the potable water demand and has had the positive spinoff in that currently only very small volumes of wastewater from the WWTWs enter the marine environment. With the closure of Arcelor Mittal the Saldanha WWTW no longer receives industrial wastewater from the plant, however the plant also no longer requires treated waste for use in the Reverse Osmosis plant. Therefore, the balance of treated wastewater currently not being used by other water users for irrigation is being discharged into the Bok river. The municipality has however identified a future user for the treated effluent and an allocation has been made available to them.

The amount of hardened (as opposed to naturally vegetated) surfaces surrounding the Bay and Lagoon have also expanded at break-neck speed in recent years, with concomitant increases in volumes of contaminated storm water running off into the Bay. The contaminant loads in stormwater is not adequately monitored (there is no monitoring of storm water quality or quantity from Saldanha or Langebaan), nor is it adequately controlled at present. The contribution to trace metal and organic loading in the Bay from these sources is thus largely unknown, and remains of concern. Disturbance from increasing numbers of people recreating in Saldanha Bay and Langebaan Lagoon is taking its toll on sensitive habitats and species, especially seagrass, water birds and fish in Langebaan Lagoon. A collaboration between Saldanha Bay municipality and Sea Harvest has initiated a project to install litter traps on stormwater drains to minimize pollution entering the bay via these waterways.

14.1.2 Dredging

Dredging interventions in the Bay in the past, particularly those associated with the iron ore terminal have been shown to have devastating impacts on the ecology of the Bay. Effects of the most recent - large scale dredging event are still discernible in the sediments and faunal communities in the Bay more than a decade after their occurrence. Likely ecological impacts arising from any future proposed dredging programmes need to be carefully considered and these need to be weighed up against social and economic benefits that may be derived from such programmes or projects. Where such impacts are unavoidable, mitigation measures applied must follow international best practice and seek to minimize impacts to the ecology of the Bay. Even relatively small dredging operations, such as those undertaken as part of the upgrade of the naval boatyard at Salamander Bay, can have very wide-reaching impacts on the Bay and Lagoon.

Historically, insufficient provision was made for buffers zones around the Lagoon and Bay with the result that development encroaches right up to the waters' edge and is now widely threatened by coastal erosion. Recently published research suggests that dredging operations conducted during the Port construction programme may be contributing to this problem as well. This research highlights the fact that much of the sediment used to build the causeway to Marcus Island was dredged from the historic ebb tide delta that existed at the mouth of Langebaan (an area where sediment derived from Langebaan Lagoon had been deposited over many thousands of years). Removal of sediment from this area has reduced the extent to which incoming waves are refracted and has increased in the wave energy density along the shoreline by around 50%. This in turn seems to be contributing to the observed erosion of the shoreline in this area.

14.1.3 Fish factories

The Department of Environmental Affairs is currently in the process of issuing Coastal Waters Discharge Permits to facilities discharging wastewater into Saldanha Bay. Sea Harvest was first issued a CWDP on 26 June 2017 (as amended subsequently to accommodate a change in discharge location and effluent composition) and further amended on 7 November 2019 to include discharge from the fish processing plant, the RO plant and the value added factory. This CWDP authorises Sea Harvest to dispose a maximum quantity of 2 190 000 m^3 per annum at a maximum daily discharge volume of 6 000 m^3. With the ongoing drought in the Western Cape, Sea Harvest reclaims potable water by means of a Reverse Osmosis plant with the intention to save municipal water and to improve effluent quality (Frank Hickley, Sea Harvest *pers. comm*., 2018). Sea Harvest is committed to meeting effluent quality thresholds and environmental monitoring requirements as stipulated in the CWDP. However, the effluent at the Sea Harvest Fish Processing Plant is currently not treated adequately to ensure minimum impact to the receiving environment. While a number of the parameters have improved, the fish processing facility is still failing to comply with the chemical oxygen demand and oil and grease concentrations prescribed in the CWDP, although both of these have improved in 2019/20 relative to the two preceding years they still exceed the allowable limits. The effluent produced by the RO plant has increased the salinity of the overall effluent dramatically and CWDP requirements were exceeded 50% of the time until the limit was increased in the amended CWDP, subsequently salinity readings are compliant 91% of the time. Additionally, significant improvements have, however been observed in terms of the ammonia nitrogen and total suspended solids concentration and the current CWDP limits are being met. Sea Harvest has been meeting the pH range prescribed in the CWDP.

14.1.4 Marine Aquaculture

Saldanha Bay is a highly productive marine environment and constitutes the only natural sheltered embayment in South Africa (Stenton-Dozey et al. 2001). These favourable conditions have facilitated the establishment of an aquaculture industry in the Bay. In January 2018, the then Department of Agriculture, Forestry and Fisheries was granted Environmental Authorisation to establish a sea-based Aquaculture Development Zone (ADZ) in Saldanha Bay and expand the total area available for aquaculture to 884 ha, located within four precincts (Small Bay, Big Bay North, Outer Bay North and South). By the end of December 2019, approximately 36% of the ADZ had been leased, but less than 60% of the actively leased area was being utilised, and this is a very dynamic value that changes constantly as new leases and rights are granted or as the economic climate changes. As of March

2020, 28 companies within the Saldanha Bay ADZ were registered on the Marine Aquaculture Right Register, of which only 15 companies were actively operational. Historic studies, as well as the State of the Bay surveys have shown that these culture operations can lead to organic enrichment and anoxia in sediments under the culture rafts and ropes. The source of the contamination is believed to be mainly faeces, decaying mussels and fouling species. DEFF specialist scientists are however conducting environmental monitoring (which includes a rapid synoptic survey of oxygen and nutrient levels in the Bay) and long term monitoring undertaken by an independent service provider. This is to ensure that the effects of aquaculture operations are well monitored and that should any thresholds be exceeded the appropriate action is taken and mitigation measures can be implemented to prevent negative impacts. International monitoring standards recommend that full macrobenthic surveys should be conducted every 3-5 years, however, the scale of the expanded ADZ is significant and the macrofaunal communities within Saldanha Bay show an inherently high level of variation. Therefore, the frequency of these monitoring surveys may need to be intensified to prevent significant ecological impacts, as well as loss to the mariculture sector itself.

14.1.5 Shipping, ballast water discharges and oil spills

Shipping traffic and ballast water discharges to the Bay are currently monitored by the Port of Saldanha. Data indicate a steady growth in the numbers of vessels visiting the Bay and a concomitant increase in the volume of ballast water discharged to the Bay. As a result, environmental impacts are increasing, including but not limited to oil spills, introduction of alien species, trace metal pollution as well as direct disturbance of marine life and sediment in the bay. Trace metal concentrations in ballast water discharged to Saldanha Bay have in the past (1996), been shown to exceed South Africa Water Guidelines. Whether this is still the case is unknown, given that the concentrations of these contaminants in ballast water discharges has not been assessed in recent years.

To address environmental impacts and risks from the discharge of ballast water, the International Convention for the Control and Management of Ship's Ballast Water and Sediments of 2004 (BWM Convention) was ratified by 30 states, including South Africa. It took almost a decade until the first Draft Ballast Water Management Bill was published in the *Government Gazette* in April 2013 (Notice 340 of 2013), aimed at giving effect to the provisions of the BWM Convention. The Draft Bill was published in the Government Gazette for comment again in 2017 but it is unknown when it will be finalised. The Bill sets out how ballast water is to be discharged, all ships are expected to have a ballast water management plan, and to keep an up to date ballast water record book. Vessels constructed after 2009 are required to be designed such that accumulation of sediments is prevented and removal is facilitated. Although no domestic legislation is currently in place to regulate ballast water discharge, the Transnet National Port Authority in Saldanha Bay has implemented a number of mechanisms to track and control the release of ballast water into the harbour.

14.1.6 Recommendations

Urgent management interventions are required to limit further degradation of the environment from the growing pressures and should focus on the following issues:

- Ensure that all discharges to the Bay, including discharges into rivers entering the marine environment, are properly licensed and monitored (both effluent volume and quality) to confirm that conditions at the edge of the mixing zone are compliant with South African Water Quality Guidelines for the Coast Zone and any other legislative requirements;

- Existing and any future increases in use of groundwater from the Langebaan Road and Elandsfontein Aquifers need to be considered very carefully, especially in the light of effects that this may have on Saldanha Bay and Langebaan Lagoon.
- Wastewater recycling should continue as wastewater production increases in the area.
- The Saldanha Bay Municipality should re-evaluate the effectiveness of shoreline erosion mitigation measures implemented in Saldanha and Langebaan taking into account possible impacts associated with dredging that was undertaken as part of the port construction operations in the 1970s and how this can be reversed.
- Coastal management (development setback) lines also need to be established around the perimeter of the Bay and Lagoon and these must allow for adequate protection of the environment and infrastructure from current and future (i.e. climate change) pressures.
- The Draft Ballast Water Management Bill (2017) needs to be finalised, promulgated and implemented as a matter of urgency; and
- Declaration of Saldanha Bay and Langebaan Lagoon as a Special Management Area in terms of ICMA should continue to be pursued.

14.2 Groundwater

While Saldanha Bay and Langebaan Lagoon receives little freshwater input via rivers or streams (surface water), groundwater input is significant and plays an important role in sustaining marsh ecosystems around the periphery of the Bay, and especially the Lagoon, and also in preventing widespread hypersaline conditions from developing in the summer months. There are two main aquifer systems from which groundwater discharges into the Bay – the Langebaan Road Aquifer System and the Elandsfontein Aquifer System with discharge to the sea occurring through separate paleo-channels. The Langebaan Road Aquifer System discharges into Saldanha Bay (Big Bay) through a northern paleo-channel while the Elandsfontein Aquifer System discharges into Langebaan Lagoon through a southern paleo-channel. Growth of the reeds *Phragmites australis* and *Typha capensis* as well as *Juncus kraussi* on the shoreline surrounding Langebaan Lagoon provide clear evidence of the significant influx of groundwater to the Lagoon, because these plants can only survive in water or damp soil, and are only able to tolerate salinity levels up to a maximum of 20–25 PSU (the salinity of the water in the lagoon is generally the same, or occasionally higher, than the 35 PSU of seawater).

The potential bigger impacts on groundwater resources surrounding Saldanha Bay and Langebaan Lagoon have been identified as (1) the agricultural sector (1 529 744 m3/a) (this registered quantity is groundwater abstraction for agriculture as at 2016 and probably increased significantly during the drought of 2015 to 2018); (2) abstraction from the Langebaan Road Aquifer wellfield (intermittently operational since 1999, however, frequently non-operational due to regular and persistent vandalism); and (3) the Hopefield wellfield (not yet operational) where it is planned to abstract 5.1 Mm^3/a and 1.8 Mm^3/a, respectively. The Langebaan Road Aquifer has been operational for 20 years but only intermittently and the long-term monitoring trend shows only slight groundwater level drawdowns. The total utilisable groundwater exploitation potential (UGEP) under normal conditions is estimated at 15.2 Mm^3/a from the SBM area, so it is important to try and reduce the impact of this nett abstraction by using Managed Aquifer Recharge methodologies and it is quite possible the wellfields will only be used in times of severe drought, so they need to be kept as “full” as possible in non-drought times. If the UGEP is adhered to, available evidence suggests that is unlikely to be an impact on the outflow to the marine environment, however, the positioning of the abstraction is crucial to ensure there is no impact on these outflows. Comprehensive groundwater monitoring and associated database within the entire region is also essential for the long-term management and preservation of the aquifers and freshwater inflows into the Langebaan Lagoon. Within the Greater Saldanha Bay area, it is imperative to ensure all groundwater abstraction above the General Authorisation limit is authorised and that the associated compliance conditions are adhered to.

Elandsfontein Exploration and Mining (Pty) Ltd/Kropz recently started mining phosphate deposits in the area of the Elandsfontein Aquifer System on the eastern side of the R27. Mining is being conducted using an open-pit strip mining method which requires that groundwater levels around the mining pit be lowered to prevent the mine pit from being flooded. Groundwater is being abstracted from a series of boreholes surrounding the mine pit but is reinjected downstream (towards the lagoon), in an effort to ensure that surrounding ecosystems (including the Lagoon) are not affected. Available evidence suggests that this activity is unlikely to impact on the lagoon, however, Kropz Elandsfontein in conjunction with the Saldanha Bay Water Quality Forum Trust (SBWQFT) elected to initiate monitoring a range of biological and physico-chemical variables associated with Langebaan Lagoon to establish an appropriate baseline against which any potential future changes in the Lagoon can be benchmarked. This includes monitoring of temperature and salinity (see below) and biota (see Chapter 9) as well as macrophytes (see Chapter 8) around the top end of the lagoon.

Monitoring of temperature and salinity at the head of the lagoon was initiated in September 2016 using a Star ODDI Salinity, Conductivity, Temperature and Depth Logger. The Star ODDI was subsequently replaced with an Aqua TROLL 200 data logger (August 2019) which has been yielding considerably better and more useful data to date. These data records show clear diurnal/tidal and seasonal trends in water temperature and salinity. The diurnal fluctuations in temperature are similar across all seasons, with temperatures increasing over the course of the day, peaking in the early afternoon, then declining through the afternoon and night , reaching a minimum at the time of sunrise each day. The trend in salinity is more interesting though, exhibiting a similar diurnal oscillation to that for temperature, but this oscillation is linked to the state of the tide (not the time of day) and changes through the year. In winter, salinity oscillates between that of normal seawater (around 35.0 PSU) at high tide and a slightly fresher state (between 32.0 and 34.0 PSU) at low tide. Salinity appears to drop as the tide recedes, and is most likely linked with outflow of freshwater from the aquifer at this time. In summer, the pattern reverses with salinity increasing from that of normal seawater (35.0 PSU) at the peak of the high tide becoming hyper-saline (39-40 PSU) as the tide recedes. It is likely that this is a function of increased evaporation at this time of year (linked to higher prevailing air temperatures), and that the water emerging from the marshes at the head of the lagoon becomes severely hypersaline as a result, and even though it is diluted by freshwater flowing out of the aquifer, this is not sufficient to bring the level below that of normal seawater. It is likely that this effect (development of hypersaline conditions) is quite localised at present (i.e. restricted to the extreme upper reaches of the lagoon only) but could become much more pervasive if freshwater outflow from the aquifer were to drop in future.

There appears to be no link between rainfall and salinity levels in the lagoon which strongly suggests that variations in salinity in the lagoon are linked with groundwater inflow as opposed to surface water inflow, which is consistent with observations made by others and points to the need for continued monitoring to track any changes over time.

14.3 Water quality

From a water quality perspective, key physico-chemical changes that have resulted from anthropogenic impacts on the Bay include modification in circulation patterns and wave exposure gradients in the Bay, leading to a reduction in water movement and exchange between the Bay and the adjacent marine environment. The SBWQFT has over the last six years monitored water temperature in Small Bay and temperature and salinity in Langebaan Lagoon. These activities are yielding valuable insights into the functioning and health of the Bay but urgently ned to be expanded to other areas and need to be extended to include a range of other parameters such as dissolved oxygen, turbidity, nutrients, chlorophyll a (as measure of phytoplankton production). As part of the environmental monitoring programme for the Saldanha ADZ (DAFF 2018), DEFF initiated monitoring of dissolved oxygen at four sites, a control and ADZ site in Small Bay and Big Bay with hourly dissolved oxygen recordings made close to the bottom (0.5 m above the seabed). DEFF have also installed a nitrate sensor at the control site within Big bay (18 m depth) that provides hourly measurements of nitrate concentration and temperature (these parameters are inversely correlated due to upwelling). DEFF have also proposed installing a fluorometer (which provides an indication of phytoplankton or at least chlorophyll concentration in the water column) in the entrance channel of Langebaan Lagoon. The ADZ monitoring programme also makes provision for collection of phytoplankton samples for calibration of the fluorometer readings. This would entail collecting discrete samples of water, sieving a portion of each sample through a 2-5 µm mesh (to extract the picoplankton component), and extracting the chlorophyll from both the screened and unscreened samples to obtain an estimate of the relative contribution from each component. DEFF are also reportedly collecting water samples on a frequent basis (a number of times a week) in the existing shellfish growing areas in the entrance to Small Bay and in North Bay as part of the South African Live Molluscan Shellfish Monitoring and Control Programme (DAFF 2019) for species identification and enumeration of phytoplankton. The ADZ monitoring programme recommends extending this sampling effort to include collection of discrete samples for size-fractionated chlorophyll analysis at least three sites that are paired as close as possible in time. The first dissolved oxygen data from this ADZ monitoring has been presented in the 2020 SOB report and have already provided some insights into the effects of bivalve mariculture on water quality. Inclusion of the nitrate and other data to the State of the Bay Monitoring Programme in the future would definitely be welcomed.

The concentrations of metals in the flesh of mussels used to be monitored by the Mussel Watch Programme (DAFF). Data are available for the period between 1997-2001 and 2005-2007 but the programme has since been discontinued. Since 2014, the SBWQFT has been collecting mussel samples from the same five sites during the field survey for trace metal analysis. The mussel samples collected from the shore and port infrastructure are analysed for the metals cadmium (Cd), copper (Cu), lead (Pb), zinc (Zn), iron (Fe) and manganese (Mn). Data on trace metals concentrations in shellfish from the mariculture farms in the Bay are also obtained from the DEFF (courtesy of the farm operators).

Concentrations of lead and cadmium in marine filter feeders in Saldanha Bay indicate that concentrations of these trace metals from some locations and samples are above published food safety limits. Concentrations of lead reported for mariculture operations lower and in nearly all cases within the food safety limits. This may be linked to higher growth rates of farmed mussels, and the fact that the cultured mussels feed on phytoplankton blooms in freshly upwelled, uncontaminated water, whilst mussels along the shore are more exposed to land-based pollutants.

Metal contamination poses a very serious risk to the health of people harvesting mussels from the shore (large qualities of shellfish are harvested and consumed by recreational and subsistence fishers from the shore of the Bay) and concentrations above regulatory limits requires management interventions to address the issue. It is vitally important that this monitoring continues in the future and that data are made available to the public. It is also imperative that this Mussel Watch Programme be revamped and possibly extended to cover other species as well (e.g. fish). As elevated trace metal concentrations within seafood is a human health concern. Signs warning of the health risks of consuming coastal mussels in this area and discouraging their collection should be posted in areas where these bivalves are easily accessible (e.g. Hoedjiesbaai).

Water samples collected from 20 stations in Saldanha Bay and Langebaan Lagoon are collected and analysed fortnightly for faecal coliform and *E. coli* concentrations courtesy of the SBWQFT and the West Coast District Municipality (WCDM). The microbial monitoring program provides evidence that water quality, from a recreational use perspective has improved at sites near the Bok River mouth but remain a cause for concern at the Hoedtjies Bay beach. With respect to mariculture, the situation in Small Bay remains a concern, with all the sites sampled along the northern shore exceeding the guideline for safe mariculture practices. Faecal coliform counts at all four sites in Big Bay were within the 80^{th} percentile limits for mariculture in 2020 and the Langebaan sites all met recreational water quality standards (and have done so for the at least the last decade at most sites).

The older DWAF water quality guidelines for recreational use have been revised following an international review of guidelines for coastal waters, which highlighted several shortcomings in those developed by South Africa. The revised guidelines (DEA 2012) are based on counts of intestinal *Enterococci* and *E. coli* and require that both types of bacteria be enumerated at least every two weeks. It is highly recommended that enumeration of Enterococci be included in the Saldanha water sampling programme in place of faecal coliforms as several studies have shown faecal coliforms and *E.coli* to be relatively poor indicators of health risks in marine waters. These organisms are also less resilient than Enterococci (and other pathogenic bacteria) so if analysis is focussed on coliforms, risk can be underestimated due to mortality occurring in the time taken between collection and analysis. Guidelines state that samples should be collected 15-30 cm below the surface, on the seaward side of a recently broken wave. Samples to be tested for *E. coli* counts should be analysed within 6-8 hours of collection, and those to be tested for intestinal Enterococci, within 24 hours. Analyses should be completed by an accredited laboratory, preferably one with ISO 17025 accreditation.

14.4 Sediments

Beach erosion in Saldanha Bay, particularly at Langebaan Beach, has been the subject of some concern in recent years as coastal developments in Langebaan and Saldanha extend right to the water's edge and are at risk from a retreating shoreline. The origins of this erosion are not clear but most experts feel that it is likely due to natural causes that may have been exacerbated as a result of the construction of the iron ore terminal and associated infrastructure (Marcus Island causeway). A number of interventions have been introduced over the years in an effort to control the erosion, and to limit the loss of sediment from these beaches. This includes construction of rock revetments along Langebaan Beach (1997-2002), construction of groynes extending perpendicularly out from the shore at Langebaan Beach (2004-2008) and construction of gabion walls on Paradise Beach. The Saldanha Bay Municipality initiated an erosion monitoring programme in 1994, designed to monitor change (erosion/accretion) in the beaches between Leentjiesklip 1 (Strandloper restaurant) and Alabama street. This entails undertaking beach surveys bi-annually - at the end of summer (Apr/May) and the end of winter (Oct/Nov) during spring low tide. This programme continues but has now been taken over by the SBWQFT. Data from this monitoring programme indicates that the seasonal patterns of erosion and accretion are complex and are to some extent reversed for the northern and southern portions of Langebaan Beach. Also, very clear in the long-term data, is the impact of the various interventions that were introduced to mitigate or control erosion on these beaches, particularly the two groynes. Overall, data suggest that the construction of the groynes was a very necessary and successful intervention and that it is very important that monitoring continue in future to confirm that this pattern continues going forwards. Additional interventions to enable reestablishment of beach habitat along the shoreline on the section north of the area currently being monitored is probably also warranted, as the shoreline here is currently made up of a rock revetment which makes access to the sea challenging for people living in this area.

Trace metal levels are mostly well within safety thresholds except for a few sites in Small Bay, where thresholds have been exceeded on a number of occasions between 2016 and 2020. In this year's survey, cadmium concentrations were noticeably high in Langebaan Lagoon of which one site even exceeded the toxicity threshold. Key areas of concern regarding trace metal pollution within Small Bay include the Yacht Club Basin, where cadmium and copper have exceeded recommended thresholds for five years in a row and enrichment factors (EF) continue to be high, as well as adjacent to the Multi-Purpose Terminal where levels of cadmium and lead are below internationally accepted guidelines, but still remain highly enriched relative to historic levels. Recent increases in the concentration of manganese around the Iron Ore Terminal are also a little concerning. Regular monitoring of trace metal concentrations is thus strongly recommended to provide an early warning of any future changes.

Sediment monitoring (particle size, total organic carbon (TOC), total organic nitrogen (TON) and trace metals) should continue to be conducted annually at the same suite of stations that have been monitored since 1999 along with additional stations added since this time (e.g. those in Langebaan Lagoon) when budg*et* allows. When budgetary constraints are in place, as in 2016, a sub-set of sites in Small Bay and Big Bay should continue to be monitored so that continuity in monitoring high impact areas is maintained. Dredging in the Bay should be avoided, if possible, and appropriate precautions need to be taken when dredging becomes necessary to ensure that suspended trace metals do not contaminate cultured and wild seafood in the Bay.

Sediment samples collected in 2020 had low Poly-aromatic hydrocarbons (PAH) levels across all sites. While the Total Petroleum Hydrocarbon (TPH) and PAH findings present no major concern, it is recommended that TPH monitoring within the vicinity of the iron ore terminal is continued annually in order to identify the frequency of occurrence of pollution incidents; like that recorded in 2014, and assess the ecological implications to the Bay.

14.5 Macrophytes

Three distinct intertidal habitats exist within Langebaan Lagoon: seagrass beds, such as those of the eelgrass *Zostera capensis;* salt marsh dominated by cordgrass *Spartina maritime* and *Sarcocornia perennis* and the dune slack rush *Juncus kraussi*, and unvegetated sandflats dominated by the sand prawn, *Callianassa krausii* and the mudprawn *Upogebia capensis* (Siebert & Branch 2005). The other major vegetation type present in the upper lagoon area, particularly where groundwater inflow occurs, are reed beds dominated by *Phragmites australis.*

The loss of seagrass beds from Langebaan Lagoon is a strong indicator that the ecosystem is undergoing a shift, most likely due to anthropogenic disturbances. Additionally, several studies have highlighted the potential for climate driven changes in water temperature and pH to alter seagrass physiology and possibly their distribution and abundance (Duarte 2002, Mead *et al.* 2013). However, information on the temperature and pH tolerance of South African seagrasses is currently lacking and warrants investigation. It is critical that this habitat and the communities associated with it be monitored in future as further reductions are certain to have long term implications, not only for the invertebrate fauna but also for species of higher trophic levels.

Saltmarshes in Langebaan are an important habitat and breeding ground for a range of fish, bird and invertebrate species (Christie 1981, Day 1981, Gericke 2008). Langebaan Lagoon incorporates the second largest salt marsh area in South Africa, accounting for approximately 30% of this habitat type in the country, being second only to that in the Knysna estuary (Adams *et al.* 1999). Long term changes in salt marshes in Langebaan Lagoon were investigated by Gericke (2008) using aerial photographs taken in 1960, 1968, 1977, 1988 and 2000. He found that overall saltmarsh area had shrunk by only a small amount between 1960 and 2000, losing on average 8 000 m^2 per annum.

The common reed *Phragmites australis* dominates the flora of the reedbeds where groundwater inflow occurs. Results of our analysis suggests that variation in reed cover over time is relatively modest and that this has remained more or less constant over the last 31 years (1989-2020, Figure 8.7). The biggest perturbations in reed cover correspond with the two largest droughts that have been experienced in the region in this period (a 1:20 year event that occurred in the period 2002-2003) and an even bigger drought that occurred recently (a 1:100 year event in the period 2015-2017). Also of interest is the fact that the extent of reed cover around Langebaan Lagoon as indicated by the satellite data is significantly greater than that inferred from aerial photography (i.e. as delineated by van der Linden 2013, Figure 8.8 - Figure 8.10).

Future efforts in this field will entail expanding this assessment to other vegetation classes (specifically seagrass and salt marsh), assessing the level of change in each vegetation class over time, and also ground truthing of each mapped vegetation class.

14.6 Benthic macrofauna

Monitoring of benthic macrofaunal communities over the period 1999-2020 has revealed a relatively stable situation in most parts of the Bay and Lagoon with the exception of 2008 when a dramatic shift in benthic community composition occurred at all sites. This shift involved a decrease in the abundance and biomass of filter feeders and an increase in shorter lived opportunistic detritivores. This was attributed to the extensive dredging that took place during 2007-2008. Aside from this Bay-wide phenomenon, localised improvements in health have been detected in the Yacht Club Basin and at Salamander Bay following construction of the boat dock. However, disturbance at the LPG site in Big Bay has resulted in reduced indices of abundance, biomass and diversity since the installation of the SPM at this site. Although highly localised, the negative impact of this development on the benthic macrofaunal community is clearly significant. Future monitoring of these indices at this site is important in order to gauge recovery in the benthos. Notable improvements in the health of benthic communities include the return of the suspension feeding sea-pen *Virgularia schultzei* to Big Bay and Langebaan Lagoon since 2004, as well as an increase in the percentage biomass of large, long lived species such as the tongue worm *Ochaetostoma capense*, and several gastropods. Certain areas of Small Bay that experience reduced water circulation patterns in (e.g. near the Small Craft Harbour and near mussel rafts) which results in the accumulation of fine sediment, organic material and trace metals (aggravated by anthropogenic inputs) still have impoverished macrofaunal communities. Results from samples collected in the vicinity of the Sea Harvest discharge pipe indicate a marked improvement on those from 2017. This would suggest that the relocation of the discharge outfall was justified and has resulted in a notable improvement in benthic ecosystem health. In order to ensure the continued improvement in the health of the Small Bay marine environment it is recommended that stringent controls are placed on the discharge of effluents into Small Bay to facilitate recovery of benthic communities and ecosystem health as a whole. The regularity (annually) and intensity of benthic macrofauna monitoring should continue at all of the current stations.

14.7 Rocky intertidal

A total of 100 taxa were recorded from the eight study sites, most of which had been found in previous survey years. The faunal component was represented by 23 species of filter-feeders, 23 species of grazers, and 14 species of predators and scavengers combined. The algal component comprised 24 corticated (foliose) seaweeds, 9 ephemerals, 5 species of encrusting algae, and 2 species of kelp. In general, rocky shore communities have remained relatively stable with only minor changes over the years.

One of the greatest threats to rocky shore communities in Saldanha Bay is the introduction of alien species via shipping and mariculture, and their potential to become invasive. Key changes in the rocky intertidal ecosystem reflect the regional invasion by the Mediterranean mussel *Mytilus galloprovincialis* and the North American barnacle *Balanus glandula* which compete for space on most of the rocky intertidal substrata in the Bay at the expense of native species. Their spread throughout the Bay has significantly altered natural community structure in the mid and lower intertidal, particularly in wave exposed areas. In 2019, *Balanus glandula* abundance was lower than in previous years with only empty shells and base plate scars left on rocks at some sites. In this year’s survey, the abundance of this barnacle had increased slightly which suggests that low abundance values in previous years may have been due to a decrease in in larval supply, but it is still unclear. The establishment of new alien species can potentially have negative impacts on native rocky shore species, and it is important that this is monitored closely through continued rocky shore surveys.

14.8 Fish

Long-term monitoring of juvenile fish assemblages by means of experimental seine-netting in the surf zone has revealed some concerning trends. A significant decline in white stumpnose abundance at all sites over the last decade suggests that the protection afforded by the Langebaan MPA was not enough to sustain the fishery at the high effort levels. A recent analysis of commercial and recreational linefish catch data and the net survey data by a team of fisheries scientists strongly recommends the implementation of additional harvest control measures, namely a reduction in the bag limit to 5 fish person^{-1} day^{-1} and an increase in the minimum size (to 30 cm TL). It is also recommended that monitoring of fish stocks, catch and effort in the Bay be intensified, and that an economic study be undertaken to assess the value of the recreational fishery and the impacts of different management options.

In the data set collected to date, the average density of commercially important fish, such as white stumpnose and harders, was much higher at Small Bay sites compared to Big Bay and Lagoon sites. Since 2011, however, estimated densities of these species were similar and low in both Big Bay and Small Bay. The juveniles of other species were historically also more abundant in Small Bay. This gives an indication of the importance of Small Bay as a nursery habitat for the fish species that support the large fisheries throughout the Bay. Small Bay is often viewed as the more developed or industrialized portion of the Bay and is considered by many as a 'lost cause'. These data provide a strong argument to stamp out such negative thinking and to continue lobbying strongly for ecologically sound management of this portion of the Bay. The collapse of the white stumpnose stock and continued poor recruitment throughout the Bay makes it even more critical that the quality of what is demonstrably the most important white stumpnose nursery habitat is improved.

The 2018 discovery of alien rainbow trout in Kraalbaai (almost certainly escapees from the pilot fish cage farming in Big Bay) is another threat to the indigenous fish fauna in the region. These predatory fish will prey on indigenous invertebrates and fish and ongoing introductions could cause ecosystem level impacts. These alien fish are highly unlikely to establish self-sustaining populations in the bay and lagoon due to the lack of suitable spawning habitat (cool, clear freshwater rivers) in the region. At the current experimental scale of fish farming, the number of escapees is not expected to be having significant impacts on indigenous fauna. However, at the proposed commercial scale finfish cage farming, the number of alien salmonids introduced into the Bay and the Lagoon via ongoing escapes will probably have significant negative effects on indigenous fauna. Given the importance of the nearshore waters of Saldanha Bay and Langebaan lagoon as nursery areas for a number of vulnerable indigenous fishery species, finfish cage farming should be restricted to the outer Bay, and mitigation measures to minimise escapes from cages should be strictly enforced.

Fish sampling surveys should be conducted annually at the same sites selected during the 2020 study for as long as possible. This sampling should be confined to the same seasonal period each year for comparative purposes.

14.9 Birds and seals

Together with the five islands within the Bay and Vondeling Island slightly to the South, Saldanha Bay and Langebaan Lagoon provide extensive and varied habitat for waterbirds. This includes sheltered deepwater marine habitats associated with Saldanha Bay itself, sheltered beaches in the Bay, islands that serve as breeding refuges for seabirds, rocky shoreline surrounding the islands and at the mouth of the Bay, and the extensive intertidal salt marshes, mud- and sandflats of the sheltered Langebaan Lagoon.

Saldanha Bay and particularly Langebaan Lagoon are of tremendous importance in terms of the diversity and abundance of waterbird populations supported. At least 56 non-passerine waterbird species commonly use the area for feeding or breeding; eleven species breed on the islands of Malgas, Marcus, Jutten, Schaapen and Vondeling alone. These islands support nationally important populations of African Penguin, Cape Gannet, Swift Tern, Kelp and Hartlaub's Gull, and four species of marine cormorant, as well as important populations of the endemic African Oystercatcher. The lagoon is an important area for migratory waders and terns, as well as for numerous resident waterbird species. Waterbirds are counted annually on all the islands (Department of Environmental Affairs: Oceans and Coasts), and bi-annually in Langebaan Lagoon (Avian Demography Unit of the University of Cape Town).

Declines in the numbers of seabirds breeding on the Saldanha Bay Islands can be attributed to a number of causes. These include (1) emigration of birds to colonies further south and east along the South African coast in response to changes in the distribution and biomass of small pelagic fish stocks, (2) starvation as a result of a decline in the biomass of sardines nationally, and particularly along the west coast over the last decade, (3) competition for food with the small pelagic fisheries within the foraging range of affected bird species, (4) predation of eggs, young and fledglings by Great White Pelicans, Kelp Gulls and Cape Fur Seals, and (5) collapse of the West Coast Rock Lobster stock upon which Bank Cormorants feed. However, because populations are so depressed, conditions at the islands in Saldanha, particularly predation by Cape Fur Seals and Kelp Gulls, have now become the major factors in driving current population decreases for many seabird species. Direct amelioration actions (*Pelican Watch*, problem seal culling) to decrease these impacts at the islands have had mixed results, with the former proving more effective than the latter. Cape Fur Seal and Kelp Gull predation continue to pose a major threat to seabird survival at the Saldanha Bay Island colonies. Current conservation initiative must continue to protect seabird populations in Saldanha Bay.

Decreasing numbers of migrant waders utilising Langebaan Lagoon reflects a global trend, which can be attributed to loss of breeding habitat and hunting along their migration routes as well as human disturbance and habitat loss on their wintering grounds. In Langebaan Lagoon, drastic population declines in four species, including the Ruddy Turnstone, Red Knot, Grey Plover, and Curlew Sandpiper signified this downward trend in summer migratory bird numbers. Most importantly, Curlew Sandpiper numbers have dropped from a pre-1990 average of just over 20 000 birds to 1 335 birds in 2019. Prior to 1990, this species accounted for almost two thirds of the total summer migratory wader numbers in the lagoon. Shrinking wader populations at Langebaan Lagoon are primarily signified by declining populations of a handful of migratory species. Conservation research and efforts should be prioritised for these species and conducted on international scale.

Locally, unfavourable conditions persisting in Langebaan Lagoon as a result of anthropogenic impacts should also be managed more effectively to protect resident and migratory waders that do arrive in the lagoon. It is highly recommended that the status of coastal and wading bird species continues to be monitored and that these data are used to inform and assesses the efficacy of management interventions aimed at halting the observed declines and supporting recovery of the regions birds.

Cape Fur Seals are amongst the largest marine top predators found in and around Saldanha Bay. They are opportunistic, generalist feeders that have been shown to benefit from human activities including utilisation of discards from fishing boats, or taking fish directly from fisherman. In addition, seals compete with seabirds, such as penguins and gannets, as well as with commercial fisheries, for small pelagic fish which form a key part of their diets. It has been suggested that the increasing numbers of seals on Vondeling island may lead to increased pressure to cull seals both from a fisheries perspective as well as to protect important seabird species on which seals are known to prey. Concerns have also been raised that, with the increased number of seals along the shores surrounding Saldanha Bay and with the addition of finfish aquaculture in the Bay, seal numbers within the Bay will likely increase, along with the occurrence of problem seals.

Although seals are likely attracted to the aquaculture sites within Saldanha Bay, chances are that their numbers will not continue to increase significantly as they are restricted to sub-adult males. Additionally, the carrying capacity of Vondeling Island appears to have been reached and the overall population within Southern Africa has remained stable over the last 30 years.

14.10 Alien invasive species

A recent update on the number of alien marine species present in South Africa reports 95 alien species as being present in this country, of which 56 are considered invasive i.e. populations are expanding and consequently displacing indigenous species (Robinson *et al.* 2016). With the recent addition of five new species – the barnacle *Perforatus perforatus* (*Biccard & Griffiths pers. comm. 2017*), the Japanese skeleton shrimp *Caprella mutica* (Peters & Robinson 2017), the South West African porcelain crab, *Porcellana africana* (Griffiths *et al.* 2018), the South American sunstar *Heliaster helianthus* and the Chilean stone crab *Homalaspis plana* (Peters & Robinson 2018) – 29 species are currently confirmed from Saldanha Bay and/or Langebaan Lagoon. All of these, except *H. helianthus*, *H. plana*, *P. perforatus* and the previously reported anemone *Sagartia ornata*, are considered invasive.

Other noteworthy invasive alien species that are present in Saldanha Bay include the Mediterranean mussel *Mytilus galloprovincialis*, the barnacle *Balanus glandula*, the Pacific mussel *Semimytilus algosus* and the Western pea crab *Pinnixa occidentalis.* The abundance of *M. galloprovincialis* on rocky shores in Saldanha Bay has been decreasing over the last few years, even to such an extent that no mussels were detected at certain sites. The reason behind this decline is, however, still not clear although numerous factors might be at play. No trend in the abundance of *B. glandula* over time is evident, although this species has shown a decrease in percentage cover at almost all sites, with no barnacles being reported from certain sites during the 2020 survey. No conclusive trend in the spread and site preference of the Western pea crab *P. occidentalis* could be established, although it does seem to flourish in deeper water habitats and occurs at lower densities close to the iron ore and multi-purpose terminals. Despite abundance and dominance of the alien crab fluctuating quite substantially at certain sites in Big and Small Bay over time, and has shown a decrease during the 2020 survey, data suggest that *P. occidentalis* is well established in the Bay. Crab populations have also shown an increase in Langebaan Lagoon, although it is restricted to only one site. The status of this crab within Danger Bay is currently not confirmed and more sampling efforts are thus needed at this site.

The discovery of five new alien species over the past six years raises concern and highlights the need for management action. An additional 19 species are currently regarded as cryptogenic in Saldanha Bay and/or Langebaan Lagoon, although comprehensive genetic analyses are urgently required to determine the definite status of these cryptogenic species.

Managing alien species within the marine environment is challenging, costly and time consuming. To ensure the efficient use of resources and desirable outcomes, management actions should be focused firstly, on managing invasive species already present in Saldanha Bay and secondly, on preventing further invasions. Both strategies present their own advantages and limitations. Research is very important for understanding biological invasions and the best way to approach their management. However, the knowledge gained from research needs to be shared with stakeholders and policy makers to implement appropriate management strategies and inform action (Foxcroft *et al*. 2020). Von der Heyden *et al*. (2016) proposed an evidence-based co-creation approach, where scientists and policy makers work together and share knowledge to inform appropriate management actions.

To ensure that resources are used efficiently, and alien species managed effectively, there is a need to know which species should be prioritised for management. One approach that has been proposed to determine this, is ranking, and managing alien species based on their impacts. Alien species are considered to represent one of the greatest threats to rocky shore communities in Saldanha Bay, owing to their potential to become invasive, and displace naturally occurring indigenous species. The Environmental Impact Classification for Alien Taxa (EICAT) ranks species based on their impacts (Blackburn *et al.* 2014), allowing management to be prioritised towards those posing the highest risk. Although it has been suggested that the EICAT be used to prioritise species of concern in Saldanha Bay, the use of this approach within the marine context is limited. This is because it depends upon knowledge of species-specific impacts in a particular area, information which is not always readily available for the majority of marine species. In South Africa, for example, impacts have only been quantified for 16% of the species known to be alien (Alexander *et al.* 2016; Robinson *et al.* 2016). In addition, impacts are context dependent and as such, impacts in one area cannot be used to infer impacts in another (Kumschick *et al.* 2014; Robinson *et al.* 2017). Considering this, studies investigating the impacts of these species in Saldanha Bay are desperately needed to prioritise management actions.

A pre-cautionary approach to prevent biological invasions is often considered the most efficient method of management. Watchlists have been identified as a useful preventative measure in the management of alien species (Faulkner *et al.* 2017). They identify species of concern that are not yet found in an area, but have the potential to arrive and establish. Watchlists are created based on a variety of factors, which include selecting species with an invasion history, pathways to the area of concern, occurring in similar climatic regions or those with biological traits that could predispose them to becoming successful invaders. Watchlists should be used together with routine monitoring, as this will increase the chances of early detection and successful eradication. Unfortunately, the lack of basic biological knowledge of species, even for large, conspicuous invaders, pose an impediment to creating such watchlists based on species traits (Swart *et al.* 2018). This further highlights the need for studies investigating the traits of alien species. In the absence of such information, invasion history, in combination with climatic matching and available pathways should be used to create watchlists.

Another pre-cautionary approach includes identifying and managing important pathways of introduction. A notable 91% of marine introductions to South Africa have been associated with shipping activities such as ballast water discharge and hull fouling. In addition, 50 of the reported 95 alien species are confined to sheltered areas such as harbours. These findings emphasise the importance of shipping as a pathway of introduction (Robinson *et al*. 2020) and further highlights the need for implementing more efficient protocols of port control to monitor vessels entering South African harbours, the treatment of hull fouling and ballast water before entering and the regular monitoring of harbours for alien species. This approach should be improved upon in Saldanha Bay, especially since it is such an important international port with high shipping traffic from around the globe. The presence of numerous alien species in Saldanha Bay with the same native range, highlights the risk of introduction from specific areas and pathways linked to Saldanha Bay. Furthermore, recreational yachts have been recognised as an important vector for the introduction of alien species in South Africa (Peters *et al.* 2019), although this vector has been overlooked in the past. International yachts frequent Saldanha Bay, thereby posing a high risk of introducing marine alien species. Unfortunately, there is currently no regulatory measures to manage alien species fouling on and potential introduction into South Africa by yachts. A recent study by Keanly and Robinson (2020) explored encapsulation as a tool to minimise and manage fouling on such vessels. This study showed promising results and similar experiments are encouraged for Saldanha Bay.

In conclusion, management actions should be focused on managing invasive species already present in Saldanha Bay by rating species based on their impacts. Secondly, efforts should also be focused on preventing further invasions. Watchlists have been identified as a useful preventative measure and are created based on selecting species with an invasion history, pathways to the area of concern, occurring in similar climatic regions or those with biological traits that could predispose them to becoming successful invaders. Another vital aspect includes identifying and managing important pathways of introduction. This should be done in combination with port control to monitor vessels entering harbours, treatment of hull fouling and ballast water before entrance and the regular monitoring of harbours. All these efforts require in depth research. This will not only contribute towards our understanding of the drivers and traits governing successful invasions, but also give insight into associated impacts and so support directed management actions to successfully control invasions and mitigate impacts.

14.11 Summary of environmental monitoring results

In summary, the environmental monitoring currently implemented in Saldanha Bay and Langebaan Lagoon (e.g. sediment, benthic macrofauna, birds, rocky intertidal, fish populations) should continue with some small adjustments or additions, however, monitoring of other environmental parameters that are not currently assessed on a regular basis (e.g. temperature, oxygen, salinity, stormwater quality) require structured, maintained monitoring to be implemented.

Table 11.1. Tabulated summary of Environmental parameters reported on in the State of the Bay: Saldanha Bay and Langebaan Lagoon.

Parameter monitored	Time period	Anthropogenic induced impact	Rating
GROUND WATER			
Aquifer and Lagoon: Physical aspects (extraction rates, volumes, recharge rates, volumes, temperature, salinity, tidal height, rainfall)	1984-2020	The total utilisable groundwater exploitation potential (UGEP) under normal conditions for the Saldanha Bay Management area is estimated at 15.2 Mm3/a which is greater than the current estimated levels of exploitation (8.33 Mm3/a). However, it is recommended that Managed Aquifer Recharge methodologies are employed in this area and that the aquifer only be used in times of severe drought and be kept as "full" as possible in non-drought times. Ongoing monitoring of water use from and water levels in the aquifer is considered essential as is the establishment of baseline conditions in Langebaan Lagoon.	
WATER QUALITY			
Physical aspects (temperature, salinity, dissolved oxygen, nutrients and chlorophyll)	1974-2000, 2010-2011, 2014-2020	Dissolved oxygen levels in bottom water in Small Bay are very much lower than they were historically or at least prior to port development. Dissolved oxygen concentration is lower in Small Bay than in Big Bay and measurements below bivalve farms are consistently lower than at control sites. This is attributed to organic loading in Small Bay and reduced flushing time. No consistent changes are evident with any other physico-chemical parameters. Anomalous water column temperature profiles (cooler water) were recorded during 2017 and 2018, corresponding with the dominance of the South Atlantic High-Pressure system during the prolonged drought.	
Current circulation patterns and current strengths	1975 vs. 2020	Reduced wave energy, and impaired circulation and rate of exchange in Small Bay. Increased wave action in parts of Big Bay and at Langebaan Beach causing coastal erosion. Increased current strength alongside obstructions (e.g. ore terminal).	
Microbiological (faecal coliform)	1999-2020	Faecal coliform counts in Small Bay frequently exceed guideline levels and although there have been improvements at some sites, others remain a concern. Big Bay and Langebaan Lagoon mostly remain within safety levels for faecal coliform pollution. However, faecal coliform may underestimate actual harmful microbiological concentrations. There is a need to monitor intestinal *Enterococci* as well.	

Parameter monitored	Time period	Anthropogenic induced impact	Rating
Trace metal contaminants in water	1997-2008, 2014-2020	Concentrations of lead in mussel flesh collected in research samples are consistently above the safety guidelines for food stuffs (this is not the case with farmed mussels). The source of the lead contamination should be investigated and addressed, and any future dredging events should be limited as far as possible owing to the likely mobilization of trace metals from sediments.	
SEDIMENTS			
Shoreline Stability	1994 - 2020	Erosion exacerbated by construction of iron ore terminal and Marcus Island causeway. Benefits are achieved by interventions to rehabilitate shoreline habitat. More beach areas should be considered for intervention.	
Particle size (mud/sand/gravel)	1974-2020	The mud fraction in the sediments in the Bay was highly elevated when the State of the Bay surveys commenced in 1999 relative to the period prior to port construction. The situation has improved considerably since this time at most sites.	
Total organic carbon (TOC)	1974-2020	Elevated levels of TOC at the Yacht Club basin and near the mariculture rafts (negative impacts) are of particular concern.	
Total organic nitrogen (TON)	1974-2020	Similar trends as for TOC. Elevated levels of TON at the Yacht Club basin and near the mariculture rafts (negative impacts) are of particular concern.	
Trace metal contaminants in sediments	1980-2020	Cadmium, lead, and copper are currently elevated considerably above historic levels. Concentrations were highest in 1999 following major dredge event. Lead, copper and nickel elevated in 2008-2016, whereas cadmium and copper increased in 2019 at Yacht Club and multi-purpose terminal, which may be related to shipping activities and maintenance dredging. However, these concentrations have declined in the current 2020 survey.	
MACROPHYTES			
Seagrasses, salt marsh, reeds and sedges	2016-2018	Seagrass (*Zostera capensis*) beds have experienced a radical reduction in size with associated fragmentation of large beds. This phenomenon has been attributed to direct and indirect anthropogenic changes such as physical disturbance, pollution, specifically eutrophication and most recently, seagrass biomass was found to be lower in warmer waters. Analyses of changes in saltmarsh cover indicate a similar but more modest (~8%) reduction in cover for this vegetation type. Reeds and sedge cover, by contrast, appears to have remained more or less constant over the last 30 years.	
BENTHIC MACROFAUNA			
Species abundance, biomass, and diversity	1999-2020	Benthic macrofaunal communities in Saldanha Bay and Langebaan Lagoon Bay are highly sensitive to dredging activities and drop dramatically immediately after each major dredging event. Macrofaunal communities are currently increasing in abundance and biomass since the last major event in 2008.	

Parameter monitored	Time period	Anthropogenic induced impact	Rating
ROCKY INTERTIDAL AND INTRODUCED SPECIES			
Impact of alien mussel and barnacle introductions	1980-2020	Alien mussel and barnacle have displaced the local mussel and other native species from much of the shore leading to decreased species diversity (negative). One new alien barnacle species found in 2014. The establishment of this species must be closely monitored.	
FISH			
Community composition and abundance	1986-2020	White stumpnose abundance and fishery landings have declined dramatically over the last decade. Abundance and diversity of fish in the Bay (e.g. elf and gobies) have also been declining in recent years, and this is of some concern.	
BIRDS			
Population numbers of key species in Saldanha Bay and islands	1977-2019	Populations of many seabirds breeding on the Saldanha Bay Islands are declining rapidly. This trend is attributed to: (1) emigration of birds to colonies further south and east along the South African coast in response to changes in the distribution and biomass of small pelagic fish stocks, (2) predation of eggs, young and fledglings by Great White Pelicans, Kelp Gulls and Cape Fur Seals; (3) starvation as a result of a decline in the biomass of sardines nationally, and particularly along the west coast over the last decade, (4) competition for food with the small pelagic fisheries within the foraging range of affected bird species, and (5) collapse of the West Coast Rock Lobster stock upon which Bank Cormorants feed	
Population numbers of key species in Langebaan Lagoon	1976-2020	Populations of migrant waders utilising Langebaan Lagoon have decreased dramatically over the last 30 years, attributed to offsite impacts on breeding grounds and local impacts (habitat changes) and disturbance in the lagoon. Numbers of resident waders have also declined and is likely due to changes in the lagoon itself, particularly increased disturbance at historically important, feeding sandflats.	
SEALS	1970 -2020	Cape Fur seal population stable over the past 30 years.	
ALIEN AND INVASIVES			
Total number of alien and invasive species in Saldanha Bay and Langebaan Lagoon	Current 2020	Twenty-nine species have been confirmed from Saldanha Bay and/or Langebaan Lagoon, of which all but four are considered invasive.	
Acorn barnacle *Balanus glandula*	2010-2020	No trend in the abundance of *B. glandula* over time, although it has shown a decrease at almost all sites, with no barnacles reported from certain sites during the 2020 survey. However, it remains one of the more abundant species on the mid-shore in Saldanha Bay and is still of significant concern.	
European mussel *Mytilus galloprovincialis*	2005-2020	Has shown a decreasing trend in abundance at some sites over the last years. Nevertheless, it remains one of the most abundant species on the mid and low shore at exposed sites in Saldanha Bay and is still of significant concern.	

Parameter monitored	Time period	Anthropogenic induced impact	Rating
Western pea crab *Pinnixia occidentalis*	2004-2020	No conclusive trend in the spread and site preference of this crab, although it has shown a decrease at certain sites during the 2020 survey. This crab is well established in the Bay and has shown an increase in Langebaan Lagoon	

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16 APPENDIX 1

The Chapter contains supplementary information for the Groundwater, Rocky Intertidal, Bird and Seals, and Alien and Invasive Species Chapters in graph format (Figure 16.1- Figure 16.6) and in table format (Table 16.1-Table 16.3).

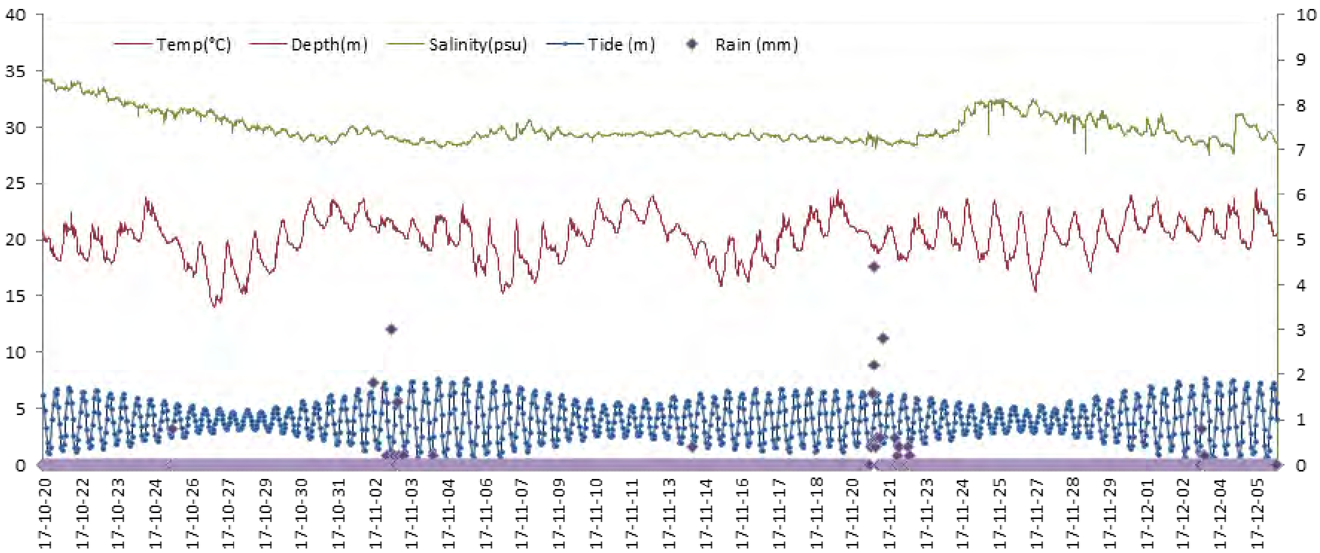

Figure 16.1. **Salinity (green line, left axis, units PSU), temperature (red line, left axis, units °C) in twenty minute intervals over a 3 month period in October 2017 through December 2017. Tidal data (dark blue points, right axis, units m) and rain (purple, right axis, units mm) are in hourly intervals over the same time period (tidal data provided by hydrographer, SANHO, rain data by WeatherSA).**

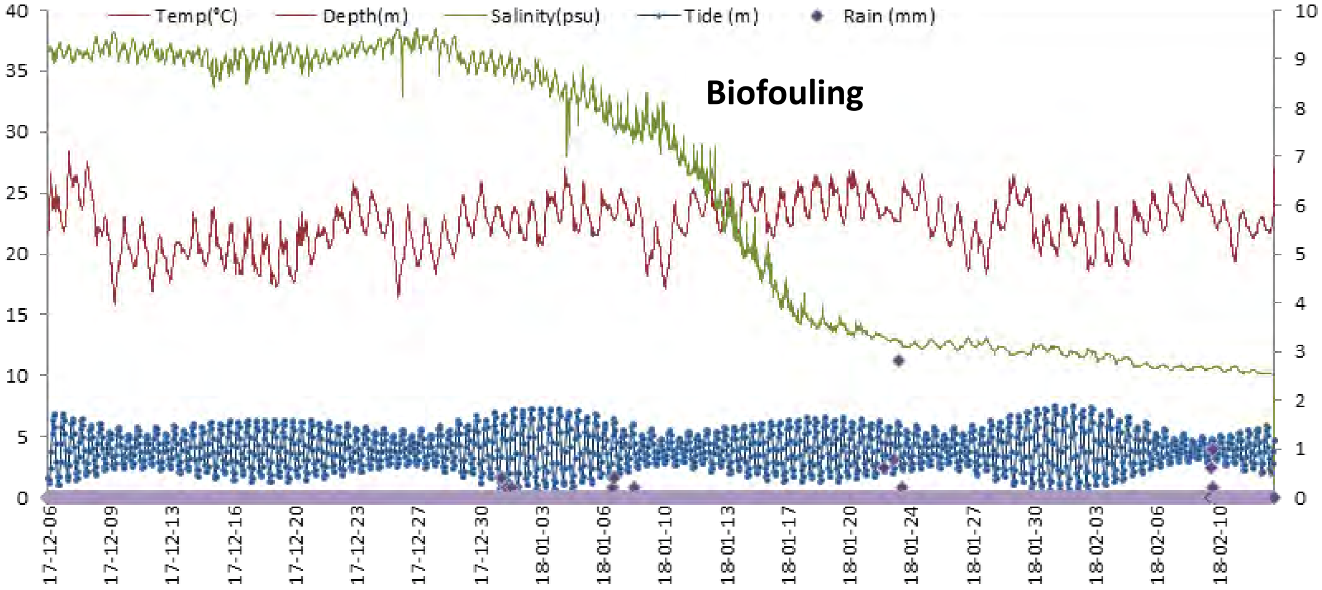

Figure 16.2. **Salinity (green line, left axis, units PSU), temperature (red line, left axis, units °C) in twenty minute intervals over a 3 month period in December 2017 through February 2018. Tidal data (dark blue points, right axis, units m) and rain (purple, right axis, units mm) are in hourly intervals over the same time period (tidal data provided by hydrographer, SANHO, rain data by WeatherSA).**

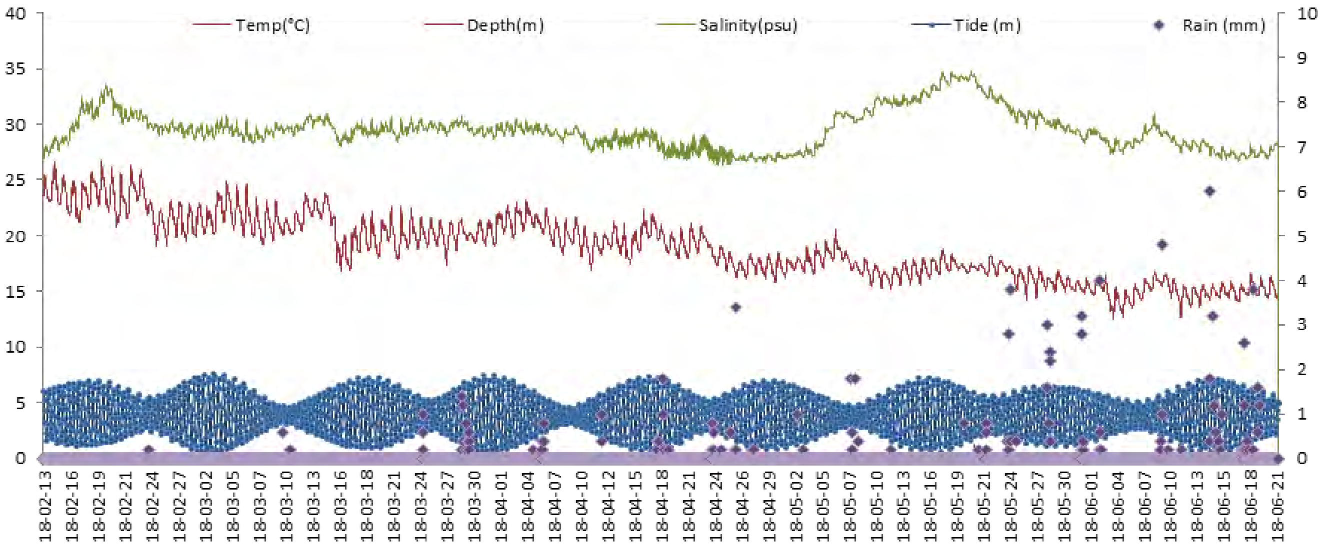

Figure 16.3. **Salinity (green line, left axis, units PSU), temperature (red line, left axis, units °C) in twenty minute intervals over a 5 month period in February 2018 through June 2018. Tidal data (dark blue points, right axis, units m) and rain (purple, right axis, units mm) are in hourly intervals over the same time period (tidal data provided by hydrographer, SANHO, rain data by WeatherSA).**

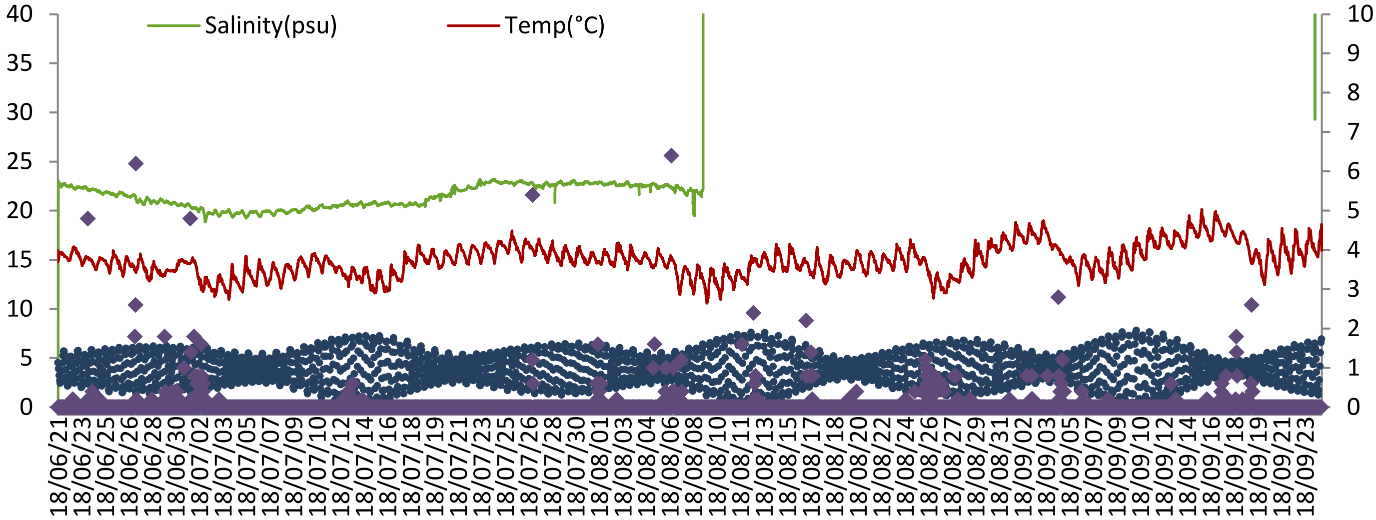

Figure 16.4. **Salinity (green line, left axis, units PSU), temperature (red line, left axis, units °C) in twenty minute intervals over a 4 month period in June 2018 through September 2018. Tidal data (dark blue points, right axis, units m) and rain (purple, right axis, units mm) are in hourly intervals over the same time period (tidal data provided by hydrographer, SANHO, rain data by WeatherSA).**

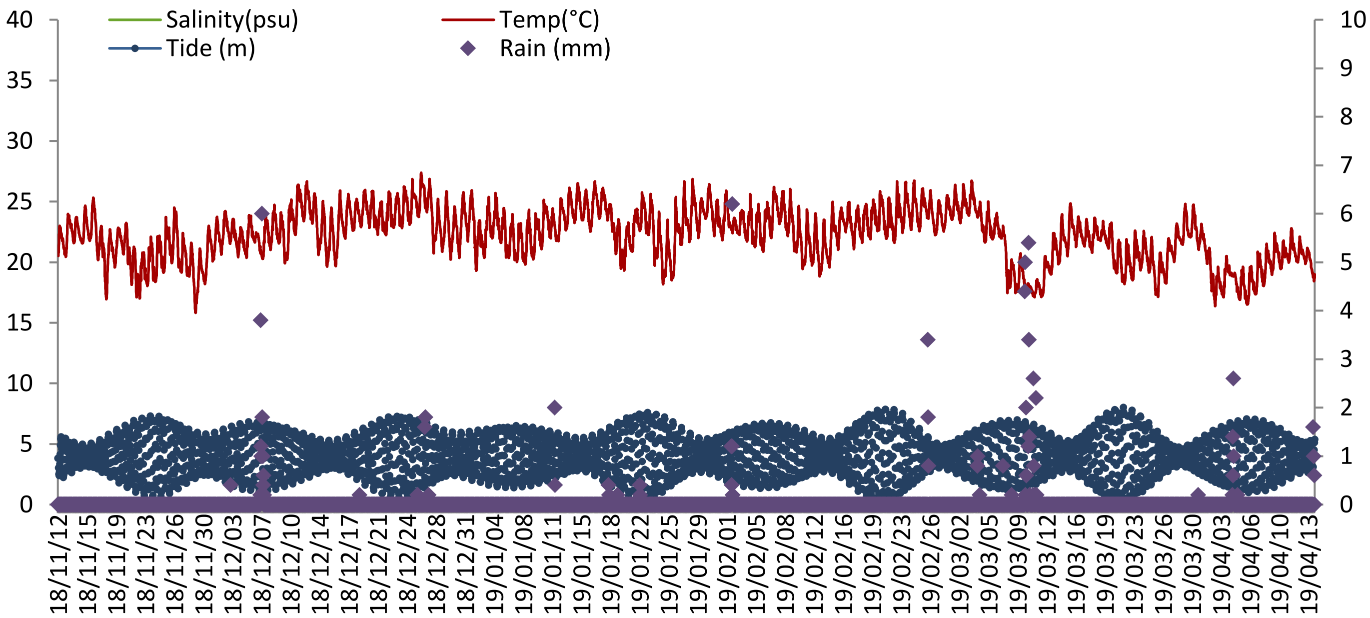

Figure 16.5. **Salinity (green line, left axis, units PSU), temperature (red line, left axis, units °C) in twenty minute intervals over a 6 month period in November 2018 through April 2019. Tidal data (dark blue points, right axis, units m) and rain (purple, right axis, units mm) are in hourly intervals over the same time period (tidal data provided by hydrographer, SANHO, rain data by WeatherSA).**

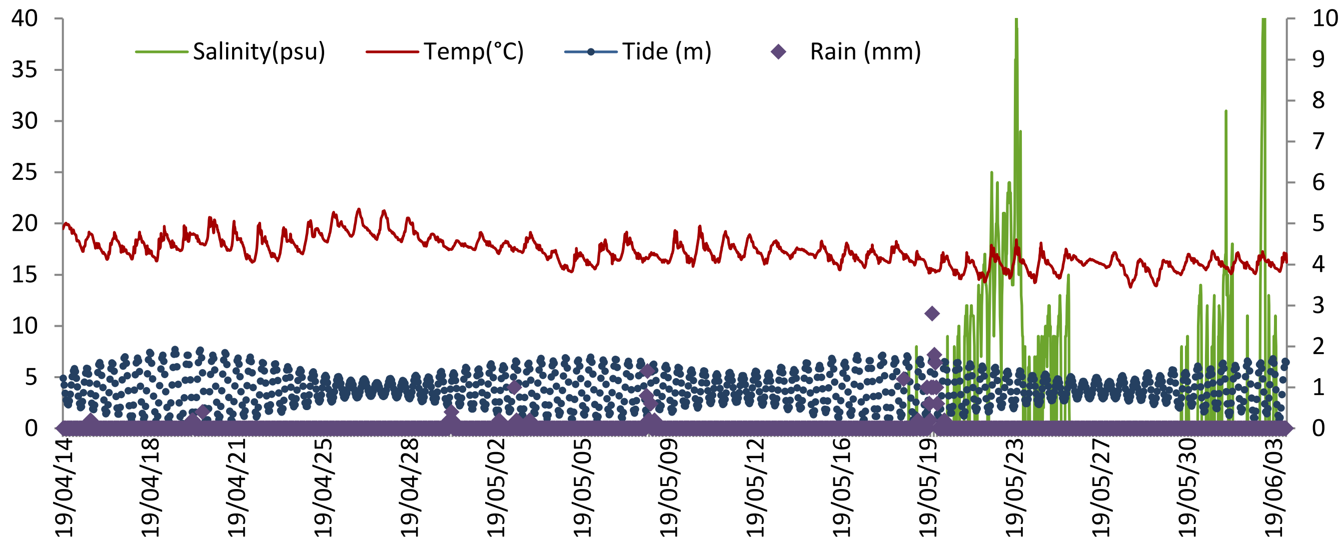

Figure 16.6. **Salinity (green line, left axis, units PSU), temperature (red line, left axis, units °C) in twenty minute intervals over a 3 month period in April 2019 through June 2019. Tidal data (dark blue points, right axis, units m) and rain (purple, right axis, units mm) are in hourly intervals over the same time period (tidal data provided by hydrographer, SANHO, rain data by WeatherSA).**

Table 16.1. Percentage cover of each species found on the rocky shores of Saldanha Bay and Langebaan Lagoon in 2019. DS = Dive School; J = Jetty; IO = Iron Ore Terminal; L = Lynch Point; M = Marcus Island; NB = North Bay; SE = Schaapen Island East; SW = Schaapen Island West. Total percentage cover for each functional group is shown in bold.

Percentage cover	DS	J	IO	L	M	NB	SE	SW
SUBSTRATE	**91,2**	**89,2**	**74,1**	**44,3**	**36,5**	**52,3**	**78,2**	**64,5**
Rock	84,27	73,87	73,07	43,48	36,47	52,34	73,49	57,36
Sand	5,58	1,02	0,49	0,54	0,00	0,00	4,75	7,11
Gravel	1,32	14,33	0,57	0,32	0,00	0,00	0,00	0,00
GRAZERS	**1,99**	**2,03**	**4,58**	**8,61**	**3,20**	**7,59**	**3,51**	**2,21**
Acanthochiton garnoti	0,02	0,02	0,36	0	0	0	0	0
Afrolittorina knysnaensis	0	0	0,28	3,85	1,05	2,06	0,00	0,04
Callochiton dentatus	0	0	0	0	0	0	0	0
Chaetopleura papilio	0	0	0	0	0	0	0	0
Cymbula compressor	0	0	0	0	0	0,02	0	0
Cymbula granatina	0,00	0,50	0,46	0,30	0,10	0,84	0,07	0,19
Cymbula miniata	0	0	0	0	0	0,03	0,0	0
Cymbula oculus	0,19	0,00	0	0	0	0,00	0,34	0
Dendrofissurella scutellum	0	0,00	0	0	0	0	0	0,00
Fissurella mutabilis	0,01	0,08	0,00	0	0,04	0	0,05	0,21
Gibbula spp.	0,00	0	0	0	0	0	0,00	0
Gibbula zonata	0	0	0,04	0	0	0	0	0,00
Haliotis midae	0	0	0	0	0	0	0,00	0
Helcion dunkeri	0	0	0	0	0	0	0	0
Helcion pectunculus	0,14	0	0	0,07	0	0	0	0
Helcion pruinosus	0	0	0	0	0	0	0	0
Ischnochiton oniscus	0,00	0	0	0	0	0	0	0
Radsia nigrovirescens	0	0	0	0	0	0	0	0
Scutellastra argenvillei	0	0	0	0,19	0,00	0,00	0,00	0,00
Scutellastra barbara	0	0	0,02	0,00	0,00	0	0	0
Scutellastra cochlear	0	0	0,00	1,09	0,00	0,74	0	0
Scutellastra granularis	0,00	0	0,24	0,41	1,44	0,61	0	0,09
Scutellastra tabularis	0,00	0	0	0,00	0	0,87	0	0
Siphonaria capensis	0,05	0	0,09	0,01	0,37	0,55	0,06	0,11
Siphonaria serrata	0,11	0	0,00	0,25	0,00	0	0,21	0,01
Parechinus angulosus	0,24	0,12	0,41	0,04	0,00	0,07	0	0,71
Parvulastra exigua	0,37	0,30	0,66	0,75	0,05	0,25	0,70	0,02
Onchidella maculata	0	0	0	0	0	0	0	0
Oxystele sinensis	0,09	0,00	0	0,00	0,00	0,01	0,00	0,00
Oxystele tigrina	0,19	0,52	0,23	0,34	0,01	0,56	1,46	0,77
Oxystele antoni	0,56	0	2	1	0	0	0,44	0
Tricolia capensis	0,00	0	0	0	0	0	0,00	0
PREDATORS	**0,63**	**0,60**	**2,85**	**0,59**	**0,42**	**2,97**	**1,28**	**3,09**
Actinia equina	0,18	0,08	0	0	0	0,50	0,00	0,00
Anthopleura michaelseni	0	0	0	0	0	0	0	0,00
Anthostella stephensoni	0	0	0	0	0,00	0	0	0,00
Anthothoe stimpsonii	0	0,00	0	0,00	0	0,14	0,00	0,01
Bunodactis reynaudi	0	0	0,30	0,15	0,33	0,18	0,21	0,16
Bunodosoma capense	0	0	0,28	0,01	0,00	0,83	0	0,00
Burnupena papyracea	0	0	0	0	0	0,00	0	0,00
Burnupena spp.	0,30	0,40	1,06	0,26	0,09	1,04	0,64	0,97
Callopatiria granifera	0,00	0	0	0	0	0	0	0,00

Percentage cover	DS	J	IO	L	M	NB	SE	SW
Clionella sinuata	0,00	0,00	0	0,00	0	0,00	0	0,00
Conus mozambicus	0	0	0	0	0	0	0	0,00
Corynactis annulata	0	0	0	0	0	0	0	1,18
Cyclograpus punctatus	0,00	0	0,02	0,03	0	0,07	0	0,05
Doris verrucosa	0	0	0	0	0	0	0	0,00
Doriopsis granulosa	0	0	0	0	0	0	0	0,00
Dromidia spp.	0	0	0	0	0	0	0	0,02
Flatworm	0	0	0	0	0	0	0	0,00
Fusinus sp.	0,02	0	0	0	0	0	0	0,00
Henricia ornata	0	0	0	0,00	0	0,13	0,02	0,09
Hermit crab	0	0	0	0	0	0	0	0,00
Hymenosoma orbiculare	0	0	0	0	0	0	0	0,00
Marthasterias glacialis	0	0	0	0	0	0	0	0,00
Nucella dubia	0,02	0	0,53	0,01	0,00	0,05	0	0,06
Nucella squamosa	0	0	0,03	0,00	0,00	0	0,00	0,00
Nudibranch	0	0	0	0	0,00	0	0	0,00
Ophiuroidea	0	0	0	0	0,00	0	0	0,00
Paguristes gamianus	0	0	0	0	0	0	0,00	0,02
Philine aperta	0	0	0	0	0	0	0	0,00
Platydromia spongiosa	0	0	0	0	0	0	0	0,00
Pseudactinia flagellifera	0,05	0	0,21	0,02	0	0	0,10	0,52
Pseudactinia sp.	0,00	0	0	0	0	0	0,00	0,00
Trochia cingulata	0	0	0	0	0,00	0	0	0,00
Volvarina capensis	0	0	0	0	0	0	0	0,00
FILTER FEEDERS	**3,14**	**3,08**	**5,07**	**10,49**	**24,53**	**11,86**	**2,52**	**3,84**
Amphibalanus amphitrite amphitrite	0	0,00	1	0	0	0	0	0
Aulacomya atra	0,26	0,05	0,23	0,37	0,04	1,90	0	0,86
Austromegabalanus cylindricus	0	0	0,02	0,00	0,00	0,14	0	0
Balanus glandula	0,26	1,68	0,52	8,03	15,68	6,69	0,01	0,00
Choromytilus meridionalis	1	0	0	0	0	0	0	0
Chthamalus dentatus	0	0	0	0	0	0	0	0
Colonial ascidian	0	0,00	0,00	0,00	0,00	0,11	0,14	0,92
Crepidula porcellana	0,37	0,07	0,15	0,01	0,01	0,13	0,16	0,02
Dendropoma corallinaceum	0	0	0	0,00	0	0,00	0	0
Dodecaceria capensis	0	0	0	0	0	0	0	0
Encrusting Bryozoa	0	0,00	0,00	0	0,00	0,00	0,00	0
Fanworm	0	0	0	0	0	0	0	0
Gunnarea gaimardi	0	0	0	0	0	0	0	0
Hemiocnus insolens	0	0	0	0	0,01	0	0	0
Hydroids	0	0	0	0,00	0	0,16	0	0
Mytilus galloprovincialis	0,00	0,00	0,88	1,74	8,47	0,97	0	0
Notomegabalanus algicola	0,00	0,17	0	0,00	0,00	0,11	0	0
Octomeris angulosa	0	0	0	0	0	0	0	0
Pentacta doliolum	0	0	0,00	0	0,00	0	0	0
Perforatus perforatus	0,00	0,22	0,00	0,00	0	0	0	0
Pyura herdmani	0	0	0,00	0	0	0	0	0
Pyura stolonifera	0,00	0,15	0,14	0,00	0	0	0	0
Roweia frauenfeldi frauenfeldi	0,35	0	0	0,03	0,00	0	0	0
Sandy tube worm	0	0,00	0	0	0	0	0	0
Spirorbis spp.	0,23	0	0,14	0,01	0	0,20	0,50	0,13

Percentage cover	DS	J	IO	L	M	NB	SE	SW
Sponge	0,42	0	0,05	0	0,29	0,78	1,59	1,86
Tetraclita rufotincta	0	0	0,00	0	0	0	0	0
Tetraclita serrata	0,00	0	0	0,02	0,00	0	0	0
Thyone aurea	0	0	1	0,01	0,02	0,00	0	0,00
Tubeworm	0	0	0	0,25	0,00	0,36	0	0,00
CRUSTOSE	**1,75**	**2,10**	**6,65**	**31,22**	**14,62**	**14,94**	**6,80**	**10,47**
Alcyonium fauri (Soft Coral)	0,00	0,00	0,00	0,00	0,00	0,32	0,00	0,00
Coralline (crustose)	0	1	3	17,58	0,11	6,02	2,81	1,04
Coralline (upright)	0	0,00	0,00	1,20	0,25	0,58	0,14	0,37
Diatoms	0,07	0,06	0,00	0,00	0,00	6,73	0,00	8,16
Hildenbrandia spp.	0,65	0,35	1,67	12,21	14	0,64	2,38	0
Ralfsia verrucosa	0,58	0,68	1,81	0,24	1	0,98	1,47	1
EPHEMERALS	**0,51**	**1,64**	**2,28**	**3,13**	**19,13**	**2,09**	**4,68**	**4,95**
Bryopsis myosurioides	0	0	0,00	0	0,00	0	0,00	0
Callithamnion collabens	0	0	0	0,00	0	0,02	0	0
Centroceras clavulatum	0	0,00	0	0	0	0,17	0	0
Ceramium spp	0,06	0,17	0,29	0,00	0,08	0,10	0,00	0,04
Cladophora spp.	0,08	0,70	0,02	0,69	0,37	0,07	0,01	0,20
Ectocarpus	0	0,00	0	0,39	0	0	0,28	0
Green turf	0	1	0	0,00	4,09	0,00	0,00	0,00
Porphyra capensis	0	0	0,00	0,20	2	0,20	3	1
Pachymenia chornia	0,00	0	0	0,00	0,00	0,00	0,00	0,00
Pachymenia orbitosa	0,02	0,00	0,00	0,10	0,07	0,21	0,24	0,14
Ulva spp.	0,36	0,21	1,56	1,75	12,23	1,31	1,29	3,48
CORTICATED	**0,81**	**1,32**	**3,01**	**1,26**	**1,64**	**4,91**	**2,87**	**10,64**
Ahnfeltiopsis complicata	0	0	0	0	0	0	0,00	0
Ahnfeltiopsis glomerata	0	0	0,00	0	0,00	0	0,00	0,07
Ahnfeltiopsis polyclada	0,00	0,16	0,00	0	0	0	0,00	0,00
Botryocarpa prolifera	0	0	0	0	0	0,00	0	0
Botryocladia paucivesicaria	0	0	0	0	0	0	0	0
Botryoglossum platycarpum	0	0	0	0	0	0	0	0
Callithamnion collabens	0	0	0,00	0	0,00	0	0	0
Carpoblepharis flaccida	0	0	0,00	0	0,00	0	0	0
Caulacanthus ustulatus	0	0,00	0,09	0,00	1,08	0,78	0,00	0
Chaetomorpha robusta	0	0	0,00	0	0	0,00	0,12	0,00
Champia compressa	0	0	0,00	0,04	0,00	0,00	0	0
Champia lumbricalis	0	0	0	0	0	2	0	0
Chondria capensis	0	0	0	0	0	0	0	0
Chordariopsis capensis	0	0	0,00	0,00	0,00	0,00	0	0
Cochlear Garden	0	0,00	0	0	0	0	0	0,00
Codium fragile fragile	0	0	0	0	0	0	0	0
Codium lucasii	0	0	0	0	0	0	0	5,14
Codium stephensiae	0,00	0,00	0	0	0	0	0	1
Colpomenia sinuosa	0	0	0	0	0,00	0	0	0
Exallosorus harveyanus	0	0	0,00	0,00	0	0,00	0,00	0,00
Gelidium pristoides	0	0	0	0	0	0	0	0
Gelidium pteridifolium	0	0	0	0,00	0	0,00	0	0
Gelidium vittatum	0	0,04	0	0	0	0,00	0,01	0
Gigartina bracteata	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00
Gigartina polycarpa	1	1	2	1	0	1	1	2

Percentage cover	**DS**	**J**	**IO**	**L**	**M**	**NB**	**SE**	**SW**
Grateloupia longifolia	0	0	0	0	0,00	0	0	0
Gymnogongrus dilatatus	0	0	0	0	0	0	0	0
Halopteris funicularis	0,00	0	0,00	0	0	0	0,00	0,00
Hypnea ecklonii	0,00	0	0	0	0	0	0	0,00
Hypnea spicifera	0	0	0	0	0	0	0	2
Laurencia glomerata	0	0	0	0	0	0	0	0
Leathesia marina	0,00	0	0	0,00	0,00	0	0,00	0
Mazzaella capensis	0	0	0	0	0	0	0	0
Neuroglossum binderianum	0	0,00	0,00	0	0	0,00	0,00	0
Nothogenia erinacea	0	0	0	0	0	0	0	0
Nothogenia ovalis	0	0	0	0	0	0	0	0
Phyllymenia belangeri	0	0	0	0	0	0	0	0
Phyllymenia capensis	0	0	0	0	0	0	0	0
Plocamium corallorhiza	0,00	0	0,00	0	0,00	0,00	0,00	0,00
Plocamium spp.	0	0	0,20	0	0	0	0,00	0,00
Polyopes constrictus	0	0	0	0	0	0	0	0
Portieria hornemanii	0	0	0	0	0	0,00	0	0
Pterosiphonia cloiophylla	0,00	0,00	0,00	0,00	0	0	0,00	0,00
Red turf	0	0	0	0	0	0	1	0
Rhodophyllis reptans	0	0,00	0	0,00	0	0,00	0,00	0,00
Rhodymenia pseudopalmata	0	0	0	0	0	0	0	0
Rhodymenia spp.	0	0	0	0	0	0	0	0
Sarcothalia radula	0	0	0	0,00	0,0	0	0,00	0,05
Sarcothalia scutellata	0	0	0	0	0,00	0,02	0,00	0,19
Sarcothalia stiriata	0	0	0,00	0	0	0	0	1
Schizymenia apoda	0	0	0	0	0	0	0,00	0,00
Splachnidium rugosum	0	0	0	0	0	0	0	0
Tayloriella tenebrosa	0	0	0,00	0	0	0	0	0
Tsengia lanceolata	0	0	0,00	0	0	0	0	0
KELP	**0**	**0**	**1,41**	**0,36**	**0,00**	**2,97**	**0,12**	**0,33**
Ecklonia maxima	0	0	1,41	0,36	0,00	2,97	0,12	0,09
Laminaria pallida	0	0	0,00	0,00	0,00	0	0,00	0,24

Table 16.2. List of non-passerine waterbird species occurring in Langebaan Lagoon (Note that this species list excludes rare vagrants, exotic species and terrestrial species) (Source: CWAC data, Animal Demography Unit at the University of Cape Town).

Common name	Scientific name	Average count	Maximum count
African Oystercatcher	*Haematopus moquini*	17	163
African Darter	*Anhinga rufa*	2	3
African Fish-Eagle	*Haliaeetus vocifer*	1	2
African Marsh-Harrier	*Circus ranivorus*	2	9
African Purple Gallinule	*Porphyrio madagascariensis*	2	2
African Rail	*Rallus caerulescens*	2	3
African Sacred Ibis	*Threskiornis aethiopicus*	112	720
African Snipe	*Gallinago nigripennis*	4	19
African Spoonbill	*Platalea alba*	23	137
Antarctic Tern	*Sterna vittata*	2	2
Arctic Tern	*Sterna paradisaea*	35	35
Bank Cormorant	*Phalacrocorax neglectus*	11	29
Bar-tailed Godwit	*Limosa lapponica*	225	3000
Black Crake	*Zapornia flavirostra*	2	2
Black-crowned Night-Heron	*Nycticorax nycticorax*	3	6
Black-headed Heron	*Ardea melanocephala*	3	29
Blacksmith Lapwing	*Vanellus armatus*	18	78
Black-tailed Godwit	*Limosa limosa*	1	1
Black-winged Stilt	*Himantopus himantopus*	37	180
Cape Cormorant	*Phalacrocorax capensis*	90	2289
Cape Shoveler	*Anas smithii*	9	45
Cape Teal	*Anas capensis*	16	90
Caspian Tern	*Hydropogne caspia*	8	53
Cattle Egret	*Bubulcus ibis*	8	45
Chestnut-banded Plover	*Charadrius pallidus*	57	581
Common Greenshank	*Tringa nebularia*	112	1175
Common Moorhen	*Gallinula chloropus*	2	5
Common Redshank	*Tringa totanus*	14	76
Common Ringed Plover	*Charadrius hiaticula*	100	548
Common Sandpiper	*Actitis hypoleucos*	4	34
Common Tern	*Sterna hirundo*	509	9658
Common Whimbrel	*Numenius phaeopus*	161	2000
Crowned Cormorant	*Microcarbo coronatus*	32	167
Crowned Plover	*Vanellus coronatus*	4	8
Curlew Sandpiper	*Calidris ferruginea*	3242	25347
Egyptian Goose	*Alopochen aegyptiaca*	15	433
Eurasian Curlew	*Numenius arquata*	82	1373
Giant kingfisher	*Megaceryle maximus*	1	1
Glossy Ibis	*Plegadis falcinellus*	28	89
Goliath Heron	*Ardea goliath*	3	3
Great Crested Grebe	*Podiceps cristatus*	2	2

Common name	Scientific name	Average count	Maximum count
Great White Egret	*Egretta alba*	1	3
Great White Pelican	*Pelecanus onocrotalus*	27	262
Greater Flamingo	*Phoenicopterus roseus*	853	8724
Greater Sand Plover	*Charadrius leschenaultii*	7	35
Grey Heron	*Ardea cinerea*	8	83
Grey plover	*Pluvialis squatarola*	707	8228
Grey-headed Gull	*Larus cirrocephalus*	6	19
Hartlaub's Gull	*Larus hartlaubii*	224	1881
Kelp Gull	*Larus dominicanus*	111	1140
Kittlitz's Plover	*Charadrius pecuarius*	53	545
Lesser Flamingo	*Phoeniconaias minor*	203	1606
Lesser Sand Plover	*Charadrius mongolus*	7	19
Little Egret	*Egretta garzetta*	24	126
Little Grebe	*Tachybaptus ruficollis*	1	1
Little Stint	*Calidris minuta*	146	858
Little Tern	*Sternula albifrons*	9	64
Malachite Kingfisher	*Alcedo cristata*	1	2
Marsh Owl	*Asio capensis*	2	5
Marsh Sandpiper	*Tringa stagnatilis*	10	55
Osprey	*Pandion haliaetus*	2	5
Pied Avocet	*Recurvirostra avosetta*	52	521
Pied Kingfisher	*Ceryle rudis*	5	16
Pink-backed Pelican	*Pelecanus rufescens*	26	26
Purple Heron	*Ardea purpurea*	1	3
Red Knot	*Calidris canutus*	963	6219
Red-billed Teal (Duck)	*Anas erythrorhyncha*	5	22
Red-knobbed Coot	*Fulica cristata*	45	277
Reed Cormorant	*Microcarbo africanus*	22	277
Ruddy Turnstone	*Arenaria interpres*	536	4587
Ruff	*Calidris pugnax*	25	237
Sanderling	*Calidris alba*	600	4950
Sandwich Tern	*Thalasseus sandvicensis*	34	1474
South African Shelduck	*Tadorna cana*	15	131
Southern Pochard	*Netta erythrophthalma*	4	4
Spur-winged Goose	*Plectropterus gambensis*	7	71
Swift Tern	*Thalasseus bergii*	36	1538
Terek Sandpiper	*Xenus cinereus*	42	266
Three-banded Plover	*Charadrius tricollaris*	6	38
Water Thick-knee	*Burhinus vermiculatus*	2	3
White-breasted Cormorant	*Phalacrocorax lucidus*	12	89
White-fronted Plover	*Charadrius marginatus*	84	473
White-winged Tern	*Chlidonias leucopterus*	4	17
Wood Sandpiper	*Tringa glareola*	4	12
Yellow-billed Duck	*Anas undulata*	51	335

Common name	Scientific name	Average count	Maximum count
Yellow-billed Egret	*Ardea intermedia*	4	31

Table 16.3. List of alien, invasive, naturalised and cryptogenic species that are likely to occur on the West Coast of South Africa or have been confirmed to occur in Saldanha Bay and Langebaan Lagoon. Region of origin and likely vector for introduction (SB = ship boring, SF = ship fouling, BW = ballast water, BS = solid ballast, OR = oil rigs, M = mariculture, F = Fisheries activities, I = intentional release) are listed. Data extracted from Mead *et al.* (2011a & b) and Robinson *et al.* 2014, and recent published and unpublished research.

Taxon	Occurrence in Saldanha/Langebaan	Status	Origin	Vector	Reference
PROTOCTISTA					
Mirofolliculina limnoriae	Likely	Alien	Unknown	SB	Mead *et al.* 2011
DINOFLAGELLATA					
Alexandrium minutum	Likely	Alien	Europe	BW	Mead *et al.* 2011
Alexandrium tamarense-complex	Likely	Alien	N Atlantic/N Pacific	BW	Mead *et al.* 2011
Dinophysis acuminata	Likely	Alien	Europe	BW	Mead *et al.* 2011
PORIFERA					
Suberites ficus	Likely	Invasive	Europe	SF	Samaai and Giboons 2005
CNIDARIA					
ANTHOZOA					
Metridium senile	Likely	Alien	N Atlantic/N Pacific	SF/OR	Mead *et al.* 2011
Sagartia ornata	**Confirmed**	**Naturalised**	**Europe**	**SF/BW**	**Robinson and Swart 2015**
ECHINODERMATA					
ASTEROIDEA					
Heliaster helianthus	**Confirmed**	**Alien**	**South American Pacific**	**SF/BW**	**Peters and Robinson 2018**
HYDROZOA					
Coryne eximia	**Confirmed**	**Invasive**	**N Atlantic/N Pacific**	**SF/BW**	**Mead *et al.* 2011**
Gonothyraea loveni	Likely	Alien	North Atlantic	SF/BW	Mead *et al.* 2011
Laomedea calceolifera	Likely	Alien	North Atlantic	SF/BW	Mead *et al.* 2011
Obelia bidentata	Likely	Naturalised	Unknown	SF/BW	Mead *et al.* 2011
Obelia dichotoma	Likely	Naturalised	Unknown	SF/BW	Mead *et al.* 2011
Obelia geniculata	Likely	Naturalised	Unknown	SF/BW	Mead *et al.* 2011

Taxon	Occurrence in Saldanha/Langebaan	Status	Origin	Vector	Reference
Pachycordyle navis	Likely	Alien	Europe	SF/BW	Mead *et al.* 2011
Pinauay larynx	Likely	Naturalised	North Atlantic	SF/BW	Mead *et al.* 2011
Pinauay ralphi	Likely	Alien	North Atlantic	SF/BW	Mead *et al.* 2011
ANNELIDA					
POLYCHAETA					
Boccardia proboscidea	**Confirmed**	**Invasive**	**Eastern Pacific**	**M**	**David and Simon 2014; CAS unpublished data**
Capitella sp.	Likely	Cryptogenic	Unknown	SF/BW	Mead *et al.* 2011
Dodecaceria fewkesi	Likely	Naturalised	North American Pacific	SF/BW	Peters *et al.* 2014
Ficopomatus enigmaticus	Likely	Invasive	Australia	SF	McQuaid and Griffiths 2014
Janua pagenstecheri	Likely	Alien	Europe	SF/BW	Mead *et al.* 2011
Neodexiospira brasiliensis	**Confirmed**	**Invasive**	**Indo-Pacific**	**SF/BW**	**Mead *et al.* 2011**
Simplicaria pseudomilitaris	Likely	Alien	Unknown	SF/BW	Mead *et al.* 2011
Polydora hoplura	**Confirmed**	**Invasive**	**Europe**	**SF/BW**	**Simon 2011; David and Simon 2014**
Polydora cf. *websteri*	Likely	Alien, in potentially open facility	Unknown	M	Simon 2015; Williams 2015
Hydroides elegans	Likely	Cryptogenic	Unknown	SF/BW	Robinson *et al.* 2016
CRUSTACEA					
CIRRIPEDIA					
Amphibalanus amphitrite amphitrite	Confirmed (AEC 2014)	Cryptogenic	Unknown	SF/BW	Mead *et al.* 2011
Amphibalanus venustus	Likely	Invasive	North Atlantic	SF	Mead *et al.* 2011
Balanus glandula	**Confirmed**	**Invasive**	**North American Pacific**	**SF/BW**	**Robinson *et al.* 2015**
Perforatus perforatus	**Confirmed**	**Alien**	**North American Pacific**	**SF/BW**	**Biccard and Griffiths (*Pers. Comm. 2017)***
COPEPOD					
Acartia (Odontacartia) spinicauda	Likely	Alien	Western North Pacific	BW	Mead *et al.* 2011
ISOPODA					
Dynamene bidentata	Likely	Invasive	Europe	SF/BW	Mead *et al.* 2011

Taxon	Occurrence in Saldanha/Langebaan	Status	Origin	Vector	Reference
Ligia exotica	Likely	Cryptogenic	Unknown	SB	Mead *et al.* 2011
Limnoria quadripunctata	Likely	Alien	Unknown	SB	Mead *et al.* 2011
Limnoria tripunctata	Likely	Alien	Unknown	SB	Mead *et al.* 2011
Paracerceis sculpta	Likely	Alien	Northeast Pacific	SF/BW	Mead *et al.* 2011
Synidotea hirtipes	Confirmed	Cryptogenic	Indian Ocean	SF/BW	Mead *et al.* 2011
Synidotea variegata	Confirmed	Cryptogenic	Indo-Pacific	SF/BW	Mead *et al.* 2011
AMPHIPODA					
Caprella equilibra	Likely	Cryptogenic	Unknown	SF/BW	Mead *et al.* 2011
Caprella mutica	Likely	Alien	North-east Asia	SF	Peters and Robinson 2017
Caprella penantis	Likely	Cryptogenic	Unknown	SF/BW	Mead *et al.* 2011
Chelura terebrans	**Confirmed**	**Invasive**	**Pacific Ocean**	**SF/SB**	**Mead *et al.* 2011**
Cerapus tubularis	**Confirmed**	**Invasive**	**North American Atlantic**	**BS**	**Mead *et al.* 2011**
Cymadusa filosa	Likely	Cryptogenic	Unknown	BS	Mead *et al.* 2011
Erichthonius brasiliensis	Likely	Invasive	North Atlantic	SF/BW	Mead *et al.* 2011
Ericthonius difformis	Likely	Alien	Unknown, northern hemisphere	SF	Peters *et al.* 2014
Ischyroceros anguipes	Likely	Invasive	North Atlantic	SF/BW	Mead *et al.* 2011
Jassa marmorata	Likely	Naturalised	North Atlantic	SF/BW	Conlan 1990; Mead *et al.* 2011
Jassa morinoi	Likely	Invasive	Eastern North Pacific	SF/BW	Conlan 1990; Mead *et al.* 2011
Jassa slatteryi	**Confirmed**	**Invasive**	**North Pacific**	**SF/BW**	**Conlan 1990; Mead *et al.* 2011**
Paracaprella pusilla	Likely	Cryptogenic	Unknown	SF/BW	Mead *et al.* 2011
Orchestia gammarella	**Confirmed**	**Invasive**	**Europe**	**BS**	**Mead *et al.* 2011**
DECAPODA					
Carcinus maenas	**Confirmed (G. Branch *pers. comm.*)**	**Invasive**	**Europe**	**SF/BW/ OR**	**Robinson *et al.* 2005**
Homalaspis plana	**Confirmed**	**Alien**	**South American Pacific**	**SF/BW**	**Peters and Robinson 2018**
Pinnixa occidentalis	**Confirmed (Anchor 2011)**	**Invasive**	**North American Pacific**	**BW**	**Clark and Griffiths 2012**

Taxon	Occurrence in Saldanha/Langebaan	Status	Origin	Vector	Reference
***Porcellana africana* (Incorrectly identified as *Porcellana platycheles*)**	**Confirmed**	**Invasive**	**North East Atlantc**	**BW**	**Griffiths *et al.* 2018**
Xantho incicus	Likely	Alien	France	M	Haupt *et al.* 2010
INSECTA					
COLEOPTERA					
Cafius xantholoma	Likely	Invasive	Europe	BS	Mead *et al.* 2011
MOLLUSCA					
GASTROPODA					
Catriona columbiana	Likely	Alien	North Pacific	SF/BW	Mead *et al.* 2011
Littorina saxatilis	**Confirmed**	**Invasive**	**Europe**	**BS**	**Mead *et al.* 2011**
Tritonia nilsodhneri	Likely	To be confirmed	Europe	SF/BW	Zsilavecz 2007
Kaloplocamus ramosus	Likely	To be confirmed	Unknown	SF/BW	Zsilavecz 2007
Thecacera pennigera	Likely	Cryptogenic	Unknown	SF/BW	Mead *et al.* 2011
Anteaeolidiella indica	Confirmed	Cryptogenic	Unknown	SF/BW	Mead *et al.* 2011
BIVALVIA					
Bankia carinata	Likely	Cryptogenic	Unknown	SB	Mead *et al.* 2011
Bankia martensi	Likely	Cryptogenic	Unknown	SB	Mead *et al.* 2011
Crassostera gigas	**Confirmed**	**Invasive**	**Japan**	**M**	**Haupt *et al.* 2010; Keightley *et al.* 2015**
Dicyathifer manni	Likely	Cryptogenic	Unknown	SB	Mead *et al.* 2011
Lyrodus pedicellatus	Likely	Alien	Unknown	SB	Mead *et al.* 2011
Mytilus galloprovincialis	**Confirmed**	**Invasive**	**Europe**	**SF/BW**	**Robinson *et al.* 2005**
Semimytilus algosus	**Confirmed**	**Invasive**	**South Pacific**	**SF/BW**	**de Greef *et al.* 2013**
Teredo navalis	Likely	Invasive	Europe	SB	Mead *et al.* 2011
Teredo somersi	Likely	Cryptogenic	Unknown	SB	Mead *et al.* 2011
BRACHIOPODA					
Discinisca tenuis	**Confirmed**	**Invasive**	**Namibia**	**M**	**Haupt *et al.* 2010; Peters *et al.* 2014**
BRYOZOA					
Bugula flabellata	Likely	Invasive	Unknown	SF	Florence *et al.* 2007
Bugula neritina	Likely	Invasive	Unknown	SF	Florence *et al.* 2007

Taxon	Occurrence in Saldanha/Langebaan	Status	Origin	Vector	Reference
Conopeum seurati	**Confirmed**	**Invasive**	**Europe**	**SF**	**McQuaid and Griffiths 2014**
Cryptosula pallasiana	**Confirmed**	**Invasive**	**Europe**	**SF**	**Mead *et al.* 2011**
Watersipora subtorquata	**Confirmed**	**Invasive**	**Caribbean**	**SF**	**Florence *et al.* 2007; Mead *et al.* 2011**
CHORDATA					
ASCIDIACEA					
Ascidia sydneiensis	Likely	Invasive	Pacific Ocean	SF	Mead *et al.* 2011; Rius *et al.* 2014
Ascidiella aspersa	Likely	Invasive	Europe	SF	Mead *et al.* 2011; Peters *et al.* 2014; Rius *et al.* 2014
Botryllus schlosseri	Likely	Invasive	Unknown	SF	Mead *et al.* 2011; Peters *et al.* 2014; Rius *et al.* 2014
***Ciona robusta* (formally known as *Ciona intestinalis*)**	**Confirmed (Picker & Griffiths 2011)**	**Invasive**	**Unknown**	**SF**	**Mead *et al.* 2011; Rius *et al.* 2014; Brunetti *et al.* 2015**
Clavelina lepadiformis	**Confirmed (Picker & Griffiths 2011)**	**Invasive**	**Europe**	**SF**	**Mead *et al.* 2011; Rius *et al.* 2014**
Cnemidocarpa humilis	Likely	Invasive	Unknown	SF	Mead *et al.* 2011
Corella eumyota	Confirmed	Cryptogenic	Unknown	SF	Mead *et al.* 2011
Diplosoma listerianum	**Confirmed**	**Invasive**	**Europe**	**SF**	**Mead *et al.* 2011; Rius *et al.* 2014**
Microcosmus squamiger	Likely	Invasive	Australia	SF	Mead *et al.* 2011; Rius *et al.* 2014
Trididemnum cerebriforme	Confirmed	Cryptogenic	Unknown	SF	Mead *et al.* 2011
PISCES					
Cyprinus carpio	Likely	Invasive	Central Asia to Europe	I	Mead *et al.* 2011
RHODOPHYTA					
Antithamnionella spirographidis	**Confirmed**	**Invasive**	**North Pacific**	**SF/BW**	**Mead *et al.* 2011**
Antithamnionella ternifolia	Likely	Cryptogenic	Australia	SF/BW	Mead *et al.* 2011
Asparagopsis armata	Likely	Invasive	Australia	Unknown	Bolton *et al.* 2011
Schimmelmannia elegans	Likely	Alien	Tristan da Cunha	BW	De Clerck *et al.* 2002
CHLOROPHYTA					
Codium fragile fragile	**Confirmed**	**Invasive**	**Japan**	**SF/BW**	**Mead *et al.* 2011**