THE STATE OF

SALDANHA BAY AND LANGEBAAN LAGOON

2019

Technical Report

September 2019

Barry Clark, Vera Massie, Ken Hutchings, Aiden Biccard, Erika Brown, Megan Laird, Kirti Gihwala, Cheruscha Swart, Anelisa Makhosonke, Safiyya Sedick, Jane Turpie and Nicolette Vermaak (Department of Water and Sanitation)

Report Prepared by:
Anchor Environmental Consultants (Pty) Ltd.
8 Steenberg House, Silverwood Close, Tokai 7945, South Africa
www.anchorenvironmental.co.za

Cover Photo: Anchor Environmental Consultants (Pty) Ltd.

Citation: Clark BM, Massie V, Hutchings K, Biccard A, Brown E, Laird M, Gihwala K, Swart C, Makhosonke A, Sedick S, Turpie J. and Vermaak N. 2019. The State of Saldanha Bay and Langebaan Lagoon 2019, Technical Report. Report No. AEC 1841/1 prepared by Anchor Environmental Consultants (Pty) Ltd for the Saldanha Bay Water Quality Forum Trust, September 2019.

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 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.

None of the above can take place without scientifically based information on the 'State of the Bay'. The Saldanha Bay Water Quality Trust has been the pioneer in this regard and has conducted a series of all-encompassing scientific tests with minimal resources over the last 20 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.

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 our Saldanha Bay and Langebaan Lagoon for future generations whilst ensuring responsible development.

Councillor André Kruger

Portfolio Chairperson: Infrastructure and Planning Services
Saldanha Bay Municipality
Chairperson Saldanha Bay Water Quality Trust

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

EXECUTIVE SUMMARY

Regular, long-term environmental monitoring is essential to identify and to enable proactive 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 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 2019 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 (residential and industrial development, dredging, coastal erosion, shipping, and 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 and birds). 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 (construction of the 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 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 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.

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 are constantly rising, especially those visiting the West Coast National Park (WCNP) (Average increase 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 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 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.

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.

With the ongoing drought in the Western Cape, however, industry and local municipalities are coming 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. Arcelor Mittal now represents the highest consumer of treated wastewater from the Saldanha Bay Wastewater Treatment Works. Arcelor Mittal constructed a Reverse Osmosis plant, which treats wastewater such that it can be used for cooling steel production equipment.

Ballast water discharge volumes are continuously increasing over time as shipping traffic increases in Saldanha Bay. The total number of ships entering the Port of Saldanha nearly doubled between 1994 and 2011 and average vessel size increased over the years. As a result, the volume of ballast water discharged almost tripled between 2000 and 2011 from 8.4 to 21.1 million tons. Since 2011, ballast water discharge per vessel has remained stable around 70 thousand tons for vessels docking at the Iron Ore Terminal. Vessels docking at the Multipurpose Terminal, however, continued increasing in size until 2014/2015 and have since stabilised with individual vessels discharging approximately 10 thousand tons. 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 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 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 affect 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.

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. A combined 430 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 is currently in the process of establishing a sea-based Aquaculture Development Zone (ADZ) in Saldanha Bay. The ADZ areas comprise four precincts (Small Bay, Big Bay North, Outer Bay North and South) totalling 420 ha of new aquaculture areas in Saldanha Bay for a total ADZ comprising 884 ha (currently farmed areas will be incorporated into the ADZ). 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 State of the Bay report. The impacts of these various activities and discharges are evaluated against their potential threat to the ecological integrity of Saldanha Bay and Langebaan Lagoon.

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 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 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 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.

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. New research has identified dredging operations conducted during the Port construction programme as making a potentially important contribution to this problem. 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. It has been 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.

Ongoing erosion monitoring of Langebaan Beach (initiated by the SBM in 1996 and continued by the SBWQFT in 2018) demonstrate that the beaches north of Langebaan are highly dynamic, with Langebaan North Beach experiencing erosion in winter and accretion in summer, while the reverse is true for Langebaan South Beach. Variability on Langebaan North Beach is also almost twice that observed at Langebaan South Beach. It is likely that this is linked to seasonal reversal of the wave climate experienced at these two sites, with wave energy at Langebaan North Beach being much more intense and peaking 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 peaks in summer (and is derived from the southerly winds blowing across the Lagoon at this time of year).

Groundwater

Saldanha Bay and Langebaan Lagoon receives little freshwater input via rivers or streams (surface water), but groundwater input is significant and plays an important role in sustaining marsh ecosystems around the periphery of the Bay, and especially at the head of the Lagoon. There are two main aquifer systems, the Langebaan Road Aquifer System and the Elandsfontein Aquifer System, that formerly were thought to be separate and these two aquifers discharge at separate locations into the bay. Recent work, however, suggests that these two aquifers may be connected. The Langebaan Road Aquifer System discharges into Saldanha Bay (Big Bay) through a northern paleo-channel, while the Elandsfontein Aquifer System discharges into the head of the Langebaan Lagoon through a southern paleo-channel. Growth of the reeds *Phragmites australis* and *Typha capensis* on the shoreline surrounding Langebaan Lagoon provide clear evidence of the significant influx of groundwater into the Lagoon. 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 typically equivalent to that of seawater- 35 PSU), providing evidence that fresh groundwater flows must be sustaining these reeds. Increasing pressure on available freshwater water in the Saldanha Bay area in recent years has resulted in attention being turned to exploitation of these groundwater resources. Historically, agriculture was the primary user of water from these aquifers but demands for water for domestic and industrial uses are increasing rapidly. The West Coast District Municipality (WCDM) operates a wellfield on the Langebaan Road Aquifer that is licenced to abstract up to 1.46 million m^3 of groundwater per annum. Abstraction of groundwater from this aquifer resulted in a localised depression of water levels in the deeper portion of this aquifer by as much as 10 m in the first few years of operation between 2005 and 2009, and concern has been expressed over how this might affecting groundwater discharge to Saldanha Bay in the future. A modest (10%) reduction in abstraction rates was implemented to address this, but it is not clear how effective this has been.

More recently, Elandsfontein Exploration and Mining (Pty) Ltd/Kropz started mining phosphate deposits in the area of the Elandsfontein Aquifer System on the eastern side of the R27. However, the process has been stalled due to a water use license appeal awaiting a Water Tribunal trial in September 2019. Should mining recommence, it will be 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 will be abstracted from a series of boreholes surrounding the mine pit and reinjected into the aquifer further away, to mitigate potential impacts on surrounding ecosystems (including the Lagoon). There is concern about the efficacy of these proposed mitigation measures and a comprehensive monitoring programme has been initiated to track the effectiveness of groundwater reinjection in minimising potential impacts on the lagoon hydrology and ecology This includes monitoring of water levels and water quality in a series of boreholes between the mine site and the lagoon edge and monitoring of salinity levels and macrofauna assemblages in the lagoon itself where to date, three years of baseline data have been collected. Some of the water quality data that has been collected to date is probably of limited value due to a series of defective instruments, however, the situation has been rectified with installation of a new, more robust water quality recording instrument.

Water Quality

Aspects of water quality (temperature, salinity and 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.

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. Currently, 15 of the 20 monitoring stations in the Bay are rated as having 'Excellent' water quality, three sites (Bok River Mouth and the beach opposite the municipal camp site in Small Bay, and Kraal Bay in Langebaan Lagoon) are rated as 'Fair', whist the Hoedtjies Bay Hotel site in Small Bay is again rated as having "Poor" water quality. The Bok River was historically the principal source of microbiological contamination in Small Bay with the impacts frequently spreading to adjacent sites. Recent efforts that focused on wastewater treatment prior to discharge and wastewater reuse appear to have resulted in some improvements with respect to recreational use at least, future monitoring will determine if this improvement is sustainable. Four of the ten monitoring sites within Small Bay did not however, meet the 80^{th} percentile faecal coliform limits for mariculture in 2019. Faecal coliform counts at all four sites in Big Bay were within the 80^{th} percentile limits for mariculture in 2019. 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. Reuse of wastewater from the Saldanha WWTW by Arcelor Mittal, which commenced in 2018, does appear to have resulted in an improvement in water quality in Small Bay and this improvement will hopefully continue to be reflected in future results.

Concentrations of trace metals in marine organisms (mostly mussels) in Saldanha Bay have historically been routinely monitored by the Department of Environmental Affairs (DEA) and by mariculture farm owners. DEA discontinued the 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 shore (particularly for lead and cadmium). Concentrations of trace metals in cultured mussels offshore are typically lower (according to data supplied by operators); although concentrations of lead and cadmium were on occasion above the limit for foodstuff prior to 2016, which was concerning. The reasons for the lower concentrations of trace metals 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 concentration 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.

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. However, 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, these have decreased again in the latest (2019) survey.

Levels of total organic carbon (TOC) and total organic nitrogen (TON) remain 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 is 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) in these areas with negative impacts on benthic communities (e.g. the Saldanha Yacht Club). 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 between 2008 and 2019 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 on 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 2019 suggest that levels dropped off slightly at many of the key sites in Small and Big Bay, however, but 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 the environment 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. Subsequent to 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 with the exceptions of a few sites in Small Bay where thresholds were exceeded on a number of occasions between 2016 and 2019. Key areas of concern regarding trace metal pollution within Small Bay include the Yacht Club Basin, where cadmium and copper exceeded recommended thresholds 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 increases.

Poly-aromatic hydrocarbons (PAH) contamination measured in the sediments of Saldanha Bay since 1999 have always been well below risk (ERL) values stipulated by NOAA 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 2019 remain unchanged from 2018 and 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, like that recorded in 2014.

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 1999-2019 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. At one point (2008) benthic fauna have been almost entirely eliminated from the Yacht Club Basin in Small Bay, owing to very high levels of trace metals and other contaminants at this site (TOC, Cu, Cd and Ni). 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 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. Future monitoring of these indices at this site is important in order to gauge recovery in the benthos.

Rocky intertidal

As a component of the ongoing State of the Bay evaluation, baseline conditions relating to rocky intertidal communities in Saldanha Bay was initiated in 2005. Eight rocky shores spanning a wave exposure gradient from very sheltered to exposed, were sampled in Small Bay, Big Bay and Outer Bay. These surveys have been repeated annually from 2008 to 2015, however, due to financial constrains no survey was conducted in 2016. In the 2019 survey, a total of 118 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, 25 species of grazers, and 20 species of predators/scavengers. The algal component comprised 33 corticated (foliose) seaweeds, ten ephemerals, five species of encrusting algae, and two 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

The 2019 seine net survey revealed some 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 2017-2019, and white stump abundance remains low throughout the system. The abundance of gobies in Small Bay has also remained low since 2007 and declines in goby abundance in Langebaan Lagoon have also occurred in recent years. The decline in gobies cannot be attributed to fishery impacts but may be related to water quality or habitat changes. Total fish diversity and overall 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 in 2016 and 2017, none were caught in 2018 or 2019, which suggests that these historic strong year classes are not yet contributing to reproductive output in significant numbers. Silversides were absent in Big Bay in 2018 and very scarce in 2019 samples. Furthermore, five species that were usually present in Big Bay surveys were absent in 2019 (False Bay klipvis, super klipvis, elf, sandsharks and pipefish) leading to the lowest diversity in 15 annual surveys with just eight species in Big Bay samples. None of these “missing” species are targeted in fisheries in the area and the reason for their absence from 2019 catches is unknown. Harders were present in Langebaan lagoon samples in similar numbers to previous surveys, but catches of all other common species, particularly gobies remained low compared to previous surveys.

Previously fish abundance at sites within or near 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. 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 (Horton 2018).

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 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, 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, however, suggest that the protection afforded by the Langebaan MPA is not be enough to sustain the fishery at the current high effort levels. Arendse (2011) found the adult stock to be overexploited using data collected during 2006-08 already, and the evidence from the seine net surveys conducted since then certainly suggests that recruitment overfishing has occurred. The annual seine net surveys can act as an early warning system that detects poor recruitment and allows for timeous adjustments in fishing regulations to reduce fishing mortality on weak cohorts and preserve sufficient spawner biomass. The consistent declining trend in juvenile white stumpnose abundance in the nursery surf-zone habitats since 2007, and the observed declines in commercial linefish CPUE, strongly supports 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. This is the fifth time Anchor Environmental are making this recommendation in the State of the Bay Report and these recommendations are now also supported by a more statistically comprehensive analysis of fishery dependent and survey data (Parker *et al.* 2017). 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).

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. At some point fishing mortality will need to be contained, if the Saldanha Bay fisheries are to remain sustainable. We think that point arrived at least six years ago for the white stumpnose fishery and recommended 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 SANParks and DAFF, and we think this approach 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 (2018) for a reduction in harder fishing effort and gear changes (increase in minimum mesh size) to facilitate stock recovery which will have socio-economic and ecological benefits.

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 popular white stumpnose fishery is 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 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

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).

Except for 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 arrival and spread of the Mediterranean mussel.

On the islands of Saldanha Bay, populations of all these species then started to decline, particularly, the penguins, gannets and kelp gulls, which have declined to 9%, 42% 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 Crowned 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.

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 key species continues to be monitored in future and that these data be made available and used as an indication of environmental conditions in the area.

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 and many other factors 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. Because 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.

Robinson *et al.* (2016) lists 89 alien species as being present in this country up until 2014, 53 of which are considered invasive i.e. population are expanding and are consequently displacing indigenous species. At least 28 alien and 42 invasive species occur along the West Coast of South Africa. 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.* The abundance of *M. galloprovincialis* on rocky shores in Saldanha Bay has been decreasing in the last few years, although the reason behind this decline is still not clear. This trend has, however, been recorded for *M. galloprovincialis* in the past. Recent studies on predator naivety found that native predators prefer native mussels to aliens and as such, are not controlling the invasive mussel population as previously thought. Instead, predators might indirectly be facilitating the invasion by these mussels by removing inter-specific competition with the native mussel. *Balanus glandula*, on the other hand, has shown an increase in abundance over time and remains one of the more abundant species on the mid-shore in Saldanha Bay. *Semimytilus algosus* was recently shown to occur exclusively sub-tidally in sheltered areas such as Saldanha Bay (Skein *et al.* 2018a). Indeed, *S. algosus* is absent in the intertidal zone in Saldanha Bay, but has previously been observed on mussel rafts in the Bay. It is therefore recommended that sub-tidal surveys are conducted to ascertain whether populations have indeed established in Saldanha Bay. Findings from this study suggest that *P. occidentalis* is now well established and slowly increasing in number over time in both Big Bay and Small Bay. At one location in Big Bay, it has shown an unexpected exponential increase in its abundance over the past decade, with numbers now exceeding 1500 individuals/m^2 at this site. In addition, it may be in the process of expanding into more exposed and deeper habitats outside of the Bay, including Danger Bay. This increase in abundance of *P. occidentalis* in the Bay and its presence again this year in Langebaan Lagoon, raises concern and highlights the need for management action. An additional 41 species are currently regarded as cryptogenic (of unknown origin and potentially introduced) but very likely introduced to South Africa.

Of these, 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 these cryptogenic species.

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. In addition to routinely monitoring changes in the population structure of these aliens throughout Saldanha Bay, in depth studies investigating pathways and biological traits associated with their invasion success and their impact upon the community structure of the surrounding native biota, are required. These will not only contribute towards our understanding of the drivers and traits governing their successful invasion, but also give insight into their associated impacts. In turn, this will support directed management actions in order to successfully control invasions and mitigate impacts.

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 also 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 maintain 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.

TABLE OF CONTENTS

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
CMP	Coastal Management Programme
COD	Chemical Oxygen Demand
CFU	Colony-Forming Unit
CSIR	Council for Scientific and Industrial Research
CWAC	Co-ordinated Waterbird Counts
CWDP	Coastal Water Discharge Permit
DAFF	Department of Agriculture, Forestry and Fisheries
DEA	Department of Environmental Affairs
DEA&DP	Western Cape Department of Environmental Affairs & Development Planning
DoE	Department of Energy
DWS	Department of Water and Sanitation
EA	Environmental Authorisation
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
FPP	Floating Power Plant
ICMA	National Environmental Management: Integrated Coastal Management Act (No. 24 of 2008)
IDZ	Industrial Development Zone
CNG	Compressed Natural Gas

LNG	Liquefied Natural Gas
LPG	Liquid Petroleum Gas
MLRA	Marine Living Resources Act (No. 18 of 1998)
MPA	Marine Protected Area
Mtpa	Million tons per annum
NEMA	National Environmental Management Act (No. 107 of 1998)
NEMBA	National Environmental Management: Biodiversity Act (No. 10 of 2004)
NOAA	National Oceanic and Atmospheric Administration
NWA	National Water Act (No. 36 of 1998)
PAH	Poly-Aromatic Hydrocarbons
PSU	Practical Salinity Unit
RWQO	Receiving Water Quality Objectives approach
SBIDZ	Saldanha Bay Industrial Development Zone
SBM	Saldanha Bay Municipality
SBWQFT	Saldanha Bay Water Quality Forum Trust
TNPA	Transnet National Ports Authority
TOC	Total Organic Carbon
TON	Total Organic Nitrogen
TPH	Total Petroleum Hydrocarbon
TSS	Total Suspended Solids
VRF	Vessel Repair Facility
WCDM	West Coast District Municipality
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 (2019) report is the 12th 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 and 2018).

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. This chapter also reports on erosion monitoring results along Langebaan Beach, which was initiated by the Saldanha Bay Local Municipality in 1996. The SBWQFT restarted this monitoring programme in 2018 after the programme was terminated in 2017.

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 that has been conducted in Saldanha Bay, Danger Bay and Langebaan Lagoon with further interpretation of the implication of the changing sediment composition over time and/or related to dredging events.

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

Chapter Nine 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 Ten 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 Eleven 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.

Chapter Twelve summarise 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 Thirteen 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 2019 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 year, 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 (2018) 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 2018 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

- A summary of the Chapter with reference to previous reports. This chapter has been updated this year and now includes erosion monitoring results along Langebaan Beach, which was initiated by the Saldanha Bay Local Municipality in 1996. The SBWQFT continued monitoring in 2018 after the programme was terminated in 2017.

Chapter 5: Groundwater

This is the third year that this new addition appears in the State of the Bay report and serves to highlight the importance of groundwater for Saldanha Bay and Langebaan Lagoon. It also presents information on the use of groundwater in this region and the 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 our own continuous water quality monitoring data.

Chapter 6: Water quality

- New information on variations in temperature, salinity, dissolved oxygen and turbidity in the Bay.
- New updated information on levels of microbial indicators (faecal coliforms and *E. coli.*) in the Bay.
- New updated information on levels of trace metals in mussels on the shoreline and offshore mariculture facilities.
- Trace metals accumulated in bivalve tissue (Bezuidenhout et al. 2015)

Chapter 7: Sediments

- New information on grain size composition and health of benthic sediment in Saldanha Bay (TOC and Nitrogen, Trace metal and hydrocarbon content).

Chapter 8: Benthic macrofauna

- New information on species composition, abundance, biomass and health of benthic macrofauna communities in Saldanha Bay and Langebaan Lagoon.

Chapter 9: Intertidal invertebrates (Rocky Shores)

- New information on species composition, abundance, biomass and health of rocky intertidal invertebrate communities in Saldanha Bay and Langebaan Lagoon.

Chapter 10: Fish

- New information on species composition, abundance, biomass and health of fish communities in Saldanha Bay and Langebaan Lagoon.

Chapter 11: Birds

- New information on species threat status, composition, abundance and health of birds breeding and feeding in Langebaan Lagoon.

Chapter 12: Alien invasive species

- New information on the number, distribution and abundance of alien invasive marine species in Saldanha Bay and Langebaan Lagoon.
- New published information on relevant alien species ecology, spread, abundance and their ability to impact biodiversity as ecosystem engineers.

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 Quality: 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.

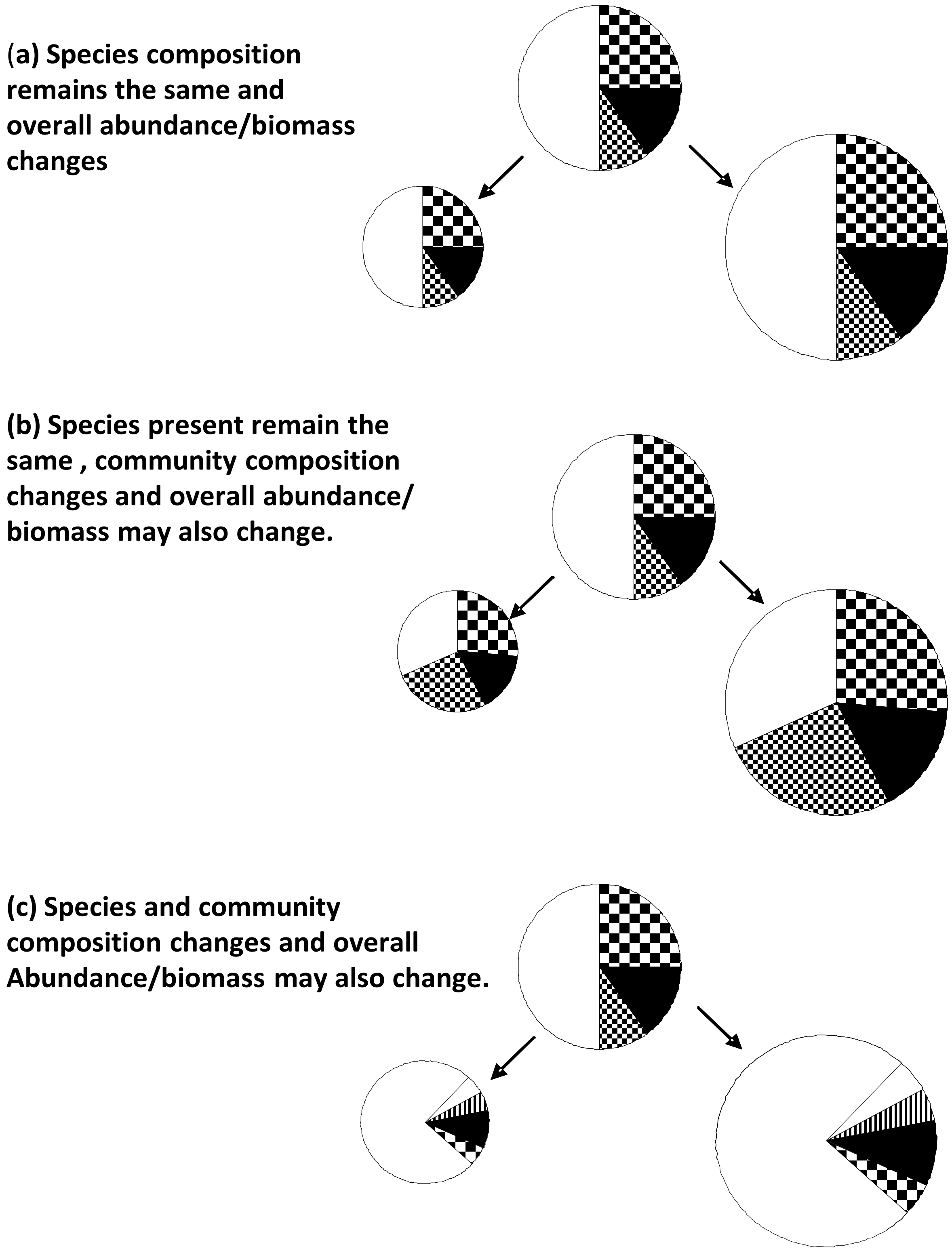


Figure 2.1. Possible alterations in abundance/biomass and community composition. Overall abundance/biomass is represented by the size of the circles and community composition by the various types of shading. After Hellawell (1986).

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.

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. With the ongoing drought in the Western Cape, however, industry and local municipalities are coming 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. Arcelor Mittal now represents the highest consumer of treated wastewater from the Saldanha Bay Wastewater Treatment Works. Arcelor Mittal constructed a Reverse Osmosis plant, which treats wastewater such that it can be used for cooling steel production equipment.

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 430 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 is establishing 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. Currently farmed areas will be incorporated into the ADZ comprising 884 ha set aside for mariculture (currently farmed areas will be incorporated into the ADZ).

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 insert reference of). 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.

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 (2021) and long-term (Beyond 2044) goals for the development of the bay are shown in Figure 3.5 and Figure 3.6. Note that updated National Port Plans were published for comment in August 2019, the final plans have, however, not been published to date.

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).

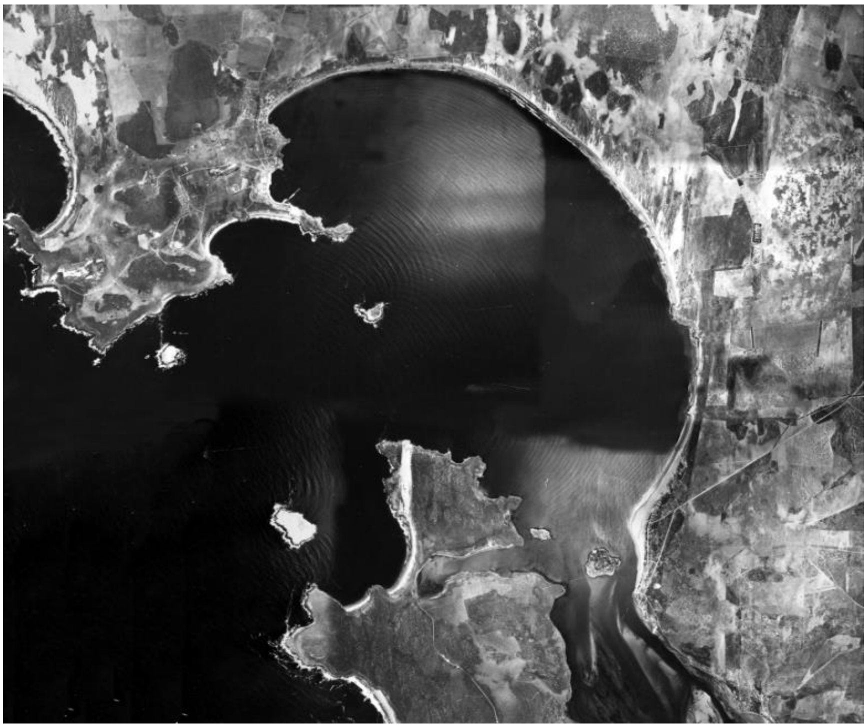


Figure 3.1 Composite aerial photo of Saldanha Bay and Langebaan Lagoon taken in 1960. (Source Department of Surveys and Mapping). Note the absence of the ore terminal and causeway and limited development at Saldanha and Langebaan.

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.

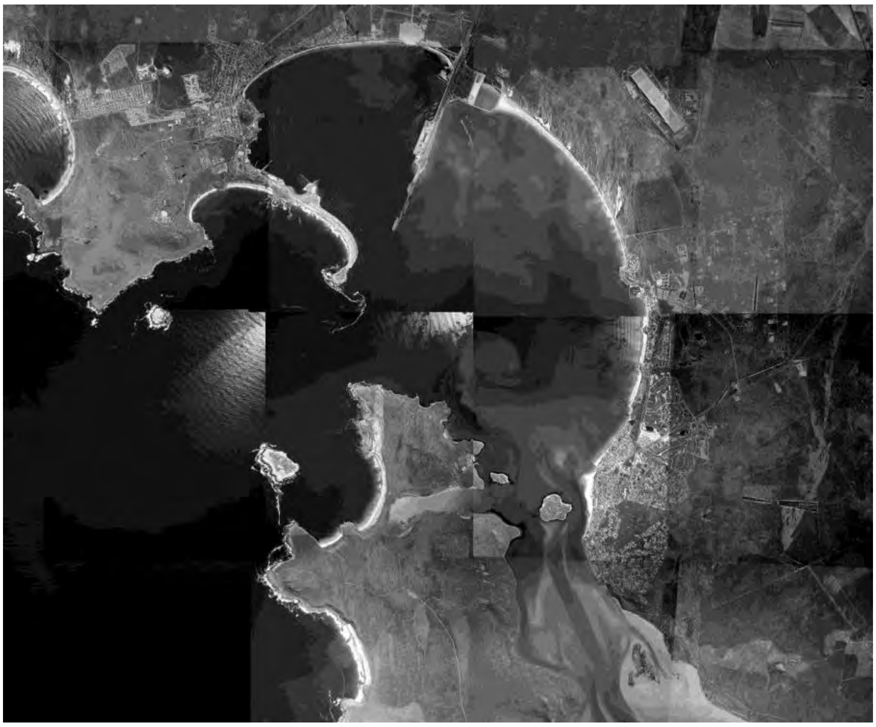


Figure 3.3. Composite aerial photo of Saldanha Bay and Langebaan Lagoon taken in 2007. (Source Department of Surveys and Mapping). Note expansion in residential settlements particularly around the town of Langebaan.

Figure 3.4. **Current layout of Transnet Saldanha Bay Port (Source: Transnet National Port Authority 2015, National Port Plans).**

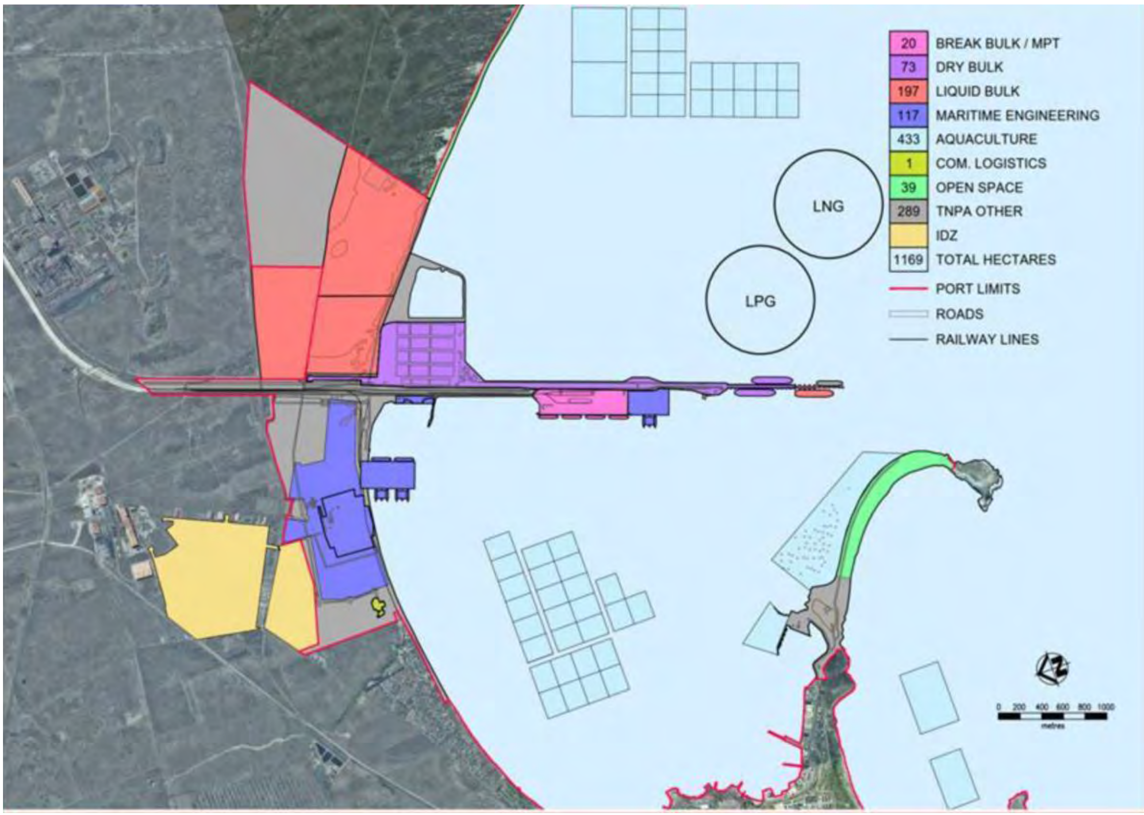


Figure 3.5. **Short term layout (2021) of Transnet Saldanha Bay Port (Source: Transnet National Port Authority 2015, National Port Plans 2015).**

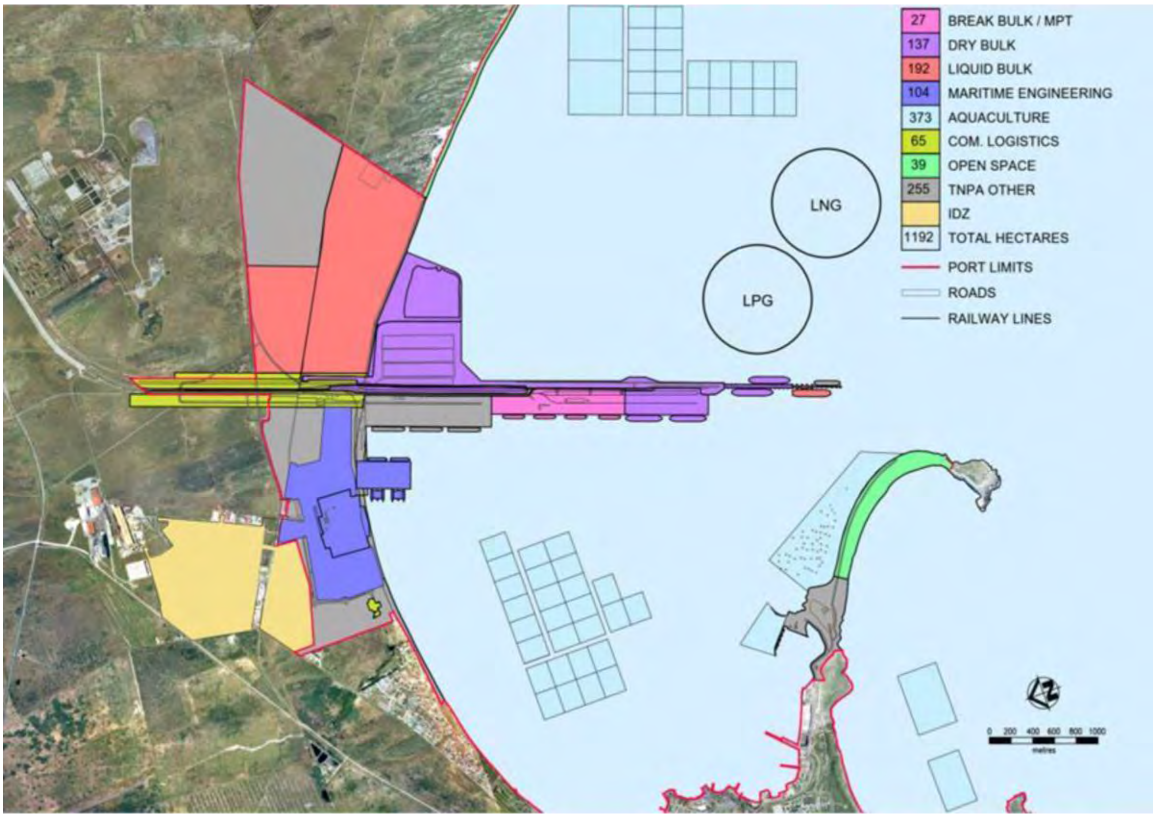


Figure 3.6. Long term layout (2044) of Transnet Saldanha Bay Port (Source: Transnet National Port Authority 2015, National Port Plans 2015).

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. **Composite aerial photograph of Langebaan showing absence of development set-back zone between the town and the lagoon. Source: Department of Surveys & Mapping, South Africa.**

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 (2005-2019) on numbers of visitors to the West Coast National Park (WCNP) indicate strong seasonal trends in numbers of people visiting 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.

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

Visitor numbers have been increasing at an average rate of 12% per annum since 2005[2], peaking in the 2016-2017 period with a total of just over 337 thousand visitors. Since then, the total number of visitors to the park has been decreasing steadily to 281 thousand visitors in 2018/2019 (Figure 3.10). The number of free guests has been increasing steadily over time and now equals the proportion of day guests. The number of international visitors has stayed relatively constant over time while popularity of overnight stays inside the park has decreased substantially after 2009, reaching lowest numbers in 2015/2016 with 2 041 guests. However, overnight visitor numbers have increased over the last two years, reaching 4633 visitors in 2017/2018. 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% of the total number of visitors to the West Coast National Park in 2018/2019.

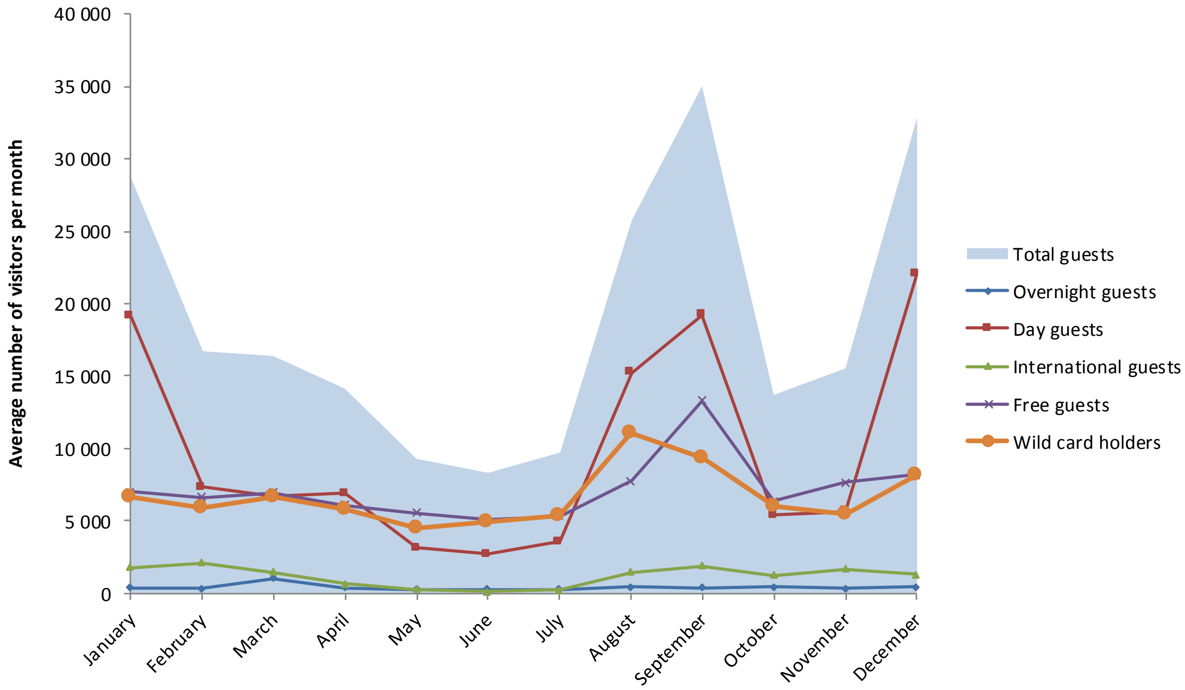


Figure 3.9. Monthly average numbers of tourists visiting the West Coast National Park between July 2005 and June 2019. Day guests include all South African visitors (adults and children), while Overnight guests refer to those staying in SANPARK accommodation. International guests include all SADC and non-African day visitors (adults and children) while the category 'Other' includes residents, staff, military, school visits, etc. Note that SANParks tourism data is now managed by national head office and the reporting structure has changed. Only in total number of guests and wild card holders are now recorded (Source: Pierre Nel, WCNP).

[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 periods from July 2005 until June 2019.

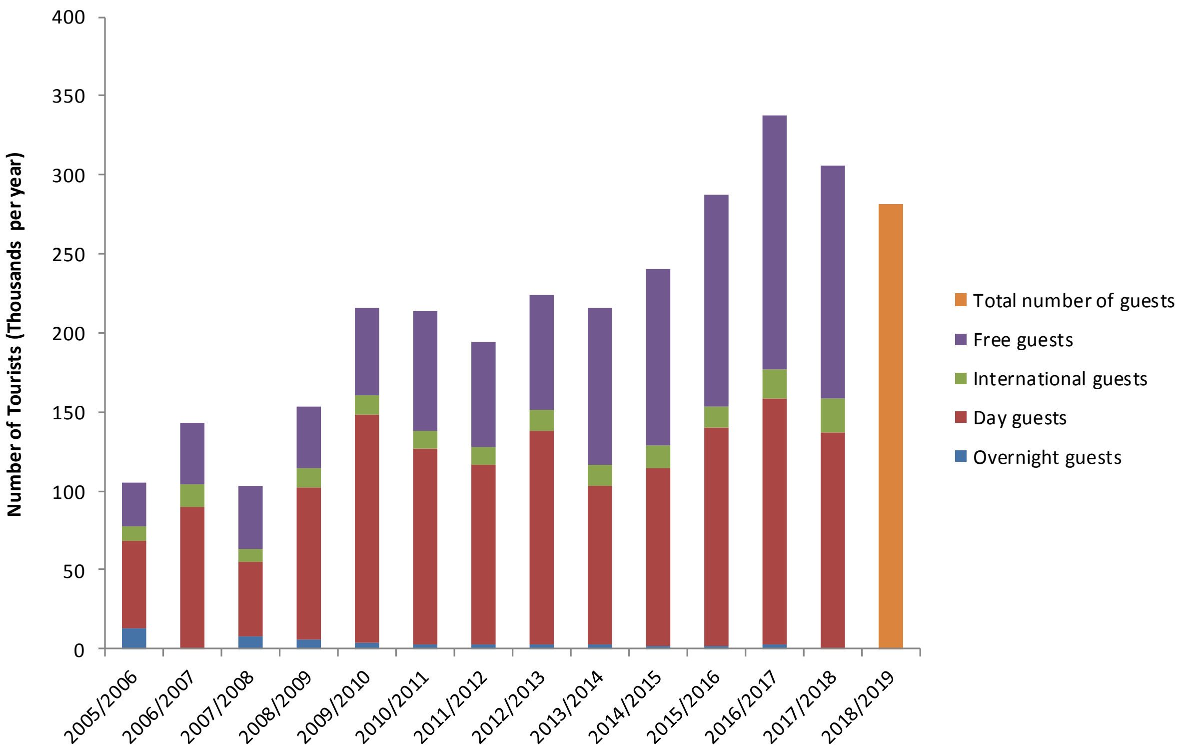


Figure 3.10. Numbers of tourists visiting the West Coast National Park in a rolling 12-month periods from July 2005 until June 2019. Day guests include all South African visitors (adults and children) while Overnight guests refer to those staying in SANParks accommodation. International guests include all SADC and non-African day visitors (adults and children) while the category 'Other' includes residents, staff, military, school visits, etc. Note that SANParks tourism data is now managed by national head office and the reporting structure has changed. Only in total number of guests and wild card holders are now recorded (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 2012-2017 IDP. The latest SDF for the Saldanha Bay Municipality (SBM) was produced in 2011 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.

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 e 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).

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 in order to transport this product globally. In 2017, 282 iron ore ships arrived and departed from the iron ore terminal in the Port of Saldanha, exporting 55.3 million tonnes of iron ore (Section 3.3).

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*)*.

Avedia Energy is in the process of developing a land based liquid petroleum gas storage facility on Portion 13 of Farm Yzervarkensrug No. 127 in Saldanha. The storage facility will 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).

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 3.2.4) (ERM 2015a and 2015b). ERM provided stakeholders with a Background Information Document in October 2015 of which excerpts and illustrations are provided below (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).

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.

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, intend to construct and operate 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 is set for 11 September 2019.

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 late 2019 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 the groundwater and benthic macrofauna chapters (Chapter 5 and 8).

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. The Environmental Impact Report has not yet been published.

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 have increased steadily from 20.7 to 57.3 million tonnes between 2003 and 2019 (note that annual metal export is calculated based on the fiscal year, i.e. April-March) (Figure 3.12).

Metal exports from the MPT have increased steadily since 2007 (Figure 3.13 and Figure 3.14). Initially only lead, copper and zinc were exported from the MPT, with lead comprising the largest proportion of the exported material in 2011 (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 tons between 2010 and 2013 before dropping by nearly half in 2014-2016. Lead exports have since recovered to approximately 60 thousand tons per annum. Zinc exports picked up in 2011, roughly equalling lead exports with an average of 62 thousand tonnes per annum over the last five years (Figure 3.14). Copper is exported in small quantities compared to all other metal ores although exports first steadily increased after 2011, peaking in 2015 at 26.7 thousand tonnes. Since then zinc exports have averaged around 22 thousand tons. 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).

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 from the MPT in Saldanha Bay only commenced in 2013 (95 thousand tonnes) and has increased by more than one third each year, totalling 4.1 million tonnes in the 2019 financial year (Figure 3.13), comprising 96% of the total metal exported from the MPT. In 2016, manganese exports from the Saldanha Bay MPT represented 15% of the total amount exported from South Africa.

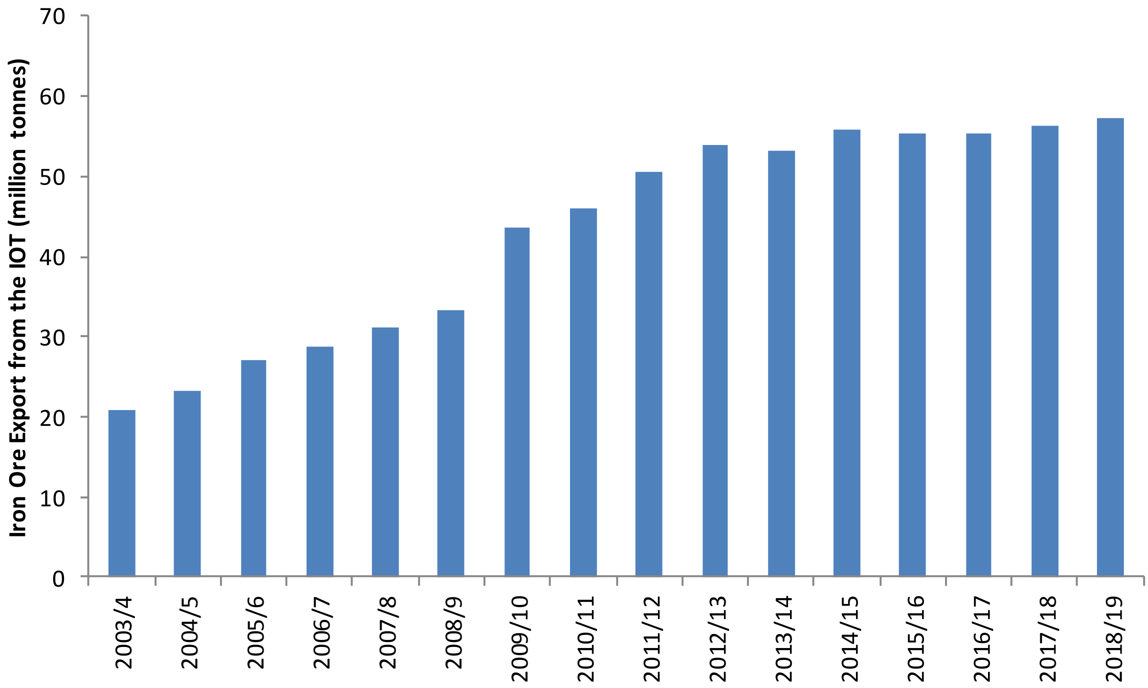


Figure 3.12 **Annual exports of iron ore from the iron ore terminal at the Port of Saldanha between April 2003 and March 2019. (Data provided by Rejean Viljoen, Transnet Port Authority 2019).**

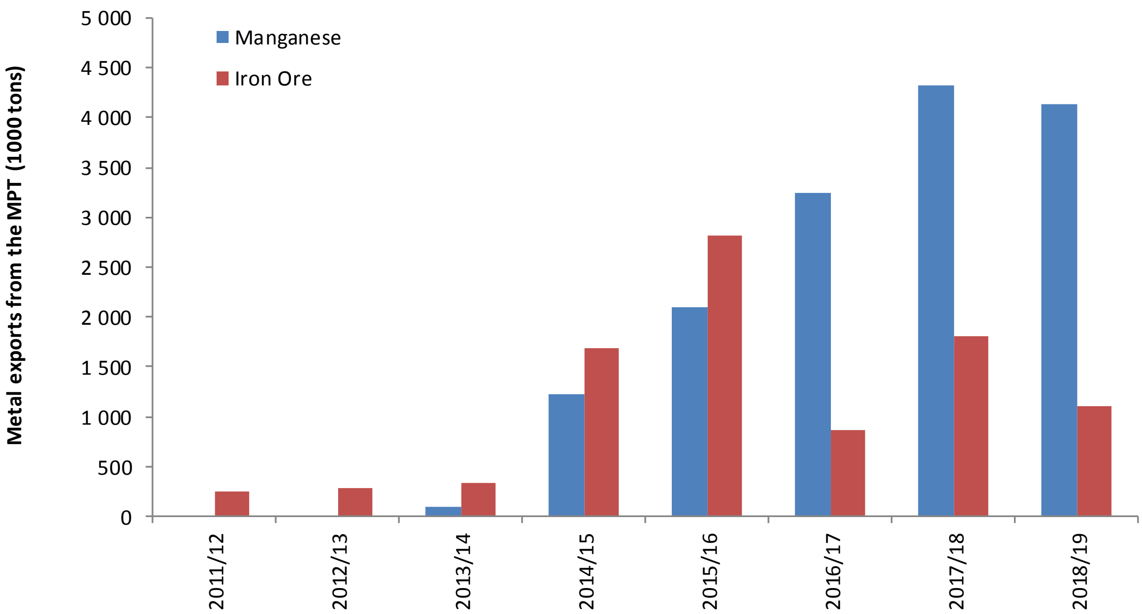


Figure 3.13 **Annual exports (April 2011 – March 2019) of manganese and iron ore from the *multi-purpose terminal* at the Port of Saldanha Bay (Data provided by Rejean Viljoen, Transnet Port Authority 2019).**

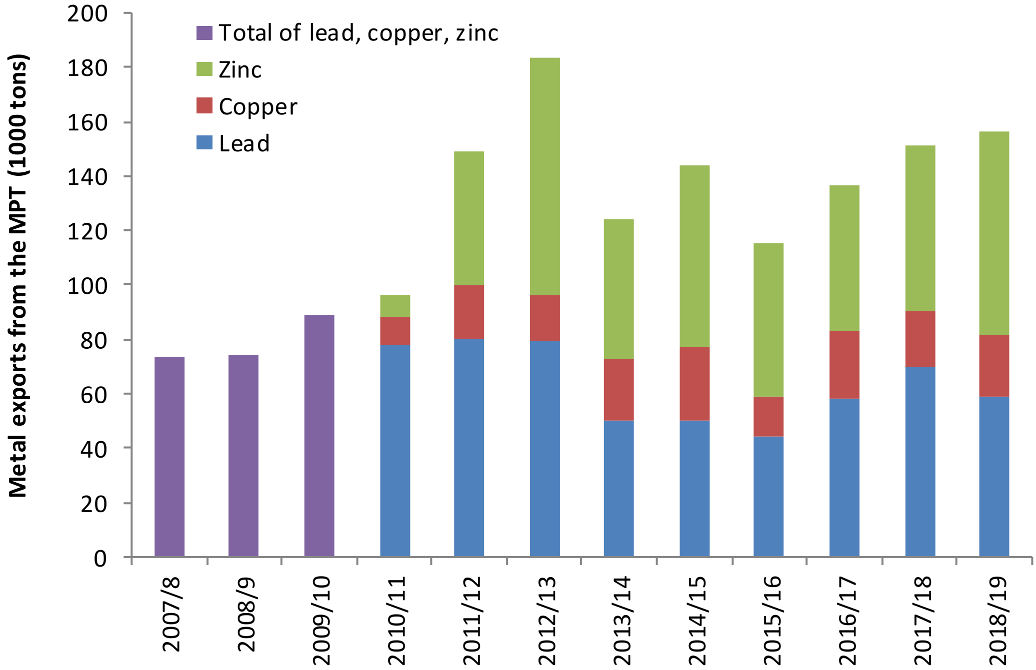


Figure 3.14 **Annual exports (April 2007 – March 2017) of lead, copper and zinc from the *multi-purpose terminal* at the Port of Saldanha Bay. Note that separate data for these commodities was only available for April 2010-March 2019 (Data provided by Rejean Viljoen, Transnet Port Authority 2019).**

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 tons of iron ore per annum. 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 tons 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.

Transnet currently holds a Provisional Air Emission License (PAEL) for the storage and handling of ore and coal at the Multi-Purpose Terminal (MPT), which was issued on 26 September 2018 and is valid for period of 12 months. According to the conditions of the PAEL, the holder of the license is entitled to an AEL when the commissioned facility has been in full compliance with conditions and requirements of the PAEL for a period of at least six months. The holder of the license may also choose to extend or renew the PAEL. It is currently not known if Transnet Port Terminals will apply for an AEL. 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 remain 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 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 consumer 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 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.

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

The most recent construction-related dredging occurred between July 2015 and October 2016, where a total of 25 000 m^3 of sediment was dredged for the expansion of the General Maintenance Quay.

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 close to 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 and zinc concentrates, and the Sea Harvest/Cold Store terminal that is dedicated to frozen fish products (Figure 3.4). There are also facilities for small vessel 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).

A recent update on the number of alien marine species present in South Africa lists 89 alien species as being present in this country, of which 53 are considered invasive i.e. population are expanding and are consequently displacing indigenous species. At least 28 alien and 42 invasive species occur along the West Coast of South Africa. 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 insert reference of this report.

Other potentially 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 insert reference for information).

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 Draft Bill has not yet been promulgated, however. 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).

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 (Figure 3.15). Average vessel size increased over the years (Figure 3.17) and as a result, the volume of ballast water discharged almost tripled between 2000 and 2011 from 8.4 to 21.1 million tons (Figure 3.16). Since 2011, ballast water discharge per vessel has remained stable around 70 thousand tons for vessels docking at the Iron Ore Terminal (Figure 3.17). Vessels docking at the Multipurpose Terminal, however, continued increasing in size until 2014/2015 and have since stabilised with individual vessels discharging approximately 10 thousand tons (Figure 3.17).

The number of vessels entering the port stabilised between 2011 and 2017 but increased steeply by almost 150 vessels in the last two years with 616 ships visiting the port between July 2018 and June 2019 (Figure 3.15). Overall, iron ore tankers contributed 51% to the observed vessel traffic and 91% to the total water discharged between July 2018 and June 2019 (Figure 3.15 and Figure 3.16). Iron ore tankers are large vessels and hold the highest quantities of ballast water.

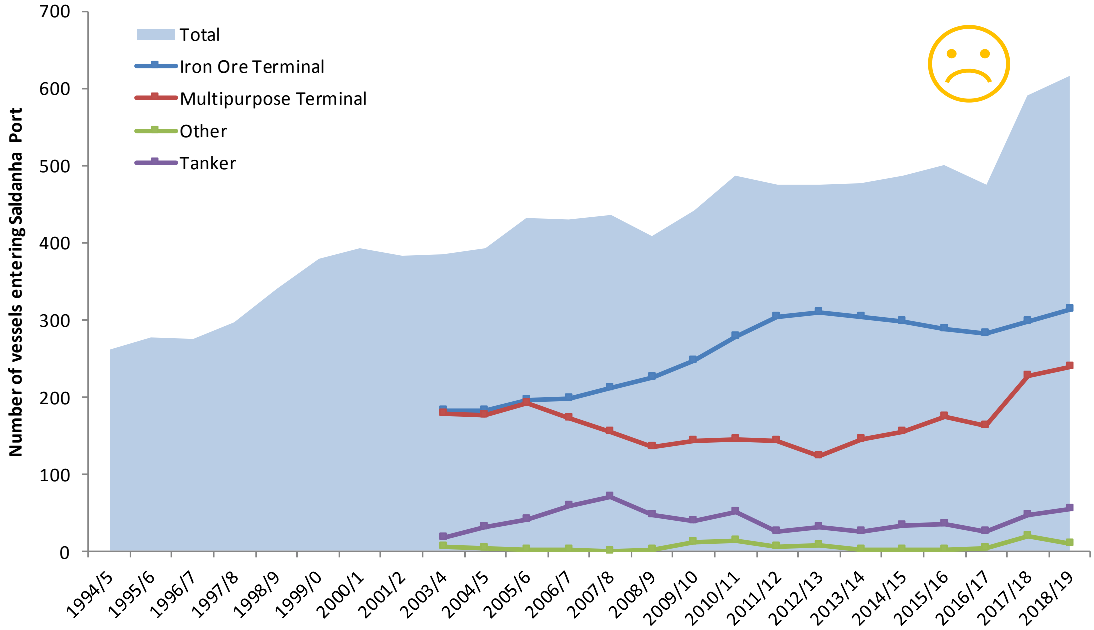


Figure 3.15. **The numbers and types of vessels entering Saldanha Port. The total number of vessels entering Saldanha Port between July 1994 and June 2019 is shown as the blue area. The numbers of vessels docking at the iron ore terminal, the *multi-purpose terminal*, tankers and other vessels are shown 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 2003-2019).**

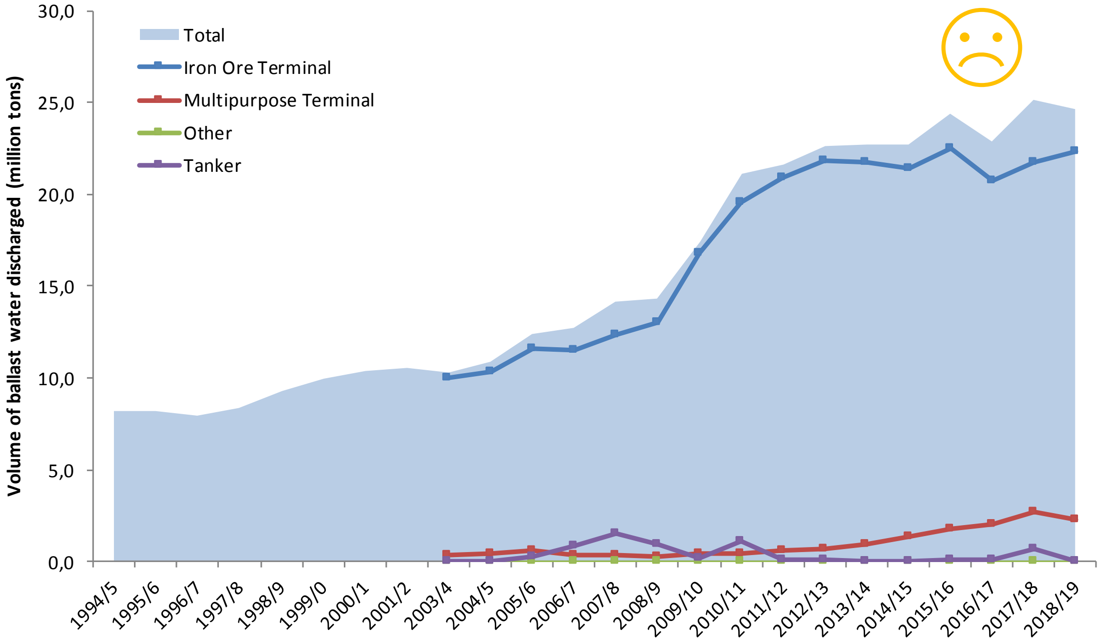


Figure 3.16 **Volumes of ballast water discharged into Saldanha Port. The total amount of ballast water discharged in Saldanha Port between the years 1994 and June 2019 is shown as the blue area. Ballast water discharged by vessels docking at the iron ore terminal, the *multi-purpose terminal*, tankers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

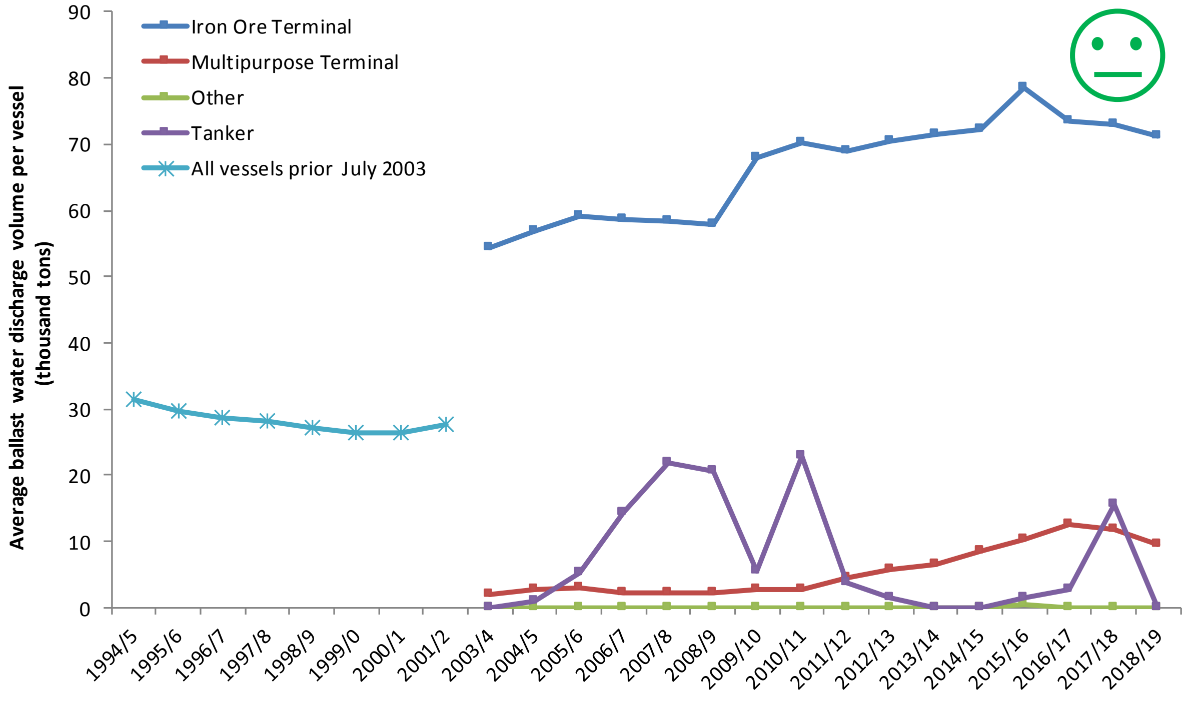


Figure 3.17 Average ballast water volumes discharged per vessel into Saldanha Port. The total amount of ballast water discharged in Saldanha Port between the years 1994 and June 2002 is shown as the blue line. Ballast water discharged by vessels docking at the iron ore terminal, the *multi-purpose terminal*, 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 onward (Sources: Marangoni 1998, Awad *et al.* 2003, Transnet-NPA unpublished data 2003-2019).

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.

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 (Make reference to WQ Chapter).

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

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

on land into coastal waters except in terms of a General Discharge Authorisation (GDA) or a Coastal 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 five years has 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 Environmental Affairs 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-high	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-high	Quarterly monitoring
Saldanha Oyster	Permit granted	Holding facility	low	N/A
Oceana Lobster Saldanha	Decision pending	Unknown (processing/holding facility?)	Unknown	N/A
Saldanha Municipality WWTW – Proposed upgrade	Pending (incomplete application)	Treated wastewater	Medium-high	N/A
Transnet Port Terminals	Permit granted	Industrial Storm Water	Medium high	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. This is clearly evidenced by the fact that the WCDM has exceeded its water allocation for the last six years. 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. 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, a recent revision of the feasibility study revealed that the Berg River may have surplus water and an application for additional allocation of water sourced from the Berg River was submitted by the WCDM. If this additional allocation is granted to the WCDM, the desalination plant will be put on hold for the next ten years.

3.6.2.3 ArcelorMittal RO plant

ArcelorMittal is 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 ArcelorMittal Saldanha operations currently require approximately 6 500 m^3/day of freshwater at present, representing approximately 25% of Saldanha Bay municipality potable water total usage. ArcelorMittal Saldanha modified its existing water treatment infrastructure to partially replace its current fresh water supply with treated municipal sewage wastewater (from the Saldanha Wastewater Treatment Works (WWTW)) and groundwater. The intention is to use 3600 m^3 wastewater from the Saldanha WWTW together with groundwater. ArcelorMittal is currently awaiting an outcome on their application for a Water Use License to abstract groundwater for their operations (ArcelorMittal, *Pers. comm.* 2019).

Under normal circumstances installation of such a plant would require an application for Environmental Authorisation in terms of the National Environmental Management Act (NEMA) – i.e. an Environmental Impact Assessment (EIA). However, owing to the prevailing drought in the Western Cape and the fact that the Saldanha Bay Municipal Area was declared a Disaster Area by the Saldanha Bay Council on 15 June 2017, the Department of Environmental Affairs and Development Planning (DEA&DP) has issued a S30A Directive which exempts private sector water provision interventions from having to undertake EIAs for projects of this nature (if their project is included on the Municipality's Water Intervention Plan which is a separate process). However, a Coastal Waters Discharge Permit (CWDP) is still required for disposal of the effluent to the marine environment in terms of the NEMA: Integrated Coastal Management Act (ICMA 2009).

The effluent produced by the Reverse Osmosis plant is near fresh and any contaminants (including harmful pathogens) are removed in the treatment process. Sludge that is produced during the groundwater and wastewater treatment process is discarded at a registered landfill site. ArcelorMittal has been granted a CWDP for the discharge of the effluent into Saldanha Bay off the “Oyster Dam” wall. Phase two of this project will involve the construction of an additional Reverse Osmosis (RO) plant, and the amendment of the current CWDP for additional discharge through their existing infrastructure.

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 grown, 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 were made available on the 2016-2017 Capital Budget for the implementation of various interventions that prevent overflow of raw sewage 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 0 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. 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.

In Saldanha Bay the most important partnership for the re-use of treated effluent was established between the SBM and ArcelorMittal Saldanha Works (Refer to Section 3.6.2.3 for more information). In June 2018 the SBM announced that all effluent, with the exception of 60 m^3 supplied to Blouwaterbaai Lodge via a pipeline system would be supplied to ArcelorMittal. Due to the agreement between ArcelorMittal and the SBM, little to no treated effluent originating from the Saldanha Bay Wastewater Treatment Works currently enters the Bok River and subsequently the marine environment. The exact volume discharged into the Bok River is currently not available.

[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
- 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 has not yet been issued (Gavin Williams, SBM, *pers. comm.*).

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. Little to no effluent has entered the marine environment as a result.

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 March 2018 however, the average daily limit was exceeded 70% of the time, reaching unprecedented levels of 3452 m^3 effluent in August 2014. It is important to note though that the WWTW plant capacity was upgraded to 5000 m^3 some time ago, which means that the effluent quality was not compromised despite regular exceedance of the legal limit. Finally, wastewater volumes treated at the Saldanha Bay WWTW decreased in 2017/2018 due to the water restrictions implemented by the SBM.

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, it is currently unknown when exactly the SBM started allocating treated effluent to different users, thereby dramatically reducing the amount of effluent that is discharged into the otherwise dry Bok River. Overall, based on the information provided by Gavin Williams and ArcelorMittal it appears that the SBM has not been producing enough wastewater to meet the demand by the various users (ArcelorMittal recently upgraded their Reverse Osmosis plant to take up to 3.6 ML of treated effluent per day). A flow meter has been installed at the Bok River discharge point; however, it is not known whether the discharge volume is recorded (this would likely be a requirement of the new water use license if this issued).

The Bok River has been dry for the last 10 years and 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 in future interpretation of water quality results must consider that very little effluent is entering the marine environment.

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 39 occasions since 2003 (23% 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). Although some improvement was evident for the period July 2016-June 2017, Faecal Coliform counts reached maximum detectable limit on four occasions (September-December 2018). No results were available for January and April 2019 (Figure 3.21). Saldanha Bay WWTW was compliant 60% of the time (improvement from 2017/18 where compliance was 50%).

Allowable limits for total suspended solids (TSS) of 25 mg/L have been exceeded 19% of the time since April 2003 (Figure 3.35). While compliance clearly improved between 2008 and 2014, the allowable limit has been exceeded 46% of the time since December 2014. Major improvements are still required to prevent exceedance of the legal limit.

Chemical oxygen demand (COD) in filtered effluent exceeded the allowable limit of 75 mg/L 25% 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 has been consistently above the legal limit since November 2017, achieving only 79% compliance. These observations are congruent with high ammonia nitrogen, faecal coliform and free chlorine levels.

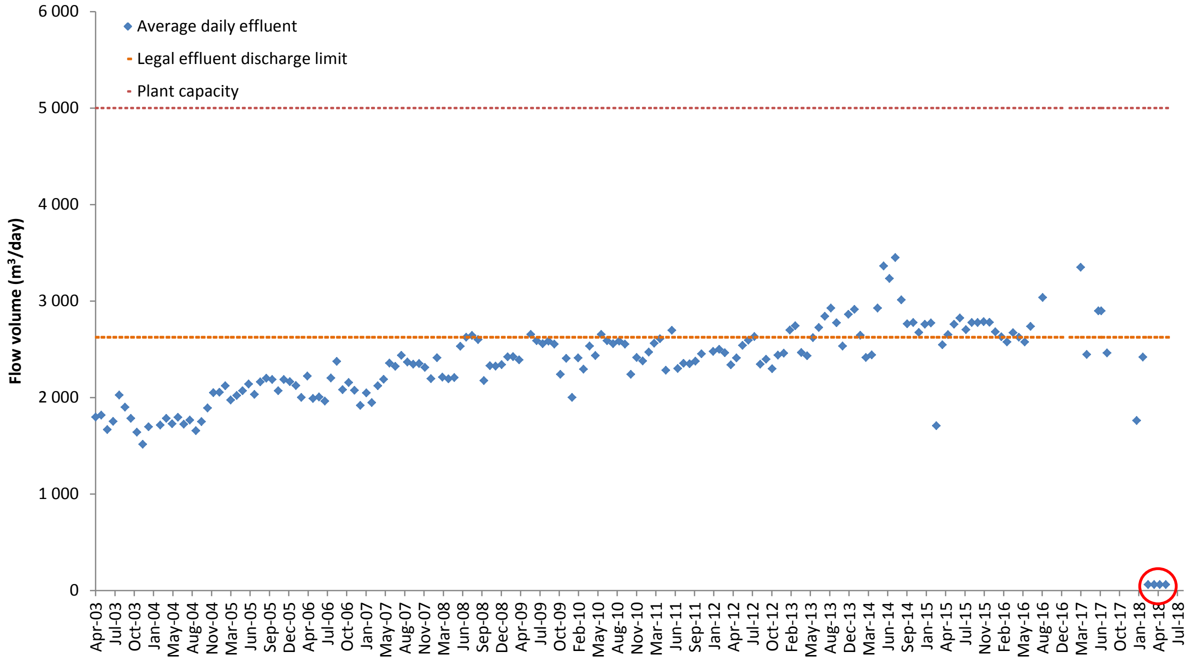


Figure 3.20. **Trend in average daily effluent (m^3/month) released from the Saldanha Wastewater Treatment Works, April 2003-June 2018. Allowable discharge limits in terms of the exemption issued by DWS under the National Water Act (No. 36 of 1998) are represented by the dashed orange line and the design 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).**

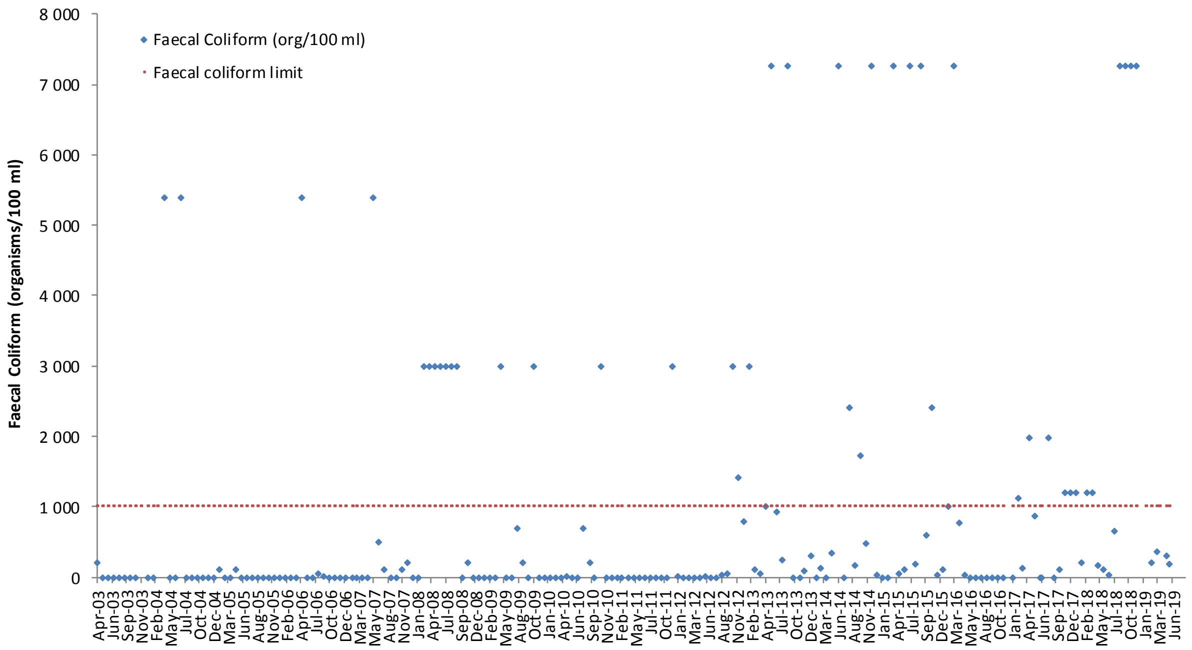


Figure 3.21 **Monthly trend in Faecal Coliforms (org/100ml) in effluent released from the Saldanha Wastewater Treatment Works, April 2003-June 2019. Allowable limits in terms of a General Authorisation under the National Water Act (No. 36 of 1998) are represented by the dashed red line (Source: Saldanha Bay Municipality).**

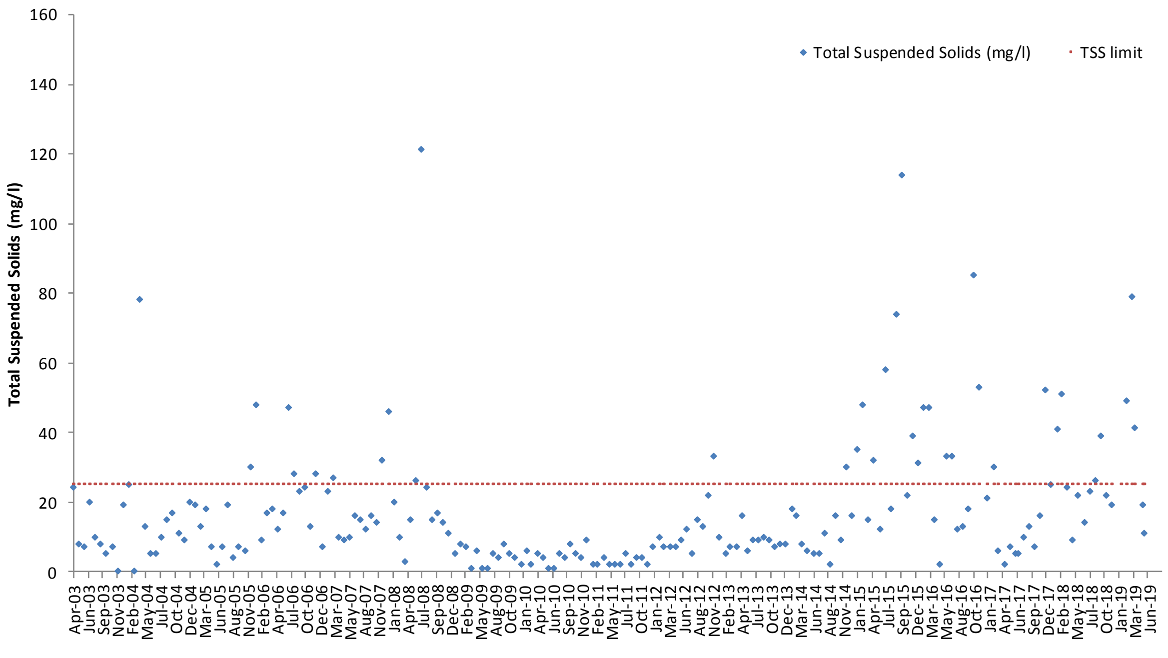


Figure 3.22 **Monthly trend in total suspended solids (mg/L) in effluent released from the Saldanha Wastewater Treatment Works, April 2003 – June 2019. Allowable limits as specified in terms of a General Authorisation under the National Water Act (No. 36 of 1998) are represented by the dashed red line (Source: Saldanha Bay Municipality).**

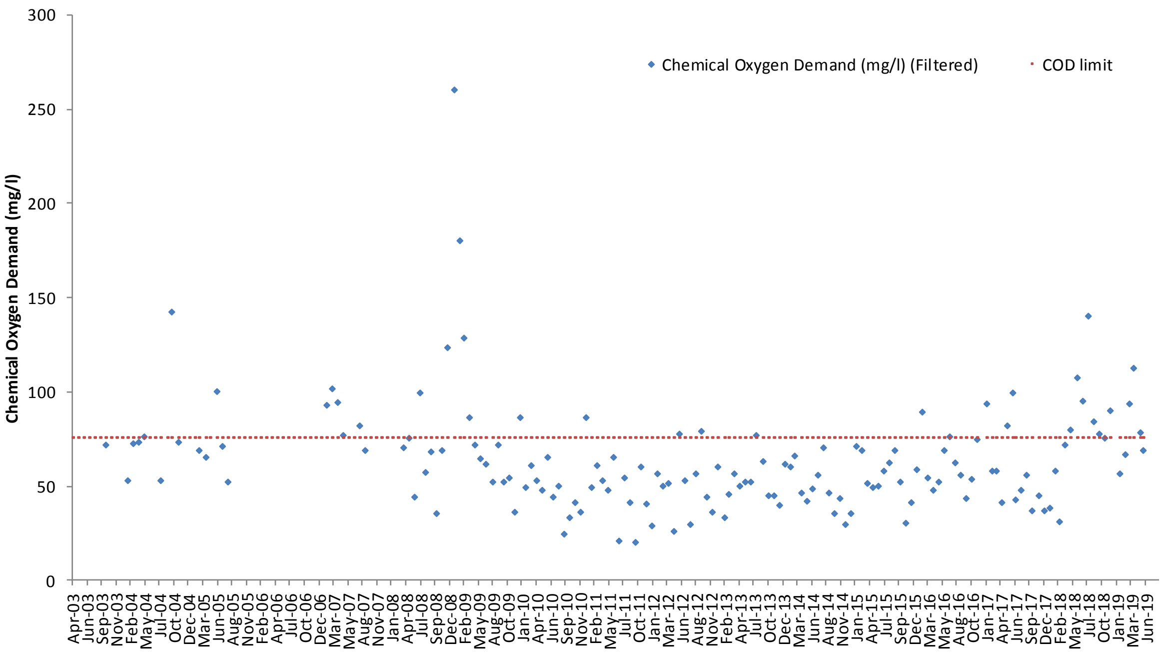


Figure 3.23 **Monthly trends in chemical oxygen demand (mg/L filtered) in effluent released from the Saldanha Wastewater Treatment Works, April 2003-June 2019. Allowable limits in terms of a General Authorisation under the National Water Act (No. 36 of 1998) are represented by the dashed red line (Source: 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 exceed the allowable limit of 6 mg/L, 81% of the time (Figure 3.24.). Ammonia levels in the effluent have not been compliant since November 2017, measuring 91.5 mg/L in October 2018, the highest concentration ever recorded. The average concentration during the period June 2018 to June 2019 was 58.2±28.4 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 15% 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 since November 2017 complement extremely high levels of ammonia nitrogen indicating the lack of bacterial treatment.

The concentration of orthophosphate in the effluent has only been measured since October 2007 showing a distinct seasonal pattern, with the highest values occurring during the summer months and lowest values in winter. This is consistent with the higher influx of visitors during summer. Orthophosphate levels have dropped since February 2013 and the allowable limit of 10 mg/L was only exceeded on eight occasions, most recently in December 2018 and February 2019 (Figure 3.26). However, concentrations have remained just below the legal limit since then.

Permissible chlorine levels of 0.25 mg/L have been exceeded 60% of the time (Figure 3.27) since 2003. However, between July 2018 and June 2019 chlorine levels improved dramatically compared to previous years, where legal limits were 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).

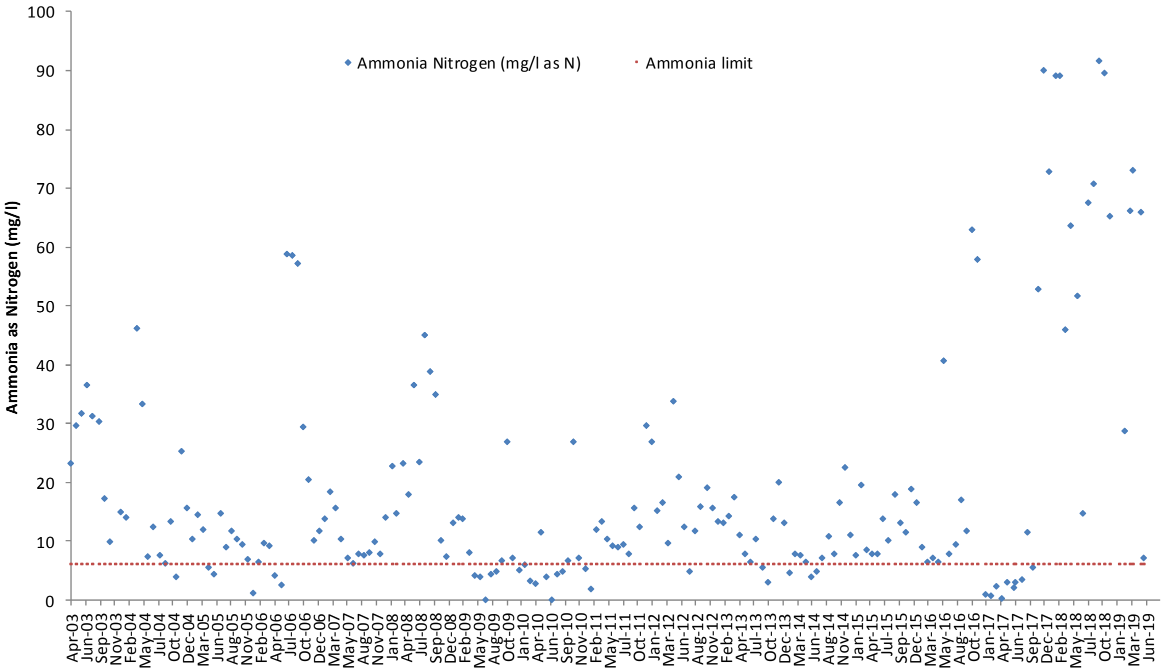


Figure 3.24. Monthly trends in Ammonia Nitrogen (mg/L as N) in effluent released from the Saldanha Wastewater Treatment Works April 2003-June 2019. Allowable limits in terms of a General Authorisation under the National Water Act (No. 36 of 1998) are represented by the dashed red line (Source: Saldanha Bay Municipality).

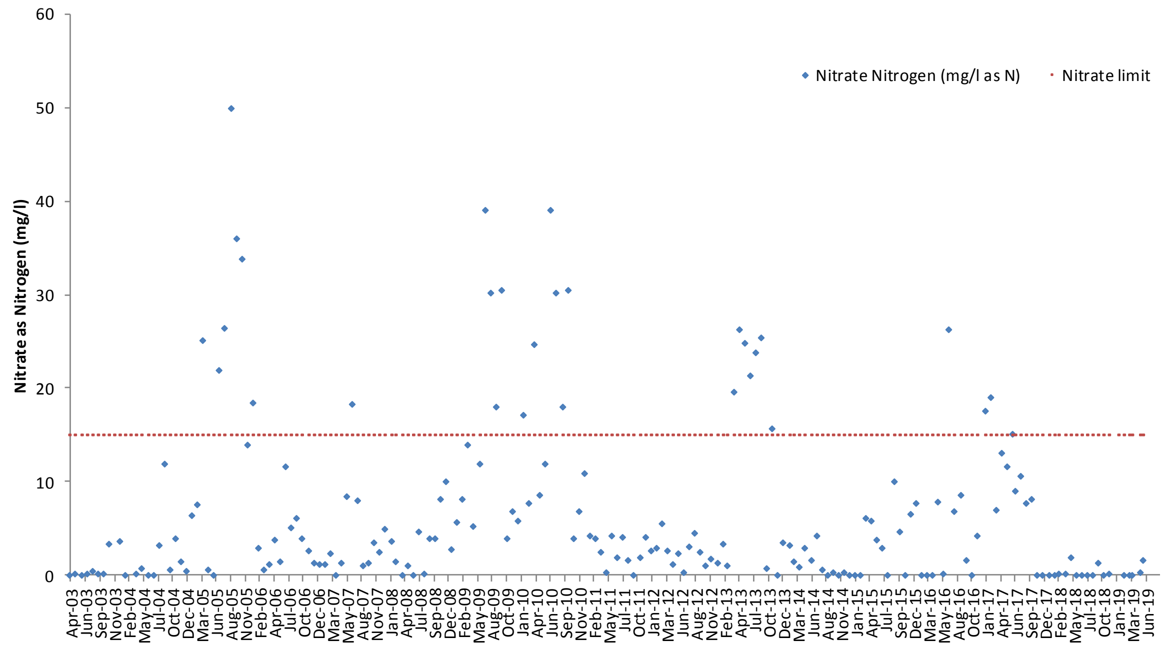


Figure 3.25 Monthly trends in Nitrate Nitrogen (mg/L as N) in effluent released from the Saldanha Wastewater Treatment Works April 2003-June 2019. Allowable limits in terms of a General Authorisation under the National Water Act (No. 36 of 1998) are represented by the dashed red line (Source: Saldanha Bay Municipality).

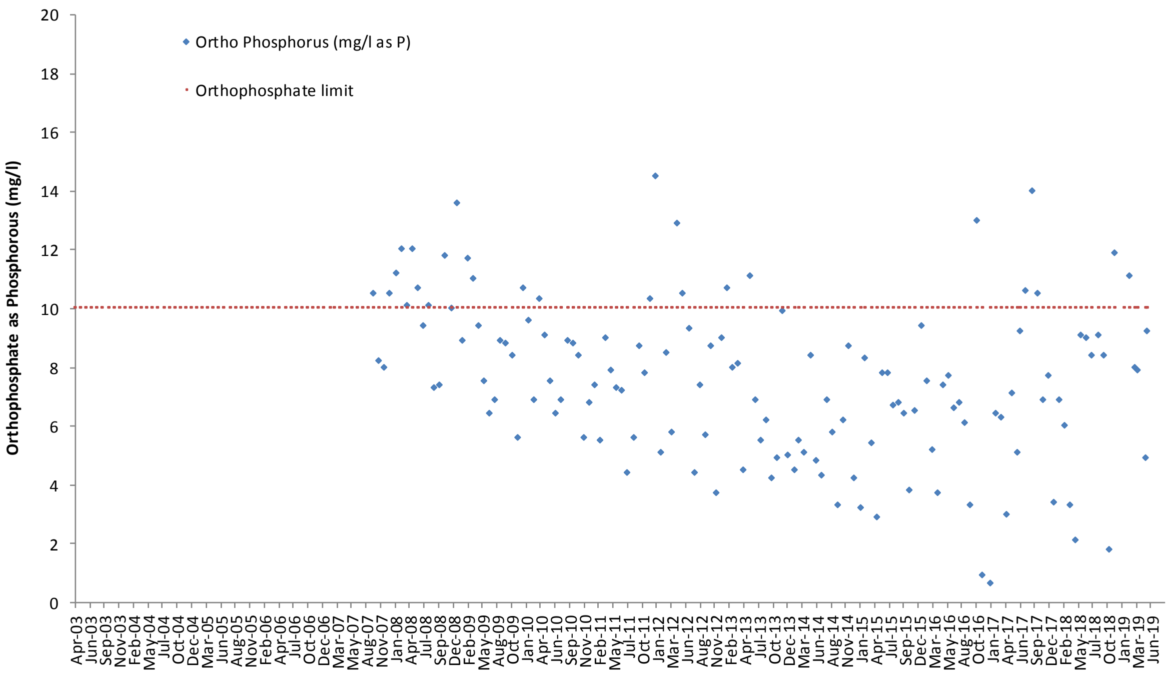


Figure 3.26 Monthly trends in Orthophosphate (mg/L as P) in effluent released from the Saldanha Wastewater Treatment Works April 2003-June 2019. Allowable limits in terms of a General Authorisation under the National Water Act (No. 36 of 1998) are represented by the dashed red line (Source: Saldanha Bay Municipality).

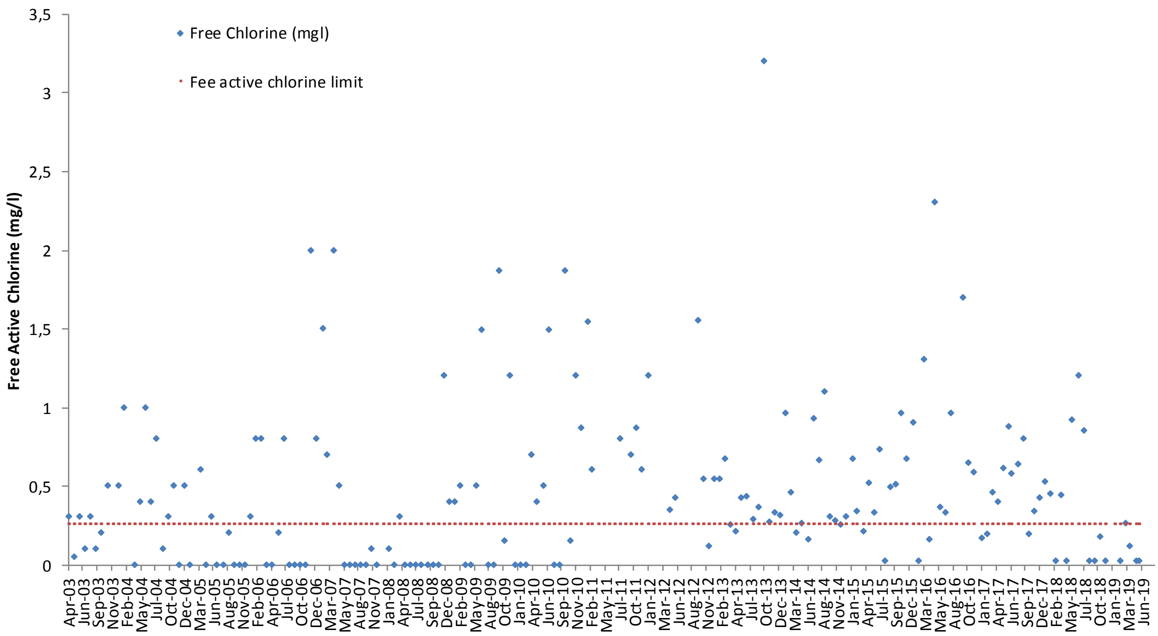


Figure 3.27 Monthly trends in Free Active Chlorine (mg/L) in effluent released from the Saldanha Wastewater Treatment Works April 2003-June 2019. Allowable limits in terms of a General Authorisation under the National Water Act (No. 36 of 1998) are represented by the dashed red line. An outlier of 12 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, since 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). According to the SBM no effluent has entered the MPA over the past few months (Quintin Williams, SBM, *pers. comm.* 2019).

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). 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. The upgrades have increased the plant capacity to 3.5 ML and further upgrades included an additional aeration basin, a new clarifier and drying beds as well as new inlet works. New upgrades commenced in September 2019, including the installation of a new clarifier, inlet works and screens with a total budget of R17 million (SBM, Gavin Williams, *pers. comm.* 2019). Future upgrades will include new infrastructure to increase the capacity of the plant to 5-7 ML (SBM, Gavin Williams, pers. comm. 2016 and 2017). 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 therefore the actual water quality of the treated wastewater entering the MPA via the illegal overflow is currently unknown (note however, that according to the SBM no effluent has entered the MPA during the past winter months). 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 3.31 - 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.) (Note that conditions changed in January 2018). 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 3.29.). The legal limit for effluent production increased to 4 485 m^3 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 was exceeded in January 2019 with an average daily flow of 4167 m^3 per day (i.e. 119% capacity).

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 (6-8 Figure 3.31). The pH of the wastewater effluent is currently more acidic than recommended for the protection of the inshore marine environment (67% of the time the pH < 7.3) (AEC 2015).

The Langebaan WWTW has also been recording electrical conductivity (in mS/m) since 2009. Electrical conductivity has decreased significantly since 2009 from values measuring up to 600 mS/m down to values fluctuating around 200 mS/m. Since electric conductivity reached lower levels in December 2014, the limit was exceeded on 13 occasions (29% of the time). Between July 2018 and June 2019, conductivity has decreased significantly, and the legal limit has not been exceeded October 2018.

COD in filtered effluent exceeded the allowable limit of 75 mg/L 31% of the time since June 2009, reaching an all-time maximum of 235 mg/L in January 2018 (Figure 3.33). While an improvement could be observed between 2015 and 2016, recent measurements show that COD is unacceptably high for the protection of the inshore marine environment. However, 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). In addition, COD has been lower in 2018/19 when compared to recent years and only one reading exceeded the recommended COD limit for the protection of the marine environment (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 with a maximum of 7258 org/100 mL frequently measured from 2013-2015. Overall, however, 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. Faecal coliform measurements have been fluctuating around the 1 000 org/100 mL mark since 2017 (Figure 3.34).

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. 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 67 occasions since 2009 (55% of the time) (Figure 3.35). Overall, TSS levels were highest at the beginning of 2015, frequently exceeding the recommended limit and reaching a maximum of 198 mg/L in March 2015. Since then TSS concentrations have been fluctuating around the 25 mg/L mark with no distinct upward or downward trend. TSS levels roughly follow the trends observed in average daily flow volumes where TSS values are higher when flow is greater.

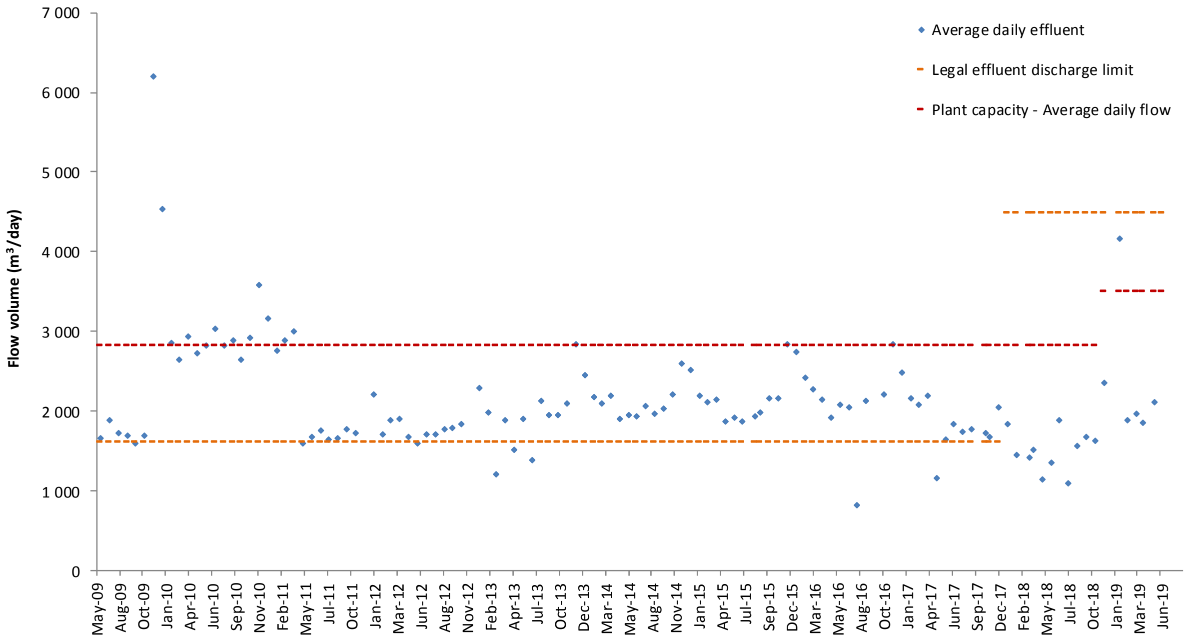


Figure 3.29. **Trends in average daily effluent volume (m³/month) released from the Langebaan Wastewater Treatment Works, June 2009 - June 2018. Allowable discharge limits in terms of the exemption issued by DWAF under the National Water Act (No. 36 of 1998) are represented by the dashed orange line and the design capacity of the plant by the red line (Source: Saldanha Bay Municipality).**

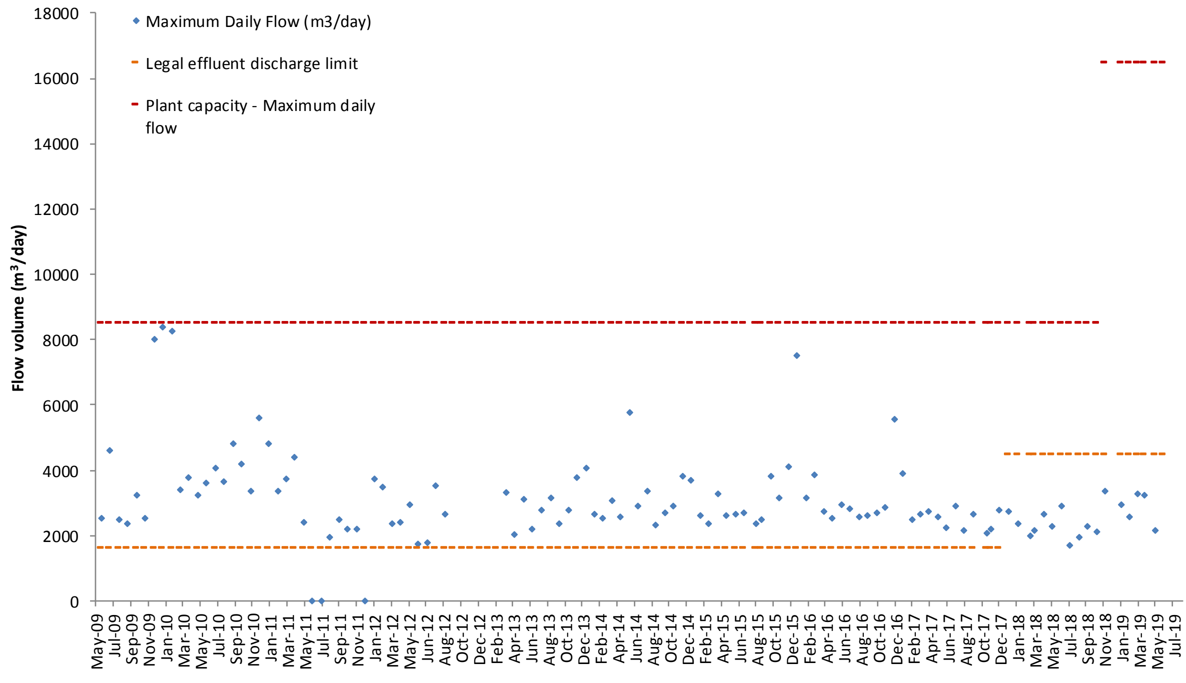


Figure 3.30. **Trends in maximum daily effluent volume (m³/month) released from the Langebaan Wastewater Treatment Works, June 2009 - June 2018. Allowable discharge limits in terms of the exemption issued by DWAF under the National Water Act (No. 36 of 1998) are represented by the dashed orange line 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 2019. 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).

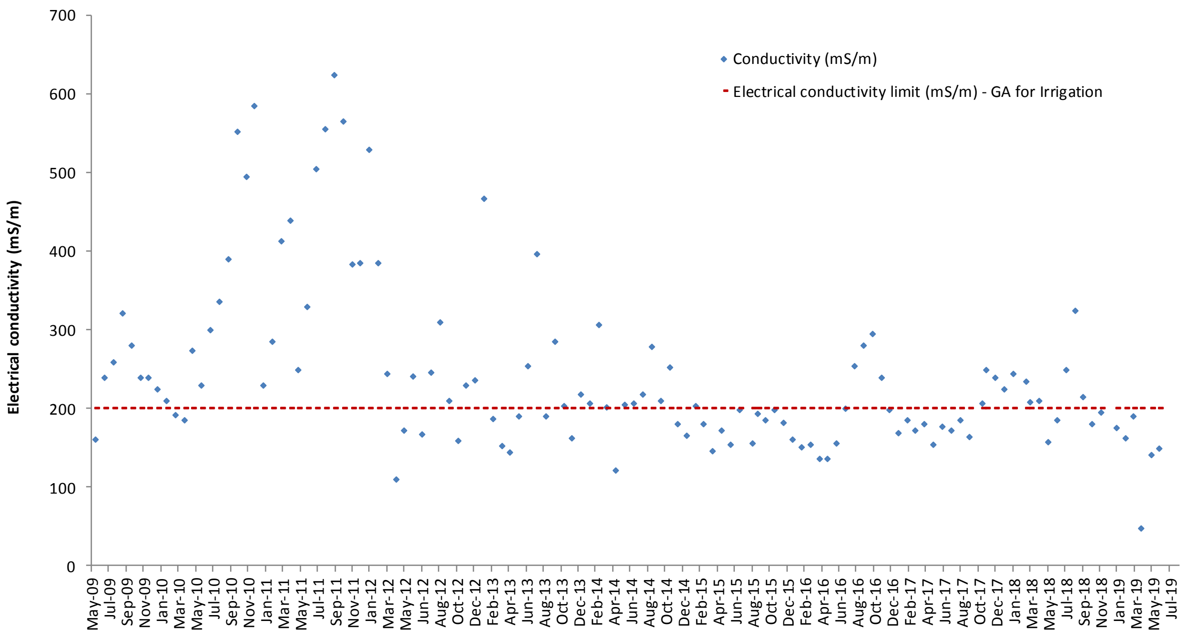


Figure 3.32. Monthly trends in conductivity of effluent from the Langebaan Wastewater Treatment Works, June 2009 - June 2019. The allowable limit in terms of the General Authorisation for irrigation purposes under the National Water Act (No. 36 of 1998) is 200 mS/m and is depicted by the red line (Source: Saldanha Bay Municipality).

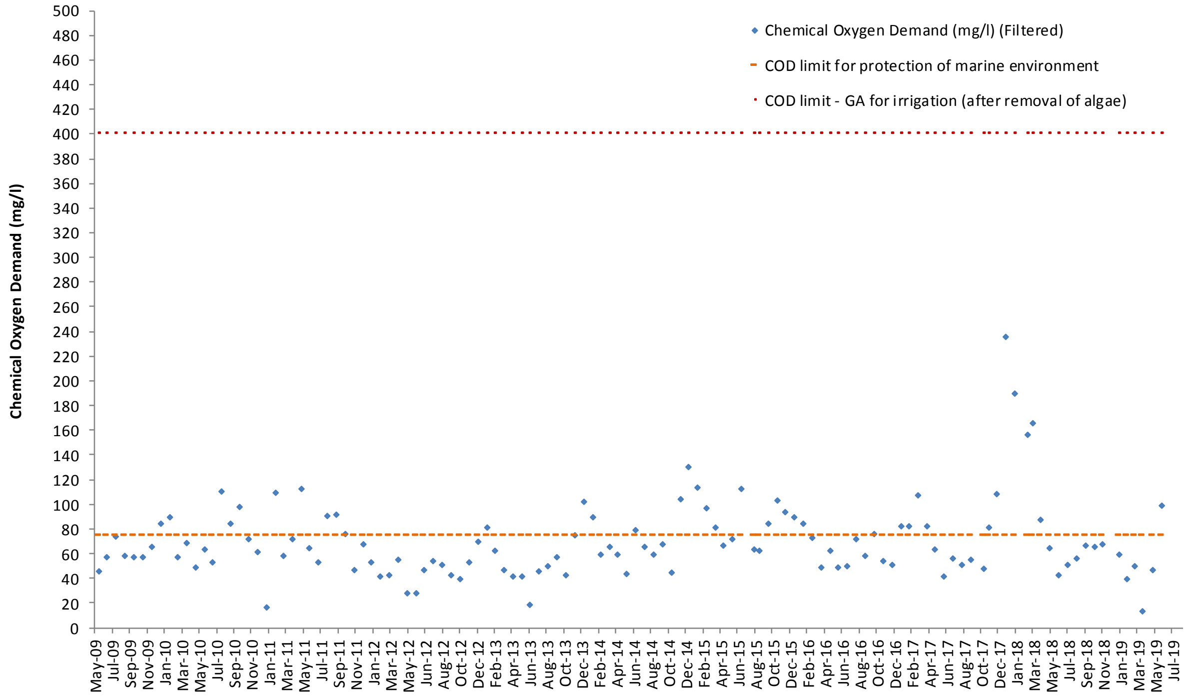


Figure 3.33 **Monthly trends in chemical oxygen demand (mg/L filtered) in effluent released from the Langebaan Wastewater Treatment Works, June 2009 - June 2019. Allowable limits as specified in terms of a General Authorisation under the National Water Act (No. 36 of 1998) are represented by the dashed red line. The recommended limit to protect marine inshore environments is shown by the orange dashed line (AEC 2015) (Source: Saldanha Bay Municipality).**

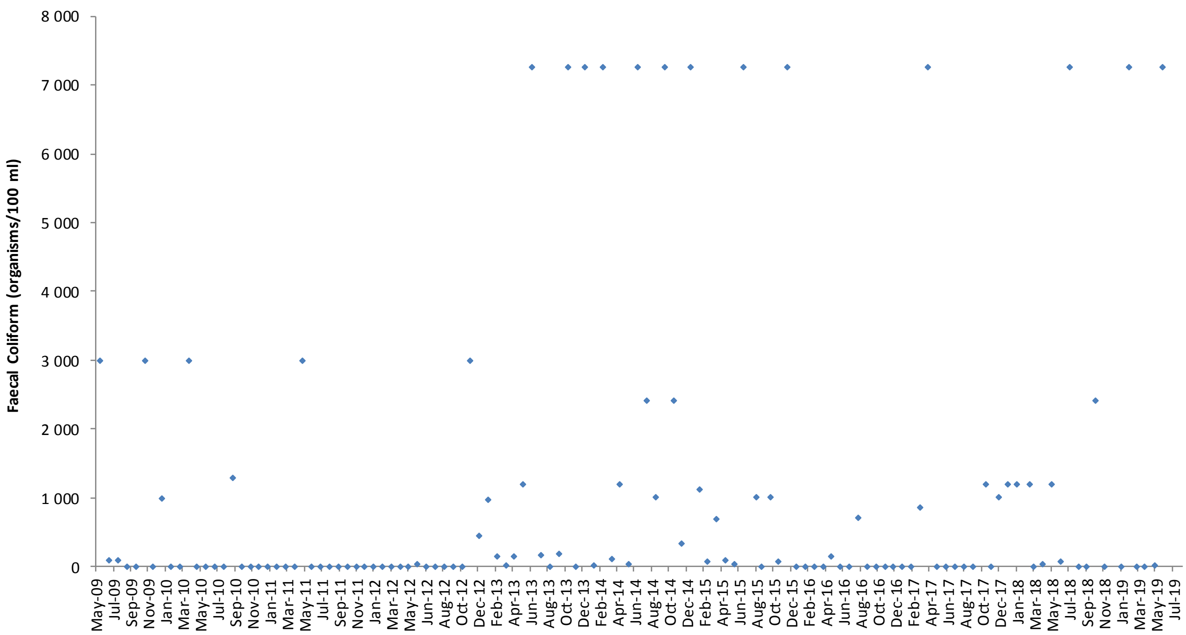


Figure 3.34 **Monthly trends in Faecal Coliforms (org/100ml) in effluent released from the Langebaan Wastewater Treatment Works, June 2009 - June 2019. The allowable limit in terms of a General Authorisation for irrigation purposes under the National Water Act (No. 36 of 1998) is 100 000 organisms per 100 mL. (Source: Saldanha Bay Municipality).**

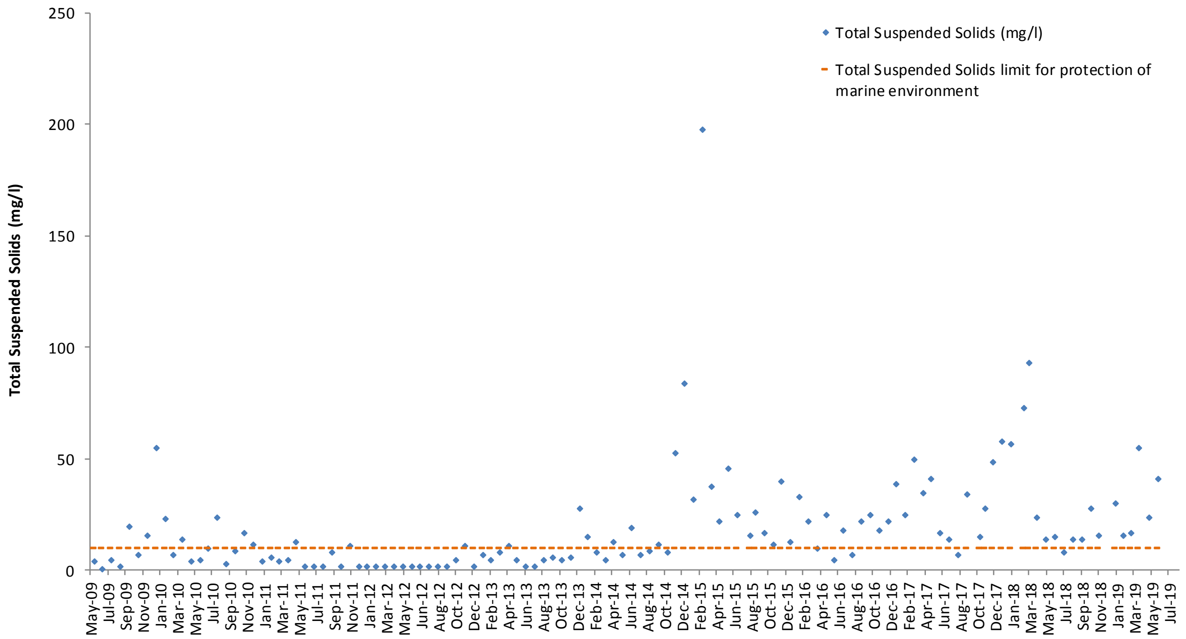


Figure 3.35 **Monthly trends in total suspended solids (mg/L) in effluent released from the Langebaan Wastewater Treatment Works, June 2009 - June 2019. The recommended limit to protect marine inshore environments is shown by the orange dashed line (AEC 2015) (Source: Saldanha Bay Municipality).**

No ammonia nitrogen limit is prescribed by the GA applicable to irrigation of wastewater. 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 between steeply between November 2012 and March 2018 from <10 mg/L to nearly 100 mg/L. Since then ammonia nitrogen concentrations have dropped significantly but are, however still grossly exceeding the recommended limit for the protection of the marine environment (3 mg/L). 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.

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. 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 53 occasions since June 2009 (44% of the time) (Figure 3.37). Although lower concentrations were recorded between April 2016 and March 2018, the concentration has increased since then. Toxic ammonia nitrogen is converted to non-toxic nitrate nitrogen by means of bacterial treatment in WWTWs. The recently observed higher levels are congruent with the lower ammonia levels in the effluent. This means that the bacterial treatment is currently more effective than during the previous 12-month cycle.

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 concentrations fluctuate in a seasonal pattern similar to that seen at the Saldanha WWTW (Figure 3.38). Orthophosphate levels have steadily increased since 2013, reaching the highest value recorded to date at 19 mg/L in May 2018. Overall, the orthophosphate concentration in the Langebaan WWTW effluent is considerably higher than 1 mg/L (92% exceedance). However, as observed with several other effluent parameters, orthophosphate levels improved significantly since November 2018, with an average of 4±5.1 mg/L (40% of readings <1 mg/L).

No free active chlorine limit is prescribed by the GA applicable to irrigation of wastewater. Free active chlorine is very 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 been fluctuating around 1.3 ±1.2 mg/L since October 2016 with no clear improvement or deterioration of the effluent quality (Figure 3.39). Readings have been consistently high in the last 12 months with an average of 1.1±1.1 mg/L. These levels are significantly higher than what would be considered acceptable discharge into the nearshore environment and more careful dosing of chlorine should be implemented.

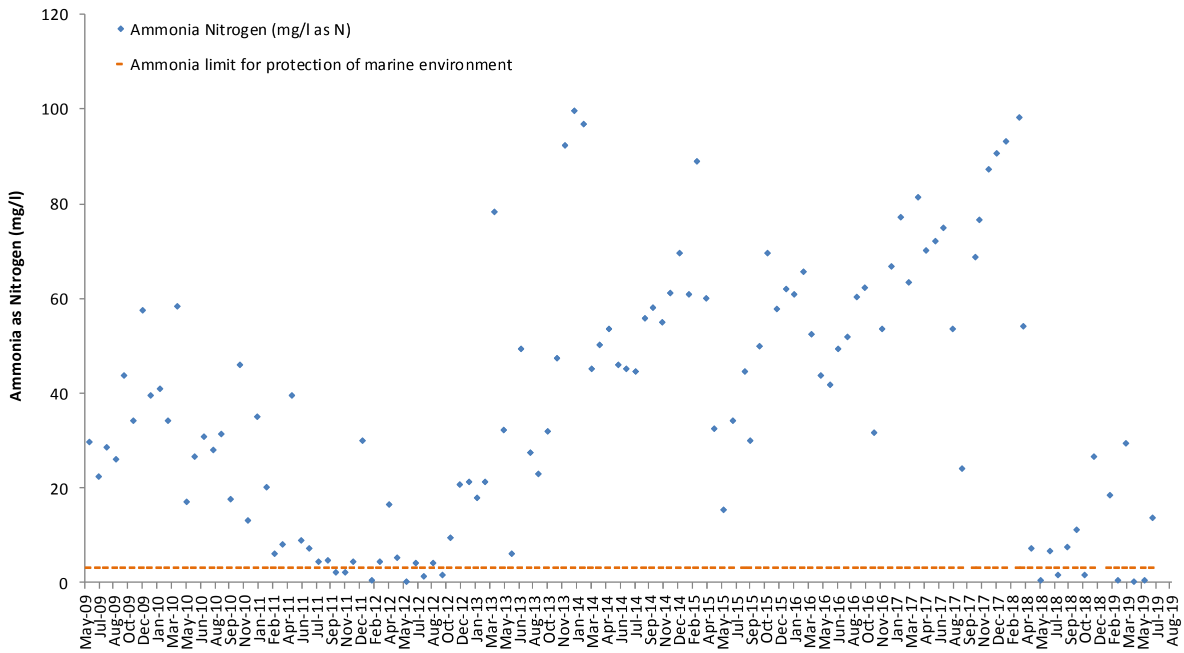


Figure 3.36. Monthly trends in Ammonia Nitrogen (mg/L as N) in effluent released from the Langebaan Wastewater Treatment Works June 2009 - June 2019. The recommended limit to protect marine inshore environments is shown by the orange dashed line (AEC 2015) (Source: Saldanha Bay Municipality).

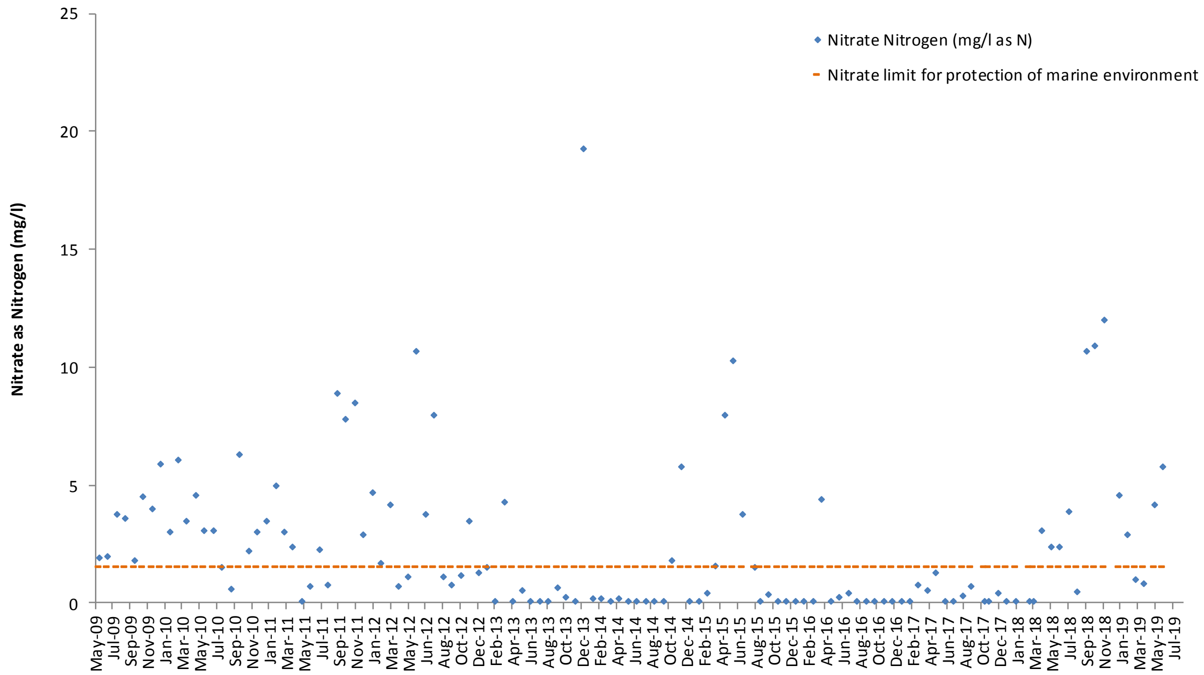


Figure 3.37 **Monthly trends in Nitrate Nitrogen (mg/L as N) in effluent released from the Langebaan Wastewater Treatment Works June 2009 - June 2019. The recommended limit to protect marine inshore environments is shown by the orange dashed line (AEC 2015) (Source: Saldanha Bay Municipality).**

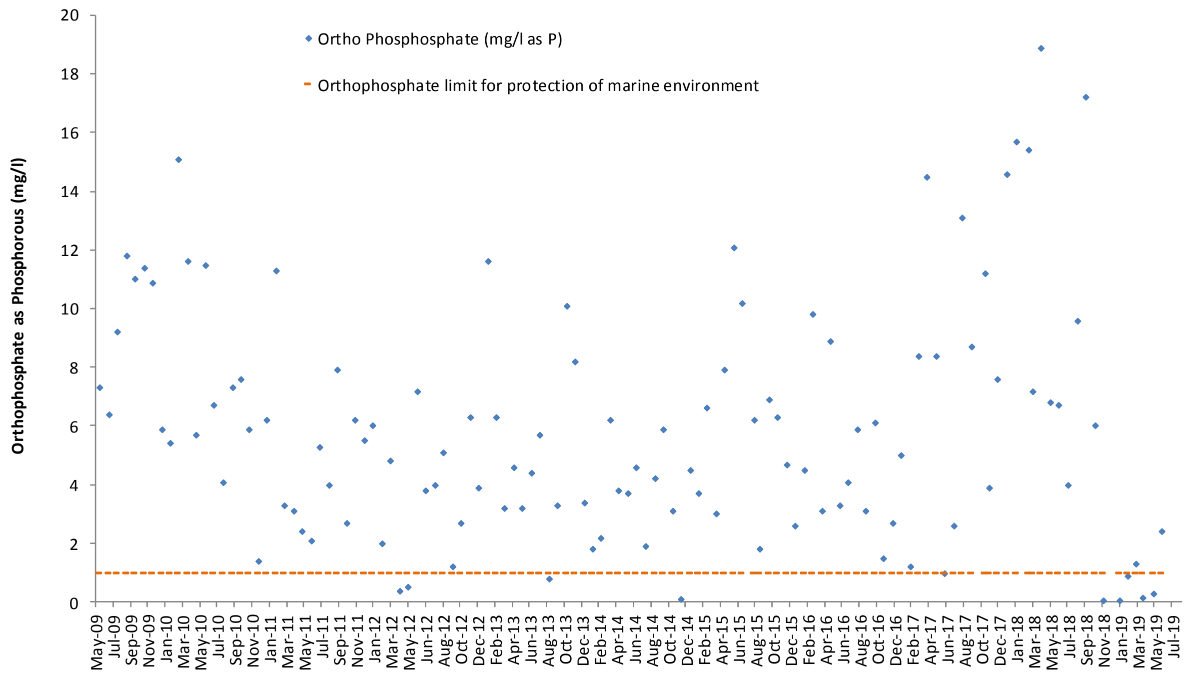


Figure 3.38 **Monthly trends in Orthophosphate (mg/L as P) in effluent released from the Langebaan Wastewater Treatment Works June 2009 - June 2019. The recommended limit to protect marine inshore environments is shown by the orange dashed line (AEC 2015) (Source: Saldanha Bay Municipality).**

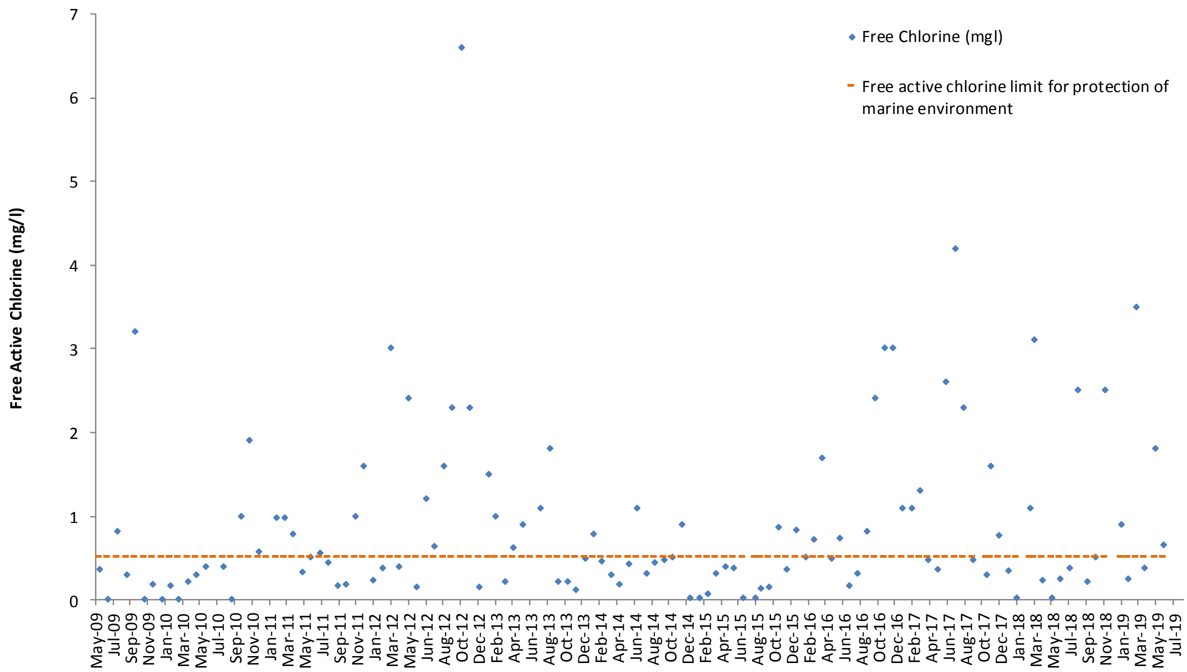


Figure 3.39 **Monthly trends in Free Active Chlorine (mg/L) in effluent released from the Langebaan Wastewater Treatment Works June 2009 - June 2019. The recommended limit to protect marine inshore environments is shown by the orange dashed line (AEC 2015) (Source: Saldanha Bay Municipality).**

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 Saldanha Bay WWTW is still experiencing difficulties in keeping water quality parameters within allowable limits and conditions as set out in the NWA (Government Gazette No. 36820, 6 September 2013). Most parameters have either worsened (Faecal coliform, ammonia nitrogen) or remained unchanged above legal limits (TSS, COD, orthophosphate) when compared to 2018/19. Chlorine levels are comparatively lower, which reduces toxicity of the effluent to the receiving environment. Toxic ammonia nitrogen is converted to non-toxic nitrate nitrogen by means of bacterial treatment in WWTWs. The recently observed lower levels of nitrate nitrogen are congruent with the higher ammonia levels in the effluent. This means that the bacterial treatment is currently not effective.

Improved effluent quality was recorded at the Langebaan WWTW for some parameters. Especially commendable are the significantly lower ammonia nitrogen and orthophosphate concentrations as well as the reduced chemical oxygen demand. Conductivity has also been consistently decreasing and has been compliant with the General Authorisation for irrigation since November 2018. Faecal coliform, TSS, and chlorine levels have remained unchanged and are currently not meeting legal requirements. The recently observed higher levels of nitrate nitrogen are congruent with the lower ammonia levels in the effluent. This means that the bacterial treatment is currently effective.

The data shows that the Saldanha WWTW is receiving greater volumes of effluent for treatment than permitted. 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. Neither capacity nor legal limit has been exceeded since. Furthermore, with the implementation of water restrictions, wastewater volumes treated by both plants have decreased to volumes that were recorded approximately 10 years ago.

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**

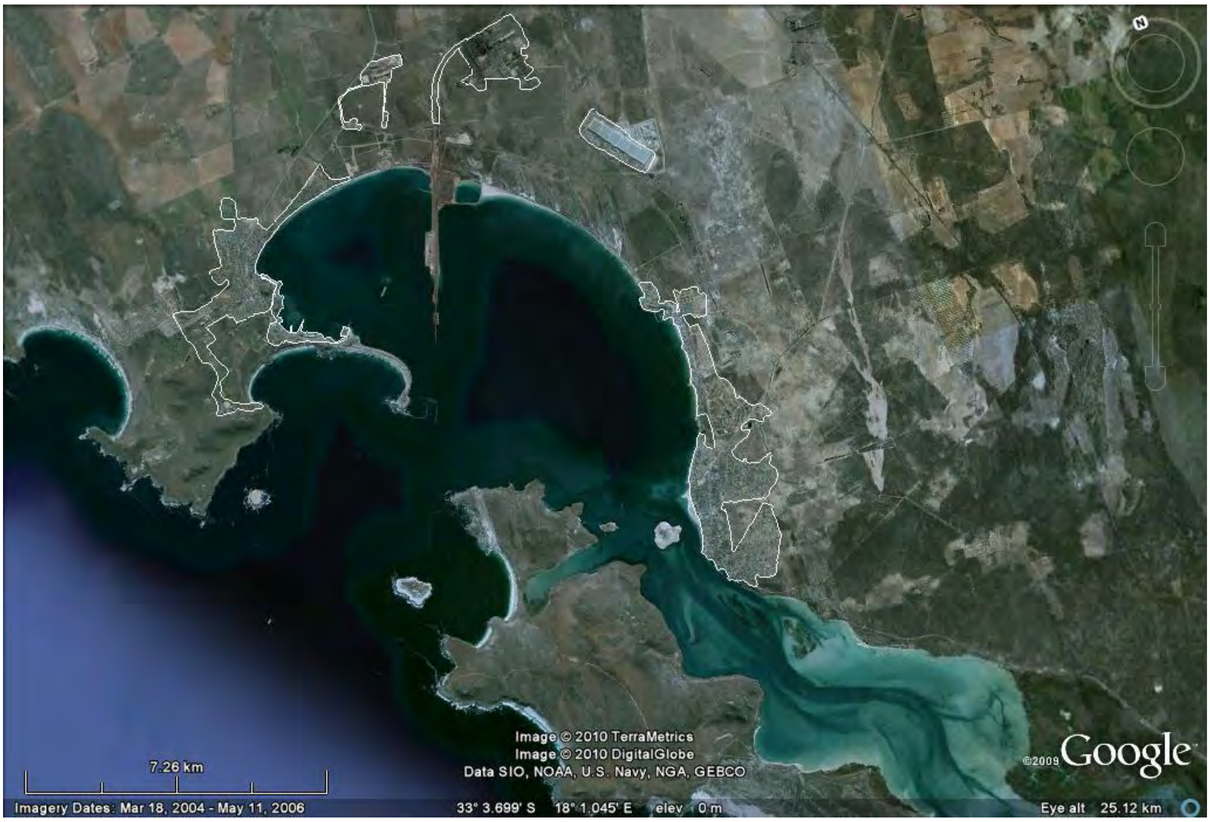


Figure 3.40. Spatial extent of residential and industrial areas surrounding Saldanha Bay and Langebaan Lagoon from which storm water runoff is likely to enter the sea (areas outlined in white). Note that runoff from the Port of Saldanha and ore terminal have been excluded as this is now reportedly all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

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.

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

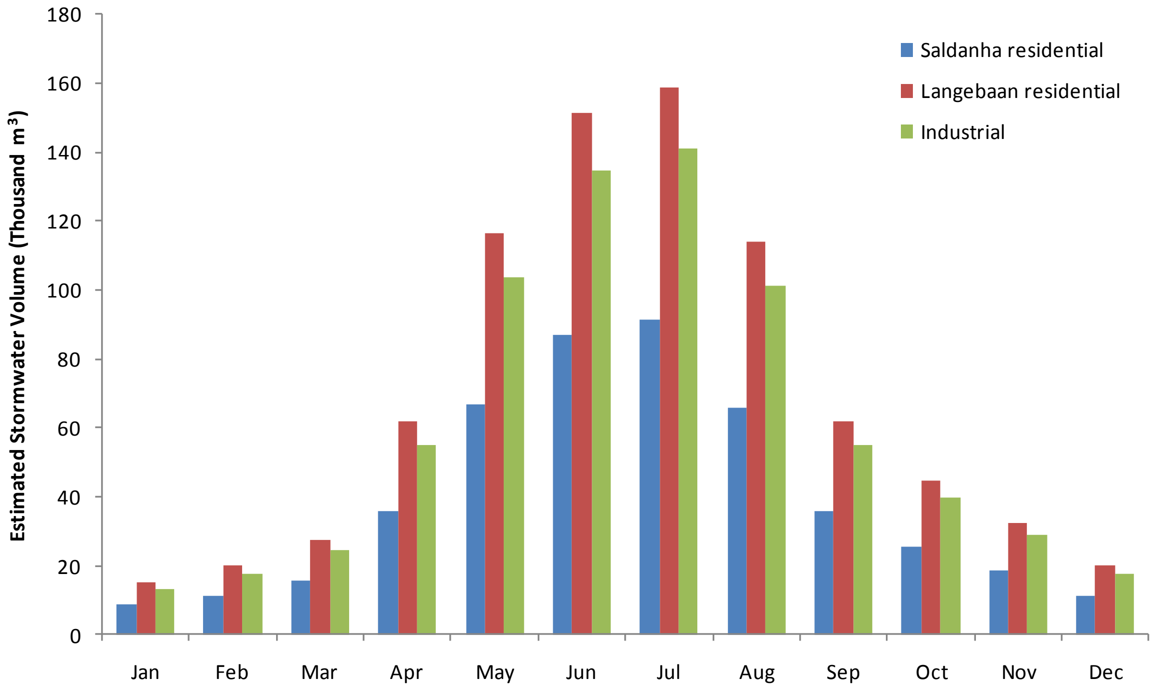


Figure 3.41. **Monthly estimated storm water volume (m^3) for Saldanha and Langebaan residential areas and industrial area. Note that runoff from the Port of Saldanha and ore terminal have been excluded as this is now reportedly all diverted to storm water evaporation ponds.**

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 has commissioned NSOVO Environmental Consulting to conduct the Basic Assessment Process (NSOVO Environmental Consulting 2017).

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.

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 3.43. Premier Fishing is currently in the process of re-commissioning and upgrading their fish processing plant.

SA Lobster Exporters 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.

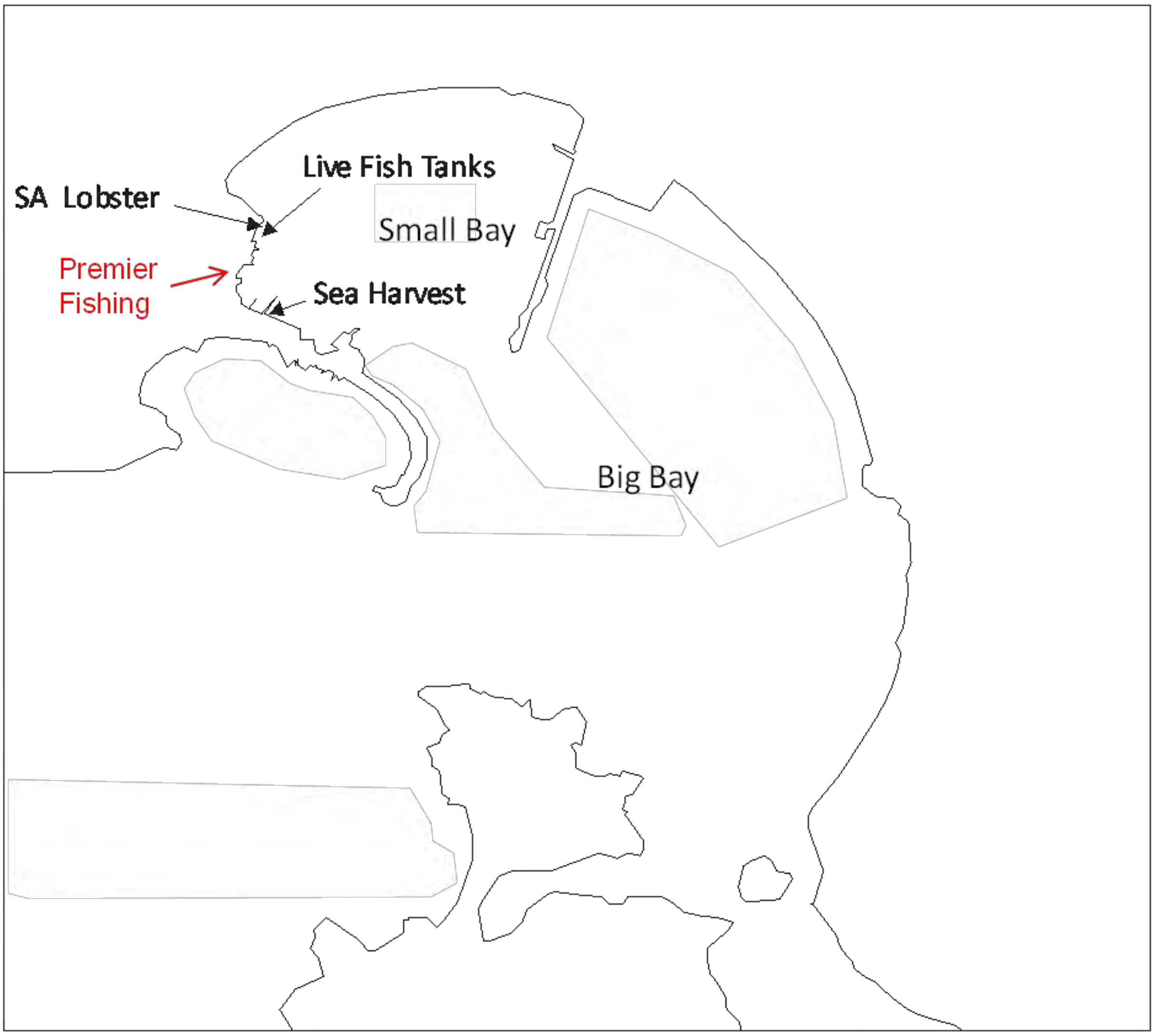


Figure 3.43. Location of seawater intakes and discharges for current and proposed seafood processing factories in Saldanha Bay. Current factories are indicated in black while the proposed Premier Fishing Fish Processing Plant is indicated in red.

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 (FFP) plant 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 suspended solids, fat, oil and grease, ammonia nitrogen, protein, and phosphate. 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).

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 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.

Coastal Waters Discharge Permit

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.

The current CWDP authorises the disposal of industrial effluent into the Saldanha Bay harbour through an existing marine outfall. This CWDP authorises 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. Sea Harvest is currently awaiting a decision on the applications for amendment submitted to the DEA on 9 July and 27 August 2018. These amendments included operational changes of the RO plant, which is 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. 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 3.44).

Figure 3.44. 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. The draft permit requires that a new monitoring plan is to be developed and implemented for the new outfall location. Note that the conditions of the amended authorisation will be included in the next edition of this monitoring report once finalised. 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.

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 has been malfunctioning relatively frequently and even fewer measurements have therefore been taken in recent years (Table 3.8). Sea Harvest had 2066 operational days since 15 July 2013 and effluent readings were only taken 42% of the time.

In the last year (July 2018-June 2019) effluent meter readings were only recorded 48% of the time due to upgrades to the plant and occasionally faulty meter (Table 3.8). Higher compliance would be desirable as on more than 50% of the days, effluent volume discharge remains unmonitored. Effluent volume readings indicate that Sea Harvest discharged more than 1152 m^3 per day 70% of the time between July 2018 and June 2019 (Table 3.8)[6]. This was anticipated due to the inability of the RO plant to process fish processing plant effluent upon installation

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.

[6] Effluent volume is calculated by subtracting the previous day's 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.

Table 3.8 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
Number of operational days	1424	2066	659	321
Number of readings	704	859	316	153
Readings taken relative to number of operational days (%)	49%	42%	48%	48%
Number of days where effluent volume was calculated[A]	571	780	305	146
Effluent volume calculated relative to number of operational days (%)	40%	38%	46%	45%
Legal daily effluent volume limit (m^3)	2000	3546	1152	1152
Exceedance of legal effluent volume limit (count)	225	137	134	102
Exceedance of legal effluent volume limit relative to number of operational days (%)	39%	18%	44%	70%

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.

Average daily effluent discharge volume was 3 285 m^3 in 2003/4, increased to 7 312 m^3 in 2006/7 and dropped to 530 m^3 in 2016/17, remaining approximately the same for 2017/2018 (603 m^3). Due to the additional effluent produced by the RO plant, average daily discharge volume tripled in the last year to above the legal limit (1693 m^3).

Estimated annual fish processing effluent volumes[7] discharged into Small Bay between July 2003 and June 2018 by Sea Harvest is shown in Figure 3.45 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 peaked at unusually high levels. It is not clear why this increase occurred, but data reporting and environmental monitoring at Sea Harvest have suffered irregularities 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. The 2018/2019 data shows that Sea Harvest is currently able to meet the new annual limit of 420 480 m^3 as specified in the CWDP conditions, despite exceedance of the daily limit of 1152 70% of the time.

[7] Average daily effluent volume was calculated by dividing the measured annual volume by the number of measurements taken.

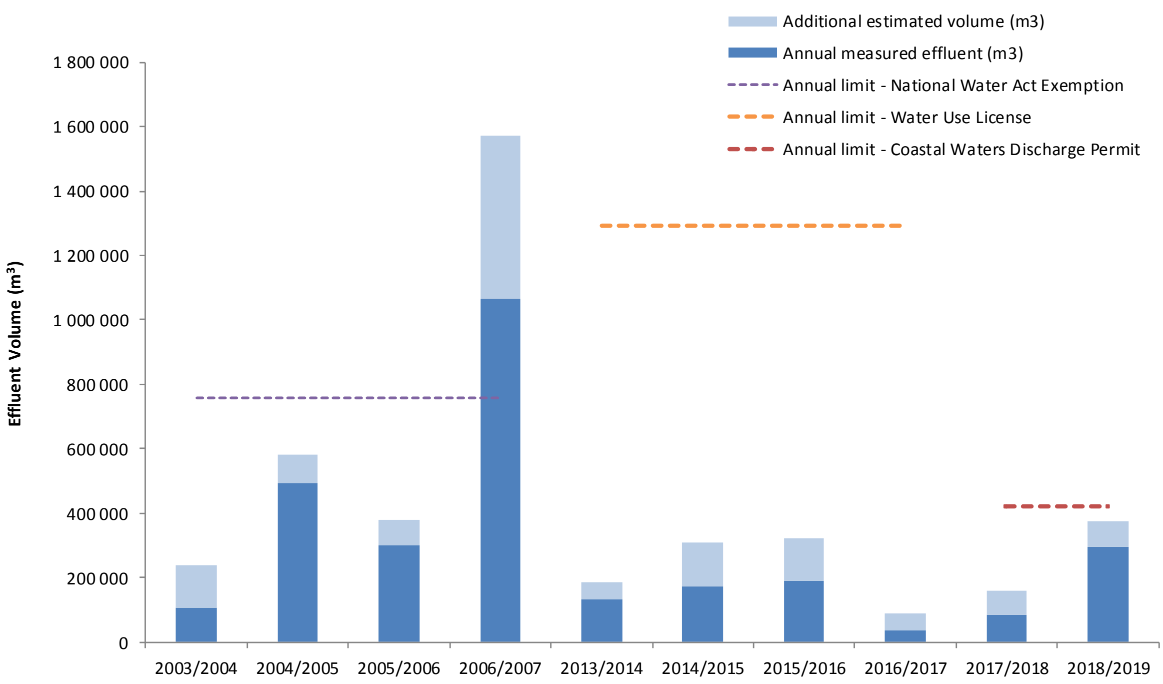


Figure 3.45 **Estimated Fresh fish processing effluent volume discharged into Small Bay per year by Sea Harvest from July 2004 - June 2019. Data was not available for the period May 2007 – August 2013. The legal annual effluent limits are indicated as dashed lines. (Source: Frank Hickley, Risk Control Manager at Sea Harvest fish Processing Plant).**

Until this CWDP was issued 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.

TSS concentrations have been extremely high and compliance with the revised General Discharge Limit of 25 mg/L was only achieved in October 2013 (14 mg/L) (Figure 3.46). Trends in TSS since 2010 suggest that concentrations fluctuate over time and it appears that peak concentrations are decreasing in magnitude (Figure 3.46). The CWDP issued on 26 June 2017 specifies a legal limit of 230 mg/L. Since July 2017, TSS concentration in the effluent exceeded the legal limit seven times, which means that Sea Harvest is compliant 73% of the time. TSS levels have decreased substantially since July 2018 and the legal limit was only exceeded once in April 2019 with 241 mg/L.

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. 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 issued on 26 June 2017 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). 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 salinity (ppt) is therefore lower than what is expected in the receiving environment (Figure 3.48). 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 receiving environment. This is likely due to the increasing use of seawater for fish processing over time. Since the implementation of the RO plant in 2018, salinity exceeds the limit specified in the CWDP (37 ppt) fluctuating around 38 ppt (±14 ppt). Maximum salinity was measured on 26 September 2019 at 75 ppt.

Sea Harvest has been measuring Chemical Oxygen Demand (COD) since November 2015. COD has consistently been extremely high, measuring on average 545±289 mg/L since July 2018, which is, however, an improvement from last year where the average was recorded as 1 121 ±618 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 when compared to last year, Sea Harvest has not been able to meet the requirements of the new CWDP (<200 mg/L) under current effluent treatment methods (Figure 3.49). 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.

Oil and grease were monitored monthly between March and December 2015 (Figure 3.50). Values always exceeded the General Authorisation limit of 2.5 mg/L, with a very high average of 27±25 mg/L, 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. Sea Harvest was compliant with the legal limit only 36% of the time July 2017. 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. COD limits were only met on four occasions during the 2018/19 monitoring period.

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 2017-19 demonstrate that the effluent has been compliant with the legal limit with the exception of one occasion when pH measured 4.8 in July 2017.

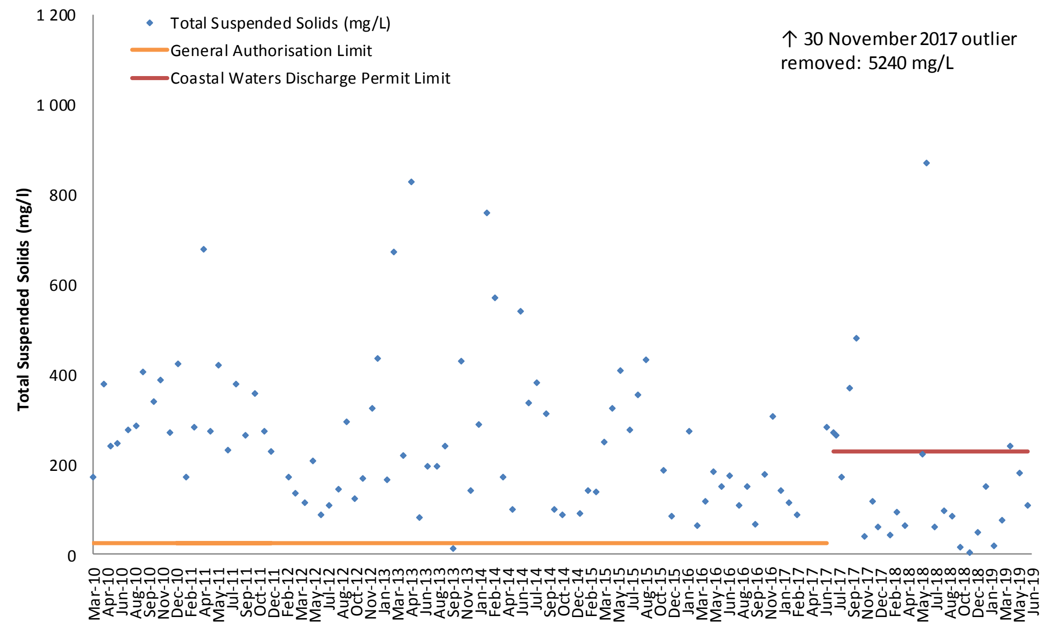


Figure 3.46 **Monthly trends in total suspended solids (TSS) (mg/L) in the effluent discharged from the Sea Harvest fresh fish processing (FFP) plant into Small Bay in the period March 2010 to June 2019 (concentration measured at the end of pipe). No data is available between April and June 2017. The orange line indicates the limit prescribed by the General Discharge Limit of the revised General and Special Standard (25 mg/L) (Government Notice No.36820 –6 September 2013). Sea Harvest was granted a Coastal Waters Discharge Permit on 26 June 2017, which prescribes a limit of 230 mg/L (depicted as the red line). (Source: Frank Hickley, Risk Control Manager at Sea Harvest fish Processing Plant).**

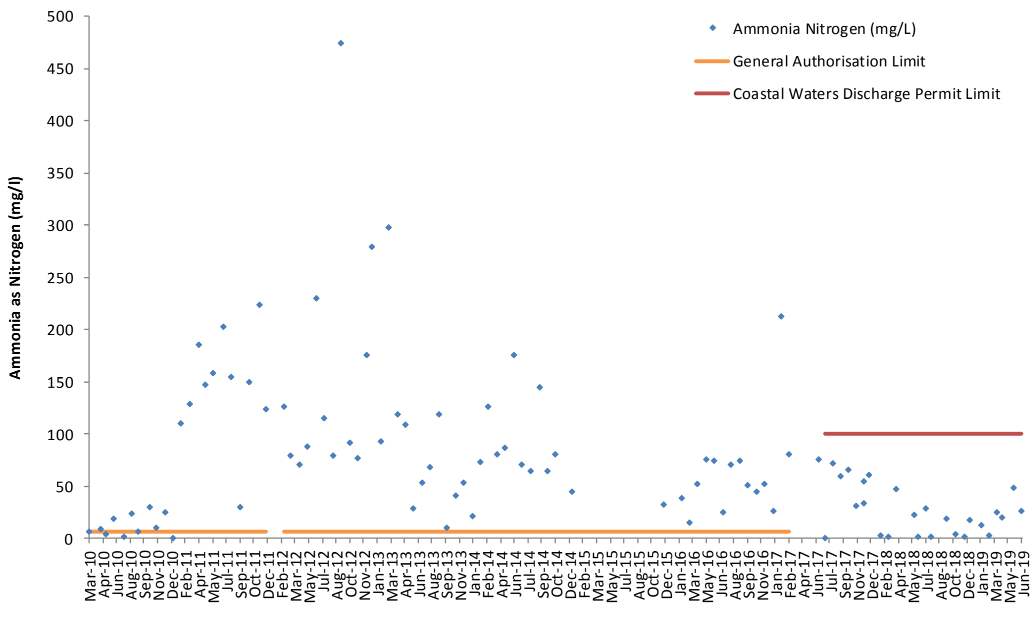


Figure 3.47 **Monthly trends in ammonia nitrogen (mg/L) in the effluent discharged from the Sea Harvest fresh fish processing (FFP) plant into Small Bay in the period March 2010 to June 2019 (concentration measured at the end of pipe). No data is available between April and June 2017. The orange line indicates the limit prescribed by the General Discharge Limit of the revised General and Special Standard (6 mg/L) (Government Notice No.36820 –6 September 2013). Sea Harvest was granted a Coastal Waters Discharge Permit on 26 June 2017, which prescribes a limit of 100 mg/L (depicted as the red line). (Source: Frank Hickley, Risk Control Manager at Sea Harvest fish Processing Plant).**

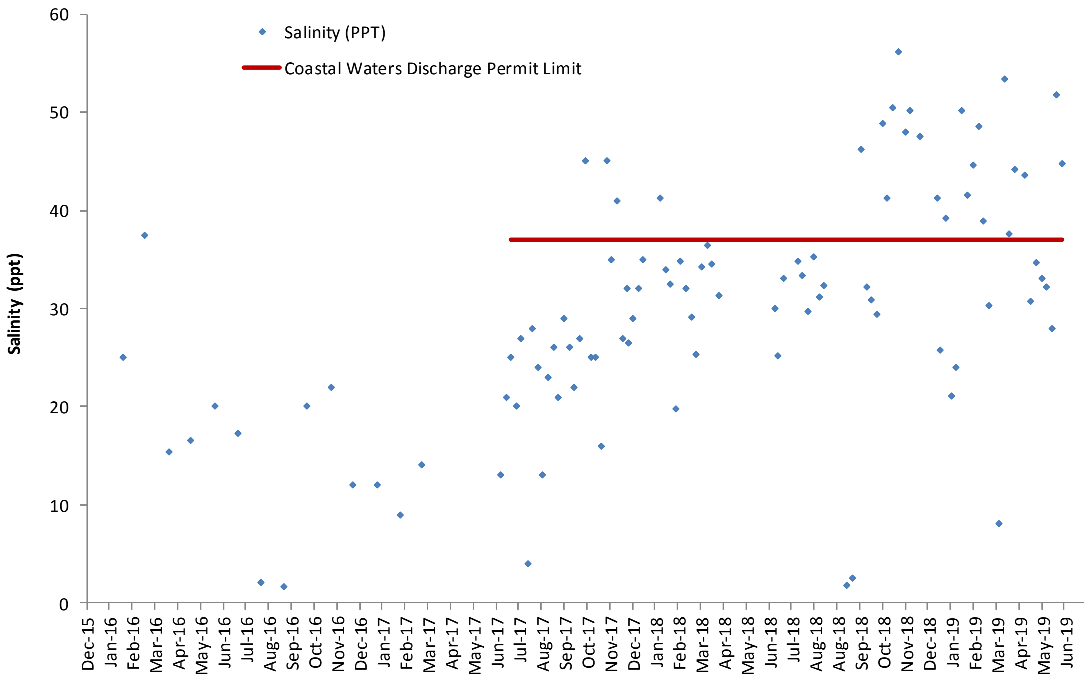


Figure 3.48 **Monthly salinity (ppt) trends in the effluent discharged from the Sea Harvest fresh fish processing (FFP) plant into Small Bay in the period January 2015 to June 2019 (concentration measured at the end of pipe). No data is available between April and June 2017. Sea Harvest was granted a Coastal Waters Discharge Permit on 26 June 2017, which prescribes a limit of 37 ppt (depicted as the red line). (Source: Frank Hickley, Risk Control Manager at Sea Harvest fish Processing Plant).**

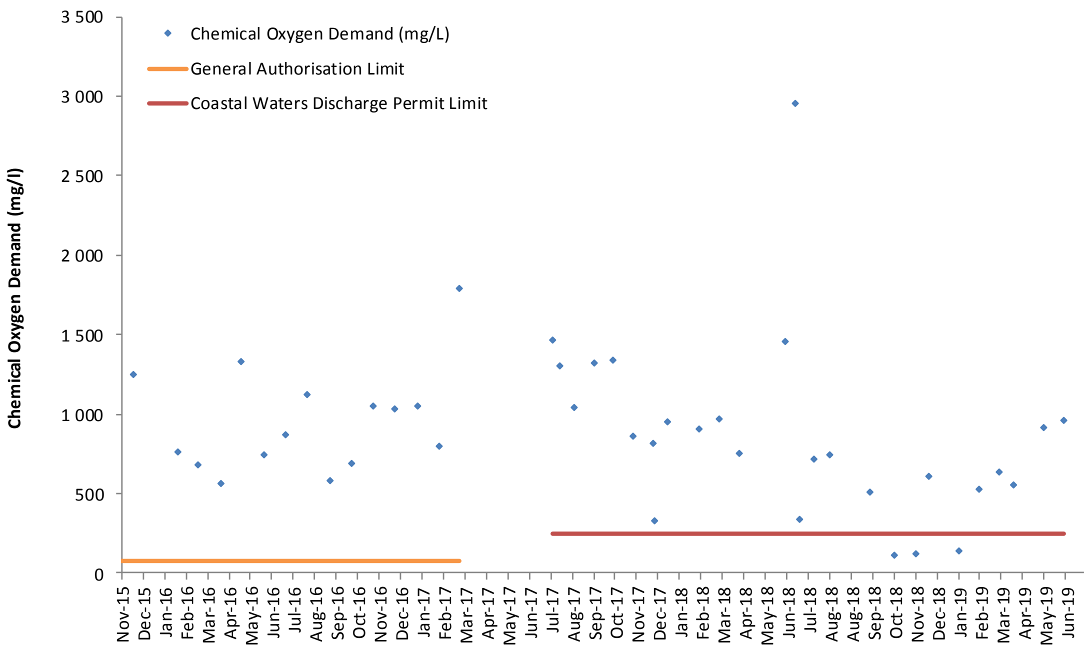


Figure 3.49 **Monthly chemical oxygen demand (COD) trends in the effluent discharged from the Sea Harvest fresh fish processing (FFP) plant into Small Bay in the period November 2015 to June 2019 (concentration measured at the end of pipe). The orange line indicates the limit prescribed by the General Discharge Limit of the revised General and Special Standard (75 mg/L) (Government Notice No.36820 –6 September 2013). Note that Sea Harvest was granted a Coastal Waters Discharge Permit on 26 June 2017, which prescribes a limit of 250 mg/L (depicted as red line). No data is available from April 2017 onward. (Source: Frank Hickley, Risk Control Manager at Sea Harvest fish Processing Plant).**

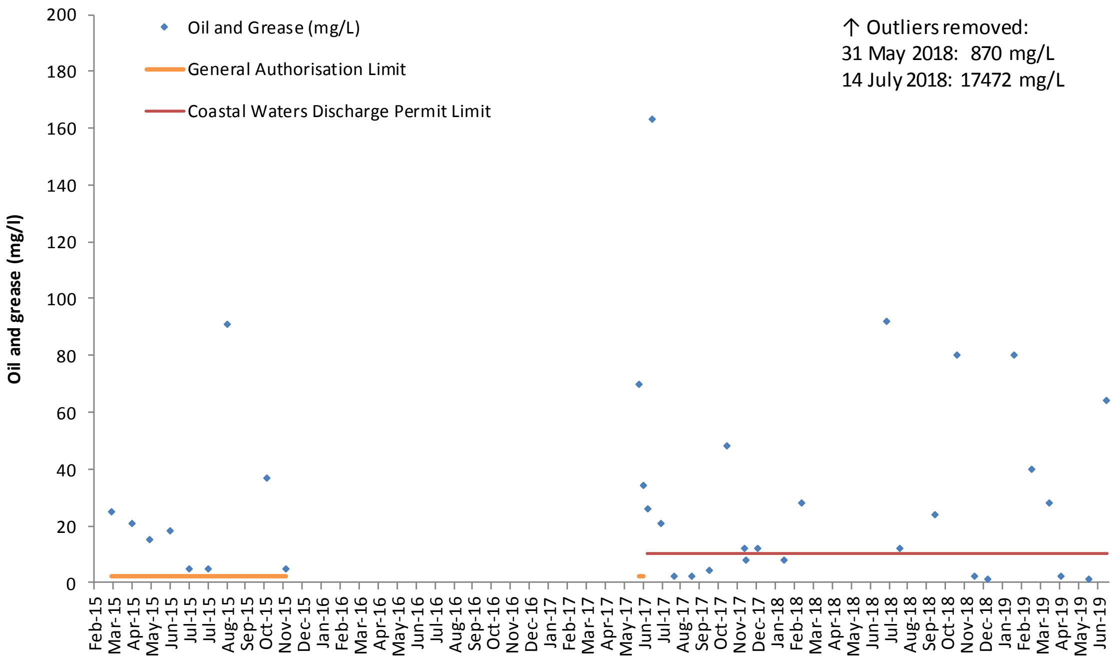


Figure 3.50 Monthly trends of oil and grease (mg/L) in the effluent discharged from the Sea Harvest fresh fish processing (FFP) plant into Small Bay in the period March to December 2015 and from June 2017 to June 2019 (concentration measured at the end of pipe). The orange line indicates the limit prescribed by the General Discharge Limit of the revised General and Special Standard (2.5 mg/L) (Government Notice No.36820 –6 September 2013). Sea Harvest was granted a Coastal Waters Discharge Permit on 26 June 2017, which prescribes a limit of 10 mg/L (depicted as red line) (Source: Frank Hickley, Risk Control Manager at Sea Harvest fish Processing Plant).

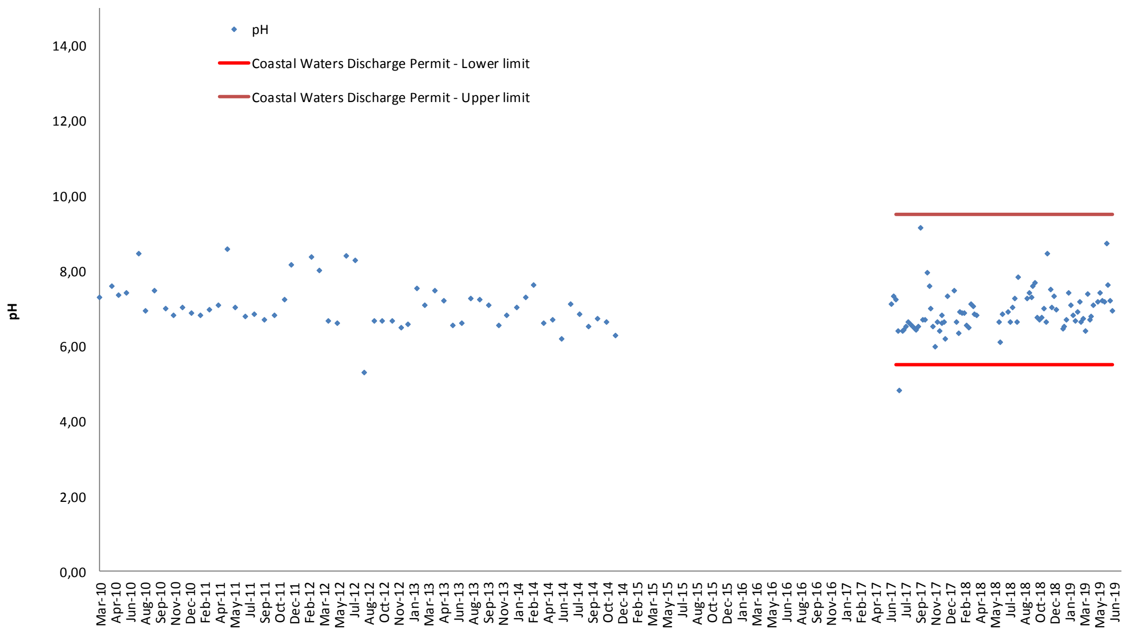


Figure 3.51 Monthly trends of the pH measured in the effluent discharged from the Sea Harvest fresh fish processing (FFP) plant into Small Bay in the period March 2010 to December 2014 and from June 2017 to June 2019. The red dashed lines indicate the lower (5.5) and upper (9.5) limits prescribed by the Coastal Waters Discharge Permit dated 26 June 2017. (Source: Frank Hickley, Risk Control Manager 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, which are on average two and three times higher than the prescribed limit. The effluent produced by the RO plant has increased the salinity of the overall effluent dramatically and CWDP requirements are currently exceeded 52% of the time. During the 2018/19 monitoring period, 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.

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 3.52). 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 insert reference of for more information). 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 insert reference of for more information on the impacts of fisheries on fish populations).

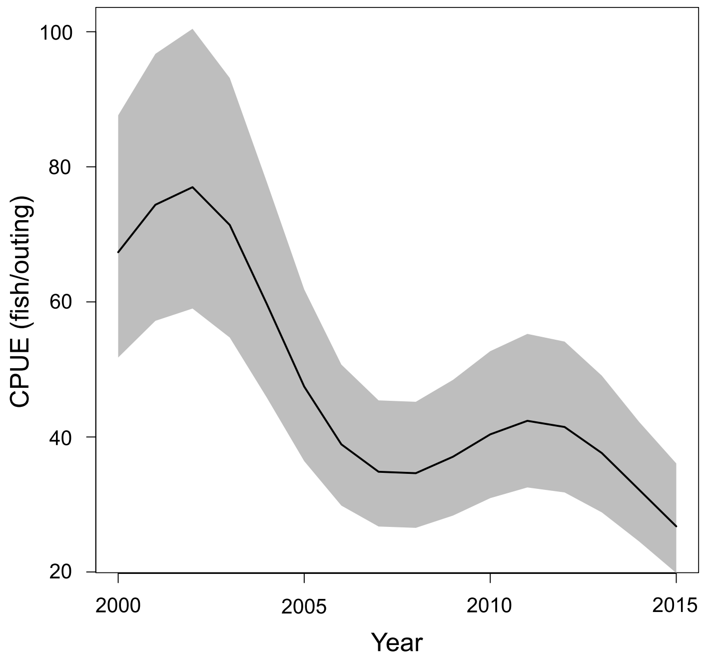


Figure 3.52 Annual Catch Per Unit Effort (CPUE) estimates (±95% Confidence Interval) of white stumpnose derived from commercial boat catches logged in the National Marine Linefish System (NMLS) database (Source: Parker et al. 2017).

3.8 Marine aquaculture

The Department of Agriculture, Forestry and Fisheries (DAFF) is currently driving accelerated development of the aquaculture sector in South Africa with the aim to create jobs for marginalised coastal communities and contribute towards food security and national income. The development of the aquacultures sector is considered a sustainable strategy to 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. A combined 430 ha of sea space are currently available for aquaculture production in Outer Bay (north and south), Big Bay and Small Bay (Figure 3.53), of which 316.5 ha have been leased to 14 individual mariculture operators (Table 3.9 and Figure 3.53.). Just over eighty percent of this available area farmers have been allocated aquaculture rights to farm mussels, oysters and finfish (Table 3.9). The DEA recently issued Environmental Authorisation to the DAFF for an Aquaculture Development Zone, which include four precincts (Small Bay, Big Bay North, Outer Bay North and South) totalling 420 ha of new aquaculture areas in Saldanha Bay. Currently farmed areas will be incorporated into the ADZ comprising 884 ha set aside for mariculture. More details on progress made in establishing the ADZ are summarised in Section 3.8.1.

Table 3.9. Details of marine aquaculture rights issued in Saldanha Bay (BB and SB refer to Big Bay and Small Bay respectively) (Sources: Aquaculture Rights Register Department of Agriculture Forestry and Fisheries November 2017, updated by individual farmers in 2019).

	Products								
Company	**Mussels**	**Oysters**	**Abalone**	**Scallops**	**Red Bait**	**Seaweed**	**Finfish**	**Area (Location*)**	**Duration of right**
Blue Ocean Mussel (previously trading as Blue Bay Aquafarm (Pty) Ltd.	x	x						52.1 ha (SB)	2017-2032
Blue Sapphire Pearls CC	x	x	x			x		10 ha (BB)	2010-2024
Imbaza Mussels (Pty) Ltd (previously trading as Masiza Mussel Farm (Pty) Ltd)	x	x		x				30 ha (SB)	2010-2024
Saldanha Bay Oyster Company (previously trading as Striker Fishing CC and West Coast Seaweeds (Pty) Ltd)		x		x				25 (BB) 10 ha (SB)	2010-2024
West Coast Aquaculture (Pty) Ltd	x	x			x			5 ha (SB) 10 ha (BB)	2010-2024
West Coast Oyster Growers CC	x	x						15 ha (BB) 15 ha (SB)	2010-2024
African Olive Trading 232 (Pty) Ltd	x							30 ha (SB)	2013-2028
Aqua Foods SA (Pty) Ltd	x	x						10 ha (BB) 10 ha (SB)	2014-2030

	Products								
Company	**Mussels**	**Oysters**	**Abalone**	**Scallops**	**Red Bait**	**Seaweed**	**Finfish**	**Area (Location*)**	**Duration of right**
Southern Atlantic Sea Farms (Pty) Ltd.	x						x	15 ha (Outer Bay - North)	2014-2029
Salmar Trading (Pty) Ltd.		x						5 ha (SB)	2016-2031
Molapong Aquaculture (Pty) Ltd.							x	1 ha (Outer Bay - south) 4.1 ha (BB)	2016-2032
Chapman's Aquaculture (Pty) Ltd	X							Outer Bay North	2016-2031
Requa Enterprises	X							15 ha (BB)	2016-2031

Figure 3.53. Mariculture concession areas in Saldanha Bay 2017 (430 ha). The total area leased to the aquaculture sector currently comprises 316.5 ha. Note that Transnet is not at liberty to disclose the names of their tenants to third parties. (Source: Transnet Property, Geo-Spatial: Western Region, Burton Siljeur).

3.8.1 Saldanha Bay Aquaculture Development Zone

With the support of finances and capacity allocated to the Operation Phakisa Delivery Unit, DAFF has been given Environmental Authorisation to establish a sea-based Aquaculture Development Zone (ADZ) in Saldanha Bay. The aim 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. Refer to AEC 2017 for a detailed description and potential impacts of the proposed ADZ.

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 (Department of Environmental Affairs) granted three separate Environmental Authorisations (EAs) for aquaculture in the Bay to the Department of Agriculture, Forestry and Fisheries (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 DEADP was issued on 7 June 2018 and stated that appeals were overturned by the Minister and that the EA was upheld. DAFF appointed an Environmental Control Officer and set up a Consultative Forum, which has 114 members thus far. The Aquaculture Management Committee (AMC) meets every two months to ensure that the implementation of the ADZ occurs in line with the requirements specified in the EA and EMPr. The DAFF recently published a "Guideline for Bivalve Production Estimates for the Saldanha Bay Aquaculture Development Zone". This document ensures that the production per annum as specified in the EA are upheld by the ADZ. Coupled with environmental monitoring, the adherence to the authorised tonnages should facilitate adaptive environmental management of the ADZ as a whole. The DAFF compiled the marine monitoring programme (Sampling Plan) and completed dispersion modelling, baseline sample collection, and completed a rapid synoptic survey of oxygen and nutrient levels in the Bay (see detail on modelling and monitoring in Section 3.8.1.1).

Various guidelines and protocols have been developed for managing the ADZ and DAFF has engaged with the Saldanha Bay Water Quality Forum Trust to combine sampling efforts. The mussel industry is in the process of applying for Marine Stewardship Council Certification with the assistance of the World Wildlife Fund, in an effort to evaluate the status of the fishery in relation to the MSC standard for sustainable fisheries.

3.8.1.1 Impacts modelling and monitoring

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 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, 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).

The 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 were developed for both dissolved nitrogen in the water column and sedimented particulate organic matter (PRDW 2017). A 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).

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

- dissolved nitrogen concentrations attributable to the fish farms will be low compared to the 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;
- based on an FCR of 1.4 (Food Conversion Ratio), 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 the monitoring programme *Protocols for environmental monitoring of the Aquaculture Development Zone in Saldanha Bay, South Africa*, 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 will be used to validate the results of the model (PRDW 2017), and to inform the proposed phased implementation and expansion of the ADZ.

The monitoring plan highlights potential impacts identified during the EIA process, including the modification of seabed by biodeposition, and of the water column dissolved oxygen and inorganic nitrogen; removal of seston[8] 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).

Proposed sampling sites are shown in Figure 3.54.

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

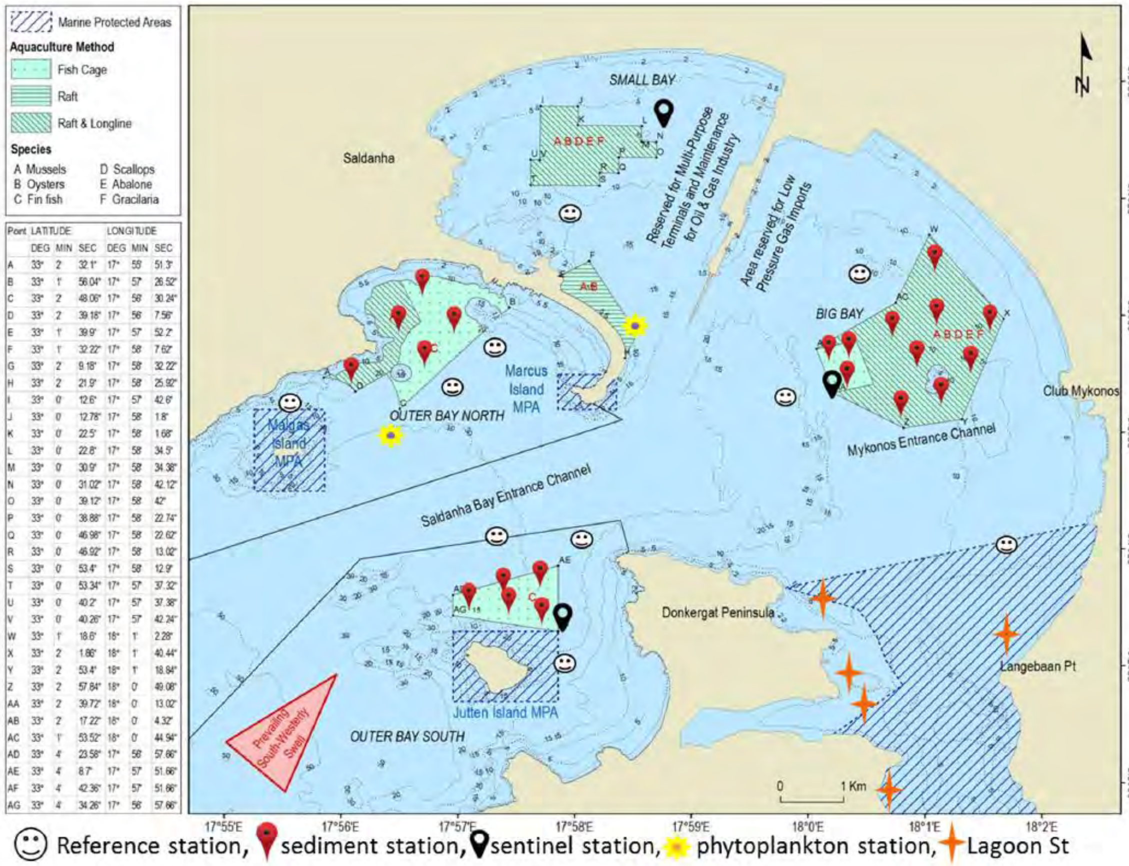


Figure 3.54 **Map of sampling station for the Saldanha Bay Aquaculture Development Zones from DAFF (2018). The monitoring program notes that these stations positions are to be finalised prior to sampling, and that the lagoon stations are from this State of the Bay monitoring programme.**

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". It is not clear whether sampling will be aligned with the State of the Bay Programme, and there is therefore some concern that this monitoring programme may not align with the long-term data collected by the SBWQFT. There is also no specification on when (time of year) the samples should be collected so this could further 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; and
- pollution by therapeutants.

It is possible that records of interactions between marine megafauna and farm infrastructure and incidences of disease on the farms that operators are required to collect may cover the first and second of these issues (although there are concerns relating to independence and objectivity of this data) there seems to be very little (if any) effort directed towards monitoring impacts of the last two aspects. The protocol does list what are referred to as “basic requirements for an effective biosecurity plan” (p33) but no specific actions are proposed. The same is true for impacts on the genetic integrity of naturally occurring biota in the Bay. 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.

A requirement for installation of sentinel and reference stations for monitoring of oxygen and temperature is also included under the “seabed” monitoring component but no details are provided as to when this should start (aside from a vague statement which says “should be operational prior to development, or as soon thereafter as possible”) and there is no indication as to how long this monitoring is to continue.

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 but again, it is not specifically stated when this will start or how long it will continue.

Specifications for operational monitoring (2) are less clear but seem to state that this will only be initiated within the respective lease areas following initiation of production. For the seabed monitoring component this is expected to include (a) collection and analysis of a set of three replicate samples at three stations (0 m, 30 m, and 60 m) in each lease area and (b) repeating the baseline survey every 3-5 years. There is concern that the interval at which the baseline 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 64-1139 in Small Bay and from 88-1403 in Big Bay (cross reference Macrofauna Chapter). As such, we recommend that the baseline 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, but it is not clear if this has (or will) be done.

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 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 3.55). No results are available as yet.

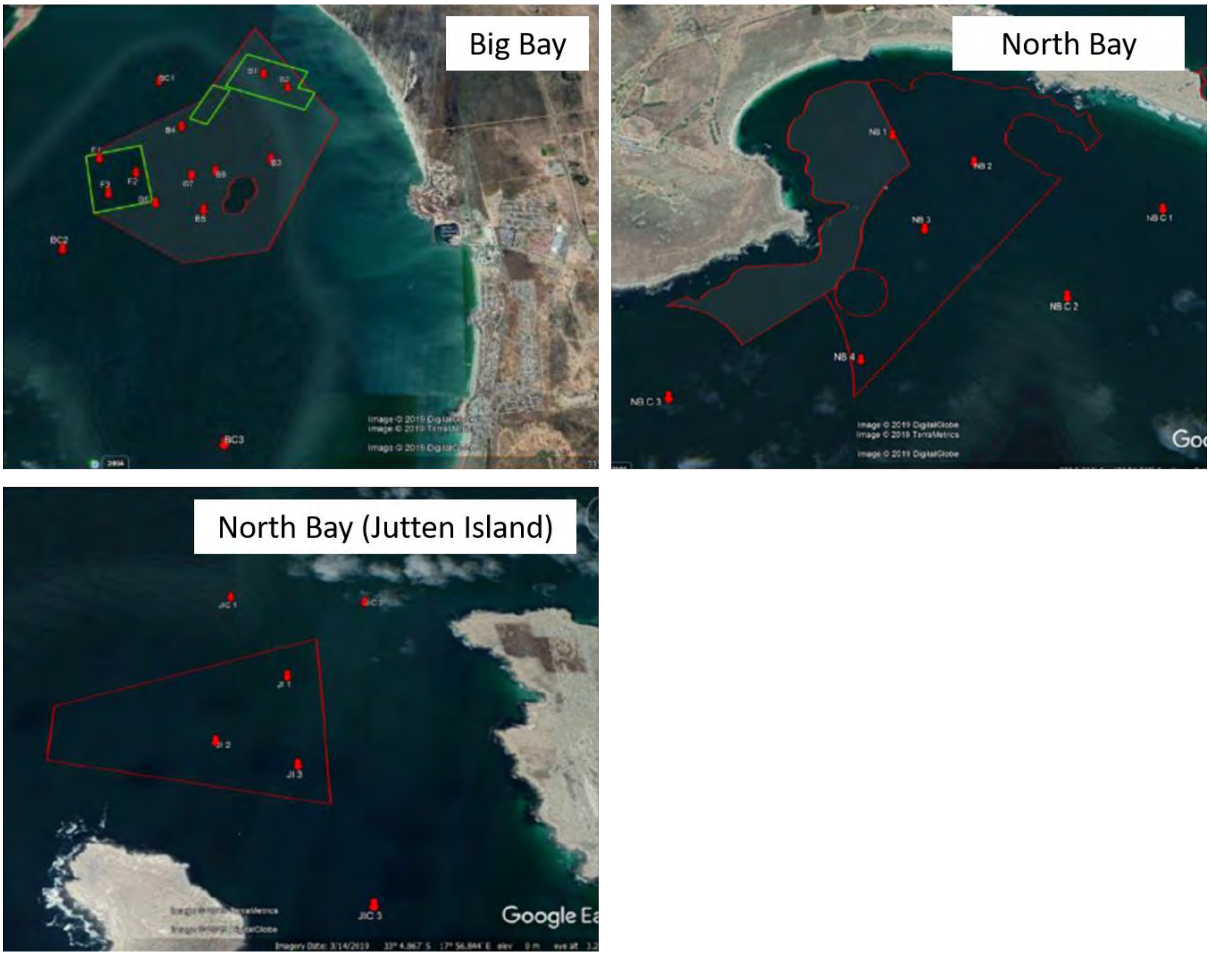


Figure 3.55 Sampling stations in Big Bay, North Bay and North Bay (Jutten Island), Saldanha Bay, for the baseline sample collection in the Aquaculture Development Zone (from Heinecken 2019).

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 (Table 3.9). 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). At the time of writing, most of the farming occurs in Small Bay and only oysters are cultured in Big Bay by the Saldanha Bay Oyster Company and West Coast Oyster Growers.

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 DAFF therefore included alien trout species in their application for EA. Consequently, the Environmental Authorisation issued to DAFF 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 of a boat. Overall mussel productivity has been increasing exponentially since 2007, peaking in 2018 at 2182 tons (Figure 3.56.). Mussel production has more than doubled 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 tons per annum since 2000. Oyster production reached a peak in 2016 at 357 tons per annum but has since decreased to 283 tons in 2018 (Figure 3.56.).

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 faeces, 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 Agriculture, Forestry and Fisheries (DAFF) will provide an indication of the environmental impact of oyster culture (DAFF 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.

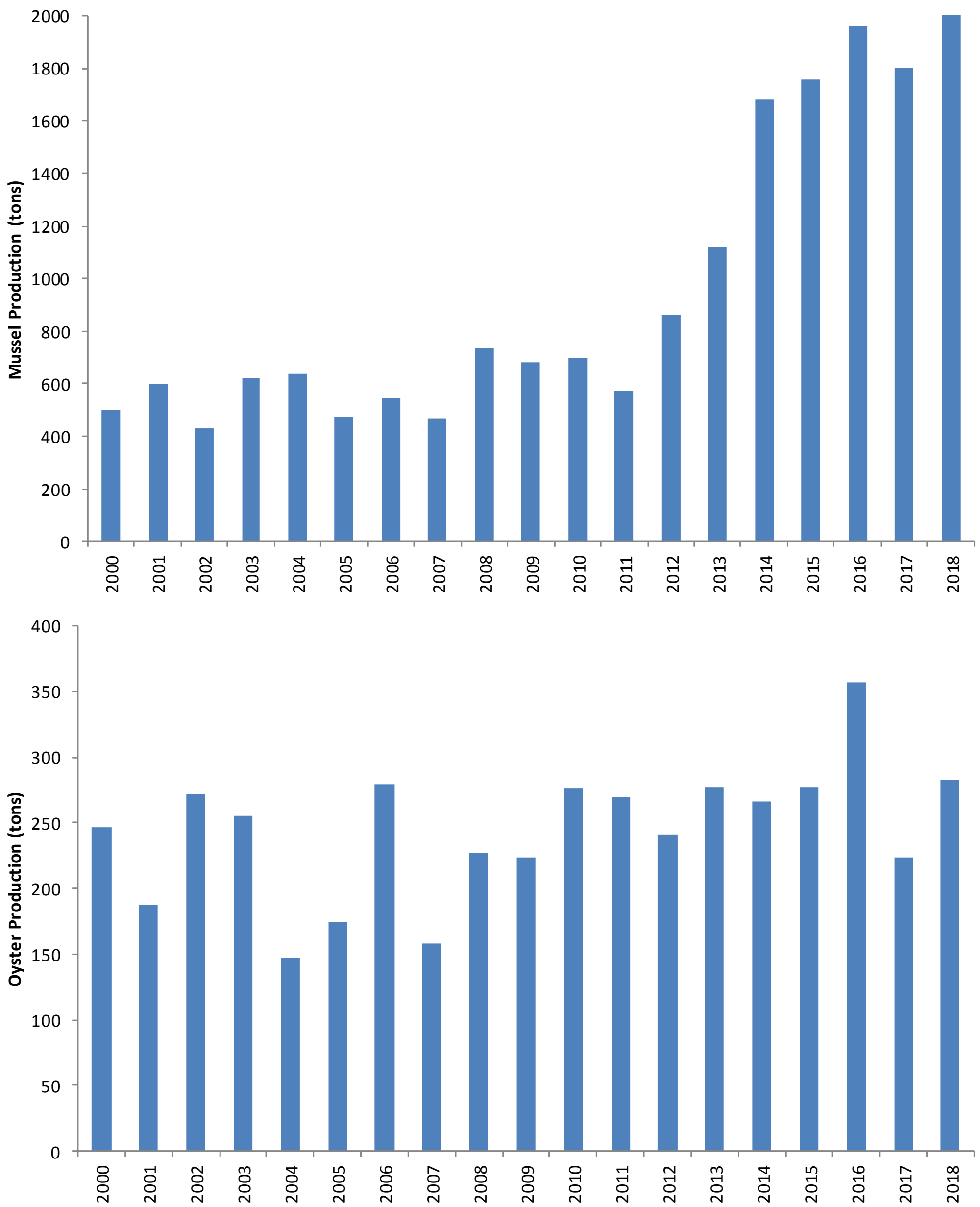


Figure 3.56. Annual mussel (top) and oyster (bottom) production (tonnes) in Saldanha Bay between 2000 and 2018 (source: Department of Agriculture, Forestry and Fisheries 2017 unpublished data, which may be subject to change).

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). Offshore 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 land-based systems.

Southern Atlantic Sea Farms attempted to pioneer Atlantic salmon in Saldanha Bay. During the pilot phase of this project, however, it was found that Small Bay 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) has experimentally been farming 50 tonnes of finfish per annum in Saldanha Bay during the last year. The experimental phase has been successful and Molapong appointed Ecosense CC to conduct a Basic Assessment process to obtain Environmental Authorisation the phased installation of sea cages on 28 ha for the production of finfish, mussels and seaweed in Saldanha Bay up to 2000 tonnes per year. Environmental Authorisation was issued on 8 June 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.

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 tons (graded) on long line. Furthermore, permission was granted to produce 1000 tons 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 not yet commenced (Andrew MacLachlan, *pers. comm.* 2019).

Operational phase environmental impacts of finfish cage culture have been well reported in international literature and include:

- Incubation and transmission of fish disease and parasites from captive to wild populations (Refer to AEC 2016 for more detail on amoebic gill disease (AGD) caused by *Paramoeba perurans* can cause high mortality, poor fish welfare and reduced growth if not treated early in the eruption phase);
- Pollution of coastal waters due to the discharge of organic wastes;
- Escape of genetically distinct fish that compete and interbreed with wild stocks that are often already depleted;
- 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;
- Physical hazard to cetaceans and other marine species that may become entangled in ropes and nets; and
- Piscivorous marine animals (including mammals, sharks, bony fish and birds) attempt to remove fish from the cages and may become tangled in nets, damage nets leading to escapes and stress or harm the cultured stock. Piscivorous marine animals may also be attracted to the cages that act as Fish Attractant Devices (FADs) and in so doing natural foraging behaviours and food webs may be altered. Farmers tend to kill problem predators or use acoustic deterrents; and
- User conflict due to exclusion from mariculture zones for security reasons.

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. While many of developments are ostensibly "land-based", a good number of them rely on ships to bring in or take away their raw material and/or processed products. While the increase in vessel traffic associated with each of these individual developments may be small in each case, they collectively contribute to the ever-increasing number of vessels visiting the Bay each year and also to the ever 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 to 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 home to a range 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 (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.

4.1.1 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: a situational analysis or status quo assessment; a vision, priority and objectives setting component; and, 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 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

4.1.2 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 an 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.

4.1.3 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 taken into account 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.

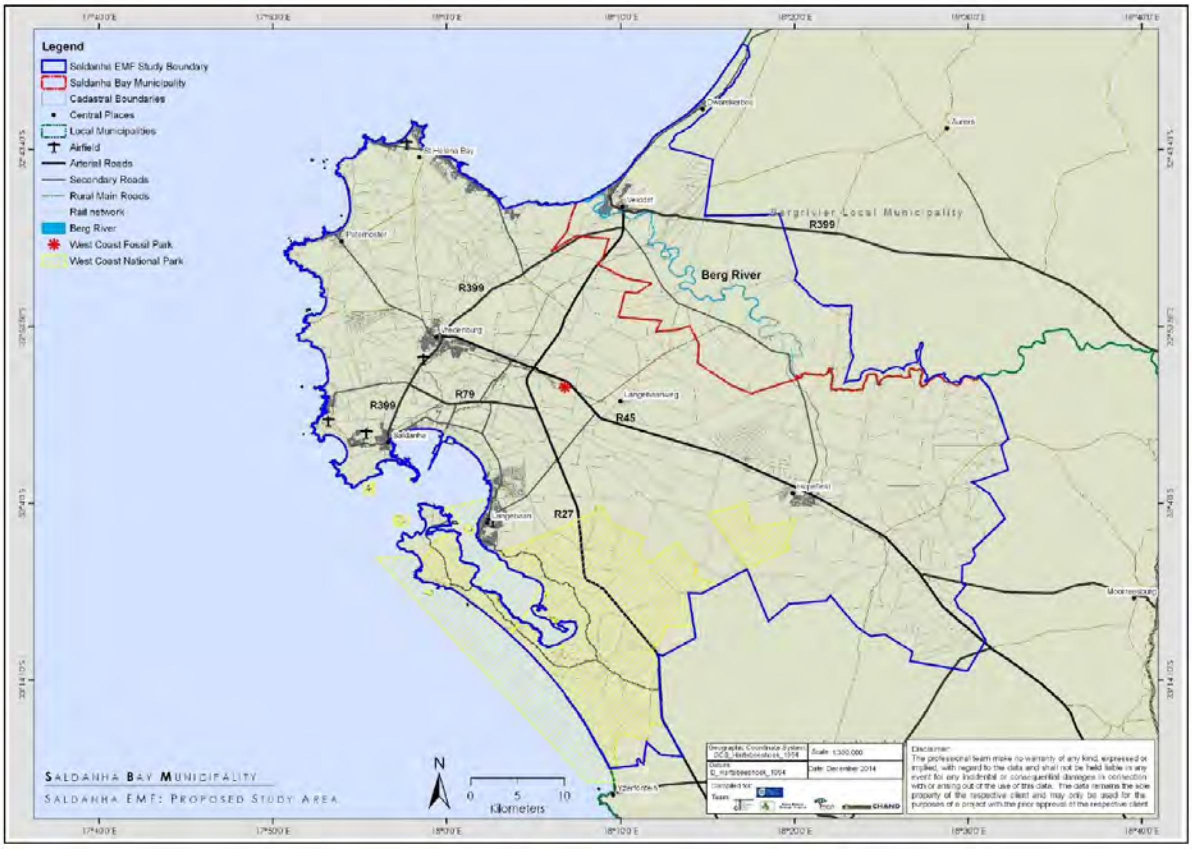


Figure 4.1 **Study Area for the Greater Saldanha Bay Environmental Management Framework (DEA&DP 2017).**

4.1.4 Generic Environmental Management Programme

DEADP compiled an Environmental Management Programme (EMPr) Key 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 (DEADP 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 this year 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.

4.1.5 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)).

4.1.6 Coastal erosion management

Beach erosion in Saldanha Bay, particularly at Langebaan Beach, has been the subject of much concern in recent years. On-going erosion for the past 30 years has been documented, with the loss of over 100 m of beach in some areas since 1960 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), as have the various ad hoc responses to these erosion problems (e.g. construction of groynes and rock revetments along Langebaan Beach, and gabion walls on Paradise Beach). Two Environmental Management and Maintenance Plans (EMMP) were drafted by Common Ground Consulting and approved by the DEA&DP, which provided guidance on strategic level erosion control and mitigation (Common Ground Consulting 2013a and b) (for more detail refer to Anchor Environmental 2013b).

A recent report by Flemming (2016) has identified dredging operations conducted during the Port construction programme as being a possible contributor to these problems (i.e. erosion of Langebaan Beach, Figure 4.2). 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 (an area where sediment derived from Langebaan Lagoon had been deposited over many thousands of years) (Figure 4.3, Figure 4.4). 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 4.5), potentially 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 4.2. Position of the original shoreline at Langebaan Beach in 1975 (Source: Flemming 2016).

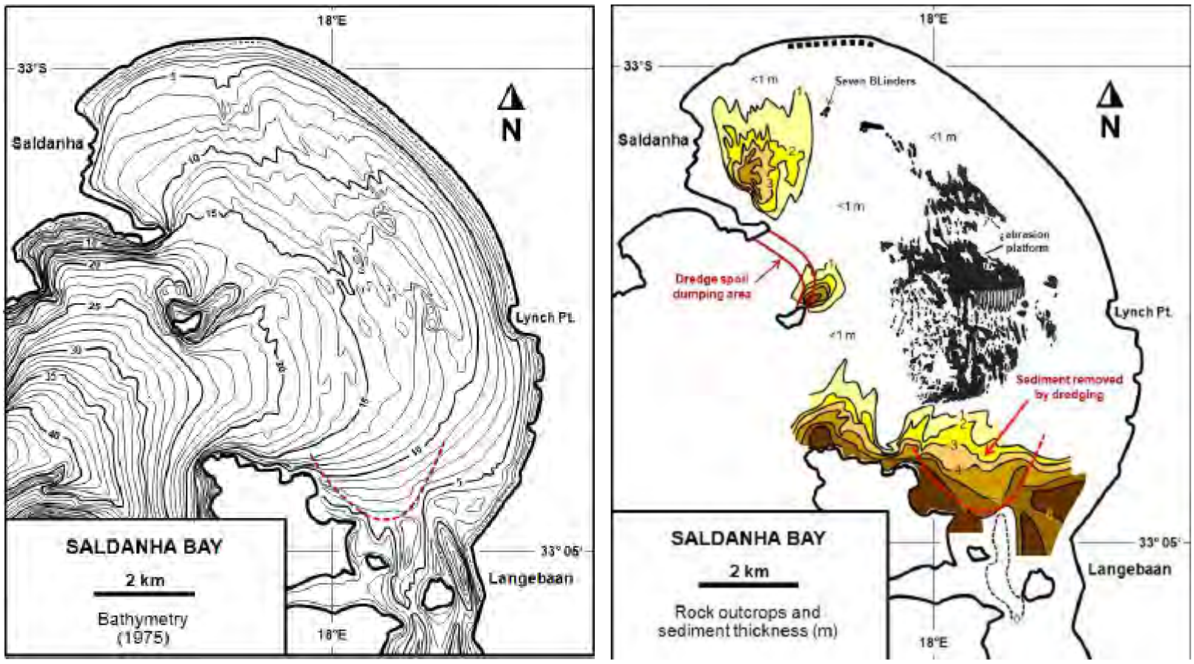


Figure 4.3. Ebb tide delta at the entrance to Langebaan Lagoon where sediment was dredged for construction of the causeway between Marcus Island and the mainland in the late 1970s. Source: Flemming (2016).

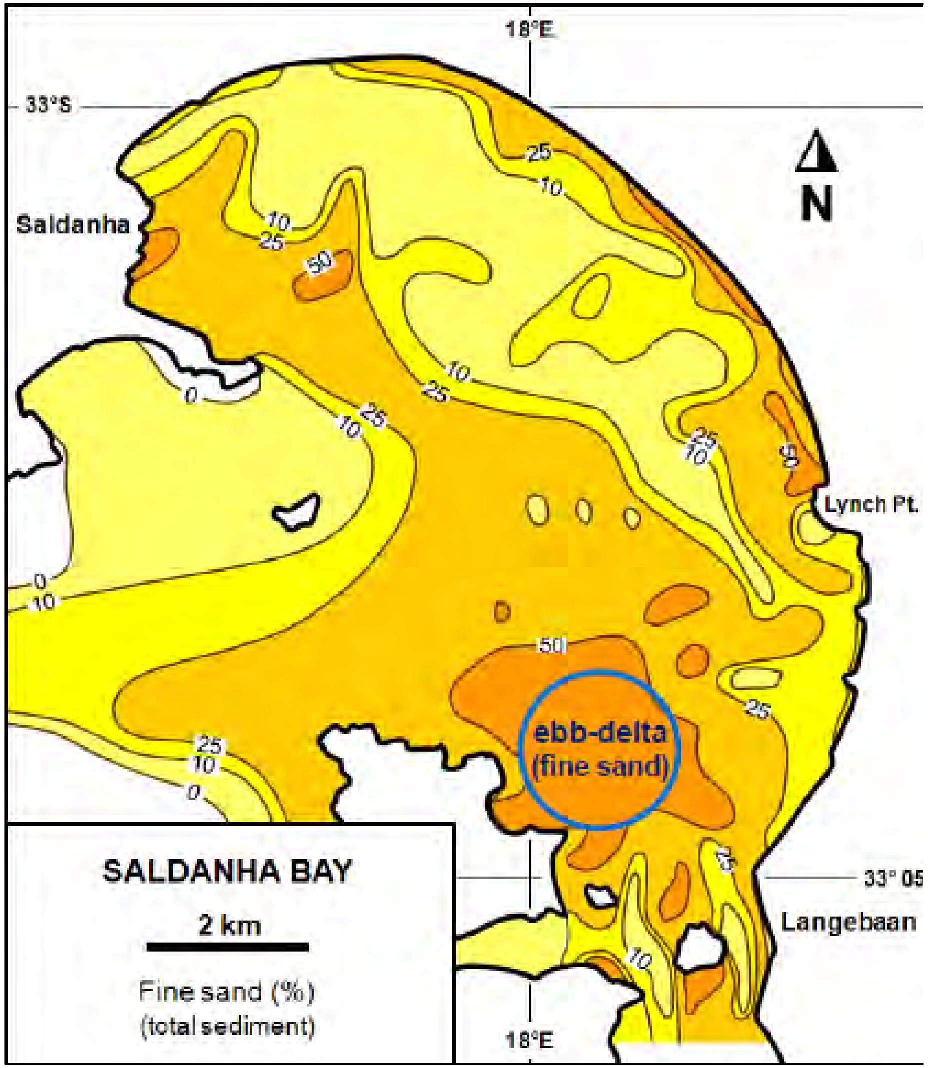


Figure 4.4. **Ebb tide delta at the entrance to Langebaan Lagoon where sediment was dredged for construction of the causeway between Marcus Island and the mainland in the late 1970s. Source: Flemming (2016).**

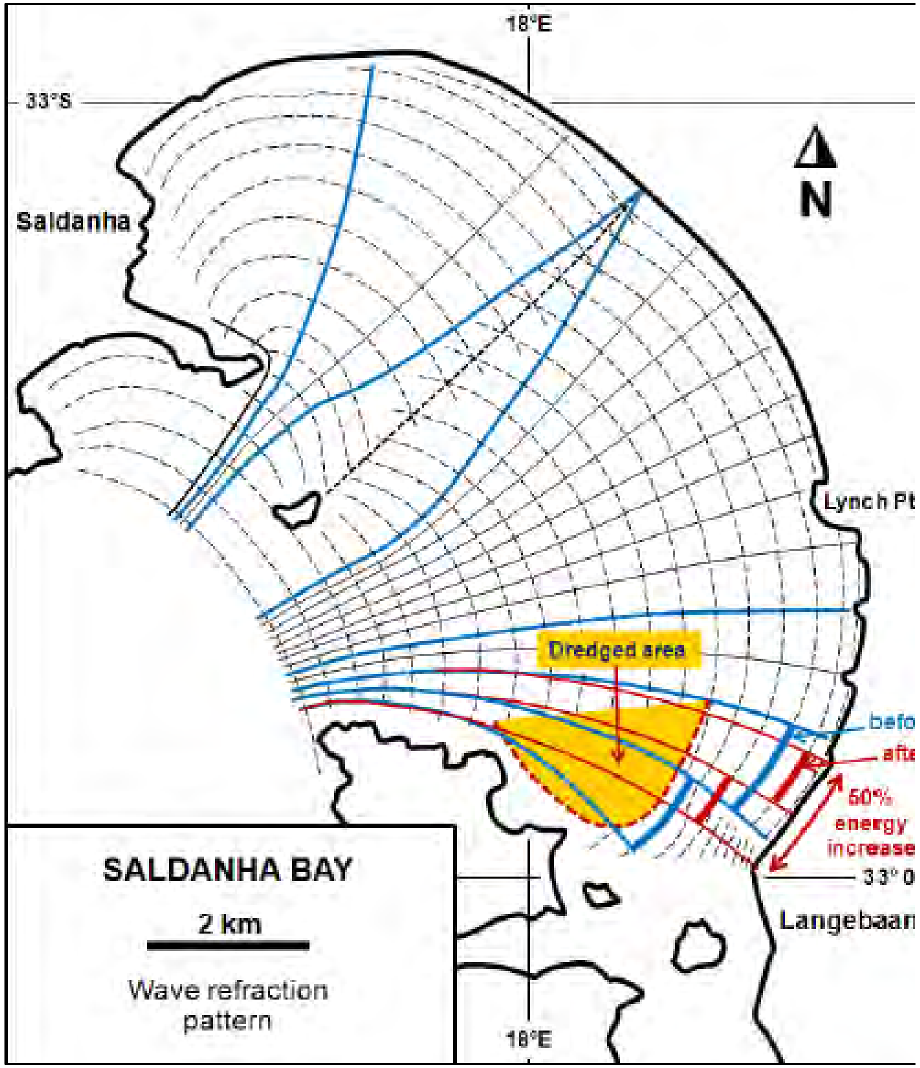


Figure 4.5. **Changes in wave refraction patterns and a consequent increase in wave energy density at the shoreline at Langebaan Beach - a result of sediment removal during the construction of the causeway linking Marcus Island with the mainland. Source: Flemming (2016).**

The Langebaan municipality agreed during 1994 to start a program to monitor change (erosion/accretion) in the beaches between Leentjiesklip 1 (Strandloper restaurant) and Alabama street as part of a beach protection investigation. This entailed undertaking beach surveys bi-annually (at the end of summer and winter) during spring low tide. Measurements were taken between the high-water mark and approximately two meters below mean sea-level across 24 transects within the study area (Figure 4.6). 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.

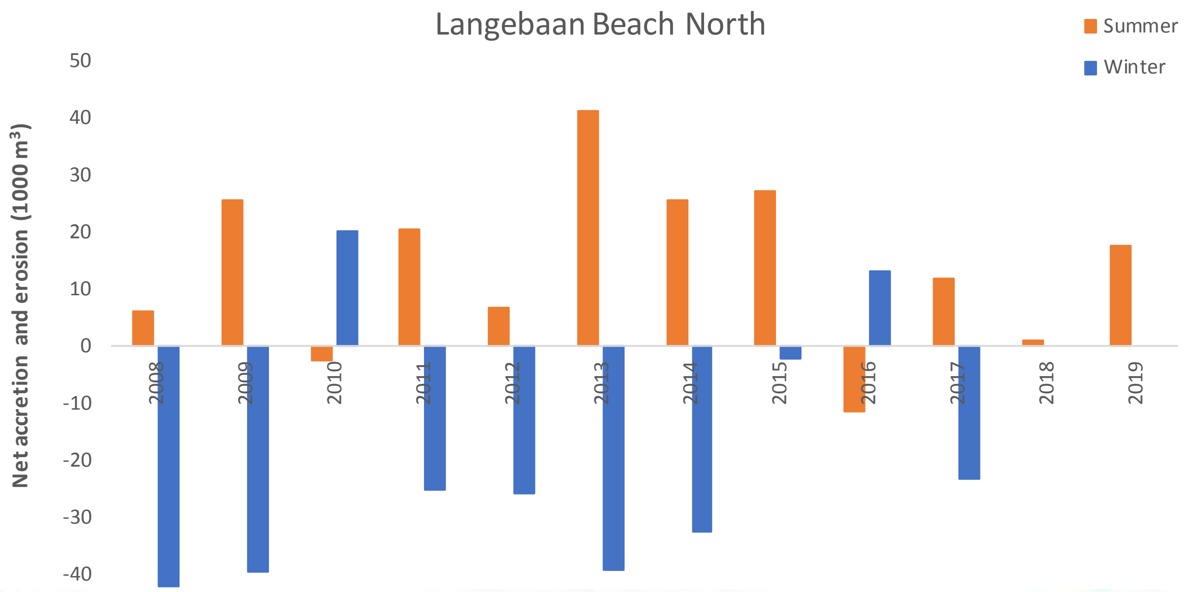
Figure 4.6. **Erosion monitoring sampling sites in Langebaan between Leentjiesklip 1 (Strandloper restaurant) and Alabama street (SBWQFT 2019).**

The interim and first progress report present spatially detailed data for the period investigated but lack visual presentation and interpretation of this data over time. The first progress report (i.e. November 2018-April 2019) (SBWQT 2019) also presents historic data on sediment accretion (gain) and erosion (loss) (m^3) (2008-2019) for the northern and southern portions of the study area.

The measured net gain/loss measured at the northern and southern portions over time is shown in Figure 4.7. What is most interesting about the data but has not been highlighted in the report is the fact that the seasonal erosion patterns are reversed for the northern and southern portions of Langebaan Beach. Langebaan Beach North generally erodes in winter and accretes in summer with only two anomalies in 2010 and 2016 (top graph in Figure 4.7). The opposite is true for Langebaan Beach South which typically erodes in summer and accretes in winter (bottom graph in Figure 4.7). The extent of the change (erosion and accretion) on Langebaan South Beach is also much reduced relative to Langebaan North Beach. Overall, in the period May 2008 and Nov 2018, Langebaan North Beach has experienced a net loss of sand amounting around 45 440 m^3, while Langebaan South Beach has experienced a net gain of sand amounting around 30 705 m^3. Overall net loss of sand is estimated at around 14 735 m^3.

It is likely that this seasonal reversal and the differences in the magnitude of the erosion are linked to seasonal reversal of the wave climate experienced at these two sites, with wave energy at Langebaan North Beach being much more intense and peaking 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 peaks in summer (and is derived from the southerly winds blowing across the Lagoon at this time of year)..

The first progress report provided data on net sediment gain and/or loss at each of the monitored transects between November 2018 and May 2019 (i.e. a summer period, refer to Table 1 of the report). This data is presented on Figure 4.8. These recent results concur with the long-term pattern, inasmuch as most of the sites on the northern portion of Langebaan Beach experienced accretion, while those on the southern portion experienced erosion. Much of the accretion that was observed was localised to the two transects immediately north of the groins (i.e. transects 8E and 10B, Figure 4.8).



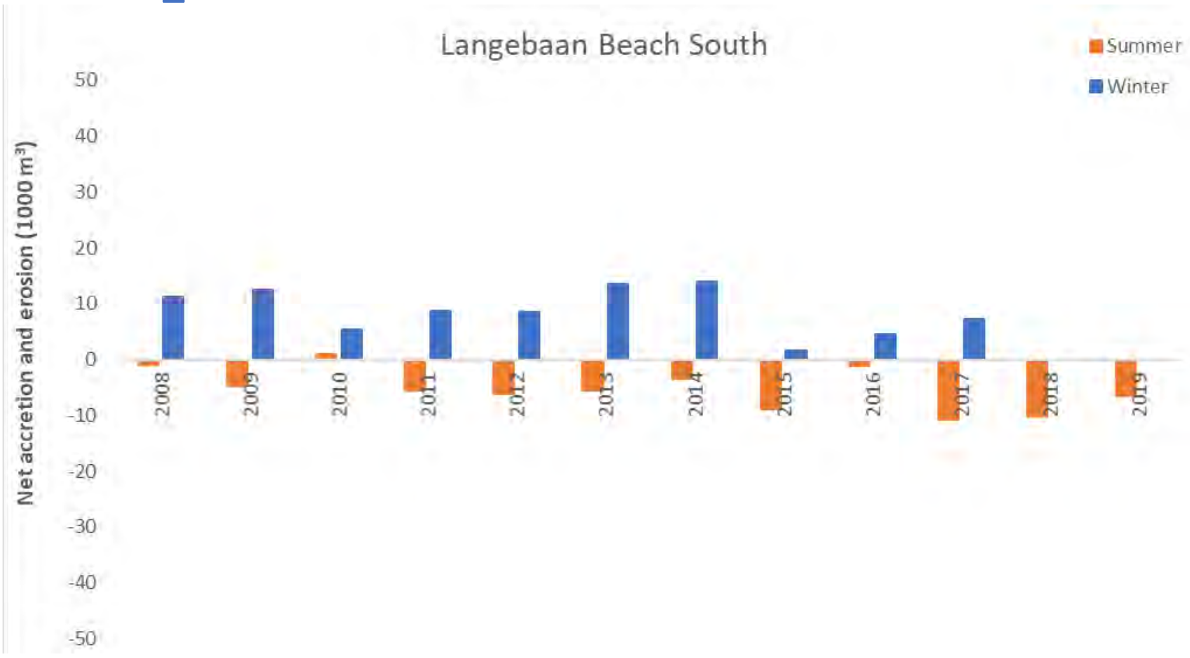


Figure 4.7. **Long-term erosion and accretion monitoring of Langebaan Beach between Leentjiesklip 1 (Strandloper restaurant) and Alabama street. Net sand accretion and erosion on Langebaan Beach North (top) and South (bottom) are shown for summer and winter between November 2017 – October 2018. Note that no data was collected in summer 2018 (Data Source: SBWQFT 2019).**

Figure 4.8. Erosion and accretion pattern between November 2018 and May 2019 at Langebaan Beach (Leentjiesklip 1 (Strandloper restaurant) to Alabama street

5 GROUND WATER

5.1 Introduction

Langebaan Lagoon is a unique 'estuary' in that it is not fed by runoff from a river but receives its fresh water from a groundwater aquifer – a paleochannel from an old river. The classification of the 16 km long Langebaan Lagoon that adjoins Saldanha Bay on the West Coast has been debated for some time. Langebaan Lagoon has many of the characteristics of an estuary. This includes the calm coastal waters that are protected from marine wave action and biota that includes many species that are typically found in estuaries. The system lacks a conventional estuarine salinity gradient because of the absence of an inflowing river. Groundwater flows into the lagoon in certain sections, however, and it is possible that it functions as a subterranean aquifer, as there is daily variation in the salinity of the wells dug along the shore of the lagoon.

At 3-4 km wide, with channels up to 5 m deep, Langebaan is much larger and deeper than conventional coastal lagoons which are usually small and shallow. Whitfield (2005) suggested that the term "coastal embayment type of estuary" be used to describe Langebaan because it does receive freshwater inflow from land drainage (input from the aquifer), and also has some typical estuarine biota. This would place the Langebaan Lagoon in a class of its own, separating it from "estuarine bays" which are fed by rivers.

The Saldanha area is in an arid area with a low average rainfall, which is facing growing pressure from industrial developments and residential growth. Equally important are projected future scenarios due to climate change and the associated potential impacts for example:

- higher mean temperatures will lead to increased evaporation and decreased water balances
- the resultant general drying trend causes increases in costs for water resources;
- mean sea level rise could lead to salt-water intrusion into ground water and coastal wetlands.

The areas that surround Langebaan Lagoon are covered mostly by natural vegetation, waterbodies or wetlands, especially the areas that fall within the West Coast National Park. The areas outside the National Park are mostly cultivated – dryland farming, with urban and industrial development in the Saldanha area (Figure 5.1). By contrast, land surrounding Saldanha Bay has been extensively transformed most for residential and industrial development. This has important implications for groundwater use, quality and recharge all of which are addressed in this chapter.

Current and potential future impacts on this valuable and sensitive resource require careful and comprehensive planning. The recent drought has highlighted the risk of relying on surface water for the water supply to the area, especially since the municipality is one of the last recipients of water from the Western Cape Water Supply System. This is exacerbated by the problems of maintaining the Misverstand Weir at the levels required to provide water when the Berg River is not flowing at its normal levels. Groundwater has the potential to provide water resources to the Saldanha area, if managed sustainably, thereby relieving the pressure on the surface water supply. The National Water Act of 1998 (Act 36 of 1998) (DWAF, 1998) considers groundwater as a national resource to be managed by the Department of Water and Sanitation in a sustainable manner, with the cooperation of the municipalities and other water users.

The Water Services Act (Act 108 of 1997) (DWAF, 1997) provides the framework for the delivery of water services by the Water Services Providers (the West Coast District Municipality and the Saldanha Bay Local Municipality in this case). These two Acts cover the legal obligations, rights, responsibilities and constraints for the sustainable development and management of the water resources in South Africa (Pietersen, 2006).

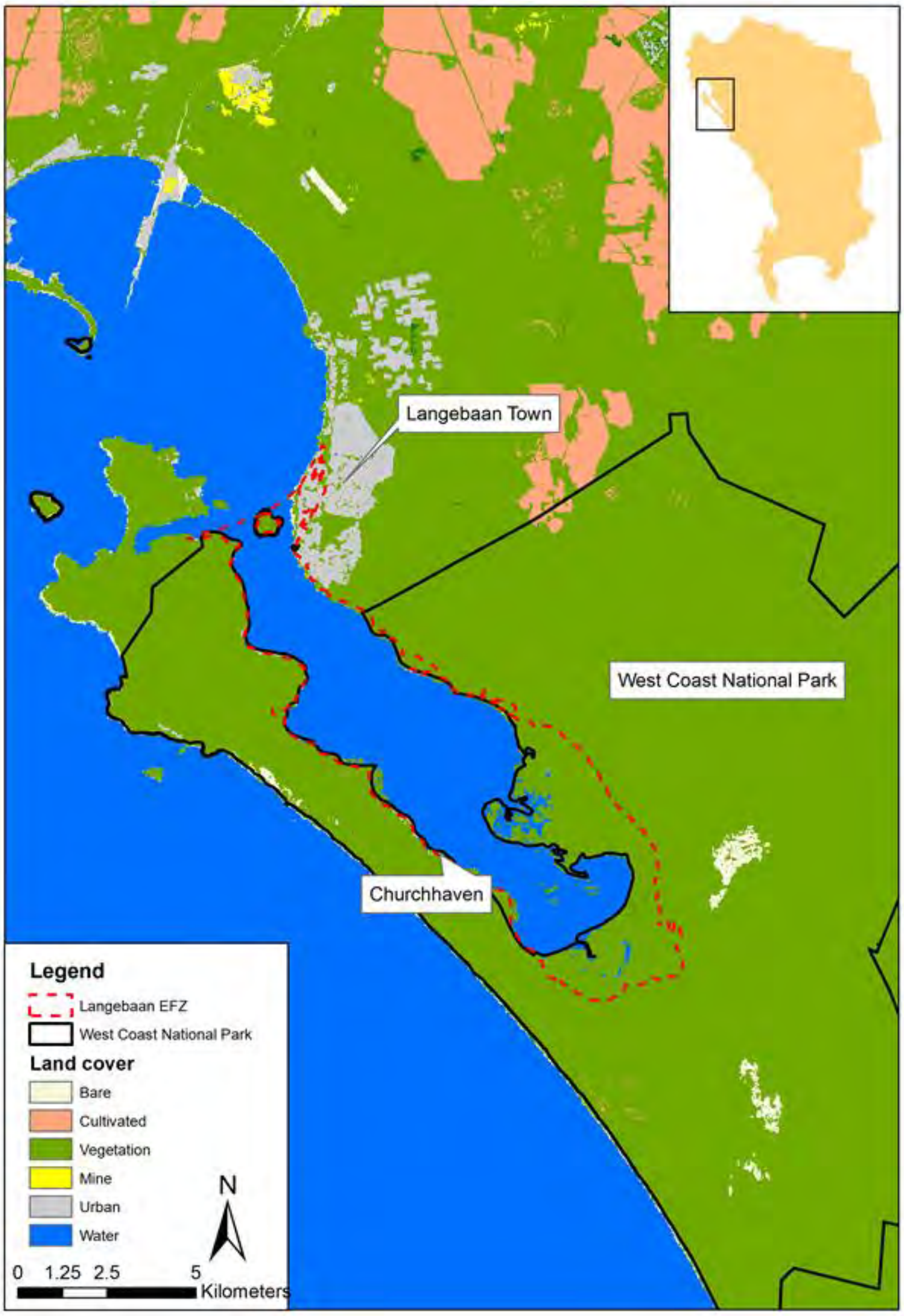


Figure 5.1 **Land use map of the area surrounding Langebaan Lagoon.**

Groundwater resides under the earth's surface in soil pore spaces and in the fractures and fissures of rock formations. A unit of rock or an unconsolidated deposit is called an aquifer when it can yield a usable quantity of water. The depth at which soil pore spaces or fractures and voids in rocks become completely saturated with water, is called a water table or water level. Groundwater is recharged by rainwater that infiltrates the subsurface to reach the saturated parts, or from surface water bodies like rivers and dams. Groundwater then flows below the surface and will eventually reach the surface again and discharge to seeps, wetlands, springs and rivers. Groundwater is abstracted for agriculture – mostly stock watering, municipal and industrial use with boreholes equipped with pumps and wind pumps. Groundwater is often cheaper, more convenient, locally available, and less vulnerable to pollution than surface water. It is often used for public water supply. Polluted groundwater is less visible than surface water, but it is more difficult to clean up. Groundwater pollution is most often the result of improper disposal of waste on land. Major sources include industrial and household chemicals and garbage landfills, excessive fertilizers and pesticides used in agriculture, industrial waste lagoons, tailings and process wastewater from mines, industrial fracking, oil field brine pits, leaking underground oil storage tanks and pipelines, sewerage sludge and septic systems. In addition to pollution, over abstraction can severely alter or irreparably damage an aquifer to such an extent that it will no longer function properly and in severe cases, cause land subsidence.

5.2 Aquifer description and climatic setting

Saldanha Bay is in the winter rainfall region of the Western Cape, where rain mostly falls between May and October. There are marked geographic variations in rainfall in this region, with the most rain falling in the south-western part, decreasing towards the Berg River in the north and east (Figure 5.2). This naturally influences recharge rates for the aquifer system in this area. Temperatures are highest during the summer months, typically peaking in February and March. This means that the potential for evapotranspiration is at its highest during the summer months.

The surface geology of the area is dominated by fine to medium grain sand and calcrete (limestone), with granite outcrops as well as the Colenso Fault System making up the important features in the geology of the area (Figure 5.3). The latter is thought to play an important role in the groundwater flow in the area, but it is not fully understood at present.

The lithostratigraphy of the Cenozoic (sand) deposits is summarized in Table 5.1. These deposits are found in paleochannels that were cut out of the basement rocks of granite and shale. These paleochannels were formed by rivers, such as the Berg River, which over time have changed direction and flow paths. A spatial classification of these deposits based on the properties of the strata and their geographical distributions is depicted below as the orientation of these paleo-channels in the basement rocks (Figure 5.4, Roberts & Siegfried 2014). This shows that the Adamboerskraal, Langebaan Road and Elandsfontein Aquifer Units are linked, and that there may be flow between these different units. The link with the Grootwater Aquifer Unit to the south of the map is inferred and the level of connection is not clear.

Data on water levels measured in the Elandsfontein Aquifer Unit also suggest that there must be a link between the Langebaan Road and the Elandsfontein Aquifer Units. There was a time delay of one to two years in the response of the water levels in the Elandsfontein Aquifer Unit to the abstraction of groundwater from the lower layer of the Langebaan Road Aquifer wellfield, so questions still remain as to the system(s) that recharge these aquifers and the extent to which they are connected.

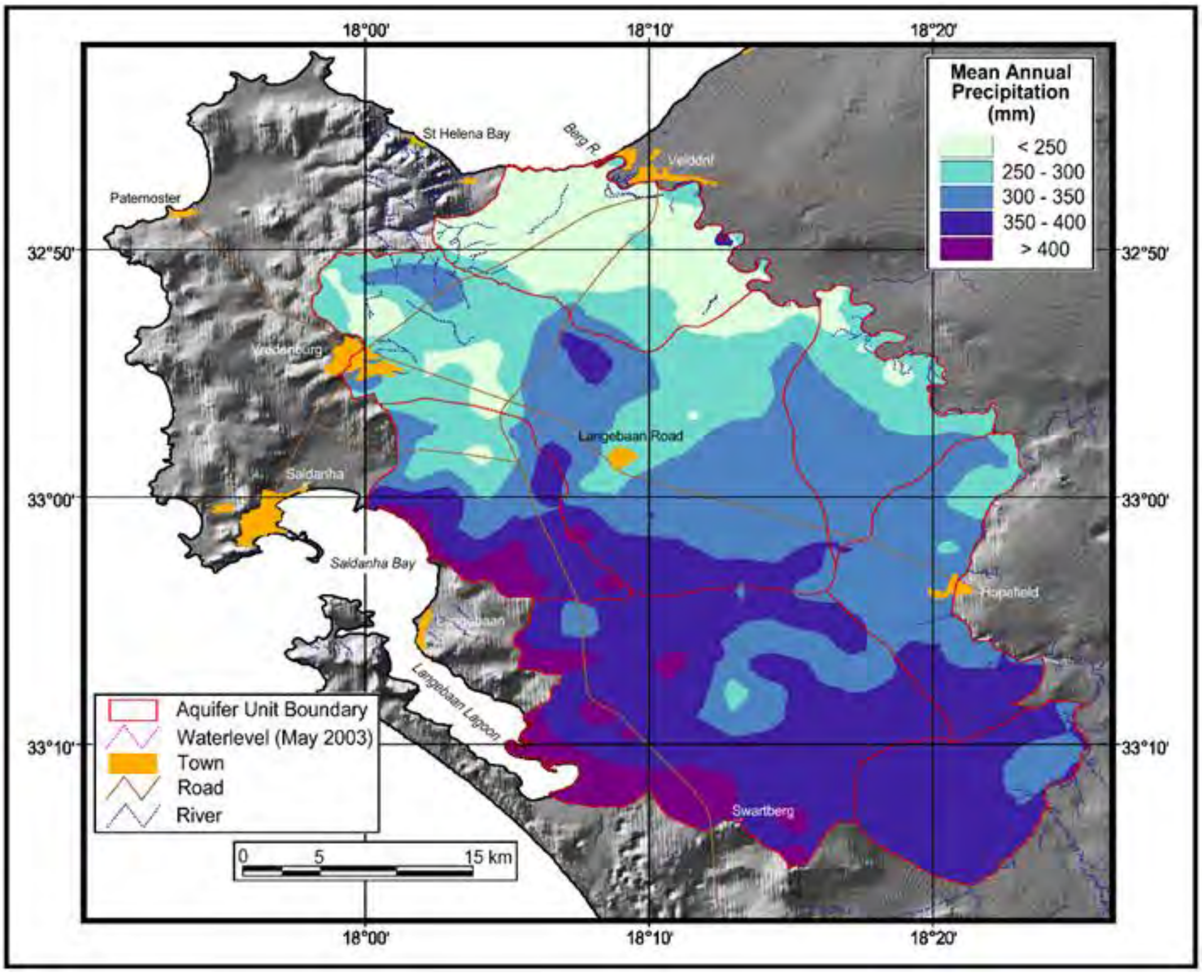


Figure 5.2 **Mean annual precipitation for the Langebaan Road/Elandsfontein aquifer unit area (Woodford and Fortuin 2003).**

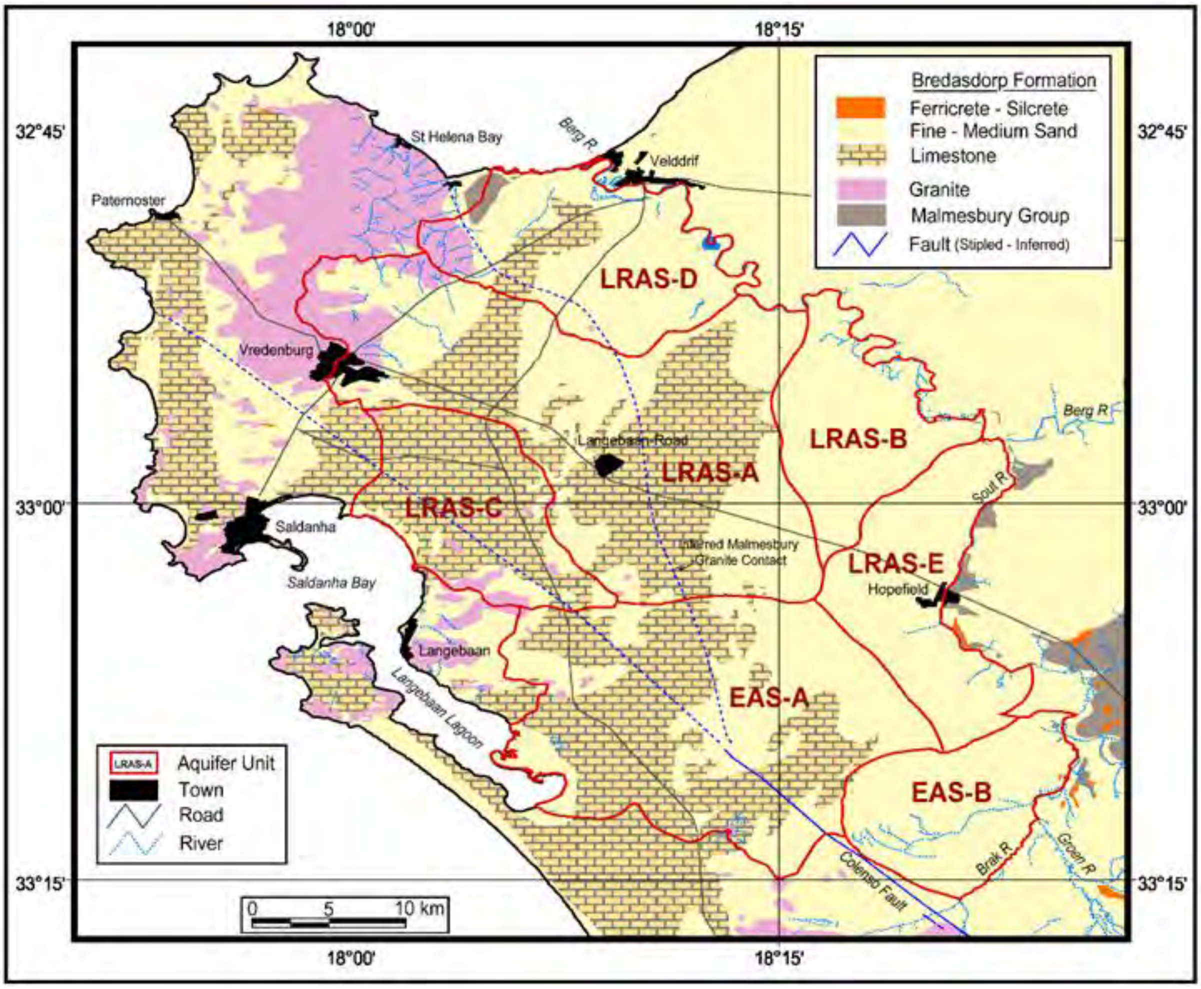


Figure 5.3 **Surface geology for the Langebaan Road/Elandsfontein aquifer unit area (Woodford and Fortuin 2003).**

Table 5.1 **Lithostratigraphy of the Cenozoic deposits (after Woodford and Fortuin, 2003).**

Epoch	Age	Lithostratigraphic Unit		Description	Depositional environment
		Formation	Member		
Holocene – Pleistocene	1.7	Bredasdorp	Witzand	Calcareous dune sands.	Aeolian.
			Langebaan	Calcretized limestone.	Aeolian.
			Velddrif	Shelly sand.	Marine.
			Springfontyn / Noordhoek	Silica to peaty sand.	Aeolian.
Pliocene	5.2	Varswater	CSM	Calcareous sands.	
			PPM (Duynefontein)	Muddy sand with pelletal phosphorite.	Marine.
			QSM	Quartzose sand.	Marine.
			SGM (Silwerstroom)	Shelly gravel.	Marine.
Late Miocene	10	'Saldanha'		Gravels.	Marine.
Miocene	22	Elandsfontyn		Predominantly coarse sand and gravel, interbedded silty, clayey and peaty layers.	Fluviatile.

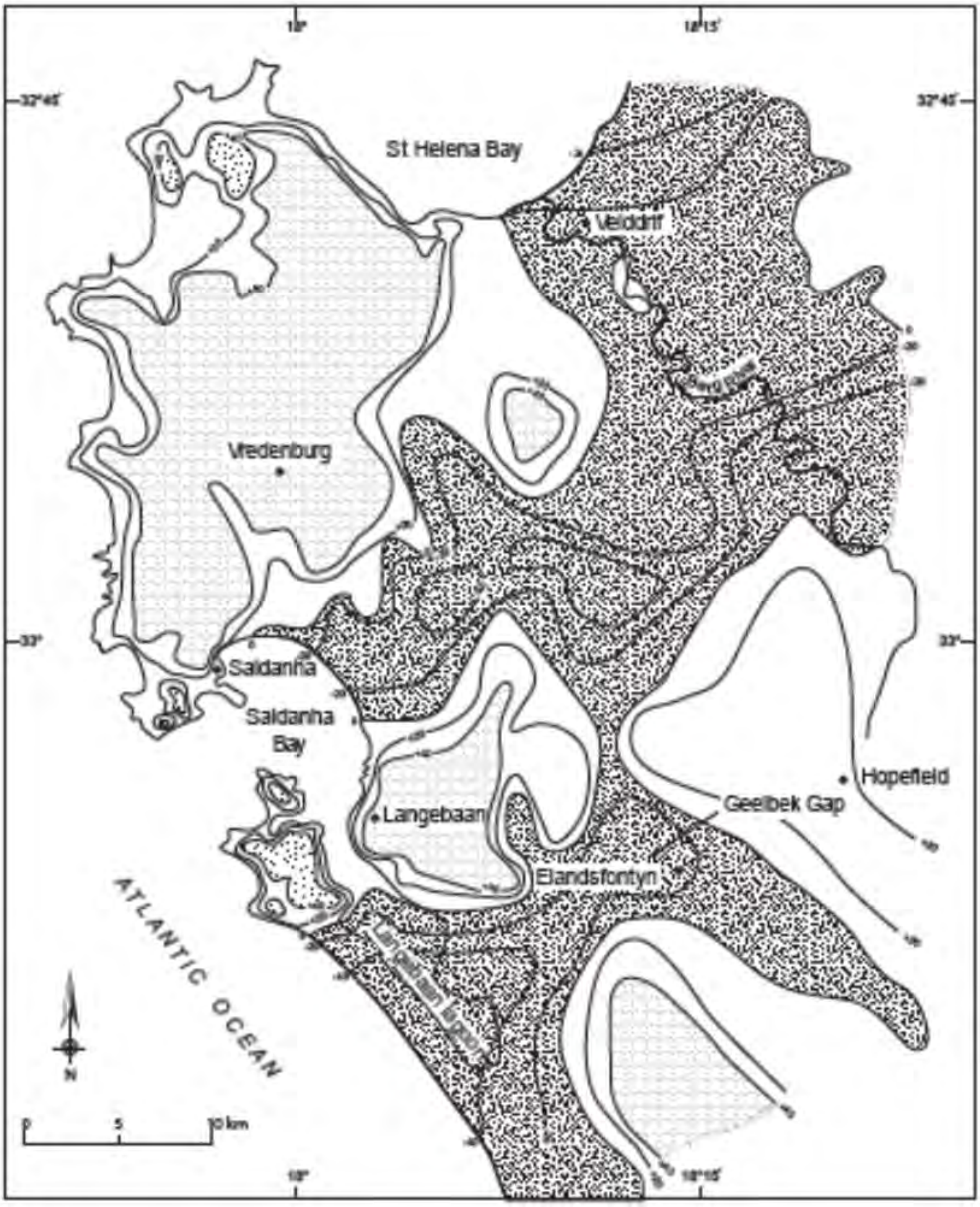


Figure 5.4 **Pre-Cenozoic basement topography of the Saldanha, Vredenburg and Velddrif sheets. Light stipple: elevations greater than +40m; heavy stipple: elevations below present sea level. Data from Rogers (1980), Timmerman (1988), Cole and Roberts (1996) and from recent drilling (Roberts and Siegfried, 2014).**

There is a cross section along the R27 coastal road in the area of Geelbek that gives an indication of the geology of the Elandsfontein paleochannel (Figure 5.5). The Elandsfontein clay layer is still present in BH4 just inside the West Coast National Park, but it is not present in the boreholes closer to the Langebaan Lagoon. The clay layer thins out near the sides of the palaeochannels (both in the Langebaan Road and Elandsfontein Aquifer Units) and may even be absent at the edges. It also seems to be absent to the west of the R27 coastal road for both aquifer units, which means that the upper and lower aquifer layers are in direct contact without a confining layer separating them. The Water Research Commission (WRC) is actively working on refining these findings through a project titled, 'Towards the Sustainable Exploitation of Groundwater Resources along the West Coast of South Africa (project K5/2744)'.

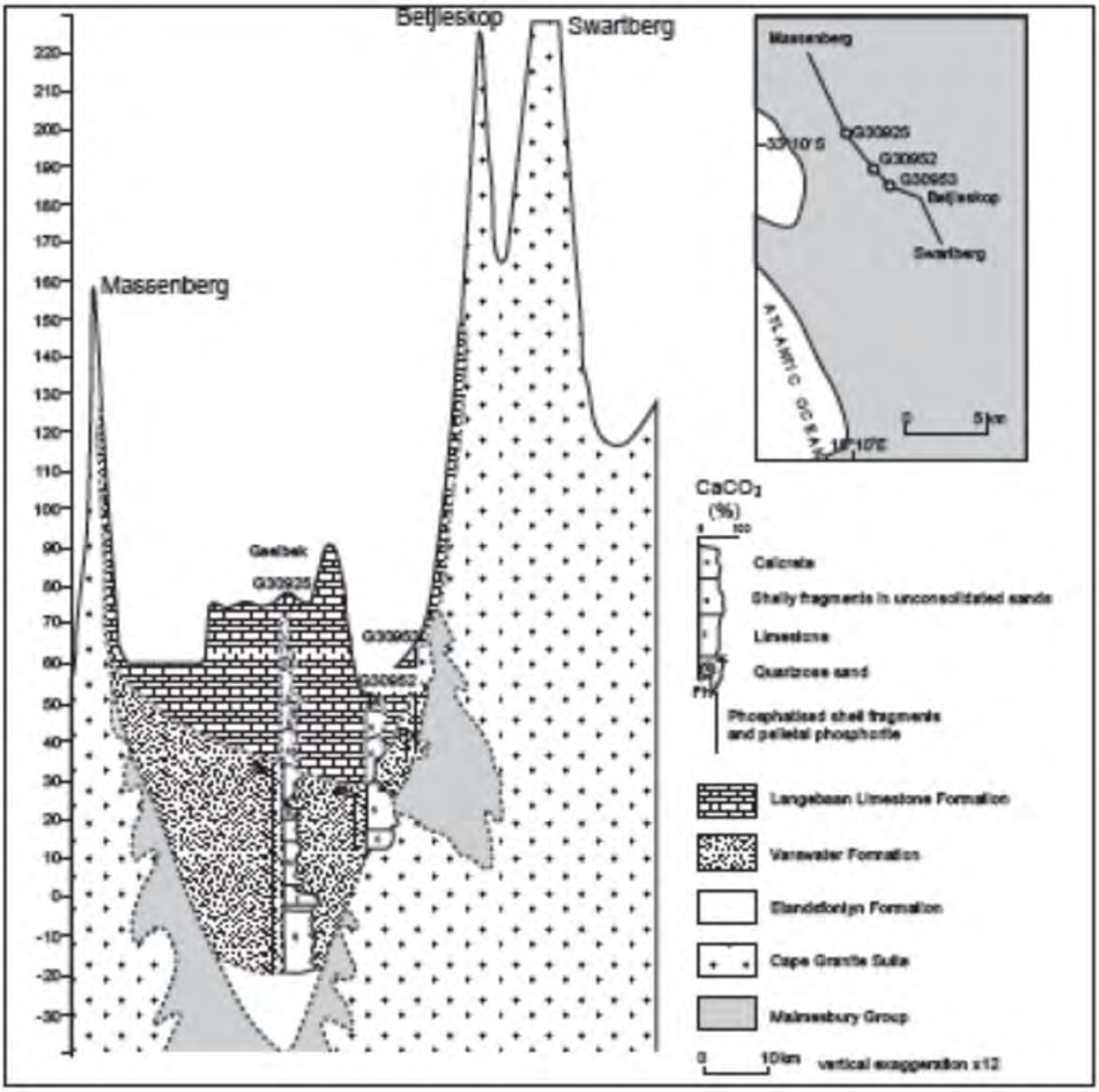


Figure 5.5 **Cross-section through Cenozoic strata on the Saldanha sheet (from Rogers 1980) (Roberts and Siegfried, 2014).**

Mean annual effective recharge from rainfall as well as water level elevation and flow in the lower confined layers of the two aquifer units, as understood by Woodford and Fortuin (2003), is depicted below (Figure 5.6). There is, however, some uncertainty on the recharge of the aquifer, as recent work by Smith (2017) and Nel (2019), has shown that recharge may not be derived from local rainfall at all (Nel *pers comm*.) The map also includes estimations provided by Woodford and Fortuin (2003) on the volumes of water that may be discharged to the surface bodies in the area. This includes discharge to the Langebaan Lagoon (3.854 Mm^3/a), Saldanha Bay (0.785 Mm^3/a), Berg River (0.730 and 0.525 Mm^3/a) and the various springs and wetlands in the area (0.394 Mm^3/a). Discharge to Langebaan Lagoon accounts for by far the greatest portion of these flows.

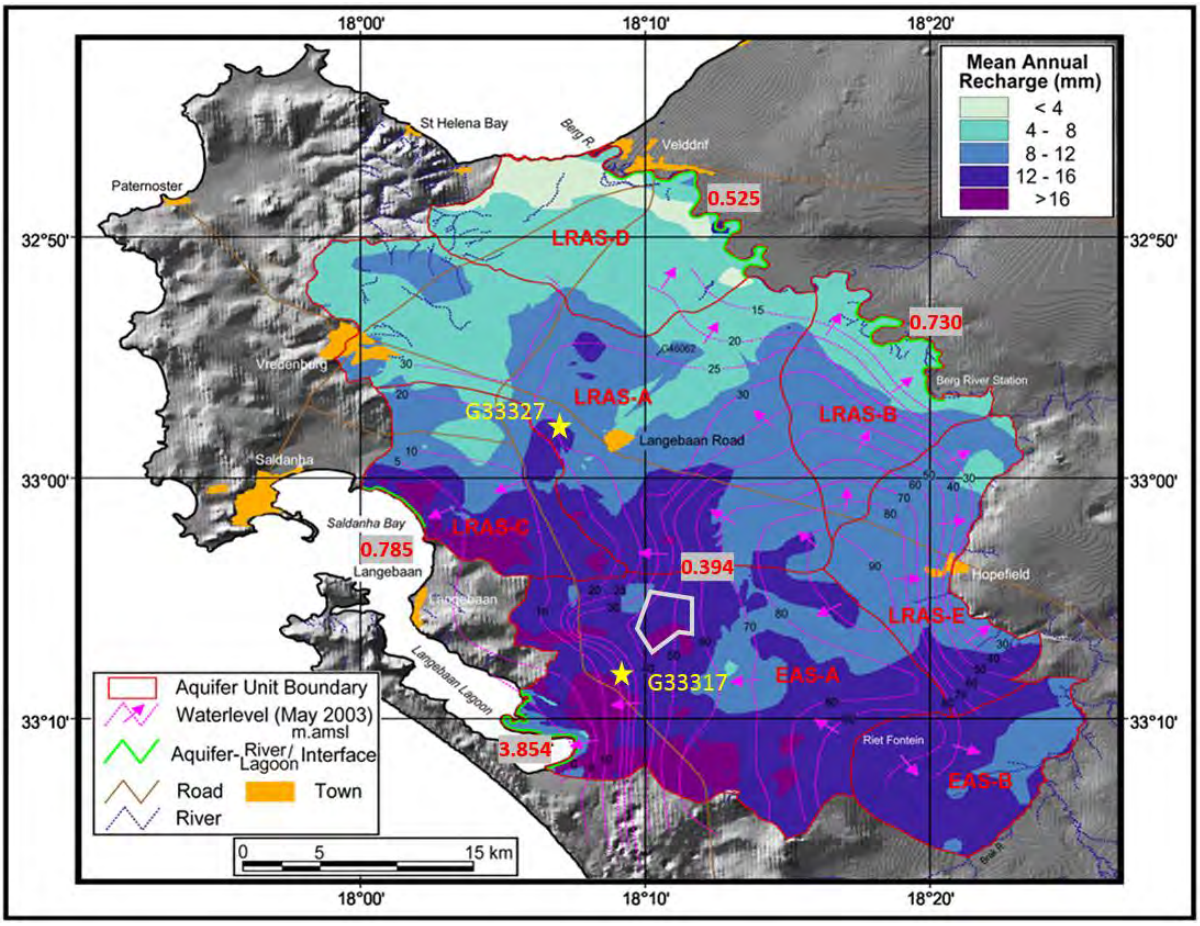


Figure 5.6 **Mean Annual Effective Recharge (mm) from rainfall and water level elevation in the Lower Confined Layers of the Langebaan Road and Elandsfontein Aquifer Units with the estimated discharge to the different surface water bodies in the area (Woodford and Fortuin, 2003) and approximate locations of Elandsfontein/Kropz (white polygon) BH 33327 and BH33317 (yellow stars).**

Recently, GEOSS has released the 2019 Strategic Environmental Assessment of the Greater Saldanha Bay Area (GSB) (Conrad and Naicker 2019) and according to the 1:500 000 scale groundwater map of Cape Town (3317) the greater Saldanha area has a range of aquifer types with varying associated yields. 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), (0.1 L/s to 0.5 L/s), and fractured (0.5 L/s to 2 L/s) (Figure 5.7) (Meyer, 2001, Conrad and Naicker 2019). Note that these classifications are based on a regional scale and boreholes do occur within the GSB with yields > 5L/s. Borehole yields from the National Groundwater Archive (NGA) have also been depicted on Figure 5.7.

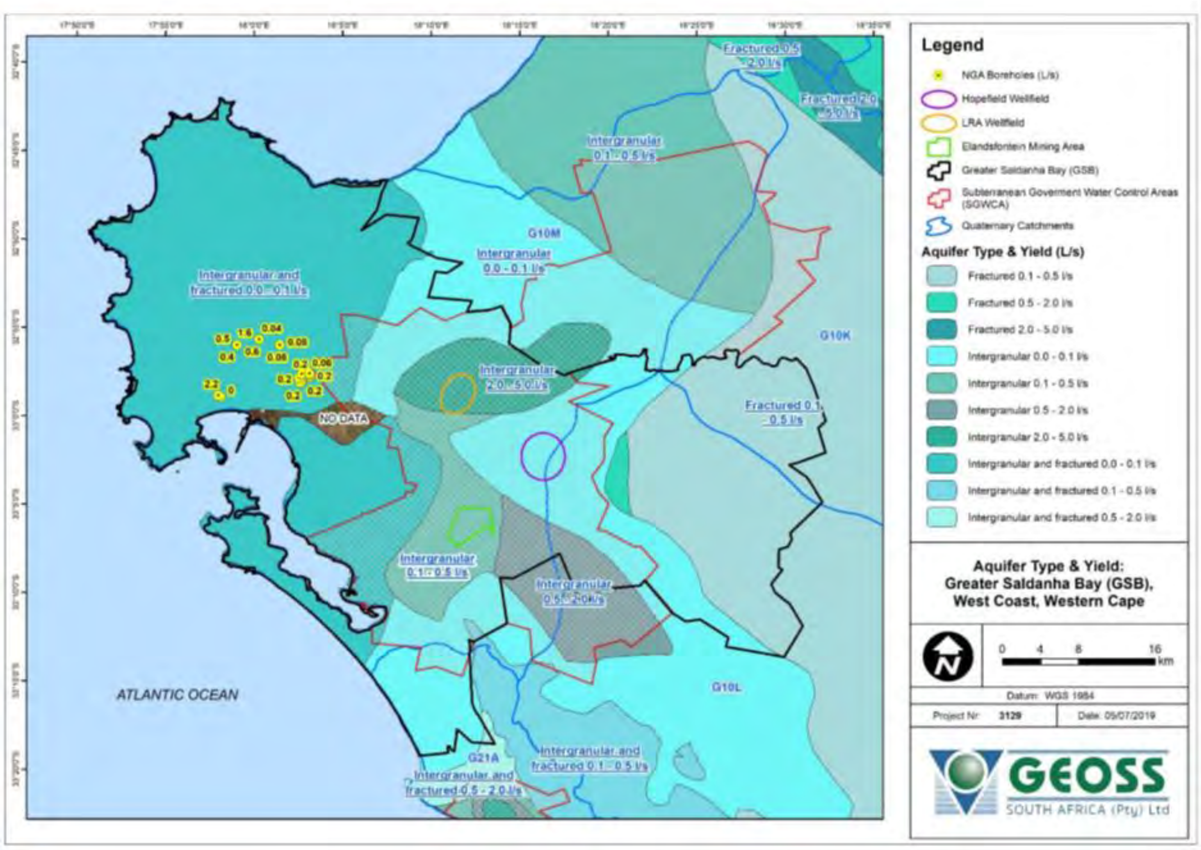


Figure 5.7 **Regional aquifer yield from the 1:500,000 scale groundwater map (Cape Town -3317) (DWAF 2000) Source: GEOSS Report 2019/05-14 Conrad and Naicker 2019.**

5.3 Groundwater use

The West Coast District Municipality (WCDM) operates a wellfield on the Langebaan Road Aquifer that is licenced to abstract 1.46 million m^3 of groundwater per annum. The operations of the wellfield began in December 1999. The abstraction of the groundwater from the aquifer resulted in a bigger decline in water levels than what was expected. Models only predicted a 5 m water level decline, but water levels declined by between 10 and 11 m from the original levels. There was a concern on how this may affect the groundwater discharge to Saldanha Bay, and the monitoring committee decided on a modest reduction of 10% in the abstraction rate.

Phosphate has been deposited in the Langebaan Road Aquifer Unit, as well as the Elandsfontein Aquifer Unit of the West Coast Aquifer System[9]. Phosphate was mined at the Chemfos mine (now the West Coast Fossil Park) from the 1960s until the 1980s, when operations ceased, and the mine was closed. Small scale phosphate mining still takes place in the area of the Langebaan Road by Geckofert. Phosphate mining again made headlines in the area when Elandsfontein Exploration and Mining (Pty) Ltd (EEM), now known as Kropz Elandsfontein, began mining phosphate in the Elandsfontein area (Farm Elandsfontein 349, situated between the town of Hopefield and the Langebaan Lagoon area). A water use authorisation was issued for the mine in April 2017, but operations had to be suspended as a result of appeals against the issuing of the authorisation. Kropz Elandsfontein hopes to recommence operations at the end of 2019.

Kropz will implement the 'roll over' mining method to access the phosphate ore, allowing the concurrent surface rehabilitation to take place. Ore will be mined at a rate of 5 Mtpa (million tons per annum) for approximately 14 years. The targeted phosphate resource lies below the natural water table. For mining to take place, the water table is lowered by extracting groundwater from the underlying aquifer via a series of boreholes upstream of the mining site, which prevents the mine pit from being flooded. The extracted ground water is then fully recharged back into the aquifer downstream of the mining activities, via a dedicated, closed system which essentially means no nett abstraction of groundwater will occur.

Processing water for the phosphate operations will largely be supplied from the Saldanha Bay Municipality. Kropz is in discussions to treat municipal effluent water for industrial reuse in order to eliminate their demand on the municipal water system. Several groundwater specialists' studies have been completed to understand and mitigate the potential impacts of mining activities on the underlying aquifer. Despite the findings of the studies, which suggest that impacts can be mitigated by means of the proposed mining methods, residual concerns have been expressed over potential impacts that the proposed phosphate mine at Elandsfontein may have on groundwater quality and flows to Langebaan Lagoon

The greatest cumulative impacts on groundwater, according to Conrad (2019), are likely to be from the agricultural sector (1.6 Mm^3/a) (this registered quantity is groundwater abstraction for agriculture as of 2016 and probably increased significantly during the drought of 2015 to 2018); abstraction from

[9] The West Coast Aquifer System consists of the Adamboerskraal Aquifer Unit, the Langebaan Road Aquifer Unit, the Elandsfontein Aquifer Unit, and the Grootwater Aquifer Unit. The different aquifer units are multilayer systems, made up of different cenozoic deposits overlaying basement rock of granite and shale.

the Langebaan Road Aquifer wellfield (intermittently operational since 1999 however frequently non-operational due to regular and persistent vandalism) and the Hopefield wellfield (not yet operational –in the final stages of construction and the water use license is still pending) where it is planned to abstract 5.1 Mm^3/a and 1.8 Mm^3/a respectively. The total utilisable groundwater exploitation potential under normal conditions is 15.2 Mm^3/a so it is important to try and reduce the impact of this nett abstraction by using Managed Aquifer Recharge (MAR) methodologies and it is 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 (Conrad and Naicker 2019). Comprehensive groundwater monitoring within the entire region is essential for the long-term management and preservation of the aquifers. Within the Greater Saldanha Bay area it is critical to ensure all groundwater abstraction above the General Authorisation limit is authorised and that the associated compliance conditions are met.

5.4 The importance of groundwater for Langebaan Lagoon

Groundwater plays a very significant role in sustaining marsh ecosystems surrounding Langebaan Lagoon (Valiela *et al.* 1990; Burnett et al. 2001). Diagnostic plants, such as *Phragmites australis* and *Juncus kraussi*, indicate significant contributions of groundwater (Adams & Bates, 1999). These communities, found in and around the lagoon, are not tolerant of sea-water salinity levels for more than a few weeks, yet have existed there for decades, despite the anticipated evaporative increase in salinity expected in the southern part of the lagoon and the low rainfall which is insufficient to maintain the reed beds. Thus, the Langebaan Lagoon, despite not having river inflow, is thought of as an estuary surrounded by a thriving wetland at the head supported by significant subsurface inflow of freshwater.

Boreholes drilled around the edge of the lagoon as well as geophysical surveys have shown a significant inland hydraulic freshwater head intruding into the lagoon (Saayman *et al*. 2004). The borehole drilling information is detailed in a report prepared by the CSIR (Weaver et al. 1998). The authors collated borehole information, yield tests and borehole construction information to be able to determine the flow rates per geological formation. Notably, the flow rates are high within the calcrete zone, although calcrete is typically a low yielding geological formation (CSIR BH3 4.3L/s). Also, of relevance are the very high flow rates of the shallow sands near to the lagoon edge (BH1 12 L/s and BH2 11L/s). The flow rates are much higher than regional estimates. The distribution of the CSIR borehole sites relative to Langebaan Lagoon and Elandsfontein are depicted below (Figure 5.8).

Figure 5.8 CSIR bore holes (BH 1 – 4) in relation to Langebaan Lagoon and Elandsfontein (depicted by white polygon).

The south-west corner of the proposed pit at Elandsfontein is 12 km away from Borehole 1 at Geelbek. To try and understand the groundwater flow dynamics at Geelbek, a schematic of the geological cross-section was drawn, initially just with borehole details as metres below ground level (i.e. keeping the ground surface level) (Weaver et al. 1998). Notably there were high groundwater flow rates within the different portions of the cross-section and the presence of clay beneath the lagoon. Of greater relevance, however, is the geological cross-section that is corrected for elevation differences (Figure 5.9). It is clear from this cross section that there are several factors that result in the higher than expected groundwater outflow in the Geelbek area. These are:

- The steep hydraulic gradient towards the lagoon from inland;
- The high flow rate (hydraulic conductivity of the calcrete) – permitting the flow of groundwater;
- The very high groundwater flow rate of the shallow sands near to the lagoon; and
- The presence of the clay beneath the lagoon, which will force the deeper groundwater flow upwards towards the lagoon.

Concern exists that any groundwater abstraction and recharge may affect this flow of fresh groundwater into the lagoon, especially when viewing the bedrock topography of the Elandsfontein aquifer. Woodford and Fortuin (2003) assessed the groundwater recharge of the area and found that high recharge occurred to the east of the lagoon in the Rietfontein area (Figure 5.6). In addition, Woodford and Fortuin (2003) assessed the groundwater flow directions in the area and found that the flow into the lagoon is directly from the east of Geelbek (and not from the north-east where the Elandsfontein site is located). Another issue is whether the groundwater flow into Geelbek is from the Upper Aquifer Unit or Lower Aquifer Unit. From the geological section (Figure 5.9), it's evident that the groundwater flow occurs in the Upper Aquifer Unit.

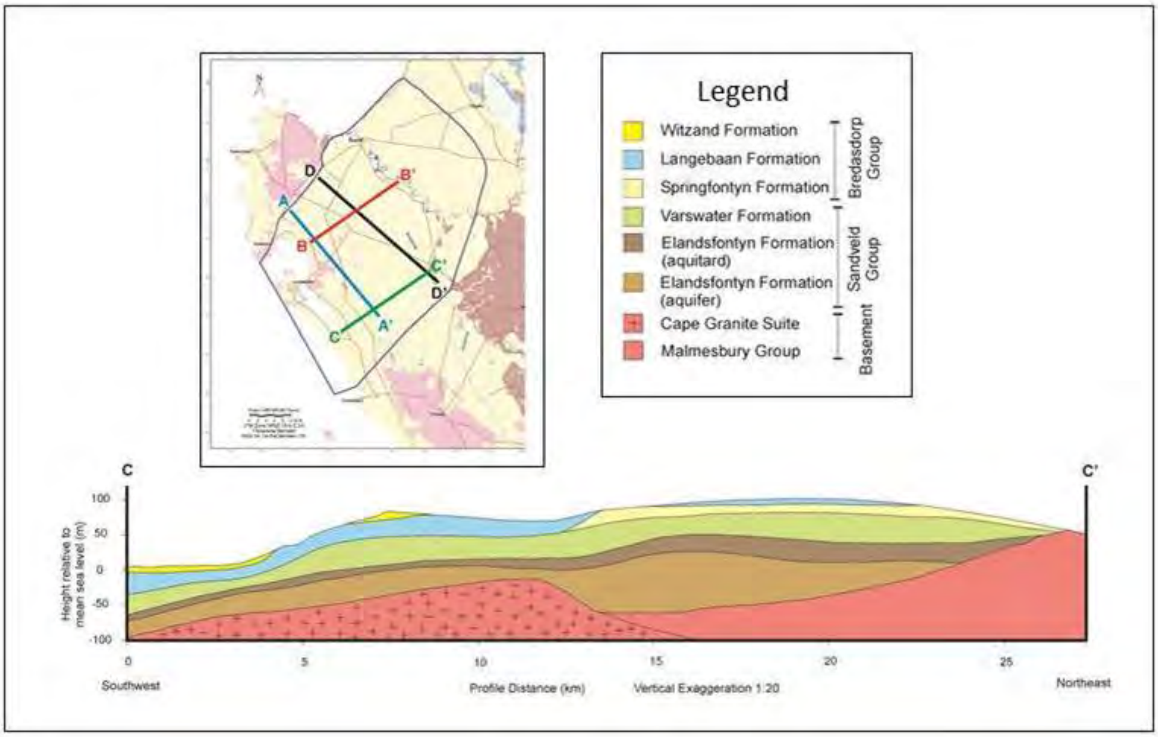


Figure 5.9 **Geological cross-section (C-C' in green) beneath the Geelbek area (DWAF 2008).**

Seyler *et al*. (2016) have attempted to estimate groundwater flow to Langebaan Lagoon using a 3D groundwater flow model SPRING, developed by delta h Ingenieurgesellschaft mbH, Germany (König 2011). They modelled a dynamic equilibrium base case prior to abstraction being initiated by the West Coast District Municipality (WCDM) from the Langebaan Road Aquifer System and the Langebaan Road wellfields, in November 1999, and a series of steady state scenarios (n = 5) designed to replicate future states of dynamic equilibrium under a range of specified abstraction regimes at the WCDM wellfield (Table 5.2).

Table 5.2 **Historic and future groundwater abstraction scenarios for the West Coast District Municipality (WCDM) and the Langebaan Road wellfields. (Source: Seyler et al. 2016.)**

Scenario	WCDM wellfield abstraction (million m3/a)	Dispersed abstraction (million m3/a)
Base case	0	4.94
Scenario 1	1.35	6.53
Scenario 2	3.5	6.53
Scenario 3	5.5	6.53
Scenario 4	7	6.53
Scenario 5	12	6.53

The volume of water abstracted from the aquifer increased from around 4.94 million Mm^3/a under the base case scenario to a combined 18.53 million Mm^3/a under Scenario 5 (Table 5.3). Impacts of these increases in abstraction on the depth of the water table for the UAU (Upper Aquifer Unit) and LAU (Lower Aquifer Unit) near the lagoon edge and outflow rates to the lagoon from each of these aquifer systems are presented in Table 5.3 and Table 5.4. Net outflow to the lagoon from the UAU and LAU barely changes under the various scenarios, dropping from around 5.7 Mm^3/a under the base case to around 5.5 Mm^3/a under Scenario 5. The model predicts no change in the water level of the UAU between the base case and the most extreme abstraction scenario modelled, and a very modest change in the water level for the LAU: <0.1 m at the waters' edge, increasing to 0.1-0.5 m, 500 m from the water's edge for Scenario 5. Thus, while the base case scenario incorporates abstraction of some 4.94 Mm^3/a from the Langebaan Road wellfield, it is likely that this corresponds closely with the reference condition.

Table 5.3 Modelled change in water level in the UAU and LAU in the vicinity of Langebaan Lagoon under different abstraction scenarios (Source Seyler et al. 2016).

	Base case	Scenario 1	Scenario 2	Scenario 3	Scenario 4	Scenario 5
Drawdown at Langebaan Lagoon LAU (m)	n/a	<0.1	<0.1	<0.1, increasing to 0.1-0.5 ~680m from water	<0.1, increasing to 0.1-0.5 ~500m from water	<0.1, increasing to 0.1-0.5 500m from water
Drawdown at Langebaan Lagoon UAU (m)	n/a	<0.1	<0.1	<0.1	<0.1	<0.1

Table 5.4 Modelled groundwater flow results for base case and future scenarios. See Table 5.2 for details on scenarios (Source Seyler *et al.* 2016).

Aquifer Flux	Base case	Scenario 1		Scenario 2		Scenario 3		Scenario 4		Scenario 5	
	Mm^3/a	Mm^3/a	% Change	Mm^3/a	% Change	Mm^3/a	% Change	$milMm^3/a$	% Change	$miMm^3/a$	% Change
Langebaan Lagoon UAU net	-0.6	-0.6	0%	-0.6	-1%	-0.6	-1%	-0.6	-1%	-0.6	-2%
Langebaan Lagoon LAU net	-5.1	-5.1	-1%	-5	-2%	-5	-3%	-5	-3%	-4.9	-4%
Langebaan Lagoon net	-5.7	-5.7	-1	-5.6	-3	-5.6	-4	-5.6	-4	-5.5	-6%

5.5 Current situation

The Department of Water and Sanitation (DWS) is monitoring about 200 boreholes in the West Coast Aquifer System. The West Coast Aquifer System is also currently the subject of a WRC project that is meant to support DWS with the management of the groundwater and the Saldanha Bay Local Municipality with water supply options. The general trend in the water levels in the area is downwards for both the upper and lower layers of the Langebaan Road and Elandsfontein Aquifer Units. This may be linked to lower rainfall in the area – a trend that goes further back than the most recent drought and give an indication of the predicted drier conditions on the West Coast as a result of climate change. Other possibilities also exist but will need additional analysis of existing data and more comprehensive research into the systems that recharge the area, the connectivity of the two aquifer systems and the lag time in recharge rates for each referred to above.

Time series data on water levels for one of these boreholes (G33327, located near the West Coast Fossil Park) is shown on Figure 5.10 This borehole was drilled in 1984 into the lower layer (Elandsfontyn Formation) of the Langebaan Road Aquifer. It was selected for this report to give an indication of how boreholes at a distance from the wellfield react to abstraction at the wellfield. Changes in water levels from the time that the borehole was drilled until June 2018, with a data gap from 1991 to 1998, are depicted on Figure 5.10. A sharp fall in water levels is evident from early 2000 (the wellfield began operations in December 1999). Some recovery took place in 2004, when the pumps at the wellfield had to go in for repairs and the wellfield was not operational for 3 months. However, water levels drop after the wellfield goes into operation again, falling lower than the level before the pumps went for repairs, even though the abstraction volume was voluntarily cut by 10%. The water levels recovered slightly in 2007 as a result of the good rain that fell that year.

Additional recharge took place in 2008 and 2009 when the CSIR experimented with an Artificial Recharge (AR) pilot test at the wellfield. The water levels dropped again when the wellfield became operational after the AR Pilot Test. This drop in the water levels was reversed in April 2013 when vandals destroyed the wiring of the wellfield. It is not clear what caused the falling water levels from August 2015, as the wellfield was apparently not properly operational and was hit by additional repeated vandalism. This will have to be investigated. The trendline correlates with the falling water levels, but the slope of the trendline shows that this drop is very gradual.

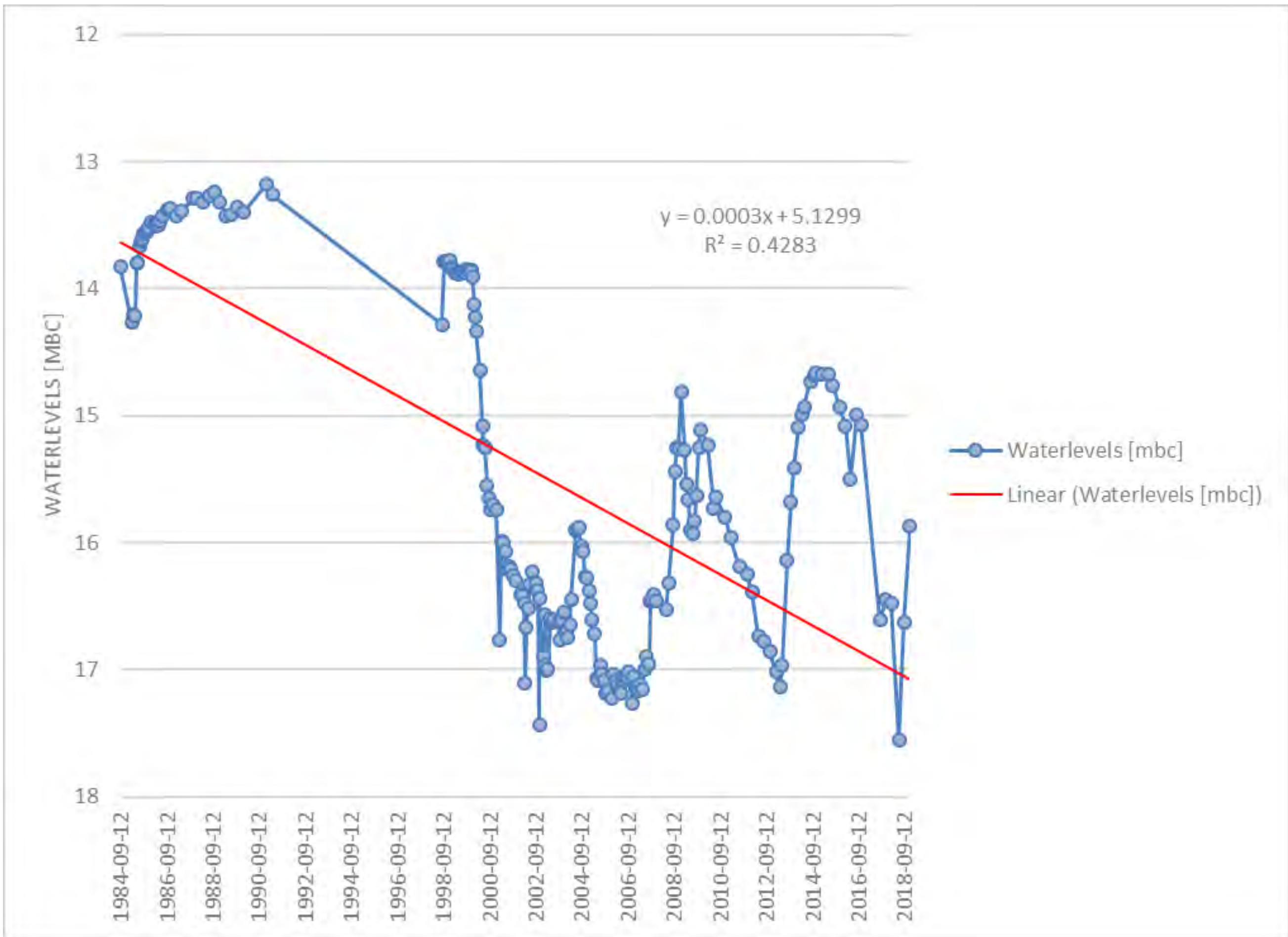


Figure 5.10 **Time series graph of water levels in the Langebaan Road aquifer as measured at borehole G33327 (Source: N. Vermaak, unpublished PhD thesis, University of the Western Cape 2019).**

Historical data from 1998 to 2018 was used to calculate percentile bands like those used for the management of dams (Figure 5.11). This is plotted in hydrological cycles, extending from the 1st of October to the 30th of September each year. Water levels for the last twelve years are plotted on top of the background. The 2012-2013 line shows is very close to the bottom of the historical record (10th percentile) indicating significant depression in water levels following several years of pumping, but that there was some recovery after this time (2013-2016) due to the vandalism of the wellfield. Water levels drop again, however, at the onset of the drought in 2017 and possible additional abstraction evident in the 2016-17 curve which drops down to the 20th percentile again. The 2017-18 curve is dipping below the 10th percentile again which is of some concern.

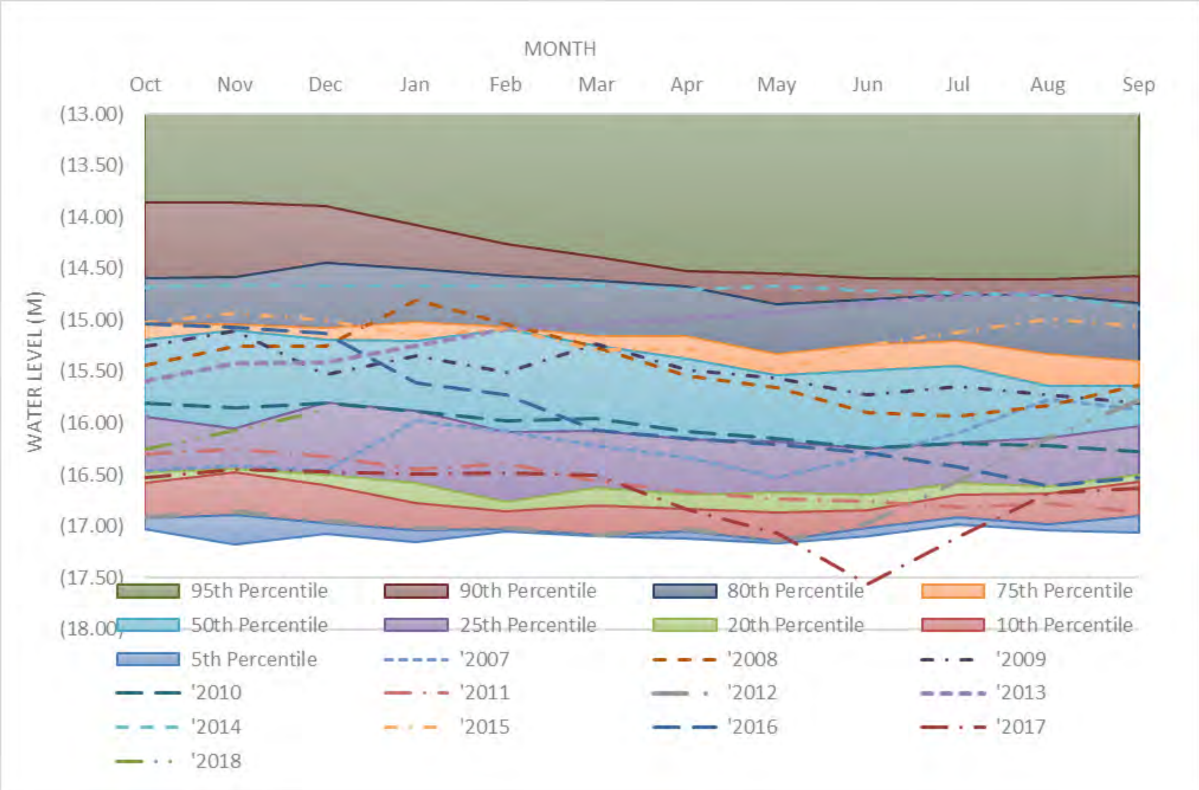


Figure 5.11 **Annual water level fluctuations for the last 12 years on the historical water level record of borehole G33327 in the Langebaan Road Aquifer from 2000 to present (Source: N. Vermaak, unpublished PhD thesis, University of the Western Cape 2019).**

A plot of water levels of borehole G33327 with the volumes pumped from the wellfield revealed a very clear picture that the abstraction at the wellfield has affected the water levels in the aquifer (Figure 5.12). It must be noted that some of the data for the abstraction at the wellfield was not available for the plotting of this graph. There is also private abstraction that has not been included, and the meter on abstraction borehole 176/1B has been removed, so it has not been possible to get any indication of the abstraction from this borehole. Under the Government Gazette that was published on the 12th of January 2018 it is now compulsory to install electronic metering devices on all boreholes to measure groundwater abstraction.

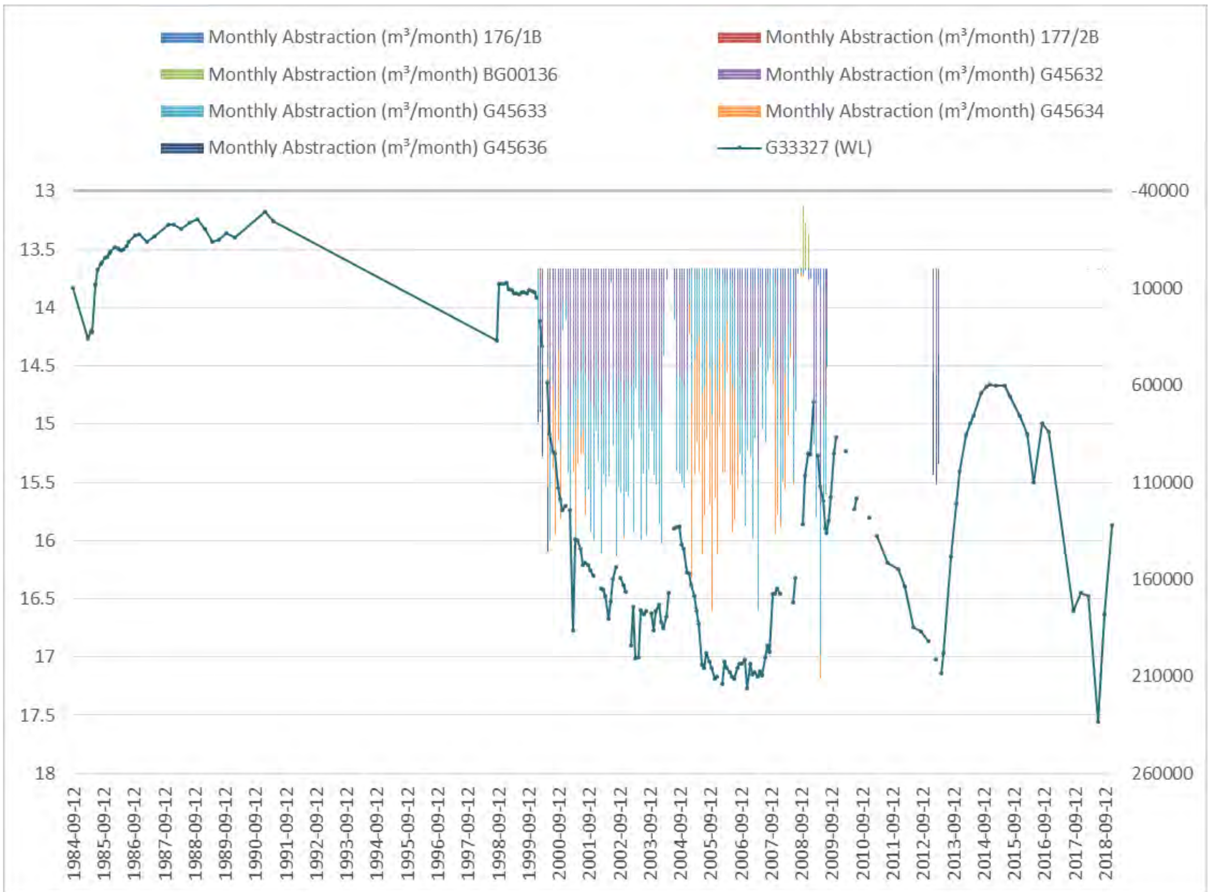


Figure 5.12 **Water levels for borehole G33327 with the abstraction record of the production boreholes at the Langebaan Road wellfield (Source: N. Vermaak, unpublished PhD thesis, University of the Western Cape 2019).**

Water levels for G33327 with rainfall as measured at some of the DWS rain gauges in the area were plotted and it is evident that there is no direct link between local rainfall and the water levels for the boreholes in the lower aquifer layer of the Langebaan Road Aquifer Unit (Figure 5.13).

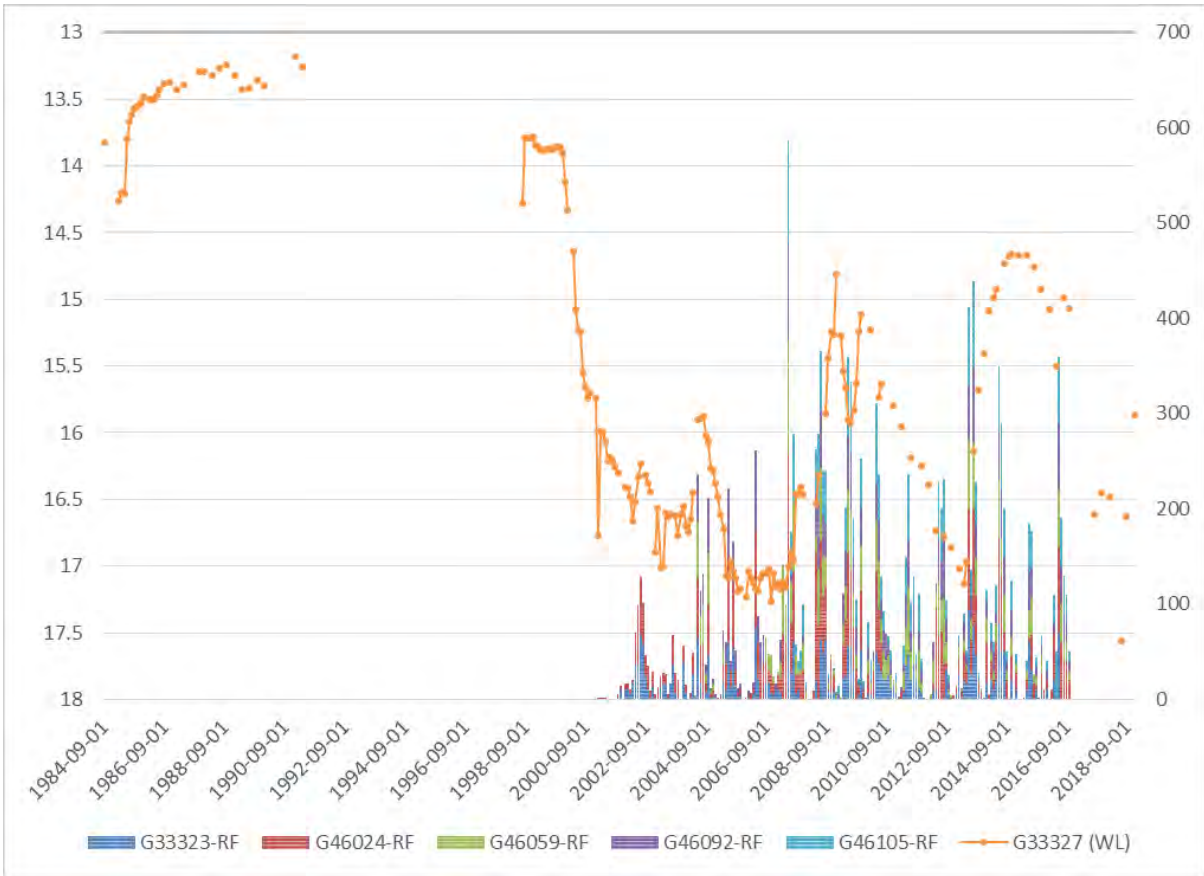


Figure 5.13 Water levels for borehole G33327 with the rainfall record as recorded by DWS rain gauges in the vicinity (Source: N. Vermaak, unpublished PhD thesis, University of the Western Cape 2019).

Water levels of borehole G33317 just south of the site of the Elandsfontein mining development are shown in Figure 5.14. These data indicate that there is a very clear link with abstractions at the Langebaan Road wellfield since around 2001. They also indicate that there is a time delay in the response to the abstraction of around 2 years. The system appears to have reached greater equilibrium over time, as the decline in the water levels slows down after 2006. It is possible that the dip in water levels in 2017 may be the result of the dewatering activities of the mine, but the levels recovered as a result of the managed aquifer recharge (MAR) taking place.

Figure 5.14 Time series graph of borehole G33317 with trendline (Source: N. Vermaak, unpublished PhD thesis, University of the Western Cape 2019).

The most recent water levels per hydrological year over a historical distribution of water levels for borehole G33317 has been plotted on Figure 5.15 below. The 2007 hydrological year was the best year for the Elandsfontein aquifer unit with water levels in the 50^{th} percentile range of the historical record. However, water levels have slowly dropped since then, with a small recovery in 2013. The 2016 line was mostly below the historical record, which shows the effects of the drought on the aquifer unit. The activities at the mine may have also contributed to the low level, with some recovery visible in the 2017 cycle. It is clear that this aquifer unit will need to be manged very carefully to prevent unacceptable harm to the system as a whole.

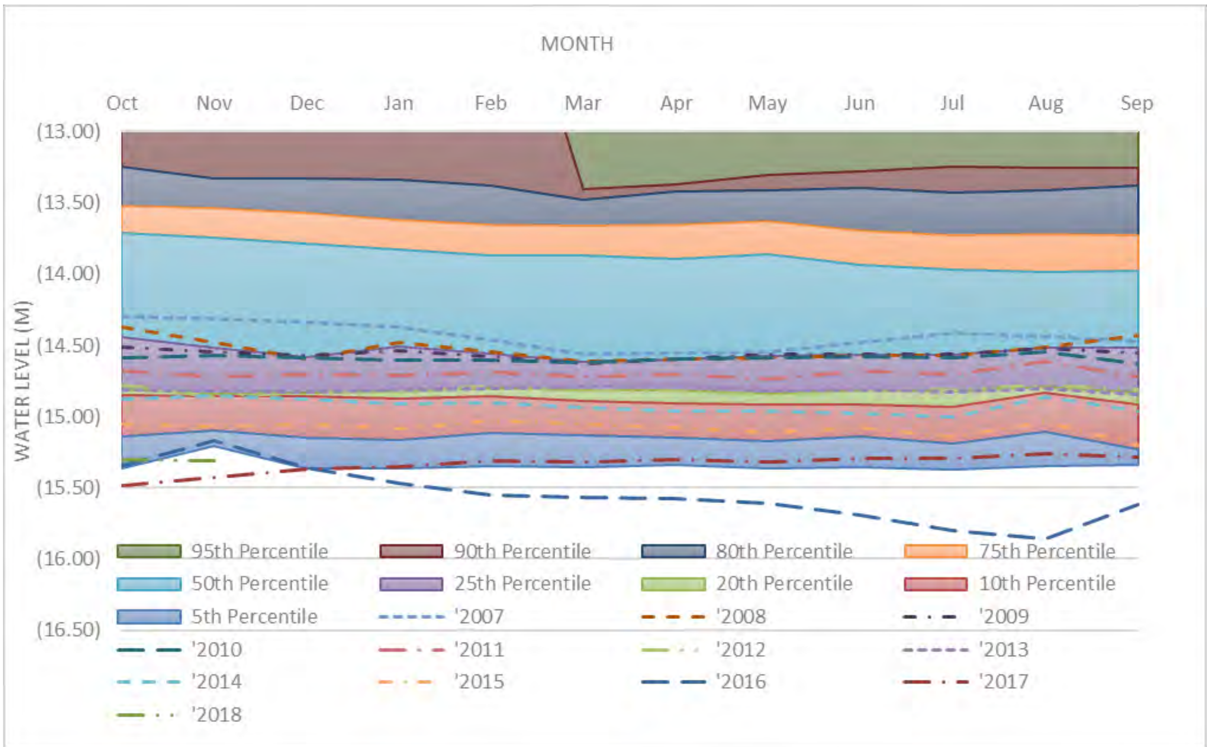


Figure 5.15 **Annual water level fluctuations for the last 12 years on the historical water level record of borehole G33317 from the year 2007 (Source: N. Vermaak, unpublished PhD thesis, University of the Western Cape 2019).**

This synopsis provided an interpretation of available data and summarized the current state of knowledge of the West Coast Aquifer System. There are, however, still critical unanswered questions particularly with respect to recharge sources and linkages between identified units. Further research is required to develop a more comprehensive understanding of the system.

5.6 Potential impacts associated with phosphate mining

The Elandsfontein phosphate deposit is 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 (Figure 5.16) (Braaf 2014). Kropz Elandsfontein intends to implement the 'roll over' mining method to access the phosphate ore, allowing the concurrent surface rehabilitation to take place. Ore will be mined at a rate of 5 Mtpa for approximately 14 years. The mining will take place in a number of discrete phases, which will reduce the overall mining footprint:

a. topsoil is removed and stockpiled;
b. overburden layer is stripped and stockpiled;
c. phosphate ore is mined; then
d. the strip is backfilled with the overburden and slimes from the plant; and
e. the topsoil returned to the strip and rehabilitated.

The targeted phosphate resource lies below the natural water table. For mining to take place, the water table is lowered by extracting groundwater from the underlying aquifer via a series of boreholes upstream of the mining site, which prevents the mine pit from being flooded. The extracted ground water is then fully recharged back into the aquifer downstream of the mining activities, via a dedicated, closed system. Processing water for the phosphate operations will largely be supplied from the Saldanha Bay Municipality. Kropz is in discussions to treat municipal effluent water for industrial reuse in order to eliminate their demand on the municipal water system.

The primary environmental impacts assessed prior to the commissioning of the mine included:

(1) the reduction of inflow of freshwater into Langebaan Lagoon causing hypersaline conditions in the lagoon resulting in negative impacts on fauna and flora sensitive to salinities in excess of normal seawater; and
(2) contamination of the groundwater as a result of the re-injection process.

A number of groundwater specialist studies have been completed to understand and mitigate these potential impacts on the aquifer (DWAF 2008, Braaf 2014, GEOSS Draft Report 2014, GEOSS Report 2017). Despite the findings of the studies, which suggest that the mining activities are highly unlikely to have any impact on the groundwater flow, residual concerns have been expressed over potential impacts that the proposed phosphate mine at Elandsfontein may have on groundwater quality and flows to Langebaan Lagoon. Kropz Elandsfontein has therefore 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 salinity and biota (benthic macrofauna) at the top of the lagoon.

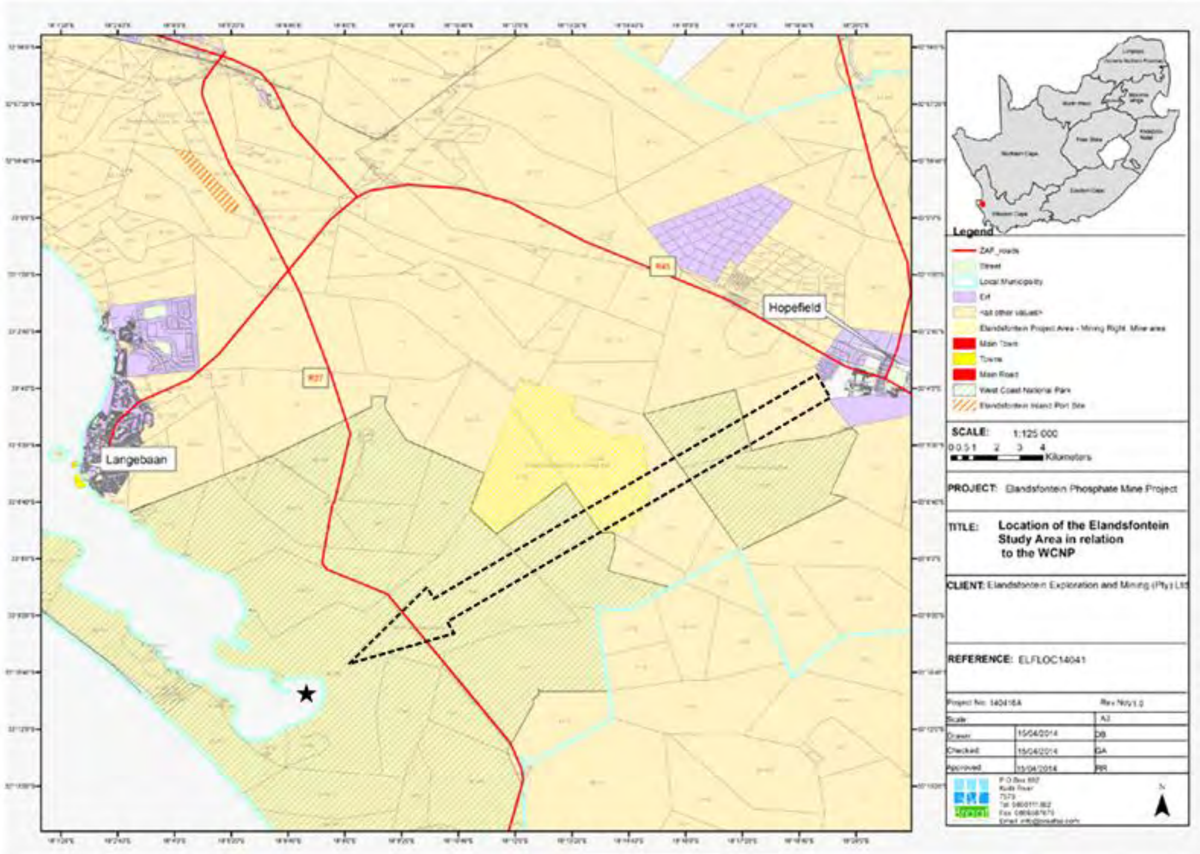


Figure 5.16 **Location of the Elandsfontein Study Area (Source: Braaf 2014). The arrow indicates direction of groundwater flow and the star location of salinity and macrofauna monitoring sites in Langebaan Lagoon.**

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 on water depth indicated a clear diurnal and bi-weekly neap-spring- tidal signal as expected but also that the instrument was exposed to the air at times on spring low tide (Figure 5.17). The tidal signal corresponded with that recorded by the South African Navy Hydrographer (SANHO) at the Port of Saldanha (Figure 5.17). The temperature signal also displayed a clear diurnal pattern, with temperatures peaking in the early afternoon and dropping to their lowest levels in the early hours of the morning as expected. Variations in tidal height seemed to have very little influence on temperature. In the initial part of the record salinity levels were fairly constant, measuring around 34.0-35.5 PSU, which is within the range for normal seawater.

However, readings soon (within the first few days) became erratic (corresponding with the first period of exposure to the air) and later dropped abruptly on at least two occasions (the first after 2 weeks and the second one month later), and remained low at around 20 PSU for the rest of the deployment period. It is believed that this was a result of exposure to the air which had introduced air bubbles into the sensor head and caused it to record inaccurately.

The instrument was cleaned, recalibrated and the mooring was moved to the deepest part of the channel, the mooring line extended, and weights added above and below the CTD to keep the instrument upright in the water and submerged at all times, regardless of tidal height. A 'check-in' retrieval and data download after one week (26 January 2017) indicated that the instrument was recording accurately at that time (Figure 5.19). Little variation in depth was evident since the instrument was now floating more or less freely on a tether, only dropping below the surface for short periods at spring high tide. Variations in temperature displayed a simple diurnal pattern as before (grey bars are included on Figure 5.19 to highlight day-night periods). Salinity displayed a low amplitude semidiurnal periodicity that seemed to be linked to tidal variation, oscillating between 35.0 and 36.0 PSU. Salinity rose as the tide receded and dropped off again when the tide turned. Normal seawater is in the range of 34.5-35.0 and this corresponds to values measured at high tide. The elevated values recorded at low tide thus, seem to be linked to saline water flowing out of the salt marshes at this location. This water would have been subjected to intense warming by the sun and loss of freshwater through evaporation is the most likely explanation for this.

The instrument was left for a further three months and retrieved again in April 2017. Data that had been collected in the preceding period was downloaded but proved to be highly erratic and showed significant "drift" (Figure 5.19). Discussion with the manufacturer resulted in the instrument being recalled and replaced. A replacement instrument was deployed in May 2017, but this was subsequently also recalled, when it was clear that this instrument was also faulty, showing significant drift in salinity levels over time (Figure 5.19).

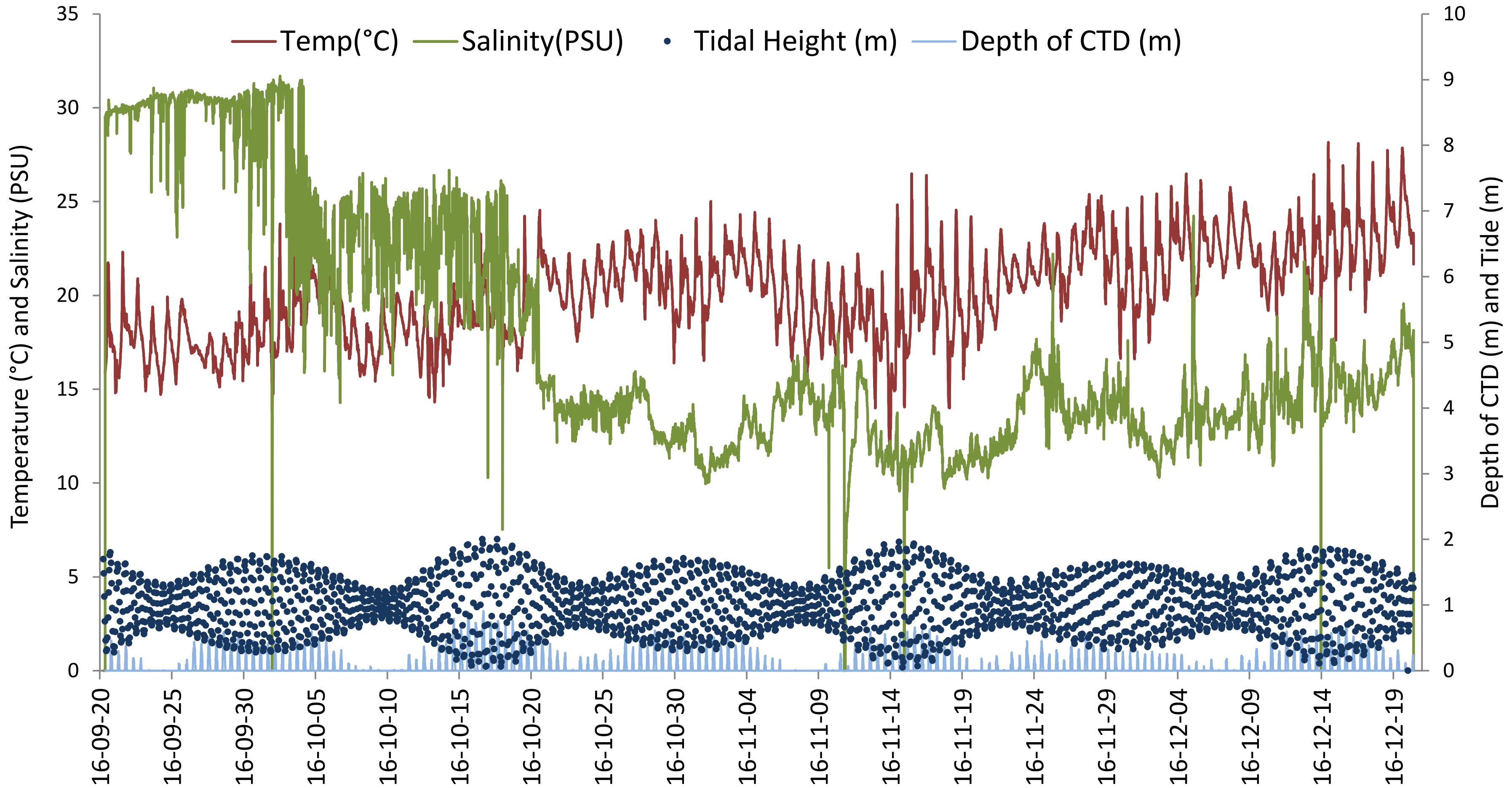


Figure 5.17 **Salinity (green line, left axis, units PSU), temperature (red line, left axis, units °C) and instrument depth (light blue) in ten minute intervals over a three month period from September through December 2016. Tidal data (dark blue points, right axis, units m) are in hourly intervals over the same time period (tidal data provided by hydrographer, SA Navy).**

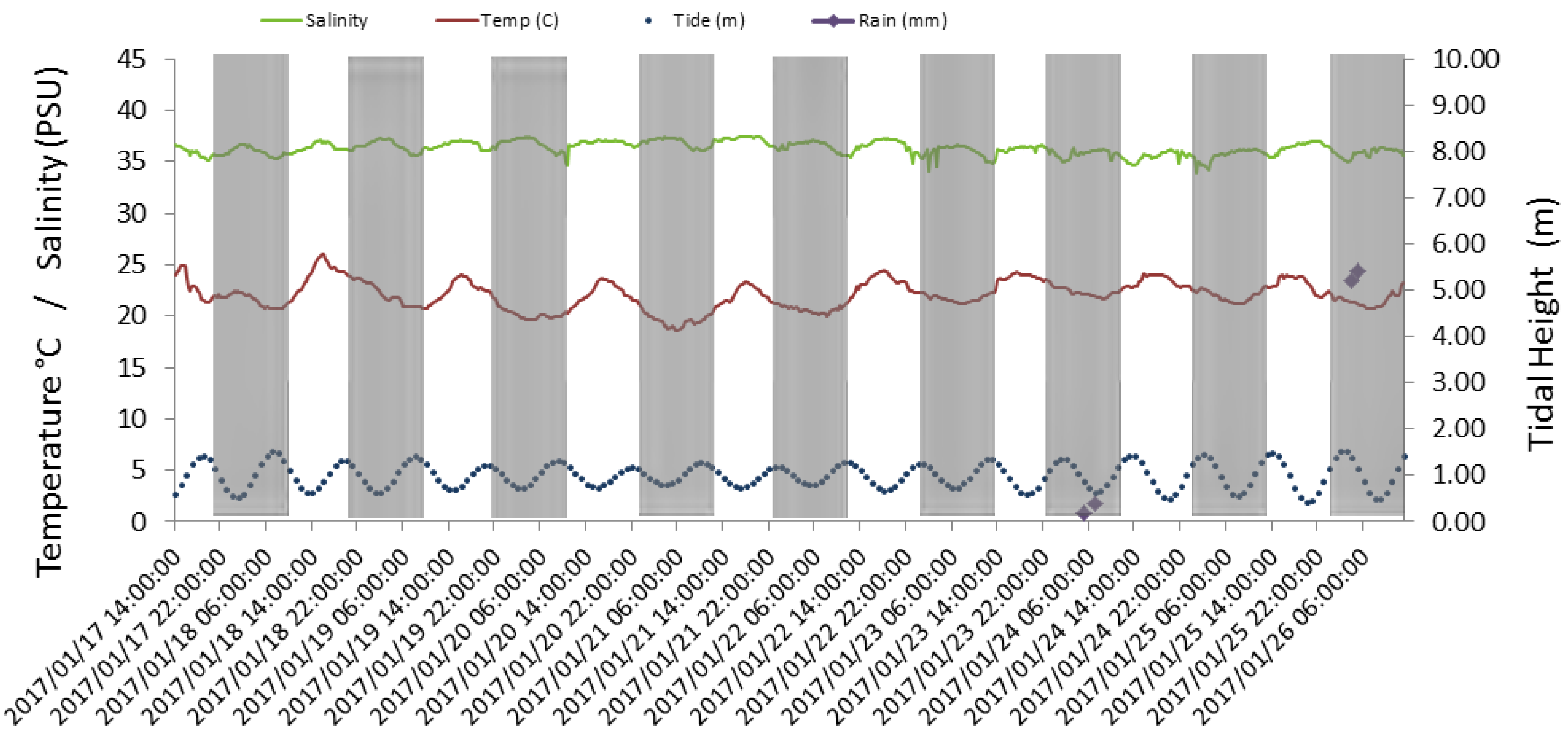


Figure 5.18 Salinity (green line, left axis, units PSU), temperature (red line, left axis, units °C) and instrument depth (light blue) in ten minute intervals over a nine day period in January 2017. Tidal data (dark blue points, right axis, units m) are in hourly intervals over the same time period (tidal data provided by hydrographer, SA Navy). Grey shaded blocks indicate day-night cycle. Rain data (purple diamonds, right axis, units mm) are in hourly intervals over the same time period (rain data provided by WeatherSA).

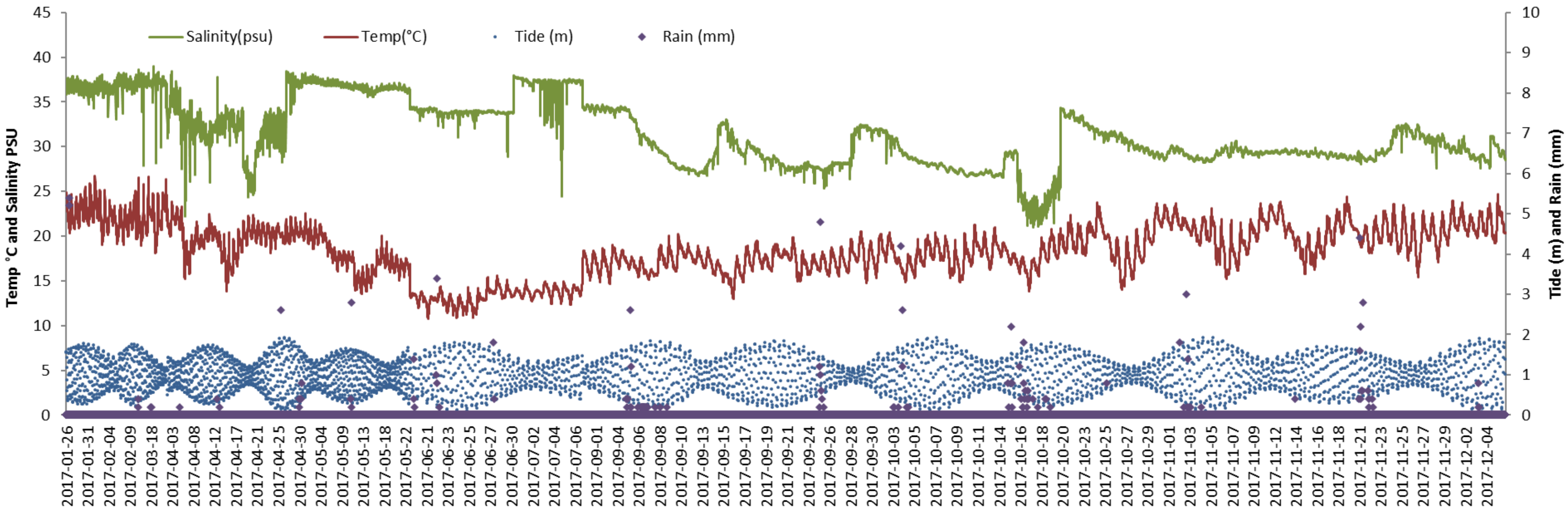


Figure 5.19 Salinity (green line, left axis, units PSU) and temperature (red line, left axis, units °C) in ten minute intervals over a one year period (2017). Tidal data (dark blue points, right axis, units m) are in hourly intervals over the same time period (tidal data provided by hydrographer, SA Navy).

Subsequent deployments (9) over the period June 2017-June 2019 produced erratic data and although temperature and depth data were realistic, the salinity data were considered largely unreliable and little correlation with tidal state or rainfall data was evident (all data collected and plotted over this period can be viewed in Appendix 1). This was thought to be due to instrument failure, exposure to air or biofouling. Due to its apparent unreliability, use of the StarOddiCTD has discontinued. This instrument has now been replaced with a new instrument from a different supplier, an Aqua Troll 200 CTD, which was deployed in July 2019.

Despite the general unreliability of the Star ODI CTD instrument, some interesting short-term variations in temperature and salinity in the lagoon were detected. These appear to be driven by semi-diurnal tidal fluctuations, diurnal (day-night) variations in air temperature and other longer-term changes that are possibly linked to changes in groundwater outflow and/or rainfall. These in turn, are confounded by biofouling episodes that affect the accuracy of the data at times. A spring high tide, represented by the maximum crest of the plotted tidal data over a two-week period, seems to be linked with a shift to sea-based salinity around 35.5 PSU. If the ambient condition in the lagoon is fresher, this will be associated with a slight decrease in salinity to that of seawater and vice versa if a hyper saline condition exists. Rain (in large volumes) may lead to a lagged decrease in salinity, which could be compounded by an outgoing tide that would draw more fresh water from the areas of ground water inflow across the CTD measuring location, or alternatively could be dampened by an incoming spring tide that is associated with a sea based 35.5 PSU as mentioned above. These scenarios could be slightly offset by seasonal and diurnal evaporation rates. For example, the highest recorded salinities (37+ PSU) coincides with the highest temperatures (25+°C) recorded in early December 2017 (Figure 5.18) and mid-February 2018 (Figure 16.2, Appendix 1). In addition to these compounding factors is the fact that the CTD became biofouled which sent the salinity reading into an erratic state with a rapid decline in salinity until the sensor was cleaned and re-set (Figure 15.1 – Figure 15.6, Appendix 1, Chapter 15).

For the new instrument (the Aqua Troll 200 CTD) the mooring configuration and location retained as described above. Data retrieved from the instrument to date suggest that measured diurnal temperature patterns and semi-diurnal tidal and salinity patterns are consistent with expectations, where temperatures increase during the day, decrease through the night, and salinity remains at or near to that of normal seawater (34.4 PSU) at high tide and freshens (32.4 PSU) with tidal outflow and consequent increase in freshwater outflow from the surrounding aquifer (Figure 5.20). Interestingly, aside from the expected salinity-tidal coupling previously described, there is again no evidence that salinity in the lagoon decreases following rainfall events which suggest these events make a very small contribution to freshwater input to the lagoon. It is considered likely that biofouling is again responsible for the abrupt downward spikes in salinity evident in the initial data record collected by the Aqua Troll 200 CTD (Figure 5.20) and steps have been taken to eliminate this. A series of biofouling experiments have been planned as part of the current deployment to better understand this issue. Also, to further explore links between salinity variations in the lagoon we will be collaborating with a research group from the University of Pretoria who are looking at groundwater, nutrient and pollutant fluxes along the west coast of South Africa, using a combination of surface ocean and groundwater measurements of radium isotopes (Humphries *et al*. pers. comm., Moore, 2010). Their first study site along the west coast is located at Geelbek and work commenced at the end of August 2019.

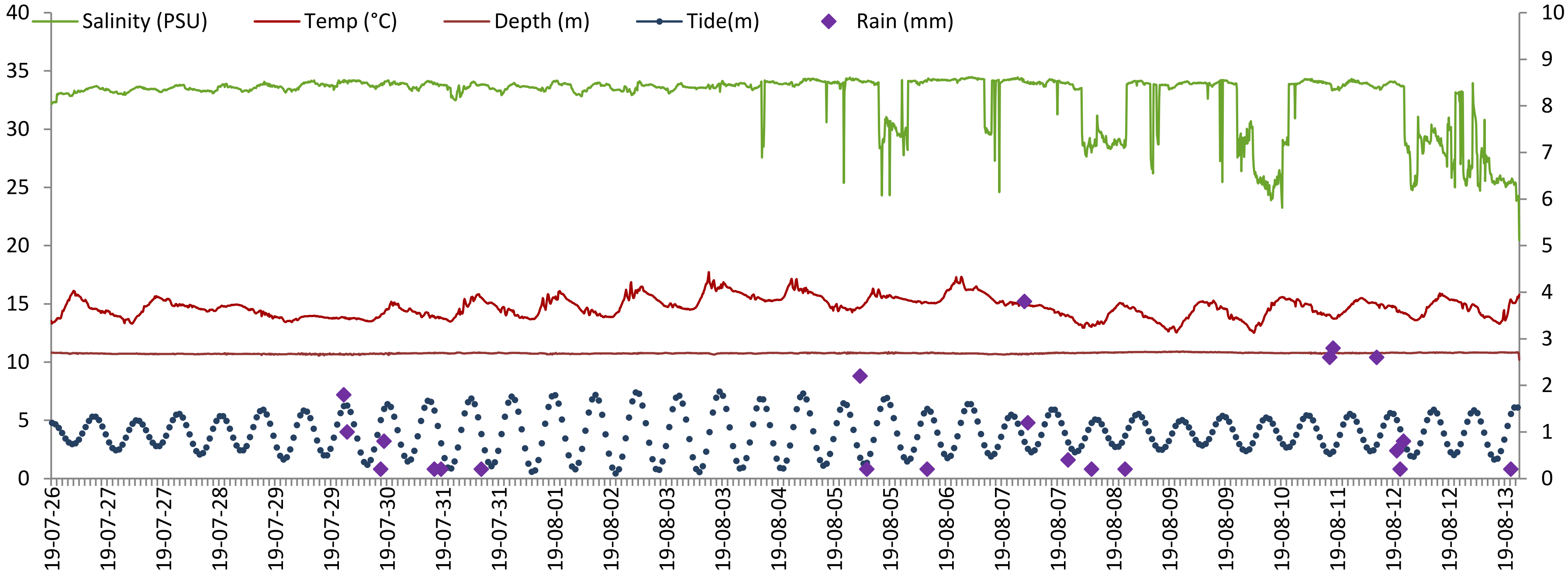


Figure 5.20 **Aqua Troll 200 CTD Salinity (green line, left axis, units PSU), temperature (red line, left axis, units °C) in ten minute intervals over a 2 month period in July 2019 through August 2019. Tidal data (dark blue points, right axis, units m) and rain (purple, right axis, units mm) are in hourly intervals over the same time period (tidal data provided by hydrographer, SANHO, rain data by WeatherSA).The red depth line was included on this plot to show the height of the newly placed instrument is consistent in the water column.**

In conclusion, the understanding of the aquifer systems and groundwater flows into Langebaan Lagoon has improved considerably in recent years. Significant knowledge gaps particularly pertaining to recharge sources and rates, aquifer unit connections and flows into surface water bodies do, however, still exist. Research into these aspects is ongoing. Monitoring of salinity in the vicinity of Geelbek, where vegetation types provide convincing evidence of groundwater input, has proved challenging. Results from three years of CTD deployments have provided erratic data with complications cause by instrument unreliability, loss of instrumentation and suspected biofouling. Clear links to tidal state and rainfall have not yet being established, but there is an indication that salinity in the upper lagoon is influenced by these factors. A new CTD instrument from a different manufacturer was deployed during June 2019 and data downloaded to date suggest that it is more reliable than the CTD previously used. It is anticipated that the drivers of salinity variation in the upper lagoon detected during previous monitoring, will be better understood with future monitoring data.

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). A trial deployment of a conductivity temperature and depth (CTD) instrument from 3 April to 13 May 2017 provided six weeks of recent data in this area. A thermistor string comprising five 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 2019 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. 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).

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). 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 weak (5-15 $cm.s^{-1}$) and tended to be clockwise (towards the NE) irrespective of the tidal state or the wind (Figure 6.1). 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 6.1).

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.

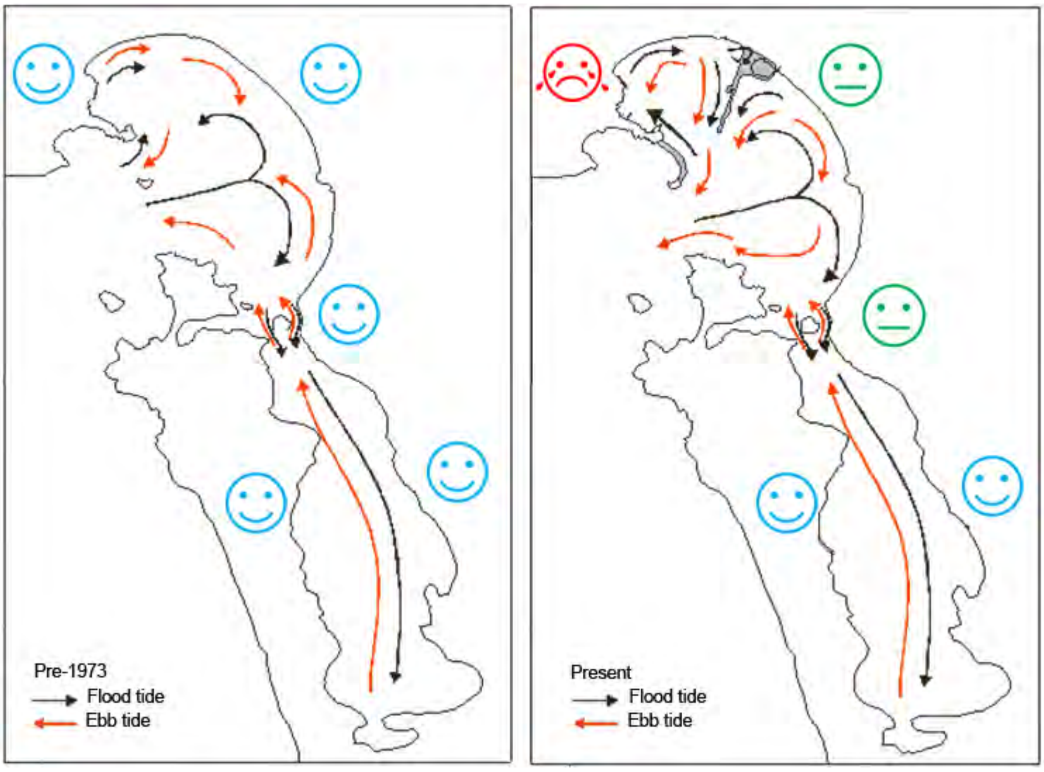


Figure 6.1 **Schematic representation of the surface currents and circulation of Saldanha Bay prior to harbour development (pre-1973) and after construction of the causeway and iron ore terminal (present) (Adapted from: Shannon & Stander 1977 and Weeks *et al*. 1991a).**

An ADCP was deployed from 7 to 10 April 2017 at Sea Harvest in Small Bay (see Figure 6.2 - 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 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 6.2 - 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 6.2 - 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 6.3). A 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 6.3). 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.

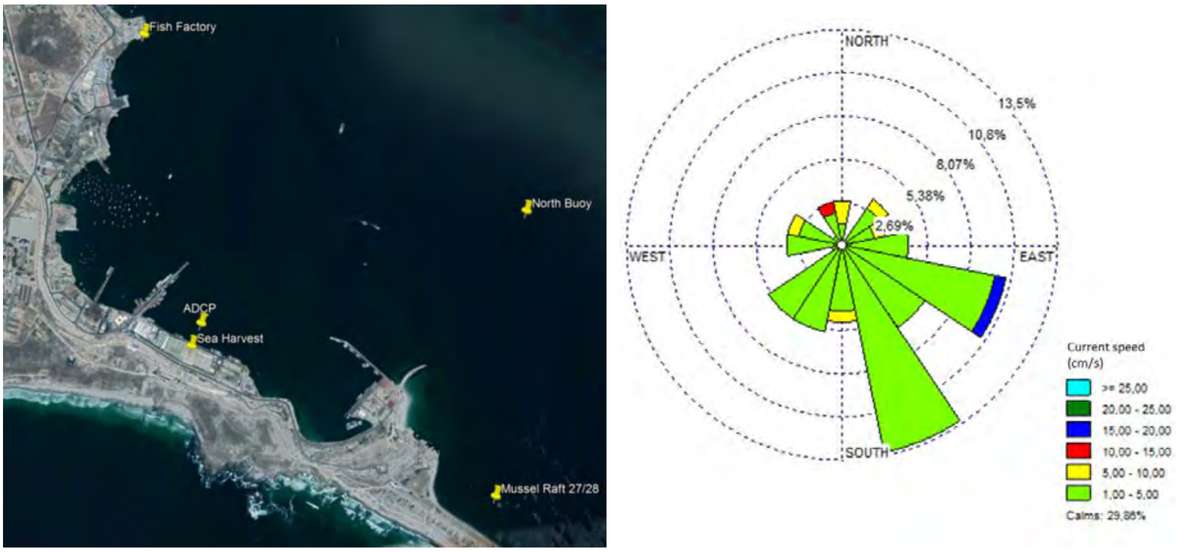


Figure 6.2. Location of the Sea Harvest ADCP (left) and current rose depicting current direction and strength at -7 m water depth (right). (Source: Wright *et al*. 2018a).

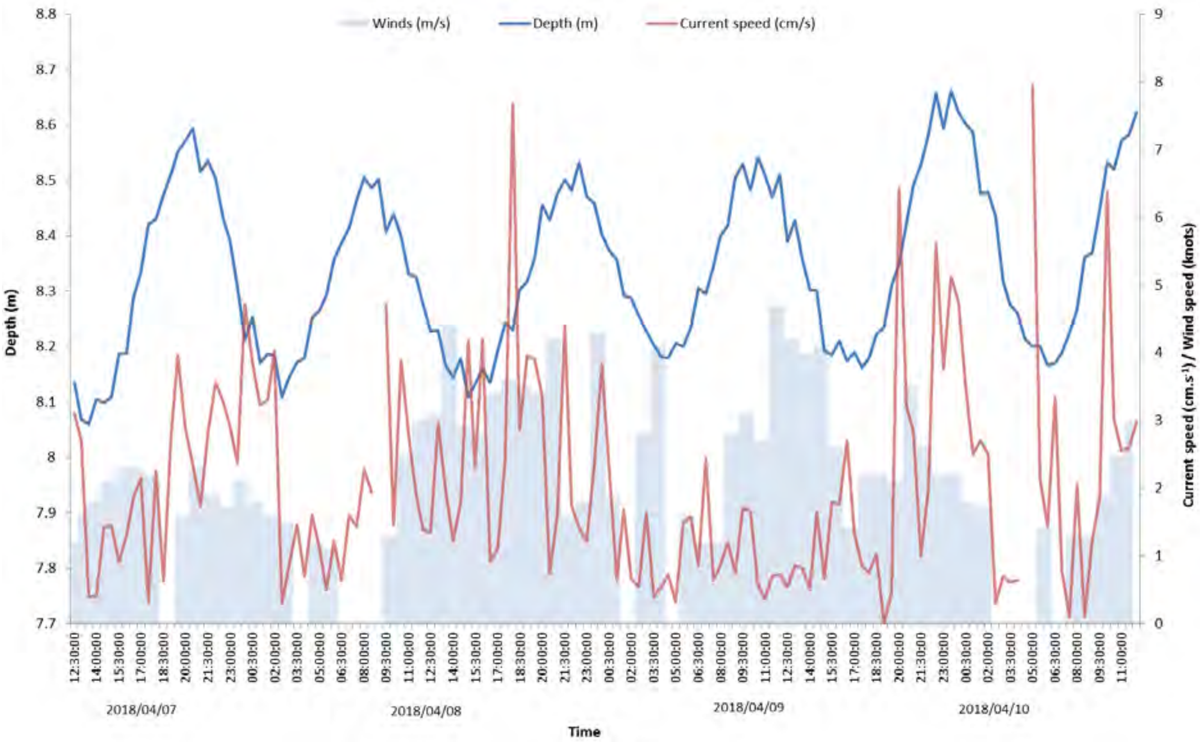


Figure 6.3. ADCP data collected every 30 minutes for depth (m), indicating the tidal cycle current speed (cm/s), and wind speed (m/s) over the four days of ADCP deployment at Sea Harvest in Small Bay in April 2018 (Source: Wright *et al*. 2018a).

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 6.4). 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 6.4). 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 6.4). 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.

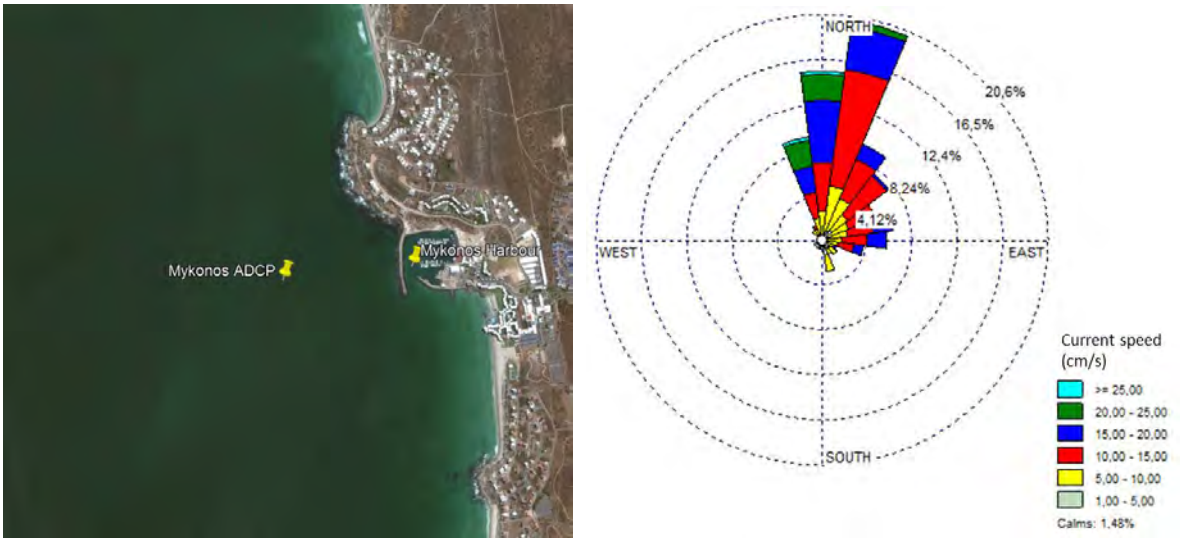


Figure 6.4. Location of the Mykonos ADCP (left) and the resulting current rose showing current direction and strength data at -8.5 m water depth (Wright *et al*. 2018b).

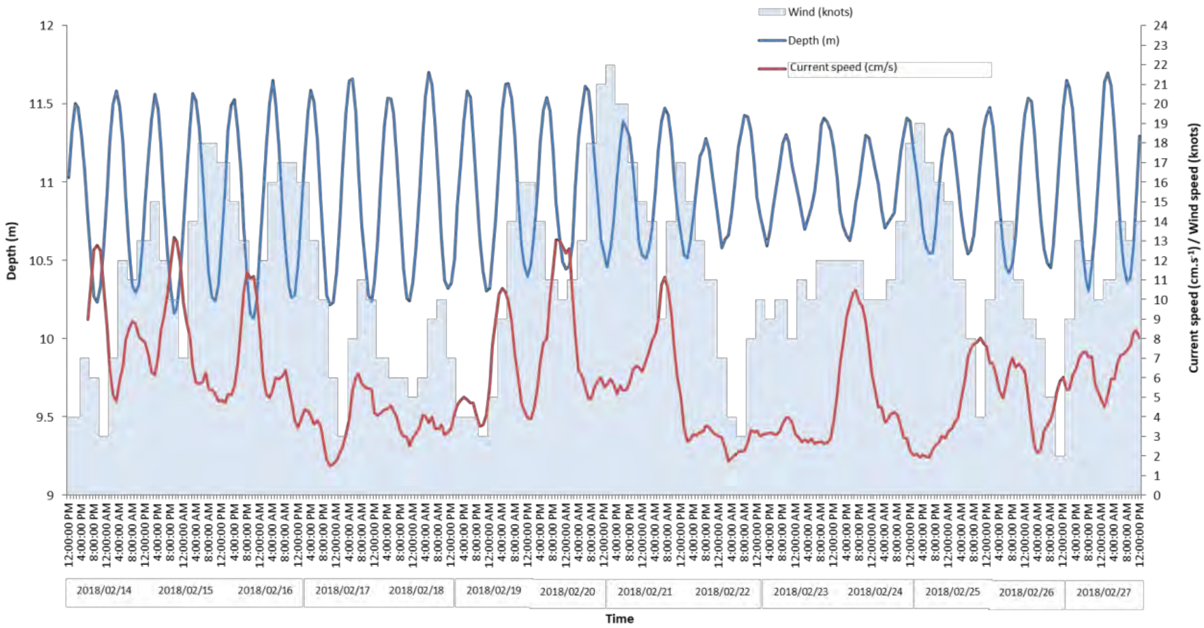


Figure 6.5. ADCP data collected every hour over a 14-day ADCP deployment period in February 2018 showing current speed (cm.s^{-1}) and wind speed (knots). Depth (m) indicates the tidal cycle (blue line). (Source: Wright *et al*. 2018b).

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 the area of 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 6.6 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 6.6 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).

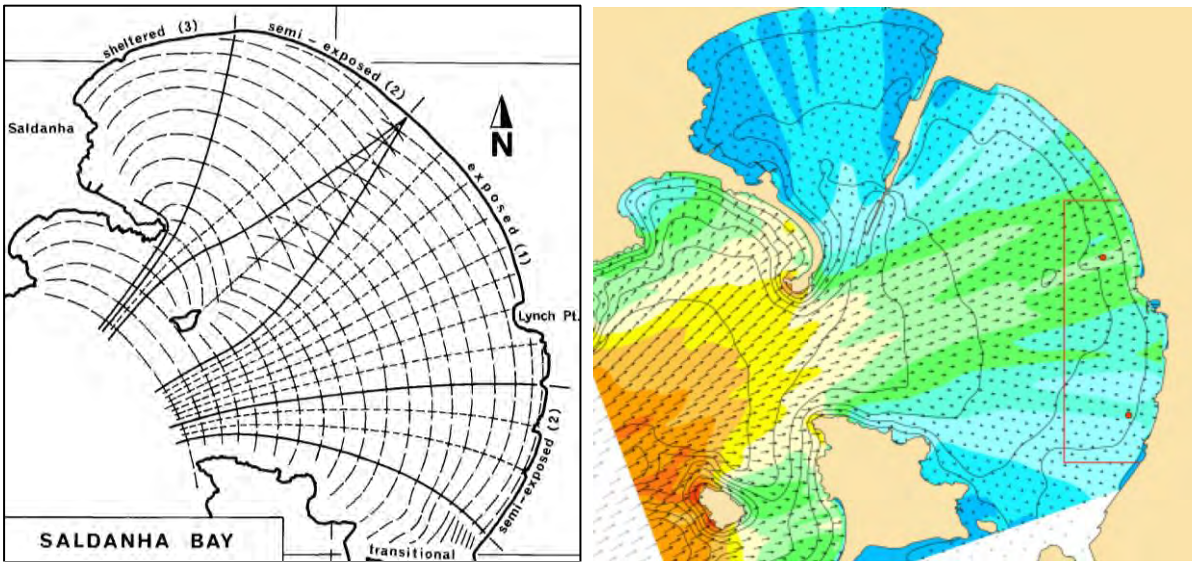


Figure 6.6 Predicted wave fields in Saldanha Bay showing wave height and direction prior to (left) and post (right) harbour development. Orange shading indicates wave heights >1.4 m, while blue shading indicates wave heights of <0.6 m (Sources: Flemming 1977 and WSP Africa Coastal Engineers 2010).

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), and now known as Department of Environmental Affairs - Oceans and Coasts (DEA-O&C). 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 6.7. 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 6.7). 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 6.7). 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 6.7).

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 6.11), 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 6.7) 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.

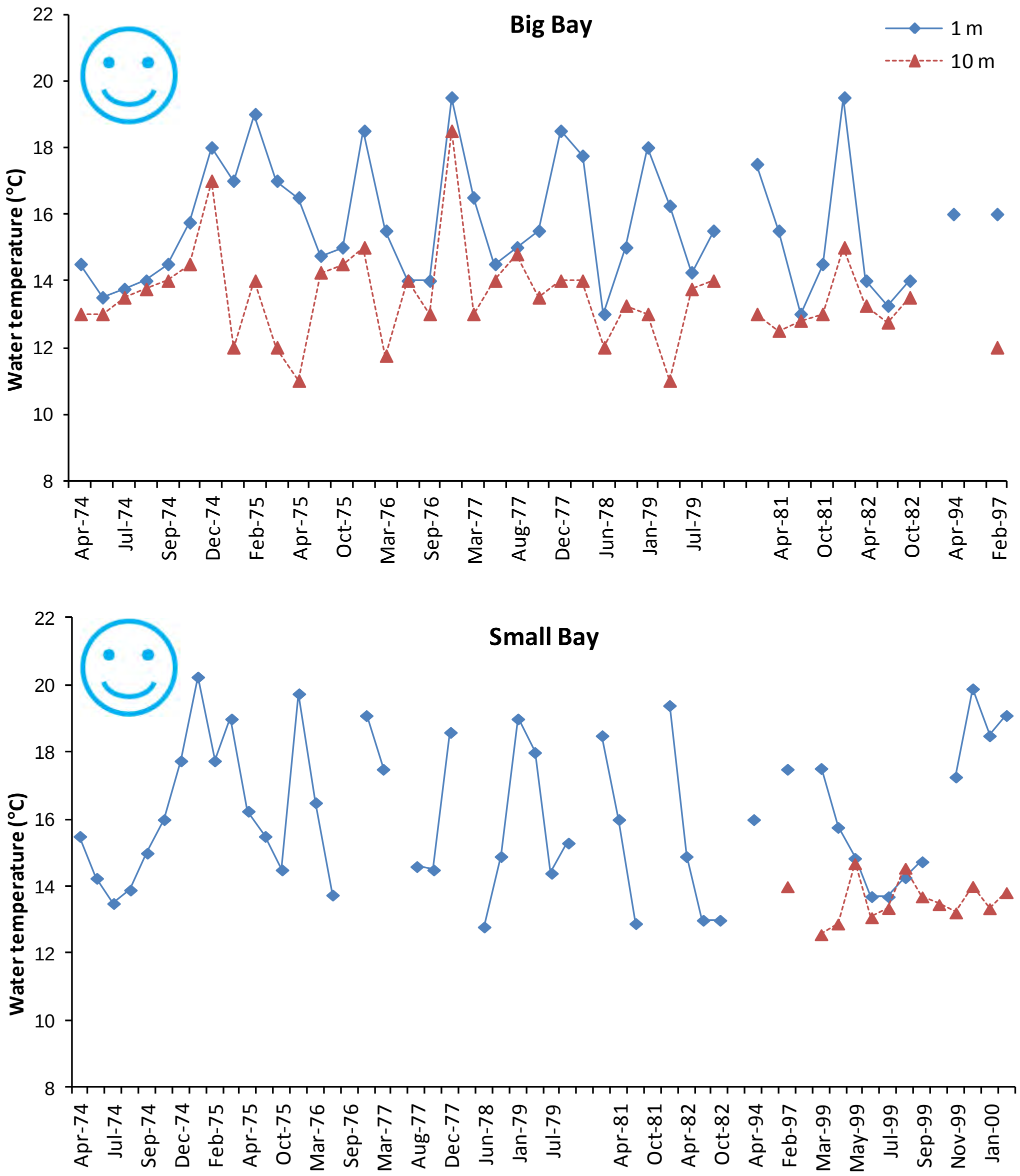


Figure 6.7 **Water temperature time series at the surface and at 10 m depth for Big Bay and Small Bay in Saldanha Bay (Data: Monteiro *et al.* 1990, Monteiro & Brundrit 1990, Monteiro *et al.* 2000 and Shannon & Stander 1977).**

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 6.7. 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 6.7). 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 6.8. 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

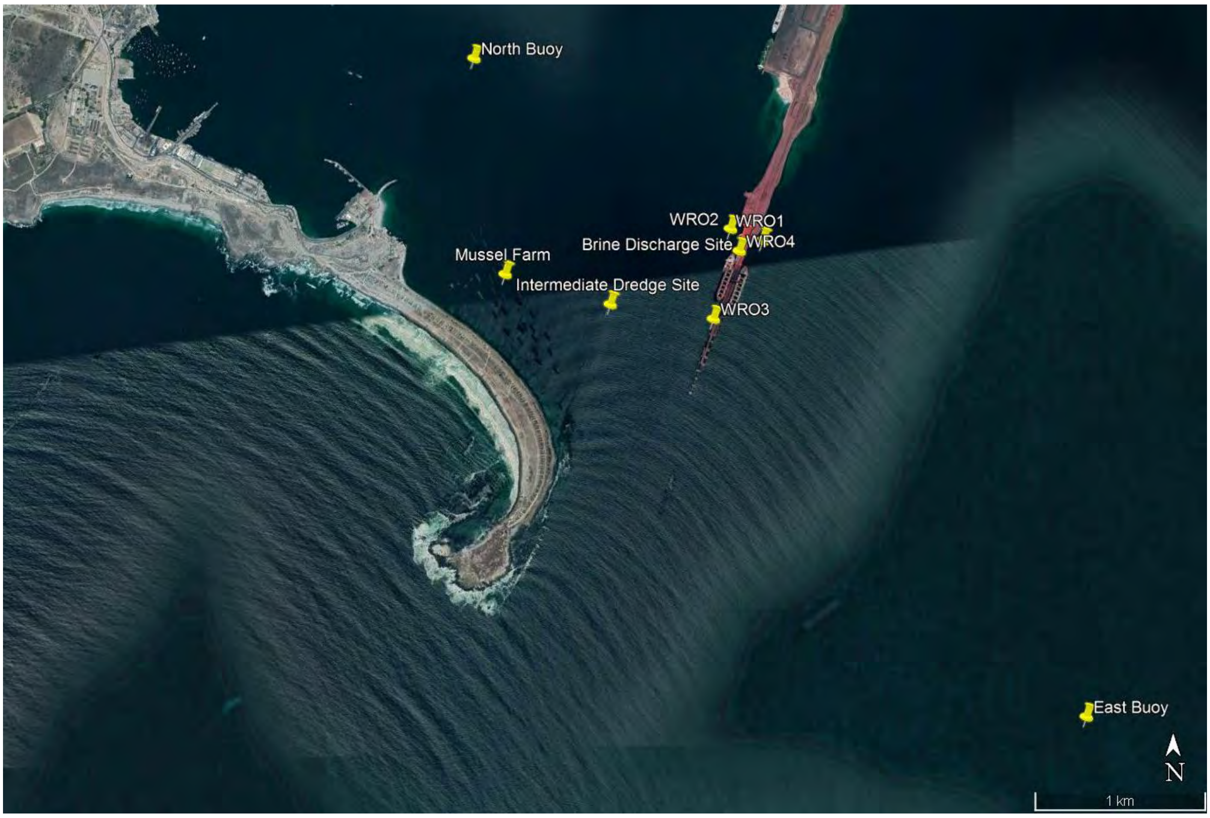


Figure 6.8 **Water quality monitoring stations adopted for the RO plant baseline survey undertaken by the CSIR (Source: van Ballegooyen *et al.* 2012).**

Examples of the temperature data from the water column profiling exercises undertaken at North Buoy are shown in Figure 6.9. 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.

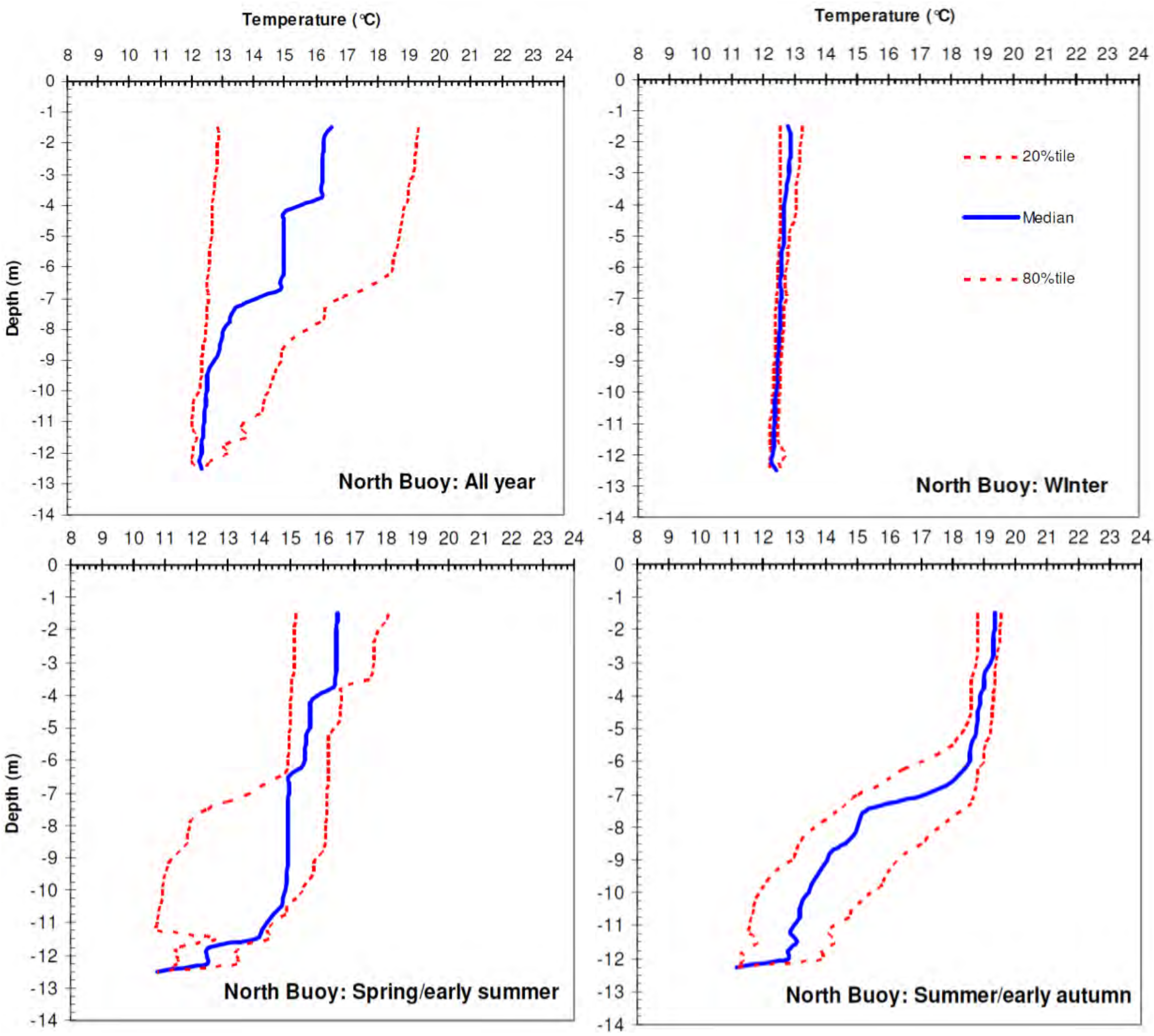


Figure 6.9 **Seawater temperature median profiles at North Buoy for all four seasons. The 20th and 80th percentile limits of the profiles are indicated by the dotted red lines (Source: van Ballegooyen *et al.* 2012).**

With a view to continuing the long-term temperature data set at North Buoy, five Vemco mini-loggers, 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 2019 are shown in Figure 6.10.

The data from 12 April 2014 to 10 April 2019 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 6.10). 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. This coincides with the extreme drought experienced in the Western Cape recently. 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. 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 2019 are, however, similar to those recorded in earlier monitoring (since 1974) (Figure 6.7). 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.

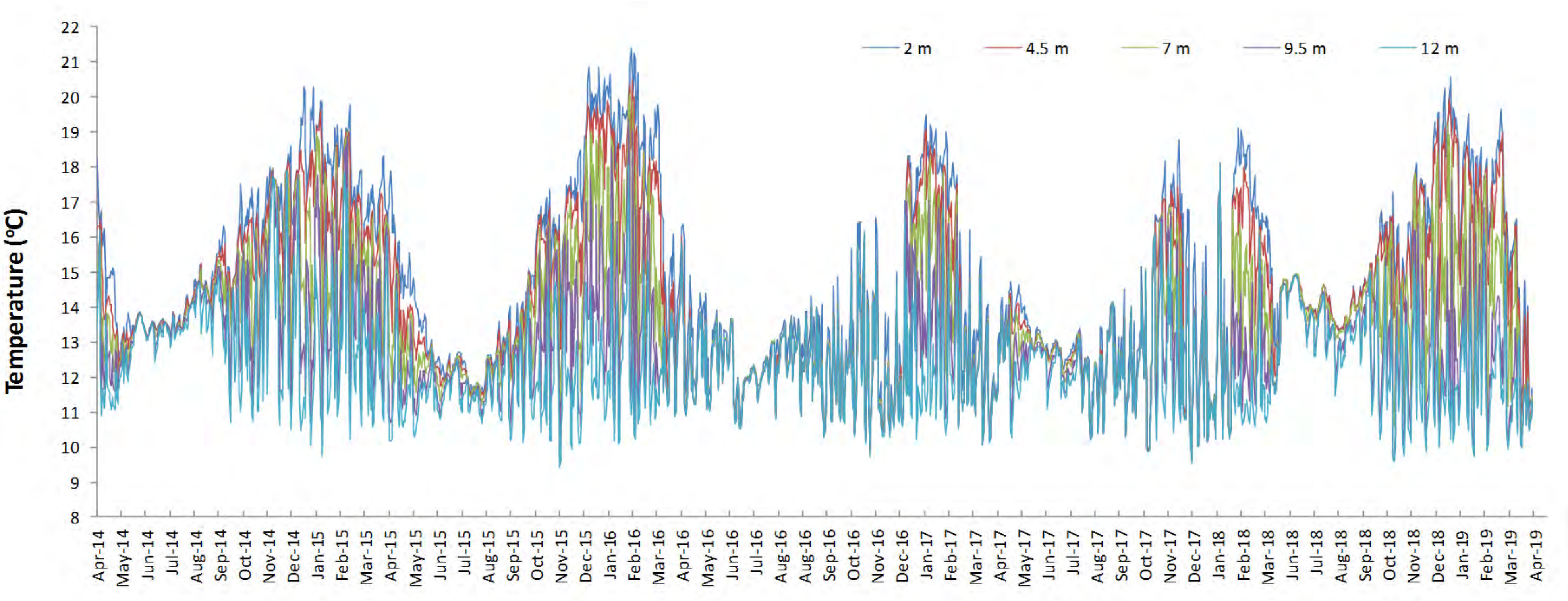


Figure 6.10 **North buoy temperature time series for the period 12 April 2014 - 11 April 2019. Temperature was recorded every hour and the average daily temperature is shown here. Note that the no data are shown for the period 10 April- 5 June 2018 whilst the instruments were out of the water.**

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 6.11. 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 6.11). 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 (Monteiro & Brundrit 1990) (Figure 6.12). 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 6.13 (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).

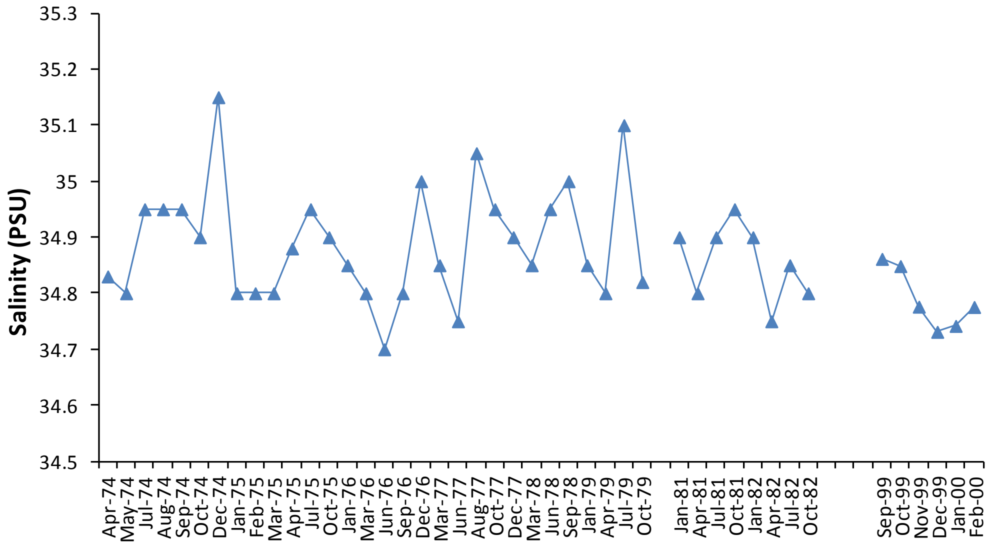
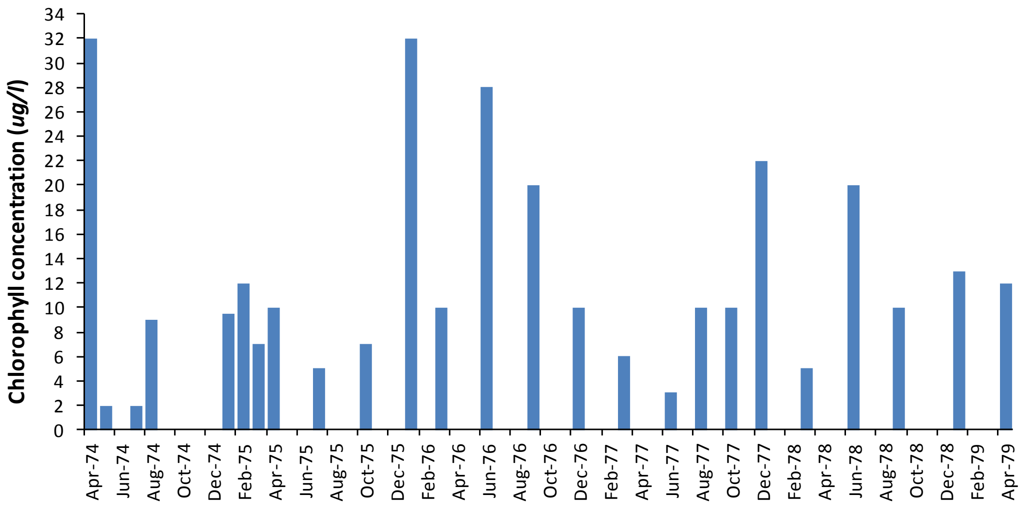


Figure 6.11 Time series of salinity records for Saldanha Bay. (Data sources: Shannon & Stander 1977, Monteiro & Brundrit 1990, Monteiro *et al*. 1990 and Monteiro *et al*. 2000).



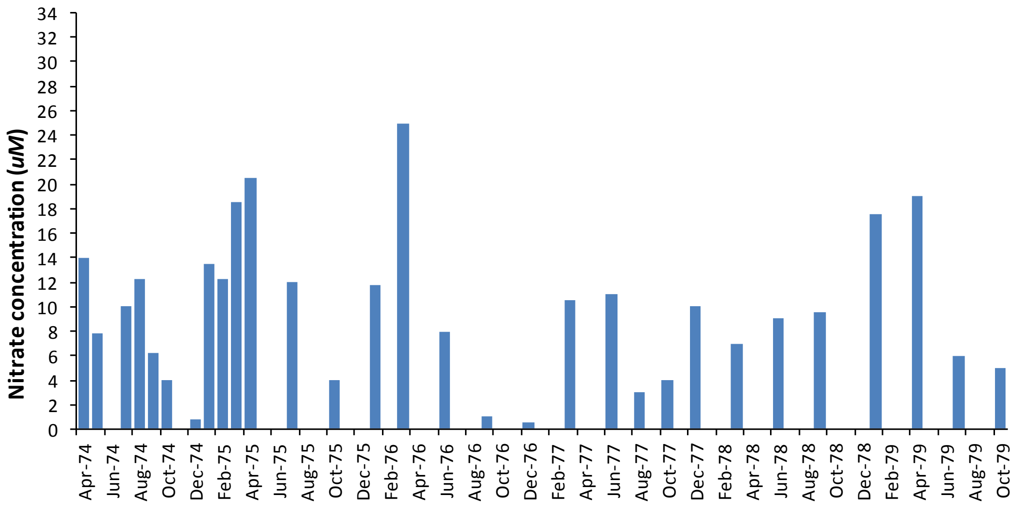


Figure 6.12 Time series of chlorophyll and nitrate concentration measurements for Saldanha Bay. (Data source: Monteiro & Brundrit 1990).

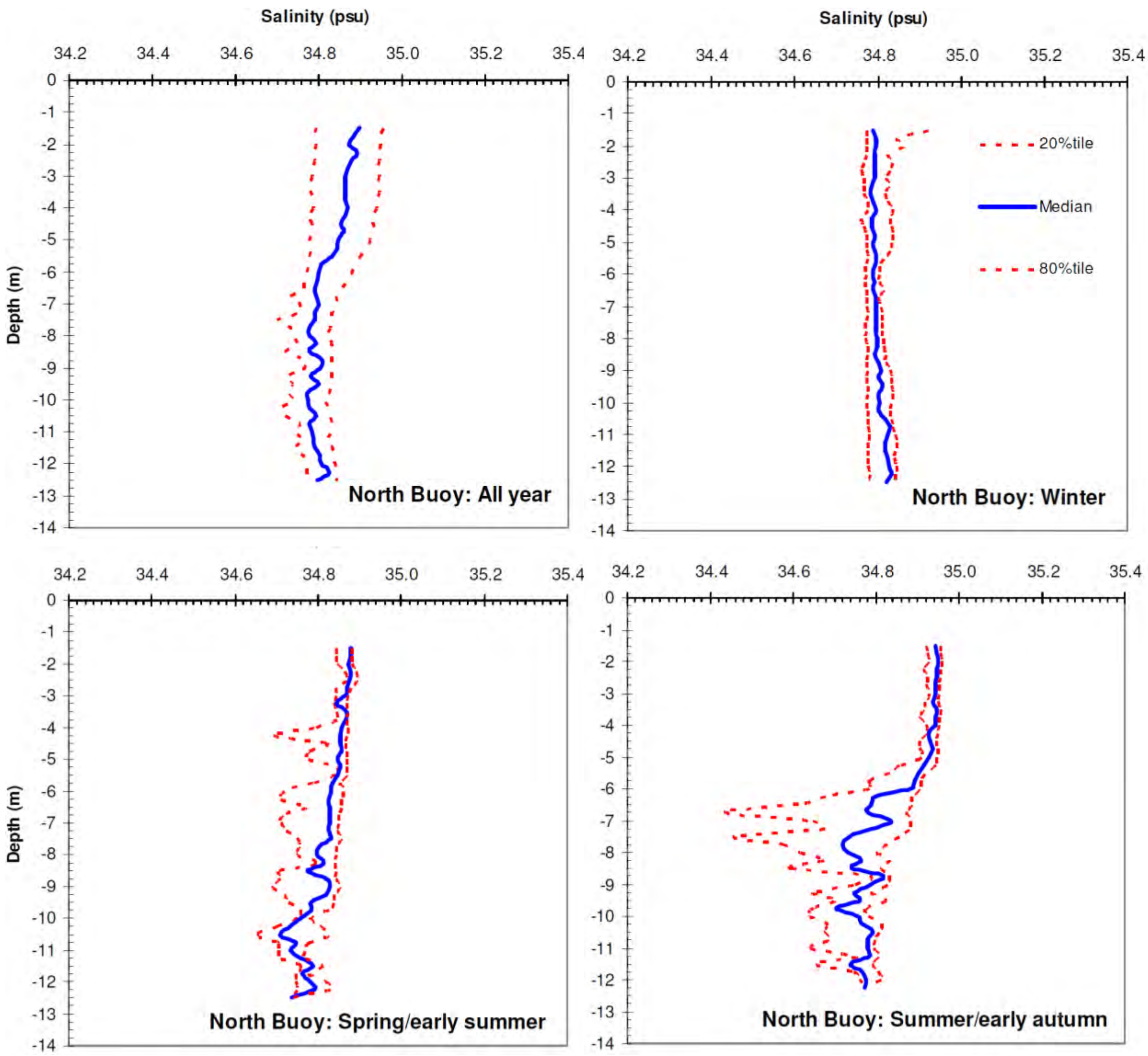
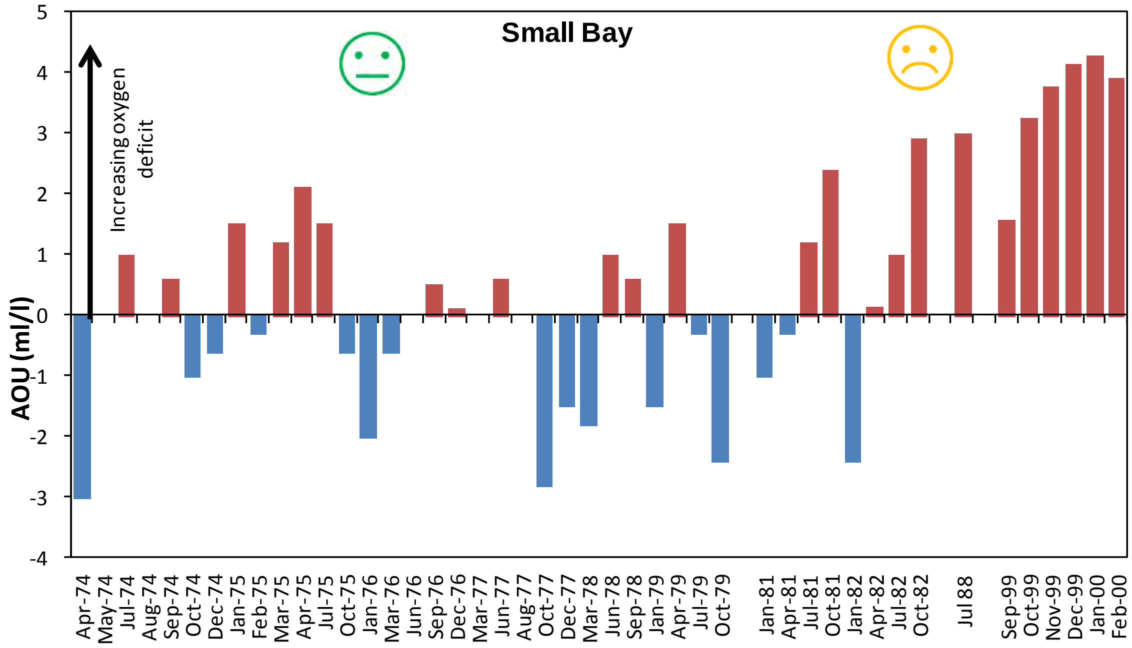


Figure 6.13 **Salinity median profiles at North Buoy in Small Bay for all seasons (winter, spring/early summer and summer/early autumn). The 20th and 80th percentile limits of the profiles are indicated by the dotted red lines. (Source: van Ballegooyen *et al.* 2012).**

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 6.14). 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 6.12.



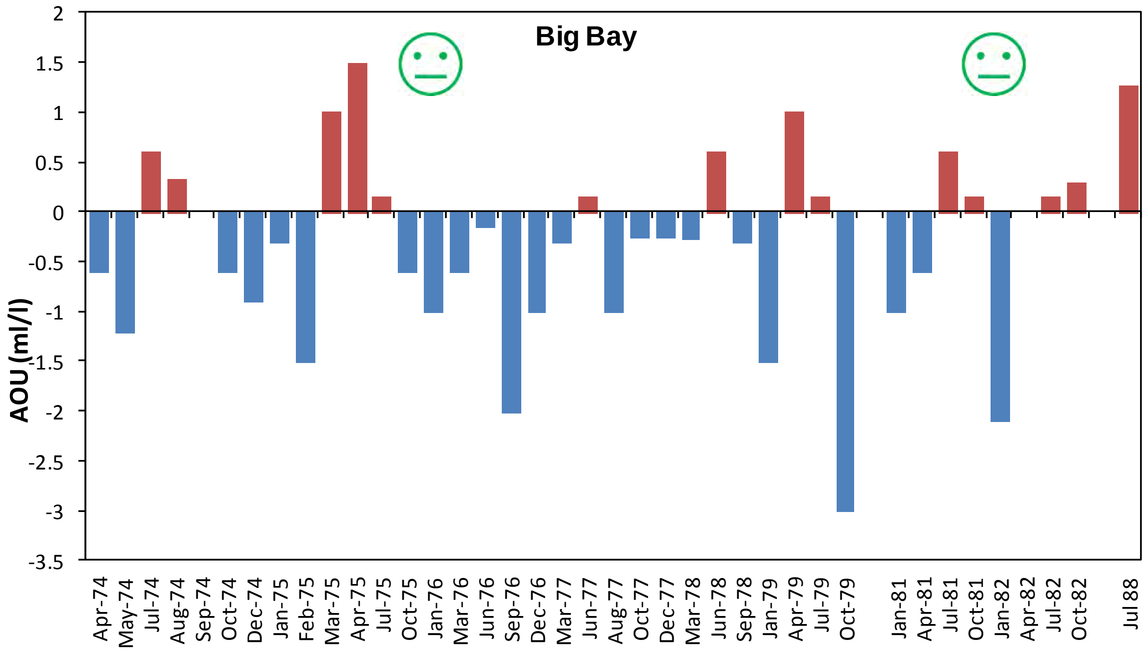


Figure 6.14 Apparent oxygen utilization time series for Small Bay and Big Bay in Saldanha Bay. Positive values in red indicate an oxygen deficit (Data sources: Monteiro *et al.* 1990 and 2000).

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 6.14). 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 6.14). 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 6.15 (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

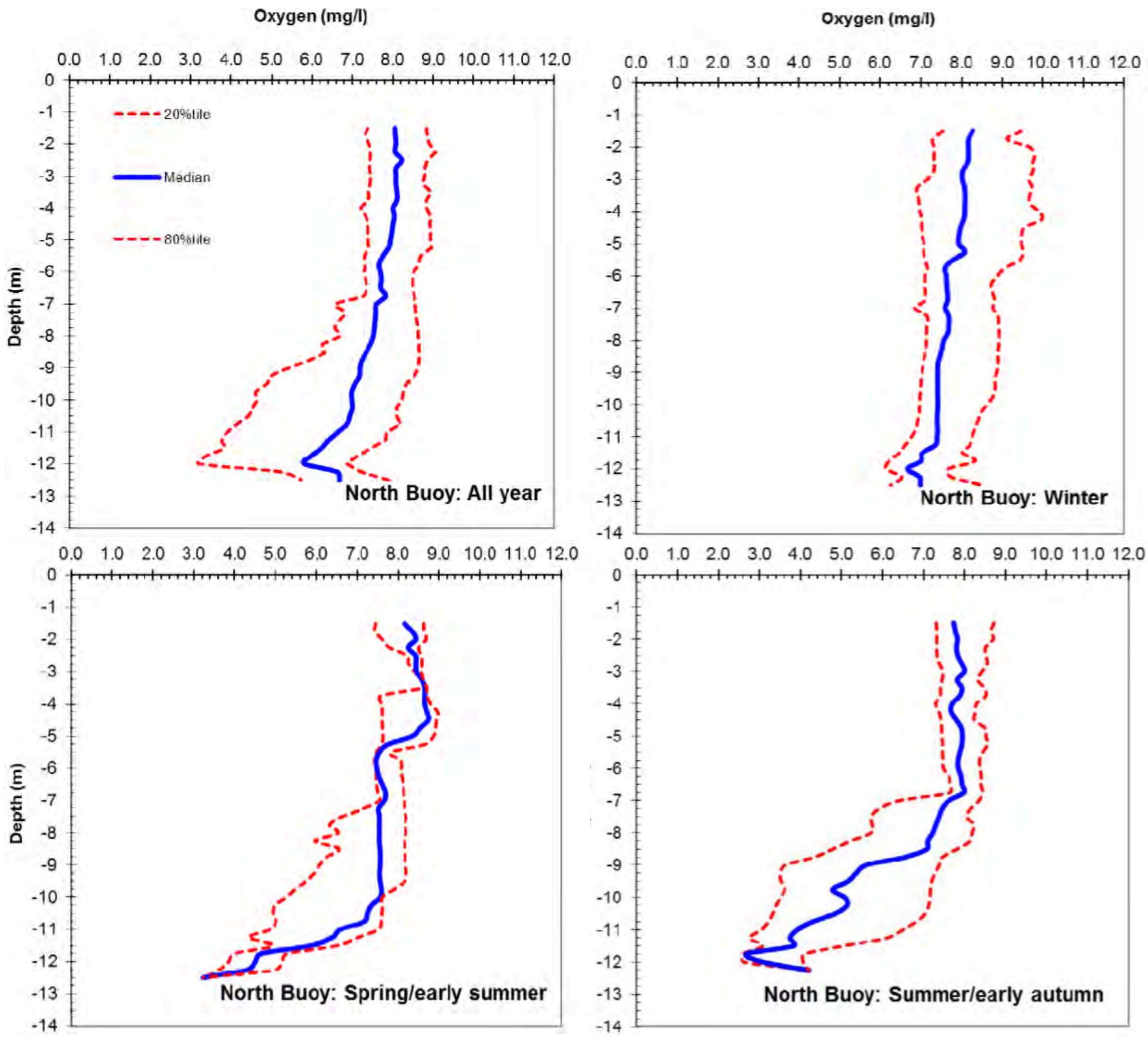


Figure 6.15 **Dissolved oxygen concentration median profiles at North Buoy for all seasons (winter, spring/early summer and summer/early autumn). The 20th and 80th percentile limits of the profiles are indicated by the dotted red lines (Source: van Ballegooyen *et al.* 2012).**

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 6.16). 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.

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 6.17). 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.

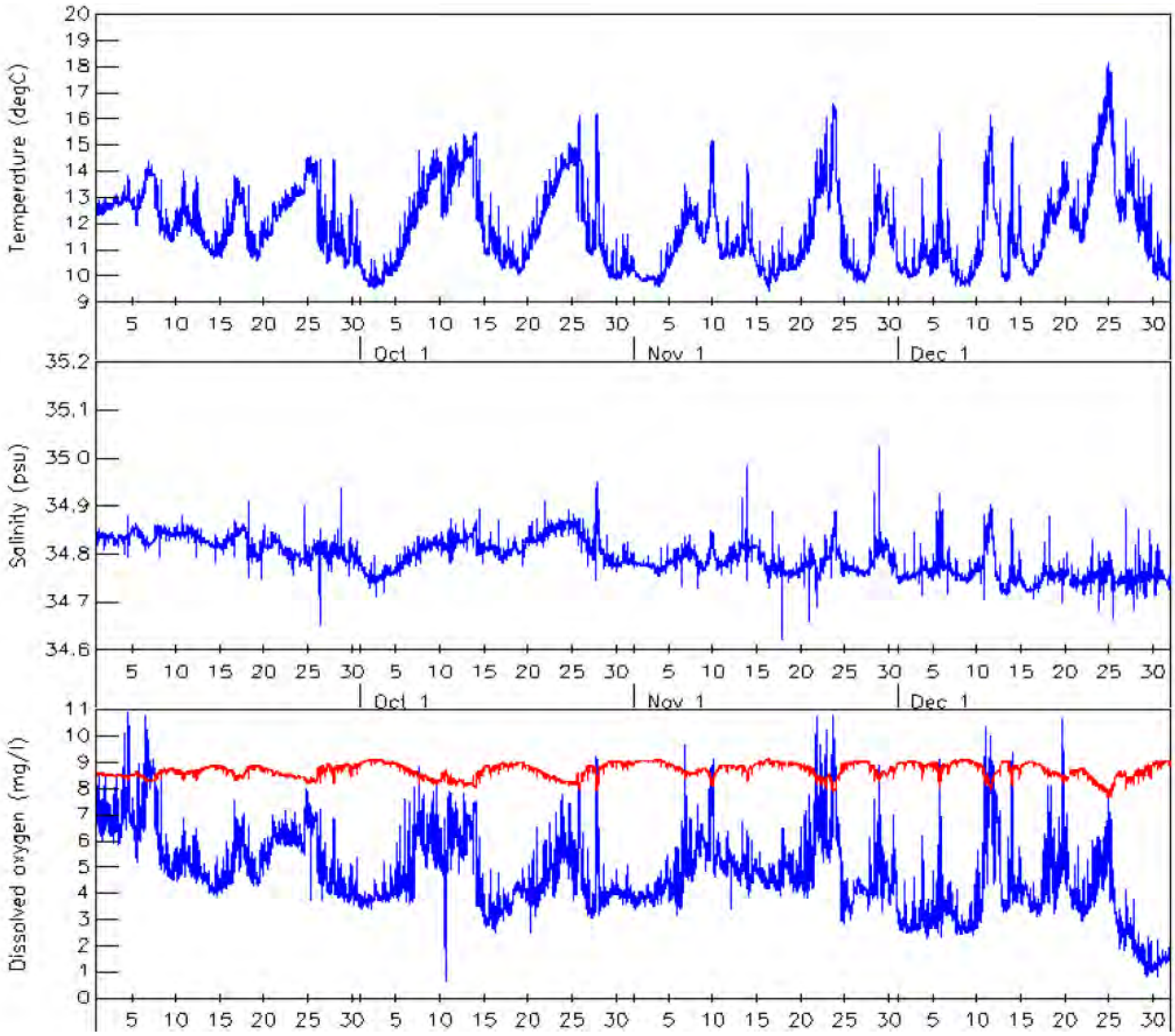


Figure 6.16. Time series of water temperature, salinity and dissolved oxygen concentrations from the mooring site (33° 01.679'S; 17° 59.143'E) for spring/early summer (Source: van Ballegooyen *et al.* 2012).

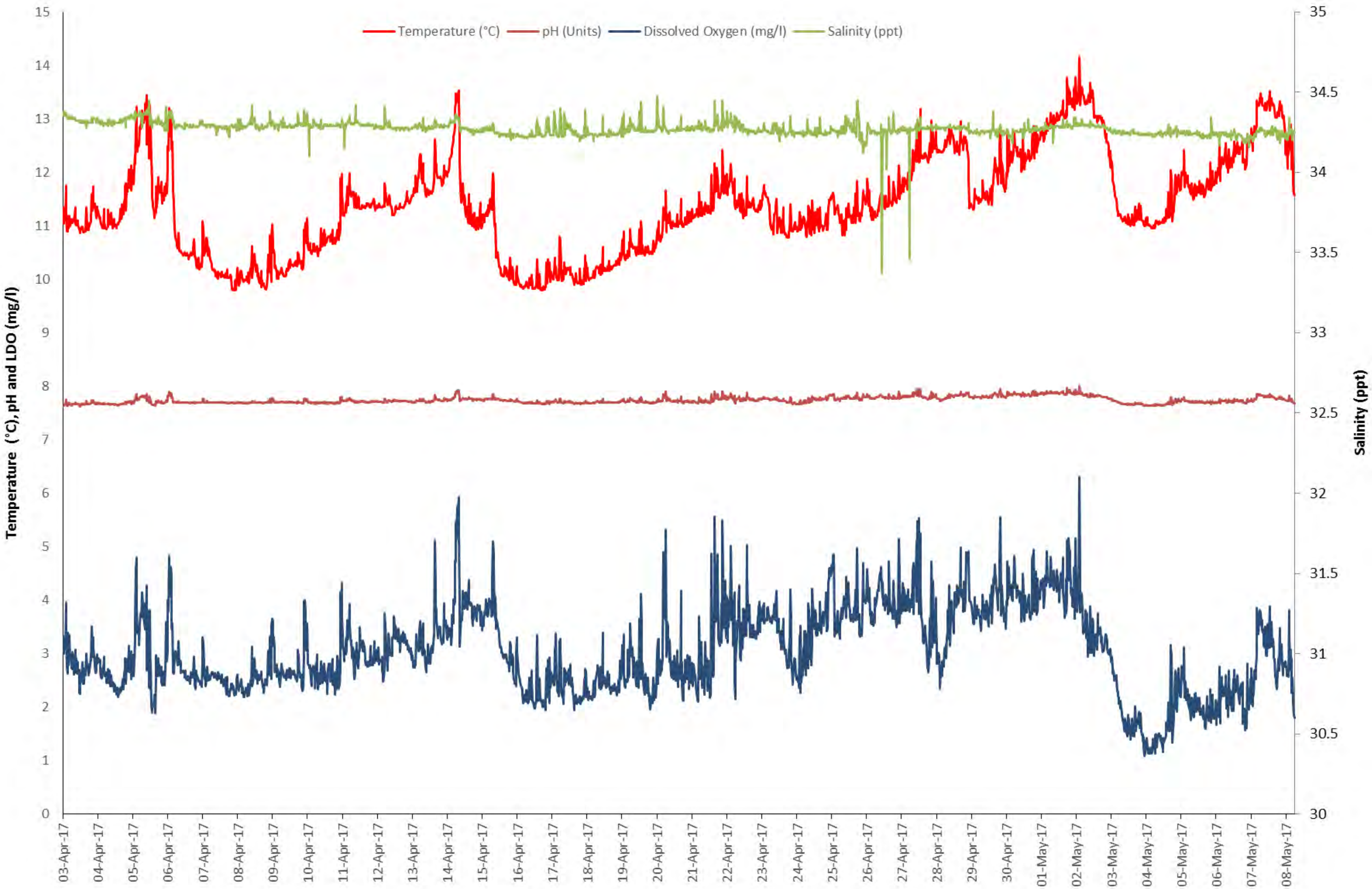


Figure 6.17. **Temperature, salinity, pH and dissolved oxygen (DO) recorded by the CTD deployed in 23 m water depth adjacent to the RO plant discharge at the base of the iron ore terminal.**

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 6.18). 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).

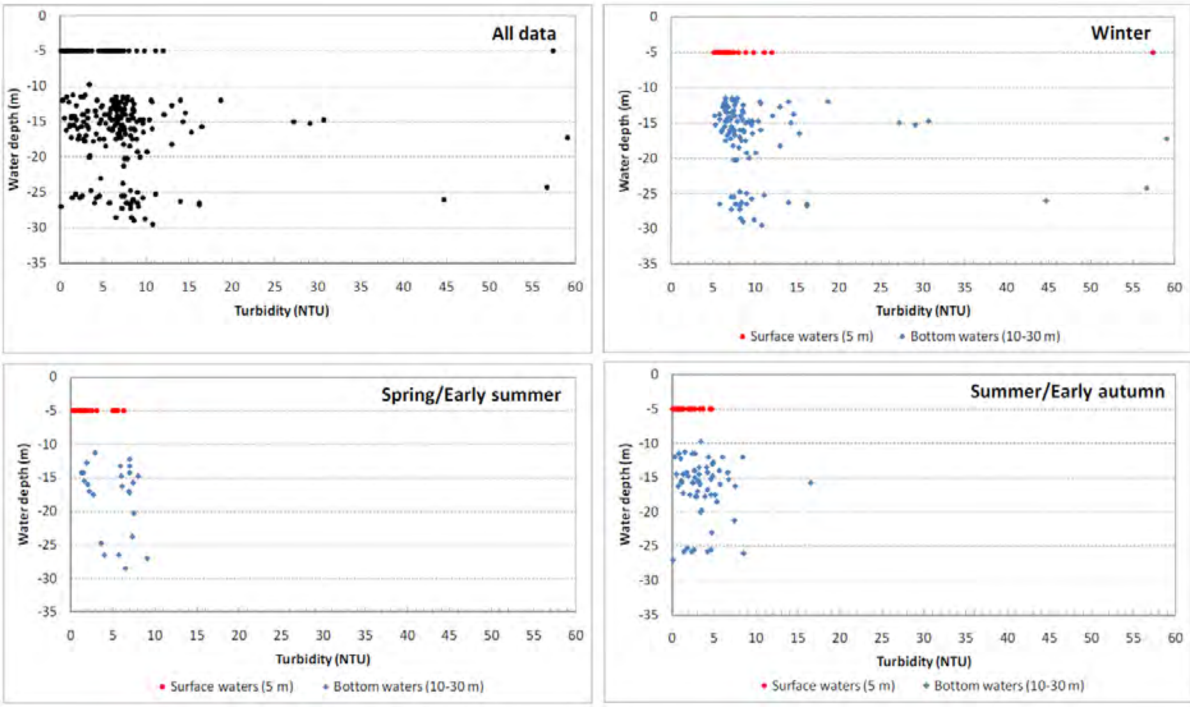


Figure 6.18 **Turbidity (NTU) plotted as a function of depth and season (Source: van Ballegooyen *et al.* 2012).**

Figure 6.19 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 breakdown 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 6.20), 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.

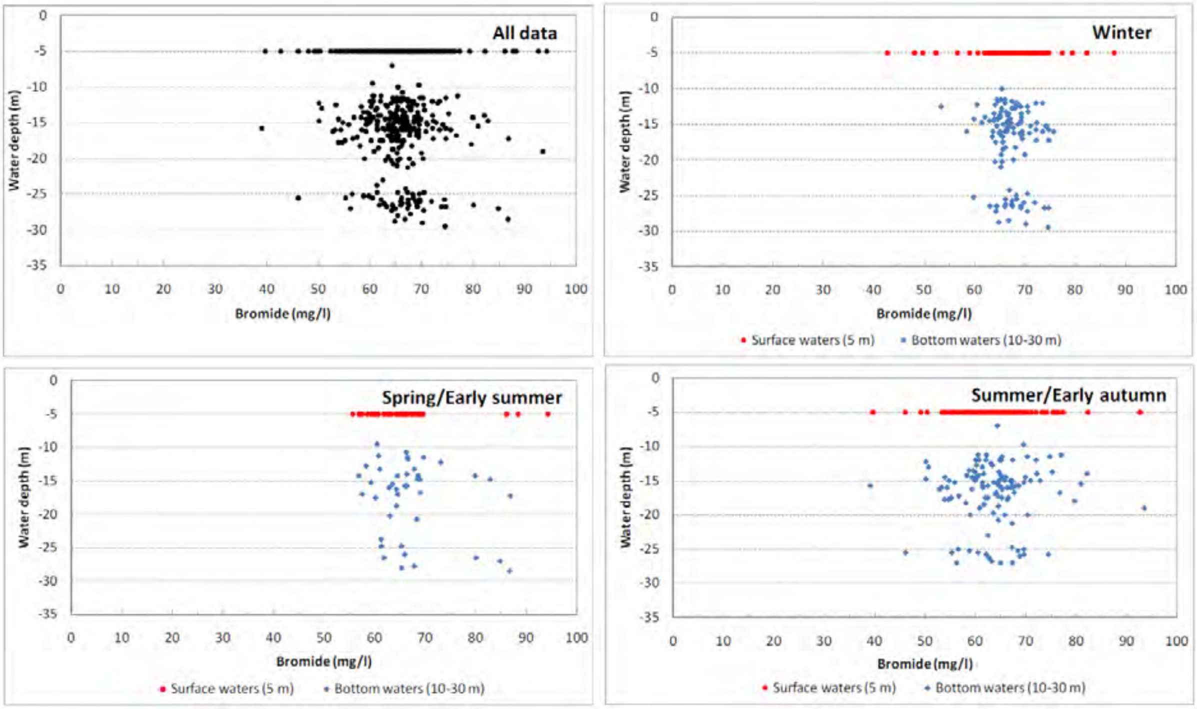


Figure 6.20 Bromide concentrations measured at all stations in winter, spring/early summer, and summer/early autumn (Source: van Ballegooyen *et al.* 2012).

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 across Europe 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 July 2019 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, this 2019 report however includes data up until end July 2019 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 6.21 and Figure 6.22 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 2019 *E. coli* data, 15 of the 20 sampled stations were categorized as having excellent water quality. The Bok River beach site that frequently has poor water quality improved slightly in 2018 and 2019 from "Poor" to "Fair", whilst water quality at the Hoedtjiesbaai site deteriorated and was again ranked as "Poor" based on the 2018 and 2019 data. Water quality at Sea Harvest (Site 3), improved from "Fair to "Excellent", whilst for the first time water quality at Kraalbaai North declined to "fair" (although the Hazen 90th percentile estimate remained well below the guideline, Figure 6.22)

It is encouraging that "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) but it is disappointing that the beaches along the northern shore of Small Bay, that are also popular swimming sites, continue to suffer from poor water quality. Considering that the majority of treated wastewater from the Langebaan Wastewater Treatment Works (WWTW) was diverted to other uses (including industrial, construction and irrigation), these results are surprising. A number of infrastructure upgrades on the plant were also completed recently. 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). Due to the lack of monitoring of treated effluent discharge volumes, it is difficult to draw a conclusion as to the source of the contamination. See Chapter **3** for further information regarding activities and discharges in the Saldanha Bay-Langebaan Lagoon System.

Reuse of wastewater from the Saldanha WWTW by Arcelor Mittal commenced in 2018 and may be responsible for the improvement in water quality near the Bok River mouth. This trend will hopefully continue into the future

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 no data were collected 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
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.
	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.
	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.
	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.
	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.
	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
	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
	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
	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
	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.
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.
	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.
	13. Mykonos - Harbour	Fair	Fair	Ex.	Ex.	Fair	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Fair	Ex.	Ex.	Good	Fair	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.
Langebaan Lagoon	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.
	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.
	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.
	18. Tooth Rock	ND	ND	ND	ND	ND	Fair	Ex.	Ex.	Ex.	Ex.	Fair	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	ND	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
	20. Kraalbaai South	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	Ex.	Ex.	Ex.	Ex.	Ex.	Ex.	ND	Ex.

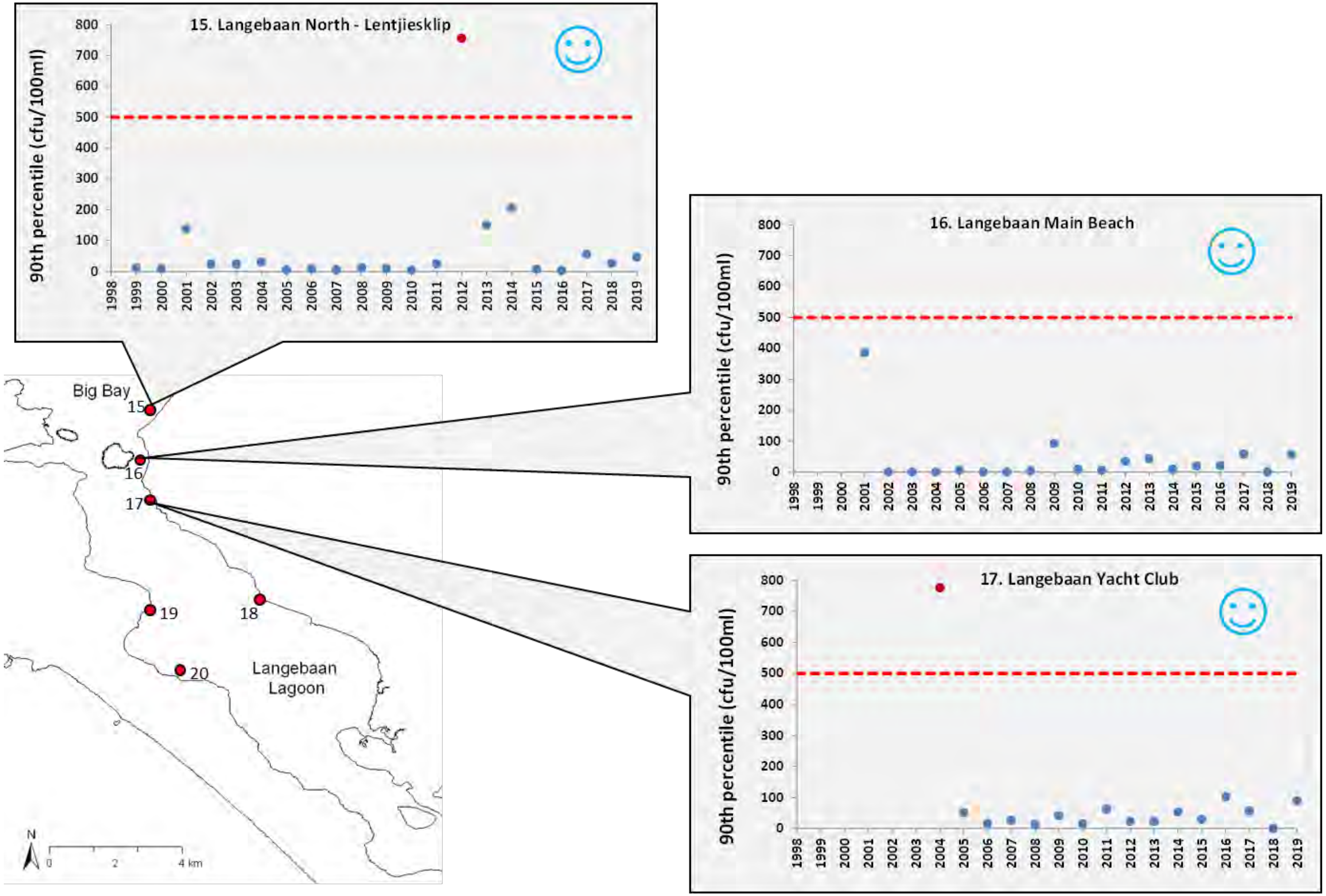


Figure 6.21 Hazen method 90th percentile values of *E.coli* counts at three of the six sampling stations within Langebaan Lagoon (Feb 1999 – Jul 2019). The red line indicates the Hazen method 90th percentile contact recreation limit of *E. coli* counts (500 colony-forming units/100 ml) above which water 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.

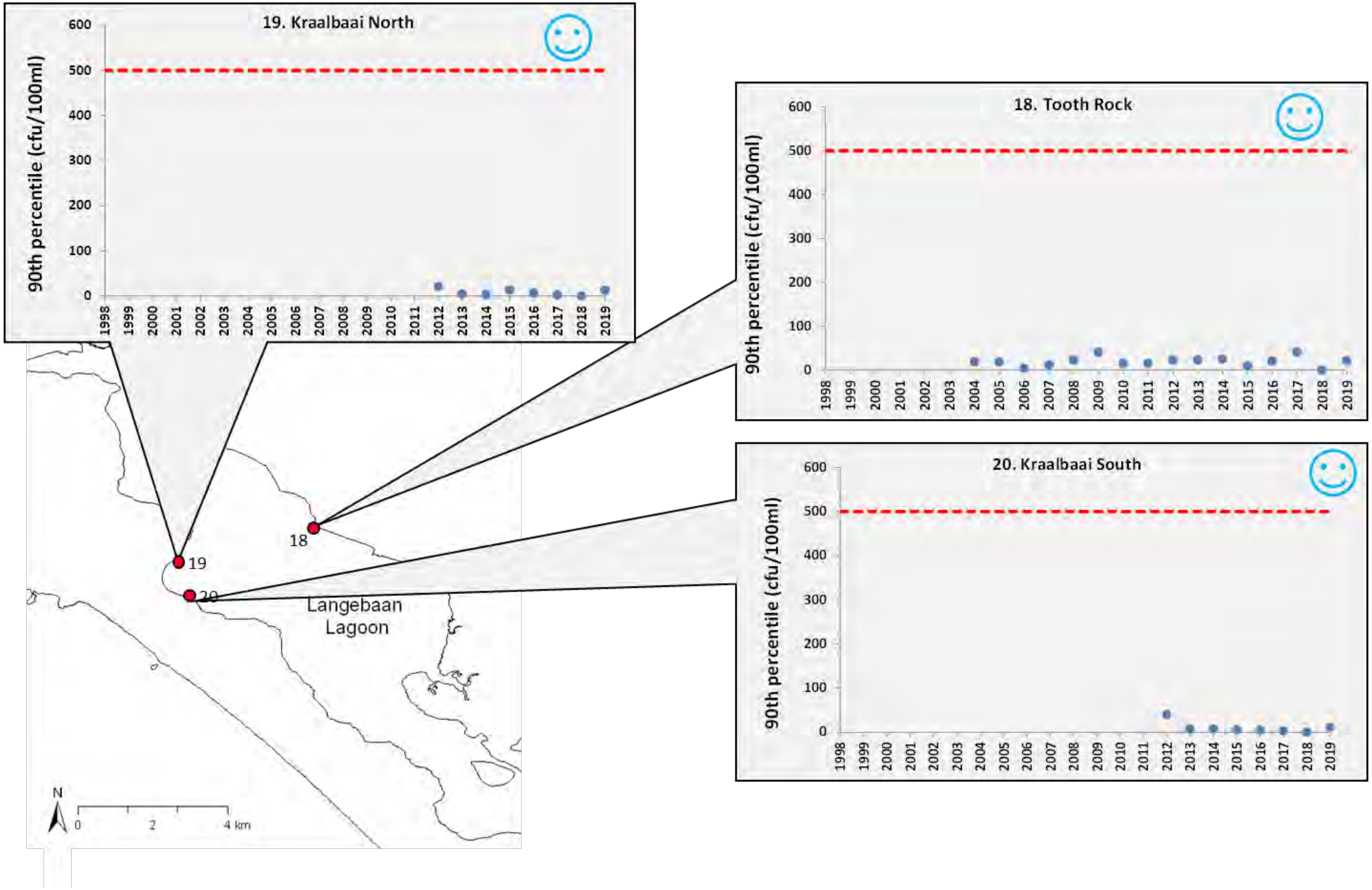


Figure 6.22 Hazen method 90th percentile values of *E.coli* counts at three of the six sampling stations within Langebaan Lagoon (Feb 1999 – July 2019). The red line indicates the Hazen method 90th percentile contact recreation limit of *E. coli* counts (500 colony-forming units/100 ml) above which water 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 6.23, Figure 6.24 and Figure 6.25). 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 several other sites elsewhere in Small Bay (the small quay and big quay at Pepper Bay), that all met the required standards in 2019. In 2019, the General Cargo Quay didn't meet the mariculture standard for the first time in the 20-year sampling history, and this result will hopefully prove anomalous. 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, which have not shown any improvement towards meeting guidelines over the last 20 years and continue to exceed the guidelines by a substantial margin (Figure 6.24 and Figure 6.25).

Although a sustained improvement in levels of compliance with mariculture WQGs has occurred since the 1999-2005 period at most sites (Figure 6.23, Figure 6.24), 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 land-based mariculture facilities 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 (data to end July so far) (Figure 6.26). 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.

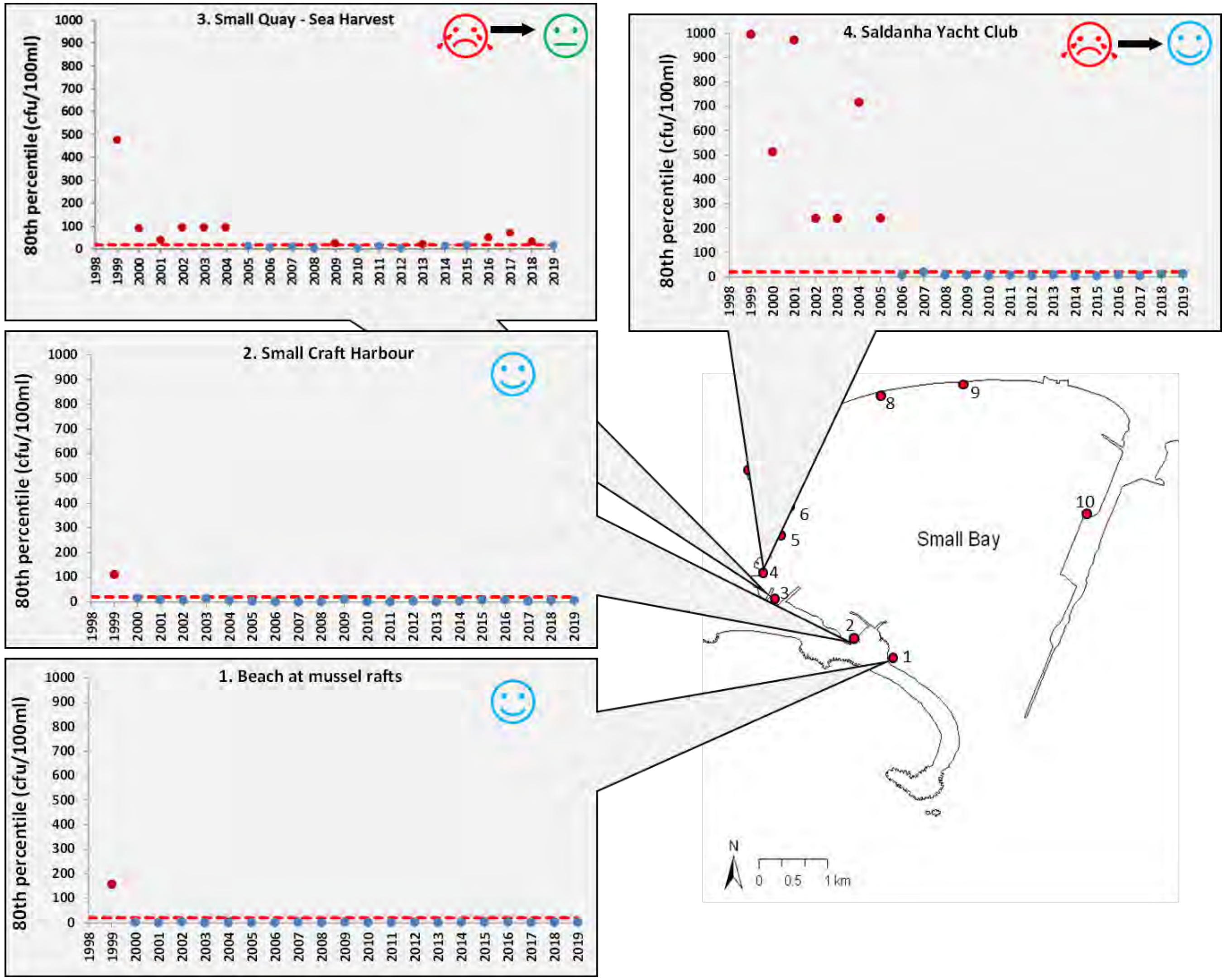


Figure 6.23 80th percentile values of faecal coliform counts at four of the 10 sampling stations within Small Bay (Feb 1999 – July 2019). The red line indicates the <u>80th percentile mariculture limit of faecal coliforms</u> (20 colony-forming units/100 ml). Red data points indicate 80th percentile values exceeding the guideline, whilst blue data points fall within the recommended guideline. The faces correspond to changes in water quality over time.

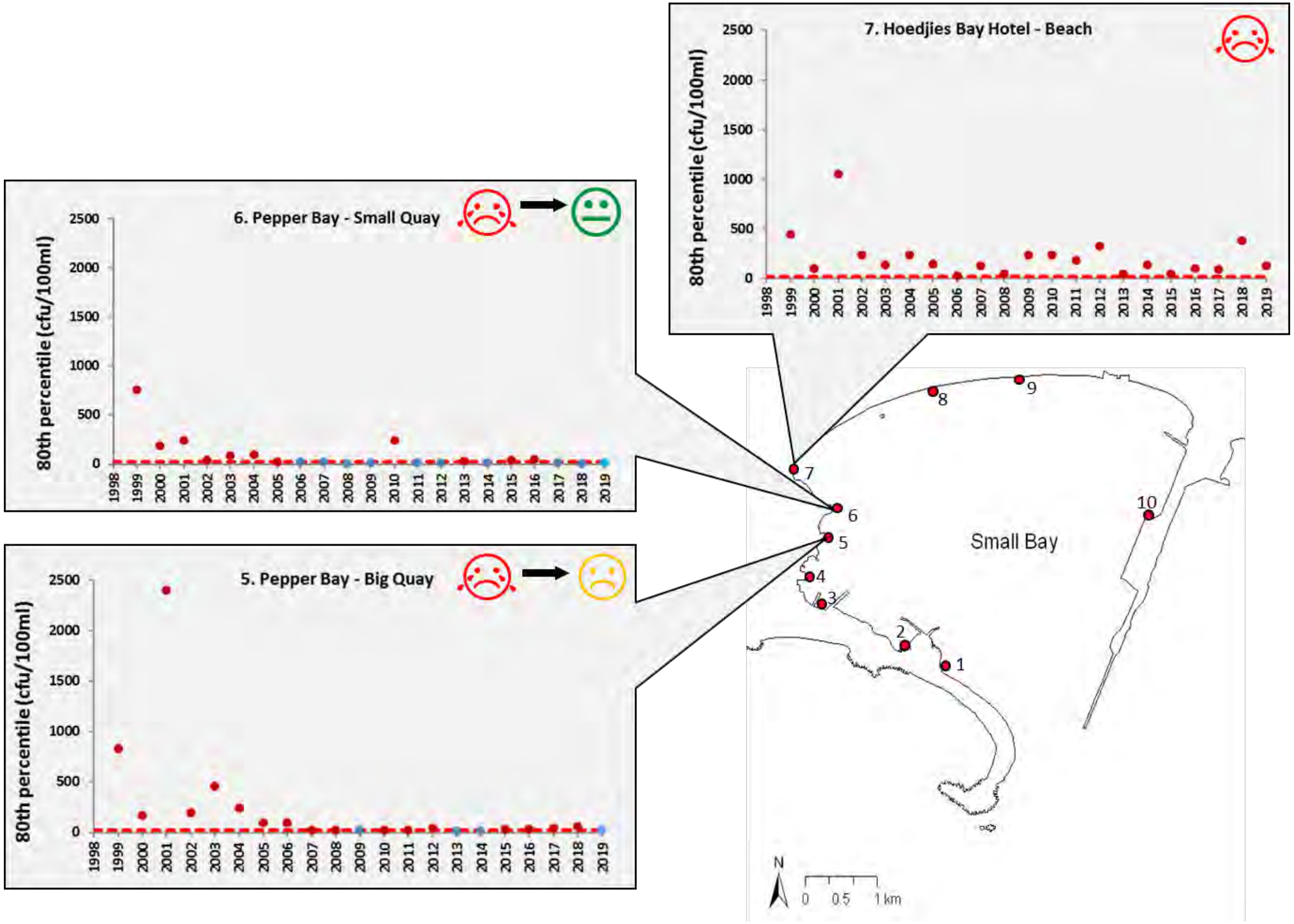


Figure 6.24 80th percentile values of faecal coliform counts at three of the 10 sampling stations within Small Bay (Feb 1999 – July 2019). The red line indicates the <u>80th percentile mariculture limit of faecal coliforms</u> (20 colony-forming units/100 ml). Red data points indicate 80th percentile values exceeding the guideline, whilst blue data points fall within the recommended guideline. The faces correspond to changes in water quality over time.

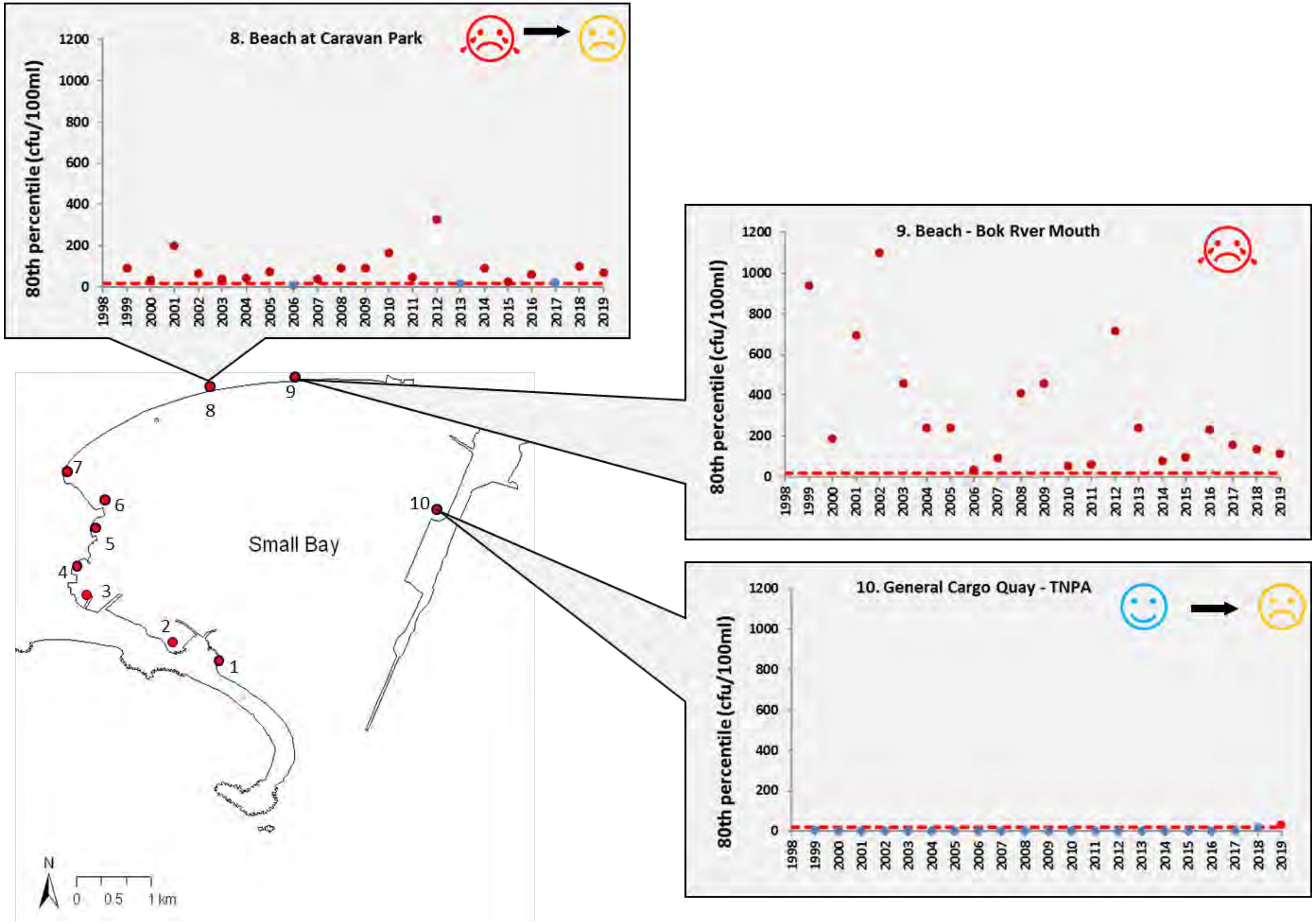


Figure 6.25 80th percentile values of faecal coliform counts at three of the 10 sampling stations within Small Bay (Feb 1999 – July 2019). The red line indicates the <u>80th percentile mariculture limit of faecal coliforms</u> (20 colony-forming units/100 ml). Red data points indicate 80th percentile values exceeding the guideline, whilst blue data points fall within the recommended guideline. The faces correspond to changes in water quality over time.

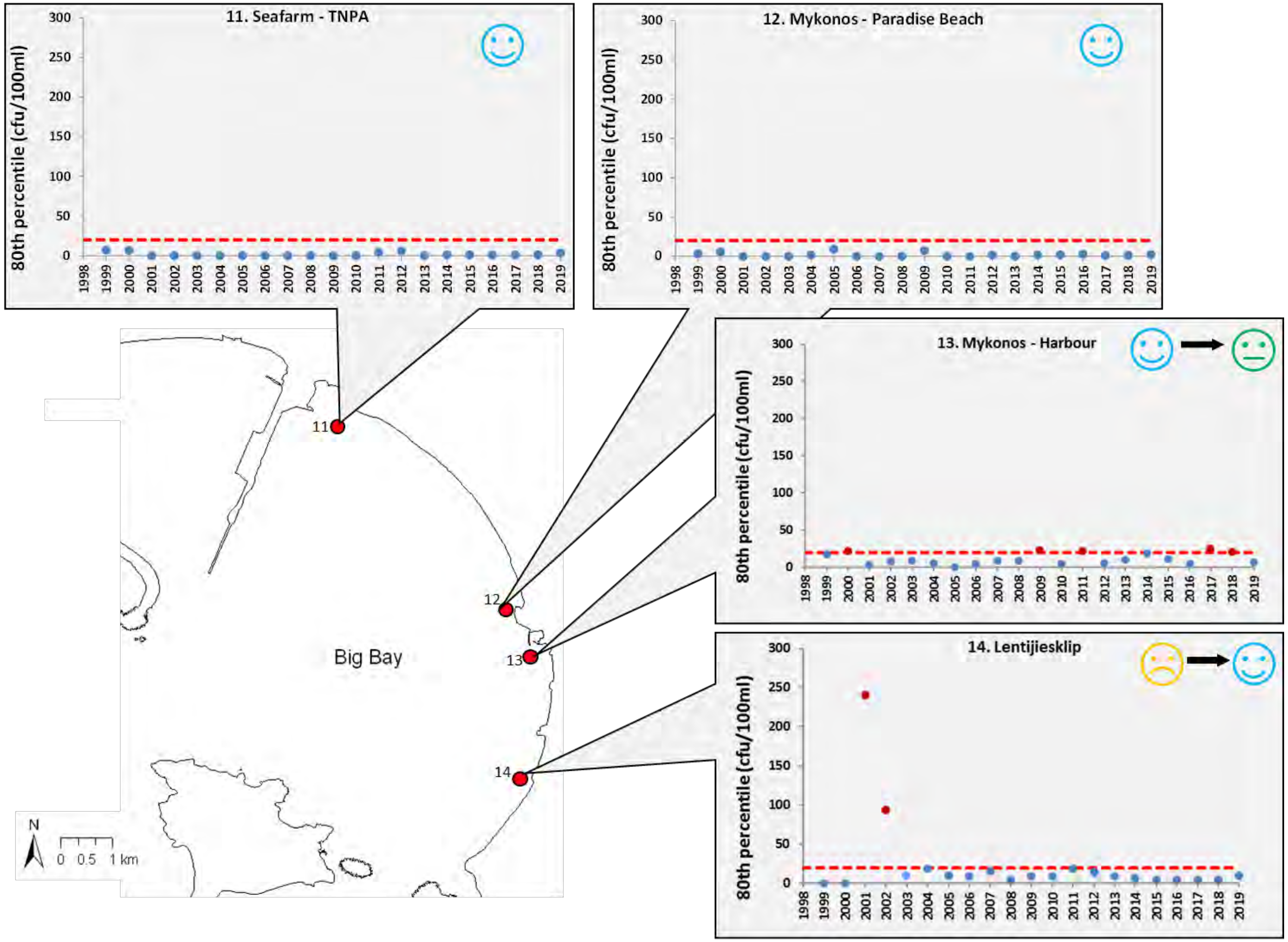


Figure 6.26 80th percentile values of faecal coliform counts at the four sampling stations within Big Bay (Feb 1999 – Jul 2019). The red line indicates the <u>80th percentile mariculture limit of faecal coliforms</u> (20 colony-forming units/100 ml). Red data points indicate 80th percentile values exceeding the guideline, whilst blue data points fall within the recommended guideline. The faces correspond to changes in water quality over time.

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 considered to be 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 dramatic 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 6.28 to Figure 6.33 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/l 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)

Trace metal levels in bivalves in the 2019 edition of the 'State of the Bay' are relative to wet weights of bivalve tissue. Mercury concentrations within mussel tissues were measured for the first time in 2016. To date, values have not exceeded the safe limit of 0.5 ppm (Figure 6.27). Lead concentrations were found to exceed the regulatory limit for foodstuffs of 0.5 ppm at Portnet and the Saldanha Bay North sites in 2019 (Figure 6.28). Lead concentration in mussel tissue collected from the other three sites in Small Bay was below the guideline limit which represents and improvement over the historical data where the guideline was frequently exceeded at these sites. Mussels collected at the Portnet site have historically had high concentrations of lead in their tissue and although values in the last five years have not been as high as historical peaks, they remain more than double the recommended level. 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 0.3 ppm to 1.7 ppm over the last five years with an average of 1.5 ppm in 2019. This indicates that the lead pollution situation in Small Bay overall has not improved. The level of lead in mussels at the Portnet site was almost 11 times the level considered safe for human consumption in 2019. This is 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 2014-2018 ranged between 0.9 and 1.6 ppm, with an average of 1 ppm recorded in 2019. Historically, the maximum value of 10.9 ppm was recorded in April 2007 at the Mussel Raft and this was the only site where the recommended level of 2 ppm was exceeded in 2018, when a concentration of 3.7 ppm was measured. In 2019 cadmium concentration in mussel tissue from all five sites sampled within Small Bay fell below the limit (Figure 6.29).

Average zinc concentrations recorded in 2019, and historically at nearly all sites, were much lower than the 150 ppm regulatory limit listed by the Canadian Authorities (Figure 6.30). This metal only rose above the limit once at the Saldanha Bay north site (165 ppm in 2016), which was also elevated in 2019 samples albeit not above the guideline Figure 6.30). Concentrations of copper 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 2019 where levels peaked at just over 3 ppm, an all-time high (Figure 6.32). 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.

Iron concentrations in mussel tissue appears to have increased over time, with generally higher concentrations recorded over the last five years compared to most historical values over the 1997-2007 period (Figure 6.33). 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. 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. 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).

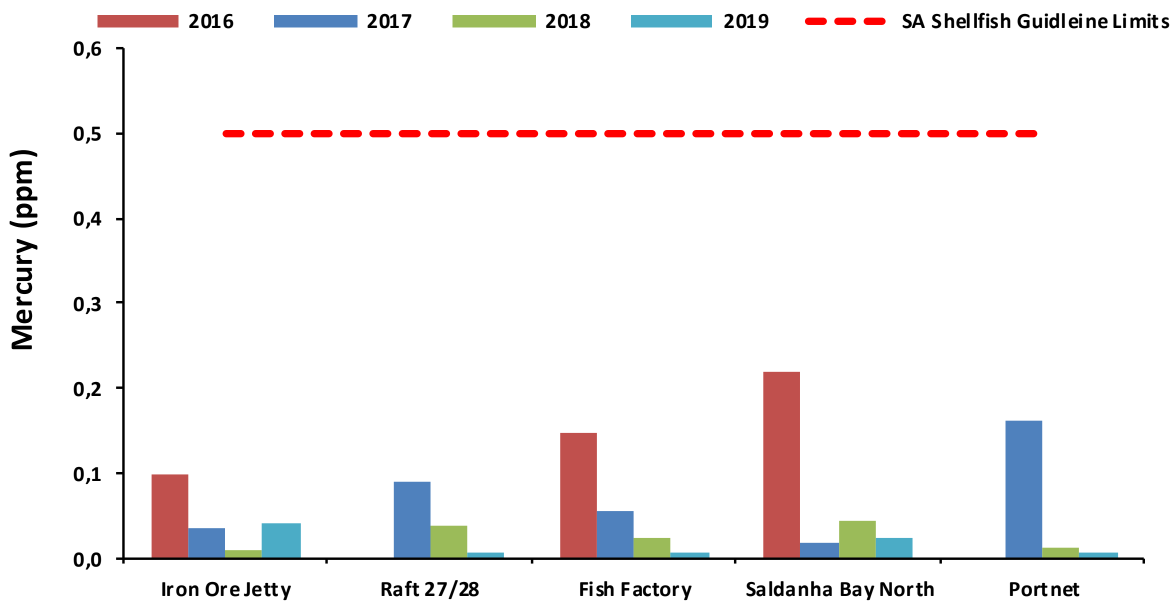


Figure 6.27 **Mercury concentrations in wet mussel flesh collected by Anchor from five sites in Saldanha Bay in autumn 2016 to 2019. The recommended maximum limit for mercury in seafood (0.5 ppm) is shown as a dotted red line.**

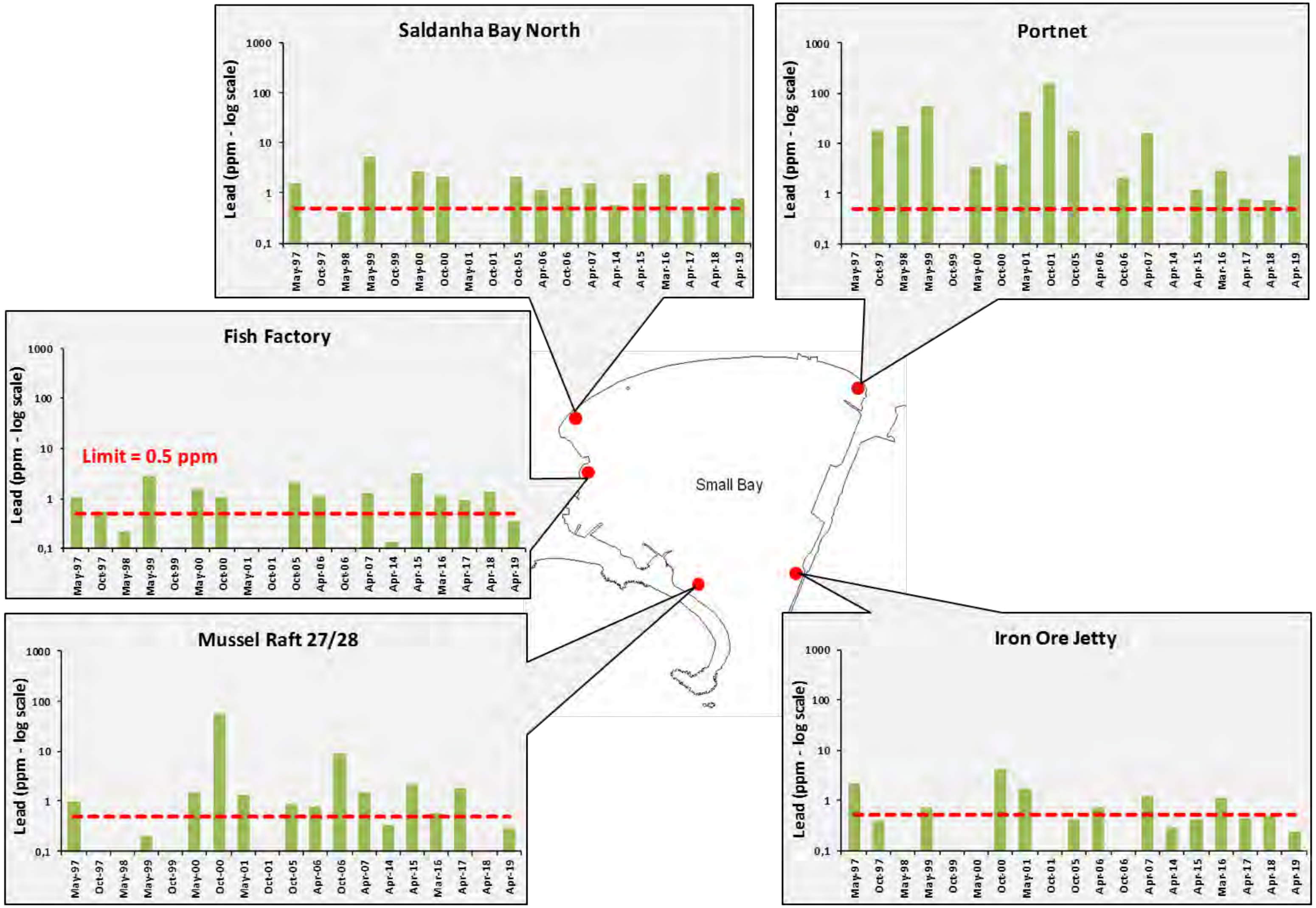


Figure 6.28 **Lead concentrations in mussels (wet weight) collected from five sites in Saldanha Bay from 1997-2007 as part of the Mussel Watch Programme (Source: G. Kiviets, Department of Environmental Affairs) and by Anchor from 2014 to 2019. The recommended maximum limit for lead in seafood (0.5 ppm) is shown as a dotted red line. Note that data are plotted on a log scale.**

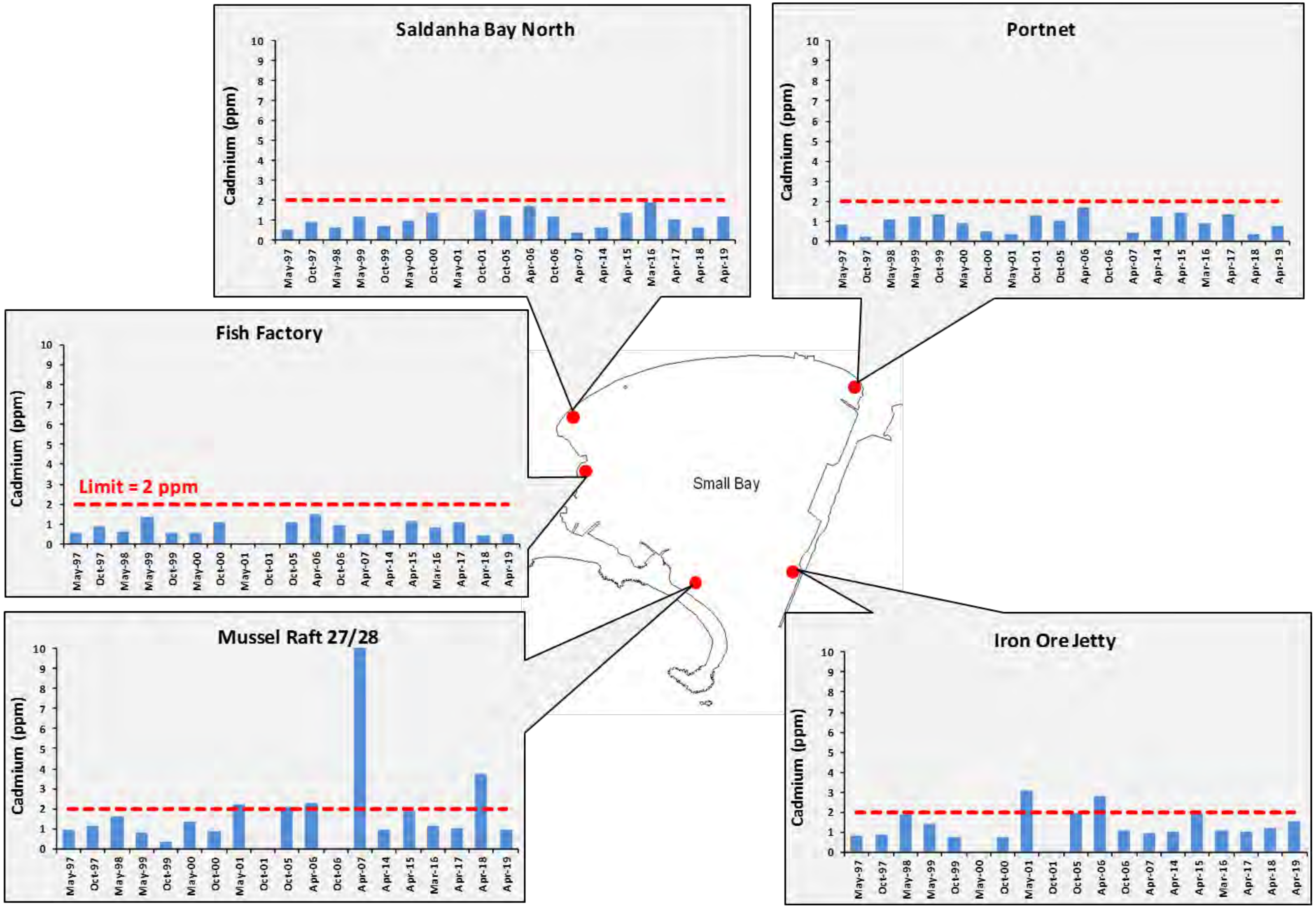


Figure 6.29 Cadmium concentrations in mussels (wet weight) collected from five sites in Saldanha Bay from 1997-2007 as part of the Mussel Watch Programme (Source: G. Kiviets, DEA) and by Anchor from 2014 to 2019. The recommended maximum limit for cadmium in seafood was reduced to 2 ppm (dotted red line) in 2018.

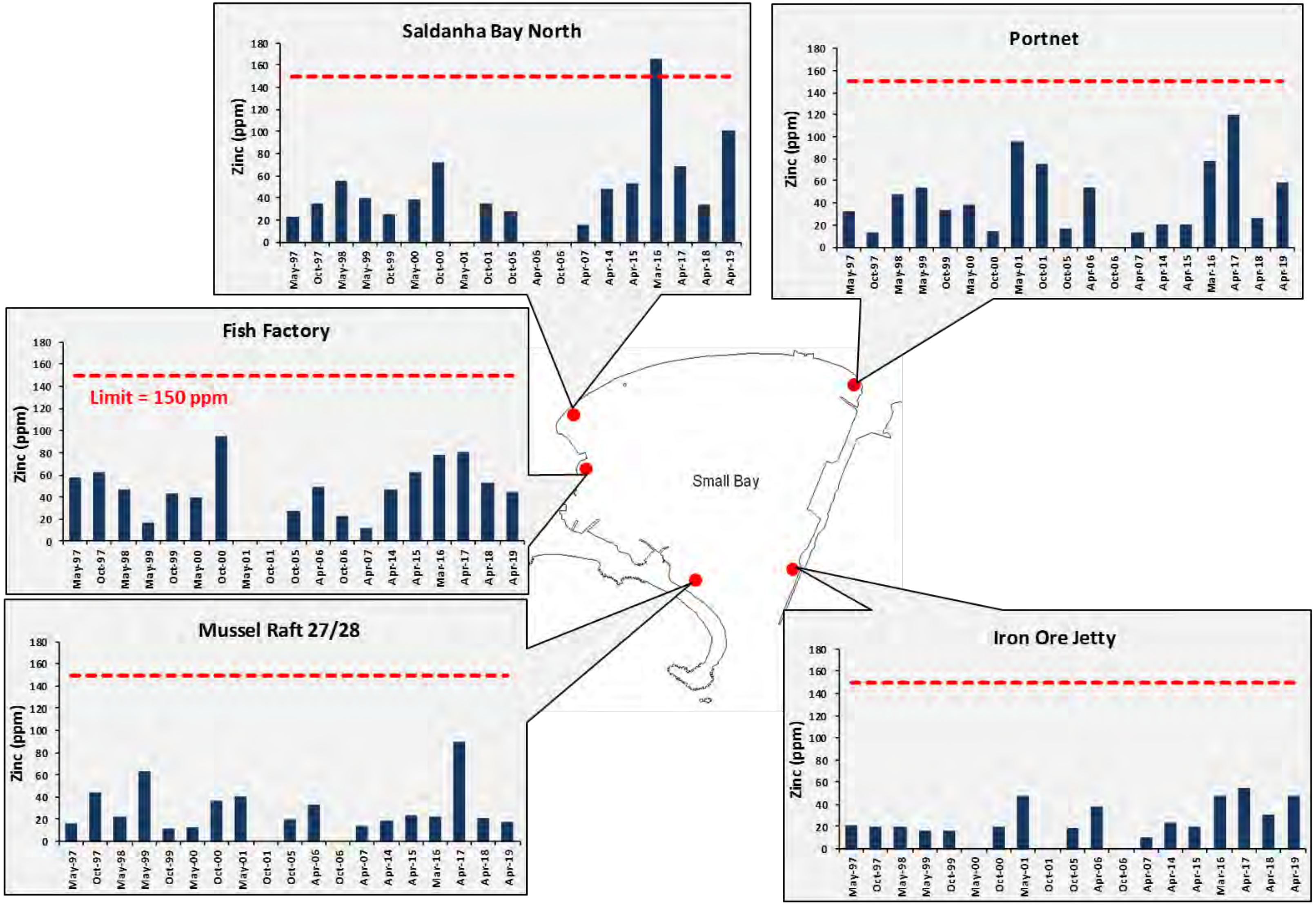


Figure 6.30 Zinc concentrations in mussels (wet weight) collected from five sites in Saldanha Bay from 1997-2007 as part of the Mussel Watch Programme (source: G. Kiviets, Department of Environmental Affairs) and by Anchor from 2014 to 2019. The recommended maximum limit for zinc in seafood (150 ppm) is shown as a dotted red line.

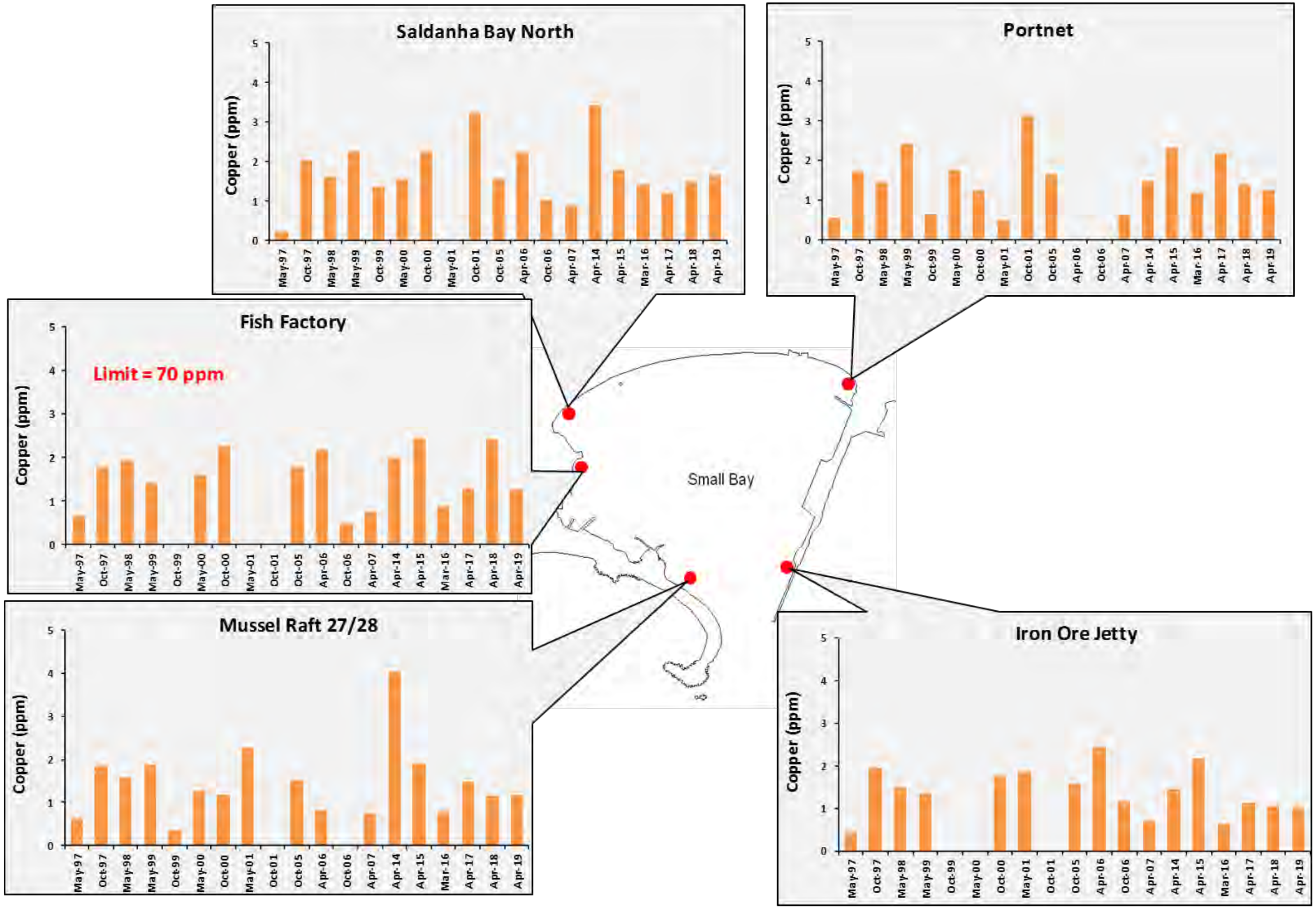


Figure 6.31 Copper concentrations in mussels (wet weight) collected from five sites in Saldanha Bay from 1997-2007 as part of the Mussel Watch Programme (Source: G. Kiviets, Department of Environmental Affairs) and by Anchor from 2014 to 2019. The recommended maximum limit for copper in seafood is 70 ppm (not indicated on graphs).

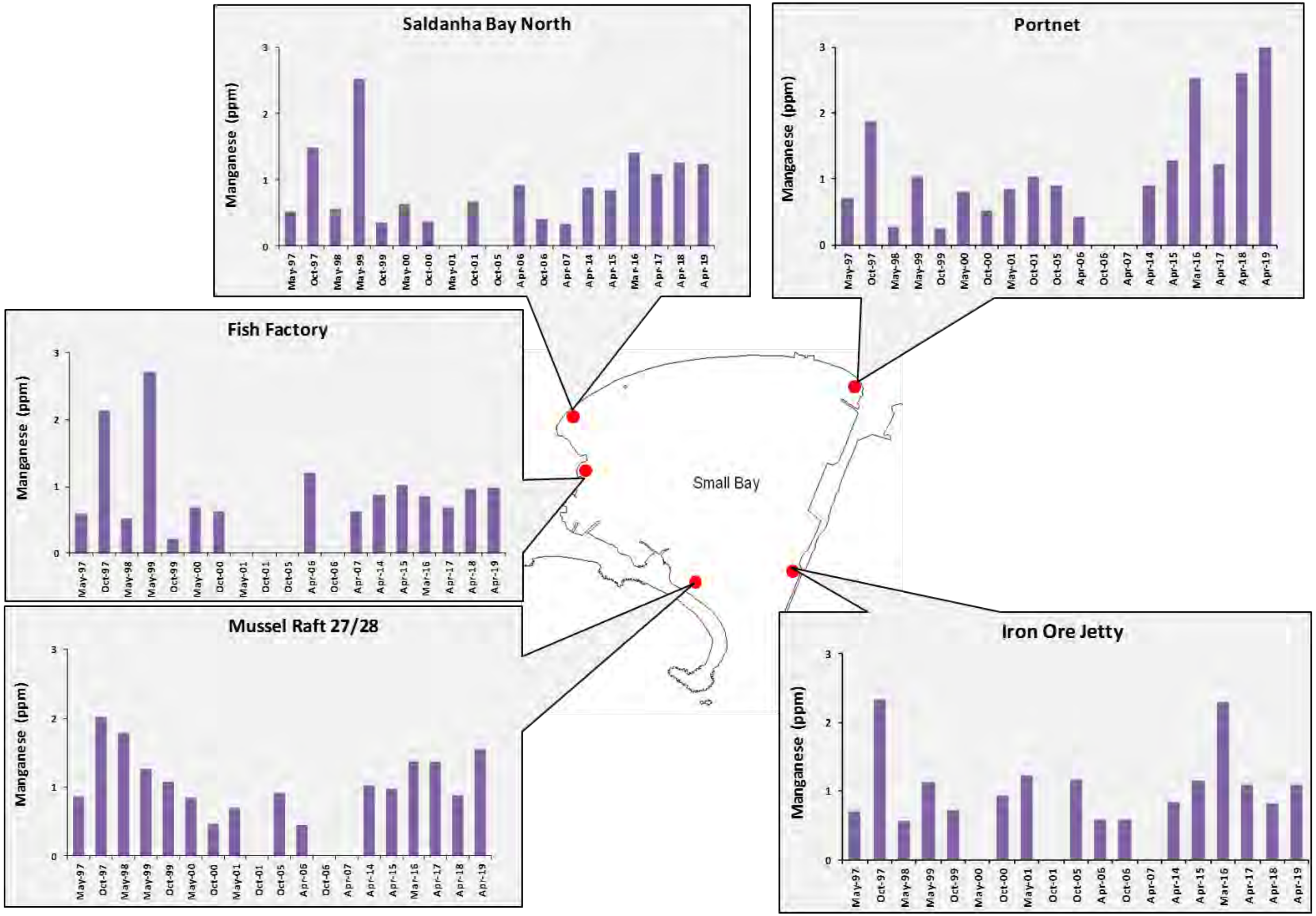


Figure 6.32 Manganese concentrations in mussels (wet weight) collected from five sites in Saldanha Bay from 1997-2007 as part of the Mussel Watch Programme (Source: G. Kiviets, Department of Environmental Affairs) and by Anchor from 2014 to 2019. No limits are specified for manganese in seafood.

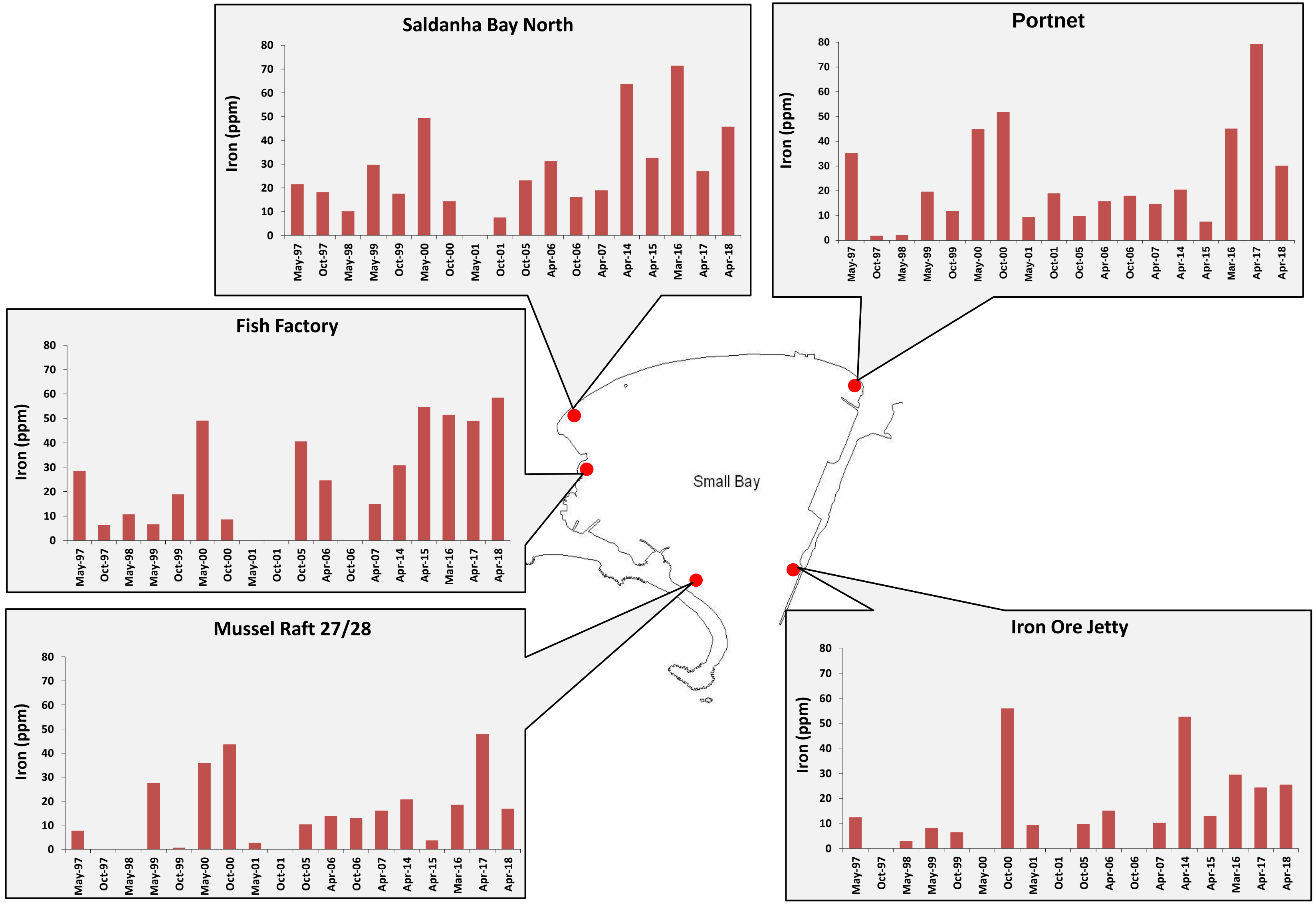


Figure 6.33 **Iron concentrations in mussels (wet weight) collected from five sites in Saldanha Bay from 1997-2007 as part of the Mussel Watch Programme (Source: G. Kiviets, Department of Environmental Affairs) and by Anchor from 2014 to 2018. No limits are specified for iron in seafood.**

6.10.2 Mariculture bivalve monitoring

A combined 430 ha of sea space are currently available for aquaculture production in Saldanha Bay, of which 316.5 ha have been leased to 14 individual mariculture operators for mussels, oysters, finfish and algae (see Chapter **3** for the layout of concession areas). Proposed expansion of the Aquaculture Development Zone (ADZ) includes an additional 1 404 ha of concessions in Outer Bay and Big Bay combined. 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 2019 are shown on

Figure 6.34, while Figure 6.35 shows data for oysters for the period 2005 to 2019. For comparative purposes, independent research data from the Mussel Watch Programme (1997-2007) and SOB monitoring (2014-2019) and from research conducted by Jacques Bezuidenhout (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 Saldanha in June 2018. Gaps in the data exist depending on the frequency of monitoring and the year each company was founded. Triangles represent data recorded from aquaculture farms, whereas circles represent data recorded during research studies. Research samples were collected from the shore, port (oil jetty, multipurpose quay, channel markers), or 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. 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 inverse was true (higher in farmed mussels) in recent samples collected from Outer Bay North (

Figure 6.34). Iron was most prevalent in mussel tissue, followed by zinc. 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 $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 two years, with the reported concentration typically much lower than that measured in research samples collected from the nearshore. 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 (see Section 6.10.1, Figure 6.28).

Data received from mussel farms showed that cadmium concentrations in Small Bay only exceeded the prescribed limit of 2 ppm once in 2015 (

Figure 6.34). 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 6.34). This is confirmed by analyses run on mussels collected in 2014 and 2015 by Bezuidenhout *et al*. (2015). In recent 2018 samples, cadmium concentrations greatly exceeded the limit at aquaculture farms in Outer Bay North. 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 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. 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 6.34). 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 6.34). Overall, data from the mussel farms discussed above suggest 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 adjected into the Bay from offshore will accumulate fewer toxins than mussels filtering 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 to the availability of phytoplankton in deeper areas of the Bay, resulting in less time for the accumulation of toxins within the mussel tissue. This pattern was, however, not observed in 2018 and 2019 mussel samples analysed for cadmium from farms in Outer Bay North, although the reasons behind this finding are still not clear.

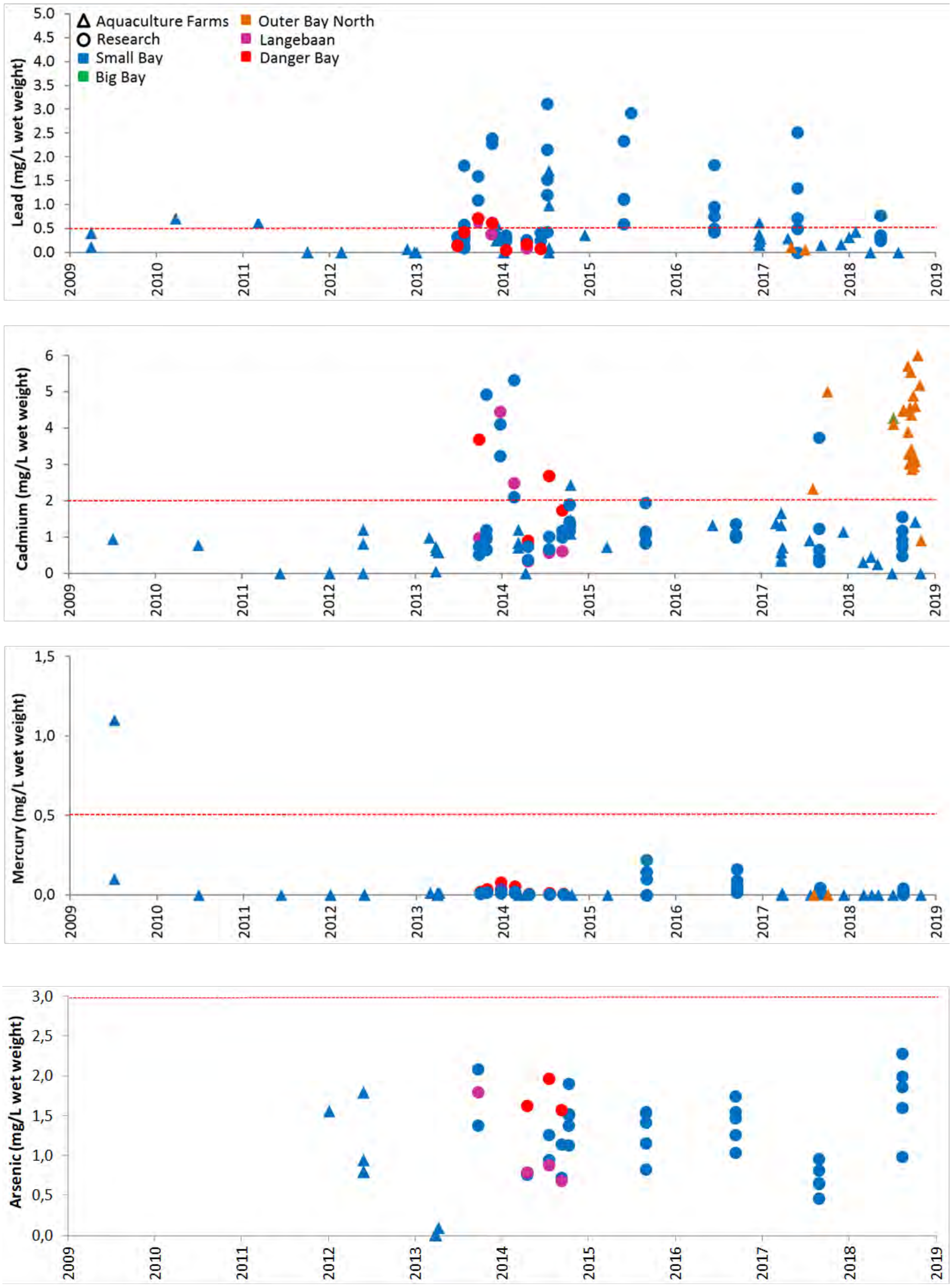


Figure 6.34 **Trace metal concentrations (wet weight) in mussel tissue provided by aquaculture facilities (triangles) and samples collected by researchers, primarily from the shore (circles).**

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 6.35). Research samples collected as part of the Anchor Oyster Monitoring Programme during 2018 and 2019 also largely show compliance with guideline levels (93%) with only two samples from Small Bay exceeding the limit (Figure 6.35). Cadmium concentration in samples of farmed oysters from Big Bay and Small Bay have mostly (97.5%) been below the guideline value of 3 ppm, with just five samples exceeding the limit (Figure 6.35). Cadmium concentration in all 29 research samples collected during 2018 and 2019 fell below 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 6.35). 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 6.35). All 31 samples analysed as part of the Anchor Oyster Monitoring Programme during 2018 and 2019 fell well below the regulatory limit for arsenic (Figure 6.35).

In general, trace metal concentrations in farmed oyster samples have largely met the regulatory limits for the four trace metals tested, with 100% compliance in all samples collected since 2016. This is also the case with samples collected as part of the Anchor Oyster Monitoring Programme, with the exception of two samples where lead concentration exceeded the limit.

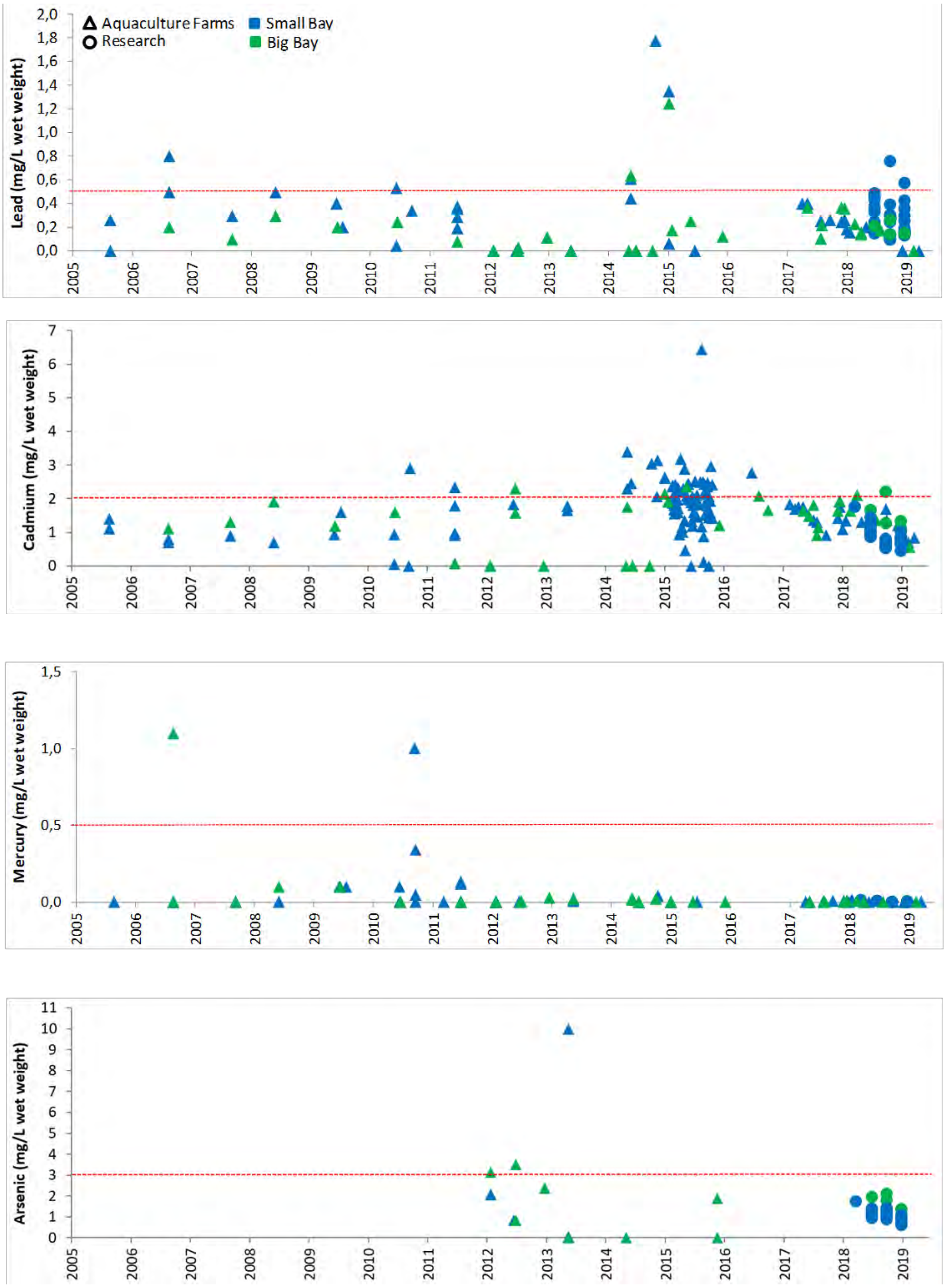


Figure 6.35 **Trace metal concentrations (wet weight) in oyster tissue provided by aquaculture facilities and the Anchor Oyster Monitoring Programme (indicated by triangles and circles respectively).**

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 conceivable 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. Given that cold, nutrient rich water influx during summer is density driven; dredging shipping channels could have facilitated this process which would be evident as a decrease in water temperature and salinity and an increase in nitrate and chlorophyll concentrations. Once again, there is little evidence of this in the available data series. Natural, regional oceanographic processes (wind driven upwelling or down-welling and extensive coast to bay exchange), rather than internal, anthropogenic causes, appear to remain the major factors affecting physical water characteristics in Saldanha Bay. The construction of physical barriers (the iron ore/oil jetty and the Marcus Island causeway) do appear to have 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 area of concern in the region of the Hoedtjies Bay Hotel. In the 2018 and 2019 data presented in this report, 15 of the 20 monitoring stations in the Bay are rated as having 'Excellent' water quality, the two beach sites in the vicinity of the Bok River Mouth are rated as 'Fair' representing an improvement over most earlier samples collected at these sites. It is a concerning that faecal coliform levels at the Hoedjiesbaai Beach remain elevated on occasion and local authorities are advised to try determining the source of this pollution. Faecal coliform counts at all four sites in Big Bay were within both the 80^{th} percentile limits for mariculture in 2019. In Small Bay however, the 80^{th} 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 (particularly for lead at the Portnet site) within Small Bay and are frequently above published guidelines for foodstuffs. In comparison, data collected by mariculture operators in Saldanha Bay show that concentrations in deeper water are lower and tend to mostly be below food safety limits (with nearly all samples collected from farmed mussel and oyster tissue in Big Bay and Small Bay since 2016 meeting the limits. Cadmium concentration in farmed mussels from Outer Bay North, however, exceeded the guidelines in 2018 and 2019 samples. Exceedance of food safety limits for lead and cadmium in mussels collected from the shore and the aquaculture farm at Outer Bay North, however, points to the need for management interventions to address this issue, as metal contamination poses a serious risk to the health of people harvesting mussels.

7 SEDIMENTS

7.1 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.

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 areaFigure 7.1). 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 7.2).

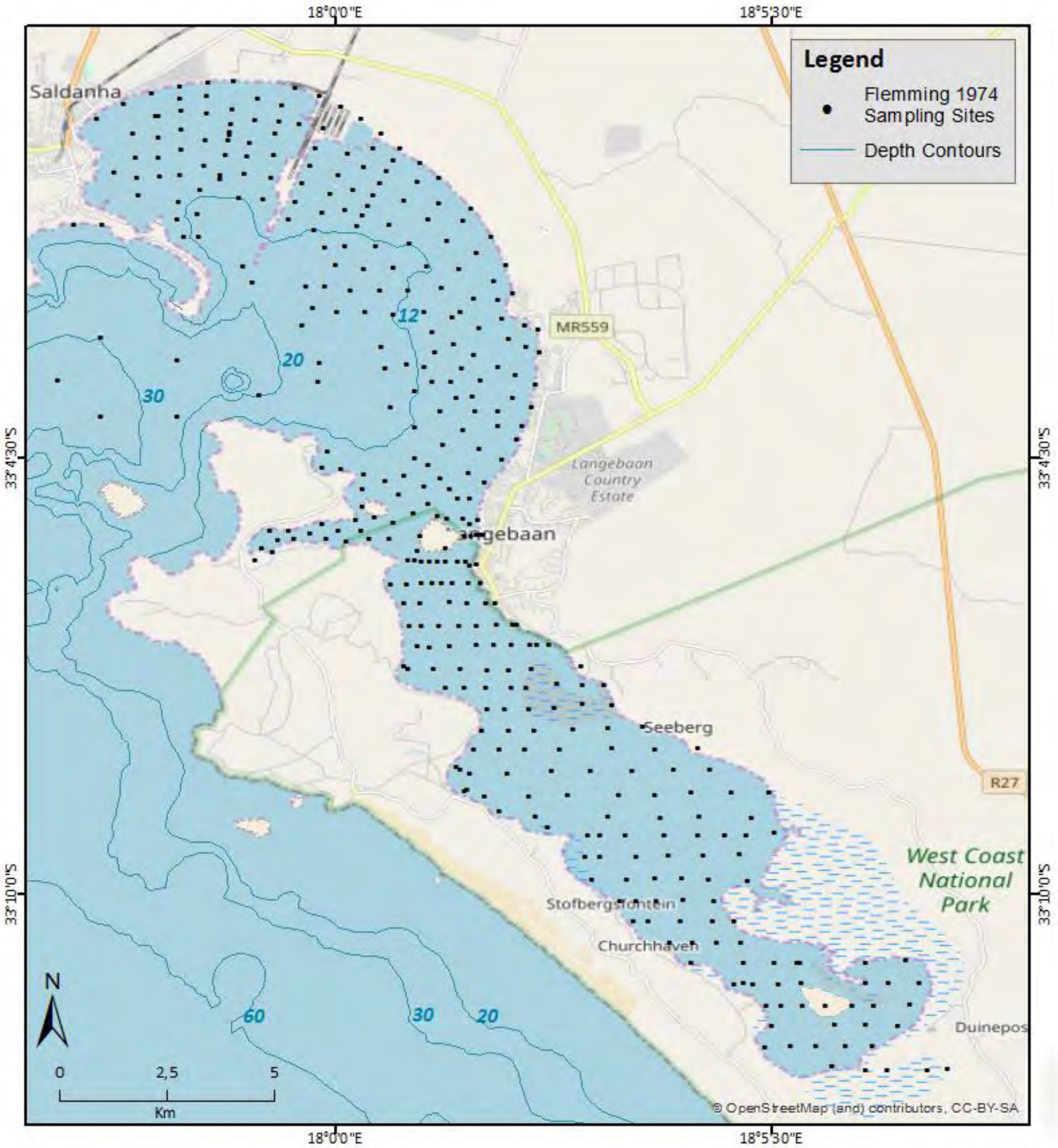


Figure 7.1 Stations sampled by Flemming (1977b) in Saldanha Bay and Langebaan Lagoon in 1974.

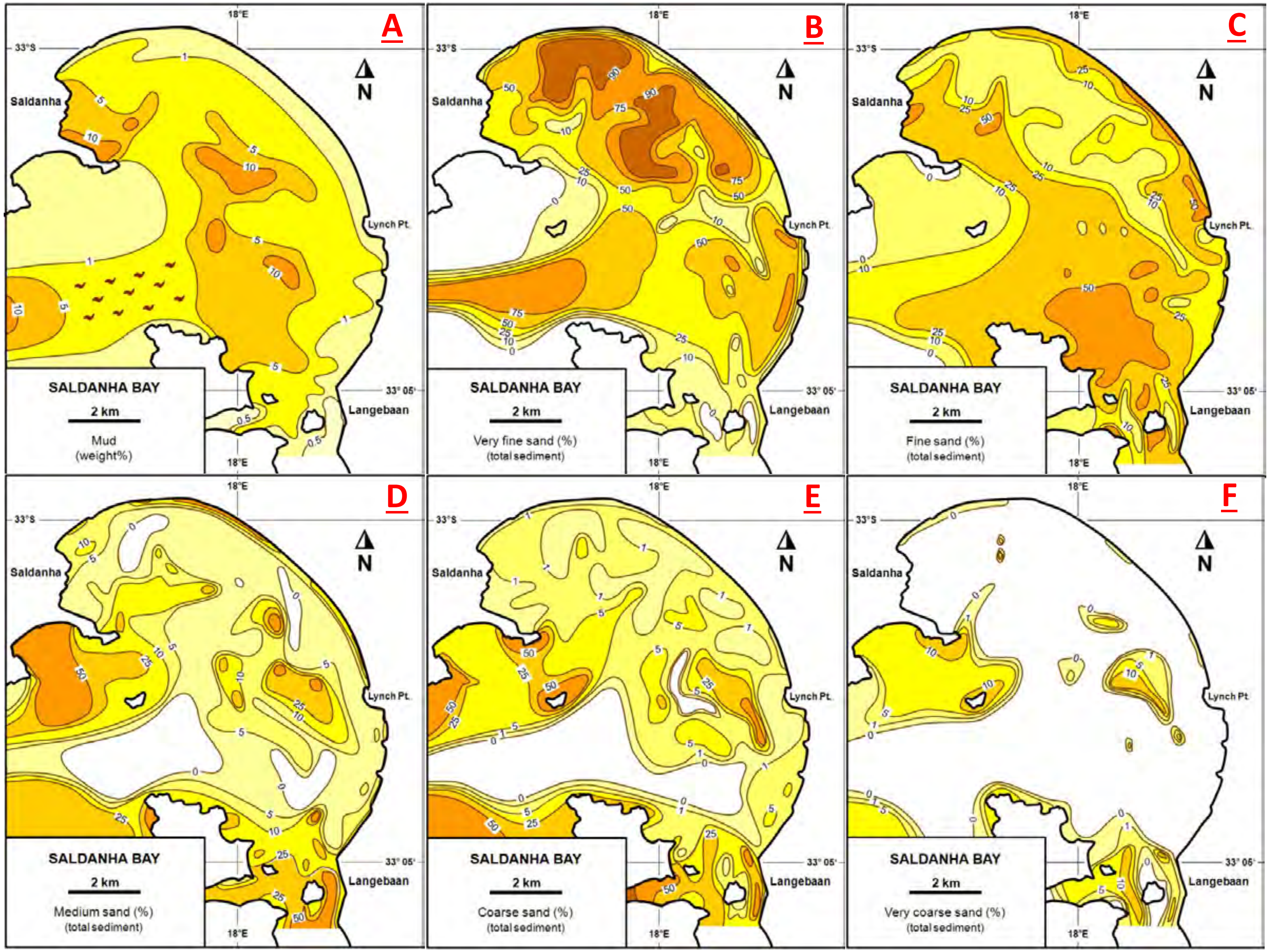


Figure 7.2 Distribution of different sediment types (% of total) in Saldanha Bay in 1975: (A) mud (<0.063 mm), (B) very fine sand (0.063-0.125 mm), (C) fine sand (0.125-0.25 mm), (D) medium sand (0.25-0.5 mm), (E) coarse sand (0.5-1.0 mm), (F) very coarse sand (1-2 mm). Source: Flemming (2015).

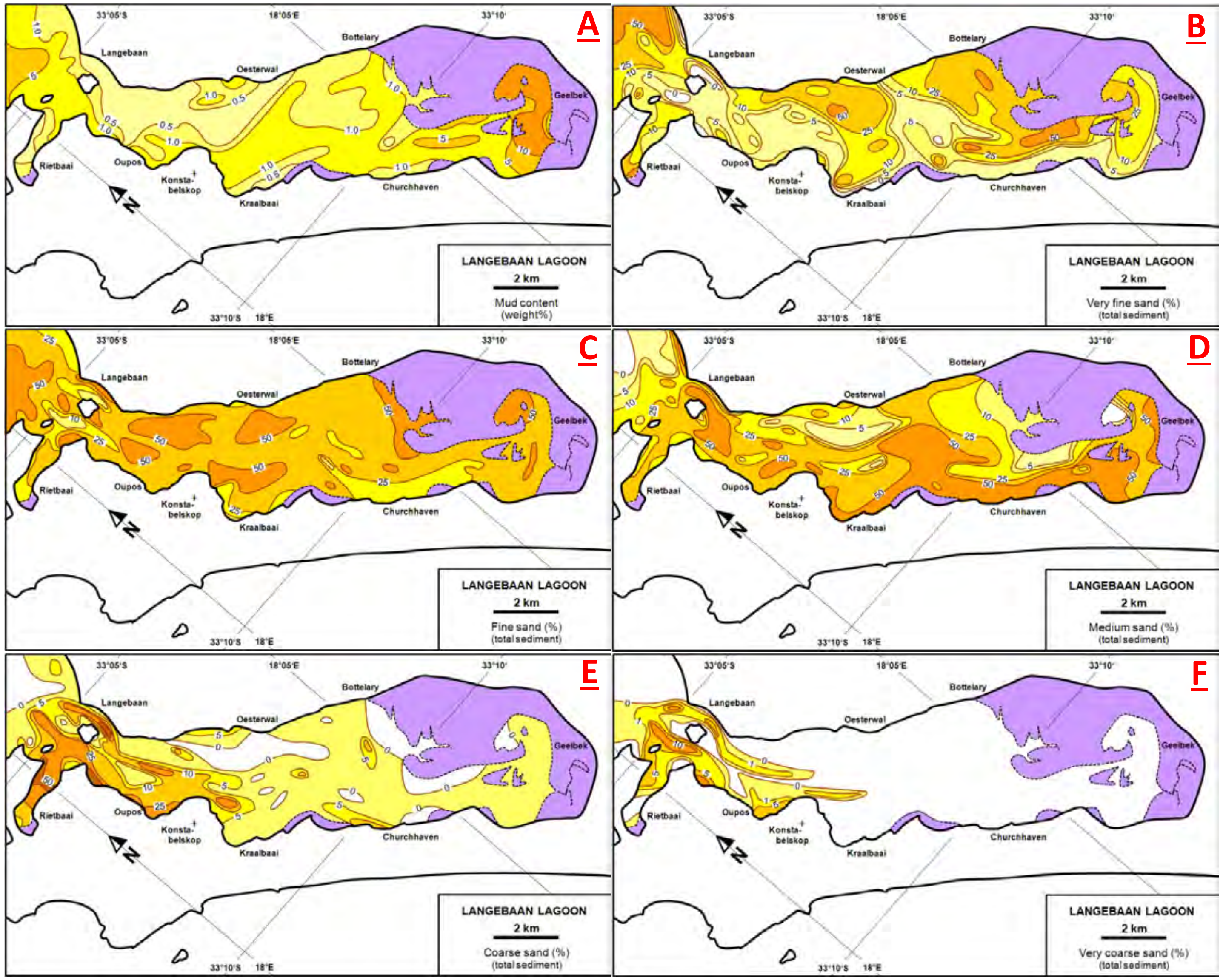


Figure 7.3 Distribution of different sediment types (% of total) in Langebaan Lagoon in 1975: (A) mud (<0.063 mm), (B), very fine sand (0.063-0.125 mm), (C) fine sand (0.125-0.25 mm), (D) medium sand (0.25-0.5 mm), (E) coarse sand (0.5-1.0 mm), (F) very coarse sand (1-2 mm). Source: Flemming (2015).

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 7.3).

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 7.5).

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 7.5, Figure 7.7) 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 7.5).

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 7.5). 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, Langebaan Lagoon and Elandsfontein in 2019 as part of the annual State of the Bay sampling programme (Figure 7.4). This

[10] Data for six key sites surrounding the iron ore terminal and in Small Bay are shown on **Error! Reference source not found.**. The reader i s referred to individual State of the Bay reports for each year for more detail on this.

included 10 sites in Small Bay, 9 in Big Bay and Langebaan Lagoon and 3 in Elandsfontein. 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 (1.42%), followed by Small Bay (1.19%) (Table 7.1). Currently, there is an overall decrease in mud percentage in both Small and Big Bay sites compared to 2017 results. No gravel (particles exceeding 2000 µm) was found across all sampling sites (Figure 7.5).

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 (ten levels: 2009, 2010, 2011, 2013, 2014, 2015, 2016, 2017, 2018 and 2019) 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_9 = 10.59, $p < 0.001$; Region: Pseudo-F_6 = 34.34, $p < 0.001$). However, there was no significant interaction between Region and Year (Region × Year: Pseudo-F_{25} = 1.07, $p > 0.05$) 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 7.6) 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 2009-2019. Sediments in Big Bay and Small Bay are mostly quite similar, but 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 7.6).

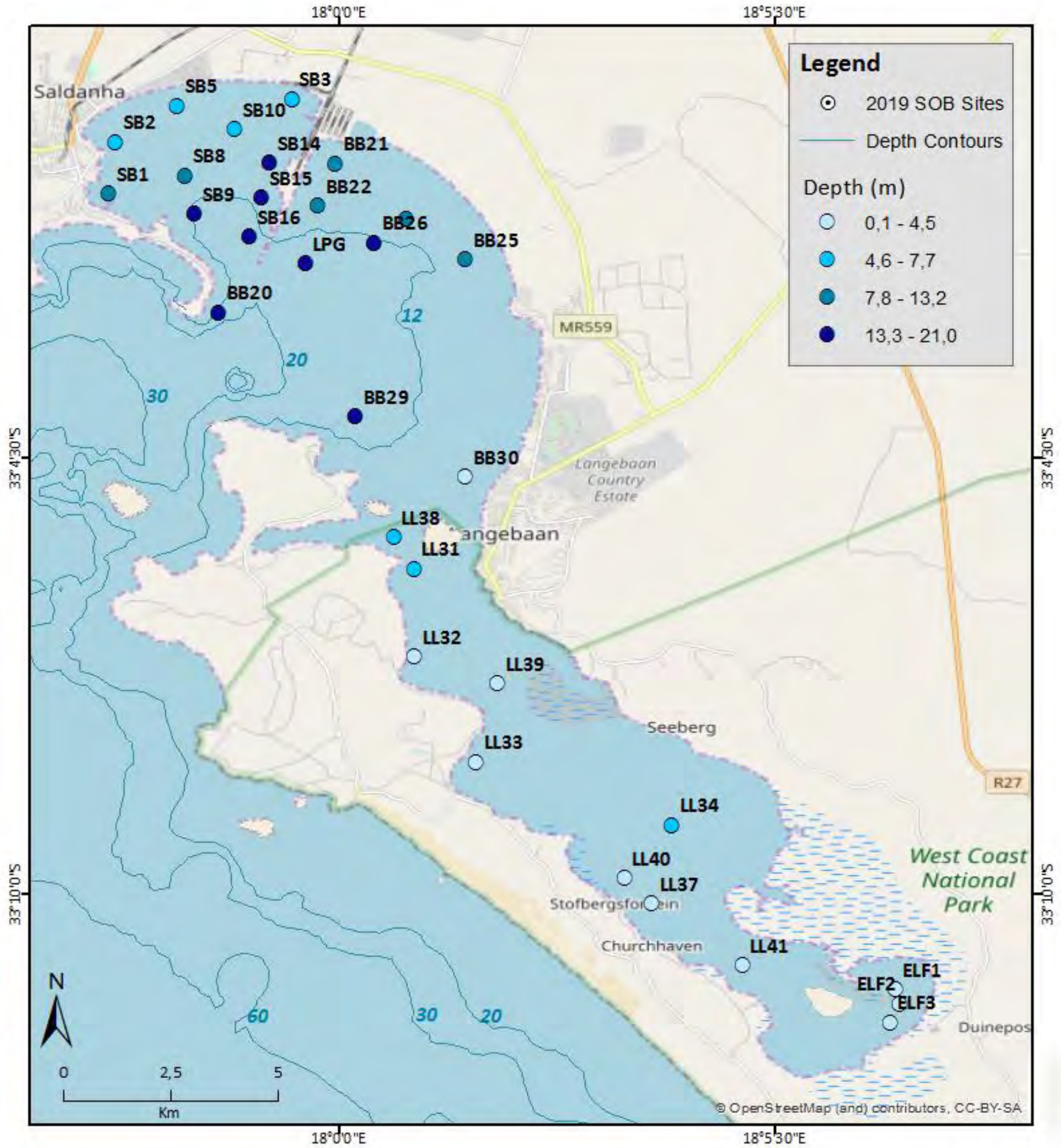


Figure 7.4 **Sediment sampling sites and respective depth ranges (m) in Saldanha Bay, Langebaan Lagoon and Elandsfontein for 2019.**

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 2019 (Particle size analysed by Scientific Services and TOC and TON analysed by the Council for Scientific and Industrial Research).

	Sample	Sand (%)	Mud (%)	TOC (%)	TON (%)	C:N
Small Bay	SB1	96.91	3.09	3.00	0.40	8.75
	SB2	99.72	0.28	0.29	0.06	5.58
	SB3	99.05	0.95	0.33	0.06	6.36
	SB5	99.97	0.03	0.10	0.03	4.01
	SB8	99.23	0.77	0.23	0.05	5.27
	SB9	98.72	1.28	0.57	0.12	5.56
	SB10	99.80	0.20	0.20	0.04	5.69
	SB14	97.88	2.12	1.74	0.26	7.79
	SB15	99.10	0.90	1.09	0.18	7.07
	SB16	97.71	2.29	0.55	0.10	6.41
	Average	**98.81**	**1.19**	**0.81**	**0.13**	**6.25**
Big Bay	BB20	99.72	0.28	0.37	0.08	5.38
	BB21	97.83	2.17	0.35	0.08	5.09
	BB22	98.51	1.49	0.35	0.07	5.80
	LPG	98.00	2.00	1.45	0.25	6.77
	BB24	98.08	1.92	0.30	0.06	5.74
	BB25	99.77	0.23	0.21	0.05	4.78
	BB26	96.78	3.22	0.60	0.11	6.35
	BB29	98.67	1.33	0.32	0.07	5.27
	BB30	99.89	0.11	0.07	0.03	2.68
	Average	**98.58**	**1.42**	**0.44**	**0.09**	**5.32**
Langebaan Lagoon	LL31	99.74	0.26	0.09	0.05	2.17
	LL32	99.89	0.11	0.05	0.06	0.93
	LL33	99.89	0.11	0.07	0.05	1.73
	LL34	99.64	0.36	0.12	0.07	2.05
	LL37	99.86	0.14	0.09	0.05	2.15
	LL38	99.09	0.91	0.19	0.08	2.76
	LL39	99.93	0.07	0.07	0.05	1.54
	LL40	99.96	0.04	0.11	0.05	2.57
	LL41	99.56	0.44	0.08	0.07	1.28
	Average	**99.73**	**0.27**	**0.10**	**0.06**	**1.91**
Elandsfontein	Eland 1	99.42	0.58	0.115	0.03	4.47
	Eland 2	99.49	0.51	0.16	0.04	4.67
	Eland 3	99.12	0.88	0.157	0.06	3.05
	Average	**99.34**	**0.66**	**0.14**	**0.04**	**4.06**

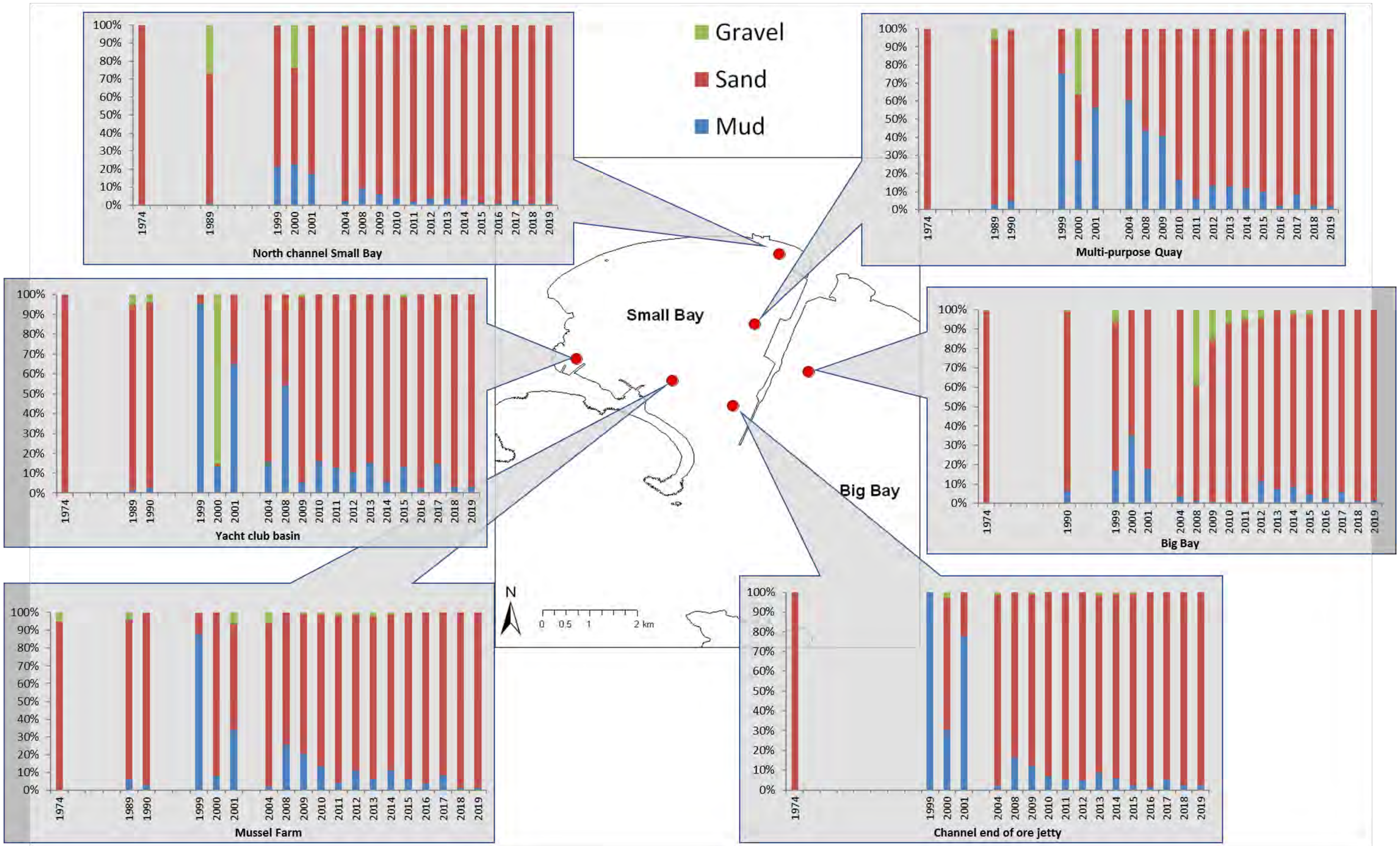


Figure 7.5 Particle size composition (percentage gravel. sand and mud) of sediments at six localities in the Small and Big Bay area of Saldanha Bay between 1974 and 2019. Data sources: 1974: Flemming (1977b). 1899-1990: Jackson & McGibbon (1991). 1999-2018: SBWQFT.

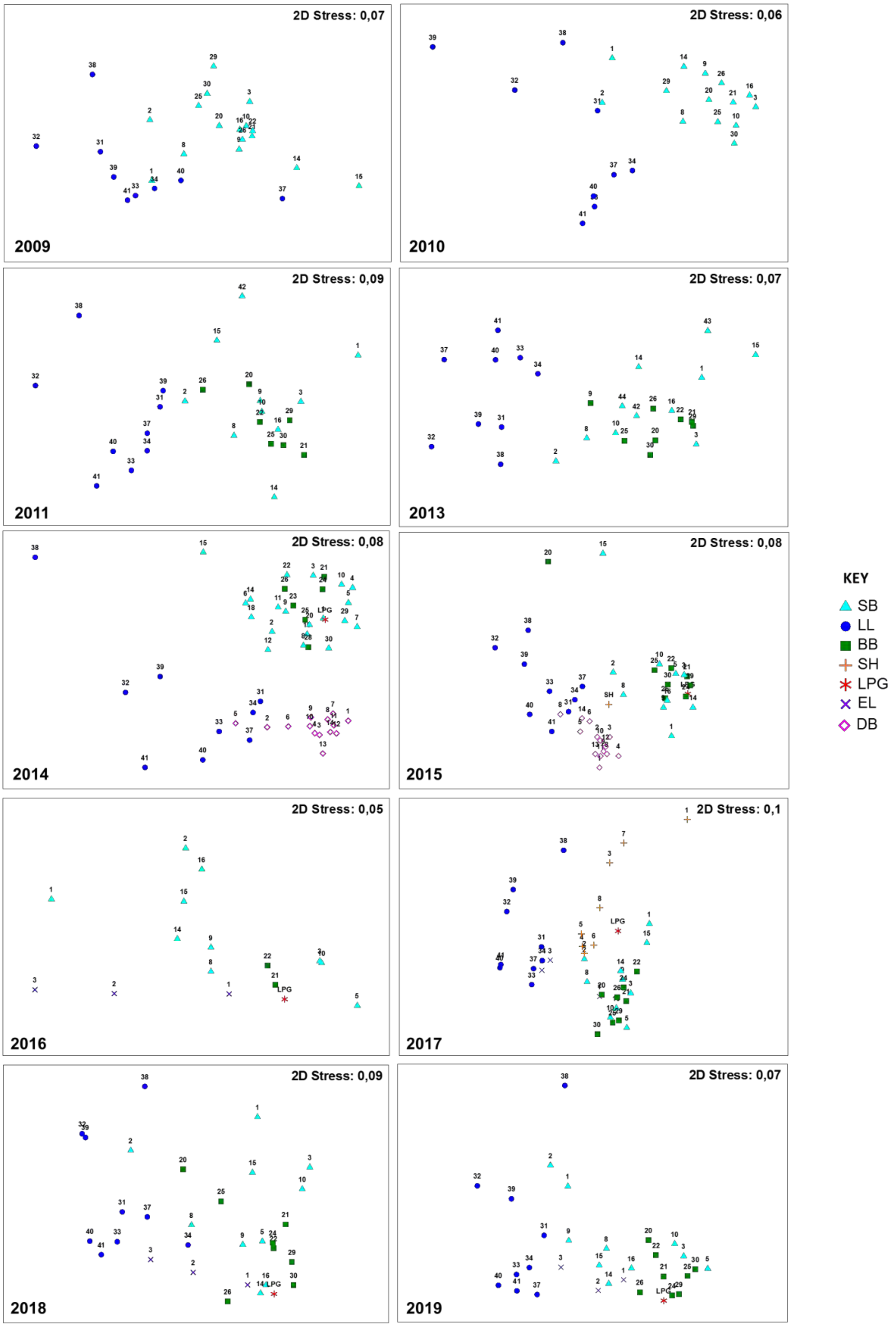


Figure 7.6 **MDS plots of particle size distribution (PSD) from samples collected at sites from Saldanha Bay. Langebaan Lagoon and Elandsfontein from 2009-2019. Each region (SB: Small Bay. LL: Langebaan Lagoon. BB: Big Bay. SH: Sea Harvest. LPG: Liquid Petroleum Gas. EL: Elandsfontein and DB: Danger 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 2019 (Figure 7.7).

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.

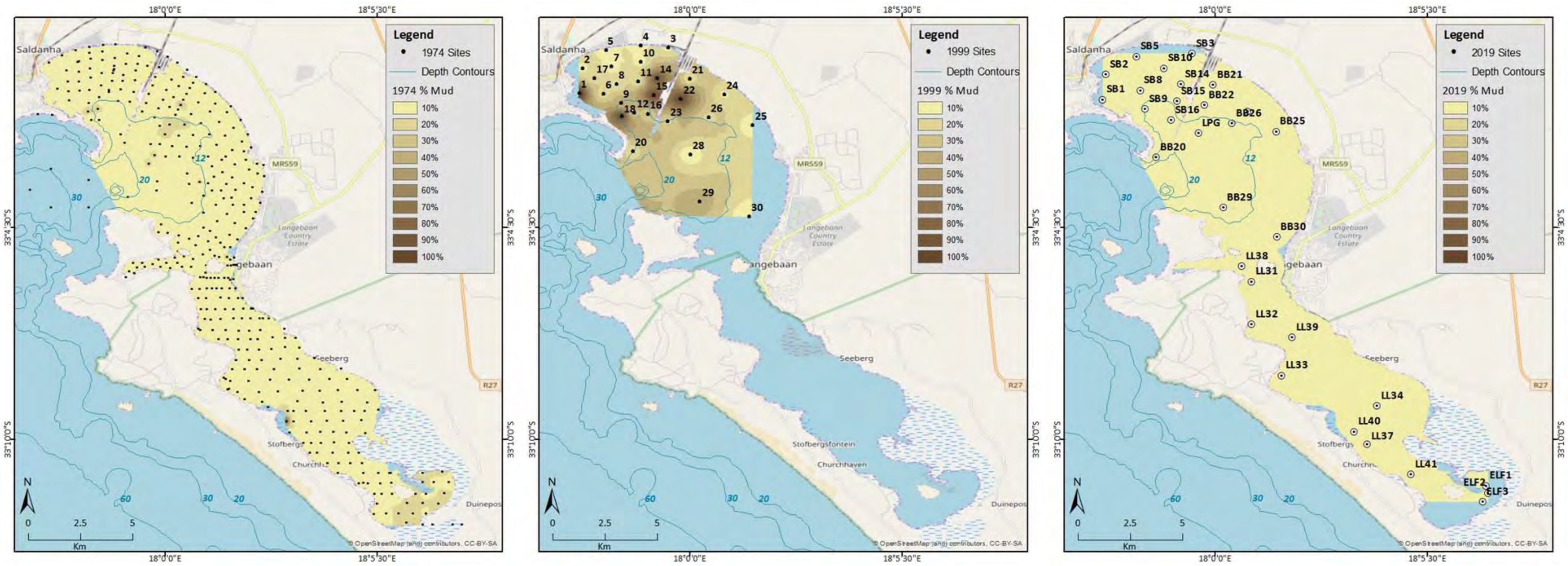


Figure 7.7 Change in the percentage mud in sediments in Saldanha Bay and Langebaan Lagoon between 1974 (left), 1999 (centre) and 2019 (right) survey results.

7.2 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 7.8 and Figure 7.9).

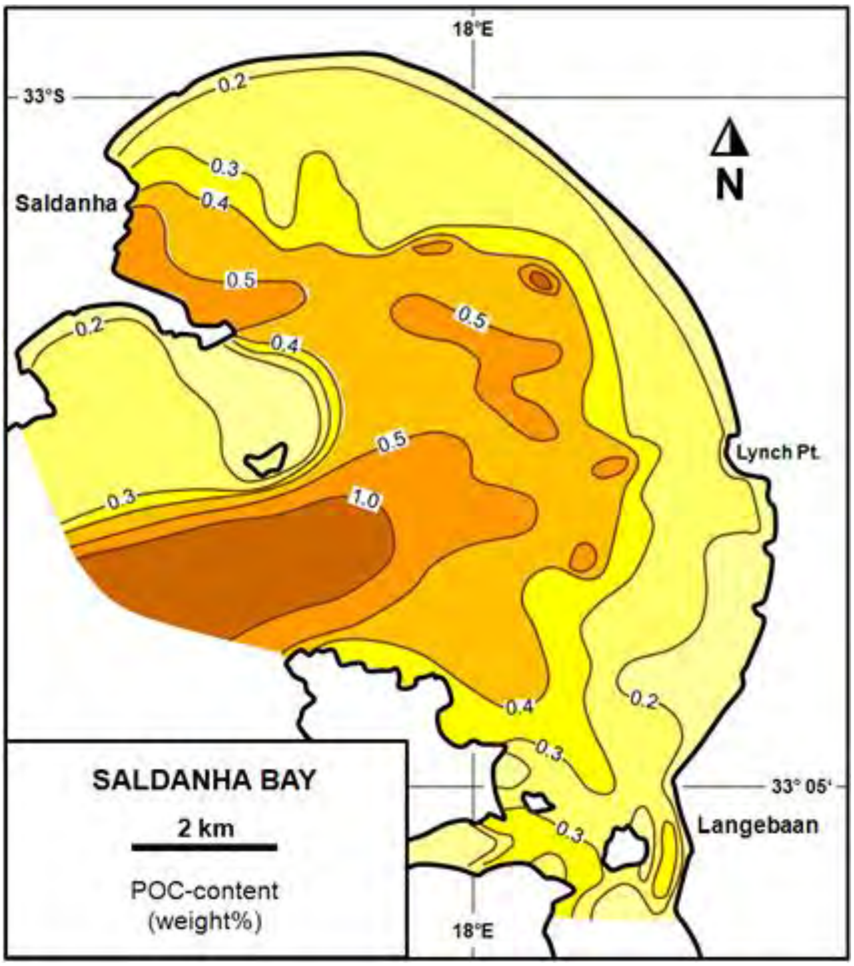


Figure 7.8 Levels of organic carbon in sediments Saldanha Bay in 1974. Source: Flemming (2015).

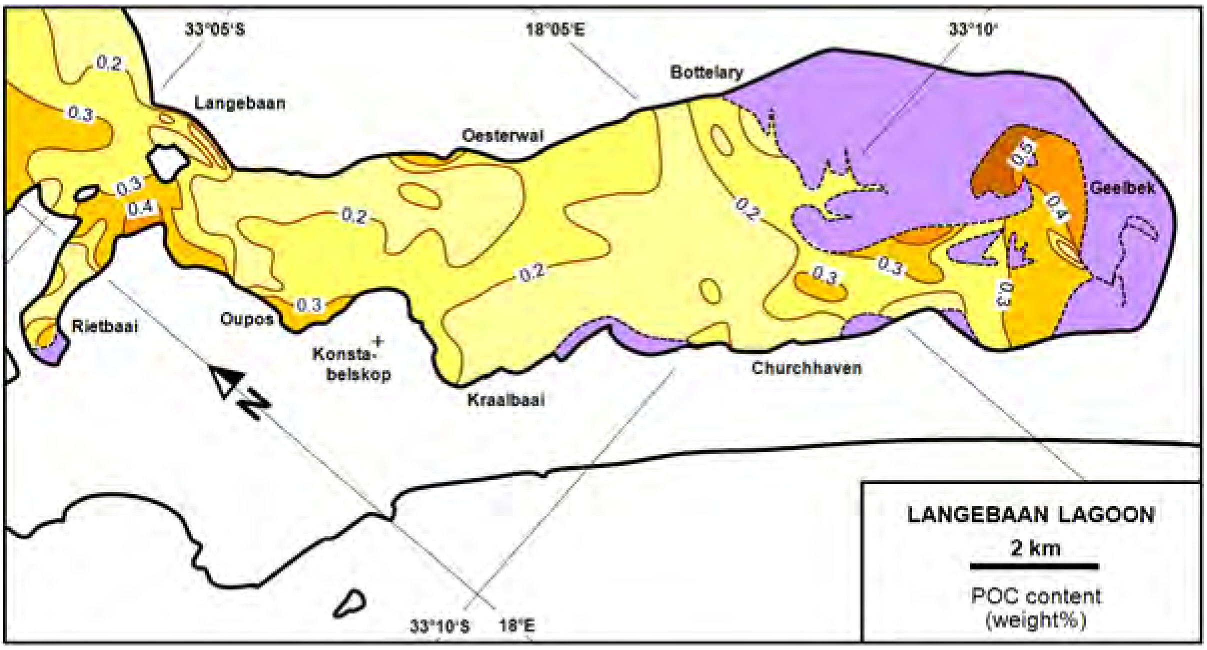


Figure 7.9 Levels of organic carbon in sediments in Langebaan Lagoon in 1974. Source: Flemming (2015).

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 7.11). By the time the next surveys had been undertaken in 1999 (CSIR 1999a, Figure 7.10, Figure 7.11) 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 7.11). 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 7.11. Data on the spatial distribution of TOC from 1999, 2018 and the most recent survey (2019) are shown in Figure 7.10. These data suggest that TOC levels have remained high between 1999 and 2018 with highest levels being recorded at the Yacht Club Basin (SB1) and Multi-Purpose Terminal (SB14). However, it is noticeable that levels have dropped slightly in the recent 2019 survey. The latter patterns are also evident on the spatial variation of TON within Saldanha Bay (Figure 7.10).

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 7.11). Levels were slightly or even considerably elevated at all sites in 2000 and 2001 (Figure 7.11). 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 7.11, see also the 2004 State of the Bay report). Results from the State of the Bay surveys conducted between 2008 and 2019 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 7.11). There was a clear increase in TON in 2018 compared to 2017 for Big Bay, but levels dropped again in 2019 (Figure 7.11). Spatial variation in TON levels recorded in the sediments in Saldanha Bay and Langebaan Lagoon in 1999, 2018 and 2019 are presented in Figure 7.10. Once again, concentrations are generally higher in Small Bay; particularly at the Yacht Club Basin and along the Iron Ore Terminal. However, 2019 TON concentrations were much lower than the 2018 survey; mirroring the patterns observed for TOC levels in the Bay (Figure 7.10). 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.

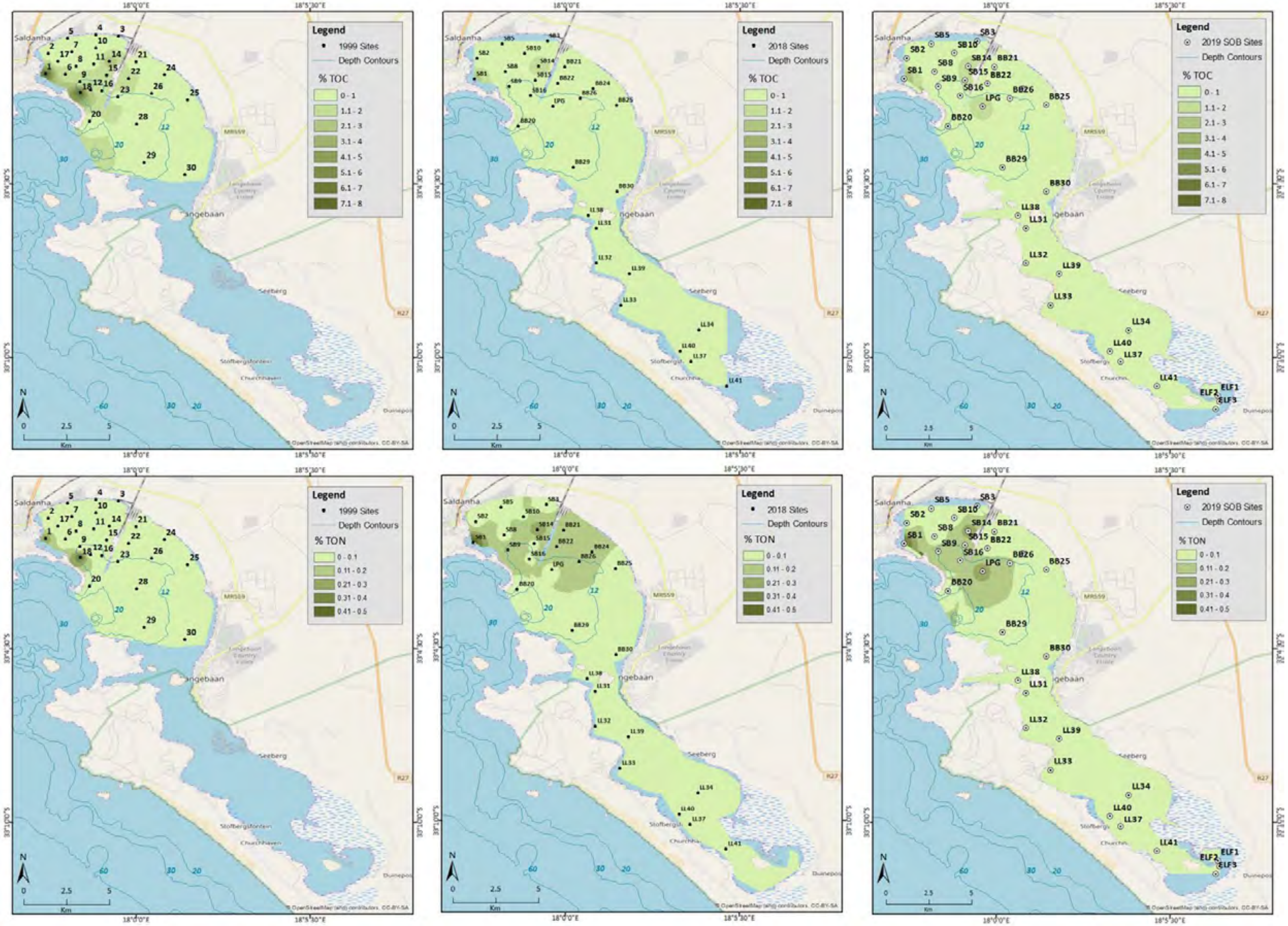


Figure 7.10 Total organic carbon (TOC) and total organic nitrogen (TON) levels in sediments in Saldanha Bay in 1999 (Source: CSIR 1999a), 2018 and 2019.

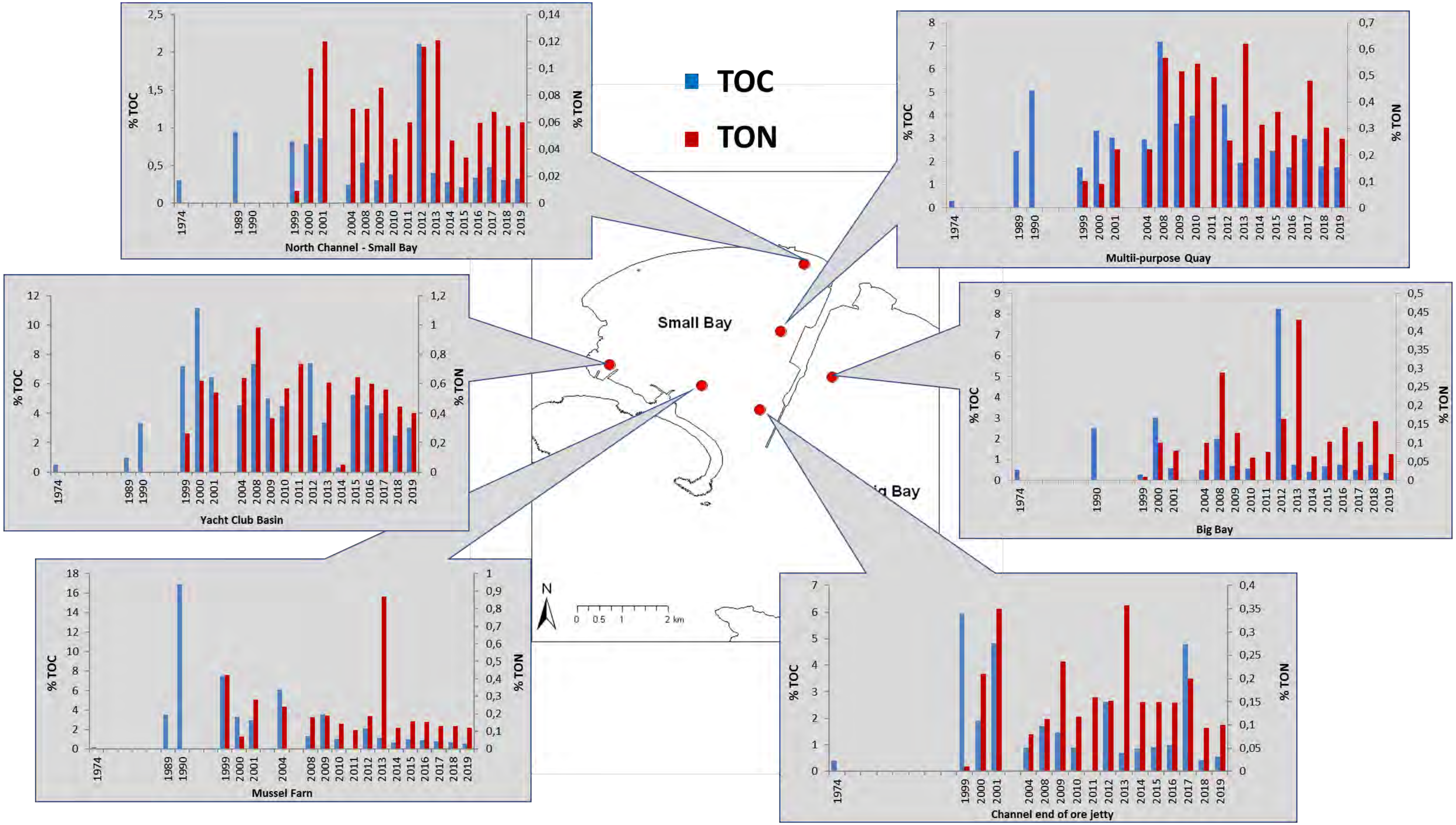


Figure 7.11 Total organic carbon and nitrogen in sediments of Saldanha Bay at six locations between 1974 and 2019. Data sources: 1974: Flemming (1977b), 1899-1990: Jackson & McGibbon (1991), 1999-2018: SBWQFT.

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 2017-2018 were highly variable. The majority of sites in 2017 were within the expected range of marine production; bar a few sites near to the Iron Ore Terminal and at the entrance to Big Bay which 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 7.12). The 2018 survey results revealed that only the Multipurpose Terminal was above the expected range; whilst majority of the sites within Small Bay as well as Big Bay and Langebaan Lagoon were below the range of marine production (Figure 7.12). The latter pattern followed through to the recent 2019 survey; however, the multipurpose ore terminal now fell within the C:N range and the Yacht Club Basin (SB1) is currently above the expected range (Figure 7.12).

There are two possible reasons for elevated C:N ratios observed in 2017; 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. 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 2017; 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 2017 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 rations.

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. Given the high inter-annual variability in the C:N ratios, interpretation that focuses on the outliers in any given year (e.g. Yacht Club Basin) is probably more informative than a temporal analysis.

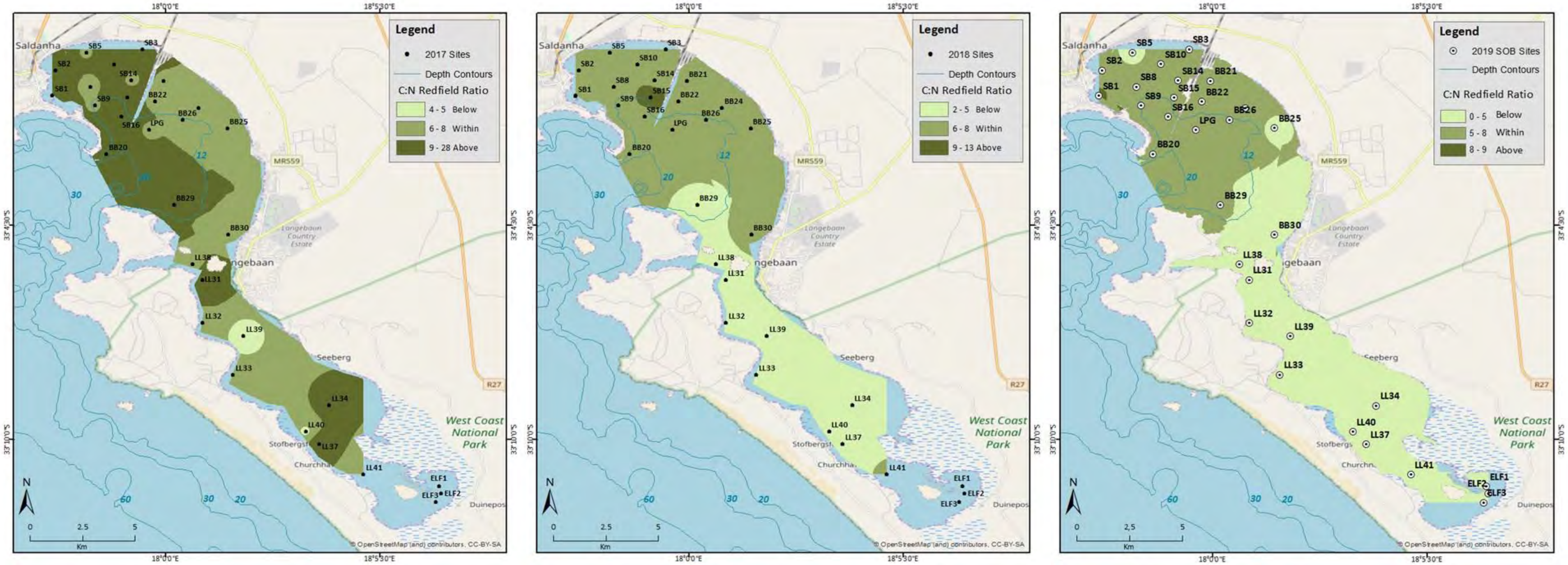


Figure 7.12 C:N ratios at different sites surveyed in Saldanha Bay, Langebaan Lagoon and Elandsfontein in 2017, 2018 & 2019 (dark green = exceeds the range expected for marine production; mild green = within the range expected for marine production and light green = below range expected for marine production).

7.3 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.

Table 7.3 Concentrations (mg/kg) of metals in sediments collected from Saldanha Bay in 2019. 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	8351	12129	3.44	49.73	13.33	17.397	68.20
	SB2	1543	3308	1.41	3.07	2.54	6.375	35.25
	SB3	1627	3042	<1.0	4.91	2.17	14.144	42.40
	SB5	648	1461	<1.0	2.08	1.07	3.608	21.71
	SB8	1594	3105	<1.0	3.23	2.56	2.63	28.97
	SB9	2577	5578	0.99	4.74	4.01	6.73	42.70
	SB10	1011	2196	<1.0	2.54	1.16	4.89	26.81
	SB14	5506	7957	1.44	15.72	6.64	38.83	53.33
	SB15	6226	10259	1.56	14.31	8.33	29.71	65.74
	SB16	2657	4638	<1.0	5.09	4.13	2.86	33.00
Big Bay	BB20	1265	1896	<1.0	2.35	1.25	<1.0	19.64
	BB21	2215	3831	<1.0	3.29	2.59	2.80	41.45
	BB22	2706	5611	1.06	4.10	3.12	4.76	57.59
	LPG	3574	5993	1.28	5.57	5.17	2.51	46.15
	BB24	3326	4247	1.20	3.66	4.16	1.81	38.96
	BB25	990	1747	<1.0	1.81	1.69	<1.0	24.90
	BB26	2466	4701	1.02	3.43	2.90	5.03	46.66
	BB29	1996	2884	<1.0	3.07	2.52	<1.0	22.73
	BB30	568	1074	<1.0	1.43	<1.0	1.07	15.65
Langebaan Lagoon	LL31	1476	2658	1.04	1.83	1.58	1.88	22.00
	LL32	891	2652	<1.0	1.02	<1.0	2.10	13.98
	LL33	730	1139	<1.0	1.03	<1.0	<1.0	9.77
	LL34	1368	1872	<1.0	1.58	1.43	<1.0	16.35
	LL37	851	1296	<1.0	1.34	1.25	<1.0	11.12
	LL38	3921	5594	1.50	3.48	4.97	1.31	45.26
	LL39	846	1655	<1.0	1.32	<1.0	<1.0	15.52
	LL40	705	967	<1.0	1.60	<1.0	<1.0	18.35
	LL41	1456	1995	<1.0	1.56	1.78	<1.0	12.55
Elandsfontein	Eland 1	1845	2631	<1.0	2.03	2.28	2.82	21.50
	Eland 2	-	-	-	-	-	-	-
	Eland 3	-	-	-	-	-	-	-

In 2019, cadmium, copper and manganese concentrations were highest and exceeded ERL guidelines in the vicinity of the Yacht Club Basin (Table 7.3). In addition, cadmium concentrations also exceeded guideline levels at the Multi-Purpose Terminal (SB14) and the LPG site in Big Bay. 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 noticeably elevated at the Yacht Club Basin and adjacent to the Multi-Purpose Terminal. 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.

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.

7.3.1 Spatial variation in trace metals levels in Saldanha Bay

7.3.1.1 Cadmium

Sediments from sites located alongside the Iron Ore Terminal within Small Bay displayed low cadmium concentrations; whereas the area within the vicinity of the Yacht Club Basin revealed the highest concentration of cadmium (Figure 7.13; 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).

7.3.1.2 Copper

Copper concentrations were highest along the Iron Ore Terminal and near the Saldanha Bay Yacht Club within Small Bay (Figure 7.13 & 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 2019 (Table 7.4).

7.3.1.3 Nickel

Nickel values measured in 2019 were elevated at the yacht club and alongside the iron ore terminal within Small Bay (Figure 7.13 & 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.

7.3.1.4 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 7.13 & 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 (48.54), however, the concentration of lead was below recommended ERL toxicity limits (Table 7.4). 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.

7.3.1.5 Manganese

Manganese concentrations were highest near the Yacht Club Basin and along the iron ore terminal within Small Bay (Figure 7.13 & 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 (Chapter 3, Section 3.3).

Table 7.4 **Enrichment factors for Cadmium. Copper and Lead in sediments collected from Saldanha Bay in 2019 relative to sediments from 1980. ND indicates no data.**

	Sample	Cd	Cu	Pb
	1980 average	**0.075**	**0.41**	**0.8**
Small Bay	SB1	45.87	121.28	21.75
	SB2	18.73	7.50	7.97
	SB3	ND	11.97	17.68
	SB5	ND	5.08	4.51
	SB8	ND	7.88	3.29
	SB9	13.21	11.57	8.41
	SB10	ND	6.18	6.11
	SB14	19.20	38.35	48.54
	SB15	20.85	34.90	37.13
	SB16	ND	12.41	3.58
Big Bay	BB20	ND	5.74	ND
	BB21	ND	8.03	3.50
	BB22	14.09	10.01	5.95
	LPG1	17.04	13.59	3.14
	BB24	15.93	8.93	2.26
	BB25	ND	4.43	ND
	BB26	13.53	8.37	6.29
	BB29	ND	7.48	ND
	BB30	ND	3.49	1.34

Table 7.5 **Normalized values for Cadmium. Copper. Nickel. Lead and Manganese in sediments collected from Saldanha Bay and Langebaan Lagoon in 2019. ND indicates no data.**

	Sample	Cd:Al	Cu:Al	Ni:Al	Pb:Al	Mn: Al
Small Bay	SB1	4.12	14.46	15.97	20.83	81.66
	SB2	9.10	2.19	16.49	41.30	228.39
	SB3	ND	ND	13.33	86.91	260.54
	SB5	ND	ND	16.56	55.70	335.22
	SB8	ND	ND	16.07	16.53	181.77
	SB9	3.85	4.79	15.57	26.10	165.69
	SB10	ND	ND	11.42	48.37	265.14
	SB14	2.62	10.92	12.07	70.52	96.86
	SB15	2.51	9.15	13.38	47.71	105.59
	SB16	ND	ND	15.54	10.77	124.19
Big Bay	BB20	ND	ND	9.89	ND	155.27
	BB21	ND	ND	11.67	12.63	187.12
	BB22	3.91	3.88	11.52	17.59	212.82
	LPG	3.58	4.36	14.47	7.03	129.13
	BB24	3.59	3.06	12.50	5.44	117.15
	BB25	ND	ND	17.04	ND	251.55
	BB26	4.12	3.38	11.75	20.40	189.23
	BB29	ND	ND	12.64	ND	113.86
	BB30	ND	ND	ND	18.81	275.66
Langebaan Lagoon	LL31	7.01	1.77	10.70	12.76	149.06
	LL32	ND	ND	ND	23.56	156.86
	LL33	ND	ND	ND	ND	133.85
	LL34	ND	ND	10.48	ND	119.47
	LL37	ND	ND	14.66	ND	130.57
	LL38	3.82	2.32	12.68	3.34	115.44
	LL39	ND	ND	ND	ND	183.42
	LL40	ND	ND	ND	ND	260.20
	LL41	ND	ND	12.21	ND	86.17

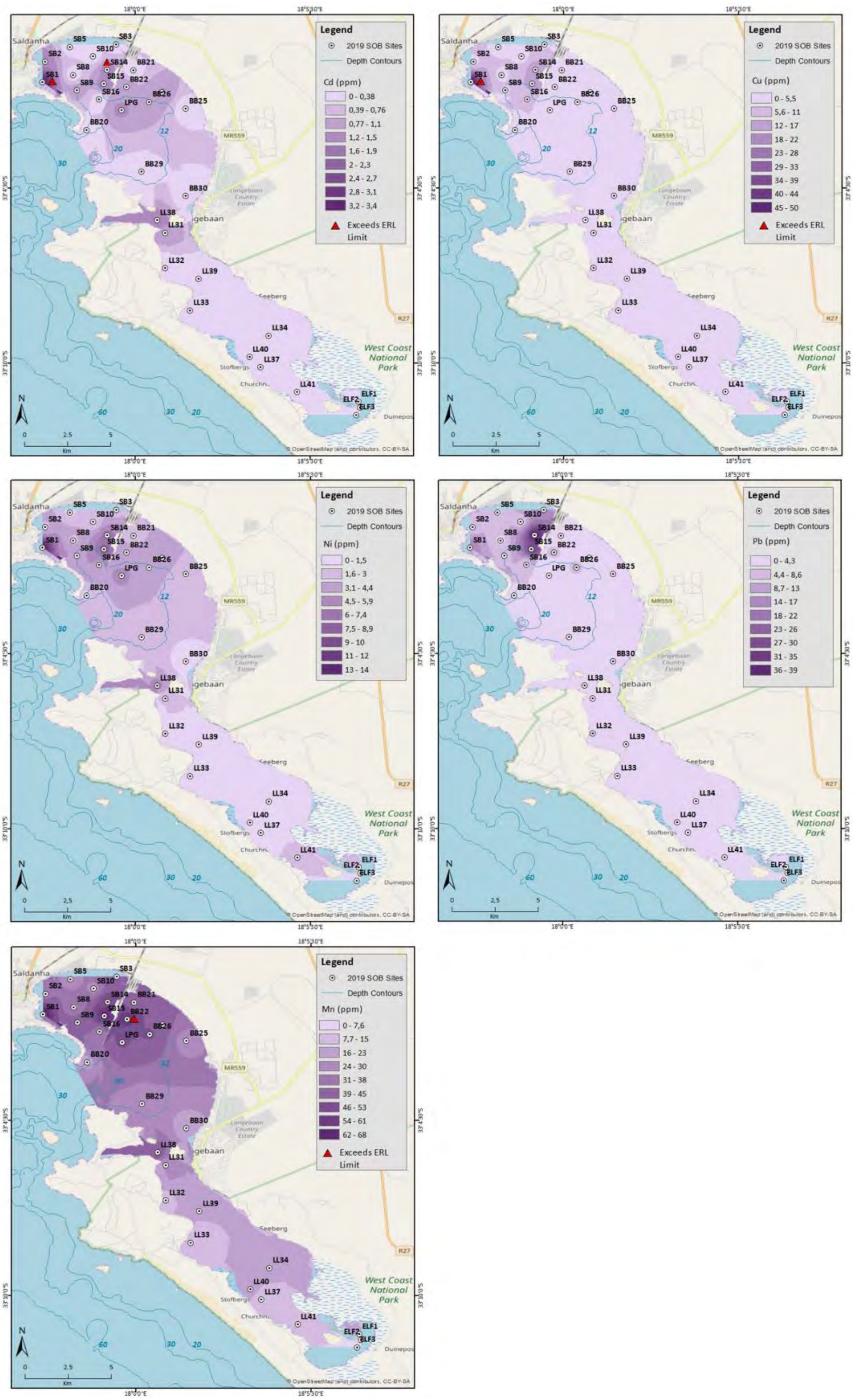


Figure 7.13 **Spatial interpolation of cadmium (Cd), copper (Cu), nickel (Ni), lead (Pb) and manganese (Mn) values measured in sediments in Saldanha Bay in 2019. Red triangles indicate sites that exceed the ERL limit.**

7.3.2 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.

7.3.2.1 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 7.14). 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. Cadmium concentrations at all other sites have remained low in recent years, however, have greatly elevated in the 2019 survey especially the site in Big Bay which is currently above the ERL toxicity threshold (Figure 7.14). The exceptions were at the Channel end ore jetty and the north channel sites in Small Bay where cadmium concentrations were very low.

7.3.2.2 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 7.15). 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 nine years. In the recent 2019 survey, there has been an elevated increased in copper concentrations across all sites, of which some are subtle (Figure 7.15).

7.3.2.3 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 7.16). 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, all six localities had an increase in nickel concentration (Figure 7.16).

7.3.2.4 Lead

The concentration of lead peaked and exceeded the ERL threshold at the Yacht Club Basin and Mussel farm site in 1999 (Figure 7.17). 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 2019 survey indicated a decline in lead concentrations at majority of the localities (Figure 7.17).

7.3.2.5 Manganese

The temporal variation in manganese concentrations in sediments around the ore terminal in Saldanha Bay is shown in Figure 7.18. 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 (Figure 7.18).

7.3.2.6 Iron

The temporal variation in the concentration of iron in sediments around the ore terminal in Saldanha Bay is shown in Figure 7.19. 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 2019 (Figure 7.19).

However, the concentration of iron at SB15 has fluctuated dramatically since 2012; but has shown an overall decrease in the last seven 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. 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.

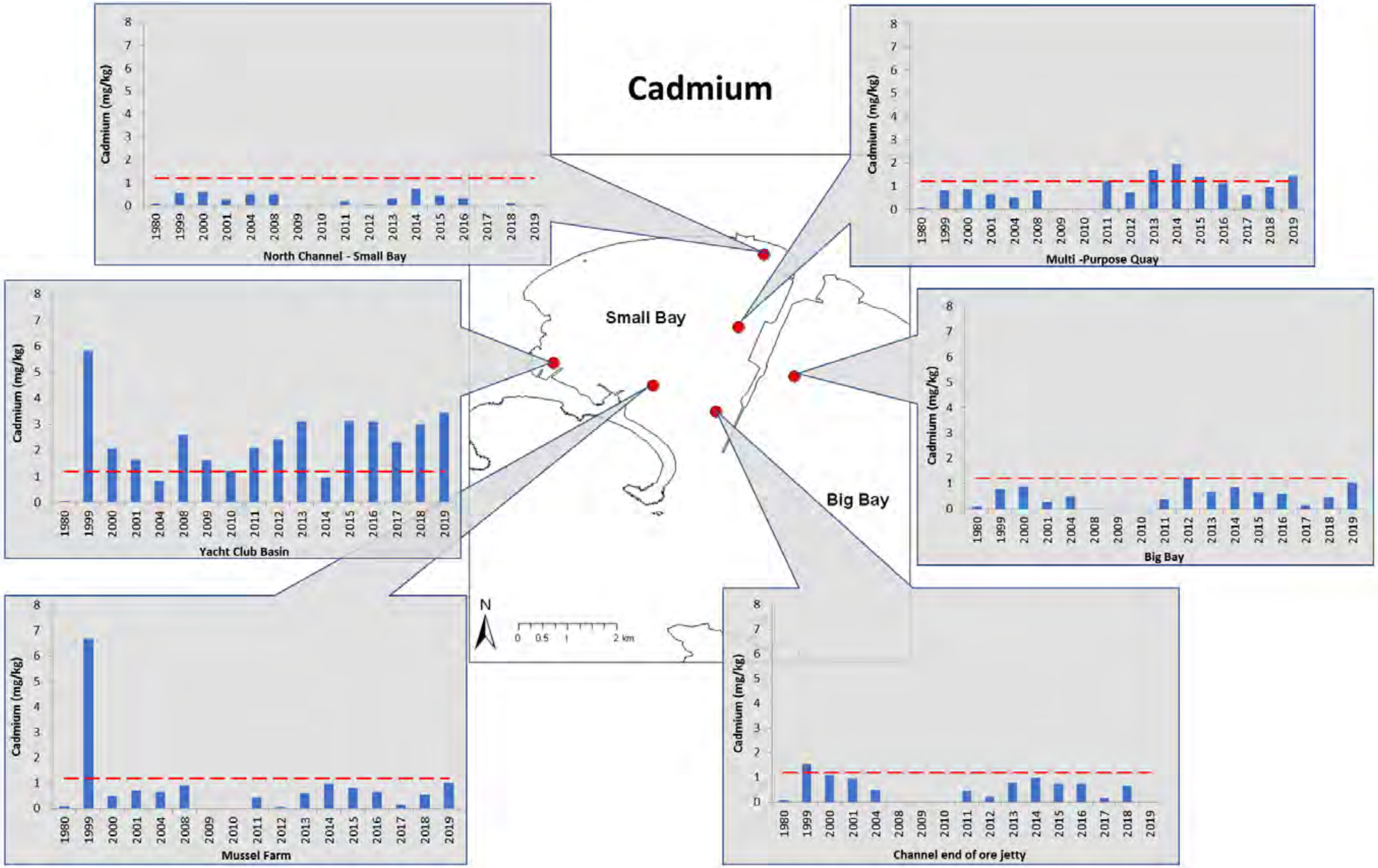


Figure 7.14 **Concentrations of Cadmium (Cd) in mg/kg recorded at six sites in Saldanha Bay between 1980 and 2019. Dotted lines indicate Effects Range Low values for sediments.**

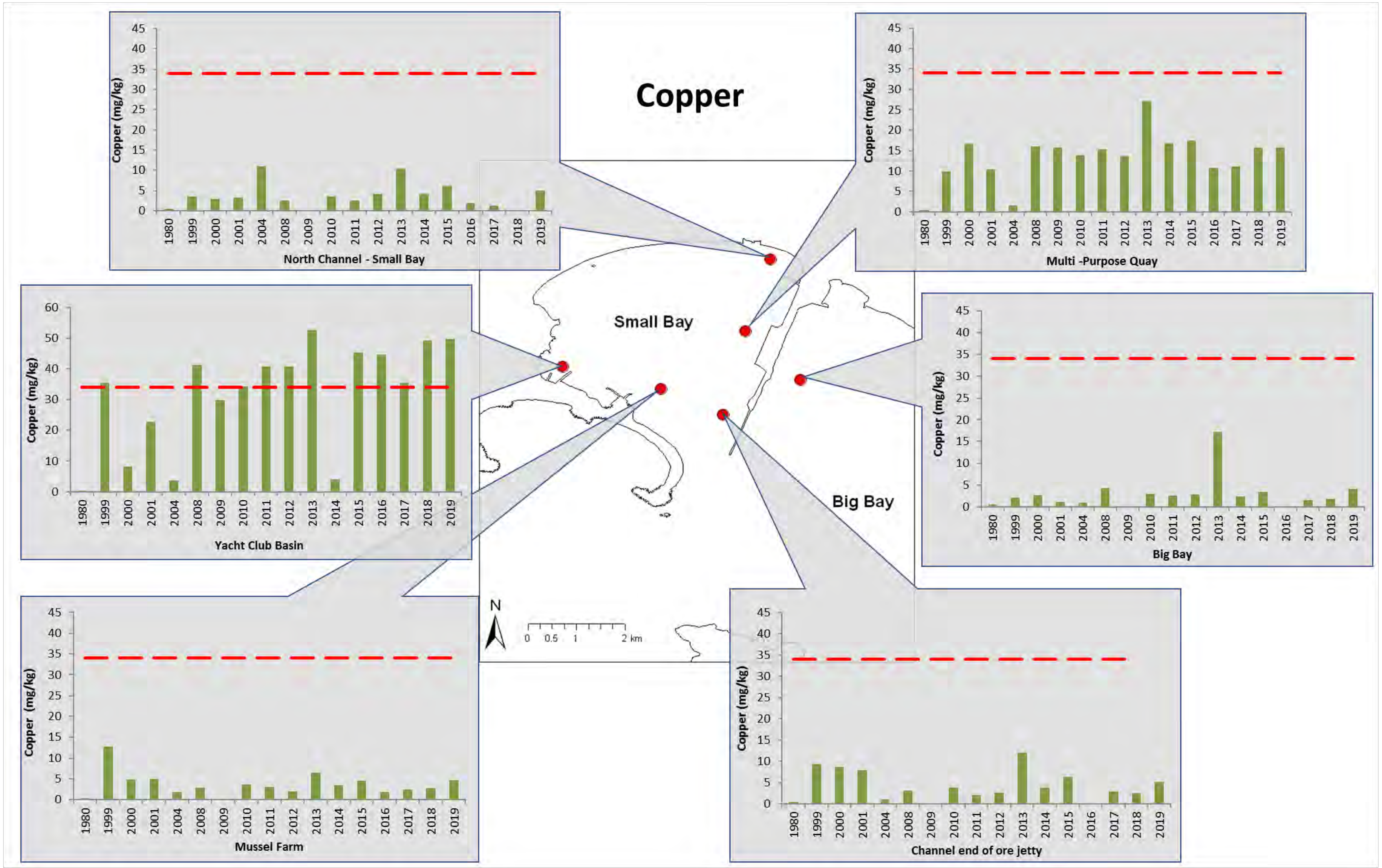


Figure 7.15 Concentrations of Copper (Cu) in mg/kg recorded at six sites in Saldanha Bay between 1980 and 2019. Dotted lines indicate Effects Range Low values for sediments.

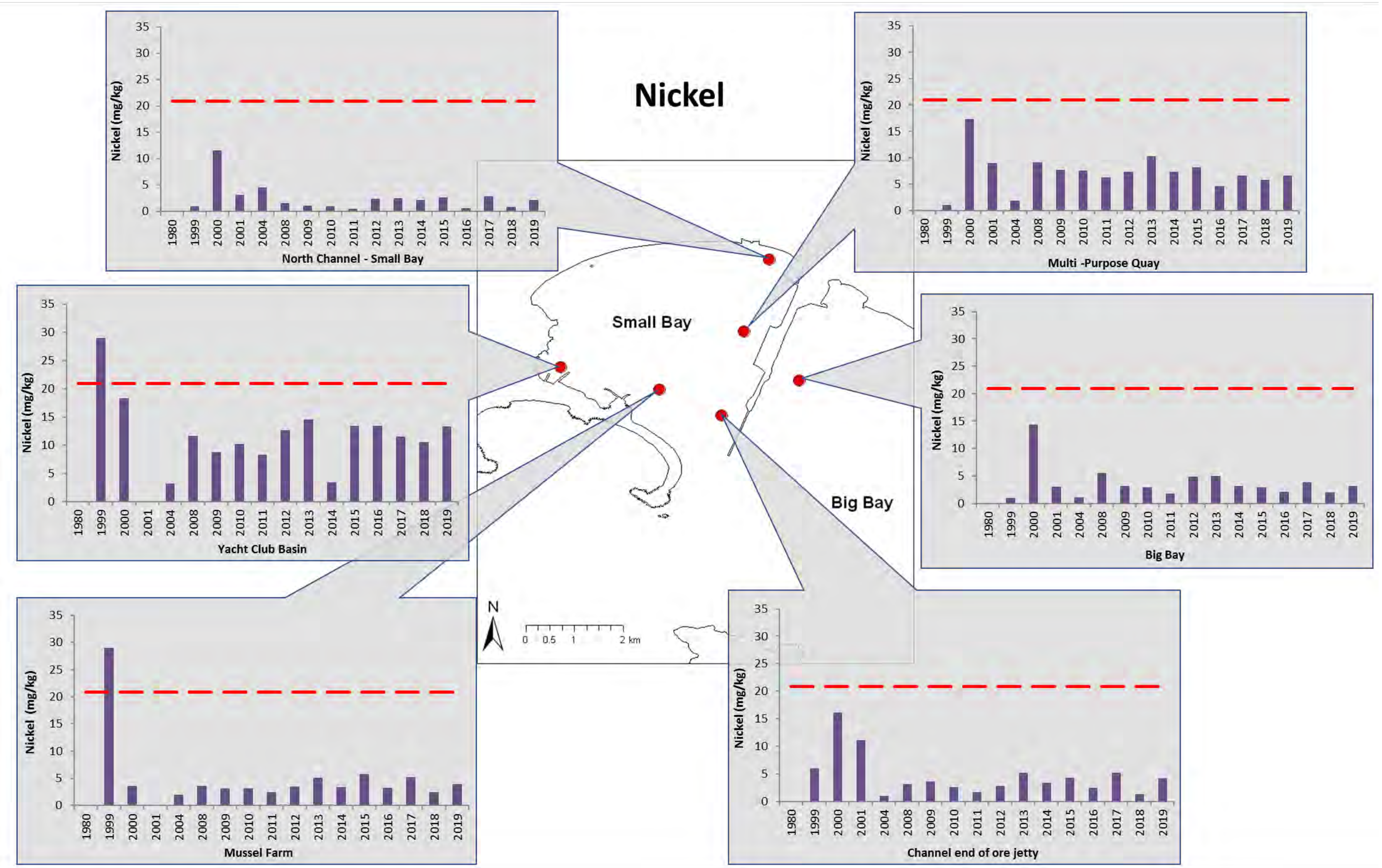


Figure 7.16 Concentrations of Nickel (Ni) in mg/kg recorded at six sites in Saldanha Bay between 1980 and 2019. Dotted lines indicate Effects Range Low values for sediments.

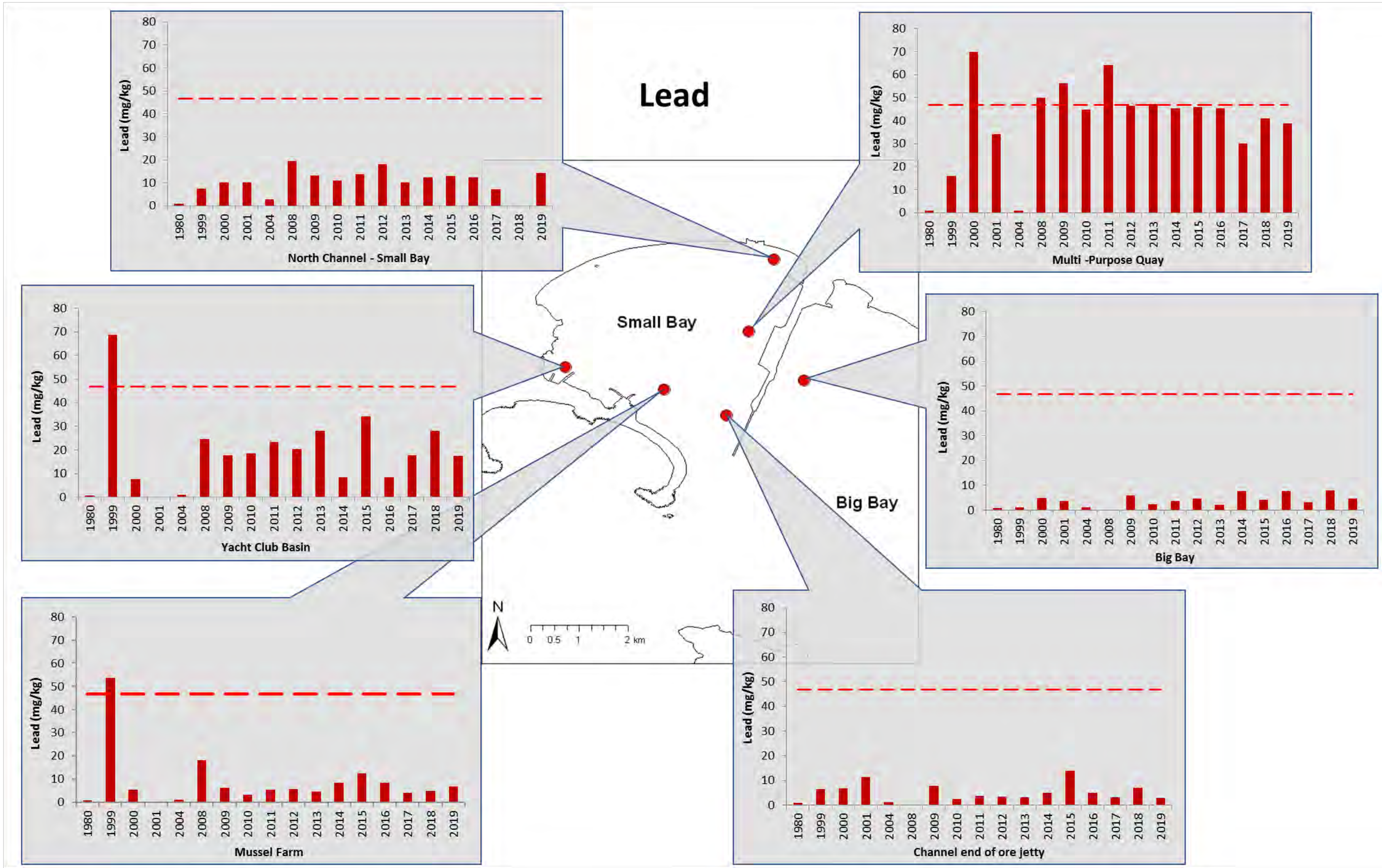


Figure 7.17 Concentrations of Lead (Pb) in mg/kg recorded at six sites in Saldanha Bay between 1980 and 2019. Dotted lines indicate Effects Range Low values for sediments.

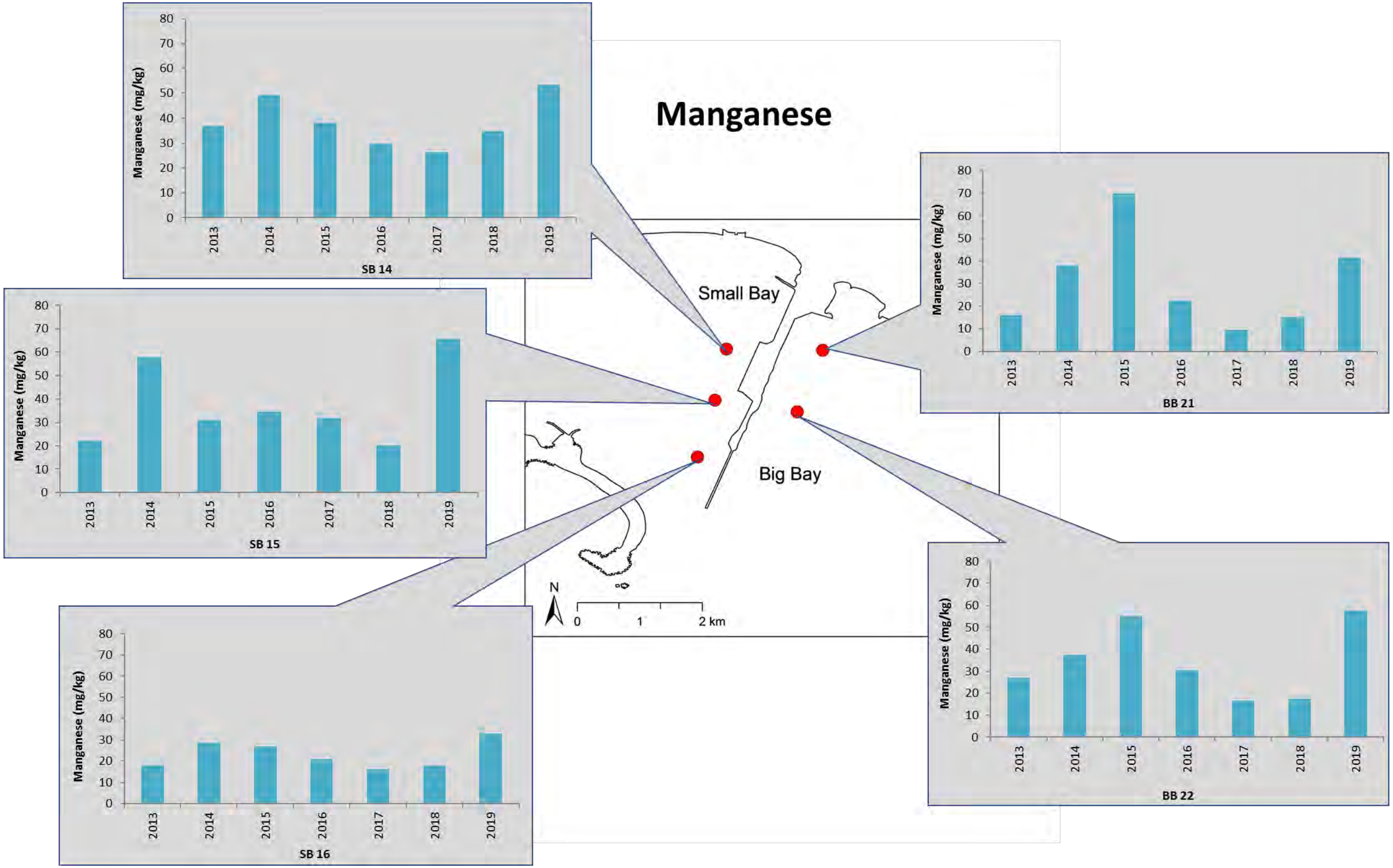


Figure 7.18 Concentration of manganese (Mn) in mg/kg recorded at five sites in Saldanha Bay between 2013 and 2019.

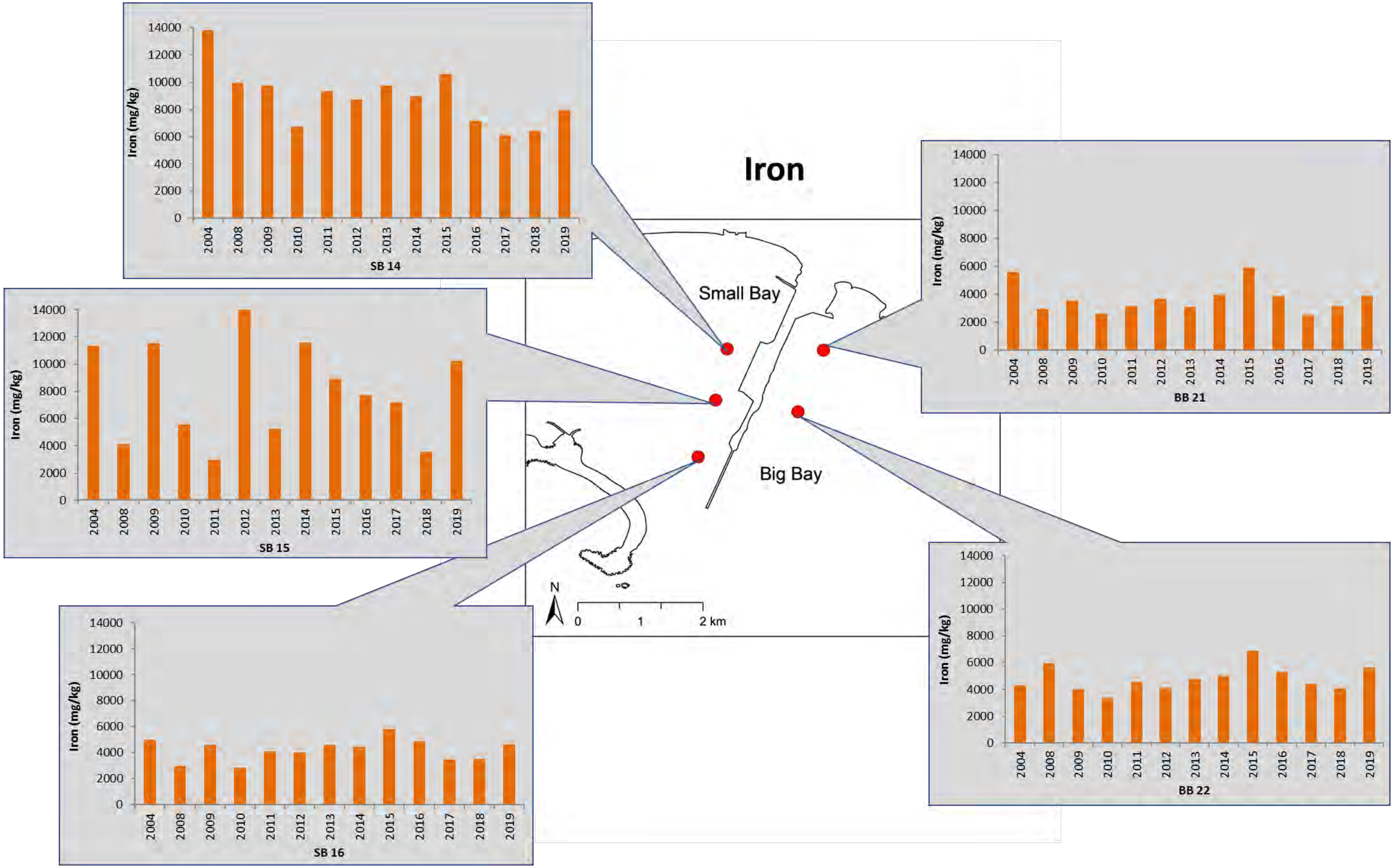


Figure 7.19 Concentrations of Iron (Fe) in mg/kg recorded at five sites in Saldanha Bay between 2004 and 2019.

7.4 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 2011-2019 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
SB14	<20	34	130	19	<38	<38	<38	<38	<38
SB15	<20	35	ND	53	<38	<38	<38	<38	<38
SB16	<20	24	28	14 649	<38	<38	<38	<38	<38
BB21	<20	20	32	20	<38	<38	<38	<38	<38
BB22	<20	17	27	<0.2	<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 2019 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 2019.

Hydrocarbon (mg/kg)	**ERL***	**ERM****	**SB14**	**SB15**	**SB16**	**BB21**	**SB22**
Acenaphthene	**0.016**	**0.5**	<0.002	<0.002	<0.002	<0.002	<0.002
Acenaphthylene	**0.044**	**0.64**	<0.002	<0.002	<0.002	<0.002	<0.002
Anthracene	**0.0853**	**1.1**	<0.002	<0.002	<0.002	<0.002	<0.002
Benzo(a) anthracene	**0.261**	**1.6**	<0.002	<0.002	<0.002	<0.002	<0.002
Benzo(a) pyrene	**0.43**	**1.6**	<0.002	<0.002	<0.002	<0.002	<0.002
Benzo(b+k) flouranthene	-	-	<0.002	<0.002	<0.002	<0.002	<0.002
Benzo (g.h.i) perylene	-	-	<0.02	<0.02	<0.02	<0.02	<0.02
Crysene	**0.384**	**2.8**	<0.002	<0.002	<0.002	<0.002	<0.002
Dibenzo (a.h) anthracene	**0.0634**	**0.26**	<0.1	<0.1	<0.1	<0.1	<0.1
Flouranthene	**0.6**	**5.1**	<0.002	<0.002	<0.002	<0.002	<0.002
Flourene	**0.019**	**0.54**	<0.002	<0.002	<0.002	<0.002	<0.002
Indeno (1.2.3-c.d) pyrene	-	-	<0.02	<0.02	<0.02	<0.02	<0.02
Naphthalene	**0.16**	**2.1**	<0.002	<0.002	<0.002	<0.002	<0.002
Phenanthrene	**0.24**	**1.5**	<0.002	<0.002	<0.002	<0.002	<0.002
Pyrene	**0.665**	**2.6**	<0.002	<0.002	<0.002	<0.002	<0.002
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 BENTHIC MACROFAUNA

8.1 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 2019 (with additional sites at Elandsfontein), are considered in this report.

8.2 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 2008-2018 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 0.24m^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 1999-2018 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.

8.3 Approach and methods used in monitoring benthic macrofauna in 2019

8.3.1 Sampling

Benthic macrofauna have been sampled at more than 30 sites in Big Bay (9 sites), Small Bay (ten sites) and Langebaan Lagoon (12 sites) since the inception of the State of the Bay monitoring programme in 2004. The localities and water depth ranges of the 2019 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 (>1 mm in size) 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 a hand-core. 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.

8.3.2 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 2019 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.

8.3.2.1 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 rate and short life-cycle 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 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.

8.3.2.2 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:

$H' = -\Sigma p_i(\log p_i)$ [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.

8.4 Benthic macrofauna 2019 survey results

8.4.1 Species diversity

Variation in species diversity (represented by the Shannon Weiner Index, H') is presented in Figure 8.1. Diversity was highest in Langebaan Lagoon (at sites LL 31, LL 33 and LL 37) and was lowest in Big Bay at the Liquid Petroleum Gas (LPG) site and at site BB 26. In Small Bay, the lowest diversity was observed near the iron ore jetty (SB 15) and in the yacht basin (SB 1). This corresponds with results from earlier surveys and is most likely attributable to the high levels of anthropogenic disturbance and the presence of elevated levels of contaminants (trace metals, organic material, etc.) in the fine 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.

8.4.2 Community structure

An ordination plot, prepared from 2019 macrofaunal abundance data, is presented in Figure 8.2. 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. 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 (SB 2, SB 3, SB 5, and SB 10) forming a separate cluster from 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 cannot tolerate high levels of disturbance are present in abundance at Elandsfontein and in Langebaan Lagoon but are largely absent from the Big Bay 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

[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.

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 2019 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 ~63% of the variation in macrofaunal abundance data, with C:N ratio and mud explaining the greatest amount (~25%) followed by very fine sand (5%) and gravel (5%).

The full model can be visualised by examining the distance-based redundancy analysis (dbRDA) ordination (Figure 8.3). The first two axes capture 50.2% of the variability in the fitted model, and 31.8% 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 C:N ratio and mud fraction clearly separated the Langebaan Lagoon sites from the Big Bay and Small Bay sites.

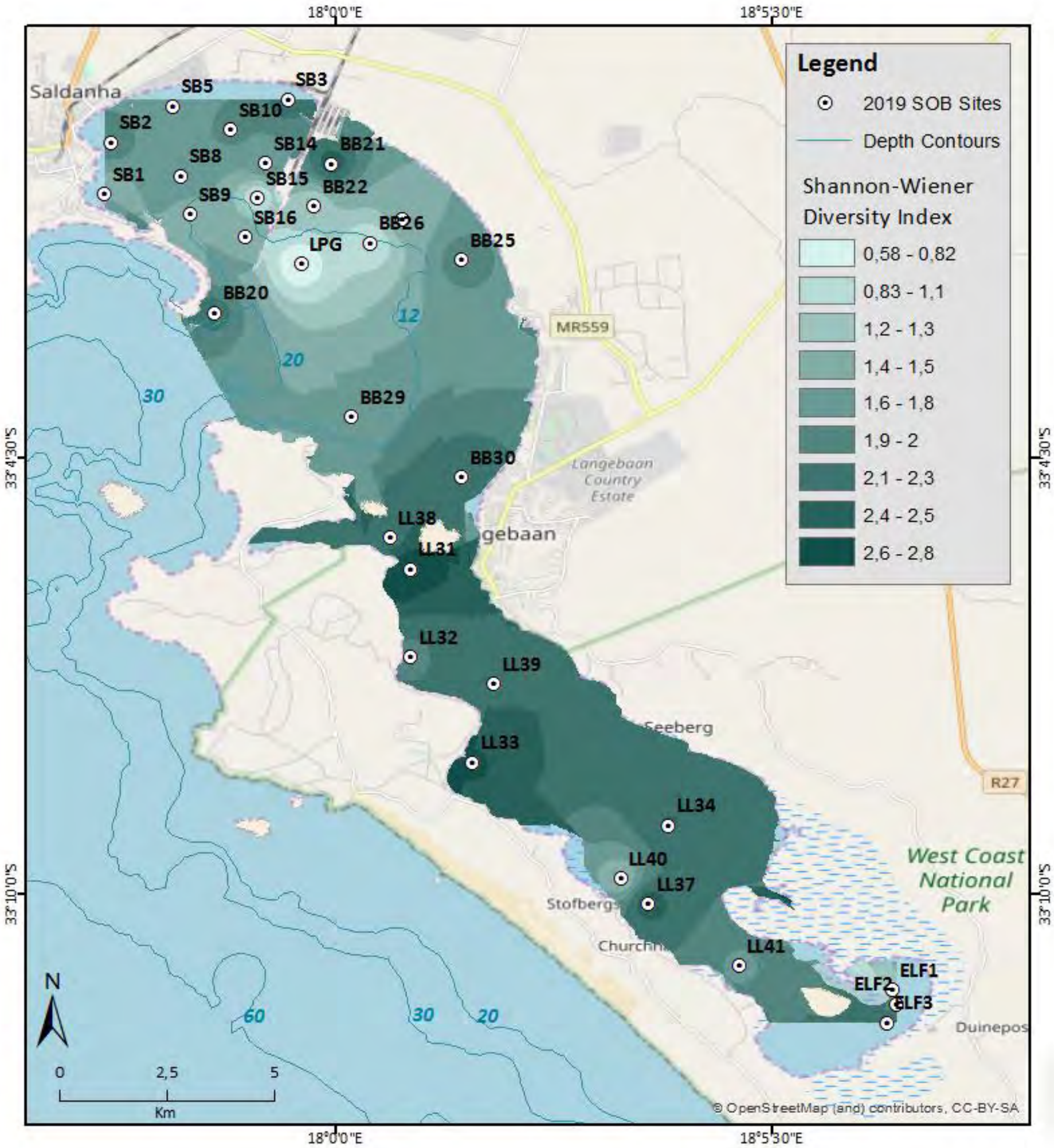


Figure 8.1 **Variation in the diversity of the benthic macrofauna in Saldanha Bay and Langebaan Lagoon as indicated by the 2019 survey results (H' = 0 indicates low diversity, H' = 2.8 indicates high diversity).**

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*, and the crown crab *Hymenosoma orbiculare* (detritivores, scavengers or predators) were more abundant in the lagoon samples.

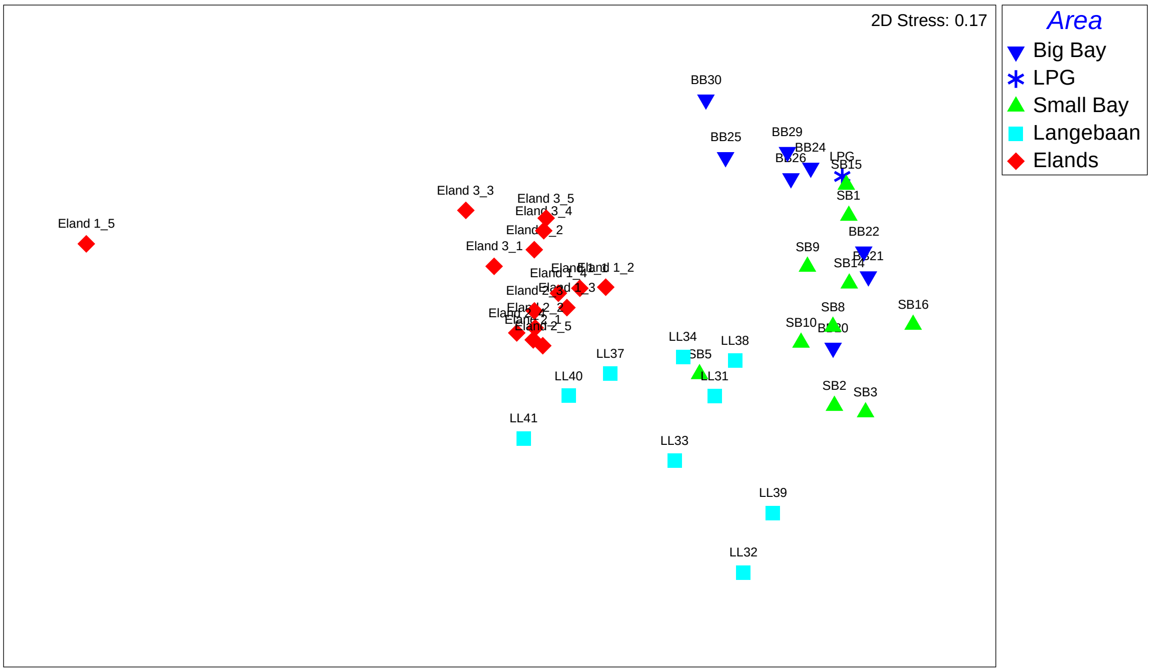


Figure 8.2 **Ordination plots showing similarity amongst sample sites based on benthic macrofauna abundance in 2019. Symbols on the ordination plots are as follows: Small Bay (SB), Big Bay (BB), Langebaan Lagoon (LL) and Elandsfontein (Elands).**

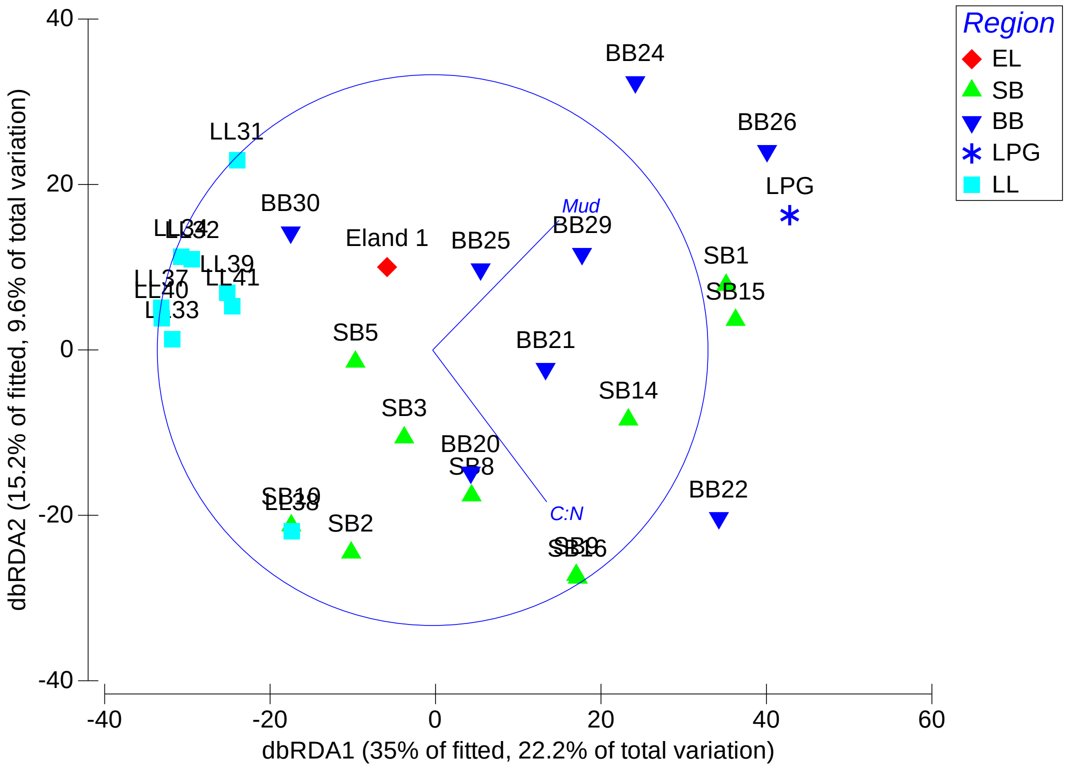


Figure 8.3 **dbRDA plot of 2019 macrofaunal abundance data. Sediment fractions, TOC, TON, C:N and trace metal concentrations were included as categorical predictors in this design. Sediment fractions were arcsine transformed prior to analysis. The blue lines are category vectors, whereby the length of 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 grimaldii*, 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, Big Bay, Langebaan Lagoon and Elandsfontein are shown in Figure 8.4. 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 and at Elandsfontein than in Small Bay and Big Bay (Figure 8.4). 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 8.4). Detritivores were numerically the most abundant group on the mudflats at Elandsfontein and in Langebaan Lagoon (Figure 8.4). 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.

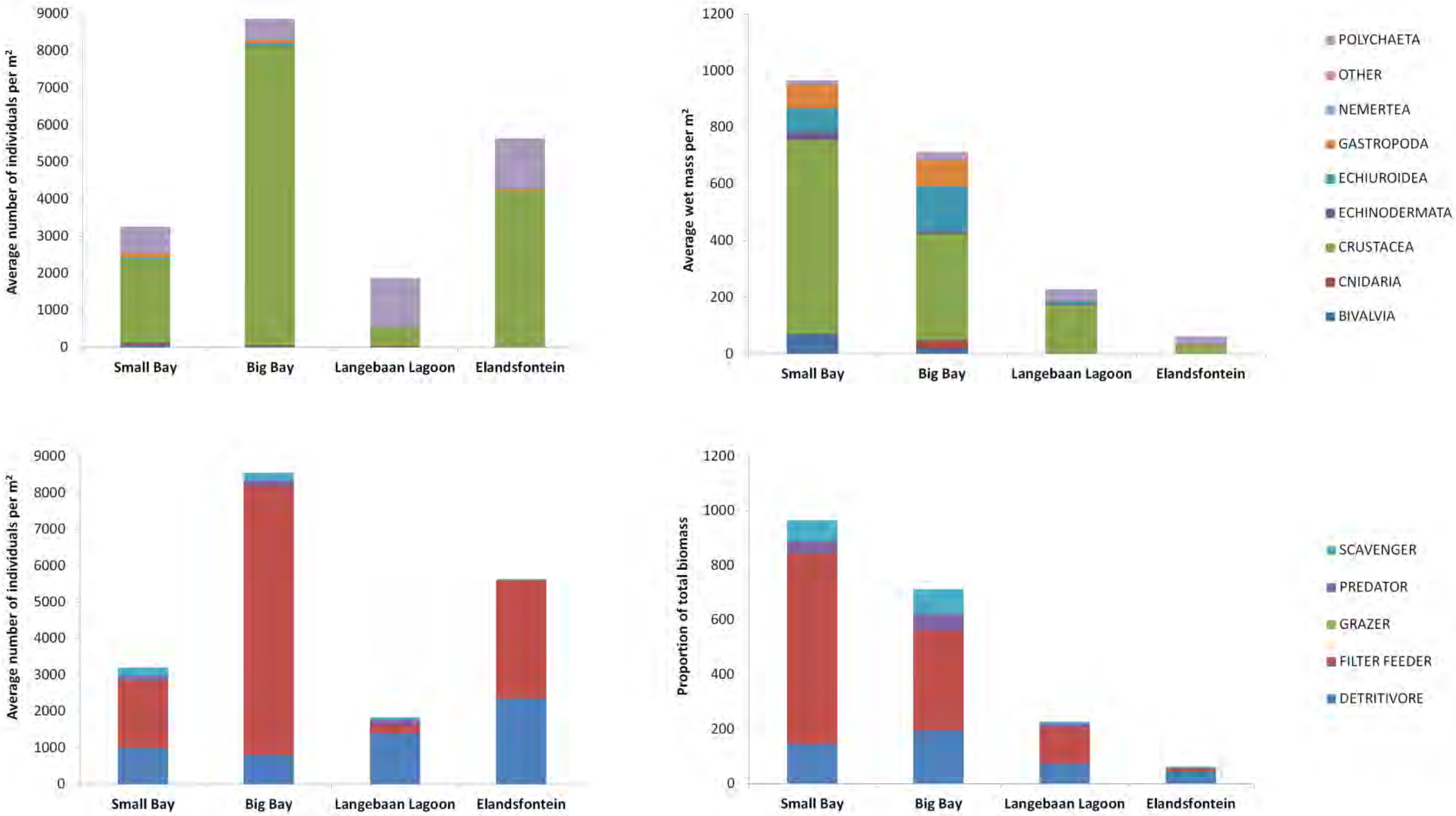


Figure 8.4 **Average abundance and biomass (g/m²) of benthic macrofauna by functional and taxonomic group in Big Bay, Small Bay, Langebaan Lagoon and Elandsfontein in 2019.**

8.5 Changes in abundance, biomass and community structure over time

8.5.1 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 2019 is shown in Figure 8.5. 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).

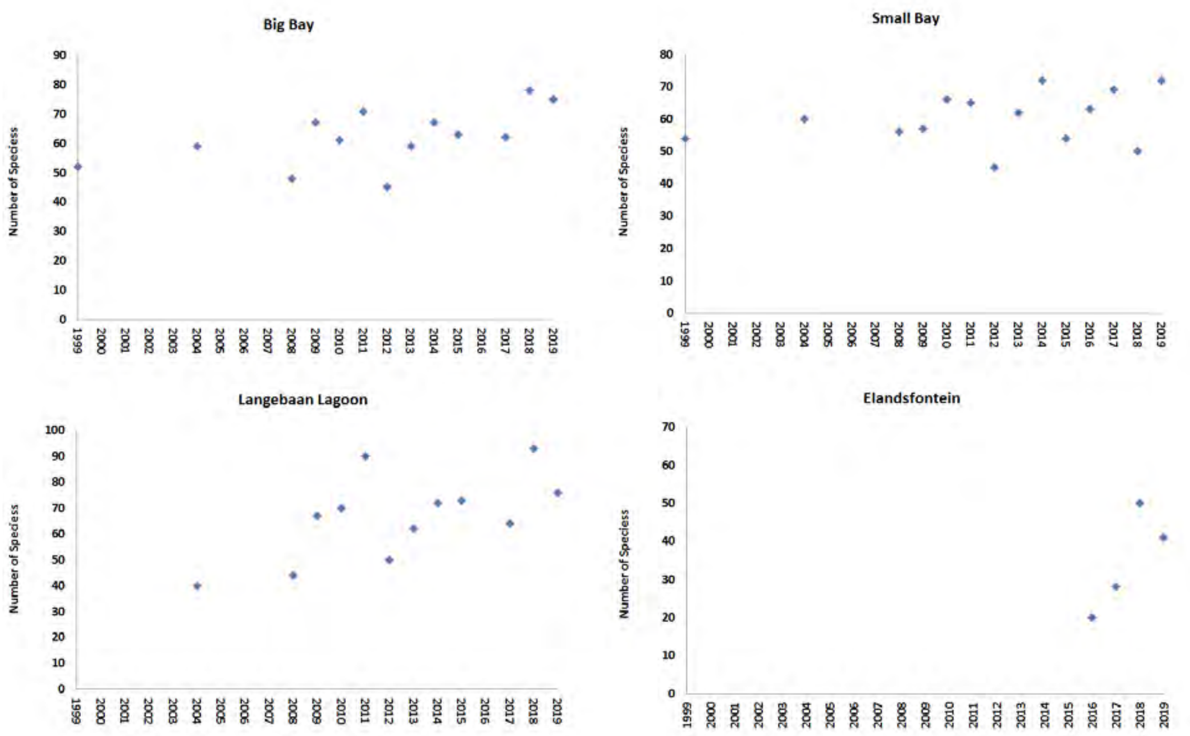


Figure 8.5 **Variation in the number of species recorded at Small Bay, Big Bay, Langebaan Lagoon (1999 – 2019) and Elandsfontein (2016 – 2019).**

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 third 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 and 2019 – this is most likely 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.

8.6 Abundance, biomass and community composition

Changes in the abundance and biomass of benthic macrofauna in Small Bay, Big Bay and Langebaan Lagoon are presented in Figure 8.6 - Figure 8.8. 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 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.

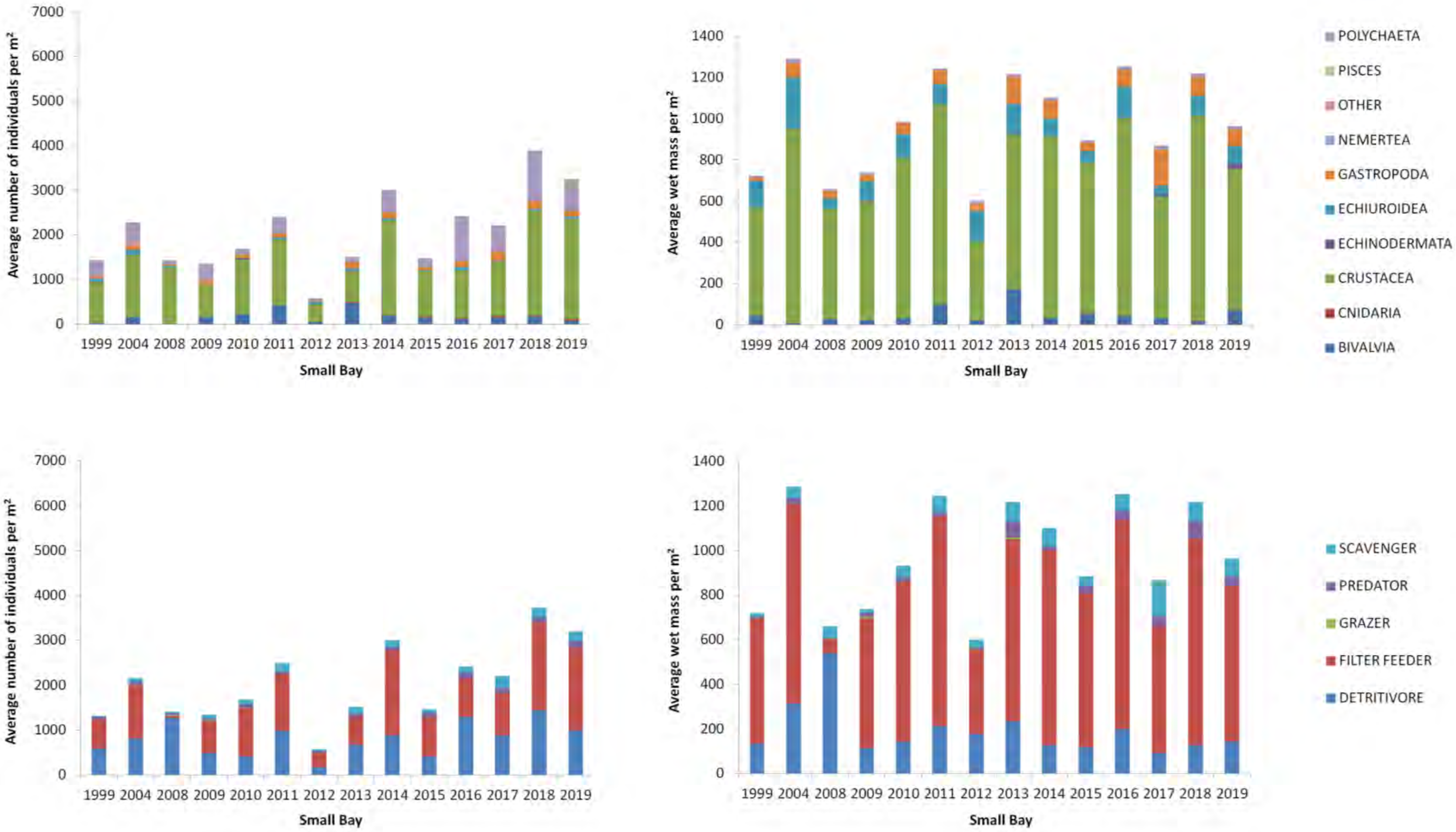


Figure 8.6 **Overall trends in the abundance and biomass (g/m²) of benthic macrofauna in Small Bay as shown by taxonomic and functional groups.**

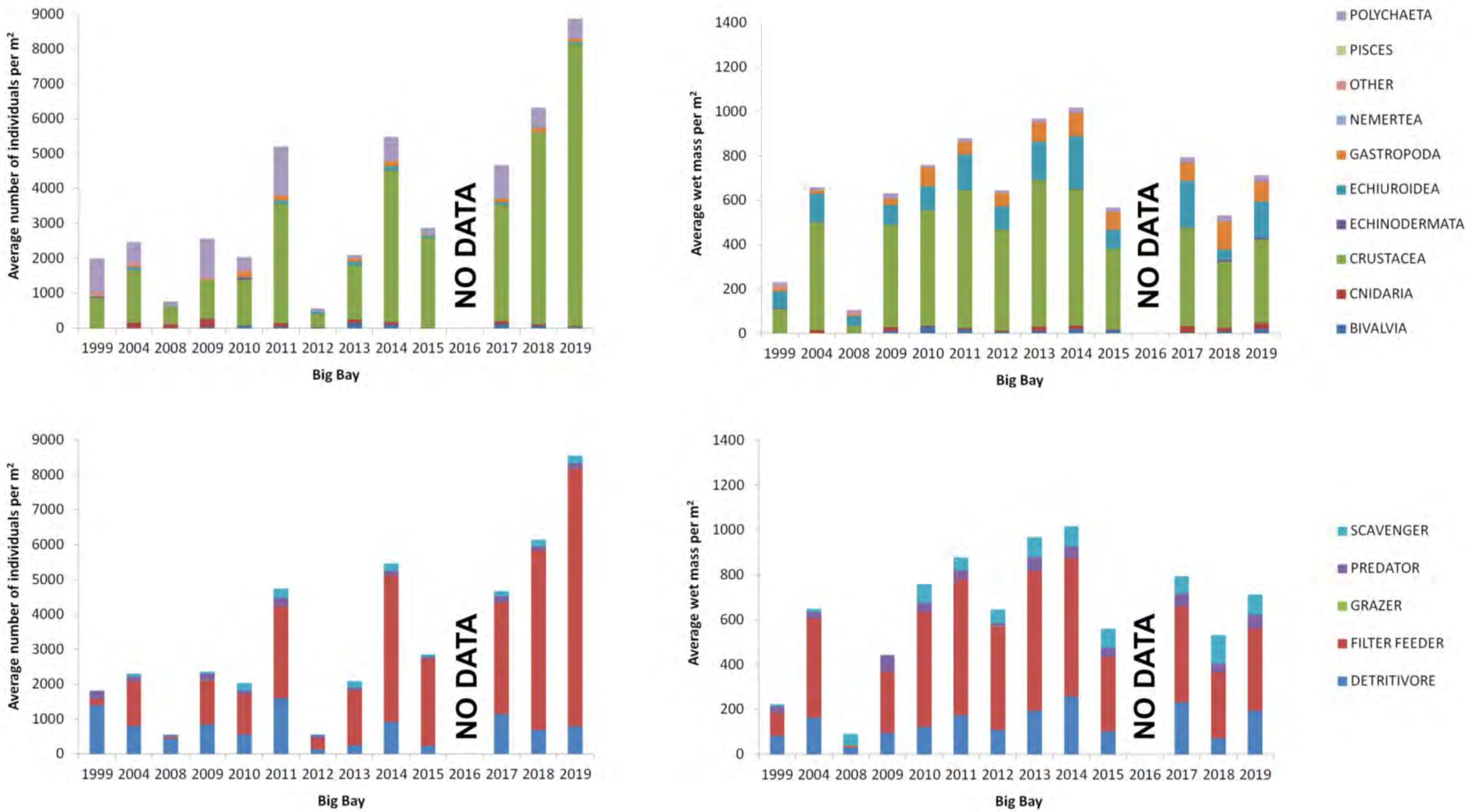


Figure 8.7 Overall trends in the abundance and biomass (g/m²) of benthic macrofauna in Big Bay as shown by taxonomic and functional groups.

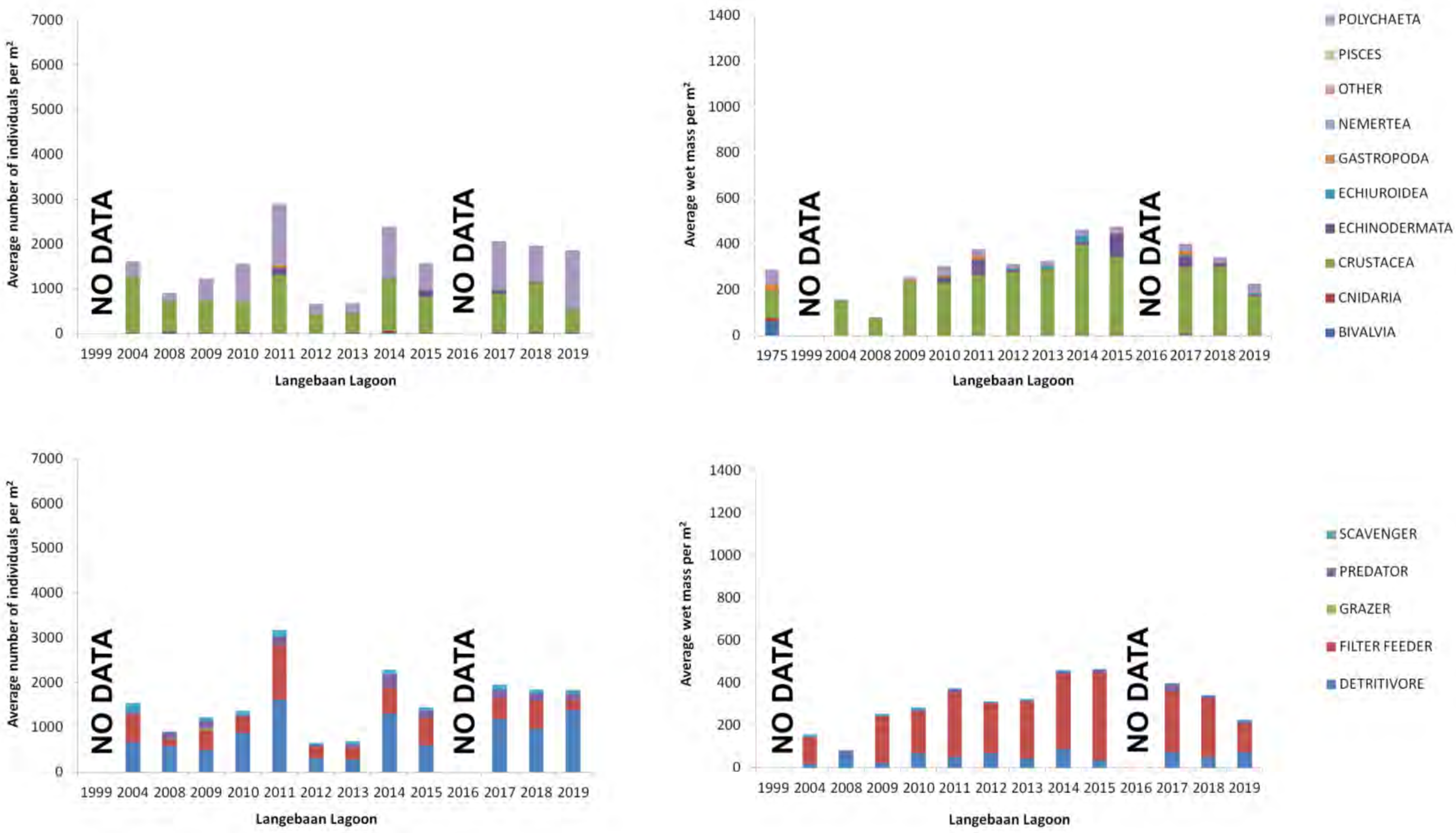


Figure 8.8 **Overall trends in the abundance and biomass (g/m²) of benthic macrofauna in Langebaan Lagoon as shown by taxonomic and functional groups.**

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.

8.7 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.

8.7.1 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 8.9). 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 2009-2019 suggests that the smaller 2009 dredging event had a limited impact with little change in macrobenthic community structure over the last ten years.

8.7.2 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 8.9). 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*.

8.7.3 Langebaan Lagoon

The 2008 samples were also outliers in the Langebaan Lagoon ordination plot (Figure 8.9). 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 2009-2018 suggests that the smaller 2009 dredging event had a limited impact with little change in macrobenthic community structure over the last ten years.

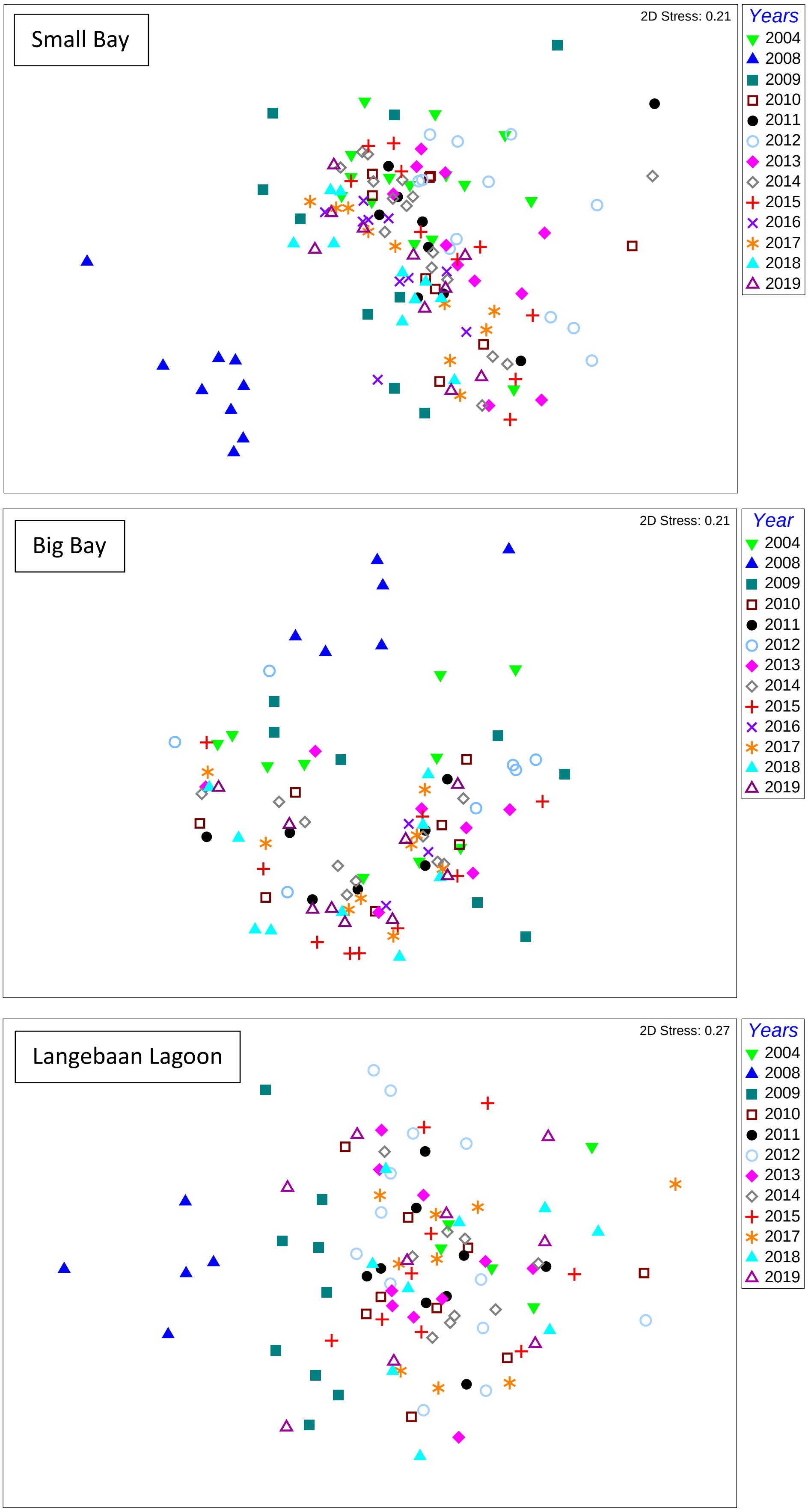


Figure 8.9. **MDS plots based on macrofaunal abundance data from samples collected in Small Bay (top), Big Bay (middle) and Langebaan Lagoon (bottom) during the period 2004-2019.**

8.8 Elandsfontein 2019 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 2019 macrofauna abundance data, are presented in Figure 8.2. 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 (77.3% dissimilarity) and in turn are most dissimilar to those in Small Bay (87.3%) and Big Bay (87.2%). 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 50 species (consisting of polychaetes, crustaceans, gastropods, bivalves, a nemertean and a cnidarian - Figure 8.10) 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 2019 (broken down into taxonomic and functional feeding groups) are shown in Figure 8.10. There does not appear to be any significant difference in mean abundance over the years, however, mean biomass was below average in 2018 and 2019. On a community composition level, the samples collected in 2018 and 2019, group separately to those collected in 2016 and 2017 (Figure 8.11). 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 8.11). A simper analysis reveals 67.25% dissimilarity between Eland_3 and Eland_1 & 2 with the amphipod, *Urothoe grimaldii*, the prawns *Upogebia africana* and *Callichirus kraussi*, and the polychaetes *Notomastus latericeus, Telothelepus capensis* and *Orbina angrapequensis* contributing >40% to this dissimilarity. This is likely to be explained by the difference in physical conditions present at each of the sites. From Figure 8.1, 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.

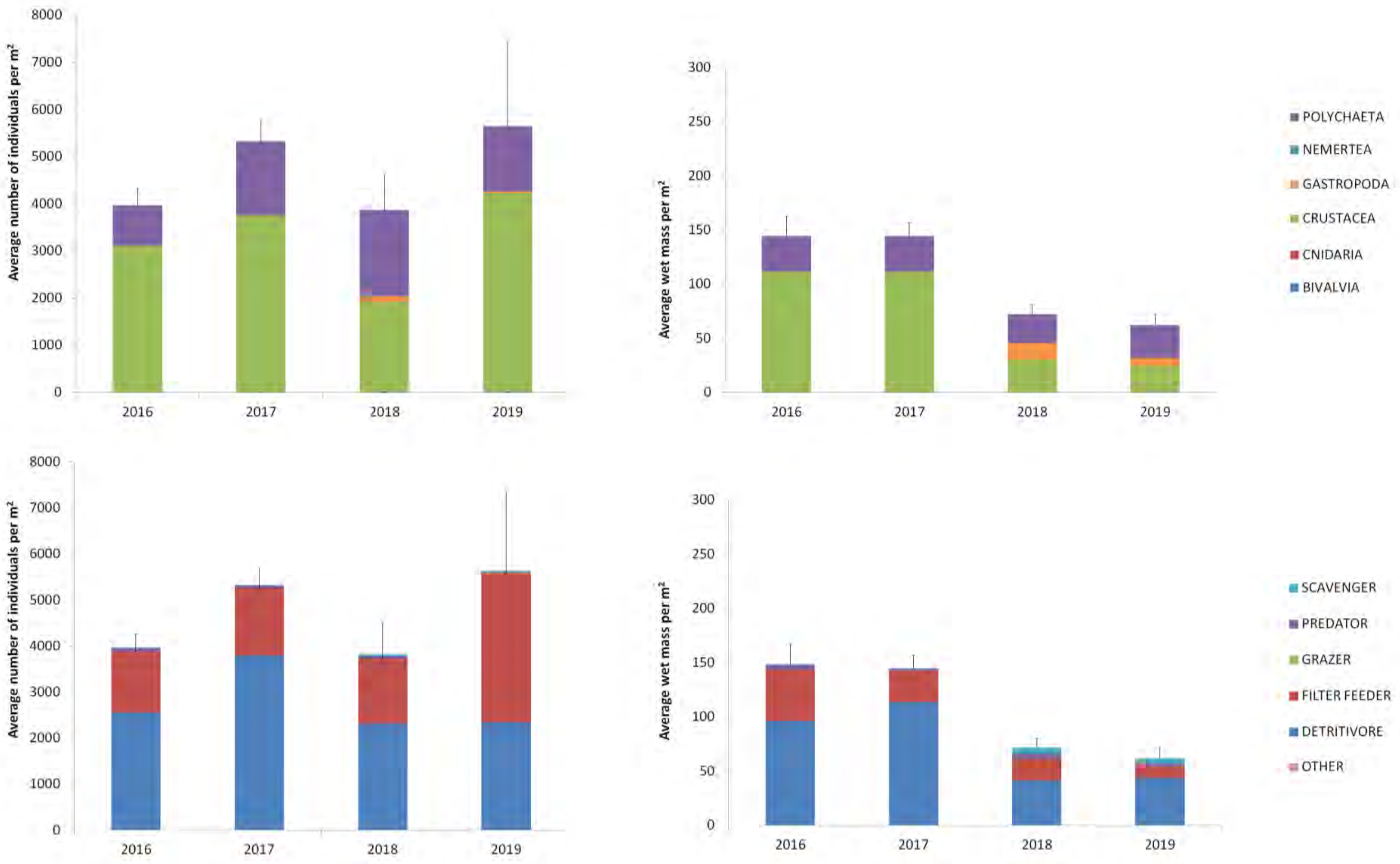


Figure 8.10 **Average abundance and biomass (g/m2) of benthic macrofauna by functional and taxonomic group from sampling sites at Elandsfontein from 2016 to 2019 - error bars are + 1 Standard Error (n=15).**

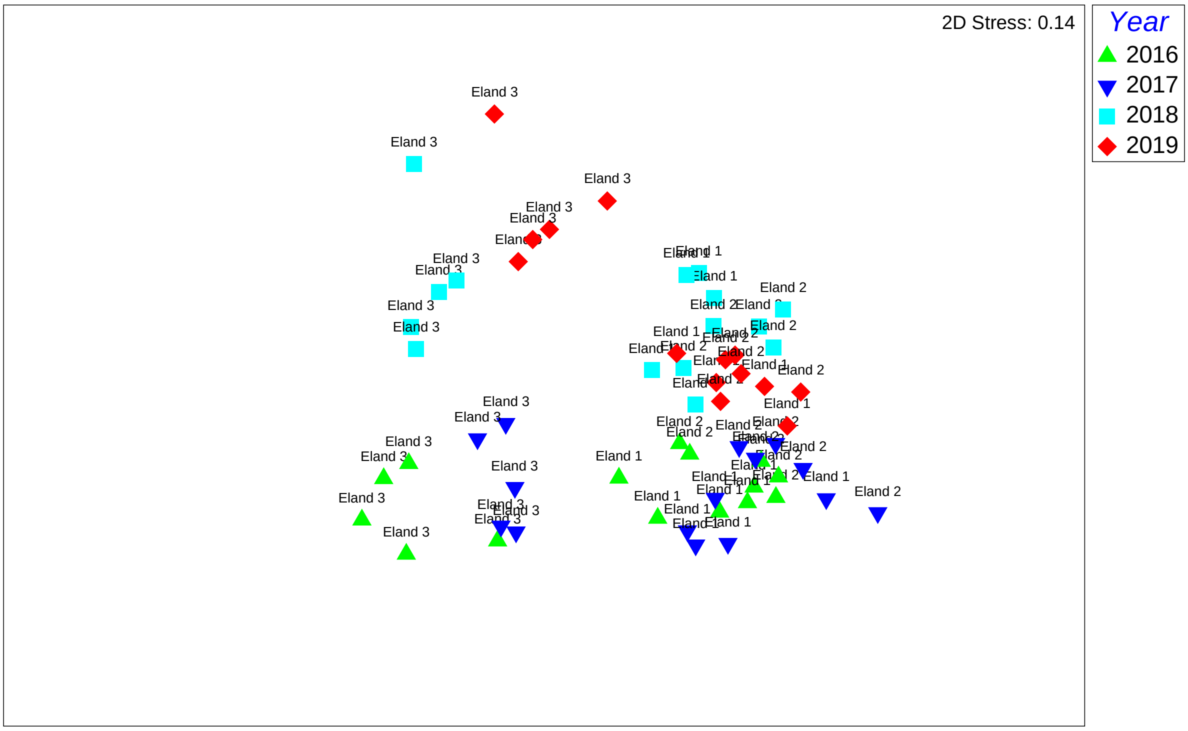


Figure 8.11 **MDS plot based on macrofaunal abundance data from samples collected at Elandsfontein from 2016 to 2019.**

8.9 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 appears 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 2019 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).

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 2010-2018) 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 2019 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.

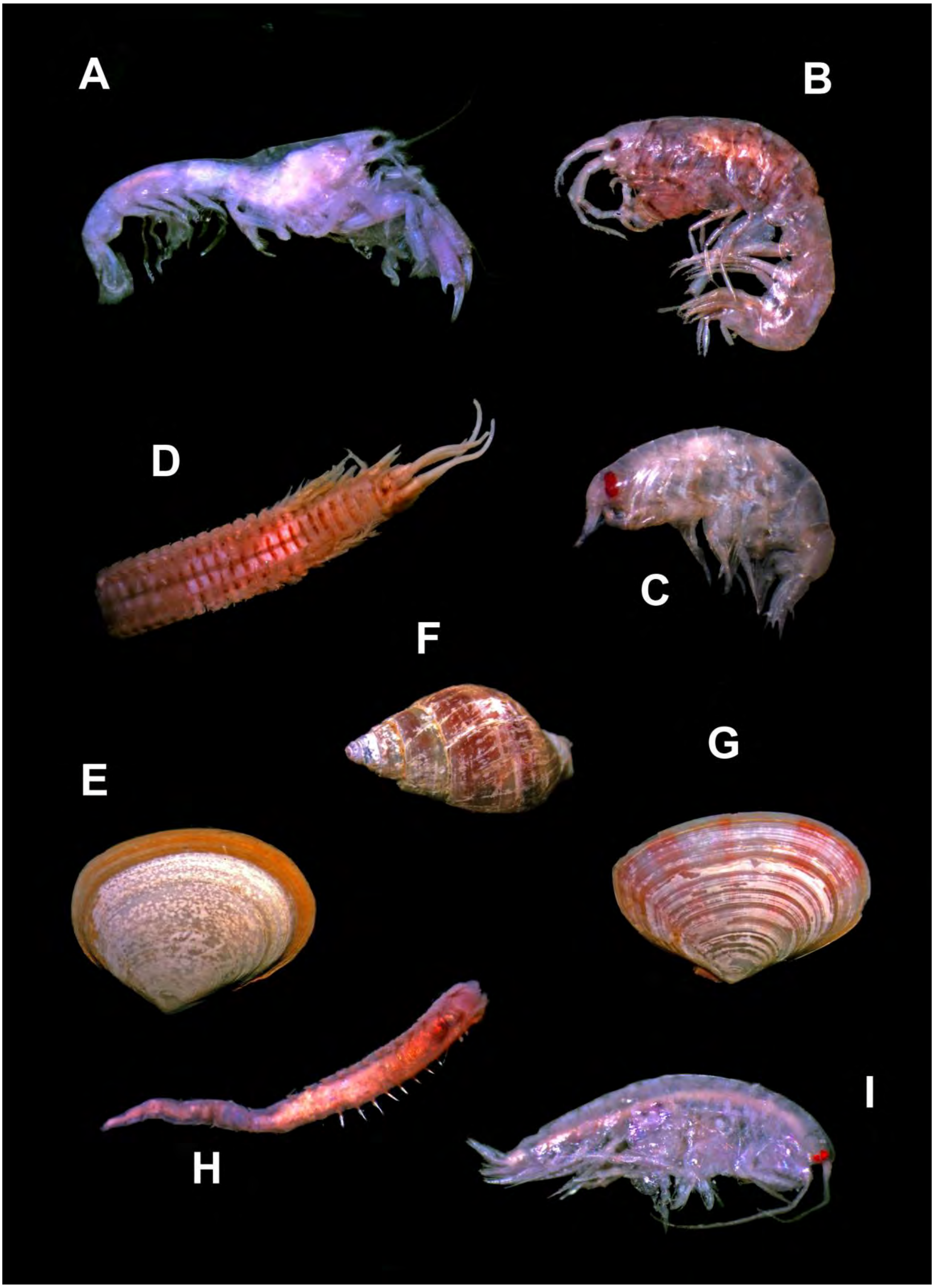


Figure 8.12. Benthic macrofauna species frequently found to occur in Saldanha Bay and Langebaan Lagoon, photographs by: Aiden Biccard. A – *Upogebia capensis*, B – *Idunella lindae*, C – *Hippomedon normalis*, D – *Diopatra monroi*, E – *Macoma c. ordinaria*, F – *Nassarius vinctus*, G – *Tellina gilchristi,* H – *Sabellides luderitzi*, I – *Ampelisca anomola.*

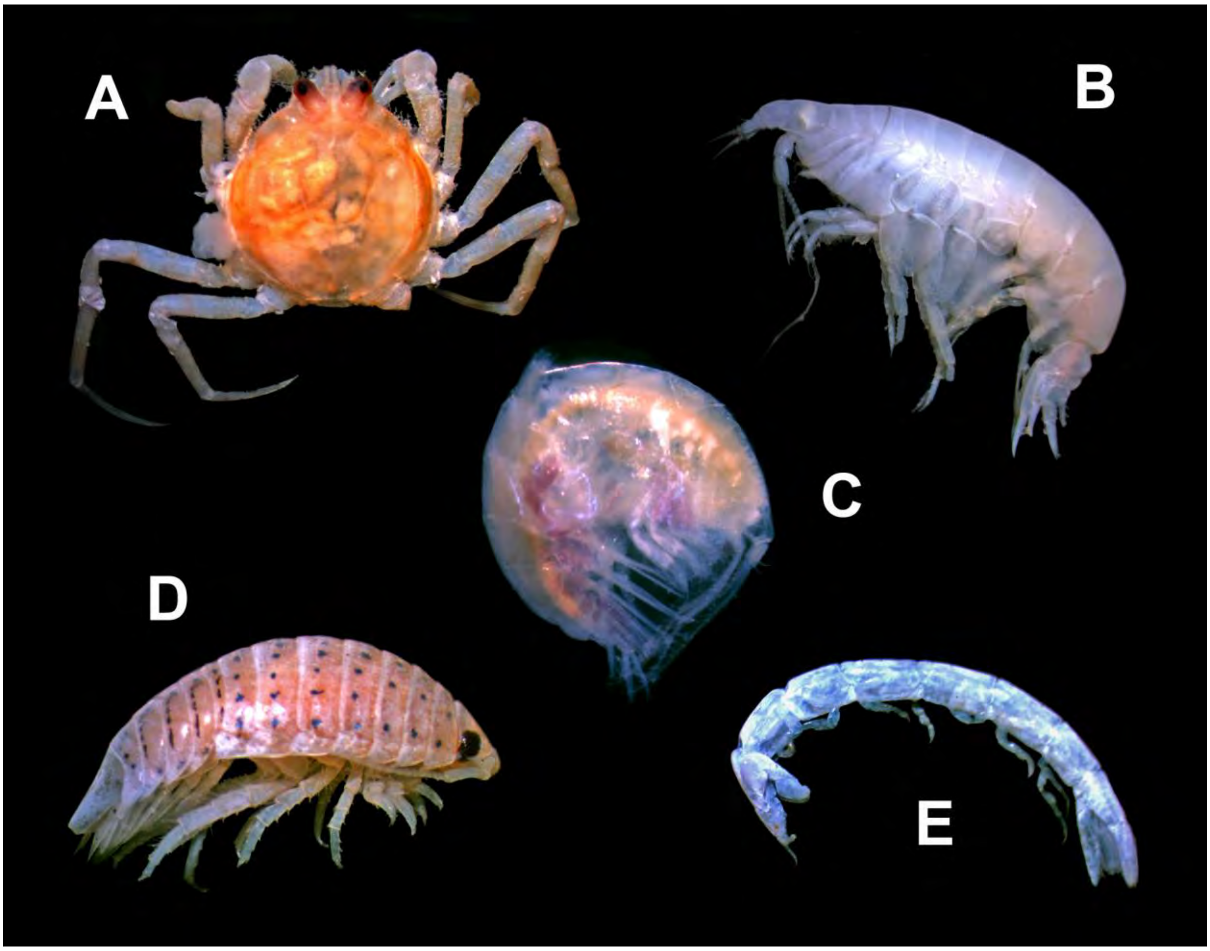


Figure 8.13. **Benthic macrofauna species frequently found to occur in Saldanha Bay and Langebaan Lagoon, photographs by: Aiden Biccard. A – *Hymenosoma obiculare,* B – *Socarnes septimus*, C – *Ampelisca palmata,* D – *Eurydice longicornis,* E – *Centrathura caeca*.**

9 ROCKY INTERTIDAL COMMUNITIES

9.1 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 12).

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 2015.

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* (Aiden Biccard *pers. comm.* 2017) and *Amphibalanus amphitrite amphitrite* were noted.

This chapter presents results from the twelfth annual monitoring survey conducted in February 2019.

9.2 Approach and methodology

9.2.1 Study sites

The locations of the eight rocky shore sampling sites are shown in Figure 9.1. 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 9.1 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 9.2). 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 9.2 **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.**

9.2.2 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. Individual mobile organisms were counted to calculate densities within the quadrat area ($0.5m^2$). The quadrat was subdivided into 171 smaller squares with 231 points to aid in the estimation of the percentage cover. 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.

9.2.3 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 4^{th}-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.

9.3 Results and discussion

9.3.1 Spatial variation in community composition

In 2019, a total of 118 taxa were recorded from all rocky shore sites, of which 68 taxa were invertebrates (58%) and 50 (42%) algae. The faunal component was represented by 23 filter feeding taxa, 25 grazers, and 20 predators/scavengers. The algal component comprised 33 corticated (foliose) seaweeds, ten ephemerals, five encrusting algae, and two 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, 2016, 2017, 2018). 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 9.3 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, 2016, 2017, 2018).

9.3.1.1 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 2019 showed that the Dive School was only 19% 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 Jetty, the Iron Ore Terminal and Schaapen West with less than 1% average cover, although densities were much higher on Marcus Island with an average of ±4% cover. Almost 23% of one of the quadrats surveyed in the high shore zone on Marcus Island was covered by this alien. On average, barren rock accounted for >80% on the high shore and algal cover was extremely sparse, except at Marcus Island which had on average 2.7% cover of *Porphyra capensis,* and at Schaapen Island West which had 2.6% cover of *Ulva* in the high shore.

9.3.1.2 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 2019 showed that the ephemeral alga *Ulva* spp., the barnacle *Balanus glandula*, the encrusting alga *Hildenbrandia* spp., the periwinkle *Oxystele antoni*, the whelk *Burnupena* spp., the cushion starfish *Parvulastra exigua*, and the limpet *Cymbula granatina* together accounted for 50% of the similarity between mid-shore sites. Algal presence was generally low in the mid shore with some ephemeral cover.

With increasing wave force across sites, the mid shores were dominated by filter feeders, particularly two alien invasive species; the mussel *M. galloprovincialis* and the barnacle *Balanus glandula*. *Balanus* was most abundant at the semi-exposed Iron Ore Terminal and the exposed North Bay with an average cover of 27% and 20% respectively. *Mytilus* was by far the most abundant at the exposed Marcus Island with an average of 21% cover in the mid shore. In contrast, neither of these species was present in substantial numbers on the mid shore at both Schaapen Island 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).

9.3.1.3 Low shore

Reflecting known zonation patterns, total biotic cover generally increased from high to low shore from an average of 14% to 53% 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 the most common.

The following species together accounted for 50% of the similarity attributed between low shore sites: encrusting coralline including *Hildenbrandia* spp.; the algae *Gigartina polycarpa*, *Ulva* spp., *Ceramium* spp., and *Cladophora* spp.; the mussels *M. galloprovincialis* and *Aulacomya atra;* the whelk *Burnupena* spp.; and colonial ascidians. *Aulacomya atra* can be found living deep down in *Mytilus* beds where they take advantage of the moisture trapped within the overlaying dense mussel matrix. In 2011, the indigenous ribbed mussel *Aulacomya atra* was fairly prominent at the low shore at Marcus Island and could locally supersede the alien mussel (Anchor Environmental Consultants 2012b) but during the 2019 survey, the ribbed mussel contributed <3% 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.

9.3.2 Temporal analysis

9.3.2.1 Temporal 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.

As previous reports showed no clear trend in diversity indices over time (Anchor Environmental 2015), temporal biotic cover data were averaged across years from 2005 to 2019 at each site (Figure 9.4). 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. Marcus Island 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 ~16% cover at Dive School to ~59% cover at 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 the iron ore terminal had the lowest values for both these indices. This indicated low overall diversity but higher variation in communities over the years, which may be an indication of disturbance.

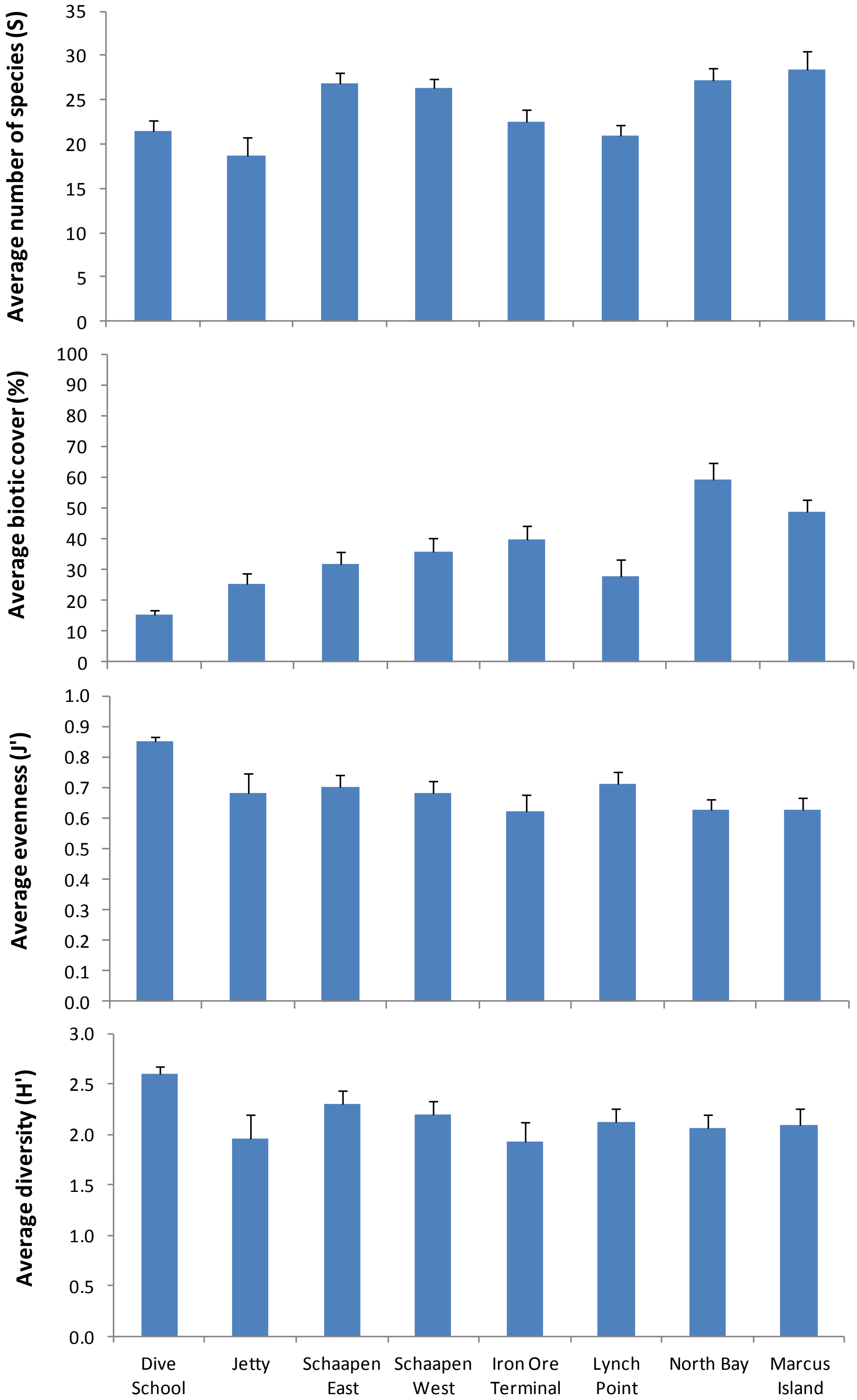


Figure 9.4 **Temporal biotic cover data from 2005 – 2019 averaged across years and displayed as biotic indices of 'species number' (S), 'biotic cover' (N), 'diversity' (d) and 'evenness' (J'). Error bars indicate standard error.**

9.3.2.2 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 9.5). 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 exposed 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, for example, samples from 2013 tend to be on the right of the cluster, while those from 2005 are on the left (Figure 9.5). 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.

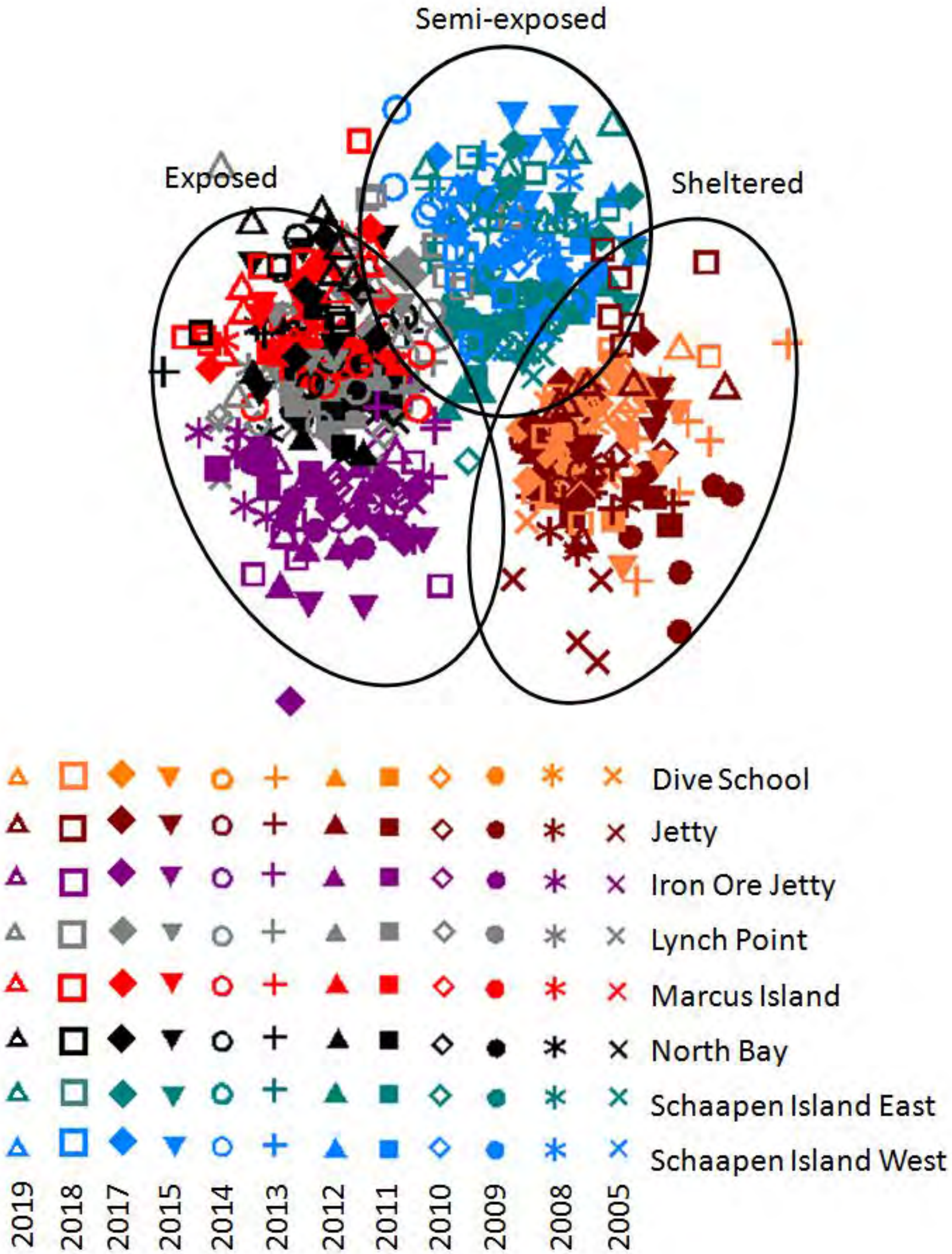


Figure 9.5 **Multi-dimensional scaling (MDS) plot of the rocky shore communities at the eight study sites from 2005 to 2019. The circles delineate a 40% similarity level and the plot has with a 2D stress of 0.23.**

9.3.2.3 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 2018 and the current dataset from 2019 are presented (Table 9.1). At most of the sites only one or two species contributed largely (>4%) to the differences in community structure between 2018 and 2019, except for Schaapen East where three species contributed and the Schaapen West where no single species contributed >4% (Table 9.1). For the latter site, the species contributing the most to the dissimilarity was listed.

Notable changes in species composition included the appearance of three species that were not present at these specific sites the previous year, the disappearance of two species, the increase in abundance of four species, and the decrease in abundance of four species (Table 9.1).

Algae were common contributors to differences between years, a result which was expected as their abundance is seasonal. Of the six algal species listed, one was an ephemeral alga, two were corticated algae, and three were 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; while the decrease in cover of *Porphyra capensis* contributed to the difference between the years at both Schaapen sites. *Gigartina* cover decreased at both Jetty and at the Iron Ore Terminal but increased at Schaapen East.

Of the four animals listed, two were filter-feeding barnacles, *Balanus glandula* and *Notomegabalanus algicola;* one was a mussel *Aulacomya atra;* and one was a limpet *Scutellastra tabularis*. *Notomegabalanus algicola* was recorded at Marcus Island in 2019 but not in 2018, while Scutellastra *tabularis* appeared at Dive School for the first time. Percentage cover of the alien barnacle *Balanus glandula* increased from 2018 to 2019 at Lynch Point but continued to decrease at North Bay (Table 9.1). 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.

Table 9.1 **SIMPER results listing the species that contribute >4% to the dissimilarity between 2018 and 2019 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	2018 %cover	2019 %cover	Ave. dissimilarity	% Contribution	Ave. dissimilarity between years
Dive School	*Scutellastra tabularis*	0	0.77	1.97	4.08	48.43
Jetty	*Hildenbrandia spp.*	0.96	0	2.92	5.26	55.59
	Gigartina bracteata	1.06	0.13	2.80	5.04	
Schaapen East	Diatoms	0	1.17	2.39	4.76	50.28
	Gigartina polycarpa	0.09	1.18	2.22	4.42	
	Porphyra capensis	1.2	0.21	2.05	4.08	
Schaapen West*	*Porphyra capensis*	1.09	0.27	1.79	3.38	52.91
Iron ore terminal	Encrusting coralline	0.28	1.71	3.12	5.72	54.51
	Gigartina bracteata	1.02	0	2.23	4.08	
Lynch Point	*Balanus glandula*	0.17	1.06	1.98	4.01	49.38
North Bay	*Balanus glandula*	1.28	0.82	2.04	4.28	47.68
Marcus Island	*Aulacomya atra*	0.39	1.47	2.40	4.71	51.06
	Notomegabalanus algicola	0	1.08	2.32	4.55	

* 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.

9.3.2.4 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 9.7 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 East 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 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.

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.

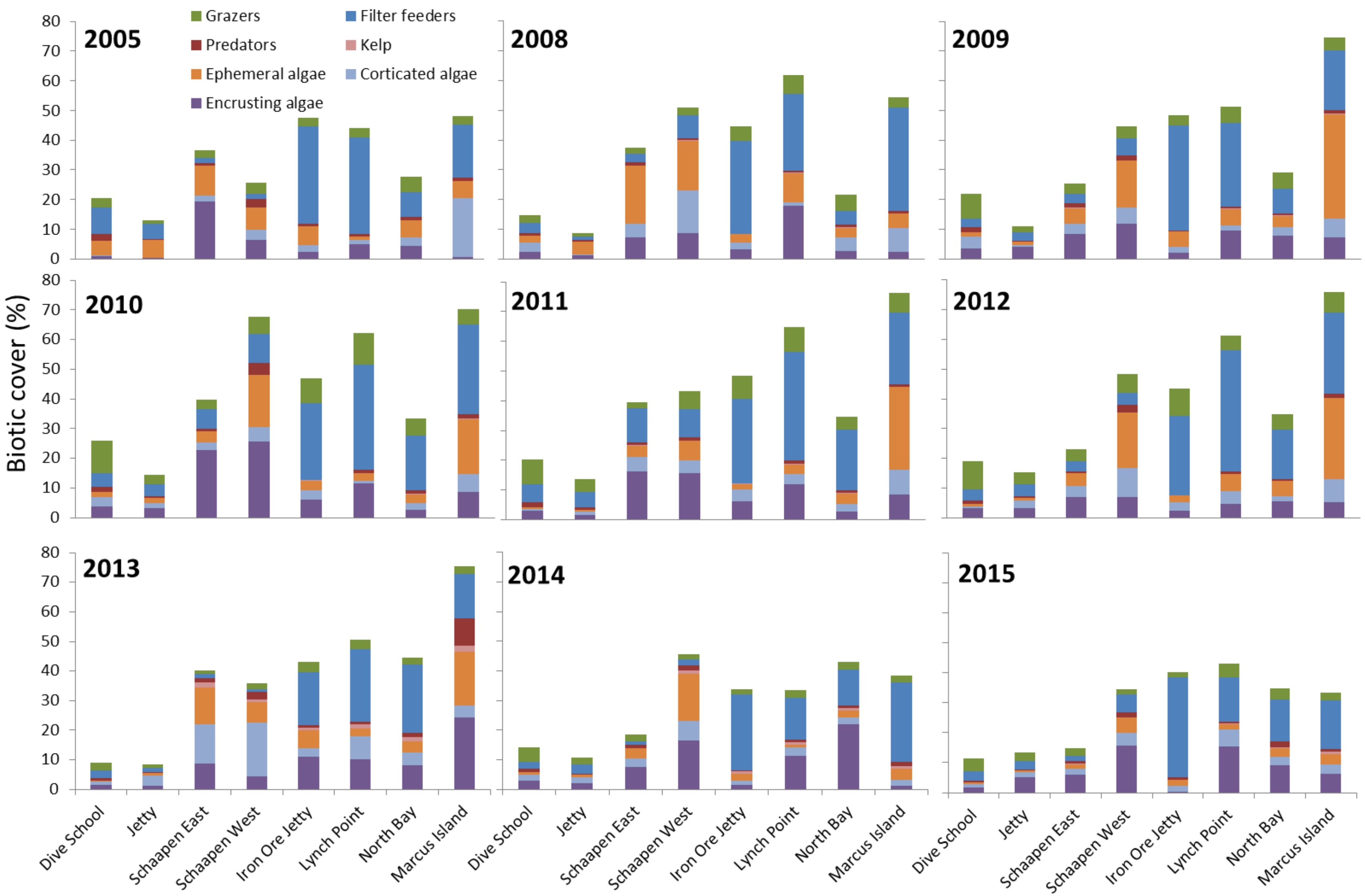


Figure 9.6 **Total percentage cover (averaged across the whole shore) of the seven functional groups at the eight study sites from 2005 to 2015.**

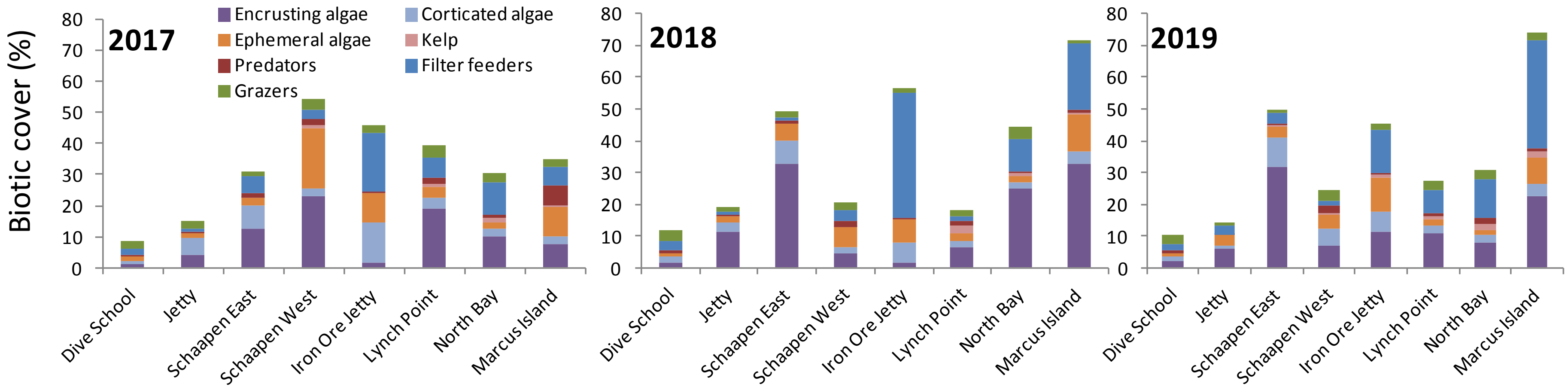


Figure 9.7 **Total percentage cover (averaged across the whole shore) of the seven functional groups at the eight study sites from 2017 to 2019.**

9.3.3 Summary of findings

In 2019, a total of 118 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, 25 species of grazers, and 20 species of predators and scavengers combined. The algal component comprised 33 corticated (foliose) seaweeds, ten ephemerals, five species of encrusting algae, and two 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, 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 appears to be declining in abundance over time with only empty shells and base plate scars left on rocks at some sites. The latter is most abundant in the mid shore zone of semi-exposed sites, but rarer at exposed sites and low shores. *Mytilus 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 12 for detailed information on invasive species).

10 FISH COMMUNITY COMPOSITION AND ABUNDANCE

10.1 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 targeting harders landed an estimated 590 tonnes valued at approximately R1.8 million during 1998-1999 (Hutchings & Lamberth 2002a). 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. These data were reported on in the “State of the Bay 2008” report (Anchor Environmental Consultants 2009).

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) effectively protects 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.

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 have to date, been resilient to rapidly increasing recreational fishing pressure (but see paragraph below on stock status). Results from angler surveys indicate that approximately 92 tonnes of white stumpnose is landed by anglers each year (Næsje *et al.* 2008). Further details of the results of these studies were reported on in the State of the Bay 2008 report (Anchor Environmental Consultants 2009). 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.*).

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 in press.). 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 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 2018). Preliminary results of this study indicate that the stock was at risk of recruitment failure at risk of recruitment failure at risk of recruitment failure (the current spawner biomass was estimated at less than 25% of the pristine level). These results are reported in a scientific paper that is undergoing review for publication in the African Journal of Marine Science. A reduction in fishing effort and increase in mesh size were recommended to help rebuild the stock (Horton 2018).

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 2008-2019 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 2019 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.

10.2 Methods

10.2.1 Field sampling

Experimental seine netting for all surveys covered in this report 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 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 Rietvlei (Figure 10.1). Large hauls were sub-sampled on site, the size of the sub-sample estimated visually, and the remainder of the catch released alive.

10.2.1.1 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 over time were analysed using a one-way ANOVA and post-hoc unequal N HSD tests in the software package STATISTICA 13. Abundance data for all sites throughout the Bay were log (x + 1) transformed to account for heteroscedacity (unequal variance) prior to analysis.

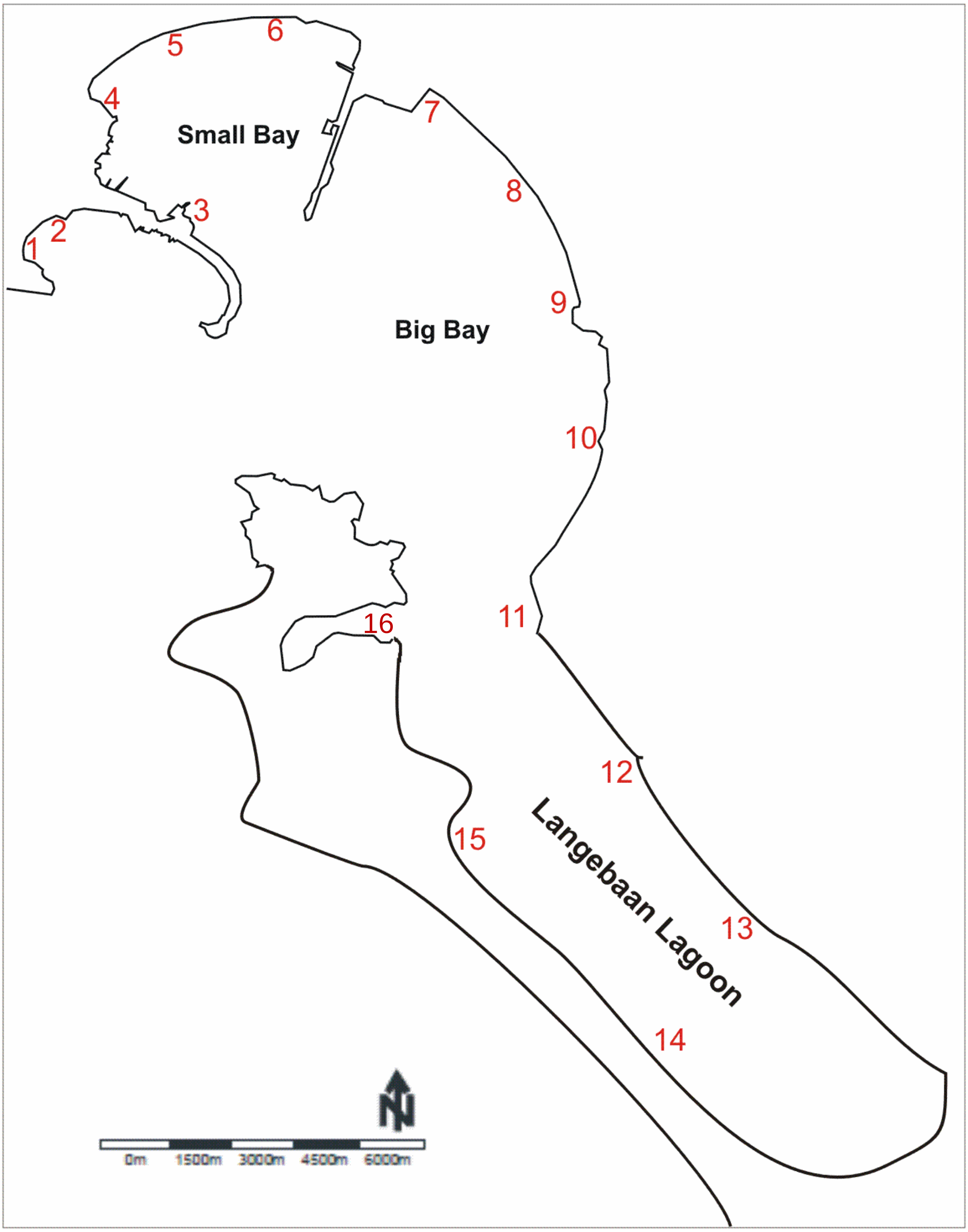


Figure 10.1. Sampling sites within Saldanha Bay and Langebaan lagoon where seine net hauls were conducted during the 2005 and 2007-2019 annual sampling events. 1: North Bay west, 2: North Bay east, 3:Small craft harbour, 4: Hoedjiesbaai, 5: Caravan site, 6: Blue water Bay, 7: Sea farm dam, 8: Spreeuwalle, 9: Lynch point, 10: Strandloper, 11: Schaapen Island, 12: Klein Oesterwal, 13: Bottelary, 14: Churchaven, 15: Kraalbaai, 16: Rietbaai.

10.3 Results

10.3.1 Description of inter annual trends in fish species diversity

The total species count in all surveys to date remains at 50 species taking into account the three different species of goby of the genus *Caffrogobius*, namely: *C. nudiceps, C. gilchristi* and *C. caffer* that have been identified in samples from the Bay. Due to the uncertainty surrounding identification of these species in earlier surveys, however, they have been grouped at the generic level for data presented reports since 2008. Considering data from all surveys conducted to date, a greater diversity of species has been captured in Big Bay (37) and in Small Bay (36), than 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 10.2.). 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 10.2.). In the 2019 samples, fish diversity in Big Bay and Langebaan Lagoon was low (only 8 species), in fact the lowest recorded in Big Bay to date. The relatively low diversity in Langebaan Lagoon was not unprecedented, with only 8 species caught in 2007 and 7 species in 2012 (although reduced sampling effort was undertaken in Langebaan during 2012). The 2019 fish diversity was well below the average recorded in previous surveys in both in Big Bay (14) and Langebaan Lagoon (11) and although it is not consistent, there does appear to a declining trend in species richness in Big Bay since 2012 (Figure 10.2.).

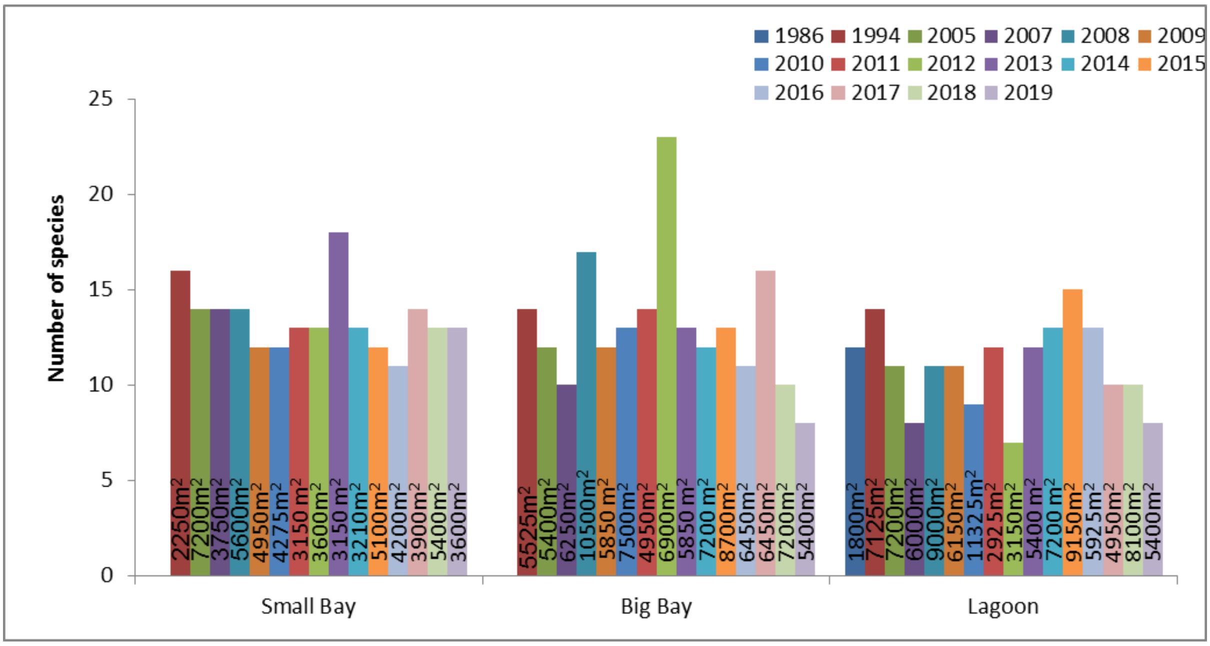


Figure 10.2. **Fish species richness during 16 seine-net surveys in Saldanha Bay and Langebaan lagoon conducted over the period 1986-2019. The total area netted in each area and survey is shown. Note: The low species richness for Langebaan lagoon during 2012 is an artefact due to low sampling effort.**

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 10.5 - Figure 10.8. 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, eight species occurred in all earlier surveys, with two additional species, namely blacktail only absent in 2015 and 2017 samples, and pipefish was absent in the 2005 and 2015 samples. Gurnard captured in all of the first six surveys, but not over the period 2011-2014 and was again absent in the 2018 and 2019 samples.

Four of the 37 species recorded in Big Bay occurred in all surveys (gurnard, Cape sole, harders and white stumpnose). Prior to this year, False Bay klipvis were only absent in the earliest 1994 survey (possibly not correctly identified), but where again absent in 2019, whilst Cape silverside has been absent from two surveys (most recently in 2018) and elf was absent for only the third time in 2019. Sand sharks were not caught in Big Bay during the 2014, 2016 and 2019 surveys, but were caught in the 2017 and 2018 surveys. The five species that were usually present in Big Bay surveys but were absent in 2019 were False Bay klipvis, super klipvis, elf, sandsharks and pipefish. None of these species are targeted in fisheries in the area and the reason for their absence from 2019 catches is unknown. 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 theses species were all present in 2019 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 10.2.).

10.3.2 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 10.3.). Harders, and to a lesser extent 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 sampling years. Overall the catches made during the 2012 survey were the lowest on record for all three areas. Over the last six years 2014-2019 the overall abundance of fish has compared favourably with earlier surveys (Figure 10.3.).

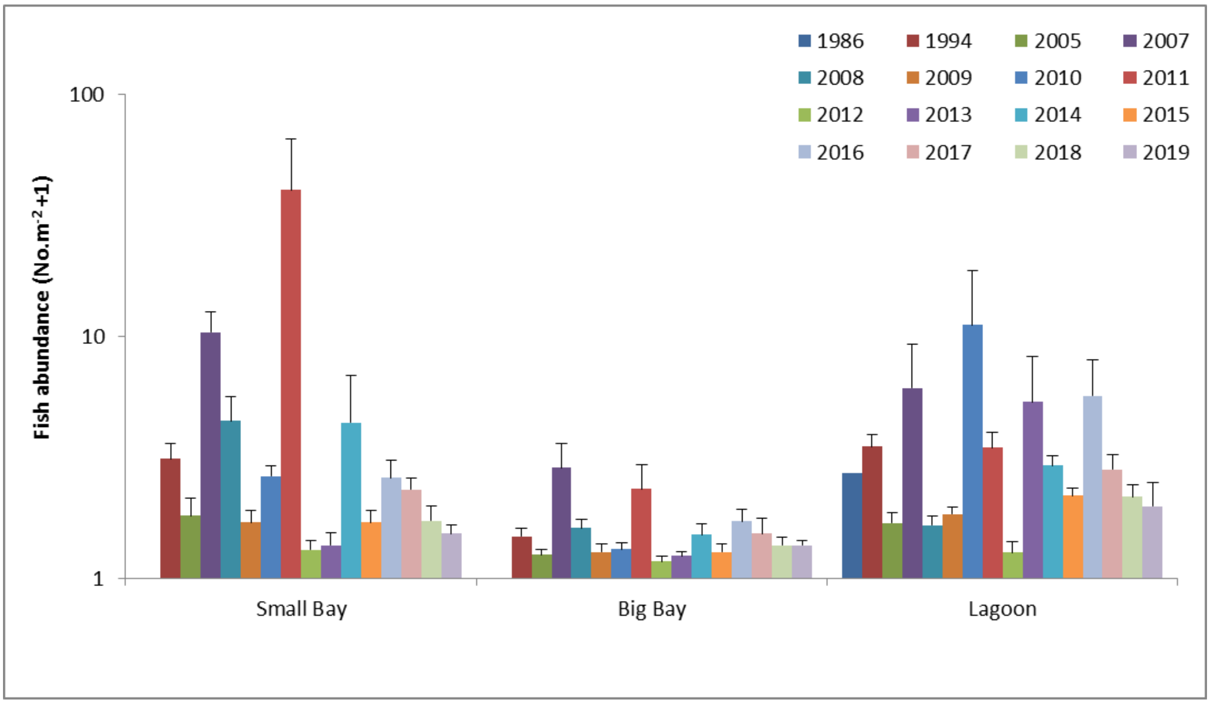


Figure 10.3. Average fish abundance (all species combined) during 16 seine-net surveys conducted in Saldanha Bay and Langebaan lagoon. (Error bars show one Standard Error of the mean). The data are transformed (x + 1) and displayed on a logarithmic axis.

Abundance of white stumpnose, nude goby and blacktail abundance 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 10.5.). 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 trend in white stumpnose and blacktail abundance over the 2012-2015 period in Small Bay appeared to have reversed with the third highest white stumpnose abundance and second highest blacktail abundance recorded in 2016 samples. Blacktail juveniles were entirely absent from Small Bay catches in 2017 and remained scarce in 2018 samples with only five individuals caught, but recovered somewhat in 2019 with 61 individuals caught. White stumpnose abundance was remained significantly down from that recorded during 2016, with only 12 individuals caught.

Within Big Bay too, average harder density observed during the 2013-2018 sampling was comparable to earlier surveys, but the abundance of the four next most common species remains low compared to earlier sampling events (Figure 10.5.). White stumpnose abundance within Big Bay over the period 2015-2018 had recovered somewhat from the very low 2013 and 2014 estimates, but crashed again in 2019 (Figure 10.5.). The strong elf recruitment in Big Bay evident in the 2016 and 2017 sampling was not repeated in 2018 or 2019, with the species absent from Big Bay samples for the only second and third times 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. 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.

With the exception of harders and silversides, the estimated abundance of the more common species in Langebaan lagoon during 2019 compared poorly with earlier surveys with substantial declines in gobies and white stumpnose (Figure 10.5.). The cause behind the observed decline in the abundance of the dominant species in Langebaan Lagoon is not known. Speculatively it could be linked to the prolonged drought in the Western Cape, with reduced freshwater run-off limiting nutrient input from the terrestrial environment. With the more "normal' rainfall experienced during the 2018 and 2019 winters future sampling will hopefully show a recovery in juvenile fish abundance in the Lagoon.

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

More concerning was the 2018 capture of three alien rainbow trout by gill net fishers in Kraalbaai, indicating that fish have escaped from the experimental fish cages in Big Bay, survived and entered the Lagoon. Trout are predators that like many fish 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 escapees from the experimental cages to date, have had a significant impact on juvenile fish abundance in the Lagoon (and elsewhere in Saldanha Bay). Trout, and other salmonids earmarked for fish cage farming in the Saldanha area also 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, indeed during the 2019 survey a workboat was busy retrieving a severely damaged fish cage that had been wrecked during a large swell the week before (Figure 10.4). It is not known if this cage was stocked with trout at the time it was damaged, and if so how man fish escaped (none were caught during the experimental netting survey and no further reports of trout in the gill net fishery have been forthcoming). 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.

Figure 10.4. Work boat busy retrieving a storm damaged fish cage in Big Bay on 12 April 2019.

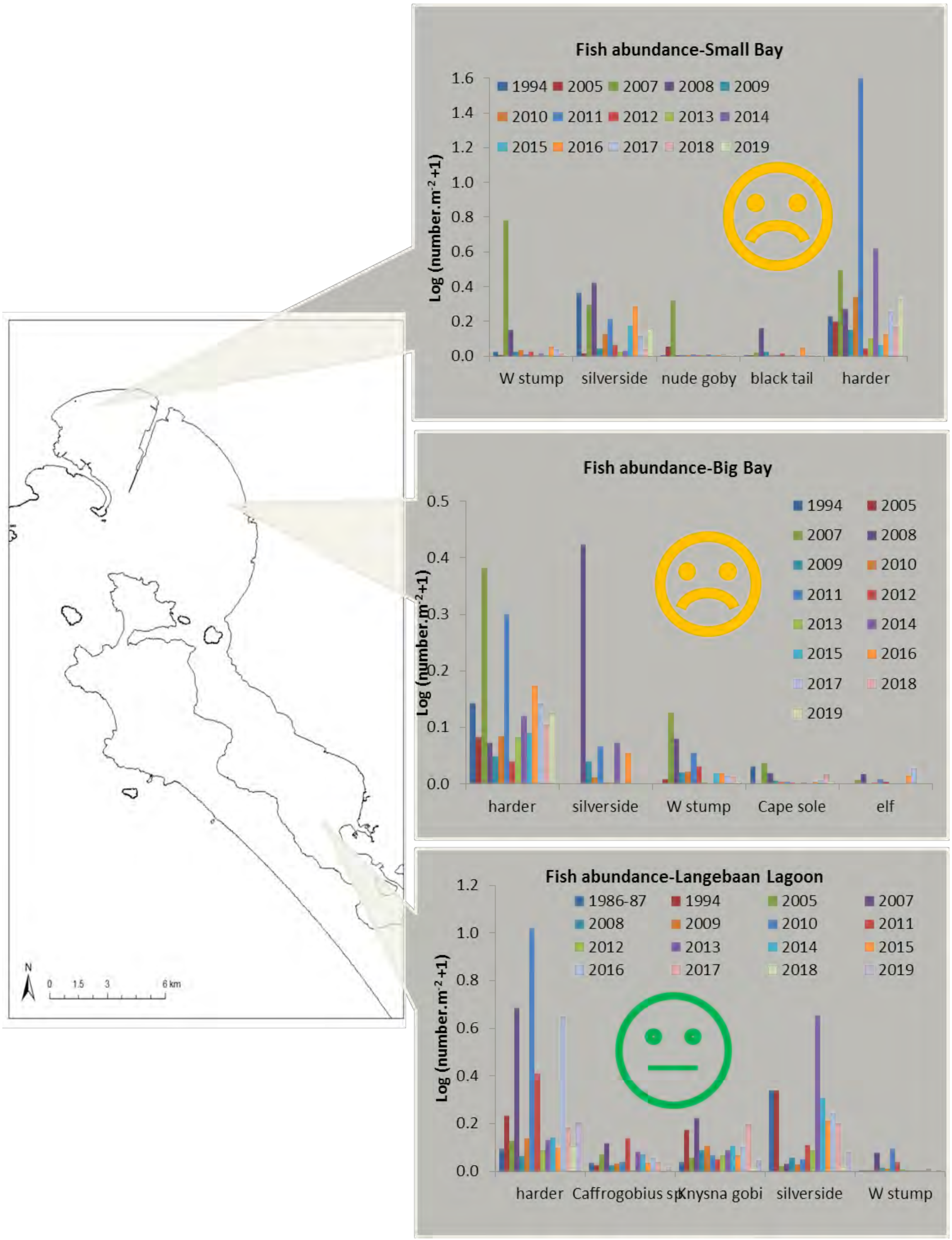


Figure 10.5. **Abundance of the most common fish species recorded in annual seine-net surveys within Saldanha Bay and Langebaan Lagoon (1986/87, 1994, 2005 & 2007-2019).**

10.3.3 Status of fish populations at individual sites sampled in 2018

The average abundance of the four most common species in catches made during all earlier surveys and the most recent 2019 survey at each of the sites sampled is shown in Figure 10.6., Figure 10.7. and Figure 10.8. 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 and white stump at Small Bay sites in the 2019 survey was lower than the long term average recorded in earlier surveys, whilst 2019 silverside catches were similar; and harder catches were greater (Bluewater bay), similar (Hoedtjiesbaai and Small craft harbour) or less (Campsite) (Figure 10.6.). At all the Big Bay sites, catches of harders during the 2019 survey were similar to the historical average with no clear spatial trend; whilst silversides were again, for the second consecutive year in the sampling record, absent from most sites (Figure 10.7.). White stumpnose catches at five of the six Big Bay sites in 2019 were less than the long-term average (Figure 10.7.). Catches of harders at Lagoon sites during 2019 were similar to the long term average; whilst goby and white stumpnose density was significantly lower at five of the six sites and similar to the long-term average only at Rietbaai (Figure 10.8). In summary, there have been significant, ongoing declines in white stumpnose density at sites throughout the system, declines in gobies at nearly all sites in Small Bay and the Lagoon; and declines in silversides at all sites in Big Bay. At most sites, harder abundance was similar to the historical average. Observed declines occurred across many sites and suggest larger system-wide impacts (i.e. fishing pressure in the case of white stumpnose) or natural fluctuations in abundance and distribution (e.g. silversides in Big Bay), rather than point source impacts.

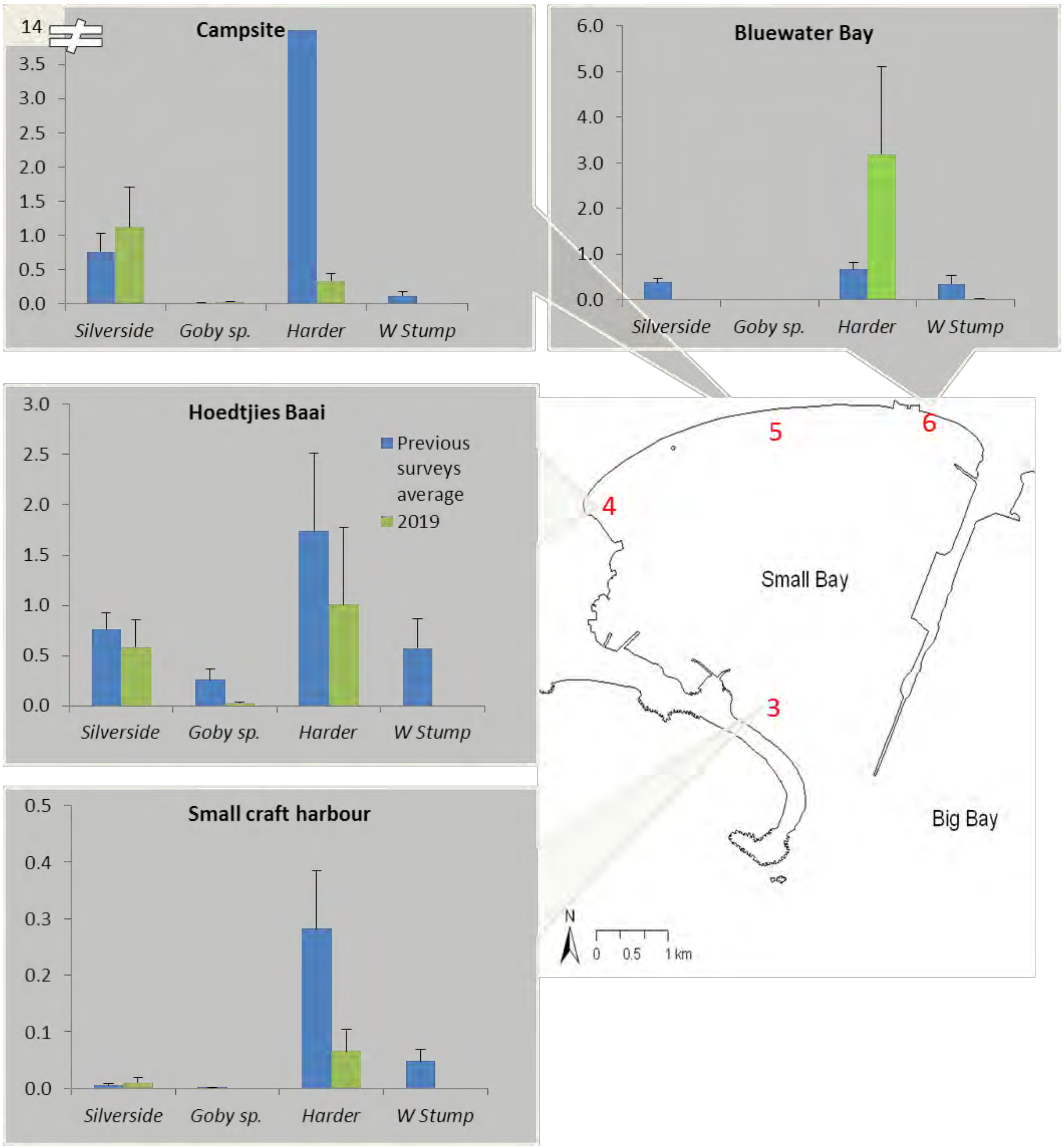


Figure 10.6. **Average abundance (No. fish.m^{-2}) of the four most common fish species at each of the sites sampled within Small Bay during the earlier surveys (1994, 2005, 2007-2018) and during the 2019 survey. Errors bars show plus 1 Standard error.**

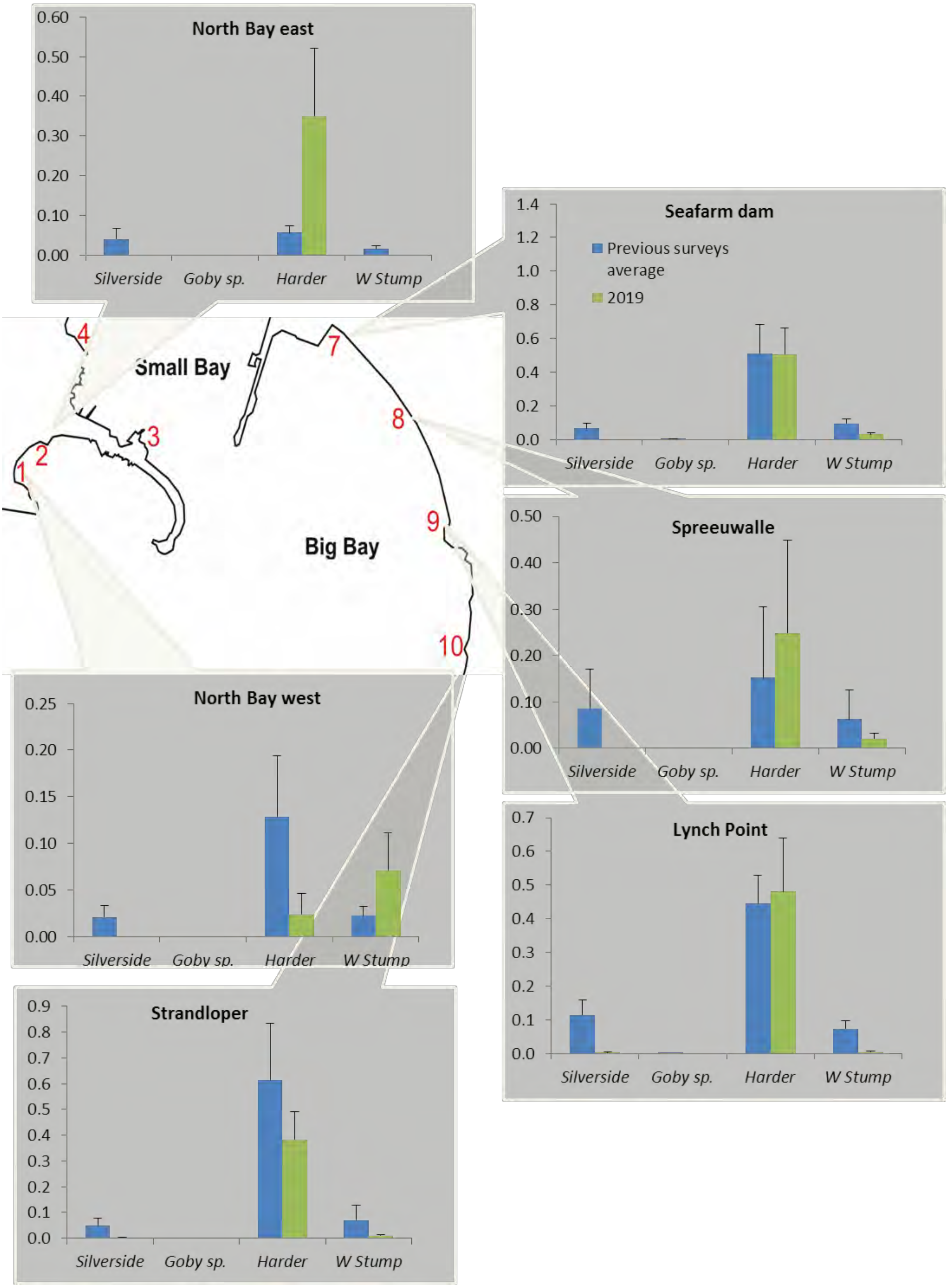


Figure 10.7. **Average abundance (#fish.m^{-2}) of the four most common fish species at each of the sites sampled within Big Bay during the earlier surveys (1994, 2005, 2007-2018) and during the 2019 survey. Errors bars show plus 1 Standard error.**

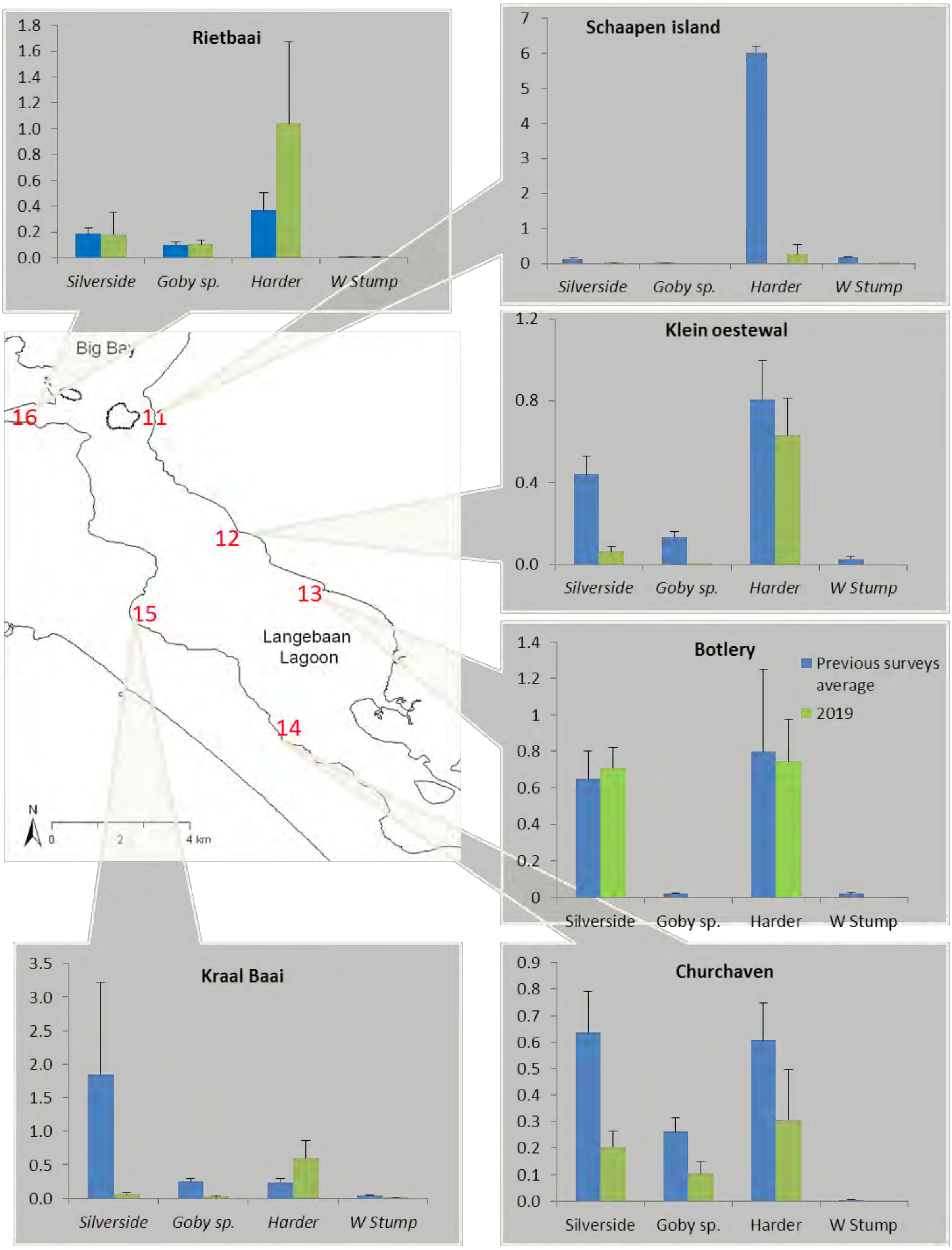
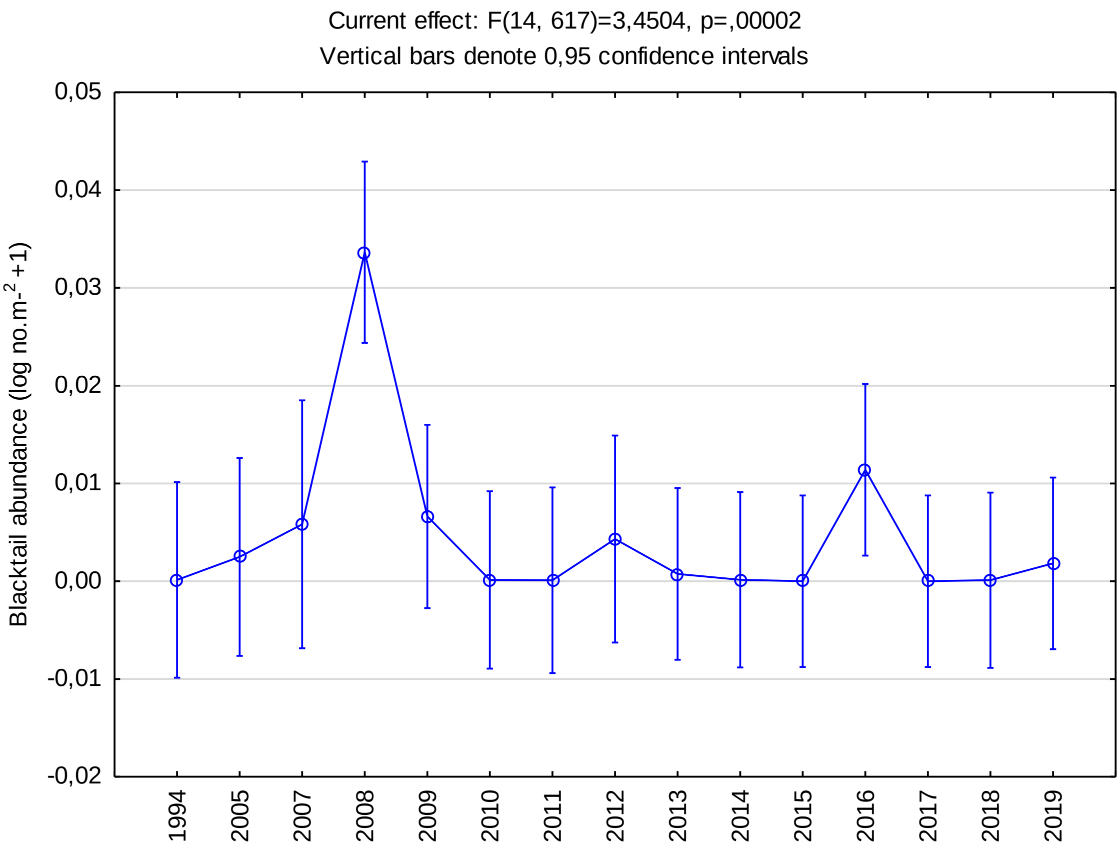


Figure 10.8. **Average abundance (#fish.m^{-2}) of the four most common fish species at sites sampled within Langebaan lagoon during the earlier surveys (1994, 2005, 2007-2018) and during the 2019 survey. Error bars show plus 1 standard error.**

10.4 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 2017-2019 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), 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. To further investigate temporal variation in recruitment of species important in the Bay's fisheries (harders, blacktail, elf and white stumpnose) univariate statistical analysis (ANOVA) was used to test for significant differences in abundance between survey years. To deal with the observed spatial variability in survey catches and to account for the fact that Saldanha Bay-Langebaan Lagoon is a single system; and different sites may be more utilized by juvenile fish in different years depending on prevailing weather and oceanography, abundance data for 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 10.9, Figure 10.10, Figure 10.11).

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 the 2018 samples and 61 in 2019 samples (Figure 10.9). Inter annual variation in the abundance of harders was greatest, with estimated abundance in 2007, 2010 and 2011 significantly greater than most other sampling events. The abundance of juvenile harders in 2019 hauls was similar to the median and only significantly less than that recorded in 2005 and 2011. Estimated white stumpnose abundance in 2007 was significantly greater than all other years, whilst the estimated abundance during 2013-2015 and 2019 surveys was less than during nearly all other survey years. Despite a small increase in abundance of juvenile white stumpnose in 2016, the 2017 and 2018-19, white stumpnose abundance estimates remained low, and were not significantly different from the abundance estimates recorded post-2008. Steentjie and elf abundance also showed 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 a zero catch in 2018 and a lower than average catch in 2019.. The spatial variability in abundance of these two species, a result of a zero catch at many sites, however, means that these differences are not statistically significant.



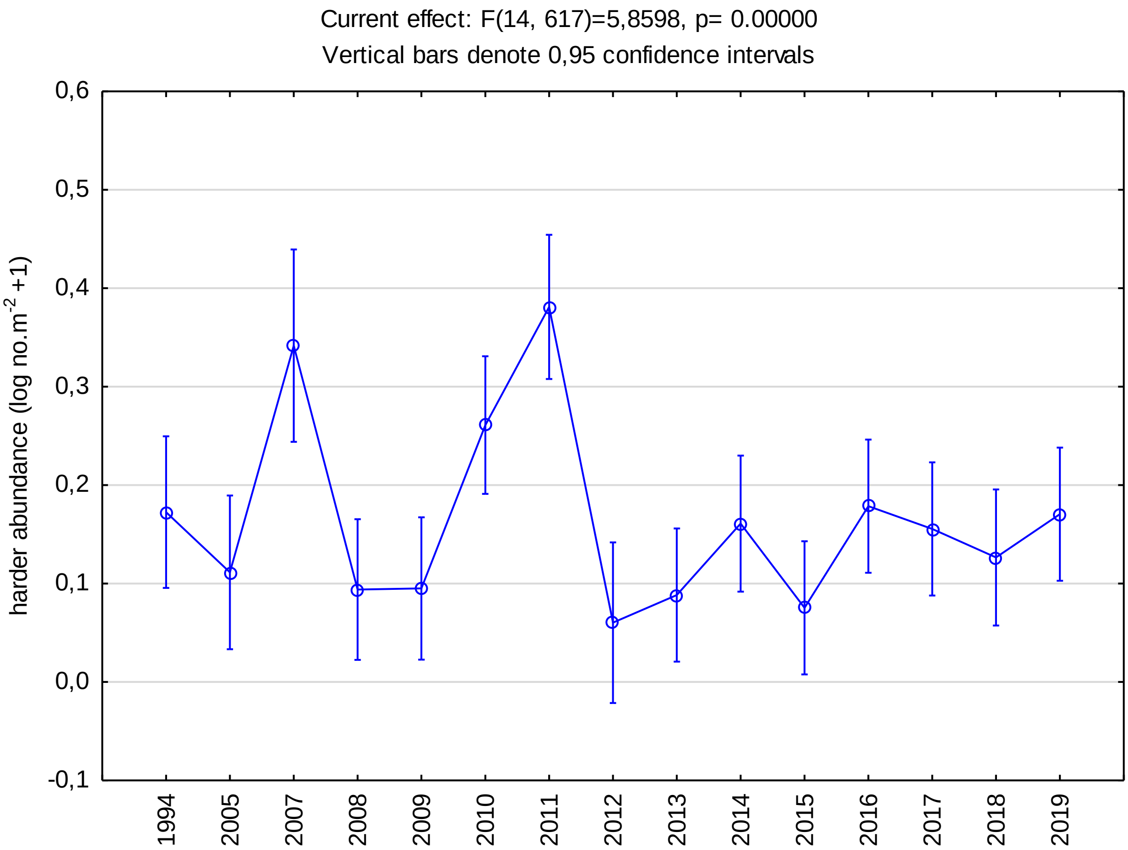


Figure 10.9. ANOVA results comparing the average annual density of blacktail (top) and harders (bottom) at all sites sampled in all surveys (1994-2019).

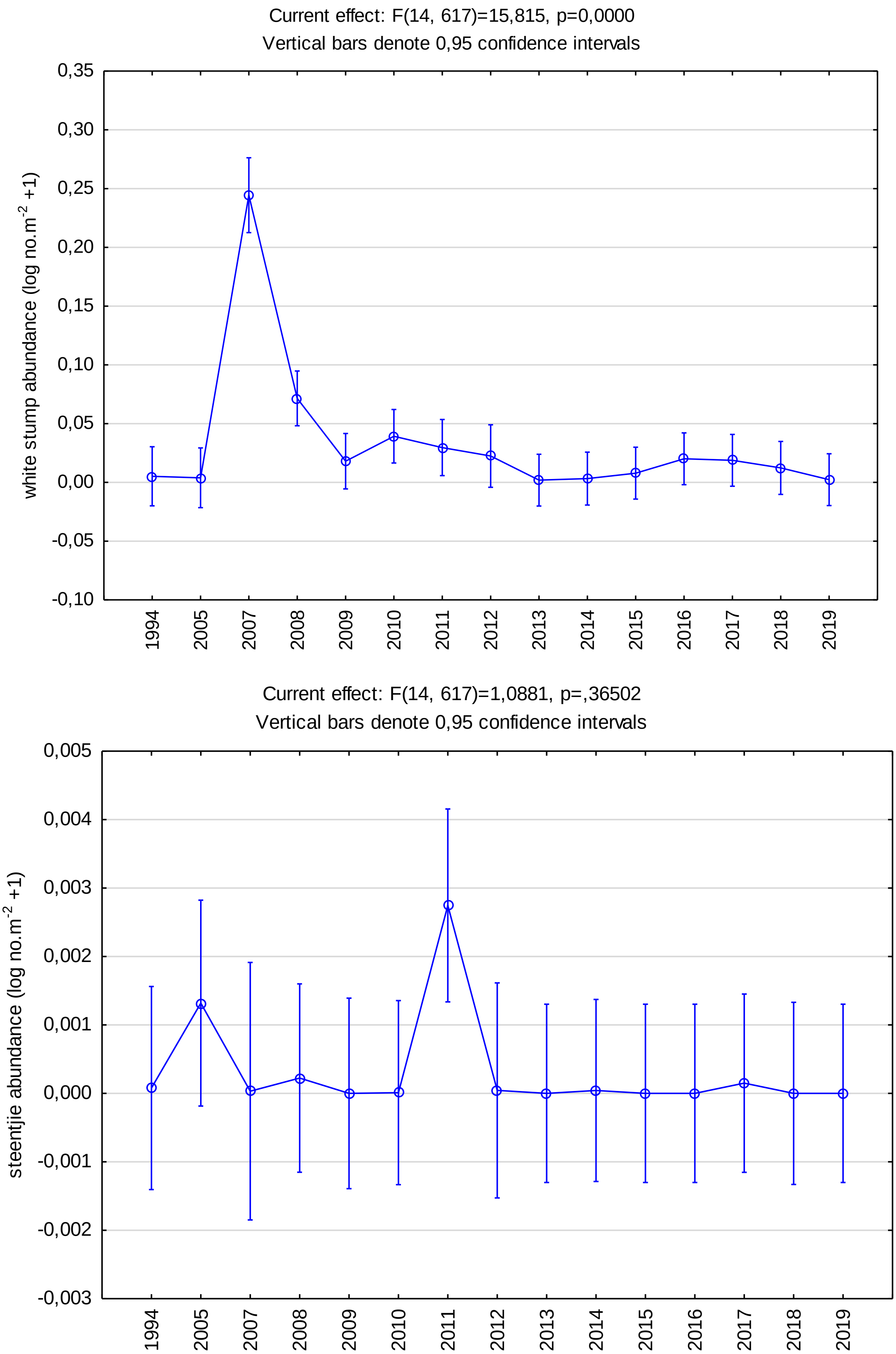


Figure 10.10. ANOVA results comparing the average annual density of white stumpnose (top) and steentjies (bottom) at all sites sampled in all surveys (1994-2019).

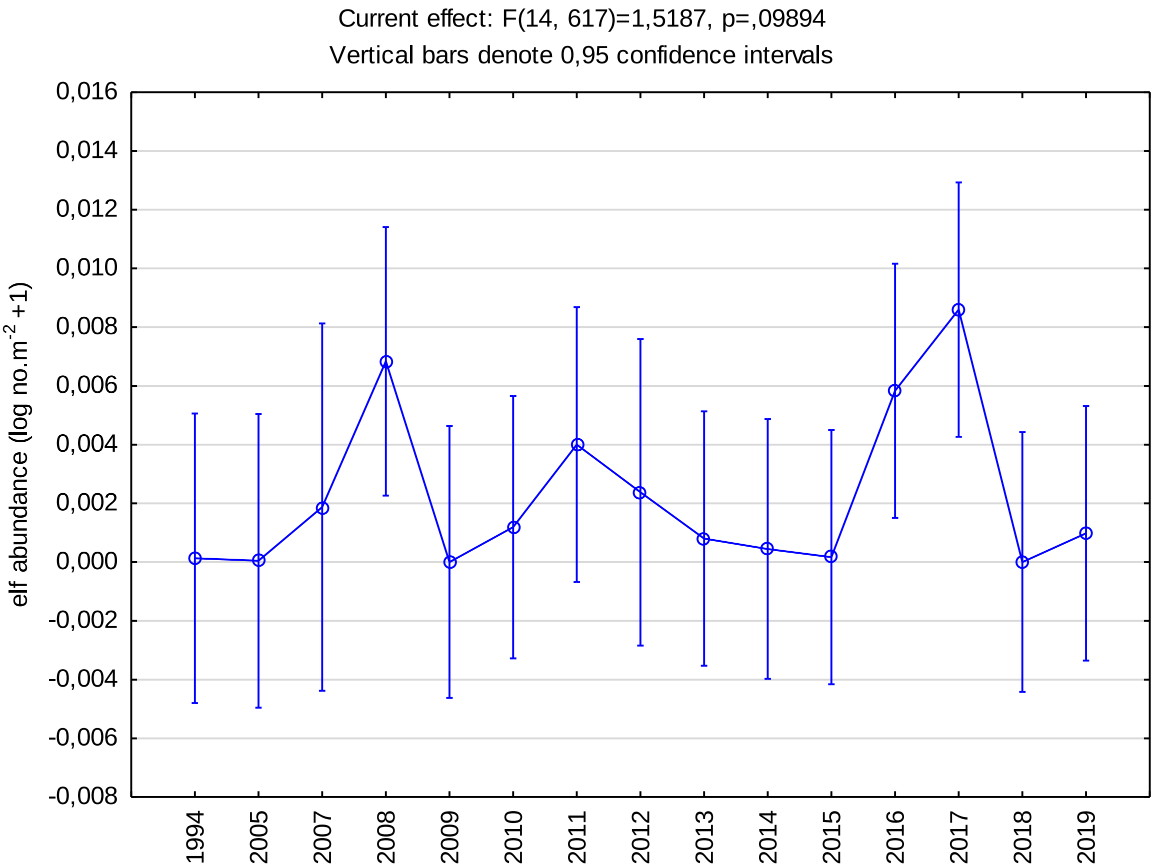


Figure 10.11. ANOVA results comparing the average annual density of elf at all sites sampled in all surveys.

10.5 Conclusion

The 2019 seine net survey revealed 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 2017-2019, 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. Fish diversity and overall abundance does not, however, show a declining trend in Small Bay 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 2018 and 2019 sampling, this suggests that these strong year classes are not yet contributing juveniles to the stock in significant numbers. Silversides were absent in Big Bay in 2018 and very scarce in 2019 samples. Furthermore, five species that were usually present in Big Bay surveys were absent in 2019 (False Bay klipvis, Super klipvis, elf, sandsharks and pipefish) leading to the lowest diversity in 15 annual surveys with just eight species in Big Bay samples. None of these “missing” species are targeted in fisheries in the area and the reason for their absence from 2019 catches is unknown. Harders were present in Langebaan lagoon samples in similar numbers to previous surveys, but catches of all other common species, particularly gobies remained low compared to previous surveys.

Previously, 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. These predatory fish will prey on indigenous invertebrates and fish and 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 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, 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, however, suggest that the protection afforded by the Langebaan MPA is not be enough to sustain the fishery at the current high effort levels. Arendse (2011) found the adult stock to be overexploited using data collected during 2006-08 already, and the evidence from the seine net surveys conducted since then certainly suggests that recruitment overfishing has occurred. The annual seine net surveys can act as an early warning system that detects poor recruitment and allows for timeous adjustments in fishing regulations to reduce fishing mortality on weak cohorts and preserve sufficient spawner biomass. The consistent declining trend in juvenile white stumpnose abundance in the nursery surf-zone habitats since 2007, and the observed declines in commercial linefish CPUE, strongly supports 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. This is the fifth time Anchor Environmental are making this recommendation in the State of the Bay Report and these recommendations are now also supported by a more statistically comprehensive analysis of fishery dependent and survey data (Parker *et al.* 2017). 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 popular white stumpnose fishery is 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 six 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 (2018) for a reduction in harder fishing effort and gear changes (increase in minimum mesh size) to facilitate stock recovery.

11 BIRDS

11.1 Introduction

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. 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 and 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. Waterbirds are counted annually on all the islands, and bi-annually in Langebaan Lagoon as part of the Collected Water Counts (CWAC) Programme conducted by the Animal Demography Unit (ADU) at the University of Cape Town (UCT).

11.2 Birds of Saldanha Bay and the islands

11.2.1 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) 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 11.2.1.1 provides data on long-term trends of each of these important seabirds and the African Oystercatcher, using the data collated by the DEA.

11.2.1.1 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 185 breeding pairs in 2018 (Figure 11.1.). 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.

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.

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 11.1.). 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 11.2). 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 11.2). 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 11.2). In the last five years however, the estimated sardine biomass along the west coast has declined dramatically, with almost none detected in the 2018 acoustic survey (Figure 11.2). Anchovy biomass too has recently declined to about 1.5 million tonnes in 2018 and the estimated biomass on the west coast is at its second lowest level since the turn of the century (Figure 11.2). 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).

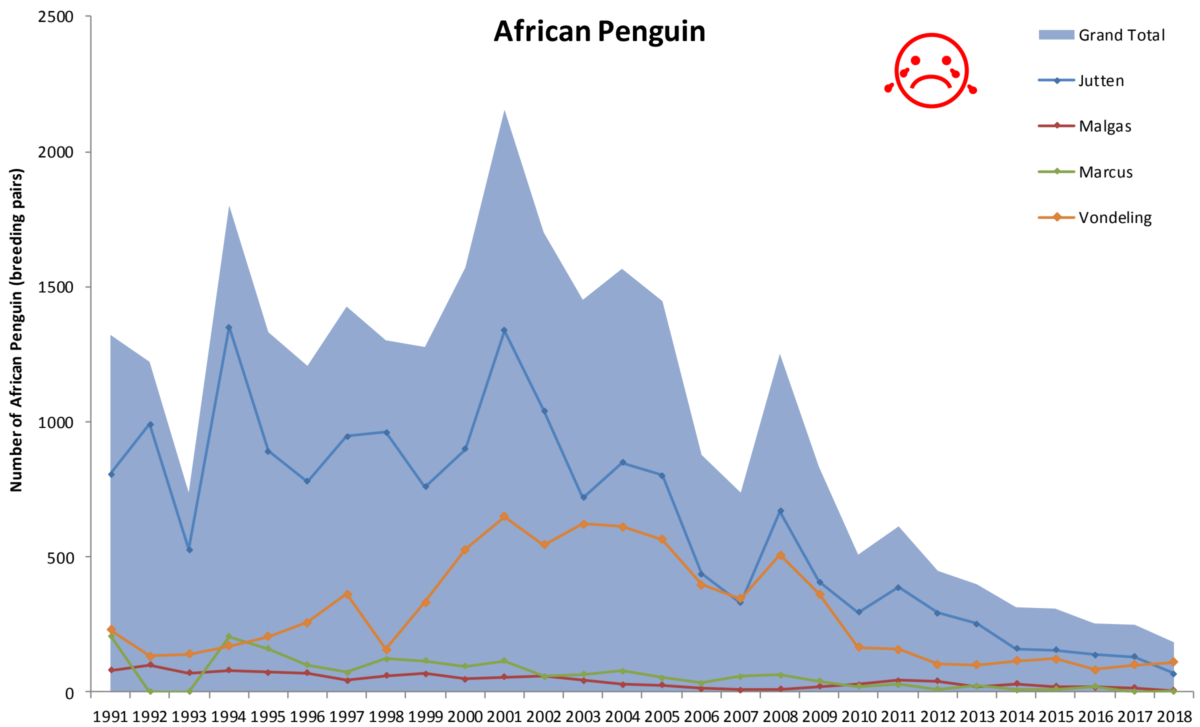
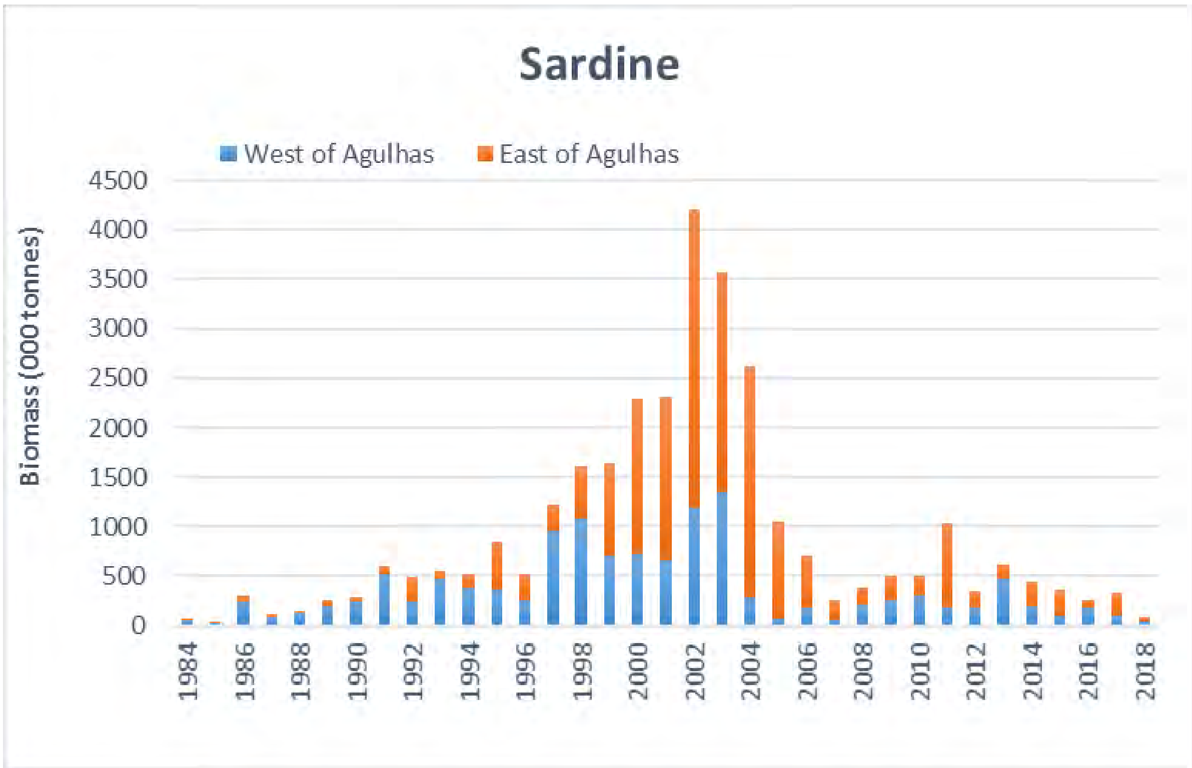


Figure 11.1. Trends in African Penguin populations at Jutten, Malgas, Marcus and Vondeling islands in Saldanha Bay from 1991-2018 measured in number of breeding pairs (Data source: Department of Environmental Affairs: Oceans & Coasts, 2019).



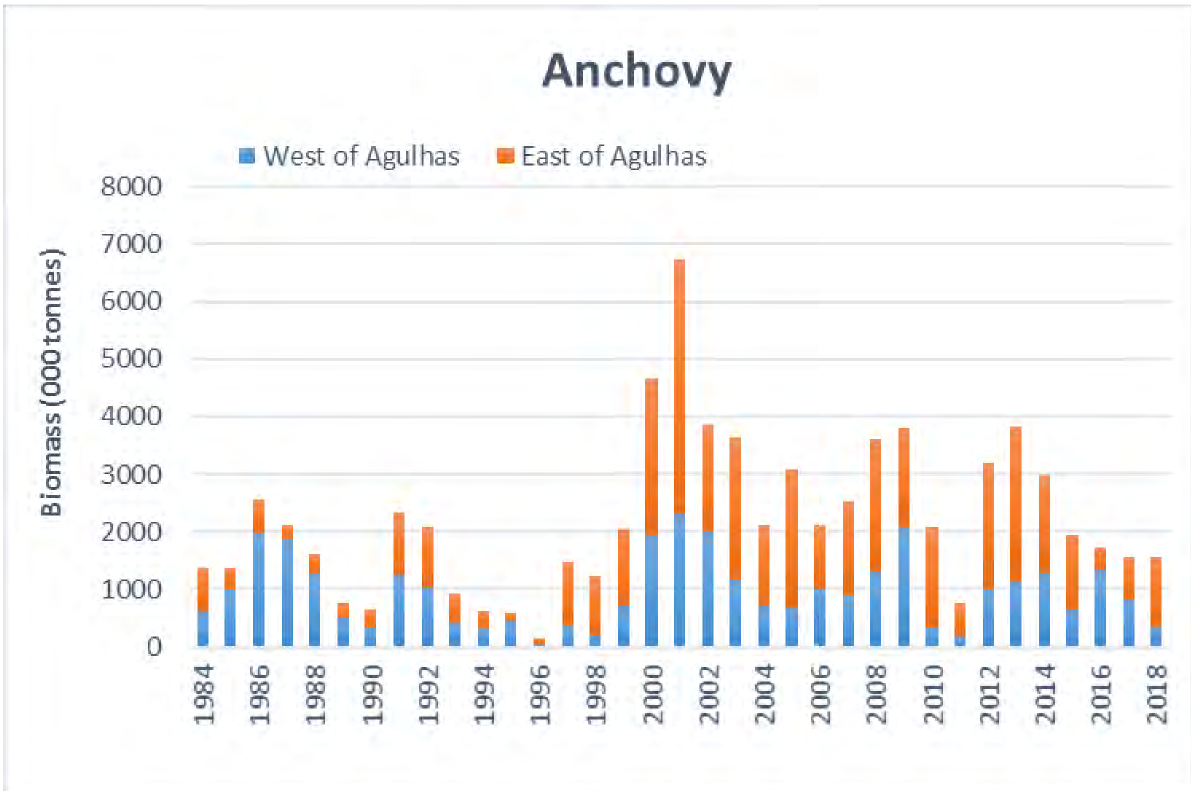


Figure 11.2. **Long term trends in the biomass of small pelagic fish (sardine and anchovy) to the west and east of Cape Agulhas based on hydro acoustic surveys conducted bi-annually from 1984-2018 (Data source: Department of Agriculture Forestry and Fisheries).**

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 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).

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. 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 2018 are the lowest on record for the eighth 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) (Figure 11.3.). 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 11.3.). Recent counts show that Kelp Gull numbers remain below those at the start of the comprehensive counting period. This reflects the continued impacts of Pelican predation as well as other anthropogenic pressures. 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 is unknown, however.

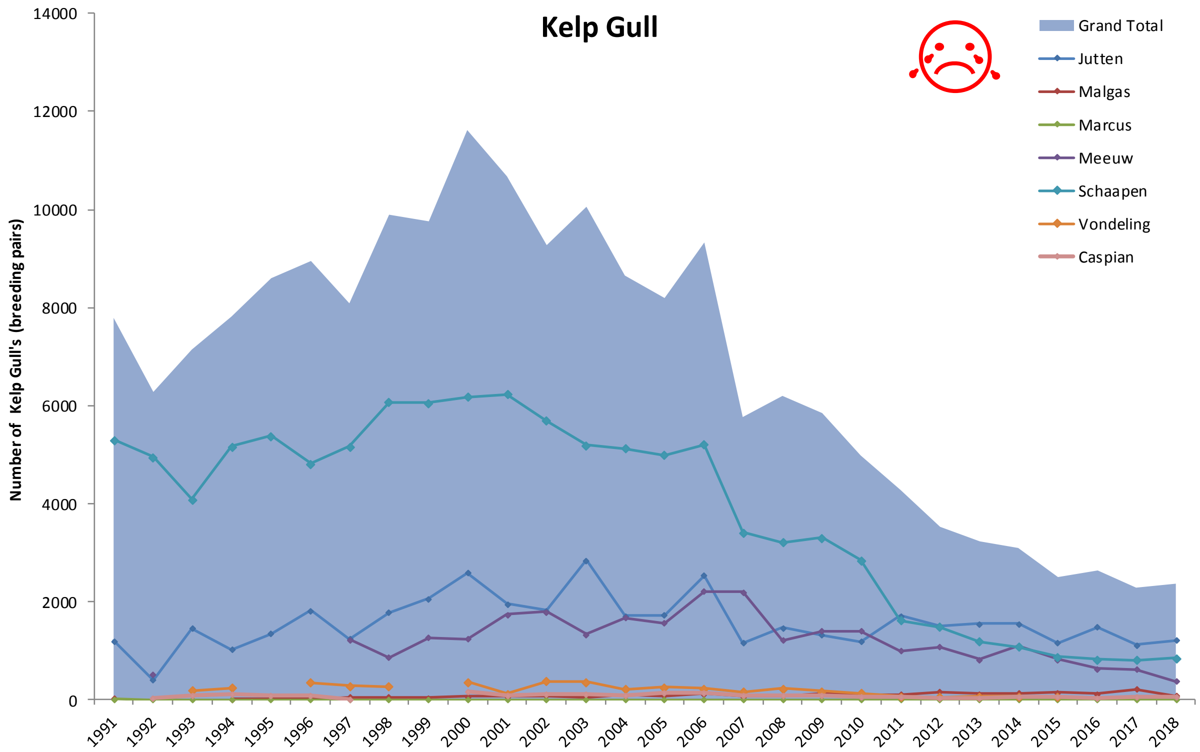


Figure 11.3. **Trends in breeding population of Kelp gulls at Jutten, Malgas, Marcus, Meeuw, Schaapen, Vondeling and Caspian Islands in Saldanha Bay from 1985 – 2018 measured in number of breeding pairs (Data source: Department of Environmental Affairs: Oceans & Coasts, 2019).**

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 for during 2012-2014. 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. Concern was recently expressed over the fact that breeding appeared to have ceased at Schaapen Island during the period 2008-2011. The number of pairs breeding on Schaapen Island did, however, recover dramatically with 925 pairs recorded in 2012 (Figure 11.4.). The total number of breeding pairs recorded in 2016 was 303 found almost exclusively on Malgas Island. Number of breeding pairs in 2018 are amongst the lowest levels on record.

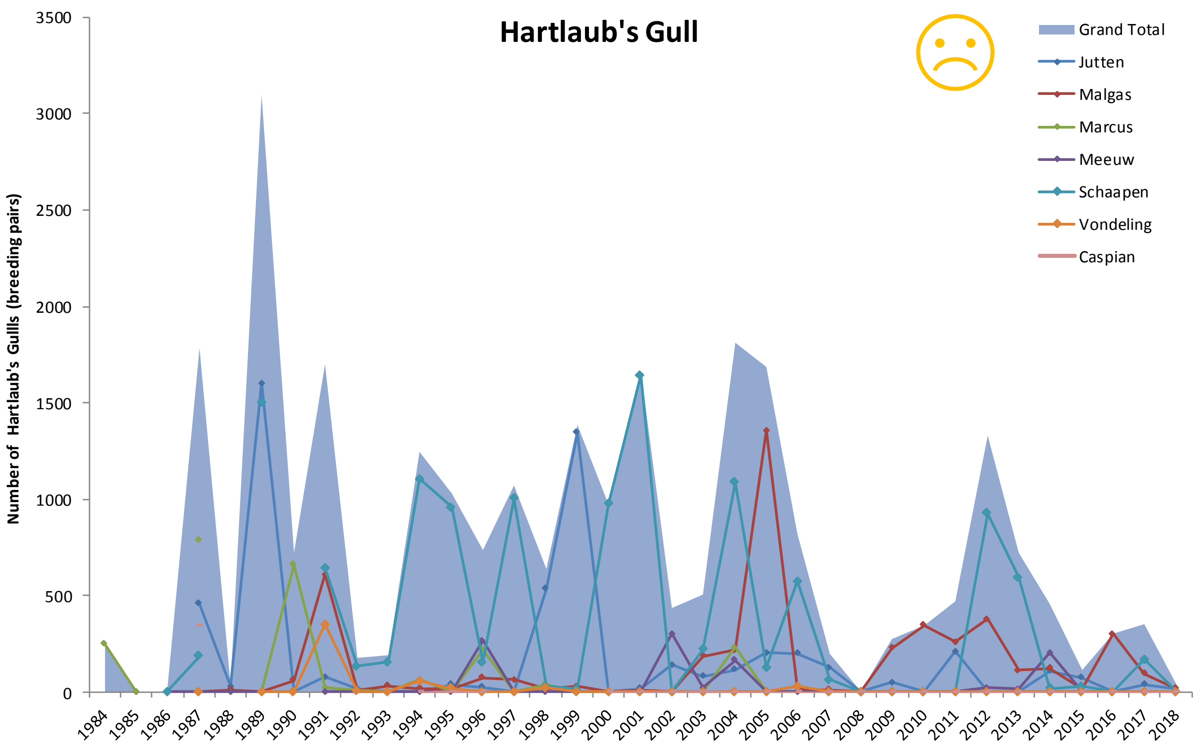


Figure 11.4. Trends in breeding population of Hartlaub's Gulls at Jutten, Malgas, Marcus, Meeuw, Schaapen, Vondeling and Caspian Islands in Saldanha Bay from 1984 – 2018 measured in number of breeding pairs (Data source: Department of Environmental Affairs: Oceans & Coasts 2019).

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 11.2). 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. It is encouraging therefore that the birds retuned again in 2011 and that numbers recorded in 2018 are amongst the highest on record (Figure 11.5.).

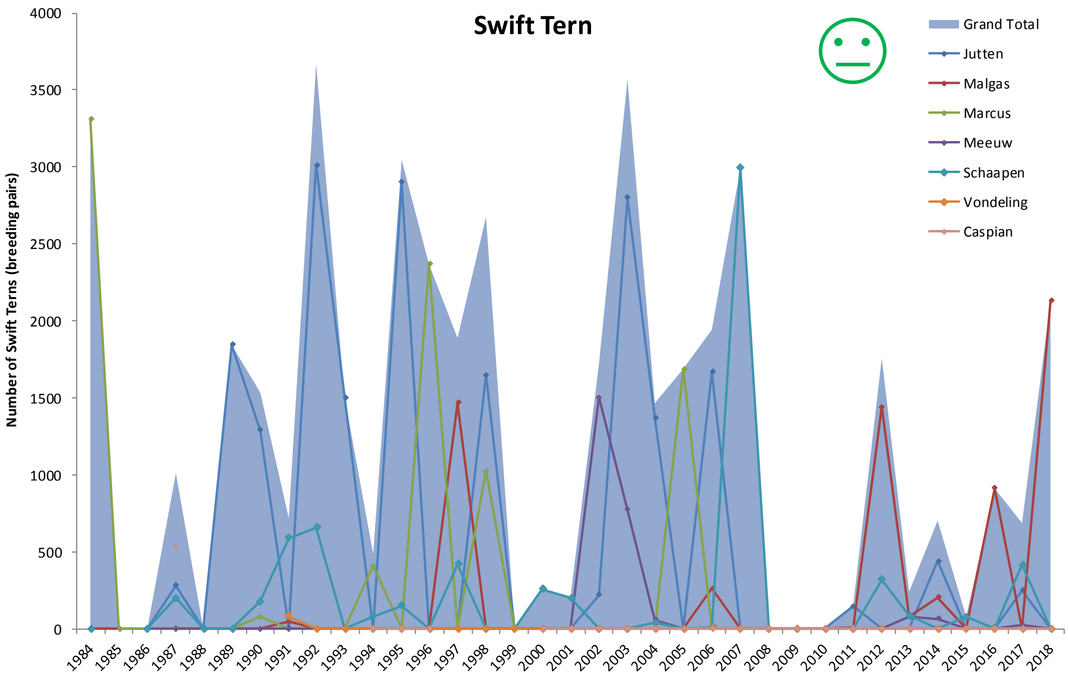


Figure 11.5. **Trends in breeding population of Swift Terns at Jutten, Malgas, Marcus, Meeuw, Schaapen, Vondeling and Caspian Islands in Saldanha Bay from 1984 - 2018 measured in number of breeding pairs (Data source: Department of Environmental Affairs: Oceans & Coasts 2019).**

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 a steady decline since 1996 (Figure 11.6.). The 2012-2018 data reveal that the breeding population on Malgas Island has fallen to record low levels. 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 fishery discards (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 11.2). 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).

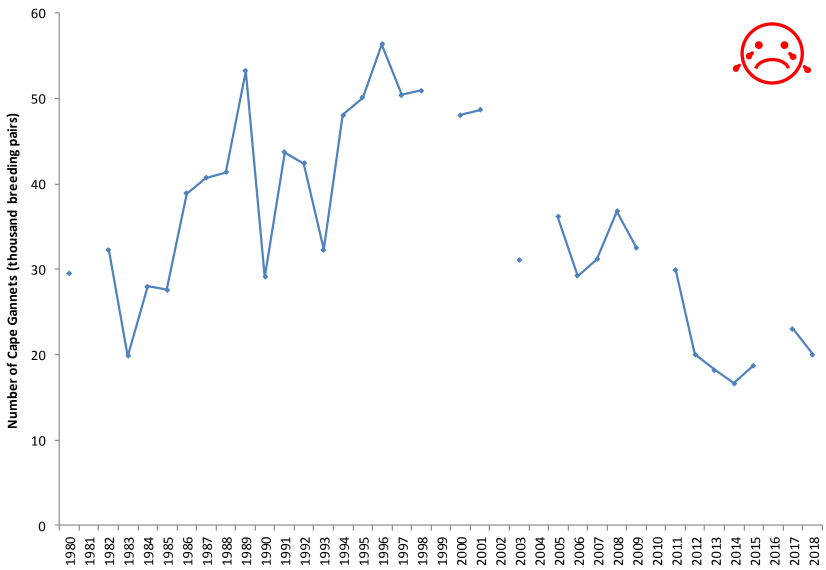


Figure 11.6. Trends in breeding population of Cape Gannets at Malgas Island, Saldanha Bay from 1980 – 2018 measured in number of breeding pairs (Data source: Department of Environmental Affairs: Oceans & Coasts).

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 uplisted 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 remains high (Figure 11.2). 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. 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 Jutten Island in recent years (Figure 11.7.). Overall, the number of breeding pairs has declined gradually since the 1990s. In 2013, a total of only 801 breeding pairs were recorded, representing the lowest level recorded to date (Figure 11.7.). Between 2013 and 2016, a short-lived recovery of breeding pairs to 9273 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 3332 in 2018, which is amongst the lowest numbers on record.

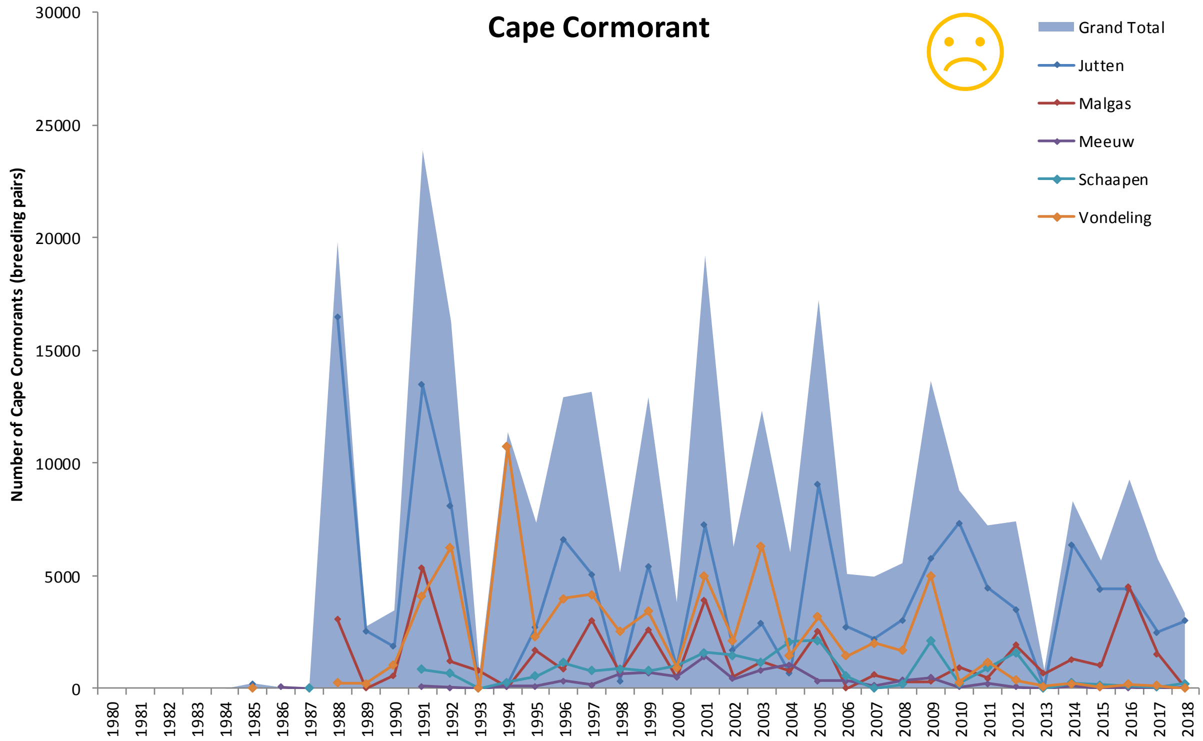


Figure 11.7. Trends in breeding population of Cape Cormorants at Jutten, Malgas, Meeuw Schaapen, and Vondeling islands in Saldanha Bay from 1980 – 2018 measured in number of breeding pairs (Data source: Oceans & Coasts, Department of Environmental Affairs 2019).

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, crustaceans and cephalopods, feeding mainly amongst kelp where they catch West Coast rock lobster, *Jasus lalandii* and pelagic goby *Sufflogobius bibarbatus* (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 3% of pristine biomass (Cockcroft *et al.* 2008, DAFF 2015). 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 93% since 1990 (Figure 11.8.). Currently numbers of breeding pairs are the lowest on record. 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.

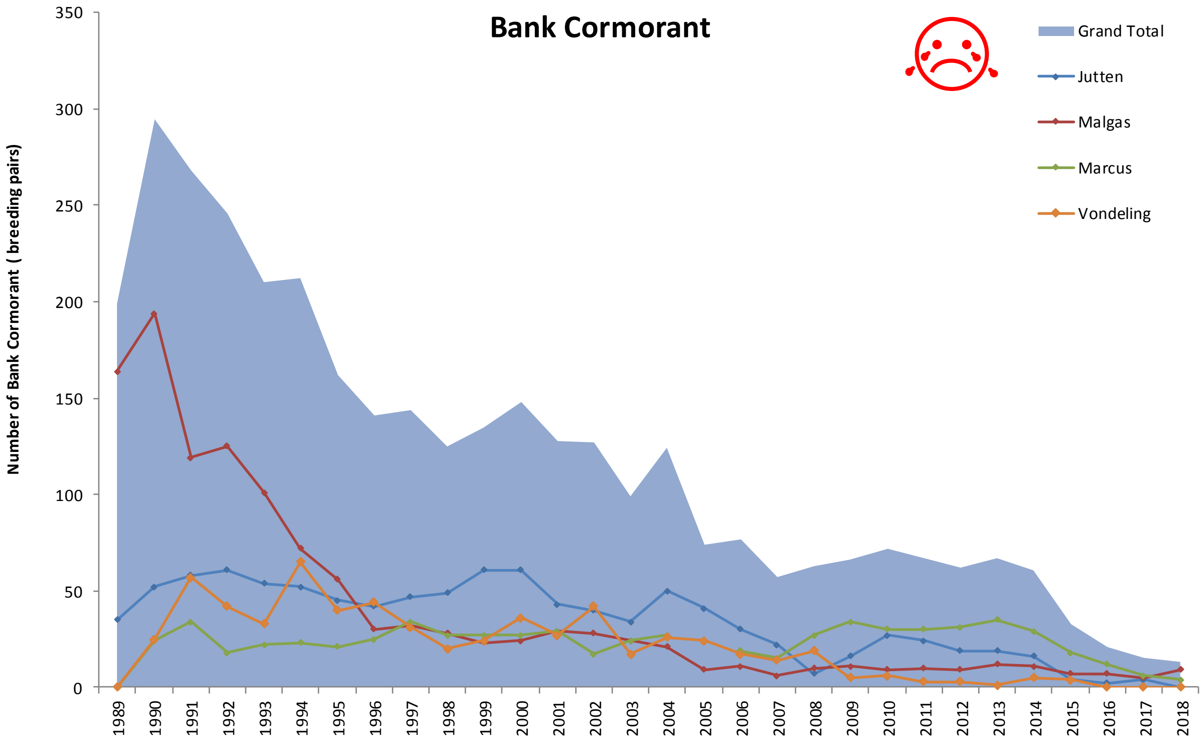


Figure 11.8. **Trends in breeding population of Bank Cormorants at Jutten, Malgas, Marcus and Vondeling islands in Saldanha Bay from 1980 – 2018 measured in number of breeding pairs (Data source:, Oceans & Coasts, Department of Environmental Affairs 2019).**

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 11.9.). Overall, numbers of breeding pairs were more or less stable until 2012 but have declined steeply since then. The last four annual counts (2015-2018) have been the lowest on record.

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 be an important source of disturbance to these birds. The substantial growth in participation in recreational water sports (particularly kite boarding) over the last decade 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.

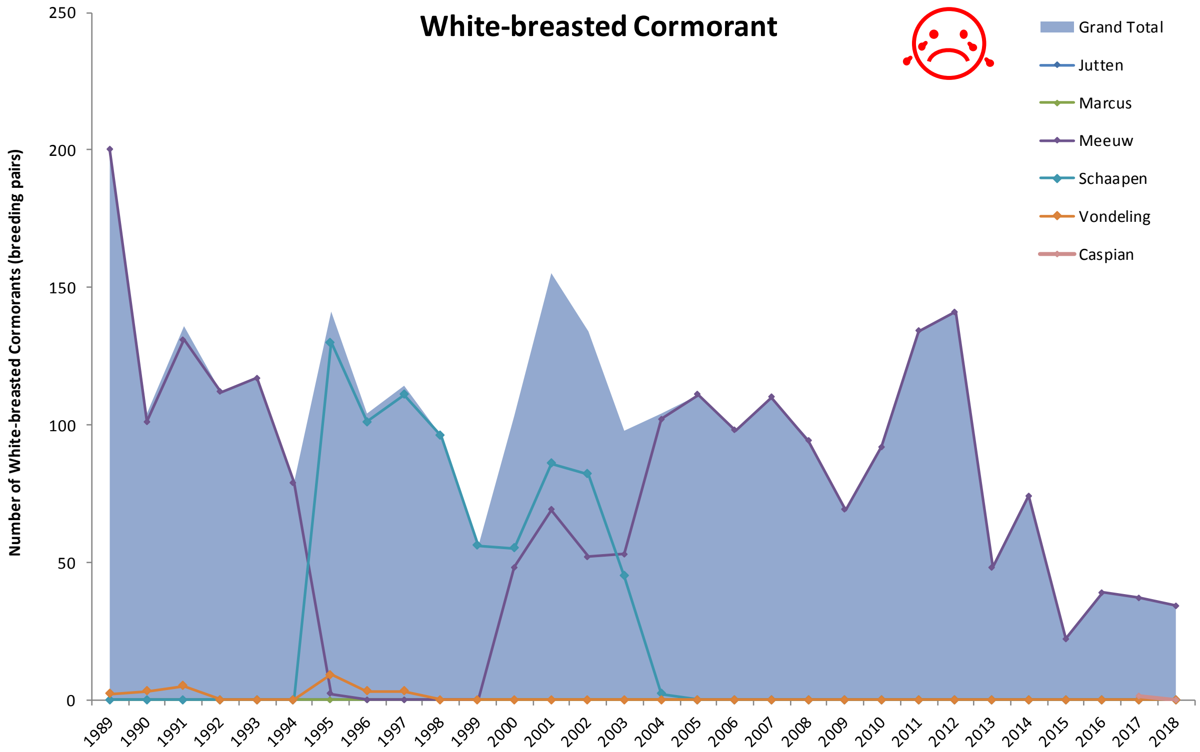


Figure 11.9. **Trends in breeding population of White-breasted Cormorants at Jutten, Marcus, Meeuw, Schaapen, Vondeling and Caspian islands in Saldanha Bay from 1980 – 2018 measured in number of breeding pairs (Data source: Department of Environmental Affairs: Oceans & Coasts 2019).**

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 11.10.). Since then, numbers have shown considerable interannual variations with an overall decreasing trend (Figure 11.10.). Currently, numbers are below average, but certainly not the lowest on record.

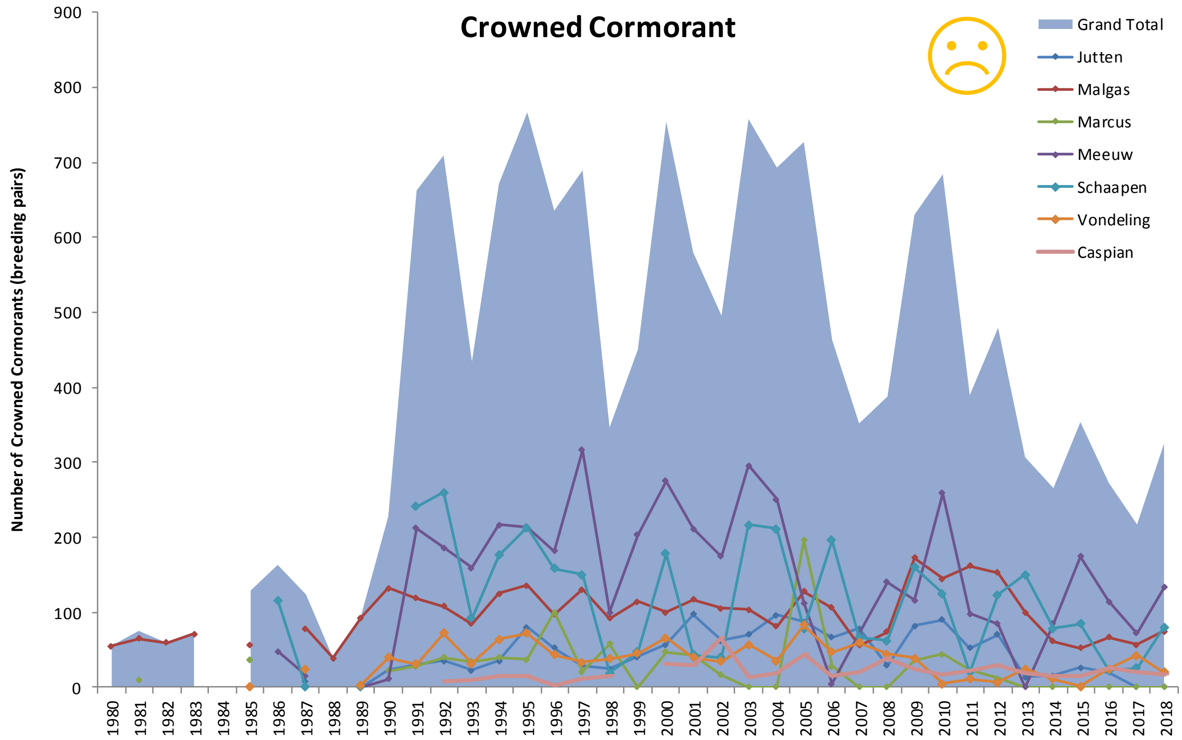


Figure 11.10. **Trends in breeding population of Crowned Cormorants at the Jutten, Malgas, Marcus, Meeuw, Schaapen, Vondeling, and Caspian islands in Saldanha Bay from 1980 – 2018 measured in number of breeding pairs (Data source: Department of Environmental Affairs: Oceans & Coasts 2019).**

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 (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 11.11.). 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 11.11.). 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 are now conducted by the DEA . Unfortunately, no data was however collected in 2017 and 2018.

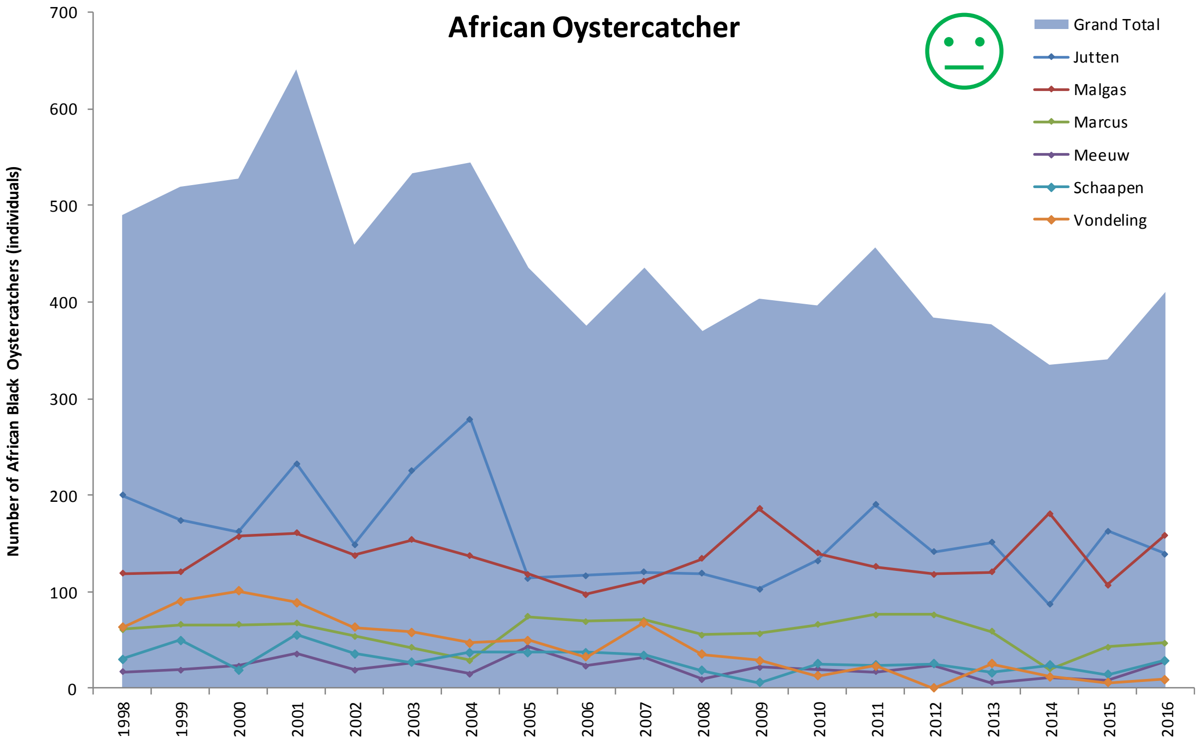


Figure 11.11. Trends in breeding population of African Oystercatchers on Jutten, Malgas, Marcus, Meeuw, Schaapen, and Vondeling Islands from 1988 - 2016. (Data source: Department of Environmental Affairs: Oceans & Coasts 2019).

11.3 Birds of Langebaan Lagoon

11.3.1 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 two decades and the more recent increases in sporting activities on the lagoon impact on some of the important feeding areas in the lagoon.

11.3.2 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 11.1). 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 11.12). Currently, waders account for about 52% of the birds in Langebaan Lagoon during summer, nearly all of these being migratory. In winter, the contribution by resident waders increases to around 9%, and numbers of wading birds increase from 24 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 14% 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 11.1 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.

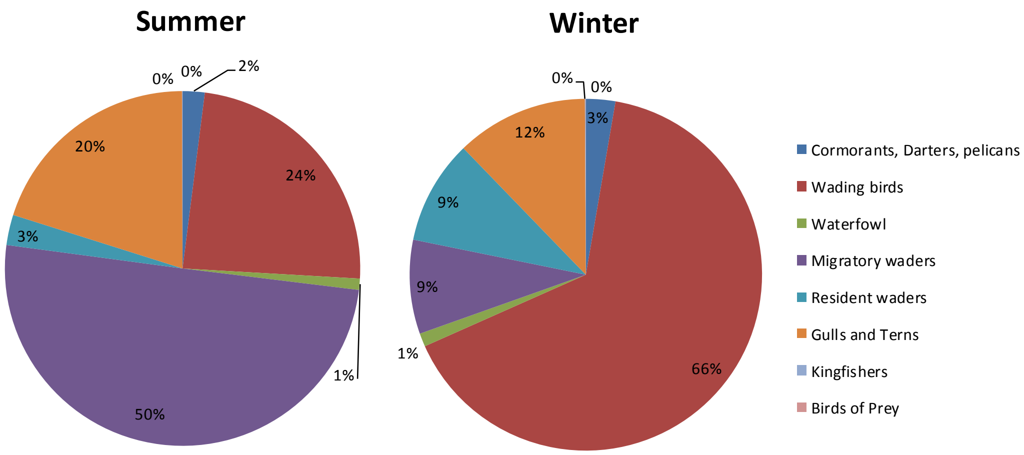


Figure 11.12 **Present average numerical composition of the waterbirds on Langebaan Lagoon during summer (left) and winter (right) and winter (2014-2019) (Data source: CWAC data, Animal Demography Unit at the University of Cape Town).**

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 ten different taxonomic orders (Table 11.1). A total of 115 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 11.1). 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. For a full species list and the average and maximum counts of non-passerine waterbirds for the period 1976-2019 see (Table 11.1).

Table 11.2. 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**

11.3.3 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 11.13). 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 2 352 individuals recorded in summer 2011. Since 2011, numbers have fluctuated around 6000 individuals (Figure 11.13 and Figure 11.14). Today (since 2011), waders make up only 29-71% of summer water bird numbers (Figure 11.13). Total numbers of bird counted in the lagoon in Summer of 2019 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 3615 migratory waders in summer 2019 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 11.15) 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 1 335 birds in 2019. 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 2013 with 1273 birds (Figure 11.14). This notwithstanding, resident bird numbers appear to be on a negative trajectory since 2007 and it remains to be seen whether bird numbers will recover.

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 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 and swimming have since been controlled within the Lagoon through zonation. Nevertheless, some important feeding areas lie within the zones that are highly utilised for recreation.

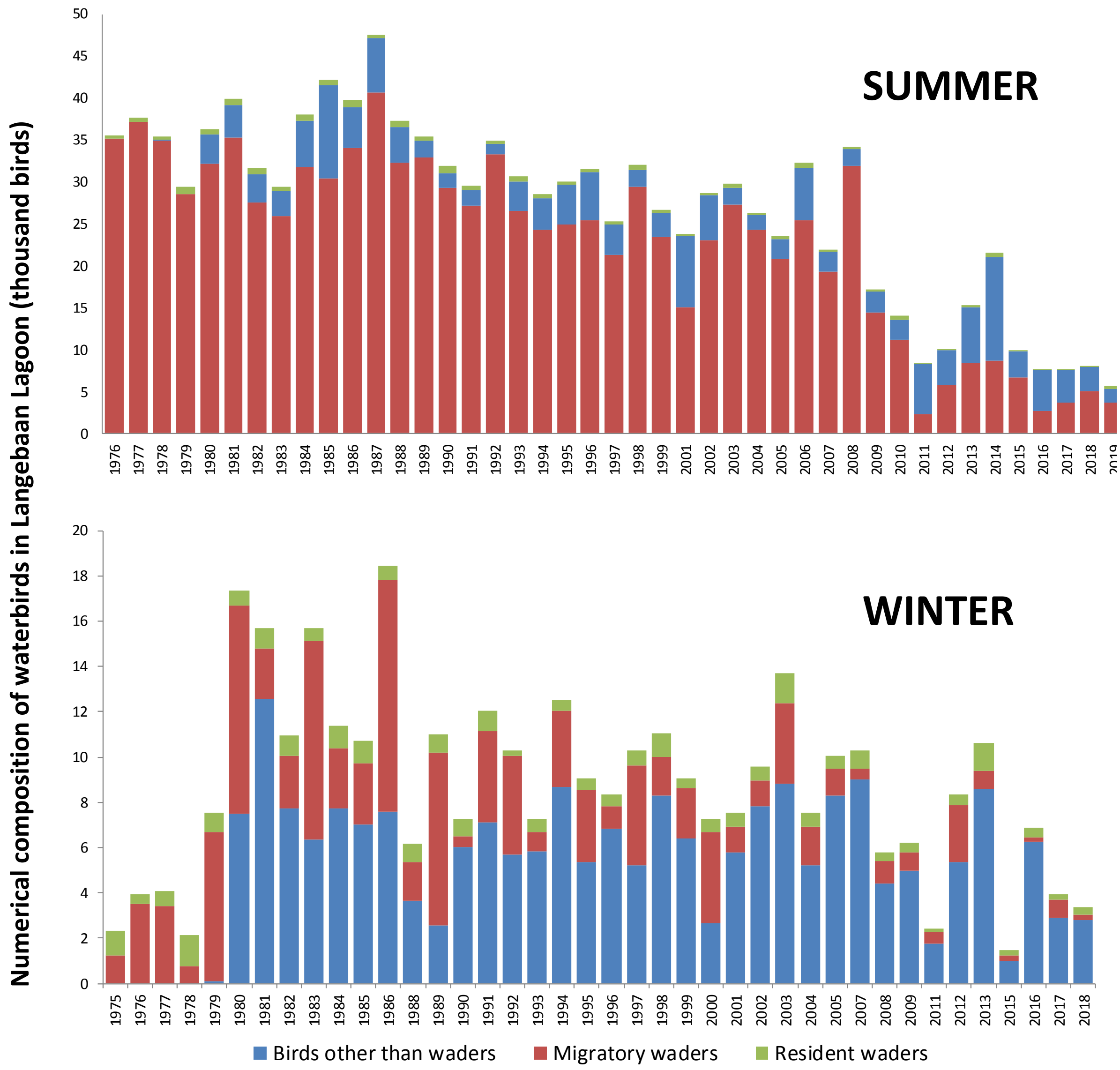
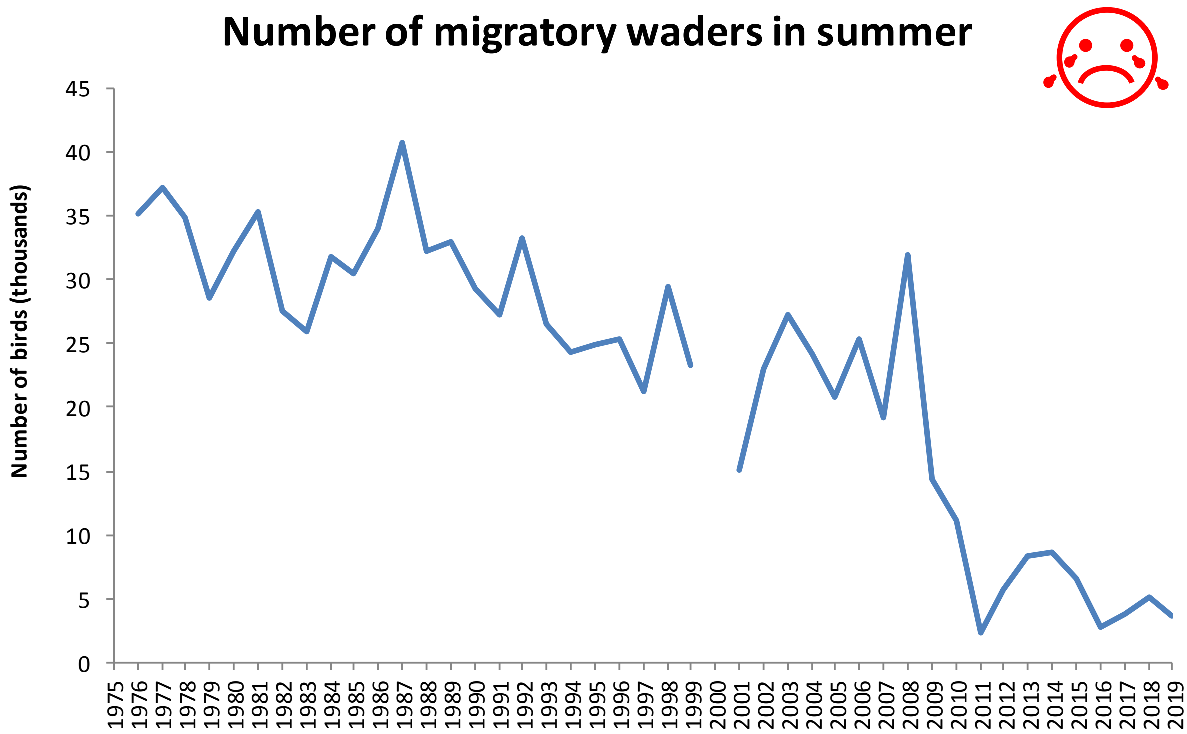


Figure 11.13. **Long-term trend in the numerical composition of waterbirds in the Langebaan Lagoon during summer (top) and winter (bottom) (1976-summer 2019). Note that no data was collected in the summer of 1975, as well as in the winter of 1987, 2006, 2010, and 2014 (Data source: Coordinated Waterbird Count data, Animal Demography Unit at the University of Cape Town 2019).**



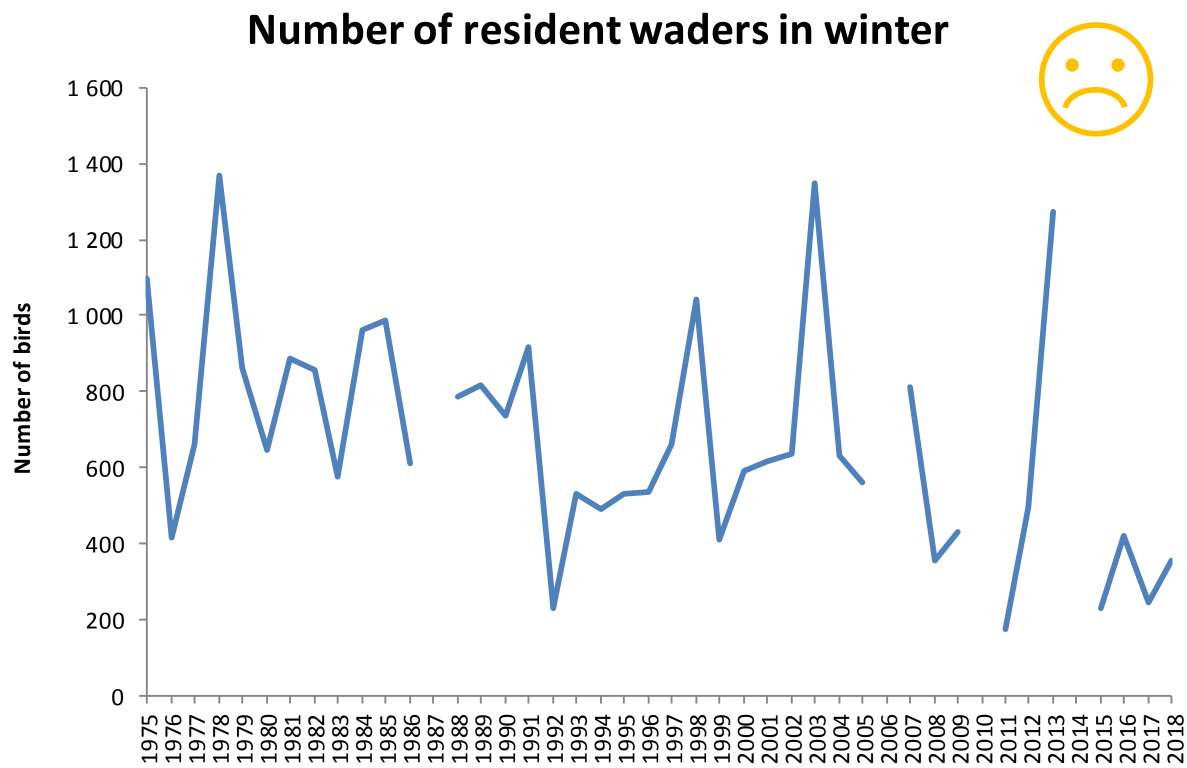


Figure 11.14. Long term trends in the numbers of summer migratory (top) and winter resident (bottom) waders on Langebaan Lagoon for the years 1976-summer 2019 (Data source: Coordinated Waterbird Count data, Animal Demography Unit at the University of Cape Town 2019).

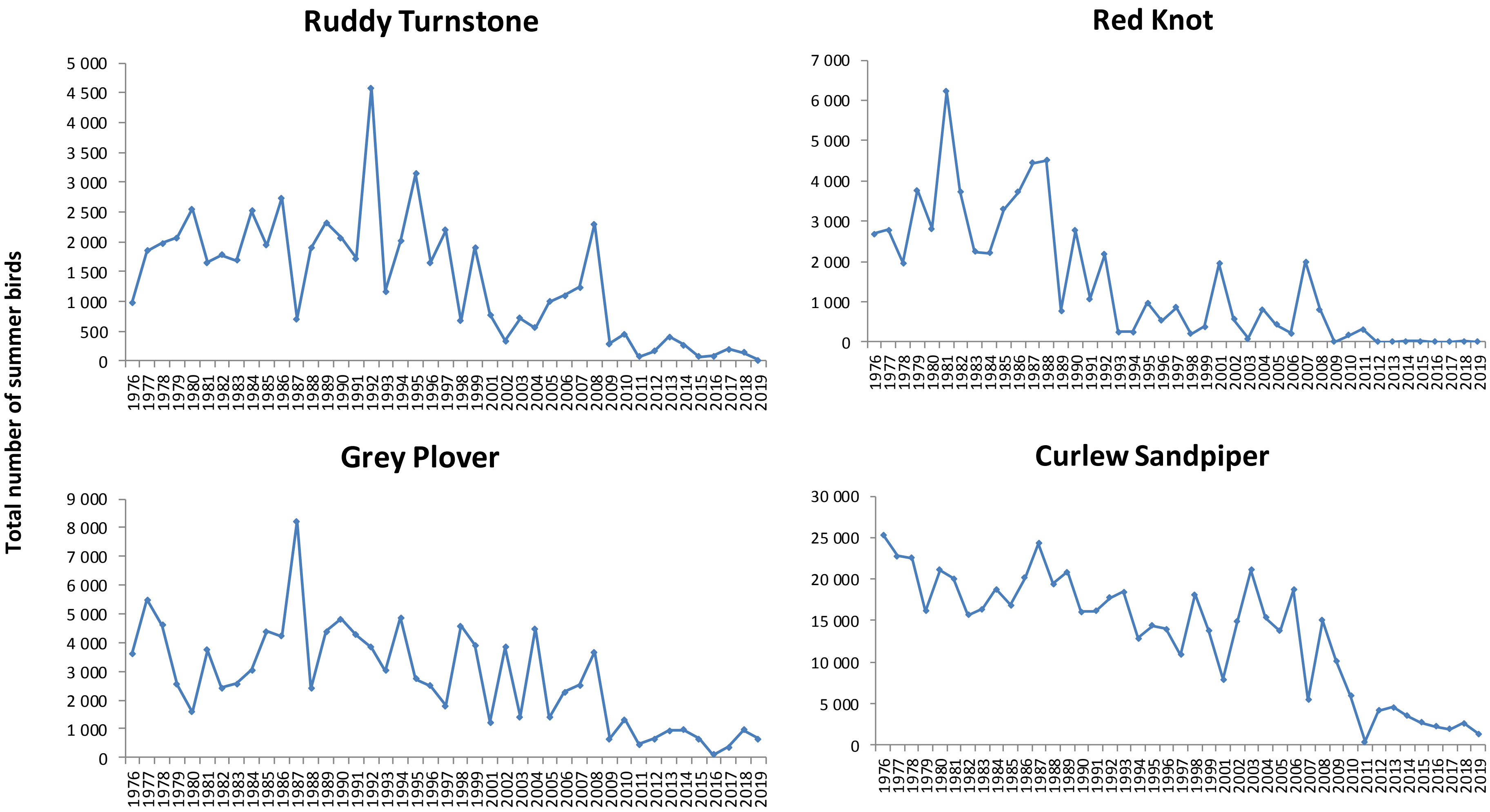


Figure 11.15 **Long-term trends in the numbers of four summer migratory waders (Ruddy Turnstone, Red Knot, Grey Plover and Curlew Sandpiper) on Langebaan Lagoon for the years 1976-2019. (Data source: Coordinated Waterbird Count data, Animal Demography Unit at the University of Cape Town 2019).**

11.4 Overall status of birds in Saldanha Bay and Langebaan Lagoon

Except for 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 arrival and spread of the Mediterranean mussel.

On the islands of Saldanha Bay, populations of all these species then started to decline, particularly, the penguins, gannets and kelp gulls, which have declined to 9%, 42% 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 Crowned 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.

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 2 635 birds in 2018. 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 may also be declining suggests that unfavourable conditions persisting in Langebaan Lagoon as a result of anthropogenic impacts may be partly to blame. Although wader numbers have not dropped below the lowest numbers as observed in 2011, it remains to be seen if winter resident wader populations remain stable, and if perhaps migratory waders are also stabilising at current levels. It is highly recommended that the status of key species continues to be monitored in future and that these data be made available and used as an indication of environmental conditions in the area.

12 ALIEN AND INVASIVE SPECIES IN SALDANHA BAY-AND LANGEBAAN LAGOON

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 and many other factors 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. Until recently, alien species were therefore 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).

By applying the above mentioned framework, marine invasion biology research published in 2016 (based on data collated up until 2014), reported 36 alien and 53 invasive marine and estuarine species occurring in South African waters (Robinson *et al.* 2016). 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) (refer to the 2017 SOB report). 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 species that occurs in South Africa (Brunetti *et al.* 2015; Robinson *et al.* 2016). When considering the West Coast of South Africa, at least 28 alien and 42 invasive species are present.

Additionally, since 2014, the presence of the barnacle *Perforatus perforates* (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. It still remains uncertain though, whether these newly discovered species should be considered alien or invasive and more research will be required to ascertain their status (Table 15.3.). In 2015 and 2017, respectively, two more species, both native to Chile, were reported from Saldanha Bay. These are 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 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. With these new additions, 28 alien species are now confirmed to be present in Saldanha Bay and/or Langebaan Lagoon, of which all but the latter two and the previously reported anemone *Sagartia ornata* are considered invasive (Table 15.3). 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. As such, this list is currently under review and being updated (*Dr Tammy Robinson pers. comm. 2019*). 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 (Sections 12.1 and 12.2). At this stage, the reason behind this decline is still not clear, although this trend has been noted for *M. galloprovincialis* in the past (Hanekom & Nel 2002; Robinson *et al.* 2007a). *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. *P. occidentalis* is now well established and has slowly been increasing in number over time in both Big Bay and Small Bay. 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.

An additional 41[12] species are currently regarded as cryptogenic (of unknown origin and potentially introduced), but very likely introduced to South Africa. Of these, 19 are likely to be found in Saldanha Bay and/or Langebaan Lagoon and six have already been identified from the Bay (Table 15.3.). Comprehensive genetic analyses are urgently required to determine the definite status of these cryptogenic species (Griffiths *et al.* 2008).

---

[12] Note: Mead *et al.* (2011a) identified 39 species as cryptogenic. Robinson *et al.* (2016) re-classified the polychaete *Hydroides elegans* as a cryptogenic (previously considered introduced). It is unknown why Mead *et al.* (2011) excluded the cryptogenic barnacle *Amphibalanus amphitrite amphitrite* in the species list despite the fact that it occurs in South African marine waters. This brings the total number of cryptogenic species to 41 to date.

Most of the introduced marine species in South Africa have been found in sheltered areas such as harbours, and are believed to have been introduced through shipping activities, for example ballast water discharge or hull fouling. 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).

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 due 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 taxa (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 in order to explore drivers behind invasion success.

Future surveys in Saldanha Bay will be used to confirm the presence of listed species and to ascertain if any additional or newly arrived introduced species are present. Current information on several key alien species in Saldanha Bay and/or Langebaan Lagoon, some of which were identified through the State of the Bay monitoring programme, are presented in the Appendix (**Table 15.3.**). Species occurrence is listed as either confirmed or likely (not confirmed from Saldanha Bay, but inferred from the regional distribution of the species). Below follows information on some of the well-known species occurring in Saldanha Bay and/or Langebaan Lagoon.

12.1 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 12.1**). 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).

Figure 12.1 Shell worm *Boccardia proboscidea* (Photo: Geoffrey Read)

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).

12.2 Acorn barnacle *Balanus glandula*

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 now the most abundant intertidal barnacle in Saldanha Bay and indeed along much of the southern west coast (Laird & Griffiths 2008). 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).

Figure 12.2 Acorn barnacle *Balanus glandula* (Photo: Prof. C.L. Griffiths)

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 12.2). *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. 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 since 2010 suggest that *B. glandula* occurs mostly on the mid shore and was most successful on the semi-exposed rocky shores sites in Saldanha Bay, with highest abundance found at the Iron ore (27%), followed by North Bay (20%), Lynch Point (14%) and Marcus Island (13%) (

Figure 12.3). There was a notable increase in the percentage cover of *B. glandula* from 0.05% in 2018 to 20% in 2019 at North Bay. Although *B. glandula* disappeared from Schaapen East and Lynch Point in 2015 and 2017 respectively, it was present in low numbers at both locations again in 2019. Following a similar trend to previous years, this species was not very abundant at the Jetty, Dive School, Schaapen Island SW or Schaapen Island SE. *B. glandula* was very abundant when it was first detected in 2010, reaching a maximum of 74% at the iron ore terminal in 2011. The abundance of this species has been fluctuating over time, most noticeably at the Iron ore terminal, Lynch Point and North Bay.

This trend may reflect a new ecosystem equilibrium as predator numbers have probably responded to the new food source and now exert some control on the abundance of the invasive species. 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. Nevertheless, it remains one of the more abundant species on the shore in Saldanha Bay and is still of significant concern.

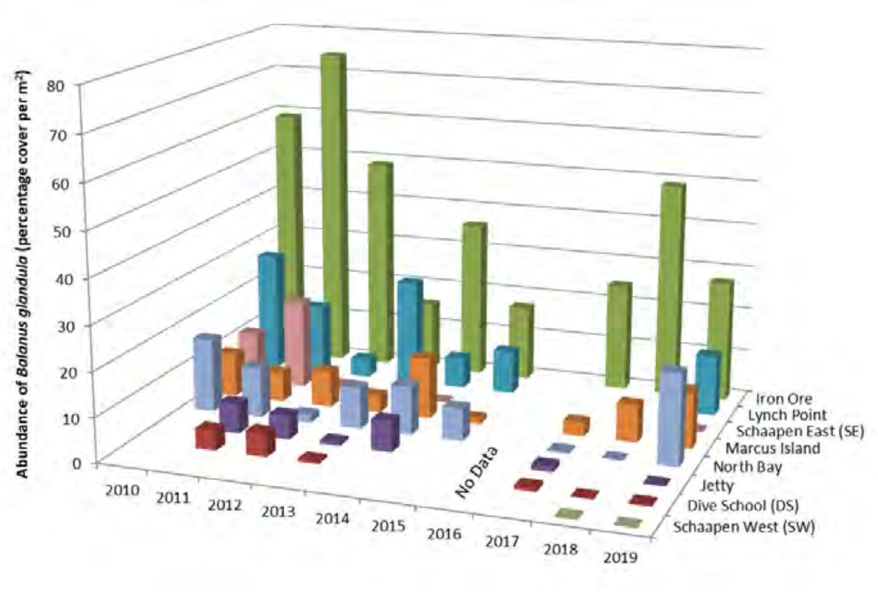


Figure 12.3. Changes in the abundance (% cover) of the acorn barnacle *Balanus glandula* at eight rocky intertidal sites on the mid shore in Saldanha Bay over the period 2010-2019. Data are shown as an average of percentage cover on the mid shore. No samples were collected 2016. Information on the locations of these sampling stations is provided in Chapter 8.

12.3 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). 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 12.4 Hitchhiker amphipod *Jassa slatteryi* (Photo: Prof. C.L. Griffiths).

12.4 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 12.5**). 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 12.5 European shore crab *Carcinus maenas.* (Photo: Prof. C.L. Griffiths).

12.5 Western pea crab *Pinnixa occidentalis*

The Western Pea crab *Pinnixa occidentalis* (**Error! Reference source not found.**) was originally described from California by MJ Rathbun in 1893, but is presently reported to occur along the whole west coast of North America from Alaska to Mexico (Ocean Biogeographic Information System 2011). The depth range distribution for this species is reported to range from 11-319 m. This species was identified in the collections from the Saldanha Bay State of the Bay surveys in 2010 (Anchor Environmental Consultants 2011), although it was previously listed as unidentified.

It appears to have established itself in the 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). The detection rate in both Big and Small Bay has been fluctuating around 40% and 20% respectively, reaching a peak in 2016, 2017 and 2019 when the species was found at 67% of the sites sampled in Big Bay (Figure 12.7). Abundance and biomass in both areas has fluctuated over time, showing no apparent upward or downward trend up to 2018 (i.e. no significant difference between the years, which is demonstrated by the overlapping standard error bars) (Figure 12.8). Sampling in 2019 suggests, however, that the abundance of *P. occidentalis* has increased by a factor of five over the last year and almost doubled in number since its peak in 2016. Although these changes are not statistically significant, it may be attributed to the large standard error bars caused by highly variable abundance at the different sites within Big Bay. In addition, crabs were only absent from three sites sampled in 2019 (BB20, BB22 and BB26), unlike most other years where it was absent from four or more sites. There was, however, a substantial increase in the biomass of *P. occidentalis* in Big Bay over the last year.

Figure 12.6 Western pea-crab Pinnixa occidentalis Photograph: Anchor Environmental Consultants).

P. occidentalis is most prevalent to the east of the iron ore and multi-purpose terminals in Big Bay towards Mykonos (site BB25, Figure 12.9). There has been an exponential increase in the abundance of crabs at this site over the past decade, with numbers exceeding 1500 individuals/m^2. It is also present in high numbers right next to the iron ore terminal (BB21 and LPG), although these numbers are lower (< 200 individuals/m^2). One of the deep water sites (BB26) has shown a decrease in numbers since 2017, with no crabs detected in this area in 2019.

In Small Bay, crabs are most abundant to the east of the terminal at the entrance of Small Bay (site SB9 Figure 12.9), where depth ranges between 13-21 m. Here, crabs exceed 600 individuals/m^2. Abundance at this site has tripled over the last year and increased exponentially over the last decade. *P. occidentalis* is present in lower numbers (<25 individuals/m^2) at sites close to the terminals (i.e. sites

SB16 and SB14, Figure 12.9). No recruitment trends of this species can be picked up from the abundance and biomass time series (Figure 12.8). *P. occidentalis* has been sporadically present in Langebaan Lagoon at three sites (LL31, LL33 and LL40) over the past decade, although abundance at these sites was very low with only four individuals recorded per square metre (Figure 12.7). This crab was again found at site LL31 (close to the mouth of the lagoon) this year, but in markedly higher numbers of 20 individuals/m^2. Although, most of the lagoon habitat may not be entirely suited for this species, which favours deeper water (>10 m) in its native range (Ocean Biogeographic Information System 2011), this site is slightly deeper than the rest of the lagoon (3.5-6 m) and is adjacent to the Big Bay sites known to support populations of this crab. Currently, this species does not appear to be spreading or increasing in density in the rest of Langebaan Lagoon.

Danger Bay was only sampled in 2014 and 2015. It is noticeable that the species was absent in the first survey, but was found in 2015 at one out of 13 sites sampled, at a density of eight animals per square metre.

In conclusion, these data suggest that *P. occidentalis* is now well established and slowly increasing in number over time in both Big Bay and Small Bay. It may also be in the process of expanding into more exposed and deeper habitats outside of the Bay, including Danger Bay. This increase in abundance in the Bay and its presence again this year in Langebaan Lagoon, raises concern and highlights the need for management action.

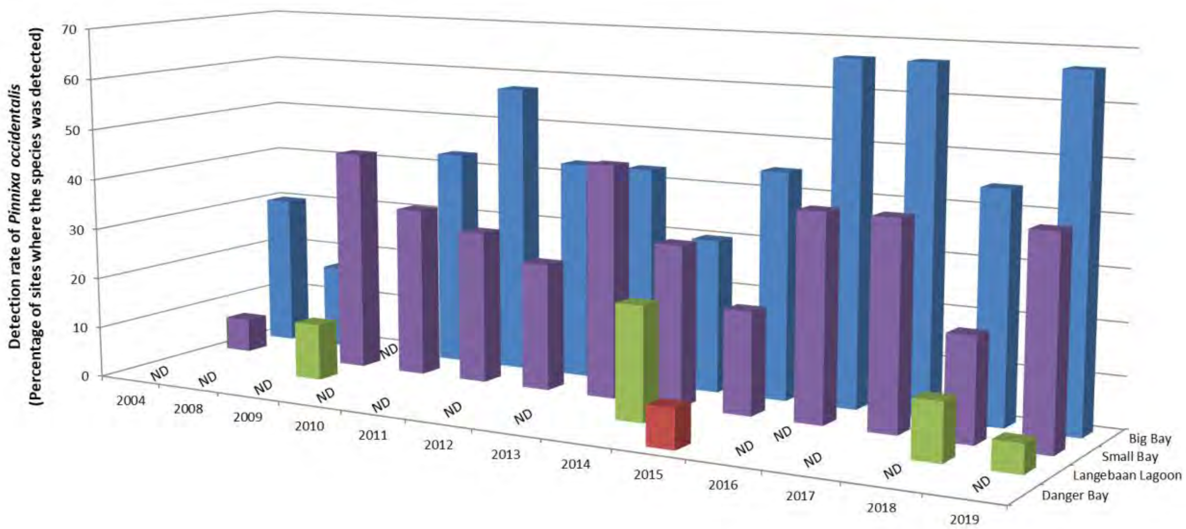


Figure 12.7 **The detection rate (percentage of sites where the species was detected) of the Western Pea crab *Pinnixa occidentalis* in Big Bay, Small Bay, Langebaan Lagoon and Danger Bay in the period 2004-2019. Note that Langebaan Lagoon and Danger Bay were first sampled in 2004 and 2014, respectively. No data were collected in the period 2005-2007. 'ND' denotes that no data was collected in the region for that year.**

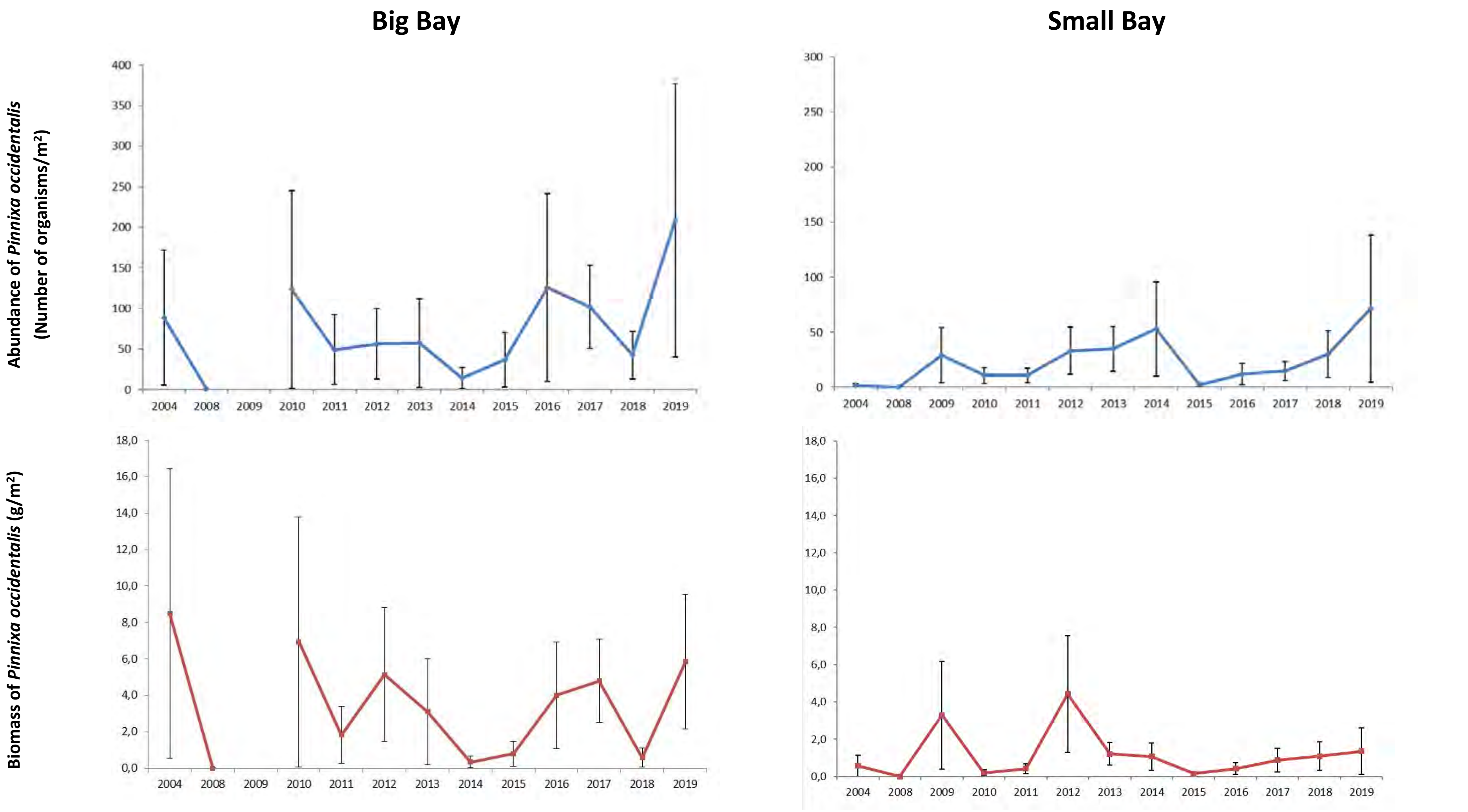
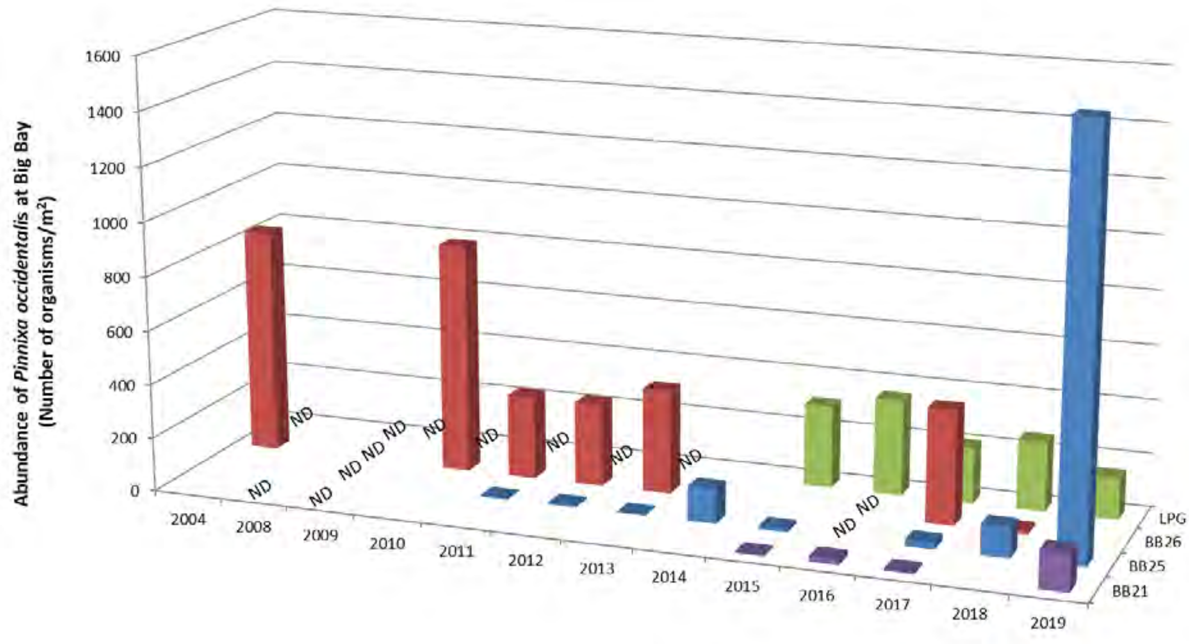


Figure 12.8 **Average abundance (top) and biomass (bottom) of the Western Pea crab *Pinnixa occidentalis* in Saldanha Bay, Big Bay (left) and Small Bay (right) from 2004-2019. No data were collected from 2005-2007. 'ND' denotes that no was data collected in the region for that year.**



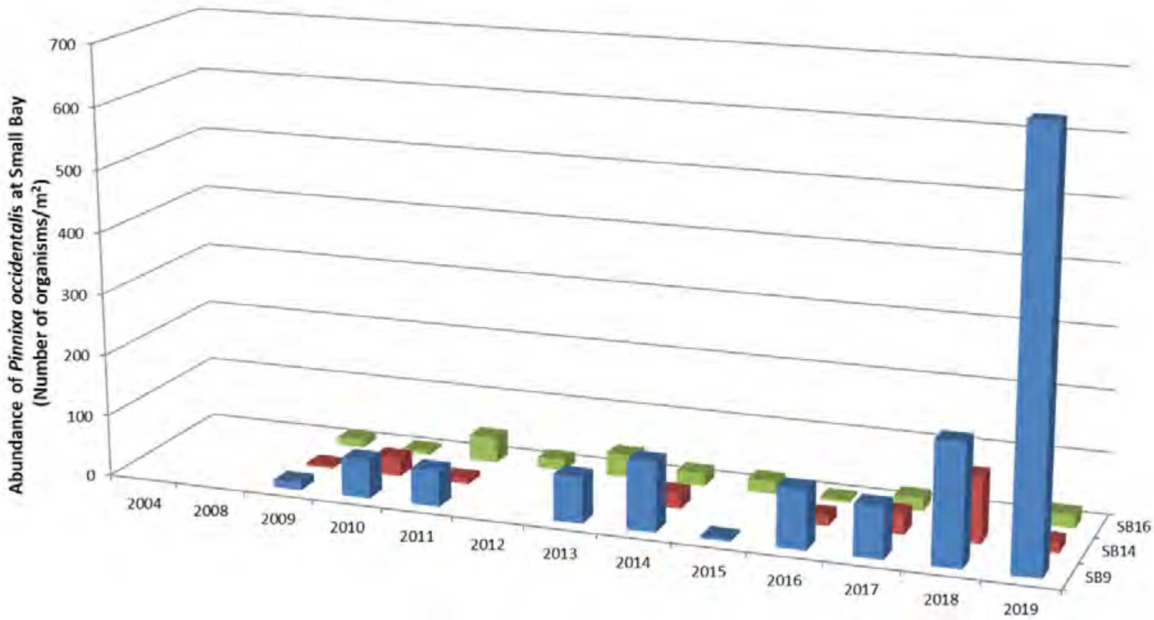


Figure 12.9 **Abundance of the Western Pea crab *Pinnixa occidentalis* in Saldanha Bay at selected sites in Big Bay (top) and Small Bay (bottom) from 2004-2019. No data were collected from 2005-2007. 'ND' denotes that no data was collected in the region for that year.**

12.6 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). 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 12.10 Lagoon snail *Littorina saxatilis* (Photo: Prof. C.L. Griffiths)

12.7 Pacific oyster *Crassostrea gigas*

Crassostrea gigas is considered native to Japan and South East Asia. *C. gigas* 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).

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. Through the use of 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) (Figure 3). 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 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 12.10).

12.8 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). 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 (**Figure 12.11**). 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 12.11 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 it is typically found on exposed rocky shores in southern Africa. *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 not clear, 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.

Data from the State of the Bay surveys suggest that *M. galloprovincialis* occurs mainly on exposed rocky shores in Saldanha Bay (i.e. Lynch Point, Marcus Island, iron ore terminal, North Bay) and is present in low numbers at the more sheltered sites (Dive School, Jetty and Schaapen Island East and West). Since the start of the surveys, up until 2015, *M. galloprovincialis* increased steeply in abundance at the exposed sites, reaching maximum abundance at Marcus Island in 2009 (37%), at Lynch Point (58%) and North Bay (23%) in 2012, and at the iron ore terminal in 2015 (40%). At exposed sites, *M. galloprovincialis* is by far the most dominant faunal species on the rocky shore, 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.

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 (Sadchatheeswaran *et al.* 2015). 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).

In more recent years, *M. galloprovincialis* abundance in Saldanha Bay has decreased, dropping to levels lower than those observed for most years at Lynch Point, Marcus Island, iron ore terminal and North Bay. As predicted in 2018, abundance of this mussel decreased again in 2019 at the iron ore site. Until recently, it was hypothesised that this trend may reflect a new ecosystem equilibrium as predator numbers have probably responded to the new food source and now exert more control on the abundance of this invasive species. However, a recent laboratory study found that when two generalist native predators, the west coast rock lobster *Jasus lalandii* and the starfish *Marthasterias africana,* were presented with a choice between the native mussels, *A. atra* and *C. meridionalis,* and the alien mussels *M. galloprovincialis and S. algosus,* they selected towards *C. meridionalis*. These findings were unexpected and suggest that native predators do not necessarily control the abundance of alien mussel species, but instead, might indirectly be facilitating their invasion by removing inter-specific competition (Skein *et al.* 2018b). This phenomenon does, however, require more investigation. The reason for this decrease is therefore still not clear, although such a sudden, unexplained decrease in abundance has been noted for *Mytilus* in the past and other factors in Saldanha Bay could potentially play a role.

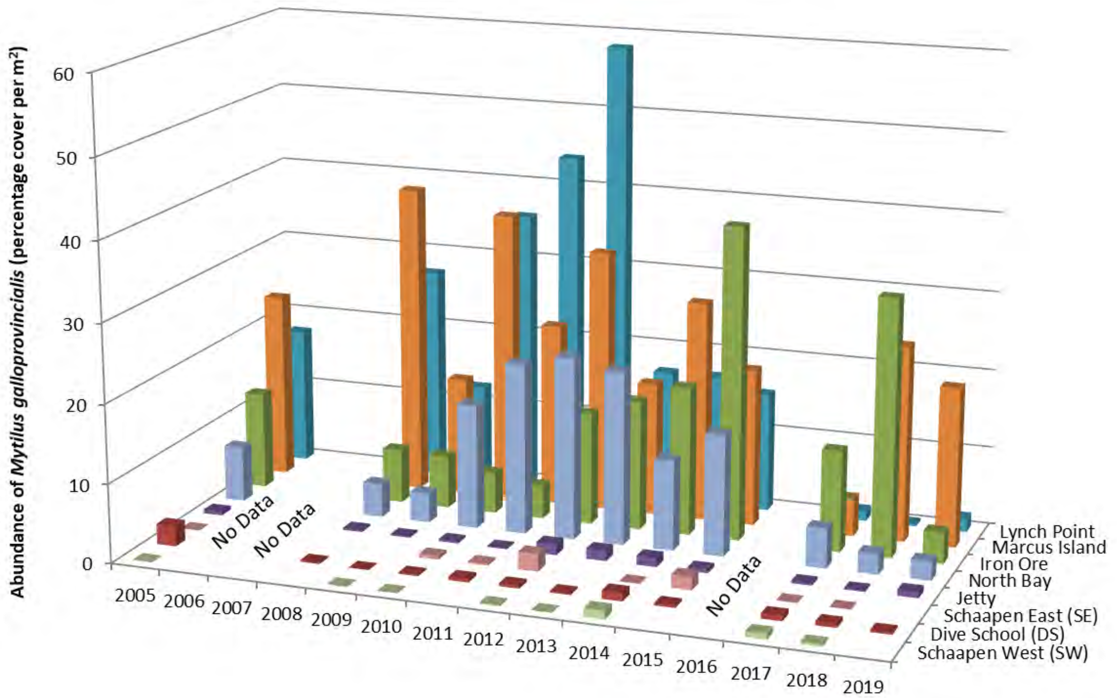


Figure 12.12. **Changes in the abundance (% cover) of the Mediterranean mussel *Mytilus galloprovincialis* at eight rocky intertidal sites in Saldanha Bay over the period 2005-2019. Data are shown as an average of percentage cover on the mid and low shore. No samples were collected in 2006, 2007 and 2016. Information of the locations of these sampling stations is provided in Chapter 8.**

12.9 Pacific South American mussel *Semimytilus algosus*

Figure 12.13 **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. 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). However, as mentioned before, the study by Skein *et al.* (2018b) found that native predators preferred the native mussels to the alien mussels which might in fact indirectly facilitate the invasion of alien mussels by removing inter-specific competition with the natives. 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.

12.10 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 12.14**). 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 12.14 Disc lamp shell *Discinisca tenuis* (Photo: Prof. C.L. Griffiths)

12.11 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 12.15**). It was originally introduced from 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 12.15 A typical aggregation of *Ciona robusta* (Photo: National Museums Northern Ireland).

12.12 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). It is believed to have been accidentally introduced from Europe prior to the 1949, probably as a fouling organism.

Figure 12.16 Jelly crust tunicate *Diplosoma listerianum* (Photo: Prof. C.L. Griffiths).

12.13 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). *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 current 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 maritime* (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 12.17 Brooding anemone *Sagartia ornata* (Photo: Prof. C.L. Griffiths)

Invaded sandy-shore areas support a higher invertebrate abundance, biomass and diversity, as well as altered community structures and 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).

12.14 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 intertidal rocky shore 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 an extended sheath and longitudinal abutment present on the inner surface of the radii and a bifid sutural edge present on the algae. 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 12.18**).

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 based on external appearance, 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).

Figure 12.18 ***Perforatus perforatus*** **(Pilsbry, 1916) (Photograph: Dr Nina Steffani)**

12.15 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*, 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 12.19**). *A. amphitrite amphitrite* is easily confused with another 'purple-pink striped' species which has not yet been identified (Biccard 2012).

Figure 12.19 ***Amphibalanus amphitrite amphitrite*** **(Photo: Prof. C.L. Griffiths)**

12.16 North West African porcelain crab *Porcellana africana*

Figure 12.20 **European porcelain crab *Porcellana africana* (Photo: Prof. C.L. Griffiths).**

The porcelain crab, *Porcellana africana,* 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 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 has 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.

12.17 South American sunstar *Heliaster helianthus*

Heliaster helianthus (Lamarck, 1816) is commonly known as the South American multiradiate sunstar. 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.

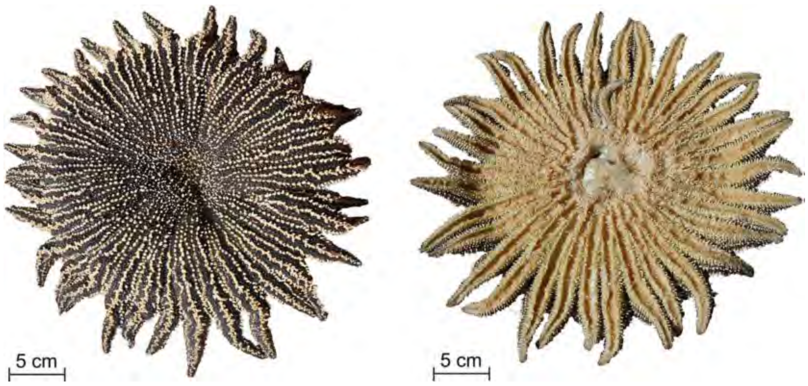


Figure 12.22 ***Heliaster helianthus*** **(Lamarck, 1816) (Photo: Dr Tammy Robinson)**

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.

12.18 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, a trait that might protect them from predation (Fernández and Castilla 2000).

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). 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.

Figure 12.21 ***Homalaspis plana*** **(H. Milne Edwards, 1834) (Photo: Dr Koebraa Peters)**

12.19 Hydrozoan *Coryne eximia*

This hydrozoan 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 12.21 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).

12.20 Tubeworm *Neodexiospira brasiliensis*

Neodexiospira brasiliensis is a small subtidal tubeworm 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 12.22 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)

12.21 Shell-boring spionid *Polydora hoplura*

Polydora hoplura is a shell-boring spionid polychaete native to Europe where it occurs from the Mediterranean to England. 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 12.23 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).

12.22 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. 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 12.24 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-bring nature, it is considered a pest and has the potential to negatively impact the economy by destroying wooden structures.

12.23 Tube-dwelling amphipod *Cerapus tubularis*

Cerapus abditus 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).

12.24 Sand-hopper *Orchestia gammarella*

Orchestia gammarella, or the sand-hopper, is a semi-terrestrial amphipod. Its native range includes Norway to the Mediterranean, as well as Madeira, Canary Islands and the Azores (Henzler and 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).

Figure 12.25 The sand-hopper *Orchestia gammarella* (Photo: Auguste Le Roux [CC BY-SA 3.0 (https://creative commons.org/licenses/by-sa/3.0)].

12.25 Bryozoan *Conopeum seurati*

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. 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 (False Bay), although proper identification is required (Mead *et al.* 2011).

Figure 12.26 The colonial bryozoan *Conopeum seurati* (Photo: De Blauwe 2009).

12.26 Bryozoan *Cryptosula pallasiana*

The bryozoan *Cryptosula pallasiana* occurs in brackish waters, where it forms pink, white or orange encrusting colonies (Occhipinti Ambrogi & d'Hondt 1981) 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).

Figure 12.27 The colonial bryozoan *Cryptosula pallasiana* (Photo: Cohen 2011).

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).

12.27 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). *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).

Figure 12.28 Red-rust bryozoan *Watersipora subtorquata* (Photo: Luis A. Solórzano in Cohen 2011).

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).

12.28 Light bulb tunicate *Clavelina lepadiformis*

Figure 12.29 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. 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.

12.29 Algae *Antithamnionella spirographidis*

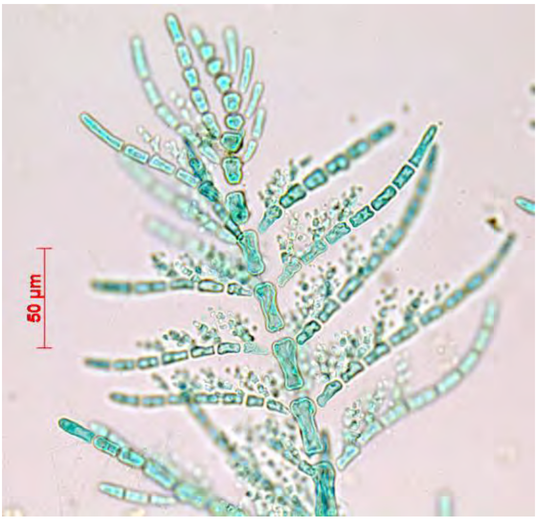


Figure 12.30 Slide of the algae *Antithamnionella spirographidis* (Photo: Anderson *et al.* 2016).

This algal species is 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.

12.30 Dead Man's Fingers *Codium fragile*

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'. 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 12.31 Dead Man's Fingers *Codium fragile* (Photo: 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.

13 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 escalations 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.

13.1 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. 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 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 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.

13.1.1 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.

The amount of hardened (as opposed to naturally vegetated) surfaces surround 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 of sensitive habitats and species, especially seagrass, water birds and fish in Langebaan Lagoon.

13.1.2 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 major 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.

13.1.3 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 issued a CWDP on 26 June 2017 (as amended subsequently to accommodate a change in discharge location and effluent composition). This CWDP authorises Sea Harvest to dispose a maximum quantity of 420 480 m^3 per annum at a maximum daily discharge volume of 1 152 m^3. Sea Harvest is committed to meeting effluent quality thresholds and environmental monitoring requirements as stipulated in the permit. 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, which are on average two and three times higher than the prescribed limit. The effluent produced by the RO plant has increased the salinity of the overall effluent dramatically and CWDP requirements are currently exceeded 52% of the time. During the 2018/19 monitoring period, 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.

13.1.4 Mariculture

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. A combined 430 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 is currently in the process of establishing 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 for a total ADZ comprising 884 ha (currently farmed areas will be incorporated into the ADZ). 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.

13.1.5 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.

13.1.6 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.

13.2 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. 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. Previously it was believed that there is little exchange of water between these two aquifer units, but this is now in question and requires confirmation. 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). Increasing pressure on available freshwater water resources in the Saldanha Bay area has resulted in attention being turned to exploitation of these groundwater resources. The West Coast District Municipality (WCDM) operates a wellfield on the Langebaan Road Aquifer that is licenced to abstract up to 1.46 million m^3 of groundwater per annum. Abstraction of groundwater from this aquifer resulted in a localised depression of water levels in the deeper portion of this aquifer by as much as 10 m in the first few years of operation between 2005 and 2009, and concern has been expressed over how this might be affecting groundwater discharge to Saldanha Bay now, and in the future. A modest (10%) reduction in abstraction rates was affected to address this but it is not clear how effective this has been.

More recently, Elandsfontein Exploration and Mining (Pty) Ltd/Kropz has 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 further away, in an effort to ensure that surrounding ecosystems (including the Lagoon) are not affected. There is concern that these mitigation measures will not effectively alleviate impacts on the lagoon, so a comprehensive monitoring programme has been initiated to confirm that potential impacts on the Lagoon hydrology and ecology are effectively mitigated by reinjection of ground water. This includes monitoring of water levels and water quality in a series of boreholes between the mine site and the lagoon edge and monitoring of salinity levels and macrofauna assemblages in the lagoon itself. It is recommended that additional monitoring points are established at the head of the lagoon as well as analysis of extent of phragmites coverage by up to date mapping and aerial photography.

Kropz Elandsfontein has not officially begun phosphate mining as their Water Use Licensing process has been appealed by the WCEPA and will be addressed in a court hearing with the Water Tribunal in early September 2019. For full details please refer to the Activities and Discharges Chapter 3, Section 3.2.7.

The artificial recharge tests that were undertaken in 2008 and 2009, as well as the evaluation of the monitoring data for the area has shown that there are still gaps in the understanding of the groundwater system in the area. A WRC project that began in 2017 is working towards gaining a better understanding of the West Coast Aquifer System, so that it would be possible to manage the system sustainably. This is of great importance as the recent drought demonstrated how necessary it is to diversify water supply options, which lead to the extension of the current wellfield and the development of a second in a portion of the Langebaan Road Aquifer System along the R45 road. Additionally, any future impacts caused by climate change will also add to the pressure and sensitivity of this system and the decisions surrounding its use. Further wellfield developments cannot be ruled out and have been advised by the Department of Water and Sanitation. The Saldanha Bay Local Municipality will also have to take over more of the monitoring in the area in order to manage the Lower Berg Aquifer System sustainably, especially since the Department of Water and Sanitation are not able to do monitoring regularly because of financial constraints and capacity issues.

13.3 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 five 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 are proposing to initiate monitoring of dissolved oxygen and temperature at two sentinel and one reference station close to the bottom (0.5 m above the seabed) in the Bay. DAFF are also proposing 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. DAFF 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 at least three sites that are paired as close as possible in time. Addition of these data to the State of the Bay Monitoring Programme 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 DAFF (courtesy of the farm operators).

Concentrations of trace metals in marine filter feeders in Saldanha Bay indicate that concentrations of trace metals are higher along the shore and are often above published food safety limits in the case of lead and cadmium. Concentrations reported for mariculture operations that are located offshore are slightly 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.

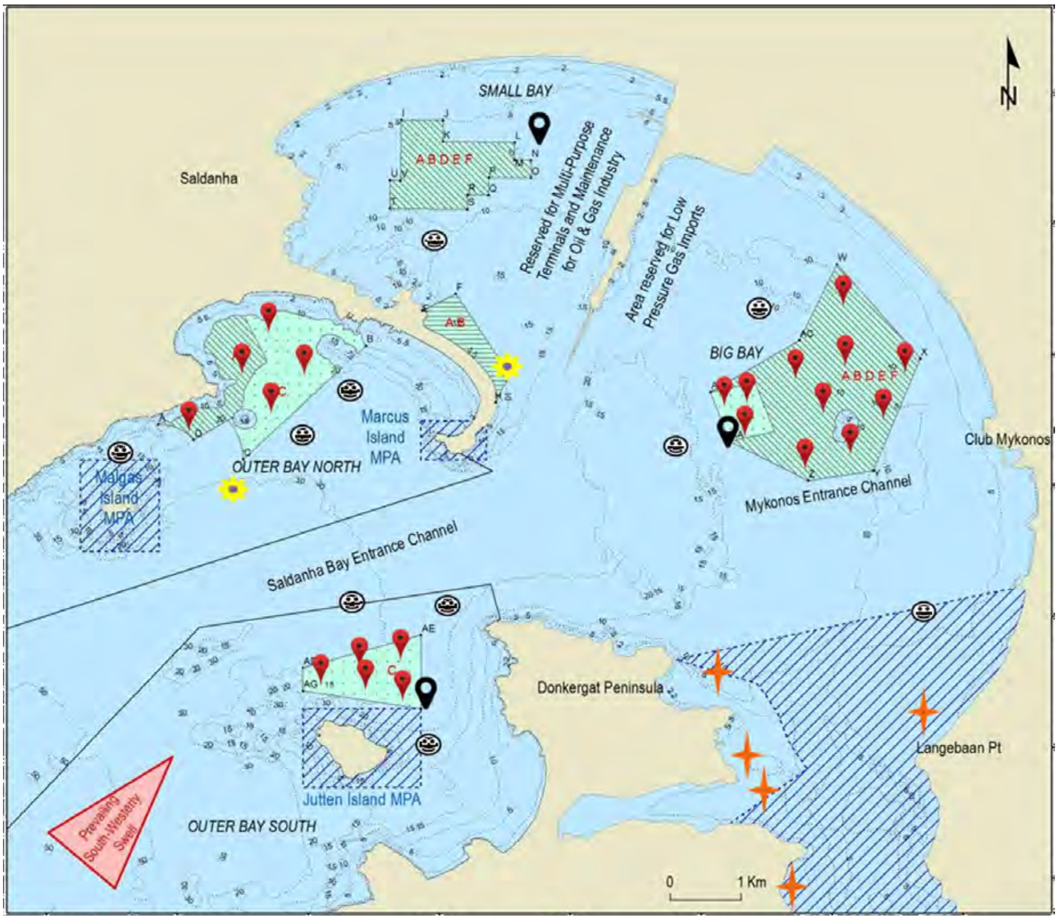


Figure 13.1. Monitoring stations for the DEFF ADZ. Locations of the sentinel water quality monitoring stations are indicated with the following symbols: .

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 three of the four sites in Big Bay were within the 80th percentile limits for mariculture in 2017 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.

13.4 Sediments

Sediment monitoring in the Bay has revealed that key heavy metal contaminants (Cd, Pb and Cu) are high at numerous sites in Small Bay, to the extent that they are almost certainly impacting on benthic fauna and possibly other faunal groups in the Bay. The recent 2019 survey revealed an overall increase in trace metal concentrations at majority of the sites sampled in comparison to 2018. The latter leaves Cd and Cu still above ERL guidelines in Small Bay (Yacht Club Basin). Enrichment factors for Cd, Pb and Cu also remain extremely high at many sites. These contaminants are typically associated with the finer sediment fraction and are highest in areas adjacent to the Iron Ore Terminal, near the Mussel Farm and the Yacht Club. It is important to note that Cd concentrations at Big Bay in 2019 are close to the ERL limit and may well be above the guideline in the future surveys.

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. Poly-aromatic hydrocarbons (PAH) were considered to pose no threat since the first survey was conducted in 1999. Recent assessment undertaken in 2019 suggested that this is still the case, however, considerable fluctuations in TPH levels have been recorded in recent years. High concentrations of TPH have been recorded at sites adjacent to the Iron Ore Terminal in the past (2014), and it is likely that this was associated with a pollution incident of some sort. TPH levels have remained the same in 2019 and present no major concern. Although, it is recommended that TPH and PAH monitoring should continue annually as a precautionary measure.

13.5 Benthic macrofauna

Monitoring of benthic macrofaunal communities over the period 1999-2019 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. 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.

13.6 Rocky intertidal

A total of 118 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, 25 species of grazers, and 20 species of predators and scavengers combined. The algal component comprised 33 corticated (foliose) seaweeds, ten ephemerals, five species of encrusting algae, and two 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 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. At this stage it is unclear if this is due to a decrease in larval supply but it does suggest that no recent introductions of contaminated ballast water have occurred in the Bay. 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.

13.7 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 has not been enough to sustain the fishery at the high (and increasing) 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 sticks, 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 and growing 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 enhanced protection of this portion of the Bay.

The concerning trend in decreasing white stumpnose 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 fresh water 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, 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 2019 study for as long as possible. This sampling should be confined to the same seasonal period each year for comparative purposes.

13.8 Birds

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 Crowned 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 key species continues to be monitored in future and that these data be made available and used as an indication of environmental conditions in the area.

13.9 Alien invasive species

A recent update on the number of alien marine species present in South Africa (up until 2014), lists 89 alien species as being present in this country, of which 53 are considered invasive i.e. populations are expanding and consequently displacing indigenous species (Robinson *et al.* 2016). At least 28 alien and 42 invasive species occur along the West Coast of South Africa. With the recent addition of five new species after 2014 – the barnacle *Perforatus perforates* (*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) – 28 species are thus confirmed from Saldanha Bay and/or Langebaan Lagoon.

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, although the reason behind this decline is still not clear. *B. glandula* has shown an increase in abundance over time and remains one of the more abundant species on the mid-shore in Saldanha Bay. *S. algosus* was absent in the intertidal zone in Saldanha Bay, but has previously been observed on mussel rafts in the Bay. It is therefore recommended that sub-tidal surveys are conducted to ascertain whether populations have indeed established in the Bay, especially since this species has the potential to spread (Skein *et al.* 2018a, b).

The Western pea crab *P. occidentalis* is now well established and has slowly been increasing in number over time in both Big Bay and Small Bay. It was also detected 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 raises concern and highlights the need for management action. An additional 41 species are currently regarded as cryptogenic, but very likely introduced to South Africa. Of these, 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 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 of these strategies present their own advantages and limitations.

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 as a way 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 89 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).

Nevertheless, 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. In light of this, studies measuring the impacts of these species in Saldanha Bay are desperately needed in order to prioritise management actions. Changes in the population structure of aliens as well as that of the surrounding native biota, should be carefully and regularly monitored as this can give insight into the impacts that these alien species have on the natives.

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.

In addition, managing pathways and associated vectors offers another approach in preventing invasions. This 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 from the same regions, highlights the risk of introduction from specific areas linked to Saldanha Bay.

In addition to routinely monitoring changes in the population structure of these aliens throughout Saldanha Bay, in depth studies investigating pathways and biological traits associated with their invasion success and their impact upon the community structure of the surrounding native biota, are required. These will not only contribute towards our understanding of the drivers and traits governing their successful invasion, but also give insight into their associated impacts. In turn, this will support directed management actions in order to successfully control invasions and mitigate impacts.

13.10 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 13.1).

Table 13.1 Tabulated summary of Environmental parameters reported on in the State of the Bay: Saldanha Bay, Danger 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 – 2018 And 2016-2019	Aquifer over extraction can have a detrimental impact as indicated in the past. Especially during times of drought, but more recently, is being monitored and managed closely in order to prevent over extraction. Baseline monitoring of the Lagoon is ongoing at present.	
WATER QUALITY			
Physical aspects (temperature, salinity, dissolved oxygen, nutrients and chlorophyll)	1974-2000, 2010-2011, 2014-2019	Dissolved oxygen levels in bottom water in Small Bay are very much lower than they were historically or at least prior to port development. This is attributed to organic loading in the 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. 2019	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)	1999-2019	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.	
Trace metal contaminants in water	1997-2008, 2014-2019	Concentrations of lead in mussel flesh collected from the shore are consistently above the safety guidelines for food stuffs (this is not the case with farmed mussels). Any future dredging events should be limited as far as possible owing the likely mobilization of trace metals from sediments.	
SEDIMENTS			
Particle size (mud/sand/gravel)	1974-2019	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.	

Parameter monitored	Time period	Anthropogenic induced impact	Rating
Total organic carbon (TOC)	1974-2019	Elevated levels of TOC at the Yacht Club basin and near the mariculture rafts (negative impacts) are of particular concern.	☹
Total organic nitrogen (TON)	1974-2019	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	1980-2019	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.	☹
BENTHIC MACROFAUNA			
Species abundance, biomass, and diversity	1999-2019	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.	😐
ROCKY INTERTIDAL AND INTRODUCED SPECIES			
Impact of alien mussel and barnacle introductions	1980-2019	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	1986-2019	White stumpnose abundance and fishery landings have declined dramatically over the last decade. Abundance and diversity of fish in the Bay and Lagoon (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	1977-2018	Populations of 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 Crowned Cormorants feed	☹
Population numbers of key species in Langebaan Lagoon	1976-2019	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	☹

Parameter monitored	Time period	Anthropogenic induced impact	Rating
		have also declined and is likely due to changes in the lagoon itself.	
ALIEN AND INVASIVES			
Total number of alien and invasive species in Saldanha Bay and Langebaan Lagoon	Current 2019	Twenty-eight species have been confirmed from Saldanha Bay and/or Langebaan Lagoon, of which all but one are considered invasive.	
Acorn barnacle *Balanus glandula*	2010-2019	Increased in abundance at most of the sites with a notably large increase at one site. Decreasing trend in abundance at some sites. 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*	2005-2019	Decreasing trend in abundance at some sites. 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.	
Western pea crab *Pinnixia occidentalis*	2004-2019	Abundance and biomass in Big Bay and Small Bay increasing over time. Also present in higher numbers in Langebaan Lagoon. This raises concern and the need for management actions	

14 REFERENCES

Aalbers SA. 2008. Seasonal, diel, and lunar spawning periodicities and associated sound production of white seabass (*Atractoscion nobilis*). *Fishery Bulletin* 106: 143–151.

Abbott R & E Bing-Sawyer. 2002. Assessment of pile driving impacts on the Sacramento blackfish (*Orthodon microlepidotus*). Draft report prepared for Caltrans District 4. San Francisco, CA, Caltrans.

Abbott R, Reyff J & G Marty. 2005. Final Report: Monitoring the Effects of Conventional Pile Driving on Three Species of Fish. Richmond, CA, Manson Construction Company.

Acuña FH, Excoffon & CL Griffiths. 2004. First record and re-description of the introduced sea anemone *Sagartia ornate* (Holdsworth 1855) (Cnidaria: Actinaria: Sargartiidae) from South Africa. *African Zoology* 39: 314–318.

Adams J & N Ngesi. 2002. An assessment of the salt marsh of the Great Brak Estuary. University of Port Elizabeth, Deparment of Botany. Report prepared for Petro SA (Pty) Ltd. November 2002.

Adams J & N Ngesi. 2002. An assessment of the salt marsh of the Great Brak Estuary.

Adams JB. 2016. Distribution and status of *Zostera capensis* in South African estuaries — A review. *South African Journal of Botany* 107: 63–73.

Adams JB & GC Bate. 1994. The effect of salinity and inundation on the estuarine macrophyte *Sarcocornia perennis* (Mill.) A.J. Scott. *Aquatic Botany* 47: 341–348.

Adams JB, Bate GC & M O'Callaghan. 1999. Primary producers. In: Allanson BR & Baird D (ed) Estuaries of South Africa. Cambridge: University Press, 91–117 pp.

Adams NJ & RA Navarro. 2005. Foraging of a coastal seabird: Flight patterns and movements of breeding Cape gannets *Morus capensis*. *African Journal of Marine Science* 27(1): 239–248.

Alexander EM, Adams R, Dick JTA & TB Robinson. 2015. Forecasting invasions: resource use by mussels informs invasion patterns along the South African coast. Invasive Species, 162(12): 2493–2500.

Anchor Environmental Consultants (AEC). 2006. State of the Bay 2006: Saldanha Bay and Langebaan Lagoon. Technical Report. Prepared for Saldanha Bay Water Quality Forum Trust. Cape Town, 93 pp. + Appendices.

Anchor Environmental Consultants (AEC). 2009. State of the Bay 2008: Saldanha Bay and Langebaan Lagoon. Technical Report. Prepared for Saldanha Bay Water Quality Forum Trust. Cape Town, 174 pp.

Anchor Environmental Consultants (AEC). 2010. State of the Bay 2009: Saldanha Bay and Langebaan Lagoon. Technical Report. Prepared for Saldanha Bay Water Quality Forum Trust. Cape Town, 213 pp.

Anchor Environmental Consultants (AEC). 2011. State of the Bay 2010: Saldanha Bay and Langebaan Lagoon. Technical Report. Prepared for Saldanha Bay Water Quality Forum Trust. Cape Town, 280 pp.

Anchor Environmental Consultants (AEC). 2012a. Re-commissioning of a Fishmeal Processing Plant, Saldanha Bay Marine Specialist Report. Prepared for SRK Consulting (South Africa) (Pty) Ltd. Cape Town, 63 pp.

Anchor Environmental Consultants (AEC). 2012b. State of the Bay 2011: Saldanha Bay and Langebaan Lagoon. Technical Report. Prepared for Saldanha Bay Water Quality Forum Trust. Cape Town, 271 pp.

Anchor Environmental Consultants (AEC). 2012c. The impact of Fe $_2O_3$ on the marine environment in Saldanha Bay. Report prepared for Transnet Port Terminal – Saldanha, October 2012.

Anchor Environmental Consultants (AEC). 2013a. Proposed Replacement of Existing Breakwater at the Salamander Bay Special Forces Regiment 4 Boat Park. 2 pp.

Anchor Environmental Consultants (AEC). 2013b. State of the Bay 2012: Saldanha Bay and Langebaan Lagoon. Technical Report. Prepared for Saldanha Bay Water Quality Forum Trust. Cape Town, 314 pp.

Anchor Environmental Consultants (AEC). 2014. . The State of Saldanha Bay and Langebaan Lagoon 2013/2014. Technical Report September 201. Prepared for the Saldanha Bay Water Quality Forum Trust. Cape Town.

Anchor Environmental Consultants (AEC). 2015a. Assessment Framework for the Management of Effluent from Land Based Sources Discharged to the Marine Environment. Anchor Environmental Consultants (Pty) Ltd, Report to the Department of Environmental Affairs.

Anchor Environmental Consultants (AEC). 2015b. The State of Saldanha Bay and Langebaan Lagoon 2014/2015. Technical Report September 2015. Prepared for the Saldanha Bay Water Quality Forum Trust. Cape Town.

Anchor Environmental Consultants (AEC). 2016. The State of Saldanha Bay and Langebaan Lagoon 2015/2016. Technical Report September 2016. Prepared for the Saldanha Bay Water Quality Forum Trust. Cape Town.

Anchor Environmental Consultants (AEC). 2017. The State of Saldanha Bay and Langebaan Lagoon 2017. Technical Report September 2017. Prepared for the Saldanha Bay Water Quality Forum Trust. Cape Town.

Anderson MJ, Gorley RN & KR Clarke. 2008. PERMANOVA+ for PRIMER: Guide to Software and Statistical Methods. PRIMER-E: Plymouth, UK, 214 pp.

Anderson RJ, Stegenga H & JJ Bolton. 2016. *Seaweeds of the South African South Coast*. World Wide Web electronic publication, University of Cape Town, http://southafrseaweeds.uct.ac.za; Accessed on 20/9/ 2019.

Angel A, Branch GM, Wanless RM & T Siebert. 2006. Causes of rarity and range restriction of an endangered, endemic limpet, *Siphonaria compressa*. *Journal of Experimental Marine Biology and Ecology* 330: 245–260.

ANZECC. 2000. Australian and New Zealand guidelines for fresh and marine water quality. Volume 2, Aquatic ecosystems. National water quality management strategy; no.4. Australian and New Zealand Environment and Conservation Council, Agriculture and Resource Management

Council of Australia and New Zealand ,Canberra, Australia. ISBN 0 642 19562 5 (www.deh.gov.au/water/quality/nwqms/introduction/).

AquaEco. 2012. Final Scoping Report - Offshore salmon farming in Saldanha Bay. Report prepared for Southern Cross Salmon Farming (Pty) Ltd.

Archambault P & E Bourget. 1996. Scales of coastal heterogeneity and benthic intertidal species richness, diversity and abundance. *Marine Ecology Progress Series* 136: 111–121 .

Arendse CJ. 2011. Aspects of the early life history and a per-recruit assessment of white stumpnose *Rhabdosargus globiceps* (Pisces: Sparidae) in Saldanha Bay with recommendations for future research and monitoring. MSc. Thesis, University of Cape Town, 120 pp.

ARUP. 2014. Transnet National Ports Authority – Saldanha Location Study for Rig or Ship Repair Facilities in the Port of Saldanha Prefeasibility (FEL2) Report. Report prepared by ARUP. Quay Oil and Gas Jetty. FEL2 Prefeasibility Report. Prepared by ARUP for Transnet, 25 pp.

ARUP. 2016. Transnet National Ports Authority – Saldanha Mossgas Quay Oil and Gas Jetty FEL2 Prefeasibility Report. Report number PDR\04\05 prepared by ARUP.

Atkinson L, Hutchings K, Clark B, Turpie J, Steffani N, Robinson T & A Duffell-Canham. 2006. State of the Bay 2006: Saldanha Bay and Langebaan Lagoon. Technical Report. Prepared for Saldanha Bay Water Quality Trust.

Attwood CG, Næsje TF, Fairhurst L & SE Kerwath. 2010. Life History parameters of white stumpnose *Rhabdosargus globiceps* (Pisces: Sparidae) in Saldanha Bay, South Africa, with evidence for stock separation. *African Journal of Marine Science* 32: 23–36 .

Awad AA, Clarke C, Greyling L, Hillard R, Polglaze J & S Raaymakers. 2003. Ballast water risk assessment, Port of Saldanha Bay, Republic of South Africa, Final Report November 2003. Global Ballast Water Management Programme. *Globallast Monograph Series* 13, 64 pp.

Awad A, Greyling L, Kirkman S, Botes L, Clark B, Prochazka K, Robinson T, Kruger L & L Joyce. 2005. *Port Biological Baseline Survey*. Draft Report. Port of Saldanha, South Africa. 41 pp.

Axelson E. 1977. A summary of the history of human settlement at Saldanha Bay. *Transactions of the Royal Society of South Africa* 42: 215–221.

Baldwin JR & JR Lovvorn. 1994. Expansion of seagrass habitat by the exotic Zostera japonica, and its use by dabbling ducks and brant in Boundary Bay, British Columbia. *Marine Ecology Progress Series* 103: 119–127.

Baptista NJA, Smith BJ & JJ Mcalliste. 2000. Heavy metal concentrations insurface sediments in a nearshore environment, Jurujuba Sound, SE Brazil. *Environmental Pollution* 109(1): 1–9.

Barahona M & SA Navarette. 2010. Movement patterns of the seastar *Heliaster helianthus* in central Chile: relationship with environmental conditions and prey availability. *Marine Biology* 157: 647–661.

Barnard KH. 1916. Contributions to the Crustacean Fauna of South Africa. *Annals of the South African Museum* 15(3): 1–302.

Barnard KH. 1951. New records and descriptions of new species of isopods and amphipods from South Africa. *Annals of Natural History* 12(5): 698–709.

Barnes RD. 1983. Invertebrate Zoology. Saunders, Philadelphia.

Bay S, Jones BH, Schiff K & L Washburn. 2003. Water quality impacts of storm water discharges to Santa Monica Bay. *Marine Environmental Research* 56(1-2): 205–223.

BCLME. 2006. The development of a common set of water and sediment quality guidelines for the coastal zone of the BCLME. Project BEHP/LBMP/03/04.

Becker G. 1971. On the biology, physiology and ecology of marine wood-boring crustaceans. Marine borers, fungi and fouling organisms of wood. *Organisaion for Economic Co-operation and Development*: 303–326.

Beckley LE. 1981. Marine benthos near the Saldanha Bay iron-ore loading terminal. *South African Journal of Zoology* 16(4): 269–271.

Bellmann MA & P Remmers. 2013. Noise mitigation systems (NMS) for reducing pile driving noise: Experiences with the "big bubble curtain" relating to noise reduction. *The Journal of Acoustical Society of America* 134(5): 4059.

Berkman PA, Garton DW, Haltuch MA, Kennedy GW & LR Febo. 2000. Habitat shift in invading species: Zebra and quagga mussel population characteristics on shallow soft substrates. *Biological Invasions* 2: 1–6.

Benkwitt R. 1982. Electrophoretic evidence for self-fertilization in tow species of spirorbid polychaetes. *Bulletin of the Southern California Academy of Sciences* 81: 61–68.

Bezuidenhout J, Dames N, Botha A, Frontasyeva MV, Goryainova ZI & D Pavlov. 2015. Trace Elements in Mediterranean mussels *Mytilus galloprovincialis* from the South African West *Coast. Ecological Chemistry and Engineering S* 22(4): 489–498.

Bhagwan J. 2012. Durban Water Recycling Project. Water Research Commission. Guidelines for Water Reuse, Appendix E International Case Studies. WRC Report number E-104.

Biccard A. 2012. Taxonomy, Systematics and Biogeography of South African Cirripedia (Thoracica). MSc. Thesis, University of Cape Town, 172 pp.

Bickerton IB. 1999. Saldanha Bay Water Quality Programme: Benthic Macrofaunal Monitoring. Council for Scientific and Industrial Research (CSIR), Cape Town, 18 pp.

Blackburn TM, Essl F, Evans T, Hulme PE, Jeschke JM, Kühn I, Kumschick S, Marková Z, Mrugała A, Nentwig W, Pergl J, Pyšek P, Rabitsch W, Ricciardi A, Richardson DM, Sendek A, Vilà M, Wilson JRU, Winter M, Genovesi P & Bacher S. 2014. A unified classification of alien species based on the magnitude of their environmental impacts. PLoS Biology 12: e1001850

Blackburn TM, Pyšek P, Bacher S, Carlton JT, Duncan RP & V Jarošík *et al*. 2011. A proposed unified framework for biological invasions. *Trends in Ecology and Evolution* 26: 333–339.

Bolton JJ, Andreakis N & RJ Anderson. 2011. Molecular evidence for three separate cryptic introductions of the red seaweed *Asparagopsis* (Bonnemaisoniales, Rhodophyta) in South Africa. *African Journal of Marine Science* 33: 263–271.

Bonvicini Pagliai AM, Cognetti Varriale AM, Crema R, Curini Galletti M & R Vandini Zunarelli. 1985. Environmental impact of extensive dredging in a coastal marine area. *Marine Pollution Bulletin* 16(12): 483–488.

Boonzaaier MK, Neethling S, Mouton A & CA Simon. 2014. Polydorid polychaetes (Spionidae) on farmed and wild abalone (*Haliotis midae*) in South Africa: an epidemiological survey. *African Journal of Marine Science* 36(3): 369–376.

Booth DB & CR Jackson. 1997. Urbanization of aquatic systems: degradation thresholds, storm water detection, and the limits of mitigation. *Journal of the American Water Resources Association* 33(5): 1077–1090.

Borges LMS, Valente AA, Palma P &L Nunes. 2010. Changes in the wood boring community in the Tagus Estuary: a case study. *Marine Biodiversity Records* 3: e41

Bornman TG, Adams JB & GC Bate. 2008. Environmental factors controlling the vegetation zonation patterns and distribution of vegetation types in the Olifants Estuary, South Africa. *South African Journal of Botany* 74: 685–695.

Bosman AL & PAR Hockey. 1986. Seabird guano as a determinant of rocky intertidal community structure. *Marine Ecology Progress Series* 32: 247–257.

Bosman AL & PAR Hockey. 1988. The influence of seabird guano on the biological structure of rocky intertidal communities on islands off the west coast of southern Africa. *South African Journal of Marine Science* 7: 61–68.

Bosman AL, Du Toit JT, Hockey PAR & GM Branch. 1986. A field experiment demonstrating the influence of seabird guano on intertidal primary production. *Estuarine, Coastal and Shelf Science* 23(3): 283–294.

Boucher C & ML Jarman. 1977. The vegetation of the Langebaan area, South Africa. *Transaction of the royal Society of South Africa* 42: 241–272.

Boussard A. 1981. The reactions of roach (*Rutilus rutilus*) and rudd (*Scardinius erythrophthalmus*) to noises produced by high speed boating. In: *Proceedings of the 2nd British Freshwater Fisheries Conference,* 188–200 pp.

Braaf O. 2014. Revised Scoping Report for a Proposed Phosphate Mine on Farm Elandsfontein No. 349, Portion 4 and a Portion of Portion 2, Malmesbury. Report prepared by Billet Trade (Pty) Ltd T/A Braaf Environmental Practitioners.

Branch GM, Griffiths CL, Branch ML & LE Beckley. 2010a. Two Oceans. David Philip Publishers, Cape Town, South Africa.

Branch GM, Odendaal F & TB Robinson. 2010. Competition and facilitation between the alien mussel *Mytilus galloprovincialis* and indigenous species: Moderation by wave action. *Journal of Experimental Marine Biology and Ecology* 383: 65–78.

Brand LE, Sunda, WG & RRL Guillard. 1983. Limitation of marine phytoplankton reproductive rates by zinc, manganese, and iron. *Limnology and Oceanography Inc*. 28(6): 1182–1198.

Bregman AS. 1990. Auditory Scene Analysis: The Perceptual Organization of Sound. Cambridge, MA, MIT Press.

Bridle AR, Crosbie PBB, Cadoret K & BF Nowak. 2010. Rapid detection and quantification of *Neoparamoeba perurans* in the marine environment. *Aquaculture* 309: 56–61.

BirdLife International 2017. *Haematopus moquini*. The IUCN Red List of Threatened Species 2017: e.T22693627A118385157. http://dx.doi.org/10.2305/IUCN.UK.2017-3.RLTS.T22693627A118385157.en. Downloaded on 15 August 2019.

BirdLife International 2018. *Morus capensis*. The IUCN Red List of Threatened Species 2018: e.T22696668A132587992. http://dx.doi.org/10.2305/IUCN.UK.2018-2.RLTS.T22696668A132587992.en. Downloaded on 15 August 2019.

Brooks KM, Mahnken C & C Nash. 2002. Environmental effects associated with marine netpen waste with emphasis on salmon farming in the Pacific Northwest. In: Stickney RR & McVey JP (eds); Responsible Marine Aquaculture CABI Publishing, New York.

Brouwer SL & MH Griffiths. 2005. Stock separation and life history of *Argyrozona argyrozona* (Pisces: Sparidae) on the South African east coast. *African Journal of Marine Science*, 27:3: 585–595.

Brown AL & RC Hill. 1995. Decision-scoping: Making environmental assessment learn how the design process works. *Project Appraisal* 10(4): 223–232.

Bulleri F & L Airoldi. 2005. Artificial marine structures facilitate the spread of a non-indigenous green alga, *Codium fragile* ssp. *tomentosoides* in the north Adriatic Sea. *Journal of Applied Ecology* 42: 1063–1072.

Brunetti R, Gissi C, Pennati R, Caicci F, Gasparini F & L Manni. 2015. Morphological evidence that the molecularly determined *Ciona intestinalis* type A and type B are different species: *Ciona robusta* and *Ciona intestinalis*. *Journal of Zoological Systematics and Evolutionary Research* 53: 186–193.

Bruun P. 1962. Sea level Rise as a Cause of Shore Erosion. *Journal of Waterways and Harbors Division:* 117–130.

Buchman MF. 1999. NOAA screening quick reference tables. 99-1, 1–12. Seattle WA, Coastal Protection and Restoration Division, National Oceanic and Atmospheric Administration. NOAA HAZMAT Report.

Burnett WC, Taniguchi M & J Oberdorfer. 2001. Measurement and significance of the direct discharge of groundwater into the coastal zone. *Journal of Sea Research* 46: 109–116.

Bustamante RH & GM Branch. 1996. Large scale patterns and trophic structure of southern African rocky shores: the roles of geographic variation and wave exposure. *Journal of Biogeography* 23: 339–351.

Bustamante RH, Branch GM & S Eekhout. 1997. The influences of physical factors on the distribution and zonation patterns of South African rocky-shore communities. *South African Journal of Marine Science* 18: 119–136.

Bustamante RH, Branch GM, Eekhout S, Robertson B, Zoutendyk P, Schleyer M, Dye A, Hanekom N, Keats D, Jurd M & C McQuaid. 1995. Gradients of intertidal primary productivity around the coast of South Africa and their relationships with consumer biomass. *Oecologia* 102: 189–201.

Caltrans. 2001. Pile installation demonstration project, fisheries impact assessment. PIDP EA 012081. San Francisco–Oakland Bay Bridge East Span Seismic Safety Project. *Caltrans Contract* 04A0148. San Francisco, CA.

Carlton JT & AN Cohen. 2003. Episodic global dispersal in shallow water marine organisms: the case history of the European shore crabs *Carcinus maenas* and *Carcinus aestuarii*. *Journal of Biogeography* 30: 1809–1820.

Carlton JT & JB Geller. 1993. Ecological roulette: the global transport and invasion of non indigenous marine organisms. *Science* 261: 78–82.

Carlton JT. 1999. The scale and ecological consequences of biological invasions in the world's oceans. In: Sandlund, OT *et al*. (Ed.), *Invasive Species and Biodiversity Management*. Kluwer Academic Publishing, Netherlands, 195–212 pp.

Carr GM & CJ Rickwood. 2008. Water Quality Index for Biodiversity Technical Development Document. Report prepared for Biodiversity Indicators Partnership, World Conservation Monitoring Centre, 219 Huntingdon Road, Cambridge CB3 0DL, United Kingdom, 64 pp.

Carr GM & CJ Rickwood. 2008. Water Quality Index for Biodiversity Technical Development Document. Report prepared for Biodiversity Indicators Partnership, World Conservation Monitoring Centre, 219 Huntingdon Road, Cambridge CB3 0DL, United Kingdom, 64 pp.

Carroll ML, Cochrane S, Fieler R, Velvin R & P White. 2003. Organic enrichment of sediments from salmon farming in Norway: environmental factors, management practices and monitoring techniques. *Aquaculture* 226(1-4): 165–180.

Castilla JC, Navarrete SA, Manzur T & M Barahona. 2013. *Heliaster helianthus*. In: Lawrence JM (ed), Starfish – Biology and ecology of the Asteroidea. The Johns Hopkins University Press, Baltimore, Maryland, 153–160 pp.

Castilla JC. 1981. Perspectivas de investigación en estructura y dinámica de comunidades intermareales rocosas de Chile central. II. Depredadores de alto nivel trófico. *Medio Ambiente* (Chile) 5: 190–215.

Castilla JC & RT Paine. 1987. Predation and community organization on Eastern Pacific, temperate zone, rocky intertidal shores. *Revista Chilena de Historia Natural* 60: 131–151.

Carter R & S Coles. 1998. Saldanha Bay General Cargo Quay Construction: Monitoring of suspended sediment distributions generated by dredging in Small Bay. CSIR Report ENVS98100, 26 pp.

Carter RA. 1996. Environmental impact assessment SSF Saldanha: the potential ecological impacts of ballast water discharge by oil tankers in the Saldanha Bay Langebaan lagoon System. CSIR report EMAS-C 96005D, Stellenbosch.

CCME. 2001. Canadian Sediment Quality Guidelines for the Protection of Aquatic Life. Canadian Environmental Quality Guidelines. Canadian Council of Ministers of the Environment, 1999, updated 2001.

Ceccherelli VU & Rossi R. 1984. Settlement, growth and production of the mussel *Mytilus galloprovincialis*. *Marine Ecology Progress Series* 16: 173–184.

CEF. 2008. Central Energy Fund –Newsroom: A state of the art oil spill protection, Tuesday, 22 April 2008. www.cef.org.za - Accessed 10 April 2009.

Cempel M & G Nikel. 2006. Nickel: A Review of Its Sources and Environmental Toxicology. *Polish Journal of Environmental Studies* 15(3): 375–382.

Chamber of Mines. 2017. Facts and Figures. 2016. Published by Chamber of Mines of South Africa in June 2017. Available [Online] at file:///C:/Users/vera.massie/Downloads/chamber-facts-figures-2016.pdf. Accessed on 18 August 2017.

Chace FA. 1956. Expédition océanographique Belge dans les eaux côtiéres africaines de l'Atlantique Sud (1948–1949). *Résultats scientifiques* 3(5): 17–43.

Christie ND & A Moldan. 1977. Effects of fish factory effluent on the benthic macrofauna of Saldanha Bay. *Marine Pollution Bulletin* 8: 41–45.

Christie ND. 1981. Primary production in Langebaan Lagoon. In: Day JH (ed) Estuarine Ecology with Particular reference to Southern Africa. Cape Town: Balkema, pp 101–115.

Çinar ME. 2013. Polychaetes (Annelida: Polychaeta) associated with *Posidonia oceanica* along the coasts of Turkey and northern Cyprus. In: Aktan Y, Aysel V (eds), First national workshop on *Posidonia oceanica* (L.) Delile on the coasts of Turkey, 19–20 September, Gökçeada, Turkey. Publication No. 39. Istanbul: Turkish Marine Research Foundation, pp 77–95.

Clark BM & CL Griffiths. 2012. Western pea crabs *Pinnixa occidentalis* Rathbun 1894 (Brachyura: Thoracotremata: Pinnotheroidea) invade Saldanha Bay, South Africa. *African Journal of Marine Science* 34: 153–156.

Clark BM. 1997. Variation in surf-zone fish community structure across a wave-exposure gradient. *Estuarine and Coastal Shelf Science* 44(6): 659–674.

Clark RB. 1986. *Marine Pollution*. Claredon Press, Oxford, 215 pp.

Clarke KR & RM Warwick. 2001. Change in Marine Communities: An Approach to Statistical Analysis and Interpretation, 2nd edition. PRIMER-E Ltd, Plymouth.

Clarke KR & RN Gorley. 2006. PRIMER v6: User Manual/Tutorial. PRIMER-E: Plymouth.

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

Cockcroft AC, van Zyl D & L Hutchings. 2008. Large-scale changes in the spatial distribution of South African West Coast rock lobsters: An overview. *African Journal of Marine Science* 30: 149–159.

Cohen A. 2011. *The Exotics Guide: Non-native Marine Species of the North American Pacific Coast*. Center for Research on Aquatic Bioinvasions, Richmond, CA, and San Francisco Estuary Institute, Oakland, CA. Revised September 2011. http://www.exoticsguide.org. Accessed 20/9/2019.

Cohen A, Zabin N & J Chela. 2009. Oyster shells as vectors for exotic organisms. *Journal of Shellfish Research* 28: 163–167.

Colan KE. 1990. Revision of the crustacean amphpod genus *Jass* Leach (Coroohioidea: Ischyroceridae). *Canadian Journal of Zoology* 68: 2031–2075.

Common Ground Consulting. 2013a. Environmental Repair, Rehabilitation, Maintenance and Management Plan for Shoreline and Beach Areas Associated with the Langebaan Beach Groynes and Rock Revetment. Prepared for Saldanha Bay Municipality. Saldanha, 18 pp.

Common Ground Consulting. 2013b. Environmental Repair, Rehabilitation, Maintenance and Management Plan for Shoreline and Beach Areas Adjacent to the Leentjiesklip Caravan Park, Alabama Street Slipway and the Melck Street Steps at Langebaan. Prepared for Saldanha Bay Municipality. Saldanha, 24 pp.

Conlan KE. 1990. Revision of the crustacean amphipod genus *Jassa* Leach (Corophioidea: Ischyroceridae). *Canadian Journal of Zoology* 68: 2031–2075.

Conrad J. 2014. Geohydrological Assessment – Updated Scoping Report - Elandsfontein, West Coast iii GEOSS Report No. 2014/08-08 02 September.

Costello MJ, Grant A, Davies IM, Cecchini S, Papoutsoglou S, Quigley D & M Saroglia. 2001. The control of chemicals used in aquaculture in Europe. *Journal of Applied Ichthyology* 17: 173–180.

Crawford RJM. 2007. Trends in the numbers of three cormorants *Phalacrocorax* spp. breeding in South Africa's Western Cape Province. In: Kirkman SP (ed.) Final Report of the BCLME (Benguela Current Large Marine Ecosystem). Project on Top Predators as Biological Indicators of Ecosystem Change in the BCLME: 173–178. Avian Demography Unit, Cape Town.

Crawford RJM. 2009. A recent increase of swift terns Thalasseus bergii off South Africa – The possible influence of an altered abundance and distribution of prey. *Progress in Oceanography* 83: 398–403.

Crawford RJM, Cooper J & PA Shelton. 1982. Distribution, population size, breeding and conservation of the Kelp Gull in southern Africa. *Ostrich* 53: 164 –177.

Crawford RJM & LG Underhill. 2003. Aspects of breeding, molt, measurements and population trend of Hartlaub's gull *Larus hartlaubii* in Western Cape, South Africa. *Waterbirds* 26: 139–149.

Crawford RJM, Altwegg R, Barham BJ, Barham PJ, Durant JM, Dyer BM, Makhado AB, Pichegru L, Ryan PG, Underhill LG, Upfold L, Visagie J, Waller LJ & PA Whittington. 2011. Collapse of South Africa's penguins in the early 21st century: a consideration of food availability. *African Journal of Marine Science* 33: 139–156.

Crawford RJM, Azwianewi M, Makhado B, Whittington PA, Randall RM, Oosthuizen WH & LJ Waller 2015. A changing distribution of seabirds in South Africa—the possible impact of climate and its consequences. *Frontiers in Ecology and Evolution* 3: 1–11.

Crawford RJM, Barham PJ, Underhill LG, Shannon LJ, Coetzee JC, Dyer BM, Mario Leshoro T & L Upfold. 2006. The influence of food availability on breeding success of African penguins *Spheniscus demersus* at Robben Island, South Africa. *Biological Conservation* 132: 119–125.

Crawford RJM, Cockcroft AC, Dyer BM & L Upfold. 2008c. Divergent trends in bank cormorants *Phalacrocorax neglectus* breeding in South Africa's Western Cape consistent with a distributional shift of rock lobsters *Jasus lalandii*. *African Journal of Marine Science* 30: 161–166.

Crawford RJM, Dundee BL, Dyer BM, Klages NT, Meyer MA & L Upfold. 2007. Trends in numbers of Cape gannets (*Morus capensis*), 1956/57–2005/06, with a consideration of the influence of food and other factors. *ICES Journal of Marine Science* 64: 169–177.

Crawford RJM, Dyer BM & RK Brooke. 1994. Breeding nomadism in southern African seabirds – constraints, causes and conservation. *Ostrich* 65: 231–246.

Crawford RJM, Dyer BM, Kemper J, Simmons RE & L Upfold. 2007. Trends in numbers of Cape Cormorants (*Phalacrocorax capensis*) over a 50 year period, 1956-57 to 2006-07. *Emu* 107: 253–261.

Crawford RJM, Makhado AB, Waller LJ & PA Whittington. 2014. Winners and losers – response to recent environmental change by South African seabirds that compete with purse-seine fisheries for food. *Ostrich* 85: 111–117.

Crawford RJM, Nel DC, Williams AJ & A Scott. 1997. Seasonal patterns of abundance of Kelp Gulls *Larus dominicanus* at breeding and non-breeding localities in southern Africa. *Ostrich* 68: 37–41.

Crawford RJM, Randall RM, Whittington PA, Waller L, Dyer BM, Allan DG, Fox C, Martin AP, Upfold L, Visagie J, Bachoo S, Bowker M, Downs CT, Fox R, Huisamen J, Makhado AB, Oosthuizen WH, Ryan PG, Taylor RH & JK Turpie. 2013. South Africa's coastal white-breasted cormorants: population trends, breeding season and movements, and diet. *African Journal of Marine Science* 35(4): 473–490.

Crawford RJM, Sabarros PS, Fairweather T, Underhill LG & AC Wolfaardt. 2008a. Implications for seabirds off South Africa of a long-term change in the distribution of sardine. *African Journal of Marine Science* 30: 177–184.

Crawford RJM, Underhill LG, Coetzee JC, Fairweather T, Shannon LJ & AC Wolfaardt. 2008b. Influences of the abundance and distribution of prey on African penguins *Spheniscus demersus* off western South Africa. *African Journal of Marine Science* 30: 167–175.

Critchley AT, Farnham WF & CH Thorp. 1997. On the co-occurrence of two exotic, invasive marine organisms: the brown seaweed *Sargassum muticum* (Yendo) Fensholt and the Spirobid tube worm *Janua* (*Neodexiospira*) *brasiliensis* (Grube), in association with the indigenous eelgrass, *Zostera marina* L. and Wrack, *Fucus serratus* L. in the south-west Netherlands and the Channel Islands, Europe*, Zoster*. *South African Journal of Botany* 63: 474–479.

Cruz-Rivera E & ME Hay. 2001. Macroalgal traits and the feeding and fitness of an herbivorous amphipod: the roles of selectivity, mixing, and compensation. *Marine Ecology Progress Series* 218: 249–266.

Cruz-Motta JJ, Underwood AJ, Chapman MG & F Rossi. 2003. Benthic assemblages in sediments associated with intertidal boulder-fields. *Journal of Experimental Marine Biology and Ecology* 285-286: 383–401.

CSIR. 1996. Environmental Impact Assessment: Proposed changes to oil transfer operations, SFF, Saldanha Bay: Specialist Study (S3(i): Effects of dredging activities on turbidity levels and shoreline stability in Saldanha Bay, prepared by Mocke G, Luger S, Schoonees KS, Smit F & Theron AK, 102pp.

CSIR. 2002. Saldanha Bay marine water quality management plan. Phase I: Situation Assessment. Report to the Saldanha Bay Water Quality Forum Trust. CSIR Report ENV-S-C, Stellenbosch.

CSIR. 2006. The development of a common set of water and sediment quality guidelines for the coastal zone of BCLME, South Africa. Prepared for Benguela Current Large Marine Ecosystem Programme. CSIR Report No. CSIR/NRE/ECO/2006/0011/C. Stellenbosch, 164 pp.

CSIR. 2006. The development of a common set of water and sediment quality guidelines for the coastal zone of BCLME, South Africa. Prepared for Benguela Current Large Marine Ecosystem Programme. CSIR Report No. CSIR/NRE/ECO/2006/0011/C. Stellenbosch, 164 pp.

CSIR 2012a. Final Environmental Impacts Assessment Report for the Proposed Construction, Operation and Decommissioning of a Seawater Reverse Osmosis Plant and Associated Infrastructure in the Saldanha Bay Region, Western Cape.

CSIR 2012b. Final Scoping Report for the Proposed Construction, Operation and Decommissioning of a Seawater Reverse Osmosis Plant and Associated Infrastructure in the Saldanha Bay Region, Western Cape. Prepared for WorleyParsons (Pty) Ltd. CSIR Report No. GWDS Stel General: 1159, Stellenbosch.

CSIR 2013a. Environmental Impact Assessment (EIA) for the Proposed Construction, Operation, and Decommissioning of the Marine Outfall Pipeline and Associated Infrastructure for Frontier Saldanha Utilities (Pty) Ltd in the Saldanha Bay Region, Western Cape. Background Information Document. Prepared for Frontier Saldanha Utilities (Pty) Ltd, 8pp.

CSIR 2013b. Environmental Impact Assessment (EIA) for the Proposed Construction, Operation and Decommissioning of the Saldanha Regional Marine Outfall Project for Frontier Saldanha Utilities (Pty) Ltd at Danger Bay in the Saldanha Bay Region. Draft Scoping Report. Prepared for Frontier Saldanha Utilities (Pty) Ltd.

CSIR 2013c. Final EIA Report for the Proposed Construction, Operation and Decommissioning of a Seawater Reverse Osmosis Plant and Associated Infrastructure in the Saldanha Bay Region, Western Cape. CSIR Report No. GWDS 1159 EMS0014, Stellenbosch.

CSIR 2015. Port of Saldanha Iron Ore Terminal Reverse Osmosis Plant Environmental Monitoring Programme: Summary of Previous Study Findings and Recommended Sampling Design. CSIR Report CSIR/NRE/ECOS/ER/2015/0004/B.

Cubit JD. 1984. Herbivory and the seasonal abundance of algae on a high intertidal rocky shore. *Ecology* 66: 1904–1917.

da Silva C, Kerwath SE, Attwood CG, Thorstad EB, Cowley PD, Økland F, Wilke CG, & TF Næsje. 2013. Quantifying the degree of protection afforded by a no-take marine reserve on an exploited shark. *African Journal of Marine Science*, 35(1): 57–66.

Dalal-Clayton B & B Sadler. 2005. *Strategic Environmental Assessment: A Sourcebook and Reference Guide to International Experience*. Earthscan, London.

David AA & CA Simon. 2014. The effect of temperature on larval development of two non-indigenous poecilogonous polychaetes (Annelida: Spionidae) with implications for life history theory, establishment and range expansion. *Journal of Experimental Marine Biology and Ecology* 461: 20–30.

David AA, Matthee CA, Loveday BR & CA Simon. 2016. Predicting the Dispersal Potential of an Invasive polychaete pest along a complex coastal biome. *Integrative and Comparative Biology* 56(4): 600 pp.

David JHM, Cury P, Crawford RJM, Randall RM, Underhill LG & MA Meyer. 2003. Assessing conservation priorities in the Benguela ecosystem, South Africa: analysing predation by seals on threatened seabirds. *Biological Conservation* 114: 289–292.

Davidson IC, Crook AC & DKA Barnes. 2004. Quantifying spatial patterns of intertidal biodiversity: is movement important? *Marine Ecology* 25: 15–34.

Day JH. 1959. The biology of Langebaan Lagoon: a study of the effect of shelter from wave action. *Transactions of the Royal Society of South Africa* 35: 475–547.

Day JH. 1974. A guide to marine life on South African shores. A.A. Balkema, Cape Town.

Day JH. 1981. The estuarine flora. In: Day JH (ed.) Estuarine Ecology with Particular reference to Southern Africa. Cape Town: Balkema, 77–99 pp.

De Blauwe H. 2009. *Mosdiertjes van de Zuidelijke bocht van de Noordzee: Determinatiewerk voor België en Nederland*. Vlaams Instituut voor de Zee (VLIZ): Oostende. ISBN 978-90-812-9003–6. 445 pp.

De Clerck O, Anderson RJ, Bolton JJ & D Robertson-Andersson. 2002. *Schimmelmannia elegans* (Gloiosiphoniaceae, Rhodophyta): South Africa's first introduced seaweed? *Phycologia* 41: 184–190.

de Greef K, Griffiths CL & Z Zeeman. 2013. Deja vu? A second mytilid mussel, *Semimytilus algosus*, invades South Africa's west coast. *African Journal of Marine Science* 35(3): 307–313.

de Moor CL & DS Butterworth. 2015. Assessing the South African sardine resource: two stocks rather than one? *African Journal of Marine Science* 37(1): 41–51.

de Ponte Machado M. 2007. Is predation on seabirds a new foraging behaviour for great white pelicans? History, foraging strategies and prey defensive responses. Final report of the BCLME (Benguela Current Large Marine Ecosystem) project on top predators as biological indicators of ecosystem change in the BCLME. Avian Demography Unit, Cape Town, 131–142 pp.

De Villiers CC & RC Hill. 2008. Environmental management frameworks as an alternative to farm-level EIA in a global biodiversity hotspot: A proposal from the Cape Floristic Region, South Africa. *Journal of Environmental Assessment Policy and Management* 10(4): 333–360.

Denny M & B Gaylord. 2002. The mechanics of wave-swept algae. *Journal of Experimental Biology* 205: 1355–1362.

Denny MW & S Gaines. 2007. Encyclopedia of Tidepools and Rocky Shores. University of California Press, 705 pp.

Department of Agriculture, Forestry and Fisheries (DWAF). 1997. Water Services Act. Government Gazette of the Republic of South Africa. 390 (18522), 19 December 1997.

Department of Agriculture, Forestry and Fisheries (DWAF). 1998. National Water Act. Government Gazette of the Republic of South Africa. 398 (19182) 26 August 1998.

Department of Agriculture, Forestry and Fisheries (DAFF). 2013. Status of South African Marine Fishery Resources 2012.

Department of Agriculture, Forestry and Fisheries (DAFF). 2014. Aquaculture Yearbook 2013 South Africa.

Department of Agriculture, Forestry and Fisheries (DAFF). 2016. Aquaculture Yearbook 2016 South Africa. Published by the Department of Agriculture, Forestry and Fisheries, Branch: Fisheries Management. ISBN: 978-0-621-46172-5.

Department of Agriculture, Forestry and Fisheries (DAFF). 2017. South African Live Molluscan Shellfish Monitoring and Control Programme. Published by the Fisheries Management Branch under the Chief Directorate of Aquaculture and Economic Development and the Directorate of Sustainable Aquaculture Management. 54 pp.

Department of Agriculture, Forestry and Fisheries (DAFF). 2018. Protocols for Environmental Monitoring of the Aquaculture Development Zone in Saldanha Bay, South Africa. A report for the Department of Agriculture, Forestry, and Fisheries produced by Dr T. Probyn. 44pp.

Department of Environmental Affairs (DEA) 2014. National Guideline for the Discharge of Effluent from Land-based Sources into the Coastal Environment. Pretoria, South Africa. RP101/2014.

Department of Environmental Affairs (DEA). 2012. South African Water Quality Guidelines for Coastal Marine Waters. Volume 2: Guidelines for Recreational Use, Cape Town.

Department of Environmental Affairs & Development Planning (DEA&DP) 2014. Environmental Authorisation: The Proposed Construction of a Liquid Petroleum Gas Handling Facility and Associated Infrastructure on Farm Yzervarkensrug 127/13, Saldanha. EIA Reference Number: E12/2/4/2/F4/16/3001/12.

Department of Environmental Affairs &Development Planning (DEA&DP) 2015. Draft *Environmental Management Framework for the Greater Saldanha Area*. Report number J-649D-13. February 2015.

Department of Environmental Affairs and Development Planning (DEA&DP) 2016. Integrating Environmental Management with Spatial Planning in Greater Saldanha Bay – Western Cape.

Department of Environmental Affairs and Development Planning (DEA&DP). 2017. Government Progress Report. Presentation given by Marlene Laros and Kobus Munro at the Saldanha Bay Water Quality Forum Trust Open Day on 14 October 2016.

Department of Environmental Affairs and Development Planning (DEA&DP). 2017. Draft Environmental Management Framework for the Greater Saldanha Area. Completed as part of the Greater Saldanha Regional Implementation Framework (GS RSIF).

Department of Environmental Affairs and Development Planning (DEA&DP). 2018. Greater Saldanha Regional Spatial Implementation Framework. Summary Report and Implementation Framework. June 2018.

Department of Agriculture, Forestry and Fisheries (DAFF). 2018. Protocols for Environmental Monitoring of the Aquaculture Development Zone in Saldanha Bay, South Africa. A report for the Department of Agriculture, Forestry, and Fisheries produced by Dr T.

Department of Water Affairs (DWA) 2013. Support to the Continuation of the Water Reconciliation Strategy for the Western Cape Water Supply System: Status Report October 2013. Prepared by Umvoto Africa (Pty) Ltd in association with WorleyParsons on behalf of the Directorate: National Water Resource Planning, South Africa.

Department of Water Affairs & Forestry (DWAF). 1995a. South African Water Quality Guidelines for Coastal Marine Waters. Volume 1 - Natural Environment. Pretoria.

Department of Water Affairs & Forestry (DWAF). 1995b. South African Water Quality Guidelines for Coastal Marine Waters. Volume 2 - Recreational Use. Pretoria.

Department of Water Affairs & Forestry (DWAF). 1995c. South African Water Quality Guidelines for Coastal Marine Waters. Volume 3. Industrial use. Pretoria.

Department of Water Affairs & Forestry (DWAF). 1995d. South African Water Quality Guidelines for Coastal Marine Waters. Volume 4. Mariculture. Pretoria.

Department of Water Affairs & Forestry (DWAF). 1997. Water Services Act. Government Gazette of the Republic of South Africa. 390 (18522), 19 December 1997.

Department of Water Affairs & Forestry (DWAF). 1998. National Water Act. Government Gazette of the Republic of South Africa. 398 (19182) 26 August 1998.

Department of Water and Sanitation (DWS) 2017. Berg Water Management Area Classification and Resource Quality Objectives: Estuary Component.

Desideri D, Meli MA, Roselli C & L Feduzi. 2009. A biomonitoring study: 210Po and heavy metals in mussels. *Journal of Radioanalytical and Nuclear Chemistry* 279: 591–600.

Dias MP, Granadeiro JP, Lecog M, Santos CD & JM Palmeirim. 2006. Distance to high-tide roosts constrains the use of foraging areas by dunlins: implications for the management of estuarine wetlands. *Biological Conservation* 131: 446–452.

Douglas-Helders GM, Dawson DR, Carson J & BF Nowak. 2002. Wild fish are not a significant reservoir for *Neoparamoeba pemaquidensis*. *Journal of Fish Disease* 25: 69–574.

du Toit M. 2004. Avian Demography Unit, Dept of Statistical Sciences, University of Cape Town (UCT). http://web.uct.ac.za/depts/stats/adu/species/bankcormorant.htm ; http://web.uct.ac.za/depts/stats/adu/species/capecormorant.htm ; http://web.uct.ac.za/depts/stats/adu/species/crcormorant.htm and http://web.uct.ac.za/depts/stats/adu/species/wbcormorant.htm

Duarte CM. 2002. The future of seagrass meadows. *Environmental Conservation* 29: 192–206.

Eckman JE. 1990. A model of passive settlement by planktonic larvae onto bottoms of differing roughness. *Limnology and Oceanography* 35: 887–901.

Ecosense Consulting Environmentalists. 2017. Basic Assessment Report for the Proposed Molapong Aquaculture Project. Report Prepared for Molapong in July 2017.

Eddy LJ. 2003. Sewage wastewater management in South Africa. MSc Thesis. Department of Geography, Environmental Management and Energy Studies (EDTs), University of Witwatersrand.

Emery NC, Ewanchuk PJ & MD Bertness. 2001. Competition and saltmarsh plant zonation: Stress tolerators may be dominant competitors. *Ecology* 82: 2471–2485.

Environmental Resources Management (ERM) 2015a. Independent Power Producers Programme: EIA for a Floating Power Plant, Port of Saldanha. Background Information Document. Report Prepared by Environmental Resources Management for the Department of Energy Republic of South Africa. October 2015, 10 pp.

Environmental Resources Management (ERM) 2015b. Independent Power Producers Programme: EIA for a Floating Power Plant, Port of Saldanha. Draft Scoping Report. Report Prepared by Environmental Resources Management for the Department of Energy Republic of South Africa. Document Code 0320754, November 2015, 114 pp.

Environmental Resources Management (ERM) 2015c. Environmental Impact Assessment Report for a Gas-fired Independent Power Plant to Support Saldanha Steel and Other Industries in Saldanha Bay, Western Cape. Draft Report version 2. Report prepared by Environmental Resources Management for Arcelor Mittal. Document Code 0315829, September 2016, 432 pp.

Eno NC, Clark RA & WG Sanderson. 1997. *Non-native species in British waters: a review and directory*. Peterborough: Joint Nature Conservation Committee. 136 pp.

EPA. 2003. After the storm. US Environmental Protection Agency. EPA 833-B-03-002. http://epa.gov/weatherchannel/storm water.html.

Erasmus T & AF De Villiers. 1982. Ore dust pollution and body temperatures of intertidal animals. *Marine Pollution Bulletin* 13(1): 30–32.

Erftemeijer PLA & RRR Lewis. 2006. Environmental impacts of dredging on seagrasses: a review. *Marine Pollution Bulletin* 52: 1553–1572.

Falco GDE, Magni P, Terasvuori LMH & Matteucci G. 2004. Sediment grain size and organic carbon distribution in the Cabras Lagoon (Sardinia, Western Mediterranean). *Chemistry and Ecology* 20: 367–377.

Faulkner KT, Robertson MP, Rouget M & JRU Wilson. 2017. Prioritising surveillance for alien organisms transported as stowaways on ships travelling to South Africa. *PLoS ONE* 12: e0173340.

Fernández M & JC Castilla. 2000. Recruitment of *Homalaspis plana* in intertidal habitats of central Chile and implications for the current use of Management and Marine Protected Areas. *Marine Ecology Progress Series* 208: 157–170.

Fey A. 1970. Peuplements sessiles de l'archipel de Glenan. I.- Inventaire: hydraires. Vie & Milieu 20: 387–413.

Flemming BW. 1977a. Distribution of recent sediments in Saldanha Bay and Langebaan Lagoon. *Transactions of the Royal Society of South Africa* 42(3–4): 317–340.

Flemming BW. 1977b. Depositional processes in Saldanha Bay and Langebaan Lagoon. Unpublished PhD Thesis, University of Cape Town, 397 pp.

Flemming BW. 2015. Depositional processes in Saldanha Bay and Langebaan Lagoon (Western Cape, South Africa). National Research Institute for Oceanology (NRIO), Stellenbosch, CSIR Research Report 362 (revised edition), 233 pp.

Flemming BW. 2016. Long-term impacts of harbour construction in Saldanha Bay (South Africa). Unpublished report, 8 pp.

Florence WK, Hayward PJ & MJ Gibbons. 2007. Taxonomy of shallow water Bryozoa from the west coast of South Africa. *African Natural History* 3: 1–58.

Fofonoff PW, Ruiz GM, Steves B, Simkanin C & JT Carlton. 2018. National Exotic Marine and Estuarine Species Information System. http://invasions.si.edu/nemesis/. Access Date: 16/9/2019.

Food and Agricultural Organization of the United Nations (FAO). 2015. Aquaculture operations in floating HDPE cages. A field handbook. Cardia F. & A. Lovatelli Fisheries and Aquaculture Technical Paper 593.

Fuentes HR. 1982. Feeding habitats of *Graus nigra* (Labridae) in coastal waters of Iquique in northern Chile*. Japanese Journal of Ichthyology* 29: 95–98.

Fuggle RF. 1977. A review of a symposium on research in the natural sciences at Saldanha Bay and Langebaan Lagoon. *Transactions of the Royal Society of South Africa* 42(3–4): 211–214.

Furlong A. 2017. West Coast Phosphate Mine Stopped – for Now. Article published in GroundUp on 15 September 2017. Available [Online]: https://www.groundup.org.za/article/west-coast-phosphate-mine-stopped-now (Accessed 16 September 2017).

Galil BS, Occhipinti-Ambrogi A & S Gollasch. 2008. Biodiversity Impacts of Species Introductions Via Marine Vessels. In: Abdulla A & Linden O (Eds) *Maritime traffic effects on biodiversity in the Mediterranean Sea Volume 1 - Review of Impacts, Priority Areas and Mitigation Measures*. Malaga, Spain: IUCN Centre for Mediterranean Cooperation, Chapter 4, 117–158 pp.

Game ET, Grantham HS, Hobday AJ, Pressey RL, Lombard AT, Beckley LE, Gjerde K, Bustamante R, Possingham HP & AJ Richardson. 2009. Pelagic protected areas: the missing dimension in ocean conservation. *Trends in Ecology and Evolution* 24: 360–369.

Ganter B. 2000. Seagrass (*Zostera* spp.) as food for brent geese (*Branta bernicla*): an overview. *Helgoland Marine Research* 54: 63–70.

Gaymer CF & JH Himmelman. 2008. A keystone predatory sea star in the intertidal zone is controlled by a higher-order predatory sea star in the subtidal zone. *Marine Ecology Progress Series* 370: 143–153.

GEOSS. 2014. Geohydrological Assessment – Updated Scoping Report - Elandsfontein, West Coast. GEOSS Report Number: 2014/08-08. GEOSS - Geohydrological & Spatial Solutions International (Pty) Ltd. Stellenbosch, South Africa.

GEOSS. 2014. Geohydrological Assessment – Updated Scoping Report - Elandsfontein, West Coast. GEOSS Report Number: 2014/08-08. GEOSS - Geohydrological & Spatial Solutions International (Pty) Ltd. Stellenbosch, South Africa.

Geraldi NR, Smyth AR, Piehler MF & CH Peterson. 2014. Artificial substrates enhance non-native macroalga and N2 production. *Biological Invasions* 16: 1819–1831.

Gericke J. 2008. Analysis of Four Decades of Changes to Sedimentary Features by means of Historical Aerial Photographs: Langebaan Lagoon and Saldanha Bay. Honours Thesis, University of Cape Town, Department of Environmental and Geographical Science.

GIBB. 2013. Port of Saldanha: Proposed Expansion of Existing Iron Ore Terminal Background Information for Public Participation. GIBB (Pty) Ltd. Prepared for Transnet.

Gibbs RJ. 1994. Metals in sediments along the Hudson River Estuary. *Environmental International* 20: 507–516.

Gibson R, Hextall B & A Rogers. 2001. Photographic guide to the sea and shore life of Britain and north-west Europe. New York: Oxford University.

Glasson J, Thérivel R & A Chadwick. 1999. *Introduction to Environmental Impact Assessment: Principles and Procedures, Process, Practice and Prospects*. UCL Press, London.

Gollasch S, Macdonald E, Belson S, Botnen H, Christensen JT, Hamer JP, Houvenaghel G, Jelmert A, Lucas I, Masson D, Mccollin T, Olenin S, Persson A, Wallentinus I, Wetsteyn LPMJ & T Wittling. 2002. Life in Ballast tanks. In: *Invasive Aquatic Species of Europe: Distribution, Impacts and Management*. Leppakoski E, Gollasch S, Olenin, S (Eds.), Academic Publishers, Netherlands, 217–231 pp.

Gordon DP & SF Mawatari. 1992. Atlas of Marine Fouling Bryozoa of New Zealand Ports and Harbours. *New Zealand Oceanographic Institute, Miscellaneous Publications* 107: 52 pp.

Gosner KL. 1978. *A field guide to the Atlantic seashore*. The Peterson field guide series. Houghton-Mifflin. Boston.

Gowen RJ & IA Ezzi. 1992. Assessment and production of the potential for hypernutrification and eutrophication associated with cage culture of salmonids in Scottish coastal waters. Dunstaffnage Marine Laboratory, Oban

Grant WS & MI Cherry. 1985. *Mytilus galloprovincialis* Lmk in southern Africa. *Journal of Experimental Marine Biology and Ecology* 90(2): 179–162.

Grant WS, Cherry MI & AT Lombard. 1984. A cryptic species of *Mytilus* (Mollusca: Bivalvia) on the west coast of South Africa. *South African Journal of Marine Science* 2: 149–162.

Green Etxabe A. 2013. *The wood boring amphipod Chelura terebrans*, University of Portsmouth 1–232 pp.

Grémillet D, Péron C, Kato A, Amélineau F, Ropert-Coudert Y, Ryan PG & L Pichegru. 2016. Starving seabirds: unprofitable foraging and its fitness consequences in Cape gannets competing with fisheries in the Benguela upwelling ecosystem. *Marine Biology* 163(2): 35.

Griffiths CL. 1974. The Amphipoda of Southern Africa Part 4: The Gammaridea and Caprellidea of the Cape Province east of Cape Agulhas. *Annals of the South African Museum* 65(9): 251–336.

Griffiths CL, Hockey PAR, Van Erkom Schurink C & PJL Roux. 1992. Marine invasive aliens on South African shores: Implications for community structure and trophic functioning. *South African Journal of Marine Science* 12: 713–722.

Griffiths CL, Roberts S, Branch GM, Eckel K, Schubart CD & R Lemaitre. 2018. The porcelain crab *Porcellana africana* Chace, 1956 (Decapoda: Porcellanidae) introduced into Saldanha Bay, South Africa. *BioInvasions Records* 7(2): 133–142.

Griffiths CL, Robinson T & A Mead. 2011. The Alien and Cryptogenic Marine Crustaceans of South Africa. In: Galil BS, Clark PF & Carlton JT (eds). *In the Wrong Place - Alien Marine Crustaceans: Distribution, Biology and Impacts Invading Nature*. Springer Series in Invasion Ecology 6.

Griffiths CL, Robinson TB & A Mead. 2008. The status and distribution of marine alien species in South Africa. In: Rilov G & Crooks J (eds.). *Marine Bioinvasions: Ecology, Conservation and Management Perspectives*. Springer, Heidelberg 204, 393–408 pp.

Griffiths CL, Robinson TB & A Mead. 2009. The status and distribution of marine alien species in South Africa: biological invasions in marine ecosystems. *Ecological Studies* 204: 393-408.

Griffiths CL, van Sittert L, Best PB, Brown AC, Clark BM, Cook PA, Crawford RJM, David JHM, Davies BR, Griffiths MH, Hutchings K, Jerardino A, Kruger N, Lamberth S, Leslie RW, Melville-Smith R, Tarr R & CD Van Der Lingen. 2004. Impacts of human activities on marine animal life in the Benguela: A historical overview. *Oceanography and Marine Biology: An Annual Review* 42: 303–392.

Grindley JR. 1977. The zooplankton of Langebaan Lagoon and Saldanha Bay. *Transactions of the Royal Society of South Africa* 42: 341–369.

Guarnieri G, Terlizzi A, Bevilacqua S & S Fraschetti. 2009. Local vs. regional effects of substratum on early colonization stages of sessile assemblages: biofouling. *Journal of Bioadhesion and Biofilm Research* 25: 593–604.

Guerra-Garcia, JM & CJ Garcia-Gomez. 2004. Polychaete assemblages and sediment pollution in a harbour with two opposing entrances. *Helgoland Marine Research* 58: 183–191.

Guichard F & E Bourget. 1998. Topographic heterogeneity, hydrodynamics, and benthic community structure: a scale-dependent cascade. *Marine Ecology Progress Series* 171: 59–70.

Hampton SL & CL Griffiths. 2007. Why *Carcinus maenas* cannot get a grip on South Africa's wave-exposed coastline. *African Journal of Marine Science* 29(1): 123–126.

Hanisak MD. 1979. Growth patterns of *Codium fragile tomentosoides* in response to temperature, irradiance, salinity, and nitrogen source. *Marine Biology* 50: 319–332.

Hanekom N & P Nel . 2002. Invasion of sandflats in Langebaan Lagoon, South Africa, by the alien mussel *Mytilus galloprovincialis*: size, composition and decline of the populations. *African Zoology* 37: 197–208.

Harris LG & AC Jones. 2005. Temperature, herbivory and epibiont acquisition as factors controlling the distribution and ecological role of an invasive seaweed. *Biological Invasions* 7: 913–924.

Haupt TM, Griffiths CL, Robinson TB & AFG Tonin. 2010. Oysters as vectors of marine aliens, with notes on four introduced species associated with oyster farming in South Africa. *African Zoology* 45(1): 52–62.

Hayward PJ & JS Ryland. 1999. *Cheilostomatous* Bryozoa. Part 2: Hippothoidea-Celleporoidea, *Synopses of the British Fauna* 14: 1–416.

He H, Chen F, Li H, Xiang W, Li Y & Y Jiang. 2009. Effect of iron on growth, biochemical composition and paralytic shellfish poisoning toxins production of *Alexandrium tamarense*. *Harmful Algae* 9(1): 98–104.

Healy B. 1997. Long-term changes in a brackish lagoon, Lady's Island Lake, South-east Ireland. Biology and Environment: *Proceedings of the Royal Irish Academy* 97B(1): 33–51.

Heck KL, Hays C & RJ Orth. 2003. A critical evaluation of the nursery role hypothesis for seagrass meadows. *Marine Ecology Progress Series* 253: 123–136.

Hedger RD, Næsje TF, Cowley PD, Thorstad EB, Attwood C, Økland F, Wilke CG & S Kerwath. 2010. Residency and migratory behaviour by adult *Pomatomus saltatrix* in a South African coastal embayment. *Estuarine, Coastal and Shelf Science* 89: 12–20.

Heggoey E, Vassenden G & P Johannessen. 2004. Investigation of the marine biological environmental conditions at a fish farm site in Radfjorden, Lindas Municipality, western Norway in 2004. *Vestbio* 1: 35 [English abstract].

Heinecken C. 2018. Report on the baseline sample collection for the Saldanha Bay Aquaculture Development Zone, South Africa. Reported prepared by Capricorn Fisheries Monitoring cc. 13 pp.

Hellawell JM. 1986. Biological indicators of freshwater pollution and environmental management. London: Elsevier Applied Science Publishers.

Hemmingra MA & CM Duarte. 2000. *Seagrass Ecology.* Cambridge: University Press.

Henschel JR, Cook PA & GM Branch. 1990. The colonization of artificial substrata by marine sessile organisms in False Bay. I. Community development. *South African Journal of Marine Science* 9: 289–297.

Henzler CM & A Ingolfsson. 2008. The biogeography of the beachflea, *Orchestia gammarellus* (Crustacea, Amphipoda, Talitridae), in the North Atlantic with special reference to Iceland: a morphometric and genetic study. *Zoologica Scripta* 37: 57–70.

Herman PMJ, Hemmingra MA, Nienhuis PH, Verschuure JM & EGJ Wessel. 1996. Wax and wane of eelgrass *Zostera marina* and water column silicon levels. *Marine Ecology Progress Series. Oldendorf* 14(1): 303–307.

Hewitt CL, Gollasch S & D Minchin. 2009. The vessel as a vector – biofouling ballast water and sediments. In: *Biological Invasions in Marine Ecosystems*, Rilov G & Crooks JA (eds.). Springer-Verlag, Berlin, 117–131 pp.

Hir ML & C Hily. 2005. Macrofaunal diversity and habitat structure in intertidal boulder fields. *Biodiversity and Conservation* 14: 233–250.

Hockey PAR & C van Erkom Schurink. 1992. The invasive biology of the mussel *Mytilus galloprovincialis* on the southern African coast. *Transactions of the Royal Society of South Africa* 48: 123–139.

Hockey PAR, Dean WRJ & PG Ryan (eds.) 2005. *Roberts Birds of Southern Africa*, 7th edition. Trustees of the John Voelcker Bird Book Fund, Cape Town, 1296 pp.

Hockey PAR. 1983. Aspects of the breeding biology of the African Black Oystercatcher. *Ostrich* 54: 26–35.

Hodgkiss IJ & KC Ho. 1997. Are changes in N:P ratios in coastal waters the key to increased red tide blooms? In: *Asia-Pacific Conference on Science and Management of Coastal Environment*. Springer, Netherlands, 141–147 pp.

Horton M. 2018 Age, growth and per-recruit assessment of the Saldanha and Langebaan stock of *Chelon richardsonii*. MSc. thesis, Department of Biological Sciences University of Cape Town.

Howarth R, Chan F, Conley DJ, Garnier J, Doney SC, Marino R & G Billen. 2011. Coupled biogeochemical cycles: Eutrophication and hypoxia in temperate estuaries and coastal marine ecosystems. *Frontiers in Ecology and the Environment* 9(1): 18–26.

Hucthings K & BM Clark. 2011. Saldanha Reverse Osmosis Desalination Plant Benthic Macrofauna and Sediment Baseline Survey Report. Report to CSIR and Transnet. Anchor Environmental Consultants Report 391/1, 38 pp.

Hughes JE, Deegan LA, Wyda JC, Weaver MJ & A Wright. 2002. The effect of eelgrass habitat loss on estuarine fish communities of Southern New England. *Estuaries* 25: 235–249.

Hughes RG & OAL Paramor. 2004. On the loss of salt marshes in south-east England and methods for their restoration. *Journal of Applied Ecology* 41: 440–448.

Hughes RN. 1979. South African populations of Littorina rudis. *Zoological Journal of the Linnean Society* 65: 119–126.

Hutchings K & SJ Lamberth. 2002a. Bycatch in the gillnet and beach-seine fisheries in the Western Cape, South Africa, with implications for management. *South African Journal of Marine Science* 24: 227–241.

Hutchings K & SJ Lamberth. 2002b. Catch and effort estimates for the gillnet and beach-seine fisheries in the Western Cape, South Africa. *South African Journal of Marine Science* 24: 205–225.

Hutchings K, Porter S, Clark BM & K Sink. 2011. Strategic Environmental Assessment: Identification of potential marine aquaculture development zones for fin fish cage culture. Report prepared for Directorate Sustainable Aquaculture Management: Aquaculture Animal Health and Environmental Interactions Department of Agriculture, Forestry and Fisheries.

IUCN. 2009. Marine Menace: Alien invasive species in the marine environment. http://wwwvlizbe/imisdocs/publications/153100pdf

Jackson LF & S McGibbon. 1991. Human activities and factors affecting the distribution of macrobenthic fauna in Saldanha Bay. *South African Journal of Aquatic Science* 17(1/2): 89–102.

Jarman ML. 1986. Conservation priorities in lowland regions of the fynbos biome. South African national scientific Programme Report No 87. CSIR, Pretoria, 55 pp.

Jensen A & B Mogensen. 2000. Effects, ecology and economy. Environmental aspects of dredging – Guide No. 6. International Association of Dredging Companies (IADC) and Central Dredging Association (CEDA), 119 pp.

Källén J, Muller H, Franken ML, Crisp A, Stroh C, Pillay D & C Lawrence. 2012. Seagrass-epifauna relationships in a temperate South African estuary: Interplay between patch-size, within-patch location and algal fouling. *Estuarine, Coastal and Shelf Science* 113: 213–220.

Keightley J, von der Heyden S & S Jackson. 2015. Introduced Pacific oysters *Crassostrea gigas* in South Africa: demographic change, genetic diversity and body condition. *African Journal of Marine Science* 37: 89–98.

Kemper J, Underhill LG, Crawford RJM & SP Kirkman. 2007. Revision of the conservation status of seabirds and seals breeding in the Benguela Ecosystem. In: Kirkman SP (ed.) Final Report of the Benguela Current Large Marine Ecosystem (BCLME), 325–342 pp.

Kensley B & ML Penrith. 1970. New recordof *Mytilidae* from the Northern South West African coast. *Annals of the South African Museum* 57(2): 15–24.

Kerry J, Hiney M, Coyne R, NicGabhainn S, Gilroy D, Cazabon D & P Smith. 1995. Fish feed as a source of oxytetracycline-resistant bacteria in the sediments under fish farms. *Aquaculture* 131: 101–113.

Kerwath SE, Thorstad EB, Næsje TF, Attwood CG, Cowley PD, Økland F & C Wilke. 2009. Crossing Invisible Boundaries: The effectiveness of the Langebaan Lagoon marine protected area as a harvest refuge for a migratory fish species in South Africa. *Conservation Biology* 23 (3): 653–661.

Kikuchi T & JM Perez. 1977. Consumer ecology of seagrass beds. In: McRoy CP, Helferich C (eds) *Seagrass Ecosystems: A Scientific Perspective.* New York: Marcel Dekker, 147–186 pp.

Kirchman D, Graham S, Reish D & R Mitchell. 1982. Bacteria induce settlement and metamorphosis of *Janua* (*Dexiospira*) *brasiliensis* Grube (Polychaeta: Spirorbidae*). Journal of Experimental Marine Biology and Ecology* 56: 153–163.

Kljaković-Gašpić Z, Herceg-Romanić S, Kožul D & J Veža. 2010. Biomonitoring of organochlorine compounds and trace metals along the Eastern Adriatic coast (Croatia) using *Mytilus galloprovincialis*. *Marine Pollution Bulletin* 60(10): 1879–1889.

Knight-Jones P, Knight-Jones EW & T Kawahara. 1975. A review of the genus *Janua*, including *Dexiospira* (Polychaeta: Spirorbidae). *Zoological Journal of the Linnean Society* 56: 91–129.

Knowles R. 1982. Denitrification. *Microbiological Reviews* 46(1): 43–70.

Knudsen FR, Schreck CB, Knapp SM, Enger PS & O Sand. 1997. Infrasound produces flight and avoidance response in Pacific juvenile salmonids. *Journal of Fish Biology* 51: 824–829.

Konings A. 2001. Malawi Cichlids in their Natural Habitat. Cichlid Press.

Kostylev VE, Erlandsson J, Ming MY & AW Gray. 2005. The relative importance of habitat complexity and surface area in assessing biodiversity: fractal application on rocky shores. *Ecology Complex* 2: 272–286.

Krug M. 1999. *Circulation through the Mouth of Langebaan Lagoon and Implications* (Doctoral dissertation, University of Cape Town).

Kruger N. 2002. Long term changes in the benthic macrofauna of Saldanha Bay. M.Sc. Thesis, University of Cape Town, 96 pp.

Kruger N, Branch GM, Griffiths CL & JG Field. 2005. Changes in the epibenthos of Saldanha Bay, South Africa, between the 1960s and 2001: An analysis based on dredge samples. *African Journal of Marine Science* 27(2): 471–477.

Krüger HR & N Peinemann. 1996. Coastal plain halophytes and their relation to soil ionic composition. *Vegetation* 122: 143–150.

Kuhne H & G Becker. 1964. Der holz-flohkrebs *Chelura terebrans* Philippi (Amphipoda, Cheluridae), Beihefte Zeitschrift Angewandte Zoologie 1: 1–141.

La Farré M, Pérez S & L Kantiani. 2008. Fate and toxicity of emerging pollutants, their metabolites and transformation products in the aquatic environment. *Trends in Analytical Chemistry* 27(11): 991–1007.

Laird MC & BM Clark. 2016. Marine Impact Assessment for the Construction of a Dedicated Ship and Rig Repair Facility at Berth 205 and a Jetty at the Existing Mossgas Quay Under Project Phakisa, Saldanha Bay. Report no. 1678/1 prepared by Anchor Environmental Consultants for CCA Environmental, 98 pp.

Laird MC & CL Griffiths. 2008. Present distribution and abundance of the introduced barnacle *Balanus glandula* Darwin in South Africa. *African Journal of Marine Science* 30(1): 93–100.

Lanntech 2017. Water Treatment Solutions: Disinfectants Peracetic Acid. Available [Online]: https://www.lenntech.com/processes/disinfection/chemical/disinfectants-peracetic-acid.htm Accessed on 16 October 2017.

Laugksch RC & NJ Adams. 1993. Trends in pelagic fish populations of the Saldanha Bay region, Southern Benguela upwelling system, 1980-1990: A predator's perspective. *South African Journal of Marine Science* 13: 295–307.

Le Roux J. 2002. Avian Demography Unit, Dept of Statistical Sciences, University of Cape Town. http://web.uct.ac.za/depts/stats/adu/species/sp324_00.htm.

Leipe T, Tauber F, Vallius H, Virtasalo J, Uscinowicz S, Kowalski N, Hille S, Lindgren S & T Myllyvirta. 2011. Particulate organic carbon (POC) in surface sediments of the Baltic Sea. *Geo-Marine Letters* 31:175–188.

Lewis JR. 1964. The Ecology of Rocky Shores. English Universities Press, London.

Lewis S, Grémillet D, Daunt F, Ryan PG, Crawford RJM & S Wanless. 2006. Using behavioural variables to identify proximate causes of population change in a seabird. *Oecologia* 147: 606–614.

Lindstrom SC & PW Gabrielson. 1989. Taxonomic and distributional notes on northeast Pacific Antithamnionaceae (Ceramiales: Rhodophyta). *Japanese Journal of Phycology* 37: 221–235.

Loewenthal D. 2007. *The Population Dynamics and Conservation of the African Black Oystercatcher* Haematopus moquini. Thesis submitted for the Degree of Doctor of Philosophy at the Percy FitzPatrick Institute of African Ornithology. Department of Zoology, University of Cape Town. September 2007.

Londoño-Cruz E & M Tokeshi. 2007. Testing scale variance in species-area and abundance-area relationships in a local assemblage: an example from a subtropical boulder shore. *Population Ecology* 49: 275–285.

Long ER, MacDonald DD, Smith SL & FD Calder. 1995. Incidence of adverse biological effects within ranges of concentrations in marine and estuarine sediments. *Environmental Management* 19: 81–97.

Ludynia K, Roux JP, Jones R, Kemper J & LG Underhill. 2010. Surviving off junk: Low-energy prey dominates the diet of African penguins *Spheniscus demersus* at Mercury Island, Namibia, between 1996 and 2009. *African Journal of Marine Science* 32: 563–572.

Ludynia K, Waller LJ, Sherley RB, Abadi F, Galada Y, Geldenhuys D, Crawford RJM, Shannon LJ & A Jarre. 2014. Processes influencing the population dynamics and conservation of African penguins on Dyer Island, South Africa. *African Journal of Marine Science*, 36(2): 253–267.

Luger S, Van Ballegooyen R & PMS Monteiro. 1999. A Modelling-driven Environmental Baseline Assessment for the Proposed New Hood Point Outfall in East London: Hydrodynamic and Water Quality Modelling Study. CSIR Report ENV-S-C 99109B. CSIR Environmentek, Stellenbosch, South Africa.

Mabin CA, Wilson JRU & TB Robinson. 2015. The Chilean black urchin, *Tetrapygus niger* (Molina, 1782) in South Africa: gone but not forgotten. *BioInvasions Records* 4: 261–264.

Mackie JA, Keough MJ, Norman JA & L Christidis. 2001. *Mitochondrial evidence of geographical isolation within Bugula dentata Lamouroux.* In: Wyse-Jackson PN, Buttler CJ & Spencer JME (eds.). Bryozoan Studies, Proceedings of 12th International Bryozoology Association Conference, Balkema, Lisse, Netherlands: 199–206 pp.

Madsen FJ. 1956. Reports of the Lund University Chile Expedition 1948–49. No. 24. Asteroidea, with a survey of the Asteroidea of the Chilean shelf. Lund Universitets Årsskrift, 53 pp.

Maggs JQ & BQ Mann. 2013. Elf (*Pomatomus saltatrix*) In: Mann BQ (Ed). Sothern African marine Linefish Species Profiles. Special publication, Oceanographic Research institute Durban: 143–144.

Maggs CA & MH Hommersand. 1993. *Seaweeds of the British Isles*. Volume 1: Rhodophyta. Part 3A: Ceramiales. London, HMSO, for Natural History Museum.

Makhado AB, Crawford RJM & LG Underhill. 2006. Impact of predation by Cape fur seals *Arctocephalus pusillus pusillus* on Cape Gannets *Morus capensis* at Malgas Island, Western Cape, South Africa. *African Journal of Marine Science* 28: 681–687.

Makhado AB, Dyer BM, Fox R, Geldenhuys D, Pichegru L, Randall RM, Sherley RB, Upfold L, Visagie J, Waller LJ, Whittington PA & RJM Crawford. 2014. Estimates of numbers of twelve seabird species breeding in South Africa. Branch Oceans and Coasts, Department of Environmental Affairs Cape Town, South Africa, 15 pp.

Makhado AB, Meÿer MA, Crawford RJM, Underhill LG & C Wilke. 2009. The efficacy of culling seals seen preying on seabirds as a means of reducing seabird mortality. *African Journal of Ecology* 47: 335–340.

Malinowski KC & Ramus J. 1973. Growth of the green alga *Codium fragile* in a Connecticut estuary. *Journal of Phycology* 9: 102–110.

Maneveldt GW, Eager RC & A Bassier. 2009. Effects of long-term exclusion of the limpet *Cymbula oculus* (Born) on the distribution of intertidal organisms on a rocky shore. *African Journal of Marine Science* 31: 171–179.

Mann BQ & S Dunlop. 2013. Blacktail *Diplodus capensis.* In: Mann BQ (ed). Southern African Marine Linefish Species Profiles. Special Publication, Oceanographic Research Institute, Durban 9, 230–232 pp.

Manuel RL. 1981. *British Anthozoa*. Academic Press, London, 241 pp.

Marangoni C. 1998. The potential for the introduction of alien phytoplankton by shipping ballast water: Observations in Saldanha Bay. M.Sc. Thesis for the Faculty of Science, University of the Witwatersrand, Johanesburg.

Martin JM & M Whitfield. 1983. The Significance of the River Input of Chemical Elements to the Ocean. *Trace Metals in Sea Water:* 265–296.

Mather AA, Stretch DD & G Garland. 2010. A simple empirical model for extreme wave run-up on natural beaches. International conference on Coastal Engineering. Shanghai, China.

McBride G & G Payne. 2009. The Hazen Percentile Calculator. National Institute of Water and Atmospheric Research (NIWA), Hamilton, New Zealand.

McClarty A, Cross J, James GM & L Gilbert. 2006. Design and construction of coastal erosion protection groynes using geocontainers, Langebaan, South Africa. Geosynthetics. In: Proceedingsof the 8th International Conference on Geosynthetics, Yokohama, Japan, Volume 2, 765–768 pp.

McClintock JB, Angus RA & FE McClintock. 2007. Abundance, diversity and fidelity of macroinvertebrates sheltering beneath rocks during tidal emersion in an intertidal cobble field: Does the intermediate disturbance hypothesis hold for less exposed shores with smaller rocks? *Journal of Experimental Marine Biology and Ecology* 352: 351–360.

McCoy ED & SS Bell. 1991. Habitat Structure: The Evolution and Diversification of a Complex Topic. In: Bell SS, McCoy ED & Mushinsky HR (eds), Habitat structure and the physical arrangement of objects in space. Chapman & Hall, New York, 3-27 pp.

McGuinness KA. 1984. Species-area relations of communities on intertidal boulders: testing the null hypothesis. *Journal of Biogeography* 11: 439–456.

McGuinness KA & AJ Underwood. 1986. Habitat structure and the nature of communities on intertidal boulders. *Journal of Experimental Marine Biology & Ecology* 104: 97–123.

McGuinness KA. 1987. Disturbance and organisms on boulders II. Causes of patterns in diversity and abundance. *Oecologia* 71: 420–430.

McKibben JR & AH Bass. 1998. Behavioral assessment of acoustic parameters relevant to signal recognition and preference in a vocal fish. *Journal of Acoustical Society of America* 104: 3520–3533.

McKinnon JG, Gribben PE, Davis AR, Jolley DF & JT Wright. 2009. Differences in soft-sediment macrobenthic assemblages invaded by *Caulerpa taxifolia* compared to uninvaded habitats, *Marine Ecology Progress Series* 380: 59–71.

McQuaid CD & GM Branch. 1984. Influence of sea temperature, substratum and wave exposure on rocky intertidal communities: an analysis of faunal and floral biomass. *Marine Ecology Progress Series* 19: 145–151.

McQuaid CD & GM Branch. 1985. Trophic structure of rocky intertidal communities: response to wave action and implications for energy flow. *Marine Ecology Progress Series* 22: 153–161.

McQuaid CD, Branch GM & AA Crowe. 1985. Biotic and abiotic influences on rocky intertidal biomass and richness in the southern Benguela region. *South African Journal of Zoology* 20: 115–122.

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

Mead A, Carlton JT, Griffiths CL & M Rius. 2011a. Introduced and cryptogenic marine and estuarine species of South Africa. *Journal of Natural History* 45: 2463–2524.

Mead A, Carlton JT, Griffiths CL & M Rius. 2011b. Revealing the scale of marine bioinvasions in developing regions: a South African re-assessment. *Biological Invasions* 13: 1991–2008.

Mead A, Griffiths CL, Branch GM, McQuaid CD, Blamey LK, Bolton JJ, Anderson RJ, Dufois F, Rouault M, Froneman PW, Whitfield AK, Harris LR, Nel R, Pillay D & JB Adams. 2013. Human-mediated drivers of change — impacts on coastal ecosystems and marine biota of South Africa. *African Journal of Marine Science* 35(3): 403–425.

Meager JJ, Schlacher TA & M Green. 2011. Topographic complexity and landscape temperature patterns create a dynamic habitat structure on a rocky intertidal shore. *Marine Ecology Progress Series* 428: 1–12.

Membrane Technology. 2013. Transnet uses RO desalination plant from Veolia Water. Membrane Technology News, April 2013 (4): 6–7 pp.

Menge BA & GM Branch. 2001. Rocky Intertidal Communities. In: Burtness MD, Gaines S & Hay ME (eds.) Marine Community Ecology. Sinauer Associates, Sunderland, 221–251 pp.

Menge BA, Daley BA, Wheeler PA, Dahlhoff E, Sanford E & PT Strub. 1997. Benthic-pelagic links and rocky intertidal communities: Bottom-up effects on top-down control? *Proceedings of the National Academy of Sciences USA* 94: 14530–14535.

Millard N. 1952. Observations and experiments on fouling organisms in Table Bay Harbour, South Africa. *Transactions of the Royal Society of South Africa* 33(4): 415–445.

Millard NAH. 1975. Monographs on the Hydroida of southern Africa. *Annals of the South African Museum* 68: 1–513.

Moldan A. 1978. A Study of the Effects of dredging on the benthic macrofauna in Saldanha Bay. *South African Journal of Science* 74: 106–108.

Monniot C, Monniot F, Griffiths CL & M Schleyer. 2001. South African ascidians. *Annals of the South African Museum* 108: 1–141.

Monteiro PMS & GB Brundrit. 1990. Interannual chlorophyll variability in South Africa's Saldanha Bay system, 1974-1979. *South African Journal of Marine Science* 9: 281–288.

Monteiro PMS & JL Largier. 1999. Thermal stratification in Saldanha Bay (South Africa) and subtidal, density-driven exchange with the coastal waters of the Benguela Upwelling System. *Estuarine and Coastal Shelf Science* 49: 877–890.

Monteiro PMS, Anderson RJ & S Woodbourne. 1997. ^{15}N as a tool to demonstrate the contribution of fish-waste-derived nitrogen to an *Ulva* bloom in Saldanha Bay, South Africa. *South African Journal of marine Science* 18: 1–10.

Monteiro PMS, Mcgibbon S & JL Henry. 1990. A decade of change in Saldanha Bay: natural or anthropogenic? *South African Journal of Science* 86: 454–456.

Monteiro PMS, Pascall A & S Brown. 1999. The Biogeochemical Status of Near-Surface Sediments in Saldanha Bay in 1999. CSIR Report ENV-S-C 99093A.

Monteiro PMS, Warwick PA, Pascall A & M Franck. 2000. Saldanha Bay Water Quality Monitoring Programme. Parts 1 & 2. CSIR.

Morales C & T Antezana. 1983. Diet selection of the Chilean stone crab *Homalaspis plana*. *Marine Biology* 77: 79–83.

Morino H. 1976. On two forms of *Cerapus tubularis*, a tube dwelling Amphipoda, from shallow waters of Japan. *Publications Of The Seto Marine Biological Laboratory* 23(1-2): 179–189.

Mouton A, Crosbie P, Cadoret K & B Nowak. 2014. First record of amoebic gill disease caused by *Neoparamoeba perurans* in South Africa. *Journal of Fish Disease* 37: 407–409.

Munday BL, Zilberg D & V Findlay. 2001. Gill disease of marine fish caused by infection with *Neoparamoeba pemaquidensis*. *Journal of Fish Disease* 24: 497–507.

Navarrete SA & T Manzur. 2008. Individual- and population-level responses of a keystone predator to geographic variation in prey. *Ecology* 89: 2005–2018.

Navarro N. 2000. Planktonic ecosystem impacts of salmon cage aquaculture in a Scottish sea loch. ICES Cooperative Research Report 240, Denmark. [Online]: http://www.ices.dk/sites/pub/Publication%20Reports/Cooperative%20Research%20Report%20(CRR)/crr263/CRR263.pdf (accessed 2018-11-07).

Næsje TJ, Attwood CG, Kerwath S, Cowley PD, Keulder F & C Arendse. 2008. Patterns and volumes of commercial and recreational harvest of white stumpnose in Saldanha Bay: an assessment of the fishery. Oral presentation at the South African Marine Science Symposium; 29-June-3rd July 2008, University of Cape Town, Cape Town.

Nel R, Coetzee PS & G Van Niekerk. 1996. The evaluation of two treatments to reduce mud worm (*Polydora hoplura* Claparede) infestation in commercially reared oysters (*Crassostrea gigas* Thunberg). Aquaculture 141: 31–9.

NERSA. 2010. Application for a License to Construct a Loading and Storage Facility for Liquefied Petroleum Gas (LPG). Public Copy after NERSA Hearing 8 Feb 2010.

Newell RC, Seiderer LJ & DR Hitchcock. 1998. The impact of dredging works in coastal waters: A review of the sensitivity to disturbance and subsequent recovery of biological resources on the sea bed. *Oceanography and Marine Biology: an Annual Review* 36: 127–78.

Newman B. 2015. CSIR. 2015. Polycyclic aromatic hydrocarbon and butyltin concentrations in sediment in Saldanha Bay, South Africa: Community Report. CSIR Report CSIR/NRE/ECOS/IR/2015/00XX/B.

Newman WA. 1982. A review of the extant taxa of the "Group of *Balanus concavus*" (Cirripedia, Thoracica) and a proposal for genus-group ranks. *Crustaceana* 43: 25–36.

Nikolaou A, Kostopoulou M, Lofrano G, Meric S, Petsas A & M Vagi. 2009. Levels and toxicity of polycyclic aromatic hydrocarbons in marine sediments. *Trends in Analytical Chemistry* 28: 653–664.

Nishihara GN & R Terada. 2010. Species richness of marine macrophytes is correlated to a wave exposure gradient. *Phycological Research* 58: 280–292.

Noe GB & J Zedler. 2001. Spatio-temporal variation of salt marsh seedling establishment in relation to the abiotic and biotic environment. *Journal of Vegetation Science* 12: 61– 74.

Nondoda SP. 2012. Macrophyte distribution and responses to drought in the St. Lucia Estuary MSc. thesis Nelson Mandela Metropolitan University (NMMU), South Africa.

Norwegian Scientific Committee for Food Safety (VKM). 2014. Panel on Animal Health and Welfare; Risk assessment of amoebic gill disease, VKM Report 2014: 11 [39 pp], ISBN nr. 978-82-8259-149-2, Oslo, Norway. Available online: www.vkm.no

NSOVO Environmental Consulting. 2017. Notive of Basic Assessment and Water Use License Application Processes for the Proposed Upgrade of Storm Water and Environmental Systems in the Port of Saldanha within the Jurisdiction of Saldanha Bay Local Municipality in Western Cape Province. Background Information Document, September 2017.

Nybakken JW. 2001. *Marine Biology: An Ecological Approach*. Fifth Ed. ed. Addison Wesley Longman, Inc. San Francisco, 579 pp.

Occhipinti Ambrogi A & JL D'Hondt. 1981. *Distribution of bryozoans in brackish waters of Italy*. In: Larwood, Gilbert P & Nielsen C (eds.). Recent and Fossil Bryozoa. Olsen and Olsen. Fredensborg.

Occhipinti-Ambrogi A & B Galil. 2010. Marine alien species as an aspect of global change. *Advances in Oceanography and Limnology* 1: 143–156.

Occhipinti-Ambrogi A. 2007. Global change and marine communities: alien species and climate change. *Marine Pollution Bulletin* 55: 342–352.

Ocean Biogeographic Information System (ORBIS) 2011. www.iobis.org. Accessed 28 July 2011.

O'Donnell MJ & MW Denny. 2008. Hydrodynamic forces and surface topography: centimeter-scale spatial variation in wave forces. *Limnology and Oceanography* 53: 579–588.

O'Donoghue CH & D de Waterville. 1935. A collection of Bryozoa from South African. *Journal of the Linnean Society* (London) 39: 203–218.

Okes NC, Hockey PA, Pichegru L, van der Lingen CD, Crawford RJM & D Grémillet. 2009. Competition for shifting resources in the southern Benguela upwelling: seabirds versus purse-seine fisheries. *Biological* Conservation, 142: 2361–2368.

Olivier D, Heinecken L & S Jackson. 2013. Mussel and oyster culture in Saldanha Bay, South Africa: Potential for sustainable growth, development and employment creation. *Food Security* 5: 251–267.

Orth RJ & KA Moore. 1983. Chesapeake Bay: An unprecedented decline in submerged aquatic vegetation. *Science* 222(4619): 51–53.

Orth RJ, Carruthers TJB, Dennison WC, Duarte CM, Fourqurean JW, Heck KL, Hughes AR, Kendrick GA, Kenworthy WJ, Olyarnik S, Short FT, Waycott M & SL Williams. 2006. A global crisis for seagrass ecosystems. *BioScience* 56: 987–996.

OSPAR. 2004. Biodiversity Series: Environmental impacts to marine species and habitats of dredging for navigational purposes. ISBN 1-904426-50-6, 22 pp.

OSPAR. 2010. Quality Status Report 2010: Status and Trend of marine chemical pollution. Accessed at OSPAR.org on 13 June 2012.

Page HM, Dugan JE, Culver CS & JC Hoesterey. 2006. Exotic invertebrate species on offshore oil platforms. *Marine Ecology Progress Series* 325: 101–107.

Paine RT, Castilla JC & J Cancino. 1985. Perturbation and recovery patterns of starfish-dominated intertidal assemblages in Chile, New Zealand, and Washington State. *American Naturalist* 125: 679–691.

Pan D, Bouchard A, Legendre P & G Domon. 1998. Influence of edaphic factors on the spatial structure of inland halophytic communities: A case study in China. *Journal of Vegetation Science* 9: 797–804.

Parker D, Kerwath S, Naesje T, Hutchings K, Clark B, Winker H, & C Attwood. 2017. When plenty is not enough: An assessment of the white stumpnose fishery of Saldanha Bay. *African Journal of Marine Science* 39:153–166.

Parsons NJ, GousIII TJ, Schaefer AM & RET Vanstreels. 2016. Health evaluation of African penguins (*Spheniscus demersus*) in southern Africa. *Onderstepoort Journal of Veterinary Research*, 83(1): 1–13.

Pavlov DF, Bezuidenhout J, Frontasyeva MV & ZI Goryainova. 2015. Differences in trace element content between non-indigenous farmed and invasive bivalve mollusks of the South African coast. *American Journal of Analytical Chemistry* 6: 1–12.

Persson LE. 1999. Growth and reproduction in two brackish water populations of *Orchestia gammarellus* (Amphipoda: Talitridae) in the Baltic Sea. *Journal of Crustacean Biology* 19: 53–59.

Peters K & TB Robinson. 2017. First record of the marine alien amphipod *Caprella mutica* (Schurin, 1935) in South Africa. *BioInvasions Records* 6(1): 61–66.

Peters K & TB Robinson. 2018. From Chile to the South African west coast: first reports of the Chilean stone crab *Homalaspis plana* (H. Milne Edwards, 1834) and the South American sunstar *Heliaster helianthus* (Lamarck, 1816) outside their natural ranges. *Bioinvasions Records* 7(4): 421–426.

Peters K, Griffiths C & TB Robinson. 2014. Patterns and drivers of marine bioinvasions in eight Western Cape harbours, South Africa. *African Journal of Marine Science* 36(1): 49–57.

Pfaff MC, Branch GM, Wieters EA, Branch RA & BR Broitman. 2011. Upwelling intensity and wave exposure determine recruitment of intertidal mussels and barnacles in the southern Benguela upwelling region. *Marine Ecology Progress Series* 425: 141–152.

Phillips DJH & PS Rainbow. 1993. Biomonitoring of Trace Aquatic Contaminants. Applied Science Publishers, Barking.

Phillips DJH. 1980. Quantitative Aquatic Biological Indicators: Their Use to Monitor Trace Metal and Organochlorine Pollution. London: Applied Science Publishers.

Phillips DJH. 1995. The chemistries and environmental fates of trace metals and organochlorines in aquatic ecosystems. *Marine Pollution Bulletin* 31(4-12): 193–200.

Pichegru L, Grémillet D, Ryan PG & RJM Crawford. 2010. Marine no-take zone rapidly benefits endangered penguin. *Biology Letters* 6: 498–501.

Pichegru L, Ryan PG, Le Bohec C, van der Lingen CD, Navarro R, Petersen S, Lewis S, van der Westhuizen J & D Grémillet. 2009. Overlap between vulnerable top predators and fisheries in the Benguela upwelling system: Implications for Marine Protected Areas. *Marine Ecology Progress Series* 391: 199–208.

Pichegru L. 2012. Increasing breeding success of an endangered penguin: Artificial nests or culling predatory gulls? Bird Conservation International, 2013. Cambridge University Press.

Pichegru L., Ryan PG, van der Lingen CD, Coetzee J, Ropert-Coudert Y & D Grémillet. 2007. Foraging behaviour and energetics of Cape gannets *Morus capensis* feeding on live prey and fishery discards in the Benguela upwelling system. *Marine Ecology Progress Series* 350: 127–136.

Picker M & CL Griffiths. 2011. Alien and Invasive Animals: A South African Perspective. Random House Struik, Cape Town, 240 pp.

Pietersen K. 2006 Multiple criteria decision analysis (MCDA): A tool to support sustainable management of groundwater resources in South Africa. *Water SA* 32(2): 119–128.

Pillay D, Branch GM, Dawson J & D Henry. 2011. Contrasting effects of ecosystem engineering by the cordgrass *Spartina maritime* and the sandprawn *Callianassa kraussi* in a marine-dominated lagoon. *Estuarine, Coastal and Shelf Science* 91: 169–176.

Pillay D, Branch GM, Griffiths CL, Williams C & A Prinsloo. 2010. Ecosystem change in South African marine reserve 1960-2009: Role of seagrass loss and anthropogenic disturbance. *Marine Ecology Progress Series* 415: 35–48.

Pilsbry HA. 1916. The sessile barnacles (Cirripedia) contained in the collections of the U.S. National Museum; including a Monograph of the American species. *Bulletin of the United States National Museum, Washington* 93: 1–366.

Pinedo S, Garcia M, Satta MP, de Torres M & E Ballesteros. 2007. Rocky-shore communities as indicators of water quality: a case study in the Northwestern Mediterranean. *Marine Pollution Bulletin* 55: 126–135.

Pitombo FB. 2004. Phylogenetic analysis of the Balanidae (Cirripedia, Balanomorpha). *Zoologica Scripta* 33(3): 261–276.

Pitta P, Apostolakia ET, Giannoulakia M & I Karakassis. 2005. Mesoscale changes in the water column in response to fish farming zones in three coastal areas in the Eastern Mediterranean Sea. *Estuarine, Coastal and Shelf Science* 65(3): 501–512.

Poggenpoel CE. 1996. The exploitation of fish during the Holocene in the south-western Cape, South Africa. M. A. Thesis, University of Cape Town: xxii+225pp.

Poluzzi A & B Sabelli. 1985. Polymorphic zooids in deltaic species populations of *Conopeum seurati* (Canu, 1928) (Bryozoa, Cheilostomata). *Marine Ecology* 6: 265–284.

Pope HR, Alexander ME & Robinson TB. 2016. Filtration, feeding behaviour and their implications for future spread: A comparison of an invasive and native barnacle in South Africa. *Journal of Experimental Marine Biology and Ecology*, 479: 54–59.

Popper AN & MC Hastings. 2009. The effects of anthropogenic sources of sound on fishes. *Journal of Fish Biology* 75: 455-489.

Ports Regulator of South Africa. 2015. Cargo Traffic Volumes Data Monitoring Report 2014/2015, Sample dates 2010/11 –2013/14. Avilable Online: http://www.portsregulator.org/images/documents/Port-Level-Cargo-Data-Audit-Report-2014-15.pdf

PRDW. 2012. Prestedge Retief Dresner Wijnberg (Pty) Ltd; General Maintenance Quay Upgrade. Report No. 1113/DST/00.

PRDW. 2017. Initial dispersion modelling for proposed finfish farming in Saldanha Bay — specialist marine modelling study. Specialist study S2001-78-RP-CE-001-R3, prepared for the Department: Agriculture, Forestry and Fisheries (DAFF) by PRDW Consulting & Port Engineers & Lwandle Technologies. 177 pp.

Price JS, Ewing K, Woo M-K & KA Kershaw. 1988. Vegetation patterns in James Bay coastal marshes. II. Effects of hydrology on salinity and vegetation. *Canadian Journal of Botany* 66: 2586–2594.

Price JS, Ewing K, Woo MK & KA Kershaw. 1988. Vegetation patterns in James Bay coastal marshes. II. Effects of hydrology on salinity and vegetation. *Canadian Journal of Botany* 66, 2586–2594.

Pullen TA. 2008. Eurotop – overtopping and methods for assessing discharge.

Puce S, Bavestrello G, Azzini F & C Cerrano. 2003. On the occurrence of *Coryneeximia* Allman (Cnidaria, Corynidae) in the Mediterranean sea. *Italian Journal of Zoology* 70(3): 249–252.

Purser J & AN Radford. 2011. Acoustic noise induces attention shifts and reduces foraging performance in three-spine sticklebacks (*Gasterosteus aculeatus*). *PLoS ONE* 6(2): e17478.

Raffaelli D & SJ Hawkins. 1996. Intertidal Ecology. London: Chapman and Hall, 356p.

Rainbow PS. 1995. Biomonitoring of heavy metal availability in the marine environment. *Marine Pollution Bulletin* 31: 183–192.

Rainbow PS. 2002. Trace metal concentrations in aquatic invertebrates: Why and so what? *Environmental Pollution* 120(3): 497–507.

Ramus J. 1971. Codium: the invader. Discovery 6: 59–68.

Read G & C Glasby. 2017. *Neanthes succinea* (Leuckart, 1847). In: Read G; Fauchald K (ed.) 2017, World Polychaeta database. Accessed through: World Register of Marine Species at http://www.marinespecies.org/aphia.php?p=taxdetails&id=130391 [accessed 30 September 2017].

Readman J, Bartocci J, Tolosa I, Fowler S, Oregioni B & M Abadulraheem. 1996. Recovery of the coastal marine environment in the Gulf following the 1991 war-related oil spills, *Marine Pollution Bulletin* 32(6): 493.

Republic of South Africa, Department of Environmental Affairs. 2018. South African Water Quality Guidelines for Coastal Marine Waters - Natural Environment and Mariculture Use. Cape Town.

Richardson DM, Pyšek P & JT Carlton. 2011. A compendium of essential concepts and terminology in invasion ecology. In: Richardson DM (ed.) Fifty years of invasion ecology. The legacy of Charles Elton. Oxford: Wiley-Blackwell, 409–420 pp.

Richardson SD, Plewa MJ, Wagner ED, Schoeny R & DM DeMarini. 2007. Occurrence, genotoxicity and carcinogenicity of regulated and emerging disinfection by-products in drinking water: A review and roadmap for research. *Mutation Research* 636(1-3): 178–242.

Rius M, Clusella-Trullas S, McQuaid CD, Navarro RA, Griffiths CL & CA Matthee *et al*. 2014. Range expansions across ecoregions: interactions of climate change, physiology and genetic diversity. *Global Ecology and Biogeography* 23: 76–88.

Roberts DL & HP Siegfried. 2014. The Geology of the Saldanha, Vredenburg and Velddrif Environs; Geological Explanation – Sheets 3317BB & 3318AA, 3217DB and 3218CA &CC Scale 1:50 000, Council of Geoscience, Pretoria.

Robertson HG. 1981. Annual summer and winter fluctuations of Palearctic and resident waders (Charadrii) at Langebaan Lagoon, South Africa, 1975-1979. In: Cooper J (ed.) *Proceedings of the Symposium on Birds of the Sea and Shore*, 1979. Cape Town: African Seabird Group, 335–345 pp.

Robinson TB & CL Griffiths. 2002. Invasion of Langebaan Lagoon, South Africa, by *Mytilus galloprovincialis*—effects on natural communities. *African Zoology* 37: 151–158.

Robinson TB & Swart C. 2015. Distribution and impact of the alien anemone *Sagartia ornata* in the West Coast National Park. *Koedoe* 57(1): 8 http://dx.doi.org/10.4102/koedoe.v57i1.1246

Robinson TB, Alexander ME, Simon CA, Griffiths CL, Peters K, Sibanda S, Miza S, Groenewald B, Majiedt P & KJ Sink. 2016. Lost in translation? Standardising the terminology used in marine invasion biology and updating South African alien species lists. *African Journal of Marine Science* 38(1): 129–140.

Robinson TB, Branch GM, Griffiths CL, Govender A & PAR Hockey. 2007b. Changes in South African rocky intertidal invertebrate community structure associated with the invasion of the mussel *Mytilus galloprovincialis*. *Marine Ecology Progress Series* 340: 163–171.

Robinson TB, Griffiths C, McQuaid C & M Rius. 2005. Marine alien species of South Africa - status and impacts. *African Journal of Marine Science* 27: 297–306.

Robinson TB, Griffiths CL & N Kruger. 2004. Distribution and status of marine invasive species in and bordering the West Coast National Park. *Koedoe* 47: 79–87.

Robinson TB, Griffiths CL, Branch GM & A Govender. 2007a. The invasion and subsequent die-off of *Mytilus galloprovincialis* in Langebaan Lagoon, South Africa: effects on natural communities. *Marine Biology* 152: 225–232.

Robinson TB, Pope HR, Hawken L & Binneman C. 2015. Predation driven biotic resistance fails to restrict the spread of a sessile rocky shore invader. Marine Ecology Progress Series 522: 169–179.

Robinson TB. 2010. Marine Ecology Report. Prepared for the Department of Public Works as part of the New Boat Park at the Langebaan 4 Special Forces Base 24 G Process and Public Participation Report (JW de Jager, Arcus GIBB Pty Ltd).

Robles C. 1982. Disturbance and predation in an assemblage of herbivorous *Diptera* and algae on rocky shores. *Oceologia* 54: 23–31.

Rogel JA, Silla RO & Ariza FA. 2001. Edaphic characterization and sediment ionic composition influencing plant zonation in a semiarid Mediterranean salt marsh. *Geoderma* 99 81–98.

Roughgarden J, Gaines S & Possingham H. 1988. Recruitment dynamics in complex life cycles. *Science* 241: 1460–1466.

Rouse S. 2011."*Conopeum seurati*". Bryozoa of the British Isles. Accessed 16/9/2019.

Ruiz JM. 2001. Effects of fish farm loadings on seagrass (*Posidonia oceanica*) distribution, growth and photosynthesis. Marine Pollution Bulletin 42(9): 749–760.

Ruiz GM, Carlton JT, Grosholz ED & Hines AN. 1997. Global invasion of marine and estuarine habitats by non-indigenous species: Mechanisms, extent, and consequences. *American Zoology* 37: 621–632.

Ruiz GM, Fofonoff P & Hines AH. 1999. Non-indigenous species as stressors in estuarine and marine communities: assessing invasion impacts and interactions. *Limnology and Oceanography* 44: 950–972.

Ryan PG. 2012. Medium-term changes in coastal bird communities in the Western Cape, South Africa. *Austral Ecology* 38: 251–259.

Ryland JS, De Blauwe H, Lord R & JA Mackie. 2009. Recent discoveries of alien *Watersipora* (Bryozoa) in Western Europe, with redescriptions of species. *Zootaxa* 2093: 43–59.

Sadchatheeswaran S, Branch GM & TB Robinson. 2015. Changes in habitat complexity resulting from sequential invasions of a rocky shore: implications for community structure. *Biological Invasions* 17(6): 1799–1816.

Sadchatheeswaran S, Branch GM, Moloney CL & Robinson TB. 2018. Impacts of alien 'ecosystem engineers' overwhelm interannual and seasonal shifts in rocky-shore community composition on Marcus Island, South Africa. African Journal of Marine Science, 40(2): 137–147.

Saldanha Bay Municipality (SBM). 2016. Saldanha Bay Municipality Integrated Development Plan 2012-2017. Revision 3, 2015/2016. Available [Online] http://www.sbm.gov.za/pages/IDP/IDP_Review_2015_2016.pdf [10 September 2016].

Saldanha Bay Municipality (SBM). 2011. *Integrated Development Plan 2006–2011*. Saldanha Bay Municipality, South Africa.

Saldanha Bay Municipality (SBM). 2014. Langebaan Stormwater Master Plan: Basic Assessment Report. Saldanha Bay Municipality, Saldanha.

Saldanha Bay Local Municipality (SBLM). 2019. Second Generation Coastal Management Programme 2019-2024.

Saldanha Bay Water Quality Forum Trust (SBWQFT) 2019a. Beach Erosion/Accretion Monitoring Program. Interim Report 2018. May 2019.

Saldanha Bay Water Quality Forum Trust (SBWQFT) 2019b. Beach Erosion/Accretion Monitoring Program. First Progress Report 2018. May 2019.

Samaai T & MJ Gibbons. 2005. Demospongiae taxonomy and biodiversity of the Benguela region on the west coast of South Africa. *African Natural History* 1: 1–96.

Sandison EE. 1950. Appearance of Elminius modestus Darwin in South Africa. *Nature* 165: 79–80.

Schaffelke B & D Deane. 2005. Desiccation tolerance of the introduced marine green alga *Codium fragile ssp. tomentosoides* – clues for likely transport vectors? *Biological Invasions* 7: 557–565.

Scheibling RE & SX Anthony. 2001. Feeding, growth and reproduction of sea urchins (*Strongylocentrotus droebachiensis*) on single and mixed diets of kelp (*Laminaria spp*.) and the invasive alga *Codium fragile ssp. tomentosoides*. *Marine Biology* 139: 139–146.

Scheibling RE & P Gagnon. 2006. Competitive interactions between the invasive green alga *Codium fragile ssp. tomentosoides* and native canopy-forming seaweeds in Nova Scotia (Canada). *Marine Ecological Progress* Series 325: 1–14.

Schils T, Clerck OD, Leliaert F, Bolton JJ & E Coppejans. 2001. The change in macroalgal assemblages through the Saldanha Bay/Langebaan Lagoon ecosystem (South Africa). *Botanica Marina* 44: 295–305.

Schuchert P. 2001 - Survey of the family Corynidae (Cnidaria, Hydrozoa). *Revue Suisse De Zoologie* 108 (4): 739–878.

Schuchert P. 2005. Species boundaries in the hydrozoan genus Coryne. *Molecular Phylogenetics and Evolution* 36: 194–199.

Schuchert P. 2017. *Moerisia maeotica* (Ostroumoff, 1896). In: Schuchert P (2015), World Hydrozoa database. Accessed through: World Register of Marine Species at http://www.marinespecies.org/aphia.php?p=taxdetails&id=247957 [accessed 30 September 2017].

Sedlak DL & U von Gunten. 2011. The Chlorine Dilemma. *Science* 331(6013): 42–43.

Serrano O, Lavert PS, Duarte CM, Kendrick GA, Calafat A, York PH, Steve A & PI Macreadie. 2016. Can mud (silt and clay) concentration be used to predict soil organic carbon content within seagrass ecosystems? *Biogeosciences* 13: 4915–4926.

Seyler H, Witthuser K & M Holland. 2016. The Capture Principle approach to Sustainable Groundwater Use Incorporating Sustainability Indictors and Decision Framework for Sustainable Groundwater Use. WRC Reviewed Draft Report prepared by Delts H Water Systems Modelling (Pty) Ltd. 1–172.

Shannon LV & GH Stander. 1977. Physical and chemical characteristics of water in Saldanha Bay and Langebaan Lagoon. *Transactions of the Royal Society of South Africa* 42: 441–459.

Shannon LV. 1966. Hydrology of the South and West Coasts of South Africa. Investigational Report, Sea Fisheries Research Institute, South Africa 58: 1–52.

Siebert T & GM Branch. 2005. Interactions between *Zostera capensis*, *Callianassa kraussi* and *Upogebia africana*: deductions from field surveys in Langebaan Lagoon, South Africa. *South African Journal of Marine Science* 27: 357–373.

Siebert T & Branch GM. 2006. Ecosystem engineers: Interactions between eelgrass *Zostera capensis* and the sandprawn *Calianassa kraussi* and their indirect effects on the mudprawn *Upogebia africana*. *Journal of Experimental Marine Biology and Ecology*. 338(2): 253–270.

Siebert T & GM Branch. 2007. Influences of biological interactions on community structure within seagrass beds and sandprawn-dominated sandflats. *Journal of Experimental Marine Biology and Ecology*. 340: 11–24.

Siegfried WR. 1977. Wading Bird Studies at Langebaan Lagoon. Interim Report submitted March 1977.

Simon CA. 2011. *Polydora* and *Dipolydora* (Polychaeta: Spionidae) associated with molluscs on the South Coast of South Africa, with descriptions of two new species. *African Invertebrates* 52: 39–50.

Simon CA. 2015. Observations on the composition and larval developmental modes of polydorid pests of farmed oysters (*Crassostrea gigas*) and abalone (*Haliotis midae*) in South Africa. *Invertebrate Reproduction & Development* 59: 124–130.

Simon CA, Ludford A & S Wynne. 2006. Spionid polychaetes infesting cultured abalone *Haliotis midae* in South Africa. *African Journal of Marine Science* 28(1): 167–171.

Simon CA, Thornhill DJ, Oyarzun F & KM Halanych. 2009. Genetic similarity between *Boccardia proboscidae* from Western North America and cultured abalone, *Haliotis midae*, in South Africa. *Aquaculture* 294: 18–24.

Simon CA & SE Van Niekerk. 2012. *Polydora hoplura*. A guide to the shell-infesting spionids of South Africa. (http://www0sun.ac.za/polychaete/) Accessed on 20/9/2019.

Simon-Blecher N, Granevitze Z & Y Achituv. 2008. *Balanus glandula*: from North-West America to the west coast of South Africa. *African Journal of Marine Science* 30(1): 85–92.

Simons RH. 1977. The algal flora of Saldanha Bay. *Transactions of the Royal Society South Africa* 42: 461-482.

Skein L, Alexander M & TB Robinson. 2018. Contrasting invasion patterns in intertidal and subtidal mussel communities. *African Zoology* 53(1):47–52.

Skein L, Robinson TB & ME Alexander. 2017. Impacts of mussel invasions on the prey preference of two native predators. *Behavioral Ecology* 29(2): 353–359.

Skinner LF & R Coutinho. 2005. Effect of microhabitat distribution and substrate roughness on barnacle *Tetraclita stalactifera* (Lamarck 1818) settlement. *Brazilian Archives of Biology & Technology* 48: 109–113.

Slabberkorn H, Bouton N, Opzeeland I, Aukje C, ten Cate Carel & N Popper. Article in Press. A noisy spring: The impact of globally rising underwater sound levels on fish. *Trends in Ecology and Evolution* 1243: 9.

SLR Consulting (Pty) Ltd. (SLR) 2016. Proposed oil and gas offshore service complex at the Saldanha Bay Industrial Development Zone. Available [Online] at http://www.ccaenvironmental.co.za/general/proposed-oil-and-gas-offshore-service-complex-at-the-saldanha-bay-idz. 20 September 2016.

Smart RM & JW Barko. 1980. Nitrogen nutrition and salinity tolerance of *Distichlis spicata* and *Spartina alterniflora*. *Ecology* 61, 630–638.

Smith K. 2017. Personal communication about unpublished research.

Sousa WP. 1979a. Disturbance in marine intertidal boulder fields: the non-equilibrium maintenance of species diversity. *Ecology* 60: 1225–1239.

Sousa WP. 1979b. Experimental investigations of disturbance and ecological succession in a rocky intertidal algal community. *Ecological Monographs* 49: 227–254.

Sousa WP. 1984. Intertidal mosaics: patch size, propagule availability, and spatially variable patterns of succession. *Ecology* 65: 1918–1935.

SRK Consulting 2009. Saldanha Air Quality Permit Basic Assessment: Air Quality Specialist Baseline Study and Impact Assessment. SRK Project Number 399449.

SRK Consulting, 2016a. Background Information Document: Project Definition and BA Process for a Proposed Sea-Based Aquaculture Development Zone in Saldanha Bay. Report prepared by SRK Consulting for the Department of Agriculture, Forestry and Fisheries. Project number 499020, June 2016.

SRK Consulting, 2016b. Project Definition Report Summary: Proposed Sea-Based Aquaculture Development Zone in Saldanha Bay. Report prepared by SRK Consulting for the Department of Agriculture, Forestry and Fisheries. Project number 499020, August 2016.

SRK Consulting, 2017. Executive Summary: Final Basic Assessment Report. BA Process for a Proposed Sea-Based Aquaculture Development Zone in Saldanha Bay. SRK Project Number: 499020. August 2017.

Staniford D. 2002. Sea cage fish farming: an evaluation of environmental and public health aspects (the five fundamental flaws of sea cage fish farming). Unpublished paper presented to the European parliament's Committee on Fisheries Public hearing on 'Aquaculture in the European Union: present situation.

Statistics South Africa 2014. Census 2011. Available at http://www.statssa.gov.za/census 2011/default.asp Accessed 15 August 2014.

Staniford D. 2002a. A big fish in a small pond: the global environmental and public health threat of sea cage fish farming. Sustainability of the Salmon Industry in Chile and the World Workshop organized by the Terram Foundation and Universidad de los Lagos in Puerto Montt, Chile.

Stebbing TRR. 1910. General Catalogue of South African Crustacea (Part V. of S.A. Crustacea, for the marine investigations in South Africa). *Annals of the South African Museum* 6(4): 281–599.

Steffani CN & GM Branch. 2003a. Spatial comparisons of populations of an indigenous limpet *Scutellastra argenvillei* and the alien mussel *Mytilus galloprovincialis* along a gradient of wave energy. *African Journal of Marine Science* 25: 195–212.

Steffani CN & GM Branch. 2003b. Temporal changes in an interaction between an indigenous limpet *Scutellastra argenvillei* and an alien mussel *Mytilus galloprovincialis*: effects of wave exposure. *African Journal of Marine Science* 25: 213–229.

Stegenga H, Bolton JJ & Anderson RJ. 1997. Seaweeds of the South African west coast. *Contributions from the Bolus Herbarium* 18: 3–637.

Stenton-Dozey JME, Jackson LF & AJ Busby. 1999. Impact of mussel culture on macrobenthic community structure in Saldanha Bay, South Africa. *Marine Pollution Bulletin* 39: 357–366.

Stenton-Dozey JME, Probyn T & A Busby. 2001. Impact of mussel (*Mytilus galloprovincialis*) raft-culture on benthic macrofauna, in situ oxygen uptake and nutrient fluxes in Saldanha Bay, South Africa. *Canadian Journal of Fisheries and Aquatic Sciences* 58: 1021–1031.

Stevcic Z. 1988. Autecological investigations of the porcelain crab *Porcellana platycheles* (Pennant) (Decapoda, Anomura) in the Rovinj area (Northern Adriatic). *Crustaceana* 55: 242–252.

Stephenson TA & A Stephenson. 1972. Life between tidemarks on rocky shores. Publishers WH Freeman and co., 425 pp.

Summers C. 2012. "Lead and cadmium in seabirds of South Africa". All Theses. 1445. https://tigerprints.clemson.edu/all_theses/1445.

Summers JK, Wade TL, Engle VW & ZA Malaeb. 1996a. Normalization of metal concentrations in estuarine sediments from the Gulf of Mexico. *Estuaries* 19: 581–594.

Summers RW. 1977. Distribution, abundance and energy relationships of waders (Aves: Charadrii) at Langebaan Lagoon. *Transactions of the Royal Society of South Africa* 42(3–4): 483–495.

Swart C, Visser V & TB Robinson. 2018. Patterns and traits associated with invasions by predatory marine crabs. *NeoBiota* 39: 79–102.

Sunda WG. 1989. Trace Metal Interactions with Marine Phytoplankton. Biological Oceanography 6(5-6): 411–442.

Takada Y. 1999. Influence of shade and number of boulder layers on mobile organisms on a warm temperate boulder shore. *Marine Ecology Progress Series* 189: 171–179.

Tarjuelo I, Posada D, Crandall K, Pascual M & X Turon. 2001. Cryptic species of *Clavelina* (Ascidiacea) in two different habitats: harbours and rocky littoral zones in the northwestern Mediterranean. *Marine Biology* 139: 1432–1793.

Taylor SR & SM McLennan. 1981. The composition and evolution of the continental crust – Rare earth element evidence from sedimentary rocks. *Royal Society of London Philosophical Transactions Series* A301: 381–398 .

Taylor SR. 1964. Abundance of chemical elements in the continental crust: a new table. *Geochimica et Cosmochimica Acta* 28(8): 1273–1285.

Thérivel R, Wilson E, Thompson S, Heaney D & D Pritchard. 1994. *Strategic Environmental Assessment*. Earthscan Publications, London.

Thompson WW. 1913. The Sea fisheries of the Cape Colony, from Van Riebeeck's days to the eve of the Union. Cape Town; Maskew Miller. 163 pp.

Tiganus D, Coatu V, Lazar L, Oros A & A Sinu. 2013. Identification of the sources of polycyclic aromatic hydrocarbons in sediments from the Romanian Balck Sea sector. *Cercetari Marine* 42: 187–196.

Tokeshi M. 1989. Feeding ecology of a size-structured predator population, the South American sun-star *Heliaster helianthus*. *Marine Biology* 100: 495–505.

Tolosa I, De Mora S.J, Fowler S, Villenenuve J.P, Bartocci J & C Cattini. 2005. Aliphatic and aromatic hydrocarbons in marine biota and coastal sediments from the Gulf and the Gulf of Oman. *Marine Pollution Bulletin* 50 (12): 1619.

Transnet National Ports Authority (TNPA). 2007. Port of Saldanha Oil Spill Contingency Plan. 51 pp.

Transnet National Port Authoristy 2015. National Port Plans 2015. Available [Online] at https://www.transnetnationalportsauthority.net/Infrastructure%20and%20Port%20Planning/Documents/National%20Port%20Plans%202015.pdf [Accessed on 5 September 2017].

Transnet National Port Authority (TNPA). 2015. TNPA Saldanha Q4 PCC Presentation. TNPA Port Consultative Committee Quarter 4. Port of Saldanha, 4 February 2015.

Tunley KL, Attwood CG, Moloney CL & L Fairhurst. 2009. Variation in population structure and life history parameters of steentjies *Spondyliosoma emarginatum*: effects of exploitation and biogeography. *African Journal of Marine Science* 31: 133–143.

Turekian KK & KH Wedepohl. 1961. Distribution of the elements in some major units of the earth's crust. *Geological Society of America Bulletin* 72: 175–192.

Turpie JK. 1995. Prioritizing South African estuaries for conservation: A practical example using waterbirds. *Biological Conservation* 74: 175–185.

Turpie JK & PAR Hockey. 1993. Comparative diurnal and nocturnal foraging behaviour and energy intake of premigratory Grey Plovers *Pluvialis squatarola* and Whim-brels *Numenius phaeopus* in South Africa. *Ibis* 135: 156–165.

Turpie JK & VC Love. 2000. Avifauna and human disturbance on and around Thesen Island, Knysna estuary: Implications for the island's marina development and management plan. Report to Chris Mulder & Associates.

Twichell SC, Meyersa PA & L Diester-Haassb. 2002. Significance of high C/N ratios in organic-carbon-rich Neogene sediments under the Benguela Current upwelling system. *Organic Geochemistry* 33: 715–722.

Tyrrell T & M Lucas. 2002. Geochemical evidence of denitrification in the Benguela upwelling system. *Continental Shelf Research* 22: 2497–2511.

Underhill LG. 1987. Waders (Charadrii) and other waterbirds at Langebaan Lagoon, South Africa, 1975-1986. *Ostrich* 58(4): 145–155.

UNEP 2008. Desalination Resource and Guidance Manual for Environmental Impact Assessments. United Nations Environment Programme, Regional Office for West Asia, Manama, and World Health Organisation, Regional Office for the Eastern Mediterranean, Cairo.

Valiela I, Costa J, Foreman K, Teal JM, Howers B & D Aubrey. 1990. Transport of groundwater-borne nutrients from watersheds and their effects on coastal waters. *Biochemistry* 10: 177–197.

van Ballegooyen RC, Mabille E, Brown S, Newman B & S Taljaard. 2012. Transnet Reverse Osmosis desalination plant, Saldanha Bay: Physico-chemical environmental baseline. CSIR Report, CSIR/NRE/ECO/ER/2012/0033/B, 146–19 pp.

van Ballegooyen RC, Steffani N & A Pulfrich. 2007. Environmental Impact Assessment: Proposed Reverse Osmosis Plant, Iron –ore Handling Facility, Port of Saldanha – Marine Impact Assessment Specialist Study, Joint CSIR/Pisces Report, CSIR/NRE/ECO/ER/2007/0149/C, 190–198 pp.

Van der Linden S. 2013. *Salt Marsh Distribution and Structure at Langebaan Lagoon*. Submitted as a partial fulfilment of the requirements for BOT450. Department of Botany, Nelson Mandela Metropolitan University, PO Box 77000, Port Elizabeth, South Africa, 6031.

van der Merwe IJ, Ferreira SLA & HL Zietsman. 2005. Growth Potential of towns in the Western Cape. Centre for Geographical Analysis, Stellenbosch University.

van Erkom Schurink C & CL Griffiths. 1991. A comparison of reproductive cycles and reproductive output in four southern African mussel species. Marine Ecology Progress Series 76(2): 123–134.

van Katwijk MM, Vergeer LHT, Schmitz GHW & JGM Roelofs. 1997. Ammonium toxicity in eelgrass *Zostera marina*. *Marine Ecology Progress Series* 157: 159–173.

Vaquer-Sunyer R & CM Duarte. 2008. Thresholds of hypoxia for marine biodiversity. *Proceedings of the National Academy of Science* 105(40): 15452–15457.

Veolia. 2017. Concepts of re-use of Saldanha Bay Domestic Waste Water. PowerPoint presentation.

Vermaak N. 2019 Unpublished PhD Thesis. Historic Review of the Langebaan Road and Elandsfontein Aquifer Systems, University of the Western Cape, Cape Town.

Viljoen R, Silomntu M, Reuther S & Jones S. 2010. Transnet Port Terminal - Bulk Terminal Saldanha: Draft Environmental Management Programme for Air Quality Permit Amendment Application.

Visser D, Goes M & P Rosewarne. 2007. Environmental Impact Assessment: Port Of Saldanha Proposed Reverse Osmosis Water Desalination Plant Groundwater Resources Impact Assessment. SRK Consulting Report.

Vieira LM, Jones MS & Taylor PD. 2014. The identity of the invasive fouling bryozoan *Watersipora subtorquata* (d'Orbigny) and some other congeneric species. *Zootaxa* 3857: 151–182.

Voellmy IK, Purser J, Simpson SD & AN Radford. 2014. Increased Noise Levels Have Different Impacts on the Anti-Predator Behaviour of Two Sympatric Fish Species. *PLoS ONE* 9(7): e102946.

Warwick RM. 1993. Environmental impact studies on marine communities: Pragmatical considerations. *Australian Journal of Ecology* 18: 63–80.

Waycott M, Duarte CM, Carruthers TJB, Orth RJ, Dennison WC, Olyarnik S, Calladine A, Fourqurean JW, Heck KL, Hughes AR, Kendrick GA, Kenworthy WJ, Short FT & SL Williams. 2009. Accelerating loss of seagrasses across the globe threatens coastal ecosystems. *PNAS* 106: 12377–12381.

Weaver J & L Fraser. 1998. Langebaan Road Aquifer Drilling and Testing of new wellfield. Project Number: JW508. CSIR, Stellenbosch.

Weaver J. & L Fraser. 1998. Langebaan Road Aquifer Drilling and Testing of new wellfield. Project Number: JW508. CSIR, Stellenbosch.

Weeks SJ, Boyd AJ, Monteiro PMS & GB Brundrit. 1991b. The currents and circulation in Saldanha Bay after 1975 deduced from historical measurements of drogues. *South African Journal of Marine Science* 11: 525–535.

Weeks SJ, Monteiro PMS, Nelson G & RM Cooper. 1991a. A note on wind-driven replacement flow of the bottom layer in Saldanha Bay, South Africa: implications for pollution. *South African Journal of Marine Science* 11: 579–583.

Weller F, Cecchini LA, Shannon L, Sherley RB, Crawford RJM, Altwegg R, Scott L, Stewart T & A Jarre. 2014. A system dynamics approach to modelling multiple drivers of the African penguin population on Robben Island, South Africa. *Ecological Modelling*, 277: 38–56.

Weller F, Sherley RB, Waller LJ, Ludynia K, Geldenhuys D, Shannon LJ & A Jarre. 2016. System dynamics modelling of the Endangered African penguin populations on Dyer and Robben islands, South Africa, *Ecological Modelling,* 327:44–56.

Wesgro. 2012. Saldanha Bay Industrial Development Zone fact sheet. Available at: http://wesgro.co.za/publications/publication-directory. Accessed 22 August 2014.

Westbrook MA. 1934. *Antithamnion spirographidis* Schiffner. *Journal of Botany* 72: 65–68.

Whitfield AK, Beckley LE, Bennett BA, Branch GM, Kok HM, Potter IC & RP van der Elst. 1989. Composition, species richness and similarity of ichthyofaunas in eelgrass Zostera capensis beds of southern Africa. *South African Journal of Marine Science* 8: 251–259.

Whittington PA, Crawford RJM, Martin AP, Randall RM, Brown M, Ryan PG, Dyer BM, Harrison KHB, Huisamen J, Makhado AB, Upfold L, Waller LJ & M Witteveen. 2016. Recent Trends of the Kelp Gull (*Larus dominicanus*) in South Africa. *Waterbirds* 39(1): 99–113.

Whittington PA, Randall BM, Wolfaardt AC, Crawford RJM, Klages NTW, Bartlett PA, Chesselet YJ & R Jones. 2005a. Patterns of movements of the African penguin in South Africa and Namibia. *African Journal of Marine Science* 27(1): 215–229.

Whittington PA, Randall RM, Wolfaardt AC, Klages NTW, Randall BM, Bartlett PA, Chesselet YJ & R Jones. 2005b. Patterns of immigration to and emigration from breeding colonies by African penguins. *African Journal of Marine Science* 27(1): 205–213.

Winston JE. 1995. Ectoproct diversity of the Indian River coastal lagoon. *Bulletin of Marine Science* 57: 84–93.

Witteveen M, Brown M & PG Ryan. 2017. Anthropogenic debris in the nests of kelp gulls in South Africa. *Marine Pollution Bulletin* 114: 699–704.

WHO. 2007. Desalination for Safe Water Supply: guidance for Health and Environmental Aspects Applicable to Desalination. Geneva.

Williams AJ, Steele WK, Cooper J & RJM Crawford. 1990. Distribution, population size and conservation of Hartlaub's Gull *Larus hartlaubii*. *Ostrich* 61: 66–76.

Williams LG. 2015. Genetic structure of pest polydorids (Annelida: Spionidae) infesting *Crassostrea gigas* in southern Africa: are pests being moved with oysters? MSc thesis, Stellenbosch University, South Africa.

Wilson JRU, Dormontt EE, Prentis PJ, Lowe AJ & DM Richardson. 2009. Biogeographic concepts define invasion biology. *Trends in Ecology and Evolution* 24: p. 586.

Windom HL. 1988. A Guide to the Interpretation of Metal Concentrations in Estuarine Sediments. Office of Water Policy. Florida Department of Environmental Protection.

Wollaston EM. 1968. Morphology and taxonomy of southern Australian genera of *Crouanieae* Schmitz (Ceramiaceae: Rhodophyta). *Australian Journal of Botany* 16: 217–417.

Woodford AC & M Fortuin. 2003a. Determination of the Groundwater Reserve and Allocations for the Langebaan Road and Elandsfontein Aquifer Systems, SRK Consulting Engineers and Scientists, October 2003, Report No. 318532, Bellville, Cape Town, 49p.

Woodford AC & M Fortuin. 2003b. Assessment of the Development Potential of Groundwater Resources for the West Coast District Municipality, specialist geohydrological report for Kwezi-V3 Consulting Engineers, as part of the project: 'Pre-Feasibility Study of Potential Water Sources for the Area served by the West Coast District Municipality', SRK Consulting Engineers and Scientists, October 2003, Report No. 318611, Bellville, Cape Town, 94p.

Woolsey S & M Wilkinson. 2007. Localized field effects of drainage water from abandoned coal mines on intertidal rocky shore seaweeds at St Monans, Scotland. *Journal of the Marine Biological Association of the United Kingdom* 87(03): 659–665.

Wright AG, Laird MC & BM Clark. 2018b. *Near Field Modelling and Assessment of Impacts for the Mykonos Desalination Plant, Club Mykonos, Langebaan*. Specialist Report no. 1785/2 prepared by Anchor Environmental Consultants (Pty) Ltd for SRK Consulting (South Africa) (Pty) Ltd. 96 pp.

Wright AG, Mostert B & BM Clark. 2018a. *Dispersion Modelling, Monitoring, and Assessment of Impacts of Reverse Osmosis Discharge into Saldanha Bay in Fulfilment of Sea Harvest Corporation (Pty) Ltd Coastal Waters Discharge Permit Conditions.* Specialist Report no. 1798/3 prepared by Anchor Environmental Consultants (Pty) Ltd for Sea Harvest Corporation (Pty) Ltd. 92 pp.

WSP Africa Coastal Engineers. 2010. Development of a methodology for defining and adopting coastal development set-back lines. Volume 1. Main report. Report prepared for Department of Environmental Affairs and Developing Planning, Western Cape Government, 81 pp.

Wuersig B & GA Gailey. 2002. Marine mammals and aquaculture: conflicts and potential resolutions. In: Stickney & J.P. McVey (eds); Responsible Marine Aquaculture CABI Publishing, New York.

Würsig B, Greene CR & TA Jefferson. 2000. Development of an air bubble curtain to reduce underwater noise of percussive piling. *Marine Environmental Research* 49: 79–93.

Wyatt ASJ, Hewitt CL, Walker DI & TJ Ward. 2005. Marine introductions in the Shark BayWorld Heritage Property, Western Australia: A preliminary assessment. *Diversity and Distribution* 11: 33–44.

ZAA. 2016. Specialist Study on the Potential Impact of the Proposed Project Phakisa Rig Repair Project in the Port of Saldanha on Hydrodynamics and Water Quality. Report prepared by ZAA Engineering Projects and Naval Architects for CCA Environmental. 37 pp.

Zedler JB, Williams P & J Boland. 1986. Catastrophic events reveal the dynamic nature of salt-marsh vegetation in southern California. *Estuaries* 9: 75–80.

ZLH Projects & Naval Architects. 2009. Pre-feasibility Study – Extension to Mossgas Quay Preliminary Development of Alternatives Report. P&NA 1264/RPT/002 Rev C.

Zsilavecz G. 2007. *Nudibranchs of the Cape Peninsula and False Bay*. Southern Underwater Research Group Press.

Zullo VA. 1992. Revision of the balanid barnacle genus *Concavus* Newman, 1982, with the description of a new Subfamily, two new genera, and eight new species. *The Paleontological Society* 27: 1–46.

15 APPENDIX

The Chapter contains supplementary information for the Groundwater, Rocky Intertidal, Birds, and Alien and Invasive Species Chapters in graph format (Figure 15.1 - Figure 15.6) and in table format (Table 15.1 - Table 15.3).

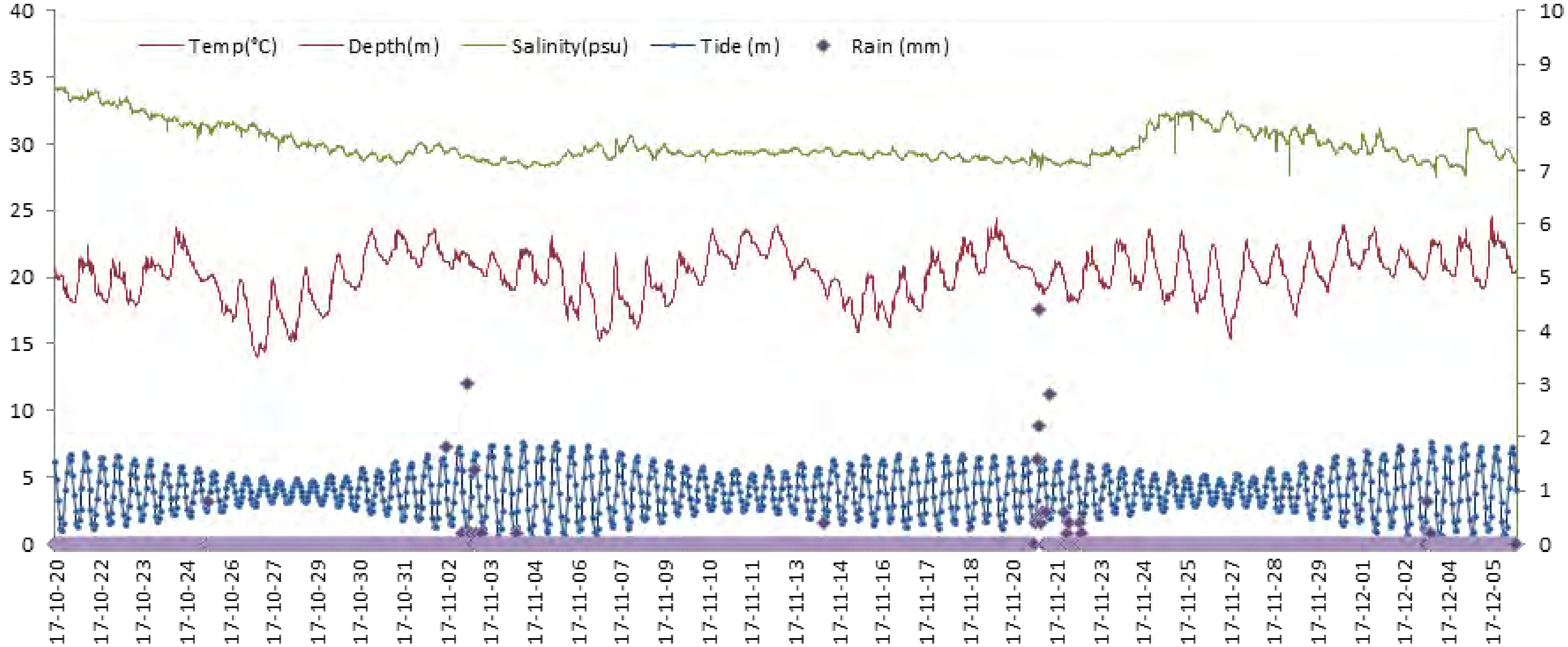


Figure 15.1 Salinity (green line, left axis, units PSU), temperature (red line, left axis, units °C) in twenty minute intervals over a 3 month period in October 2017 through December 2017. Tidal data (dark blue points, right axis, units m) and rain (purple, right axis, units mm) are in hourly intervals over the same time period (tidal data provided by hydrographer, SANHO, rain data by WeatherSA).

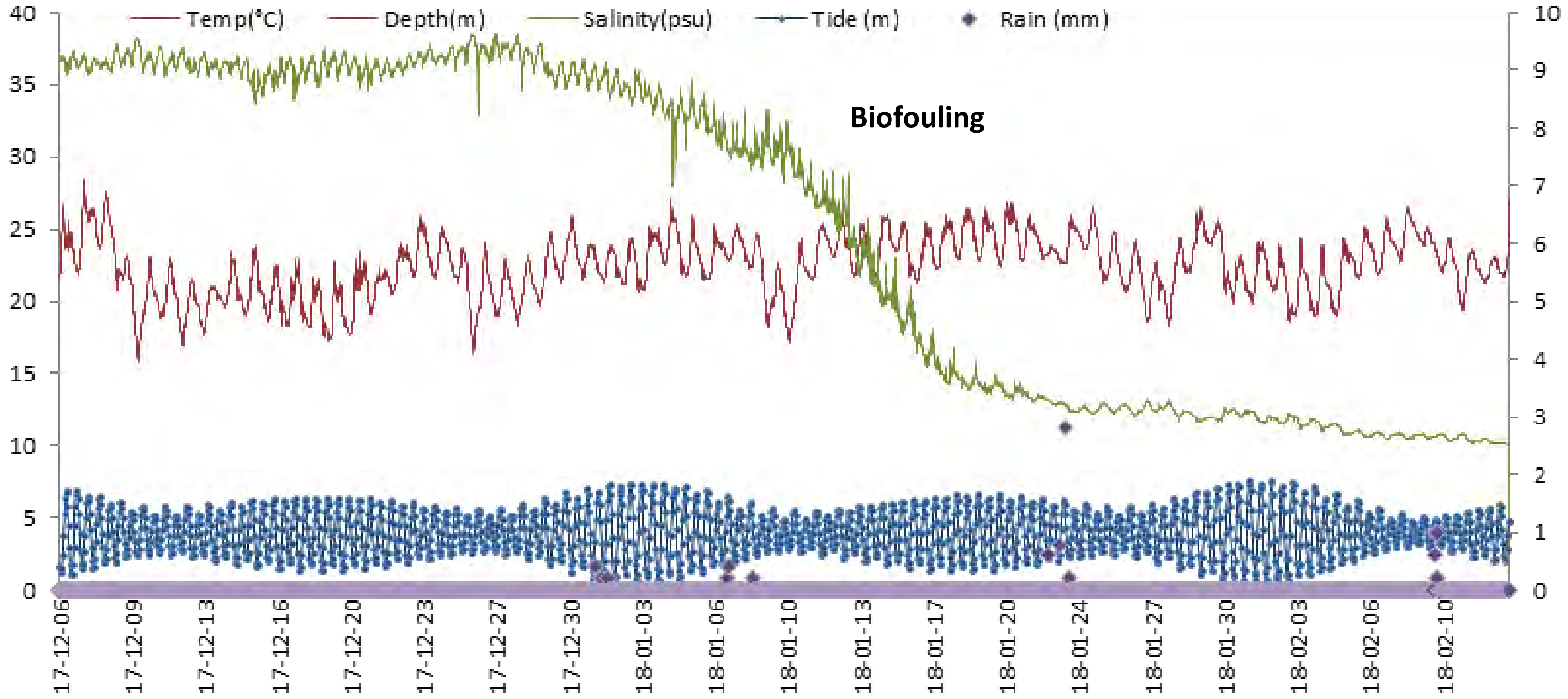


Figure 15.2 Salinity (green line, left axis, units PSU), temperature (red line, left axis, units °C) in twenty minute intervals over a 3 month period in December 2017 through February 2018. Tidal data (dark blue points, right axis, units m) and rain (purple, right axis, units mm) are in hourly intervals over the same time period (tidal data provided by hydrographer, SANHO, rain data by WeatherSA).

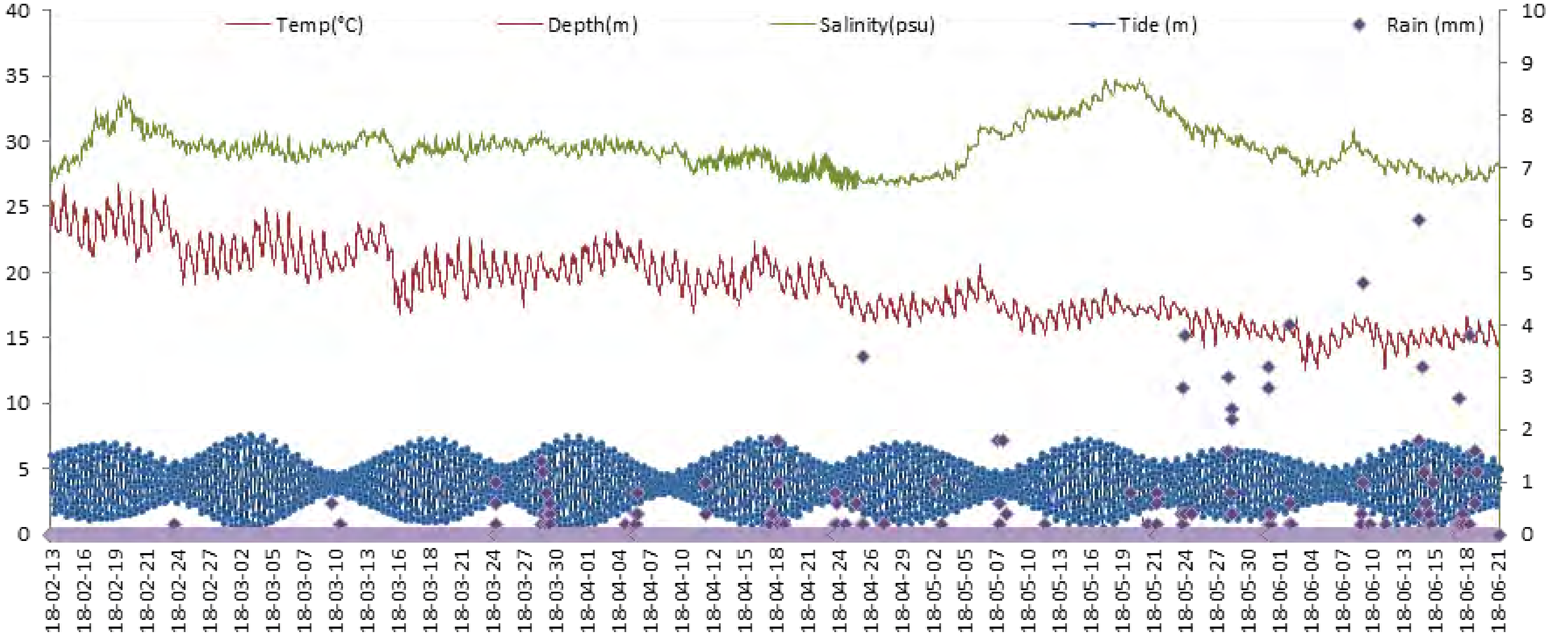


Figure 15.3 Salinity (green line, left axis, units PSU), temperature (red line, left axis, units °C) in twenty minute intervals over a 5 month period in February 2018 through June 2018. Tidal data (dark blue points, right axis, units m) and rain (purple, right axis, units mm) are in hourly intervals over the same time period (tidal data provided by hydrographer, SANHO, rain data by WeatherSA).

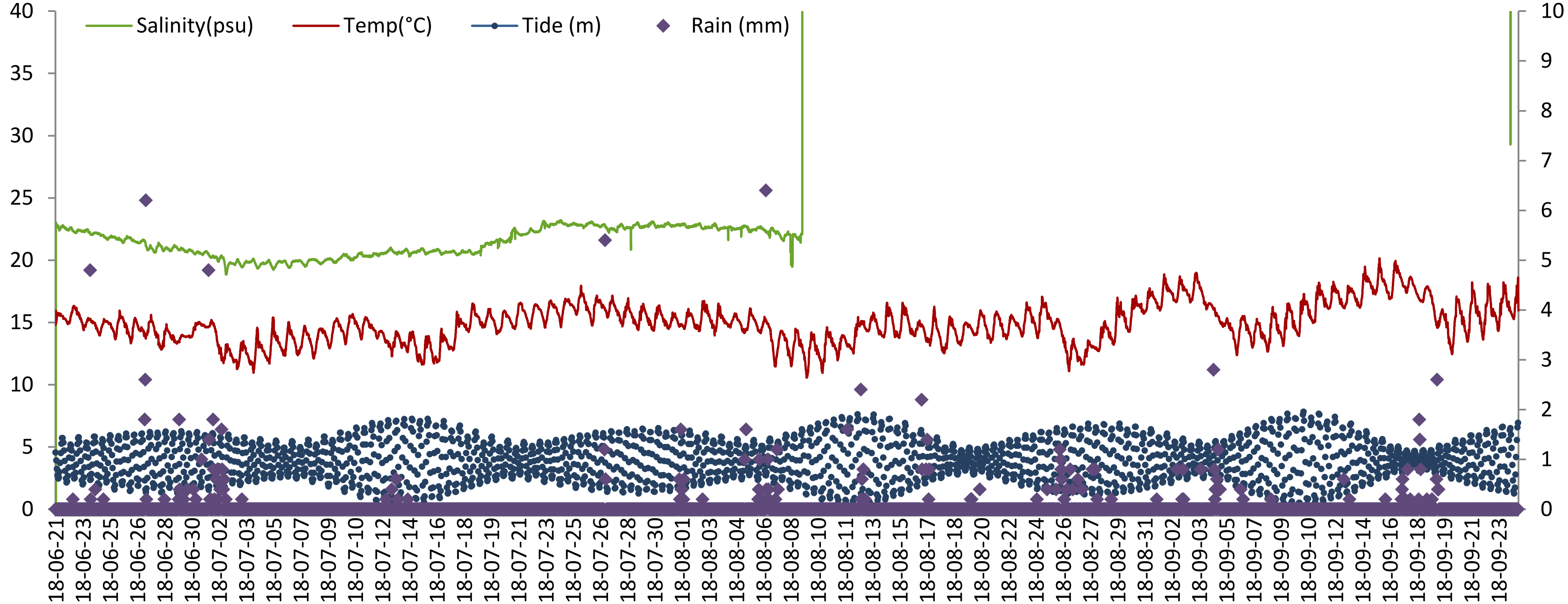


Figure 15.4 **Salinity (green line, left axis, units PSU), temperature (red line, left axis, units °C) in twenty minute intervals over a 4 month period in June 2018 through September 2018. Tidal data (dark blue points, right axis, units m) and rain (purple, right axis, units mm) are in hourly intervals over the same time period (tidal data provided by hydrographer, SANHO, rain data by WeatherSA).**

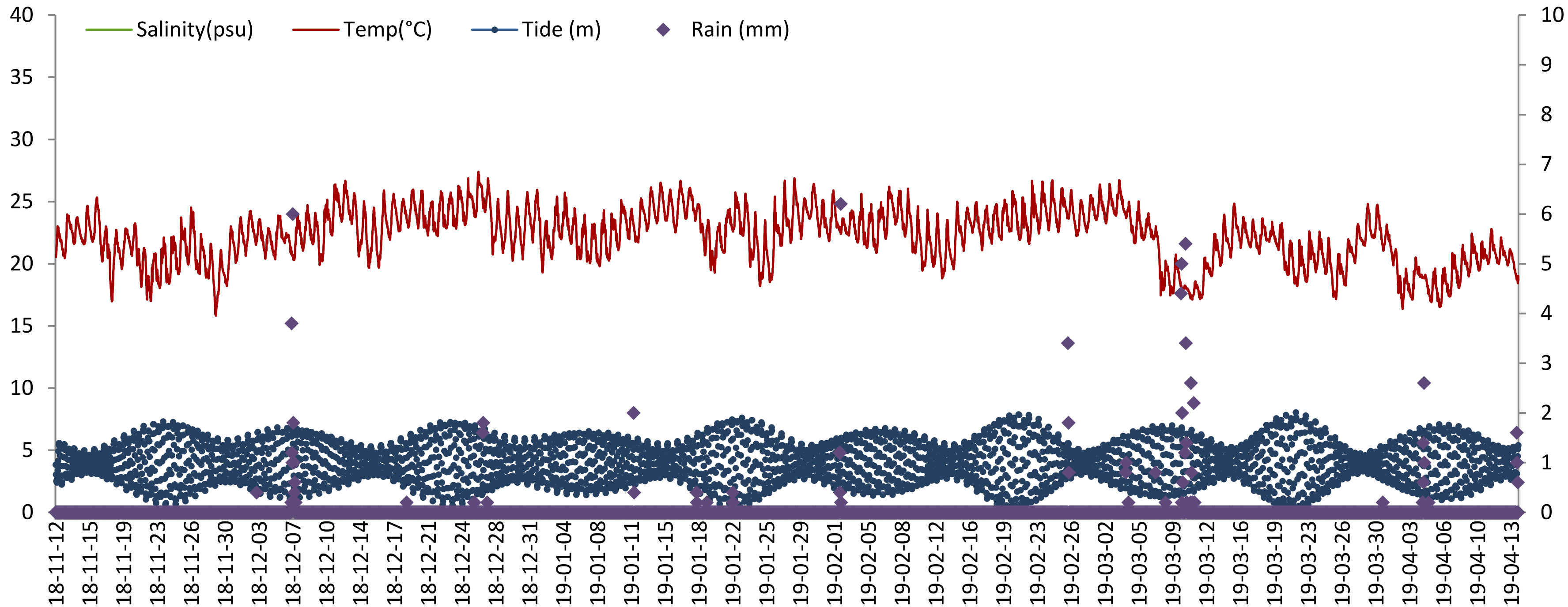


Figure 15.5 **Salinity (green line, left axis, units PSU), temperature (red line, left axis, units °C) in twenty minute intervals over a 6 month period in November 2018 through April 2019. Tidal data (dark blue points, right axis, units m) and rain (purple, right axis, units mm) are in hourly intervals over the same time period (tidal data provided by hydrographer, SANHO, rain data by WeatherSA).**

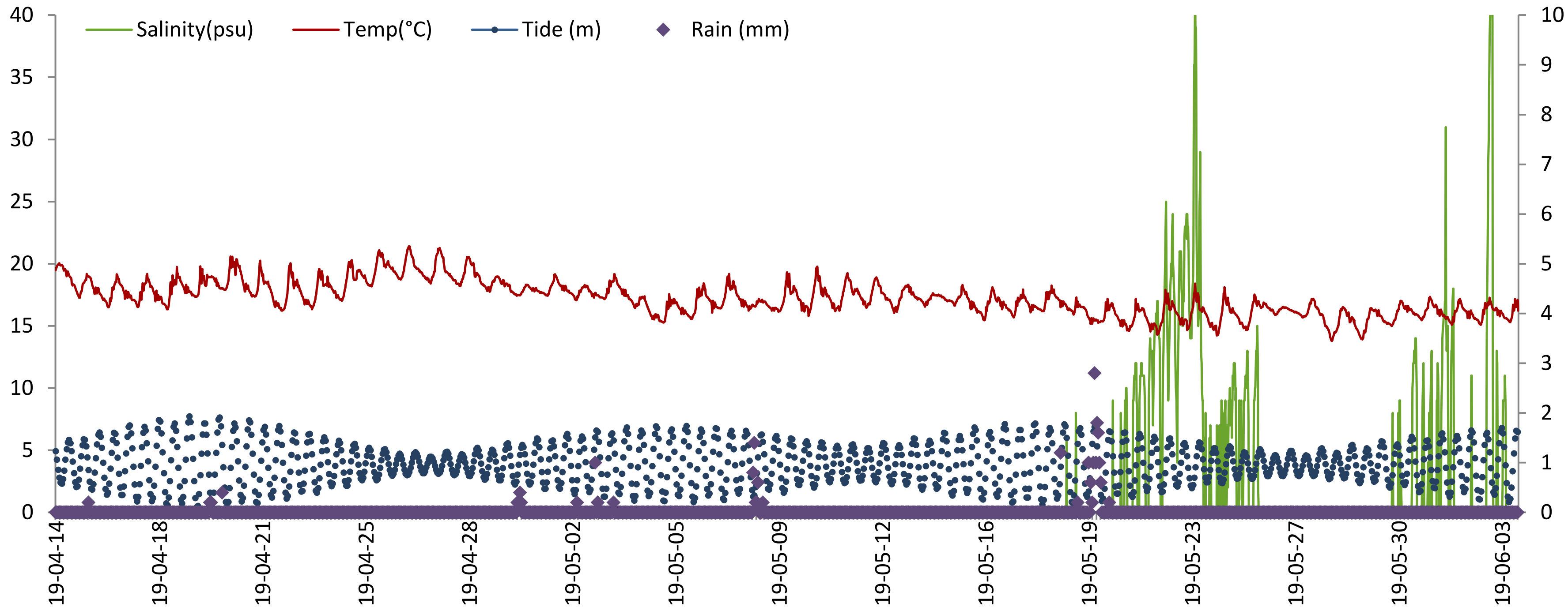


Figure 15.6 **Salinity (green line, left axis, units PSU), temperature (red line, left axis, units °C) in twenty minute intervals over a 3 month period in April 2019 through June 2019. Tidal data (dark blue points, right axis, units m) and rain (purple, right axis, units mm) are in hourly intervals over the same time period (tidal data provided by hydrographer, SANHO, rain data by WeatherSA).**

Table 15.1 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	**89.5**	**85.5**	**54.7**	**72.4**	**26.1**	**69.0**	**50.1**	**75.4**
Rock	87.60	71.86	54.50	72.29	26.06	68.96	36.61	69.09
Sand	0	0	0	0	0	0	13.48	6.35
Gravel	1.94	13.66	0.25	0.12	0	0	0	0
GRAZERS	**2.76**	**1.33**	**1.81**	**3.04**	**2.32**	**3.01**	**1.27**	**3.41**
Acanthochiton garnoti	0.16	0.03	0.03	0	0	0	0	0
Afrolittorina knysnaensis	0	0	0.82	0.39	0.57	0.55	0.02	0.05
Callochiton dentatus	0	0	0	0	0	0	0	0
Chaetopleura papilio	0	0	0	0	0	0	0	0
Chiton nigrovirescens	0	0	0	0	0	0	0	0
Cymbula compressor	0	0	0	0	0	0.04	0	0
Cymbula granatina	0.14	0.19	0.04	0.02	0.32	0.13	0.40	0.09
Cymbula miniata	0	0	0	0	0	0.01	0.0	0
Cymbula oculus	0.07	0.01	0	0	0	0.02	0.03	0
Dendrofissurella scutellum	0	0.02	0	0	0	0	0	0.09
Fissurella mutabilis	0.02	0.01	0.01	0	0.02	0	0.04	0.05
Gibbula spp.	0.03	0	0	0	0	0	0.01	0
Gibbula zonata	0	0	0.01	0	0	0	0	0.08
Haliotis midae	0	0	0	0	0	0	0.02	0
Helcion dunkeri	0	0	0	0	0	0	0	0
Helcion pectunculus	0.44	0	0	0.01	0	0	0	0
Helcion pruinosus	0	0	0	0	0	0	0	0
Ischnochiton oniscus	0.01	0	0	0	0	0	0	0
Scutellastra argenvillei	0	0	0	0.20	0.27	0.09	0.02	0.10
Scutellastra barbara	0	0	0.11	0.13	0.06	0	0	0
Scutellastra cochlear	0	0	0.02	0.95	0.03	0.24	0	0
Scutellastra granularis	0.05	0	0.21	0.33	0.85	0.72	0	0.11
Scutellastra tabularis	0.39	0	0	0.32	0	0.28	0	0
Siphonaria capensis	0.15	0	0.01	0.05	0.04	0.40	0.04	0.12
Siphonaria serrata	0.07	0	0.22	0.06	0.02	0	0.10	0.20
Parechinus angulosus	0.18	0.04	0.07	0.02	0.01	0.16	0	0.19
Parvulastra exigua	0.34	0.25	0.09	0.02	0.03	0.09	0.32	0.65
Onchidella maculata	0	0	0	0	0	0	0	0
Oxystele tigrina	0.30	0.38	0	0.05	0.09	0.23	0.07	1.36
Oxystele antoni	0.39	0.39	0.17	0.50	0.02	0.04	0.17	0.32
Tricolia capensis	0.01	0	0	0	0	0	0.03	0
PREDATORS	**0.99**	**0.29**	**0.65**	**0.93**	**0.94**	**2.15**	**0.78**	**2.24**
Actinia equina	0.16	0.02	0	0	0	0.19	0.01	0.00
Anthopleura michaelseni	0	0	0	0	0	0	0	0.00
Anthostella stephensoni	0	0	0	0	0.01	0	0	0.21
Anthothoe stimpsonii	0	0.05	0	0.10	0	0.11	0.01	0.25
Bunodactis reynaudi	0	0	0.02	0.02	0.16	0.10	0.03	0.05
Bunodosoma capense	0	0	0.04	0.08	0.10	0.56	0	0.00
Burnupena papyracea	0	0	0	0	0	0.16	0	0.00
Burnupena spp.	0.55	0.21	0.14	0.59	0.56	0.81	0.28	1.48
Callopatiria granifera	0.02	0	0	0	0	0	0	0.00
Clionella sinuata	0.03	0.01	0	0.01	0	0.01	0	0.00

Percentage cover	DS	J	IO	L	M	NB	SE	SW
Conus mozambicus	0	0	0	0	0	0	0	0.00
Corynactis annulata	0	0	0	0	0	0	0	0.00
Cyclograpus punctatus	0.05	0	0.02	0.02	0	0.08	0	0.01
Doris granulosa	0	0	0	0	0	0	0	0.00
Doris verrucosa	0	0	0	0	0	0	0	0.00
Dromidia spp.	0	0	0	0	0	0	0	0.00
Flatworm	0	0	0	0	0	0	0	0.00
Fusinus sp.	0.03	0	0	0	0	0	0	0.00
Henricia ornata	0	0	0	0.02	0	0.04	0.08	0.03
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	0.01	0	0.07	0.02	0.03	0.09	0	0.05
Nucella squamosa	0	0	0.32	0.00	0.04	0	0.01	0.00
Nudibranch	0	0	0	0	0.02	0	0	0.00
Ophiuroidea	0	0	0	0	0.01	0	0	0.00
Paguristes gamianus	0	0	0	0	0	0	0.02	0.00
Philine aperta	0	0	0	0	0	0	0	0.00
Platydromia spongiosa	0	0	0	0	0	0	0	0.00
Pseudactinia flagellifera	0.03	0	0.03	0.07	0	0	0.16	0.02
Pseudactinia sp.	0.09	0	0	0	0	0	0.17	0.14
Trochia cingulata	0	0	0	0	0.01	0	0	0.00
Volvarina capensis	0	0	0	0	0	0	0	0.00
FILTER FEEDERS	**2.12**	**2.59**	**13.51**	**7.21**	**34.23**	**12.26**	**3.03**	**1.75**
Amphibalanus amphitrite amphitrite	0	0.01	0	0	0	0	0	0
Aulacomya atra	0.23	0.08	0.12	0.61	7.31	1.71	0	0.77
Austromegabalanus cylindricus	0	0	0.50	0.05	0.41	0.13	0	0
Balanus glandula	0.12	0.11	9.16	4.56	5.66	6.83	0.01	0.09
Choromytilus meridionalis	0	0	0	0	0	0	0	0
Chthamalus dentatus	0	0	0	0	0	0	0	0
Colonial ascidian	0	0.07	0.28	0.07	0.25	0.14	0.16	0.09
Crepidula porcellana	0.22	0.13	0.01	0.01	0.07	0.05	0.11	0.05
Dendropoma corallinaceum	0	0	0	0.01	0	0.07	0	0
Dodecaceria capensis	0	0	0	0	0	0	0	0
Encrusting Bryozoa	0	0.02	0.29	0	0.01	0.09	0.51	0
Fanworm	0	0	0	0	0	0	0	0
Gunnarea gaimardi	0	0	0	0	0	0	0	0
Hydroids	0	0	0	0.07	0	0.20	0	0
Mytilus galloprovincialis	0.17	0.50	2.66	1.19	13.53	1.62	0	0
Notomegabalanus algicola	0.02	0.01	0	0.10	4.90	0.61	0	0
Octomeris angulosa	0	0	0	0	0	0	0	0
Pentacta doliolum	0	0	0.02	0	0.01	0	0	0
Perforatus perforatus	0.18	0.05	0.02	0.02	0	0	0	0
Pseudocnella insolens	0	0	0	0	1.73	0	0	0
Pyura herdmani	0	0	0.04	0	0	0	0	0
Pyura stolonifera	0.05	0.02	0.28	0.25	0	0	0	0
Roweia frauenfeldi	0.28	0	0	0.05	0.05	0	0	0
Sandy tube worm	0	1.60	0	0	0	0	0	0
Spirorbis spp.	0.01	0	0.07	0.16	0	0.49	0.59	0.05
Sponge	0.57	0	0.02	0	0.13	0.24	1.65	0.54

Percentage cover	DS	J	IO	L	M	NB	SE	SW
Tetraclita serrata	0	0	0.03	0	0	0	0	0
Thyone aurea	0.28	0	0	0.04	0.15	0	0	0
Tubeworm	0	0	0	0.05	0.01	0.09	0	0.16
CRUSTOSE	**2.19**	**5.94**	**11.41**	**10.81**	**22.47**	**8.05**	**31.79**	**7.03**
Coralline (crustose)	0.36	2.10	10.62	7.59	6.19	3.33	11.35	1.39
Coralline (upright)	0	0	0	0.95	0.48	0.24	0.53	0.13
Diatoms	0	0.26	0.09	1.15	15.07	3.14	7.34	5.41
Hildenbrandia spp.	0.90	2.44	0.61	0.84	0.73	0.73	11.96	0.09
Ralfsia verrucosa	0.93	1.14	0.08	0.27	0	0.62	0.60	0
EPHEMERALS	**0.77**	**3.21**	**10.83**	**2.01**	**7.94**	**1.64**	**3.52**	**4.12**
Bryopsis myosurioides	0	0	0.51	0	0.12	0	0.09	0
Callithamnion collabens	0	0	0	0.07	0	0.02	0	0
Centroceras clavulatum	0	0.02	0	0	0	0.04	0	0
Ceramium spp.	0.03	0.29	0.74	0.16	0.15	0.18	0.05	0.16
Cladophora spp.	0.03	1.15	0.15	0.01	0.58	0.08	0.23	0.01
Green turf	0	0.57	0	0.02	0	0	0.01	0
Porphyra capensis	0	0	0	1.24	1.24	0.30	0.06	0.05
Pachymenia chornia	0	0	0.05	0.02	0	0.01	0	0
Pachymenia orbitosa	0.02	0	0	0.14	0.19	0.23	0.06	0.13
Ulva spp.	0.68	1.18	9.43	0.34	5.66	0.78	3.02	3.77
CORTICATED	**1.63**	**1.13**	**6.35**	**2.40**	**4.06**	**2.18**	**9.18**	**5.45**
Ahnfeltiopsis complicata	0	0	0	0	0	0	0.10	0
Ahnfeltiopsis glomerata	0	0	0.92	0	0.13	0	0.90	0.19
Ahnfeltiopsis polyclada	0.02	0.17	0.31	0	0	0	0.03	0.02
Botryocarpa prolifera	0	0	0	0	0	0.02	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	0.17	0	0.01	0	0	0
Carpoblepharis flaccida	0	0	0.44	0	0.11	0	0	0
Caulacanthus ustulatus	0	0.02	0.03	0.09	0.65	0.47	0.27	0
Champia compressa	0	0	0.01	0	0	0.02	0.36	0.02
Champia lumbricalis	0	0	0.07	0.01	0.45	1.07	0	0
Chondria capensis	0	0	0	0	0	0	0	0
Chordariopsis capensis	0	0	0	0	0	0	0	0
Cochlear Garden	0	0	0.01	0.15	0.01	0.03	0	0
Codium fragile fragile	0	0.08	0	0	0	0	0	0.07
Codium lucasii	0	0	0	0	0	0	0	0
Codium stephensiae	0	0	0	0	0	0	0	0.14
Colpomenia sinuosa	0.06	0.17	0	0	0	0	0	0
Exallosorus harveyanus	0	0	0	0	0.01	0	0	0
Gelidium pristoides	0	0	0.10	0.14	0	0.07	0.21	0.05
Gelidium pteridifolium	0	0	0	0	0	0	0	0
Gelidium vittatum	0	0	0	0.16	0	0.08	0	0
Gigartina bracteata	0	0.06	0	0	0	0.02	1.02	0
Gigartina polycarpa	1.18	0.59	1.87	1.56	1.42	0.07	4.08	3.13
Grateloupia belangeri	0	0	0	0	0	0	0	0
Grateloupia longifolia	0	0	0	0	0	0	0	0
Gymnogongrus dilatatus	0	0	0	0	0.09	0	0	0
Halopteris funicularis	0	0	0	0	0	0	0	0
Hypnea ecklonii	0.14	0	0.04	0	0	0	0.07	0.46

Percentage cover	**DS**	**J**	**IO**	**L**	**M**	**NB**	**SE**	**SW**
Hypnea spicifera	0.03	0	0	0	0	0	0	0.23
Laurencia glomerata	0	0	0	0	0	0	0	0
Leathesia marina	0	0	0	0	0	0	0	0
Mazzaella capensis	0.02	0	0	0.05	0.11	0	0.01	0
Neuroglossum binderianum	0	0	0	0	0	0	0	0
Nothogenia erinacea	0	0.01	0.06	0	0	0.03	0.23	0
Nothogenia ovalis	0	0	0	0	0	0	0	0
Phyllymenia capensis	0	0	0	0	0	0	0	0
Plocamium corallorhiza	0	0	0	0	0	0	0	0
Plocamium spp.	0.01	0	1.41	0	0.63	0.01	0.50	0.02
Polyopes constrictus	0	0	0.13	0	0	0	0.56	0.04
Portieria hornemanii	0	0	0	0	0	0	0	0
Pterosiphonia cloiophylla	0	0	0	0	0	0.02	0	0
Red turf	0.16	0.02	0.58	0.10	0	0	0.25	0.51
Rhodophyllis reptans	0	0	0	0	0	0	0	0
Rhodymenia pseudopalmata	0	0.02	0	0.10	0	0.04	0.02	0.05
Rhodymenia spp.	0	0	0	0	0	0	0	0
Sarcothalia radula	0	0	0	0	0	0	0	0
Sarcothalia scutellata	0	0	0	0.05	0.0	0	0.50	0.16
Sarcothalia stiriata	0	0	0	0	0.42	0.22	0.04	0.33
Schizymenia apoda	0	0	0.12	0	0	0	0	0
Splachnidium rugosum	0	0	0	0	0	0	0.01	0.05
Tayloriella tenebrosa	0	0	0	0	0	0	0	0
Tsengia lanceolata	0	0	0.02	0	0	0	0	0
KELP	**0.0**	**0.0**	**0.0**	**2.4**	**0.2**	**0.8**	**0.0**	**0.1**
Ecklonia maxima	0	0	0.53	0.78	0.22	1.73	0.34	0.37
Laminaria pallida	0	0	0.17	0.41	1.75	0	0.02	0.19

Table 15.2 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 15.3. **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

Taxon	Occurrence in Saldanha/ Langebaan	Status	Origin	Vector	Reference
Obelia geniculata	Likely	Naturalised	Unknown	SF/BW	Mead *et al.* 2011
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	To be confirmed	North American Pacific	SF/BW	Biccard and Griffiths (*Pers Comm. 2017)*

Taxon	Occurrence in Saldanha/ Langebaan	Status	Origin	Vector	Reference
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
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
Ischyrocerus 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

Taxon	Occurrence in Saldanha/ Langebaan	Status	Origin	Vector	Reference
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
Porcellana africana (Incorrectly identified as *Porcellana platycheles*)	Confirmed	Confirmed	North East Atlantc	BW	Griffiths *et al.* 2018
Xantho incicus	Likely	Alien	France	M	Haupt *et al.* 2010
<u>INSECTA</u>					
COLEOPTERA					
Cafius xantholoma	Likely	Invasive	Europe	BS	Mead *et al.* 2011
<u>MOLLUSCA</u>					
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

Taxon	Occurrence in Saldanha/ Langebaan	Status	Origin	Vector	Reference
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
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

Taxon	Occurrence in Saldanha/ Langebaan	Status	Origin	Vector	Reference
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